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

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Chapter 4

Assessing the impact of compensation flows on fisheries populations

4.1. Introduction

In the Rivelin, Loxley and Hipper, the predominant salmonid species is brown trout (*Salmo trutta*), and so this became a focus for this study. Since the advent of the Water Framework Directive (WFD) there has been increased awareness of other species. In each of the study rivers there is a population of Bullheads (*Cottus gobio*) and Brook Lamprey (*Lampetra planeri*) which are both listed in annexes IIa and Va of the Habitats Directive, Appendix III of the Bern Convention, and as Long List Species in the UK Biodiversity Action Plan. Thus, the consideration of fisheries issues includes all aspects of the fisheries populations within the rivers, but with a particular focus on brown trout as the main fish biomass. Brown trout is a species which although originally native to Europe is now found around the world, flourishing in unpolluted, well oxygenated streams, rivers, reservoirs and lakes (Elliot, 1994). The brown trout found in these systems generally spawn around October, with the fry emerging from the redd in spring. The trout in these systems do not appear to migrate. Their ubiquitous nature in upland Millstone Grit catchments in general and in particular means that this species is a primary focus of assessing the potential impacts of altering compensation flows.

This chapter aims to investigate the impacts of compensation flows on fish populations. It has four objectives:

1) to establish the structure and quality of fish populations in each of the rivers;
2) to assess natural variability in these populations in order to be able to distinguish the impacts of any changes in discharge on instream ecology;
3) to test the hypothesis that differences in fish population can be related to the different levels of compensation flow in the rivers; and
4) to test the hypothesis that the fish populations are changed by the alteration of the compensation flow regime.

Section 2 addresses a key debate in fisheries science: the exact mechanism with which fisheries populations are controlled, and reviews literature relevant to this topic. The methodology involved in the collection of the fisheries data and the data analysis methods used are outlined in section three. The results section is divided into two sub-sections. The first sub-section addresses the structure and quality of the fisheries populations under the pre-change conditions. This involved comparing the Rivelin and Loxley in order to investigate the impacts of the pre-change compensation flows. It also considers the control catchment. Finally, a section addresses the variability issues present within fisheries data (perhaps caused by the inherent variability within the systems outlined in Chapter 2; Figure 1.2). The second sub-section results section focuses on assessing the impacts on the fisheries populations of altering compensation flows. A discussion follows, the aim of which is to synthesise the results of the two results sections and also to link the results back to the scientific literature presented in section two in order to provide a context for the results. The limitations and options for future work are also outlined in the discussion. A final section provides conclusions.

4.2. Literature review

4.2.1. Introduction

Abundance of stream-dwelling salmonids is influenced by two broad categories of process (Milner *et al.*, 2003):

1) density-dependent feedback mechanisms, such as territorial competition or limited food availability, which can be said to truly regulate abundance; and
2) density-independent processes (such as climate), which act to determine abundance and, because they can have large effects on survival, may obscure the underlying density-dependent process.

It is an assumption of this work that fish populations within the rivers are habitat limited. This leads to the hypothesis that fish populations will be altered by the changes in habitat resulting from altering compensation flows. Thus, the first section of this review focuses on density dependence theory, and the mechanisms at work. All of the research related to density dependence which is applicable to this research has been focussed (e.g. Elliott, 1987; Elliott and Hurley, 1998) on Salmonids, in particular sea and brown trout (*Salmo trutta*). Therefore, this section will focus on the density dependence of Salmonids. Evidence for the presence of density dependence will be presented. As was apparent in the previous chapter, despite being regulated, these rivers are not 'closed' systems. They are subject to a degree of natural variability. Under such conditions, the tenets of density dependence become weaker, and the ways in which this happens will also be addressed.

4.2.2. Density dependence

Density-dependent mortality is thought to operate only for comparatively short periods of the life cycle, during critical stages, when regulation is achieved by competition for limited resources (Milner *et al.*, 2003). Given the territorial nature of juvenile salmonids and their requirement for food, there is clearly a limit to the number of fish that any stream can support. At low spawning densities, because competition is limited, reproduction is efficient and the number of juveniles produced is closely proportional to the spawning level

(Milner *et al.*, 2003). As spawning numbers increase so does competition amongst the fry[1] and density-dependent factors begin to restrict the population as the carrying capacity is approached (Milner *et al.*, 2003).

Elliott (1987) and Elliott and Hurley (1998) examined a high density anadromous[2] and a low-density resident[3] populations in two nearby UK streams and showed that in the high density, sea trout population, recruitment (i.e. the number of juveniles entering the population to create a new year-class) was regulated by a dome-shaped density dependent mortality acting on the younger juveniles related to the initial abundance of eggs. However, recruitment regulation in the juvenile stage was absent in the low-density, resident population where simple proportionate survival occurred in the early life stages. In both populations, density-dependent mechanisms accounted for just under half the variation in recruitment between generations, implying that density-independent factors were responsible for the remaining variation (Elliott, 1994; Elliott and Hurley, 1998).

Elliott *et al.* (1997) also found strong evidence for density-dependent population regulation and identified extreme outliers, all these being for year-classes affected by the summer droughts. It was concluded that survivor density at different stages in the life cycle was (density) dependent on egg density at the start of each year-class (Elliott *et al.*, 1997). There was also a significant density-dependent relationship between total egg production by the surviving progeny and egg density at the start of each year class (Elliott *et al.*, 1997). In a long-term study of sea trout, Elliott (1993a) showed that regulation of population size was achieved through density dependent mortality operating over a short critical period (30-70 days) when the fry dispersed from the redds. Thereafter, survival was shown to be proportionate, influenced by density-independent factors. Further studies have demonstrated the occurrence of density-dependent mortality during the early stages of free swimming life. Examples in the United Kingdom include Gee *et al.* (1978); Buck and Hay (1984); Elliott (1984); and Gardiner and Shackley (1991). In addition, Elliott (1987) described the resident trout population in Wilfin Beck and concluded that it was not

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[1] Fry is a term commonly used to refer to a brown trout which is under one year old.
[2] Anadromous trout are those which spend part of their life cycle at sea; and part in rivers.
[3] Resident trout is a term used to define one which does not migrate.

regulated by density dependent mortality although, as will be seen, the method by which this population is regulated remains obscure.

However, where conditions are variable and frequently severe, or the habitat has been degraded, in most years egg input will be appreciable but survival of the intragravel stages will be so low that the population density of fry and parr will be too low for density-dependent mortality to operate (Crisp, 1993). In such populations, midsummer population densities will be generally low, very variable between years and with total failure of recruitment (at least locally) in some years (Crisp, 1993). Crisp *et al.*, (1974) suggested that populations in part of the upper Tees might be of this type. The most likely mechanism for reducing the survival of the intra-gravel stages in these Pennine streams is washout during spates (Crisp *et al.*, 1984; Crisp, 1989). Before density-dependent mortality occurs in the Cow Green streams (egg inputs and survival to swim-up), it is clear that the initial population density must exceed 10 fish m^{-2} (Crisp, 1993). For early months of free swimming fry Crisp (1993) found populations consistent with two densities:

1) At initial densities below about $1.5/m^2$, trout parr have very few encounters with one another, do not show marked territoriality and forage widely. This has two consequences: (i) the increased activity and wandering leads to a predominantly downstream shift of population and to relatively high downstream dispersal rates; and (ii) foraging is highly efficient and gives high growth rates.
2) At some higher density (probably 3-5 swim-up fry m^{-2}) the fish become increasingly territorial and become limited, in terms of food supply, to that produced in or drifting through their territory. Growth rates then vary very little with population density, unless or until density becomes so high as to produce a significant shortage of food or of some other essential resource. In the Cow Green streams there was no evidence of this latter mechanism, at least up to densities of 10 swim-up fry m^{-2}. As initial population density increases, downstream dispersal occurs earlier so that the dispersed fish are healthy and are able to establish themselves elsewhere. In addition, the importance of dispersal relative to mortality, as a cause of loss, increases with initial population density.

There are a vast number of studies which demonstrate density dependent losses from populations (Gee *et al*, 1978; Egglishaw and Shackley, 1980; Gardiner and Shackley, 1991;

Crisp, 1993; Crozier and Kennedy, 1995; Jonsson *et al.*, 1998; Nordwall *et al.*, 2001; Cattaneo *et al.*, 2002; Dumas and Prouzet, 2003). In a recent review, Milner *et al.* (2003) conclude that the evidence for density-dependent regulation of abundance in the salmonid life cycle is overwhelming.

Growth rate might be expected to be influenced by fish density, because it represents the individuals success in acquiring energy through food (Milner *et al.*, 2003). But results on this point have been inconsistent, which may partly reflect species differences as well as the circumstances of individual studies (Milner *et al.*, 2003). Elliott (1984) could not detect a relationship between population density and growth of brown trout in Black Brows Beck. For sea trout in Black Brows Beck, mean growth rate and mean size were found to be independent of density, although variability in size was inversely density-dependent during the critical period (early stages of free-swimming life) for sea trout (Elliott, 1994). Gardiner and Shackley (1991) showed that growth was density-dependent in salmon over the first growing season, but Gee *et al.* (1979) were unable to demonstrate density dependent growth (expressed as production/biomass ratios) in salmon in the River Wye. Crisp (1993) found that estimated instantaneous growth rate per day of 0 group fish from swim-up to August and from swim-up to October was inversely related to the natural logarithm of August population density, and this was most apparent for August densities of <0.15 fish m^{-2}.

Gibson (1993), in a review of salmon production, reported inconsistencies between various authors, in the effect of density on growth, and suggested that this may have been due to differences in food availability and/or habitat between different studies. Gibson (1993) noted that over a range of streams varying greatly in productivity, high growth was positively associated with high density in relatively rich sites, but, in a single stream type, higher density usually gave lower growth. Comparisons should thus only be made under similar habitat and productivity conditions (Milner *et al.*, 2003).

This review has shown that density dependence can and does occur in brown trout populations. Where it does occur, increasing the capacity of the system may reduce any density dependence and allow population growth. Hence, habitat may be limiting: however, as argued in the next section, habitat limitation is more likely to be associated with density independence as the range of possible population limiting factors commonly holds populations below the their density dependent cases.

4.2.3. Habitat-limitation of brown trout.

The relative importance of endogenous feedback mechanism versus environmental factors in the dynamics of animal populations is a long-standing, but not fully resolved yet, issue in ecology (Lobon-Cervia and Rincon, 2004). Endogenous regulation has been considered self-evidently necessary for population persistence and temporal stability, and empirical evidence of its occurrence in wild populations has been steadily accumulating (Sinclair, 1989; Turchin, 1995, 1999). In contrast, the role of density independent factors (e.g. environmental variability) is less well understood (Ricklefs and Miller, 2000). However, density-dependent and density-independent factors need not be considered mutually exclusive (Harrison and Cappuccino, 1995). There are a number of factors which could impact upon habitat limitation (Figure 4.2.1).

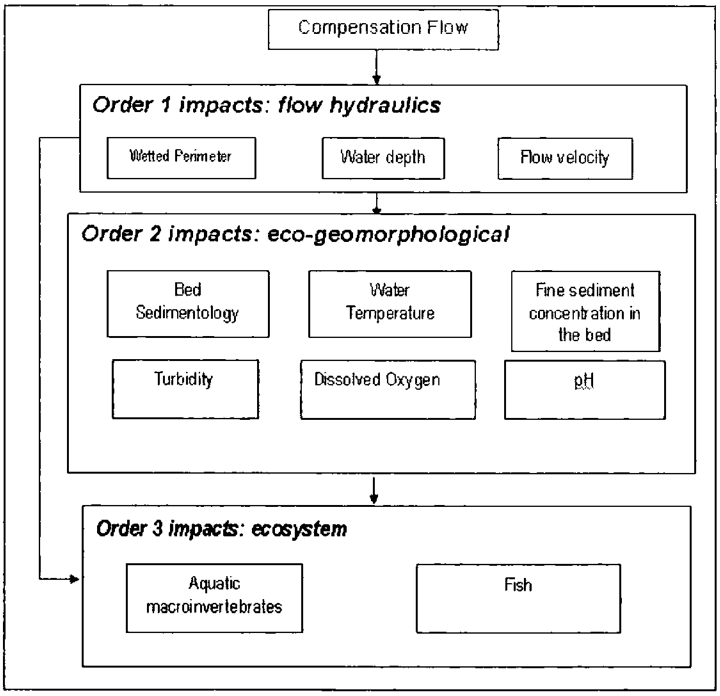

Figure 4.2.1. Figure to show the potential impacts of compensation flow releases (adapted from Petts, 1984).

Thus, habitat limitation factors include a wide range of variables (Figure 4.2.1) that cause sometimes extensive, but unpredictable mortality at any stage in the life cycle and at any density (Milner *et al.*, 2003). Thus, habitat limitation acts so as either: (i) to reduce the extent of density dependence; and/or (ii) to control the point at which a population becomes density dependent.

4.2.3.1. Depth

One of the first factors which may impact upon the density dependence or otherwise of brown trout is the depth of the water produced by the interaction of the discharge and topography. Kennedy and Strange (1982) found considerable niche segregation of each age class in relation to water depth. They found that trout fry were negatively correlated with water depth. It is not yet known if fry are significantly more abundant in shallow flow as a result of choice or competition (Kennedy and Strange, 1982). However, the range of fry habitat diversity in the presence of older age classes found by Kennedy and Strange (1982) suggests that deeper water could be colonised by greater numbers of fry in the absence of competition from older fish. Kennedy and Strange (1982) found that 1+ brown trout were significantly positively correlated with depths of 15.1–30cm, and >30cm, whilst significantly negatively correlated with shallow depths of 0-15cm. The fact that older trout prefer deeper habitats than younger specimens has been highlighted in many works (Bagliniere and Champigneulle, 1982; Kennedy and Strange, 1982; Heggenes, 1989; Greenberg *et al.*, 1996)

Elso and Giller (2001) found that there was a proportionally greater number of fish in deeper pools than in shallower ones in summer and autumn, suggesting that trout use the available habitat (i.e. the pool) as a three dimensional space in which an increase in the third dimension (depth) leads to a proportionally greater number of fish per unit area. Furthermore, the density of fish (fish m^{-2}) was correlated significantly and positively with water volume in autumn and summer but not in winter and spring (Elso and Giller, 2001). While in summer and autumn large pools supported proportionally more fish than small ones, in winter and spring the number of fish in large pools was proportionally smaller than in small pools (Elso and Giller, 2001). The study by Elso and Giller (2001) highlighted the importance of the riffle-pool sequences in river morphology on brown trout populations, with pools of different morphology, depth, and physical characteristics being differentially suitable for trout.

4.2.3.2. Velocity

As with the depth, the velocity distribution within each of the rivers will determine whether the population is habitat limited or not. A velocity tolerance range at spawning of 15 to 90 cm/s, with an optimal range of 40 to 70 cm/s, was assumed by Raleigh *et al.*, (1986). Heggenes and Traaen (1988) showed that brown trout larvae entering the free-feeding stage are sensitive to water velocities for a few weeks and are not able to resist velocities above 25cm/s. Although their sensitivity declines as they grow, they are only able to withstand velocities of up to 50cm/s, on average, after two months. Less extreme water velocities, although not displacing fish downstream, may drive young trout to seek nonoptimal habitats, with higher energy costs or lower food availability, therefore possibly leading towards mortality (Elwood and Waters, 1969).

Vismara *et al.* (2001) calculated water velocity suitability curves for both adult and juvenile trout which showed optimum values for low current velocities (<20 cm/s). As velocity increased above the optimum, juvenile preference decreased, whereas the curve for adults remained relatively constant (Vismara *et al.*, 2001).

4.2.3.3. Substrate

Most salmon locate their redds in areas of accelerated flow, so that downwelling currents will force streamflow into and through the substrate (Alonso *et al.*, 1996). Many authors report that salmonids prefer sites of downwelling at the tails of pools (Burner, 1951; Stuart, 1953; Hoopes, 1972). Brown trout prefer gravel with a diameter of about 1.0 to 7cm for spawning substrate (Frost and Brown, 1967), but utilise gravel from 0.3 to 10cm (Raleigh *et al.*, 1986).

Once the eggs have been deposited, the intragravel stage represents a critical phase of the life cycle, as during this period they are susceptible to damages from high levels of fine sediments (Alonso *et al.*, 1996) which reduce intragravel flows. The latter must be sufficient to remove the toxic metabolic wastes produced by the eggs (Alonso *et al.*, 1996). High levels of fines reduce the permeability of the gravel bed, resulting in lower intragravel flows and as reduced interchange of water between the main stream flow and the intragravel flow (Alonso *et al.*, 1996). Fry raised at lower dissolved-oxygen levels generally have lower survival rates, emerge later, and are usually less able to compete with other fish raised in higher levels of dissolved oxygen (Alonso *et al.*, 1996). Excessive amounts of fine sediments (which carry organic matter) exert a sediment oxygen demand that reduces the availability of oxygen to salmonid eggs and entrap the fry within the gravel as they try to emerge (Alonso *et al.*, 1996). Raleigh *et al.* (1986) stated that the optimal spawning gravel conditions for brown trout are assumed to be <5% fines; >30% fines are assumed to result in low survival of embryos and emerging fry. But in regulated systems, fine sediment deposition rates are less important because of the role of the reservoirs in sediment attenuation.

Substrate composition is of further importance as smaller fish tend to select areas with cobble substrate but larger fish prefer areas with boulders (Greenberg *et al.*, 1996). So in order to maintain a balanced population of trout a balance must be struck between areas which are cobbled and areas which contain boulders.

4.2.3.4. Flow variability

It was seen in the chapter 3 that its not just the interaction of discharge and topography which is providing habitat variability, but temporal variability is provided in the study rivers by tributary input and reservoir overtopping. Therefore an understanding of the temporal aspects of streamflow and habitat is essential to designing water management schemes intended to protect, enhance or restore riverine fish populations (Stalnaker *et al.*, 1996).

Functional links between recruitment and hydrological factors have been described for a variety of stream-living populations of brown trout (Soloman and Paterson, 1980; Jensen and Johnsen, 1999; Spina, 2001). Poff and Allan (1995) found that hydrologically variable river habitats (high coefficient of variation of daily flows, moderate frequency of spates) in Wisconsin and Minnesota, U.S.A., tended to support fish that are feeding and habitat generalists while stable habitats were characterised by a higher proportion of specialists. Bain *et al.* (1988) found that extreme flow variability acts to impose functional habitat homogeneity. The reduction and elimination of shallow and slow flow loving species under fluctuating habitat conditions indicates that this group was not able to persist effectively in its particular microhabitat even though that microhabitat always existed (Bain *et al.*, 1988). They found that without the functional availability of shallow, slow, shoreline areas, the stream environment become one general type of usable habitat, which Bain *et al.* (1988) found to be dominated by generalist species.

Lobon-Cervia and Rincon (2004) found that population size appeared largely determined by recruitment, whilst the abundance of the parental stock was a poor predictor of recruitment. They also found that survival rates were maximised in years of intermediate discharge and attained similarly low magnitudes in years of lower and higher discharge. They therefore surmised that environmental variability in the form of year to year variation in discharge was the major determinant of population dynamics with endogenous density-dependent mechanisms playing, at most a limited role. Environmental variability operated through a limiting factor namely the availability of suitable micro-habitats for juveniles shortly after emergence, which in turn resulted from the interaction between a regional phenomenon such as rainfall (i.e. water discharge) and local stream channel morphology (Lobon-Cervia and Rincon, 2004). Whilst the year-to-year variation in the regional factor determines the strength of the annual recruitment, the physical structure of the site introduces small-scale, yet functionally relevant spatial heterogeneity into the process (Lobon-Cervia and Rincon, 2004). Maximum recruitment appeared to occur at the most frequent discharge conditions and extremes at both ends of the spectrum resulted in decreased recruitment (Lobon-Cervia and Rincon, 2004).

Lobon-Cervia and Mortensen (2005) used 17 successive years of data to show that population size of juveniles were chiefly influenced by annual recruitment that in turn, was determined by stream discharge and annual temperature. These patterns matched those previously highlighted for a resident population located >2500km away and emphasised the importance of environmental (climatic) variability as a major regulating agent of population size in brown trout. Patterns found in brown trout in a stream in Norway relating discharge with recruitment and recruitment with year-class strength fully matched the patterns previous found in the Spanish stream and suggests the occurrence of similar mechanisms underlying temporal variations in the numerical abundance of these two populations (Lobon-Cervia and Mortensen, 2005). However, a major divergence between the two populations concerns the shape of the recruitment-discharge relationships. In the Norweigan stream, recruitment declined curvilinearly with increasing discharge. In the Spanish stream, recruitment described a two phase trajectory with increasing discharge up to a point at which the direction of the trajectory switched towards a continuous decline of recruitment with increasing discharge (Lobon-Cervia and Mortensen, 2005).

However, Jowett and Duncan (1990) found no significant relationship between brown trout abundance and flow variability, over 130 sites across New Zealand. Furthermore, Einum (2005) suggests that the findings of Lobon-Cervia and Rincon (2004) should be treated cautiously. Einum (2005) points out that in Lobon-Cervia and Rincon (2004), 51-77% of the variation in recruit densities could be explained by variation in the amount of available juvenile habitat caused by variation in discharge. It should be no surprise that detection of density dependence from traditional time-series analyses under such circumstances is difficult (Einum, 2005). Such environmental forcing of densities was claimed to indicate that endogenous regulation has little importance for the dynamics of the population (Einum, 2005). However, in the terminology of Berryman (1999, 2004), this is an example of fluctuations in the limiting factor, and under such circumstances models for the process of population regulation and detection of such regulation should be based on the ratio of population size to its limiting factor (Einum, 2005). Thus, if the brown trout population in question is regulated through density dependent survival due to space limitation during early juvenile stages, loss rates over the period from egg to 4-month-old juveniles should increase as the ratio of egg density to amount of suitable juvenile habitat increases (Einum, 2005). Einum (2005) found that it was the presence and not the absence of endogenous regulation that causes the population abundance to track environmentally induced variation in the limiting factor.

4.2.3.5. Discharge immediately after emergence

Elliott (1993a) found that regulation of population was achieved through density dependent mortality operating over a short period (30-70 days) when the fry dispersed from the redd. But, many authors think that density independent factors are more important at that time.

Cattaneo *et al.* (2002) found that hydrology only constrains trout dynamics during the critical emergence period, after which cohort interactions regulate the 0+ density (as water level increased 0+ abundance decreased) (Heggenes and Traaen, 0+ vulnerable). No relationships were found between the 1+ fish and the hydrological variables. Adult fish were not related to any hydrological variables, but were highly related to the abundance of 1+ fish the previous year (Cattaneo *et al.*, 2002). They also found that the model relating the 1+ fish density to the 0+ fish density had a slope significantly smaller than one, therefore suggesting density dependence within the 0+ cohort survival. However, the exact mechanism that links high discharges to reduced 0+ density is not well identified (Cattaneo *et al.*, 2002). A possible cause is the flushing of the 0+ fish as a consequence of their ability to maintain their stream position or to find shelters when the water velocities increase during high discharges (Cattaneo *et al.*, 2002).

Surprisingly, Cattaneo *et al.* (2002) found a significant negative relationship between the 0+ fish at year n and the 1+ fish the previous year, and they do not think this relationship has been found elsewhere. They speculated two reasons for this relationship:

1) The microhabitat used by the 0+ fish largely overlaps that used by 1+ fish, after which a change in microhabitat use appears when the fish becomes 2+. At the time of their sampling, the 1+ fish of the previous year just changed their microhabitat use, but there could have been some interactions (competition or predation) with the 0+ fish that searched for key positions.
2) High densities within a reach were mainly due to the 0+ and 1+ fish, so it is possible that high densities of 1+ fish might disturb spawning, and therefore lead to a low 0+ cohort the following year.

Lobon-Cervia (2004) looked at patterns of spatial variation in the population dynamics of brown trout across Rio Esva by using the residuals from stock-recruitment relationships as indices of survival rates of spawner-to-recruit (STR), spawner-to-cohort size (STC), and spawner-to-spawner (STS). The analyses produced highlighted the importance of discharge during, or just after, trout emergence as a major determinant of recruitment whose effects are reflected in the population over the lifetime and emphasised the synchrony between environmental processes and brown trout dynamics (Lobon-Cervia, 2004).

Spina (2001) used data from an 11 year study of a population of brown trout to show that peak discharge at the time of incubation was inversely related to year-class strength (density of young of the year estimated in the fall), which suggests that scouring of nests is a source of temporal variation in the density of the young of the year. Relationships between cohort (individuals spawned in the same year) age-groups provide a clue that discharge-related year-class strength might influence the population dynamics of a year-class over time (Spina, 2001). The results of this study are consistent with those of other investigators reporting inverse relationships between fish abundance and discharge (Spina, 2001). Some researchers have found negative relationships between discharge at the time of incubation and salmonid abundance (Thorne and Ames, 1987) and survival (Holtby and Healey, 1986); others have observed extremely depressed abundance of young of the year of various resident trout species following relatively high discharge events during incubation (Seergrist and Gard, 1972; Erman *et al.*, 1988; Hayes, 1995). Depressed abundance of brown trout young of the year following high discharge events has not always been observed (Lobon-Cervia, 1996). The finding that suggests that lower discharges lead to more young of the year should be interpreted cautiously because low discharges are believed to affect spawning success adversely (Davies, 1991) and limit salmonid abundance (Gibson and Myers, 1988). Although discharge at the time of incubation is a parsimonious explanation for the temporal variation in the year-class strength of brown trout, the regression results do not necessarily imply that the observational variables are related in a causal manner (Spina, 2001).

Discharge-limited survival rates of the youngest juveniles have been described for a variety of populations in the UK (Solomon and Paterson, 1980; Milner *et al.*, 1993; Elliott *et al.*, 1997).

4.2.3.6. Temperature

Brown trout showed significantly faster growth in scarcely covered or uncovered sections compared with more shaded sections (Reyes-Gavilan *et al.*, 1995). Shading can negatively affect fish growth by decreasing light availability for autochthonous primary production, thus diminishing benthos abundance (Angermeier and Karr, 1983; Behemer and Hawkins, 1986); by decreasing water temperature (e.g. Brown and Krygier, 1970); or by impairing the efficiency of prey capture (Wilzbach and Cummins, 1986) and reducing the effective time for food searching, given that salmonids are visual feeders (e.g. Wankowski and Thorpe, 1979).

Saltveit (1990) found that brown trout were smaller beneath a cold water discharge into the river when compared with brown trout sampled upstream of the cold water discharge.

4.2.3.7. Low Flows

Cowx *et al.* (1984) looked at the effects of a drought (1976) on brown trout populations. A comparison of recruitment between the 1976 and 1977 spawning years showed that recruitment of trout was similar (Cowx *et al.*, 1984). The only detrimental effect on the Afon Dulas of the 1976 drought was the elimination of the young-of-the-year salmon (Cowx *et al.*, 1984). Some compensation for this loss to the Dulas population was indicated by evidence of increased recruitment during 1977, in that the numbers of juvenile salmon surviving beyond the fry stage was greater in 1977 than in 1975. On the other hand, the trout population which showed no unusual reduction in numbers as a result of the drought, maintained a similar level of recruitment in 1977 as in 1975. Ricker (1954), LeCren (1973) and Gee *et al.* (1978) suggested that salmonid population density is regulated by density dependent mechanisms of which the most likely is aggressive territorial behaviour. The loss of the 1976 year class, therefore, probably resulted in reduced territorial aggression amongst the parr present in 1977, and led to increased survival of the fry which hatched during 1977 (Cowx *et al.*, 1984).

In a broader study of the of the rivers of the upper Severn catchment Cowx *et al.* (1981) observed that the failure of the 1976 year class of salmon was restricted to the unregulated Afon Dulas. In the regulated streams Vyrnwy and Clywedog, where the summer ambient temperatures were depressed by the release of reservoir bottom water, no evidence of the elimination of the 1976 year class of salmon could be found. It has already been shown that growth in freshwater was poor for year classes affected by the summer droughts of 1976, 1983 and 1984 (Elliott, 1985; 1993b), but a detailed assessment of the effects of droughts on growth rates has yet to be made.

Brooker *et al.* (1977) thought that a decrease in oxygen concentration was the main cause of high mortalities in adult salmon, and was caused by the death and decay of the submerged macrophyte *Rannuculus fluitans* Lam., the decay under drought conditions being accelerated by high water temperatures. Cowx *et al.* (1984) attributed mass mortality (of Salmon parr but not trout) to the 1976 drought to the Salmon parr being being less tolerant of sustained higher water temperatures, but subsequent experimental work has shown salmon parr to have a higher thermal tolerance than trout (Elliott, 1991). However, no mass mortalies from the drought of 1976 in Black Brows Beck were reported in the study by Elliott *et al.* (1997).

In Black Brows Beck, Elliott *et al.* (1997) estimated parameters of a Ricker stock-recruitment model for different life stages of sea trout (*Salmo trutta*) population and demonstrated that outliers of this model generally corresponded with years that experienced a summer drought. The 0+ and 1+ stages were the more sensitive and were negatively affected by summer droughts, possibly by reduction of available suitable habitat.

4.2.4. Bullheads

Much less research has been conducted on the bullheads compared to the trendy salmonids such as brown trout and Atlantic salmon. Because of this, our current knowledge on the biology of the bullhead (*Cottus gobio* L.) stems largely from comparatively near-natural running waters (Smyly, 1957; Mills and Mann, 1983; Marconato and Bisazza, 1988; Gaudin and Caillere, 1990). The Bullhead is common in upland Millstone Grit streams, and is the only freshwater cottid found in the UK (Smyly, 1957). The Bullhead rarely exceeds 15cm in length and a weight of 28g (Maitland and Campbell, 1992). The Bullhead has a large head and a dorso-ventrally flattened tapering body adapted to life on the bottom of flowing waters, amongst other adaptations (Tomlinson and Perrow, 2003). Bullheads are found predominantly in stony streams with moderate flow and cool, oxygen rich water (Tomlinson and Perrow, 2003).

Habitat analyses indicate that the minimum depth of patches selected by bullheads is 10cm (Roussel and Bardonnet, 1996) (From: Fischer and Kummer, 2000). Based on a literature review, Stahlberg-Meinhardt (1994) specifies the average density of bullheads in suitable habitats of small streams in Europe to be in the range of 50-100 ind/m^2 (from: Fischer and Kummer, 2000). These values generally ignore or under-represent juvenile stages (Fischer and Kummer, 2000). Mills and Mann (1983) who included one-year-old bullhead, reported densities of up to 2100 ind/m^2 .

4.2.5. Brook Lamprey

The brook Lamprey is a primitive jawless fish resembling an eel. The ammocoetes are only to be expected where the current is sluggish (Hardisty, 1944). They are often found in eddies or backwaters below obstructions such as fallen trees and projecting roots, or on bends in the stream, in accumulations of silt covered by a layer of decaying leaves, sticks and debris (Hardisty, 1944; personal observation). Although the type of mud inhabited by the larvae is said to vary, the deposit in which they were most numerous consisted of fine silt blackened by decaying organic matter (Hardisty, 1944).

4.2.6. Fish populations in a river regulation context

There has been no previous study such of the type reported herein. However, there has been a number of water resource studies which have investigated the abundance and growth rates of brown trout and to some extent Bullheads in relation to altering flow. Many have concentrated on fisheries population following the implementation of impoundment, and others have been in a more general water resource context. These will be discussed below.

Crisp *et al.* (1983) looked at the effects of river regulation on bullheads and brown trout beneath Cow Green Reservoir, in the upper Tees valley, and compared it with an unregulated control tributary. A major feature of the effects of regulation in the Tees was substantial reduction of diel water temperature fluctuations (Crisp *et al.*, 1983). They found that in experimental releases of 3.66 to 6.7m/s during very hot weather in July the water temperature in the Tees below the confluence of Maize Beck fluctuated between 11.5 and 18.5°C over a 6-day period, with daily means of 12.1-15.3°C. In contrast, the temperature of Maize Beck (unregulated) during the same period ranged from 13.1-27.5°C with daily means of 17.0-20.3°C. Crisp *et al.*, (1983) found no difference in observed or back-calculated lengths of brown trout between either of the sites or any periods (i.e. the implementation of regulation did not appear to make any difference to growth rates).

Crisp *et al.* (1983) observed instantaneous rate of growth (in length of a trout weighing 18.6g which can be derived from Crisp (1977)), was found to be significantly higher after regulation than before. This metric is useful as it includes a larger number of fish and covers the full range of available fish lengths. But this increase was only very small. Artmitage (1978) showed that the standing crop of benthos increased in the Tees after regulation and this suggests an increase in available food for fish, though estimates of benthic production and information in availability and acceptability to the fish would be required to prove this. However, it is evident that from the results of Crisp *et al.* (1983) that conditions in the Tees beneath Cow Green reservoir have become more favourable for the fish possibly, but not necessarily, as a result of increased food supply. Whatever the resource concerned, it is clear that, as temperature substantially limits growth rate for trout, the improved conditions have been explored through an increase in mean population density and biomass rather than through a marked improvement of the growth of individuals (Crisp *et al.*, 1983). Crisp *et al.*, (1983) found that regulation (for 5 years after regulation commenced) in the Tees led to a significant increase in mean minimum population of brown trout, whereas no such change could be demonstrated in the unregulated Maize Beck. The fact that the estimated minimum population densities in the Tees after regulation showed a negative correlation with discharge may indicate either that a permanent increase in density was masked by sampling limitations during high discharges, or that fish moved into the reach during low discharges and out during high (Crisp *et al.*, 1983). This mobility of fisheries species is thus both a blessing and a curse in studies such as this one. If, the habitat quality within the study reaches has improved we want fish to move into the new habitat space, however, we are unable to sample the whole river and it is possible that the fish may just move to a different part of the river, hence leading to a observed decrease in density. Thus, a number of sampling sites must be used to try and lessen the impact of this.

The before and after comparisons in the Tees were examined in the light of similar comparisons made in the unregulated Maize Beck (Crisp *et al.*, 1983). The use of Maize Beck as a 'control' was valuable in ensuring that changes observed in the Tees were probably, in part at least, a result of regulation rather than being solely a result of some phenomenon (e.g. a series of mild winters) which was nor causally related to regulation, and which might be expected to have similar effects both in the regulated and unregulated streams (Crisp *et al.*, 1983).

Solomon and Paterson (1980) investigated the influence of natural and regulated streamflow on survival of brown trout (*Salmo trutta* L.) in a chalkstream. They found that numbers of 0+ trout in October and annual survival of fish over two years old, were closely correlated with mean April flow (discharge) in a groundwater fed stream. A pilot flow regulation scheme, which enhanced flows from May to September in a drought year, failed to increase survival. The pumping scheme (May-September 1976) turned a very low discharge spring into an average discharge summer; yet the 0+ population was the lowest of all the years studied, indicative of a drought situation.

The relationship of 0+ numbers with discharge with apparent independence from number of eggs, is suggestive of density dependent competition for a resource which is limited by discharge (Solomon and Paterson, 1980). It is not consistent with a mechanism operating on survival of eggs or young fish before emergence from the gravel; in that case, numbers of 0+ fish would not be independent of the numbers of eggs (Solomon and Paterson, 1980). It is suggested that the influence of discharge on 0+ on the Candover operates by rendering greater areas of the stream suitable for young fish territories, and possibly by increased velocities decreasing the area defended by each fish (Solomon and Paterson, 1980). From the apparent timing of this influence (before May) it would seem to operate at the time of establishing initial territories (Solomon and Paterson, 1980).

Cowx and Gould (1989) looked at the effects of imposing stream regulation on brown trout in the upper Severn catchment. In the regulated Afon Clywedog, juvenile recruitment of brown trout decreased steadily following the utilisation of the impoundment for regulation (Cowx and Gould, 1989). The decline was considered the response to the variable and rapidly changing discharge regime adopted in the river (Cowx and Gould, 1989). Spawning success and juvenile survival in the Vyrnwy was relative stable suggesting the flow regime in this river was less devastating (Cowx and Gould, 1989). This contrasts with the findings of Crisp *et al.* (1983), which indicated enhanced trout populations with similar regulation discharges on the River Tees, north-east England.

Harpman *et al.* (1993) looked at the changes in fish populations after altering releases from Taylor Park and Blue Mesa reservoir system on the Taylor river. Prior to 1975 the reservoirs in this system operated in a "fill and spill" fashion , in which, inflows from the Taylor river were stored until reservoir capacity was reached and the remaining inflows overflowed (Harpman *et al.*, 1993). Little water was released during other periods and the recorded flow in the Taylor river reached zero on a number of occasions (Harpman *et al.*, 1993). He reported that Nehring (1988) discovered a significantly increased brown trout population following the institution of a new flow regime. Harpman *et al.* (1993) argued that the two key components of the new flow regime were firstly, increased minimum flows during the critical winter months (which supports a higher brown trout population from year to year) and secondly, reduced fluctuations in the autumn and winter (which improved spawning success and increased recruitment).

Aass *et al.* (1989) found that the brown trout population of a Norweigan river showed no response to a 93% reduction in winter discharge. Harris *et al.* (1991) showed that a four year period of a five-fold increase in the minimum flow did not alter the brown trout population in a small stream on the west coast of the USA.

Milner *et al.*, 2003 proposed that the abundance of stream-dwelling salmonids is influenced by two broad categories of process:

1) density dependent feedback mechanisms, such as territorial competition or limited food availability, which can be said to truly regulate abundance;
2) density-independent processes (such as climate), which can act to determine abundance and, because they can have large effects on survival, may obscure the underlying density-dependent processes.

If these two broad categories are related back to the compensation flow problem, perhaps the first category can be though of as acting when base compensation flow alone is acting upon the systems. Thus assuming that there were no other factors influencing the abundance of brown trout, it perhaps would be expected at baseflow that the trout populations would be both density dependent and habitat limited.

However, Chapter 2 has explored the importance of flow variability in each of the study sites; and numerous references in this section have highlighted the influence of flow variability (e.g. Jowett and Duncan, 1990) and timing of the variability (e.g. Elliott, 1993a) as influencing the density dependence of the trout in the stream. Such processes will mean that the trout populations within the study rivers are unlikely to be density dependent outside of the early emergence period where density dependence has been seen in salmonids in many instances (e.g. Gee *et al.*, 1978; Buck and Hay, 1984; Elliott, 1984; Gardiner and Shackley, 1991).

The flow variability and any other factors which impact upon the trout population (e.g. substrate; pollution) will make detecting the impacts of compensation flows difficult. To determine whether a population of trout is density dependent requires a specially designed sampling system, and this is not really of interest in this work. What is of interest from a fisheries perspective is firstly whether the habitat within any of the systems is limiting and secondly whether there are any changes to the fisheries populations when the compensation flows are altered (and so the perhaps the habitat altered).

4.3. Methods

This section introduces and justifies the data collection and analysis methods. The first sub-section outlines electrofishing methodology used and introduces the study sites. The second sub-section explains and justifies the data analysis method used: population estimates, length-frequency histograms; growth rates and HABSCORE (Wyatt *et al.*, 1995; Barnard and Wyatt, 1995).

4.3.1. Data collection

Fisheries surveys were conducted at 21 sites on the three study rivers (Table 1) in August and September for four sampling years: 2002; 2003; 2004; and 2005. The surveys were conducted during this same period each year, so that the 0+ trout would be able to survive the electrofishing process. Four years data were collected in order to have two years pre change data and two years post change data, which will allow characterisation of the sites and an examination of the potential impacts of compensation flow regimes.

The same quantitative electrofishing method was used in each year. It involved four operatives with a single anode electric fishing apparatus. The equipment comprised a 3.0. kVA generator powering an 'Electracatch' control box to produce a smooth DC output. Stop nets were positioned at the top and bottom of the site (with a site length of around 50m in each case), and the site was fished from the bottom up, three consecutive times, with each catch being kept separate. During the fishing exercise, as may fish as possible were caught in dip nets by the operatives who were positioned either side, and downstream of the anode. Following completion of each survey all fish were identified to species level, the fork length[4] was measured, and a number of scales were removed from Brown Trout (*Salmo trutta*) to allow calculation of growth rates.

As this method of electrofishing requires four field operatives and much specialist equipment, the Environment Agency organised and provided three out of four members on each of the teams. The scales were measured at the National Fisheries Laboratory. All subsequent data analysis was performed by the author.

The same length of river was sampled in each sampling year, apart from at the Coppice Cottages site on the Rivelin in 2005 where, due to some minor engineering works the exact same reach could not be fished in 2005, but a reach very nearby was fished instead. Due to the lack of temporal continuity, the 2005 data from this site was removed from any competitive or statistical analysis. Unfortunately, in 2002, no scales were taken from trout at the Hospital site (Rivelin), Olive Cottages site (Loxley) and Somershall Park site (Hipper), due to field error.

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[4] The fork length of a brown trout is the length from the nose of the trout to the area where the two forks of the tail meet.

River	Site Code	Site name	NGR
Rivelin	R1	Rivelin Mill	SK290873
Rivelin	R2	Hospital	SK297874
Rivelin	R3	U.S. Coppice Cottages	SK303876
Rivelin	R4	Coppice Cottages	SK304877
Rivelin	R5	Mill Dam	SK309876
Rivelin	R6	Glen Bridge	SK312878
Rivelin	R7	Roscoe Bridge	SK318883
Rivelin	R8	Allotments	SK321883
Rivelin	R9	Havelock Dam	SK324888
Loxley	L1	Stacey Lane	SK287905
Loxley	L2	Loxley Fisheries	SK288903
Loxley	L3	Storrs Bridge Works	SK293901
Loxley	L4	Old Wheel Farm	SK295898
Loxley	L5	Rowell Lane	SK299895
Loxley	L6	Olive Cottage	SK304895
Loxley	L7	Proroll	SK309894
Loxley	L8	Fiat Garage	SK316896
Loxley	L9	Pinegrove	SK319095
Hipper	H1	Somershall Lane	SK354700
Hipper	H2	Somershall Park	SK355701
Hipper	H3	Playing Fields	SK358702

Table 4.3.1. Table to show the name and location of the fisheries sites surveyed.

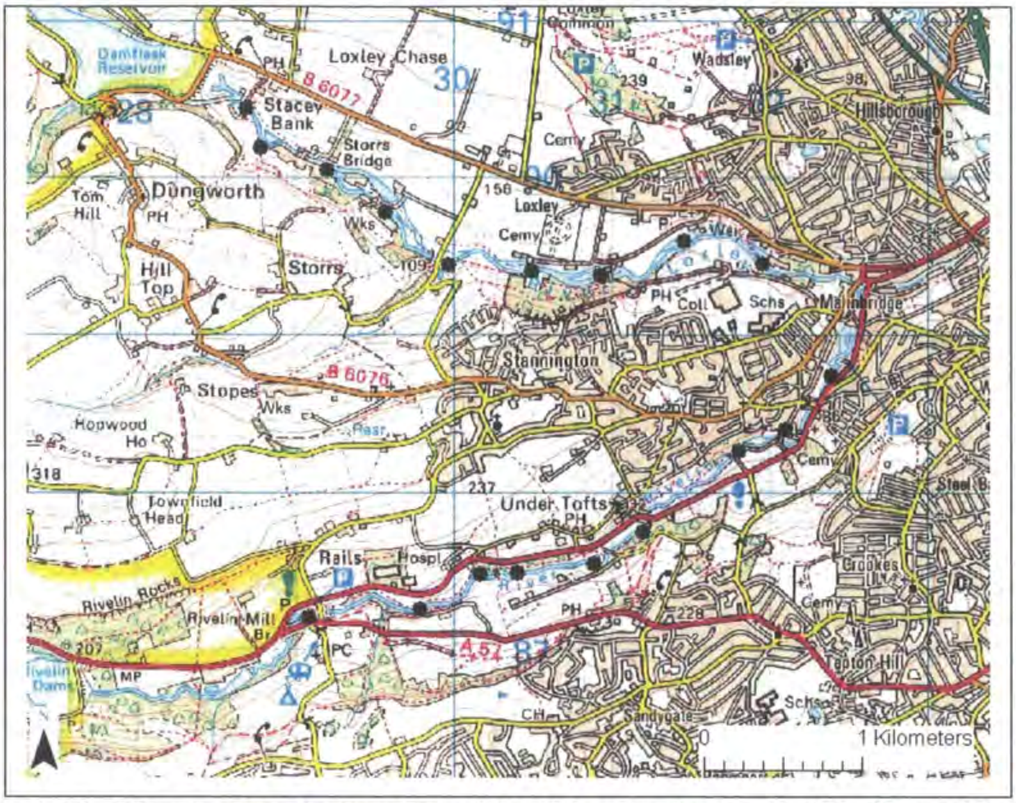


Figure 4.3.1. Map to show the location of the fisheries sites on the Rivelin and Loxley.

4.3.2. Data analysis methods

Population estimates

Gaining an understanding of the populations of brown trout in the rivers is useful for a number of reasons:

1) to establish a baseline trout population from which any changes in trout population can be assessed;
2) to establish whether the pre-change compensation flows had any effect on the trout populations in either of the rivers;
3) To assess whether the size of the trout populations changed as a result of altering the compensation flows.

The population estimates of brown trout at each site were calculated from estimates of absolute abundance based on the three-catch removal method using the Maximum Likelihood Method (Carle and Strub, 1978). The removal method assumes that changes in population size occur only through capture, and the probability of capture is equal for all individuals in a population during the removal sequence (Carle and Strub, 1978). Population estimates of minor species were not calculated as catches in the second and third runs were often greater than the first run, which contradicts one of the main assumptions of depletion sampling: that the population is reduced on each sampling run.

Density estimates (number of trout per $100m^2$) were calculated from the population estimates in order to standardise the populations to allow comparison between the rivers. The areas used for the standardisation were the areas found with the HABSCORE analysis, with the 2002 and 2003 population estimates standardised with respect to the 2003 HABSCORE data (i.e. the pre-change HABSCORE providing the pre-change area) and the 2004 and 2005 data standardised using the 2005 (post-change) HABSCORE data. The average widths recorded at the time of the fisheries surveys were very variable, but the width estimates provided by the HABSCORE data were more accurate.

The density was calculated for three age size classes, each with different habitat requirements: 0+; >0+ (<20cm); and >20cm trout. The densities were calculated in this manner for two reasons:

1) these age/size categories are those used by the HABSCORE analysis and hence allow for comparison; and
2) each of these age/size classes has different habitat requirements and so any variations in the densities of trout of these age size classes may represent an impact of the existing compensation flows and/or impacts from altering the compensation flows.

