Field and model assessment of the ecological impacts of redesigning compensation flow releases

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Field and Model Assessment of the Ecological Impacts of Redesigning Compensation Flow Releases

Volume One

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Thesis submitted for the degree of Doctor of Philosophy

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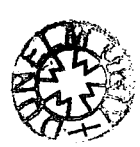
2007



Abstract

Rivers around the world have been regulated for a variety of reasons. Historically the flows released from these reservoirs have been set with minimal attention to ecology. The aim of this study was to use both field and modelling techniques to assess the ecological implications for altering compensation flow regimes in upland millstone grit catchments. A simple paired-catchment approach was chosen. The study catchments used were the regulated Rivelin and Loxley (tributaries of the River Don), and the unregulated Hipper. Before the 1st of April 2004, the steady state compensation releases from the Rivelin and Loxley were set at 2.6 Thousand Cubic Metres per Day (TCMD) and 28 TCMD respectively. After the 1st of April 2004 the compensation flows were altered on the Rivelin and Loxley to 8.6 TCMD and 22 TCMD. This study used three broad scales of enquiry: macroinvertebrates; fisheries and modelling.

Detailed macroinvertebrate surveys were conducted over a four year period (2002-2005), with samples taken in the spring, summer and autumn of each of those years. Surber samples were taken, along with associated environmental variables. Data analysis was conducted using both univariate and multivariate techniques. The invertebrate results showed that the role of flow variability is key within both of the regulated systems. The influence of flow variability is different on each of the study rivers, and the influence changes due to the alteration in compensation flows.

Fisheries surveys were also conducted over this four year study period (2002-2005). Nine sites were surveyed yearly on the Rivelin and Loxley; and three on the Hipper. The fish populations in each of the rivers are dominated by brown trout (*Salmo trutta*). The importance of the interaction of reach scale morphology and discharge was evident in the fisheries populations with the Rivelin having similar observed densities of brown trout as the Loxley. Altering the compensation flow regimes appears to lead to an increase in the growth rate of older trout within the Rivelin.

A two-dimensional hydraulic model was used to simulate the distributions of depths and velocities on four study reaches (two on the Rivelin; two on the Loxley). The model was calibrated to three separate calibration data sets. Subsequently, steady state simulations were conduced for the pre and post change compensation flows in each of the reaches. This output was subsequently linked to a fuzzy-logic based habitat model in order to generate predictions of available habitat. Predictions of available habitat were generated for four life stages of brown trout and for functional feeding groups of macroinvertebrates.

Acknowledgments

Firstly, I would like to thank Stuart Lane for firstly giving me the opportunity to do a PhD and his careful supervision and patience over the past few years. Despite his hectic schedule, he has always had time for my questions. Its been a privilege.

Secondly, I would like to acknowledge the fudnig for the project provided by Yorkshire Water and the Environment Agency. On a personal level I would also like to thank everyone involved in the compensation review group from the Environment Agency and Yorkshire Water. Particular thanks go to Martin Christmas for providing a great link between myself and the Environment Agency and likewise Mark Tinsdeall for Yorkshire Water. Also to Brian Helmsley-Flint for his valuable comments on some macroinvertebrate work.

A big thank-you to all the people I've dragged out paddling in the rivers (in no particular order): Dave Milledge, Mark Smith, Rob Dunford, Mike Lim, Rob Thomas, Amy Tavendale amongst others. Also, thanks to everyone from the EA involved in all the fisheries surveys.

Thanks to all my friends and office mates in both Durham and Leeds, for keeping me sane (and plied with vegetables – you know who you are!), especially for the last few months.

Finally and most importantly, thanks to my parents, without whose influence and guidance I would not be here. Their support throughout the last four years has been unwavering (as always).

Outline of responsibilities

Due to this work being conducted as part of a wider project with the Environment Agency and Yorkshire Water it was thought necessary to define the work conducted by the author specifically. The section below outlines the work undertaken in for the macroinvertebrate and fisheries work; and the details of who conducted the work added.

All of the invertebrate surveys conducted from 2002-2006 were conducted by the author. I did training in macroinvertebrate identification to the species level, however, it became apparent early on in this project, that to identify all the invertebrates as well as all the other work which needed to be done was impossible. Therefore, the invertebrates were identified by Environment Agency approved consultant, following the Environment Agency quality control protocols. All data analysis was conducted by me.

As the fisheries surveys needed four people, they were organised by the Environment Agency, with myself assisting at most of the sites. The scales taken from the trout were read and measured by the National Fisheries Laboratory. HABSCORE surveys were undertaken by trained Environment Agency staff. All data analysis was conducted by me.

All modelling work was conducted by me.

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Chapter 1

Introduction

1.1. Aims and objectives

The aim of this PhD is to use both field and modelling techniques to assess the ecological implications of altering compensation flow regimes in upland Millstone Grit catchments. The PhD has four objectives:

1) to review the potential impacts of sustained compensation flows upon instream ecology (notably macroinvertebrates and fish) in relation to current debates regarding ecological function;
2) to design and to implement field-scale experiments that allow comparison of compensation flow impacts both between catchments and through flow changes within a specific catchment;
3) to assess the potential for developing generic modelling approaches for predicting the impacts of compensation flow changes upon instream ecology; and
4) to synthesise the results from 1 to 3 to develop a conceptual model for how compensation flows impact upon ecosystem form and function.

1.2. Context

Rivers throughout the world have been regulated for a number of purposes (hydroelectric power, water resource management, flood mitigation etc). Rivers within the United Kingdom have long been under pressure from water users, resulting in abstraction from and regulation of the systems. The upland Millstone Grit rivers of the Ridings area of Yorkshire are no exception, with 26 dams constructed on the River Don catchment alone (Firth, 1999). Historically, the flows released from these reservoirs have been set with minimal attention to ecological requirements. In the last 30 years, environmental awareness has increased, and the Water Framework Directive introduced across Europe,

there has been a movement towards defining minimum flow requirements for ecology downstream of reservoirs.

The EU Water Framework Directive (WFD) requires that water bodies, including rivers and lakes reach Good Ecological Status by 2015. The biological elements include the composition and abundance of aquatic flora, and invertebrate and fish fauna (Old and Acreman, 2005). The hydromorphological elements supporting these biological elements include hydrological regime, river continuity, and morphological conditions (Old and Acreman, 2005). The chemical and physio-chemical elements supporting the biological elements include thermal, oxygenation and nutrient conditions (Old and Acreman, 2005). All of these elements may be influenced by the regulation of a river downstream of an impoundment.

In unregulated streams Vannote *et al.* (1980) introduced the "River Continuum Concept", which proposed that understanding of the biological strategies and dynamics of river systems requires consideration of the gradient of physical factors formed by the drainage network. Thus energy input, and organic matter transport, storage, and use by macroinvertebrate functional feeding groups may be regulated by fluvial geomorphic processes (Vannote *et al.*, 1980). The patterns of organic matter use may be analogous to those of physical energy expenditure proposed by geomorphologists (e.g. Leopold and Maddock, 1953). Further, the physical structure coupled with the hydrological cycle form a templet (Southwood, 1977) for biological responses and result in consistent patterns of community structure and function and organic matter loading, transport, utilisation and storage along the length of the river (Vannote *et al.*, 1980). Reservoirs impact on this continuity by creating a spatial disturbance to the continuum, which can be viewed as a discontinuity. Ward and Stanford (1983a, 1983b) describe the Serial Discontinuity Concept (SDC).

In the SDC the biophysical responses were predicted in terms of 'discontinuity distance' and, the upstream or downstream shift of a variable induced by regulation, and by 'parameter intensity', the extent of departure from the natural or reference condition (Stanford and Ward, 2001). From this basic construct, Ward and Stanford then made a series of graphical predictions about the responses of a suite of key biophysical variables in

a river continuum context. The SDC was later expanded to include lateral and vertical connectivity (Ward and Stanford, 1995), based upon the notion that flow regulation, and channel revetment alter exchange of water and materials between river channels and floodplains in relation to natural or reference conditions (Stanford and Ward, 2001). In its expanded form, the SDC incorporates the idea that the relative strengths of lateral, vertical and longitudinal interactive pathways change along the stream corridor (Stanford and Ward, 2001). They envisioned that the relative ecological importance of longitudinal, lateral and vertical pathways likely is a natural function of position or location within the river corridor. Longitudinal connectivity would be most important in the constrained headwaters, the vertical dimension would be most developed in the often braided, middle reaches, and the lateral dimension would be most well developed in the meandering channel of the piedmont and coastal plains (Stanford and Ward, 1995). Stream regulation in all its forms was predicted to compromise or constrain all of these indicative pathways Stanford and Ward, 2001).

The effects of altering baseflow alone are synthesised in Figure 1.1 using a traditional framework first suggested by Petts (1984). The first level of impacts (order 1) indicate that changing the compensation flow will lead to changes in the flow hydraulics of the river, i.e. changes in wetted perimeter, water depth and flow velocity. This in turn will lead to a number of eco-gemorphological impacts (such as bed sedimentology, pH) which are termed order two impacts. Changes in both order one and order two variables could both lead to changes in the macroinvertebrate and fish populations of the river.

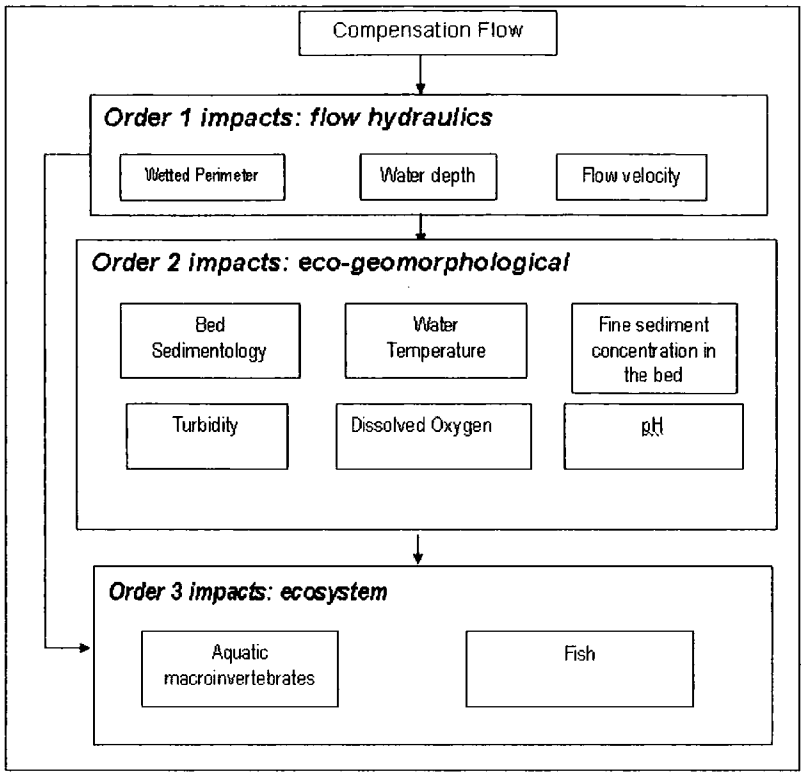

Figure 1.1. Flow diagram to show the potential impacts of altering the compensation flow in the rivers. Adapted from Petts (1984).

The maintenance of minimum flows is only one of a range of considerations necessary for the protection or rehabilitation of fluvial hydrosystems and two important components need to be added to the simple formula in Figure 1.1. First, regulated rivers are still subject to some flow variation related both to reservoir overtopping and to tributary inputs. Because fluvial processes maintain a dynamic mosaic of channel habitat structures (Leopold *et al.*, 1964), creating patchy and shifting distributions of environmental factors that sustain diverse biotic assemblages, hydrological variation is now recognised as a primary driving force within river ecosystems (e.g. Sparks *et al.*, 1990; Schlosser, 1991; Sparks, 1995; Stanford *et al.*, 1996). Each of these types of variability will be superimposed upon the base compensation flows, and will impact differently on different organisms and different life stages of those organisms. As most aquatic species possess life history traits that enable individuals to survive and reproduce within a certain range of environmental variation (Townsend and Hildrew, 1994; Stanford *et al.*, 1996). Indeed, hydrological variation is now recognised to play a major part in structuring biotic diversity within river ecosystems as it controls key habitat conditions within the river channel, and hyporheic

zones (e.g. Poff and Ward, 1989; Townsend and Hildrew, 1994; Richter *et al.*, 1996; Stanford *et al.*, 1996).

Richter *et al.* (1997) suggest that accumulated research on the relationship between hydrological variability and river integrity overwhelmingly suggest a natural flow paradigm, which states: the full range of natural intra- and inter-annual variation of hydrological regimes, and associated characteristics of timing, duration, frequency and rate of change are critical in sustaining the full native biodiversity and integrity of aquatic ecosystems. Numerous streamflow characteristics are important for the maintenance of riverine habitats and biological diversity, including: the seasonal patterning of flow; timing of extreme conditions; the frequency, predictability, and duration of floods, droughts and intermittent flow; daily seasonal, and annual flow variability, and rates of change (Resh *et al.*, 1988; Poff and Ward, 1989; Walker *et al.*, 1995; Richter *et al.*, 1996).

Second, the movement of water across the landscape will influence the ecology of rivers across a broad range of temporal and spatial scales (Vannote *et al.*, 1980; Sparks, 1995). The shape and size of river channels, the distribution of riffle and pool habitats and stability of substrate are principally determined by the interaction between the flow regime and local geology and landform (e.g. Frissel *et al.*, 1986; Cobb *et al.*, 1992; Newbury and Gaboury, 1993). In turn, this complex interaction between flows and physical habitat is a most important determinant of the distribution abundance and the diversity of stream and river organisms (Schlosser, 1982; Poff and Allan, 1995; Ward *et al.*, 1999). Furthermore, the impact of hydrological variability can only be understood with respect to the variability of habitat in space through time.

The majority of what aquatic ecologists know or believe about the biotic consequences of flow alteration has been derived from comparisons of dammed versus undammed rivers (e.g. Copp, 1990); from inferences drawn from (relatively short-term) observations of flow and fluvial processes (Petts, 1979, 1980; Lyons *et al.*, 1992); or from measured differences in fish or invertebrate communities at increasing distances downstream from dams (invertebrates: Voelz and Ward, 1991; Moog, 1993; fish: Kinsolving and Bain, 1993). Virtually all such studies have statistical weaknesses that limit inferences regarding causation between biota and flow (e.g. Kinsolving and Bain, 1993; Richter *et al.*, 1996),

because flow perturbations cannot be replicated or randomly assigned to experimental units (Hurlbert, 1984; Carpenter, 1989; Carpenter *et al.*, 1989; Stewart-Oaten *et al.*, 1992).

Herein lies the difficulty for instream ecology studies such as this one. The literature, as cited above, tells us that hydrological variability is very important for ecosystems in general. The impact of this variability is influenced by channel geometry over a number of scales, and with the variability impacting upon different organisms and different life stages differently. However, this variability cannot be controlled (as it is driven by environmental forcing combined with catchment characteristics). Therefore, this variability needs to be included when assessing invertebrate and fish populations, recognise that the impacts of flow variability cannot be disentangled from any potential compensation flow impacts, as changing compensation flow magnitude will change the depth of storage in the reservoir, the magnitude and frequency of overtopping and hence instream hydrological variability.

1.3. Explanation of objectives

Objective one: to review the potential impacts of sustained compensation flows upon instream ecology (macroinvertebrates and fish) in relation to current debates regarding ecological function;

This objective is needed in order to establish the academic background for this study. Literature reviews will be provided for macroinvertebrates (Chapter 3) and fish (Chapter 4). There are three reasons why these are provided: (i) to enhance the context and the justification for the study; (ii) to justify the methodologies used, as they must be consistent and logical with known organism constraints; and (iii) to provide information against which to relate the results presented, highlighting the position of this work within the academic literature.

Objective two: to design and to implement field-scale experiments that allow comparison of compensation flow impacts both between catchments and through flow changes within a specific catchment;

A field-based approach was taken so that any changes to instream ecology could be observed in the real world, and any future modelling efforts could be informed by this fieldwork. The case-study sites chosen will be introduced and justified in Section 1.4. The potential impacts of existing compensation flows will be manifest in two ways: (i) within river characteristics; and (ii) differences between rivers with differing compensation flow inputs. Any impacts of altering the compensation flows could also be observed either by observing within river changes through time as the flows are adjusted or by changes in the relationship between nearby catchments in response to the flow change. Therefore, a fieldwork survey needed to be designed which incorporated sampling under the pre and post-change compensation releases, on two adjacent catchments. A control catchment was also used to enhance further the investigation of the between river differences.

Objective three: to assess the potential for developing generic modelling approaches for predicting the impacts of compensation flow changes upon instream ecology.

Despite the logic of the field-based methodology outlined above, the associated costs can be very high. The costs of fieldwork escalate when there is a need to investigate similar problems in other catchments (as there is in this case), or more than one aspect of instream ecology. One approach which can be used to lessen such fieldwork costs is modelling. When developing a model, large amounts of field data may have to be collected so that the model can be run, calibrated and validated. However, once the model has been calibrated and validated for the original rivers, it may be applied to other catchments with appropriate thought. The amount of field data then required depends on the structure of any given model. One further advantage of modelling approaches is that they can often include many aspects of instream ecology, which otherwise would be expensive to sample using a fieldwork approach. The final advantage of a modelling approach is that it should have the ability to make predictions and so aid the decision maker.

Objective four: to synthesise the results from 1 to 3 to develop a conceptual model for how compensation flows impact upon ecosystem form and function.

The results produced by investigating Objectives two and three will be site specific, but they will have significance in relation both to general instream ecology questions and also the question of how compensation flows impact upon ecosystem form and function. There are many catchments other than the study catchment, in which compensation flows are used to maintain streams. In these catchments, regulatory bodies such as the Environment Agency need to know: (i) whether there any ecological impacts from the current compensation flow levels; and (ii) if there are situations where the compensation flows could be changed, either for operational reasons by the water company or for the purpose of improving instream ecology. There must therefore be a conceptual model which allows assessment of the impacts of the existing compensation flows, or the potential impacts of altering the compensation flows. Not only will this conceptual model aid decision makers and regulatory bodies in understanding of the impacts of compensation flows, it will also aid in the contextualisation of the site-specific results presented in this thesis.

1.4. Case study site and justification

A catchment scale case-study approach was adopted because the impact of existing and altering compensation could be felt across the entire catchment. Compensation flows were altered on only two study rivers because:

1) it maximises the cost-effectiveness for the ecological sampling, thereby allowing a number of scales of variation to be included in the sampling processes (temporal, river-scale variation; reach-scale variation); and
2) it is difficult to change the compensation flow with regard to consulting and advertising the change to riparian land-owners and interested parties.

It is hoped that the work conducted on this example study site can be transferred to other nearby catchments, by means of both the conceptual and ecological models developed.

There were a number of criteria which had to be met when choosing the study sites. First, the compensation flows had to be able to be changed with relative ease, as some changes in compensation flows require Acts of Parliament, and so it was preferable to avoid such a

process if possible. Second, the case-study rivers must be representative of upland Millstone Grit catchments. Third, hydrological data (rainfall, discharge, overtopping data) must be available for the catchment.

The rivers chosen as the case-study sites were the Rivelin and Loxley which are eastward flowing impounded tributaries of the River Don. The two rivers join at Malinbridge, approximately two kilometres upstream of the confluence with the River Don in the Owlerton district of northwest Sheffield. What makes the Rivelin-Loxley a convenient system is the fact that the compensation flows are set by law at 30.6 Thousand Cubic Metres per Day (TCMD) at the confluence of the two rivers. It is therefore possible (with the minimum of legal wrangling) to alter the compensation flows in each of the rivers for a trial such as this so long as the total compensation flow at the confluence is 30.6 TCMD. Furthermore, the proximity of the two rivers, makes this an ideal twinned-catchment study. The impacts of altering the compensation flows can therefore be assessed both in terms of an increase and decrease in compensation flows in adjacent rivers which are assumed to have identical prevailing environmental conditions.

The compensation flows of the Rivelin and Loxley were changed on the 1st April 2004. Before the 1st April 2004, the compensation flow was set at 2.6 TCMD on the Rivelin and 28 TCMD on the Loxley. After this date, the compensation flow was set at 8.6 TCMD on the Rivelin and 22 TCMD on the Loxley. The geology of both catchments is made up of exposed Millstone Grit at the western end of the catchment and Coal Measures overlying the Millstone Grit in the eastern half of the catchment. There is also a control catchment, which is the River Hipper, an unregulated river draining a predominantly rural catchment to the west of Chesterfield. Further information on the characteristics of the study catchments and their histories will be provided in Chapter 2.

1.5. Thesis structure

Figure 1.2 maps the structure adopted to this thesis and the links between chapters. The second chapter of this thesis will discuss the history of impoundment, impacts of impoundment on the hydrology of the Rivelin and Loxley, and examine in detail the discharge records of each these rivers. This chapter will also introduce the concept of discharge variability in regulated systems such as these, with reservoir overtopping data also used.

The following two chapters of this thesis cover the results from the ongoing monitoring of invertebrates (Chapter 3) and fish (Chapter 4) respectively. Each of these chapters is similarly structured (Figure 1.2), with a literature review focussing on the aspect of ecosystem of interest with respect to flow regulation, flow alteration and flow variability. Four main research aims are then addressed in each of these chapters: (i) establishment of the structure and quality of the populations; (ii) assessment of the natural variability of the populations; (iii) establishment of whether the existing compensation flows are having an impact; and (iv) assessment of any change to the populations resulting from altering the compensation flows. Both chapters end with a discussion, in which the methodology is assessed, and links between the presented results and existing literature outlined.

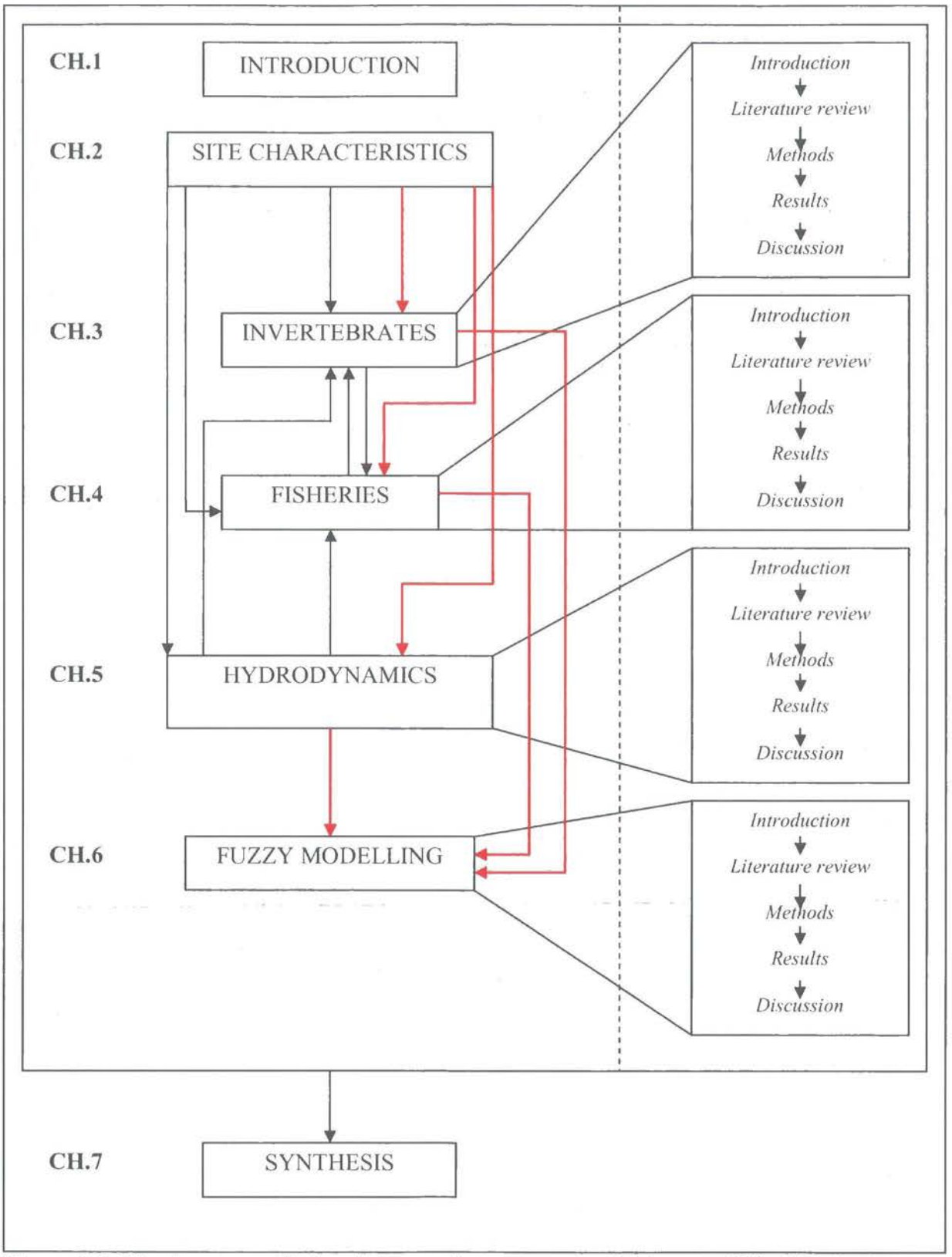

Figure 1.2. Thesis map outlining the overall structure and linkages within the thesis. Black arrows indicate information flows in reality whilst the red arrows indicate data flow within the thesis.

The results of the hydraulic modelling are presented in Chapter 5 (Figure 1.2). Again, the chapter starts with a section outlining the aims and objectives of the chapter, which is followed by a literature review and critique of hydraulic modelling techniques in order to establish a precedent for the methodology used. Subsequently the methodology is presented in detail in Section 5.3. The first results section constitutes a discussion of the calibration of the model and discusses model sensitivity to parameterisation and mesh density. The next results section addresses model reliability in terms of the effects of varying model parameters on the distribution of depths and velocities at one of the study sites. The third results section assesses the primary impacts of altering the compensation flows (i.e. do the distributions of velocity and depth change with the alteration in compensation flows). This is followed by a discussion where the results presented are assessed and linked to the literature review cited in the review.

The first section of Chapter 6 reviews the literature pertaining to the modelling of habitat based on hydraulic predictions (Figure 1.2). The second section identifies, justifies and explains in detail the fuzzy approach taken in this chapter. The first of the results sections addresses the sensitivity of the fuzzy rules to uncertainties in the predicted velocities and depths. The next two sections address the predicted habitat available for both macroinvertebrate guilds and brown trout in terms of the pre and post change discharges, whilst validating (or otherwise) the model with respect to the results presented in Chapters 3 and 4 respectively (information flow shown in Figure 1.2).

Chapter 7 is a synthesis of the previous three chapters. It is the purpose of this chapter to tie together the threads of previous four chapters in order to provide a perspective on this habitat study as well as a reflection of broader conclusions. The extent to which the three disciplines complement each other and the extent of integration achieved in this thesis will also be discussed. Results from this work and the work presented in the existing literature will be compared. The weaknesses and problems associated with the methodologies used in this research will also be included in this synthesis. A final section provides final conclusions.

Chapter 2

Study site characteristics

2.1. Introduction

The rationale behind the choice of the study sites was introduced in Section 1.0. It is the purpose of this chapter to provide further information and justification of the study sites, the importance of which is highlighted by the number of linkages between this chapter and subsequent chapters illustrated in Figure 1.2. This will be achieved by addressing four aims:

a) to introduce and justify the choice of study site;
b) to establish the general catchment characteristics and to illustrate why these are representative of other catchments;
c) to review the historical dimensions of use and flow regulations in the catchment; and
d) to characterise the time periods when the ecological results reported in chapters 3 and 4 were made.

This chapter is split into two main sections. The first section assesses the geomorphology of the systems, firstly in terms of the upstream regulation of the Rivelin and Loxley; secondly, the instream regulation and history of the study rivers; thirdly the rainfall inputs into the respective systems and finally the contemporary geomorphology is assessed. These are the large scale controlling factors upon which the compensation flows will be superimposed. The second section of this chapter focuses on the hydrology of the systems assessing the influence of impoundment on the hydrology of the Rivelin and Loxley; and introducing the interaction of geomorphology and hydrology.

2.2. Geomorphology of the systems

This section aims to outline the large-scale geomorphological features of each of the systems. This section begins with an outline of the upstream regulation present on the Rivelin and Loxley. The second section addresses the instream regulation and history of each of the systems. The third section presents the rainfall impacts on each of the systems and the final section provides information on the contemporary geomorphology of the systems. Each of these is a large-scale factor upon which compensation flow effects are superimposed.

2.2.1. Upstream regulation

Figure 2.1 shows the location of impoundments on the Rivelin and Loxley, and Table 2.1 summarises their characteristics. With six reservoirs on the Rivelin and four on the Loxley, it is clear that these rivers have been impacted heavily on by regulation. The length of time that regulation has been operational extends back to the nineteenth century, with the Rivelin regulated since 1834 and the Loxley since 1859. The reservoirs from which the compensation flows are released are shown in bold in Table 2.1.

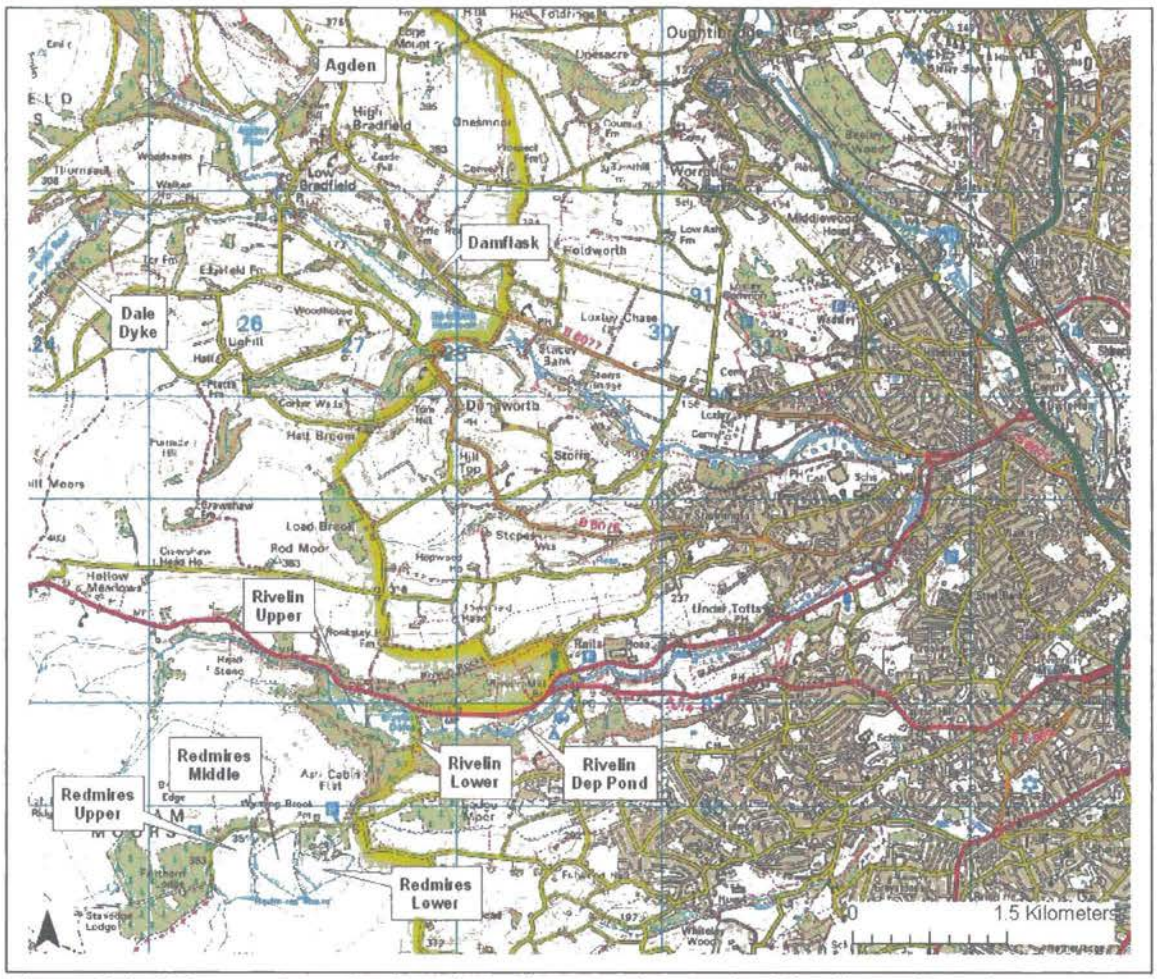

Figure 2.1. Map to show the location of reservoirs on the Rivelin and Loxley.

RIVELIN					
Name	**Grid Ref**	**Year Built**	**Storage Capacity (TCM)**	**Max Depth (m)**	**Surface Are (hectares)**
Redmires Upper	SK 250 855	1854	1423	13.5	23
Redmires Middle	SK 264 855	1836	784	11.5	19
Redmires Lower	SK 268 855	1849	566	12.5	12
Rivelin Upper	SK 271 868	1848	220	9.1	4
Rivelin Lower	SK 277 867	1848	525	14.6	12
Rivelin Dep. Pond	**SK 287 869**	**1869**	**36**	**6.1**	**1.6**
LOXLEY					
Name	**Grid Reference**	**Year Built**	**Storage Capacity (TCM)**	**Max Depth (m)**	**Surface Are (hectares)**
Strines	SK 232 905	1869	2059	N/A	22
Dale Dyke	SK 243 917	1875	2118	N/A	25
Agden	SK 261 923	1869	2541	N/A	25
Damflask	**SK 284 907**	**1896**	**5037**	**N/A**	**47**

Table 2.1. Table to show some characteristics of the Rivelin and Loxley reservoirs (Adapted from Firth, 1999). Maximum depth data not available for the Loxley. The reservoirs from which the compensation flows are released are shown in bold.

The Loxley has not only been impacted upon by the presence of regulation, weirs and mills. A much more dramatic man-induced phenomenon has also had a large impact upon the system. On the 11th March 1864, the Bradfield reservoir, situated 8 miles to the west of Sheffield burst its banks. It was a large reservoir covering around 76 acres and contained 691, 000, 000 gallons of water (Draper, 1995). The gap in the reservoir created during the burst was at least 110 yards wide at the top, and 70 feet deep. It was said that the velocity of the flood was 18 mph (Draper, 1995). In the early stages of the flood, trees were eroded, there was substantial valley side erosion, and large rocks transported by through the river. One stone alone entrained by the flood was said to have weighed 60 tons (Draper, 1995). Figure 2.2 illustrates the catastrophic nature of the flood and the large amount of sediment deposited. It is probable that this event had a long-term impact on valley morphology.

Figure 2.2. Figure to show Rowell's Wheel after the flood. From Draper (1995).

2.2.2. Instream regulation

2.2.2.1. Rivelin

Very little is known of the extent to which the waters of the Don and its tributaries were used by man in the middle-ages (Crossley, 1989). However, by the 1800s, the Rivelin became documented in greater detail. During the industrial revolution, the Rivelin was used as a power source to drive numerous mills. By the late 1840s, the Rivelin was described as very open in aspect, as trees were probably in very short supply, having been removed for fuel for the various mills and their shade would not have been tolerated in view of the fact that the interiors of the mills were badly lit (Robinson, 1989). In 1863, the Rivelin was seen as a place of great beauty: "running immediately below this is the beautiful stream and ravine of the Rivelin, which in its course, passes through some of the most delightful scenery it is possible to conceive, at one place passing under a rustic bridge, in others flowing tortuously among the grey rocks in the bed of the stream, and delighting the ear with its gentle murmurs" (Zimmerman, 1863).

Local poet Ebenezer Elliott wrote:

"Beautiful River! Goldenly Shining,
Where, with the cistus, woodbines are twining.
Birklands around thee, Mountains above thee,
Rivelin wildest! Do I not love thee?"

In order to provide a reliable source of water for the mills, some water from the Rivelin was redirected to mill-ponds using weirs. Even though many of the ponds only held enough water for one day of operation, they protected the mill owners from variations in discharge due to other operations upstream. Mill ponds are now considered to be an important habitat resource for invertebrates, plants and amphibians (Wood and Barker, 2000). Table 2.2 provides information on the mills. The abundance of the weirs is an important geomorphological feature of the Rivelin, limiting changes in flow vertically and to some extent laterally.

Mill Name	Grid Ref	Year Built	Information
Rivelin Mill	SK 291 873	1600	This mill was used for the grinding of corn and was owned by the Earl of Shrewsbury.
Upper Coppice Wheel	SK 293 873	1736	
Second Coppice Wheel	SK 294 873	1736	Locally known as the Darwin wheel. It included a grinding mill and a wire drawing mill.
Third Coppice wheel or Paper Mill	SK 295 873	1758	This mill had the longest tail goit in the valley. The reason for this is that the wheel pit was set below the level of the river and to stop backwatering stopping the wheel the water had to be taken a long way down the river before being reintroduced. The mill ground cutlery until 1794, but by 1814 the mill was converted to a paper mill.
Frank Wheel	SK 300 874	1737	Used to grind cutlery in its early years, but by 1854 it had been converted to a paper mill.
Wolf or Rocher Wheel	SK 302 875	1722	One of the biggest dams in the valley, the Wolf wheel measured 15ft by 6ft.
Swallow Wheel	SK 305 875	1692	
Plonk Wheel	SK 307 876	1737	The mill could have been abandoned as early as 1814.
Hind (or Iron) Wheel	SK 309 876	1581	The oldest dam in the valley having first being recorded in 1581 and having been rebuilt around 1820.
Upper Cut Wheel	SK 312 878	Unkown	The dam for this wheel was long and thin and was said to be used for the hire of rowing boats.
Nether Cut or New Wheel	SK 313 879	1718	
Little London Wheel	SK 314 880	1752	
Holme Head Wheel	SK 315 881	1794	The holme head has some of the best examples of a pen trough and wheel spindle still remaining, and there is a good example of an overflow, which still exists.
Roscoe Weir	SK 317 883	1725	Used for the better part of its life as a grinding hull for fender and saw.
New Dam		1853	The last dam to be built in the valley.
Spooner Wheel	SK 321 884	1794	Used mainly for grinding of scythes, saws, files and cutlery, part of the complex was used for the forging of knives.
Rivelin Bridge Wheel (Hollins Bridge)	SK 322 885	1794	Used mainly as a cutlers mill for most of its working life, but in its latter years was also used for grinding optical glass.
Walkley bank tilt (Havelock Dam)	SK 324 888	1750	Initially a cutlers wheel, it was converted to a tilt forge in 1764, and then a wire mill in 1901.
Mousehole Forge	SK 325 891	1628	Became known worldwide for the manufacture of its famous anvils. Converted to an iron forge from a lead mill in 1664. Anvils from the Mousehole won gold medals in 1851 at the great exhibition at Crystal Palace.

Table 2.2. Table to show the history and location of mills on the Rivelin.

2.2.2.2. Loxley

The Loxley is possibly the most industrialised of Sheffield's rivers, but again little is known of the river until the 19th century. In the mid 19th century the Loxley was described as follows: "About three miles above the point of its confluence with the Don, the valley through which it flows opens out in a style of great interest and beauty: steep rugged cliffs, strongly broken or gently undulating ground, precipitous banks, on the one hand, or on the other, sloping fields; the whole space on the south and rocky side being more or less, clothed with the native Birch, or other wood" (Holland, 1837).

The details of mills on the Loxley are presented in Table 2.3. There is more literature on the mills of the Rivelin than the mills of the Loxley and so this table is less complete. The impacts of the mills and the weirs will be the same on the Loxley as on the Rivelin. As with the Rivelin, the weirs on the Loxley are not vertical and are possibly passable by brown trout.

Mill Name	Grid Ref	Year Built	Information
Damflask or Dunham Park or Whitman Wheel	SK 278 909	1750	
Stacey Wheel	SK 286 905	1749	
Old Wheel or Loxley Plane Wheel	SK 295 898		
Rowell Bridge Wheel	SK 299 895	1734	
Storrs Mill	SK 296 891		
Loxley Wine Mills (Storrs Brook)	SK 299 895	1693	
Olive Wheel	SK 304 895		
Cliff Wheel	SK 308 894	1737	
Lower Cliff or Boggey Or Low Matlock Wheel	SK 309 894	1732	
Ashton Carr Wheel	SK 312 895	1557	
Green Wheel	SK 314 897		
Glass Tilt	SK 315 897		
Broadhead Wheel	SK 316 897		
Wisewood Scythe Wheels or Forge	SK 320 895		
Wisewood Forge and Rolling Mill (Bradshaw Wheel)	SK 324 895		
Malinbridge Grinding Wheel or Corn Mill	SK 325 894		

Table 2.3. Table to show the location and history of mills on the Loxley.

2.2.3. Rainfall inputs

2.2.3.1. Rivelin and Loxley

A further catchment characteristic, which must be considered, is the rainfall inputs, as this is a primary control on the augmentation of the rivers. In this analysis, Redmires Filters rain gauge was used to represent both the Rivelin and the Loxley catchments, as it had a record that extends from 1974 to the present day and is proximal to each of the rivers. Figure 2.3 shows the variability between years in terms of annual rainfall, but a very slight increase in rainfall from 1974-2002.

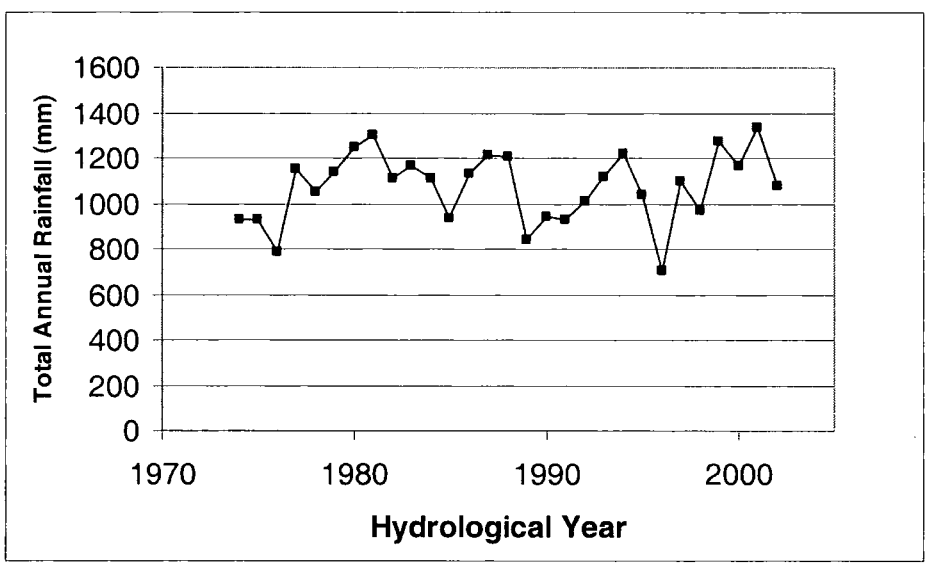

Figure 2.3. Graph to show the total annual rainfall at Redmires Filters rain gauge, from 1974-2002.

Figure 2.4 shows a general increase in the percentage of days on which rainfall exceeded 20mm from 1974 to 2002. The most interesting aspect of both Figures 2.3 and 2.4 shows the between year variability both in terms of the total annual rainfall and the number of intense rainfall events. It would be expected that if these rivers were unregulated, this variability would be conveyed into variations in the discharge of the rivers.

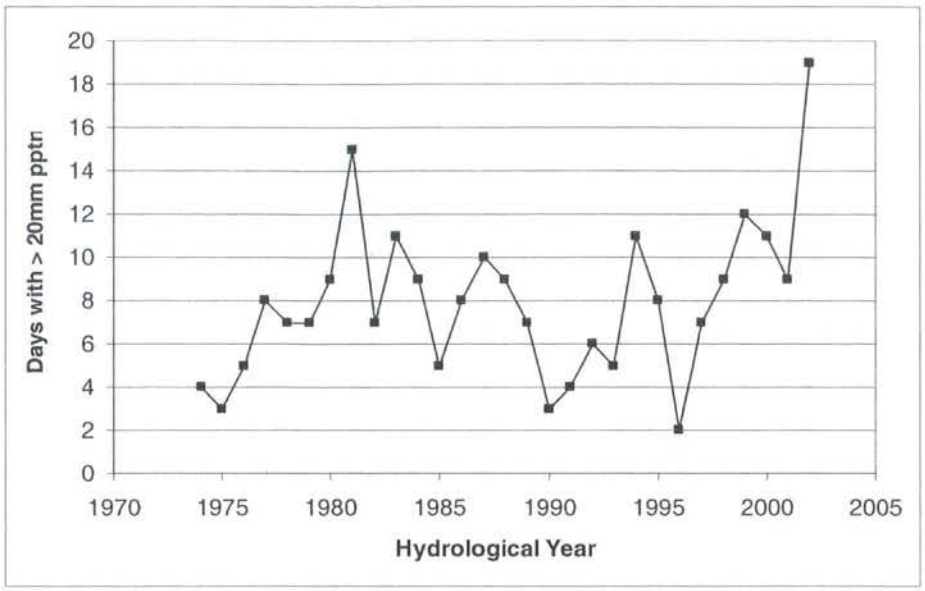

Figure 2.4. Graph to show the number of days when rainfall exceeded 20mm at the Redmires Filters rain gauge from 1974-2002.

2.2.3.1. Hipper

For the Hipper, rainfall records from Upper Linacre (SK 338726) rain gauge were used, which again is proximal to the river. These records stretch from 1961 to the present day. Figure 2.5 shows the variation between the years in terms of annual rainfall, and also displays very little trend.

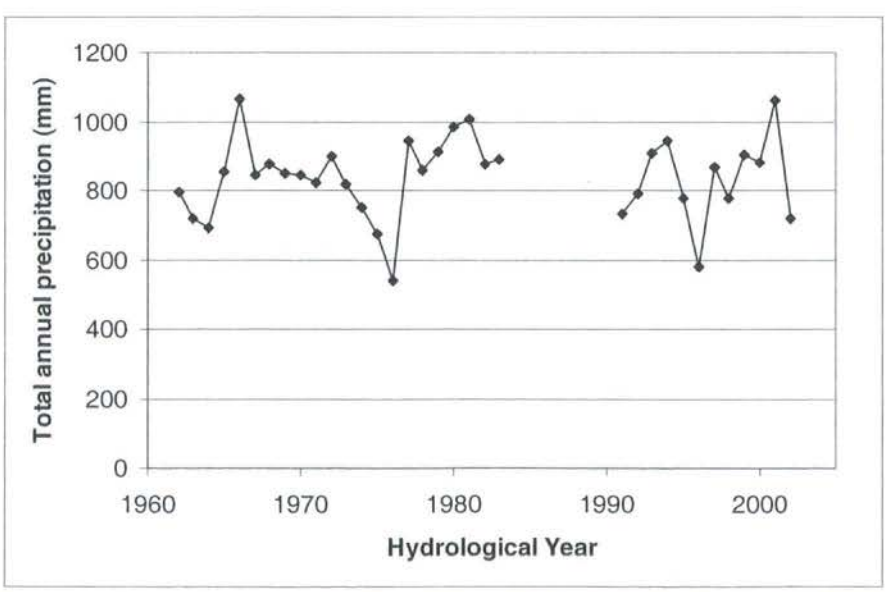

Figure 2.5. Graph to show the total annual rainfall from 1962-2002 at the Upper Linacre Filter raingauge.

As with the Redmires Filters rainfall data, further analyses were performed, as one of the main drivers of hydrological variability are large rainfall events (especially in unregulated catchments). Figure 2.6 shows an upwards trend in the number of days when the rainfall exceeded 20mm. This indicates an increase in the intensity of the rainfall within the catchment, which would also suggest (combined with the variability in total annual rainfall) an increased hydrological variability in the system. The Hipper is unguaged so no direct comparisons of discharge can be made.

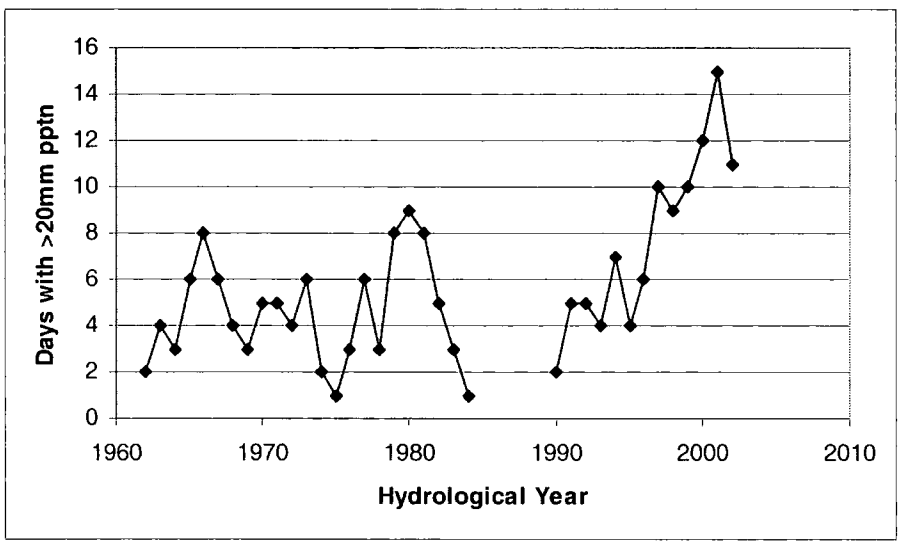

Figure 2.6. Figure to show the percentage of days where rainfall exceeded 20mm at Linacres rainguage (1962-2002).

2.2.4. Contemporary geomorphology

Another point of interest when introducing the study catchments is to discuss the catchment characteristics as, via a number of processes (e.g. geology controls on bedrock; riparian land use etc), they are a key control on the rivers. A survey was conducted on each of the rivers in November 2002 in order to assess the contemporary geomorphology. This took the form of recording instream and riparian attributes at around 40 cross sections along the length of the river. The audit sheet used was slightly modified from that proposed by Thorne (1998). In an effort to include all the weirs whilst retaining a sensible number of study cross-sections some of the riffle dominated areas of the Rivelin may not have been sampled.

2.2.4.1. Rivelin

The geology of the Lower Don catchments comprises Carboniferous Millstone Grit and Westphalian Series. An important feature of any river is the riparian land-use, as any activities undertaken in the areas surrounding the channel itself may have an impact upon the river. Figure 2.7 shows that the predominant land-use around the Rivelin is set-aside. The set-aside land on the Rivelin contains a large proportion of vegetation and trees, which provide shade and a source of allochthonous material for the Rivelin.

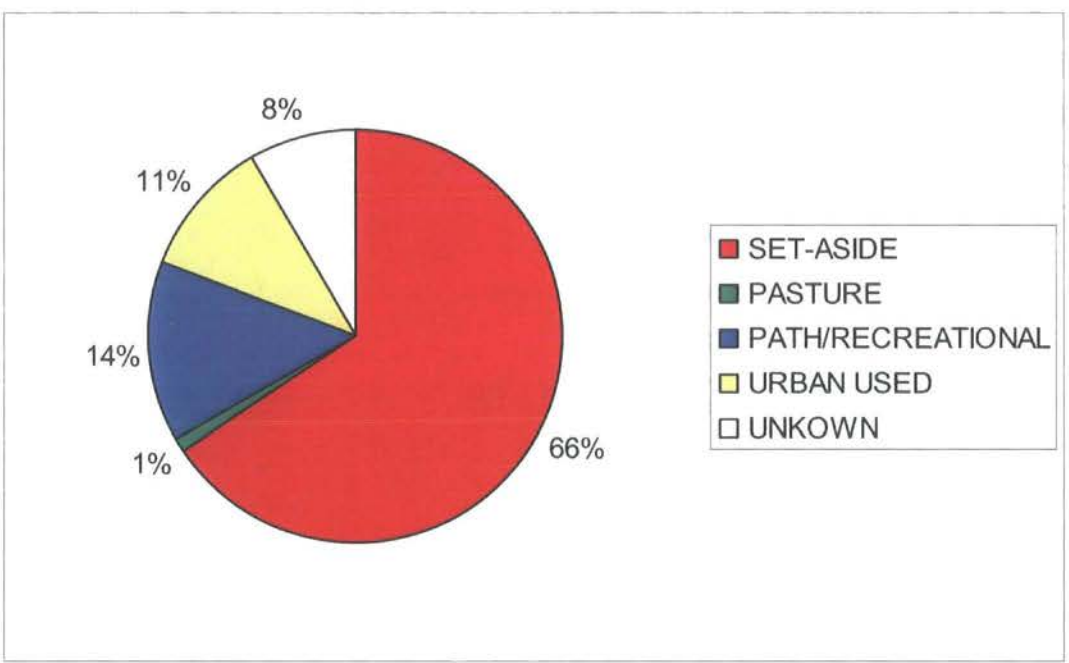

Figure 2.7. Pie- chart to show the proportions of riparian land-use sampled during the geomorphological survey on the Rivelin.