Length frequency histograms

The purpose of using the length frequency histograms was to help analyse the structure of the bullhead populations present at each of the sites:

1) to establish a baseline length of bullhead population from which any changes in bullhead population can be assessed;
2) to establish whether the pre-change compensation flows had any effect on the length frequency of the bullhead populations in either of the rivers; and

3) to assess whether the length frequency distributions of bullhead populations changed as a result of altering the compensation flows.

Length frequency distributions were created for the Bullhead populations for the 2003, 2004 and 2005 sampling years. Unfortunately, no Bullhead length data were recorded during the 2002 survey, due to changes in Environment Agency policy. However, no other aspect of the surveys changed.

Growth rates

Growth rates can be seen as representing an individual's success in acquiring energy through food (Milner *et al.*, 2003). Thus, the growth rates were used to assess for any potential impacts of the existing compensation flows, and any potential impacts from changing the compensation flows.

The age and growth of brown trout at each site were determined by the interpretation and counting of annual growth checks (annuli) which appear on the scales of the fish (Bagenal and Tesch 1978). These are formed during periods of faster and little or no growth, with the latter generally occurring during the winter months in temperate regions. When large numbers of scale samples were collected in surveys, sub-sampling of a representative size range was carried out to the Environment Agency Management System (EA-MS) (Britton 2003).

Scales from each individual fish were examined under a microfiche projector and the fish were aged by counting the number of annuli, taking care to note any false checks. More than one scale was examined to ensure correct interpretation of the annuli. The total scale radius and scale radius to each annuli was measured from the nucleus to the scale edge. The analysis of the data involved assessment of the relationship between the length of the fish, scale radius to annuli and total scale radius (Dahl-Lea method; Francis, 1990):

$$Li = (Si/Sc) \times Lc \qquad (4.3.1)$$

Where: Li is the length (mm) at age *i*; Si is the scale radius at length Li; Lc is the length at capture; and Sc is the scale radius at capture. For each brown trout, the length at age was calculated from the scale radius to each annuli at each age using equation 1. This gives a length at age *i* for each fish, and hence back-calculates the length of a given fish for all its previous ages. This increases the number of fish within the data set. The Dahl-Lea method was used because it is robust at small sample sizes such as those found in this study.

Considering the average length for the age of the fish captured in each year, includes the impacts of the cumulative nature of growth rate. As a consequence of this, the growth rate data were not analysed in terms of sampling year, but by separating the growth data into cohorts, and from there the growth periods can be compared directly. From these data, the percentage of standard growth (standard growth values given by the National Fisheries Laboratory (Table 4.3.2.)) can also be converted. The percentage growth was also standardised in such a way as to negate the effects of poor or good growth at the previous ages.

$$Percentage\ growth = \left(\frac{Amount\ grown\ in\ last\ year}{Amount\ \sup posed\ to\ grow\ in\ that\ year} \right) * 100 \qquad (4.3.2)$$

For example, the percentage growth rate of one year old trout is calculated as such: (Observed length at one year old / 80)*100, with 80mm being the standard growth rate for trout at one year old. For two year old trout the growth rates calculated as follows ((Observed length at 2 years old/Observed length at one year old)/(150-80))*100, with 150mm being the standard growth rate for two year old trout. This method allows us to compare the rivers in terms of percentage growth rates, and also in terms of standard growth rates.

Age of Brown trout	Standard growth rate (mm)
1	80
2	150
3	210
4	262
5	307

Table 4.3.2. Table to show the standard growth rates of brown trout, as provided by the National Fisheries Laboratory.

An example of the tabular output provided with this process can be seen in Table 4.3.3. The cohort year can be seen across the top in grey, with the age of brown trout shown in the left hand column. The 'growth periods' are colour-coded. For example, the one-year old trout of the 2002 cohort grew in the 2002-2003 period; whilst the two year old trout of the 2000 cohort grew in the 2001 -2002 period. By using this method and percentages of standard growth rates it is should be possible to disentangle the good and bad years for the different cohorts.

R1	1996	1997	1998	1999	2000	2001	2002
1	112.8 (1)		87.8 (1)	97.5 (3)	94.9 (8)	83.8 (9)	81 (26)
2	92 (1)		100.4 (1)	128.9 (3)	102.7 (5)	84.4 (9)	
3	86 (1)		117 (1)	71.7 (3)	38.2 (2)		
4	111.3 (1)		24.8 (1)				

Table 4.3.3. Example table to show the percentage of average growth rate for each cohort found at R1. Yellow = 2002-2003 growth period; green 2001– 2002 growth period. (Number of fish used in calculation shown in brackets).

HABSCORE data collection and outputs

HABSCORE is a system of salmonid habitat measurement and evaluation based on empirical estimates of fish density based upon combinations of site and catchment features (Milner *et al.*, 1998). Using the information from three HABSCORE questionnaires, the software produces a series of outputs, which includes estimates of the expected populations (the Habitat Quality Score, HQS) and the degree of habitat utilisation (Habitat Utilisation Index, HUI), for the three brown trout species/age combinations (Wyatt *et al.*, 1995). The three age/size class groups used for brown trout are: (i) 0+ trout; (ii) >0+(<20cm) trout; and (iii) >20cm trout.

The methodology of habitat collection and completion of the relevant form (HABform), along with completion of river catchment information (MAPform) and fisheries information (FISHform), are documented by Barnard and Wyatt (1995).

To collect information for HABSCORE analysis a questionnaire on the habitat found at each site was completed alongside with the 2003 and 2005 fisheries surveys. The aim of this is to create separate HABSCORE databases for both the pre and post change data. This should allow both characterisation of the pre-change fish populations and also the comparison of the pre and post change trout data.

The HQS value is a measure of the habitat quality expressed as the expected long-term average density of fish (in numbers/100m^2). The HQS is derived from habitat and catchment features and assumes that neither water quality nor recruitment are limiting the populations. The HQS is used in this chapter as an indicator of the potential of the site, against which, the observed size of populations may be compared. There are upper and lower confidence limits for the HQS in numbers/100m^2. The confidence limits given should enclose the average observed density for a site on 90% of occasions. The probability of getting an observed density lower than the lower confidence limit by chance alone is therefore 5%.

The HUI is a measure of the extent to which the habitat is utilised by the salmonids. It is based on the difference between the 'observed' density and that which would be expected under 'pristine' conditions (i.e. the HQS). When the 'observed' density and the HQS are exactly the same, the HUI takes the value of one; HUI values less than one will occur when the observed densities are less than expected. There are upper and lower 90% confidence limits for the HUI, expressed as a proportion. An upper HUI confidence interval <1 indicates that the observed population was significantly less than that expected under pristine conditions. Conversely, a lower HUI confidence interval > 1 indicates that the observed population was significantly higher than that expected under pristine conditions. In this chapter a HUI value of >1 is taken to mean that the habitat is limiting i.e. there are more fish wanting to live there than there is habitat available. If the HUI is less than one, this is taken to be indicative of a fish population which is limited by factors other than physical habitat availability.

4.4. Results

The presentation of the results in this chapter is split into two sections. Firstly, the Rivelin and Loxley are compared to assess the structure and quality of fish populations before the compensation flow change. The results from the Rivelin and Loxley are then compared with the results from the control catchment. The final sub-section of the first results section assesses the variability in the fisheries populations. The second results section focuses on the impacts of altering the compensation flows, by first comparing the Rivelin and Loxley, and then introducing the results from the Hipper.

4.4.1. Paired-catchment analysis: differences in the structure and quality of fish populations before compensation flow change

4.4.1.1. Comparison of the Rivelin and Loxley

The first focus of this sub-section is a comparison of the Rivelin and Loxley. By comparing the fisheries populations present in the Rivelin and Loxley, this section aims to establish a baseline for change and establish any potential impacts of the pre-change compensation flows.

Brown trout

Figure 4.4.1 shows that the highest predicted densities (Habitat Quality Score –HQS) on the Rivelin were for 0+ trout. The 0+ HQS bar chart displays a bell-shaped tendency with the highest densities found in the middle reaches of the Rivelin. It can also be seen that the predicted densities of >0+ (<20cm) are lower than for the 0+ trout, with the predicted densities of the >20cm trout being lowest of all. There is also a much higher habitat quality for the 0+ trout than for the other age/size classes on the Loxley (Table 4.4.1). There are no statistically significant differences between the HQS scored for the Rivelin and Loxley for any of the age/size classes. What can also be seen from Figure 4.4.1 is that there are wide 90% confidence intervals on the HQS estimates produced by HABSCORE for both rivers, which make distinguishing between sites impossible statistically.

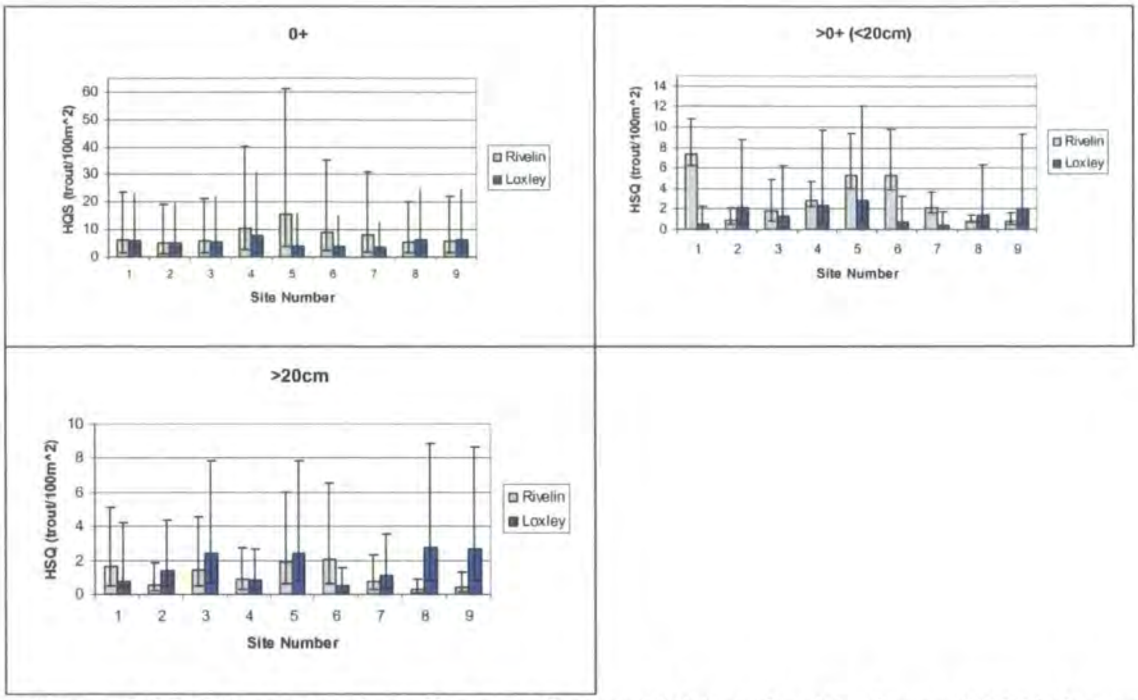


Figure 4.4.1. Figure to show the pre-change Habitat Quality Score (HQS) for each age/size class of trout on the Rivelin and Loxley. HQS upper and lower confidence intervals have been shown.

Rivelin	Loxley
HQS O+ > HQS >0+(<20cm)	HQS O+ > HQS >0+(<20cm)
HQS 0+ > HQS >20cm	HQS 0+ > HQS >20cm
HQS >0+(<20cm) > HQS >20cm	HQS >0+(<20cm) = HQS >20cm

Table 4.4.1. Table to show the ANOVA output and direction when comparing the age/size classes using the HQS scores. 95% statistical significance.

Figure 4.4.2 shows that the highest observed 0+ densities were found at R4 in 2002, but no clear temporal pattern is shown. The 0+ densities do appear to follow a bell-shaped distribution, with the highest densities of 0+ trout found at R2, R3 and R4. A similar distribution is observed in the >0+(<20cm) trout densities. Figure 4.4.2 also shows that for all sites, the density of >0+(<20cm) trout were higher in 2002 than in 2003. The >20cm densities display no spatial variation and it appears that >20cm trout density may be slightly higher in 2002 compared with 2003. There are no statistically significant differences between the 2002 and 2003 densities for any of the age/size classes. But Table 4.4.2 shows that the most abundant age/size class statistically is the >0+(<20cm) trout.

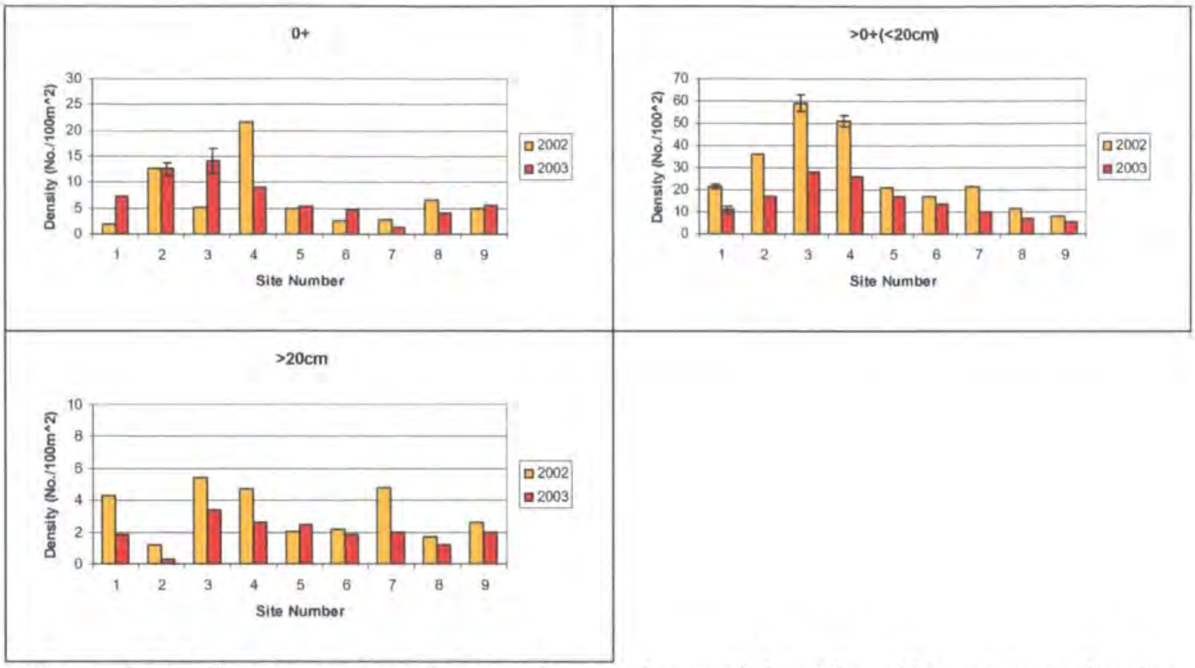


Figure 4.4.2. Graphs to show the pre-change observed densities of brown trout for the three main age/size classes for the Rivelin. Site one is the uppermost and site 9 the lowest. Error bars showing the Carle-Strub variance. Scales vary to aid visualisation.

	2002	2003
0+ vs >0+(<20cm)	Diff (0+ < >0+(<20cm)	Diff (0+ < >0+(<20cm)
0+ vs >20cm	No diff	Diff (0+ > >20cm)
>0+(<20cm) vs >20cm	Diff >0+(<20cm) > >20cm	Diff >0+(<20cm) > >20cm

Table 4.4.2 Table to show the results of an ANOVA analysis comparing the densities of brown trout of the different age/size classes for the Rivelin. Differences are statistically significant to the 95% significance level.

Figure 4.4.3 and Table 4.4.3 show that, generally, the highest density age/size class in the Loxley is the >0+(<20cm) class. Apart from high densities of 0+ trout at L1 and L2 in 2002, densities of 0+ trout are generally lower than those of >0+(<20cm) trout. There does appear to be a downstream decrease in the density of 0+ trout, whilst this is not apparent for the two older age/size classes. There is also no clear pattern of temporal variation in any of the age/size classes.

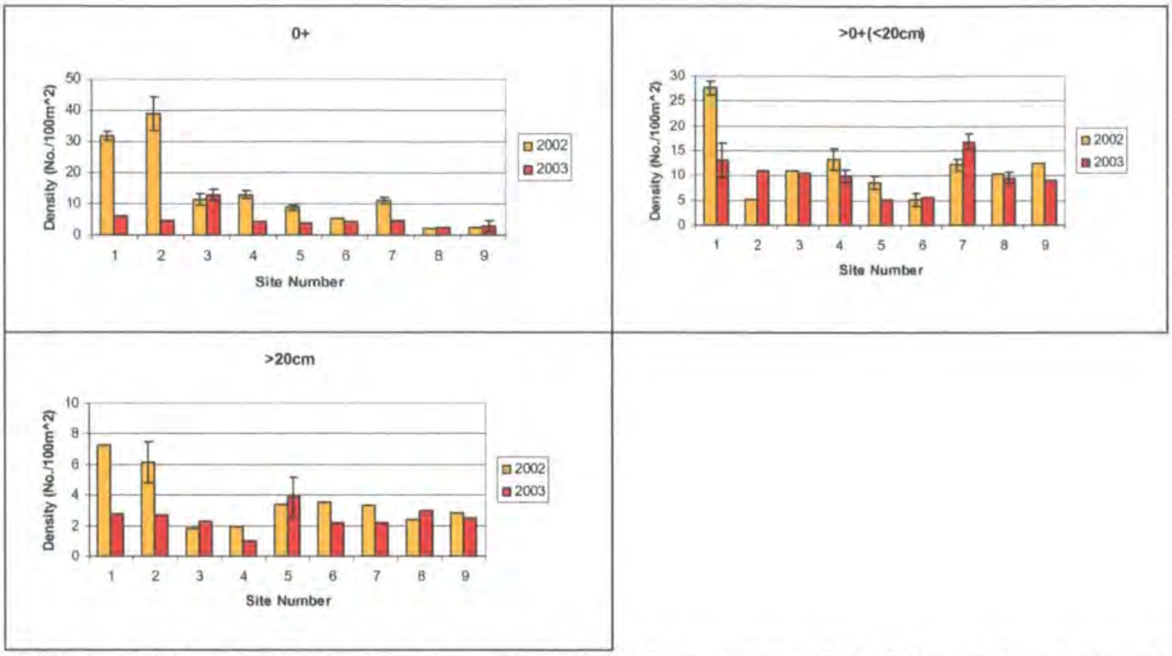


Figure 4.4.3. Figure to show the pre-change observed density of brown trout at each site on the Loxley for three age/size classes. Site one is the uppermost and site 9 the lowest. Error bars showing the Carle-Strub variance. Scales vary.

	2002	2003
0+ vs >0+(<20cm)	No Diff	Diff (0+ < >0+(<20cm)
0+ vs >20cm	Diff (0+ > >20cm)	Diff (0+ > >20cm)
>0+(<20cm) vs >20cm	Diff >0+(<20cm) > >20cm	Diff >0+(<20cm) > >20cm

Table 4.4.3. Table to show the results of an ANOVA analysis comparing the densities of brown trout of the different age/size classes for the Loxley. Differences are statistically significant to the 95% significance level.

In 2002 there were large 0+ trout densities for the first two sites on the Loxley, whilst the 0+ densities in the Rivelin peaked slightly further downstream. There was much less difference in 0+ recruitment between the Rivelin and Loxley in 2003, compared with the highly variable densities found in 2002. Figures 4.4.2 and 4.4.3 also show that the Rivelin appears to have higher densities of >0+ (<20cm) than the Loxley. Again, the largest amount of difference between the Rivelin and Loxley densities were found in the 2002 samples. An ANOVA test showed that the Rivelin and Loxley had statistically significant different >0+ (<20cm) trout populations in 2002. The densities found at sites R2, R3 and R4, are very important to the Rivelin, especially in 2002. Figures 4.4.2 and 4.4.3 show that the Rivelin and Loxley have similarly low densities of >20cm trout, with no clear pattern as to which river has the highest densities. The lack of visual difference and the existence of only one statistically significant difference between the trout densities on the Rivelin and Loxley indicate that there is no clear suggestion that the differences in compensation flows are having an impact on the densities of trout found in the rivers.

Despite (or perhaps because of) the large intra-river variability in terms of density there are statistically no difference between the 2002 densities and 2003 densities in either the Rivelin or the Loxley (Table 4.4.4). The same lack of difference indicates that the rivers are not different in terms of their inter-year variation.

	0+	**>0+ (<20cm)**	**>20cm**
R 2002 vs R 2003	Not diff	Not diff	Not diff
L 2002 vs L 2003	Not diff	Not diff	Not diff

Table 4.4.4. ANOVA results from the density data comparing the sampling seasons within the rivers for each age/size class of brown trout. Differences are statistically significant to the 95% significance level.

Table 4.4.5 shows the Habitat Utilisation Index (HUI) values for each site on the Rivelin. It shows that the HUI index of 0+ trout is less than one (indicating a lower than expected density) for all Rivelin sites below site R5. Table 4.4.5 also shows the densities of >0+ (<20cm) trout are always greater than expected. For 6 out of 9 of the sites the densities are significantly higher than those expected under pristine conditions. Also, the HUI for >20cm trout was greater than one for seven out of nine sites, with three sites having densities of >20cm trout significantly greater than those expected under pristine conditions. This indicates that the Rivelin has survival of trout from 0+, but perhaps the Rivelin doesn't have the deeper water habitat for the fish, hence leading to over use of habitat for the >0+ (<20cm) and >20cm trout.

Table 4.4.5 shows that there is a lot of variation in the Habitat Utilisation Index (HUI) for 0+ trout on the Loxley. For 6 out of the 9 sites the HUI was greater than one, but the densities were not statistically better than that expected under pristine conditions, so perhaps there are there other factors impacting on the population. As on the Rivelin, the HUI for the >0+ (<20cm) trout were greater than one for each site. There are more statistically significant HUI values on the Rivelin compared to the Loxley (six compared to five), which may indicate that habitat may be more limiting on the Rivelin than on the Loxley for >0+(<20cm) trout. All but one site on the Loxley had a HUI for >20cm trout of greater than one, and two sites had a >20cm trout population which was significantly better than that expected under pristine conditions.

It thus appears that habitat for 0+ trout is not limiting, as evidenced by the low HUI values. If the HUI values were high, it indicates a large population using a given habitat, but a low HUI appears to indicate that other factors are influencing the 0+ populations of both rivers. This may have implications when interpreting the modelling results in Chapter 6 in that, an increase in theoretical 0+ habitat may not actually improve the densities of 0+ present in the Rivelin. However, the high HUI values could indicate that habitat is limiting the densities of >0+(<20cm) trout on the Rivelin.

	0+	**>0+ (<20cm)**	**>20cm**			**0+**	**>0+ (<20cm)**	**>20cm**
R1	1.14	2.11	1.75		L1	2.33	**34.57**	**5.84**
R2	2.54	**26.10**	0.91		L2	2.60	4.14	2.96
R3	1.22	**22.64**	2.65		L3	2.21	**7.81**	0.87
R4	1.31	**13.26**	**3.79**		L4	0.93	5.31	1.59
R5	0.3	3.66	1.18		L5	1.48	2.31	1.5
R6	0.38	2.85	0.97		L6	1.23	**7.43**	**5.38**
R7	**0.09**	**7.41**	**4.15**		L7	2.15	**37.75**	2.42
R8	0.84	**9.93**	4.70		L8	0.3	**8.25**	1.59
R9	0.83	**8.87**	**5.51**		L9	0.99	4.9	1.83

Table 4.4.5. Table to show the Habitat Utilisation Index (HUI) values for the Rivelin and Loxley. Statistically significant differences from what would be expected under pristine conditions are shown in bold.

	Rivelin	**Loxley**
0+ vs >0+(<20cm)	Diff 0+ < >0+(<20cm)	Diff 0+ < >0+(<20cm)
0+ vs >20cm	Diff 0+ < >20cm	No diff
>0+(<20cm) vs >20cm	Diff >0+(<20cm) > >20cm	Diff >0+(<20cm) > >20cm

Table 4.4.6 Table to show the results of an ANOVA analysis comparing the HUI values of the different age/size classes for the Rivelin and Loxley. Differences are statistically significant to the 95% significance level.

There are no statistically significant differences between the Rivelin and Loxley (ANOVA, 95% significance) using either the HQS or HUI for any of the age/size classes. This further enhances the hypothesis that the trout densities on the Rivelin and Loxley are similar.

For the purpose of analysing the growth rates of brown trout in the Rivelin, as discussed in section 4.3, the trout captured were aged and their lengths back-calculated using the Dahl-lea method. The average length at age for each cohort was then calculated. From this data, the percentage of average growth rate was calculated for each cohort. This data allows us to explore the structure and quality of the fish growth rates on the Rivelin.

Table 4.4.7 shows that the growth rates of 0+ trout in the 2001 and 2002 cohorts are generally slower than those found for any other cohort. This, combined with the fact that the trout growing in the 2002-2003 growth period grew more slowly than for any of the other growth periods indicates that this period may not have been a very good one for brown trout growth in the Rivelin. Table 4.4.7 also shows that the growth rates of brown trout in the Rivelin tend to decrease as the age of the trout increase. It was seen earlier that the HUI values for >0+(<20cm) and >20cm trout on the Rivelin were generally greater than one, with many trout populations significantly better than those expected under pristine conditions. It was hypothesised at that point that this could be due to the river being habitat limited for those age groups. If the density of these older trout is too great for the habitat available, perhaps a by-product of this is the slower growth rates for larger trout, due to increased competition for food.

Unlike on the Rivelin, it appears that on the Loxley there is no decrease in growth rate with increasing age of trout (Table 4.4.8). When the HUI values for >0+(<20cm) and >20cm trout on the Loxley were examined above it was seen that the values were similar to those on the Rivelin. It was thus hypothesised that the physical habitat may be limiting for older trout in these rivers. The fact that the Loxley has faster growth rates for older brown trout could be due to the greater density of invertebrates found compared with the Rivelin (as seen in Chapter 3). The 2001 and 2002 cohorts generally have a slower growth rate than the earlier cohorts at one year old. This is also seen in the Rivelin and so could be due to environmental factors rather than biotic factors. It is also apparent that there is no clear difference between the 2001-2002 and 2002-2003 growth periods.

R1	**1996**	**1997**	**1998**	**1999**	**2000**	**2001**	**2002**
1	112.8 (1)		87.8 (1)	97.5 (3)	94.9 (8)	83.8 (9)	81 (26)
2	92 (1)		100.4 (1)	128.9 (3)	102.7 (5)	84.4 (9)	
3	86 (1)		117 (1)	71.7 (3)	38.2 (2)		
4	111.3 (1)		24.6 (1)				
R2	**1996**	**1997**	**1998**	**1999**	**2000**	**2001**	**2002**
1					1	69.6 (5)	94.1 (28)
2					2	95.9 (5)	
R3	**1996**	**1997**	**1998**	**1999**	**2000**	**2001**	**2002**
1			106.9 (1)	100.1 (4)	92.7 (5)	88.4 (25)	90.9 (37)
2			166.7 (1)	112.9 (4)	96.1 (5)	83.8 (15)	
3			77.7 (1)	69.8 (4)	64.3 (2)		
4			74.8 (1)				
R4	**1996**	**1997**	**1998**	**1999**	**2000**	**2001**	**2002**
1				117.1 (1)	100.3 (7)	86.3 (21)	85.4 (22)
2				109.5 (1)	105.1 (7)	81.5 (16)	
3				71.0 (1)	101.9 (2)		
R5	**1996**	**1997**	**1998**	**1999**	**2000**	**2001**	**2002**
1					92.9 (2)	95.9 (16)	91.2 (21)
2					96.3 (2)	87.2 (8)	
R6	**1996**	**1997**	**1998**	**1999**	**2000**	**2001**	**2002**
1				105.4 (5)	100.6 (5)	75.8 (16)	88.1 (21)
2				83.7 (5)	81.7 (5)	78.2 (13)	
3				76.4 (5)	61.3 (2)		
4				112.2 (1)			
R7	**1996**	**1997**	**1998**	**1999**	**2000**	**2001**	**2002**
1				115.5 (3)	96.7 (11)	93.6 (18)	87.9 (23)
2				100.3 (3)	83.1 (11)	78.8 (10)	
3				67.6 (3)	60.7 (5)		
R8	**1996**	**1997**	**1998**	**1999**	**2000**	**2001**	**2002**
1				109.6 (2)	110.3 (10)	92.2 (16)	92.6 (18)
2				112.4 (2)	98.2 (10)	77.1 (11)	
3				88.0(2)	87.0 (3)		
R9	**1996**	**1997**	**1998**	**1999**	**2000**	**2001**	**2002**
1			83.1 (1)	89.6 (4)	91.8 (9)	80.1 (12)	74.6 (17)
2			112.3 (1)	89.9 (4)	101.3 (9)	87.7(10)	
3			70.5 (1)	81.7 (4)	55.3 (4)		
4			93.0 (1)				

Table 4.4.7. Table to show the percentage of average growth rate for each cohort found at each site on the Rivelin. Yellow = 2002-2003 growth period; green 2001– 2002 growth period. (Number of fish used in calculation shown in brackets).

L1	**1997**	**1998**	**1999**	**2000**	**2001**	**2002**
1		107.7 (2)	99.0 (6)	107.6 (10)	96.2 (18)	93.7 (21)
2		125.6 (2)	99.8 (6)	100.7 (10)	108.0 (9)	
3		99.5 (2)	93.9 (6)	121.8 (2)		
4		102.0 (2)	142.1 (1)			
L2	**1997**	**1998**	**1999**	**2000**	**2001**	**2002**
1	102.6 (1)	109.3 (2)	112.8 (7)	97.1 (19)	89.6 (15)	87.1 (30)
2	129.0 (1)	89.8 (2)	109.6 (7)	130.1 (19)	121.0 (12)	
3	136.8 (1)	128.0 (2)	97.6 (7)	113.4 (11)		
4	78.9 (1)	100.9 (2)	60.6 (3)			
L3	**1997**	**1998**	**1999**	**2000**	**2001**	**2002**
1			95.4 (4)	104.2 (4)	97.2 (11)	86.1 (26)
2			96.8 (4)	120.0 (4)	93.5 (5)	
3			75.4 (4)	160.4 (2)		
L4	**1997**	**1998**	**1999**	**2000**	**2001**	**2002**
1		140.9 (1)	100.5 (4)	89.8 (11)	89.1 (10)	86.0 (26)
2		130.9 (1)	95.9 (4)	97.4 (4)	92.3 (8)	
3		35.2 (1)	91.0 (4)	91.0 (1)		
4		81.3 (1)				
L5	**1997**	**1998**	**1999**	**2000**	**2001**	**2002**
1		110.6 (3)	99.3 (9)	95.8 (5)	92.3 (18)	92.3 (14)
2		98.4 (3)	93.0 (9)	95.4 (5)	99.3 (15)	
3		78.5 (3)	91.7 (9)	83.8 (3)		
4		67.9 (3)	52.8 (3)			
L6	**1997**	**1998**	**1999**	**2000**	**2001**	**2002**
1			83.9 (1)	91.7 (2)	93.8 (10)	91.3 (25)
2			113.3 (1)	110.9 (2)	101.8 (10)	
3			101.7 (1)	117.8 (2)		
4			117.3 (1)			
L7	**1997**	**1998**	**1999**	**2000**	**2001**	**2002**
1		102.7 (3)	95.6 (7)	88.9 (4)	87.6 (27)	88.2 (31)
2		85.0 (3)	108.1 (7)	88.1 (4)	98.1 (21)	
3		99.3 (3)	76.6 (7)	99.3 (2)		
4		65.4 (3)	148.1 (1)			
L8	**1997**	**1998**	**1999**	**2000**	**2001**	**2002**
1		69.4 (1)	109.8 (7)	101.3 (11)	84.0 (18)	86.5 (24)
2		109.0 (1)	95.6 (7)	96.4 (11)	97.5 (14)	
3		81.0 (1)	74.4 (7)	73.1(4)		
4		26.7 (1)				
L9	**1997**	**1998**	**1999**	**2000**	**2001**	**2002**
1			96.7 (3)	96.0 (6)	83.0 (28)	82.5 (13)
2			87.7 (3)	101.4 (6)	88.2 (23)	
3			93.8 (3)	81.2 (3)		

Table 4.4.8. Table to show the percentage of average growth rate for each cohort found at each site on the Loxley site. Yellow = 2002-2003 growth period; green 2001–2002 growth period. (Number of fish used in the calculation in brackets).

When the growth rates of the different growth periods were analysed using an ANOVA test it was seen that there was no statistically significant difference between the Rivelin and Loxley in terms of one year old trout in either of the 2001-2002 or 2002-2003 growth periods. Only the latest two growth periods were included in the ANOVA analysis in order to maintain the number of sites in the analysis. Table 4.4.8 shows that there is a statistically significant difference between the Rivelin and Loxley in terms of trout growing to two years old in the 2002-2003 period, but no difference in the 2001-2002 period. It can also be seen that there is a statistically significant difference between the Rivelin and Loxley in terms of the growth to three years old for both growth periods. This reinforces the fact the older trout appear to grow faster in the Loxley compared with the Rivelin.

One year old	**Rivelin vs Loxley**
2001/2002	No diff
2002/ 2003	No diff
Two years old	
2001/2002	No diff
2002/ 2003	Diff (L>R)
Three years old	
2001/2002	Diff (L>R)
2002/ 2003	Diff (L>R)

Table 4.4.8. Table to show the results of the ANOVA output when comparing the percentage of standard length at age for the two pre-change growth periods. Differences are statistically significant to the 95% significance level.

It is also interesting to compare whether the growth rates of the different growth periods are different for either the Rivelin and/or Loxley (Table 4.4.9). Table 4.4.9 shows that the only age/size class to grow at different rates in the two years was the two year old trout in the Rivelin. This indicates that the 2002-2003 growth rate for two year old trout in the Rivelin was unusual, and this could have caused the difference with the Loxley for that growth period.

One year old	Rivelin	Loxley
2001/02 vs 2002/03	No diff	No diff
Two years old		
2001/02 vs 2002/03	Diff (01/02>02/03)	No diff
Three years old		
2001/02 vs 2002/03	No diff	No diff

Table 4.4.9. Table to show the ANOVA output from comparing the growth rates between growth periods in the Rivelin and Loxley. Differences are statistically significant to the 95% significance level.

Because the percentage of standard growth rate was used in the ANOVA analysis, it is also possible to compare the growth rates of the different age/size classes within each growth period (Table 4.4.10). Table 4.4.10 shows that, for the Rivelin, the growth rates for each age were different for the 2001-2002 period, whilst the growth rates for the Loxley were no different for the one and three year old trout. There is less difference between the growth rates of the different ages in the 2002-2003 period.

	Rivelin		Loxley	
	2001-2002	2002-2003	2001-2002	2002-2003
1 vs 2	Diff (2>1)	No diff	Diff (2>1)	Diff (2>1)
1 vs 3	Diff (1>3)	Diff (1> 3)	No diff	No diff
2 vs 3	Diff (2>3)	No diff	Diff (2>3)	No diff

Table 4.4.10. Table to show the output from the ANOVA analysis comparing the growth rates of the different ages. Differences are statistically significant to the 95% significance level.

Bullheads

No Bullhead length data were collected in the 2002 survey at any sites, therefore the pre-change Bullhead data consist of one years sample data. No Bullhead populations were found in the uppermost three sites on the Rivelin, with stone loach being dominant. At the R4 site, the population was dominated by adult Bullheads, but contained evidence of 0+ recruitment (Figure 4.4.4). A much larger Bullhead population was found at R5, dominated by adult Bullheads, but with a good 0+ population. A population of similar size was found at R6, with the same dominance of adult bullheads with good recruitment of 0+ Bullheads. The R7 and R8 sites contained a slightly smaller overall population than the previous two sites and had a reduced abundance of 0+ Bullheads. The second smallest Bullhead population was found at the R9, with this site having the lowest 0+ recruitment of any site on the Rivelin. The population of Bullheads appear to decrease in abundance with increasing distance downstream, and the 0+ recruitment also appears to decrease in the same way.

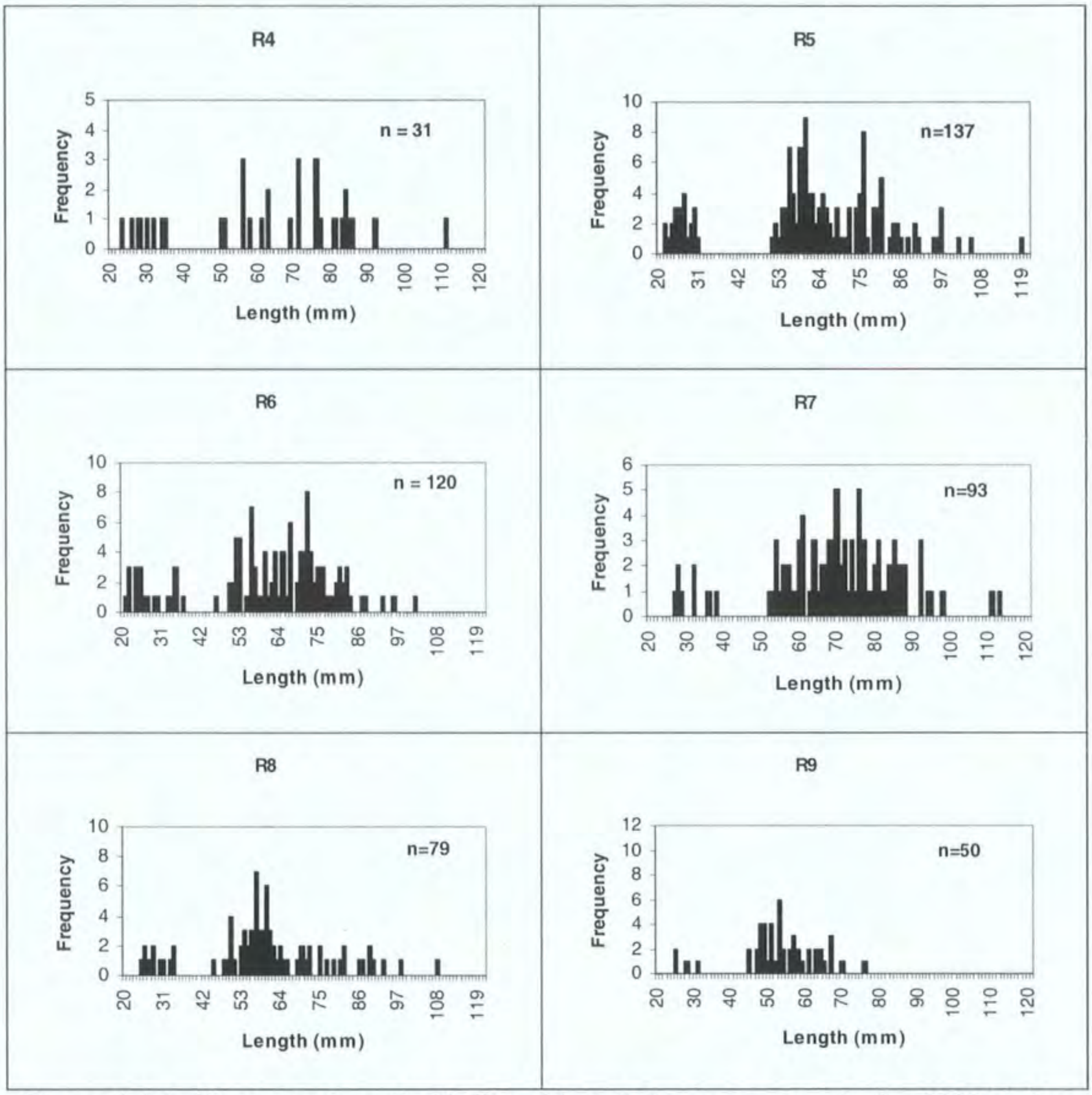
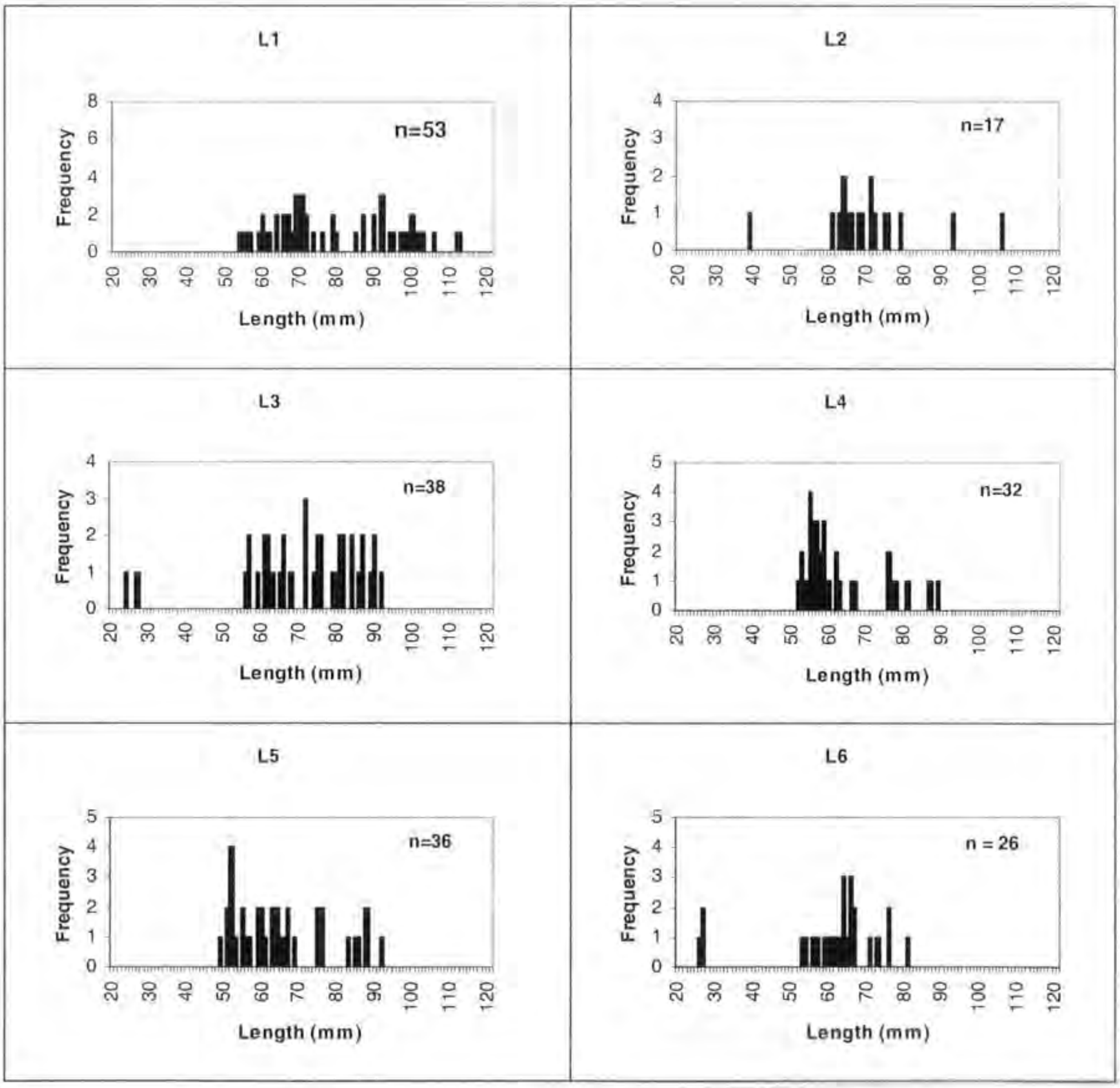


Figure 4.4.4. Figure to show the Bullhead length frequency graphs for 2003 on the Rivelin.

Unfortunately as with the Rivelin, Bullheads were only counted and measured from 2003 onwards, and so this section – contextualising the Loxley can only use that data. At L1, there was a mature adult population present, but no evidence of juvenile recruitment (Figure 4.4.5). A smaller population downstream at L2 contained one juvenile and an adult population. Two juveniles were found at L3, along with a dominant adult population. A similar sized population was found at L4 and L5 except there was no evidence of juvenile recruitment. At L6 and L7 there was evidence of juvenile recruitment but the adult population was still dominant. A much smaller Bullhead population was found at L8, with one juvenile found amongst eight fish. A larger population was found at L9 with both evidence of juvenile recruitment and a dominant adult population. There appears to be a strong adult population of Bullheads within the Loxley. However, there are not as many juveniles were captured as there was on the Rivelin. Hopefully, the sampling sites are not inclusive of the Bullhead rearing grounds on the Loxley and it is not simply that 2003 was an appalling year for Bullhead recruitment on the Loxley.



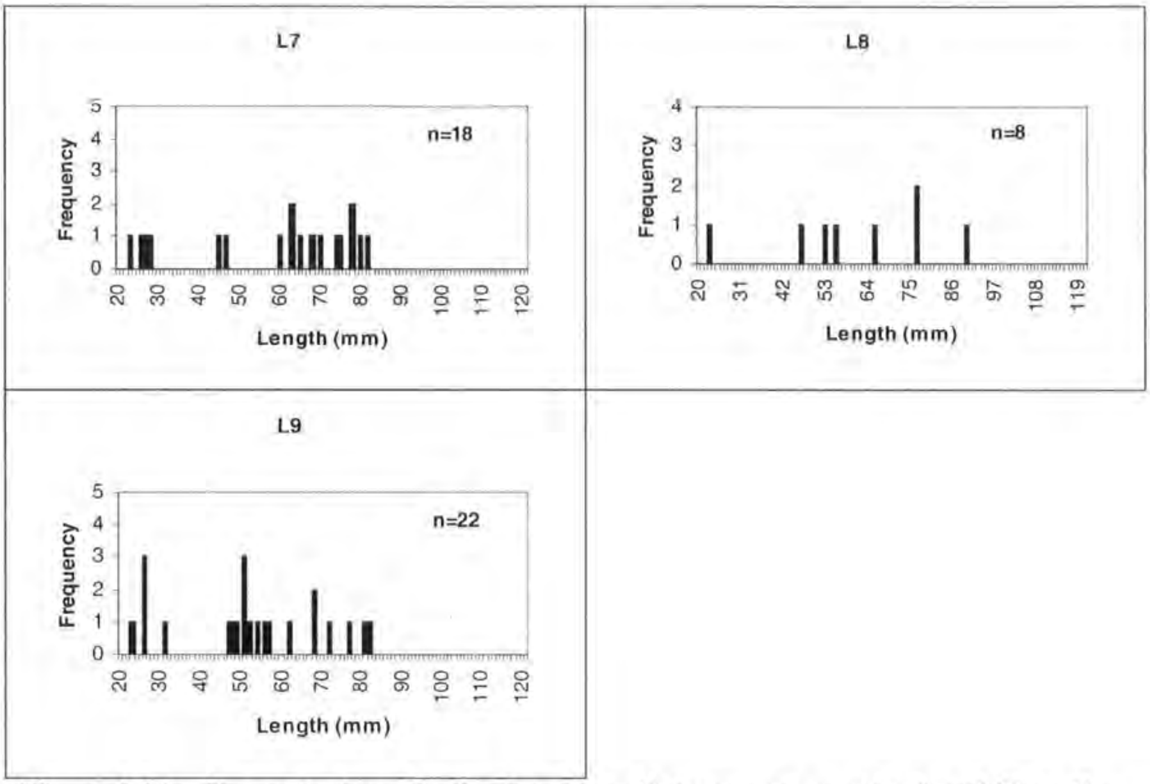


Figure 4.4.5. Figure to show the Bullhead length frequency graphs for 2003 on the Loxley.