It follows therefore, that there are generally high levels of canopy cover on the Rivelin, with the majority of cross-sections having a canopy cover of greater than 35% (Figure 2.8). Only 11 out of 57 cross-sections had a canopy cover of less than 35%. Raleigh *et al.* (1986) suggested that the minimum cover for good adult brown trout habitat was 35%. This indicates good conditions for brown trout in terms of shade provided by canopy cover in the Rivelin.

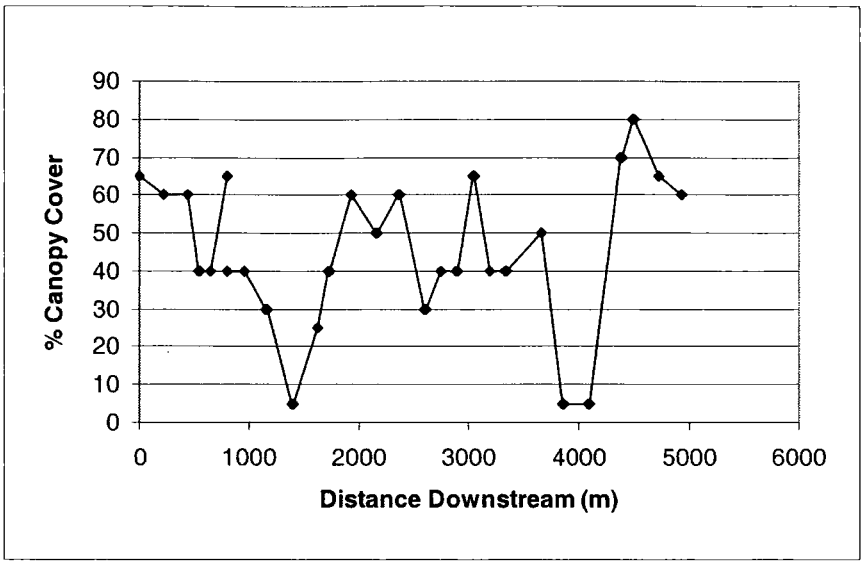

Figure 2.8. Graph to show variation in percentage of canopy cover sampled during the geomorphological survey in the Rivelin.

2.2.4.2. Loxley

The Loxley shares the same geology as the Rivelin: Carboniferous Millstone Grit and Westphalian Series. The predominant land-uses surrounding the Loxley are urban wasteland, pasture and recreational (Fig. 2.9). The presence of a large amount of urban wasteland (Fig. 2.10) is important. These sites are remnants of more recent heavy industry, and have non-permeable ground-surfaces which will lead to a rapid delivery of runoff to the Loxley. More importantly, these sites are abandoned, and it is unknown what chemicals could be washed into the Loxley from them. As well as the potential for pollution from present day rainfall events, the industries using these sites in the past may have polluted the river to such an extent that the Loxley may still be recovering. Also in these areas of 'urban wasteland', there is a very little vegetation cover and so the channel has less shade than the Rivelin, as evidenced by Figure 2.11 where the upstream areas of the Loxley have less canopy cover. However, levels of canopy cover were still above 35% for 18 out of 31 sampled sites.

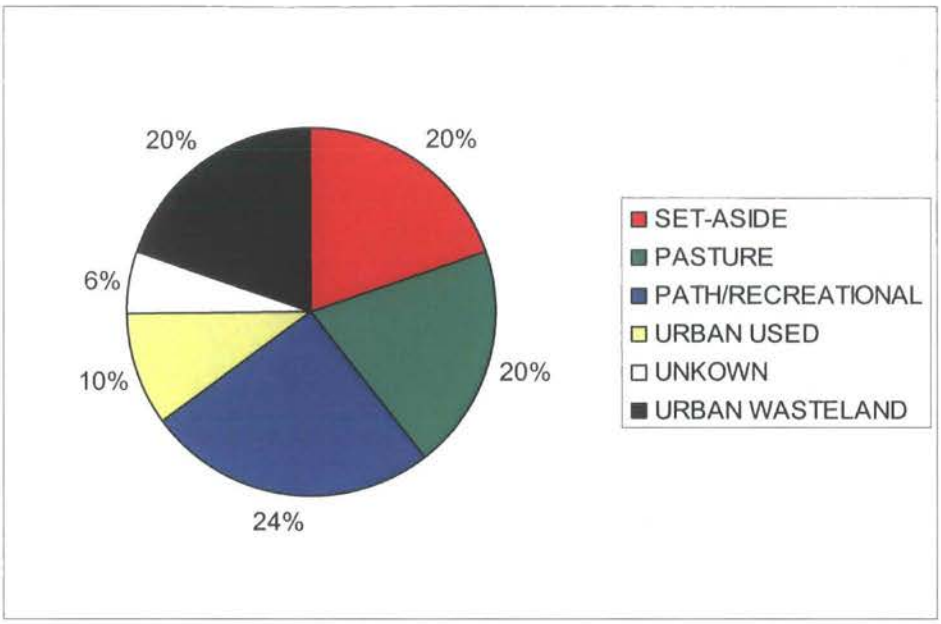

Figure 2.9. Pie-chart to show the proportion of riparian land-use sampled during the geomorphological survey on the river Loxley.

Figure 2.10. Photograph to show an example of the urban wasteland in the upper reaches of the Loxley.

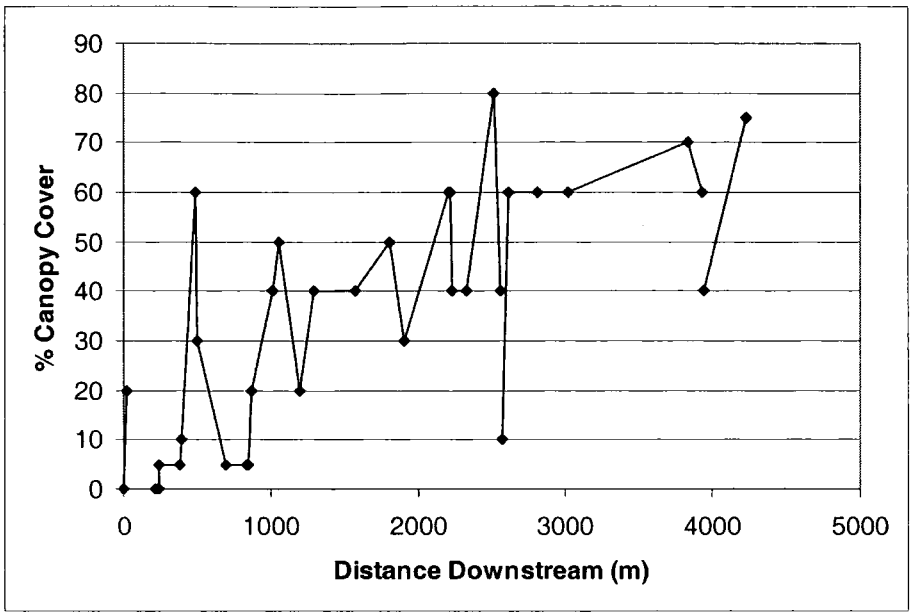

Figure 2.11. Graph to show percentage canopy cover sampled during the geomorphological survey along the Loxley.

2.2.4.3. Hipper

There has been much less written about the history of the River Hipper than that of the Rivelin and Loxley. The length of the river Hipper used is much less than that of the Rivelin and Loxley as it was thought desirable to assess the River upstream of Chesterfield in the same way that the Rivelin and Loxley are upstream of Malinbridge (Figure 2.12). As on the Rivelin and Loxley, a gemorphological survey was conducted in November 2002, in order to assess the contemporary geomorphology of this river.

Figure 2.12. Figure to show the location of the River Hipper, and the area of the river used in this study.

The dominant land-uses surrounding the Hipper are set-aside and recreational (Figure 2.13). This means that although the areas surrounding Hipper may have been modified by humans it was not for the purposes of industry. Figure 2.14 shows that due to the nature of the riparian land-use around the Hipper, there always appears to be canopy cover apart from the areas immediately downstream of the weir, which is where the some urbanisation impacts are noted.

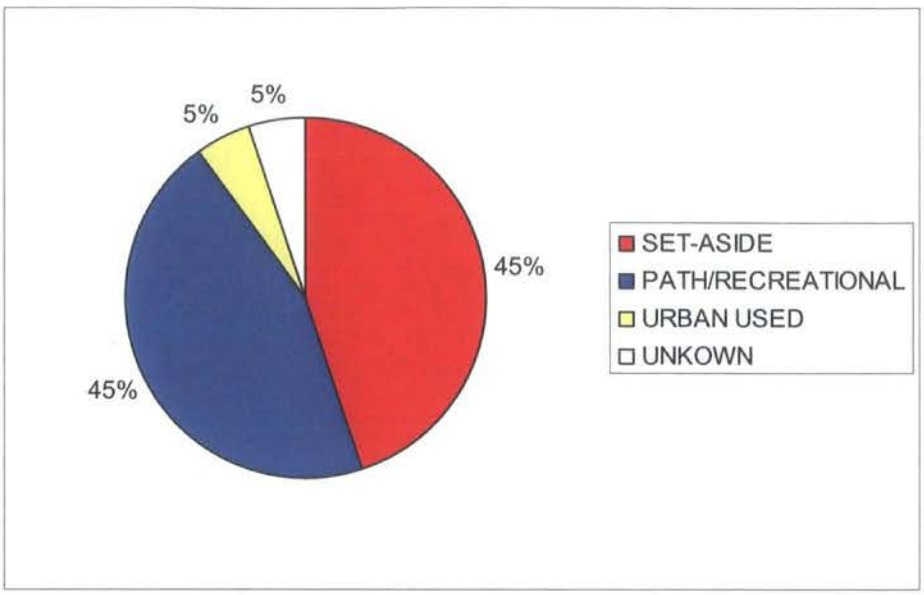

Figure 2.13. Pie chart to show the proportions of riparian land-use sampled during the geomorphological survey on the river Hipper.

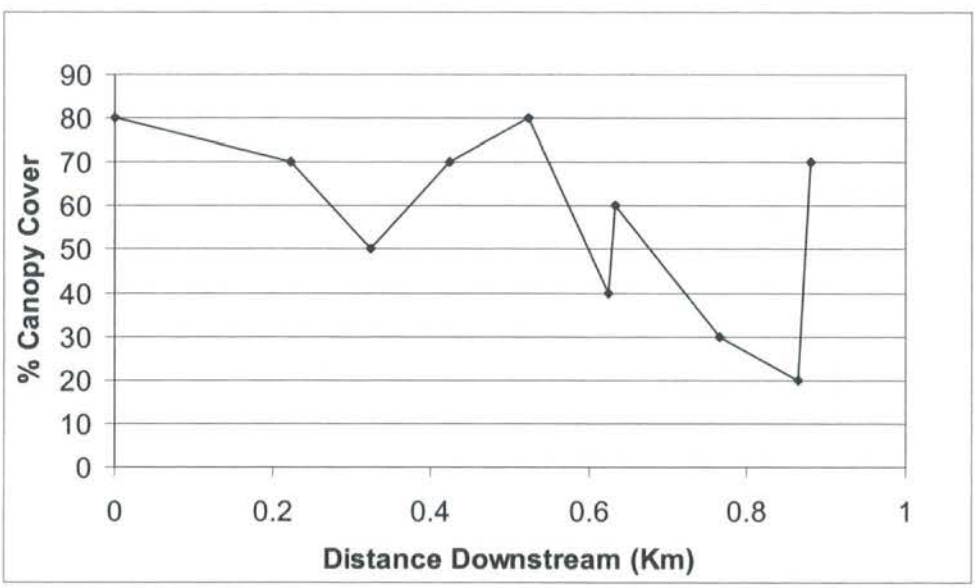

Figure 2.14. Graph to show the percentage of shade provided by the overhead canopy on the Hipper, sampled during the geomorphological survey.

2.3. Hydrological characteristics

2.3.1. Reservoir inputs

As stated in the introduction, the compensation flows in the Rivelin and Loxley were changed on the first of April 2004. This section highlights any changes in hydrology over that period and introduces the importance of hydrological variation in regulated streams. Hydrological variation plays a major part in structuring the biotic diversity within river ecosystems as it controls key habitat conditions within the river channel, the floodplain, and hyporheic zones (e.g. Poff and Ward, 1989; Townsend and Hildrew, 1994; Richter *et al.*, 1996; Stanford *et al.*, 1996). In this section, the hydrology of the systems for the entire flow record is examined. The information used in this section is derived from overtopping events on the compensation reservoirs and two gauging stations (on the Rivelin and Loxley) (Figure 2.15). Unfortunately the Hipper is not gauged.

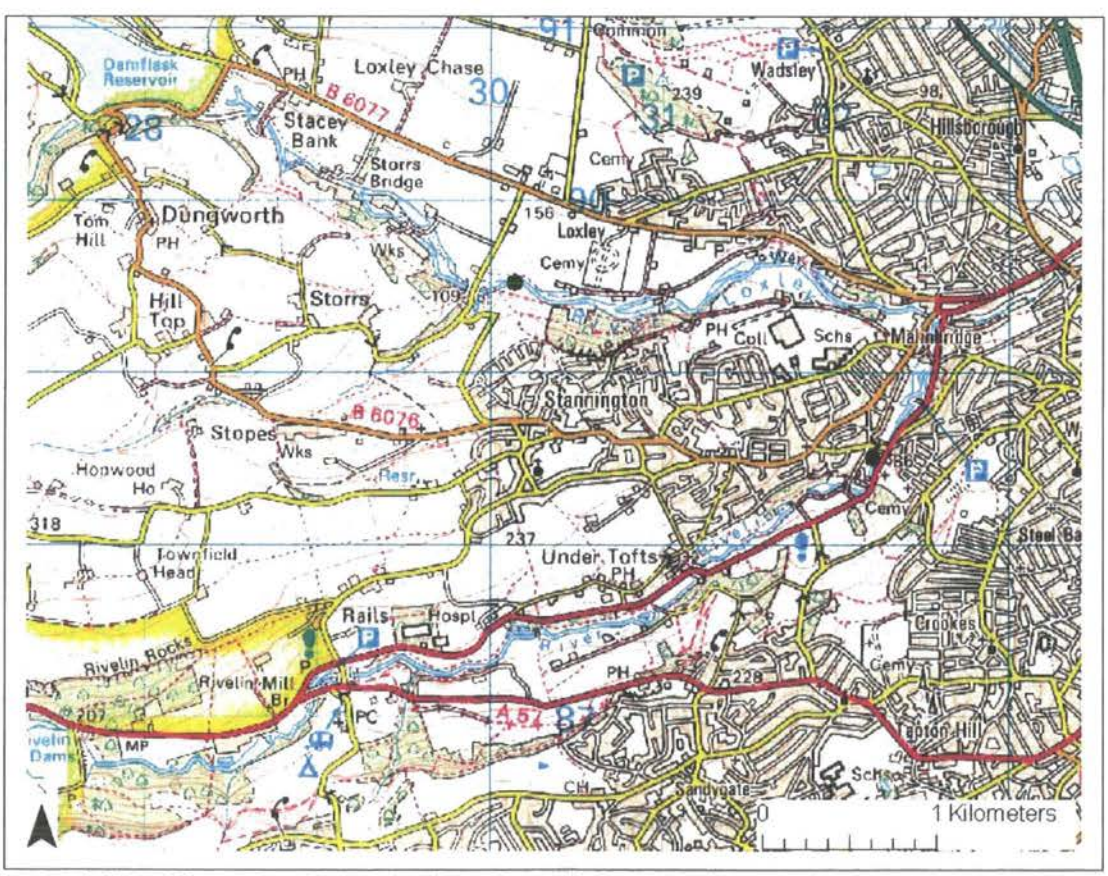

Figure 2.15. Figure to show the location of the gauging stations on the Rivelin and Loxley.

Table 2.4 shows that the Rivelin compensation reservoir is much smaller than the Loxley compensation reservoir. This means that the storage capacity of the Rivelin compensation reservoir during times of high rainfall intensity is much smaller than that of the Loxley reservoir. The reservoir level and amount of overtopping data for both compensation flow reservoirs were provided by Yorkshire Water. These variables were sampled every seven days by Yorkshire water staff. Figure 2.16 shows that the Rivelin compensation reservoir is almost always full, whereas a distinct seasonal variation can be seen in the levels of the Loxley reservoir. The apparent over-utilisation of the Rivelin reservoir's storage has led to a greater frequency of overtopping events on the Rivelin when compared to the Loxley (Figure 2.17).

	Total Storage Capacity (TCM)
Rivelin	36
Loxley	5037

Table 2.4. Table to show the total storage capacity of the Rivelin and Loxley compensation flow reservoirs.

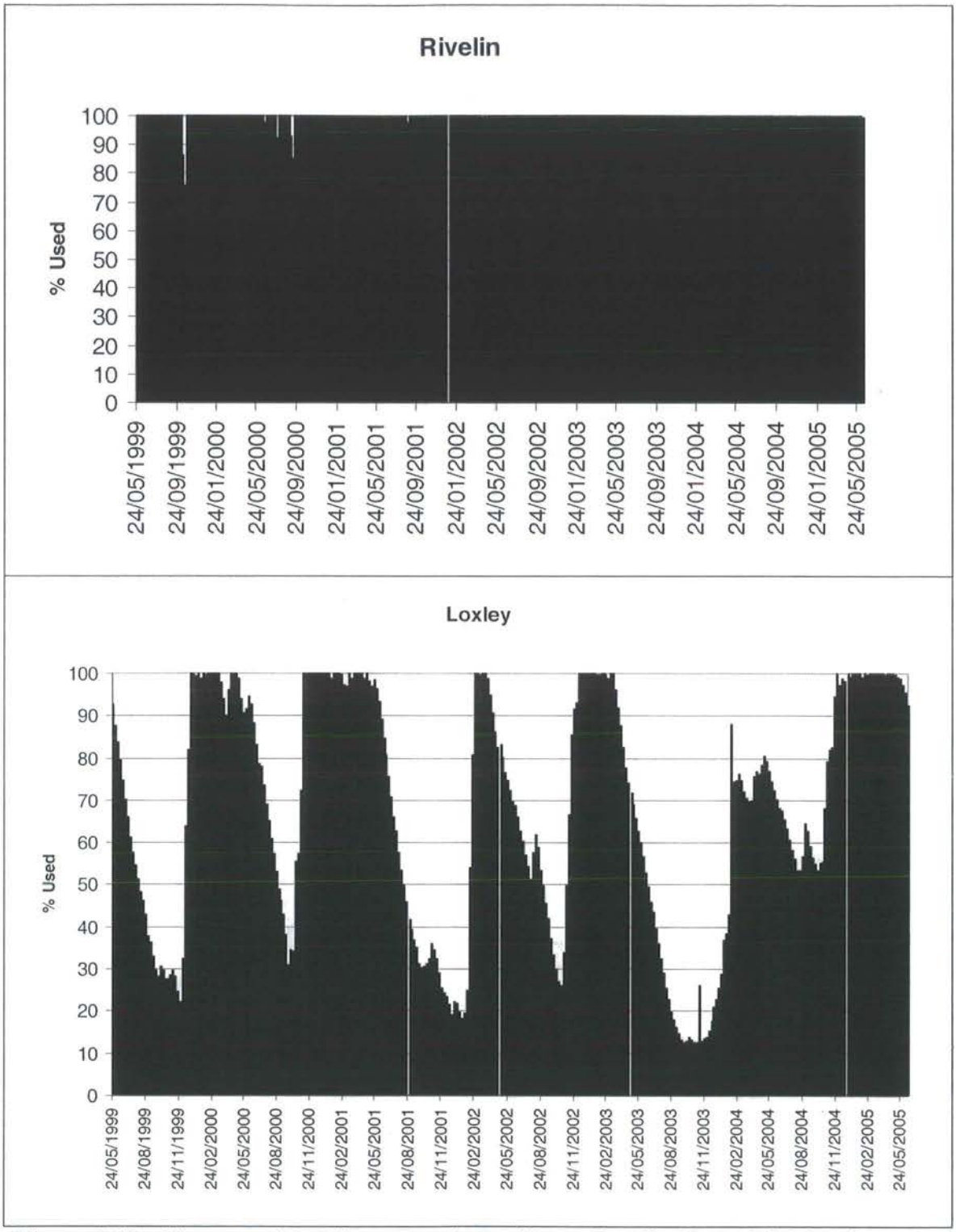

Figure 2.16. Figure to show the reservoir stocks on the Rivelin and Loxley from the 24/5/1999 to 14/5/2005.

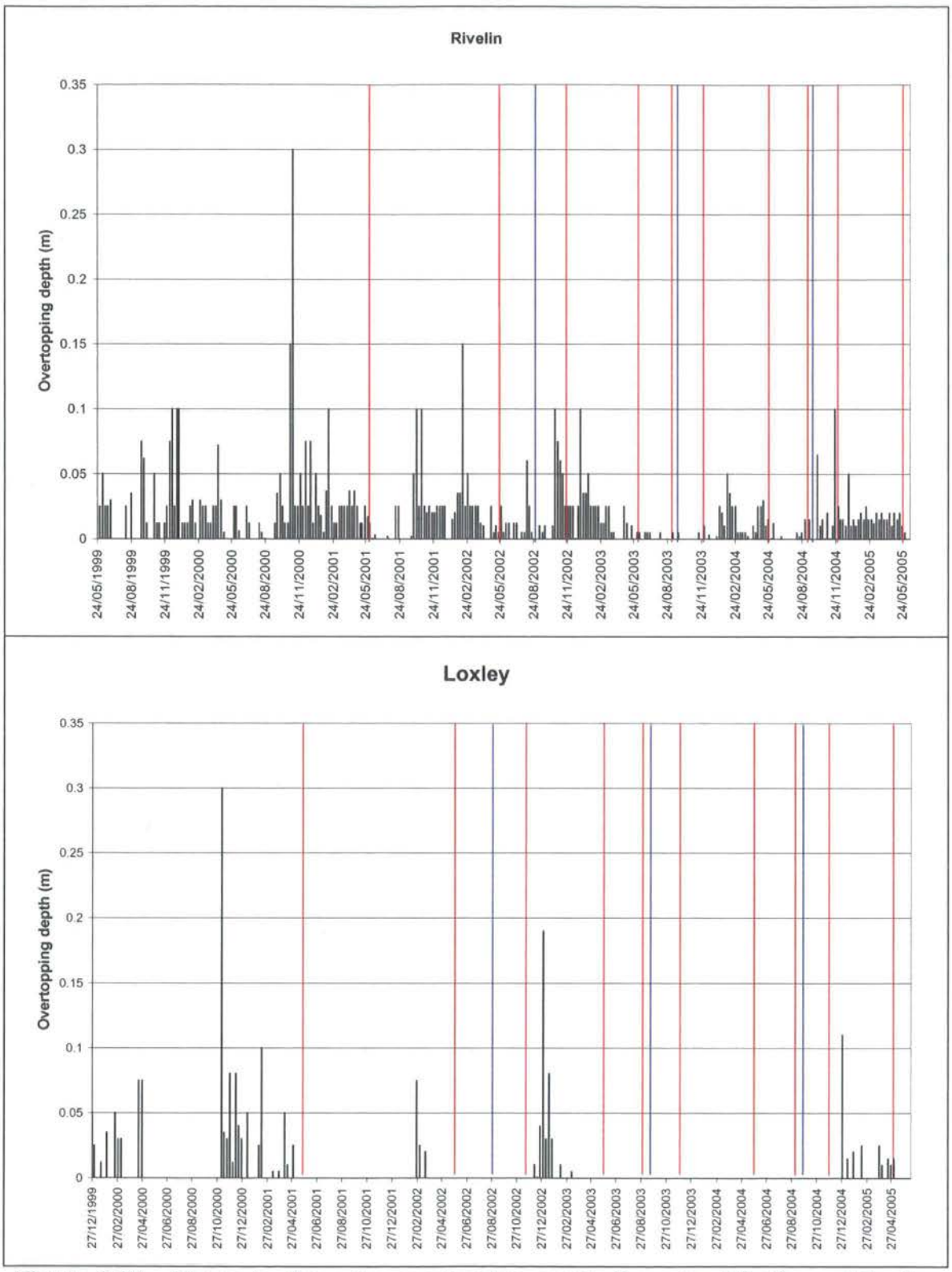

Figure 2.17. Figure to show the overtopping events from the Rivelin and Loxley compensation flow reservoirs from June 1999 to June 2005. Red lines highlight the timings of the invertebrate sampling; blue lines represent the fisheries sampling.

Table 2.5 shows that when the number of overtopping events are compared, the Rivelin is subject to a greater number of overtopping events than the Loxley. During the post-change period there are proportionally fewer overtopping events in the Rivelin compared with the pre-change period. The greater compensation flow released from the reservoir could have lowered the level of the reservoir sufficiently to slightly reduce the amount of overtopping. However, it can be seen that the reservoir levels at the Rivelin compensation reservoir do not appear to be any lower during the post-change period (Figure 2.16).

	Number of overtopping events
Rivelin	
Pre change (24/5/1999 to 31/3/2004)	186
Post-change (1/4/2004 to 24/5/2005)	53
Loxley	
Pre-change (24/5/1999 to 31/3/2004)	40
Post-change (1/4/2004 to 24/5/2005)	27

Table 2.5. Table to show the number of overtopping events on the Rivelin and Loxley, for the pre and post change periods.

2.3.2. Discharge

What is apparent from the discussion above is that there is some inherent variability present in the Rivelin and Loxley systems despite their being regulated. This is also seen in Figure 2.18. The change in baseflow after compensation adjustment (from 1st April 2004) is also evident in Figure 2.18, with a slight increase in the Rivelin discharge and corresponding decrease in discharge on the Loxley.

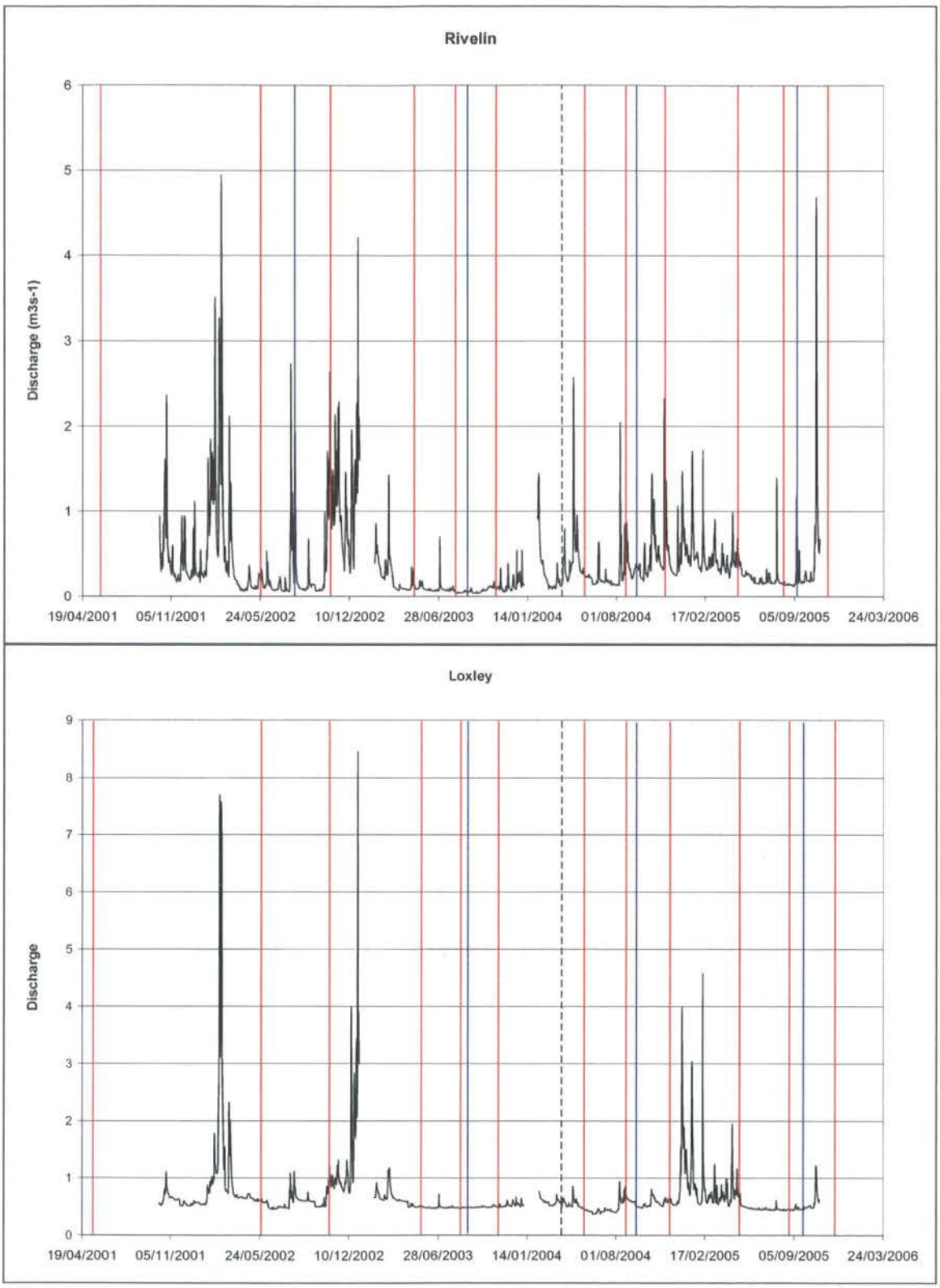

Figure 2.18. Daily averaged discharge hydrograph for the Rivelin and Loxley from 9/10/2001 to 1/11/05. Red lines highlight the timings of the invertebrate sampling; blue lines represent the fisheries sampling.

In order to gain a further understanding of the discharge characteristics of the Rivelin and Loxley, standard summary statistics were calculated (Table 2.6). In terms of the pre-change statistics it can be seen that the Rivelin has a lower minimum and maximum discharge than the Loxley. Neither river spends very long at minimum discharge which indicates that the discharge of both rivers is almost permanently augmented by tributaries and overtopping. What is interesting is that the Loxley appears to spend more time at base flow levels than the Rivelin, which is more augmented. The picture of greater variability on the Rivelin is further enhanced by the median variation statistics where a much greater percentage of days were spent above the various median flow thresholds.

Table 2.6 shows that the minimum flow has increased under the post-change conditions on the Rivelin, as has the median discharge. It is interesting therefore the Rivelin spends less time near the minimum discharge, but spends more time below minimum +20%. There has been a decrease in variability in the Rivelin. Whether this is due to the increase in median flow causing buffering or due to the increased compensation flow reducing the amount of overtopping is unknown, but Figure 2.16 shows that despite the increased compensation flow released from the Rivelin reservoir there does not appear to have been a lowering of the level of the Rivelin reservoir.

The impact of altering the compensation flows is also evident on the Loxley with a lowering of the minimum discharge and median discharge. There is also a decrease in the variability of the Loxley discharge (median), so this is indicating that there is a decrease in the variables forcing variability in this area. Therefore, the reduction in variability in the Rivelin may be partly due to natural forcing rather as well as altering the compensation flow releases.

	Rivelin		Loxley	
	Pre-change	Post-change	Pre-change	Post-change
Minimum Q	0.04 m^3/s	0.12 m^3/s	0.46 m^3/s	0.37 m^3/s
Maximum Q	4.95 m^3/s	4.69 m^3/s	8.45 m^3/s	4.58 m^3/s
St. Dev	0.54 m^3/s	0.35 m^3/s	0.71 m^3/s	0.35 m^3/s
Median Q	0.145 m^3/s	0.29 m^3/s	0.58 m^3/s	0.52 m^3/s
Q95 (low)	0.085 m^3/s	0.16 m^3/s	0.48 m^3/s	0.42 m^3/s
Q5 (high)	3.51 m^3/s	2.32 m^3/s	7.31 m^3/s	1.95 m^3/s
% days at min	0.66%	0.17%	0.33%	0.52%
% days <min +10%	1.77%	0.52%	26.19%	3.62%
% days <min +20%	3.65%	5.52%	41.66%	11.55%
% days > 2 times median Q	28.62%	12.24%	5.08%	5.52%
% days > 3 times median Q	20.22%	5%	3.09%	2.07%
% days > 5 times median Q	13.59%	1.38%	1.66%	0.86%
% days > 7 times median Q	8.95%	0.69%	0.66%	0.34%
% days > 9 times median Q	6.30%	0.34%	0.66%	0%

Table 2.6. Table to show the summary statistics for the Rivelin and Loxley pre and post change discharge hydrographs.

The flow duration curves presented in Figure 2.19 show that for the pre-change discharges on the Rivelin the receding limb of the curve is angled rather than flat as on the Loxley. This further indicates that the Rivelin is subject to greater discharge augmentation than the Loxley. When the pre and post-change curves are compared for the Rivelin, it can be seen that the receding limb of the curve is no longer angled and is much flatter, thus resembling the Loxley curves. This decrease in augmentation is supported by the overtopping data presented in Figure 2.17, and the summary statistics presented in Table 2.6. It therefore appears that the increase in compensation flow releases on the Rivelin has led to a decrease in the frequency of overtopping and so therefore a decrease in augmentation. The Rivelin has now developed a discharge more similar to that of the Loxley. The Loxley curve does appear to have shifted over to the left for the post-change discharges when compared to the pre-change discharges. However, little noticeable change has happened to the 'curviness' of the graph.

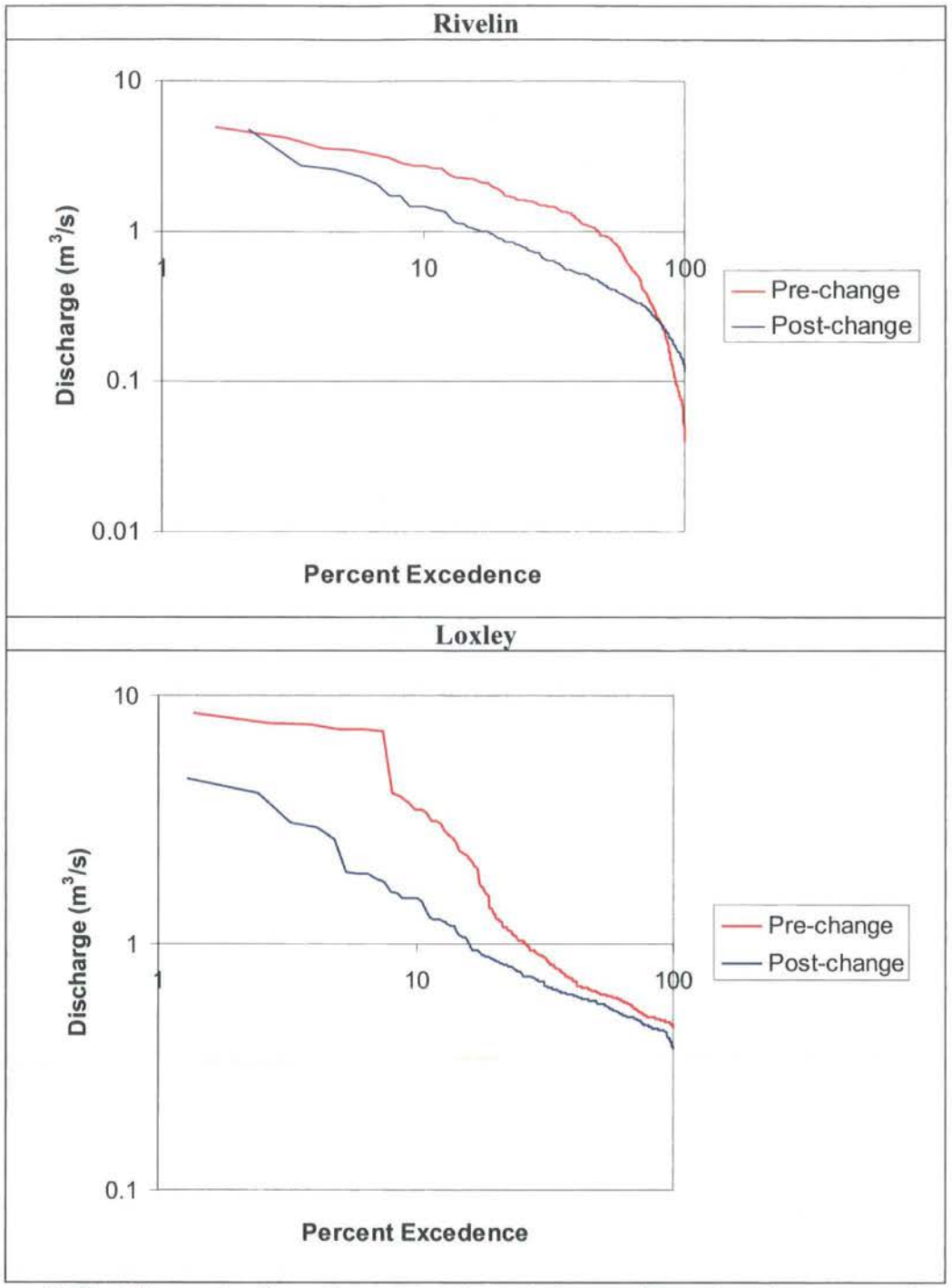

Figure 2.19. Figure to show the pre and post change flow duration curves for the Rivelin and Loxley.

The above section has shown that the Rivelin is subject to greater flow variability than the Loxley, and spends less time at or near base flow than the Loxley. It appears that after the change in compensation flows, there is a proportional decrease in overtopping from the Rivelin reservoir. This leads to an apparent decrease in augmentation in the Rivelin compared with the pre-change discharge. The discharge of the Loxley was lowered, but with no perceptible change to the amount of augmentation.

2.3.3. Interaction of geomorphology and hydrology

2.3.3.1. Rivelin

It can be seen from Fig. 2.20 that there are large fluctuations in water width and depth along the Rivelin. This is due to the instream regulation. Weirs force water to back up behind the structure hence causing an increase in width and depth in many cases. The gaps in the record are caused by weirs as the depth of water over the weir top was not measured. It can be seen that many of these data gaps are preceded by a sharp increase in water depth, indicating that the presence of a weir leads to an increase in water depth.

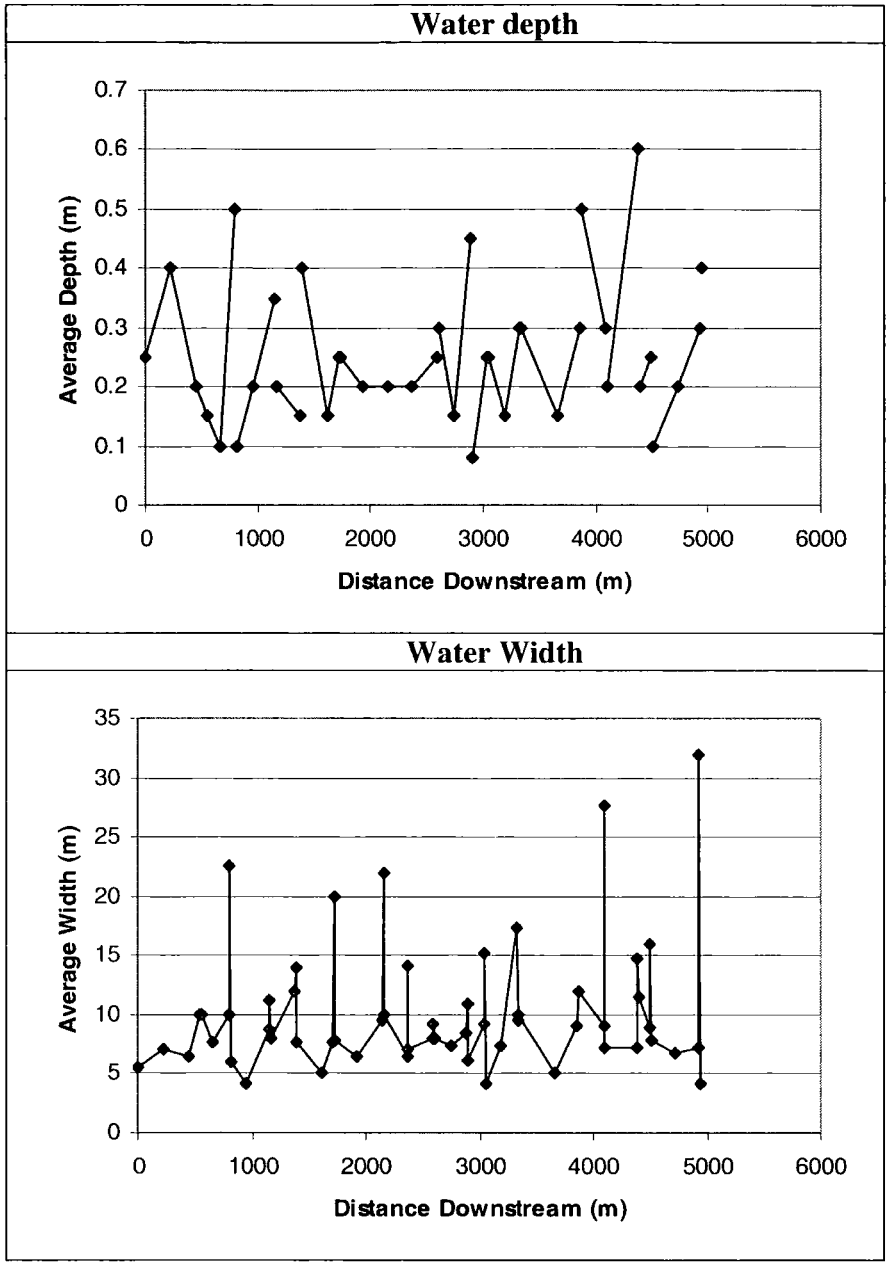

Figure 2.20. Graphs to show the average water width and depth sampled along the Rivelin during the geomorphological survey.

The flow types on the Rivelin are dominated by pools, runs and glides (Figure 2.21). The abundance of pools can be explained by weir effects, as many 'pools' in the Rivelin occur where water backs up behind weirs. Pools and runs are essential flow types for brown trout, with older trout preferring to spend time in pools, and the younger trout exhibiting a preference for shallower, faster flows. There does not appear to be a large proportion of riffle habitat in the cross-sections sampled, which is interesting, as the riffle habitat is a

very important habitat for spawning brown trout. Brown trout are thought to use the head of riffles as location of their spawning redds (Burner, 1951; Stuart, 1953; Hoopes, 1972).

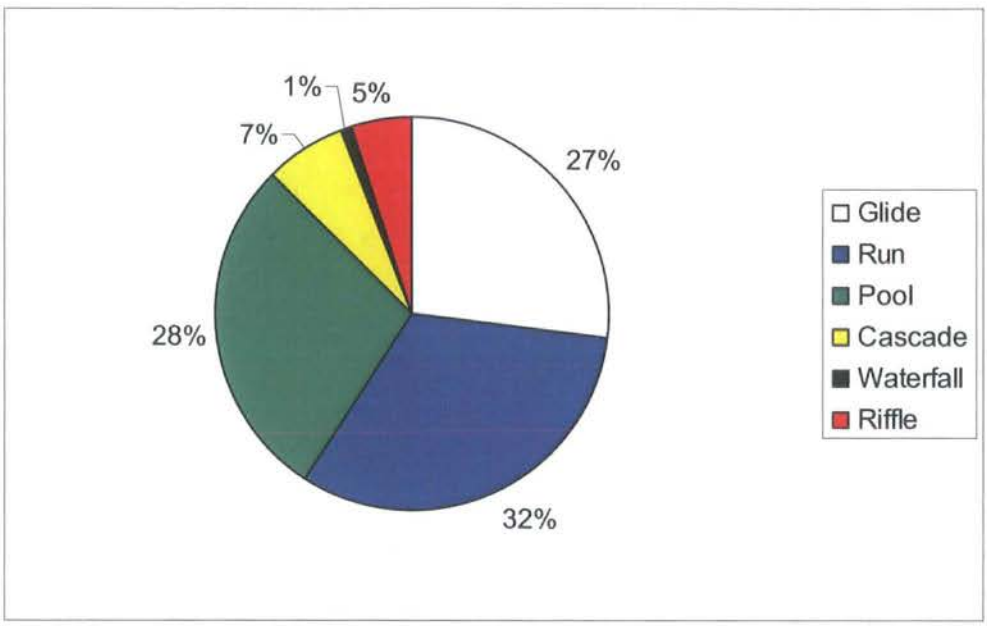

Figure 2.21. Pie-chart to show the proportions of flow types present sampled during the geomorphological survey on the Rivelin.

A high percentage of riverbanks in the Rivelin system are artificial (Table 2.7). This is related to the frequency of weirs in the system that have artificial banks in order to control the water. Also, the ponds which are fed by off-takes from the weirs are often in close proximity to the river itself, and this leads to the need for artificial structures to retain the water. The large proportion of artificial banks enhances the image of the Rivelin as a river which has been heavily impacted upon by human activities. However, even though a large percentage of the banks on the Rivelin are artificial, this does not stop the banks being undercut: as Table 2.7 shows, 48% of cross-sections on the Rivelin had at least one bank which was undercut.

	Percentage
Artificial Banks	88%
Natural Banks	12%
Bank undercut	48%
Bank not undercut	52%

Table 2.7. Table to show the bank statistics from the geomorphological survey on the Rivelin.

2.3.3.2. Loxley

The water widths and depths in the Loxley also appear to display a large amount of variation (Figure 2.22). This is again largely due to the presence of weirs in the system. It can be seen in Figure 2.22 that before every data gap there is a sharp increase in water depth. This is due to water backing up behind the weir. Generally, the Loxley is 5-10m wide.

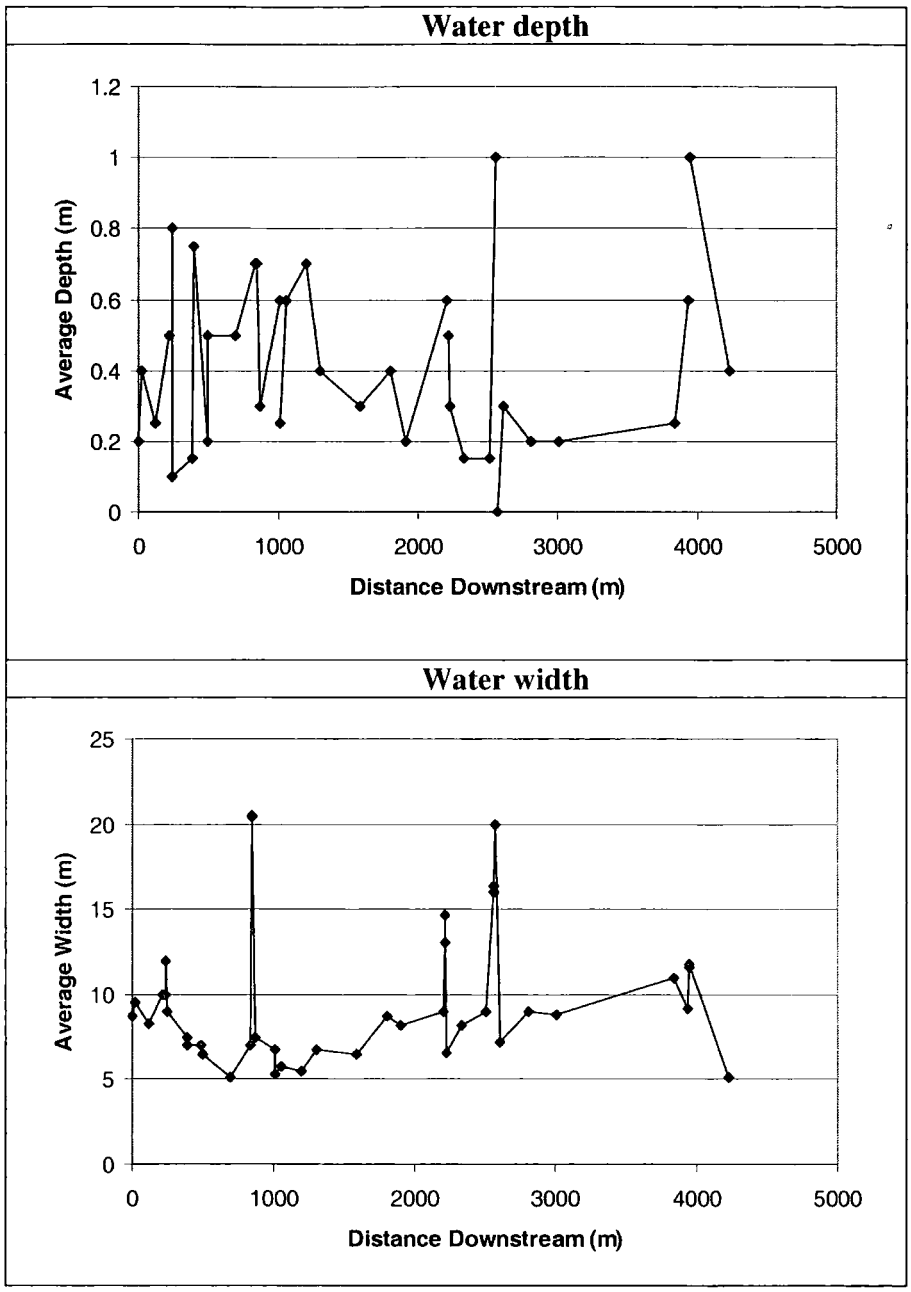

Figure 2.22. Graph to show average water width and depth sampled during the geomorphological survey along the Loxley.

The predominant flow types in the Loxley are run, glide, pool and riffle, with run and pool the two most abundant flow types. The presence of each of these flow types in almost equal proportions (Figure 2.23) indicates that, in terms of flow type, there is a good variety of instream flow habitat in the Loxley.

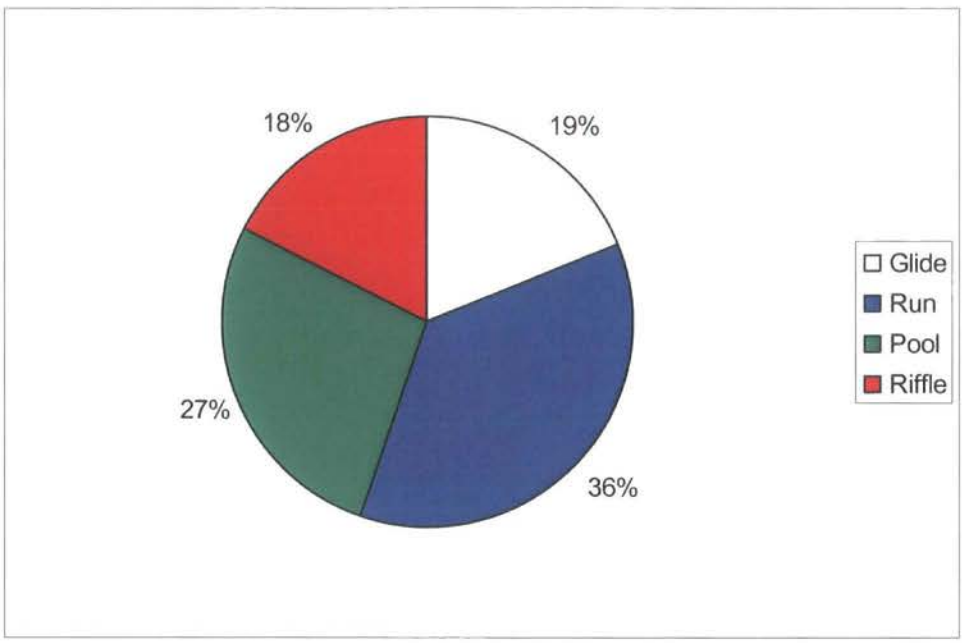

Figure 2.23. Pie-chart to show the proportion of flow types sampled during the geomorphological survey in the Loxley.