Other fish species

A number of Brook Lamprey (*Lampetra planeri*) were found in the upper reaches of the Rivelin (Table 4.4.11). Although no Lamprey were found in 2002, this is not to say they were not present in the river, as this species are difficult to electrofish (with the anode having to be held static for upwards of a minute to tease them out). The abundances listed in Table 4.4.11 cannot be taken as true estimates of population as the habitats were not fished to exhaustion, due to the difficulty in fishing this species. The main aim was to ascertain the presence of a population.

	2002	**2003**
R1	0	1
R2	0	5
R3	0	0
R4	0	0

Table 4.4.11. Table to show the abundance of brook lamprey (*Lampetra planeri*) in the Rivelin in 2002 and 2003.

Table 4.4.12 shows that in the higher reaches of the Rivelin, where Bullheads were absent there were healthy populations of Stone loach (*Barbatulus barbatulus*). As the populations of Bullheads increased were larger in the downstream sites, the populations of Stone loach appeared to decrease.

	2002	2003
R1	No data	59
R2	No data	18
R3	No data	36
R4	No data	8
R5	No data	15
R6	No data	12
R7	No data	15
R8	No data	8
R9	No data	1

Table 4.4.12. Table to show the number of Stone Loach *(Barbatulus barbatulus)* captured in the Rivelin sites in 2002 and 2003.

Stone loach (*Barbatulus barbatulus*) were only caught at L7 in 2003, although, like Bullheads, they were not actively caught and measured in 2002. Avery small number of Perch, Roach, Grayling and Pike were caught during sampling, and it is thought that these species are escapees from the mill ponds which feed off and back into the main river, and many of which are used for recreational fishing purposes.

4.4.1.2. Comparison with the control catchment

Unlike the Rivelin and Loxley, the River Hipper has remained unregulated, and as such can be used as a control to investigate the effects of regulation on upland Millstone Grit catchments such as the Rivelin and Loxley.

Brown trout

Figure 4.4.6 shows that the predicted Habitat Quality Score (HQS) is higher for 0+ trout than any other age/size class and the predicted densities of >0+ (<20cm) were higher than those for the >20cm trout. There appears to be little variation between the sites in terms of the HQS for each of the age/size classes, with the size of the 90% confidence intervals produced by HABSCORE make detecting differences in terms of either inter-site or inter-age/size class difficult.

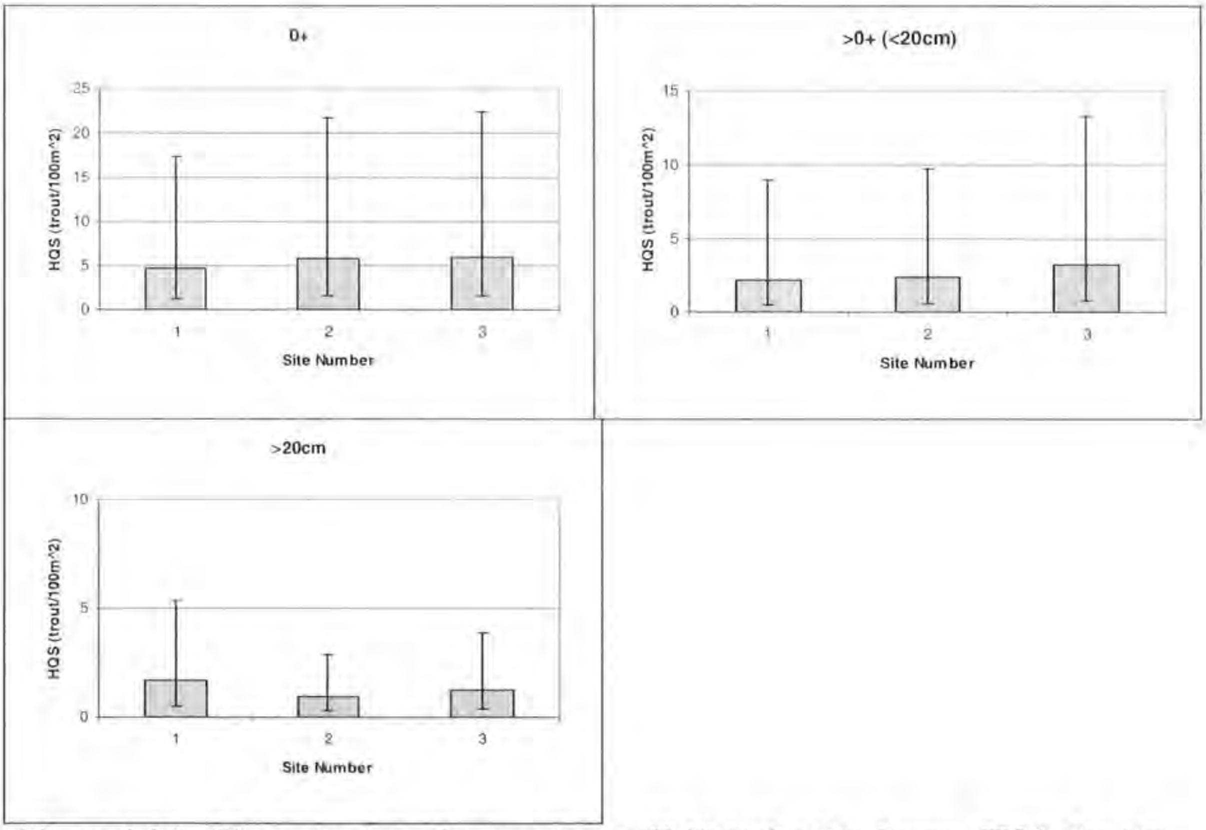


Figure 4.4.6. Figure to show the pre-change Habitat Quality Scores (HQS) for the three Hipper sites. 90% error bars included. Scales not the same.

Figure 4.4.7 shows that there was better 0+ recruitment in 2002 for two of the Hipper sites, but with 2003 having marginally better recruitment than 2002 at H3. H3 had lower 0+ recruitment than the two other sites especially in 2002. The densities of >0+ (<20cm) trout were again higher in 2002 for the first two sites, but lower for the remaining site, but they generally appeared stable in both space and time. The densities of >20cm trout were higher in 2002 for each of the sites, with the difference being largest at H1 and H2.

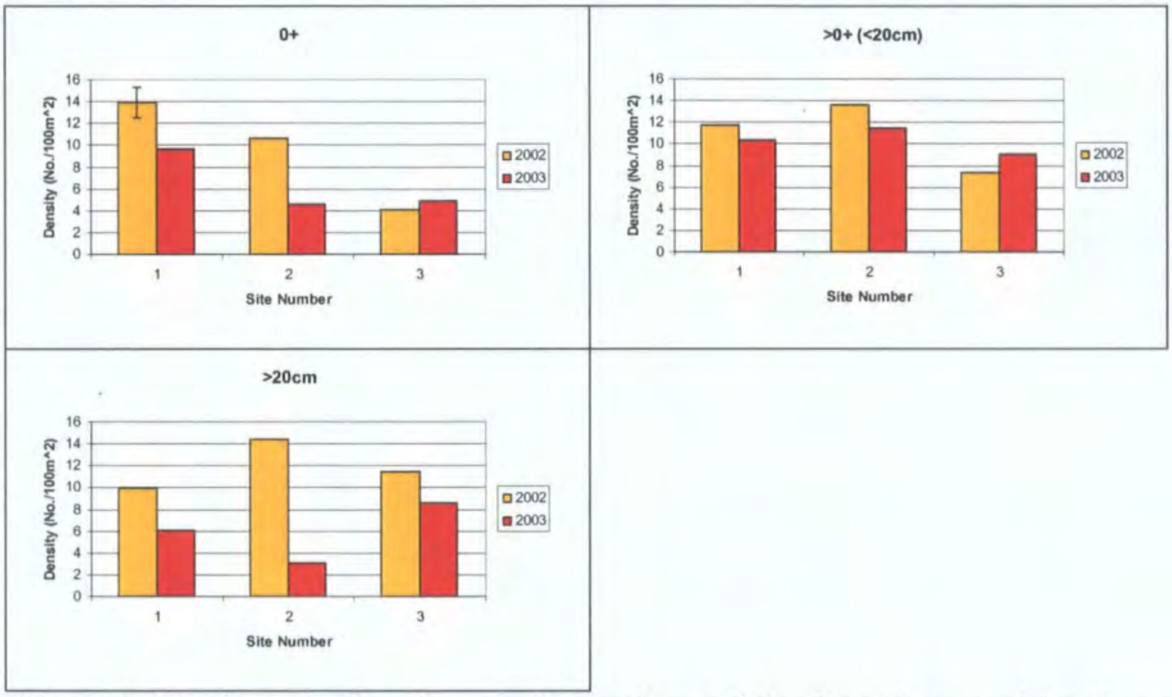
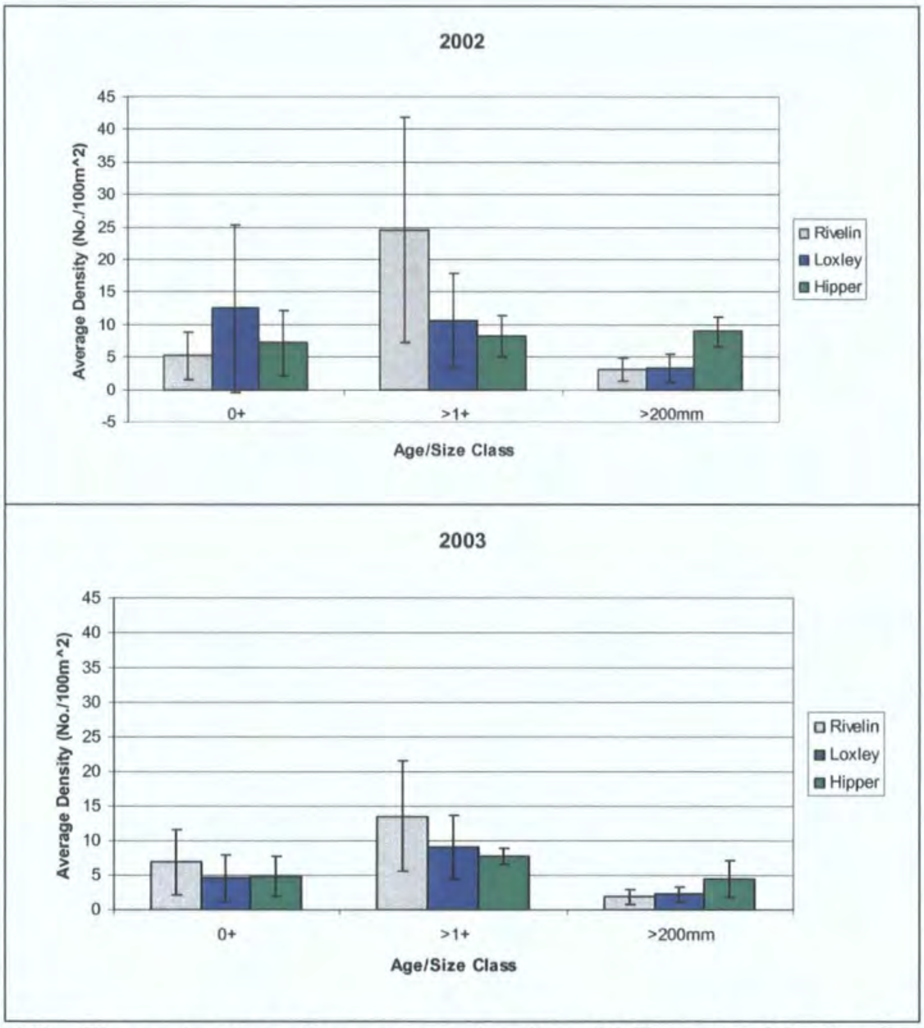


Figure 4.4.7. Figure to show the pre-change observed density of three age/size classes of brown trout at each site on the Hipper.

Table 4.4.13 shows that for 0+ trout, sites H1 and H2 have HUI values greater than one, but H3 has HUI values of less than one, perhaps indicating a slight under-use of habitat at site H3. Table 4.4.13 also shows that the HUI values for >0+ (<20cm) are all greater than one. This is interesting because similar results were found for the Rivelin and Loxley, and HABSCORE may be underestimating the habitat suitability of these rivers for >0+ (<20cm) trout. The densities of >20cm trout found at each of the sites is significantly higher than that expected under pristine conditions.

Figure 4.4.8 shows that the Hipper had higher densities of >20cm trout than the Rivelin or Loxley in 2002 and 2003 (but the standard deviation is overlapping in 2003). Figure 4.4.9 shows that there is no difference between the Rivelin, Loxley and Hipper in terms of the HQS for >20cm trout.



2002

2003

Figure 4.4.8. Figure to show the average density of BT on the Rivelin, Loxley, and Hipper in 2002 and 2003. Standard deviation as error bars.

As well as the density prediction of the Habitat Quality Score (HQS), the Habitat Utilisation Index (HUI) value produced by the HABSCORE program, gives an indication of how well the habitat is being used. Figure 4.4.9 shows that the higher density of >20cm trout in the Hipper has translated to a higher HUI for that age/size class for the Hipper than for the Rivelin and Loxley. Again, the intra-river variation leads to large standard deviations and so it is difficult to differentiate between the rivers. Although the Loxley appears to have lower densities of >0+ (<20cm) trout the HUI appears to be slightly higher than the Rivelin, indicating that more trout are surviving in less than ideal habitat. Figure 4.4.9 suggests that, if there is a difference in the habitat structure and habitat use between these regulated and a de-regulated systems, the HABSCORE system is not able to detect it.

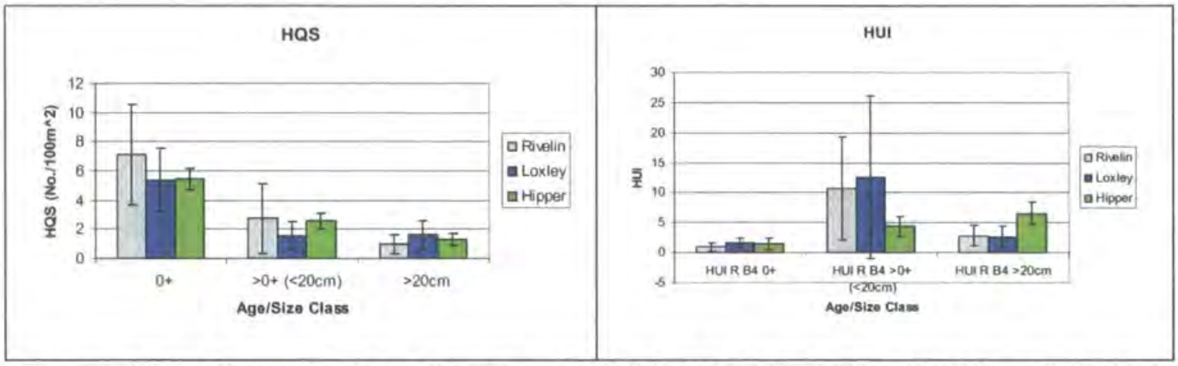


Figure 4.4.9. Figure to show the average Habitat Quality Score (HQS) and Habitat Utilisation Index (HUI) in each of the rivers for the HABSCORE data. Standard deviation as error bars

Bullhead

Bullheads were not counted and measured in 2002, so the discussion below only refers to 2003. The population at H1 in 2003 was dominated by the adult population but there were four 0+ Bullheads also captured (Figure 4.4.10). A much greater 0+ abundance was seen at H2, but the population was still dominated by adult Bullheads. The recruitment at H3 was much reduced, with the adult population again dominant.

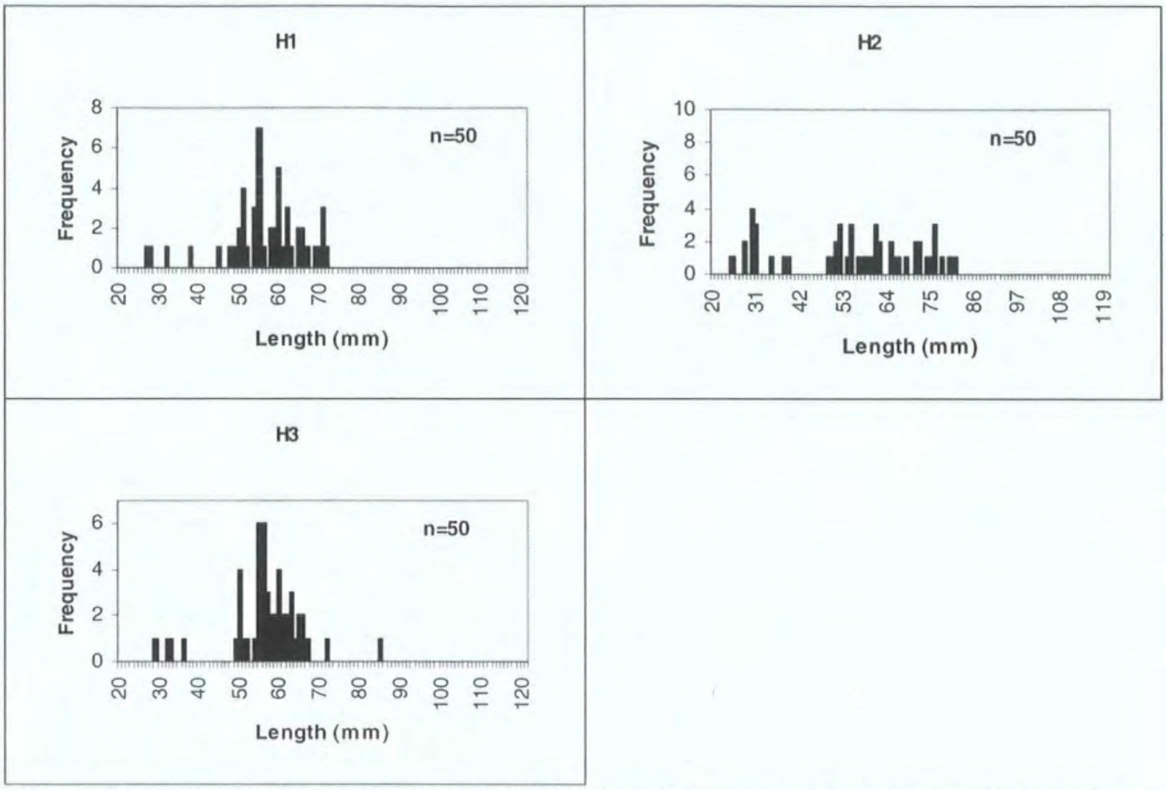


Figure 4.4.10. Figure to show the Bullhead length frequency graphs for 2003 on the Hipper.

Other Fish Species

A number of native crayfish have been caught on the Hipper during this survey at both H1 and H3, one was caught at each site in 2003. One Stone Loach was caught at H1 in 2003, but again this species were not caught and measured in 2002. A number of Perch were also caught throughout the sampling years.

4.4.1.3. Assessing the variability in the fisheries populations

There are four main types of variability inherent within fisheries populations sampled for this study: (i) between rivers; (ii) within rivers (between sites); (iii) between years; and (iv) within samples.

The between river variability has been addressed in the previous section, with the Rivelin and Loxley being compared, and the Hipper being compared to the two regulated rivers. However, the remaining three types of variation may impact upon not only the ability to compare the rivers, but also the ability to detect changed to the fisheries population caused by altering the compensation flow regimes. The variability inherent in the hydrology of the Rivelin and Loxley systems was presented in chapter two. It is the aim of this section to assess whether this hydrological variability has transferred to the fisheries population.

River scale variation

What is of interest in this section is to discover whether trends in the Rivelin, Loxley and Hipper can be observed at the river scale by averaging the site data across the rivers or whether variation dwarfs this. The error bars in Figure 4.4.11 shows that there is a large amount of variation in the average density of each of the age/size classes for each of the rivers. The size of these error bars means that we can have little confidence in stating the differences between each of the years within the rivers. However, where larger differences occur (i.e. when comparing the >0+ (<20cm) populations for the Rivelin and Hipper), it may well be possible to detect differences. It was seen above that the two uppermost sites on the Loxley had very high densities of >0+ (<20cm) trout in 2002. This has obviously had an effect on both the average and standard deviation. However, it does appear that when averaging out densities across an entire river, there is too much variation of the within the river systems. An ANOVA test was conducted on the density data above, and no differences could be found between the sampling years on the rivers using the density data despite or perhaps due to the variations pointed out above (Table 4.4.4).

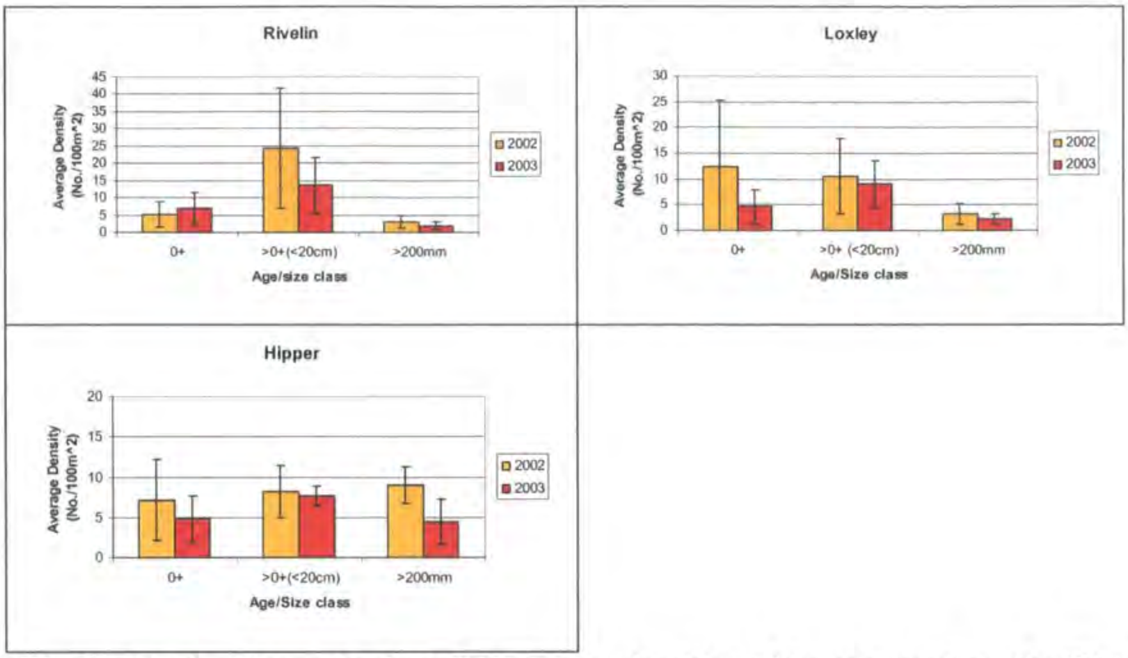


Figure 4.4.11. Figure to show the average densities of brown trout on the Rivelin, Loxley and Hipper in 2002 and 2003. Standard deviation included as error bars. Scales vary.

Figure 4.4.12 shows that there is also a large amount of within-river variability in terms of the average length at age for each cohort in the Rivelin, Loxley and Hipper. The graphs presented here are cohort plots averaged across all the river sites, and a sense of the within-river variability can be gained by looking at the standard deviation error bars on Figure 4.4.12. On the Rivelin, many of the error bars are as large as ±50mm, and as such it is difficult to determine many significant differences between the cohorts in terms of their average length at age. The 1996 data must be treated with care as it is derived from one fish. Figure 4.4.12 shows that for the Loxley, the inter-site variation portrayed in the error bars appears larger than that on the Rivelin. As such it remains difficult to detect any patterns or differences when the cohort growth rates are averaged across the Loxley. Taking account of the error bars Figure 4.4.12 shows that the lengths of one and two year old trout are relatively consistent across each of the cohorts. A similar magnitude of error bars is seen on the unregulated Hipper.

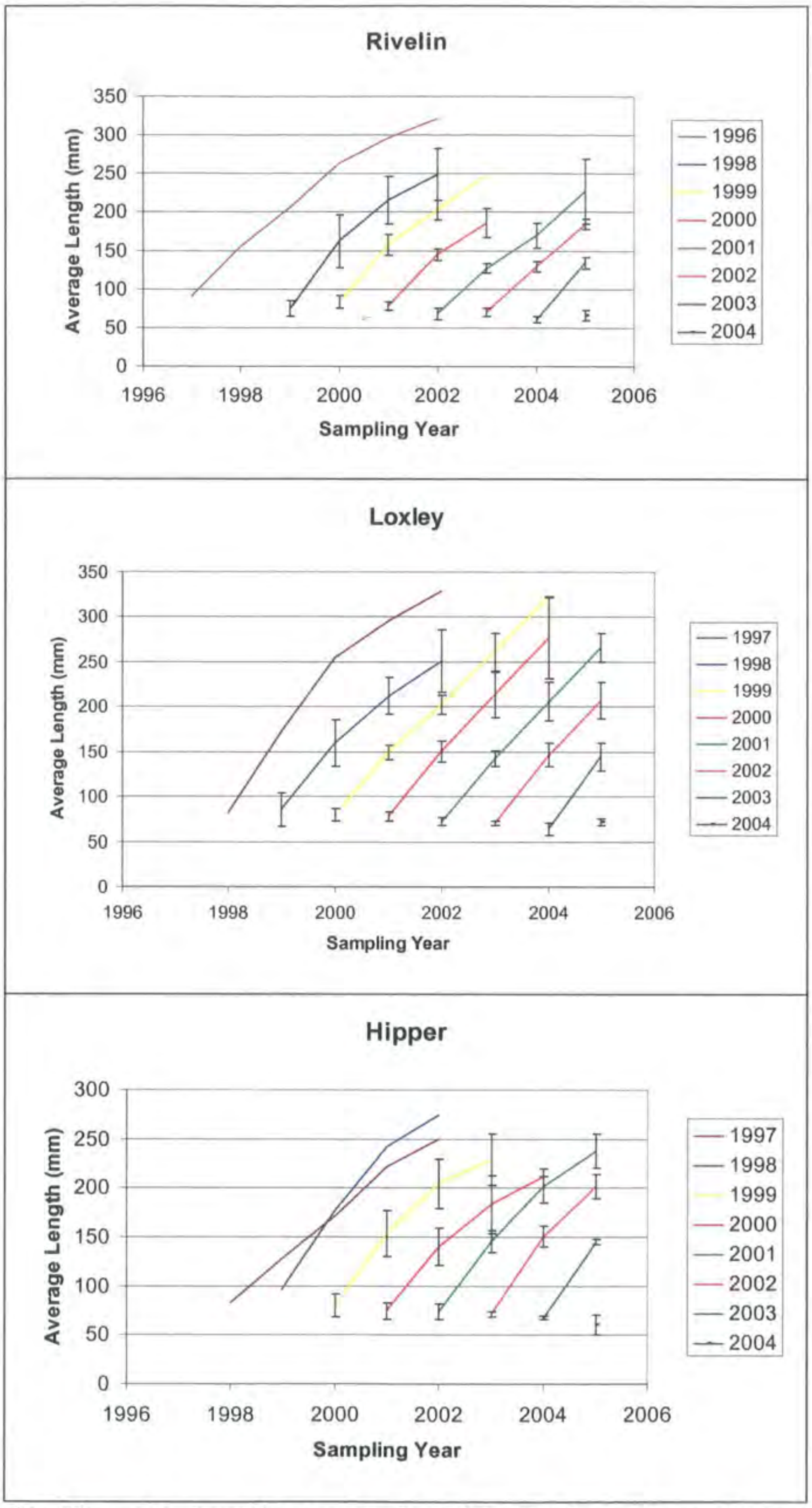


Figure 4.4.12. Figure to show the average length at age for each pre-change cohort on the Rivelin, Loxley and Hipper. Error bars showing the standard deviation of the average length at age.

It may have been convenient for the purposes of detecting the impacts of existing and changing compensation flow regime, if the many sites on each river could be averaged. However, it appears that due to the large amount of intra-site variation, this will not be possible in this case, and the aims outlined in section 4.1 will have to be addressed at the site scale.

Between -year variation

From the analysis of the river-averaged data above, it appears that there is a large amount of within-river variability present in these systems. However, it is also valuable to examine the extent to which the fish populations varies between 2002 and 2003 in order to establish whether there is scope for assessing the impacts of existing and changing compensation flows at the site scale. It may also help to establish if the large variation is being caused by an outlying site (or year), and it may be of use to remove that if present. But with only two years baseline data, few conclusions can be ascertained with any certainty.

Figure 4.4.2 shows that, on the Rivelin, for five out of the nine sites, the 0+ densities were higher in 2003 than in 2002. Figure 4.4.2 also shows that for R4 the density of 0+ trout is much larger in 2002 than in 2003. The presence of this site will lead to a greater standard deviation when averaging the 0+ densities. It appears that there is a clear trend in the >0+ (<20cm) trout with 2002 having higher densities than 2003. It can also be seen that the more upstream sites have a larger difference than the downstream sites, which then brings a spatial element into the mix. But also, it will be difficult for the sites with the large variation between the 2002 and 2003 data to determine any changes which are due to the altering of the compensation flow regimes. The ANOVA results presented previously show that there is no statistical difference between densities of any age/size class when comparing the two sampling seasons on the Rivelin. This means that there is a lack of statistically significant variation, and if some differences do occur after the change in compensation flows it could be attributed to that.

Figure 4.4.3 shows that for 0+ trout, there is a big difference between the 0+ populations in 2002 and 2003 at L1 and L2. The differences are much smaller at the rest of the sites. The large difference between these two years will lead to problems when averaging. For the

>0+ (<20cm) trout there is a much higher density in 2002 compared to 2003 at the L1 site. There are three sites where the 2003 >0+ (<20cm) densities are higher than those found in 2002. The largest differences between 2002 and 2003, this time for the >20cm trout were again found at L2 and L1, the differences for the remaining sites being less pronounced. As on the Rivelin, the ANOVA results presented previously show that there is no statistical difference between densities of any age/size class when comparing the two sampling years. This means that there is a lack of statistically significant variation, and if some differences do occur after the change in compensation flows it could again be attributed to that.

Figure 4.4.7 shows that for two of the sites on the Hipper, the 2002 population has higher densities of 0+ trout than the 2003 population, with H2 having the largest difference. The 0+ densities for the playing field site are much less variable than the two other sites. There is much less difference between the 2002 and 2003 data for H1 and H2 for the >0+ (<20cm) trout than for the 0+ trout, indicating a much more stable population. However, H3 had higher densities for the 2003 population than the 2002 population. For the >20cm trout densities the 2002 density is always higher than the 2003 density. However, there is a very large difference between the 2002 and 2003 density for H2, with smaller differences for the H1 and H3 sites. Unfortunately, due to the small number of sampling sites, no statistics could be conducted.

Within-sample variability

There is within sample variability present in the density estimates, as illustrated by Figures 4.4.2, 4.4.3 and 4.4.7, as the error bars represent the intra-sample error in estimating the population from the triple-catch data using the Carle-Strub (1978) Maximum Likelihood Method. This error has to be thought of more as a sampling error than a product of natural variation. Despite the quantitative nature of the fisheries surveys, not all fish will be captured by the electrofishing method, hence the necessity of using the Maximum-Likelihood Method.

In general the intra-sample variability in terms of the population estimates is relatively low on the Rivelin, with very little error for any of the sites in any of the sampling seasons. There is more intra-sample variability in the Loxley, especially in the >0+ (<20cm) age/size class where the error bars overlap for four out of nine sites. There was very little intra-sample variation in the Hipper sites for either 2002 or 2003 (Figure 4.4.7).

There is also intra-sample variation in the average length at age data due to natural variation in growth rate. It is assumed from the cohort growth tables that there will be an error associated with every average length at age, as a number of fish are sampled. Due to natural variation not all fish grow at the same rate, but the average of our sample can be thought of as indicative of growth rate. Due to the size of these error bars, it is difficult to distinguish between the average lengths at age of the different cohorts.

4.4.1.4. Summary

There appears to be no noticeable impact of the pre-change compensation flows upon fish populations, despite the larger flow in the Loxley. The fact that older brown trout habitat is available in the Rivelin is despite the very low compensation flow will be a consequence of the interaction between discharge and reach-scale morphology. This lack of difference in habitat is manifest in a lack of difference in the density of brown trout between the Rivelin and Loxley. What is also apparent is the similar way in which brown trout use the habitat in the two rivers. Both rivers do not appear to be habitat limited for 0+ trout, but appear to become habitat limited for the older trout, as evidenced by the high Habitat Utilisation Index (HUI) for this age/size class in each of the river.

However, the Loxley has faster growing older brown trout than the Rivelin. This is despite the habitat quality for the older trout being no different between the two rivers and the density of the older trout being similar. This could be attributed to a higher density of invertebrates present in the Loxley when compared to the Rivelin (as was seen in Chapter 3). It can also be seen that the 2001 and 2002 cohorts grow more slowly on the Rivelin and Loxley to one year old than any of the other cohorts, but this is not the case in the Hipper.

The Hipper was used as a control catchment in this study, and displayed considerable variation of habitat quality, trout density and habitat use, as expected in a natural river. It was also evident that, by using fewer sites on the Hipper, it was impossible to compare with the Rivelin and Loxley using statistics, thus lessening its utility as a control.

4.4.2. Assessing the impacts of altering the compensation flow releases

4.4.2.1. Comparing the Rivelin and Loxley

Brown trout

As with the pre-change data, Figure 4.4.13 shows that the confidence intervals of the Habitat Quality Score (HQS) produced by HABSCORE are large. As such, few significant differences are seen between the pre and post-change habitat on the Rivelin. However, on the Rivelin, the HQS for 0+ trout is lower for seven out of the eight sites, with the only site at which the 0+ HQS increased with the post change conditions was R9. There are less distinctive differences between the pre and post change HQS for the >0+ (<20cm) trout, with the HQS being very similar for three of the sites and higher for the pre change data for three sites. The HQS scores for >20cm trout remain very low for the Rivelin despite the increase in discharge (Table 4.4.15). An ANOVA analysis found no statistically significant differences between the pre and post change HQS on the Rivelin.

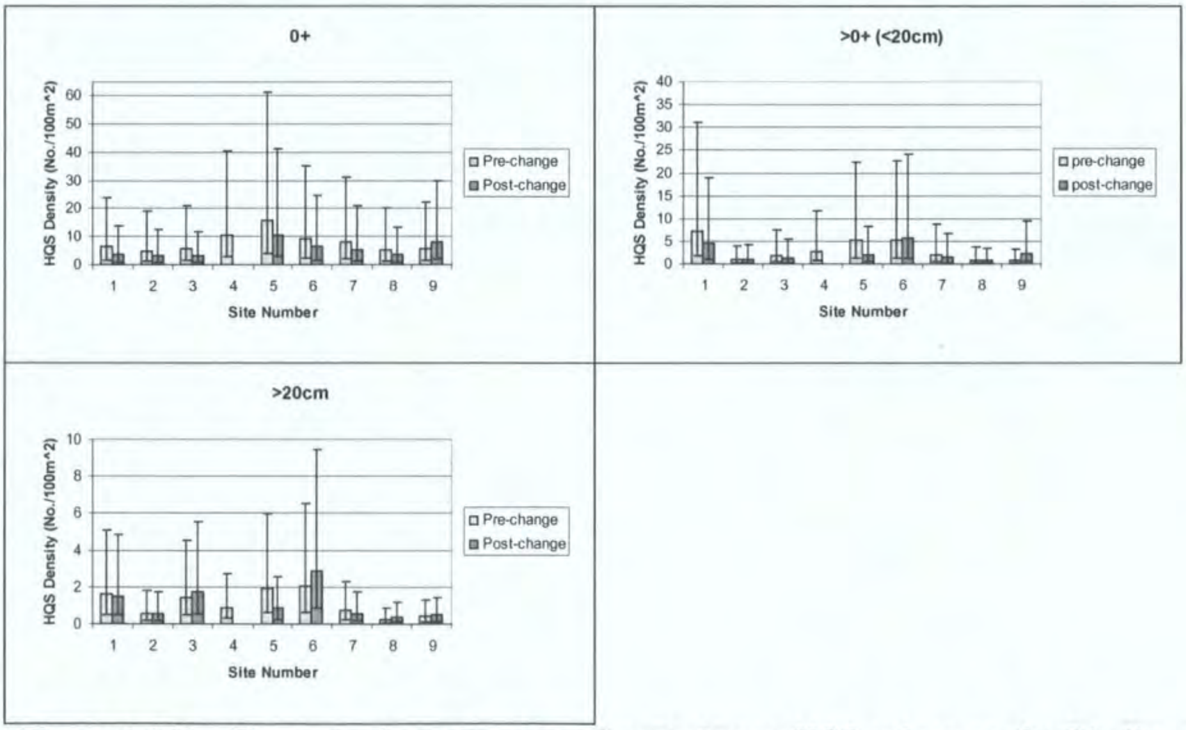


Figure 4.4.13. Figure to show the pre-and post change HQS values on the Rivelin. Scales vary to aid visualisation.

Pre-change	Post-change
HQS O+ > HQS >0+(<20cm)	HQS 0+ > HQS >0+(<20cm)
HQS 0+ > HQS >20cm	HQS 0+ > HQS >20cm
HQS >0+(<20cm) > HQS >20cm	HQS >0+(<20cm) = HQS >20cm

Table 4.4.15. Table to show the results of the ANOVA analysis comparing the HQS of the three age/size classes for both pre and post change data. Differences are statistically significant to the 95% significance level.

In terms of 0+ trout habitat the first three sites on the Loxley have very similar HQS scores for both the pre and post change data (Figure 4.4.14). For the remaining sites, it is difficult to detect a pattern between the pre and post change HQS scores, with three sites having higher HQS scores for the pre-change data and three sites displaying the opposite. It can also be seen that the 90% confidence intervals provided by HABSCORE are so large that none of these between site differences can be considered statistically significant. The predicted HQS for the >0+ (<20cm) trout are lower than that for 0+ trout for both the pre and post change data (Table 4.4.16). The confidence intervals are also very large for the >0+ (<20cm) trout, but it appears that for six of the nine sites the HQS scores for >0+ (<20cm) trout are higher for the post-change data than for the pre change data. This is perhaps counterintuitive in that you may have thought that a decrease in discharge may decrease the habitat quality for juvenile trout, who prefer deeper water to the 0+ trout. This highlights the importance of reach-scale morphology in the relationship between brown trout habitat and discharge. There is no clear pattern with the >20cm trout HQS, but the HQS is higher in the post-change than the pre-change data for five sites. Further indication in the variation of the HQS scores (whilst ignoring the confidence intervals) is given by an ANOVA analysis which showed that there are no statistically significant differences between the pre and post change HQS scores for any of the age/size classes.

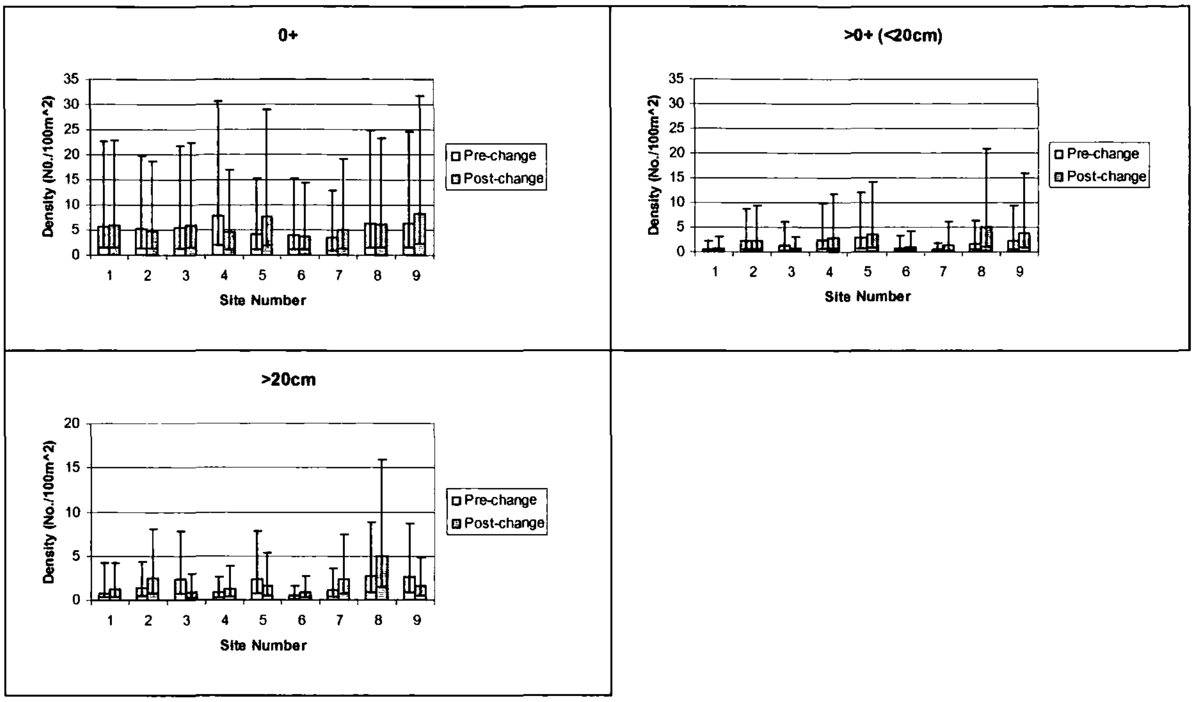


Figure 4.4.14. Figure to show the pre and post change values of HQS on the Loxley. Scales vary to aid visualisation.

Pre-change	Post-change
HQS 0+ > HQS >0+(<20cm)	HQS 0+ > HQS >0+(<20cm)
HQS 0+ > HQS >20cm	HQS 0+ > HQS >20cm
HQS >0+(<20cm) = HQS >20cm	HQS >0+(<20cm) = HQS >20cm

Table 4.4.16. Table to show the ANOVA output and direction when comparing the age/size classes using the pre and post-change HQS scores on the Loxley. 95% statistical significance.

A striking feature of the 0+ trout density is that the population present in the Rivelin in 2004 is generally lower than for any of the other sampling years, having the lowest densities for eight out of the nine sites with densities showing an improvement in 2005. The 0+ fish captured in 2004 would have been spawned under the very low flow conditions of 2003, and perhaps the lack of rainfall in 2003 did not provide high flows in the autumn to prompt spawning (e.g. Campbell, 1977). In terms of the spatial variation of 0+ densities on the Rivelin, Figure 4.4.15 shows that the upstream sites generally have a higher density than the lower sites, maintaining the bell shaped curve established in section 4.4.1. This pattern appears to be present for the 2005 samples, but the low densities present in 2004 don't adhere to this pattern.

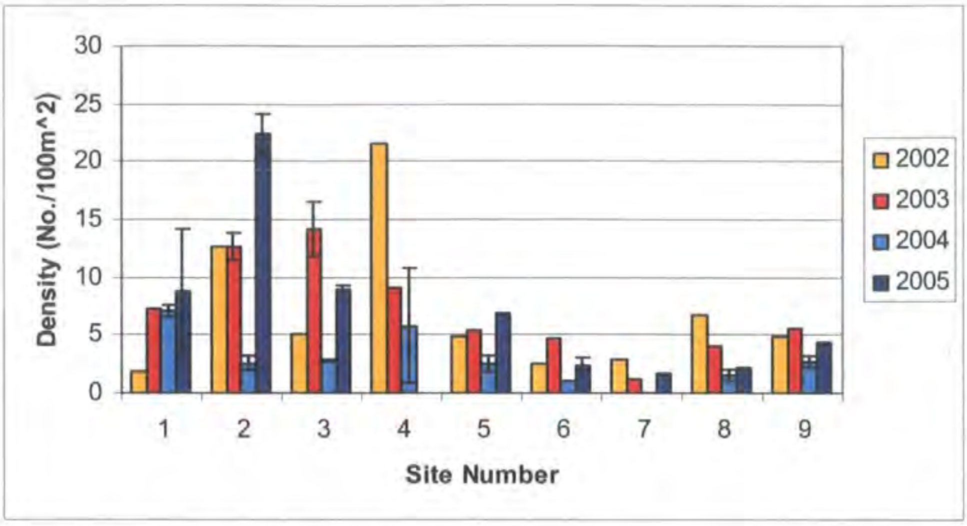


Figure 4.4.15. Figure to show the observed density of 0+ BT on the Rivelin for all sampling seasons. Error bars showing the Carle-Strub variance.

Figure 4.4.16 shows that the densities of >0+(<20cm) trout found in 2004 and 2005 on the Rivelin were generally lower than or similar to those found in 2003. Five sites seemingly having higher densities in 2003 than in 2004 or 2005 hence leading to a statistically significant difference in each case (Table 4.4.18). The densities of >0+(<20cm) trout were always higher in 2002 than in either 2004 or 2005. This is perhaps surprising as increasing the volume of water would have been thought to increase the available habitat for this age/size class. Figure 4.4.16 also shows that the HQS for >0+(<20cm) trout is also lower post-change. This again perhaps highlights the interaction of discharge and topography in determining the habitat availability. In terms of spatial variation it appears that the highest densities of >0+ (<20cm) trout were found at the R2 and R3, with this pattern largely holding for each of the sampling years.

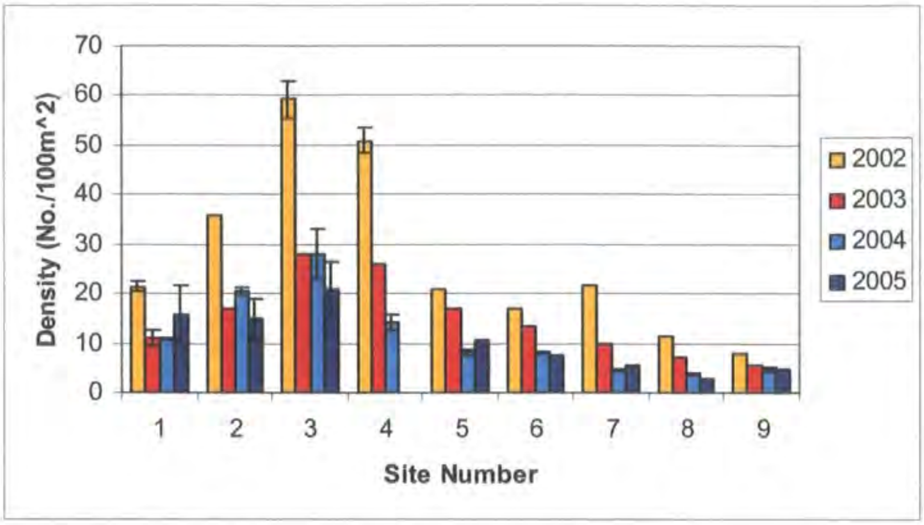


Figure 4.4.16. Figure to show the observed density of >0+ (<20cm) trout in the Rivelin for all sampling seasons. Error bars showing the Carle-Strub variance.

Figure 4.4.17 shows that the 2003 samples have the lowest or joint-lowest densities of >20cm at seven out of nine sites, with 2004 and 2005 generally having better than or equal to densities of trout. This is in contrast to the pattern displayed by the two smaller age/size classes, in which 2004 and 2005 often had the lowest densities, and despite there being no noticeable increase in HQS along the Rivelin. In terms of a spatial pattern, the highest densities of >20cm trout were found in the R3 and R4. However, unlike the densities of 0+ and >0+ (<20cm) trout, the densities of >20cm do not decrease to such a great extent towards the downstream sections of the Rivelin.

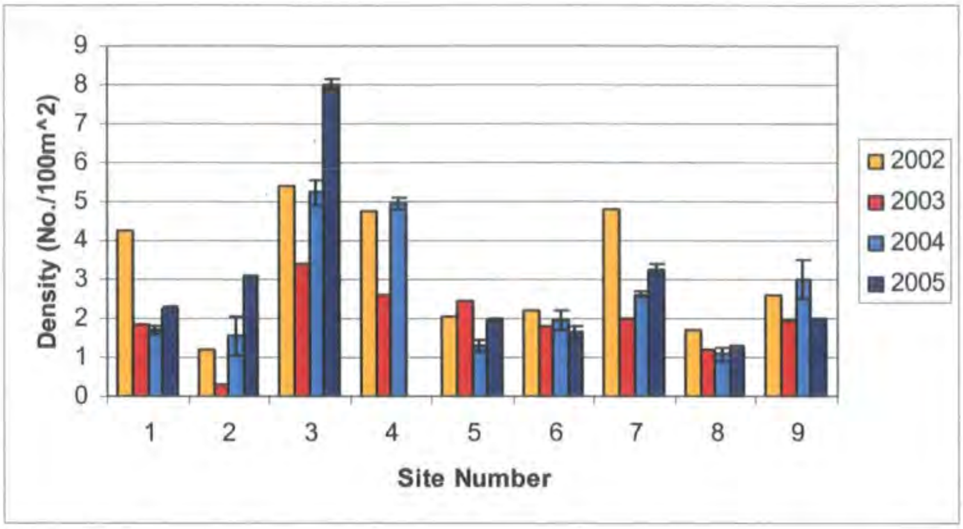


Figure 4.4.17. Figure to show the observed density of >20cm trout on the Rivelin for all sampling seasons. Error bars showing the Carle-Strub variance.

Table 4.4.18 shows that in terms of 0+ trout density there is only a statistically significant difference between the 2003 and 2004 sampling years. Table 4.4.18 shows that there is also a statistically significant difference between the 2002 and 2004 years and the 2002 and 2005 years for >0+ (<20cm) trout. It was seen in Figure 4.4.16 that the 2002 densities of >0+(<20cm) trout were generally higher than any other year and it appears they were significantly better than those found in 2004 and 2005. This is strange, because it might have been thought that increasing the discharge in the Rivelin would have increased the available habitat for the >0+ (<20cm) trout and hence led to an increase in density. The poor >0+(<20cm) trout population could be due to the poor 0+ recruitment in 2004. The lack of change in older >20cm trout is further evidenced by the lack of statistically significant differences between the sampling years. This could be due to a lack of change in habitat or that older fish may take longer to respond to any habitat change.

	2002	**2003**
0+ vs >0+(<20cm)	Diff (0+ < >0+(<20cm)	Diff (0+ < >0+(<20cm)
0+ vs >20cm	No diff	Diff (0+ > >20cm)
>0+(<20cm) vs >20cm	Diff >0+(<20cm) > >20cm	Diff >0+(<20cm) > >20cm
	2004	**2005**
0+ vs >0+(<20cm)	Diff (0+ < >0+(<20cm)	No diff
0+ vs >20cm	No diff	No diff
>0+(<20cm) vs >20cm	Diff >0+(<20cm) > >20cm	Diff >0+(<20cm) > >20cm

Table 4.4.17. Table to show the results of an ANOVA analysis comparing the densities of brown trout of the different age/size classes for the Rivelin, for all sampling seasons. Differences are statistically significant to the 95% significance level.

	0+	**>0+ (<20cm)**	**>20cm**
2002 vs 2003	Not diff	Not diff	Not diff
2002 vs 2004	Not diff	Diff (2002>2004)	Not diff
2002 vs 2005	Not diff	Diff (2002>2005)	Not diff
2003 vs 2004	Diff (2003>2004)	Not diff	Not diff
2003 vs 2005	Not diff	Not diff	Not diff
2004 vs 2005	Not diff	Not diff	Not diff

Table 4.4.18. Table to show the ANOVA analysis looking for differences between the different sampling years on the Rivelin using the densities of various age/size classes. Differences are statistically significant to the 95% significance level.

The 0+ recruitment of the four uppermost sites on the Loxley does appear to be much lower in 2005, when compared to the previous sampling years (Figure 4.4.18). It also appears that there is no real spatial or temporal pattern of 0+ trout density on the Loxley (apart from the very high densities found at sites one and two). The 2004 brown trout populations are not as poor as in the Rivelin. Perhaps the Loxley was less affected by the dry 2003 than the Rivelin which relies more heavily on overtopping and tributary inputs. The impact of a low-rainfall period may be more keenly felt on the Rivelin, where the compensation flow remains lower than that of the Loxley, whereas the Loxley may be buffered against reduction in overtopping/tributary inputs to a certain extent by the larger compensation flow. This phenomenon was also observed in the invertebrate populations (Chapter 3).

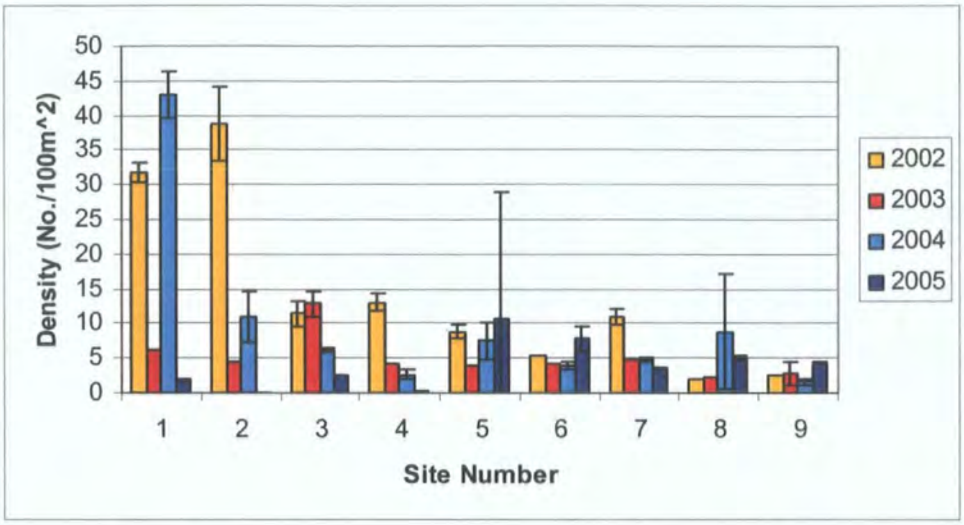


Figure 4.4.18. Figure to show the observed densities of 0+ trout on the Loxley for each sampling season. Error bars showing the Carle-Strub variance.

Figure 4.4.19 shows that it is very difficult to differentiate between the densities of >0+(<20cm) brown trout sampled pre-change and the densities sampled post-change. There are only two sites (L4 and L9) where there appears to be a noticeable difference between the pre and post-change densities. But it is also worth noting that only at one site did the post-change densities exceed the pre-change densities (L2) thus suggesting that altering the compensation flows has not improved the densities of >0+(<20cm) trout. In terms of spatial variation, the middle reaches in the Loxley appear to have slightly lower densities of >0+ (<20cm) trout than the other reaches, and this is maintained during the post-change data.

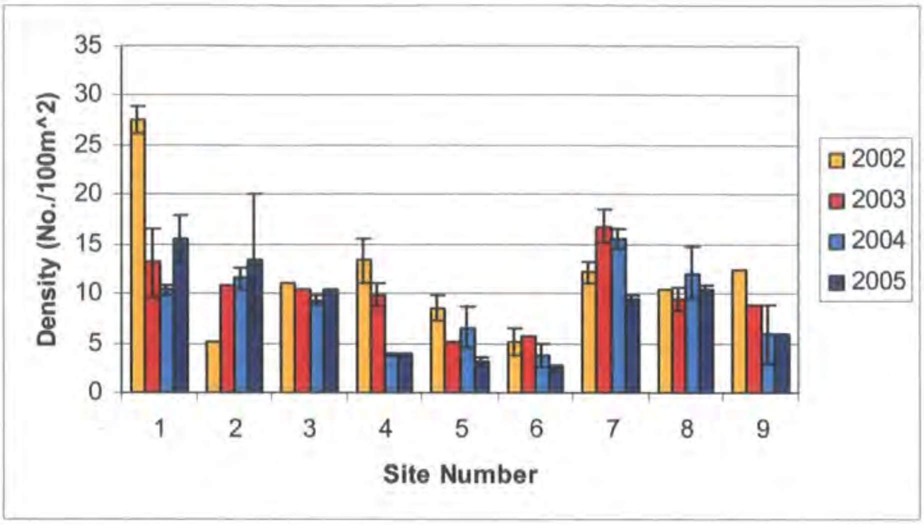


Figure 4.4.19. Figure to show the observed density of >0+ (<20cm) brown trout on the Loxley for each sampling season. Error bars showing the Carle-Strub variance.

Figure 4.4.20 shows a large increase in >20cm trout density at L2 for the post-change data. Apart from that one site, the pre-and post-change trout densities are relatively indistinguishable (with perhaps a slight increase in density at L7 and L8). At only two sites was the density found at either 2004 or 2005 lower than that found in 2003. This lack of decrease in >20cm trout density is counterintuitive as it could be thought that a decrease in discharge would lead to a decrease in the density of larger trout who prefer deeper water. Figure 4.4.14 also showed an increase in HQS at six out of nine sites. Figure 4.4.20 also shows that three of the later sites (L6, L7 and L8) display the main increases in >20cm trout density. Figure 4.4.14 shows that these sites also displayed an increase in HQS. Again, this is perhaps a counterintuitive result, which further enhances the need to consider the interaction of reach-scale morphology and discharge in setting compensation flow levels.

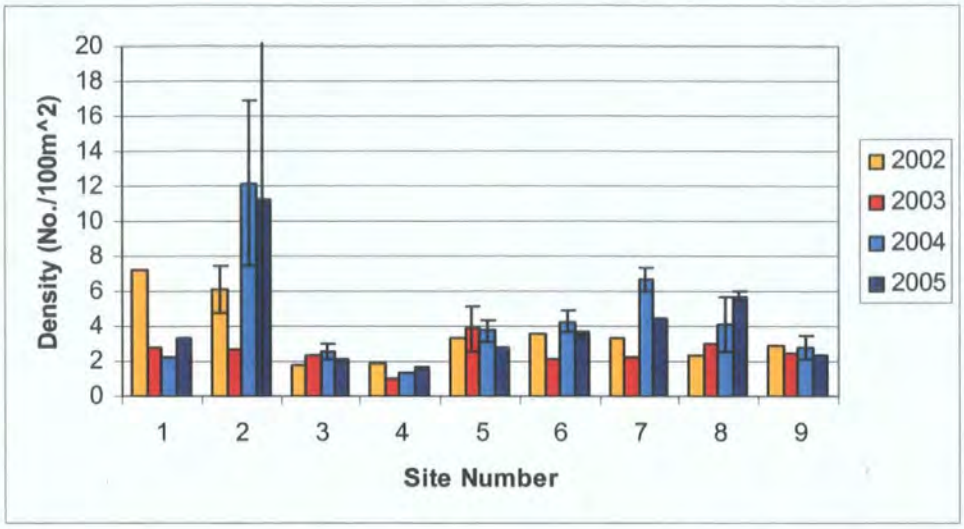


Figure 4.4.20. Figure to show the observed >20cm trout density on the Loxley for all sampling seasons. Error bars showing the Carle-Strub variance.

The discussions above have shown that there are few distinctive and coherent patterns present in the density data for the Loxley. This pattern (or lack of) is further enhanced by the results presented in Tables 4.4.19 and 4.4.20. It is shown that there is only a statistically significant difference between the 2002 and 2005 sampling years in terms of the density of 0+ trout. This was one of the few themes to come out of the above discussion and it appears that the 2005 recruitment of 0+ trout was not as successful as that in 2002, but it was not different to the lower density 0+ populations present in 2003. As of two years of data, there does not appear to have been a net improvement or degradation of the Loxley in terms of brown trout density.

	2002	2003
0+ vs >0+(<20cm)	No Diff	Diff (0+ < >0+(<20cm)
0+ vs >20cm	Diff (0+ > >20cm)	Diff (0+ > >20cm)
>0+(<20cm) vs >20cm	Diff >0+(<20cm) > >20cm	Diff >0+(<20cm) > >20cm
	2004	**2005**
0+ vs >0+(<20cm)	No Diff	Diff (0+ < >0+(<20cm)
0+ vs >20cm	No Diff	No Diff
>0+(<20cm) vs >20cm	Diff >0+(<20cm) > >20cm	Diff >0+(<20cm) > >20cm

Table 4.4.19. Table to show the results of an ANOVA analysis comparing the densities of brown trout of the different age/size classes for the Loxley for all sampling seasons. Differences are statistically significant to the 95% significance level.

	0+	>0+ (<20cm)	>20cm
2002 vs 2003	not diff	not diff	not diff
2002 vs 2004	not diff	not diff	not diff
2002 vs 2005	Diff (2002>2005)	not diff	not diff
2003 vs 2004	not diff	not diff	not diff
2003 vs 2005	not diff	not diff	not diff
2004 vs 2005	not diff	not diff	not diff

Table 4.4.20. Table to show the ANOVA analysis comparing the Loxley at different sampling seasons using the density data. Differences are statistically significant to the 95% significance level.

Table 4.4.21 shows that the only times the Rivelin and Loxley were statistically significantly different in terms of the densities of trout was for >0+ (<20cm) trout in 2002. The lack of statistically significant differences for either the pre or post change data indicates that the relationship between the Rivelin and Loxley in terms of trout density has not changed. It was also seen that there was no statistically significant difference between the Rivelin and Loxley for either the pre or post change HQS and HUI scores, further indicating the lack of change.

	0+	>0+ (<20cm)	>20cm
R 2002 vs L 2002	not diff	Diff (R>L)	not diff
R 2003 vs L 2003	not diff	not diff	not diff
R 2004 vs L 2004	not diff	not diff	not diff
R 2005 vs L 2005	not diff	not diff	not diff

Table 4.4.21. Table to show the comparison of the Rivelin and Loxley for the density of brown trout for each of the age/size classes. Statistical significance at 95%.

In terms of habitat use, Table 4.4.22 shows that for 0+ trout the populations at R7 remained significantly lower than that expected under pristine conditions. There is no clear pattern of change of the HUI values, with four out of eight sites registering a decrease in 0+ HUI and the opposite for the remaining sites. Table 4.4.22 shows that for >0+ (<20cm) trout six sites in the pre-change data have trout populations significantly better than those expected under pristine conditions, whereas there were only three sites using the post-change data (with one of those sites being unable to be compared). Six out of eight sites had lower >0+(<20cm) HUI with the post change data when compared with the pre change data. This perhaps indicates that the habitat has become slightly less limiting for >0+(<20cm) brown trout in the Rivelin under the post-change conditions. The HUI for >20cm trout is better for the post-change data, with four out of eight sites being significantly better than that expected under pristine conditions compared with three with the pre-change data. A lack of clear pattern is apparent in that half the sites register an increase in HUI whilst the other half register a decrease. This indicates that the >20cm trout habitat has not become any more or less limiting than it was pre-change. This could again be due to morphology-flow interactions limiting the impact of the change in compensation flow, as it is the morphology of the channel which determines the habitat created by the flow. Perhaps the morphology is such that any change in discharges does not impact greatly upon >20cm brown trout habitat availability. The lack of clear difference between the pre- and post change data was further highlighted with an ANOVA analysis revealing no statistically significant differences between the pre and post change data using the HUI and HQS for each age/size class.

	0+ Trout		**>0+ (<20cm) trout**		**>20cm trout**	
	Pre-change	Post-change	Pre-change	Post-change	Pre-change	Post-change
R1	1.14	2.13	2.11	2.95	1.75	1.29
R2	2.54	2.3	**26.1**	**17.51**	0.91	**3.96**
R3	1.22	1.58	**22.64**	**18.36**	2.65	**3.78**
R4	1.31	1.21	**13.26**	**10.38**	**3.79**	**5.45**
R5	0.3	0.4	3.66	4.65	1.18	1.91
R6	0.38	0.24	2.85	1.39	0.97	0.62
R7	**0.09**	**0.14**	**7.41**	3.12	**4.15**	**5.15**
R8	0.84	0.52	**9.93**	3.96	**4.7**	3.1
R9	0.83	0.43	**8.87**	2.08	**5.51**	**5.25**

Table 4.4.22. Table to show the HABSCORE HUI values for both the pre and post-change data on the Rivelin. Statistically significant HUI values are in bold.

	Pre-change	**Post-change**
0+ vs >0+(<20cm)	Diff 0+ < >0+(<20cm)	Diff 0+ < >0+(<20cm)
0+ vs >20cm	Diff 0+ < >20cm	Diff 0+ < >20cm
>0+(<20cm) vs >20cm	Diff >0+(<20cm) > >20cm	Diff >0+(<20cm) = >20cm

Table 4.4.23. Table to show the results of an ANOVA analysis comparing the HUI values of the different age/size classes for the pre and post change data on the Rivelin. Differences are statistically significant to the 95% significance level.

Table 4.4.24 shows that for the Loxley 0+ trout HUI, the post-change HUI is lower than the pre-change value for seven out of nine sites. This indicates a lowering of 0+ trout density in relation to the overall habitat. This could indicate an increase in the habitat available which was not followed by an increase in 0+ densities (as evidenced by Figure 4.4.15), thus further indicating that the 0+ populations on the Loxley are not habitat controlled. As with the Rivelin, the age/size class with the most statistically significant HUI values was the >0+ (<20cm) trout. However, there were two sites for which the >0+ (<20cm) trout populations were significantly better than those expected under pristine condition in the pre-change data, that were not so for the post-change data. For seven of the nine sites, the actual HUI value was lower for the post-change data, and these were not the same seven sites as for the 0+ trout. This again indicates a reduction in habitat limitation. For the >20cm trout, one of the sites at which the trout population was significantly better than that expected under pristine conditions, was not so under the post-change conditions. The actual HUI values were lower for the post-change data for six out of the nine sites, again indicating a reduction in habitat use. When the HUI values for the before and after data were analysed using ANOVA, there was a statistically significant difference between the pre and post change HUI values for the 0+ trout, but none for the older age/size classes. This indicates a lowering of 0+ trout density but also a lowering (in general) of habitat use on the Loxley for the post change data when compared with the pre change data.

	0+ Trout		**>0+ (<20cm) trout**		**>20cm trout**	
	Pre-change	Post-change	Pre-change	Post-change	Pre-change	Post-change
L1	2.33	1.18	**34.57**	**13.98**	**5.84**	1.64
L2	2.6	0.38	4.14	5.73	2.96	4.72
L3	2.21	0.69	**7.81**	**15.46**	0.87	2.63
L4	0.93	0.2	5.31	1.36	1.59	1.16
L5	1.48	1.17	2.31	1.34	1.5	1.96
L6	1.23	1.47	**7.43**	3.22	**5.38**	**4.44**
L7	2.15	0.81	**37.75**	**8.94**	2.42	2.35
L8	0.3	1.1	**8.25**	2.28	1.59	0.98
L9	0.99	0.31	4.9	1.56	1.83	1.61

Table 4.4.24. Table to show the HUI values on the Loxley, for both the pre-and post-change data. Statistically significant (95%) values in bold.

	Pre-change	**Post-change**
0+ vs >0+(<20cm)	Diff 0+ < >0+(<20cm)	Diff 0+ < >0+(<20cm)
0+ vs >20cm	No diff	Diff 0+ < >20cm
>0+(<20cm) vs >20cm	Diff >0+(<20cm) > >20cm	No diff

Table 4.4.25. Table to show the results of an ANOVA analysis comparing the HUI values of the different age/size classes for the Rivelin and Loxley. Differences are statistically significant to the 95% significance level.

Table 4.4.26 shows that the 2003-2004 growth period (red) was relatively bad (especially for one year old trout) on the Rivelin for all age groups, at its best growth rates were similar to those found in the 2002-2003 (yellow) period. The 2003-2004 period was characterised hydrologically as a period of low rainfall and low reservoir overtopping (as was seen in Chapter 2). But Table 4.4.26 also shows that the 2004-2005 growth rates were similar to, if not better than (for two and three year old trout), than those found in the pre-change growth periods. This indicates that, perhaps the slight increase in >20cm trout habitat as observed in Figure 4.4.15, has not been converted into slightly greater densities but increased growth rates of older brown trout. There is a lot of variability in the growth data, making it difficult to determine the impacts of altering the compensation flow regimes.

The growth rates of brown trout in the Loxley do not appear to have been affected by the alteration of the compensation flows. Table 4.4.27 shows that the growth rate to one year old for the 2003 cohort was slower than for any of the other cohorts, but this phenomenon was also seen in the Rivelin and so perhaps can be attributed to regional forcing. The growth rates in 2004-2005 are generally good, with the 2003-2004 growth period appearing to have the slowest growth rates. Again, the slow growth rate in the 2003-2004 growth period was also highlighted as a slow growth period in the Rivelin as well.

R1	1998	1999	2000	2001	2002	2003	2004
1	87.8 (1)	97.5 (3)	94.9 (8)	83.8 (9)	81 (26)	75 (17)	87.6 (6)
2	100.4 (1)	128.9 (3)	102.7 (8)	84.4 (9)	100.9 (14)	105.6 (6)	
3	117 (1)	71.7 (3)	38.2 (2)	116.3 (1)	81.7 (6)		
4	24.6 (1)						
R2	**1998**	**1999**	**2000**	**2001**	**2002**	**2003**	**2004**
1				69.6 (5)	94.1 (28)	73.3 (23)	87.4 (6)
2				95.9 (5)	77.2 (13)	111.4 (9)	
3				69.8 (3)	98.3 (3)		
R3	**1998**	**1999**	**2000**	**2001**	**2002**	**2003**	**2004**
1	106.9 (1)	100.1 (4)	92.7 (5)	88.4 (25)	90.9 (37)	80.0 (18)	95.6 (7)
2	166.7 (1)	112.9 (4)	96.1 (5)	83.8 (15)	82.3 (19)	116.5 (8)	
3	77.7 (1)	69.8 (4)	64.3 (2)	57.6 (9)	101.5 (4)		
4	74.8 (1)			206.2 (1)			
R4	**1998**	**1999**	**2000**	**2001**	**2002**	**2003**	**2004**
1		117.1 (1)	100.3 (7)	86.3 (21)	85.4 (22)	76.4 (16)	81.4 (7)
2		109.5 (1)	105.1 (7)	81.5 (16)	88.3 (11)	107.7 (6)	
3		71.0 (1)	101.9 (2)	61.7 (4)	107.8 (5)		
R5	**1998**	**1999**	**2000**	**2001**	**2002**	**2003**	**2004**
1			92.9 (2)	95.9 (16)	91.2 (21)	82.8 (12)	79.5 (8)
2			96.3 (2)	87.2 (8)	76.7 (8)	92.4 (6)	
3				41.2 (5)	86.7 (4)		
R6	**1998**	**1999**	**2000**	**2001**	**2002**	**2003**	**2004**
1		105.4 (5)	100.6 (5)	75.8 (16)	88.1 (21)	71.2 (14)	87.4 (6)
2		83.7 (5)	81.7 (5)	78.2 (13)	69.6 (9)	97.1 (7)	
3		76.4 (5)	61.3 (2)	56.5 (7)	88.3 (2)		
4		112.2 (1)		73.4 (1)			
R7	**1998**	**1999**	**2000**	**2001**	**2002**	**2003**	**2004**
1		115.5 (3)	96.7 (11)	93.6 (18)	87.9 (23)	81.4 (8)	72.7 (3)
2		100.3 (3)	83.1 (11)	78.8 (10)	96.9 (15)	96.9 (6)	
3		67.5 (3)	60.7 (5)	72.6 (5)	78.5 (6)		
R8	**1998**	**1999**	**2000**	**2001**	**2002**	**2003**	**2004**
1		109.6 (2)	110.3 (10)	92.2 (16)	92.6 (18)	78.3 (16)	80.8 (5)
2		112.5 (2)	98.2 (10)	77.1 (11)	86.5 (4)	113.5 (7)	
3		88.0 (2)	87.0 (3)	54.7 (5)	86.4 (3)		
4				118.5 (1)			
R9	**1998**	**1999**	**2000**	**2001**	**2002**	**2003**	**2004**
1	83.1 (1)	89.6 (4)	91.8 (9_	80.1 (12)	74.6 (17)	67.7 (10)	69.7 (7)
2	112.3 (1)	89.9 (4)	101.3 (9)	87.7 (10)	85.6 (10)	97.4 (4)	
3	70.5 (1)	81.7 (4)	55.3 (4)	114.0 (4)	104.6 (5)		
4	93.0 (1)						

Table 4.4.26. Table to show the percentage growth rate for each cohort for each site on the Rivelin. Green = 2001-2002 growth period; yellow = 2002-2003; red = 2003 – 2004; blue = 2004 – 2005. The number of fish used in the calculation shown in brackets.

L1	1998	1999	2000	2001	2002	2003	2004
1	107.7 (2)	99.0 (6)	107.6 (10)	96.2 (18)	93.7 (21)	84.9 (17)	92.6 (10)
2	125.6 (2)	99.8 (6)	100.7 (10)	108.0 (9)	119.2 (9)	121.4 (4)	
3	99.5 (2)	93.9 (6)	121.8 (2)	95.2 (3)	80.2 (4)		
4	102.0 (2)	142.1 (1)	86.4 (1)				
L2	**1998**	**1999**	**2000**	**2001**	**2002**	**2003**	**2004**
1	109.2 (2)	112.8 (7)	97.1 (19)	89.6 (15)	87.1 (30)	80.6 (20)	90.3 (6)
2	89.8 (2)	109.6 (7)	130.1 (19)	121.0 (12)	113.5 (18)	129.1 (7)	
3	128.0 (2)	97.6 (7)	113.4 (11)	152.1 (7)	136.3 (7)		
4	100.9 (2)	60.6 (3)	103.9 (7)				
L3	**1998**	**1999**	**2000**	**2001**	**2002**	**2003**	**2004**
1		95.4 (4)	104.2 (4)	97.2 (11)	86.1 (26)	72.8 (13)	85.6 (10)
2		96.8 (4)	120.0 (4)	93.5 (5)	90.3 (12)	99.5 (4)	
3		75.4 (4)	160.4 (2)	81.9 (3)	74.8 (4)		
4			148.7(1)				
L4	**1998**	**1999**	**2000**	**2001**	**2002**	**2003**	**2004**
1	140.9 (1)	100.5 (4)	89.8 (11)	89.1 (10)	86.0 (26)	87.0 (11)	92.1 (6)
2	130.9 (1)	95.9 (4)	97.4 (4)	92.2 (8)	110.2 (12)	98.4 (6)	
3	35.2 (1)	91.0 (4)	91.0 (1)	133.6 (1)	103.6 (3)		
4	81.3 (1)			86.5 (1)			
L5	**1998**	**1999**	**2000**	**2001**	**2002**	**2003**	**2004**
1	110.6 (3)	99.3 (9)	95.8 (5)	92.3 (18)	92.3 (14)	91.3 (11)	97.2 (8)
2	98.4 (3)	93.0 (9)	95.4 (5)	99.3 (15)	141.6 (5)	123.8 (6)	
3	78.5 (3)	91.7 (9)	83.8 (3)	134.5 (5)	117.0 (2)		
4	67.9 (3)	52.8 (3)		96.9 (1)			
L6	**1998**	**1999**	**2000**	**2001**	**2002**	**2003**	**2004**
1		83.9 (1)	91.7 (2)	93.8 (10)	91.3 (25)	86.2 (9)	80.8 (9)
2		113.3 (1)	110.9 (2)	101.8 (10)	108.0 (15)	141.3 (4)	
3		101.7 (1)	117.8 (2)	95.4 (5)	95.2 (5)		
4		117.3 (1)	75.5 (1)	153.9 (1)			
L7	**1998**	**1999**	**2000**	**2001**	**2002**	**2003**	**2004**
1	102.7 (3)	95.6 (7)	88.9 (4)	87.6 (27)	88.2 (31)	68.0 (14)	92.1 (92.1)
2	85.0 (3)	108.1 (7)	88.1 (4)	98.1 (21)	90.6 (16)	108.3 (4)	
3	99.3 (3)	76.6 (7)	99.3 (2)	77.5 (8)	118.5 (5)		
4	65.4 (3)	148.1 (1)	45.6 (1)	117.9 (1)			
L8	**1998**	**1999**	**2000**	**2001**	**2002**	**2003**	**2004**
1	69.4 (1)	109.8 (7)	101.3 (11)	84.0 (18)	86.5 (24)	81.8 (15)	86.5(8)
2	109.0 (1)	95.6 (7)	96.4 (11)	97.5 (14)	110.4 (15)	102.1 (5)	
3	81.0 (1)	74.4 (7)	73.1 (4)	104.4 (2)	69.4 (5)		
4	26.7 (1)						
L9	**1998**	**1999**	**2000**	**2001**	**2002**	**2003**	**2004**
1		96.7 (3)	96.0 (6)	83.0 (28)	82.5 (13)	67.9 (12)	91.5 (8)
2		87.7 (3)	101.4 (6)	88.2 (23)	93.3 (7)	105.2 (6)	
3		93.8 (3)	81.2 (3)	80.7 (8)	109.8 (4)		

Table 4.4.27. Table to show the percentage growth rate for each cohort on the Loxley for all sites. Green = 2001-2002 growth period; yellow = 2002-2003; red = 2003 – 2004; blue = 2004 – 2005.

As was highlighted in Tables 4.4.26 and 4.4.27, the growth rates to one year old are poor for both the Rivelin and Loxley for 2003/2004. The extent to which this period is characterised hydrologically by low flows has been discussed above. It is also interesting on the Rivelin that 2004/2005 growth period on the Rivelin is providing faster growth rates for the older brown trout, thus suggesting that altering the compensation flows may have helped the growth rates of older trout.

Table 4.4.28 shows that for the Loxley one year old trout there is also a statistically significant difference between the growth rates in 2003-2004 and each of the other sampling years. The 2003/2004 'drought' did not manifest itself in the Loxley in terms of 0+ recruitment as on the Rivelin, but the young of the year trout on the Loxley did grow more slowly during this period. This further indicates that this year may have just been a bad year due to natural variation, and not related to alteration of the compensation flows as the 2004-2005 growth rates are indistinguishable from the pre-change growth periods.

It can also be seen from Table 4.4.28 that, as was the case in the Rivelin, there are fewer differences in growth rates with the older trout, with faster growth for two year old trout occurring in 2004-2005 than 2002/2003. There are no statistically significant differences between any of the growth periods in terms of growth rates of three year old trout. Thus, despite the growth rate for older trout appearing to increase on the Rivelin, the Loxley shows no change.

	Rivelin	Loxley
One year old		
2001/02 vs 2002/03	No diff	No diff
2001/02 vs 2003/04	Diff (01/02>03/04)	Diff (01/02>03/04)
2001/2 vs 2004/05	No diff	No diff
2002/03 vs 2003/04	Diff (02/03>03/04)	Diff (02/03>03/04)
2002/03 vs 2004/05	No diff	No diff
2003/04 vs 2004/05	No diff	Diff (04/05>03/04)
Two years old		
2001/02 vs 2002/03	Diff (01/02>02/03)	No diff
2001/02 vs 2003/04	No diff	No diff
2001/2 vs 2004/05	No diff	No diff
2002/03 vs 2003/04	No diff	No diff
2002/03 vs 2004/05	Diff (04/05>02/03)	Diff (04/05>02/03)
2003/04 vs 2004/05	Diff (04/05>03/04)	No diff
Three years old		
2001/02 vs 2002/03	No diff	No diff
2001/02 vs 2003/04	No diff	No diff
2001/2 vs 2004/05	Diff (04/05>01/02)	No diff
2002/03 vs 2003/04	No diff	No diff
2002/03 vs 2004/05	Diff (04/05>02/03)	No diff
2003/04 vs 2004/05	No diff	No diff

Table 4.4.28. Table to show the ANOVA output when comparing sampling seasons using the percentage of standard growth rate on the Rivelin and Loxley. Statistically significance at the 95% level.

When the growth rates for the four main growth periods were used in an ANOVA test to differentiate between the Rivelin and Loxley, there was no difference between the Rivelin and Loxley for each of the growth periods for 0+ trout, further indicating the stable, non-habitat limited nature of the 0+ trout populations. Table 4.4.29 shows that for the two year old trout, there was a difference between the Rivelin and Loxley for the 2003-2004 growth period, but no difference for the 2004-2005 period. The growth rate to three years old was different in the Rivelin and Loxley for the first three growth periods, but not different for the 2004-2005 period. This could be due to the increased discharge in the Rivelin, making the conditions better for older trout.

One year old	Rivelin Vs Loxley
2001/2002	No diff
2002/ 2003	No diff
2003/2004	No diff
2004/2005	No diff
Two years old	
2001/2002	No diff
2002/ 2003	Diff (L>R)
2003/2004	Diff (L>R)
2004/2005	No diff
Three years old	
2001/2002	Diff (L>R)
2002/ 2003	Diff (L>R)
2003/2004	Diff (L>R)
2004/2005	No diff

Table 4.4.29. Table to show the ANOVA results from comparing the Rivelin and Loxley using percentage of standard growth rates for all growth periods. Differences are statistically significant to the 95% significance level.

It is interesting when the growth rates of the different ages of trout for the four main growth periods are compared (Table 4.4.30). Often on the Rivelin, the younger trout were growing more quickly than the older tout, whilst the reverse was true for the Loxley. This perhaps indicates that relatively speaking there is a greater availability of habitat for younger trout than older trout in the Rivelin, and vice versa on the Loxley. Table 4.4.30 also shows that two year old trout grew faster than one year old trout in the post change samples, perhaps indicating an increase in habitat availability for older trout under the post-change flows.

Rivelin	2001-2002	2002-2003	2003-2004	2004-2005
1 vs 2	Diff (2>1)	No diff	Diff (2>1)	Diff (2>1)
1 vs 3	Diff (1>3)	Diff (1> 3)	No diff	No diff
2 vs 3	Diff (2>3)	No diff	No diff	Diff (2>3)
Loxley	**2001-2002**	**2002-2003**	**2003-2004**	**2004-2005**
1 vs 2	Diff (1<2)	Diff (1<2)	Diff (1<2)	Diff (1<2)
1 vs 3	No diff	No diff	Diff (1<3)	No diff
2 vs 3	Diff (2>3)	No diff	No diff	No diff

Table 4.4.30. Table to show the output from the ANOVA analysis comparing the growth rates of the different ages for each growth period for the Rivelin and Loxley. Differences are statistically significant to the 95% significance level.

Bullheads

Figure 4.4.21 shows that none of the top three sites on the Rivelin contained a population of Bullheads. Most subsequent sites appears to have good adult populations in most of the sampling years, with again most sites having a noticeable 0+ population as well. The main exception to this is sites R5 and R6 where the overall population is much lower in 2004 and 2005 compared with 2002, with a reduction in 0+ populations as well. On the whole it appears that the Bullhead population has not been affected by the changing of the compensation flow regimes to any great extent (apart from at R5 and R6).

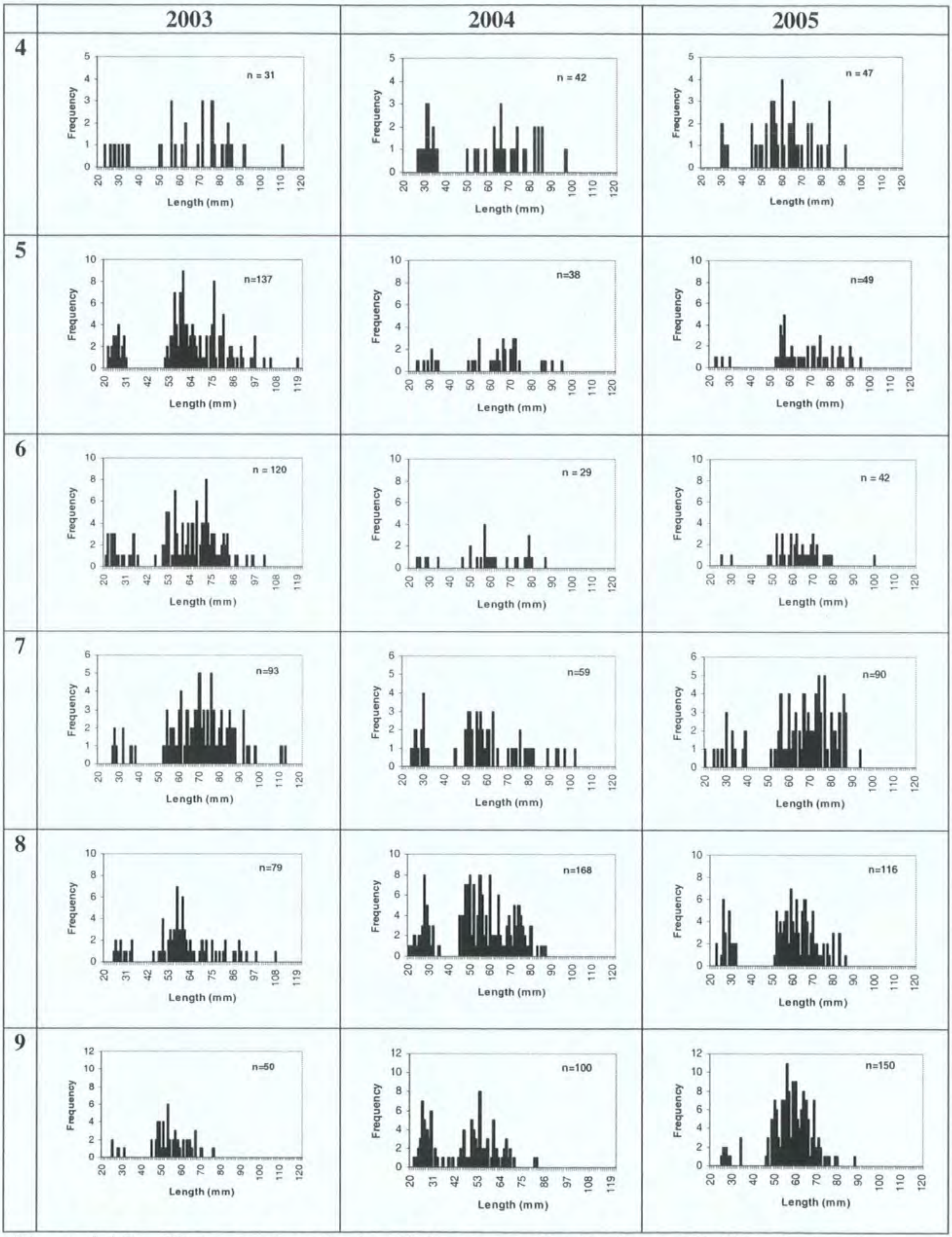
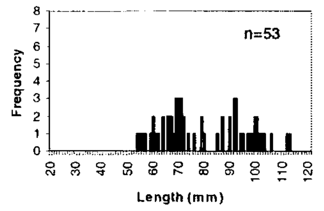
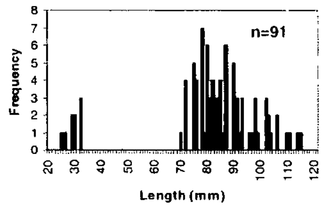
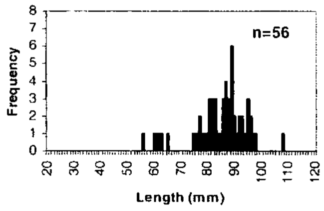
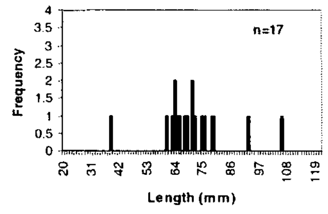
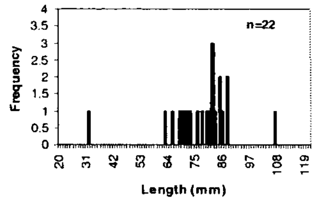
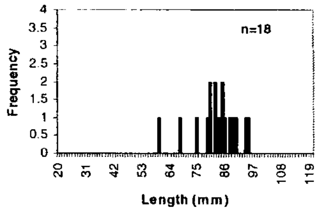
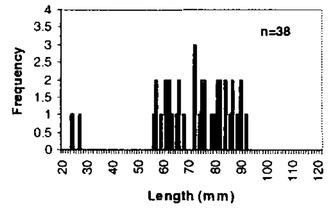
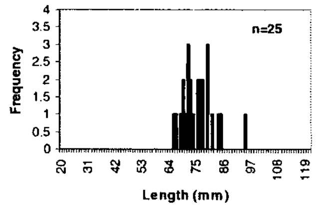
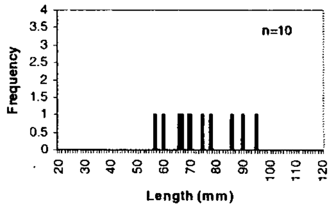


Figure 4.4.21. Figure to show the length frequency distributions of Bullheads at each of the sampling sites on the Rivelin for 2003, 2004 and 2005.

Figure 4.4.22 shows that despite there being a healthy adult population at every site on the Loxley for each of the sampling years, the level of juvenile recruitment was very low. A number of sites even had two out of three years with no evidence of juvenile recruitment within the Bullhead population. However, the fact that the adult populations had survived into 2004 and 2005 with the poor recruitment (at some sites) in the preceding years, indicates that the sites chosen for this study may have just missed the juvenile rearing grounds for the Bullheads. It is also apparent that the Bullhead population decreases as a function of distance downstream, this could be because the cumulative effects of the industry surrounding the Loxley will increase with distance downstream and the Bullhead species are thought to be relatively sensitive to pollution. The Bullhead populations do not appear to have been affected by altering the compensation flows in the Loxley.

	2003	2004	2005
1			
2			
3			

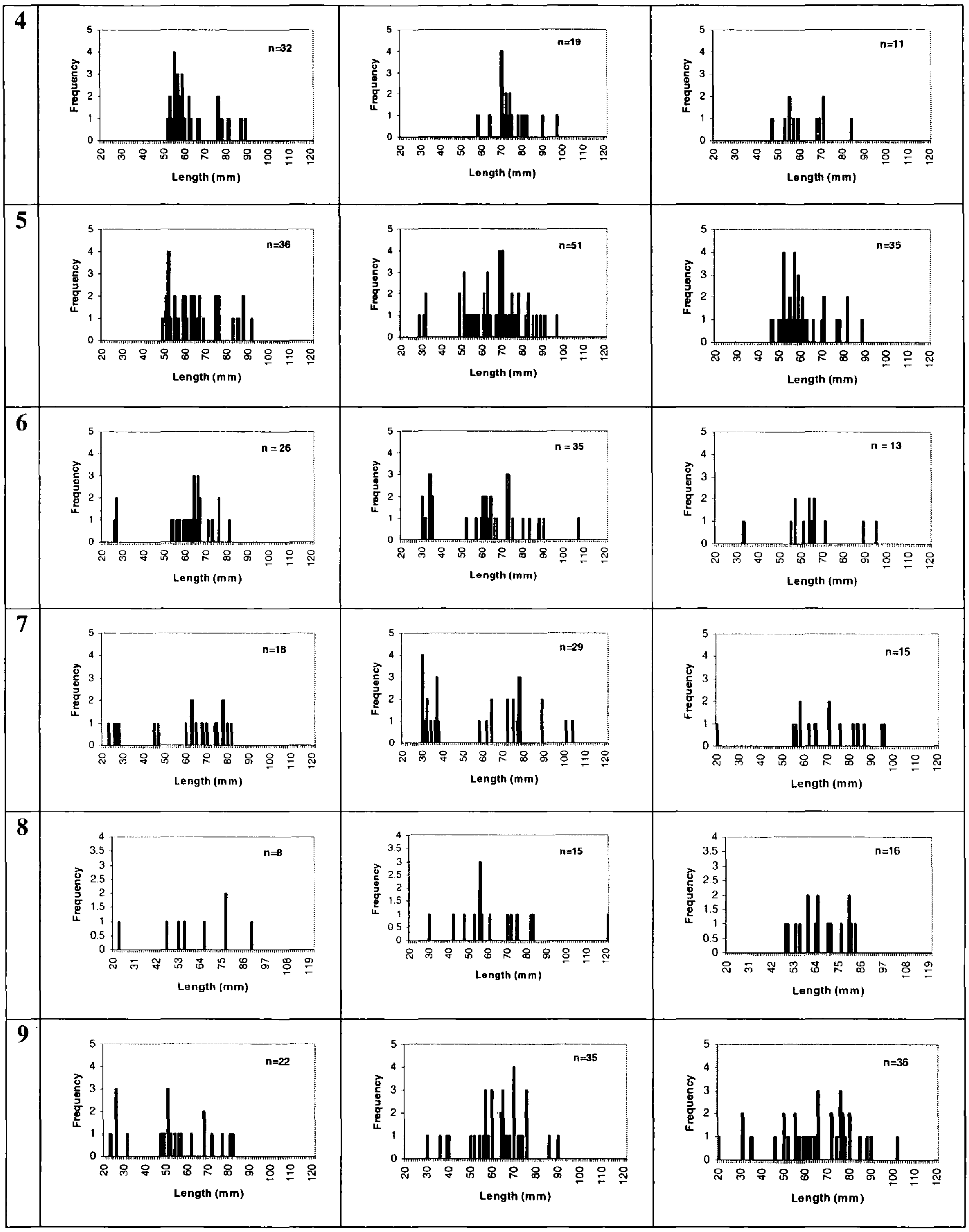


Figure 4.4.22. Figure to show the length frequency distributions of Bullheads at each of the sampling sites on the Loxley for 2003, 2004 and 2005.