Table 2.8 shows that at 72% of cross-sections on the Loxley there were artificial banks present. Even though Table 2.8 shows that 72% of cross-sections on the Loxley had artificial banks, undercutting was observed at 40% of the cross-sections. Such undercutting provides habitat for mature brown trout.

	Percentage
Artificial Banks	72%
Natural Banks	28%
Bank undercut	40%
Bank not undercut	60%

Table 2.8. Table to show the output from the gemorphological survey on the banks of the Loxley.

2.3.3.3. Hipper

The measured water depth on the Hipper study reach ranges from 20-40cm. However, the data gap which can be seen in Figure 2.24 is the product of a very large weir pool, the depth of which could not be ascertained. For the remainder of the study reach the water depth of the Hipper remains relatively constant. Figure 2.24 also shows that the wetted width of the Hipper also remains relatively constant, until the large weir.

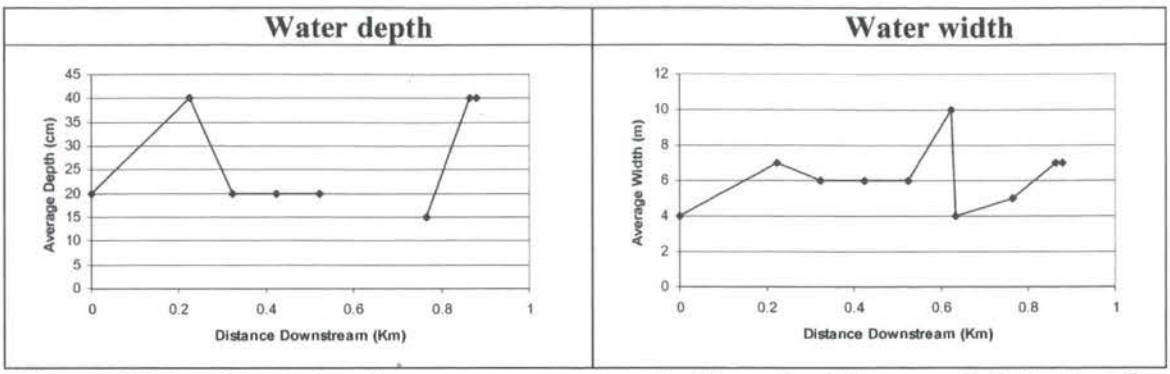

Figure 2.24. Graph to show average water depth and width sampled during the geomorphological survey along the river Hipper.

Figure 2.25 shows that the flow types which dominate in the Hipper are glide, run and pool, with very little of the river classified as riffle, (with similar implications as on the Rivelin). They are present in almost equal proportions. This again indicates that there is a good variety of habitat available for both invertebrates and fish.

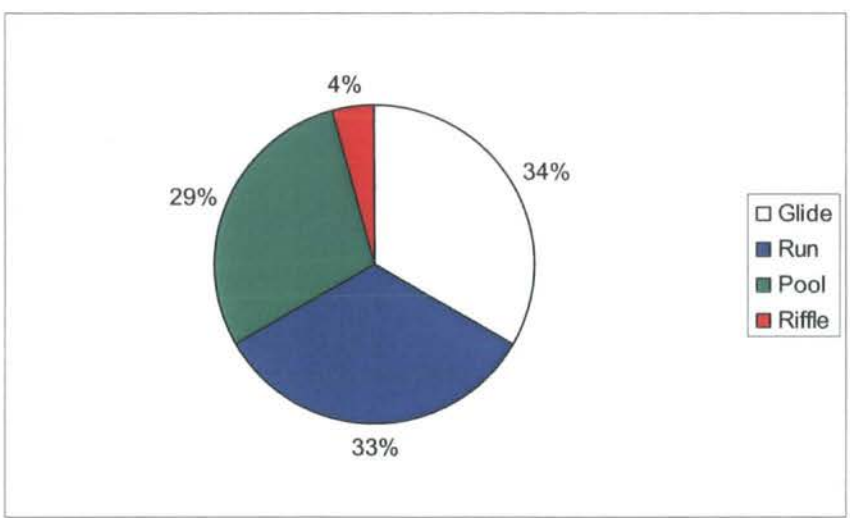

Figure 2.25. Pie-chart to show the flow types present on the Hipper as surveyed on the geomorphological survey.

Every cross-section surveyed on the Hipper had at least some form of artificial bank present (Table 2.9). This means that the Hipper is very much influenced by anthropogenic activities, especially the section through the playing fields. As a result, only 30% of cross-sections on the Hipper displayed any undercutting. This means that this aspect of brown trout habitat is limited in this stretch of river.

	Percentage
Artificial Banks	100%
Natural banks	0%
Undercut Banks	30%
Not undercut banks	70%

Table 2.9. Table to show the bank statistics from the Hipper geomorphological survey.

2.3.3.4. Summary

Each of the study rivers has a number of weirs which impact upon both the width and depth of the rivers, but also upon the life cycle and mobility of aquatic organisms. The influence of weirs is also felt in the bank characteristics, with each river having a large percentage of artificial banks, which are mostly due to the walls used to constrain the river in the weir systems, and walls used to constrain the mill ponds. Despite the high proportions of artificial banks, each of the rivers had a presence of undercut banks which provide habitat for older brown trout.

2.4. Hydrology with respect to ecological sampling

This section characterises the hydrological conditions prior to sampling the fisheries and macroinvertebrates because the preceding hydrological conditions may impact greatly on the ecology sampled. This is done in the form of metrics characterising the properties of discharge, rainfall and overtopping events for a period of interest before the samples were taken. These tables will be referred back to frequently in the two subsequent chapters.

2.4.1. Invertebrates

Table 2.10 shows that the timescale over which the metrics are calculated for the macroinvertebrate samples are of a short term nature (of the order of a month or less). This is because of a number of reasons:

1) invertebrates have a relatively short life cycle (compared to brown trout), and so metrics averaged out over a year would not be suitable, in the space of a year an

individual would have been through a number of life stages, which have different habitat requirements; and

2) invertebrates are relatively immobile compared to fish, and it is thought that a monthly timescale would reflect conditions present at invertebrate sampling

Table 2.10 shows that most of the invertebrate sampling on the Rivelin were conducted under augmented conditions. The only exceptions being the samples taken in summer and autumn 2003 where, due to a very low rainfall period, the Rivelin remained relatively unaugmented. For the 2004 and 2005 samples, rainfall and overtopping events remain commonplace and the discharge is well above baseflow levels.

Table 2.11 shows that for the Loxley samples, there were fewer overtopping events, and that for the majority of the invertebrate samples, the river was augmented by rainfall. Again, the low rainfall in 2003, means that the Loxley was generally non-augmented throughout 2003. Throughout 2004, there were no overtopping events due to the level of Damflask being reduced by the dry 2003 (Figure 2.17) and it can be seen that the Loxley is only slightly augmented by rainfall through this period.

	RU	RU	RU	RU	RU	RU
Season	Sp. 01	Sp.02	Aut02	Sp.03	Su.03	Aut.03
Date	21/5/01	21/5/02	28/11/02	23/5/03	29/8/03	8/12/03
Time since last overtopping measurement	0	1	3	4	25	7
28 day rainfall depth	96.1mm	80.2mm	128.3mm	46.4mm	0mm	0mm
Discharge on the day	No data	0.27	0.38	0.11	0.06	0.08
Indicator conclusions	Probably augmented flow	Augmented flow	Augmented flow	Slightly augmented flow	Base flow	Base flow

	RU	RU	RU	RU	RU	RU
Season	Sp. 04	Su.04	Aut04	Sp.05	Su.05	Aut.05
Date	19/5/04	24/8/04	29/11/04	18/5/05	9/9/05	18/11/05
Time since last overtopping measurement	2	1	6	4	Not known	Not known
28 day rainfall depth	84.6mm	156.8mm	80.2mm	52.6mm	94.2mm	158.6mm
Discharge on the day	0.30	0.41	0.40	0.27	0.30	No data
Indicator conclusions	Augmented flow	Augmented flow	Augmented flow	Augmented flow	Augmented flow	

	RD	RD	RD	RD	RD	RD
Season		Sp.02	Aut02	Sp.03	Su.03	Aut.03
Date		24/5/02	28/11/02	23/5/03	3/9/03	8/12/03
Time since last overtopping		5	3	4	30	7
28 day rainfall depth		105.3mm	128.3mm	46.4mm	0mm	0mm
Discharge on the day		0.26	0.38	0.11	0.06	0.08
Indicator conclusions		Augmented flow	Augmented flow	Slightly augmented flow	Baseflow	Baseflow

	RD	RD	RD	RD	RD	RD
Season	Sp. 04	Su.04	Aut04	Sp.05	Su.05	Aut.05
Date	18/5/04	24/8/04	30/11/04	18/5/05	9/9/05	22/11/05
Time since last overtopping	1	1	1	2	Not known	Not known
28 day rainfall depth	90mm	156.8mm	76.8mm	52.6mm	94.2mm	96mm
Discharge on the day	0.33	0.41	0.38	0.27	0.30	No data
Indicator conclusions	Augmented flow	Augmented flow	Augmented flow	Augmented flow	Augmented flow	Augmented flow

Table 2.10. Table to show the discharge/overtopping/rainfall conditions immediately before macroinvertebrate sampling conducted on the Rivelin.

	LU	LU	LU	LU	LU	LU
Season		Sp.02	Aut02	Sp.03	Su.03	Aut.03
Date		21/5/02	22/11/02	22/5/03	3/9/03	8/12/03
Time since last overtopping		64	249	73	177	273
28 day rainfall depth		80.2mm	169.4mm	46.4mm	0mm	0mm
Discharge on the day		0.62	0.89	0.5	0.48	0.5
Indicator conclusions		Slightly augmented flow	Slightly augmented flow	Pretty much baseflow	Baseflow	No flow variability
	LU	**LU**	**LU**	**LU**	**LU**	**LU**
Season	Sp. 04	Su.04	Aut04	Sp.05	Su.05	Aut.05
Date	19/5/04	24/8/04	30/11/04	18/5/05	13/9/05	18/11/05
Time since last overtopping	435	535	629	16	Not known	Not known
28 day rainfall depth	84.6mm	156.8mm	76.8mm	52.6mm	89.9mm	158.6mm
Discharge on the day	0.47	0.74	0.63	0.50	0.45	Not known
Indicator conclusions	Baseflow	Augmented by rain	Augmented by rain	Baseflow	Baseflow	Augmented flow
	LD	**LD**	**LD**	**LD**	**LD**	**LD**
Season	Sp. 01	Sp.02	Aut02	Sp.03	Su.03	Aut.03
Date	21/5/01	21/5/02	21/11/02	22/5/03	3/9/03	9/12/03
Time since last overtopping	21	64	248	73	177	274
28 day rainfall depth	96.1mm	80.2mm	169.4mm	46.4	0mm	0mm
Discharge on the day	No data	0.62	0.92	0.50	0.48	0.50
Indicator conclusions		Slightly rainfall augmented	Rainfall augmented	Baseflow	Baseflow	Baseflow
	LD	**LD**	**LD**	**LD**	**LD**	**LD**
Season	Sp. 04	Su.04	Aut04	Sp.05	Su.05	Aut.05
Date	19/5/04	19/8/04	30/11/04	17/5/05	9/9/05	18/11/05
Time since last overtopping	435	530	629	17	Not known	Not known
28 day rainfall depth	84.6mm	117.6mm	76.8mm	54.6mm	94.3mm	158.6mm
Discharge on the day	0.47	0.58	0.63	0.50	0.51	Not known
Indicator conclusions	Little augmentation	Little augmentation	Little augmentation	Little augmentation	Little augmentation	Augmented

Table 2.11. Table to show the discharge/overtopping/rainfall conditions immediately before macroinvertebrate sampling conducted on the Loxley.

2.4.2. Fisheries

Table 2.12 shows that the metrics chosen for the fisheries were of a longer term nature, this was for a number of reasons:

1) fish have longer life cycles and so will spend longer periods of time in certain types of habitat; and
2) the fish were sampled and analysed on a yearly basis (especially growth rates), and so it makes sense to characterise the hydrology over that year.

The metrics were calculated for each site for each sampling season and the range of values presented. In most cases, it appeared inconsequential as to where in the sampling season the reaches were fished. Table 2.12 shows that for the Rivelin, 2004 had the lowest year averaged discharge despite the increase in compensation flows implemented on the 1st April 2004. The augmentation due to rainfall in 2002 and 2003 lead to a higher discharge. The 2004 fisheries surveys also had the lowest average discharge on the Loxley, and 2005 had a lower discharge than 2002 and 2003, but was still subject to some augmentation. It will be interesting to analyse the fisheries data from 2004, so as to try and determine the effects of dry spells in these regulated rivers.

Rivelin				
Year	**2002**	**2003**	**2004**	**2005**
Dates	2/9/02 to 10/9/02	26/8/03-3/10/03	26/8/04-3/9/04	22/8/05-30/8/05
Number of overtopping events in the year	43-44	37	30-32	40-41
Average Q (9/10/01 to sample)	0.44-0.45	0.38-0.39	0.24	0.38-0.39
Total yearly rainfall	1112m to 1145mm	754-758mm	751-752mm	960-929mm
Indicator conclusions	Wet	Normal	Dry	Wet
Loxley				
Year	**2002**	**2003**	**2004**	**2005**
Date	3/9/05-12/9/05	3/9/03-10/9/03	10/9/04-21/9/04	11/8/05-6/9/05
Number of overtopping events in the year	3	9	3	9
Average Q (9/10/01 to sample)	0.79-0.80	0.76	0.52	0.67-0.68
Total yearly rainfall	1115-1154mm	741-754mm	756-799mm	983-933mm
Indicator conclusions	Wet	Normal	Dry	Normal

Table 2.12. Table to show the characteristics of the year before the fisheries populations were sampled on the Rivelin and Loxley.

2.5. Comparison and conclusions

Both the Rivelin and Loxley share the same geology. The riparian land use of all three of the rivers contained a large proportion of set-aside, and recreational land. Typically, this land contained large amounts of vegetation, and provided a reasonably dense canopy cover for all three rivers. However, the exception to this generalisation is the upper reaches of the Loxley, where there is a predominance of urban wasteland (Figure 2.9). As a consequence of this, there is also a lack of canopy cover in the upper reaches of the Loxley study area (Figure 2.10). Almost everywhere else, in all three catchments, the canopy cover afforded by the riparian vegetation was greater than the 35% level suggested by Raleigh *et al.* (1986) as being the minimum for good adult brown trout habitat.

On the Rivelin, 86% of cross-sections contained some artificial banks, whereas on the Loxley, 72% of cross-sections had artificial banks. On the Hipper, 100% of cross-sections

examined contained some form of artificial banks. This indicates the extent of human influence on all three of the systems. However, the high percentage of artificial banks on the Rivelin could be due to the greater number of weirs than on the Hipper and Loxley. Despite having a greater percentage of artificial banks, the Rivelin has the greatest percentage of undercut banks (48%). The Loxley, which has fewer artificial banks than the Rivelin also has fewer undercut banks than the Rivelin (40%). One of the most important implications of this survey relates to availability of cover for fish in the rivers, as bank-derived cover provides an essential refuge for mature brown trout.

It is apparent that there is hydrologic variability present on the Rivelin and Loxley despite them being regulated. The amount of variation is greater in the Rivelin than in the Loxley and this can largely be attributed to the Rivelin compensation reservoir being smaller than the Loxley compensation reservoir. It can also be seen that after the change in compensation flow, the amount of variability in the Rivelin appears to decrease proportional to the Loxley variability. It appears that the post-change Loxley discharge regime remains relatively similar to its pre-change regime, whilst the Rivelin has changed both in terms of compensation flows and variability.

There is significant variation of the depth and width of water in all three rivers in the downstream direction due to the presence of weirs. Weirs cause a reduction in velocity and hence an increase in depth. The wetted widths of the Rivelin appear to display more variation than those of the Loxley. This could be due to the greater number of weir structures in the Rivelin. It was also evident that the Loxley was generally slightly wider than the Rivelin.

In the Rivelin, glide, run and pool dominate the flow type, whereas in the Loxley, riffle is more abundant. This is strange as one might expect the river which has the smallest discharge (e.g. the Rivelin) to have a greater abundance of riffle type flow than the river with the larger discharge. This could be due to the number of weirs on the Rivelin. Due to the fact that weirs are important geomorphological features, they must be included in any geomorphological sampling programme. As discussed earlier in the effort to include all the weirs whilst retaining a sensible number of study cross-sections some of the riffle dominated areas of the Rivelin may not have been sampled.

The hydrological conditions present for the ecological sampling were also discussed. The Rivelin and Loxley had similar patterns. It was seen that 2003 was a very dry year, with little augmentation of flow for the invertebrate samples, and the 2004 fisheries samples were also affected by this low rainfall period.

All of the geomorphological and hydrological information presented above will have an influence on the invertebrates, fisheries and hydrodynamics of the system (Figure 1.2). The information shown above will also be used to help interpret the results of Chapters 3, 4 and 5.

Chapter 3

Assessing the impacts of compensation flows on macroinvertebrates

3.1. Introduction

Aquatic macroinvertebrates occupy a unique position within the structure of lotic ecosystems (Gore *et al.*, 2001). As processors of organic material (either allochthonous or autochthonous), the invertebrates serve as the critical link between primary production and the fish community (Gore *et al.*, 2001). Numerous studies have shown an interaction between discharge and invertebrate communities (e.g. Gore, 1977, 1978). Thus, assessing the potential impact of altering compensation flow regimes on the macroinvertebrate populations of the rivers has two rationales: (i) it provides a diagnostic approach for assessing if there is a biological impact in its own right; and (ii) assessment of changes in macroinvertebrate populations may be of importance in relation to sustaining fish communities. Thus, the aims of this chapter are:

1) to establish the structure and quality of the macroinvertebrate populations in each of these rivers;
2) to assess the natural variability in these populations in order to be able to distinguish the impacts of any changes in compensation from what would be expected as normal fluctuations;
3) to test the hypothesis that existing differences in macroinvertebrate populations can be related to the different levels of compensation flow in the two rivers; and
4) to assess any impact from changing the compensation flow regimes on the macroinvertebrate communities in the two rivers.

Section 3.2 reviews the literature that links invertebrates and discharge. As was established in Chapter 2 there is a difference between the Rivelin and Loxley in terms of hydrological variability. The literature review will therefore also focus on the importance of hydrologic variability (as introduced in Chapter 2) to macroinvertebrate communities and, as

hydrologic variability essentially constitutes a disturbance, the literature pertaining to disturbance models is also reviewed. The field and data analysis methodologies employed are presented in section 3.3, with both univariate and multivariate data analysis methods explained. Section 3.4 discusses the evidence in relation to the four aims above. Section 3.5 synthesises this chapter in relation to the literature identified in section 3.2.

3.2. Literature review

3.2.1. Introduction

As discussed in section one, the potential impacts of altering compensation flow regimes can be summarised into three orders of impacts, with both order one and order two impacts potentially influencing the macroinvertebrate communities within rivers. Firstly, this section will highlight ways in which the parameters shown in Figure 3.2.1 can impact upon the invertebrate communities in a steady state situation. However, Chapter 2 shows that the Rivelin, Loxley and Hipper are not steady state systems and invertebrate populations are not only influenced by the habitat: both habitat disturbances and species interactions have to be considered.

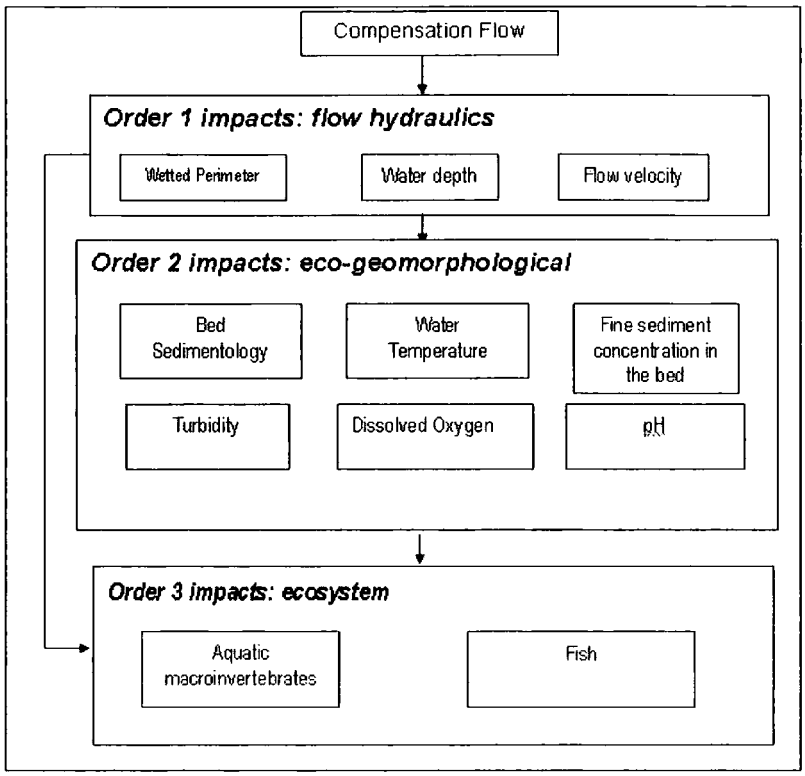

Figure 3.2.1. Flow diagram to show the potential impacts of altering the compensation flow in the rivers. Adapted from Petts (1984).

3.2.2. Controls on macroinvertebrate populations

In steady state terms, three sets of controls on macroinvertebrate populations have been identified: flow velocity; substrate; and habitat type.

Extence *et al.* (1999) created the Lotic Index for Flow Evaluation (LIFE), which linked qualitative and semi-quantitative patterns in macroinvertebrate communities to prevailing flow regimes, largely cast in terms of flow velocity. They found it to be very effective in encapsulating ecological response to changing flow patterns in a range of river types. The positive response occurs despite the fact that the flow data used in LIFE may not necessarily be the flows to which benthic macroinvertebrates are normally exposed because of the complex interactions that exist between river hydraulics, habitat morphology and habitat composition (Extence *et al.*, 1999). The method can thus be used to summarise the multiple effects of flow on invertebrate populations, much as biotic indices have historically been used to integrate water quality effects (Extence *et al.*, 1999).

In a large river, Rempel *et al.* (2000) identified a positive association of Reynolds number (Re) with collector-filterers, grazers and predators. Collector-gatherers were positively associated with Coarse Particlate Organic Matter (CPOM) and negatively associated with mean velocity. Shredders were also correlated significantly with CPOM. They also found that invertebrate density was not found to be related to mean grain size, but was positively correlated with organic matter when both variables were entered in the model simultaneously. The positive correlation of filterers with Re, (in their case a depth-averaged velocity measure), rather than near-bed conditions was surprising given the suggestion that near-bed flow patterns in deep water may be more closely related to substratum characteristics than mean velocity (Davis and Barmuta, 1989). Rempel *et al.* (2000) believe that in deep water, where filterers were most common, near-bed hydraulic conditions are tightly coupled with the main flow profile because flow resistance results only from bottom drag.

Substrate factors will, to some extent, be correlated with hydraulic variables, and the two are commonly identified together as factors that influence community composition, and the abundance and distribution of the component populations (Statzner *et al.*, 1988; Cobb *et al.*, 1992; Quinn and Hickey, 1994). Rempel *et al.* (2000) note that this interaction makes distinguishing between hydraulics and substratum effects very difficult: grain size composition of the channel substratum influences the distribution of many benthic taxa (e.g. Gurtz and Wallace, 1984); but this is largely determined by the competence of past and present flows (Leopold, 1994); and influences spatial patterns of flow velocity and depth through grain (roughness) and grain organisation (geometry) effects, which, in turn, creates fine-scale patterns of near bed flow that influence organic matter retention and the distribution of benthic organisms (e.g. Culp *et al.*, 1983; Hart *et al.*, 1996).

However, flow and substrate dynamics can also be important, Cobb *et al.* (1992) found that benthic insects were negatively correlated with increasing discharge and substrate particle movement. The impact of flood events on benthic invertebrate densities differed between sites as a function of differences in substrate stability. They estimated the percentage of bed material moving during floods of different magnitudes and found that invertebrates at sites with less stable beds were impacted most by floods. Rivers with unstable substrates tend to be characterised by low species diversity, and the biota present often have life history or

behavioural characteristics of frequently disturbed environments (Sagar, 1986; Cobb *et al.*, 1992).

Fine sediment suspension and deposition affects benthic invertebrates in four ways (Wood and Armitage, 1997): 1) by altering substrate composition and changing the suitability of substrate for some taxa; 2) by increasing drift due to sediment deposition or substrate instability; 3) by affecting respiration due to the deposition of silt on respiration structures or low oxygen concentrations associated with silt deposits; and 4) by affecting feeding activities by impeding filter feeding due to an increase in suspended sediment concentrations, reducing the food value of periphyton and reducing the density of prey items.

Kaller and Hartmann (2004) used: simple linear regression to test relationships between substrate size classes and metrics; and nested ANOVA to test macroinvertebrate differences amongst Appalachian streams. Consistent negative relationships between the finest particles (<0.25mm) and Ephemeroptera, Plecoptera and Trichoptera (EPT) taxa richness were observed. In seasons of normal hydrology, EPT taxa scores significantly decreased ($p < 0.05$) in streams where fine substrate particles exceeded 0.8-0.9% of the riffle substrate composition. In drought seasons, fine sediment (<0.25mm) exceeded 0.8-0.9% in most surveyed streams, lowering macroinvertebrate diversity in all streams.

A review by Vinison and Hawkins (1998) found that the average size of individual stones does not appear to affect taxa richness consistently. Studies have shown that more taxa occur on larger than on smaller substrates (Erman and Erman, 1984; Friberg *et al.*, 1977; Quinn and Hickey, 1990; Williams and Mundie, 1977; Wright *et al.*, 1984). Other studies have found the reverse: more taxa can occur on smaller than on larger substrates (Hawkins and Furnish, 1987; Wise and Molles, 1979). These differences do not appear to be a function of differences in the range of substrate sizes examined (Vinison and Hawkins, 1998). In contrast, each of the six studies reviewed by Vinison and Hawkins (1998) that examined relationships between species richness and substrate type showed that different types of substrates support different numbers of taxa, with more complex substrate supporting more complex taxa. Thus, what seems to matter in relation to substrate is a combination of substrate complexity and substrate stability. A more complex habitat will

provide a greater range of ecological niches. A more stable habitat will prevent species washout.

Habitat type generally refers to the larger scale variability of velocity, depth and substrate and linked to longitudinal and planform changes in river morphology. These larger scale variations in habitat have also been linked to macroinvertebrates. Scarsbrook and Townsend (1993) found higher taxa richness in riffles than in pools in a stable stream and no difference between the habitats in an unstable stream. In a review of 17 studies that sampled both pool and riffle habitats, Logan and Brooker (1983) found no significant differences in richness at several taxonomic levels between the two habitats. Similarly, Jenkins *et al.* (1984) showed that there were no substantial differences in richness among eight habitat types. But eight other studies evaluated by Vinson and Hawkins (1998) did find significant inter-habitat variation in richness (Angradi, 1996; Brown and Brussock, 1991; Grubaugh *et al.*, 1996; McCulloch, 1986; Thorp, 1992; Wohl *et al.*, 1995). Delucchi (1988) also found a difference between the invertebrates in pools and riffles, irrespective of flow permanence. In their investigation of species assemblages and mesohabitats, Pardo and Armitage (1997) found that the mesohabitat scale provides a convenient structural ecological unit to examine the functional dynamics of faunal communities in rivers.

The 'evaluation of surface flow biotopes' (no flow; no perceptible flow; smooth boundary turbulent flow; rippled flow; unbroken standing wave; broken standing wave; chute and free fall) represents a field assessment system for the bankside measurement of surface flow character in a stream and river (Harper *et al.*, 2000). This system provides a rapid yet accurate way of measuring the link between a river's hydrological regime and its physical habitat pattern (Padmore, 1997; Newson *et al.*, 1998; Padmore *et al.*, 1999). An alternative field assessment system for measuring habitat structure of river channels has also been developed. Sixteen 'functional habitats' have been shown to accommodate the river's overall macro-invertebrate biodiversity (Harper *et al.*, 1992; Kemp *et al.*, 1999).

This section has shown that, in steady state terms there are three main controls on invertebrate populations: (i) flow velocity; (ii) substrate; and (iii) habitat type. Extence *et al.* (1999) provided a direct link between flow velocity and macroinvertebrates, whilst

Rempel *et al.* (2000) examined the interaction of depth, velocity and the invertebrate functional feeding groups. Further complication arises from the interaction of substrate and flow velocity, with Cobb *et al.* (1992) highlighting the importance of substrate stability. Fine sediment suspension and deposition also affects invertebrates in a number of ways. There are confused messages as to the impact of substrate size on invertebrate richness, but what appears to matter to invertebrates in relation to substrate is a combination of substrate complexity and stability. It appears that in general, different flow habitat types support different macroinvertebrate communities.

3.2.3. Disturbance and species interactions

Ecological research has shown that there is no single factor explanation for all patterns of species diversity (Huston, 1994). In looking at macroinvertebrate populations in relation to variability, there are two main hypotheses as to what the major controls are: the intermediate disturbance hypothesis; and the dynamic equilibrium hypothesis; with competitive equilibrium (in its purest form) occurring rarely (Huston, 1994).

A disturbance occurs when potentially damaging forces are applied to habitat space occupied by a population, community, or ecosystem (Lake, 2000). The magnitude of the forces may be such that organisms may be killed or displaced, consumable resources (e.g. living space and food) may be depleted, and habitat structure may be degraded or destroyed (Lake, 2000). Disturbances should be defined by the nature of their damaging (abiotic) properties, especially the intensity and forms of their forces, along with parameters such as frequency, predictability, spatial extent, and temporal duration (Lake, 2000).

Connell (1978) proposed that the highest levels of species diversity were maintained at some 'intermediate' frequency or intensity of disturbance. At high rates of disturbance, diversity is reduced because some species are unable to recover from mortality (Connell, 1978). At low rates of disturbance, diversity is reduced by competitive exclusion as the dominant species eliminate poorer competitors (Huston, 1994).

Huston (1994) noted that the intermediate disturbance hypothesis has been criticised as being too simplistic to account for the structure of communities (McGuinness, 1987) and for being difficult to falsify (Juhasz-Nagy, 1993). The independent disturbance hypothesis is also dependent on a trade-off between colonisation and competitive ability (Collins and Glen, 1997), which may not be realistic for assemblages of highly mobile species, such as stream invertebrates. Huston (1994) argues that it should not be surprising that virtually all the evidence in support of the 'Intermediate Disturbance Hypothesis' comes from systems with high growth rates of competitive displacement such as the intertidal zone (e.g. Paine, 1966; Sousa, 1979), the crests of coral reefs (e.g. Connell, 1978), and algae growing in shallow water (e.g. Lubchenco, 1978). Likewise, the 'Hump-backed Productivity-Diversity Curve' (Grime, 1973, 1979) is most likely to be found in communities with intermediate to high frequencies of disturbance, such as herbaceous plant communities (Huston, 1994). At very low frequencies of disturbance, the relationship between productivity and species diversity is usually monotonically decreasing (Huston, 1994).

Huston's (1979; 1994) dynamic equilibrium model offers a broader range of predictions than the intermediate disturbance hypothesis. Depending on rates of competitive exclusion and the rates of population growth, species diversity can peak at low, high or intermediate rates of disturbance (McCabe and Gotelli, 2000). The interaction is represented graphically in Figure 3.2.2. In this dynamic model, species diversity can be reduced by either of the two processes (Figure 2.1). On one side of the diagonal of maximum diversity, diversity is reduced by competition, while on the other side of the diagonal, diversity is reduced by the inability of populations to recover from disturbances (Huston, 1994). The first of these processes is the competitive displacement that eventually leads to competitive exclusion and local extinction (Huston, 1994). Species diversity is reduced by this mechanism in situations where the effect of disturbance is insufficient to prevent competitive exclusion (Huston, 1994). The second process that reduced diversity is the local extinction of populations that are growing too slowly to recover from the disturbances (Huston, 1994). Thus, this conceptual model is based upon variation in the relative importance of different processes that influence species diversity (Huston, 1994). Local processes, such as competition dominate and reduce diversity under conditions of high growth rates and low disturbance frequencies (Huston, 1994). Other local processes such as mortality, reproductive failure, and local extinction, are important under conditions of low growth

rates and high disturbance frequencies (Huston, 1994). This approach clearly demonstrates that the regulation of species diversity is not an issue of local processes versus regional processes, but of the expression of regional processes under some conditions and the dominance of local processes under other conditions (Huston, 1994). The most significant consequence of this model for understanding species diversity is the concept that the same absolute change in frequency (or intensity) of disturbance can have totally different effects on species diversity depending on the rate of competitive displacement (Huston, 1994) (i.e. it is locally dependent).

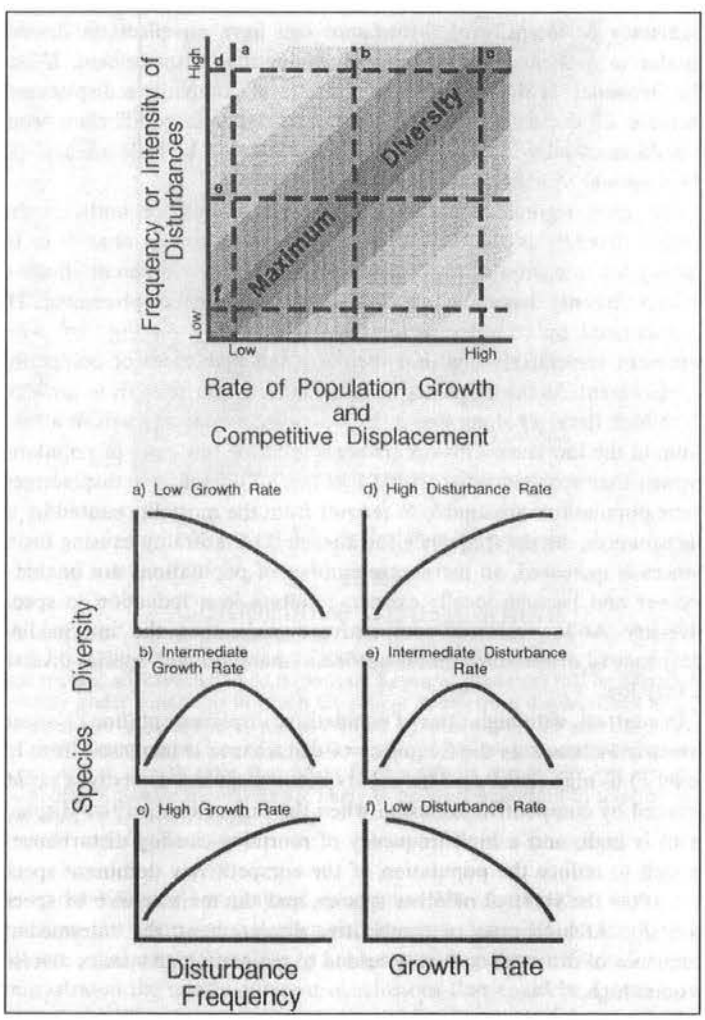

Figure 3.2.2. Three-dimensional representation of the predictions of the Dynamic Equilibrium Model for species diversity in relation to the dynamic equilibria between different rates of competitive displacement (correlated with growth rate, productivity etc) and different frequencies of mortality-causing disturbances. From, Huston (1994).

McCabe and Gotelli (2000) found that macroinvertebrate abundance in disturbed sections were lower than in undisturbed sections. However, species richness was significantly higher in disturbed reaches than undisturbed reaches. This pattern is predicted by Huston's (1979, 1994) dynamic-equilibrium model when the species populations are growing rapidly and have high rates of competitive exclusion (McCabe and Gotelli, 2000). In most other studies, experimental disturbance has been found to decrease species density and was followed by a rapid recovery to control or predisturbance levels. This pattern was consistent across disturbance types including stone turning (Englund, 1991; Palmer *et al.*, 1995) and substrate kicking and raking (Doeg *et al.*, 1989; Lake *et al.*, 1989). Numerous other authors have found that richness appears to be consistently and negatively related to disturbance intensity or frequency over the range of disturbances examined (Death and Wintebrourn, 1995; Robinson and Minshall, 1986; Scarsbrook and Townsend, 1993). None of these studies either alone or collectively support the idea that levels of disturbance promote high taxa richness in streams (Vinison and Hawkins, 1998).

3.2.4. Disturbance in riverine ecosystems

Hydrology is a primary control on the ecological quality of rivers, through its influence on geomorphology, flow regime, water quality and thus habitat availability (Gilvear *et al.*, 2002). The shape and size of river channels, the distribution of riffle and pool habitats and the stability of substrate are all largely determined by the interaction between the flow regimes and the local geology (Frissel *et al.*, 1986; Cobb *et al.*, 1992; Newbury and Gaboury, 1993). The flow regime of a river, together with its influence on water quality and sediment movement, create what are called biotope's and hence a 'habitat template' for aquatic flora and fauna (Gilvear *et al.*, 2002). Two aspects of flows that have been identified as important to benthic biota are flow variability, as defined by the coefficient of variation of flows (Jowett and Duncan, 1990), and the frequency of flood events (Fisher and Grimm, 1988, 1991; Biggs, 1996). Furthermore, Richter *et al.* (1997), suggest that accumulated research on the relationship between hydrological variability and river integrity overwhelmingly suggests a *natural flow paradigm,* which states: *the full range of natural intra- and inter-annual variation of hydrological regimes, and associated characteristics of timing, duration, frequency and rate of change, are critical in sustaining the full native biodiversity and integrity of aquatic ecosystems.*

There are now many conceptual frameworks and empirical studies linking flow properties to species traits, species densities and community functioning in streams (e.g. Hildrew and Townsend, 1987; Resh *et al.*, 1988; Poff and Ward, 1989; Jowett and Duncan, 1990; Poff and Allan, 1995; Poff, 1996; Richter *et al.*, 1996, 1997; Poff *et al.*, 1997; Clausen and Biggs, 1997, 1998; Biggs *et al.*, 1998). These studies have considered both high flow variability and low flow variability impacts.

Individual high flow events have been shown to reduce greatly biomass and change the species structure of invertebrates (Grimm and Fisher, 1989; Scrimgeour and Winterbourn, 1989; Cobb *et al.*, 1992). Thus, negative correlations have been found between the frequency of disturbance events and the density and diversity of certain invertebrate groups (Scarsbrook and Townsend, 1993; Death and Winterbourn, 1995; Biggs *et al.*, 1999). Habitats with an intermediate frequency of high flow disturbance have been shown to have the highest total densities and diversity of benthic invertebrates across some disturbance gradients (Death and Winterbourn, 1995; Clausen and Biggs, 1997; Townsend *et al.*, 1997). The frequency of high flow disturbance is now used as a primary axis of conceptual habitat models depicting variations in the structure of benthic communities among temperate perennial streams (e.g. Hildrew and Townsend, 1987; Townsend and Hildrew, 1994; Biggs *et al.*, 1998).

Townsend and Hildrew (1994) proposed for river habitats a template with axes of temporal heterogeneity and spatial heterogeneity. They assumed that temporal variation would bear a relationship with the disturbance regime to which organisms are subjected, while increased spatial heterogeneity would ameliorate or modify the influence of disturbances by provision of refugia within which survival was more likely. The results presented by Wood *et al.* (2000) support the view that high flows (or the lack thereof) are of major importance to benthic communities.

Flood events can influence both the biomass and taxonomic structure of invertebrates (McElravy *et al.*, 1989; Scrimgeour and Winterbourn, 1989; Jowett and Duncan, 1990; Scarsbrook and Townsend, 1993). Indeed, several authors have suggested that a measure of flood disturbance should be used as the primary axis of the habitat template for

invertebrates (Hildrew and Townsend, 1987; Townsend, 1989; Townsend and Hildrew, 1994; Townsend *et al.*, 1997).

Townsend *et al.* (1997) found that more disturbed communities contained a significantly higher percentage of individuals possessing the following traits: small size, high adult mobility, habitat generalist, clinger, streamlined/flattened and with two or more life stages outside the stream. Townsend and Hildrew (1994) also made the point that while resilience /resistance traits may be expected to predominate in highly disturbed areas, a mixture of resilience/resistance and non resilience/resistance traits would be expected in more stable locations, since species with resilience/resistance traits would not necessarily be excluded from the latter. Townsend *et al.* (1997) also found that refugia appear to ameliorate the effect of spates. Palmer *et al.* (1995) found that for Chironomids and copepods, species composition and relative abundance in refugium patches changed more in response to the spate than in the non-refugium patches. An influx of individuals from just a few species for each group was responsible for the change in assemblage structure. This suggests that resistance and resilience of the species composition of the community apparently are no greater in refugium patches than in non-refugium patches.

Death and Winterbourn (1995) found that species richness and density were markedly higher at the most stable sites, but species evenness peaked at sites with an intermediate disturbance. Their species evenness results suggested that competitive exclusion may be occurring patchily and that Huston's dynamic equilibrium model may have some validity, at least at the level of the patch. Competitive exclusion may be occurring at the most stable sites. In their study, species number declined as overall stream stability decreased. Of the 20 environmental variables examined, overall stability was the single best predictor of the number of species at a site (Death and Winterbourn, 1995). This is consistent with the results of Robinson and Minshall (1986) who found that the number of species on artificial substrata decreased as the frequency of disturbance increased.

In contrast Reice (1985) found no impact of disturbance frequency on species number or "diversity" (i.e. the Shannon index) in experimentally disturbed baskets of substrate. Lake *et al.* (1989) also found that similar numbers of species recolonised disturbed patches whether they had been disturbed once or three times prior to monitoring community

recovery. Death and Winterbourn (1995) suggest that increasing disturbance frequency in streams acts on species diversity primarily by reducing the time available for recolonisation following disturbance events.

Clausen and Biggs (2000) identified four key aspects of hydrology based on covariance among the sites through a principal components analysis. These groups were:

1) the size of the river (central tendency of magnitude);
2) the overall variability of flow (including magnitude of high and low flows);
3) the volume of high flows; and
4) the frequency of high flow events.

Many of the variables in group two were found to correlate significantly with periphyton diversity and invertebrate density and species richness (Clausen and Biggs, 1997). The relative magnitudes of high and low flows and the total time that the flow is high are factors likely to have important influences on biota through the intensity of habitat destruction associated with both drying during low flows and washout/destruction during high flows (e.g. Sagar, 1983, 1986; Delucchi, 1987).

Clausen and Biggs (1997) found that both the average flow conditions and some measure of variability were significantly related to most of the biological variables. These individual hydrological variables were more strongly correlated to the biological measures than the composite principal components. Thirty-one of their thirty-four flow variables were correlated with total invertebrate density, whereas only four variables correlated with diversity (Clausen and Biggs, 1997). Clausen and Biggs (1997) found a positive/curvilinear relationship between invertebrate densities and flood disturbance frequency.

Density of invertebrates was most strongly correlated with measures of flood variables/overall flow variability (Clausen and Biggs, 1997). Most previous studies have focused on the effects of single disturbances on invertebrates (Scrimgeour and Winterbourn, 1989; Quinn and Hickey, 1990), or disturbance of only two streams (Sagar, 1986; Grimm and Fisher, 1989; Scarsbrook and Townsend, 1993), and have generally concluded that invertebrate density increases with time since disturbance and thus densities

will be higher in less disturbed habitats. Death and Winterbourn (1995) recently extended this understanding by analysing invertebrate density over four seasons at eleven sites which covered a gradient in habitat suitability. They found that density decreased linearly with increasing instability. This conflicts directly with the results of Clausen and Biggs (1997), but a direct comparison may not be valid because Death and Winterbourn (1995) did not express their results as a function of absolute disturbance frequency. Whilst the mechanism to explain a reduction in invertebrate density at high flood frequency is simple and well known (shear stress removal of invertebrates either directly as physical abrasion /crushing by moving gravels, or indirectly as part of periphyton mats), it is more difficult to explain the increase in density from low to moderate flow frequencies (Clausen and Biggs, 1997).

However, community diversity did not show a similar pattern for Clausen and Biggs (1997). Diversity only correlated with four hydrological variables, of which three were flood variables. This only explained 14% of the variance. Diversity decreased weakly with increasing flood frequency (Clausen and Biggs, 1997). This trend does agree with previous studies which have demonstrated relatively lower diversity with higher disturbance (Sagar, 1986; Scarsbrook and Townsend, 1993).

Gibbins *et al.* (2001) found clear seasonal differences in invertebrate communities within each stream studied. They found that in two of their study streams, communities found in the spring were more variable from one year to the next, than those in summer or autumn. Such invertebrate community variability may be influenced by the variable nature of the hydrological and hydrochemical regimes in the months preceding spring sample collection (Gibbins *et al.*, 2001). Gibbins *et al.* (2001) also found that CCA ordinations using indices of hydrological and hydrochemical variation over preceding time periods were more successful at explaining temporal variation in invertebrate community structure than those using conditions on the day of sampling or average conditions over the preceding time period.

Jowett and Duncan (1990) found significant differences in flow variability between two major invertebrate community groups. The first group, described by Quinn and Hickey (1990) as "clean water" fauna dominated by mayflies and caddisflies, were more common

in rivers with low flow variability. The second group, characterised by molluscs, Chironomids or Oligochaeta, were more common in rivers with more variable flows.

Wood *et al.* (2000) looked at changes in the macroinvertebrate community response to flow variations in the a small groundwater dominated stream in the south east of England, over a six year period (a period which included the final year of the 1988-1992 drought). Their study highlighted the importance of stream discharge during February –May, which corresponds to the period of sustained high discharge in most English chalk streams (Berrie, 1992) and relates to maximum groundwater levels. They found a strong correlation between flow and the total number of individuals which may reflect the greater wetted bed area, and associated greater habitat diversity, during summers following wet winters. But this may also have been due to the warmer late-winter weather and spring temperatures within the benthic and hyporheic zones, and the higher benthic productivity associated with the higher groundwater flux. Indices of high discharge (or their absence), 4-7 months prior to sampling (i.e. winter-spring), were found to be the most important variables for describing the late summer community (Wood *et al.*, 2000).

A difficulty for stream ecologists is choosing appropriate flow variables from the plethora that are available to represent elements of the flow regime, for instance, over 50 different variables have been used in only six broad-scale studies of how flow regimes may influence in-stream biota (Poff and Ward, 1989; Jowett and Duncan, 1990; Poff, 1996; Richter *et al*, 1996; Clausen and Biggs, 1997).

The FREx (frequency of 'median*x' flow events; where x is a multiplier) variables directly depict the frequency with which biota in the rivers are probably being disturbed, or destroyed, by high flows (Clausen and Biggs, 2001). Clausen and Biggs (1997) selected the flood frequency (FRE3), where a flood is defined as flows higher than three times the median flow, as the most ecologically useful *overall* flow variable in New Zealand streams because it explained a significant amount of variance in four out of the six benthic community measures, and it had a clear mechanism of control of the biota which was commensurate with current stream ecosystem theory. Invertebrates had an increasing/curvilinear relationship with FRE3 (Clausen and Biggs, 1997).

Sagar (1986) demonstrated that major reductions in benthic invertebrate densities occurred when flows increased to three times the median discharge in a large New Zealand river. It was reported in Clausen and Biggs (1997) that, in an experimental study, Rutledge *et al.* (1992) found that invertebrate density peaked in the channels which were subjected to flow perturbations of three times their 'normal' flow at a frequency of 25/year compared to the channels where flows were either kept constant or had threefold flow perturbations at frequencies of 12 or 52/year. They also found (as did Clausen and Biggs, 1997) that invertebrate species richness varied little over a wide range in disturbance frequencies in the streams.

Gibbins *et al.* (2001) found that (by using Canonical Correspondence Analysis) the frequency of flows exceeding three times the median discharge (FRE3) proved the most useful single index of flow variability in four small Cairngorm streams. The significance of FRE3 (Clausen and Biggs, 1997; Gibbins *et al.*, 2001) within the ordinations provides further evidence of the importance of maintaining natural flow variation (e.g. Richter *et al.*, 1997).

Hildrew *et al.* (1984), Hildrew and Townsend (1987), Minshall (1988), Poff and Ward (1989), Townsend (1989) and Townsend and Hildrew (1994) tried to develop conceptual models that more thoroughly integrated the effects of abiotic and biotic processes on stream communities. For example Hildrew and Townsend (1987) suggested richness would be highest at intermediate levels of disturbance but more noticeably so under conditions of high productivity. Minshall (1988) speculated that species richness would be greatest in streams intermediate in both stability and harshness. Poff and Ward (1989) argued that biological structure in streams should be influenced by four hydrologic factors (intermittency, flood frequency, flood predictability, and flow predictability) and that invertebrate richness should be highest under conditions of high flow predictability, intermediate under conditions of high flood predictability, and low under conditions of either high flood frequency or intermittency.

Cowx *et al.* (1984) looked at the effects of drought on macroinvertebrate populations in the upper River Severn. Two main effects of the drought were observed: an initial reduction in abundance during the drought (around 40%) and a change in community structure in the

following year (Cowx *et al.*, 1984). In terms of changes in community structure, the numbers of Plecoptera and Coleoptera were reduced in 1977 compared with 1976, although this decline was counterbalanced by large increases in the numbers of Chironomidae and Simuliidae larvae, which are well-known opportunist groups (Cowx *et al.*, 1984). One of the major effects of drought on invertebrates is the overall decrease in the potentially colonisable habitats due to the reduction in width and depth of the river (Cowx *et al.*, 1984). The reduction in wetted bed area will also have long-term implications since many adult insects lay eggs in fast flowing or broken water (Sawyer, 1950) and if suitable areas are reduced subsequent populations will be affected (Hynes, 1958). This effect was evident in the Afon Dulas, and the seasonal maximum abundances of the major insect species, all showed considerable decrease in 1977 from their previous levels of abundance (Cowx *et al.*, 1984). The reduction was thought to be more acute because the severe drought overlapped the normal period of hatching (Cowx *et al*, 1984). Follow up surveys in 1978 and 1979 showed that, despite restrictions on the mode of recolonisation, the invertebrate fauna had fully recovered from the drought both in terms of taxonomic diversity and numbers of animals (Cowx *et al.*, 1984). These observations suggest that the invertebrate fauna of rivers can recover quite quickly from the effects of drought, usually within the span of several life cycles (Cowx *et al.*, 1984).

Extence (1981) found that the drought conditions present in 1975 and 1976 led to an increase in invertebrate populations. However, it was also found that a number of inidvidulas of certain groups such as caddisfly larvae and prosobranch molluscs (Potamopyrgus, Valvata, Bithynia) were eliminated from the river during the drought. The increased invertebrate population could be due to increased food supply as algal growth would have been large due to the sunny conditions. Another factor contributing to increased invertebrate production during the drought could have been raised water temperatures, stimulating faster growth and earlier reproductive activity (Extence, 1981).

Wood and Petts (1999) investigated the impacts of the 1988-1992 drought on a chalk stream, where despite riverine habitats being severely degraded in 1992, the recovery was dramatic in the following three years. They found that few taxa were eliminated during the drought suggesting the presence of refugia from which recolonisation takes place.