Other Fish Species

A number of Brook Lamprey were found in the upper reaches of the Rivelin (Table 4.4.31). The abundances listed in Table 4.30 cannot be taken as true estimates of population as the habitats were not fished to exhaustion, due to the difficulty in fishing this species. The main aim was to ascertain the presence of a population. What is encouraging from these results is that not-only is the Rivelin providing habitat for these species, but altering the compensation flows does not appear to have removed the habitat or the populations.

	2002	2003	2004	2005
R1	0	1	5	3
R2	0	5	9	6
R3	0	0	1	8
R4	0	0	0	4

Table 4.4.31. Table to show the abundance of brook lamprey (*Lampetra planeri*) in the Rivelin for each sampling season.

In the higher reaches of the Rivelin, where Bullheads were absent there were healthy populations of Stone loach (*Barbatulus barbatulus*) (Table 4.4.32). As the populations of Bullheads increased were larger in the downstream sites, the populations of Stone loach decreased. Altering the compensation flow regimes does not appear to have affected the Stone Loach populations.

	2002	2003	2004	2005
R1	No data	59	76	65
R2	No data	18	17	45
R3	No data	36	26	21
R4	No data	8	6	0
R5	No data	15	1	0
R6	No data	12	3	5
R7	No data	15	3	0
R8	No data	8	5	3
R9	No data	1	2	0

Table 4.4.32. Table to show the number of Stone Loach (*Barbatulus barbatulus*) captured in the Rivelin sites for each sampling season.

On the Loxley, Stone loach were only caught at L7 in 2003, and L8 in 2004 and 2005. A very small number of Perch, Roach, Grayling and Pike were caught through the sampling efforts, and it is thought that these species are escapees from the mill ponds which feed off and back into the main river, and many of which are used for recreational fishing purposes. It is thus impossible to detect the impacts of the compensation flow regimes from this data.

4.4.2.2. Comparing the Rivelin and Loxley with the Hipper

Brown trout

First, 2004 and 2005 populations found in the Hipper are examined, and then compared with the existing populations. What is then of interest is to determine whether changes present in the Rivelin and Loxley are present in the Hipper and hence potentially disentangle the twin influences of natural variability and compensation flow alterations. As there has been no change in compensation flows in the Hipper, the terms pre and post change are redundant. Therefore, the 2002 and 2003 samples will be addressed as 'phase I' and the 2004 and 2005 samples as 'phase II'.

Figure 4.4.23 shows that for 0+ trout there is very little difference between phase I and phase II Habitat Quality Score (HQS). There is a larger difference between the phase I and phase II for the >0+ (<20cm), especially at H3, where the phase II HQS is higher than the phase I HQS. But there is no clear pattern for >0+ (<20cm) trout, as for H2, the phase I HQS is higher than the phase II values. A similar lack of consensus is seen for the >20cm trout, where for two of the sites the HQS is higher for the phase I data, and the opposite for the remaining site. What is also interesting in Figure 4.4.23 is that the HQS score for H2 is always higher for the phase I data, whilst the HQS for H3 is always higher for the phase II data. The extent of the 90% confidence limits produced by the HABSCORE program are such that there is no statistically significant difference between the phase I and phase II HQS values for any of the age/size classes. No ANOVA analysis was performed, due to the small number of sampling sites present on the Hipper.

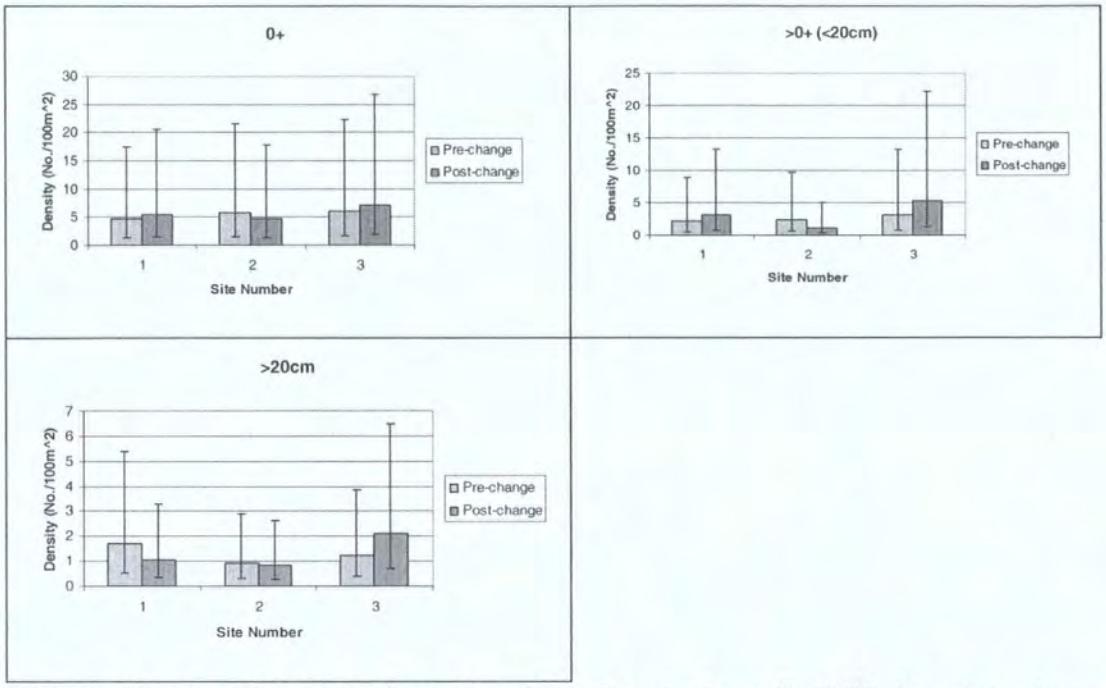


Figure 4.4.23. Figure to show the phase I and phase II HQS for each site on the Hipper.

Figure 4.4.24 shows that 2003 and 2005 had very similar 0+ trout densities at each site. The lowest 0+ densities were found in 2004. For >0+ (<20cm) trout 2004 and 2005 densities were very stable, with more variation in the 2002 and 2003 densities. The highest densities of >20cm trout were found in the 2002 samples, with lots of variation between the remaining sampling seasons. This data shows no clear patterns in the trout density data, which we would expect to see replicated in the Rivelin and Loxley.

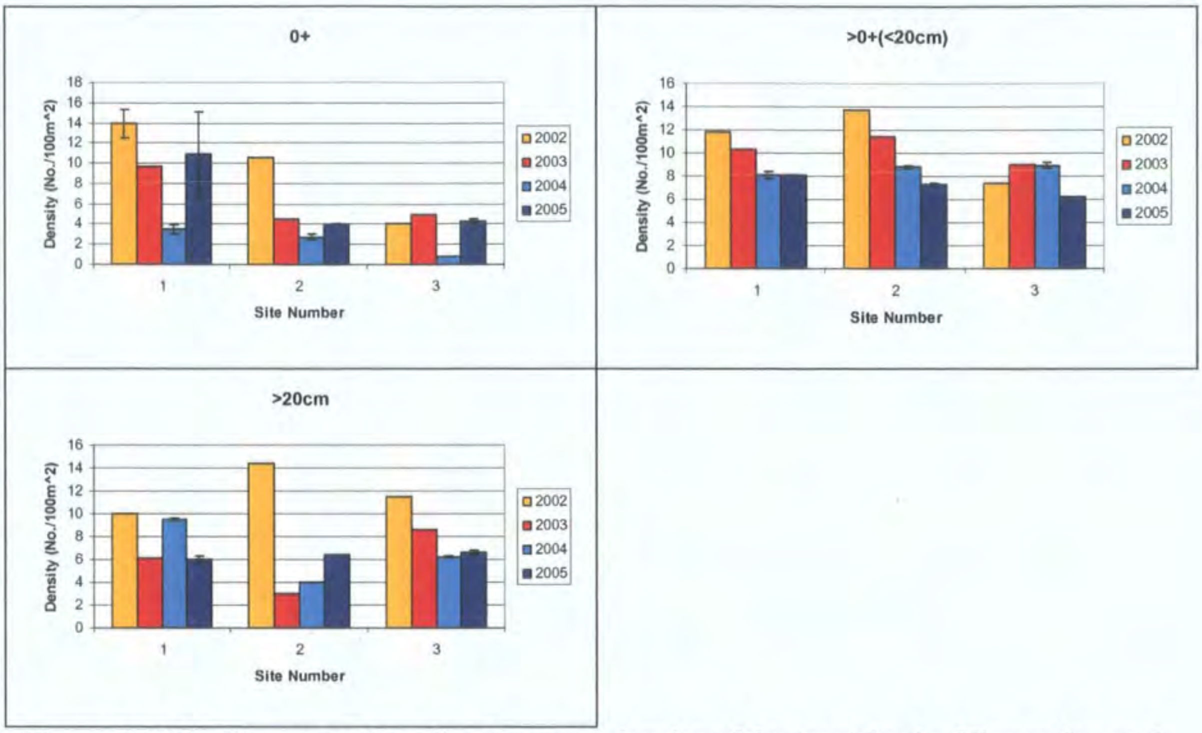


Figure 4.4.24. Figure to show the observed density of 0+ trout in the Hipper for each age/size class for all the sampling seasons. Error bars showing the Carle-Strub variance.

When the HUI values for the phase I and phase II HABSCORE analysis are analysed, it can be seen that there is a reduced 0+ trout HUI value for each of the sites (Table 4.4.33). Table 4.4.33 also shows that for two sites there is a lower >0+ (<20cm) trout HUI in the phase II data compared to phase I. However for H2, the >0+ (<20cm) trout HUI is higher for the phase II data, and the phase II data is classified as significantly better than that expected under pristine conditions whilst the phase I data is not. For the phase I data, each of the sites is significantly better than that expected under pristine conditions (for >20cm trout), and at H1, the HUI value is higher for the phase II data and so the trout density remains significantly higher than that expected under pristine conditions. However, for H2 and H3 sites the HUI value for the phase II data is lower than for the phase I data, with H3 no longer significantly better than expected under pristine conditions. There appears to be no clear variation in HUI in terms of differentiating between phase I and phase II conditions on the Hipper.

	0+ Trout		>0+ (<20cm) trout		>20cm trout	
	Pre-change	Post-change	Pre-change	Post-change	Pre-change	Post-change
H1	2.51	1.12	5.16	2.54	**4.53**	**7.27**
H2	1.23	0.7	5.41	**6.94**	**7.14**	**5.4**
H3	0.75	0.26	2.55	1.41	**8.01**	3.05

Table 4.4.33. Table to show the HUI for the Hipper, using both the pre-and post-change data.

Table 4.4.34 shows that there is no clear pattern in growth rates on the Rivelin. What is interesting is that the 2003-2004 growth period, which is statistically significantly slower on the Rivelin and Loxley, appears not be any different on the Hipper. This is the opposite of what one would expect, as it would be thought that the unregulated river may be impacted upon more greatly by low flows than regulated streams with a constant discharge. The large amount of variation, however is to be expected from an unregulated stream subject to greater hydrological variation than a regulated stream.

H1	**1998**	**1999**	**2000**	**2001**	**2002**	**2003**	**2004**
1	120.8 (1)	98.7 (9)	94.7 (16)	89.2 (15)	92.9 (21)	82.7 (22)	85.0 (7)
2	115.1 (1)	100.7 (9)	96.6 (16)	96.8 (12)	110.1 (12)	108.1 (8)	
3	107.4 (1)	78.3 (9)	60.7 (10)	76.6 (6)	104.9 (3)		
4	62.0 (1)	45.7 (3)	61.1 (2)	63.1 (1)			
H2	**1998**	**1999**	**2000**	**2001**	**2002**	**2003**	**2004**
1		86.3 (1)	82.1 (1)	85.3 (15)	85.4 (26)	85.9 (13)	79.8 (6)
2		90.5 (1)	76.7 (1)	99.9 (15)	102.1 (16)	113.2 (4)	
3		86.3 (1)	59.7 (1)	106.8 (6)	83.9 (6)		
4		44.3 (1)		92.4 (3)			
H3	**1998**	**1999**	**2000**	**2001**	**2002**	**2003**	**2004**
1		115.2 (9)	103.3 (16)	102.9 (15)	89.1 (21)	85.2 (22)	61.9 (1)
2		124.1 (9)	105.7 (16)	108.4 (12)	127.1 (8)	109.3 (9)	
3		88.9 (9)	96.1 (8)	101.0 (7)	67.8 (2)		
4		49.6 (1)		48.9 (3)			

Table 4.4.34. Table to show the percentage growth rate for each cohort at the each site on the Hipper.

Bullheads

Figure 4.4.25 shows that a large Bullhead population is maintained throughout the study periods for each of the study sites. The 0+ population generally appears to make up a small proportion of the population, but its presence remains constant at each site.

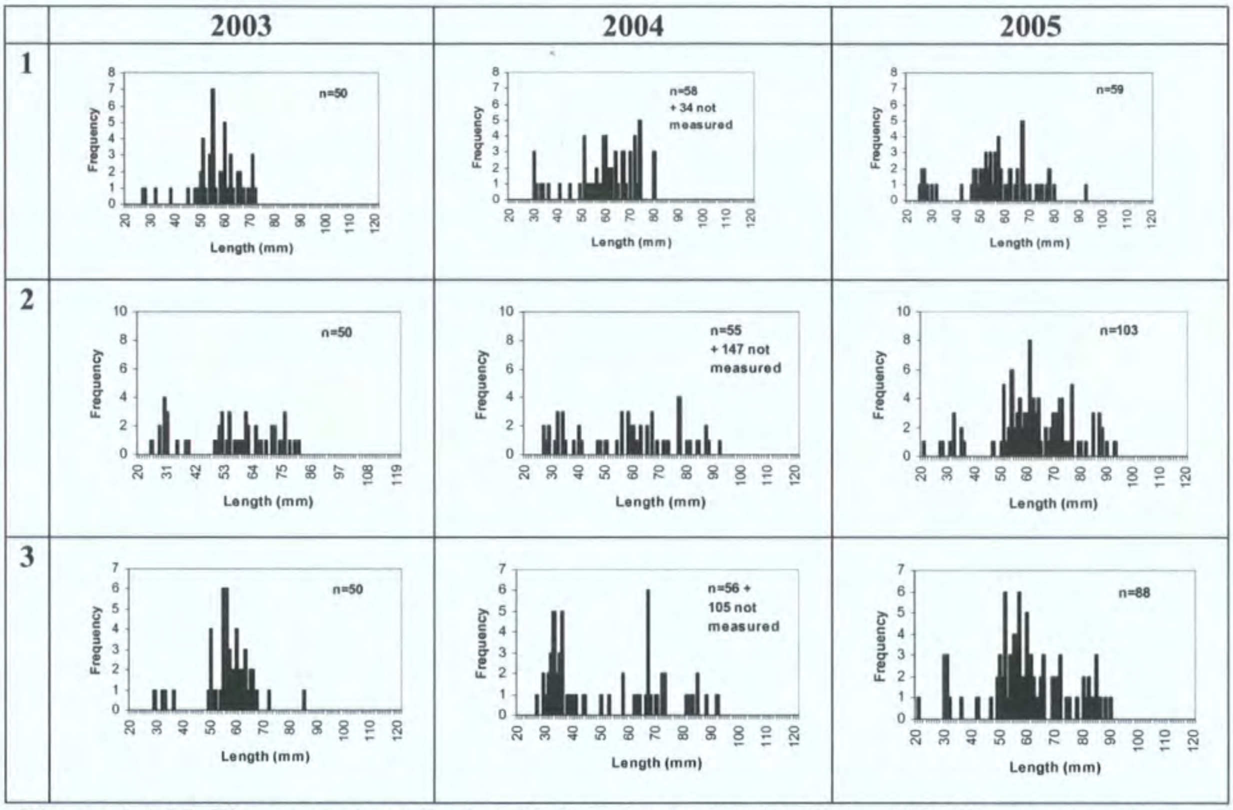


Figure 4.4.25. Figure to show the length frequency distributions of Bullheads at each of the sampling sites on the Hipper for 2003, 2004 and 2005.

4.4.2.3. Summary

This section has shown that the growth period of 2003-2004 is of key importance to the dynamics of both the Rivelin and Loxley. In this period, the Rivelin suffered both poor 0+ trout densities, and a slow growth rate, whilst the Loxley merely displayed a slower growth rate for trout up to one year old. This indicates that the Loxley was affected by this dry period, although not to the extent of the Rivelin. This could be due to the fact that the lower compensation flows present in the Rivelin are less of a buffer to the removal to the removal of overtopping and tributary inputs than the larger compensation flows in the Loxley. It also highlights the importance of using trout growth rates in restoration studies such as this one. This period of reduced rainfall appeared to have no impact on the densities or growth rate of trout in the Hipper.

The importance of calculating growth rates is again highlighted when older trout within the Rivelin do appear to be growing faster in the 2004-2005 growth period than for some of the pre-change growth periods. No change was seen in the Loxley growth rates. When a faster growth rate was seen in the Loxley under pre-change conditions it was attributed to the higher density of invertebrates in the Loxley than the Rivelin. If the growth rates of the older Rivelin trout have increased due to the increase in compensation flow the faster growth rate in the Loxley could be due to its greater compensation flow.

In summary, the fish populations of the Loxley appear to have changed little, but the growth rate of older brown trout may have increased on the Rivelin after the change in compensation flows. The populations of Bullheads on each of the rivers remain unchanged. Also, the alteration of the compensation flows did not negatively impact upon the habitat or abundance of brook lamprey in the Rivelin.

4.5. Discussion and conclusions

It was seen in the literature review that Milner *et al.* (2003) proposed two major controls on brown trout abundance:

1) density-dependent feedback mechanisms, such as territorial competition or limited food availability; and
2) density independent processes (such as climate) which act to determine abundance and, because they can have large effects on survival, may obscure the underlying density dependent process.

The density-independent processes outlined in (2) will serve to make it harder to detect the impacts of both the existing compensation flow regimes and altering the compensation flow regimes. What was argued at the end of the literature review was that, as interesting as the debate on density dependence is from a theoretical fish stock standpoint, in studies such as this one, the key information required by the decision makers is not whether the trout populations are density dependent, but whether they are habitat limited.

So perhaps, in structuring this discussion, it is possible to alter the two major controls slightly, by assuming that 1) applies to habitat limitation; and 2) are the factors stopping habitat limitation from occurring. An understanding of the controls on the trout population is vital for three reasons:

1. to help in setting the baseline from which change can be judged;
2. to assess whether discharge is controlling the fisheries populations on the Rivelin, Loxley and Hipper; and
3. to aid in the interpretation of the modelling results, as if habitat is not limiting for a given age/size class of brown trout, the importance of the modelling output for this age/size class is diminished.

4.5.1. Periods of no augmentation

This is the period when the rivers will be at or nearest to the compensation flow discharges. Hence, if the populations within the rivers are habitat limited, this is the period in which you may expect to observe some changes.

The manifestation of the 2003-2004 dry period in the fisheries populations is interesting as the Rivelin had very poor 0+ densities in 2004 whilst the Loxley did not. It appears that the potential decrease in habitat due to a decrease in augmentation led to a decrease in 0+ trout densities. This highlights the dependence on flow augmentation in the Rivelin as, due to the smaller compensation flow, the ecosystem is more reliant on those overtopping events. The Loxley, with its larger discharge can be thought of as being buffered and so was seemingly protected from the impact of the 2003-2004 low rainfall period. Cowx *et al.*, (1984) showed that the 1976 drought caused a failure of the 1976 year class of salmon but not trout on the unregulated Afon Dulas, whereas Cowx *et al.* (1981) reported no failure of the 1976 year class was found in regulated streams. Solomon and Paterson (1980) did find that a low discharge led to a low 0+ trout population.

However, it was also seen that the growth rates to one year old in the 2003-2004 period were slower than for any other time during the study in both the Rivelin and Loxley. Elliott (1985; 1993b) showed that growth was poor for sea trout classes affected by the summer droughts of 1976, 1983 and 1984. Therefore, it appears that despite the regulation of the Rivelin and Loxley, the low rainfall period of 2003-2004 did impact upon the trout populations within the rivers. What is also interesting is that the unregulated Hipper did not display any reduction in 0+ trout density in 2004, or a slower growth rate in the 2003-2004 period. This is the opposite of that shown in Cowx *et al.* (1981).

In their study, Elliott (1997) suggested that the negative impacts of drought on 0+ trout was because that age group are more sensitive, possibly by reduction of available suitable habitat. During the low rainfall period, there was not only a reduction in the frequency of overtopping but also a reduction in discharge augmentation via tributaries. This could have caused a reduction in suitable 0+ habitat in the Rivelin, which is more reliant upon the augmentation despite its increase in discharge. The Loxley, with its larger discharge could have been buffered against the reduction in augmentation. Furthermore, temperatures during the low flow periods are likely to have been elevated, so therefore, as well as reducing the impact of the lack of augmentation, the higher compensation flow in the Loxley may have reduced the impact of the elevated air temperatures on stream temperature.

What also must be considered is the lack of increased flows during the autumn 2003 which may have not prompted the trout to spawn (e.g. Campbell, 1977). The trout in the Rivelin, which are accustomed to variability and perhaps accustomed to spawning prompts may not have spawned due to the lack of high flows in the autumn, whereas trout in the Loxley, which would be less accustomed to autumn spates may well have spawned anyway. No impact of this reduced augmentation period could be seen on trout older than one year old, again agreeing with the findings of Elliott (1997).

4.5.2. Disentangling density independent processes

It is the aim of this section to see whether it is possible to detect the influences of compensation flow changes during periods where compensation flows are not dominant.

Figures 4.4.2 and 4.4.3 shows that the Rivelin and Loxley contained very similar densities of 0+ trout in 2002 and 2003, with each river having some sites of very high density 0+ trout, but with the majority of sites having 0+ trout densities of $<10/100m^2$. There was no statistically significant difference between the Rivelin and Loxley in terms of 0+ trout density. This density is well below the threshold for density dependence as suggested by Crisp (1993). Table 4.5.1 shows that the HUI values for the Rivelin were predominantly less than one (especially in the lower reaches), with the 0+ populations found at R7 under the pre-change flow regime being significantly less than that expected under pristine conditions. The 0+ HUI values in the Loxley show a tendency to be greater than one, but at no sites were the 0+ populations greater than those expected under pristine conditions. The Loxley also displayed the same downstream decrease in 0+ trout HUI as seen on the Rivelin, apart from at L4, which was near some heavy industry. There was no statistically significant difference between the Rivelin and Loxley in terms of the 0+ trout HUI values

It was hypothesised in Section 4.4 that these lower HUI values indicate that the habitat for 0+ trout in either river may not be limiting. This could be due to a number of factors. The first of which could be a lack of suitable spawning habitat on both the Rivelin and Loxley leading to not enough 0+ trout utilising the habitat available. However, the availability of spawning habitat within each of the rivers has not been quantified in the field study, but it will be addressed in the ecological modelling presented in Chapter 6. Secondly, preferable habitat for 0+ trout could be available outside the reaches fished, hence fewer 0+ trout present in inferior habitat. Thirdly, it is natural for brown trout to spawn after a high flow event in the autumn, which acts not only as a prompt but also serves to clean the gravels (e.g. Campbell, 1977). But with the rivers having constant compensation releases, perhaps this leads to a lower spawning rate than the habitat could provide and hence not producing enough 0+ trout to fill the available habitat. Finally, the 0+ trout could face competition from the high densities of >0+(<20cm) trout for the habitat available (a situation observed by Cattaneo *et al.*, 2002).

Rivelin	**0+ Trout**		**>0+ (<20cm) trout**		**>20cm trout**	
	Pre-change	Post-change	Pre-change	Post-change	Pre-change	Post-change
R1	1.14	2.13	2.11	2.95	1.75	1.29
R2	2.54	2.3	**26.1**	**17.51**	0.91	**3.96**
R3	1.22	1.58	**22.64**	**18.36**	2.65	**3.78**
R4	1.31	**1.21**	**13.26**	**10.38**	**3.79**	**5.45**
R5	0.3	0.4	3.66	4.65	1.18	1.91
R6	0.38	0.24	2.85	1.39	0.97	0.62
R7	**0.09**	**0.14**	**7.41**	3.12	**4.15**	**5.15**
R8	0.84	0.52	**9.93**	3.96	**4.7**	3.1
R9	0.83	0.43	**8.87**	2.08	**5.51**	**5.25**
Loxley	**0+ Trout**		**>0+ (<20cm) trout**		**>20cm trout**	
	Pre-change	Post-change	Pre-change	Post-change	Pre-change	Post-change
L1	2.33	1.18	**34.57**	**13.98**	**5.84**	1.64
L2	2.6	0.38	4.14	5.73	2.96	4.72
L3	2.21	0.69	**7.81**	**15.46**	0.87	2.63
L4	0.93	0.2	5.31	1.36	1.59	1.16
L5	1.48	1.17	2.31	1.34	1.5	1.96
L6	1.23	1.47	**7.43**	3.22	**5.38**	**4.44**
L7	2.15	0.81	**37.75**	**8.94**	2.42	2.35
L8	0.3	1.1	**8.25**	2.28	1.59	0.98
L9	0.99	0.31	4.9	1.56	1.83	1.61

Table 4.5.1. Table to show the HUI values on the Rivelin and Loxley, for both the pre- and post-change data. Statistically significant (95%) values in bold.

It was seen that there was no difference between the Rivelin and Loxley in the growth rate to one year old in either year, or any difference between the two growth rates in 2002 and 2003 for either river. The growth rates to one year old were always less than that taken as standard by the national fisheries laboratory for both the Rivelin and the Loxley. This indicates that despite the general under use of the habitat by these young trout, the amount of food that they have access to may be limited and as a consequence their growth rates are slower than expected. The slow growth rate combined with the low(ish) densities perhaps suggest that the physical habitat (as sampled by HABSCORE) may not be limiting, but the availability of food may be limiting the growth and abundance of the 0+ trout, as observed by Cattaneo *et al.* (2002). Crisp (1995) pointed out the potential (but unquantifiable) loss of nutrients and food to the biota downstream of a reservoir.

The absence of change in post change densities of 0+ brown trout in either the Rivelin or Loxley, suggests that altering the compensation flows has not affected the 0+ trout populations to detectable levels as yet. Furthermore, the way in which the 0+ trout use the available habitat also did not change, with low HUI values still indicating an underuse of habitat. But when the data from the 2003-2004 dry period are considered, it shows that there is a decrease in 0+ density in the Rivelin, and a decrease in growth rates to one year old in both the Rivelin and the Loxley, thus indicating that the 0+ populations may be habitat limited to a certain extent. However, two years pre-change and two years post change is insufficient data to address this question properly.

One possible competitor with the 0+ trout for the available food and/or habitat may be juvenile trout. Tables 4.4.17 and 4.4.19 show that the densities of >0+(<20cm) trout are greater than any other age/size class in both the Rivelin and Loxley. But what is more interesting are the results shown in Table 4.5.1. that show the greater than one HUI values for >0+(<20cm) trout at every sampling site on the Rivelin and Loxley. Numerous sites on each river had >0+(<20cm) trout populations which were significantly better than those expected under pristine conditions. It was hypothesised in the previous section that such high HUI values could indicate that the systems are habitat limited for >0+(<20cm) trout.

What is interesting about such high abundance of >0+(<20cm) trout is that it means that despite the low HUI of the 0+ trout, many are surviving to be juveniles and hence making the habitat limiting for >0+(<20cm) trout. This could mean that the fisheries survey sites have not included the rearing areas for the 0+ trout, and when the trout are older they migrate to the better habitat for the older trout. The HQS scores for 0+ trout on both the Rivelin and Loxley were significantly higher than those for >0+(<20cm) trout so it appears that this is not the case.

The >20cm HQS scores or actual BT densities for the Rivelin and Loxley could not be differentiated statistically in either 2002 or 2003, which is a surprise as it may be thought that the Loxley must have better habitat available for older trout due to the greater discharge. The average HQS is slightly higher for the Loxley than the Rivelin though, but the densities are very similar. This further highlights the importance of the interaction between the discharge and the topography of the river reach, as it is this which will determine the distributions of velocities and depths within the reach. This provides further justification for the use of two-dimensional modelling in studies such as this, as it encompasses a spatial component not seen in one-dimensional models such as Maddock *et al.* (2001); Gibbins and Acornley, (2000) and Gibbins *et al.* (2001).

Section 4.4 showed that the growth rates for older trout in the Loxley are faster than for those in the Rivelin. The ANOVA analysis showed that the two year old fish grew significantly faster in the Loxley than in the Rivelin in 2002/2003; and three year old trout grew significantly faster in the Loxley than the Rivelin in both sampling years. This may be due to two reasons. Firstly the Loxley has a greater discharge (Chapter 2) possibly providing a slightly deeper, and therefore better habitat (this is hinted at with a slightly higher >20cm HQS for the Loxley). Secondly, the Loxley has a greater number of invertebrates / sample than on the Rivelin (Chapter 3). The abundance of food for trout may help make the growth rates faster, as growth rate represents the individuals success in acquiring energy through food (Milner *et al.*, 2003).

Comparison of growth rates of the different ages of trout for the various growth periods (Table 4.5.2) shows that for the Rivelin, the slowest growth rates observed were for one year old trout, with two year old trout growing faster than any of the ages. This is interesting because it shows that despite the high densities of >0+(<20cm) trout the growth rates are still the fastest of any of the age groups. This agrees with the findings of Gee *et al.* (1979), where growth rates appeared unrelated to density. Although, Gibson (1993) generalised that in a single stream type, higher density usually gave lower growth. In the Loxley, it can be seen that the growth rates are faster for older trout than younger trout (with one exception). This perhaps indicates that the higher discharge in the Loxley provides a greater habitat, so despite a similar density the quality of habitat in terms of food availability may be better.

Rivelin	2001-2002	2002-2003	2003-2004	2004-2005
1 vs 2	diff (2>1)	no diff	diff (2>1)	diff (2>1)
1 vs 3	diff (1>3)	diff (1> 3)	no diff	no diff
2 vs 3	diff (2>3)	no diff	no diff	diff (2>3)
Loxley	**2001-2002**	**2002-2003**	**2003-2004**	**2004-2005**
1 vs 2	diff (1<2)	diff (1<2)	diff (1<2)	diff (1<2)
1 vs 3	no diff	no diff	diff (1<3)	no diff
2 vs 3	diff (2>3)	no diff	no diff	no diff

Table 4.5.2. Table to show the output from the ANOVA analysis comparing the growth rates of the different ages for the Rivelin and Loxley.

4.5.3. The impact of altering the compensation flows

As was explained in Section 4.2, there has never been a paired catchment study conducted into the role of compensation flows such as this one conducted before with most of the studies relating to reservoir construction or heavily varying HEP systems. Hence the results presented in this chapter are difficult to relate back to existing literature.

The picture painted by the post-change data was obscured somewhat by the low rainfall period of 2003-2004. When the final years sampling was considered it can be seen that there were no changes to the brown trout densities in both the Rivelin and Loxley. This is consistent with Harris *et al.* (1991) who showed that a four year period of a five fold increase in the minimum flow did not alter the brown trout populations in a small stream in the U.S.A.

When the growth rates are considered, the 2004-2005 period provided similar growth rates to the pre-change period in the Loxley, and faster growth rates in the Rivelin for two and three year old trout. It is thought perhaps that the greater discharge in the Rivelin, despite the densities of trout remaining similar, could have improved habitat conditions for the older trout which led to the improved growth rates, perhaps making the pools slightly deeper (e.g. Elso and Giller, 2001). This is interesting when compared back with the slightly faster growth rates of the Loxley older trout in the pre-change populations, a phenomenon which could have been attributed to the greater invertebrate density on the Loxley, but perhaps this could now be attributed to the greater discharge in the Loxley.

4.5.4. Limitations

Crisp *et al.* (1983) highlighted the fact that fish can move in and out of study reaches depending on the habitat available. This is a key problem for a study such as this as, the importance of the interaction of reach-scale morphology and flow is well established; any changes in discharge may have different impacts in different areas of the river. As trout are a relatively mobile species, it may be possible for better habitat to be created outside the study reaches and for the population to move there. This may create a situation where there is a seeming decrease in trout density. This situation may act in reverse, as the study sites may be the areas of trout refugia during low flows; and a reduction in discharge could then lead to an apparent increase in density. In order to try and counteract this problem, nine sites were sampled along the length of the river.

It must be stated that due to the nature of the project, only two years pre change and two years post-change fisheries surveys are able to be included. Section 4.1.3 showed the natural variability within the systems, and highlighted the difficulty in being sure of changes/influences on the brown trout populations. Furthermore, for the first of the post-change surveys, the 0+ trout sampled were spawned under the pre-change conditions. Added to this the low rainfall nature of 2003-2004, the post-change impacts have largely been estimated from one years data, and hence conclusions must be made carefully. This is a limitation of this fisheries chapter.

Crisp *et al.*, (1983) found a control catchment useful in distinguishing between natural events and regulation impacts in the Cow Green system, but financial and time reasons meant that, only three sites were surveyed on the Hipper. This caused two main problems;

1) the lack of replication with the sampling structure on the Rivelin and Loxley meant that the Hipper could not be included in statistical analyses; and

2) the large amount of natural variation inherent in the Hipper meant that it was difficult see with any certainty patterns of variation displayed by only three sites.

Furthermore, the Hipper remained un-gauged, and it was therefore difficult to associate the biotic and abiotic factors occurring in the river.

4.5.5. Final conclusions

Fisheries surveys were conducted on the Rivelin, Loxley and Hipper over a four year period; encompassing two years pre change and two years post change data. The 2003-2004 period was highlighted as one where potentially habitat limitation could occur, and it was seen that there were reductions in 0+ trout density in the Rivelin and reductions in growth rates in both rivers. This perhaps indicates that the lack of augmentation in this period reduced the habitat available and led to these changes – a form of habitat limitation for 0+ trout in these rivers. These results illustrate the influence of site characteristics on fisheries populations (Figure 1.2).

These analyses also showed that >0+(<20cm) trout appeared to be habitat limited. This was evidenced by the very high HUI scores in both the Rivelin and Loxley. The older trout grew more quickly in the Loxley than in the Rivelin. Changing the compensation flows appeared to lead to an increase in growth rate for older trout in the Rivelin, hence enhancing the theory that >0+(<20cm) trout may also be habitat limited within these systems.

Chapter 5

Hydrodynamic modelling of upland Millstone Grit streams

5.1. Introduction

The previous chapters have shown that the impact of compensation flow releases on the ecology of the system can be detected in both in the pre and post-change scenarios. However, in both Chapters 3 and 4, it was difficult to link changes in ecology to changes in discharge, but there are links between the hydrodynamics and ecology in these systems (Figure 1.2). We know that the discharges of the rivers have changed (Chapter 2), and that the ecology of the rivers has changed at least to some extent. However, changes in flow might not cause organism response. As Figure 5.1.1 shows, it is the patterns of usable depth and velocity that matter and these will depend on the magnitude of discharge change in relation to reach slope, reach geometry and bed roughness. Simulating organism response to changing environmental conditions is not easy, and this is addressed in more detail in Chapter 6. Of particular note here is the fact that such simulations will only predict organism response in the special case of habitat limitation and if the model contains all dimensions and scales (in space and time) of possible organism response. In this chapter, the focus is on usable depth and velocity and hence is strictly an assessment of whether or not the degree of habitat limitation (velocity and depth) changes due to simulated flow changes. This can be investigated using a hydraulic model. There are two further reasons why hydraulic habitat modelling may be of interest. First, a problem which faces many decision makers in the real world is that there is not always the money for the kind of extensive invertebrate and fisheries sampling (over and above the routine sampling conducted by the Environment Agency) conducted in this study. Habitat modelling may then be used in order to assess whether the compensation flows need to be altered in the special case that habitat flow is limited. Second, if changing the compensation flows leads to a net increase in ecological value or the compensation flows need to be changed for other reasons (perhaps water resources) then modelling could be used to simulate the potential impacts of the different flow scenarios. Modelling can allow the investigation of impacts on habitat availability for a number of species and different life stages.

There are a number of approaches which can be taken when wanting to model the hydraulics of rivers. The first is a one-dimensional approach, which is based on the St. Venant equations for width and depth averaged flow. However, such approaches suffer from the lack of spatial variability inherent in their output. It was seen in Chapters 3 and 4 that both spatial flow variability and the interaction of reach scale geomorphology were very important to the ecology of the streams. Such features can be incorporated into higher dimensional models (2D and 3D).

The approach taken in this chapter is to use a two-dimensional model to simulate the distributions of velocities and widths in the study rivers under a variety of discharges. It was adopted for this study in order to incorporate the spatial heterogeneity of the streams (the importance of which was seen in the invertebrates) and the impact of reach scale morphology interaction with discharge (as seen in the fisheries section).

The primary purpose of this chapter is to investigate the capability of two-dimensional hydrodynamic modelling in upland Millstone Grit catchments such as the study sites considered herein. The secondary purpose is to assess the potential impacts of altering the compensation flows on the hydrodynamics of the rivers in terms of the primary variables, velocity, depth and wetted area. As such there are three aims which will be addressed in this chapter: i) to assess whether a 2-D finite element model accurately predicts the hydrodynamics of upland Millstone Grit streams; ii) to explore the primary controls on predicted hydrodynamics, including mesh resolution and model parameters; and iii) to assess the primary impacts of altering the compensation flows.

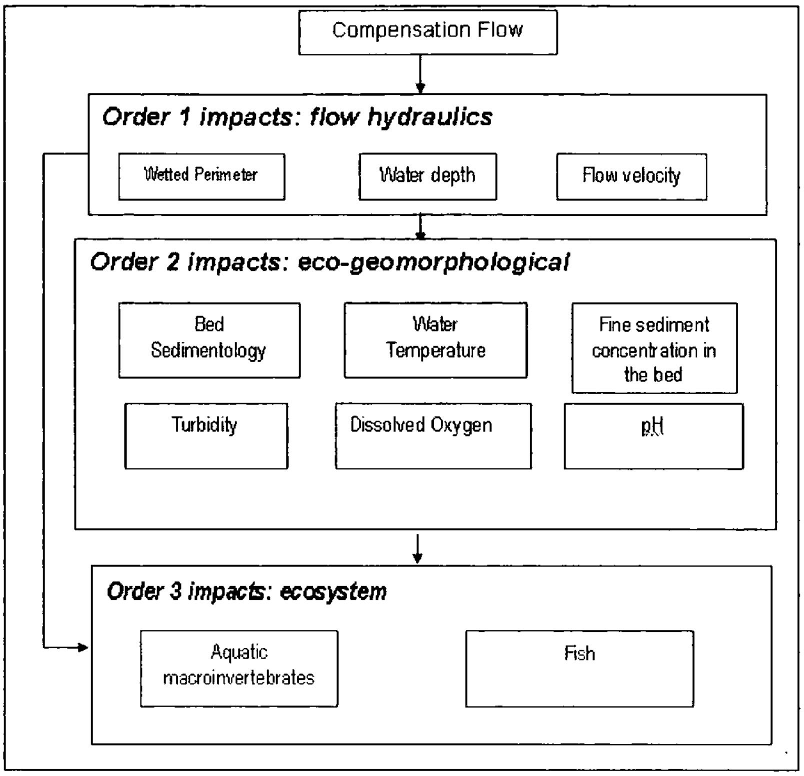


Figure 5.1.1. Figure to show the potential impacts of compensation flow releases (adapted from Petts, 1984).

This chapter begins with a literature survey, which is structured around the basic steps in the modelling process. The methods used in this chapter are then explained and justified. The first results section constitutes a discussion of the calibration of the model and includes an assessment of model sensitivity to parameterisation and mesh density. The next results section addresses the 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 change in compensation flows). This is followed by a discussion where the important results are highlighted and linked back to the literature cited in Section 5.2. A final section provides conclusions.

5.2. Literature review

There are a number of steps to any modelling study. Figure 5.2.1 outlines the key processes which can be followed in any modelling process. It is the purpose of this review to review the hydraulic modelling literature with respect to each of these steps. There are a potentially a number of hydraulic models which could be used in a situation such as this one, and each of these will be addressed at various points in this review.

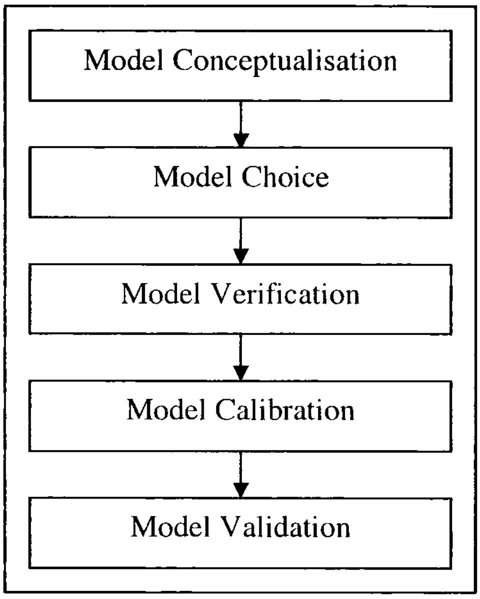


Figure 5.2.1. Figure to show the major stages in the modelling process. Adapted from Lane (2003).

5.2.1. Model conceptualisation

Figure 5.1.1 shows that the main controls on habitat for macroinvertebrates and fish are wetted area, depth and velocities. These parameters are therefore what are of interest to model. There are a number of factors which will influence the distributions of depths and velocities: primarily slope, channel geometry and bed roughness. Given the target of modelling, the next step is to identify the means to get to it. This leads onto the next step in Figure 5.2.1, in that we must choose the right model for the task. The next section will outline various modelling options, before discussing their advantages and disadvantages.

5.2.2. Model choice

There a number of methods which can be used to model the hydraulics of rivers, and it is the purpose of this section to outline each of these methods, and highlight their advantages and disadvantages.

5.2.2.1. Non-hydraulically correct approaches

Traditionally, instream flow models or methodologies have been simplistic. The simplest, such as the 'Montana Method' (Tennant, 1976) prescribe environmental flows as regimes based upon the average daily discharge or the mean annual flow (MAF). In general, 10% of MAF was recommended as a minimum instantaneous flow to enable most aquatic organisms to survive; 30% MAF was recommended to sustain good habitat; 60%-100% MAF provides excellent habitat; and 200% MAF was recommended for flushing flows. Such approaches have obvious shortcomings, the most serious being the elimination of ecologically important flow extremes and a lack of attention to flow timing (Richter *et al.*, 1997).

The aim of a study by Lamouroux and Capra (2002) and a similar study by Lamouroux and Souchon (2002) was to determine whether the habitat predictions of conventional instream habitat models (PHABSIM) could be predicted from average characteristics of these reaches, in a wide variety of stream reaches and at different discharge rates. Using non-linear mixed effect models, they demonstrated that the outputs of instream habitat models (habitat values for three trout life stages) are predictable from average characteristics of reach (discharge, depth, width and bed particle size). These variables explained up to 89% of variance for adult brown trout and 86% of habitat value variability.

5.2.2.2. One-dimensional approaches

One-dimensional hydraulic models are generally used as part of the Physical HABitat SIMulation (PHABSIM) software, The starting point of 1D approaches is the treatment of the basic flow equations (Lane and Ferguson, 2005). Consider a reach of river that can be described by a cross-section of area A, with an average velocity in that section of v. Thus, the discharge (Q) through that section is $Q=vA$. Both v and A can vary as a function of downstream distance (x). If the flow is steady ($dQ/dx=0$), mass conservation gives:

$$0=-\frac{d(vA)}{dx}+i=-v\frac{dA}{dx}-A\frac{dv}{dx}+i \qquad (5.2.1)$$

where i is the input from (or if negative, the loss to) storage per unit distance downstream. If the flow is unsteady, equation (5.2.1) becomes:

$$\frac{\partial A}{\partial t}=-v\frac{\partial A}{\partial x}-A\frac{\partial v}{\partial x}+i \qquad (5.2.2)$$

The same analysis applies to momentum conservation (Lane and Ferguson, 2005). In principle, for an incompressible fluid, they state that the rate of change of momentum through time at a point will be a function of the spatial change of momentum plus sources (the driving forces):

$$\frac{\partial(Av)}{\partial t}=-\frac{\partial(Av^2)}{\partial x}+sources \qquad (5.2.3)$$

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

$$\frac{\partial(Av)}{\partial t}+\frac{\partial(Av^2)}{\partial x}=-Ag\frac{\partial h}{\partial x}+gA(S_o-S_f) \qquad (5.4.4)$$

where h is mean flow depth, S_o is the bed slope of the channel (defining the potential energy term) and S_f is the friction slope (defining the friction term). If these equations are derived from the full 3D form of the Navier-Stokes equations, then it becomes clear that he friction term in equation (5.4.4) is not just representing the effects of boundary resistance, but a whole set of other processes (Lane and Ferguson, 2005). These result in the extraction of momentum from the mean downstream flow (i.e. Av) and its transformation into flow components that are variable in the cross-stream and vertical directions (i.e. dispersion processes associated with secondary circulation) and through time (i.e. turbulence processes) (Lane and Ferguson, 2005). The friction term is commonly defined under the assumption that the flow is locally uniform (Lane and Ferguson, 2005). This allows uniform flow equations to be used, such as the Darcy-Weisbach equation:

$$S_f = \frac{v^2 f}{8gR} \qquad (5.2.5)$$

where R is the hydraulic radius and f is a 'friction parameter'. However, f is actually representing more than just friction such that, in most situations, its relationship to the actual surface roughness will be uncertain (Lane and Ferguson, 2005). The above equations are commonly used in relation to flood risk modelling and there are a number of commercially available packages which do this (MIKE-11, ISIS, HEC-RAS).

As outlined above, a one-dimensional approach looks at the stream as a number of cross-sections (Ghanem *et al.*, 1996). Each cross-section is described by cross-section average variables. Thus, for each cross-section there exits one representative stage and one representative cross-section average velocity (Ghanem *et al.*, 1996). Hydraulic principles are applied to relate these cross-sections to each other and solve for the cross-section average variables. The hydraulic modelling in PHABSIM is performed in two stages. Firstly a stage-discharge relationship is obtained via one of three ways; the use of a stage-discharge regression model (IFG-4); a uniform flow model (MANSQ); or a gradually varied flow model (WSP) (Milhous *et al.*, 1989). Once a stage-discharge relationship has been obtained, the programme IFG-4 is used to estimate the velocity distribution across each cross-section (Ghanem *et al.*, 1996). In PHABSIM, the cross-section is divided into cells and verticals and the velocity at each vertical is determined from measured velocity values at different discharges at that vertical (Ghanem *et al.*, 1996). The vertical is not tied in any way to the other verticals around it through hydrodynamic principles. Therefore, this approach to velocity determination should be more correctly termed a zero-dimensional approach, which relies solely on interpolation from measured values and not on physical principles (Ghanem *et al.*, 1996).

Leclerc *et al.* (1995) note that a PHABSIM type treatment will not produce reliable results for areas of river less than 10m^2. Hence, the limited data make habitat description difficult at a scale relevant to fish (Leclerc *et al.*, 1995). Moreover, PHABSIM models are sometimes difficult to calibrate (Osbourne *et al.*, 1988; Ghanem *et al.*, 1996) and they cannot be easily applied to areas that are frequently uncovered during low flow periods (Leclerc *et al.*, 1995).

A further limitation of one-dimensional models is the width-averaging of the flow calculations (Lane and Ferguson, 2005). This could be perfectly adequate for canals or channelised rivers with trapezoidal cross-sections, but means that 1-D models cannot represent the bar-pool-riffle topography commonly found in natural rivers nor the associated local lateral variation in flow (Lane and Ferguson, 2005). This is a greater issue in these upland Millstone Grit streams which are highly heterogeneous (as explored in Chapters 3 and 4), and therefore spatial variation in flow may also be of importance.

Furthermore, one dimensional models do not permit extrapolation outside of the measured discharge interval (Leclerc, *et al.*, 1995). This is obviously of interest in a study such as this one, as the discharges which need to be modelled are often outside the existing discharges (i.e. both rivers are rarely at baseflow, and the Loxley discharge was lowered below any previous recorded discharge). As such a model, with which, discharges from outside the measured ranges could be simulated would be preferable.

5.2.2.3. Two-dimensional approach

Realising the limitations of the one-dimensional approach, Ghanem *et al.* (1996) developed a two-dimensional model for simulating the flow of water in fish habitats. For the one-dimensional model, the depth and roughness data were used to calculate cell conveyances, which were then used to calculate the velocity distribution across the river (Ghanem *et al.*, 1996). For the two-dimensional model, the simulation was performed by considering only the topographic features of the domain together with the assumed constant roughness height (Ghanem *et al.*, 1996). The one-dimensional and two-dimensional models were then compared, and the authors preferred two-dimensional models on a number of counts: i) they gave a better representation of flow, as they account for the two-dimensional flow features and are able to simulate complex flow patterns such as recirculation and transverse water slopes; and ii) they may require less office work than a one-dimensional model as the topographic measurements could be easily adapted.

The importance of spatial variation of hydraulics and in particular the role of boulders in producing this spatial variability of flow was highlighted by Crowder and Diplas (2000a, 2000b, 2002). Two-dimensional hydraulic model results suggest that at base flow conditions channel complexity in the form of exposed boulders and bedrock outcrops plays a significant role in creating localised flow patterns of potential biological importance (Crowder and Diplas, 2002). Sensitivity analyses show that reducing element sizes in the vicinity of obstructions and banks is crucial in modelling the spatial flow patterns created by meso-scale topographic features (Crowder and Diplas, 2000a). Leclerc *et al.* (1995) reported that 2-D models allow a more spatially detailed description of fish habitat than do traditional methods. This feature is particularly useful for studies on territorial fish such as salmonids, which defend territories that rarely exceed 4m^2 (Grant and Kramer, 1990). Leclerc *et al.* (1995; 1996) found that the advantage of using a 2-D approach is that it recognises that as some habitat becomes less suitable as flow depth and/or velocity locally rises, so other habitats become more suitable. Another advantage of the 2-D models is enhanced accuracy in estimating the physical variables provided by either a better representation of the field data and greater reliability in calibrating Manning's roughness coefficients (Leclerc *et al.*, 1995).

Derivation of the basic equations that govern shallow open channel flow begins with the assumption that a fluid can be considered a continuum (Tritton, 1988), where the details of the molecular structure only appear in constituent equations that recognise the effects of molecular processes on the overall flow characteristics (Lane, 1998). Although Newton's Laws of motion provide a basis for the study of motion of particles or rigid bodies, they need modification for the application to the continuum that represents a fluid (Lane, 1998). The Eulerian transformation of Newton's laws begins by considering a unit of fluid of some arbitary volume (Lane, 1998). Through time, this fluid will move through space and the constitution of the unit will change.

Using these principles it is possible to obtain two key equations (White, 1974) for a single fluid unit from Newton's laws of motion (i) the law of conservation of mass for an incompressible fluid in Eulerian form (Lane, 1998):

$$\frac{\partial u}{\partial x}+\frac{\partial v}{\partial y}+\frac{\partial w}{\partial z}=0 \qquad (5.2.6)$$

where, u, v and w are the components of velocity in the x, y (planform) and z vertical directions respectively; and (ii) the Navier-Stokes momentum equations for an incompressible fluid:

$$\frac{\partial}{\partial t}(\rho u)+\frac{\partial}{\partial x}(\rho u^2)+\frac{\partial}{\partial y}(\rho uv)+\frac{\partial}{\partial z}(\rho uw)-2\rho u\xi\sin\Phi+\frac{\partial p}{\partial x}-\frac{\partial \tau_{xx}}{\partial x}-\frac{\partial \tau_{xy}}{\partial y}-\frac{\partial \tau_{xz}}{\partial z} \qquad (5.2.7a)$$

$$\frac{\partial}{\partial t}(\rho v)+\frac{\partial}{\partial x}(\rho uv)+\frac{\partial}{\partial y}(\rho v^2)+\frac{\partial}{\partial z}(\rho vw)-2\rho v\xi\sin\Phi+\frac{\partial p}{\partial y}-\frac{\partial \tau_{xy}}{\partial x}-\frac{\partial \tau_{yy}}{\partial y}-\frac{\partial \tau_{yz}}{\partial z} \qquad (5.2.7b)$$

$$\frac{\partial}{\partial t}(\rho w)+\frac{\partial}{\partial x}(\rho uw)+\frac{\partial}{\partial y}(\rho vw)+\frac{\partial}{\partial z}(\rho w^2)-\rho g+\frac{\partial p}{\partial z}-\frac{\partial \tau_{xz}}{\partial x}-\frac{\partial \tau_{yz}}{\partial y}-\frac{\partial \tau_{zz}}{\partial z} \qquad (5.2.7c)$$

where ρ is the density of water, ζ is the angular rotation of the earth, Φ is latitude, p is pressure, g is the acceleration due to gravity,

$$\frac{\tau_{ij}}{\rho} = \mu\left(\frac{\delta v_j}{\delta x_i} + \frac{\delta v_i}{\delta x_j}\right) \qquad (5.2.8)$$

and μ is the coefficient of viscosity for a Newtonian fluid (the molecular viscosity). For shallow open flow these equations need some modifications and additions (Lane, 1998). First, in the case of shallow water, it is generally assumed that the vertical scale is much smaller than the horizontal scale and the boundary layer extends throughout the water depth. This allows Equation 2c to be simplified to the hydrostatic pressure distribution [see Rodi *et al.* (1981) for justification],

$$\frac{\partial p}{\partial z} = -\rho g \qquad (5.2.9)$$

Assuming density is constant with depth, and integrating over depth, Equation (3) becomes (Lane, 1998):

$$p = \rho g\left(h + z_b\right) \qquad (5.2.10)$$

where h is the water depth and z_b is the bottom elevation. This allows the pressure gradients in Equations (2a) and (2b) to be redefined, and assuming no horizontal elevation in density, Equations (2a) and (2b) become (Lane, 1998):

$$\frac{\partial}{\partial t}(u) + \frac{\partial}{\partial x}\left(u^2\right) + \frac{\partial}{\partial y}(uv) + \frac{\partial}{\partial z}(uw) - 2u\Omega\sin\Phi + g\frac{\partial h}{\partial x} + g\frac{\partial z_b}{\partial x} + \frac{1}{\rho}\frac{\partial \tau_{xx}}{\partial x} - \frac{1}{\rho}\frac{\partial \tau_{xy}}{\partial y} - \frac{1}{\rho}\frac{\partial \tau_{xz}}{\partial z} = 0 \qquad (5.2.11a)$$

$$\frac{\partial}{\partial t}(v) + \frac{\partial}{\partial x}(uv) + \frac{\partial}{\partial y}\left(v^2\right) + \frac{\partial}{\partial z}(vw) - 2v\Omega\sin\Phi + g\frac{\partial h}{\partial y} + g\frac{\partial z_b}{\partial y} + \frac{1}{\rho}\frac{\partial \tau_{xy}}{\partial x} - \frac{1}{\rho}\frac{\partial \tau_{yy}}{\partial y} - \frac{1}{\rho}\frac{\partial \tau_{yz}}{\partial z} = 0 \qquad (5.2.11b)$$

Secondly, it is necessary to recognise that there will be special conditions at both the (free) water surface and the bottom since water particles will be unable to cross either boundary (Lane, 1998). At the bottom, assuming that it is solid, the normal velocity components must disappear, that is:

$$u\frac{\partial z_b}{\partial x} + v\frac{\partial z_b}{dy} - w = 0 \qquad (5.2.12)$$

where z_b is the bottom elevation.

It is possible to show that shallow open flow satisfies, at every point in space, Equations 5.2.6. and 5.2.11, modified in the vicinity of boundaries by equations 5.2.10-5.2.12 (Lane, 1998). However, direct application of these equations results in a computational problem (Lane, 1998). To solve the equations with sufficient accuracy, a three dimensional computation grid is required, which must be large enough to cover the area of interest but with spacing smaller than the smallest turbulent motion (Lane, 1998). The computations must be unsteady, utilising a time step smaller than that associated with the smallest eddys. In computational fluid dynamics, the most common solution to this problem is a semi-empirical analysis of the effects of turbulent motions on the mean flow properties, which involved a technique called Reynolds (1895) averaging.

Two dimensional treatments begin by depth averaging equations (5.2.6) and (5.2.11), with the aim of calculating the horizontal distribution of depth average velocity components □ and $\bar{V}$ (Rodi *et al.*, 1981), where capital letters indicate depth-averaged quantities and overbars indicate time-averaged quantities. These are defined by (Lane, 1998):

$$V_i = \frac{1}{h - z_b}\int_{z_b}^{h} v_i \partial z \qquad (5.2.13)$$

Applying Equation (5.2.13) to Equation (5.2.6), the depth-averaged form of the law of conservation of mass becomes:

$$\frac{\partial h}{\partial t} + \frac{\partial}{\partial x}\left[(h - z_b)\bar{U}\right] + \frac{\partial}{\partial y}\left[(h - z_b)\bar{V}\right] = 0 \qquad (5.2.14)$$

The depth-averaged Navier-Stokes momentum equations become, for each of the x and y components, and ignoring the Coriolis term (Lane, 1998):

x-component momentum equation

$$\frac{\partial}{\partial t}\left(\overline{U}\right)+\frac{\partial}{\partial x}\left(\overline{U}^{2}\right)+\frac{\partial}{\partial y}\left(\overline{U}\overline{V}\right)=-g\frac{\partial h}{\partial x}-g\frac{\partial z_b}{\partial x}-\frac{1}{\rho}\frac{\partial}{\partial x}\left(\overline{\tau}_{xx}\right)+\frac{1}{\rho\left(h-z_b\right)}\frac{\partial}{\partial y}\left(\overline{\tau}_{xy}\right)-\frac{\tau_{bx}}{\rho\left(h-z_b\right)}+ \\ \frac{1}{\rho}\frac{\partial}{\partial x}\int_{z_o}^{h}\rho\left(\overline{u}-\overline{U}\right)^{2}\partial z+\frac{1}{\rho\left(h-z_b\right)}\frac{\partial}{\partial y}\int_{z_o}^{h}\rho\left(\overline{u}-\overline{U}\right)\left(\overline{v}-\overline{V}\right)\partial z \quad (5.2.15)$$

y-component momentum equation

$$\frac{\partial}{\partial t}\left(\overline{V}\right)+\frac{\partial}{\partial x}\left(\overline{U}\overline{V}\right)+\frac{\partial}{\partial y}\left(\overline{V}^{2}\right)=-g\frac{\partial h}{\partial y}-g\frac{\partial z_b}{\partial y}-\frac{1}{\rho}\frac{\partial}{\partial x}\left(\overline{\tau}_{xy}\right)+\frac{1}{\rho\left(h-z_b\right)}\frac{\partial}{\partial y}\left(\overline{\tau}_{yy}\right)-\frac{\tau_{by}}{\rho\left(h-z_b\right)}+ \\ \frac{1}{\rho\left(h-z_b\right)}\frac{\partial}{\partial x}\int_{z_o}^{h}\rho\left(\overline{u}-\overline{U}\right)\left(\overline{v}-\overline{V}\right)\partial z+\frac{1}{\rho\left(h-z_b\right)}\frac{\partial}{\partial y}\int_{z_o}^{h}\rho\left(\overline{v}-\overline{V}\right)^{2}\partial z \quad (5.2.16)$$

The dispersion terms are a product of vertical non-uniformity in the velocity field (which is inevitable because the velocities tend to fall to zero at the channel bottom) and represent deviations from the depth-averaged velocities within a vertical profile (Lane and Ferguson, 2005). Rodi (1980) and Rodi *et al.* (1981) note that the physical meaning of the dispersion terms is similar to the turbulent stress terms, in that both represent gradients of transport of momentum. From measurements of depth averaged quantities it is usually not possible to distinguish between the turbulent and dispersion contributions to the momentum transport (Lane and Ferguson, 2005). As with the Reynolds shear stresses, no additional equations arise during the depth averaging process and determination of those terms requires either knowledge of the secondary flow field or a model of their effects on mean flow properties (Lane, 1998).

Research into the performance of 2D numerical codes has emphasised the need for careful choice of the mesh discretisation method. There are three main types of discretisation: finite difference; finite volume and finite element.

Finite difference schemes are based upon the solution of the mass and momentum equations in differential form, which are approximated by a system of linear algrebraic equations where the values of variables at the grid nodes are the unknowns (Ferziger and

Peric, 1999). Each term of the partial differential equation at a particular node is replaced by a finite difference approximation. The differences may be evaluated in a backward, central or forward manner (Lane and Ferguson, 2005).

Finite volume schemes uses the integral form of the conservation equations rather than the differential form (Lane and Ferguson, 2005). The domain is divided into control volumes with a computational node at each volume centre, and integrals that apply to both the surface and the volume of the control. In a similar way to finite differences, an algebraic equation has to be determined for each control volume which requires approximation of the surface and volume integrals using quadrature formula (Ferzinger and Peric, 1999). The finite difference and finite volume schemes assume a structured grid.

Unstructured grids have a triangular, rather than quadrilateral, shape which makes them easier to fit to a boundary. This provides considerable flexibility for the study of reach-scale flows (Ferziger and Peric, 1999) and a direct link to finite element methods. Finite element solutions are similar to finite volume methods in that they use volumes, but these are subject to weights that provide continuity across element boundaries. The non-structured aspect of the finite element option is reputed to offer a better adaptability to local flow and riverbed features characterised by higher variable gradients, and consequently more accurate results (Lane and Ferguson, 2005).

Recent studies have highlighted the potential of two-dimensional and three-dimensional habitat models rather than the one-dimensional approach of PHABSIM (Bovee, 1996; Hardy, 1996). A clear understanding and an accurate representation of the hydraulic characteristics of a fish habitat are necessary, not only to model the physical features of the habitat (Ghanem *et al.*, 1996). In natural streams, flow phenomena such as eddies and recirculation are encountered which cannot be resolved using a zero-or one dimensional approach, which assumes a priori a flow direction perpendicular to the cross-section (Ghanem *et al.*, 1996).

Even if given a justifiable treatment of bottom stresses and dispersion terms, and with proper discretisation careful thought need to be given to the acceptability of depth-averaging (Lane and Ferguson, 2005). It depends in the first place on the scale and aims of the study. Lane *et al.* (1999) compared a depth averaged and a fully 3-D solution and found little difference in predictive ability for horizontal velocity components of velocity, as judged by levels of explained variance with respect to check data. However, the depth-averaged predictions were biased in relation to the check data as compared with the 3-D predictions, implying that additional calibration of the depth-averaged model was required to get the same levels of agreement (Lane *et al.*, 1999). The use of a three-dimensional model would provide useful data on near bed flow velocities (especially interesting for macroinvertebrates) but this is offset by the extra computational cost of using a three dimensional scheme.

5.2.3. Model verification

Coupled with the application of 2D codes to open channel flow studies is the need to consider how these codes may be verified and validated with respect to particular applications (Hardy *et al.*, 2003). Roache (1997) regarded verification as 'solving the equations right' and validation as 'solving the right equations'. Lane and Richards (2001) again use 'verification' for correct solution of the associated equations but define 'validation' as correct determination of parameters predicted by the equations. Hardy *et al.* (2003) use 'verification' for tests of the ability of the discrete computational scheme to provide an accurate solution of the underlying equations, and 'validation' for tests of the plausibility of the model as a whole. Verification involves checking for minimisation of coding errors as well as errors associated with both spatial and temporal discretisation of the numerical solution (Hardy *et al.*, 2003). If these errors have been minimised then validation will yield the error due to specification of system geometry, initial conditions, boundary conditions and parameters, as well as to process representation decisions (e.g. over turbulence closure) (Hardy *et al.*, 2003).

A crucial aspect of spatial discretisation is the choice of mesh resolution (Lane *et al.* 2005). In the development of hydraulic models, there has been a trend among many modellers to increase the spatial resolution (the number of cells representing the spatial area of interest) in the expectation of improved insights into temporal and spatial processes (Hardy *et al.*, 1999). However, the spatial resolution at which a model is applied affects the solution of the equations and thus the simulation result (Hardy *et al.*, 1999). The hypothesis that a models predictive ability increases as the spatial and temporal resolution increases, stems from three avenues of thought (Hardy *et al.*, 1999): i) expected improvements in solution stability as the grid spacing tends towards the true continuum level; ii) the ability of high resolution models to facilitate complex, and thereby more realistic parameterisation of the code; and iii) a closer correspondence of field measurement model scales.

One approach for assessing the choice of mesh resolution is the Grid Convergence Index (GCI) (Roache, 1994, 1997, 1998). The GCI is an index of the uncertainty associated with the solution at a particular grid resolution (the mesh uncertainty), based on comparison with the solution at another resolution (Lane *et al.*, 2005). It uses the theory of generalised Richardson extrapolation, which assumes that, within a certain radius of convergence, the discrete solution for some flow variable converges monotonically at all points in the continuum as the grid spacing tends towards zero (Lane *et al.*, 2005).

Hardy *et al.* (1999) in their investigation on floodplains found that: i) spatial resolution directly affects bulk flow characteristics; ii) spatial resolution directly affects the inundation extent although it may be an effect of the loss of topographic information; iii) spatial resolution has a greater effect than the typical calibration parameter, friction, in altering the hydraulic simulations; this indicates that initial model set up needs to be carefully considered and the transfer of parameter values should not occur; and iv) the spatial resolution has a dramatic effect on the internal results.

As such, Hardy *et al.* (1999) recommended the construction of at least four meshes of different spatial resolutions to ascertain the envelope of response to spatial resolution. Lane *et al.* (1999) argue that as mesh resolution is increased there is the progressive introduction of geometric variability that is associated not with the bed topography, but with the way it has been sampled. The result is that it may be impossible to obtain a solution that is mesh independent.

Crowder and Diplas (2000b) hypothesised that mesh refinement, particularly near the banks, may significantly impact wetting and drying processes and velocities near the bank. Consequently, even channel topography which can be accurately described with spot elevations taken every 20m may require element size much smaller than 20m x 20m

(Crowder and Diplas, 2000b). Sensitivity analyses are also needed to properly calibrate a model. If calibration is performed without a sensitivity analysis, the adjusted channel roughness and eddy viscosity values may be compensating for low numerical accuracy and not variations in actual roughness and eddy viscosity values (Crowder and Diplas, 2000b). For example, increasing roughness coefficients near channel banks to compensate for using a coarse mesh may result in unrealistic roughness coefficients near the banks and thus inappropriate velocity values (Crowder and Diplas, 2000b). Similarly, when obstructions are not included in a models bathymetry data, the boulders are viewed simply as channel roughness rather than as part of the channel. Any local effects the boulders create are diffused throughout the modelled stream section via roughness and eddy viscosity values (Crowder and Diplas, 2000b).

Pasternack *et al.* (2006) used the Finite Element Surface Water Modelling System (FESWMS) to look at error propagation for velocity and shear stress prediction using 2-D models for environmental management. In their study, they found that the error in 2-D model predictions of depth, velocity and shear velocity over well-mixed double washed gravel averaged 21%, 29% and 31% respectively. These accuracies reflect the very challenging field conditions on a carefully constructed geomorphic unit with complex 3-D features by design. They found that depth error prediction was directly attributable to error in the DEM and thus it was not primarily an error of the 2-D model itself. More than half the error in velocity was in turn caused by depth error, and then 90% of the shear velocity error was caused in the velocity error. Thus the single most important control of accuracy in 2D model predictions is DEM quality (Pasternack *et al.*, 2006) (i.e. precision, accuracy and resolution).

Pasternack *et al.* (2006) used a bed survey with a resolution of 1 point per 1.14m^2, which is quite high relative to the previously published efforts to and above that specified to capture typical gravel bed morphology (Brasington *et al.*, 2000), apart from Lane *et al.* (1994). Unlike sands, gravel beds can have significant interlocking grain friction that is capable of sustaining complex pebble cluster morphologies, depressions and bars at scales of 0.001-1m (Pasternack *et al.*, 2006). It is now apparent that reducing the error of 2D model predictions at individual nodes from the 20% to 30% range to the <10% range must require higher survey point densities than 1 point every 1.2m^2 (Pasternack *et al.*, 2006).

5.2.4. Model calibration

In 1-D models, calibration is provided by altering the roughness of the channel. In a 1-D model, the roughness parameter is an effective parameter, in that it represents the effects of turbulence, secondary circulation and boundary friction in a single parameter (e.g. Manning's *n*) (Lane *et al.*,2005). Thus, the resemblance to data that can be measured in the field is poor, but it has proved to be an effective and sensitive parameter (Romanowicz *et al.*, 1996).

In 2D models, the calibration parameters are Manning's roughness and eddy viscosity. The issue of how to specify the boundary condition on a river bed (roughness) is probably the most challenging of all boundary condition issues (Lane and Ferguson, 2005). In the case of a 3D model, the roughness parameter only contributes directly to the bottom grid cell, commonly as a control upon the elevation at which the velocity becomes zero within the grid cell (i.e. a roughness height) (Lane and Ferguson, 2005). In the 2D scenario, as the 3D equations are averaged over the flow depth, the bottom shear stress appears in the momentum equations explicitly, as a source term, i.e. it affects every cell, rather than being a condition that only affects boundary cells (Lane and Ferguson, 2005). It is commonly assumed that the shear stress can be expressed as a square law of the depth-averaged velocity (*U*) using:

$$\tau_0 = k\rho U^2 \qquad (5.2.17)$$

The parameter *k* is then expressed in terms of a roughness parameter, such as Chezy's C or Manning's *n*. For uniform flow it is readily found that equation 5.2.13 is equivalent to the Darcy-Weisbach equation (Equation 5.2.5) with *k=f/8* (Lane and Ferguson, 2005). Similarly, comparison with the Chezy and Manning equation gives:

$$k = \frac{g}{C^2} \qquad (5.2.18)$$

$$k = \frac{gn^2}{h^{1/3}} \qquad (5.2.19)$$

C (or *f* or *n*) can thus be regarded as an empirical parameter that determines the effective roughness required to get the correct relationship between shear stress and velocity (Lane and Ferguson, 2005).

Uniform values of Manning's n have been used by Shimizu and Itakura (1989); Tingsanchali and Mahewswaran (1990); Shimizu *et al.* (1990). However, most notably in coarse grained channels, C_f will be spatially variable, reflecting spatial variation in grain size and sedimentological structure. An alternative method (e.g. Dietrich and Whiting, 1989) involves specification of a roughness height, but both *n* and roughness height specifications demand information on grain size. There may or may not be some attempt to correct estimates to correct estimates based on grain size for the increase in roughness associated with sedimentological structures, sediment transport or the form of drag associated with reach-scale topography, such that friction is used as one of the key calibration parameters (Lane, 1998). In general (e.g. Dietrich and Whiting, 1989; Nelson and Smith, 1989) spatial variations in the roughness coefficient have tended to be ignored, and a characteristic grain size for each reach is specified.

An analogy with the 1-D case was provided by Lane and Ferguson (2005): in one dimensional models, roughness parameterisation is required in order to get the channel conveyance correct; in 2D models, it is needed in order to get the depth-averaged velocity and local water depth correct. It is an effective roughness parameter that may show little resemblance to measured values of variables (e.g. bed grain size) that might in part control the roughness (Lane and Ferguson, 2005).

A more physically based alternative to the traditional 1D friction parameters to estimate *k* is to assume that the law of the wall holds throughout the full flow depth (Lane and Ferguson, 2005). The law of the wall is given by:

$$U = \frac{1}{\kappa}\sqrt{\frac{\tau_0}{\rho}}\ln\left(\frac{z}{z_0}\right) \qquad (5.2.20)$$

where □is von Karman's constant; z is the height above the bed; and z_0 is the height above the bed at which the velocity becomes zero. The depth-averaged value of velocity (U) is given by integration of equation (5.2.20) across the water depth (h) as (Lane and Ferguson, 2005):

$$U = \frac{1}{\kappa}\sqrt{\frac{\tau_0}{\rho}}\ln\left(\frac{h}{e \cdot z_0}\right) \qquad (5.2.21)$$

$$\tau_0 = \left(\frac{\kappa}{\ln\left(\frac{h}{e \cdot z_0}\right)}\right)^2 \rho \bar{U}^2 \equiv k\rho\bar{U}^2 \qquad (5.2.22)$$

This gives the classic square law resistance velocity relationship assumed above (Lane and Ferguson, 2005). The derivation of equation (5.2.20) (e.g. Richards, 1982) is based upon the Boussinesq approximation in which shear stress is assumed to be proportional to a local strain rate (the vertical variation of velocity with elevation above the bed) and a constant of proportionality (the eddy viscosity). Research by Prandtl (1952) showed that the eddy viscosity is itself proportional to the local strain rate and a mixing length term related to von Karman's constant and distance above the bed (Lane and Ferguson, 2005). This leads to equation (5.2.20) without any assumption of the relationship between shear stress and velocity, and is an important justification for the form of equation (5.2.17) (Lane and Ferguson, 2005). Equation (5.2.17) also allows parameters like n to be expressed in terms of bed roughness height, which has a clearer physical meaning, as the elevation above the bed at which the velocity becomes zero (Lane and Ferguson, 2005):

$$n = \frac{\kappa h^{1/6}}{\sqrt{g}\ln\left(\frac{h}{e \cdot z_0}\right)} \qquad (5.2.23)$$

Equation 5.2.21 has an important property: it demonstrates that the roughness height parameter n depends on the local flow depth (h) as well as roughness height (z_o).

5.2.5. Model validation

Conventional validation is based upon the premise that when a model fails to predict independent data adequately, something must be wrong (Luis and McLaughlin, 1992). However, when model predictions are correct, the model is not necessarily valid, as it is possible for an invalid model to provide an adequate representation of some aspects of reality (Lane *et al.*, 2005). Bearing this in mind, model validation should consider (Flavelle, 1992): i) the extent to which variability in the observations in explained by the predictions, which can vary from 0% (imprecise or poor fit) to 100% (precise or good fit); ii) the extent to which predictions agree with observations, which can vary from prefect equality (accurate or unbiased) to perfect inequality (inaccurate or biased); and iii) the extent to which the predictions provide sufficiently reliable information for them to be accepted when there are no check data (e.g. when boundary conditions are different from those used when the check data was collected).

Very few comparisons between two and three dimensional numerical models and field velocity data have been undertaken (Bates and Anderson, 2001). Lane *et al.* (1995) compared a two dimensional depth averaged model to velocities measured in a proglacial stream and found correlations between observed and measured data of 0.72 for downstream velocities and 0.57 for cross-stream velocities. Using an improved velocity data set (measured using Acoustic Doppler Velocimetry) for a gravel bed river confluence, Lane *et al.* (1999) found improved correlations between observations and model predictions. Similarly, Nicholas and Smith (1999) attempted to validate a 3D model against two dimensional velocities measured using an Electromagnetic Current Meter (ECM) and found a correlation of 0.88 with the vector product of the downstream and cross-stream components of velocity).

The description of both the accuracy and precision of a model simulation is crucial to establishing how model predictions may be used (Lane *et al.*, 2005). Good precision and good accuracy is the ideal case. If the precision is good but there is bias, then it is likely that the patterns of model predictions are good but their quantitative magnitudes are not (Lane *et al.*, 2005). An empirical correction factor could be used in this instance (Lane *et al.*, 2005). If the precision is poor, the model is less likely to be of use, even if the accuracy is good, as it implies substantial local uncertainty (Lane *et al.*, 2005). At best, such a model may be used in general qualitative terms, coupled with the associated reporting of model uncertainty (Lane *et al.*, 2005).

5.3. Methods

Figure 5.3.1 conceptually outlines the complete modelling approach taken in this work including the habitat treatment described in Chapter 6. The basic approach is to use hydrodynamic simulations of wetted width, velocity and depth to drive fuzzy modelling of habitat availability for both fish and macroinvertebrates. However, the objective of this chapter is to examine the hydrodynamic modelling results. Therefore, the area of interest, in terms of the overall modelling structure, is highlighted in black in Figure 5.3.1. The first aspect of the methodology to be examined is the model structure (5.3.1), as it is this which defines the data needs of the following section (5.3.2), which addresses the fieldwork conducted to meet the data requirements of the model. The final section will address the data analysis tools used in the analysis of the simulation results.

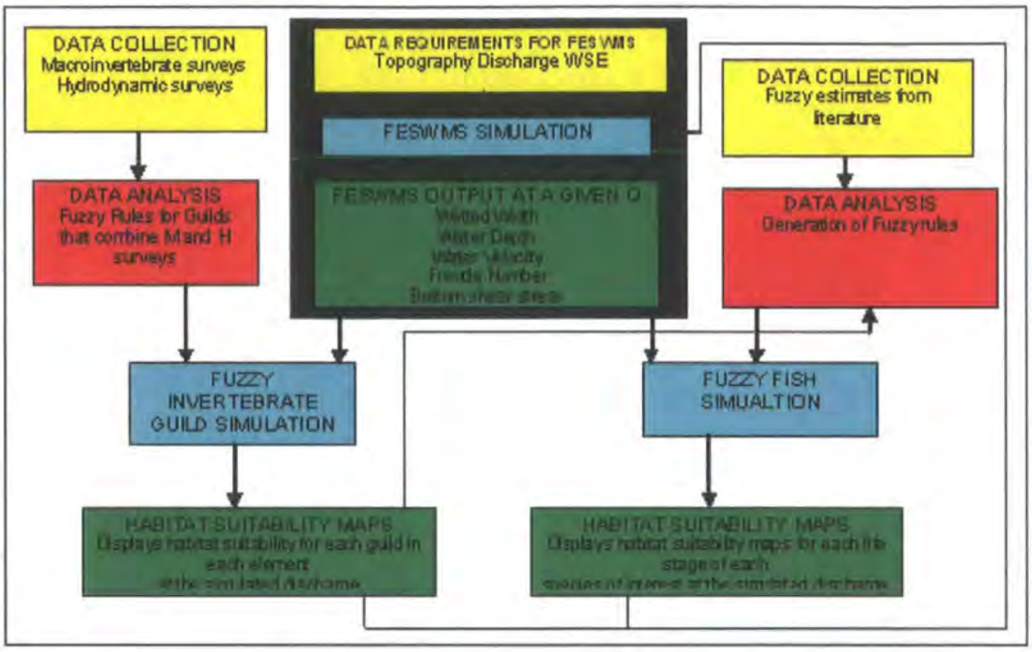


Figure 5.3.1. Figure to show the flow of information through the modelling process. WSE = water surface elevation; Q = discharge.

5.3.1. Finite Element Surface Water Modelling System (FESWMS)

A two dimensional scheme was chosen for a number of reasons. The first was a desire to include spatial heterogeneity into the hydraulic modelling process. The study rivers contain many boulders and as was demonstrated by Crowder and Diplas (2000a; 2000b; 2002), boulders are very important in creating localised flow patterns of potential biological importance. The importance of spatial heterogeneity in these rivers was demonstrated in Chapter 3, where there was rarely a statistically significant difference between central and marginal invertebrate samples in terms of the measured environmental variables.

The second reason was to include the interaction of reach scale morphology and discharge, which was found to be important for the brown trout populations (especially in the Rivelin). The use of a one-dimensional model would not incorporate such features into the hydraulic output. The third reason why a 2-D scheme was chosen is that it is computationally cheaper than using a 3D scheme. This is despite the fact that using a 3D scheme would produce interesting near bed flows, it would be too computationally expensive for this study. Using a 2D scheme also allows many simulations to be run with a variety of parameter sets, which firstly allow a hydraulic sensitivity analysis to be conducted, but also more interestingly allow the sensitivity of the habitat predictions to be assessed (Chapter 6).

The hydrodynamics of the systems are simulated using a 2-D, finite element, depth-averaged hydrodynamic model called FESWMS. The model was developed by the U.S. Department of Transportation, Federal Highway Administration (Froehlich, 2002). The flow model within FESWMS applies the finite element method to solve steady-state and time-dependent systems of equations that describe two-dimensional depth averaged water flow. It is not a widely used model, however there have been three recent studies which involved the use of FESWMS (Pasternak *et al.*, 2004, 2006; Wheaton *et al.*, 2004). The primary reasons for choosing FESWMS are two fold. First, it retains an explicit time dependence which means that the spatial extent of inundation does not need to be known *a priori*. Rather, in response to specification of channel geometry and boundary conditions (roughness, eddy viscosity, flow rate, stage at outlet) it determines the spatial patterns of inundation. This requires the use of a wetting-drying algorithm as the model normally initialises on a complete inundation and is then dried to give the correct flow-stage condition. Once initialised, the model can be simulated with changing flows. Wetting-drying algorithms are notoriously unstable (Lane, 1998) and FESWMS was chosen as one with a reliable and relatively stable algorithm. Second, as the code is finite element, it provides for a better representation of complex surface geometry, especially where there are individual boulders.

Depth averaged velocity components in the horizontal x and y coordinate directions respectively, are defined as follows (Froehlich, 2002):

$$U = \frac{1}{H}\int_{z_b}^{z_s} u.dz \qquad V = \frac{1}{H}\int_{z_b}^{z_s} v.dz \qquad (5.3.1)$$

where H = water depth, z = vertical direction, z_b = bed elevation, $z_s = z_b + H$ = water surface elevation, u = horizontal velocity in the x direction at a point along a vertical coordinate, and v = horizontal velocity in the y direction at a point along the vertical coordinate.

Equations depth averaged water surface flow are found by integrating the three dimensional mass and momentum transport equations with respect to the vertical coordinate from the bed to the water surface, considering vertical velocities and accelerations to be negligible. The vertically-integrated mass transport equation or continuity is (Froehlich, 2002):

$$\frac{\partial z_w}{\partial t}+\frac{\partial q_1}{\partial x}+\frac{\partial q_2}{\partial y}=q_m \qquad (5.3.2)$$

where q_1 = UH = unit flow rate in the x direction, q_2 = VH = unit flow rate in the y direction, q_m = mass inflow or outflow rate per unit area, and water mass density □ is considered constant throughout the modelled region. Equations describing momentum transport in the x and y directions, respectively, are as follows (Froehlich, 2002):

$$\frac{\partial q_1}{\partial t}+\frac{\partial}{\partial x}\left(\beta\frac{q_1^2}{H}+\frac{1}{2}gH^2\right)+\frac{\partial}{\partial y}\left(\beta\frac{q_1 q_2}{H}\right)+gH\frac{\partial z_b}{\partial x}+\frac{H}{\rho}\frac{\partial P_a}{\partial x}-\Omega q_2$$

$$+\frac{1}{\rho}\left[\tau_{bx}-\tau_{sx}-\frac{\partial(H\tau_{xx})}{\partial x}-\frac{\partial(H\tau_{xy})}{\partial y}\right]=0$$

and (5.3.3)

$$\frac{\partial q_2}{\partial t}+\frac{\partial}{\partial x}\left(\beta\frac{q_1 q_2}{H}\right)+\frac{\partial}{\partial y}\left(\beta\frac{q_1^2}{H}+\frac{1}{2}gH^2\right)+gH\frac{\partial z_b}{\partial y}+\frac{H}{P}\frac{\partial P_a}{\partial y}+\Omega q_1$$

$$+\frac{1}{\rho}\left[\tau_{by}-\tau_{sy}-\frac{\partial(H\tau_{yx})}{\partial x}-\frac{\partial(H\tau_{yy})}{\partial y}\right]=0$$

where □= isotropic momentum flux correction coefficient that accounts for the variation of velocity in the vertical direction, g = gravitational acceleration, □= water mass density, P_a = atmospheric pressure at the water surface, □ = Coriolis parameter, τ_{bx} and τ_{by} = bed shear stresses acting in the x and y directions, and τ_{sx} and τ_{sy} = surface shear stresses acting in the x and y directions, respectively, and τ_{xx}, τ_{xy}, τ_{yx}, τ_{yy} = shear stresses caused by turbulence where, for example, τ_{xy} is the shear stress acting in the x direction on a plane that is perpendicular to the y direction.

The directional components of bed shear stress are computed as follows (Froehlich, 2002):

$$\tau_{bx} = \rho c_f m_b \frac{q_1\sqrt{q_1^2 + q_2^2}}{H^2} \qquad \tau_{by} = \rho c_f m_b \frac{q_2\sqrt{q_1^2 + q_2^2}}{H^2} \qquad (5.3.4)$$

where c_f = dimensionless bed-friction coefficient, and

$$m_b = \sqrt{1 + \left(\frac{\partial z_b}{\partial x}\right)^2 + \left(\frac{\partial z_b}{\partial y}\right)^2} \qquad (5.3.5)$$

is a factor that accounts for increased shear stress caused by a sloping bed. Bed friction coefficients c_f are given by (Froehlich, 2002):

$$c_f = \frac{gn^2}{\phi_n^2 H^{1/3}} \qquad (5.3.6)$$

where n = Manning's roughness coefficient, □ = 1.0.

Appropriate flow resistance coefficients can be estimated using references such as Chow (1959). We will be using representative values of roughness in order to get the most 'accurate' simulations: i.e. we calibrate roughness so as to give the correct water surface elevations and velocities due to a given flow (Lane and Ferguson, 2005).

Depth averaged lateral shear stresses caused by turbulence are computed using Boussinesq's eddy viscosity concept whereby the turbulent stresses, like viscous stresses, are considered proportional to gradients of the depth averaged velocities (Froehlich, 2002). The stresses are computed as follows (Froehlich, 2002):

$$\tau_{xx} = \rho v_t\left(\frac{\partial U}{\partial x}+\frac{\partial U}{\partial x}\right),\ \tau_{xy} = \rho v_t\left(\frac{\partial U}{\partial y}+\frac{\partial V}{\partial x}\right),\tau_{yy} = \rho v_t\left(\frac{\partial V}{\partial y}+\frac{\partial V}{\partial y}\right) \qquad (5.3.7)$$

where v_t = depth-averaged kinematic eddy viscosity or turbulent exchange coefficient, which is considered isotropic.

Eddy viscosity is related to eddy diffusivity for heat or mass transfer □ as (Froehlich, 2002):

$$\Gamma_t = \frac{v_t}{\sigma_t} \qquad (5.3.8)$$

where □ = an empirical constant called the Prandtl number (for diffusion of heat) or Schmidt number (for diffusion of mass). Considering turbulent exchange of mass and momentum to be similar (that is, □ = 1), eddy viscosity in natural open channels can be related to bed shear velocity and depth by (Froehlich, 2002):

$$V_t = (0.6 \pm 0.3)u_* H \qquad (5.3.9)$$

Relating eddy viscosity to the scales of motion being resolved by a mesh and the local deformation field, and Kinematic eddy viscosities are calculated in FESWMS as follows (Froehlich, 2002):

$$v_t = v_{t0} + c_{\mu 1}u_* H + c_{\mu 2}|J|\sqrt{\left(\frac{\partial U}{\partial x}\right)^2 + \left(\frac{\partial V}{\partial y}\right)^2 + \frac{1}{2}\left(\frac{\partial U}{\partial y}+\frac{\partial V}{\partial x}\right)^2} \qquad (5.3.10)$$

where v_{t0} = base kinematic eddy viscosity, $\mathrm{C}_{\mu 1}$, $\mathrm{C}_{\mu 2}$ = dimensionless coefficients, $|J|$ = determinant of the jacobian matrix of element coordinate transformations, which provides pointwise measures of element area. Comparing expressions for v_t shows that $c_{\mu 1} \square 0.6 \pm 0.3$ in natural channels when $v_{t0} = 0$ and $c_{\mu 2} = 0$; and that $c_{\mu 2} \square 0.1$ when $\mathrm{v}_{t0} = 0$ and $c_{\mu 1} = 0$. Constant eddy viscosities are assigned by spedifying $c_{\mu 1} = c_{\mu 2} = 0$, and $v_{t0} > 0$. This was done in this study.

As discussed in section 2, the topography of a given reach is represented in the form of connected subregions called elements that are in the shape of either triangles or quadrilaterals (Froehlich, 2002). This assemblage of elements forms a mesh. The position and geometry of the elements in a mesh are defined by node points at the element vertices, midside points, and for nine-node quadrilaterals, at their centres (Froehlich, 2002). Solution variables are defined at the nodes, and continuous approximations of quantities are made within elements using interpolation functions and the node point values (Froehlich, 2002).

Node points within FESWMS become dry when calculated water surface elevations are lower than their bed elevations. Elements that contain at least one dry node are turned off at the start of an iteration and are not included in the calculations (Froehlich, 2002). All elements that are turned off are checked at the start of an iteration to see if they can be turned back on as previously dry nodes become wet (that is, as the water surface rises above the bed), and boundary conditions are modified (Froehlich, 2002). Adjustment of boundaries in this way allows a finite element network to be constructed without too much concern for the limits of inundation (Froehlich, 2002). However, solution stability can be affected adversely by elements switching on and off, especially if the elements in transition are comparatively large and only small portions of those elements actually dry (Froehlich, 2002).

By introducing the concept of element storativity, partially dry elements can be retained in calculations when solving the governing equations (Froehlich, 2002). Bed storativity coefficients, $\square_b$, are ratios of changes in stored water per unit area with respect to changes in water elevation and are calculated as follows (Froehlich, 2002):

$$\lambda_b = \begin{cases} 1; if z_w \geq z_b + \zeta \\ a + (1-a)\left(\dfrac{z_w - z_b + b}{\zeta + b}\right); if z_b + \zeta > z_w > z_b - b \\ a; if z_b - b \geq z_w > z_b - \eta_b \zeta \\ 0; z_w \leq z_b - \eta_b \zeta \end{cases} \tag{5.3.11}$$

where □= storativity depth; a = minimum element storativity; η_b = storativity depth factor; and (Froehlich, 2002):

$$b = \zeta\left[\frac{1-(1+2\eta_b)a}{1-a}\right] \tag{5.3.12}$$

is the depth below z_b at which □$_b$ = a. Element storativity □$_b$ for z_b – b □z_w □$z_b - \eta_b\zeta$. The coefficient □$_b$ represents the ability of elements to store water when water depth is less than □ and is shown in Figure 5.3.2 as a function of z_w and □ (Froehlich, 2002).

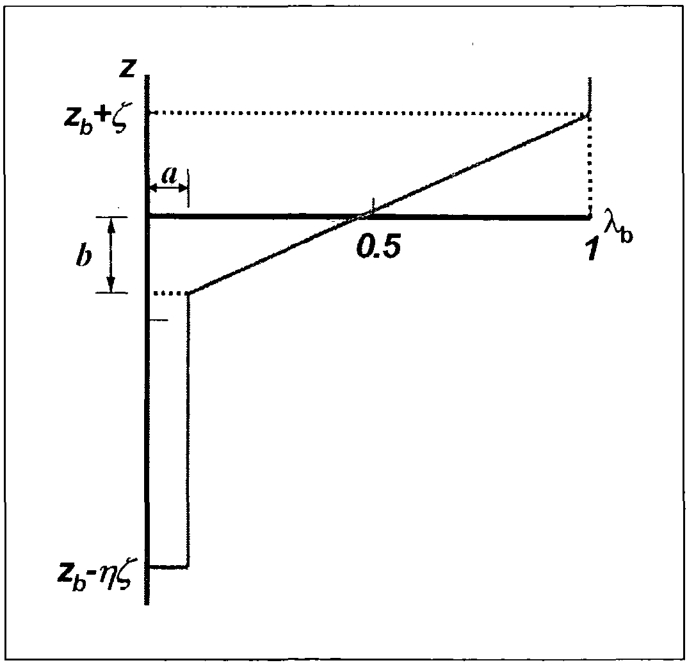


Figure 5.3,2. Figure to show the element storativity coefficient λ_b as a function of water surface elevation z_w, storativity depth ζ, minimum storativity a, and storativity depth factor η_b. From, Froehlich (2002).