3.2.5. Habitat, disturbance and compensation flows

There have been no studies conducted on a twinned catchment basis with steady state compensation flow releases such as this one but there have been a number of studies looking at slightly different water resource problems.

Maddock *et al.* (2001) studied the reallocation of compensation flows in the upper Derwent catchment, Derbyshire. The invertebrate fauna of the rivers in that region were dominated by Ephemeroptera and Diptera, which is fairly typical of upland, soft-water streams with little or no organic pollution. Below abstraction points, biotic indices and other community analyses showed that the community overall was slightly reduced but with some species in greater abundance than upstream. Species with a preference for higher velocities and depths were found in lower abundance downstream of the abstraction points (Maddock *et al.*, 2001).

Gibbins *et al.* (2001b), studied the Kielder catchment in the North East of England, where again the compensation flows had also been set without taking the ecological needs of the river into account. These were to be altered including a transfer from the river Tyne to the Wear. Species level analysis revealed that the abundance of some invertebrate animals did change during transfers to the River Wear, although changes were not detected at all sites or during all the transfer releases. There was no evidence of either long or short term changes in invertebrate community structure or abundance. Hence, there was no reason for concern over the impact of transfers on the food base of the River Wear fish stocks. The River Wear regularly experiences natural flood events of a much greater magnitude than transfers, even during the summer period (Gibbins, 1996), and it was thought that this might help to explain the resistance of invertebrate communities to transfer-induced flow changes."

Gore (1977) took invertebrate samples to determine the distributions and abundances of the benthic fauna after various reservoir manipulations in a prairie river. He found that in the upper cold water section, influenced by hypolimnial discharge from the reservoir, was impoverished in insect fauna and dominated by the molluscs *Physa* and *Sphaerium*. The

lower warm water sections of the river contained two communities determined primarily by turbidity and periphyton cover (Gore, 1977). The upper warm water area was dominated by *Strophopteryx* and hydropsychid caddis larvae. The lower river was dominated by *Cheumatopsyche*. The summer fauna, in the warm water area was dominated by short-lived mayfly species (Gore, 1977). During the summer of 1975, the cold water section was invaded by many insects due to warming of the area when no hypolimnion was formed in the reservoir (Gore, 1977). Invasion was apparently due to increased thermal fluctuations which caused diapause eggs to hatch and influenced the upstream migration of older nymphs and larvae (Gore, 1977).

When taken with the above review of steady state and disturbance issues, it is possible to synthesise the role of compensation flow releases in relation to macroinvertebrate populations. Compensation flows tend to have the following hydrological effects: (i) they will provide a sustained low flow that reduced exposure to drought except in the special case that a decision is taken to lower the compensation release during an extreme drought; (ii) they will dampen the exposure of the river bed to extreme flow events, eliminating them when the dam is only at compensation flow and dampening them when the dam is overtopping[1]; (iii) this dampening will be reduced with distance from the dam as unregulated tributary inputs progressively increase in importance; and (iv) there will be an interaction between the magnitude of the compensation flow (i.e. (i)) and the flow variability related to (ii) and (iii), with larger compensation flows being less impacted upon by high flows, whether from tributaries or overtopping; and (v) the sensitivity to magnitude in (iv) can only be evaluated with respect to how a change on flow causes a change in the spatial distribution of velocity and depth (and by implication, wetted area), as determined by channel slope, geometry and roughness. The latter point is particularly important as it means that discussions of flow are incomplete without reference to the particular channel under consideration; the same flow conditions could have different results in different places. In other words, temporal disturbance will be conditioned by the spatial patchiness of the habitat under consideration both because this determines the amount of reachable refugia available as the flow increases or falls, but also because this patchiness determines how the nature of the refugia change as the flow varies.

[1] It can be shown that a full reservoir will still dampen extreme flow delivery events as a result of the time it takes to transmit delivered flow through the reservoir.

In terms of this research, this discussion creates a number of problems. First, it implies that the generic conditions as to how compensation flows should be changed to deliver habitat improvements are likely to be limited. Indeed, some of the differences in findings reported above may arise from the conditioning effect of habitat patchiness upon flow effects. Second, it implies that within the study streams, particular attention has to be given to the choice of representative reaches and their location with respect to possible tributary inputs. Third, despite an apparently simple research design based upon a paired catchment analysis, temporal variability in flow due to overtopping and/or tributary inputs means that the results must be considered in the context of flow variability unrelated to the compensation flow releases. Finally, whilst the focus of this chapter is evaluation of macroinvertebrate response to redesigning compensation flows, it will be necessary to explore macroinvertebrate response to changing spatial patterns of velocity and depth and this is addressed in Chapter 5 through a modelling approach. Throughout this chapter, we assume that the proposed compensation flow changes lead to mainly primary impacts (i.e. on velocity and depth) rather than impacts on water quality, substrate or habitat type.

3.3. Methods

3.3.1. Field data collection

3.3.1.1. Macroinvertebrate sampling techniques

There are a number of macroinvertebrate sampling techniques which have been widely used. Each of them has advantages and disadvantages in terms of their efficiency of sampling, ease of sampling and the costs involved. It is the purpose of this section to review the sampling techniques available in wadeable streams such as the study site, in order to contextualise the choice of samplers used. The two sampling techniques used in this study are kick and Surber sampling.

Both kick and Surber sampling only sample the upper part of the stream bed. Adkins and Winterbourn (1999) showed that when comparing invertebrate densities in freeze cores with Surber samples, the latter substantially underestimated the abundance of some taxa, even in the top 10cm of stream bed. Furthermore, the Surber sampler will not sample any invertebrates which live at a depth >10cm in the substrate (Adkins and Winterbourn, 1999). However, the process of freeze coring is a very expensive sampling method, and the use of such a method would put the feasibility and transferability of this research into question.

A comparison of kick sampling and Surber sampling by Storey *et al.* (1991), found that each kick sample consistently collected more individuals than Surber sampling. This should be expected due to the larger surface area of stream bed sampled by a single kick sample (Storey *et al.*, 1991). However, total species richness in Surber samples exceeded that of the kick samples on 84% of occasions. They also detected significantly higher numbers of low-occurrence taxa in the Surber samples when compared with the kick samples, and this may reflect the greater effort per unit area expended in Surber sampling (Storey *et al.*, 1991).

Results from Storey *et al.* (1991) suggested that substratum type influences the effectiveness of the kick methods. The substratum in their riffle reaches in headwater

streams was dominated by highly indented lateritic cobbles, which provide many small retreats. Storey *et al.* (1991) proposed that the heterogeneous habitat of the headwater streams influenced the effectiveness of the kick method, as evidenced by the greater number of low-occurrence taxa in Surber samples. Hornig and Pollard (1978) proposed that the kick technique will sample the more easily dislodged and highly mobile taxa, whereas the Surber method, being more intensive, will take cryptic and closely adherent taxa.

The inability of kick samples to detect low-occurrence taxa is of concern in environmental impact assessments (Storey *et al.*, 1991), as a key part of such assessments is the detection of rare and potentially endangered species. The findings of Storey *et al.* (1991) are that kick samples will not adequately sample the rare taxa. Thus the initial use of replicate Surber sampling, particularly in areas which have not been previously sampled is advisable to detect them. It is these rare taxa which may display the influence of water stress upon the systems, and allows us to test whether the Rivelin has too little discharge and the Loxley too great a discharge.

As a result of the above, it was decided to use a combination of both kick and Surber samples in this work.

3.3.1.2. Kick sampling

Kick sampling data have been collected by the Environment Agency over the last 15 years for each of the rivers. Kick sampling is the standard Environment Agency macroinvertebrate sampling method in wadeable rivers. It is logical to continue to collect kick samples as part of this research as: (i) it adds data to the Environment Agency's macroinvertebrate database; and (ii) the use of kick samples is of importance with regard to the transferability of this work, as kick samples are the standard benthic fauna sampling technique in wadeable upland Millstone Grit streams, allowing the results from this research to be compared with those from other catchments. In this thesis, kick samples will be used only to gain an understanding of historical macroinvertebrate variability, and are not used for compensation flow assessment.

The macroinvertebrate data record used for the Rivelin extends from December 1989 to the 1st of April 2004. The River Loxley record covers from October 1985 to the 1st of April 2004. The Hipper kick sampling data covers from April 1990 to the 1st of April 2004 (Table 3.3.1). The data were supplied by the Environment Agency, with taxa identified to the family level. The abundance of a given species was indicated by a letter rather than a number (A = 1-9; B=10-99; C=100-999). As software such as PRIMER is not able to handle non-numeric data, the letter A was given a value 1, the letter B a value of 2 and so on.

Site Name	Number of Samples	Span of sampling
RIVELIN		
Rivelin Hollins Bridge (RHB)	15	1/12/1989 – 1/4/2004
Rivelin Downstream (RD)	5	24/5/2002 – 1/4/2004
Rivelin Mill (RM)	27	12/4/1996 – 1/4/2004
Rivelin Upstream (RU)	4	23/4/2002 – 1/4/2004
Rivelin – Upstream of Dams (RUSD)	15	18/7/1990 – 1/4/2004
LOXLEY		
Loxley A61 (LA61)	38	29/10/1985 – 1/4/2004
Loxley at Malinbridge (LMB)	7	19/4/1990 – 1/4/2004
Loxley Rowell Lane Bridge (LRB)	19	5/5/1992 – 1/4/2004
Loxley Downstream (LD)	4	24/4/2002 – 1/4/2004
Loxley Upstream(LU)	5	21/5/2002 – 1/4/2004
Loxley Upstream of Dams (LUSD)	12	17/9/1991 – 1/4/2004
HIPPER		
Hipper at Chesterfield (H@CHEST)	13	9/4/1990 – 1/4/2004
Hipper at Somershall Park (H)	5	28/5/2002 – 1/4/2004
Hipper downstream of Brookside Beck (H D/S BB)	9	4/6/1992 – 1/4/2004

Table 3.3.1. Table to show the span of sampling and number of samples included in the kick sample analysis.

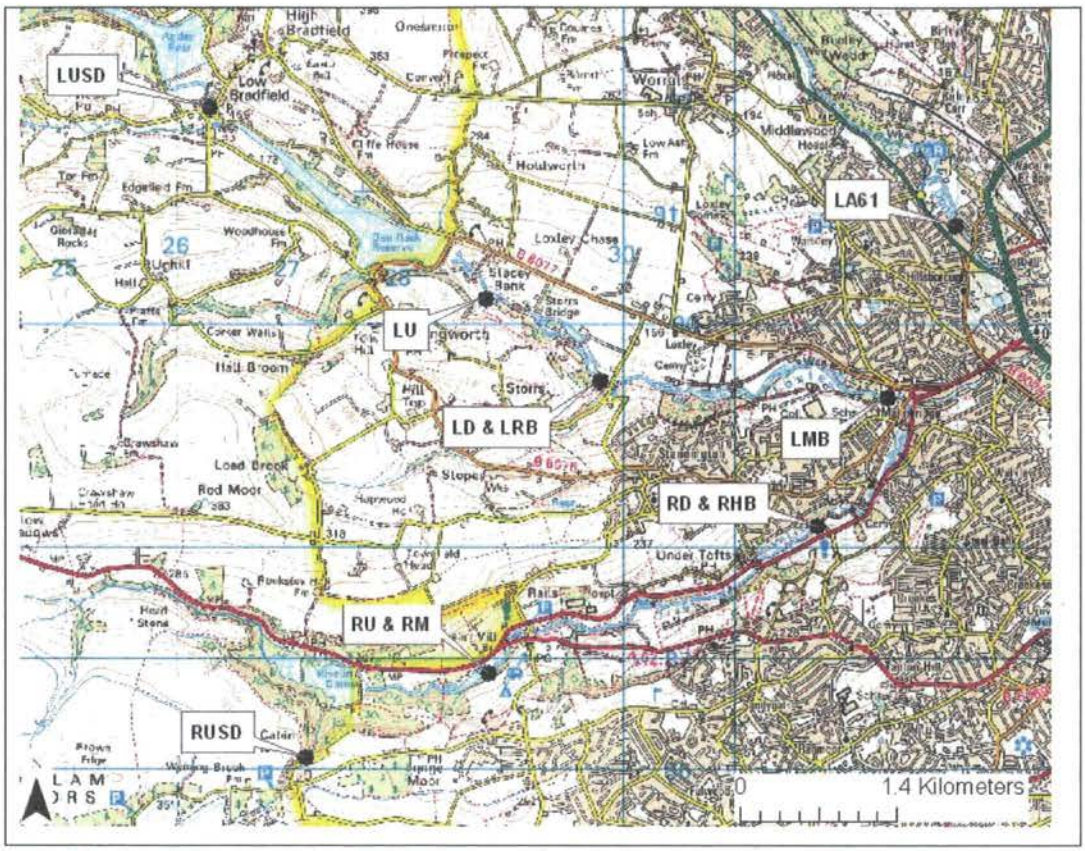

Figure 3.3.2. Figure to show the location of all invertebrate sampling sites.

3.3.1.3. Surber sampling

The macroinvertebrate sampling survey specifically designed for this project was undertaken in spring 2001, spring 2002, and autumn 2002 to autumn 2005 (in the spring, summer and autumn), with the exception of that of summer 2002. The data in spring 2001 were collected at two sites (RU and LD) as part of a preliminary study by the Environment Agency (unpublished report). The spring 2002 sampling season was conducted as part of a Masters dissertation. On each occasion the sampling was conducted at five locations: Rivelin Mill (RU) and Roscoe Bridge (RD) on the River Rivelin; downstream of weir (LU) and Rowell Lane Bridge (LD) on the River Loxley, and Hipper Somershall Park (H) on the River Hipper. U indicates upstream and D indicates downstream. The sites were assessed for representiveness using the results from the geomorphological survey.

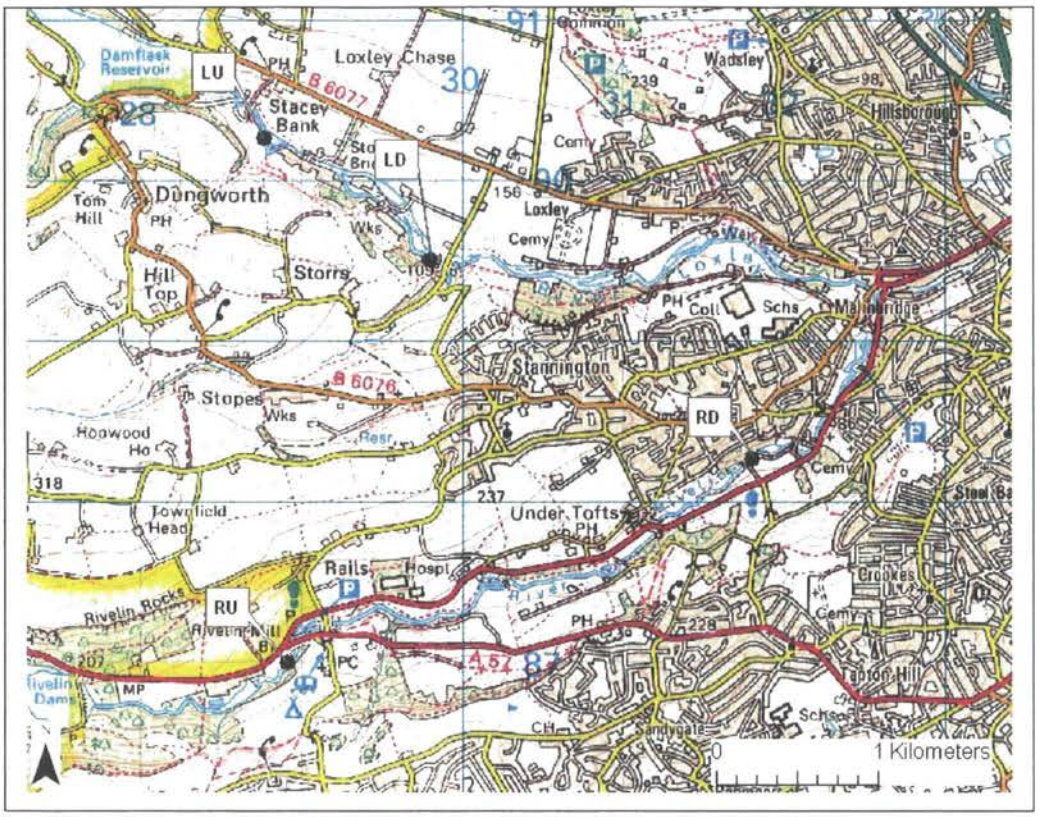

Figure 3.3.2. Map to show the location of the Surber sampling sites.

A Surber sampler (Surber, 1970) (0.25m^2 area base with a 0.5mm diameter mesh size) was used to sample discrete areas of bed within each of the rivers. Discrete areas of the bed were focussed upon so that:

1) the marginal and central areas of the channel could be sampled independently; and
2) a suite of environmental variables could be measured at the specific sampling locations.

Five replicates were obtained, with a 10m spacing so as to estimate spatial variability of the macroinvertebrate populations (Figure 3.3.3). The sampling location was selected initially so that a riffle and a pool would be included in the 40m sampling length. As was discussed in the literature review, a number of studies have found inter-habitat variation in macroinvertebrate richness (Angradi, 1996; Brown and Bussock, 1991; Grubaugh *et al.*, 1996; McCulloch, 1986; Thorp, 1992; Wohl, 1995), whereas some studies have not (e.g. Jenkins *et al.*, 1984). Sampling started from the same location on each visit.

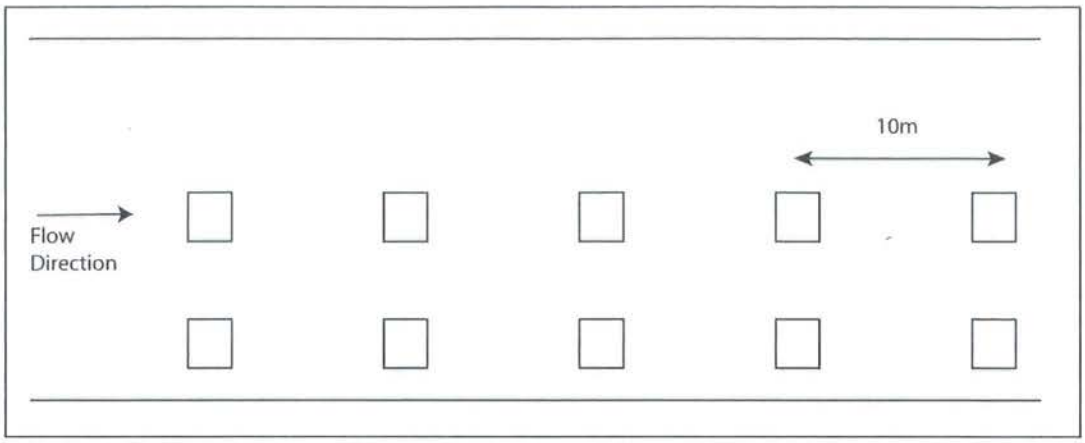

Figure 3.3.3. Figure to show a plan view of the Surber sampling structure. The location of the Surber samples being represented by the squares. Not to scale.

The procedure with the Surber sampler followed Surber (1970). Large stones were first picked out and scrubbed into the collecting net to remove attached invertebrates. The substratum was then vigorously disturbed to a depth of around 10cm, for around two minutes and the current allowed to sweep the sediment and organic matter containing the resident macroinvertebrate fauna into the net. The contents of the net were then placed in a labelled plastic bag for storage and transport to the laboratory. Once in the laboratory, the samples were preserved in Formalin before being sorted and identified to species level. The RU and LD samples in spring 2002 were sorted and identified by the author, with the remaining samples identified by consultants. In each case the Environment Agency data quality control rules were used.

After the Surber sample was taken, a suite of environmental variables were also measured at that same location:

- Water depth. Recorded in cm.

- Bed substrate. The substrate was recorded in terms of percentages of boulders and cobbles, pebbles and gravel, sand and silt and clay. The size classes (using the Wentworth scale) of the substrate had to be estimated by hand, as measuring the substrate before the sample was taken would disturb the invertebrates. This enabled the calculation of the mean substratum size category (MSUB), which is a standard

measure used by the Environment Agency and in the River Invertebrate Prediction and Classification System (RIVPACS).

$$MSUB = \frac{(17.75B - 3.25P - 2.00S + 8.00C)}{100} \qquad (3.3.1)$$

where:

B = % boulders and cobbles

P = % pebbles and gravel

S = % sand

C = % silt and clay

- Flow velocity. Measurements were taken using an electromagnetic flow meter at 40% of the flow depth up from the bed, as it was assumed to be the depth-integrated velocity (Richards, 1982). At each site, at least three flow velocity replicates were recorded in order to ensure the precision of the velocity measurements, with the three replicates having to be within ±10% of each other.

- Flow type. The flow type was recorded as one of the following types, using the same flow types as for the Environment Agency kick sampling protocol:
  - No perceptible flow (0): Stagnant water.
  - Pool (1): Moving water with smooth water surface.
  - Glide (2): Moving water with signs of upwelling on the surface.
  - Run (3): Moving water with slight broken surface, very little white water.
  - Riffle (4): Moving water with broken surface, abundance of white water.

The same procedure was then undertaken for the marginal sample (around 1m from the bank) at the same cross section, and for every subsequent cross-section moving upstream. After the above procedure was completed, a three minute kick sample (following Environment Agency protocol) was taken from the same area of the channel sampled during the Surber sampling, but care was taken to avoid the areas directly sampled with the Surber sampler.

3.3.2. Data analysis methods

Both univariate and multivariate techniques were used in the analysis of the data outlined above. Univariate methods are ones which collapse the full set of species counts for a sample into a single coefficient, such as a diversity index. Multivariate methods are those which base their comparisons of two (or more) samples on the extent to which these samples share particular species, at comparable levels of abundance. Both of these analyses will be used to address the aims identified in section 3.1, and the use of each method will be justified in the following section.

For all of the following analyses, the species by sample matrix generated by the Surber sampling was aggregated to the family level before any analysis was conducted. This was done for a number of reasons: (i) for the ease of comparability between the kick and Surber sample results; and (ii) when the invertebrates were being identified, not all invertebrates were identified to the species level, when large numbers of a particular family were captured. As such, on many occasions, there would be a given species and a large number of family sp. Either the family sp. data could have been aggregated into the known species (where often there was more than one) without exact knowledge of numbers, or the family sp. counts could have been introduced into the species list as a measurement artefact. It was therefore thought better to aggregate to the family level to remove this error.

3.3.2.1. Univariate 1: Measures of Diversity

A large part of most studies on macroinvertebrates appears to focus on the diversity of the invertebrate community (Section 3.2). Diversity indices can be considered to have the following advantages: (i) they are strictly quantitative, dimensionless, and lend themselves to statistical analysis (Cook, 1976); (ii) most are relatively independent of sample size (Pinder *et al.*, 1987); and (iii) no assumptions are made as to the relative tolerances of individual species which may be very subjective (Pinder *et al.*, 1987).

One of the most commonly used diversity indices is the Shannon Index, and it is used in four ways in this study: (i) to detect any differences in diversity between rivers and/or sites; (ii) to assess within sample variability; (iii) to asses the natural variability of this index

through the sampling seasons; and (iv) to assess any changes to the diversity of macroinvertebrate populations after the change in compensation flow releases.

Only one index was assessed, as even if we looked at a selection and found one was effective at addressing the aims outlined above, there would be no guarantee as to the transferability of that given index. The Shannon diversity index could only be calculated for the Surber samples themselves as the historical macroinvertebrate data from the Environment Agency came in the form of abundance codes (A= 1-9; B = 10-99; C=100-999) rather than actual abundances. The Shannon index was calculated from (Shannon and Weaver, 1949):

$$H' = \sum_{i=1}^{s} \frac{n_i}{N} \left| \log \frac{n_i}{N} \right| \qquad (3.3.2)$$

Where:

H' = Diversity

N_i = Count of individuals of species i

N = Total count of all individuals of all species

It is widely accepted that the Shannon diversity index is not without its faults. Many other diversity indices exist, and while it is not the aim of this thesis to examine the effectiveness of diversity indices in detecting the influence of water stress, it was examined to a certain extent in Mould (2002, unpublished Masters thesis). That work inferred the effectiveness of diversity indices in terms of detecting differences between central and marginal samples (assuming that there was a difference). As such Simpson's index and Margalef's index found more differences between central and marginal samples. However, the purpose of using the Shannon index in this report is to provide a measure of the quality of the macroinvertebrate populations at the various sites that is comparable with Environment Agency protocols.

3.3.2.2. Univariate 2: The Lotic Invertebrate Flow Evaluation (LIFE) index.

Altering the compensation flow regimes may affect flow velocities in the rivers. Different organisms have different velocity preferences. Hence it is of interest to look at which organisms are present, in terms of velocity preferences. The Lotic Invertebrate Flow Evaluation (LIFE) index (Extence *et al.*, 1999) was seen as important as it calculates an index with regard to invertebrates estimated flow velocity preferences. Three questions can be addressed by using the LIFE index: (i) can we detect differences between the rivers and/or sites in terms of the velocity preferences for the pre-change discharges?; (ii) what is the natural variability within the systems in terms of the flow preferences of the invertebrates present?; and (iii) do the flow velocity preferences of the macroinvertebrates change with the compensation flow alteration.

The LIFE index is calculated as follows (Extence *et al.*, 1999):

$$LIFE = \frac{\sum fs}{n} \qquad (3.3.3)$$

Where,

Σfs = The sum of individual taxon flow scores for the whole sample

n = The number of taxa used to calculate Σfs

The LIFE calculation involves individual flow scores (*fs*) for each scoring taxon present in a sample being obtained from the matrix shown in Table 3.3.3, using the estimated abundances (Table 3.3.2) and the defined flow group associations, given in appendices to the paper of Extence *et al.* (1999). In this study, the analysis was performed at the family level.

Category	Estimated Abundance
A	1-9
B	10-99
C	100-999
D	1000-9999
E	10,000+

Table 3.3.2. Table to show the macroinvertebrate abundances used in the LIFE index system. From Extence *et al.* (1999).

	Flow groups	Abundance categories			
		A	B	C	D/E
I	Rapid	9	10	11	12
II	Moderate/fast	8	9	10	11
III	Slow/sluggish	7	7	7	7
IV	Flowing/standing	6	5	4	3
V	Standing	5	4	3	2
VI	Drought resistant	4	3	2	1

Table 3.3.3. Table to show the scores (fs) for the different abundance categories of taxa associated with flow groups I-VI. From Extence *et al.* (1999).

Not only is the LIFE index calculated for a number of kick samples, the LIFE velocity preference groups are used to infer habitat preferences of families and species throughout the chapter, with the groups and corresponding flow velocities outlined in Table 3.3.4. The LIFE scores were not calculated for any of the Surber samples.

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

Table 3.3.4. Table to show the LIFE groups and corresponding velocity preferences and ecological associations, from Extence *et al.* (1999).

3.3.2.3. Multivariate data analysis

The multivariate analysis has three aims: (i) to establish where between site differences are greater than within site differences; (ii) to isolate which species contribute most importantly to particular sites; and (iii) to establish what (if any) relationship exists between the measured environmental variables and the invertebrate community.

Rather than calculating a score or an index as with the univariate methods, multivariate methods use the species by sample array as the base for the analysis. The multivariate work conducted for this thesis used the PRIMER software. The starting point for many of the analyses within PRIMER is the concept of similarity between any pair of samples in terms of the biological communities they contain (Clarke and Warwick, 2001). Similarity matrices are calculated using a similarity coefficient, which is an algebraic measure of how close the abundance levels are for each species, averaged over all species, and defined such that 100% represents a total similarity and 0% a complete dissimilarity (Clarke and Warwick, 2001). The similarity coefficient used was the Bray-Curtis coefficient (Bray and Curtis, 1957). The Bray-Curtis coefficient was used because it satisfies all of the following, practically desirable criteria (Clarke and Warwick, 2001): (i) it takes the value 100 when two samples are identical; (ii) it takes the value 0 when two samples have no species in common; (iii) a change of measurement unit does not affect its value; (iv) its value is unchanged by inclusion or exclusion of a species which is jointly absent from the two samples; (v) inclusion (or exclusion) of a third sample, C in the data array makes no difference to the similarity between samples A and B; and (vi) it has the flexibility to register differences in total abundance for two samples as a less-than-perfect similarity when the relative abundances for all species are identical.

The similarity between the *j*th and the *k*th samples can be defined as follows;

$$S_{jk} = 100\left\{1 - \frac{\sum_{i=1}^{p}\left|y_{ij} - y_{ik}\right|}{\sum_{i=1}^{p}\left(y_{ij} + y_{ik}\right)}\right\} \qquad (3.3.4)$$

Here y_{ij} represents the entry in the ith row and jth column of the data matrix, i.e. the abundance for the ith species in the jth sample (i = 1, 2, ..., p; j = 1, 2, ..., n). Similarly, y_{ik} is the count for the ith species in the kth sample.

In many species-list data sets, the similarities could be unduly dominated by the counts of the most dominant species (Clarke and Warwick, 2001). Transformations can therefore be used as a way of defining the balance between contributions from common and rarer species in the measure of similarity of two samples. Different transformations can be performed depending on the aspect of the invertebrate community which is of interest.

As explored in section 3.1, there are two main sources of data used in this chapter. Firstly, the historical species sample data supplied by the Environment Agency contained taxa identified to the family level, and the abundance of a given species was indicated by a letter rather than a number (A = 0 – 9, B = 10 – 99, C = 100 – 999, D = 1000 – 9999). As software such as PRIMER is not able to handle non-numeric data, the letter A was given a value 1, the letter B a value of 2 and so on. The same changes were made to the kick sample data collected specifically for this project.

For the Surber samples a full species by sample matrix is available. However, in these data, the numerical dominance of the more abundant species may merely cloud the more subtle community patterns which need to be investigated in this paper. Therefore, for the analysis of the Surber samples, the counts of Oligochaeta and Chironomids (dominant species in these rivers) were removed from the matrix (section 3.4.1.1). Furthermore, a log(x+1) transform was performed on the data. This further down-weighted the importance of the more abundant species in the species lists and reduce the clustering of common and abundant taxa at the centre of the ordination plot (Castella *et al.*, 1995). It was hoped that by using this 'stronger' transformation, more subtle differences in community structure can be teased out, in order to assess the impact of compensation flows on the invertebrate community. Examples of studies which have used this transformation include: Wood *et al.* (2000); and Smith *et al.* (2003).

The above discussion of transformations and Bray-Curtis similarity applies solely to the biotic data. As discussed in section 3.1, a suite of environmental variables were also

collected at the same locations as the Surber samples, and it is possible to create a triangular dissimilarity matrix of the environmental variables, and analyse them using the Euclidean distance algorithm. This is done so that the relationship between the sites and rivers can be assessed for the environmental variables, as with the biotic data, and as such enhance our understanding of the relationship between habitat and the invertebrate populations.

The natural distance between any two points in space is referred to as a Euclidean distance. The equivalent of a species by sample matrix can be created using the measured environmental variables, which are listed as rows instead of the species names. In terms of a species abundance matrix, the Euclidean distance between samples j and k is defined algebraically as:

$$d_{jk} = \sqrt{\sum_{i=1}^{p} (y_{ij} - y_{ik})^2} \qquad (3.3.5)$$

This formula creates a similarity matrix similar to that created by the Bray-Curtis algorithm, and means that the environmental data can be treated in the same way as the biotic data. This opens up many data analysis possibilities. The data was normalised before the Euclidean algorithm was used, in order to remove the influence of the different sampling units on the analysis.

In order to test statistically between groups of samples within the multivariate framework, the ANOSIM routine was used. The groups of samples are user-defined, in that groups can be separate years, sites, or rivers. For this routine the null hypothesis (H_o) is that there are no differences in community structure between groups. There are four reasons why this is of interest: (i) identifying differences between sites and rivers will help to further establish the structure and quality of the invertebrate populations (aim 1); (ii) comparing the invertebrate communities between the sampling seasons gives a further insight into the natural variability of the systems (aim 2); (iii) it will help to analyse the extent to which the Rivelin had too little discharge and/or the Loxley too great a discharge in the a priori scenario (aim 3), by detecting differences between the rivers and sites; and (iv) it will allow

us to assess whether differences between rivers and sites remain/are created after the alterations to the compensation flows (aim 4).

In order to examine H_0 there are three main steps (for more details see Clarke and Green, 1988):

1) A test statistic is computed reflecting the observed differences between sites, contrasted with differences among replicates within sites. The test is based upon the corresponding rank similarities between samples in the underlying triangular similarity matrix. If $\bar{r}_W$ is defined as the average of all rank similarities among replicates within sites, and $\bar{r}_B$ is the average of rank similarities arising from all pairs of replicates between different sites, then a suitable test statistic is

$$R = \frac{(\bar{r}_B - \bar{r}_W)}{0.5M} \qquad (3.3.6)$$

where $M = n(n-1)/2$ and n is the total number of samples under consideration. The denominator is such that: (a) R can never technically lie outside the range (-1, 1); (b) R = 1 only if all replicates within sites are more similar to each other than replicates from other sites; and (c) R is approximately zero if the null hypothesis is true, so that similarities between and within sites will be the same on average.

R will usually fall between 0 and 1, indicating some degree of discrimination between sites. An R substantially less than zero is unlikely since it would correspond to similarities across different sites being higher than those within sites. Such an occurrence is more likely to indicate an incorrect labelling of samples. The R statistic itself is a useful comparative measure of the degree of separation of sites, and its value is at least as important as its statistical significance. As with univariate tests, it is perfectly possible for R to be significantly different from zero yet inconsequentially small if there are many replicates at a given site.

2) Recompute the statistic under permutations of the sample labels. Under the null hypothesis H_0 there will be little effect on the value of R on average if the labels

identifying which replicates belong to which sites are arbitrarily rearranged. This is the rationale for a permutation test of H_o; all possible allocations of the labels are examined and the R statistic recalculated for each.

3) Calculate the significance level by referring the observed value of R to its permutation distribution. If H_o is true, the likely spread of values of R is given by the random arrangements, so that if the true value of R looks unlikely to have come from this distribution there is evidence to reject the null hypothesis.

The ANOSIM routine was used on both biotic and abiotic similarity matrices.

Whilst the ANSOIM technique focussed on detecting differences between groups of samples, it is also of interest is to discover what species are important to each of the sites. The SIMPER routine (Clarke, 1993) is used to do this. There are four reasons why this is of interest: (i) once the important species have been identified, they will help to further establish the structure and quality of the invertebrate populations (aim one); (ii) it will also be possible to use these techniques to see if the species important to a given site varies with time, and so gives a further insight into the natural variability of the systems (aim 2); (iii) it will help to analyse the extent to which the Rivelin had too little discharge and/or the Loxley too great a discharge in the a priori scenario (aim 3), by identifying the important families within each of the rivers; and (iv) to assess whether the species important to each of the sites changed as a result of the alteration of compensation flows.

The fundamental information on the multivariate structure of an abundance matrix is summarised in the Bray-Curtis similarities between samples. By disaggregating these, one can identify the species responsible for particular aspects of the multivariate picture (Clarke, 1993). For Bray-Curtis dissimilarity δ_{jk} between two samples j and k, the contribution from the ith species, $\delta_{jk}(i)$, could simply be defined as the ith term in the summation of the equation below (Clarke, 1993):

$$\delta_{jk}(i) = \frac{100|y_{ij} - y_{ik}|}{\sum_{i=1}^{p}(y_{ij} - y_{ik})} \qquad (3.3.7)$$

$\delta_{jk}(i)$ is then averaged over all pairs (j, k), with the j in the first and k in the second group, to give the average contribution $\overline{\delta}_i$ from the ith species to the overall dissimilarity $\overline{\delta}$ between groups 1 and 2. Typically, there are many pairs of samples (j, k) making up the average $\overline{\delta_i}$, and a useful measure of how consistently a species contributes to $\overline{\delta_i}$ across all such pairs is the standard deviation SD(δ_i) of the $\delta_{jk}(i)$ values. If $\overline{\delta_i}$ is large and SD(δ_i) small (and thus the ratio $\overline{\delta_i}$/SD(δ_i) is large), then the ith species not only contributes much to the dissimilarity between groups 1 and 2 but it also does so consistently in inter-comparisons of all samples in the two groups. It is thus a good discriminating species.

For each group comparison (be it comparing sampling seasons at a given site, or comparing the sites for a given season), the output consists of a list of species and their cumulative dissimilarity to differentiate between the groups, with the taxa having the largest dissimilarity coefficient at the top of the list. The taxa are then listed in the order of decreasing magnitude of the dissimilarity coefficient. This routine can also be used to assess the similarity of species within a given set of samples, and as such these are considered to be important species as they contribute most to the statistical similarity within the given set of samples. The SIMPER routine was run for each sampling site in each sampling season so as to get a subset of important species for each season at each sampling site.

It is also of interest to assess to what extent any environmental variables affect the community structure of the macroinvertebrate population in the rivers for a number of reasons: (i) details of the relationship between the biotic and abiotic data are important in terms of trying to determine whether the pre-change compensation flow levels had an impact on the invertebrate communities; and (ii) if the relationship between the measured environmental variables and the invertebrate communities changed after the change in compensation flows, it may be an indication of that alteration in compensation flows is impacting upon the invertebrate communities.

To this end the BIOENV routine was used (Clarke and Ainsworth, 1993). The intuitive premise adopted in BIOENV is that if the suite of environmental variables responsible for structuring a community were known, then samples having rather similar values for these

variables would be expected to have rather similar species composition (Clarke and Ainsworth, 1993). The matching of biotic to environmental variables can take place as outlined schematically in Fig. 3.3.4. Combinations of the environmental variables are considered at steadily increasing levels of complexity, i.e. k variables at a time (k = 1,2,3,...., v).

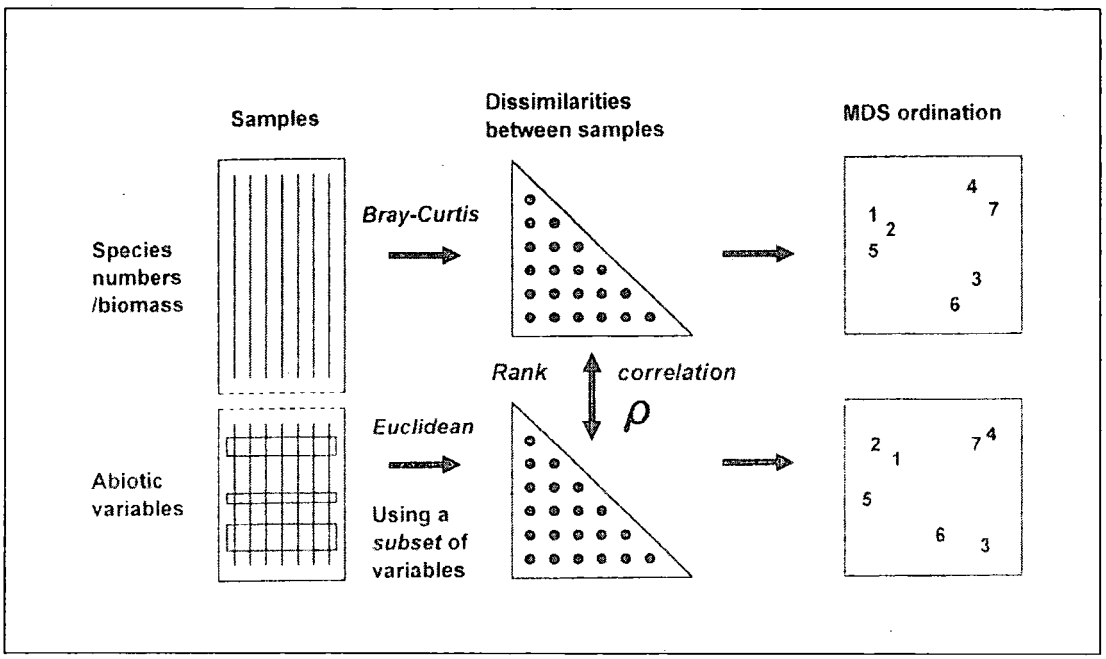

Figure 3.3.4. Schematic diagram of the BIO-ENV procedure: selection of the abiotic variable subset maximising rank correlation (ρ) between biotic and abiotic (dis)similarity matrices. From Clarke and Warwick (2001).

An ordination based on this abiotic information would group sites in the same way as for the biotic plot. If key environmental variables were omitted, the match between the two plots would deteriorate and, by the same token, the match will also worsen if abiotic data which are irrelevant to the community structure are included. Thus the link between biotic and environmental data can be analysed. The matching coefficients are defined between the (unravelled) elements of the respective rank similarity matrices $\{r_i;\ i = 1, ..., N\}$ and $\{s_i;\ i = 1,.....,N\}$, where $N=n(n-1)/2$ and n is the number of samples. The ranks are compared using Spearman's rank correlation.

$$\rho_s = 1 - \frac{6}{N(N^2-1)} \sum_{i=1}^{N} \frac{(r_i - s_i)^2}{r_i + s_i} \qquad (3.3.8)$$

The constant terms are defined such that, ρ lies in the range (-1,1), with the extremes of ρ = -1 and +1 corresponding to the cases where tow sets of ranks are in complete opposition or complete agreement (Clarke and Warwick, 2001). Values of ρ around zero correspond to the absence of any match between the two patterns.

3.3.2.4. Guild Analysis

Aquatic macroinvertebrates can be thought of as belonging to one or more functional feeding groups (e.g. Cummins, 1973). These are of interest as it will introduce an element of functionality and habitat use into the analysis and this is of interest for four reasons; (i) in terms of analysing the structure and quality of macroinvertebrate populations in the rivers, by using the feeding guild information will allow us to examine the habitat use in a more functional manner; (ii) variations in the proportions of functional feeding groups available will provide an insight into natural variability in the streams; (iii) the proportions of functional feeding groups may be able to illustrate if the pre-change compensation flow regimes had any impact upon the Rivelin and Loxley; and (iv) by the same token as (3), any changes detected in the proportions of the functional feeding guilds when the pre-and post change data is compared could perhaps be linked back to the alteration of the compensation flow releases.

As with the multivariate and univariate analyses (and for the same reasons), the guild analysis was conducted at the family level. The guilds were created after an extensive literature survey (Table 3.3.5). They were allocated as a function of the entire family, and not just the species of that family which were found in the study rivers. This will help with the transferability of the study.

FAMILY	Feeding Group	Reference
EPHEMEROPTERA		
BAETIDAE	Collector-gatherer	Elliott *et al.* (1988)
CAENIDAE	Collector-gatherer	Elliott *et al.* (1988)
EPHEMERELLIDAE	Collector-gatherer	Elliott *et al.* (1988)
HEPTAGENIIDAE	Collector-gatherer	Elliott *et al.* (1988)
LEPTOPHLEBIIDAE	Collector-gatherer	Elliott *et al.* (1988)
EPHEMERIDAE	Collector-filterer	Elliott *et al.* (1988)
PLECOPTERA		
LEUCTRIDAE	Shredder	Hawkins and Seddell (1981); Fenoglio (2005)
CHLOROPERLIDAE	Predator	Hawkins and Seddell (1981); Fenoglio (2005)
NEMOURIDAE	Shredder	Hawkins and Seddell (1981); Heino (2000); Fenoglio (2005).
PERLODIDAE	Predator	Hawkins and Seddell (1981)
TAENIOPTERYGIDAE	Scraper	Pretty *et al.* (2005), but only for *Brachyptera risi*
CAPNIDAE	Shredder	Pretty *et al.* (2005)
TRICHOPTERA		
PSYCHOMYIIDAE	Scraper	Edington and Hildrew (1995)
RHYACOPHILIDAE	Predator	Cummins (1973);
GLOSSOSOMATIDAE	Scraper	Cummins (1973); Fenoglio (2005)
ODONTOCERIDAE	Predator	Dangles (2000)
SERICOSTOMATIDAE	Shredder	Dangles (2000).
LEPTOCERIDAE	Shredder	Cummins (1973)
LIMNEPHILIDAE	Shredder	Cummins (1973)
HYDROPSYCHIDAE	Filterer	Hynes (1961); Fenoglio (2005); Bis *et al.* (2000); Hawkins and Seddell (1981)
POLYCENTROPODIDAE	Predator	Cummins (1973)
LEPIDOSTOMATIDAE	Shredder	Hawkins and Seddell (1981); Fenoglio (2005)
HYDROPTILIDAE	Scraper	Hawkins and Seddell (1981)
BERAEIDAE	Collector-gatherer	Fenoglio (2005)
GOERIDAE	Scraper	Cummins (1973)
PHRYGANEIDAE	Shredder	Cummins (1973)
DIPTERA		
SIMULIIDAE	Filterer	Pretty *et al.* (2005)
EMPIDIDAE	Predator	Pretty *et al.* (2005)
TIPULIDAE	Shredder	Heino (2000); Bis *et al.* (2000); Fenoglio (2005)
CERATOPOGONIDAE	Predator	Heino (2000); Bis *et al.* (2000); Hawkins and Seddell (1981)
PSYCHODIDAE	Collector-gatherer	Hawkins and Seddell (1981)
DIXIDAE	Collector- gatherer	Heino (2000)
COLEOPTERA		
ELMIDAE	Scraper	Cummins (1973)
DYTISCIDAE	Predator	Cummins (1973); Fenoglio (2005)
HYDRAENIDAE	Collector- gatherer	Fenoglio (2005)
HYDROPHILIDAE	Collector- gatherer	Cummins (1973)
HYDRACARINA	Predator	Merritt and Cummins (1978); Hawkins and Seddell (1981); Heino (2000); Bis *et al.* (2000)
OLIGOCHAETA	Collector-gatherer	Hawkins and Seddell (1981); Heino (2000); Bis

		et al. (2000).
HIRUDINEA		
ERPOBDELLIDAE	Predator	Bis *et al.* (2000)
GLOSSIPHONIIDE	Predator	Heino (2000); Seaby *et al.* (1996)
CRUSTACEA		
GAMMARIDAE	Shredder	Goodyear and McNeil (1999); Heino (2005)
ASELLIDAE	Shredder	Goodyear and McNeil (1999); Bis *et al.* (2000)
GASTROPODA		
ANCYLIDAE	Scraper	Fenoglio (2005)
PLANORBIIDAE	Scraper	Heino (2000); Bis *et al.* (2000)
LYMNAEIDAE	Scraper	Heino (2000, 2005); Fenolglio (2005)
BIVALVIA		
SPHAERIIDAE	Filterer	Heino (2000, 2005); Bis *et al.* (2000)
TRICLADIDA		
PLANARIIDAE	Predator	Dangles (2002)
GASTROPODA (ASWELL)		
PHYSIDAE	Scraper	Goodyear and McNeil (1999)
MEGALOPTERA		
SIALIDAE	Predator	Heino (2000, 2005); Bis *et al.* (2000); Hawkins and Seddell (1981).
HEMIPTERA		
GERRIDAE	Collector-gatherer	Cummins (1973)

Figure 3.3.5. Figure to show the feeding guilds of the families present in the Rivelin, Loxley and Hipper. Guilds based on all species within the families. All families not listed are counted as unknown.

3.3.3. Summary

Table 3.3.6 shows a summary of the aims of this chapter, and the data sets and analysis methods used in addressing each of the aims. This section has outlined the sampling methodology, in which both the kick and Surber sample techniques are critiqued and explained. Both univariate and multivariate data analysis methods will be used to analyse the data in subsequent section and the purpose and methodology of each technique has been explained.

AIM	DATA SETS	ANALYTICAL METHODS
1. To establish the structure and quality of the macroinvertebrate populations in the rivers.	- EA kick samples - Surber samples -Environmental data	- Shannon diversity - LIFE index -Guilds - SIMPER - BIOENV
2. Assessing the natural variability of the data.	-Surber samples	-Shannon diversity -ANOSIM -SIMPER
3.. Detecting differences in macroinvertebrate populations can be related to the discharges in the streams.	- EA kick samples - Surber samples - Environmental data	- LIFE index - ANOSIM - SIMPER - BIOENV
4. Assessing the impacts of altering the compensation flows on the invertebrate community.	-Surber samples -Environmental data	-Shannon diversity -LIFE score -Guilds -ANOSIM -SIMPER -BIOENV

Table 3.3.6. Table to show a summary of the data analysis techniques used in this chapter.

3.4. Results

This section presents the results of the macroinvertebrate data analysis. It is separated into three sections. The first section assesses the quality and structure of the invertebrate communities within the three study rivers using the pre-change data only. The second section assesses the variability of the invertebrate populations at a number of scales within each of the rivers, with regard to distinguishing any potential changes in invertebrate community due to the change in compensation flow from that caused by natural variability. The third and final section examines whether there has been an impact on the invertebrate communities from altering the compensation flows. In each section, a combination of univariate and multivariate data analysis methods will be used.

3.4.1. Are there differences between the structure and quality of the macroinvertebrate populations with the pre-change compensation flows?

This section assesses the difference between the structure and quality of the macroinvertebrate populations with the pre-change compensation flows. This will establish a baseline from which any (potential) changes in invertebrate populations can be judged. It also addresses aim three directly. The first section will provide support for the removal of Chironomidae and Oligochaeta from the samples. The data analysis proper is reported first

with the presentation of the kick sample results. For the Surber samples the reporting begins with a description of the abundance and family density and then diversity indices and presence/absence of families, and the similarities/dissimilarities between the sites and sampling seasons. The important species at each of the sites are then highlighted, the proportions of the various feeding guilds discussed and finally the relationship between the biotic and abiotic data addressed using the BIOENV routine.

3.4.1.1. Kick samples

Removal of Chironomids and Oligochaeta

A first point of interest when attempting to characterise macroinvertebrate populations can be to look at which families are important to which site (Table 3.4.1). When the SIMPER routine was used to analyse the important species within each of the kick sampling sites on the Rivelin, Loxley and Hipper, it shows that Chironomidae and Oligochaeta are important to each of these rivers. These two species, despite their obvious importance to these systems, are very tolerant to stress. More subtle features of the invertebrate community, which may give answers to the questions of interest may be masked by arbitrary changes in these two families. The decision was therefore made to remove these families from the Surber sample analysis.

	RUSD	RM_EA	RU	RD	RHB
Cumulative similarity	53.19	51.02	51.81	51.01	51.18
Chironomidae	*	*	*	*	*
Oligochaeta	*	*	*	*	*
Elmidae (II*)	*	*	*	*	*
Seriscostomatidae (II)				*	*
Hydropsychidae (II)		*	*	*	*
Baetidae (II*)		*	*		*
Heptageniidae (I*)		*	*		*
Gammaridae (II)		*	*	*	
Rhyacophilidae (I)	*	*			
Leuctridae (II*)	*				
Nemouridae (IV*)	*				
Limnephilidae (IV*)				*	
Hydracarina			*		

	LUSD	LU	LRB_EA	LD	LMB	LA61
Cumulative similarity	54.25	56.80	52.25	55.99	55.23	52.48
Chironomidae	*		*	*	*	*
Oligochaeta	*	*	*	*	*	*
Baetidae (II*)	*	*	*			*
Elmidae (II*)	*					*
Empididae						*
Asellidae (IV)						*
Gammaridae (II)		*	*		*	
Rhycophilidae (I)			*		*	
Hydropsychidae (II)		*	*			
Leuctridae (II*)	*					
Nemouridae (IV*)	*					
Tipulidae (IV*)	*					
Polycentropodidae (IV*)		*				
Sericostomatidae (II)		*		*		
Sphaeriidae (IV*)				*		
Simuliidae (II)				*		

	H D/D BB	H	H@CHEST
Cumulative similarity	52.90	51.17	54.08
Oligochaeta	*	*	
Chironomidae	*	*	
Erpobdellidae (IV)	*		*
Asellidae (IV)			*
Baetidae (II*)	*	*	
Elmidae (II*)	*	*	
Sericostomatidae (II)	*	*	
Simuliidae (II)	*	*	
Tipulidae (IV*)	*	*	
Heptageniidae (I*)		*	

Table 3.4.1. Table to show the similarity output from the SIMPER routine for the invertebrate kick samples taken on each of the rivers. The families in the table contributed at least 60% of the cumulative similarity. LIFE flow groups for each of the families are given in brackets. Site codes as per Table 3.3.1 and Figure 3.3.1.

LIFE scores

The first analysis was conducted using the historical kick sample data, to provide an historical context for the more recent Surber sampling programme. An accepted method of judging the velocity preferences of an invertebrate community present in a river is to calculate the LIFE index (Extence *et al.*, 1999) of a given sample. This index was calculated for the kick samples. Table 3.4.2 shows that, generally, the Rivelin has a higher LIFE score than the Loxley. This is not what one would expect when examining the two

rivers as the Loxley has a higher discharge than the Rivelin, and it would be expected to have a higher velocity. This is the first indication that what may be important to the systems is not necessarily the base compensation flows themselves.

	LIFE		LIFE		LIFE
U/S Dams	7.82 (0.27)	U/S Damflask	7.15 (0.18)	H D/S BB	7.47 (0.14)
RM	7.76 (0.33)	LU	7.65 (0.45)	H	7.54 (0.08)
RU	7.48 (0.35)	LRB	7.54 (0.3)	H @ CHEST	6.86 (0.14)
RD	7.62 (0.24)	LD	7.30 (0.24)		
Hollins Bridge	7.55 (0.26)	Malinbridge	7.02 (0)		
		A61	7.40 (0.41)		

Table 3.4.2. Table to show the average LIFE scores at each kick sampling site on the Rivelin, Loxley and Hipper. Standard errors in brackets.

Drought

For one site on each river there was an increase in the frequency of the Environment Agency kick sampling programme in the years 1996-1998. For 9 months (4th of January 1996 to the 4th October 1996) during 1996, a drought order was in place both upon the lower Rivelin reservoir and Damflask reservoir (Table 3.4.3). Unfortunately there is no record of the reservoir levels at this time. This means that although the drought order was in place, there is no way of knowing when or if it was used. However, what can be assumed is that the discharge of the Rivelin and Loxley may have been reduced due to a decrease in frequency of overtopping events and a decrease in contributions from augmenting tributaries which flow into the rivers downstream of the reservoir.

Reservoir level	Reduction in Compensation flow
> 80%	0
>40% < 80%	½
< 40%	2/3

Table 3.4.3. Table to show the drought order placed on the lower Rivelin and Damflask reservoirs from January to October 1996.

When the Rivelin Mill data was considered, an ANOSIM analysis revealed a statistically significant difference (to a $p<0.01$) between the 1996, 1997 and 1998 data (as a group) and the remaining data (a separate group). Applying, the SIMPER routine to the Rivelin Mill

kick sampling site data (site situated just downstream of the reservoir (Figure 3.3.1)), shows that when comparing the families which contribute up to 50% of the cumulative similarity of the two groups of samples (Table 3.4.4), a number of taxa are common to both groups (these taxa are in **bold**). When the families which discriminate between the two sites are considered, there are no families from the similarity columns in the results. This may indicate that although the basis of the invertebrate population remains the same, peripheral changes to the population did occur. Table 3.4.4 also shows that most of the best discriminating families prefer slower flows.

Similarity within each of the groups		**Dissimilarity between the two groups**
Rivelin Mill 96	**Rivelin Mill POST 96**	**Rivelin Mill 96 Vs Rivelin Mill POST 96**
Cumulative Similarity = 51.84	*Cumulative Similarity = 54.65*	*Cumulative Dissimilarity = 51.98*
Hydrobiidae (IV*)	**Hydropsychidae (II)**	Hydrobiidae (IV*)
Elmidae (II*)	**Oligochaeta**	Leuctridae (II*)
Hydropsychidae (II)	**Elmidae (II*)**	Sphaeriidae (IV*)
Baetidae (II*)	**Chironomidae**	Planariidae (IV*)
Chironomidae	**Baetidae (II*)**	Limnephilidae (IV*)
Oligochaeta	**Heptageniidae (I*)**	Planorbiidae (IV)
Heptageniidae (I*)	**Gammaridae (II)**	Simuliidae (II)
Gammaridae (II)	Rhyacophilidae (I)	Hydraenidae (IV*)
	Leuctridae (II*)	Tipulidae (IV*)
		Polycentropodidae (IV*)
		Leptophlebiidae (II*)
		Empididae
		Leptoceridae (IV*)
		Glossiphoniide (IV)
		Taeniopterygidae (II*)
		Chloroperlidae (I)

Table 3.4.4. The SIMPER output investigating the drought year at the Rivelin Mill kick sampling site, showing the species which contribute up to the 60% cumulative similarity for each of the groups of samples, and the species which contribute up to 60% of the cumulative dissimilarity between the two groups. Family LIFE flow groups are given in brackets.

At the Loxley site (Rowell Lane), a statistically significant difference (to a $p<0.01$) was also found between the samples collected in the period 1996, 1997, 1998 and those collected outside that period. When the species composition of the drought and non drought samples at Rowell Lane are compared (Table 3.4.5) it shows that Chironomidae, Oligochaeta, and Baetidae are common to drought and non-drought samples. The non-drought species are characterised by species with a preference for higher flow velocities

than in the drought conditions. This indicates that during the droughts the flow velocities in the rivers are slower and hence improving the habitat for slow-flow loving invertebrates.

Similarity within each of the groups		**Dissimilarity between the two groups**
Rowell Bridge NOT 96	**Rowell Bridge 96**	**Rowell Bridge NOT 96 Vs Rowell Bridge 96**
Cumulative similarity = 50.65	*Cumulative Similarity = 53.70*	*Cumulative Dissimilarity = 50.72*
Chironomidae	**Chironomidae**	Hydrobiidae (IV*)
Oligochaeta	Hydrobiidae (IV*)	Hydracarina
Baetidae (II*)	**Oligochaeta**	Leuctridae (II*)
Rhyacophilidae (I)	**Baetidae** (II*)	Elmidae (II*)
Gammaridae (II)	Empididae	Leptoceridae (IV*)
	Hydropsychidae (II)	Polycentropodidae (IV*)
	Polycentropodidae (IV)	Limnephilidae (IV*)
		Rhyacophilidae (I)
		Gammaridae (II)
		Chironomidae
		Nemouridae (IV*)
		Lymnaeidae (IV*)
		Glossiphoniidae (IV)

Table 3.4.5. Table of the SIMPER output comparing the samples taken in 1996 and those taken outside of that period for the Loxley at Rowell Bridge. LIFE flow group of families are given in brackets.

No statistical difference was found between the two periods in the sites upstream of the dam in either of the rivers, so the drier summers plus the possible change in compensation flows may have had an impact upon the invertebrate population downstream of the reservoirs.

The above examination of the drought samples suggested that the core species seemed to remain the same, with the differences in species communities driven by changes in peripheral families. The families which were best discriminators between the drought and non-drought samples are characterised by invertebrate families with a preference for slow flow velocities. Table 3.4.6 shows that the drought causes changes in the abundance of the different LIFE classes, in each of the rivers. On both rivers during the drought period, the average abundance of the fast flow flow-loving families decreased, whilst the average abundance of slow-flow loving families increased. This suggests that changes to the discharges in the Rivelin and Loxley could alter the invertebrate communities within these rivers.

	AVERAGE – non augmented flow	AVERAGE –augmented flow
RM-EA		
I (>100cm/s)	1.00	1.50
II (20-100cm/s)	1.10	1.50
III (<20cm/s)	N/A	N/A
IV (slow and standing water)	1.35	1.13
LRB-EA		
I (>100cm/s)	1.20	1.85
II (20-100cm/s)	0.67	1.29
III (<20cm/s)	N/A	N/A
IV (slow and standing water)	1.47	0.94

Table 3.4.6. Table to show the average abundances (A=1, B=2) of the best discriminating species (using the SIMPER routine) between the non-augmented and augmented flow samples.

3.4.1.2. Environmental variables

Even in a compensation flow system, detecting compensation flow impacts is complicated by superposition of natural variability on the compensation flows over two very different scales of variability: (i) reservoir overtopping on a weekly to monthly scale leading to low frequency variability with higher 'baseflows'; and (ii) storm event scale, non impounded tributaries with high frequency flow variability, possibly augmented if the reservoir is overtopping.

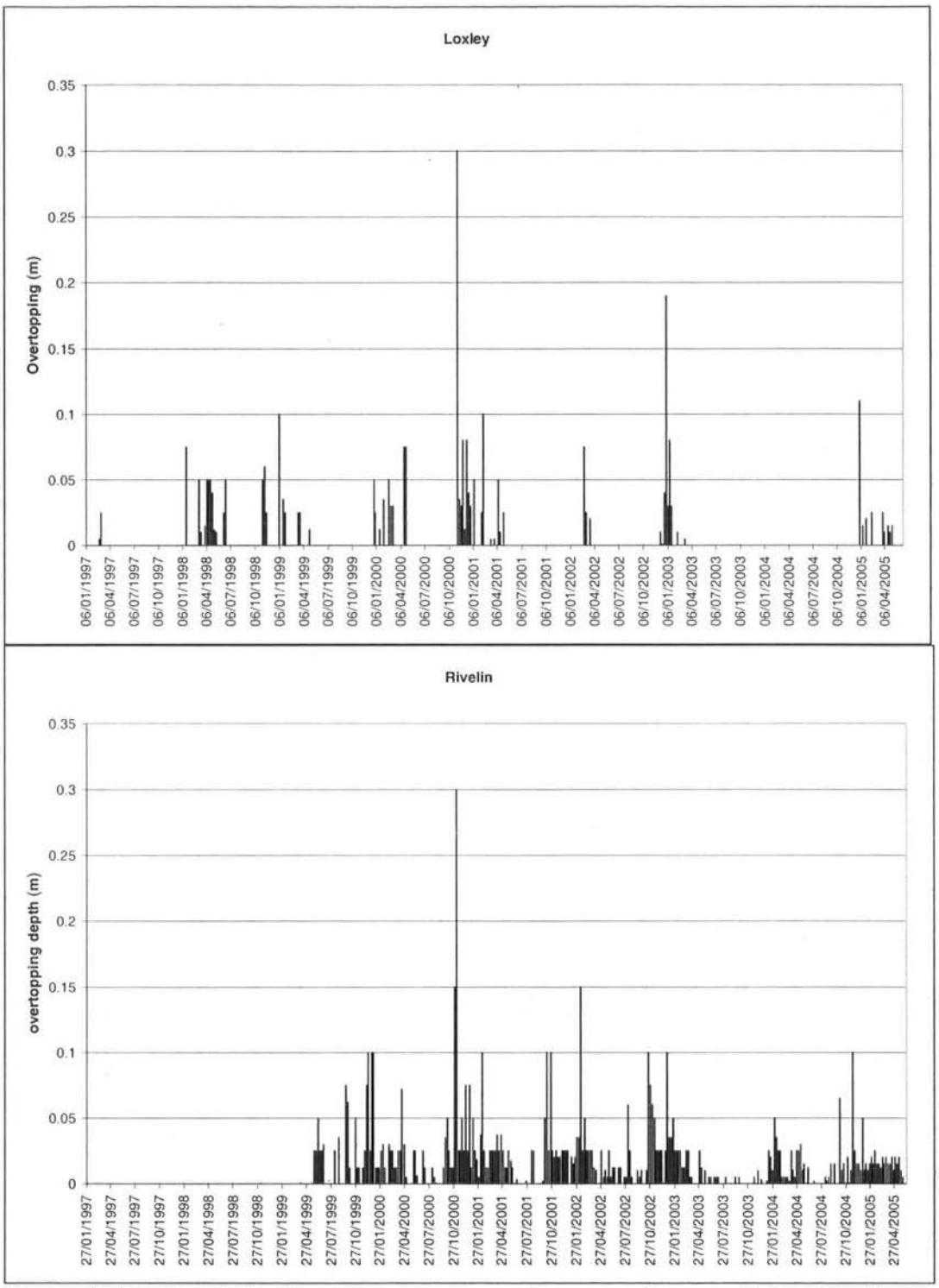

Figure 3.4.1. Graph to show the occurrence of overtopping events into the Loxley and Rivelin from January 1997 to May 2005.

Figure 3.4.1 shows that there are more overtopping events in the Rivelin than in the Loxley. As a consequence of the low rainfalls seen in 2003, there were no overtopping events from Damflask reservoir (the Loxley) and there were fewer overtopping events from the Rivelin reservoir. This is a period of time that is less affected by natural variability than other times during the study period, and as such is the ideal time to try and detect any impacts of the compensation flow regimes in the Surber samples. Table 2.6 shows that for both the Rivelin and Loxley, the spring 2003 samples were slightly augmented, the summer autumn 2003 samples were not augmented, whilst the remaining samples did appear to be subject to some form of flow augmentation.

The environmental variables can also be analysed to compare the sampling years at a given site. Table 3.4.7 shows that there is more difference between the environmental variables recorded on the Rivelin and Hipper at various seasons when compared to the Loxley. It may be expected that differences in the environmental variables would occur in the Hipper (as it is unregulated, and hence subject to greater variations in discharge), but what is interesting is the abundance of differences in the Rivelin and the lack of differences in the Loxley. This could mean that in terms of compensation flow releases, the Loxley's higher compensation release may be smoothing out natural variability in the environmental data. It was evident that the three differences found at LD involved comparing augmented and non-augmented samples.

However, in the Loxley, there are differences between the sampling seasons in the LU and LD biotic data (Table 3.4.18). The fact that the invertebrate populations within the Loxley can be statistically different despite there being no difference between the environmental variables suggests that the invertebrate community may operate largely independently of the environmental variables in the Loxley, and more so than in the Rivelin. The lack of differences in the Loxley suggests that the Loxley is less flow limited, as compared with the augmented rivers.

RU	Sp.02	Aut.02	Sp.03	Su. 03	Aut.03
Sp.02		0.20	0.29	0.07	0.14
Aut.02			0.37	0.39	0.27
Sp.03				0.11	0.20
Su.03					0.10
Aut.03					
RD	Sp.02	Aut.02	Sp.03	Su. 03	Aut.03
Sp.02		0.07	0.26	0.72	0.59
Aut.02			0.23	0.62	0.52
Sp.03				0.18	0.16
Su.03					-0.04
Aut.03					
LU	Sp.02	Aut.02	Sp.03	Su. 03	Aut.03
Sp.02		-0.1	-0.05	-0.07	-0.02
Aut.02			-0.10	-0.08	-0.06
Sp.03				-0.05	-0.03
Su.03					-0.09
Aut.03					
LD	Sp.02	Aut.02	Sp.03	Su. 03	Aut.03
Sp.02		0.05	-0.05	0.06	0.17
Aut.02			0.06	0.31	0.45
Sp.03				-0.05	0.02
Su.03					-0.07
Aut.03					
H	Sp.02	Aut.02	Sp.03	Su. 03	Aut.03
Sp.02		0.57	0.02	0.17	0.16
Aut.02			0.37	0.75	0.80
Sp.03				0.15	0.14
Su.03					0.03
Aut.03					

Table 3.4.7. Table to show the R values following an ANOSIM analysis of the measured environmental variables comparing sampling seasons at a site. Grey colouration indicates a statistically significant difference to the 95% significance level.

SP.02	RU	RD	LU	LD	H
RU		0.34	0.21	0.31	0.15
RD			0.01	0.26	0.24
LU				0.28	0.24
LD					0.60
H					

Aut.02	RU	RD	LU	LD	H
RU		0.14	0.20	0.38	0.36
RD			-0.04	0.15	0.4
LU				0.20	0.21
LD					0.53
H					

SP.03	RU	RD	LU	LD	H
RU		0.13	0.46	0.61	0.46
RD			0.29	0.37	0.23
LU				0.29	0.34
LD					0.45
H					

Su. 03	RU	RD	LU	LD	H
RU		0.13	0.81	0.65	0.61
RD			0.53	0.54	0.31
LU				0.64	0.61
LD					0.62
H					

Aut03	RU	RD	LU	LD	H
RU		0.25	0.68	0.55	0.30
RD			0.30	0.31	0.03
LU				0.43	0.54
LD					0.48
H					

Table 3.4.8. Table to show the ANOSIM output comparing the measured environmental variables of the sampling sites for each sampling season. Grey colouration indicates a statistically significant difference to the 95% significance level.

3.4.1.3. Surber Samples

Invertebrate density and family numbers

A first step in many invertebrate studies is to examine the abundance of invertebrates found at each of the sites. Figure 3.4.2 shows that the highest densities were found at LU. For each site, there are large error bars, and so distinguishing between sites and seasons is difficult. When the sampling seasons are compared at each of the sites, Table 3.4.9 shows that there was very little between season variation at the Rivelin sites, but a greater amount of variation at LD, with the differences in densities involving spring and summer 2003. Table 3.4.10 shows that generally there is a difference between the sites in a given season.

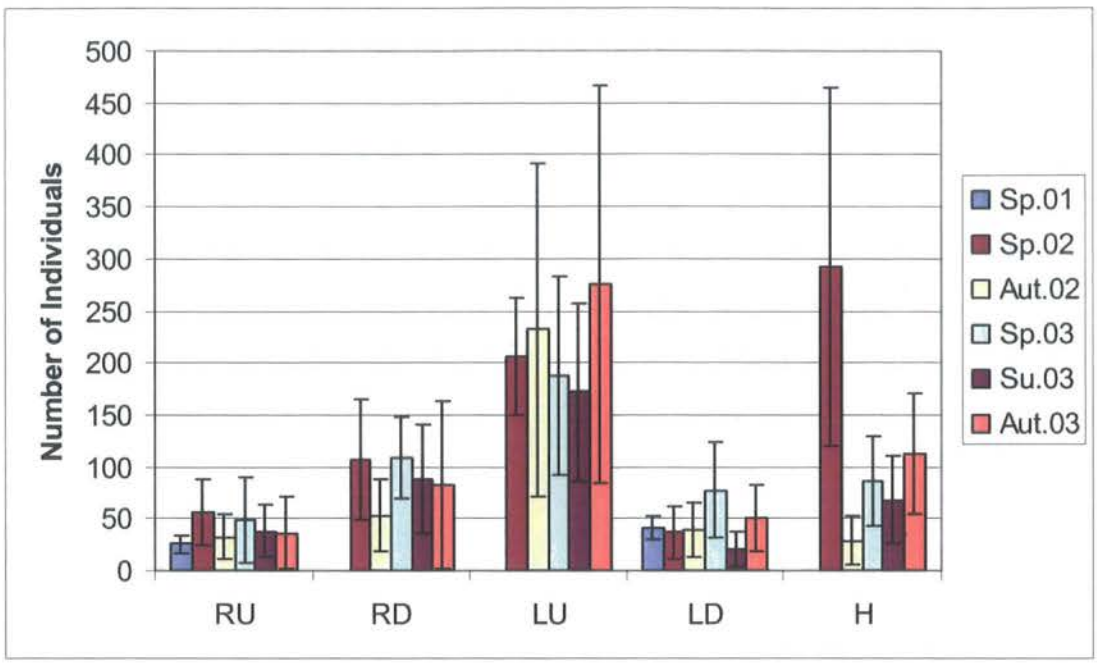

Figure 3.4.2. Figure to show the average number of individuals found at each of the sites in the pre-change samples. Error bars to show the standard deviation.

RU	Sp.01	Sp02	Aut02	Sp03	Su03	Aut03
SP.01		8.70	0.24	0.51	2.24	0.84
Sp.02			4.85	2.77	2.17	2.02
Aut.02				0.09	0.66	0.25
Sp.03					0.15	0.03
Su.03						0.03
Aut.03						

RD	Sp.01	Sp02	Aut02	Sp03	Su03	Aut03
Sp.01						
Sp.02			5.08	2.06	0.01	0.02
Aut.02				11.35	3.02	1.10
Sp.03					1.03	0.90
Su.03						0.04
Aut.03						

LU	Sp.01	Sp02	Aut02	Sp03	Su03	Aut03
Sp.01						
Sp.02			0.22	0.29	1.13	1.21
Aut.02				0.56	1.08	0.31
Sp.03					0.15	1.70
Su.03						2.45
Aut.03						

LD	Sp.01	Sp02	Aut02	Sp03	Su03	Aut03
Sp.01		0.24	0.03	6.31	9.52	0.92
Sp.02			0.06	6.32	2.61	1.26
Aut.02				5.34	3.51	0.80
Sp.03					13.76	2.37
Su.03						6.93
Aut.03						

H	Sp.01	Sp02	Aut02	Sp03	Su03	Aut03
Sp.01						
Sp.02			22.81	13.36	15.79	9.73
Aut.02				13.91	6.66	18.16
Sp.03					0.87	1.32
Su.03						3.76
Aut.03						

Table 3.4.9. Table to show the results of an ANOVA (F statistic) analysis comparing the sampling seasons at each of the sites using the number of individuals sampled. Grey colouration indicates a statistically significant difference to the 95% significance level.

Sp.01	RU	RD	LU	LD	H
RU				12.51	
RD					
LU					
LD					
H					

Sp.02	RU	RD	LU	LD	H
RU		4.03	52.32	2.50	17.75
RD			35.43	15.65	13.72
LU				76.79	2.21
LD					21.40
H					

Aut.02	RU	RD	LU	LD	H
RU		3.30	15.74	0.87	0.00
RD			11.92	1.02	3.44
LU				14.12	15.81
LD					0.95
H					

Sp.03	RU	RD	LU	LD	H
RU		22.38	23.41	6.46	9.75
RD			5.81	2.72	1.54
LU				10.78	9.36
LD					0.18
H					

Su.03	RU	RD	LU	LD	H
RU		7.31	22.65	3.08	3.73
RD			7.06	14.62	0.83
LU				30.05	11.81
LD					10.56
H					

Aut.03	RU	RD	LU	LD	H
RU		2.82	15.29	1.03	12.91
RD			8.74	1.32	0.93
LU				13.50	6.72
LD					8.74
H					

Table 3.4.10. Table to show the ANOVA (F statistic) output comparing the sampling sites at a given season, using the abundance of invertebrates. Grey colouration indicates a statistically significant difference to the 95% significance level.

When the number of families collected in each Surber sample are considered, Figure 3.4.3 shows that the sites are generally similar. Spring 2003 appears to have a higher number of families than many of the other sampling seasons (especially in LU and LD). Table 3.4.11 shows that when the sampling seasons are compared statistically, there were no differences found between the sampling seasons at either of the Rivelin sites, with some differences being found at the Loxley and Hipper sites. Table 3.4.12 shows that there are some statistically significant differences found between the sites, mostly involving LD and H.

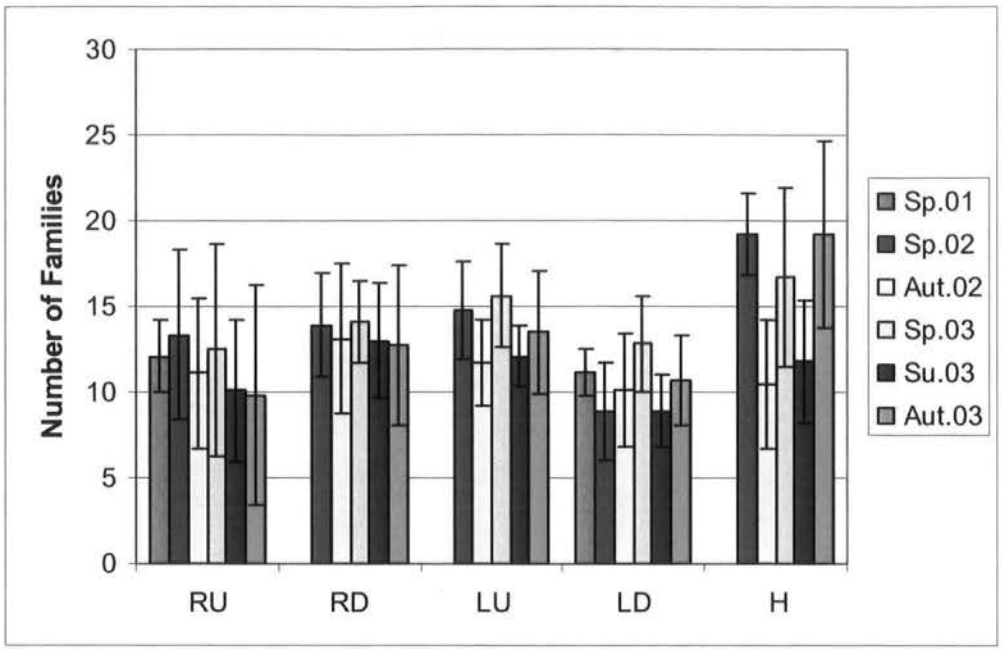

Figure 3.4.3. Figure to show the average number of families captured per sample at each site in each sampling season. Error bars to show the standard deviation.

RU	Sp.01	Sp02	Aut02	Sp03	Su03	Aut03
SP.01		0.59	1.30	1.02	1.84	1.15
Sp.02			2.11	1.82	2.52	1.95
Aut.02				0	0	0.01
Sp.03					0	0
Su.03						0.02
Aut.03						

RD	Sp.01	Sp02	Aut02	Sp03	Su03	Aut03
Sp.01						
Sp.02			0.05	0.23	0.12	0.2
Aut.02				0.4	0	0.04
Sp.03					0.71	0.71
Su.03						0.03
Aut.03						

LU	Sp.01	Sp02	Aut02	Sp03	Su03	Aut03
Sp.01						
Sp.02			6.77	0.37	6.40	0.81
Aut.02				10.18	0.17	1.73
Sp.03					10.09	2.04
Su.03						1.23
Aut.03						

LD	Sp.01	Sp02	Aut02	Sp03	Su03	Aut03
Sp.01		4.85	0.79	3.01	7.81	0.19
Sp.02			0.76	9.61	0.00	2.19
Aut.02				3.94	0.95	0.21
Sp.03					12.62	3.06
Su.03						2.95
Aut.03						

H	Sp.01	Sp02	Aut02	Sp03	Su03	Aut03
Sp.01						
Sp.02			38.25	1.86	29.48	0
Aut.02				9.19	0.63	17.20
Sp.03					5.91	1.08
Su.03						12.79
Aut.03						

Table 3.4.11. Table to show the ANOVA (F statistic) output from comparing the sampling seasons using number of families captured per sample at each site. Grey colouration indicates a statistically significant difference to the 95% significance level.

Sp.01	RU	RD	LU	LD	H
RU				1.56	
RD					
LU					
LD					
H					

Sp.02	RU	RD	LU	LD	H
RU		0	0.46	5.75	9.43
RD			0.93	11.65	20.58
LU				21.38	13.92
LD					76.69
H					

Aut.02	RU	RD	LU	LD	H
RU		1.98	0.82	0	0.06
RD			0.78	3	2.03
LU				1.53	0.72
LD					0.06
H					

Sp.03	RU	RD	LU	LD	H
RU		3.77	6.59	1.69	6.75
RD			1.52	1.24	2.01
LU				4.70	0.33
LD					4.28
H					

Su.03	RU	RD	LU	LD	H
RU		2.97	1.96	0.67	0.96
RD			0.57	10.89	0.6
LU				13.59	0.06
LD					4.90
H					

Aut.03	RU	RD	LU	LD	H
RU		1.33	2.52	0.17	12.36
RD			0.19	1.41	8.17
LU				4.04	7.61
LD					19.72
H					

Table 3.4.12. Table to show the ANOVA (F statistic) output comparing each of the sites at each sampling season using the number of families captured per sample. Grey colouration indicates a statistically significant difference to the 95% significance level.

Diversity Indices and Family Presence/Absence

The Shannon diversity index was calculated for each of the Surber samples. It could not be calculated for the kick samples because of the format of the data provided by the Environment Agency. As a rough guide to the meaning of the index, Wihlm and Dorris (1968) state that a Shannon value of greater than three is characteristic of an unpolluted water body, values in the range of 1 to 3 were characteristic of a mildly polluted water body and values of less than one of heavily polluted conditions. However, in a study such as this, a better definition may be stressed as we are not interested in pollution per se.

Figure 3.4.4 shows that almost all of the samples collected on the Rivelin, Loxley and Hipper can be considered indicative of mildly polluted conditions. This is hardly surprising, since the Loxley has been subjected to heavy industry along much of its length until 20 or so years ago, with the upstream Surber sampling site (LU) being located above the industry and the downstream site situated below some of the industry. However, although the Rivelin was a site for steel working mills throughout the 1800's and early 1900's the industrial usage of this river ended before that of the Loxley. In this case, the

downstream Loxley signal may be reflecting the legacy of heavy industrialisation, but the Rivelin is reflecting other forms of stress, possibly related to flow.

Aside from characterising the macroinvertebrate populations of the study rivers as mildly polluted, few clear patterns are shown in Figure 3.4.4. No-one river has a clearly higher diversity, and the error bars make it generally difficult to distinguish between both sites and seasons. The ANOVA results presented in Tables 3.4.13 and 3.4.14 show that most between season differences were found at LU and H; with most inter-site differences involving the Hipper. Little consistent pattern is seen, which, indicates a range of variability at a number of scales which will be addressed.

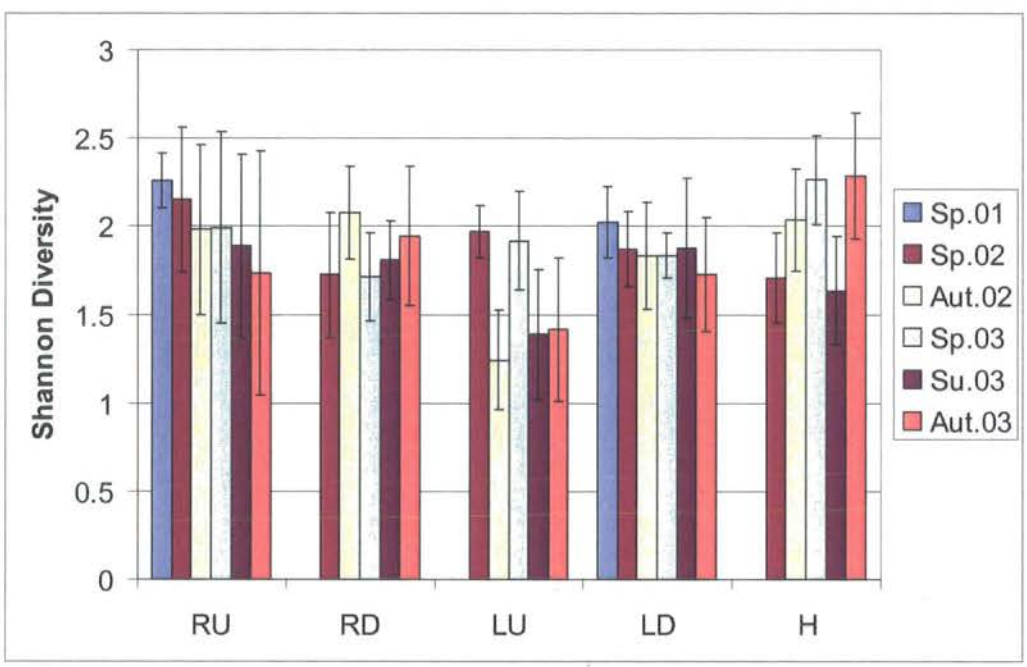

Figure 3.4.4. Graph to show the Shannon diversity of the Surber samples for each site in each pre-change sampling season. Error bars showing the standard deviation of the diversities of each set of Surber samples.

RU	Sp.01	Sp02	Aut02	Sp03	Su03	Aut03
SP.01		0.26	3.63	4.47	4.67	5.39
Sp.02			2.0	1.68	1.87	2.93
Aut.02				0.09	0.12	0.02
Sp.03					0.32	0
Su.03						0.3
Aut.03						

RD	Sp.01	Sp02	Aut02	Sp03	Su03	Aut03
Sp.01						
Sp.02			6.89	0	0.53	1.94
Aut.02				9.82	6.04	0.76
Sp.03					0.74	2.38
Su.03						0.91
Aut.03						

LU	Sp.01	Sp02	Aut02	Sp03	Su03	Aut03
Sp.01						
Sp.02			51.60	0.25	21.26	16.34
Aut.02				28.69	0.95	1.23
Sp.03					13.07	10.34
Su.03						0.03
Aut.03						

LD	Sp.01	Sp02	Aut02	Sp03	Su03	Aut03
Sp.01		2.8	2.69	6.36	1.11	5.98
Sp.02			0.09	0.21	0	1.31
Aut.02				0	0.07	0.55
Sp.03					0.1	0.89
Su.03						0.8
Aut.03						

H	Sp.01	Sp02	Aut02	Sp03	Su03	Aut03
Sp.01						
Sp.02			7.19	23.55	0.33	17.14
Aut.02				3.5	8.97	2.96
Sp.03					24.58	0.03
Su.03						18.87
Aut.03						

Table 3.4.13. Table to show the output from ANOVA (F statistic) pre-change comparisons of the Shannon diversity for each sampling season at each sampling site. Grey colouration indicates a statistically significant difference to the 95% significance level.

Sp.01	RU	RD	LU	LD	H
RU				8.46	
RD					
LU					
LD					
H					

Sp.02	RU	RD	LU	LD	H
RU		6.92	2.07	4.05	8.52
RD			4.55	1.47	0
LU				1.48	7.63
LD					2.3
H					

Aut.02	RU	RD	LU	LD	H
RU		1.28	4.28	0.04	0.94
RD			46.64	3.59	0.1
LU				20.17	38.66
LD					2.23
H					

Sp.03	RU	RD	LU	LD	H
RU		0	0.58	0.21	4.3
RD			2.89	1.72	23.62
LU				0.76	8.28
LD					22.81
H					

Su.03	RU	RD	LU	LD	H
RU		0.19	6.09	0	1.7
RD			9.47	0.22	2.02
LU				8.05	2.66
LD					2.24
H					

Aut.03	RU	RD	LU	LD	H
RU		0.68	1.59	0	4.96
RD			8.7	1.78	4.12
LU				3.64	25.85
LD					13.33
H					

Table 3.4.14. Table to show the ANOVA (F statistic) output comparing the Shannon diversity of each sampling site for each sampling season. Grey colouration indicates a statistically significant difference to the 95% significance level.

The family list for the Rivelin and Loxley was examined, and the families which were present in one of the rivers but absent from the other are listed in Table 3.4.15. Table 3.4.15 shows that there were more invertebrate families present in the Rivelin than the Loxley, perhaps indicating a more diverse population. Table 3.4.15 also shows that more 'extra' families in the Rivelin are members of the Ephemeroptera, Plecoptera and Trichoptera (EPT) groups, which are considered pollution intolerant invertebrates. This indicates a 'better' macroinvertebrate population in the Rivelin compared to the Loxley. There could be two reasons for this:

a) the industrial history of the Loxley impacting upon the invertebrates; and
b) the flow regime in the Rivelin is preferable for macroinvertebrates than that of the Loxley.

Table 3.4.16 shows that when the species lists of the two Loxley sites are compared, it appears that a) may not be the correct answer as if the populations at LD were pollution controlled, there could have been more taxa found at LU, relatively unaltered by pollution compared to LD, and the taxa found at LD may have been more pollution tolerant. As it is, there are families which are present or otherwise, but no clear inference can be drawn.

IN LOXLEY NOT RIVELIN	IN RIVELIN NOT LOXLEY
Lepidosomatidae (II)	Ephemeridae (II*)
Muscidae	Taeniopterygidae (II*)
Fanniidae	Odontoceridae (I)
Syrphidae (V)	Beraeidae (II)
Staphylinidae	Goeridae (I)
Asellidae (IV)	Curculionidae
Daphnidae	Scirtidae (IV*)
Collembola	Carabidae
Oribatei	Erpobdellidae (IV)
	Physidae
	Chrysopidae
	Hydridae
	Pyralidae

Table 3.4.15. Table to show the families which are present in one of the rivers but not the other. LIFE flow groups in brackets where appropriate.

IN LU not LD	IN LD not LU
Muscidae	Perlodidae (I)
Hydraenidae (IV*)	Glossosomatidae (II*)
Hydrophilidae (IV*)	Psychodidae
Staphylinidae	Faniidae
Collembola	Stratiomyiidae
Oribatei	Syrphidae
	Daphniidae

Table 3.4.16. Table to show the families present in one of the Loxley sites but not the other. LIFE flow groups in brackets where appropriate.

Comparison of species similarity within and between sites

This section compares the invertebrate populations in the rivers to see whether differences exist between the Rivelin, Loxley and Hipper, using the replication of the Surber samples as a basis for statistical analysis. For this purpose, the upstream and downstream sites for each of the Rivelin and Loxley are combined so as to increase the effective numbers of replicates. Table 3.4.17 shows that (using the multivariate ANOSIM test) that the three rivers are statistically different in terms of their invertebrate population, apart from the Rivelin and Hipper in the spring 2002 sampling season. An interesting point to observe here is that the Rivelin appears to be more similar to the Hipper than the Loxley, as evidenced by the lower r values (as the higher the r value, the greater the difference).

	HISTORICAL	SPRING 2001	SPRING 2002	AUTUMN 2002	SPRING 2003	SUMMER 2003	AUTUMN 2003
Rivelin Loxley	0.27	0.58	0.23	0.45	0.42	0.43	0.34
Rivelin Hipper	0.40		0.15	0.37	0.20	0.27	0.19
Loxley Hipper	0.27		0.62	0.78	0.60	0.51	0.51

Table 3.4.17. Table to show the R values from the ANOSIM inter-River analysis using the biotic data (grey colouring indicates a statistically significant difference to the 95% level).

Table 3.4.18 shows that for all sites, the invertebrate community sampled in each season was different to every other season apart from autumn 2002 and autumn 2003 at the LU sampling site. The ANOSIM routine also revealed a statistically significant difference between each sampling site in every sampling season (Table 3.4.19). This indicates a large amount of difference between the sites, and implies that they must be treated separately in all subsequent analyses. The fact that the sites and sampling seasons are all found to be statistically significantly different with the multivariate analyses, shows how poor the Shannon index is for use in this study.

Sp.01	RU	RD	LU	LD	H
RU				0.58	
RD					
LU					
LD					
H					

Sp.02	RU	RD	LU	LD	H
RU		0.21	0.40	0.28	0.43
RD			0.61	0.57	0.49
LU				0.69	0.98
LD					0.90
H					

Aut.02	RU	RD	LU	LD	H
RU		0.25	0.69	0.55	0.58
RD			0.80	0.42	0.50
LU				0.54	0.96
LD					0.87
H					

Sp.03	RU	RD	LU	LD	H
RU		0.29	0.55	0.56	0.32
RD			0.82	0.76	0.53
LU				0.72	0.71
LD					0.77
H					

Su.03	RU	RD	LU	LD	H
RU		0.59	0.89	0.67	0.71
RD			0.87	0.66	0.3
LU				0.78	0.90
LD					0.80
H					

Aut.03	RU	RD	LU	LD	H
RU		0.30	0.53	0.39	0.51
RD			0.78	0.40	0.47
LU				0.56	0.89
LD					0.61
H					

Table 3.4.18. Table to show the ANSOIM output comparing each of the sites for each of the sampling seasons, using the invertebrate community from the Surber samples. Grey colouration indicates a statistically significant difference to the 95% significance level.

RU	Sp.01	Sp.02	Aut.02	Sp.03	Su. 03	Aut.03
Sp.01		0.36	0.27	0.51	0.58	0.27
Sp.02			0.32	0.15	0.44	0.37
Aut.02				0.44	0.30	0.12
Sp.03					0.36	0.35
Su.03						0.26
Aut.03						
RD	Sp.01	Sp.02	Aut.02	Sp.03	Su. 03	Aut.03
Sp.01						
Sp.02			0.61	0.18	0.64	0.67
Aut.02				0.80	0.49	0.26
Sp.03					0.57	0.79
Su.03						0.45
Aut.03						
LU	Sp.01	Sp.02	Aut.02	Sp.03	Su. 03	Aut.03
Sp.01						
Sp.02			0.74	0.19	0.83	0.79
Aut.02				0.49	0.66	0.16
Sp.03					0.62	0.51
Su.03						0.62
Aut.03						
LD	Sp.01	Sp.02	Aut.02	Sp.03	Su. 03	Aut.03
Sp.01		0.66	0.53	0.87	0.85	0.75
Sp.02			0.59	0.37	0.78	0.72
Aut.02				0.73	0.54	0.05
Sp.03					0.76	0.78
Su.03						0.35
Aut.03						
H	Sp.01	Sp.02	Aut.02	Sp.03	Su. 03	Aut.03
Sp.01						
Sp.02			0.86	0.31	0.7	0.75
Aut.02				0.75	0.71	0.53
Sp.03					0.49	0.46
Su.03						0.64
Aut.03						

Table 3.4.19. Table to show the output from an ANOSIM analysis comparing each of the sampling seasons for each of the Surber sampling sites using the biotic data. Grey colouration indicates a statistically significant difference to the 95% significance level.

Analysis of important species

As the sites appear to be statistically significantly different, this section examines which species drive these differences using the SIMPER routine. Table 3.4.21 shows the important species for the RU site (these species contribute up to 60% of cumulative similarity as defined by the SIMPER routine). It shows that there are no families common

to all three of the spring samples, but it shows that Leuctridae and Leptoceridae are common to two out of the three spring sets of samples. When the spring 2001 samples are discounted for the benefit of comparison, *Seretella ignita* and Elmidae are important to both the spring 2002 and spring 2003 sampling seasons. It is interesting that these are generally fast-flow loving species. There are also no families common to both the autumn samples. A further indication of a very variable invertebrate population is given by the LIFE scores of each of the species, which also show no clear seasonal pattern at this site. However, both Leptoceridae and Elmidae are important for four out of the six sampling seasons, but with no clear seasonal pattern, indicating inter-season variation in the importance of these two families. The Elmidae populations at this site are dominated by *Limnius volckmari* and *Elmis aenea*, which have a preference for faster flow conditions. The two main Leptocerid populations were *Athripsodes bilineatus* and *Mystacides azurea,* which have very different flow velocity preferences. A further indication of the within-year variation present at the RU site, is that the most important species in spring 2001 and autumn 2002 were *S. torrentium* and *Gammarus pulex* respectively, and both of these families were not found to be important for any of the other sampling seasons. Also five species are found to be important for various seasons, and not found to be important for any other sampling season.

	Sp.01	Sp.02	Aut.02	Sp.03	Su.03	Aut.03
Chloroperlidae						
Siphonoperla torrentium (I)	*					
Hydropsychidae						
H. siltalai (II)	*					*
H. instabilis (II)	*					
Leuctridae						
L. inermis (I)	*	*				
L. hippopus/moselyii (I)		*				
L. geniculata (II)		*				
L. fusca (II)		*				
Heptageniidae						
Rithrogena spp. (I)	*					
R.semicolorata (I)			*			
Ecdyonurus torrentis (I)			*			
Seriscostomatidae						
S. personatum (II)	*					*
Leptoceridae						
Leptoceridae sp. (IV)	*					
Athripsodes spp. (II*)			*		*	
Mystacides spp. (IV)			*			
Athripsodes bilineatus (II)				*		
Mystacides azurea (IV)				*	*	
Leptophlebiidae						
P. submarginata (II)	*					*
Ephemerellidae		*				
S. ignita (II)				*		
Elmidae						
L. volckmari (II)		*	*	*	*	
E. aenea (II)		*	*	*	*	
Oulimnius sp. (IV)		*	*	*	*	
Gammaridae						
G. pulex (II)		*	*			
Polycentropodidae						
P. flavomaculatus (II)		*	*			
P.kingi (II)		*				
P. conspersa (II)		*				
Crynus trimaculatus (IV)			*			
Empididae		*				
Ephemeridae						
E. danica (II)			*		*	
Hydracarina				*		
Caenidae						
C. horaria (IV)						*
Limnephilidae						
P. latipennis (II)						*

Table 3.4.21. Table to show the SIMPER within sample similarity output for each season at RU, with the listed species contributing at least 60% of the cumulative similarity. (LIFE flow groups given in brackets).

At the RD site (Table 3.4.22), the invertebrate community is more stable than that seen at the RU site. Elmidae (predomiantly *Elmis aenea* and *Limnius volckmari*) is found to be an important family for the RD site in each of the sampling seasons. It can also be seen that three families are common to each of the spring sampling seasons (Ephemerellidae, Leuctridae and Elmidae). This is interesting, as Ephemerellidae and Elmidae were found to be important for both the spring 2002 and spring 2003 for the RU site. The same species of Leuctridae were found in each of the spring sampling seasons. *S. ignita* was only important in the two spring samples, and Leuctridae was also only important in the two spring samples and summer 2003. This indicates that these fast-flow loving species were not as important to the macroinvertebrate communities in the autumn as they were earlier in the year. Table 3.4.22 shows that Elmidae and *S. personatum* are important for both of the autumn sampling seasons, with *S. personatum* being important for four out of the five sampling seasons. Hydracarina is important for each of the 2003 sampling seasons (Table 3.4.22), without being important for either of the 2002 sampling seasons. The more stable compensation dominated flows present in 2003 compared to 2002 (discussed in some detail later) may well be a factor in this emergence. When the LIFE scores of the important species at RD are considered, there is no clear pattern, with no distinct difference between the seasons and sampling years. However in general, the important species are faster flow loving families than the RU site. The invertebrate population at RD is dominated by a few important species, with some of these species being season dependent, others not, and the important species at RD are seemingly more 'stable' than those at RU.

	Sp.02	Aut.02	Sp.03	Su.03	Aut.03
Ephemerellidae					
S. ignita (II)	*		*		
Leuctridae					
L. hippopus (I)	*		*	*	
L. geniculata (II)	*		*	*	
L. fusca (II)	*		*	*	
Sericostomatidae					
S. personatum (II)	*	*		*	*
Elmidae					
E. aenea (II)	*	*	*	*	*
L. volckmari (II)	*	*	*	*	*
Oulimnius spp. (IV)					*
Rhyacophilidae					
R. dorsalis (I)	*				
R. obliterata (I)	*				
Heptageniidae					
Ecdyonurus spp. (I)		*			
R. semicolorata (I)		*			
Polycentropodidae					
P. flavomaculatus (II)		*			
P. kingi (II)		*			
Hydropsychidae					
H. siltalai (II)		*			
H. pellucidula (II)		*			
Hydracarina			*	*	*
Limnephilidae					
P. latipennis (II)					*
Ephemeridae					
E. danica (II)					*

Table 3.4.22. Table to show the SIMPER within sample similarity output for each season at RD, with the listed species contributing at least 60% of the cumulative similarity. (LIFE flow groups given in brackets).

Table 3.4.23 shows that for the LU site, Gammaridae (*Gammarus pulex*) is important for each of the sampling seasons at this site, and Hydropsychidae is important for four out of five sampling seasons. The important species for both spring samples are exactly the same. What is also of interest when the spring samples are examined is that Ephemerellidae and Baetidae are important only in the spring and are not important in any other sampling season. This matches with the pattern of *S. ignita* importance found in both the Rivelin sites, and again Leuctridae was also only found to be important in the spring and summer samples (with *L. geniculata* and *L.fusca* being the only Leuctridae species present). The three important families for the autumn 2002 samples are also important to the autumn 2003 samples, which again indicates a very stable group of important families for the LU

site. Two new species of Hydropsychidae (*H. angustipennis* and *H. instabilis*) were found in the autumn 2003 samples which were not found in the autumn 2002 samples. Seriscostomatidae is only found to be important in the autumn samples. A further indication of the stable nature of the important species found at LU is that only one family is found to be important in only one season (Sphaeriidae in summer 2003). In summer 2003 is there is a presence of slow flow preferring families (Sphaeriidae sp.), perhaps indicating greater habitat available for these invertebrates. Again this may be related to the lower flows present in 2003 compared to 2002 (see Chapter Two).

	Sp.02	Aut.02	Sp.03	Su.03	Aut.03
Ephemerellidae					
S. ignita (II)	*		*		
Gammaridae					
G. pulex (II)	*	*	*	*	*
Leuctridae					
L. geniculata (II)	*		*	*	
L. fusca (II)	*			*	
Hydropsychidae					
H. siltalai (II)	*	*	*		*
H. pellucidula (II)		*			*
H. angustipennis (II)					*
H. instabilis (II)					*
Baetidae					
B. rhodani (II)	*		*		
B. scambus (II)	*				
Sericostomatidae					
S. personatum (II)		*			*
Sphaeriidae (IV*)				*	
Polycentropodidae					
P. flavomaculatus (II)				*	
P. kingi (II)					*

Table 3.4.23. Table to show the SIMPER within sample similarity output for each season at LU, with the listed species contributing at least 60% of the cumulative similarity. (LIFE flow groups given in brackets).

When the important species for LD are considered, Table 3.4.24 shows that Leuctridae (always the fast flow loving species of) are found in each of the spring samples, *Gammarus pulex* is present in two of the three sampling years, but with the spring 2002 and 2003 samples having no further common families. There is much greater variation in important families in the spring than found at LU. However, Leuctridae was only important in the spring samples, which is a very similar pattern to LU. There is a much greater agreement

in important species between sampling years for the autumn samples, with three families found to be important in both years (Limnephilidae, Simuliidae, and Seriscostomatidae). One Limnephilidae species was common to both autumn samples (*P. latipennis*), whereas *P. cingulatus* was present in autumn 2002 and not 2003; and *L. rhombicus* was present in 2003 and not in 2002. *P. latipennis* was the dominant Limnephilidae species in both sampling seasons. *Limnephilus rhombicus* prefers slower flowing water than *P. cingulatus*, and again this may be due to the lower flows found in summer 2003 (where there appeared to be an increase in importance of slower flow preferring species). Seriscostomatidae (*S. personatum*) was found to be important in each of the 2003 samples, and only in the 2003 samples.

	Sp.01	Sp.02	Aut.02	Sp.03	Su.03	Aut.03
Hydropsychidae						
H. siltalai (II)	*		*			
H. pellucidula (II)			*			
Leuctridae						
L. hippopus/moselyii (I)	*	*				
L. inermis (I)	*	*				
L. geniculata (II)		*		*		
L. fusca (II)		*		*		
Baetidae						
Baetis rhodani (II)	*					
Gammaridae						
G. pulex (II)	*	*	*			
Polycentropodidae						
P. flavomaculatus (II)		*				*
P. kingi (II)		*				
Ephemerellidae						
S.ignita (II)		*				
Limnephilidae						
P. latipennis (IV*)			*			*
P. cingulatus (II)			*			
Limnephilus rhombicus (IV)						*
Simuliidae (II)			*			*
Empididae				*		
Hydracarina				*	*	
Sericostomatidae						
S. personatum (II)				*	*	*
Spaheriidae (IV*)					*	*
Leptoceridae						
Mystacides spp. (IV)					*	
Lymnaeidae						
Lymnea peregra (IV)					*	

Table 3.4.24. Table to show the SIMPER within sample similarity output for each season at LD, with the listed species contributing at least 60% of the cumulative similarity. (LIFE flow groups given in brackets).

It appears that the spring samples found at LD are somewhat less stable than those found at LU in terms of the important species, but the autumn important species show a similar level of stability. This is perhaps a facet of the increased flow variability experienced at the more downstream site due to tributaries which are not present above the upstream site. This may well be an impact of impoundment. When the LIFE scores of the important species at LD are examined, it can again be seen, that the flow preferences of important species in summer and autumn 2003, are generally slower than the preceding sampling seasons. This again could be due to the lower discharge during 2003.

Table 3.4.25 shows that for the Hipper, Elmidae is important for each of the sampling seasons, and the most abundant species of this family being the faster flow preferring species of *L. volckmari* and *E. aenea*. *S. ignita*, Leuctridae, Baetidae and Elmidae are important for both of the spring sampling seasons. The same species of Baetidae are found in both the spring samples, but *Leuctra nigra* and *L. inermis* are found in the spring 2003 are not present in the spring 2002 samples. *Leuctra nigra* is the hairiest of the Leuctridae nymphs, and is generally found in silty substrates (Hynes, 1970). Its presence in 2003 may be due to the reduced flows present in that time period. Ephemerellidae is only found to be important in the spring samples.