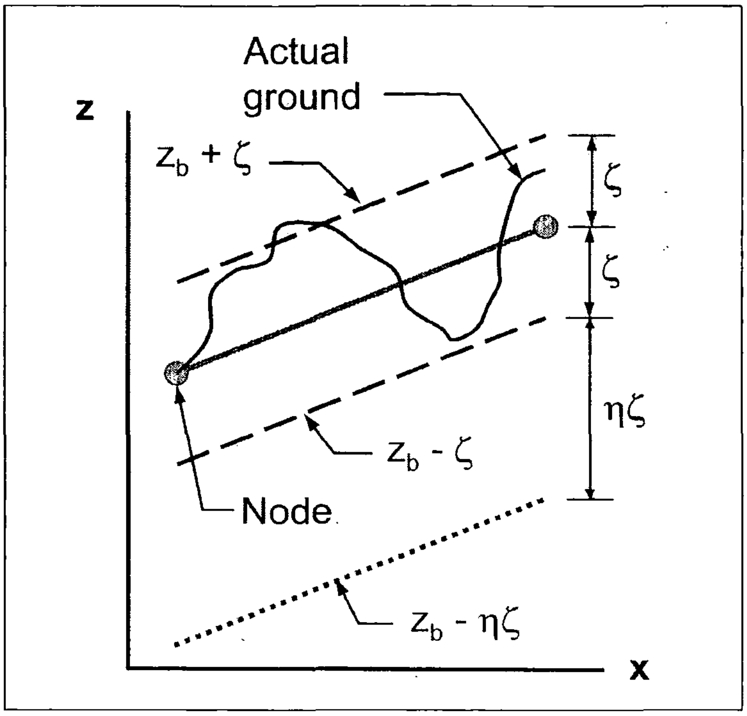


Figure 5.3.3. Figure to show the variation of bed elevation within an element showing storativity depth ζ. From, Froehlich (2002).

Element storativity is implemented in computations by replacing water depth $H = z_w - z_b$ with an effective water depth H_{eff}, which is given by (Froehlich, 2002),

$$H_{eff} = \int_{z_b-\eta_b\zeta}^{z_w} \lambda_b dz \qquad (5.3.13)$$

The expression for the bed storativity □$_b$ assures that $H_{eff} = H$ when z_w □z_b + □ Storativity depth □depens on ground surface variability within an element as shown in Figure 5.3.3. Storativity depths vanish for elements having perfectly planar surfaces (Froehlich, 2002). However, non-zero storativity depths might be assigned even for perfectly planar elements to keep them from being turned off when only small sections of them are dry (Froehlich, 2002). Therefore, use of element storativity is beneficial because of both physical and computational reasons (Froehlich, 2002). Froehlich (2002) stated that numerical experiments showed that assigning $a = 0.01$ and $\eta_b = 3$ provides a good means of controlling element transition from wet to dry states.

5.3.2. Data collection

There are three pieces of information needed to run FESWMS: 1) topographic data; 2) discharge; and 3) downstream water surface elevation.

The approach taken for the 2-D hydrodynamic modelling was reach-scale (around 10 x channel widths) because the computational time required for modelling longer reaches would have been too great, and the nature of the rivers (closely wooded) meant that surveying longer reaches would have been very difficult. In each study reach, at least one pool-riffle sequence was included. Two sample reaches were selected for each river (Figure 5.3.4). These were situated close to, but not on top of the macroinvertebrate sampling sites, as it was thought preferable not to disturb the macroinvertebrate sites during the detailed surveying process.

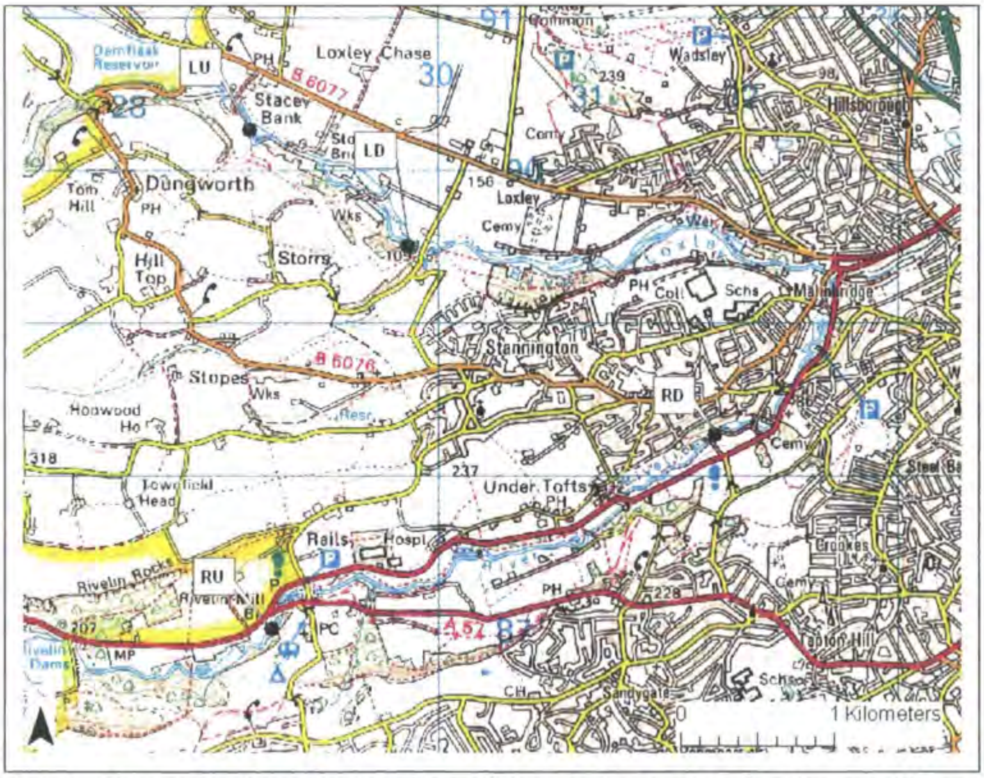


Figure 5.3.4. Figure to show the location of the study reaches for the modelling.

The surveying process for collecting the topographic information was conducted in a number of stages. The bulk of the surveying was conducted in spring 2005, with some further work conducted in autumn 2005 when there were fewer leaves on the trees. Data collection used a Leica 1200 robotic EDM. For each site, there were a number of features which were surveyed (in each case, surveying breaks of slope): lateral cross sections at 1m spacing; plus bottoms of banks; tops of banks; breaks of banks; and boulders (in order to ensure a representation of the spatial variability of flow). Meshes were created using the Surfer software (Golden Software Inc.). The Kriging option within surfer was used (with the bottom of bank data set as breaklines) in order to produce meshes with regular node spacing. For each site, two mesh resolutions were created (0.25m and 0.15m), in order to investigate the potential influence of mesh resolution on hydraulic predictions (e.g. Hardy *et al.*, 1999). The 0.25m spacing Digital Elevation Models (DEM) for each of the sites can be seen in Figures 5.3.5 to 5.3.7. A 15m 'spin-up' zone of consistent topography was used for each mesh in order to aid model convergence, and can be seen in each of the DEMs below. This area was ignored from any analysis of the simulations.

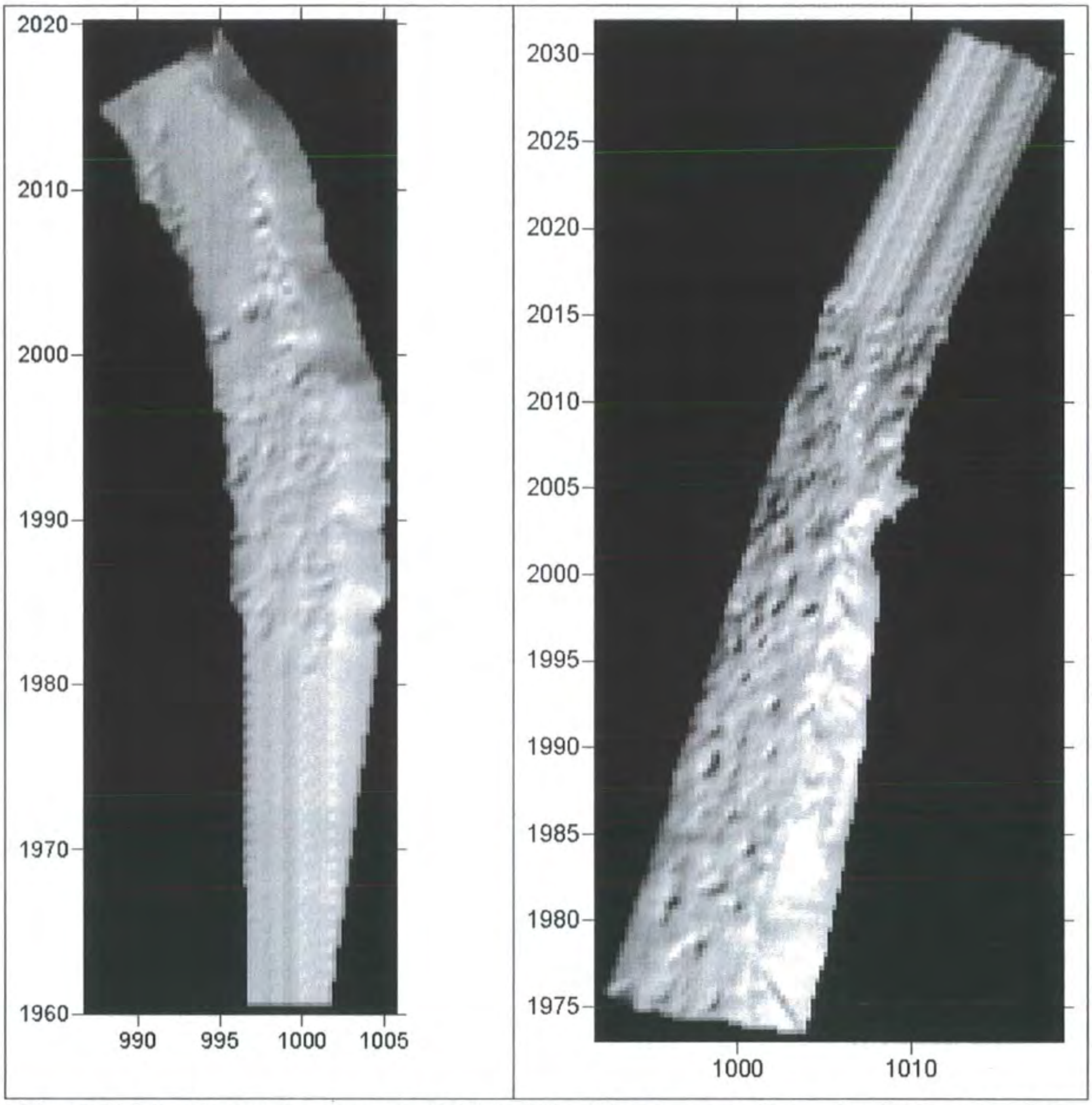


Figure 5.3.5. Figure to show the DEM of the Rivelin modelling sites (RU on the left; RD on the right).

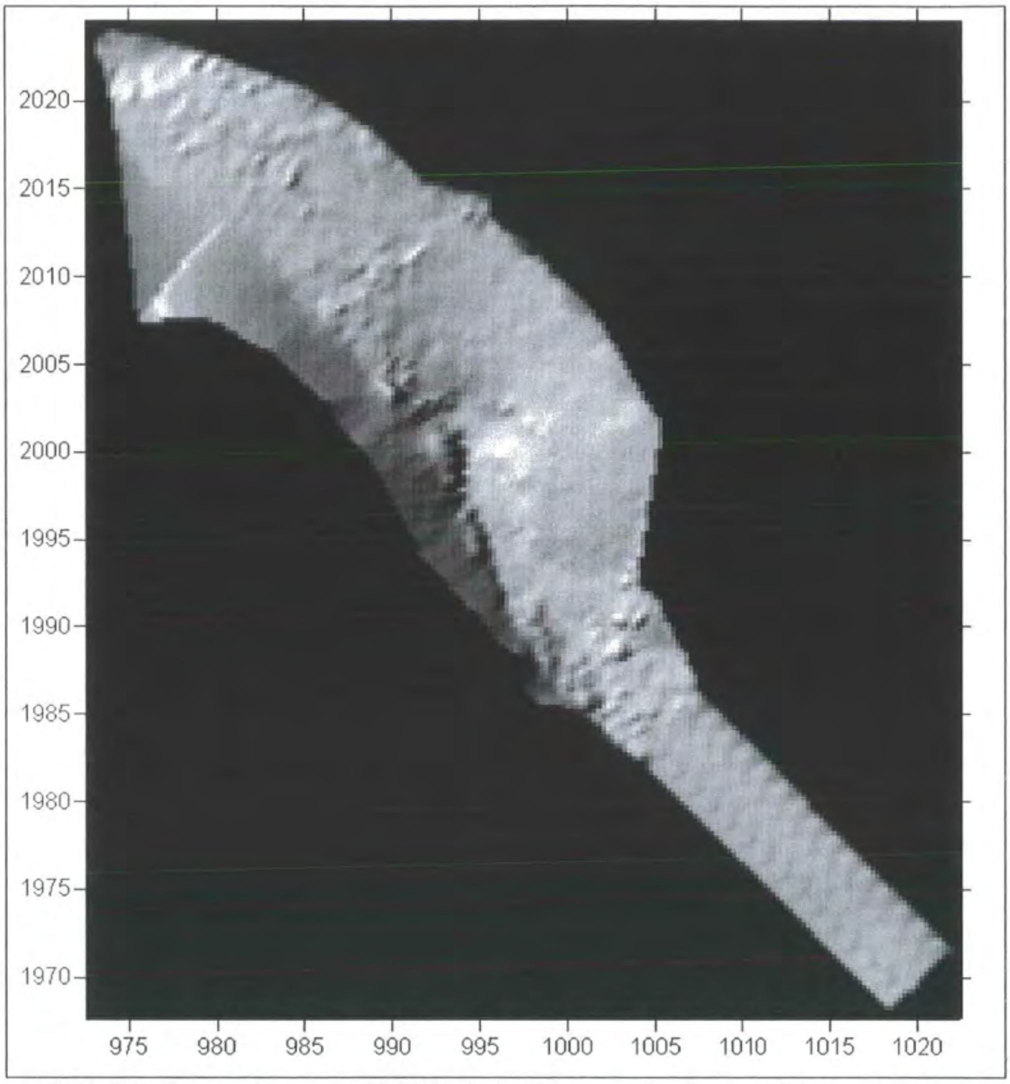


Figure 5.3.6. Figure to show the DEM of the Loxley Upstream (LU) modelling site.

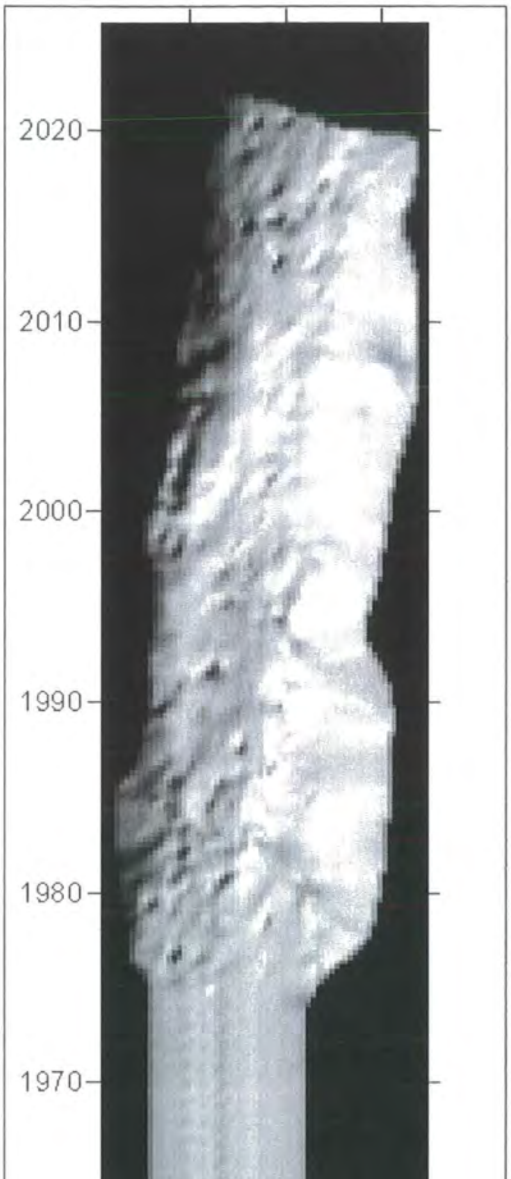


Figure 5.3.7. Figure to show the DEM of the Loxley Downstream (LD) sampling site.

The second piece of information needed is the discharge entering each of the reaches. This was based upon the velocity-area method (Richards, 1982). Depth and depth averaged velocity were measured at intervals along a cross-section, ensuring no more then 10% of discharge passed through a given ‘segment’. The depth-averaged velocity was measured using an electromagnetic flow meter at 40% of the water depth up from the bed, and three 30s averaged readings were taken.

The final piece of information needed to run the 2-D model is the water surface elevation at the downstream boundary nodestring. Obviously, the two most important discharges which needed to be simulated for each of the sites are the pre- and post change compensation releases. Both of these were difficult as a result of potential tributary and overtopping effects. Thus, the discharge was measured and the associated water surface elevations recorded from a water surface edge survey, undertaken using the EDM. The water surface elevation was then scaled by the change in discharge required to give the pre and post compensation flows:

$$WSE_u = WSE_k - \left(Depth_k \left(1 - \left(\frac{Q_u}{Q_k} \right) \right) \right) \qquad (5.3.14)$$

where, WSE_u = the water surface elevation needed. WSE_k = known water surface elevation, $Depth_k$ = known average depth at the downstream boundary nodestring, Q_u = discharge for the unknown water surface elevation, Q_k = discharge for the known water surface elevation. The assumption above is a linear relationship between depth and discharge which is likely to be acceptable up to the point at which all roughness elements become submerged. At this point relative roughness should start to fall and changes in discharge would also involved changes in velocity. The simulated discharges were not sufficient to submerge all boulders.

Table 5.3.1 shows the discharges and water surface elevations used as boundary conditions in the two dimensional model. The values in bold in Table 5.3.1 shows values which were measured in the field. The values in bold are known compensation release, whilst the remaining values are those calculated using equation 5.3.14. The pre change discharge is of interest as it will be used to asses any impact of the 'existing' compensation flows on the habitat available for brown trout and macroinvertebrates in each of the study rivers. The post-change compensation discharge will be used to asses the impacts of altering the compensation flows on the study rivers. The remaining discharges to be simulated are the three discharges used for the three validation tests which will be run on the model.

	Site: RU		**Site: RD**		**Site: LU**		**Site: LD**	
	Q (m^3/s)	**WSE**	**Q (m^3/s)**	**WSE**	**Q (m^3/s)**	**WSE**	**Q (m^3/s)**	**WSE**
Pre-change comp flow	*0.030*	99.280	*0.03*	99.670	*0.324*	99.390	*0.324*	99.723
Pre-change survey measurements	**0.127**	99.530	**0.127**	99.780	**0.3708**	99.430	**0.585**	99.820
Post-change comp flow	*0.099*	99.460	*0.099*	99.747	*0.255*	99.315	*0.255*	99.700
Post-change survey measurements	**0.150**	**99.590**	**0.187**	**99.852**	**0.269**	**99.330**	**0.477**	**99.780**
Post-change Depths and velocities	**0.236**	99.790	**0.1723**	99.830	**0.293**	99.360	**0.497**	99.787

Table 5.3.1. Table to show the discharges and water surface elevations of the simulations run. Q = discharge; WSE = downstream water surface elevation. Measured values in bold, compensation releases in italics.

A simulation is said to be converging when the residuals (or errors) in the equations decrease as the iterative solution proceeds (Lane *et al.*, 2005). The solution is said to have converged when the sum of the absolute values of the residuals for that variable falls below a pre-specified tolerance (Lane *et al.*, 2005). The exact definition of the criterion is ambiguous in fluvial applications, with the actual values rarely, if ever defined. One approach is to set the tolerance for mass and momentum flux residuals to 0.1% of the inlet flux (Lane *et al.*, 2005). However, achieving the level of convergence for fluvial problems can be difficult especially at high resolution meshes (Lane *et al.*, 2005). Indeed, especially where the bed topography and/or the associated flow fields is complex, achieving this level of convergence may require good initial approximations. This work used a unit flow convergence value of 0.001m^3/s; and a unit depth convergence of 0.001m. Under these convergence parameters three simulations did not converge:

- Rivelin upstream site; 0.25m spacing mesh; pre-change compensation flow;
- Loxley downstream; 0.15m spacing mesh; pre-change compensation flow; and
- Loxley downstream; 0.15m spacing mesh; post-change compensation flow.

Lane *et al.* (2005) also discuss the role of model relaxation in model convergence. Relaxation is a technique that can be used to increase or decrease the convergence rate of (or even achieve convergence when the problem would diverge) by slowing down the rate at which variables may change during the iteration procedure. The relaxation factor in this study was maintained at a value of one.

5.3.3. Data analysis methods

Before the change in compensation flows on the 1st of April 2004, the waters edge was surveyed at each of the sampling sites at a 15cm resolution. This allows the potential for calibration of the model to pre-change flows. This was repeated for the post change discharge. In order to contextualise these measurements in terms of the flow regime, the discharge was measured before each measurement.

Also, for the post change discharge, 60 point velocity magnitude and depth measurements were made in each reach for the purpose of model validation. The velocity magnitudes were measured using an electromagnetic flow meter, sampling at 40% of water depth above the bed (i.e. assumed to be the average of the velocity and therefore commensurate to model predictions) for 30 seconds, with the positions mapped with the EDM. This comparison of measured and predicted velocity has been used in many similar studies (e.g. Czernuszenko and Rylov, 2000; Meselhe and Sotiropolous, 2000; Sanjiv and Marelius, 2000; Sofialidis and Prinos, 2000; Lane *et al.*, 1995; 1999). At least three velocity measurements were taken at each location so as to assess the precision of the measurements. As with the waters edge measurements in order to contextualise these measurements in terms of the flow regime, the discharge was measured before the data were collected.

5.3.3.1. Waters edge validation

The observed waters edge and simulated waters edge were compared using a MATLAB programme. Comparing the accuracy and precision of the simulated waters edge is of interest as one of the key river characteristics which may change with the alteration of the compensation flows is the wetted area of the rivers. The amount of wetted area influences the amount of habitat available for both fish and invertebrates. Therefore, the ability of the model to predict the wetted area is of interest.

The principal of the validation technique is outlined in Figure 5.3.3. It shows that □x represents the lateral difference between the measured and predicted waters edge. The value of □x is calculated perpendicular to the flow direction for each measured water edge point. From each of these measurements, the mean error (bias/accuracy), and standard deviation of the error (precision) are calculated.

In the context of this study, accuracy is defined by the level of bias in a chosen ensemble of predictions. It is desirable for the bias to be zero. Measuring ensemble bias is insufficient as the same level of bias could result from elevation errors with very different variances. Hence the standard deviation of error is also calculated.

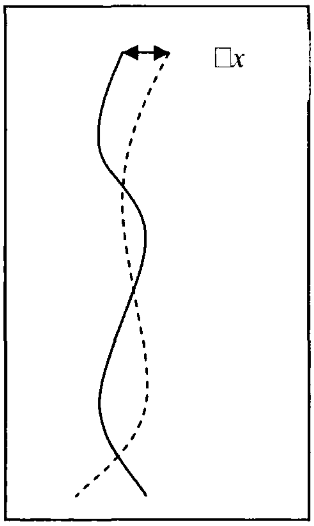


Figure 5.3.8. Diagram to illustrate the principal behind the horizontal waters edge validation technique. Solid line = measured; dotted line =predicted.

5.3.3.2. Point velocity and depth validation

Further validation was provided by comparing observed velocity and depth data with simulations with corresponding boundary conditions (again using the MATLAB programme). It is of interest to analyse the ability of FESWMS to predict depths and velocities and these are key order 1 characteristics (Figure 5.1.1.). It is the spatial distribution of velocities and depths which impact upon the fish and invertebrates and this is a strength of the 2D approach when compared to the 1D approach. Therefore comparing the 60 spatially separated values of velocity and depth will provide a valuable insight into model performance.

For each measured velocity, the error was defined as the observed velocity minus the predicted velocity. The mean error is calculated by averaging the error across the sixty observations, and an estimation of precision is provided by the standard deviation of the errors. The same procedure occurs with the point depth measurements.

5.4. Model calibration

As was explored in the literature review, there are two main parameters which can influence the predictions of a 2D finite element model such as FESWMS: roughness; and eddy viscosity. It is the purpose of this calibration section to find the best parameter set with respect to known datum. This calibration process also incorporates a sensitivity analysis (Crowder and Diplas, 2000b), in which the extent of roughness and eddy viscosity impacts are investigated. As explained above, two river reaches were selected on both the Rivelin and Loxley, with them being labelled upstream (U) and downstream (D) respectively. Two mesh resolutions were also used, 0.25m (1) and 0.15m (2). This design allows us to test the model sensitivity to parameterisation and mesh resolution, the latter of which is an important current debate within the modelling community (Hardy *et al.*, 1999; Crowder and Diplas, 2000b; Pasternak *et al.*, 2006). Therefore, the calibration of the model will be discussed for both of the mesh resolutions at each of the sites. As discussed in section 5.2, the model was calibrated with reference both to waters edge (for two discharges) and the point depth and velocity data (for one discharge). The Figures for this section can be found in Appendix A.

5.4.1. Rivelin upstream (RU)

Figures 5.4.1 and 5.4.2 (Appendix A) show that for the pre-change waters edge horizontal position the best accuracy attained was around 0.17±0.20m, with around 0.9±0.14cm attained with the post change data. Such accuracy values can be considered good as it would be difficult to get an accuracy better than the mesh resolution (0.25m). Both Figures also show that the main control on the simulation output was eddy viscosity, as a large vertical deviation of accuracy values (for a given Manning's roughness) can be seen in the Manning's roughness plots. Little such variation seen in the eddy viscosity plots indicating that for the same roughness values, many different simulation outputs can be produced by altering the eddy viscosity. In the pre-change data (Figure 5.4.1) a levelling out of the influence of eddy viscosity is seen in the left hand side of the plots, indicating that the level of bias may not improve whatever parameter set used. The post-change data shows that the accuracy starts to decrease at the very lowest eddy viscosities (Figure 5.4.2).

When the waters edge position calibration results for the 0.15m spacing mesh (RU-2) are examined it can be seen that the pre-change simulation calibrates to around 0.14±0.19m, and the post change simulation calibrates to around 0.5±0.11cm (Figures 5.4.3 and 5.4.4). Again, compared to the mesh resolution of 0.15m the accuracy is good. Again, it can be seen that eddy viscosity appears to exert a greater influence over the simulations than Manning's roughness, although not to the same extent as in the 0.25m mesh, which indicates that the increased spatial resolution is decreasing the parameter influence. Figures 5.4.3 and 5.4.4 also shows that again, the accuracy appears to worsen at the very lowest values of eddy viscosity and Manning's roughness. It should be noted that the optimum eddy viscosity value is lower on the finer mesh and the higher (post-change) discharge (Table 5.4.1). Both of these results are as expected. On the finer mesh, there will be better quantification of the spatial gradients of velocity so meaning that less energy needs to be represented through the turbulence stresses. As discharge increases, shear velocity will decrease, as the primary sensitivity to changing discharge will be changing depth until all roughness elements are submerged. This should also reduce eddy viscosity.

Figure 5.4.5 shows that for the point depth validation data at RU1, as the roughness and eddy viscosity values decrease, the accuracy of the simulations with respect to measured data increases. Despite this, the model is still over predicting depths to around 0.11±0.06m. What is also apparent from Figure 5.4.5 is that when eddy viscosities get below a value of one, despite the accuracy still improving, the standard deviation starts to increase. This same phenomenon can also be seen in the point velocity validation (Figure 5.4.6), but below eddy viscosities of around 0.1. Figure 5.4.6 also shows that the standard deviation for the point velocity validation is higher than that for the depth validation, and gives predictions of 0.06±0.15m/s.

Despite the higher resolution mesh at RU2, the predictions of point depths could still only be calibrated to around 0.12±0.06m. Figure 5.4.7 also shows that there appears to be little difference between the influence of Manning's roughness and eddy viscosity on the predictions. The predictions do however have standard deviation values which again appear to increase at the lowest values of Manning's roughness and eddy viscosity (as seen on the coarser mesh). Figure 5.4.8 shows that the velocity predictions have higher standard deviation values indicating more scatter in the relationship between simulated and observed velocities (0.05±0.15m/s). Again, in this denser mesh, there appeared to be little difference between the influence of Manning's roughness and eddy viscosity, and the precision dramatically worsened at the lowest values of roughness and eddy viscosity.

The above results show that eddy viscosity is the most important model parameter in all of the calibration tests apart form the point depth and velocity calibration at RU2. No significant impact of mesh resolution was found with either of the calibration methods (but the denser mesh may have produced more accurate predictions for waters edge). For both calibration methods, the best values of eddy viscosity are lower in the denser mesh than in the coarser mesh. This may be due to the increased spatial representation as discussed above, this picture is further enhanced by the lower eddy viscosity values in the higher discharge simulations.

Mesh: RU-1	**Manning's Roughness**	**Eddy Viscosity**	**Accuracy**	**Standard Deviation**
Pre waters edge	0.0005	0.55	0.167m	0.198m
Post waters edge	0.003	0.5	0.102m	0.150m
Point depths	0.0001	0.01	0.118m	0.061m
Point velocities	0.0001	0.01	-0.056m/s	0.153m/s
Mesh: RU-2				
Pre waters edge	0.0001	0.3	0.142m	0.192m
Post waters edge	0.0001	0.09	0.049m	0.111m
Point depths	0.0001	0.005	0.120m	0.062m
Point velocities	0.0001	0.005	-0.055m/s	0.157m/s

Table 5.4.1. Table to show the best parameter sets and the values of accuracy and precision predicted, for each of the validation tests at RU.

5.4.2. Rivelin downstream (RD)

Figure 5.4.9 (Appendix A) shows that for the pre-change data, the waters edge position could be calibrated to around 0.21±0.32m. There is a general increase in agreement between measured and observed data with the lowering of both Manning's roughness and eddy viscosity. At the very lowest values of Manning's roughness and eddy viscosity however, this relationship appears to break down, with more variation being present. This suggests that the level of agreement may not be able to be increased by further altering the parameter set, and so the difference may be purely due to other reasons, the mesh perhaps. Figure 5.4.10 shows that, for the post-change data, the model could be calibrated to around 0.17±0.23m. Compared to the mesh resolution of 0.25m these accuracy values of both the pre and post change horizontal waters edge calibration can be considered good.

For the denser mesh, Figures 5.4.11 and 5.4.12 show that the waters edge position for the pre-change data can be calibrated to around 0.25±0.32m and around 0.21±0.24m for the post-change data. However, it can be seen that altering the values of Manning's roughness and eddy viscosity does not appear to alter the agreement between predicted and observed to any great extent. Again, it appears that the eddy viscosity is more influential on the validation results than the Manning's roughness, but, the importance of eddy viscosity in influencing the model simulations appears to be reduced in the denser mesh compared with the coarser mesh. The finer mesh has worse accuracy and precision than the coarser mesh.

Figure 5.4.13 shows for the point depth validation data at RD1, that the model could be calibrated to 0±0.03cm. It again appears that eddy viscosity is having a greater influence on simulation output than Manning's roughness. The standard deviation appears to increase at the very lowest eddy viscosity values, with an inflexion being found at eddy viscosities of 0.4. The model could be calibrated to 0±0.23m/s for the point velocities. An inflexion point also occurs in the precision plot for the point velocity validation plots (Figure 5.4.14), again at an eddy viscosity value of around 0.4. The eddy viscosity again appeared more important than the manning roughness value in the calibration process.

Figure 5.4.15 shows that for the denser mesh (RD2) the point depth validation data can be predicted to 0±0.03m, with again eddy viscosity being more influential than Manning's roughness. The increasing standard deviation at the lowest eddy viscosity and Manning's roughness values can again be seen, with the inflexion point and at eddy viscosities of around 0.2. Figure 5.4.16 shows that again, the velocities are generally being under predicted (0±0.22m/s), with larger standard deviation values than the depth validation. A similar standard deviation inflexion point can again be seen at eddy viscosity values of around 0.4.

It was seen in the above section that altering the model parameters did not appear to make a great deal of difference to the waters edge calibration at either of the study discharges. When the point depth and velocity calibration is considered, eddy viscosity was once again the most important model parameter. An inflexion point in the standard deviation values was also observed, with the precision of the simulations decreasing at eddy viscosities of around 0.4 for both mesh resolutions. When the mesh resolutions were compared, the waters edge calibration was better for the finer mesh, but the calibration results for the point depth and velocities were similar. As with the upstream site on the Rivelin (RU) the 'best' eddy viscosities were higher in the coarser mesh than the finer mesh.

Mesh: RD-1	**Manning's Roughness**	**Eddy Viscosity**	**Accuracy**	**Standard Deviation**
Pre waters edge	0.0001	0.4	0.209m	0.319m
Post waters edge	0.001	0.78	0.176m	0.230m
Point depths	0.0005	0.2	0m	0.034m
point velocities	0.0001	0.105	0m/s	0.226m/s
Mesh: RD-2				
Pre waters edge	0.009	1.05	0.252m	0.327m
Post waters edge	0.001	0.87	0.218m	0.244m
Point depths	0.0005	0.13	0m	0.031m
Point velocities	0.0001	0.06	0m/s	0.221m/s

Table 5.4.2. Table to show the best parameter sets and the values of accuracy and precision predicted, for each of the validation tests at RD.

5.4.3. Loxley upstream (LU)

Figures 5.4.17 and 5.4.18 (Appendix A) show that at LU-1, altering the eddy viscosity and Manning's roughness did not appear to impact upon the waters edge validation results to any great extent, with the pre-change boundary conditions being calibrated to 0.33±0.49m and the post-change boundary conditions calibrated to 0.14±0.20m. Which, compared with the mesh resolution of 0.25m are not bad. What is also interesting is that in an opposite trend of each of the Rivelin sites, the accuracy appeared to decrease when the Manning's roughness and eddy viscosity values were lowered. Again, the eddy viscosity appears to have greater influence on the waters edge validation results than the Manning's roughness.

For the denser mesh, Figures 5.4.19 and 5.4.20 show that the pre-change simulations could be calibrated to around 0.47±0.55m whilst the post-change simulations could be calibrated to around 0.17±0.25m in terms of waters edge. It was seen that in both cases, a reduction in Manning's roughness and eddy viscosity lead to a better agreement between measured and predicted values; this is the opposite of what was seen in the coarser mesh. There was however, a slight decrease in accuracy at eddy viscosities of around 4, wich could account for the inverse relationship between mean difference and both eddy viscosity and Manning's roughness in LU-1. The eddy viscosities appeared again to have a greater influence than the Manning's roughness.

When the point depth validation data is considered for LU-1, the depth can be calibrated to around 0.01±0.05m (Figure 5.4.21). There again appears to be more influence of eddy viscosity than the Manning's roughness on the calibration results. There is no inflexion point in the standard deviation values as there was in the Rivelin. Perhaps this is due to the characteristics of the site, or because the eddy viscosities were unable to be lowered any lower than 1.8, and the inflexion points in the Rivelin were found at eddy viscosities of less than one. When the point velocities are examined it can be seen that the velocities could be calibrated to -0.08±0.18m/s (Figure 5.4.22). The greater influence of eddy viscosity is again seen (Figure 5.4.22). No inflexion point in the standard deviation values was observed.

Figure 5.4.23 shows that for the denser mesh (LU2) the depths can be calibrated to 0±0.05m. Figure 5.4.23 also shows no inflexion point in the standard deviation values for the point depth and velocity validation. The eddy viscosity is again more important than the Manning's roughness. Figure 5.4.24 shows that when the point velocity validation data is considered there is a lot more variation in the standard deviation values. With decreasing values of Manning's roughness and eddy viscosity the velocity could be calibrated to 0.06±0.19m/s.

For the first time a decrease a decrease in model accuracy was caused by reducing the values of the model parameters, however, this was only observed for one of the mesh resolutions (LU1). In agreement with the calibration results presented for the Rivelin, eddy viscosity was again the more important model parameter. What is interesting is that there did not appear to be an inflexion point in the standard deviation data (apart from maybe at LU2 using the point velocity). Table 5.4.3 shows that there is very little difference between the best calibration results between the two meshes, with perhaps the waters edge calibration being slightly better in the coarser mesh. Table 5.4.3 also shows that the best parameter sets have a lower eddy viscosity in the finer mesh than in the coarser mesh, and this again may be due to the increased spatial resolution (however no discharge effect could be seen).

Mesh: LU-1	**Manning's Roughness**	**Eddy Viscosity**	**Accuracy**	**Standard Deviation**
Pre waters edge	0.01	2.2	0.332m	0.490m
Post waters edge	0.03	2.75	0.143m	0.196m
Point depths	0.011	1.83	0.015m	0.053m
Point velocities	0.011	1.83	-0.076m/s	0.179m/s
Mesh: LU-2				
Pre waters edge	0.025	3.4	0.464m	0.546m
Post waters edge	0.02	2.15	0.169m	0.254m
Point depths	0.02	0.85	0.002m	0.046m
Point velocities	0.02	0.5	-0.058m/s	0.187m/s

Table 5.4.3. Table to show the best parameter sets and the values of accuracy and precision predicted, for each of the validation tests at LU.

5.4.4. Loxley downstream (LD)

Figures 5.4.25 and 5.4.26 (Appendix A) show that for both the pre change data altering the values of eddy viscosity and Manning's roughness appears to have little influence on the accuracy of the predictions. The accuracy is much degraded than that for any of the previous sites (pre change = 0.43±0.39m; post change = 0.49±0.53m). For the first time, the waters edge calibration results are much worse than the mesh resolution (0.25m). Altering the parameter set does appear to have a slightly greater influence in the post-change data, with eddy viscosity seemingly having a greater influence than Manning's roughness (especially in the post-change waters edge horizontal position calibration).

Figures 5.4.27 and 5.4.28 show that only one roughness value was used in the parameterisation, to ensure convergence of the model for LD2. It was difficult to reduce the water surface elevation by lowering the values of Manning's roughness and eddy viscosity. The model could be calibrated to 0.62±0.51m and 0.73±0.59m respectively for the pre and post change waters edge horizontal position. Again, the waters edge calibration results are much worse than the mesh resolution (0.15m). There also appears to be a large amount of scatter in the validation values compared with eddy viscosity, which further enhances the lack of influence of parameters on the wetted area of the simulation in this reach.

When the point depths are considered, it can be seen that the depth can be calibrated to around 0.15±0.07m (Figure 5.4.29) at LD-1, with the accuracy values levelling out at the lowest values of eddy viscosity and roughness. The eddy viscosity again appeared to exert a greater influence than the Manning's roughness. The point velocities can be calibrated to -0.06±0.18m/s using the LD-1 mesh, but with higher standard deviation values than for the depth predictions (Figure 5.4.30). Again, eddy viscosity is the most important parameter, but altering the parameters appears ineffective at this site.

Unfortunately, only one roughness value could be used due to simulation convergence issues for the point depth validation (Figure 5.4.31). The point depths could be calibrated to around 0.17±0.07m, by decreasing the eddy viscosity. Velocities are again under predicted, and again to around –0.07±0.17m/s. The precision values for the velocity validation are once again worse than for the point depth validation values (Figure 5.4.32). However, altering the model parameters appeared to have no real influence, only altering point depths by 2cm and point velocities by 1cm/s.

The results presented above show that altering the model parameters does not appear to impact upon the model simulations to any great extent. The worst calibration results are found on the denser mesh, and the two meshes have similar best values of model parameters for each of the calibration tests.

Mesh: LD-1	**Manning's Roughness**	**Eddy Viscosity**	**Accuracy**	**Standard deviation**
Pre waters edge	0.035	5.2	0.434m	0.386m
Post waters edge	0.026	2.77	0.487m	0.531m
Point depths	0.034	4.8	0.154m	0.068m
Point velocities	0.034	4.7	-0.068m/s	0.176m/s
Mesh: LD-2				
Pre waters edge	0.028	4.4	0.623m	0.513m
Post waters edge	0.03	5.25	0.732m	0.585m
Point depths	0.03	5.5	0.170m	0.075m
Point velocities	0.03	5.5	-0.073m/s	0.173m/s

Table 5.4.4. Table to show the best parameter sets and the values of accuracy and precision predicted, for each of the validation tests at LD.

5.4.5. Summary

Generally, apart from at the Rivelin upstream (RU) site, altering the Manning's roughness and eddy viscosity did not change the accuracy of the waters edge horizontal positions to any great extent. This perhaps indicates that the horizontal waters edge errors are controlled by mesh errors. The point depth and width calibration showed that the eddy viscosity was a much more important model parameter than Manning's roughness. This was evidenced by the greater vertical distribution of validation points at a given roughness compared with the eddy viscosity.

In general, the lower the parameter values were set, the better the accuracy of the simulation. In a number of cases, despite the level of agreement increasing in terms of accuracy, the standard deviation increased, creating an inflexion point. This was generally evident in the Rivelin sites and not the Loxley sites. This indicates an increasing amount of scatter in the relationship between predicted and observed data. This is important because not only is good accuracy required of model predictions, but they must also be precise.

In terms of general agreement (Table 5.4.5) it was generally relatively good. If the waters edge calibration results can be thought of as standardised by the average width of the river (for example; Rivelin 6m; Loxley 8m), then a 20m horizontal error on the Rivelin is of the order of 3.3%; and a 50cm error on the Loxley is of the order of 6.3%. In general, when compared to the mesh resolution, the model could predict horizontal waters edge relatively accurately. The Loxley generally has worse accuracies and precisions than the Rivelin in terms of horizontal waters edge position. The point depth validation data displayed good accuracies at two sites (RD and LU), with much worse accuracies found at the two remaining sites, but the precision values were consistent across each of the sites. The precision of the velocity predictions were much worse than those of the depth predictions, indicating that spatial variability was not included to a great enough extent during the survey process.

Relatively little difference can be seen between the calibration results for the two mesh resolutions used (0.25m and 0.15m). However, the eddy viscosities used in the best parameter set were generally lower in the more detailed mesh, indicating a greater inclusion of spatial variability within that mesh. However, Table 5.4.5 shows that the precision values for the velocity predictions are no better in the more detailed mesh. This indicates that spatial variation is included to a similar extent in both the meshes, and both meshes are subject to the same errors. For these upland Millstone Grit rivers, it appears that a mesh resolution of 0.25m produces similar results to that of a 0.15m spacing mesh.

At RU, the eddy viscosity values for the best parameter sets were lower in the simulations which had the highest discharge. This was attributed to the fact that as discharge increases, shear velocity will decrease, as the primary sensitivity to changing discharge will be changing depth until all roughness elements are submerged. This should decrease the required eddy viscosity. Therefore, it could perhaps have been expected for the Loxley sites to have lower values of Manning's roughness and eddy viscosity in their best parameter sets than the Rivelin, due to the greater discharge in the Loxley. However, this was not the case.

What this section has highlighted is the difference in calibration characteristics at each of the sites. Each site has different values of Manning's roughness and eddy viscosity in the best parameter sets, and so calibrating this depth-averaged 2D model for each study reach is crucial.

	Pre-change waters edge position	**Post-change waters edge position**	**Point Depth**	**Point Velocity**
RU-1	0.17±0.20m	0.09±0.14m	0.12±0.06m	-0.06±0.15m/s
RU-2	0.14±0.19m	0.05±0.11m	0.12±0.06m	-0.06±0.15m/s
RD-1	0.20±0.31m	0.17±0.23m	0±0.03m	0±0.22m/s
RD-2	0.25±0.33m	0.22±0.24m	0±0.03m	0±0.22m/s
LU-1	0.33±0.49m	0.14±0.20m	0.01±0.05m	-0.08±0.17m/s
LU-2	0.47±0.55m	0.17±0.25m	0±0.05m	-0.06±0.19m/s
LD-1	0.43±0.39m	0.49±0.53m	0.15±0.07m	-0.06±0.18m/s
LD-2	0.62±0.51m	0.73±0.58m	0.17±0.07m	-0.07±0.17m/s

Table 5.4.5. Table to show a summary of the 'best' calibration results.

5.5. Reliability analysis

Ultimately, model error as expressed in perimeter position, depth and velocity is less important than the implications of that error for habitat assessment. This section focuses on this propagation of error for the RU site. Therefore, depth and velocity cumulative frequency plots for a selection of Manning's roughness and eddy viscosity values were plotted. A cross-section of model parameter sets covering the complete range of parameter sets simulated was used. More simulations were conducted but the output not included in this analysis so as not to clutter the graphs. As it is the distribution of velocities and depths which is of interest, the boundary conditions used will be those used for the point depth and velocity calibration. An effort will be made to link any changes to potential brown trout habitat requirements (Table 5.5.1). For the purpose of comparing available good habitat, two parameter sets will be compared directly. The lowest parameter set (green line) provides the best calibration results with the point depth and velocity data. The second parameter set was chosen arbitrarily as it has higher values of Manning's roughness and eddy viscosity, so as to provide an illustration of the potential changes in habitat occurring from changing Manning's roughness and eddy viscosity.

	Values
Nursery (0+) Velocity	0.15-0.2m/s
Nursery (0+) depth	0..2-0.3m
Rearing (>0+(<20cm))Velocity	0.2-0.4m/s
Rearing (>0+(<20cm)) Depth	0.5-0.75m

Table 5.5.1. Table to show some selected habitat preferences of brown trout. These values are the 'good' habitat values for the fuzzy rules in Chapter 6.

In terms of predicted velocities all of the simulations have similar proportions of the slower flowing water (Figures 5.5.1 and 5.5.2). But Figures 5.5.1 and 5.5.2 show that there are greater differences in the proportions of faster flow present between the different parameter sets. It can be seen that the lower values of Manning's roughness and eddy viscosity leads to faster flow. This is seen for both mesh resolutions. Conversely, it can be seen that altering the model parameters leads to greater variation within the lower depths, with little difference found between the simulations for the higher depths. For the 0.25cm mesh, 70% of depths recorded were less than 0.6m for each of the simulations. In the 0.15cm mesh, a similar pattern is seen, apart from the simulation with the highest parameter values. It can be seen in both meshes that the once the parameter values have been lowered to a certain break point (n=0.0005;Vo=0.05), little difference in the distribution of depths is notable.

Table 5.5.2 shows that by decreasing Manning's roughness and eddy viscosity, thereby reducing depth and increasing velocity, the available velocity habitat for 0+ brown trout decreases (only at RU1), but the available good velocity habitat for >0+(<20cm) trout increases at both sites. Less impact can be observed in the depth distributions, with there being a slight increase in 0+ trout habitat availability with decreasing Manning's roughness and eddy viscosity, but no change in the >0+(<20cm) trout habitat.