Leuctridae is again found to be important only in the spring and summer samples as was the case in the Rivelin and Loxley. Baetidae and Elmidae are important for both autumn samples. In both autumn samples only one species of the Baetidae family was found (*Baetis rhodani*), which is in contrast to the more numerous species of the Baetidae family found in the spring samples. When the LIFE scores for the important species are examined, it appears that the flow velocity preferences of important species are slower in summer and autumn 2003, than in any of the other previous sampling season.

	Sp.02	Aut.02	Sp.03	Su.03	Aut.03
Ephemerellidae					
S. ignita (II)	*		*		
Leuctridae					
L. geniculata (II)	*		*	*	
L. fusca (II)	*		*	*	
L. hippopus (I)	*		*		
L. nigra (II)			*		
L. inermis (I)			*		
Baetidae					
B. scambus (II)	*		*		
B. rhodani (II)	*	*	*		*
B. muticus (II)	*		*		
Simuliidae (II)	*				*
Elmidae					
L. volckmari (II)	*	*	*	*	*
E. aenea (II)	*	*	*	*	*
Oulimnius (IV)	*			*	
Heptageniidae					
Ecdyonurus spp. (I)		*			
R. semicolorata (I)		*			
Tipulidae (IV*)		*		*	
Hydracarina			*		
Chloroperlidae					
S. torrentium (I)			*		
Sericostomatidae					
S. personatum (II)					*
Limnephilidae					
P. latipennis (II)					*
Erpobdellidae					
E. octoculata (IV)					*
Nemouridae					
Nemoura avicularis (IV)					*
A. sulcicollis (II)					*

Table 3.4.25. Table to show the SIMPER within sample similarity output for each season at H, with the listed species contributing at least 60% of the cumulative similarity. (LIFE flow groups given in brackets).

In terms of important species as determined by the SIMPER routine, it appears that the site with the most variation in important species was RU, which has a large number of important families, and a number of families which were important in only one sampling season. The downstream site on the Rivelin (RD) had a more stable 'set' of important macroinvertebrates, but with similar important species as the upstream site. The reverse pattern was seen in the Loxley, with the upstream site (LU) having a more stable set of important macroinvertebrates than the downstream site. This could be due to the lack of tributaries creating hydrologic variability at the more upstream Loxley site. It was also

seen that in the two Loxley sites and the Hipper, that the important species in summer and autumn 2003 preferred slower flows than in previous sampling seasons, this could be due to the lack of augmentation during that period.

Guild Analysis

The percentage of invertebrates which are members of a particular feeding guild was also calculated for each of the sampling sites as explained in section 3.3.2.4. Figure 3.4.5 shows that the RU site shows little clear pattern and no great variation. The percentage of invertebrates that are predators remains relatively stable across each of the sampling seasons, apart from spring 2001, which has a larger predator population than the following seasons, and this led to a much smaller scraper population. Despite having more variable patterns of important species, it appears that the percentages of feeding guilds do not change greatly at RU.

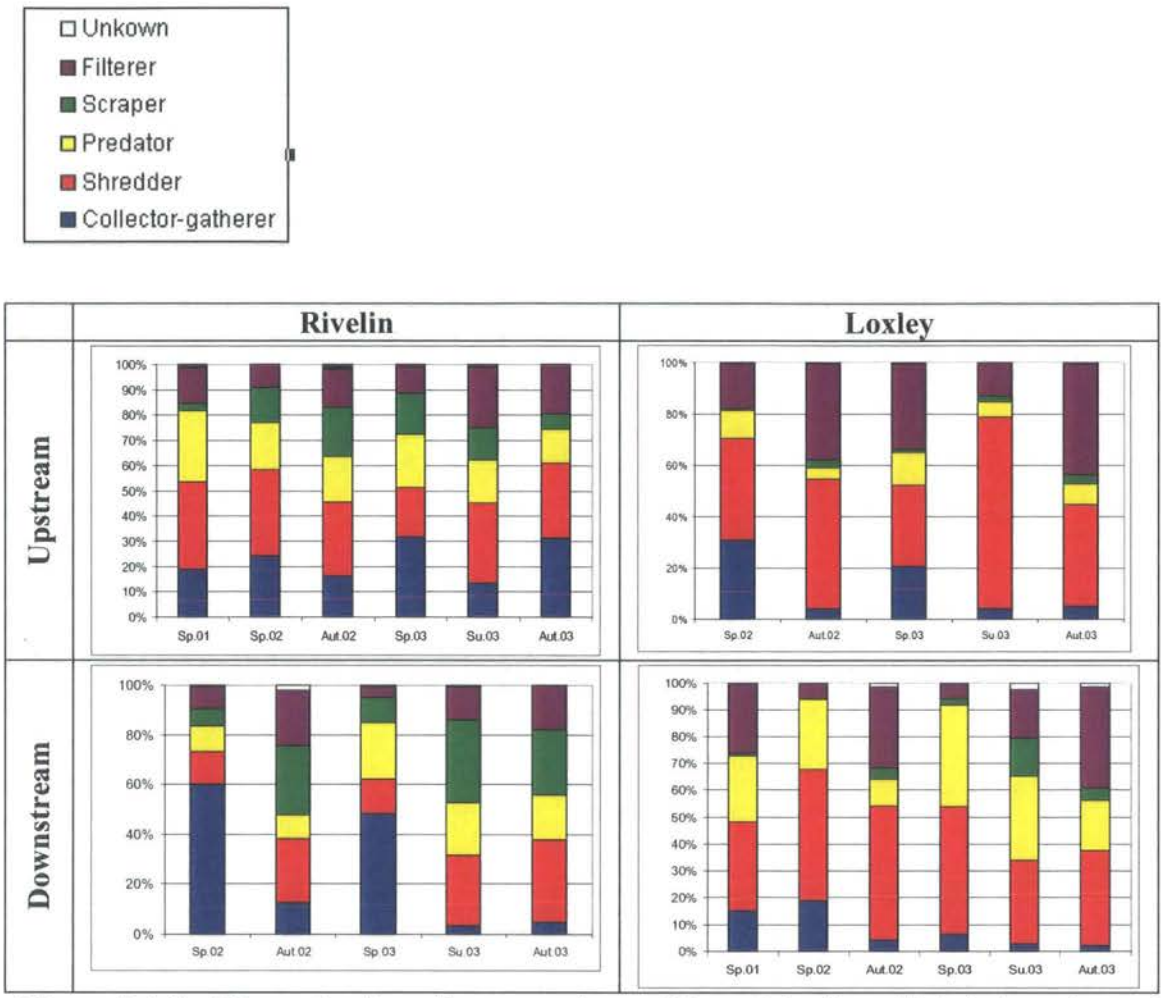

Figure 3.4.5. Figure to show the percentage of invertebrates which belong to each feeding guild for each pre-change sampling season for each sampling site on the Rivelin and Loxley.

There is much more variation in the guilds representing the RD sampling site than at RU (Figure 3.4.5). It shows that in spring 2002, the population is dominated by the collector-gatherer guild, with relatively similar populations of the remaining feeding guilds. The autumn 2002 population saw a large decrease in the percentage of collector gathering invertebrates, with the bulk of the population evenly divided between the filterer, scraper and shredder guilds. The second spring sample had a large percentage of collector-gatherer invertebrates, as with spring 2002. The populations of shredder, filterer and scraper invertebrates decreased from autumn 2002, whilst the proportion of predators increased. The populations sampled in summer and autumn 2003 had very similar proportions of the feeding guilds. The percentage of collector-gatherer invertebrates was considerably

reduced from the spring 2003 samples. The summer and autumn 2003 samples were largely dominated by shedder and scraper invertebrates.

At LU, the spring samples again showed the highest percentage of collector-gatherer invertebrates, but in smaller proportions than those found at RD (Figure 3.4.5). The feeding guilds proportions were relatively similar for both of the spring samples at the LU site, with the spring 2002 having slightly higher proportions of collector-gatherers and shredders than in 2003, but with a higher proportion of filterers found in 2003 when compared with 2002. Very small proportions of the populations were scraper invertebrates in both years. The two sampling years of autumn samples also had very similar proportions of feeding guilds. Each sampling season had a low proportion of collector gatherer invertebrates, with the population dominated by shredder and filterer invertebrates. The summer 2003 samples were dominated by the shredder guild, with a lower proportion of filterer invertebrates than for any other sampling season. It must also be noted that the proportions of predators and scrapers remained relatively constant across the whole sampling programme.

Figure 3.4.5 shows that for the LD site, there is more variation in percentage guilds in the spring samples when compared with the LU site (this matches the SIMPER analysis). The spring 2002 and 2003 samples have relatively similar proportions of feeding guilds with the population dominated by shredder invertebrates, but with the 2003 population having a slightly higher predator population and the 2002 samples having a slightly higher collector-gatherer proportion than in 2003. The spring 2001 samples had a larger percentage of filterer invertebrates than either of the two later spring samples. It can again be seen that the two autumn samples have very similar percentages of feeding guilds, with the populations dominated by shredder and filtering invertebrates. The proportion of predator invertebrates was higher in autumn 2003 compared with autumn 2002. The summer 2003 samples were dominated by shredder and predator invertebrates, but also contained the largest percentage of scraper invertebrates of any of the sampling seasons.

The most important feeding guild in both the Loxley sites is Shredder, and that Predators are much more important at LD than LU, and there also appears to be a greater variation in the proportions of feeding guilds present at the downstream site. There were however some

similarities between the two Loxley sites, with both sites having relatively consistent patterns through the seasons. In both cases, filtering invertebrates became more important in the autumn than for any other sampling season.

When the feeding guilds present in the Hipper are considered, Figure 3.4.6 shows that the two spring sets of samples had relatively similar proportions of feeding guilds, but with spring 2003 having a slightly higher proportion of predator invertebrates than spring 2002. In each of the spring samples, the highest percentage was found in the collector-gatherer guild. Figure 3.4.6 shows that the autumn 2002 and 2003, samples had relatively similar proportions of filterer, scraper and predator invertebrates. The autumn 2002 samples had a much higher percentage of the collector-gatherer guild than in 2003, but the autumn 2003 samples have a higher percentage of the shredder guild than the 2002 samples. The summer 2003 samples had the smallest proportion of the collector-gatherer guild of any sampling season, and was dominated by the shredder guild. This increase in importance of shredder invertebrates in autumn was also seen at LU.

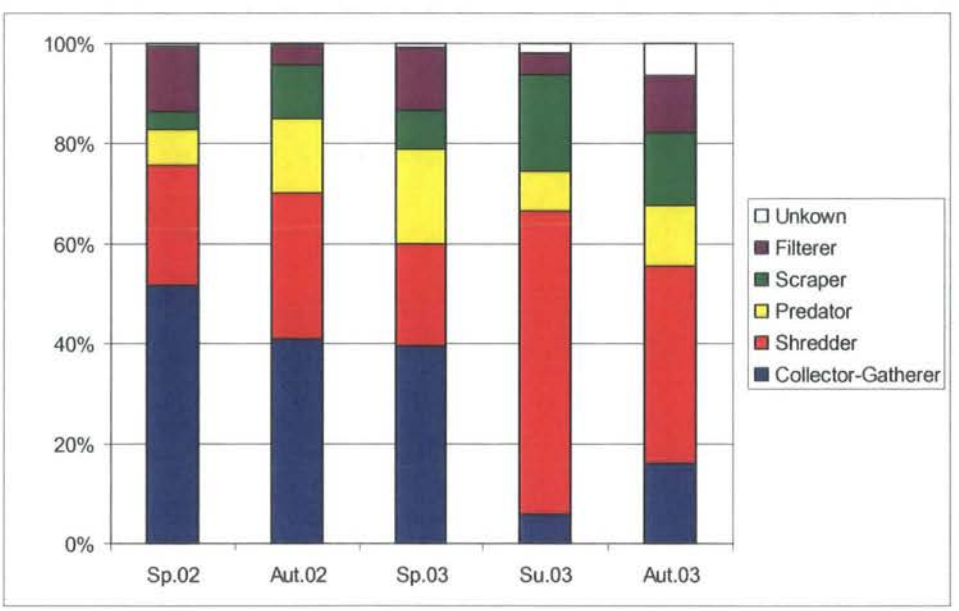

Figure 3.4.6. Figure to show the percentage of invertebrates which belong to each feeding guild for each pre-change sampling season at the H site.

The guild analysis for the Rivelin has shown that the variability displayed in the guilds is the opposite of the displayed in the important species analysis. In the important species analysis, RU displayed the greatest variation, in the guild analysis, RD displayed the

greatest variation. In the Loxley, the guilds followed the same pattern as the important species analysis with the greatest variation present at the downstream site. Figure 3.4.5 shows that the shedder and filterer guilds are very important on the Loxley, but to a lesser extent, on the Rivelin. The greater importance of the shredder guild in the Loxley, is of interest as shredding invertebrates generally have a preference for slower flowing water. The scraper guild is important for the Rivelin, but few are found in the Loxley. Figure 3.4.6 shows that the Hipper is generally dominated by the collector-gatherer and shredder guilds.

Relationship between biotic and abiotic data: BIOENV

The BIOENV routine was used to examine directly the relationship between the biotic and abiotic data to assess any linkages (section 3.3.2.3). The results are presented in terms of a list of environmental variables which provided the best Spearman's rank correlation with the biotic data. These environmental variables are considered important. At RU, Table 3.4.26 shows that depth is an important environmental variable for both spring sets of samples, and flow velocity is important for both autumn sets of samples. Table 3.4.26 also shows that higher Spearman's rank correlations are found in the spring samples than at other times of year, which suggests that environmental variables have more influence on the invertebrate community at that time of year at RU.

SPRING 2002	AUTUMN 2002	SPRING 2003	SUMMER 2003	AUTUMN 2003
RU				
Spearman's Rank Correlation = 0.585	*Spearman's Rank Correlation = 0.337*	*Spearman's Rank Correlation = 0.718*	*Spearman's Rank Correlation = 0.445*	*Spearman's Rank Correlation = 0.459*
Width	Substrate	Depth	Depth	Depth
Depth	Flow Velocity		Flow Type	Flow Velocity
Flow Type				
RD				
Spearman's rank Correlation = 0.471	*Spearman's rank Correlation = 0.434*	*Spearman's rank Correlation = 0.356*	*Spearman's rank Correlation = 0.656*	*Spearman's rank Correlation = 0.613*
Depth	Substrate	Flow Type	Flow Type	Flow Type
Flow Type	Flow Velocity	Substrate	Substrate	Flow Velocity
Substrate				
Flow Velocity				
LU				
Spearman's rank correlation = 0.473	*Spearman's rank correlation = 0.142*	*Spearman's rank correlation = 0.658*	*Spearman's rank correlation = 0.482*	*Spearman's rank correlation = 0.248*
Flow Type	Depth	Flow Type	Depth	Depth
	Substrate	Substrate	Flow Type	Flow Type
LD				
Spearman's rank correlation = 0.645	*Spearman's rank correlation = 0.463*	*Spearman's rank correlation = 0.673*	*Spearman's rank correlation = 0.158*	*Spearman's rank correlation = 0.529*
Width	Flow Velocity	Depth	Flow Type	Flow Velocity
Substrate		Substrate	Flow Velocity	
Flow Velocity				
H				
Spearman's rank correlation = 0.155	*Spearman's rank correlation = 0.100*	*Spearman's rank correlation = 0.051*	*Spearman's rank correlation = 0.162*	*Spearman's rank correlation = 0.512*
Width	Flow Type	Depth	Depth	Flow Type
Flow Velocity			Flow Type	Substrate

Table 3.4.26. Table to show the output from the BIOENV procedure for each site at each pre-change sampling season, comparing the invertebrate communities and measured environmental variables.

Table 3.4.26 shows the relative importance of flow type, substrate and flow velocity to the matching of biotic and abiotic data at RD, and the lack of influence of width and depth. What can also be seen from Table 3.4.26 is that flow type and substrate were found to be important in both spring lists, and flow velocity is common to both the autumn lists.

The BIOENV analysis for LU (Table 3.4.26) reveals that there are much lower Spearman's rank correlation values for the autumn analysis than the spring analysis. This indicates a weaker relationship between the measured environmental variables and the invertebrate community at that time of year. Despite the low Spearman's rank correlation values, depth is present in each of the autumn lists and so it appears that depth may be a factor in determining the invertebrate composition of LU in the autumn. In the spring samples (with higher correlations) flow type is important in both years.

Table 3.4.26 shows that no single environmental variable is common to the output for all seasons at LD. Flow velocity is in the output for 4 out of the 5 seasons. Substrate occurs in both of the spring lists, suggesting that this is an important factor affecting the invertebrate community in the spring. The spring sampling seasons have higher Spearman's rank correlation between the biotic and abiotic data than the other seasons and summer 2003 has the lowest correlation value. Table 3.4.26 also shows that flow velocity only occurs individually in both autumn lists, which suggests it is an important environmental variable at LD in the autumn.

For the Hipper Surber site, no flow type data were included in the spring 2002 analysis due to data gaps. Table 3.4.26 shows that no environmental variable occurs in the list for every season, but flow type appears to be of importance for both autumn sets of samples. This BIOENV analysis (Table 3.4.26) has very low Spearman's rank correlation values for all seasons apart from Autumn 2003, indicating a poor relationship between the invertebrate community and measured environmental variables at this site. This further enhances the theory that to some extent the invertebrate populations at H operate independently from the environmental variables sampled for this study, and that the measured environmental variables are not limiting as in the Rivelin and Loxley.

Table 3.4.27 indicates that the Loxley sites appear to have the lowest Spearman's rank correlation. When averaged the Loxley sites do have smaller Spearman's rank correlations. The difference between LD and the two Rivelin sites is very small whilst the average spearman's rank correlation is much lower at LU than at LD. This is further indication that tributaries do provide some hydrologic variation in the Loxley, but only at the downstream site, as there are no tributaries upstream of the LU site.

SITE	AV. SPEARMAN'S RANK CORRELATION
RU	0.51
RD	0.51
LU	0.40
LD	0.49
H	0.20

Table 3.4.27. Table to show the average spearman's rank correlation values produced from the BIOENV analysis at each of the sites.

The Loxley samples appear to have lower correlation values than the Rivelin. They also display more variation as to which environmental variable is important than the Rivelin (Table 3.4.27). This further shows a level of detachment between the Loxley (especially at LU) invertebrate populations and the environmental variables sampled in this study, as a higher Spearman's rank correlation values indicates a better correlation of biotic and abiotic data.

3.4.1.4. Summary

Data Analysis Method	Brief results
Environmental variables: ANOSIM	More between season variability in R than L.
Av. Invertebrate density	Most dense at LU; RD>RU. Lots of between season variability at each of the sites.
Av. No. Families	All sites similar. Large amount of between site variation and between season variation.
Av. Shannon Diversity	Few clear patterns – either from graph or ANOVA analysis
Family Lists	More families found in the Rivelin than the Loxley
Invertebrate samples: ANSOIM	Sampling sites statistically significantly different, both between seasons and between sites.
Important Species: SIMPER	Most variability in important species at RU. LU has the most 'stable' set of important species In the Loxley and Hipper, the important species in late 2003 preferred slower flows.
Feeding Guilds	RU very stable proportions; RD very variable feeding guilds. Shredders important for the Loxley. Scrapers important for the Rivelin.
BIOENV	Higher correlation values in the Rivelin. Loxley invertebrates more detached. Flow type appears important to each river.

Table 3.4.28. Table to summarise the data analysis performed and a brief synopsis of their results.

This section has demonstrated the importance of Chironomids and Oligochaeta to the study systems, and explained the reasons behind their removal from subsequent analysis. The kick samples also demonstrated that the LIFE scores were generally higher in the Rivelin than in the Loxley. The importance of flow regime was first observed when it was seen that the 1995 drought caused some changes to the invertebrate populations, but mostly in peripheral species.

The importance of flow regime was further highlighted when the measured environmental data were analysed, with greater differences seen on the Rivelin (Table 3.4.28). It was then seen that LU, had the highest densities of invertebrates, with the Shannon diversity displaying little coherent pattern. The analysis of the family lists showed that the Rivelin contained more families than the Loxley, and many of these families were Ephemeroptera, Plecoptera, and Trichoptera (EPT).

When the Surber samples were analysed using multivariate analyses, it was seen that there was generally a difference between sites and seasons. The most variable important species list were found at RU, with much more consistent important species lists at the other sites. It was seen however, that RU had stable proportions of functional feeding guilds. Both Loxley sites contained large proportions of shredder macroinvertebrates. The BIOENV analysis revealed that a range of measured environmental variables are of importance, but also that the Rivelin invertebrates appear to have a closer relationship with the measured environmental variables than the Loxley.

The results presented above are pointing to the fact that the Loxley may currently have too great a discharge. This is evidenced by the higher density of invertebrates in the Loxley than the Rivelin, the lack of difference between sampling seasons in terms of the environmental data, the LIFE scores being higher on the Rivelin than the Loxley, the importance of the shredder guild to the Loxley macroinvertebrates and the lower correlation values between environmental and invertebrate data using the BIOENV routine. This is perhaps surprising as it may be thought that the more water (within reason) that a river receives the better, however, this does not appear to be the case. This is also compounded by the fact that the Loxley (and in particular LU) is subject to less variation than the

Rivelin, as overtopping events are less frequent due to the larger size of the Loxley compensation flow reservoir.

3.4.2. Variability in invertebrate populations

The previous chapter has shown that there are differences between the Rivelin, Loxley and Hipper, some of which are perhaps caused by compensation flows. However, there are a number of sauces of variability which may mask these patterns: (i) within-sample variability; (ii) within site (marginal – central) variability; (iii) within river variability; and (iv) temporal variability.

Not only is an understanding of this variability desirable to contextualise the previous section but, an understanding of each of these types of variability is vital in terms of analysing the post-change macroinvertebrate data so any changes which may occur can be attributed to the alteration of the compensation flows and not due to natural variation, and each of these will be addressed in turn in this section.

3.4.2.1. Within-sample variability

In order to investigate the within-sample variability the standard deviation of the Shannon diversity was calculated for each site at each sampling season, as it gives an indication of the variety of macroinvertebrate populations found within a given sample. Figure 3.4.7 shows that the RU samples had the largest standard deviation indicating a wide variation in the diversity of the macroinvertebrate populations, thus suggesting a wide variety of habitat within the study site. The standard deviation at RD was always lower than that at RU, suggesting less within-site variability. The standard deviations of the two Loxley sites are of a similar level to that of RD, which indicates less variation in the macroinvertebrate habitat in the Loxley and the bottom of the Rivelin, when compared with the upper reaches of the Rivelin.

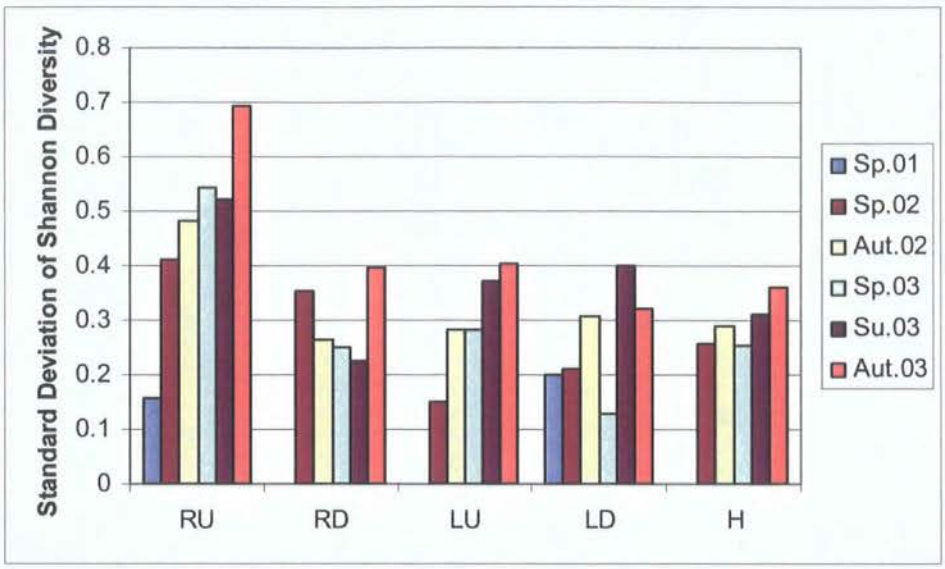

Figure 3.4.7. Standard deviation of the Shannon diversity of the Surber samples at each of the sites in each of the pre-change sampling seasons.

3.4.2.2. Marginal/central variability

Table 3.4.29 shows that there is generally very little difference between the invertebrate communities in the centre and margins of the rivers. It thus appears that the spatial heterogeneity of upland Millstone Grit rivers such as the Rivelin and Loxley means that there is no distinct difference between central and marginal samples. The topography of these rivers is so complex that various flow conditions could occur at any point in the channel. This is further supported by Table 3.4.30, which shows that the environmental variables sampled in the centre and margins were rarely different. What can also be seen from Tables 3.4.29 and 3.4.30 is that where there were statistically significant differences found using the environmental variables, there was no difference between the biotic data at the centre and margins of the channel, and vice versa. Again, this indicates some detachment of the abiotic and biotic.

SITES	SP. 01	SP. 02	AUT. 02	SP. 03	SU. 03	AUT. 03
RU_C – RU_M	0.88	-0.02	0.08	0.03	0.04	-0.19
RD_C – RD_M		-0.09	0.14	-0.06	0.08	-0.10
LU_C – LU_M		-0.16	-0.09	-0.06	-0.08	0.14
LD_C – LD_M	0.19	0.08	0.37	0.17	0.20	0.12
H_C – H_M		0.07	-0.03	0.00	0.04	0.02

Table 3.4.29. Table to show the ANOSIM output for comparing the biotic data in the central and marginal Surber samples for each of the pre-change sampling seasons. Grey colouration indicates a statistically significant difference to the 95% significance level.

SITES	SP. 01	SP. 02	AUT. 02	SP. 03	SU. 03	AUT. 03
RU_C – RU_M		0.36	0.17	-0.06	0.08	-0.09
RD_C – RD_M		0.20	0.06	-0.18	-0.09	-0.06
LU_C – LU_M		-0.08	0.11	-0.16	-0.22	-0.13
LD_C – LD_M		0.01	0.25	-0.12	0.38	-0.11
H_C – H_M		-0.15	-0.16	-0.17	-0.12	-0.02

Table 3.4.30. Table to show the ANOSIM output for comparing the measured environmental variables from central and marginal Surber samples for each of the pre-change sampling seasons. Grey colouration indicates a statistically significant difference to the 95% significance level.

When the SIMPER routine is used to asses the important families within the central and marginal samples using similarity (for RU in spring 2001), there are no common species between the two groups of samples. Table 3.4.31 shows that Gammaridae is the best discriminatory family. It can also be seen that apart from *Ephemera vulgata*, the best discriminatory species are found in running water (velocity: 20 – 100cm/s) or higher.

RU -Central	RU Marginal	RU_C versus RU_M
Cumulative similarity = 55.37	*Cumulative similarity = 56.39*	*Cumulative dissimilarity = 50.47*
Chloroperlidae (I)	Gammaridae (II)	Gammaridae (G. pulex (II))
Heptageniidae (I*)	Ephemeridae (II*)	Ephemeridae (E. vulgata (III))
Hydropsychidae (II)	Sericostomatidae (II)	Limnephilidae
Baetidae (II*)	Limnephilidae (IV*)	Baetidae (B. rhodani (II))
	Leptoceridae (IV*)	Rhyacophilidae (R. dorsalis, (I))
		Heptageniidae
		Leuctridae (L. hippopus/moselyii, (I); L. inermis, (I))
		Leptoceridae

Table 3.4.31. Table to show the output from the SIMPER analysis for RU for the central and marginal samples of the spring 2001 Surber samples. Species included contribute to 60% of the cumulative similarity or dissimilarity LIFE flow groups given in brackets.

The central and marginal samples for the same sampling seasons were also compared for the LD autumn 2002 samples. Table 3.4.32 shows that the Limnephilidae family is an important family for both the central and marginal samples. Furthermore, Table 3.4.32 shows that the Limnephilidae family is a good discriminator between the central and marginal samples. Two species of Limnephilidae were found in the central samples (*P. cingulatus* (n=1) and *P. latipennis* (n= 18)) whilst only *P. latipennis* (n=7) was found in the margins. Gammaridae can also be seen as an important discriminatory family, due to the fact that the abundance of *Gammarus pulex* was 44 in the central samples and only 6 in the marginal samples. Both species of the Hydropsychidae family which have been found at this site were present in both the central and marginal samples, but with a difference in abundance being driven by a greater abundance of *H. siltalai* in the central samples (35) as opposed to the marginal samples (16). As *H. siltalai* is a filter feeding invertebrate with a preference for flows of 20-100cm/s, it appears that in this case there may well be a greater abundance of this habitat in the central samples compared with the margins. The Leptoceridae sub-family Mystacides sp. was found in greater numbers in the marginal samples (11) than in the central samples. This is further evidence for the existence of

slower flowing water in the margins as the Mystacides genus has a preference for slow and standing waters.

AUTUMN 2002		
Group LD C Cumulative similarity = 63.29	**Group LD M** Cumulative similarity = 50.84	**Groups LD C & LD M** Cumulative Dissimilarity = 54.70
Hydropsychidae (II)	Limnephilidae (IV*)	Gammaridae (G. pulex (II))
Gammaridae (II)	Simuliidae (II)	Hydropsychidae (H. siltalai (II); H. pellucidula (II))
Limnephilidae (IV*)		Simuliidae (II)
		Sericostomatidae (S. personatum (II))
		Limnephilidae (P. cingulatus (II); P. latipennis (II))
		Leptoceridae (Mystacides sp. (IV))
		Empididae (Chelifera; Hemerodromia; Clinocera)

Table 3.4.32. Table to show the output from the SIMPER analysis for LD for the central and marginal samples of the autumn 2002 Surber samples. Species included contribute to 60% of the cumulative similarity or dissimilarity LIFE flow groups given in brackets.

There appears that there are no consistent differences between the central and marginal samples in either the Rivelin or the Loxley, in terms of invertebrate populations or measured environmental variables. This is probably due to the large amount of spatial heterogeneity present in these upland Millstone Grit streams.

3.4.2.3. Spatial variability (intra-river)

In terms of quantifying the effects of altering the discharge regimes upon the rivers, the within river spatial variability must also be of interest. This is one of the reasons why two Surber sites were used on both the Rivelin and Loxley, and the kick sample data were also used. What was shown in the ANOSIM analysis of the surber sample data for all rivers is that all of the sites are different indicating a large amount of spatial variability in terms of the invertebrate populations present at the upstream and downstream sites. The differences between the sites will be further explored in Section 3.4.3.

3.4.2.4. Temporal variability

Table 3.4.19 showed that there was always a statistically significant difference between the invertebrate communities for each sampling season at a site, indicating a large amount of temporal variability in the communities at each of the sites. In the bar graph of the Shannon diversity at each site through time (Figure 3.4.4), there are few annual or seasonal trends. However, Figure 3.4.7 shows that the standard deviation of the Shannon diversity of the surber samples appears higher in the autumn than in any of the other sampling seasons. This indicates a greater variation in the diversity of the invertebrate populations sampled within the autumn compared with the other sampling seasons.

Tables 3.4.21 to 3.4.25 showed that there was a large amount of seasonal variation in the important species at each of the sites. There are a number of families which appear to be important in only one or two sampling seasons. Therefore, if, when the analysis is conducted upon the post-change data, different families are seen to be important to the sites, it may be an indication of the change in flow regime altering the invertebrate populations in the Rivelin and Loxley.

Figure 3.4.5 showed that there were often large seasonal variations in the proportions of the functional feeding guilds found at some of the sites. For example, at RD and LU, there was a higher proportion of collector gatherers in spring samples than for any other time of year. If the proportions of the functional feeding groups were to change in these seasons, perhaps this could be seen as an indication of an impact of altering compensation flow regimes.

3.4.2.5. Summary

Four scales of variability were assessed within this section: (i) within-sample variability; (ii) marginal/central variability; (iii) intra-river variability; and (iv) temporal variability. The largest within sample variability (measured by the standard deviation of Shannon diversity) was seen at RU, with the remaining sampling sites having similar within sample variability. There was generally very little difference between the central and marginal samples in terms of either invertebrate communities or measured environmental variables, thus indicating a large amount of spatial heterogeneity. There was also a large amount of

within river variability, with the invertebrate communities at the five sampling sites being different within each of the sampling seasons. The invertebrate communities at each of the sampling sites were also different when the sampling seasons were compared, indicating a large amount of temporal variability in the invertebrate populations in these rivers. Seasonal variation was also observed in the important species and guild analyses.

3.4.3. Is there an impact from altering the compensation flow regimes?

This section assesses the impact of altering the compensation flows on the macroinvertebrate communities in the Rivelin and Loxley, by introducing data collected in 2004 and 2005 and comparing it to the pre-change data. As with the previous sections, a number of univariate and multivariate methods will be used. The first section will address whether the environmental variables in the rivers have changed, and the second section displays the invertebrate density and family abundance. The third section discusses the diversity indices and analyses the family lists for each of the sites. The fourth section displays the changes to the functional feeding groups. The fifth section then compares the important species for the pre and post change samples. The final section analyses the extent to which the relationship between the invertebrates and measured environmental variables was altered or not.

3.4.3.1. Environmental variables

The obvious first point of interest is to assess whether the environmental conditions on the Rivelin and Loxley have changed due to the alteration of the compensation flow releases. When the sampling sites are compared for each of the seasons, Table 3.4.33 shows that there are marginally more similarities between sampling sites in the post change data compared to the pre change data.

SP.02	RU	RD	LU	LD	H
RU		0.34	0.21	0.31	0.15
RD			0.01	0.26	0.24
LU				0.28	0.24
LD					0.60
H					

Aut.02	RU	RD	LU	LD	H
RU		0.14	0.20	0.38	0.36
RD			-0.04	0.15	0.4
LU				0.20	0.21
LD					0.53
H					

SP.03	RU	RD	LU	LD	H
RU		0.13	0.46	0.61	0.46
RD			0.29	0.37	0.23
LU				0.29	0.34
LD					0.45
H					

Su. 03	RU	RD	LU	LD	H
RU		0.13	0.81	0.65	0.61
RD			0.53	0.54	0.31
LU				0.64	0.61
LD					0.62
H					

Aut03	RU	RD	LU	LD	H
RU		0.25	0.68	0.55	0.30
RD			0.30	0.31	0.03
LU				0.43	0.54
LD					0.48
H					

Sp.04	RU	RD	LU	LD	H
RU		0.29	0.28	0.36	0.51
RD			0.03	0.15	0.39
LU				0.08	0.08
LD					0.28
H					

SU04	RU	RD	LU	LD	H
RU		0.32	0.20	0.31	0.59
RD			0.01	0.225	0.28
LU				0.16	0.05
LD					0.51
H					

Aut04	RU	RD	LU	LD	H
RU		0.24	0.23	0.35	0.34
RD			0.07	0.22	0.30
LU				0.27	0.03
LD					0.27
H					

Sp05	RU	RD	LU	LD	H
RU		0.203	0.268	0.567	0.447
RD			0.077	0.558	0.245
LU				0.594	0.341
LD					0.425
H					

Su.05	RU	RD	LU	LD	H
RU		0.14	0.31	0.45	0.42
RD			-0.03	0.22	0.12
LU				0.10	0.12
LD					0.45
H					

Aut05	RU	RD	LU	LD	H
RU		0.11	0.34	0.36	0.60
RD			0.00	0.11	0.18
LU				0.29	0.07
LD					0.43
H					

Table 3.4.33. Table to show the ANOSIM output comparing the measured environmental variables of the sampling sites for each sampling season. Grey colouration indicates a statistically significant difference to the 95% significance level.

As discussed in section 4.1, in general for the pre-change samples, the Rivelin sites have a greater number of statistically significant differences than the Loxley sites. Table 3.4.34 shows that by reducing the Loxley's compensation flows, the number of differences in the Loxley sites has increased. There are now differences between the pre-and post-change samples (mostly spring and summer 2005), in both LD and LU. What is also noticeable is

that when the post-change samples are compared for both LD and LU, each sampling season is generally different, which is the complete opposite of that seen for the pre-change samples. When the Rivelin sites are considered, it was seen in section 4.1 that there was generally a difference between the pre-change samples in the Rivelin sites. This was attributed to the lower compensation flow, allowing the impact of variation to be greater than for the Loxley, where the higher compensation flow was hypothesised to dampen the impact of overtopping events. What can now be seen from Table 3.4.34 is that the post-change samples within the two Rivelin sites have a much reduced number of differences when compared with the pre-change differences. It appears that the increased compensation flow in the Rivelin is dampening the impacts of the overtopping events and also decreasing their number (as seen in Chapter 2). The interesting question now is whether this has impacted upon the macroinvertebrate community of the Rivelin and Loxley.

RU	Sp02	Aut02	Sp03	Su03	Aut03	Sp04	Su04	Au04	Sp05	Su05	Au05
Sp.02		0.20	0.29	0.07	0.14	0.20	0.192	0.16	0.115	0.29	0.23
Aut.02			0.37	0.39	0.27	0.09	0.13	0.17	0.064	0.36	0.19
Sp.03				0.11	0.20	0.19	0.023	0.01	0.056	0.00	0.07
Su.03					0.1	0.18	0.119	0.06	0.104	0.28	0.21
Aut.03						0.13	0.118	0.09	0.116	0.18	0.16
Sp.04							-0.06	-0.05	-0.02	0.07	-0.04
Su.04								-0.1	-0.07	-0.02	-0.07
Aut.04									-0.07	-0.01	-0.07
Sp.05										0.03	-0.03
Su.05											-0.04
Aut.05											

RD	Sp02	Aut02	Sp03	Su03	Aut03	Sp04	Su04	Au04	Sp05	Su05	Au05
Sp.02		0.07	0.26	0.72	0.6	0.12	-0.03	0.05	0.217	0.14	0.03
Aut.02			0.23	0.62	0.52	0.02	0.157	0.25	0.437	0.08	0.05
Sp.03				0.18	0.16	0.30	0.474	0.33	0.209	0.03	0.11
Su.03					-0.04	0.60	0.853	0.62	0.451	0.42	0.43
Aut.03						0.46	0.68	0.47	0.255	0.31	0.29
Sp.04							0.073	0.08	0.271	0.16	0.03
Su.04								0.03	0.281	0.28	0.08
Aut.04									0.209	0.23	-0.02
Sp.05										0.31	0.12
Su.05											0.09
Aut.05											

LU	Sp02	Aut02	Sp03	Su03	Aut03	Sp04	Su04	Au04	Sp05	Su05	Au05
Sp.02		-0.1	-0.05	-0.06	-0.02	0.09	-0.05	-0.03	0.055	0.26	-0.04
Aut.02			-0.10	-0.08	-0.06	0.12	-0.021	0.01	0.087	0.10	-0.02
Sp.03				-0.05	-0.03	0.22	0.008	0.05	0.097	0.12	0.04
Su.03					-0.09	0.10	-0.051	-0.02	0.053	0.18	-0.05
Aut.03						0.10	-0.055	-0.04	-0.01	0.12	-0.03

Sp.04							-0.024	0	0.037	0.36	0.00
Su.04								-0.06	-0.04	0.20	-0.06
Aut.04									-0.09	0.29	-0.04
Sp.05										0.33	0
Su.05											0.25
Aut.05											
LD	**Sp02**	**Aut02**	**Sp03**	**Su03**	**Aut03**	**Sp04**	**Su04**	**Au04**	**Sp05**	**Su05**	**Au05**
Sp.02		0.05	-0.05	0.06	0.17	-0.04	-0.034	-0.02	0.752	0.29	-0.03
Aut.02			0.06	0.31	0.45	0.01	0	0.02	0.81	0.16	0.08
Sp.03				-0.05	0.02	-0.05	-0.06	-0.06	6.82	0.30	-0.05
Su.03					-0.07	0.04	0.033	0.03	0.743	0.49	-0.02
Aut.03						0.11	0.122	0.10	0.725	0.51	0.02
Sp.04							-0.08	-0.08	0.566	0.12	-0.05
Su.04								-0.1	0.58	0.16	-0.08
Aut.04									0.544	0.17	-0.05
Sp.05										0.66	0.66
Su.05											0.16
Aut.05											
H	**Sp02**	**Aut02**	**Sp03**	**Su03**	**Aut03**	**Sp04**	**Su04**	**Au04**	**Sp05**	**Su05**	**Au05**
Sp.02		0.57	0.02	0.17	0.16	0.05	0.44	0	0.311	-0.04	0.01
Aut.02			0.37	0.75	0.81	0.58	-0.04	0.40	0.691	0.56	0.49
Sp.03				0.15	0.14	-0.06	0.217	-0.06	0.168	0.04	-0.06
Su.03					0.03	-0.02	0.633	0.07	-0.02	0.10	0.18
Aut.03						0	0.742	0.09	0.224	0.07	0.18
Sp.04							0.443	-0.02	0.033	0.01	-0.04
Su.04								0.23	0.612	0.47	0.33
Aut.04									0.083	0.03	0.01
Sp.05										0.22	0.25
Su.05											0.06
Aut.05											

Table 3.4.34. Table to show the inter-season ANOSIM analysis of the measured environmental variables for each of the sampling sites. Grey colouration indicates a statistically significant difference to the 95% significance level.

3.4.3.2. Density and family abundance

Figure 3.4.8 shows that LU appears to maintain its high density of invertebrates despite the change in compensation flows. However, there are two seasons (summer 04 and summer 05) in which the density is noticeably lower than other sampling seasons at LU. There also appeared to be a season on season decrease in the average number of individuals found a sample at RD. At the three remaining sites there appears to be no systematic change to the average density of invertebrates sampled. When the sampling seasons are compared using ANOVA (Table 3.4.35), for sites RU, RD and LU, the number of individuals captured in summer 2005 are statistically significantly lower than those found in most of the sampling seasons. At LD, the differences were much more distributed, whilst the number of

invertebrates found in spring 2002 at the H site were much higher than those found at any other time. As with the pre change samples (Table 3.4.36), there is generally a statistically significant difference between sampling sites for a given season. In general, the sites which were most similar were RU, RD and LD and H.

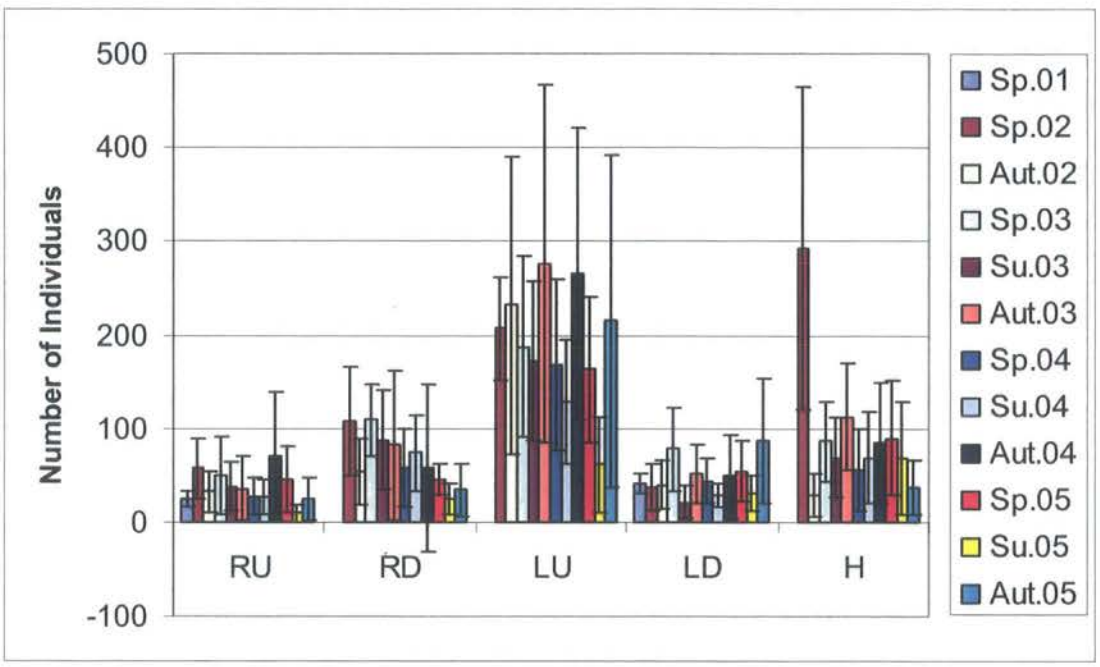

Figure 3.4.8. Figure to show the average number of individuals captured in each sample for each of the sampling sites. Error bars show the standard deviation.

RU	Sp.01	Sp02	Aut02	Sp03	Su03	Aut03	Sp04	Su04	Au04	Sp05	Su05	Au05
SP.01		8.70	0.24	0.51	2.24	0.84	0.09	0	4.2	3.16	17.38	0.02
Sp.02			4.85	2.77	2.17	2.02	6.06	6.89	0.28	0.55	19.49	6.80
Aut.02				0.09	0.66	0.25	0.04	0.13	3.20	1.56	6.43	0.22
Sp.03					0.15	0.03	0.22	0.37	2.40	0.73	4.97	0.48
Su.03						0.03	1.10	1.51	1.94	0.34	11.49	1.63
Aut.03							0.44	0.64	2.00	0.42	5.44	0.76
Sp.04								0.04	3.60	2.09	7.13	0.10
Su.04									3.90	2.53	6.44	0.02
Aut.04										0.98	7.75	4.03
Sp.05											9.99	2.64
Su.05												3.96
Aut.05												

RD	Sp.01	Sp02	Aut02	Sp03	Su03	Aut03	Sp04	Su04	Au04	Sp05	Su05	Au05
Sp.01												
Sp.02			5.08	2.06	0.01	0.02	2.86	0.57	0.93	13.63	31.38	15.45
Aut.02				11.35	3.02	1.10	0.10	1.55	0.02	0.38	5.47	1.74

Sp.03					1.03	0.90	7.76	3.79	2.78	22.39	39.77	23.94
Su.03						0.04	1.89	0.42	0.86	5.87	13.15	8.00
Aut.03							0.67	0.08	0.42	1.99	4.92	3.14
Sp.04								0.72	0.00	0.85	5.79	2.32
Su.04									0.29	4.27	12.83	6.47
Aut.04										0.17	1.31	0.61
Sp.05											8.34	1.18
Su.05												0.90
Aut.05												

LU	**Sp.01**	**Sp02**	**Aut02**	**Sp03**	**Su03**	**Aut03**	**Sp04**	**Su04**	**Au04**	**Sp05**	**Su05**	**Au05**
Sp.01												
Sp.02			0.22	0.29	1.13	1.21	1.29	7.94	1.30	2.02	37.34	0.02
Aut.02				0.56	1.08	0.31	1.19	3.54	0.24	1.48	10.38	0.05
Sp.03					0.15	1.70	0.22	2.56	1.84	0.39	13.70	0.18
Su.03						2.45	0.01	1.61	2.80	0.06	12.54	0.46
Aut.03							2.59	5.29	0.01	2.97	11.83	0.54
Sp.04								1.22	2.95	0.02	10.54	0.54
Su.04									6.57	1.14	6.44	2.05
Aut.04										3.48	15.65	0.47
Sp.05											12.14	0.70
Su.05												6.87
Aut.05												

LD	**Sp.01**	**Sp02**	**Aut02**	**Sp03**	**Su03**	**Aut03**	**Sp04**	**Su04**	**Au04**	**Sp05**	**Su05**	**Au05**
Sp.01		0.24	0.03	6.31	9.52	0.92	0.12	5.43	0.28	1.51	2.30	4.61
Sp.02			0.06	6.32	2.61	1.26	0.42	0.75	0.54	1.82	0.38	4.94
Aut.02				5.34	3.51	0.80	0.15	1.35	0.31	1.26	0.79	4.37
Sp.03					13.76	2.37	4.44	10.89	2.08	1.78	9.36	0.12
Su.03						6.93	5.97	1.46	3.26	8.05	1.40	9.13
Aut.03							0.33	4.24	0.02	0.05	3.12	2.34
Sp.04								3.07	0.09	0.67	1.92	3.68
Su.04									1.77	5.25	0.05	7.28
Aut.04										0.10	1.38	2.22
Sp.05											3.97	1.89
Su.05												6.61
Aut.05												

H	**Sp.01**	**Sp02**	**Aut02**	**Sp03**	**Su03**	**Aut03**	**Sp04**	**Su04**	**Au04**	**Sp05**	**Su05**	**Au05**
Sp.01												
Sp.02			22.81	13.36	15.79	9.73	17.44	15.41	12.62	12.15	15.03	21.02
Aut.02				13.91	6.66	18.16	3.18	5.71	6.75	8.84	3.59	0.62
Sp.03					0.87	1.32	2.32	0.68	0.00	0.03	0.63	8.61
Su.03						3.76	0.37	0.00	0.46	0.85	0.00	3.44
Aut.03							5.93	3.28	1.01	0.71	2.87	13.25
Sp.04								0.37	1.32	1.98	0.21	1.26
Su.04									0.38	0.71	0.00	3.06
Aut.04										0.03	0.38	4.41
Sp.05											0.68	5.92
Su.05												1.93
Aut.05												

Table 3.4.35. Table to show the ANOVA (F statistic) output from comparing the sampling seasons using number of individuals at each sample site. Grey colouration indicates a statistically significant difference to the 95% significance level.

Sp.01	RU	RD	LU	LD	H
RU				12.51	
RD					
LU					
LD					
H					

Sp.02	RU	RD	LU	LD	H
RU		4.03	52.32	2.50	17.75
RD			35.43	15.65	13.72
LU				76.79	2.21
LD					21.40
H					

Aut.02	RU	RD	LU	LD	H
RU		3.30	15.74	0.87	0.00
RD			11.92	1.02	3.44
LU				14.12	15.81
LD					0.95
H					

Sp.03	RU	RD	LU	LD	H
RU		22.38	23.41	6.46	9.75
RD			5.81	2.72	1.54
LU				10.78	9.36
LD					0.18
H					

Su.03	RU	RD	LU	LD	H
RU		7.31	22.65	3.08	3.73
RD			7.06	14.62	0.83
LU				30.05	11.81
LD					10.56
H					

Aut.03	RU	RD	LU	LD	H
RU		2.82	15.29	1.03	12.91
RD			8.74	1.32	0.93
LU				13.50	6.72
LD					8.74
H					

Sp.04	RU	RD	LU	LD	H
RU		4.67	22.71	2.81	3.73
RD			11.89	0.99	0.01
LU				17.38	12.12
LD					0.67
H					

SU.04	RU	RD	LU	LD	H
RU		11.81	21.63	0.21	7.03
RD			4.77	11.49	0.06
LU				21.20	5.11
LD					6.54
H					

Aut.04	RU	RD	LU	LD	H
RU		0.13	13.21	0.69	0.24
RD			13.48	0.08	0.62
LU				17.94	11.55
LD					2.13
H					

Sp.05	RU	RD	LU	LD	H
RU		0	18.72	0.29	3.88
RD			21.80	0.54	4.93
LU				16.64	5.50
LD					2.67
H					

Su.05	RU	RD	LU	LD	H
RU		7.30	10.23	10.69	9.08
RD			4.67	0.52	4.64
LU				3.23	0.07
LD					3.43
H					

Aut.05	RU	RD	LU	LD	H
RU		0.81	11.20	7.87	1.37
RD			9.95	5.19	0.07
LU				4.55	9.56
LD					4.47
H					

Table 3.4.36. Table to show the ANOVA (F statistic) output for using the number of individuals captured per sample to compare sampling sites at a given season. Grey colouration indicates a statistically significant difference to the 95% significance level.

Figure 3.4.9 shows that there appears to be a season on season decrease in the average number of families sampled at both RU and RD, with no perceptible changes at LU, LD or H. When the ANOVA analysis is considered (Table 3.4.37), it was seen that, at RU and

LU, the number of families captured in the summer 2005 season was significantly different to those captured for any other sampling season. For the other sites, the differences were more evenly distributed, with the Hipper appearing to have the most statistically significant differences. Table 3.4.38 shows that there was no clear pattern when the differences between the sites were analysed.

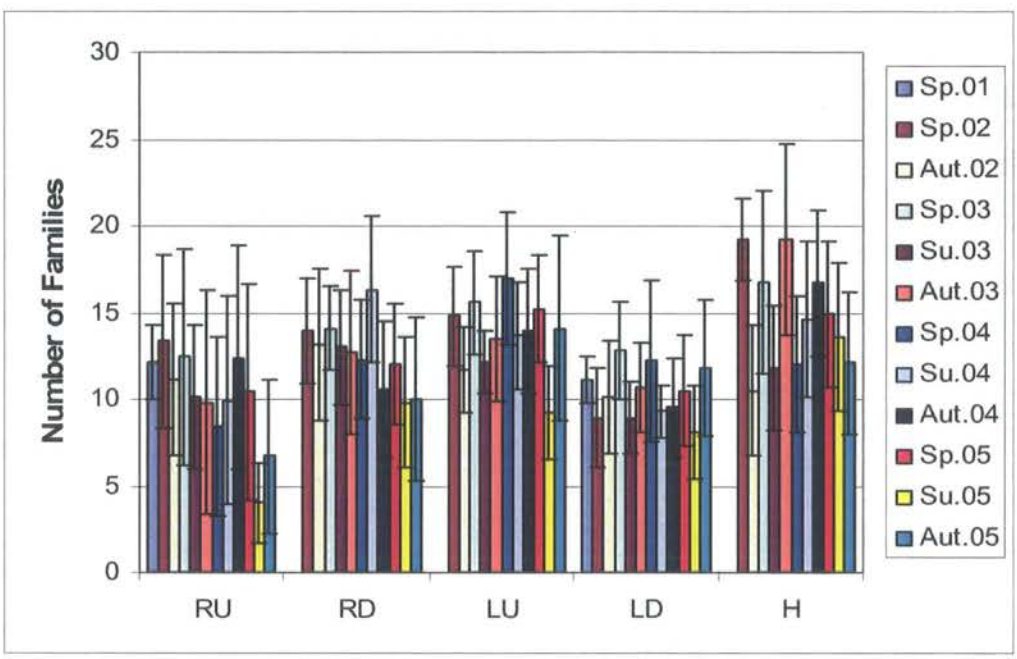

Figure 3.4.9. Graph to show the average number of families captured per sample at each site in each sampling season. Error bars show the standard deviation.

RU	Sp.01	Sp02	Aut02	Sp03	Su03	Aut03	Sp04	Su04	Au04	Sp05	Su05	Au05
SP.01		0.59	1.30	1.02	1.84	1.15	4.38	1.19	0.02	0.66	67.95	12.09
Sp.02			2.11	1.82	2.52	1.95	4.69	2.0	0.17	1.42	26.67	9.58
Aut.02				0	0	0.01	0.46	0	0.81	0.02	10.45	2.23
Sp.03					0	0	0.39	0	0.72	0.02	8.22	1.87
Su.03						0.02	0.66	0.01	0.90	0.02	16.67	3.14
Aut.03							0.29	0	0.81	0.04	7.22	1.57
Sp.04								0.36	2.33	0.61	6.08	0.62
Su.04									0.8	0..03	8.45	1.84
Aut.04										0.49	15.03	5.29
Sp.05											9.25	2.33
Su.05												2.95
Aut.05												

RD	Sp.01	Sp02	Aut02	Sp03	Su03	Aut03	Sp04	Su04	Au04	Sp05	Su05	Au05
Sp.01												
Sp.02			0.05	0.23	0.12	0.2	0.65	2.82	3.28	1.01	5.65	3.85
Aut.02				0.4	0	0.04	0.20	2.77	1.80	0.38	3.26	2.34

Sp.03					0.71	0.71	1.81	2.05	5.70	2.44	9.22	6.08
Su.03						0.03	0.21	3.77	2.16	0.43	4.05	2.74
Aut.03							0.05	3.27	1.18	0.14	2.34	1.67
Sp.04								5.37	1.05	0.04	2.39	1.57
Su.04									9.73	6.16	13.22	10.03
Aut.04										0.70	0.21	0.10
Sp.05											1.83	1.18
Su.05												0.01
Aut.05												

LU	**Sp.01**	**Sp02**	**Aut02**	**Sp03**	**Su03**	**Aut03**	**Sp04**	**Su04**	**Au04**	**Sp05**	**Su05**	**Au05**
Sp.01												
Sp.02			6.77	0.37	6.40	0.81	2.14	0.68	0.39	0.09	20.28	0.14
Aut.02				10.18	0.17	1.73	13.73	2.57	2.58	7.78	4.70	1.69
Sp.03					10.09	2.04	0.84	1.95	1.33	0.09	25.25	0.61
Su.03						1.23	13.60	2.00	2.03	7.42	8.01	1.28
Aut.03							4.51	0.02	0.06	1.29	9.24	0.09
Sp.04								4.54	3.53	1.34	27.99	1.98
Su.04									0.02	1.17	12.01	0.04
Aut.04										0.75	11.01	0.01
Sp.05											21.15	0.32
Su.05												6.78
Aut.05												

LD	**Sp.01**	**Sp02**	**Aut02**	**Sp03**	**Su03**	**Aut03**	**Sp04**	**Su04**	**Au04**	**Sp05**	**Su05**	**Au05**
Sp.01		4.85	0.79	3.01	7.81	0.19	0.51	7.88	2.47	0.3	9.90	0.29
Sp.02			0.76	9.61	0.00	2.19	3.65	0.15	0.22	1.41	0.42	3.60
Aut.02				3.94	0.95	0.21	1.36	0.49	0.19	0.08	2.22	1.11
Sp.03					12.62	3.06	0.12	12.29	6.71	2.97	14.78	0.43
Su.03						2.95	4.18	0.24	0.28	1.78	0.55	4.29
Aut.03							0.79	2.2	0.95	0.02	4.87	0.55
Sp.04								3.51	2.41	0.91	5.81	0.04
Su.04									0.04	1.17	1.52	3.57
Aut.04										0.54	1.25	2.22
Sp.05											3.34	0.67
Su.05												6.08
Aut.05												

H	**Sp.01**	**Sp02**	**Aut02**	**Sp03**	**Su03**	**Aut03**	**Sp04**	**Su04**	**Au04**	**Sp05**	**Su05**	**Au05**
Sp.01												
Sp.02			38.25	1.86	29.48	0	0	8.14	2.63	7.98	13.31	22.63
Aut.02				9.19	0.63	17.20	0.77	4.90	11.99	6.15	3.01	0.84
Sp.03					5.91	1.08	5.12	0.92	0	0.72	2.11	4.77
Su.03						12.79	0.01	2.37	7.78	3.17	1.06	0.03
Aut.03							11.45	4.21	1.30	3.90	6.56	10.84
Sp.04								1.90	6.62	2.57	0.77	0
Su.04									1.15	0.02	0.26	1.70
Aut.04										0.91	2.68	6.12
Sp.05											0.48	2.31
Su.05												0.65
Aut.05												

Table 3.4.37. Table to show the ANOVA (F statistic) output using the number of families sampled to compare each the sampling seasons at each site. Grey colouration indicates a statistically significant difference to the 95% significance level.

Sp.01	RU	RD	LU	LD	H
RU				1.56	
RD					
LU					
LD					
H					

Sp.02	RU	RD	LU	LD	H
RU		0	0.46	5.75	9.43
RD			0.93	11.65	20.58
LU				21.38	13.92
LD					76.69
H					

Aut.02	RU	RD	LU	LD	H
RU		1.98	0.82	0	0.06
RD			0.78	3	2.03
LU				1.53	0.72
LD					0.06
H					

Sp.03	RU	RD	LU	LD	H
RU		3.77	6.59	1.69	6.75
RD			1.52	1.24	2.01
LU				4.70	0.33
LD					4.28
H					

Su.03	RU	RD	LU	LD	H
RU		2.97	1.96	0.67	0.96
RD			0.57	10.89	0.6
LU				13.59	0.06
LD					4.90
H					

Aut.03	RU	RD	LU	LD	H
RU		1.33	2.52	0.17	12.36
RD			0.19	1.41	8.17
LU				4.04	7.61
LD					19.72
H					

Sp.04	RU	RD	LU	LD	H
RU		3.93	17.97	2.98	3.08
RD			8.35	0	0.03
LU				6.37	8.40
LD					0.01
H					

SU.04	RU	RD	LU	LD	H
RU		7.60	3.16	0.09	3.92
RD			2.47	24.47	0.76
LU				16.41	0.27
LD					12.48
H					

Aut.04	RU	RD	LU	LD	H
RU		0.56	0.41	1.67	3.09
RD			3.84	0.50	11.07
LU				9.10	2.55
LD					19.55
H					

Sp.05	RU	RD	LU	LD	H
RU		0.50	4.71	0	3.58
RD			4.66	1.01	2.84
LU				11.16	0.03
LD					7.04
H					

Su.05	RU	RD	LU	LD	H
RU		17.44	21.81	13.64	40.19
RD			0.17	1.35	4.51
LU				0.83	7.71
LD					12.08
H					

Aut.05	RU	RD	LU	LD	H
RU		2.63	11.49	7.46	8.08
RD			3.37	0.87	1.15
LU				1.22	0.90
LD					0.03
H					

Table 3.4.38. Table to show the ANOVA (F statistic) comparison of sampling sites for a given season using the number of families in a sample. Grey colouration indicates a statistically significant difference to the 95% significance level.

3.4.3.3. Diversity Indices and Family Presence/Absence

This section assesses whether the diversity of the macroinvertebrate populations has been changed by the alteration in compensation flow regimes. This will be addressed (as in the previous sections) using Shannon diversity. Figure 3.4.10 shows that there is no obvious difference between the pre and post change samples in terms of Shannon diversity. The size of the error bars that show the standard deviation of the Shannon diversity again (Section 3.4.2) mean that little difference can be found between the sampling sites in any of the sampling seasons.

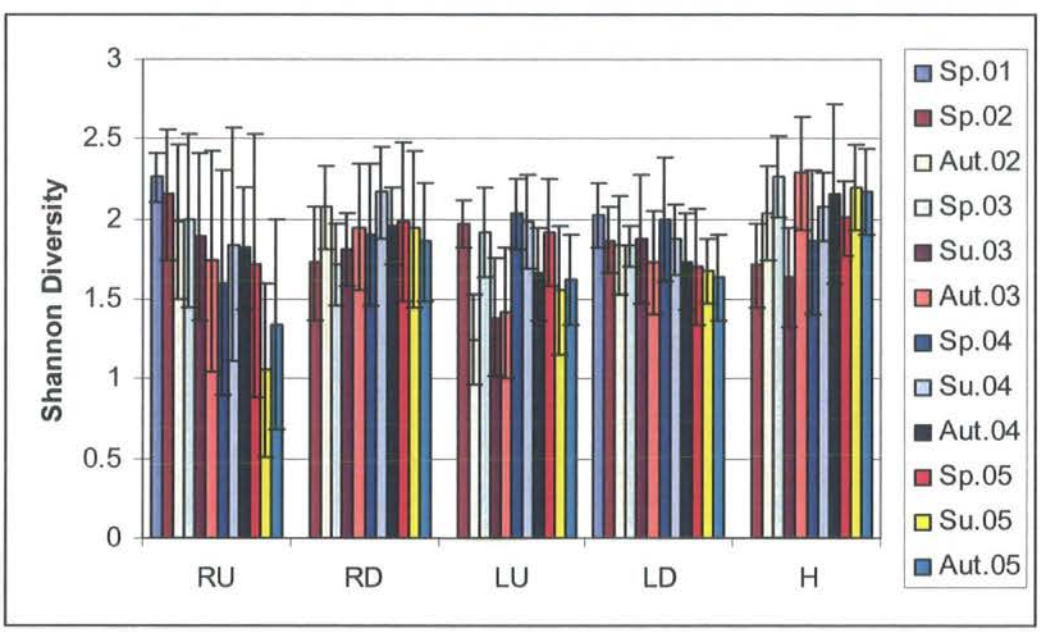

Figure 3.4.10. Figure to show the average Shannon diversity for each site for each sampling season. Standard deviation shown in the error bars.

For RU, the diversity was lower in the spring and summer 2005 samples than it was for any other sampling season. Table 3.4.39 shows that when the different sampling seasons were compared using the ANOVA test, many of the statistically significant differences for the RU site were provided by these two samples. There appears to be no great change in Shannon diversity values following the change in compensation flows. The RD site on the Rivelin, also has relatively similar diversities in both the post-change samples and pre-change samples. Table 3.4.39 shows that this site has fewer statistically significant

differences than RU in both the pre and post change samples perhaps indicating a more stable invertebrate community (perhaps dampened by less hydrologic variation?).

Table 3.4.39 shows that the average diversity of the invertebrates at LU was relatively similar in the pre and post change samples. In the pre-change samples, the lowest diversity at LU was found in the two autumn samples and the summer samples, and this pattern is largely reproduced in the post-change samples, with the lowest diversity found in autumn 2004, summer 2005 and autumn 2005. Table 3.4.39 shows that LU had the most statistically significant differences between the sampling seasons than any other site, highlighting the inter-season variability in diversity. It must be noted that many of these differences involved autumn 2002 which had a very low diversity. However, many of the statistically significant differences also involved the summer and autumn 2005 samples, which indicates that the diversity sampled was lower than previously sampled. There were no differences between autumn 2003 and any of the later autumn samples.

The diversity of invertebrates at the LD site showed a season on season decrease in the post change samples. When Table 3.4.39 is considered it shows that there is generally very little difference between the sampling seasons in terms of the Shannon diversity, apart from when the spring 2001 data is involved. It is curious that each of the remaining differences involved either or both of summer and autumn 2005. This tallies with the findings from LU indicating that the diversity sampled in summer and autumn 2005 was lower than in many of the previous sampling seasons.

RU	Sp.01	Sp02	Aut02	Sp03	Su03	Aut03	Sp04	Su04	Au04	Sp05	Su05	Au05
SP.01		0.26	3.63	4.47	4.67	5.39	8.37	3.16	11.27	4.22	45.09	18.8
Sp.02			2.0	1.68	1.87	2.93	4.9	1.61	3.85	2.51	25.73	11.42
Aut.02				0.09	0.12	0.02	0.3	0.03	0.02	0.04	5.9	1.92
Sp.03					0.32	0	0.12	0.13	0.13	0	4.67	1.33
Su.03						0.3	1.06	0.03	0.11	0.32	12.13	4.19
Aut.03							0.19	0.11	0.1	0.01	5.97	1.74
Sp.04								0.56	0.74	0.11	3.77	0.74
Su.04									0.01	0.14	7.46	2.63
Aut.04										0.13	13.09	4
Sp.05											4.4	1.25
Su.05												1.12
Aut.05												

RD	Sp.01	Sp02	Aut02	Sp03	Su03	Aut03	Sp04	Su04	Au04	Sp05	Su05	Au05
Sp.01												

Sp.02			6.89	0	0.53	1.94	1.1	10.11	3.22	1.96	1.42	0.83
Aut.02				9.82	6.04	0.76	1.16	0.56	1.14	0.27	0.6	2.35
Sp.03					0.74	2.38	1.29	14.05	4.64	2.26	1.64	1.02
Su.03						0.91	0.34	9.79	1.97	1.02	0.6	0.14
Aut.03							0.06	2.08	0	0.04	0	0.26
Sp.04								2.55	0.12	0.15	0.04	0.05
Su.04									3.19	1.02	1.61	4.45
Aut.04										0.03	0.01	0.49
Sp.05											0.04	0.41
Su.05												0.18
Aut.05												

LU	**Sp.01**	**Sp02**	**Aut02**	**Sp03**	**Su03**	**Aut03**	**Sp04**	**Su04**	**Au04**	**Sp05**	**Su05**	**Au05**
Sp.01												
Sp.02			51.60	0.25	21.26	16.34	0.58	0.02	9.16	0.22	9.56	11.8
Aut.02				28.69	0.95	1.23	48.38	32.6	10.47	23.29	4.08	8.88
Sp.03					13.07	10.34	1.03	0.26	4.19	0	5.6	5.56
Su.03						0.03	22.45	15.83	3.31	11.10	0.96	2.51
Aut.03							17.82	12.75	2.33	8.91	0.59	1.69
Sp.04								0.17	10.65	0.88	11.11	13.15
Su.04									6.22	0.25	7.52	7.84
Aut.04										3.35	0.44	0.08
Sp.05											4.78	4.45
Su.05												0.18
Aut.05												

LD	**Sp.01**	**Sp02**	**Aut02**	**Sp03**	**Su03**	**Aut03**	**Sp04**	**Su04**	**Au04**	**Sp05**	**Su05**	**Au05**
Sp.01		2.8	2.69	6.36	1.11	5.98	0.05	2.6	6.41	6	15.24	13.6
Sp.02			0.09	0.21	0	1.31	0.77	0	1.35	1.6	4.49	4.76
Aut.02				0	0.07	0.55	1.02	0.09	0.54	0.79	1.91	2.42
Sp.03					0.1	0.89	1.51	0.2	0.91	1.17	4.46	4.5
Su.03						0.8	0.44	0	0.8	1.04	2.02	2.52
Aut.03							2.7	1.27	0	0.04	0.21	0.51
Sp.04								0.74	2.76	2.99	5.3	5.77
Su.04									1.31	1.56	4.23	4.56
Aut.04										0.05	0.27	0.61
Sp.05											0.04	0.21
Su.05												0.15
Aut.05												

H	**Sp.01**	**Sp02**	**Aut02**	**Sp03**	**Su03**	**Aut03**	**Sp04**	**Su04**	**Au04**	**Sp05**	**Su05**	**Au05**
Sp.01												
Sp.02			7.19	23.55	0.33	17.14	0.83	12.18	5.26	7.12	17.34	15.49
Aut.02				3.5	8.97	2.96	1.12	0.13	0.36	0.07	1.67	1.16
Sp.03					24.58	0.03	6.18	3.14	0.31	5.51	0.33	0.64
Su.03						18.87	1.66	13.81	6.61	8.92	18.85	17.09
Aut.03							5.56	2.5	0.39	4.27	0.4	0.68
Sp.04								1.95	1.72	0.83	4.2	3.57
Su.04									0.17	0.52	1.21	0.73
Aut.04										0.61	0.04	0.01
Sp.05											2.87	2.13
Su.05												0.05
Aut.05												

Table 3.4.39. Table to show the output from ANOVA (F statistic) comparisons of the Shannon diversity for each sampling season at each sampling site. Grey colouration indicates a statistically significant difference to the 95% significance level.

The RD site only had a higher average Shannon diversity than RU in two out of the five pre-change samples, whereas RD had a higher diversity than RU in all six of the post-change samples. This perhaps suggests that the site at which greater variability occurs has swapped due to the change in compensation flows, i.e. the increased discharge dampens variability at the uppermost site, whilst the tributaries still provide sufficient hydrologic variation to the downstream site. However, Table 3.4.40 shows that the only time when RD is statistically significantly (ANOVA, 95%) more diverse than RU is in the summer and autumn 2005 samples.

In the pre-change samples it was clear that LU displayed the greatest amount of variation of average diversity, whilst LD had more stable values. The two sites were statistically significantly different in autumn 2002 and summer 2003 (Table 3.4.40). The LD post change data appeared to show a greater variation in average Shannon diversity (with its year on year decrease). Because of this, the two sites were not statistically significantly different in terms of Shannon diversity.

In the pre-change samples H displayed a very variable average of Shannon diversity. In the post-change samples, this site displayed a lot less variation than in the pre change samples, and also had the highest Shannon values for four out of six sampling seasons, and was second highest for the one other.

Sp.01	RU	RD	LU	LD	H
RU				8.46	
RD					
LU					
LD					
H					

Sp.02	RU	RD	LU	LD	H
RU		6.92	2.07	4.05	8.52
RD			4.55	1.47	0
LU				1.48	7.63
LD					2.3
H					

Aut.02	RU	RD	LU	LD	H
RU		1.28	4.28	0.04	0.94
RD			46.64	3.59	0.1
LU				20.17	38.66
LD					2.23
H					

Sp.03	RU	RD	LU	LD	H
RU		0	0.58	0.21	4.3
RD			2.89	1.72	23.62
LU				0.76	8.28
LD					22.81
H					

Su.03	RU	RD	LU	LD	H
RU		0.19	6.09	0	1.7
RD			9.47	0.22	2.02
LU				8.05	2.66
LD					2.24
H					

Aut.03	RU	RD	LU	LD	H
RU		0.68	1.59	0	4.96
RD			8.7	1.78	4.12
LU				3.64	25.85
LD					13.33
H					

Sp.04	RU	RD	LU	LD	H
RU		1.29	3.46	2.38	0.95
RD			0.73	0.24	0.04
LU				0.09	1.23
LD					0.51
H					

Su.04	RU	RD	LU	LD	H
RU		1.74	0.34	0.01	0.98
RD			1.93	6.65	0.62
LU				0.95	0.64
LD					4.5
H					

Aut.04	RU	RD	LU	LD	H
RU		0.9	1.12	0.3	2.48
RD			6.16	3.23	1.09
LU				0.34	6.26
LD					4.4
H					

Sp.05	RU	RD	LU	LD	H
RU		0.78	0.51	0	1.16
RD			0.13	2.07	0.02
LU				1.85	0.48
LD					4.87
H					

Su.05	RU	RD	LU	LD	H
RU		14.54	5.48	11.42	35.44
RD			3.72	2.5	2.14
LU				0.74	18.07
LD					24.64
H					

Aut.05	RU	RD	LU	LD	H
RU		4.81	1.56	1.75	13.92
RD			2.64	2.44	4.8
LU				0.01	19.93
LD					20.19
H					

Table 3.4.40. Table to show the ANOVA (F statistic) output comparing the Shannon diversity of each sampling site for each sampling season. Grey colouration indicates a statistically significant difference to the 95% significance level.

A further look at any potential changes in macroinvertebrate communities can be gained by looking at the family lists for each site and comparing the pre and post change lists. The most noticeable change at RU was that Polycentropodidae was found in 30 out of 62 samples in the pre-change samples, but was not present in any of the post-change samples.

A number of other families were also found to be present in the pre-change samples but were absent from the post-change samples: Glossosomatidae, Goeridae, Dixidae, and Ostracoda. However, these families were only present in a small number of samples. There were also a number of families which were present in the post-change samples but absent from the pre-change samples: Syrphidae (0-1); Hydrophilidae (0-5); Scirtidae (0-3); Erpobedellidae (0-1); Asellidae (0-5); Lymnaeidae (0-1); Zontidae (0-1); Dolichopodidae (0-1); Curculionidae (0-1); Chyorbidae (0-1); Collembola (0-1) and Valvatidae (0-1). What is interesting about these species is that: (1), they were only found in low numbers of samples, indicating that these species have not developed into dominant species; and (2) that each of the families only found in the post-change samples have preferences for slow and standing water. This perhaps indicates a decreased velocity at RU despite the increase in discharge, which could be due to a roughness submergence effect.

When the same analysis was performed on RD, a large reduction in the number of samples in which Polycentropodidae was present was seen (24 – 1). This matches the pattern seen at RU. Two families were found to be present only in the pre-change samples: Taeniopterigidae (2-0); Carabidae (1-0). However, the low sample numbers indicate that this is not a major alteration to the invertebrate community. There was a noticeable increase in the number of samples with Glossiphonidae present (2-15) in the post-change samples. A number of families were also found to be present only in the post change samples: Hydrophilidae (0-1); Beraeridae (0-1); Stratiomyidae (0-1); Muscidae (0-4); Asellidae (0-10); Crangonyctidae (0-1); Collembola (0-1); Entombryoidea sp. (0-2); Sminthuridae (0-1) and Oribatei (0-3). What is interesting from the above is that again, the 'new' families are generally only found in a small number of samples. However, the presence of Asellidae in 10 post-change samples is interesting as a similar pattern was found for this family at RU. In the two Rivelin sites, the main changes appear to have been the large decrease in abundance of Polycentropodidae and an increase in the number of peripheral species in the post-change samples.

At LU, there is no large decrease in the number of samples in which Polycentropodidae is present. However, there were a number of families which were present only in pre-change samples: Hydraenidae (1-0); Staphylinidae (1-0); Hydrobiidae (1-0); Collembola (1-0) and Oribatei (1-0). The low numbers of samples perhaps indicate that there has been little

change to the LU community in terms of its species list. A number of families were found to be present in more post-change samples than pre-change samples: Ceratopogonidae (5-20); Glossiphonnidae (5-23); and Sialidae (1-7). There were also a number of families which were found to be present only in the post-change samples: Ephemeridae (0-1); Perlodidae (0-1); Philopotomidae (0-1); Psychodidae (0-1); Syrphidae (0-1); Bibionidae (0-1); Zontidae (0-1); Hydridae (0-2); Corixidae (0-1) and Veliidae (0-1). The new species pretty much appear to be Diptera or low importance taxa (in BMWP terms).

When the samples collected at LD are considered it can be seen that there is no large decrease in the number of samples in which Polycentropodidae is present. There were a number of families which were present only in the pre-change samples: Chloroperlidae (8-0); Perlodidae (3-0); Glossosomatidae (3-0); Lepidosomatidae (4-0); Psychodidae (1-0); Stratiomyidae (1-0); Syrphidae (1-0) and Daphniidae (1-0). Only four families were found to be present only in the post-change samples: Hydrophilidae (0-9); Scirtidae (0-1); Chaorbidae (0-1); and Chrysopidae (0-2). This indicates a relatively unchanged macroinvertebrate population at this site, as compared with the upstream Loxley site displayed greater variation, with the emergence to importance of three new species.

The Hipper site (H) provides a comparison against which to compare the Rivelin and Loxley. Over the same sampling periods, there was no reduction in the number of samples in which Polycentropodidae was found. A number of families were found to be present only in the pre-change samples: Beraeridae (1-0); Chyoridae (7-0); Curculionidae (1-0); and Collembola (1-0). Also, a number of families were found in only the post-change samples: Hydrophilidae (0-3); Faniidae (0-1); Stratiomyidae (0-1); Culcidae (0-2); Dolichopodidae (0-2); Pescicolidae (0-1); Corixidae (0-1); Mesovelidae (0-1); Geriidae (0-1); and Veliidae (0-1). Again, most of the changes to the HSP invertebrate list appear to be in peripheral species.

3.4.3.4. Comparison of species similarity within and between sites

Multivariate techniques may also be of use in assessing the impacts of compensation flow changes, and it is first of interest to assess whether the sampling sites remain different. Table 3.4.41 shows that each of the sites are statistically significantly different for each of

the post-change sampling seasons. It was also found that each of the rivers were statistically significantly different using the ANOSIM routine. This indicates that the Rivelin and Loxley have not become more similar through the altering the compensation flow regime.

Sp.01	RU	RD	LU	LD	H
RU				0.58	
RD					
LU					
LD					
H					

Sp.02	RU	RD	LU	LD	H
RU		0.21	0.40	0.28	0.43
RD			0.61	0.57	0.49
LU				0.69	0.98
LD					0.90
H					

Aut.02	RU	RD	LU	LD	H
RU		0.25	0.69	0.55	0.58
RD			0.80	0.42	0.50
LU				0.54	0.96
LD					0.87
H					

Sp.03	RU	RD	LU	LD	H
RU		0.29	0.55	0.56	0.32
RD			0.82	0.76	0.53
LU				0.72	0.71
LD					0.77
H					

Su.03	RU	RD	LU	LD	H
RU		0.59	0.89	0.67	0.71
RD			0.87	0.66	0.3
LU				0.78	0.90
LD					0.80
H					

Aut.03	RU	RD	LU	LD	H
RU		0.30	0.53	0.39	0.51
RD			0.78	0.40	0.47
LU				0.56	0.89
LD					0.61
H					

Sp.04	RU	RD	LU	LD	H
RU		0.45	0.74	0.55	0.50
RD			0.93	0.60	0.59
LU				0.52	0.94
LD					0.80
H					

SU.04	RU	RD	LU	LD	H
RU		0.31	0.52	0.52	0.47
RD			0.82	0.77	0.52
LU				0.64	0.95
LD					0.89
H					

Aut.04	RU	RD	LU	LD	H
RU		0.35	0.58	0.70	0.60
RD			0.90	0.78	0.72
LU				0.83	0.96
LD					0.94
H					

Sp.05	RU	RD	LU	LD	H
RU		0.48	0.66	0.60	0.59
RD			0.86	0.42	0.51
LU				0.63	0.91
LD					0.68
H					

Su.05	RU	RD	LU	LD	H
RU		0.44	0.44	0.58	0.70
RD			0.76	0.74	0.57
LU				0.61	0.862
LD					0.88
H					

Aut.05	RU	RD	LU	LD	H
RU		0.17	0.62	0.66	0.55
RD			0.7	0.74	0.33
LU				0.66	0.92
LD					0.92
H					

Table 3.4.41. Table to show the ANSOIM output comparing each of the sites for each of the sampling seasons using the Surber sample biotic data. Grey colouration indicates a statistically significant difference to the 95% significance level.

As it is established that the sites remain different, it is of interest to asses any inter-seasonal differences for each of the sites. This will give an indication as to whether the invertebrate community has changed because of the alteration of the compensation flows. Table 3.4.42 shows that the invertebrate communities at the different sampling seasons at each of the sites are generally statistically significantly different. It was seen in section 4.1 (Table 3.4.9) that there was only one set of pre-change samples which were similar (LD; autumn 2002 and autumn 2003). There were however, more similarities presented in the post-change data, with the RU site having a number of similar populations among the post-change data. The LU site, which had no similarities in the pre-change data, had three sets of samples where no difference could be detected: autumn 2003-autumn 2004; autumn 2003 – autumn 2005; autumn 2004 – autumn 2005. Every sampling season was different to each other at the RD and H sites.

It is interesting that the two more upstream sites on the Rivelin and Loxley were the sites which had fewer statistically significant differences than the two downstream sites, with these similarities mostly occurring in the post-change data. For the RU site there is a correspondence of the biotic and abiotic data as Table 3.4.42 shows that there are fewer statistically significant differences between sampling seasons in both the abiotic and biotic data in the post-change samples. At the LU site, there are more differences between the environmental variables in the post-change data, but there were three sets of seasons which were not statistically significant. The detachment of environmental variables and the biotic community at LU, first discussed in section 4.1, appears to continue at least to some extent under the post-change conditions.

When the multivariate output from Table 3.4.41 is compared to the ANOVA analysis of the Shannon diversity index (Table 3.4.39) it can be seen that the Shannon diversity index is not able to detect differences between sampling seasons at a site.

RU	Sp01	Sp02	Aut02	Sp03	Su03	Aut03	Sp.04	Su 04	Aut04	Sp 05	Su 05	Aut 05
Sp.01		0.36	0.27	0.51	0.58	0.27	0.43	0.36	0.37	0.42	0.41	0.3
Sp.02			0.32	0.15	0.44	0.37	0.49	0.25	0.39	0.37	0.53	0.39
Aut.02				0.44	0.30	0.12	0.29	0.13	0.15	0.29	0.32	0.21
Sp.03					0.36	0.35	0.24	0.26	0.42	0.23	0.58	0.39
Su.03						0.26	0.08	0.3	0.45	0.22	0.6	0.41
Aut.03							0.17	0.27	0.07	0.27	0.26	0.09
Sp.04								0.18	0.25	0.05	0.48	0.27
Su.04									0.1	0.1	0.22	0.11
Aut.04										0.16	0.19	0.03
Sp.05											0.41	0.24
Su.05												0.08
Aut.05												

RD	Sp01	Sp02	Aut02	Sp03	Su03	Aut03	Sp.04	Su 04	Aut04	Sp 05	Su 05	Aut 05
Sp.01												
Sp.02			0.61	0.18	0.64	0.67	0.65	0.69	0.73	0.47	0.68	0.64
Aut.02				0.80	0.49	0.26	0.55	0.47	0.30	0.61	0.31	0.21
Sp.03					0.57	0.79	0.65	0.81	0.84	0.43	0.76	0.80
Su.03						0.45	0.48	0.17	0.48	0.34	0.44	0.56
Aut.03							0.30	0.30	0.13	0.45	0.25	0.24
Sp.04								0.43	0.30	0.29	0.44	0.43
Su.04									0.27	0.47	0.33	0.45
Aut.04										0.42	0.1	0.19
Sp.05											0.34	0.46
Su.05												0.19
Aut.05												

LU	Sp01	Sp02	Aut02	Sp03	Su03	Aut03	Sp.04	Su 04	Aut04	Sp 05	Su 05	Aut 05
Sp.01												
Sp.02			0.74	0.19	0.83	0.79	0.69	0.73	0.87	0.55	0.57	0.83
Aut.02				0.49	0.66	0.16	0.63	0.85	0.32	0.45	0.57	0.35
Sp.03					0.62	0.52	0.39	0.58	0.56	0.16	0.46	0.61
Su.03						0.62	0.63	0.45	0.76	0.42	0.23	0.58
Aut.03							0.35	0.75	0.10	0.39	0.54	0
Sp.04								0.72	0.54	0.21	0.56	0.42
Su.04									0.77	0.63	0.39	0.67
Aut.04										0.40	0.65	0.09
Sp.05											0.37	0.41
Su.05												0.53
Aut.05												

LD	Sp01	Sp02	Aut02	Sp03	Su03	Aut03	Sp.04	Su 04	Aut04	Sp 05	Su 05	Aut 05
Sp.01		0.66	0.53	0.87	0.85	0.75	0.71	0.90	0.92	0.43	0.81	0.96
Sp.02			0.59	0.37	0.78	0.72	0.5	0.73	0.92	0.32	0.71	0.92
Aut.02				0.73	0.54	0.05	0.53	0.86	0.26	0.56	0.74	0.38
Sp.03					0.76	0.78	0.41	0.73	0.96	0.23	0.81	0.94
Su.03						0.35	0.52	0.33	0.49	0.62	0.43	0.47
Aut.03							0.36	0.79	0.13	0.59	0.75	0.24
Sp.04								0.59	0.59	0.25	0.66	0.60
Su.04									0.69	0.48	0.24	0.73
Aut.04										0.79	0.61	0.17

Sp.05											0.60	0.82
Su.05												0.53
Aut.05												
H	**Sp01**	**Sp02**	**Aut02**	**Sp03**	**Su03**	**Aut03**	**Sp.04**	**Su 04**	**Aut04**	**Sp 05**	**Su 05**	**Aut 05**
Sp.01												
Sp.02			0.86	0.31	0.7	0.75	0.86	0.88	0.83	0.58	0.69	0.86
Aut.02				0.75	0.71	0.53	0.68	0.65	0.46	0.70	0.44	0.35
Sp.03					0.49	0.46	0.57	0.67	0.62	0.24	0.51	0.77
Su.03						0.64	0.63	0.47	0.67	0.43	0.2	0.71
Aut.03							0.52	0.66	0.34	0.52	0.42	0.42
Sp.04								0.52	0.62	0.48	0.45	0.69
Su.04									0.51	0.62	0.18	0.65
Aut.04										0.50	0.23	0.27
Sp.05											0.42	0.72
Su.05												0.20
Aut.05												

Table 3.4.42. Table to show the ANOSIM output comparing each of the sampling seasons at each of the sampling sites using the Surber sample biotic data. Grey colouration indicates a statistically significant difference to the 95% significance level.

3.4.3.5. Feeding guilds

Figure 3.4.11 shows that for RU, the proportion of feeding guilds were relatively consistent through each of the pre-change sampling seasons. What is very noticeable is the much greater amount of variation in the post-change guild proportions. The proportion of invertebrates which are predators appears to have decreased in the post-change samples with the exception of spring 2004 and spring 2005. As in the pre-change data, the percentage of invertebrates which are shredders remains relatively constant, with the variations driven by changes in the remaining guilds, particularly collector-gatherer. There appears to be little seasonal pattern in the variation of the feeding guilds at RU.

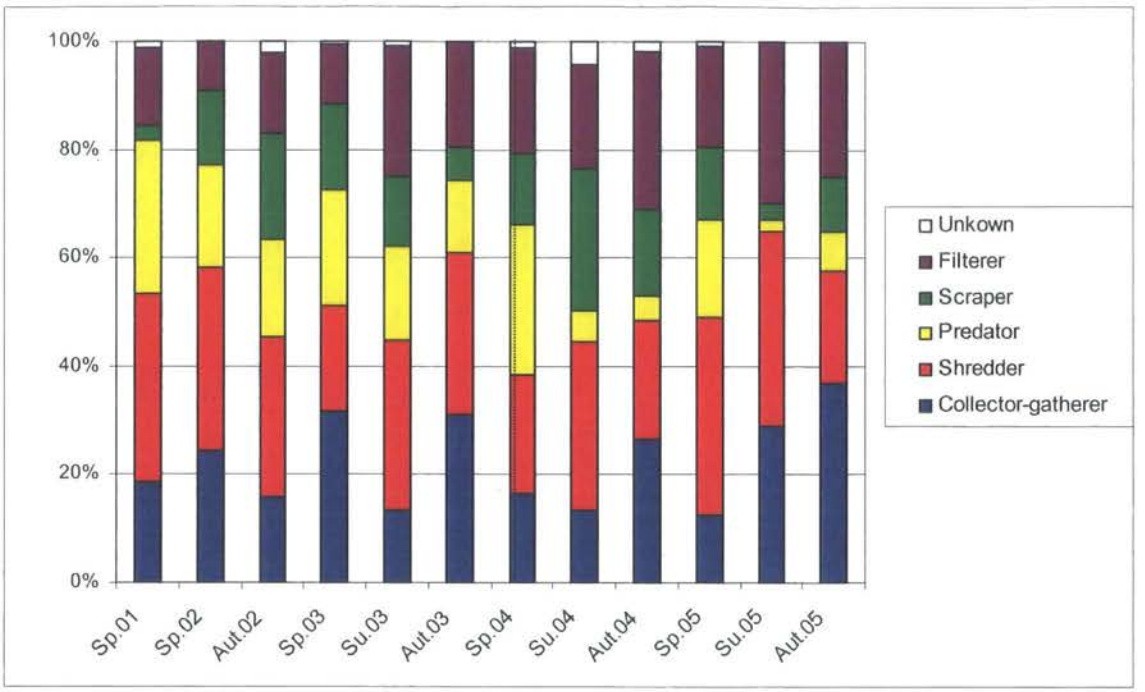

Figure 3.4.11. Figure to show the percentage of invertebrates which belong to each feeding guild for each sampling season at RU.

The feeding guild proportions at RD displayed a great deal of variation in the pre-change samples (Table 3.4.12), much of this was down to the large collector-gatherer populations in spring 2002 and 2003, which were not found in any other sampling seasons. The two autumn sets of samples also appeared to be relatively similar. In the post-change data, the populations are not dominated by collector gatherers in the same way as in spring 2002 and 2003. This leads to much less variation in the proportions of feeding guilds. There are no obvious seasonal patterns present in the post-change data in Figure 3.4.12.

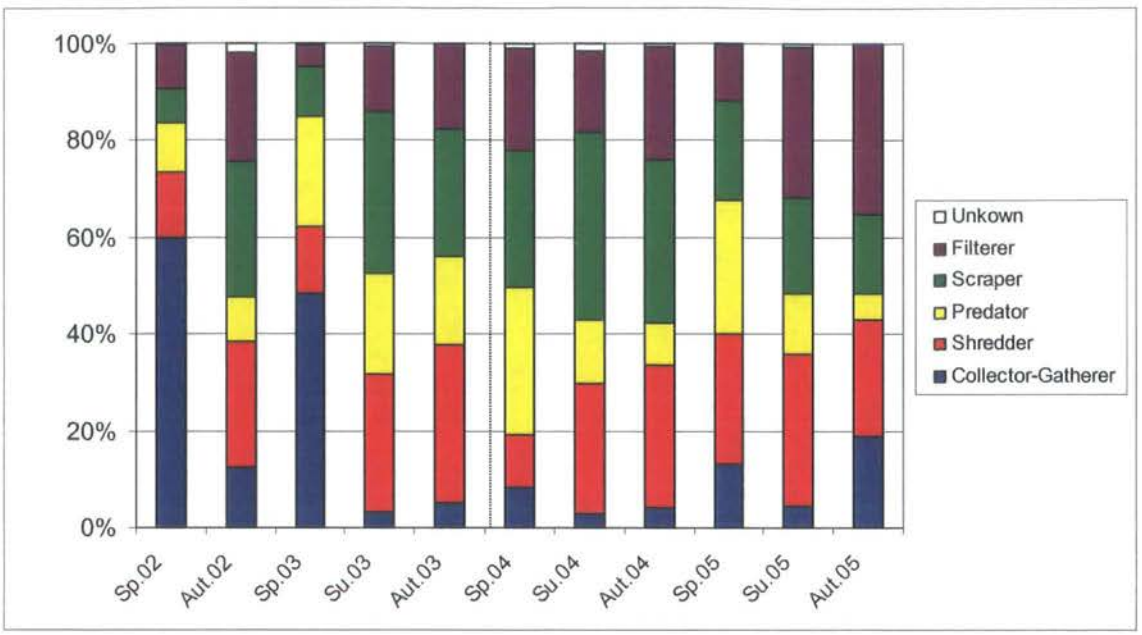

Figure 3.4.12. Figure to show the percentage of invertebrates which belong to each feeding guild for each sampling season at RD.

Figure 3.4.13 shows that for LU, in the pre-change samples, the populations were largely dominated by shredder and filterer invertebrates, with the highest proportions of collector-gatherers being found in the spring samples. The dominance of the shredder and filterer guilds continues with the post-change samples, apart from the spring 2004 and spring 2005 samples where the percentage of predators is higher than the percentage of filterer invertebrates. There is no increase in collector-gatherer invertebrates in the spring samples post-change, and that the samples are dominated by shredding invertebrates in all seasons apart from spring 2004, autumn 2004 and spring 2005. The small proportion of scraper invertebrates persisted. These results indicate a community which is relatively stable in terms of the feeding guilds present, and one which has not been changed by the alteration in compensation flows.

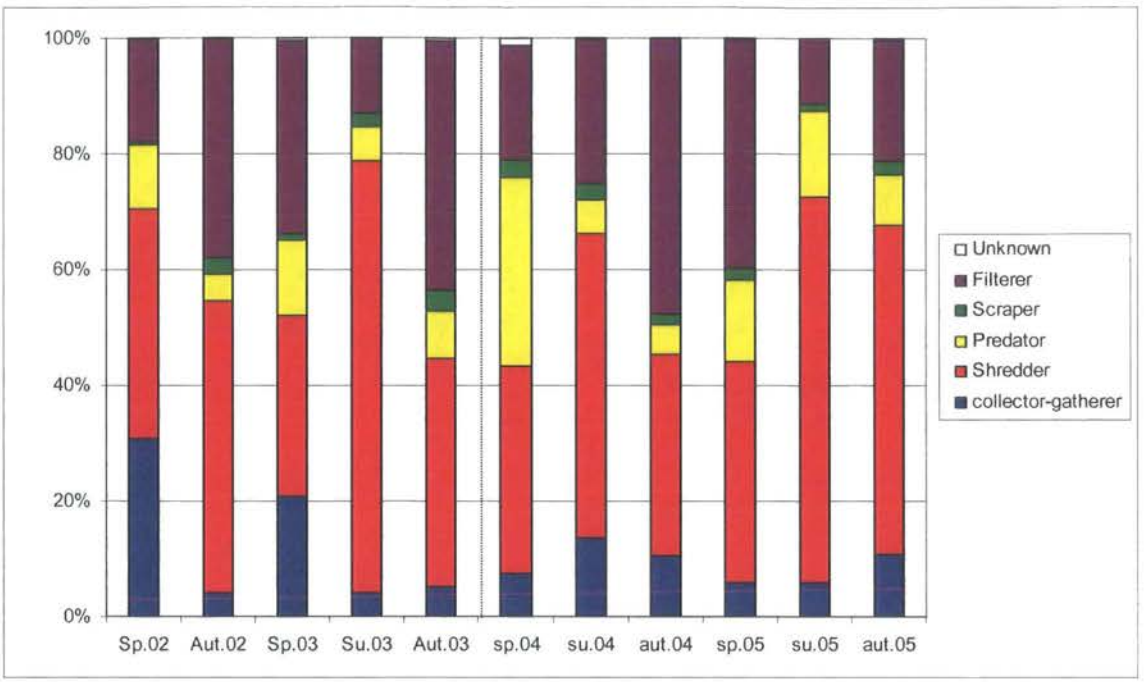

Figure 3.4.13. Figure to show the percentage of invertebrates which belong to each feeding guild for each sampling season at LU.

Figure 3.4.14 shows that at the LD site for the pre-change samples, each of the samples had a large proportion of shredder invertebrates. The largest proportion of collector-gatherer invertebrates were found in spring 2001 and spring 2002. Figure 3.4.14 shows that the two pre-change autumn samples had the lowest proportions of predator invertebrates of any of the pre-change sampling seasons, with this decrease being compensated for with an increase in the percentage of filterers. In the post-change samples, the importance of the shredder guild to this site is not diminished, and the proportion of collector-gatherer invertebrates remains relatively low. The lowest post-change percentages of scrapers and highest post-change percentages of predators were found in the two spring samples. However, the two post-change summer samples, contained the highest proportion of scraper invertebrates found in any of the sampling seasons. The proportion of filtering invertebrates appeared to be slightly smaller than for the pre-change samples. Again, the highest percentages for this guild were found in the autumn samples. These results, coupled with those from LU, indicate that the invertebrate communities of the Loxley have been relatively unchanged by the alteration in compensation flow, at least in terms of the proportions of feeding guilds present.

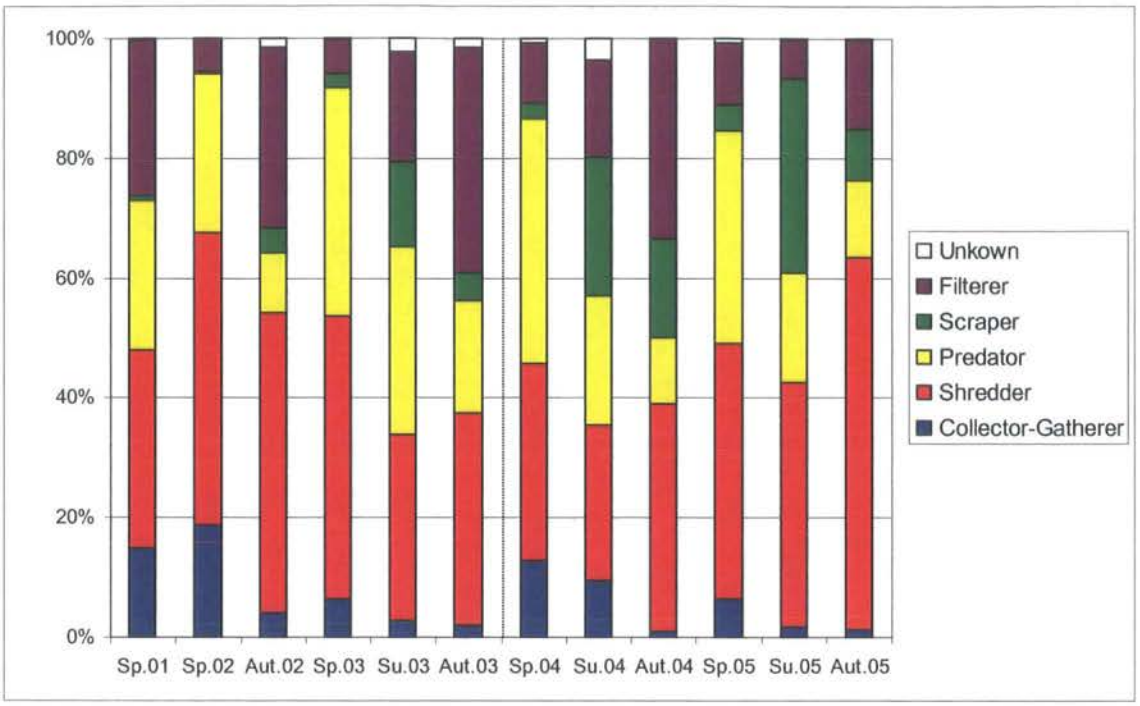

Figure 3.4.14. Figure to show the percentage of invertebrates which belong to each feeding guild for each sampling season at LD.

Figure 3.4.15 shows that the H site is dominated by collector gatherer invertebrates in the spring 2002, autumn 2002 and spring 2003 samples, with a much reduced proportion found in the two remaining samples, which contained an increased percentage of shredder invertebrates. When the post-change samples are considered it can be seen that there is a clear seasonal variation in the proportion of collector-gatherer invertebrates at this site, with the spring samples having the highest percentages and summer samples the lowest (as was found in the pre-change samples). There is also a decrease in the percentage of filterer invertebrates present and an increase of scraper invertebrates when compared to the pre-change samples. The highest percentages of scraper invertebrates were found in the post-change summer and autumn samples. Similar proportions of filterer invertebrates were found in the post-change samples as in the pre-change samples apart from in the autumn 2004 samples where a larger increase was seen. The percentage of predators was larger in spring 2004 than for any other sample, but the percentage of predators in the remaining post-change samples remained consistent and of a similar magnitude to the pre change samples.

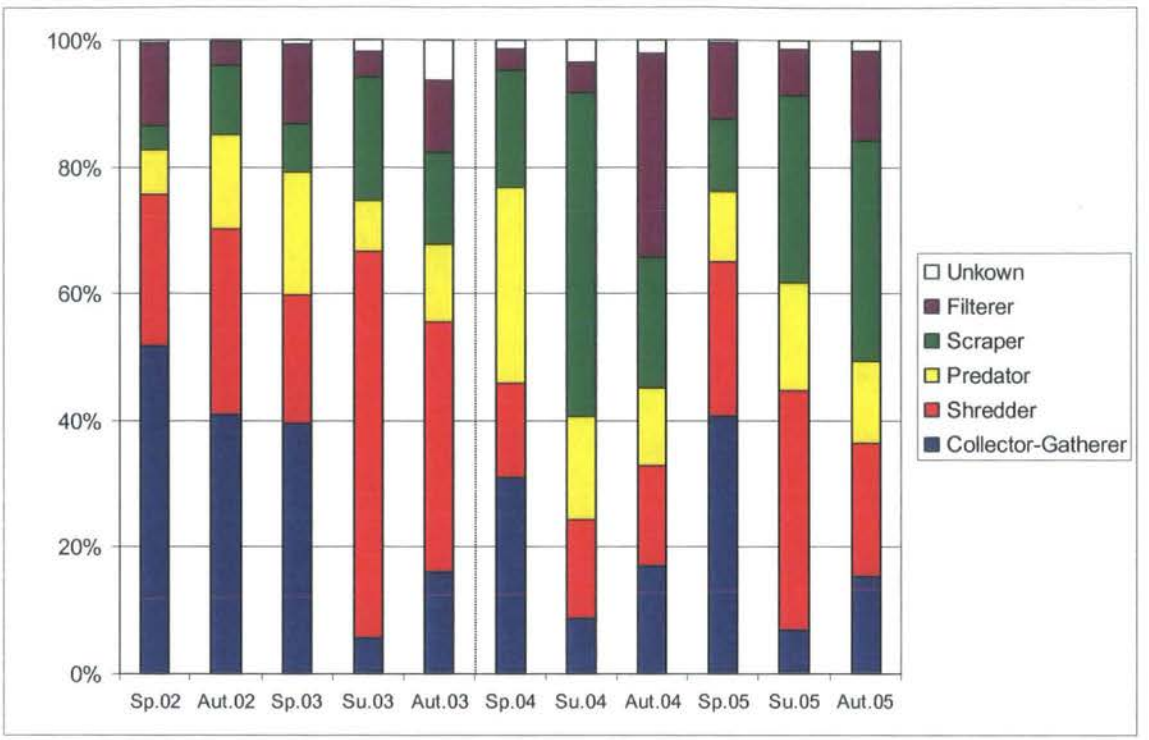

Figure 3.4.15. Figure to show the percentage of invertebrates which belong to each feeding guild for each sampling season at H.

The above section shows that for both of the Rivelin sites there was not a great deal of change in the proportions of feeding guilds present. The proportions of feeding guilds present at RU, appeared to increase in variability in the post-change samples compared to the pre-change samples, whereas the proportions of feeding guilds present at RD, appeared to display much less variation in the post-change samples than the pre-change samples. At both Loxley sites, very little change could be ascertained, indicating a very stable invertebrate population little influenced by the change in compensation flows. The feeding guilds present at LU were still dominated by shredder invertebrates, but greater proportions of predators were found in the post-change samples. The shredder guild also remained very important at LD, with perhaps a slight decrease in the proportion of filtering invertebrates found in the post-change samples.

3.4.3.6. Alterations with central/marginal dynamics

Section 4.1 showed that there was generally very little difference between the central and marginal samples at any of the sites. Table 3.4.43 shows that there are a greater number of statistically significant differences between central and marginal samples in the post-change data. What is immediately apparent from Table 3.4.43 is that all the differences in the post-change samples occur in the summer and autumn. It is also interesting that three out of the five sites had statistically significantly different central and marginal samples in the autumn 2005 sampling season. Table 3.4.43 shows no clear influence of augmentation, with some seasons involved in differences being augmented and others not, as it may have been expected that a decrease in augmentation may lead to greater lateral heterogeneity in macroinvertebrates as the margins get less water or vice versa if invertebrates use the margins as refugia in periods of augmentation.

SITES	SP. 01	SP. 02	AUT. 02	SP. 03	SU. 03	AUT. 03
RU	0.88	-0.02	0.08	0.03	0.04	-0.19
RD		-0.09	0.14	-0.06	0.09	-0.1
LU		-0.16	-0.09	-0.06	-0.08	0.14
LD	0.19	0.08	0.37	0.17	0.20	0.12
H		0.07	-0.03	0.00	0.04	0.02
	SP.04	**SU. 04**	**AUT.04**	**SP. 05**	**SU.05**	**AUT. 05**
RU	-0.1	-0.04	0.19	0.09	0.43	0.44
RD	-0.11	0.12	0.20	-0.17	-0.01	0.06
LU	0.00	-0.1	0.00	-0.02	0.19	0.02
LD	0.18	0.34	0.15	0.15	0.14	0.29
H	-0.13	0.01	0.0	-0.05	-0.16	0.43

Table 3.4.43. Table to show the ANOSIM output for comparing the central and marginal Surber samples biotic data for each of the sampling seasons. Grey colouration indicates a statistically significant difference to the 95% significance level.

3.4.3.7. Analysis of important species

This sub-section investigates the extent to which the important species of the various sites have been changed by altering the compensation flows. Table 3.4.44 shows that for the RU site, there are no species which are common to every post-change sampling season. However, Leptoceridae, and Elmidae are common to the two post-change spring samples; *Hydropsyche siltalai* and *Gammarus pulex* common to the two summer samples; and

Leptophlebiidae, *Hydropsyche siltalai* and Gammaridae common to the two autumn samples (Table 3.4.44). However, there is only one common Leptoceridae species (*Athripsodes cinereus)* between the two post-change spring samples, with the spring 2005 sample containing two 'new' Leptoceridae species. The presence of *Mystacides azurea* was only one of three species of importance to the post-change samples which had a preference for slow flow. Autumn 2004 can be noted for the presence of *Gammarus lacustris* which has a preference for stagnant waters. *Hydropsyche siltalai* was found to be important in five out of the six post-change samples. Thus it appeared that the important species for the RU site were dominated largely by a 'hard-core' of *Hydropsyche siltalai*, *Limnius volckmari*, *Elmis aenea* and *Gammarus pulex*. It was also evident that the important families at RU in the post-change samples were all fast-flow loving apart from Leptoceridae.

When compared to the pre-change samples it appears that the important species for the post-change samples are much more consistent than for the pre-change samples, which could be another indication of the increased dampening of the post-change compensation flow. There were no common species between the two autumn pre-change samples, yet there were three common species in the two post-change autumn samples. It was also seen that a number of families have markedly reduced in importance. Leuctridae were important for two out of three of the pre-change spring samples, and yet they were not found to be important for any post-change sample. The Polycentropodidae family was found to be important for two seasons in the pre-change samples, and were largely absent from any post-change samples. It was also seen that in the pre-change samples, Leptoceridae was important for four out of six sites, but this family was only important for two out of six post-change samples. It was also apparent that Gammaridae and Hydropsychidae increased in importance in the post-change samples. It would have been of interest to compare the important species for pre and post change samples when there was no augmentation of the flow. However, Table 2.10 in Chapter 2 shows that none of the post-change samples were without tributary or overtopping inputs.

	Sp 2001	Sp.2002	Aut.2002	Sp.2003	Su.2003	Aut.2003	Sp.2004	Su.04	Aut.04	Sp.05	Su.05	Aut.05
Hydropsychidae												
H. siltalai (II)	*					*		*	*	*	*	*
H. instabilis (II)	*											
Leuctridae												
L. inermis (I)	*	*										
L. hippopus/moselyii (I)		*										
L. geniculata (II)		*										
L. fusca (II)		*										
Heptageniidae												
Rithrogena spp. (I)	*											
R.semicolorata (I)			*									
Ecdyonurus torrentis (I)			*									
Seriscostomatidae												
S. personatum (II)	*					*						
Leptoceridae												
Leptoceridae sp. (IV)	*											
Athripsodes spp. (II*)			*		*							
Mystacides spp. (IV)			*									
Athripsodes bilineatus (II)				*			*					
Athripsodes cinereus (II)							*			*		
Mystacides azurea (IV)				*	*		*					
Leptophlebiidae												
P. submarginata (II)	*					*			*			*
P.cincta (II)									*			
Ephemerellidae		*										
S. ignita (II)				*								
Elmidae												
L. volckmari (II)		*	*	*	*		*	*	*	*		
E. aenea (II)		*	*	*	*		*	*	*	*		
Oulimnius sp. (IV)		*	*	*	*			*	*	*		
Gammaridae												
G. pulex (II)		*	*					*	*		*	*
G. lacustris (II)									*			
Polycentropodidae												
P. flavomaculatus (II)		*	*									
P.kingi (II)		*										
P. conspersa (II)		*										
Crynus trimaculatus (IV)			*									
Empididae		*								*		
Ephemeridae												
E. danica (II)			*		*		*	*				
Hydracarina				*			*					
Caenidae												
C. horaria (IV)						*						
Limnephilidae												
P. latipennis (II)						*						
Ancylidae												
A. fluviatilis (II)								*				

Table 3.4.44. Table to show the distilled SIMPER within sample similarity output for all samples at RU (species are those which contribute up to 60% of the cumulative similarity). LIFE flow groups are given in brackets. Post-change samples highlighted red.

Table 3.4.45 shows that, for RD, the most important family is Elmidae, as it is found to be important for each of the post-change sampling seasons (dominated by the fast-flow loving *Limnius volckmari* and *Elmis aenea*). Seriscostomatidae were found to be important for five out of six sampling seasons. For the post-change samples: Hydracarina, Elmidae and Empididae are important to the spring samples; *S. personatum*, Elmidae and Sphaeriidae are important to summer samples; and *S. personatum*, Elmidae and *Ephemera danica* are important to autumn samples. The Empididae family were only found to be important for the spring samples. Most of the important families for the post-change samples were generally fast flow loving families, with the summer samples seemingly having more important species with a preference for a slower flow velocity.

The importance of Elmidae and Seriscostomatidae to the invertebrate community at RD was highlighted in Section 3.4.1, and it is interesting that they have remained important to the invertebrate community despite the change in compensation flows. There were a number of smaller changes however, with Leuctridae, which was important for three out of five pre-change samples only being important for one out of six post-change samples, and Ephemerellidae, which was important to two out of five pre-change samples being important for none of the post-change samples. The change in importance of Leuctridae is interesting as a similar phenomenon occurred at the RU site. What is also of interest is the emergence of the important family Empididae, as this was not important to any sampling season in the pre-change period. There does appear to have been less change at this site when compared to RU, as perhaps the impacts of altering the compensation flows is somewhat dampened by tributaries feeding into the Rivelin.

	Sp.02	Aut.02	Sp.03	Su.03	Aut.03	Sp.04	Su.04	Aut.04	Sp.05	Su.05	Aut.05
Ephemerellidae											
S. ignita (II)	*		*								
Leuctridae											
L. hippopus (I)	*		*	*							
L. geniculata (II)	*		*	*					*		
L. fusca (II)	*		*	*					*		
L.inermis (I)									*		
Sericostomatidae											
S. personatum (II)	*	*		*	*	*	*	*		*	*
Elmidae											
E. aenea (II)	*	*	*	*	*	*	*	*	*	*	*
L. volckmari (II)	*	*	*	*	*	*	*	*	*	*	*
Oulimnius spp. (IV)					*		*	*		*	*
Rhyacophilidae											
R. dorsalis (I)	*										
R. obliterata (I)	*										
Heptageniidae											
Ecdyonurus spp. (I)		*									
R. semicolorata (I)		*									
Polycentropodidae											
P. flavomaculatus (II)		*									
P. kingi (II)		*									
Hydropsychidae											
H. siltalai (II)		*									*
H. pellucidula (II)		*									*
H. instabilis (II)											*
Hydracarina			*	*	*	*	*		*		
Limnephilidae											
P. latipennis (II)					*			*			
Potomophylax sp. (II)										*	
Hydrataphylax infumatus (II)										*	
Ephemeridae											
E. danica (II)					*			*	*		*
Simuliidae (II)						*					
Spaheriidae											
Pisidium sp. (IV*)							*			*	
Empididae						*			*		
Leptoceridae											
Mystacides spp. (IV)							*				
Ancylidae											
A. fluviatilis (II)							*				
Leptophlebiidae											
P. submarginata (II)											*

Table 3.4.45. Table to show the distilled SIMPER within sample similarity output for all samples at RD (species are those which contribute up to 60% of the cumulative similarity). LIFE flow groups are given in brackets. Post-change samples highlighted red.

Table 3.4.46 shows that, for LU, *Gammarus pulex* is important for all post-change samples. Table 3.4.46 shows that Polycentropodidae, *Gammarus pulex*, *S. personatum*, and Hydropsychidae are important to both spring samples, with *H. angustpiennis* present in the spring 2005 samples and not the 2004 samples, and *Polycentropus kingi* present in the spring 2004 samples and not the spring 2005 samples. *Gammarus pulex*, *L. fusca*, *L. geniculata* and *S. personatum* are important to each of the summer samples. It can also be seen that *Gammarus pulex*, Baetidae and Sphaeriidae are important to the autumn samples, *Baetis rhodani* present in both spring samples, but with *Centroptilum luteolum* only found in the autumn 2005 samples. It is interesting to note that Baetidae were only found to be important in the autumn samples. It can also be seen in Table 3.4.46 that there are no clear seasonal variations in the LIFE scores of the important species found at LU.

Section 3.4.1 showed that for the pre-change samples, the important species at LU were very consistent, and this pattern continues through to the post-change data. Gammaridae remains important to each of the sampling seasons. The families which were important in the autumn for the pre-change samples (Gammaridae, Hydropsychidae, and Seriscostomatidae) were then found to be important in the spring for the post-change samples. Polycentropodidae also became important in the post-change spring samples, and this family was only found to be important in the summer 2003 and autumn 2003. This is interesting as the emergence of Polycentropodidae as an important species on the Loxley is the reverse of what occurred on the Rivelin. The emergence of Sphaeriidae as an important species in the autumn may also be of interest as it is a slow flow loving family. It is interesting to note that Baetidae was found to be important to the LU invertebrate community in the spring and summer for the pre-change samples, yet is only found to be important in the autumn post-change. Thus, despite the family remaining important to the site, the community has changed a little, as the time of year at which Baetidae is important appears to have changed. It was seen in the pre-change samples that the important species in the summer samples appeared to prefer slower flow than any other time of year, yet this was not apparent in the post-change samples.

	Sp.02	Aut.02	Sp.03	Su.03	Aut.03	Sp.04	Su.04	Aut.04	Sp.05	Su.05	Aut.05
Ephemerellidae											
S. ignita (II)	*		*				*				
Gammaridae											
G. pulex (II)	*	*	*	*	*	*	*	*	*	*	*
Leuctridae											
L. geniculata (II)	*		*	*			*		*	*	
L. fusca (II)	*			*			*		*	*	
L. hippopus (I)									*		
L. inermis (I)									*		
Hydropsychidae											
H. siltalai (II)	*	*	*		*	*		*	*		
H. pellucidula (II)		*			*	*		*	*		
H. angustipennis (II)					*			*	*		
H. instabilis (II)					*	*			*		
Baetidae											
B. rhodani (II)	*		*					*			*
B. scambus (II)	*										
Centroptilum luteolum (III)											*
Sericostomatidae											
S. personatum (II)		*			*	*	*		*	*	*
Sphaeriidae (IV*)				*			*	*			*
Polycentropodidae											
P. flavomaculatus (II)				*		*			*		*
P. kingi (II)					*	*					
P. conspersa (II)						*			*		
Empididae						*					

Table 3.4.46. Table to show the distilled SIMPER within sample similarity output for all samples at LU (species are those which contribute up to 60% of the cumulative similarity). LIFE flow groups are given in brackets. Post-change samples highlighted red.

When the downstream Loxley site (LD) is considered, Table 3.4.47 shows that no family is common to all of the sampling seasons for the post-change samples. However, Hydracarina and Empididae are important to both spring samples. *Lymnea peregra*, *Leuctra fusca*, *Leuctra geniculata* Sphaeriidae and Hydracarina are common to the summer samples; and *Potomophylax latipennis*, Sphaeriidae and *L. peregra* are important to the autumn samples. It is interesting to observe that Empididae is only found to be important in the spring samples, whilst *L. peregra* is important only in the summer and autumn samples. It is also apparent from Table 3.4.47 that there are no seasonal patterns in the flow velocity preferences of the important macroinvertebrates.

When compared with the important spring species in the pre-change samples (Table 3.4.47), the important species for the post-change samples are different. It is also interesting that the two important species in the pre-change samples are generally not found to be of importance in the post-change samples. The important species in the summer samples remained relatively unchanged, whereas only Limnephilidae remained important to the autumn samples. Sphaeriidae became important to the invertebrate population at LD in the later pre-change samples, and its importance remained in the post-change samples. Sphaeriidae has a preference for slow and standing waters, which ties in with a small shift in the flow velocity preferences of the important species in the post-change samples compared with the pre-change samples.

	Sp.01	Sp.02	Aut.02	Sp.03	Su.03	Aut.03	Sp.04	Su.04	Aut.04	Sp.05	Su.05	Aut.05
Hydropsychidae												
H. siltalai (II)	*		*									
H. pellucidula (II)			*									
Leuctridae												
L. hippopus/moselyii (I)	*	*										
L. inermis (I)	*	*									*	
L. geniculata (II)		*		*				*		*	*	
L. fusca (II)		*		*				*			*	
Baetidae												
Baetis rhodani (II)	*									*		
Gammaridae												
G. pulex (II)	*	*	*								*	*
Polycentropodidae												
P. flavomaculatus (II)		*				*						
P. kingi (II)		*										
Ephemerellidae												
S.ignita (II)		*						*				
Limnephilidae												
P. latipennis (IV*)			*			*	*		*			*
P. cingulatus (II)			*									
Limnephilus rhombicus (IV)						*						
Chaetopteryx villosa (II)							*					
H. radiatus (II)							*					
Simuliidae (II)			*			*			*			
Empididae				*			*			*		
Hydracarina				*	*		*	*		*	*	
Sericostomatidae												
S. personatum (II)				*	*	*	*					*
Spaheriidae (IV*)					*	*		*	*	*	*	*
Leptoceridae												
Mystacides spp. (IV)					*							
Lymnaeidae												
Lymnea peregra (IV)					*			*	*		*	*
Hydrophilidae												
Helophrous sp. (V*)							*					

Table 3.4.47. Table to show the distilled SIMPER within sample similarity output for all samples at LD (species are those which contribute up to 60% of the cumulative similarity). LIFE flow groups are given in brackets. Post-change samples highlighted red.

Table 3.4.48 shows that Elmidae remains important to the invertebrate community at every sampling season at H for the post-change data. Table 3.4.48 shows that *Limnius volckmari*, *Elmis aenea* and *Baetis rhodani* are important to both spring samples; *Limnius volckmari*, *Elmis aenea*, *Erpobdella octoculata*, and Tipulidae are important to both summer samples; and *Limnius volckamri*, *Erpobdella octoculata*, *Agapetus fuscipes* and *Glossosoma boltoni* are important to both autumn samples. There appears to be no distinct seasonal variation or pattern in the flow velocity preferences for the important species found at each of the sites.

The most interesting difference between pre and post change samples at H is the emergence of Erpobdellidae as an important family at this site. This family prefers slow flowing water and this agrees with the impression that the important species in the post-change communities seem to have a preference for slower flows than the pre-change communities. However, there was some consistency between the two sets of samples, with Elmidae remaining generally very important, Baetidae remaining important for the spring samples, and Tipiulidae being important for every summer sampling season. However, the important species for the autumn samples have altered, with Baetidae not being important for only one of the post-change autumn samples, and Glossosomatidae and Erpobedellidae becoming important species for the autumn in the post-change samples. From the above analysis, it does appear as though some change in important species (as defined by SIMPER) could occur over this time period without the complication of impoundment. Therefore, it may be difficult to tell for certain if changes to the important species of the Rivelin and Loxley are due to the alteration of the compensation flows.

	Sp.02	Aut.02	Sp.03	Su.03	Aut.03	Sp.04	Su.04	Aut.04	Sp.05	Su.05	Aut.05
Ephemerellidae											
S. ignita (II)	*		*								
Leuctridae											
L. geniculata (II)	*		*	*					*		
L. fusca (II)	*		*	*					*	*	
L. hippopus (I)	*		*								
L. nigra (II)			*								
L. inermis (I)			*						*		
Baetidae											
B. scambus (II)	*		*				*				
B. rhodani (II)	*	*	*		*	*	*	*	*		
B. muticus (II)	*		*								
Simuliidae (II)	*				*			*			
Elmidae											
L. volckmari (II)	*	*	*	*	*	*	*	*	*	*	*
E. aenea (II)	*	*	*	*	*	*	*	*	*	*	*
Oulimnius (IV)	*			*				*		*	
Heptageniidae											
Ecdyonurus spp. (I)		*									
R. semicolorata (I)		*									
Tipulidae (IV*)		*		*			*	*		*	
Hydracarina			*								
Chloroperlidae											
S. torrentium (I)			*						*		
Sericostomatidae											
S. personatum (II)					*	*					
Limnephilidae											
P. latipennis (II)					*						
Erpobdellidae											
E. octoculata (IV)					*	*	*	*		*	*
Nemouridae											
Nemoura avicularis (IV)					*						
A. sulcicollis (II)					*						
Ancylidae											
A. fluviatilis (II)							*				
Glossosomatidae											
Agapetus fuscipes (II)								*			*
Glossosoma boltoni (II)								*			*
Glossosoma conformis (II)								*			
Agapetus sp. (II)										*	
Hydropsychidae											
H. siltalai (II)											*

Table 3.4.48. Table to show the distilled SIMPER within sample similarity output for the post-change samples at H (species are those which contribute up to 60% of the cumulative similarity). LIFE flow groups are given in brackets. Post-change samples highlighted red.

Table 3.4.44 shows that the important species in the post-change samples appear to be more consistent when compared with the pre-change samples at RU. Leuctridae and Leptoceridae appeared to decrease in importance whilst Gammaridae and Hydropsychidae

increased in importance. Leuctridae was also seen to decrease in importance at RD, but there was generally less change seen at this site compared with the upstream site. The increased consistency of the important species at RU, could be due to the decreased overtopping during the post-change period, and RD will be less effected by this decrease due to the effects of tributary inputs. It was shown that the important species at LU remained consistent with the pre-change species, whilst the there was a change in important species between the pre and post change samples at LD. This could be due to the lower compensation flow level increasing the impact of tributary variability at LD. Some changes in important species were observed in the unregulated Hipper, with the emergence to importance of Erpobdellidae.

3.4.3.8. Relationship between biotic and abiotic data: BIOENV

The way the biotic and abiotic data interact may give us more of an insight into whether changes have been driven by the compensation flow changes or by other factors. As in section 3.4.1, this will be investigated using the BIOENV routine within PRIMER.

Table 3.4.49 shows that, at RU for the pre-change data, the highest Spearman's rank correlation values were found in the spring samples, and depth was found to be important in four out of the five sampling seasons. The importance of depth at this site continues in the post-change samples, with it being present in four out of the six sampling seasons. There is no obvious pattern in the Spearman's rank correlation values for the post-change data, but the two autumn samples do have very similar values, but pertaining to different environmental variables. It can also be seen from Table 3.4.49 that width is seemingly very important for the 2004 samples and flow velocity important for the 2005 samples.

Spring 2002	Autumn 2002	Spring 2003	Summer 2003	Autumn 2003
Spearman's Rank Correlation = 0.585	Spearman's Rank Correlation = 0.337	Spearman's Rank Correlation= 0.718	Spearman's Rank Correlation = 0.445	Spearman's Rank Correlation= 0.459
Width	Substrate	Depth	Depth	Depth
Depth	Flow Velocity		Flow Type	Flow Velocity
Flow type				

Spring 2004	Summer 2004	Autumn 2004	Spring 2005	Summer 2005	Autumn 2005
Spearman's Rank Correlation = 0.468	Spearman's Rank Correlation = 0.599	Spearman's Rank Correlation =0.277	Spearman's Rank Correlation= 0.430	Spearman's Rank Correlation = 0.155	Spearman's Rank Correlation = 0.280
Width	Width	Width	Depth	Depth	Flow Velocity
Flow Type	Depth	Depth	Substrate	Flow Velocity	
Substrate	Flow Type		Flow Velocity		

Table 3.4.49. Table to show the best BIOENV output from the RU site for every sampling season, comparing the invertebrate community and measured environmental variables.

Table 3.4.50 shows the relative importance of flow type, substrate and flow velocity to the matching of biotic and abiotic data at RD, with width and depth seemingly less important. It is also evident from Table 3.4.50 that in the pre-change samples flow-type and substrate were found to be of importance in both spring lists, and flow velocity is common to both the autumn lists. There is no clear patterns to the Spearman's rank correlation in the pre-change samples at this site. Table 3.4.50 shows that for the post-change samples the Spearman's rank correlation is of a similar size as that of the higher correlations in the pre-change data and was more consistent than that found in the pre-change data (apart from Autumn 2005). Table 3.4.50 appears to show that width remains relatively unimportant, whereas depth is the most important environmental variable in the post-change data, being important for both summer and autumn post-change samples. Depth was not important in either of the post-change spring samples, whereas flow type was found to be important for both spring samples. Flow velocity was also found to be important for both post-change summer samples, with depth and substrate important to both autumn samples.

Spring 2002	Autumn 2002	Spring 2003	Summer 2003	Autumn 2003
Spearman's Rank Correlation = 0.471	Spearman's Rank Correlation = 0.434	Spearman's Rank Correlation= 0.356	Spearman's Rank Correlation = 0.656	Spearman's Rank Correlation= 0.613
Depth	Substrate	Flow Type	Flow Type	Flow Type
Flow Type	Flow Velocity	Substrate	Substrate	Flow Velocity
Substrate				
Flow Velocity				

Spring 2004	Summer 2004	Autumn 2004	Spring 2005	Summer 2005	Autumn 2005
Spearman's Rank Correlation = 0.633	Spearman's Rank Correlation =0.648	Spearman's Rank Correlation = 0.506	Spearman's Rank Correlation= 0.605	Spearman's Rank Correlation =0.640	Spearman's Rank Correlation = 0.320
Flow Type	Depth	Depth	Width	Depth	Depth
	Flow Velocity	Flow Type	Flow Type	Flow Velocity	Substrate
		Substrate			Flow Velocity

Table 3.4.50. Table to show the best BIOENV output from the RD site for every sampling season comparing the invertebrate community and measured environmental variables.

Table 3.4.51 shows that for the LU site, the lowest Spearman's rank correlation values were found in the autumn samples. It can also be seen that flow type and depth are very important to the pre-change samples. When the post-change samples are considered, the Spearman's rank correlation values appear to be higher in the post-change data than in the pre-change data. This increase in Spearman's rank correlation could indicate a decrease in the detachment of biotic and abiotic variables at LU, which may have been caused by the decrease in base compensation flow.

Spring 2002	Autumn 2002	Spring 2003	Summer 2003	Autumn 2003
Spearman's Rank Correlation = 0.473	Spearman's Rank Correlation = 0.142	Spearman's Rank Correlation=0.658	Spearman's Rank Correlation = 0.482	Spearman's Rank Correlation= 0.248
Flow Type	Depth	Flow Type	Depth	Depth
	Substrate	Substrate	Flow Type	Flow Type

Spring 2004	Summer 2004	Autumn 2004	Spring 2005	Summer 2005	Autumn 2005
Spearman's Rank Correlation = 0.624	Spearman's Rank Correlation = 0.652	Spearman's Rank Correlation = 0.647	Spearman's Rank Correlation= 0..340	Spearman's Rank Correlation =0.334	Spearman's Rank Correlation = 0.722
Flow Velocity	Width	Flow Type	Width	Width	Flow Type
	Substrate		Depth	Depth	
			Flow Type	Flow Type	
			Substrate	Substrate	

Table 3.4.51. Table to show the best BIOENV output from the LU site for every sampling season comparing the invertebrate community and measured environmental variables.

Table 3.4.52 shows that for the downstream Loxley site (LD), flow velocity can be thought of as important for four out of the five pre-change sampling seasons. It can also be seen that the highest Spearman's rank correlation values were found in the spring samples, with a much lower correlation found in the summer 2003 samples. What can also be seen in Table 3.4.52 that substrate is important for both spring samples, and flow velocity is important to both autumn sets of samples. When the post change samples are considered, there appears to be no obvious changes when compared to the pre-change results. Flow type is important for five out of six post-change sampling seasons at this site, and flow velocity important in four out of six sampling seasons. Substrate and width are found only to be important in one of the sampling seasons. Thus, it appears that flow velocity and flow type are the most important environmental factors in the post-change samples, which is interesting as the emergence of flow type as an important environmental variable also occurred at LU. Overall there appeared to be no systematic change of the relationship between biotic and abiotic variables at LD.

Spring 2002	**Autumn 2002**	**Spring 2003**	**Summer 2003**	**Autumn 2003**
Spearman's Rank Correlation = 0.645	Spearman's Rank Correlation = 0.463	Spearman's Rank Correlation= 0.673	Spearman's Rank Correlation = 0.158	Spearman's Rank Correlation= 0.529
Width	Flow Velocity	Depth	Flow Type	Flow Velocity
Substrate		Substrate	Flow Velocity	
Flow Velocity				

Spring 2004	**Summer 2004**	**Autumn 2004**	**Spring 2005**	**Summer 2005**	**Autumn 2005**
Spearman's Rank Correlation = 0.610	Spearman's Rank Correlation = 0.564	Spearman's Rank Correlation = 0.396	Spearman's Rank Correlation= 0.256	Spearman's Rank Correlation = 0.374	Spearman's Rank Correlation = 0.557
Depth	Flow Velocity	Width	Flow Type	Depth	Depth
Flow Type		Flow Type		Flow Type	Flow Type
Substrate				Flow Velocity	Flow Velocity
Flow Velocity					

Table 3.4.52. Table to show the best BIOENV output from the LD site for every sampling season comparing the invertebrate community and measured environmental variables.

When the relationship between the environmental and biotic data is considered for H, the unregulated control catchment, Table 3.4.53 shows that for the pre-change data, the Spearman's rank correlation values are generally very low apart from in autumn 2003. Table 3.4.53 also shows that flow type is important for three out of five sites (and was not

included in the spring 2002 data analysis due to a data gap). The post-change samples display a much better correlation between the biotic and abiotic data, but with no clear seasonal pattern. Table 3.4.53 also shows that the importance of flow type persists in the post-change samples. Apart from flow type there were no environmental variables which were common to any of the sampling seasons in the post-change data. However, depth was found to be important for three out of the six sampling seasons.

Spring 2002	**Autumn 2002**	**Spring 2003**	**Summer 2003**	**Autumn 2003**
Spearman's Rank Correlation = 0.155	Spearman's Rank Correlation = 0.1	Spearman's Rank Correlation= 0.051	Spearman's Rank Correlation = 0.162	Spearman's Rank Correlation= 0.512
Width	Flow Type	Depth	Depth	Flow Type
Flow Velocity			Flow Type	Substrate

Spring 2004	**Summer 2004**	**Autumn 2004**	**Spring 2005**	**Summer 2005**	**Autumn 2005**
Spearman's Rank Correlation = 0.202	Spearman's Rank Correlation = 0.261	Spearman's Rank Correlation = 0.558	Spearman's Rank Correlation= 0.464	Spearman's Rank Correlation = 0.599	Spearman's Rank Correlation = 0.348
Flow Type	Depth	Depth	Depth	Flow Type	Flow Type
	Flow Type	Flow Type	Flow Type	Substrate	
		Flow Velocity	Substrate		
			Flow Velocity		

Table 3.4.53. Table to show the best BIOENV output from the H site for every sampling season comparing the invertebrate community and measured environmental variables.

Table 3.4.54 shows that the correlation between measured environmental variables and the invertebrate communities decreases in the post-change samples at RU, but increases at LU. This could be due to the changes in flow variability at each of these sites.

	Pre-change	**Post-change**
RU	0.51	0.37
RD	0.51	0.56
LU	0.4	0.55
LD	0.49	0.46
H	0.196	0.41

Table 3.4.54. Table to show the average Spearman's rank correlation calculated using the BIOENV routine for the pre and post change samples at each site.

3.4.3.9. Summary

Data Analysis Method	Brief results
Environmental variables: ANOSIM	Between site differences remain. Fewer between season differences in the Rivelin sites in the post-change samples. More between season differences in the Loxley in the post-change samples.
Av. Invertebrate density	No change. LU remains highest. RD>RU
Av. No. Families	No clear pattern.
Av. Shannon Diversity	No clear pattern.
Family Lists	Polycentropodidae lost from the Rivelin. No large changes to the Loxley. Erpobdellidae becomes important to the Hipper.
Invertebrate samples: ANSOIM	All sites still different. Sampling seasons still largely statistically different.
Important Species: SIMPER	RU = important species become more stable. RD = more variation in the important species list. LU = not much change. LD = some changes to the important species list (more slow-flow loving species). Erpobdellidae becomes important to the Hipper.
Feeding Guilds	RU = increase in guild variability. RD = less guild variability in the post-change samples. Little change in the Loxley – shredders still important.
BIOENV	RU = no change in correlation values RD = higher correlation in post-change samples LU = higher correlation in post-change samples LD = no change in correlation values

Table 3.4.55. Table to show the data analysis techniques and summarise the results from Section 3.4.3.

This section has shown that both the rivers have changed in terms of the measured environmental variables, with the Loxley apparently undergoing greater variability under the post-change compensation flow regime. However, few consistent changes could be determined in terms of average invertebrate density per sample, average number of families per sample, or Shannon diversity of the invertebrate community. The family list analysis showed that Polycentropodidae was much less abundant in the post change sampled compared with the pre change samples on the Rivelin. On the Loxley, any changes in the family list were in species which occurred in only a few samples.

The ANOSIM analysis of the invertebrate data showed that the sampling sites remained statistically significantly different, and the sampling seasons (compared within each site) also remained largely statistically significantly different. This indicates that the

invertebrate communities have not changed to too greater extent. However, when the important species were analysed (using the SIMPER routine) for the post-change sampling seasons, it was apparent that changes have occurred to the invertebrate communities. At RU, the important species have become more stable compared with the large amount of variation seen in the pre-change samples, whilst less change was observed at RD, with perhaps a greater amount of variability introduced in the post-change samples. Very little change was observed in the important species list for LU, whilst at LD, more slow-flow loving species came to importance.

The pattern displayed in the guilds appeared to be the opposite of that displayed in the important species. The proportion of guilds present at RU became more variable in the post-change samples, and this is despite the decrease in augmentation from the Rivelin compensation reservoir. Whilst greater variability in the RD important species was observed, the proportions of feeding guilds at this site became more stable in the post-change samples. However, little change was seen in the proportions of feeding guilds present in the two Loxley sampling sites, with shredders still being very important.

The BIOENV analysis showed that there were higher post-change Spearman's rank correlations for RD and LU, whilst the correlation at RU decreased and that of LD changed little. Flow type appeared to remain of importance.

3.5. Discussion and conclusions

There were a number of aims which were used to investigate the invertebrate populations, as introduced in section 3.1. However, the two main questions which need to be discussed in further detail are: (i) can we detect an impact of pre-change flows?; and (ii) what are the impacts of altering the compensation flows. Each of these questions will be addressed in turn.

3.5.1. The impact of the pre-change compensation flows

As was discussed in Section 3.4.1, both the Rivelin and Loxley are impacted upon by two scales of flow variability: high frequency; and low frequency. Due to the nature of the pre-change compensation releases, there appears to be a gradient of variability within the study sites. Figure 3.5.1 shows the Huston (1994) dynamic equilibrium diagram annotated with a hypothesised gradient of flow variability (which essentially constitutes a disturbance). The site with the least flow variability, due to its proximity to the compensation reservoir, and the lack of overtopping from the dam is LU. The LD site will be subject to some flow variation, due to tributary inputs, but these will perhaps be dampened somewhat by the larger compensation flow releases in the Loxley. The next most variable site is RU, due to a combination of lower compensation flow releases, and proximity to a more frequently overtopping reservoir. The site subject to the greatest variability should be RD, where there are both overtopping impacts and unregulated tributary inputs.

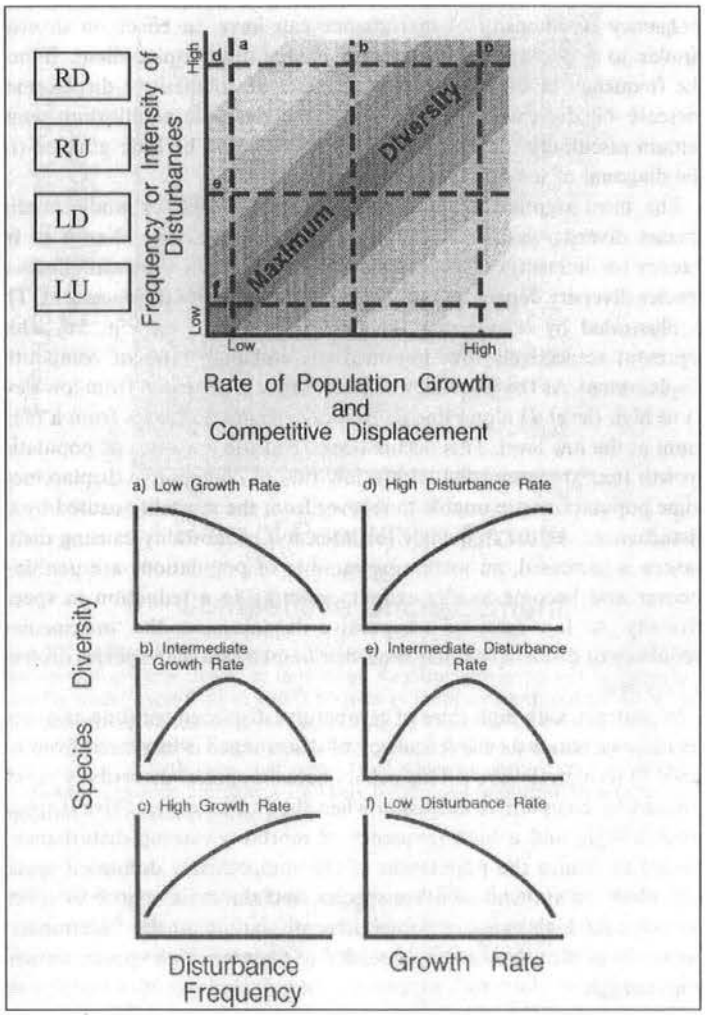

Figure 3.5.1. Three-dimensional representation of the predictions of the Dynamic Equilibrium Model for species diversity in relation to the dynamic equilibria between different rates of competitive displacement (correlated with growth rate, productivity etc) and different frequencies of mortality-causing disturbances. Adapted from, Huston (1994).

The Rivelin contained a greater number of families than the Loxley, but the Loxley appeared to have a greater density of invertebrates per Surber sample (LU in particular). This finding agrees with McCabe and Gotelli (2000) in that the river subject to the greatest disturbance has the highest number of taxa, whilst the site with the least disturbance was the one with the highest density of invertebrates. When the numbers of families/sample is considered, there appears to be little difference between the Rivelin and Loxley. This suggests that disturbance may be useful in regulated rivers such as these in order to maintain species diversity. It is interesting that LU had the highest densities of invertebrates per sample as it is probably the site subject to the least hydrological

variability, with it being close to the Loxley reservoir and hence not subject to tributary augmentation. This agrees with the findings of Englund (1991); Doeg *et al.* (1989); Jowett and Duncan (1990) and Lake *et al.* (1989) who found a decrease in invertebrate density with increased disturbance, but disagrees with the findings of Clausen and Biggs (1997) who found a positive relationship between invertebrate densities and flood disturbance frequency.

The important species analysis showed that on the Rivelin, the most variable important species lists were found at the upstream sampling site. This could be because of its proximity to the compensation reservoir meaning that it is more impacted upon by overtopping events than the downstream site (allied with it only being added to the small compensation flow). Perhaps the added rainfall and tributary inputs will dampen the variability caused by the overtopping events at the downstream Rivelin site. The important species sets were more stable at each of the Loxley sites.

The reverse pattern was seen in terms of the feeding guilds at the Rivelin, with the invertebrates sampled at RU having more stable proportions of the various feeding guilds and RD having much more variable proportions. This is strange, as one would expect the site with the more consistent set of 'important' species to have a more stable proportions of the functional feeding guilds. This indicates that RD may have an abundance of habitat for each of the guilds, and that despite the important species remaining constant, variations in the peripheral species provide the variation in the feeding guilds.

A problem when trying to further interpret Figure 3.5.1 with respect to the study rivers is that we have not quantified the rate of population growth or the rate of competitive displacement. Therefore, it is impossible for us to say whether the invertebrates in these upland Millstone Grit streams follow the intermediate disturbance hypothesis or the dynamic equilibrium hypothesis. It appears in general that the more variable Rivelin is better?

It is also difficult to quantify the flow variability for each of the sampling sites separately. This is because there was only one gauging station present in each of the rivers, and these are located just beneath the downstream invertebrate sampling sites. On the Loxley, it

appears that LU is subject to much less variation than LD. But in the Rivelin, the picture is more confused, as the invertebrate populations at RD appear to be more stable than those found at RU. It would be logical to assume that RD would be subject to the most variation as the greater overtopping events would combine with tributary inputs at the downstream site. However, it does appear that the invertebrate populations are more variable at the upstream site, so this site may be more susceptible to overtopping variation due to the low compensation flow levels; and the variability at the downstream site is dampened by tributary inputs.

The Rivelin samples also appeared to have a better relationship with the measured environmental variables than the Loxley. The Rivelin has more invertebrate families present than the Loxley, but, the greater variability present in the Rivelin flow has perhaps led to a closer relationship between the macroinvertebrate community and the measured environmental variables.

There was a large amount of variation in the environmental variables in the BIOENV output. It therefore appears that a combination of environmental variables must impact upon the invertebrate community and it is not always the same environmental variables. Table 3.5.1 interestingly highlights the importance of the steady state parameters discussed in section 3.2.2, and the lack of correlation with width on the macroinvertebrate communities. It appears that flow type may be the most important environmental variable in these upland Millstone Grit rivers, which is interesting as it is generally created from a combination of flow velocity, depth and substrate. Differences in invertebrate richness have been found between pools and riffles (Angradi, 1996; Brown and Brussock, 1991; Grubaugh *et al.*, 1996; McCulloch, 1986; Thorp, 1992; Wohl, 1992). It may have been interesting to statistically investigate using a multivariate methodology the invertebrate communities. However, the invertebrate sampling was designed as to statistically test between the central and marginal samples rather than the flow type.

	Flow Velocity	Depth	Flow Type	Substrate	Width
RU	2	4	2	1	1
RD	3	1	4	4	0
LU	0	3	4	2	0
LD	4	1	1	2	1
H	1	2	3	1	1
Total	**10**	**11**	**14**	**10**	**3**

Table 3.5.1. Table to show the number of times each measured environmental variable was included in the BIOENV output for each of the sampling sites in the pre-change samples.

A variety of environmental conditions provides a diverse habitat for macroinvertebrates. It appears that the Loxley suffers from a homogenisation of habitats (hence the lower Shannon diversity values compared to RU and the smaller standard deviation of those values) due to having too great a discharge. It is possible that after the reduction in discharge the availability of habitats for different invertebrate groups will increase. It is (obviously) the diversity of the habitat available in the river which is of importance, not the amount of water in the system, and this emphasises the need to factor river channel boundary shape into these sort of deliberations.

This question of what the characteristics of the rivers are when not influenced by overtopping can addressed by assessing the impact of the 1995/1996 drought on the invertebrate communities in the Rivelin and Loxley by using the Environment Agency kick sampling data. Unfortunately, no such data were available for the Hipper, as it would have been of interest to compare the impact of droughts on the regulated and unregulated rivers. In both the Rivelin and Loxley, there was a different species composition in the drought and non-drought samples. In both cases, the changes were in the peripheral species (often slow-flow preferring) rather than key species in the river. This indicates that there is good refugia for the most important species in the rivers, as observed in the literature by Cowx *et al.* (1984) and Wood and Petts (1999). It was also seen that the average abundances of species which preferred faster flowing were lower in the drought samples compared with the non-drought samples. This indicates the importance of overtopping/tributary inputs in regulated rivers in influencing the invertebrate communities in the rivers.

In general, the sites were different, and the invertebrate communities found at the sites in the various sampling seasons were also different. When the environmental variables were considered, there were many differences found between the sampling seasons (at a site) on the Rivelin, with much fewer on the Loxley. This appears to indicate that the Loxley was being homogenised by the lower number of overtopping events and the higher discharge dampening the impact of any variability provided by rainfall inputs. This could account for the larger number of families present in the Rivelin compared with the Loxley. This hints at a detachment of the environmental variables and the invertebrate communities in the Loxley, as despite the environmental variables not changing, the invertebrate populations were. However, this is not to say that because the environmental variables change and the invertebrate communities change in the Rivelin does not mean that the changing environmental variables causes the change in the invertebrate communities.

The importance of spatial variability of flow in the study rivers was highlighted by the few statistically significant differences found between the central and marginal samples. This spatial variability is present in terms of both the measured environmental variables and the macroinvertebrate community. This is of great importance when modelling the rivers is considered. It will be important to use an approach which incorporates this spatial variation (such as a two-dimensional model) in preference to a zero or one dimensional model which will average habitat conditions across a given cross-section. This highlights the importance of the interaction of reach-scale morphology and discharge in creating habitat. Having found the lack of difference between central and marginal samples, any future investigation may be better off sampling duplicates of different flow types, so that the impact of compensation flow alteration on the invertebrate communities within each of the flow types could be examined.

The kick sample analysis revealed that the LIFE scores on the Rivelin were generally higher than those found on the Loxley. This seemingly disagrees with the findings of Extence *et al.* (1999) who, when developing the LIFE index found that the rivers with higher discharge had higher LIFE scores. The finding also enhances the importance of the interaction between reach scale morphology and discharge in determining the velocity patterns within the respective streams. It could be this interaction that causes higher velocities in the Rivelin compared to the Loxley, which perhaps led to the higher LIFE

scores present within the Rivelin. Indeed, these rivers are roughness dominated. Where roughness is high, increases in flow should be manifest primarily as increases in depth, although the opposite of this is seen in Chapter 5 when the discharges are changed. Whilst an increase in depth will lead to a decrease in relative roughness (i.e. roughness standardised by flow depth), the absolute roughness will remain, meaning that the increase in flow is accounted for with a decrease in velocity. The modelling chapter will investigate this.

It is also seen that the Loxley invertebrate populations also contain a large proportion of shredding invertebrates. In a large river, Rempel *et al.* (2000) found that shredders were correlated with coarse particulate organic matter (CPOM). Although the presence or otherwise of CPOM was not tested directly in this study, the presence of this substance may be prevented by fast flowing water. Slower velocities may be associated with the greater discharge, and so allowing the presence of the CPOM.

The apparent ineffectiveness of the Shannon diversity index in determining differences between invertebrate communities is interesting. If we assume that the results of the ANOSIM analysis reflect the true differences between the sites and seasons, when the Shannon values are compared using ANOVA, fewer differences are detected. This may be due to the Shannon index being generally ineffective in situations such as this, with another index being able to distinguish such differences. However, it was not the purpose of this study to try and find a diversity index which is effective in determining the effects of water stress. This result does highlight however, that multivariate analyses are much more effective at determining differences between sites and seasons than a given diversity index.

3.5.2. What are the impacts of altering the compensation flows?

The impact of the alteration in compensation flows on the characteristics of the discharge of the two rivers was addressed in Chapter 2. It was seen that the Rivelin was subject to decreased augmentation in the post-change period, with the increase in compensation flow apparently leading to a decrease in reservoir overtopping. Little change occurred in the overtopping pattern of the Loxley compensation reservoir.

Figure 3.4.33 shows that the measured environmental variables on the Rivelin became more similar following the alteration in compensation flows whereas they became generally more different on the Loxley. A number of reasons for this were hypothesised: (i) the increased discharge on the Rivelin homogenises the environmental variables to a greater extent than for the pre-change flows; (ii) the increased discharge released from the Rivelin compensation flow reservoir led to a decrease in the amount of overtopping (this was evidenced by Table 2.9), and hence there would lead to less environmental variability; and (iii) the decreased discharge in the Loxley, led to the variability provided by both overtopping and rainfall events having a greater impact on the measured environmental variables in the Loxley.

Further evidence for changes to the environmental variables is provided by Table 3.5.1 which shows that there are seemingly changes to the Spearman's rank correlations (derived from BIOENV) at the two upstream sites. The Spearman's rank correlation at RU decreased in the post-change samples, perhaps indicating a slight detachment of the invertebrate communities from the measured environmental variables. This could be due to the decreased overtopping which occurs from the Rivelin reservoir in the post-change period, and the increased compensation flow dampening the impact of the flow variability. There was a corresponding increase in Spearman's rank correlation at the upstream Loxley site. There was no increase in overtopping from the Loxley reservoir, but the impact of that overtopping there was could have been increased by the decrease in the buffering afforded by the compensation flows.

	Pre-change	Post-change
RU	0.51	0.37
RD	0.51	0.56
LU	0.4	0.55
LD	0.49	0.46
H	0.196	0.41

Table 3.5.2. Table to show the average Spearman's rank correlation calculated using the BIOENV routine for the pre and post change samples at each site.

When the family lists for each of the sites were examined, there was a large decrease in the number of samples in which Polycentropodidae were present in the Rivelin, but no such change in the Loxley. Apart from this, all other changes to the family lists were in families which were only found in one or two samples.

The invertebrate communities found at each of the sites remained statistically significantly different in the post-change samples, as with the pre-change samples. This suggests that the change in compensation flows had no effect in terms of making the sites any more similar. When the sampling seasons were compared for each of sites, the two upstream sites (Rivelin Upstream and Loxley Upstream) had fewer statistically significant differences than the downstream sites. This perhaps indicates that the downstream sites are subject to greater natural variability from rainfall events.

The guild analysis revealed much more variation in the post-change proportion of guilds present at RU, and a more stable proportion of guilds found at RD. There was little change seen to the proportions of the feeding guilds at either site on the Loxley. This is an indication of the lack of change of the invertebrate community on the Loxley, and the small amount of change which has occurred in the Rivelin. The persistence of the importance of shredders in the Loxley is also of interest, as the reduction in compensation flows does not appear to have removed the habitat available for shredding invertebrates.

The important species in the Rivelin again displayed the opposite pattern to that shown of the functional feeding groups. As the proportions of the functional feeding groups became less consistent at RU, the important species became more consistent. At RD and LU, there was very little change in the important species, which further enhances the picture of the Rivelin changing slightly, but the Loxley hardly changing at all. At LD, the two important

species in the pre-change samples are generally not found to be of importance in the post-change samples. This could be due to a greater impact of tributary variability because of the lower dampening compensation flow.

There was no perceptible change in the relationship between the invertebrate communities and the measured environmental variables, with flow type remaining important (Table 3.5.2). This reinforces the potential usefulness of using flow type as a key informer of invertebrate habitat in these upland Millstone Grit rivers.

	Flow Velocity	**Depth**	**Flow Type**	**Substrate**	**Width**
RU	3 (2)	4 (4)	2 (2)	2 (1)	3 (1)
RD	3 (3)	4 (1)	3 (4)	2 (4)	1 (0)
LU	1 (0)	2 (3)	4 (4)	3 (2)	3 (0)
LD	4 (4)	3 (1)	5 (1)	1 (2)	1 (1)
H	2 (1)	3 (2)	6 (3)	2 (1)	0 (1)
Total	**13 (10)**	**16 (11)**	**20 (14)**	**10 (10)**	**8 (3)**

Table 3.5.2. Table to show the number of times each measured environmental variable was included in the BIOENV output for each of the sampling sites in the post-change samples (pre change sample values in brackets).

This is significant in that it appears that it is difficult to attach changes in the macroinvertebrate to the alteration of the compensation flows. It was seen that the invertebrate communities of the Loxley did not change to any great extent, but the communities present in the Rivelin did change in terms of important species and functional feeding groups. However, attaching causality (to the alteration of the compensation flows) to the changes in the invertebrate communities has proved impossible. This is not helped by the relatively short timescale over which the field monitoring has been conducted. Despite the short life cycle of aquatic invertebrates, the range of discharge variability measured during two years pre change data were not enough to contextualise the rivers fully, and two years post change data are not enough for assessing long term changes to the invertebrate community.

3.5.3. Final conclusions

The first main conclusion of this chapter is that the impacts of compensation flow releases are augmented both by reservoir overtopping and tributary inputs. It was seen each of the sites was impacted upon by varying degrees of variability. The results of this are manifest in the Rivelin containing a greater number of invertebrate species than the Loxley; the Rivelin (especially the upstream site) having the greatest variation in Shannon diversity; and the Loxley upstream site having the greatest invertebrate density/sample. This variability is superimposed upon the compensation flows.

It does appear that the higher compensation flows on the Loxley impacts upon the macroinvertebrate population. The first mechanism by which this occurs is by the larger compensation in the Loxley appearing to dampen the impact of the flow variability, creating much more uniform environmental conditions through time. The second mechanism is by the deeper water provided by the higher discharge apparently creating greater slow flow habitat within the Loxley, evidenced by the abundance of shredding invertebrates; and the lower LIFE scores than on the Rivelin. This is because higher discharges are mainly manifest as increases in depth due to the high relative roughness of the river.

When the post-change invertebrate communities were compared with the pre-change invertebrates, there was generally very little change. It appears that the comparatively small change in the Loxley compensation flow may have precluded any change in any facet of the invertebrate community. There were some small changes in the Rivelin, especially at the upstream site. However, despite the compensation flows on the Rivelin increasing by over 100%, the changes in invertebrate community were minute. This could be due to the interaction of reach-scale geomorphology and discharge creating patchy habitats within the rivers, meaning that changes in habitat are not linearly equated to increases in discharge. This increases the importance of the modelling approach developed in Chapters 5 and 6, as illustrated by the linkages shown in Figure 1.2.