	Mesh: RU1		**Mesh: RU2**	
	0.001, 0.2 (red)	0.0001;0.01 (lgreen)	0.001;0.7 (red)	0.00005; 0.005 (lgreen)
Velocity				
0+	21%	15%	15%	15%
>0+(<20cm)	16%	41%	3%	43%
Depth				
0+	13%	14%	8%	15%
>0+(<20cm)	51%	51%	51%	51%

Table 5.5.2.. Table to show the cumulative percentage change of velocities and depths for two of the age/size classes of brown trout.

Lowering the Manning's Roughness and eddy viscosity has the effect of decreasing depth and increasing velocity within the study reach. The differences in velocity predictions between the two chosen parameter sets was the key difference in terms of habitat impacts. Little impact was seen in the depth habitat available. As with the findings of the previous section, the influence of Manning's roughness and eddy viscosity on model predictions is such that careful attention must be paid to the calibration of the model at each of the study sites.

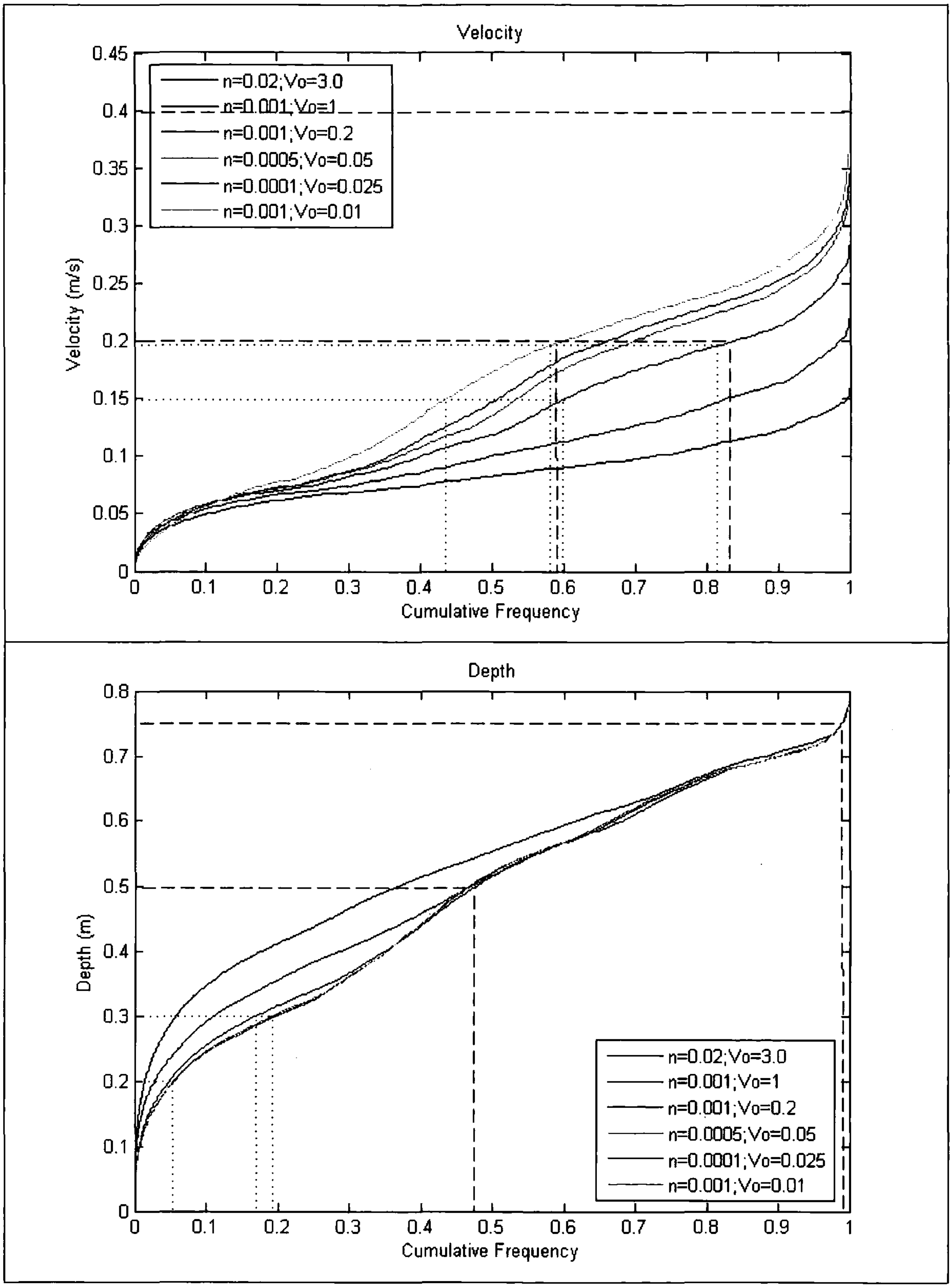


Figure 5.5.1. Figure to show the cumulative frequency distributions for RU-1 at various parameter combinations. Dotted line is 0+ habitat; dashed line >0+(<20cm).

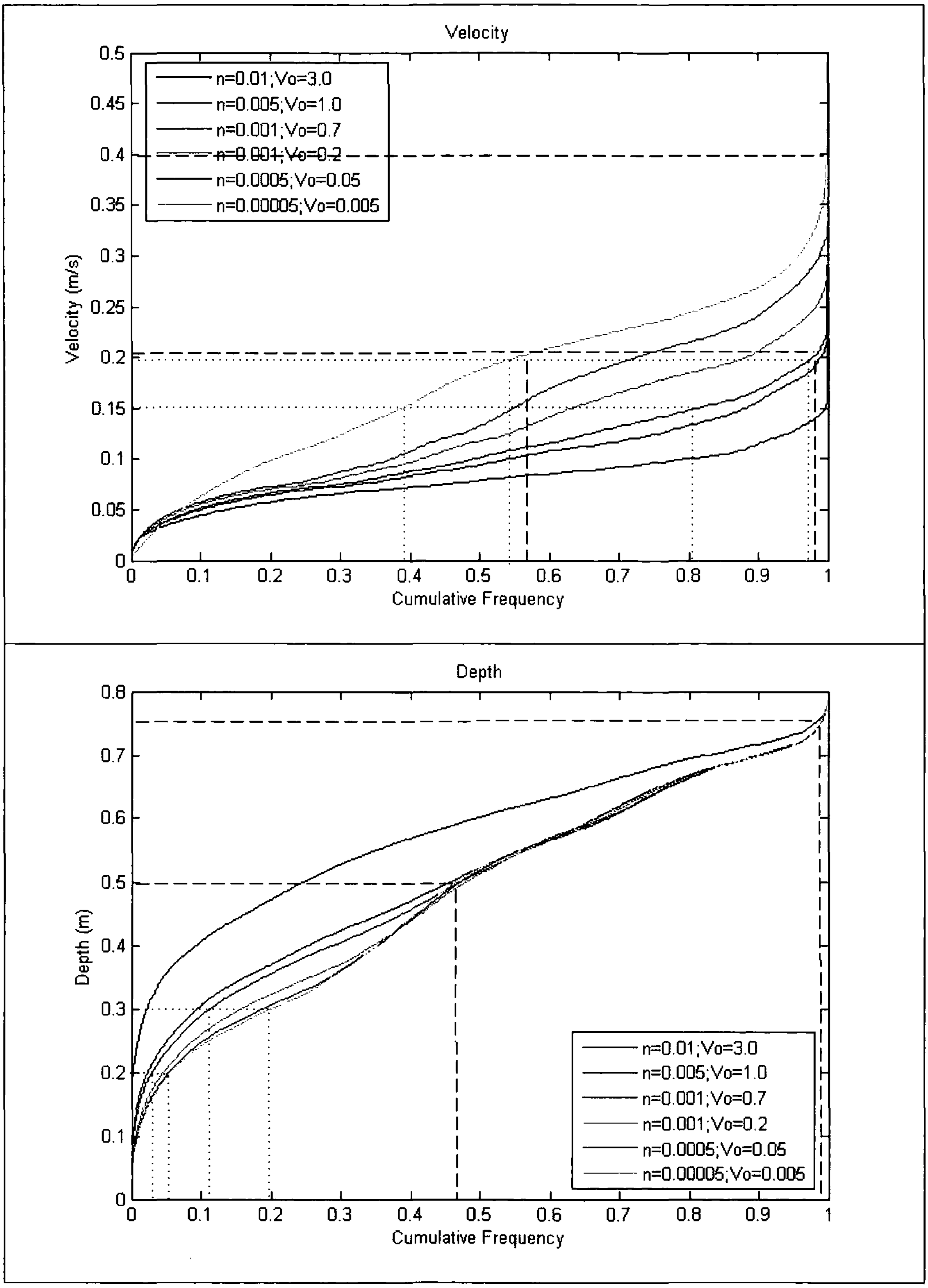


Figure 5.5.2. Figure to show the cumulative frequency distributions for RU-2 at various parameter combinations. Dotted line is 0+ habitat; dashed line >0+(<20cm).

5.6. The modelled impacts of altering compensation flows on the hydrodynamics of the rivers

This section analyses the potential impacts of altering the compensation flows on the hydrodynamics of the study streams. This will be done via steady state simulations comparing the pre-change and post-change compensation release levels. Steady state simulations were used for two reasons. Firstly, they are computationally efficient, allowing for the modelling of two sites per river and run simulations using multiple parameter sets. Secondly and most importantly, is the fact that these steady-state discharges are what can be controlled by the regulatory body (in this case Yorkshire Water). This is the only information we know for sure about the future flows within these rivers, as no flow variability is included in Yorkshire Water's releases. The amount of augmentation is highly variable and more importantly unpredictable and so is ignored from this analysis. The depth and velocity cumulative frequency graphs for the pre and post change compensation flows are compared. In order to provide further insight, the good habitat requirements (Table 5.5.1) for 0+ and >0+(<20cm) trout are mapped onto these cumulative frequency graphs.

5.6.1. Rivelin upstream (RU)

Table 5.6.1 shows the parameter values used that resulted in the most accurate validation results. The parameter sets which are of most interest are the ones which provided the best validation results for the point depth and velocity data, as in this section we are most interested in the distributions of depths and velocities rather than the wetted area of the river. Table 5.6.1 also shows the closest parameter sets available for the post-change simulations for RU-1 and RU-2. Unfortunately, the simulations for the pre-change conditions for RU-1 did not converge and as such, the distributions of depth and velocities cannot be compared.

Mesh: RU-1	**Manning's roughness**	**Eddy viscosity**
Best Point depths	0.0001	0.01
Best point vels	0.0001	0.01
Pre-change	N/A	N/A
Post-change	0.0001	0.05
Mesh: RU-2		
Best Point depths	0.0001	0.005
Best point vels	0.0001	0.005
Pre-change	0.0001	0.09
Post-change	0.0001	0.025

Table 5.6.1. Table to show a summary of the parameter sets which produce the best validation values for the various methods at RU-1 and RU-2.

Figure 5.6.1 shows that the post-change velocities at RU are higher than the pre-change velocities. There is a relatively constant difference between the pre and post change simulations until velocities of around 0.4m/s, but then there are more higher velocities found in the post-change data. Figure 5.6.1 also shows that the depths present at RU increase under the post-change discharge, with the greater depths appearing to increase in abundance more than the shallower depths (the sort of cone shape). In terms of how this relates to fish habitat, it can be seen that there is an increase in 0+ and >0+(<20cm) good velocity habitat with the post-change data (Table 5.6.2). There is a slight increase in the good depth habitat available for 0+ trout with the new discharge. The entire reach has depths of less than 50cm under both flow scenarios, and hence the percentage of good depth habitat for >0+(<20cm) trout is zero. This increase in good velocity habitat availability for >0+(<20cm) trout tallies with the faster growth rates for that age size class in 2005 in the Rivelin.

	Mesh:RU1		**Mesh: RU2**	
	Pre-change	**Post-change**	**Pre-change**	**Post-change**
Velocity				
0+	N/A	N/A	13%	14%
>0+(<20cm)	N/A	N/A	20%	25%
Depth				
0+	N/A	N/A	22%	29%
>0+(<20cm)	N/A	N/A	0%	0%

Table 5.6.2. Table to show the percentage depth and velocity habitat availability for 0+ and >0+(<20cm) trout for the pre and post change conditions at RU.

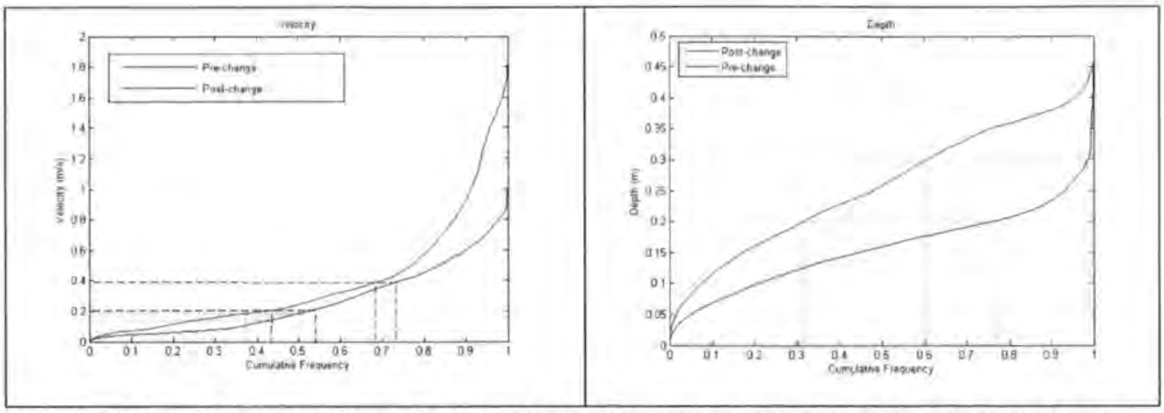


Figure 5.6.1. Figure to show the cumulative frequency output comparing the best pre- and post change simulations for RU-2. Dotted line is 0+ habitat; dashed line >0+(<20cm).

5.6.2. Rivelin downstream (RD)

Table 5.6.3 shows that the best values of roughness and eddy viscosity are different when validating the point depths and point velocities respectively for both mesh resolutions at RD. Table 5.6.3 also shows the nearest parameter sets for the pre- and post change simulations.

Mesh: RD-1	Manning's Roughness	Eddy viscosity
Best Point depths	0.0005	0.2
Best point vels	0.0001	0.105
Pre-change	0.0005	1.35
Post-change	0.0005	0.52
Mesh: RD-2		
Best Point depths	0.0005	0.13
Best point vels	0.0001	0.06
Pre-change	0.005	0.61
Post-change	0.0001	0.28

Table 5.6.3. Table to show a summary of the parameter sets which produce the best validation values for the various methods at RD-1 and RD-2.

Figure 5.6.2 shows that the pre-change values of both depth and velocity appear to be lower than the post-change data. Figure 5.6.2 also shows that generally the change in distribution of depths and velocities is very similar for both mesh resolutions. There also appears to be a larger difference between the pre- and post change output in terms of velocity distributions than the changes in depth. As at the RU site, the difference in velocities is greatest at the higher velocities. The denser mesh does appear to produce a slightly greater difference in depths between the pre and post-change simulations than the coarser mesh. For the pre-change conditions, only around 3-8% of the velocities available were good for >0+(<20cm) trout, whereas around 14-16% is suitable in the post-change simulations (Table 5.6.4). This again may be linked to the faster growth rates found in 2005 in the Rivelin. There is an increase in 0+ good velocity habitat for both mesh resolutions of around 10%, when the pre and post change simulations are compared. In terms of good depth habitat, there is a more good depth habitat available for 0+ brown trout with the post change compensation flow. Under both the pre and post change conditions, all depths were less than 50cm.

	Mesh: RD1		**Mesh: RD2**	
	Pre-change	**Post-change**	**Pre-change**	**Post-change**
Velocity				
0+	6%	16%	4%	15%
>0+(<20cm)	3%	14%	8%	16%
Depth				
0+	30%	38%	23%	36%
>0+(<20cm)	0%	0%	0%	0%

Table 5.6.4. Table to show the percentage depth and velocity habitat availability for 0+ and >0+(<20cm) trout for the pre and post change conditions at RD.

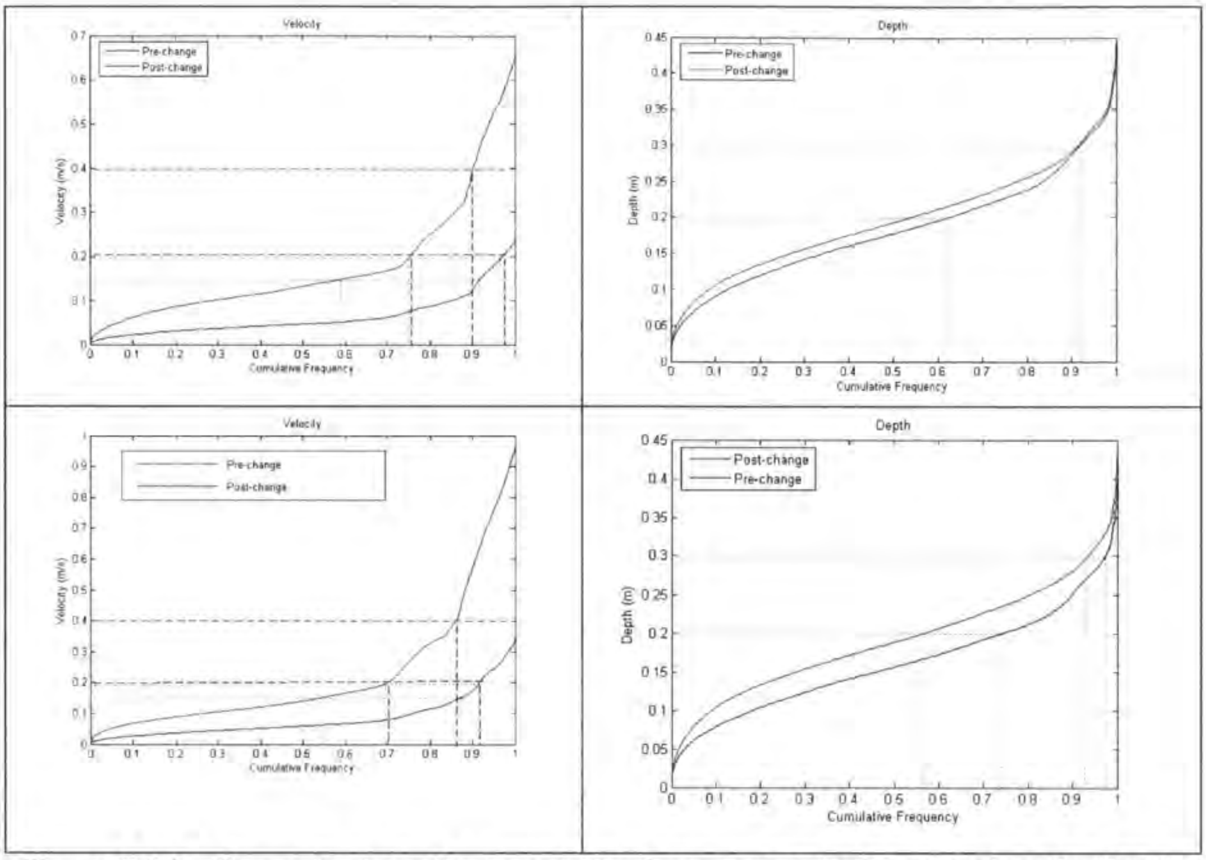


Figure 5.6.2. Figure to show the cumulative frequency output comparing the 'best' pre-and post change simulations for RD-1 and RD-2. RD-1 are the top 2. Dotted line is 0+ habitat; dashed line >0+(<20cm).

5.6.3. Loxley upstream (LU)

Table 5.6.5 shows the parameter sets used in the simulations which produced the best validation results. Table 5.6.5 also shows that the closest parameter sets for the pre and post-change conditions are relatively close for LU-1 but slightly different for LU-2.

Mesh: LU-1	Manning's Roughness	Eddy viscosity
Best Point depths	0.011	1.83
Best point vels	0.011	1.83
Pre-change	0.005	1.8
Post-change	0.01	2.21
Mesh: LU-2		
Best Point depths	0.02	0.85
Best point vels	0.02	0.5
Pre-change	0.015	2.75
Post-change	0.02	1.6

Table 5.6.5. Table to show a summary of the parameter sets which produce the best validation values for the various methods at LU-1 and LU-2.

Figure 5.6.3 shows that there appears to be much less difference between the pre and post change conditions than seen in the two Rivelin sites. This could be due to the fact that the change in discharge is proportionally smaller in the Loxley compared to the Rivelin. The greatest amount of difference was seen between the pre-and post change velocities at LU-2, where the greater spatial detail may be leading to a more accurate response of velocity to a change in discharge and water surface elevation (again more noticeable in the faster velocities). Table 5.6.6 shows no change in the availability of good velocity habitat for 0+ brown trout, but perhaps a slight increase in good velocity habitat availability for >0+(<20cm) trout. No changes in habitat availability could be observed in the depths.

	Mesh: LU1		Mesh: LU2	
	Pre-change	**Post-change**	**Pre-change**	**Post-change**
Velocity				
0+	10%	8%	9%	7%
>0+(<20cm)	41%	44%	43%	48%
Depth				
0+	14%	14%	14%	12%
>0+(<20cm)	32%	32%	32%	31%

Table 5.6.6. Table to show the percentage depth and velocity habitat availability for 0+ and >0+(<20cm) trout for the pre and post change conditions at LU.

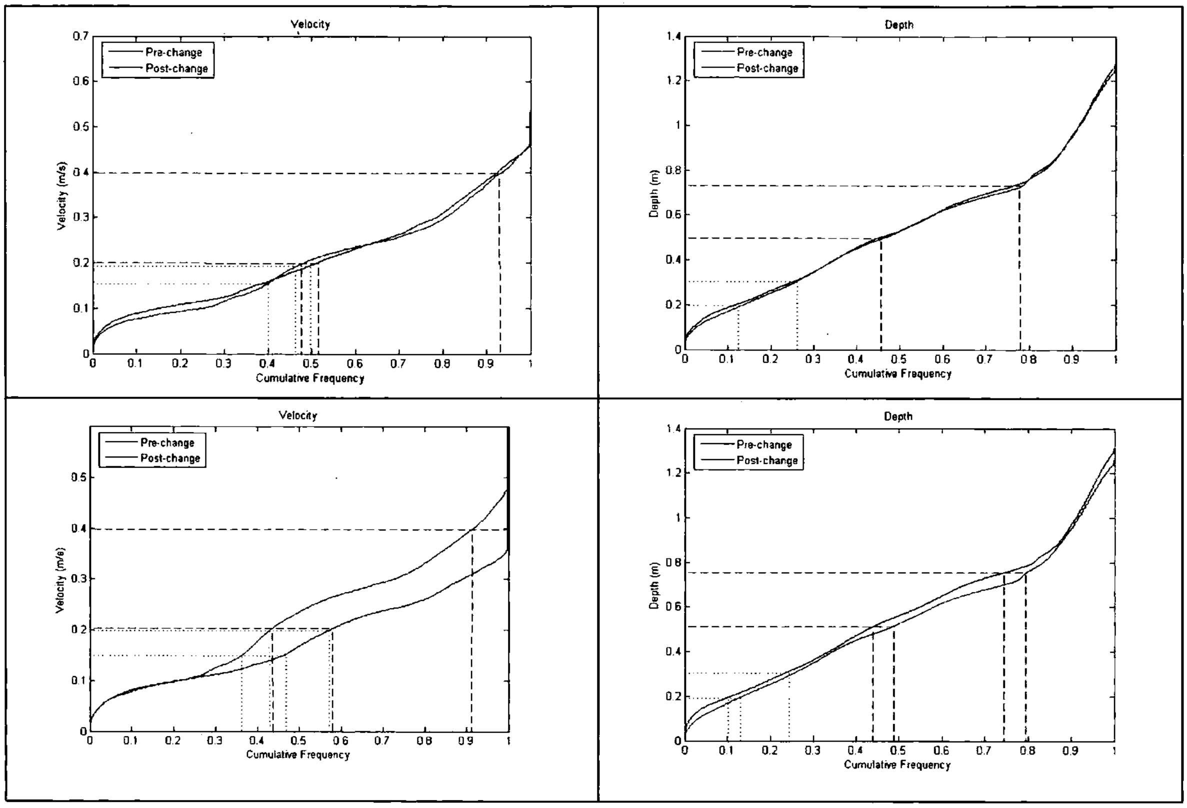


Figure 5.6.3. Figure to show the cumulative frequency output comparing the best pre- and post change simulations for LU-1 and LU-2. LU-1 at top. Dotted line is 0+ habitat; dashed line >0+(<20cm).

5.6.4. Loxley downstream (LD)

Table 5.6.7 shows the best parameter set for both the point depth and velocity validation were the same for LD-1. Section 5.4 did highlight the potential errors in calibration of this model and in particular, this reach and boundary conditions. Unfortunately, due to unknown reasons, the author was unable to get both the pre and post change simulations to converge for LD-2.

Mesh: LD-1	Manning's Roughness	Eddy viscosity
Best Point depths	0.034	4.8
Best point vels	0.034	4.7
Pre-change	0.035	4.93
Post-change	0.025	5.4
Mesh: LD-2		
Best Point depths	0.03	5.5
Best point vels	0.03	5.5
Pre-change	N/A	N/A
Post-change	N/A	N/A

Table 5.6.7. Table to show a summary of the parameter sets which produce the best validation values for the various methods at LD-1 and LD2.

As stated above, Figure 5.6.4 only shows the results from LD-1. Figure 5.6.4 shows that again, the velocities appear to become lower with a reduction in discharge. This leads to no change in the availability of good 0+ trout habitat, but a decrease in good habitat for >0+(<20cm) trout. However, there appears to be very little change in the depth distribution, and this is equated to no depth habitat change for either age/size class.

	Mesh: LD1		Mesh: LD2	
	Pre-change	**Post-change**	**Pre-change**	**Post-change**
Velocity				
0+	21%	21%	N/A	N/A
>0+(<20cm)	29%	11%	N/A	N/A
Depth				
0+	41%	41%	N/A	N/A
>0+(<20cm)	20%	20%	N/A	N/A

Table 5.6.8. Table to show the percentage depth and velocity habitat availability for 0+ and >0+(<20cm) trout for the pre and post change conditions at LD.

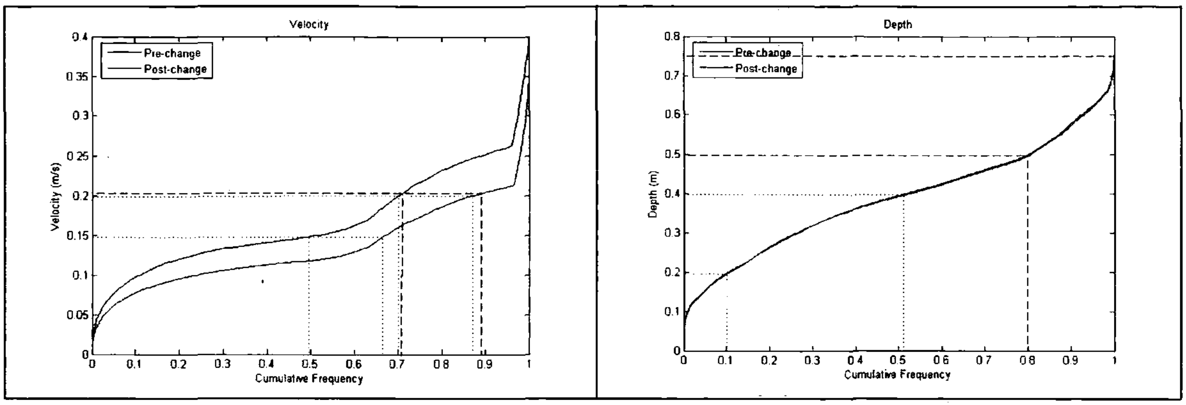


Figure 5.6.4. Figure to show the cumulative frequency output comparing the best pre- and post change simulations for LD-1. Dotted line is 0+ habitat; dashed line >0+(<20cm).

5.6.5. Summary

When the pre and post change discharges on the Rivelin were considered, the post-change discharge predicted higher depths and velocities than the pre-change discharge. The alterations in depth distributions were generally not very large, but greater difference was seen in the velocity distributions (especially at higher velocities). This is the opposite of what is expected in roughness dominated rivers such as these, as was discussed in Chapter 3. This resulted in increased good habitat availability for both 0+ and >0+(<20cm) trout, especially in terms of velocity habitat. At both sites, all depths were less than 50cm, which meant that there was no good depth habitat for >0+(<20cm) trout for either of the compensation flow scenarios, but there were >0+(<20cm) trout nearby as seen in Chapter 4. This highlights the importance of less than perfect habitat when modelling habitat availability.

Due to the proportionally smaller change in discharge, the alterations in the distribution of depths and velocities were smaller on the Loxley than on the Rivelin. The impact of altering compensation flows on the depth in both the study sites was minimal, but the impact on the distribution of velocities was larger and more confused. At Loxley upstream (LU), there appeared to be an increase in velocity under the post-change flows, which led to an increase in the availability of >0+(<20cm) trout habitat. At the Loxley downstream site, there appeared to be a decrease in velocity under the post-change flows, which led to a decrease in >0+(<20cm) trout habitat availability. This result indicates the importance of including reach-scale morphology in the modelling methodology, and the virtue of having two modelling sites per river in order to further capture the spatial variation.

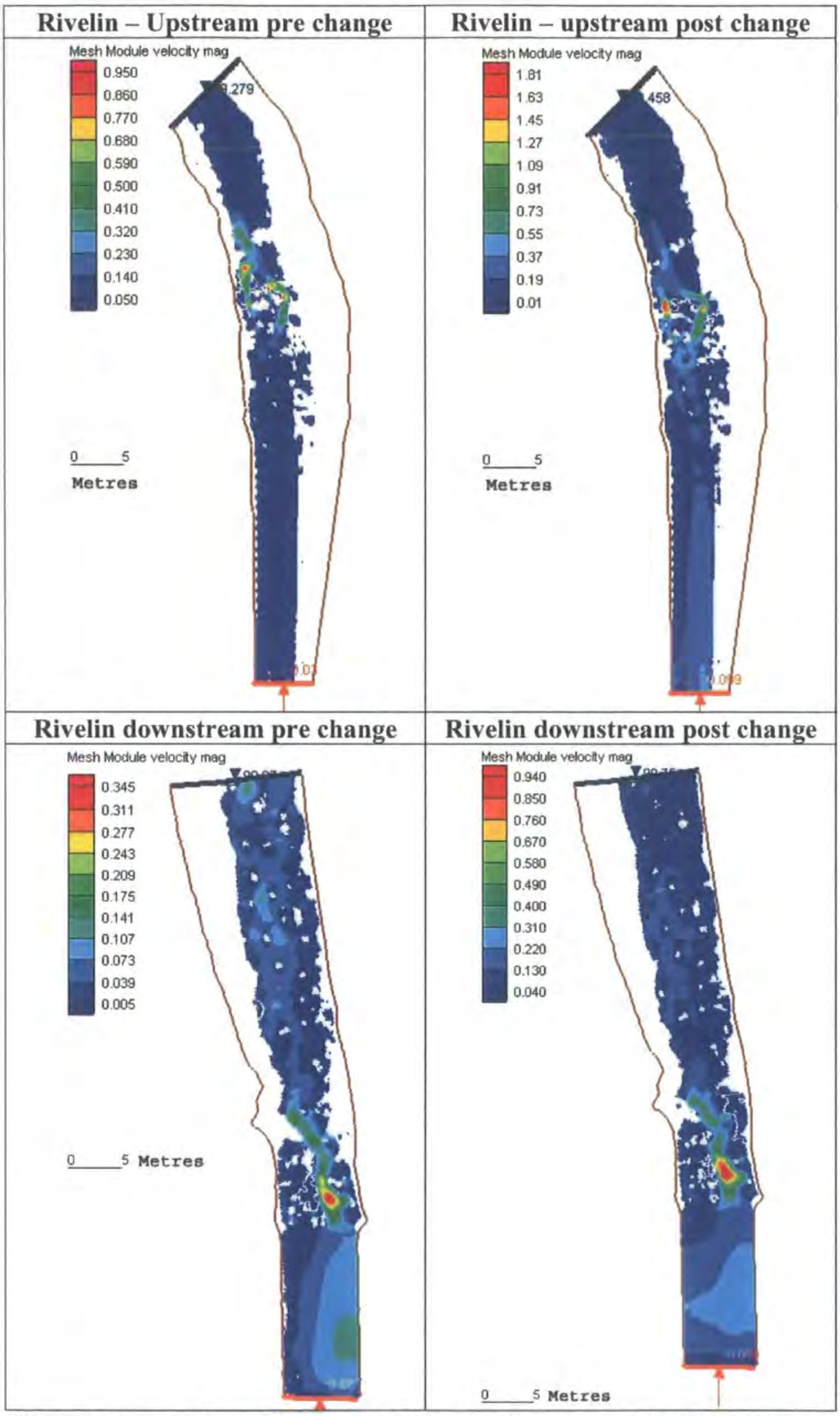
5.7. Discussion and conclusions

The aims of this chapter were as follows: i) to assess whether a 2D finite element model can accurately predict the hydrodynamics of an upland Millstone Grit stream; ii) to explore the primary controls on predicted hydrodynamics, including mesh resolution and model parameters; and ii) to assess the primary impacts of altering the compensation flows. It is the purpose of this section to work through each of the aims in turn and to assess both the effectiveness of the work in evaluating the aims, and link the work produced back to the existing literature.

5.7.1. Assessing whether a 2D finite element model can accurately predict the hydrodynamics of an upland Millstone Grit stream

Chapters 3 and 4 noted the importance of both spatial heterogeneity and reach-scale morphology in terms of affecting the habitat availability for both invertebrates and brown trout. As was outlined in section 5.3 the use of one dimensional models would not have allowed this to have been accounted for in the hydraulic modelling. Longitudinal reach-scale morphology would have been included in the output if a 1D model was used, but it would not have been able to represent the bar-riffle-pool topography nor the associated lateral local variation in flow (Lane and Ferguson, 2005). Leclerc *et al.* (1995) note that a PHABSIM type treatment will not produce reliable results for river areas <10m^2, and as the study rivers are <10m wide, this is undesirable.

Figure 5.7.1 shows that for both pre and post-change simulations for each of the study reaches there is a large amount spatial heterogeneity and influence of reach-scale morphology. The influence of boulders on the flow pattern is greater in the Rivelin than in the Loxley, and the influence of boulders appears to decrease with the increase in discharge in the Rivelin. This agrees with the findings of Crowder and Diplas (2000a, 2000b, 2002) in highlighting the importance of boulders as providers of spatial heterogeneity. At the Loxley upstream modelling site, there is perhaps an indication of boulders becoming more significant in the lower discharge (post change) simulation, by the evidence of patchy areas of higher velocities, but none of these boulders break the water surface. What is also apparent is that the Rivelin, with its lower discharge, appears to have a greater amount of spatial heterogeneity than the Loxley, where again, it can perhaps be thought of as being homogenised by the greater discharge.



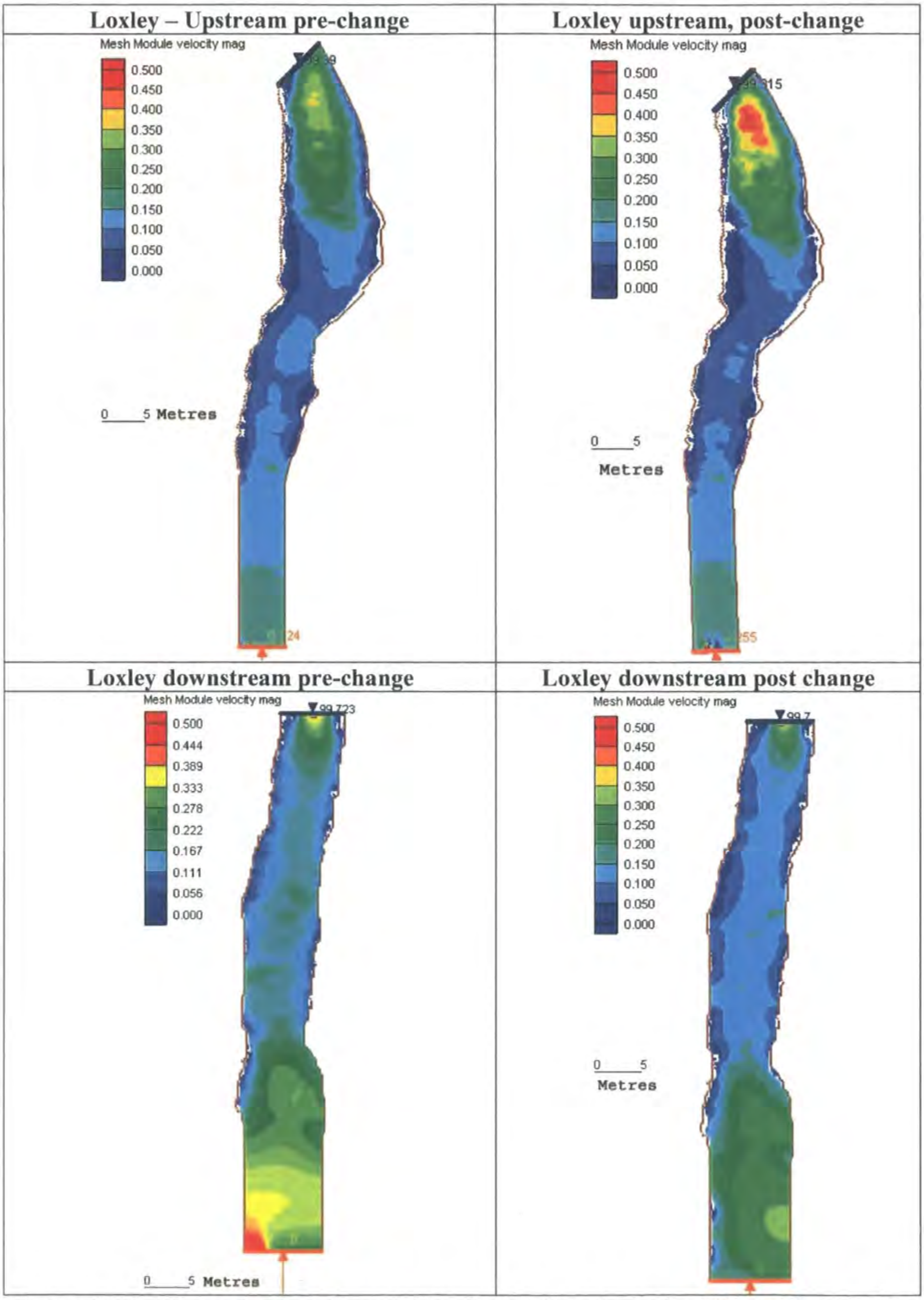
Rivelin – Upstream pre change

Rivelin – upstream post change

Rivelin downstream pre change

Rivelin downstream post change

Figure 5.7.1. Figure to show the distributions of velocity in each of the Rivelin reaches for the post-change conditions (RU2; RD2).



Loxley – Upstream pre-change

Loxley upstream, post-change

Loxley downstream pre-change

Loxley downstream post change

Figure 5.7.2. Figure to show the velocity distributions in each of the Loxley reaches for the post-change conditions.

In the calibration process, the lower the parameter values used, the better the accuracy of agreement found between predicted and observed data (for both waters edge and point depth and velocity). In a number of cases, despite the level of agreement increasing in terms of accuracy, the standard deviation increased, creating an inflexion point. This was generally evident in the Rivelin sites and not the Loxley sites. This indicates an increasing amount of scatter in the relationship between predicted and observed data. This is important because it is both the accuracy and precision of a model simulation which is crucial to establishing how model predictions may be used. Good precision and accuracy is the ideal case (Lane *et al.*, 2005). If the precision is good but there is bias, then it is likely that the patterns of model predictions are good, but their quantitative magnitudes are not (Lane *et al.*, 2005). If precision is poor, the model is less likely to be of use, even if the accuracy is good, as it implies substantial local uncertainty.

In terms of general agreement (Table 5.7.1) it was generally good. If the waters edge calibration results can be thought of as standardised by the average width of the river (for example; Rivelin 6m; Loxley 8m), then a 20cm horizontal error on the Rivelin is of the order of 3.3%; and a 50cm error on the Loxley is of the order of 6.3%. When compared to the mesh resolutions used in this study, the model predictions of horizontal waters edge are generally good. The Loxley generally has worse accuracies and precisions than the Rivelin in terms of horizontal waters edge position. The point depth validation data displayed good accuracies at two sites (RD and LU), with much worse accuracies found at the two remaining sites, but good precision values were consistent across each of the sites. The precision of the velocity predictions were much worse than those of the depth predictions, and there are a few possible reasons for this.

Both Lane *et al.* (1999) and Nicholas and Smith (1999) point out that high levels of unexplained variance may be inevitable in some gravel bed rivers with high relative roughness where it may be difficult to apply sufficiently detailed topographic information to the model. This may limit the models ability to resolve flow around individual grains and grain clusters that have a significant effect on the flow velocity distribution. If the model topographic boundary condition was smoothed (as in this study), the small-scale roughness was parameterised using a roughness coefficient (Lane *et al.*, 1999). However, the empirical data used to assess the models ability to predict flow in a junction zone are 'contaminated' by the actual roughness effects, and accordingly there remain fundamental limits to the ability of the data to provide an adequate basis for model validation (Lane *et al.*, 1999). Lane *et al.* (1999) also demonstrate how some of the poor correspondence between model predictions and field observations of two-dimensional flow patterns of depth averaged velocity can be attributed to problems of accurately orientating an electromagnetic current meter into the flow.

However, in this work, we have taken validation to mean the comparison of predicted and observed data but, Lane and Richards (2001) suggest that model validation must involve more than simply comparing model predictions with independent empirical data as a model is normally closed to create a system that is fundamentally different to that which has been measured. For the purposes of ecological modelling, it may be that the approach of comparing predicted values to observed values may be the correct one, as it will be the errors in the wetted area, depth and velocity predictions which will propagate errors through the ecological modelling.

Pasternack *et al.* (2006) mused as to what constituted a validated model, but decided that their model was validated. In the case of ecological modelling, as long as the errors in wetted area, velocity and depth predictions can be quantified, the model can be called 'valid'. As once the errors are quantified they can be taken into account when the ecological modelling is conducted. The above section shows that calibration is essential to this process, and calibration must be conducted separately for each of the study reaches.

	Pre-change waters edge position	**Post-change waters edge position**	**Point Depth**	**Point Velocity**
RU-1	0.17±0.20m	0.09±0.14m	0.12±0.06m	-0.06±0.15m/s
RU-2	0.14±0.19m	0.05±0.11m	0.12±0.06m	-0.06±0.15m/s
RD-1	0.20±0.31m	0.17±0.23m	0±0.03m	0±0.22m/s
RD-2	0.25±0.33m	0.22±0.24m	0±0.03m	0±0.22m/s
LU-1	0.33±0.49m	0.14±0.20m	0.01±0.05m	-0.08±0.17m/s
LU-2	0.47±0.55m	0.17±0.25m	0±0.05m	-0.06±0.19m/s
LD-1	0.43±0.39m	0.49±0.53m	0.15±0.07m	-0.06±0.18m/s
LD-2	0.62±0.51m	0.73±0.58m	0.17±0.07m	-0.07±0.17m/s

Table 5.7.1. Table to show a summary of the best calibration results for each of the meshes.

5.7.2. Exploring the primary controls on predicted hydrodynamics

Two main points about mesh resolution came out of the model calibration presented in Section 5.4. First, little difference can be seen in the calibration results from the two mesh resolutions. Second, the eddy viscosities used in the best parameter set were generally lower in the more detailed mesh, indicating a greater inclusion of spatial variability within the more detailed mesh. However, Table 5.4.5 shows that the precision values for the velocity predictions are no better in the more detailed mesh. This indicates that for both the 0.15m and 0.25m mesh, spatial variability was not included to a great enough extent. This is probably due to the surveying.

Table 5.7.2 shows the number of survey points recorded and the number of nodes present in each mesh. This comparison is somewhat skewed as the mesh density remained constant on the banks, but the survey density was more concentrated on within channel features. However it does serve to illustrate how much more dense the meshes were than the survey data recorded. This is evidenced in the standard deviation values for the point velocity validation, where no difference is seen between the two mesh resolutions. The high precision values are due to the lack of spatial variation in the mesh, and it can be seen that the source of this lack of spatial variation is the topographic survey.

If the least dense mesh resolution is already at a greater spatial scale than the survey conducted, this means that increasing the mesh density from there will not be enhancing the topographic picture at all, but just producing topography with facets of the mesh generation algorithm. Lane *et al.* (1999) note that as mesh resolution is increased there is the progressive introduction of geometric variability that is associated not with the bed topography but the way in which it has been sampled.

Site	No. Data Points surveyed	No of nodes in 0.25m mesh	No of nodes in 0.15m mesh
RU	3567	26597	73041
RD	3123	28915	80641
LU	4298	46750	82128
LD	3033	25114	73149

Table 5.7.2. Table to show the number of data points surveyed for each river and the number of nodes present in each of the meshes.

Although Hardy *et al.* (1999) suggested the use of four mesh resolutions in modelling studies, only two were used in this study due to limits of both computational power and time. Judging from the results of the model calibration for the two mesh resolutions, it would probably not be interesting to run any simulations on a more detailed mesh, having already established the fact that we have been constrained by the limitations of the topographic survey. It may be interesting to explore the potential of coarser meshes, and the extent to which they can accurately simulate flow properties, as this will also have the effect of decreasing the computational time.

Section 5.4 incorporated a sensitivity analysis of each of the meshes and their response to alterations of Manning's roughness and eddy viscosity. It showed that generally, a reduction in the values of Manning's roughness and eddy viscosity produced a more accurate agreement between the measured and predicted data. The eddy viscosity appeared to be the most important model parameter.

In some cases, lowering the eddy viscosity and Manning's roughness to its very lowest lead to a levelling out or a reduction in the accuracy, thus indicating a mesh controlled error. Further evidence of the control of mesh errors on the model simulations was provided by the waters edge horizontal position calibration. This showed that for three out of the four modelling sites, altering the model parameters appeared to have little effect, whilst having an effect on the point velocity and depth calibration. This perhaps indicates that the errors seen in the waters edge horizontal position calibration are due to mesh errors. This agrees with the findings of Pasternack *et al.* (2006) who found that that most of the errors produced by their use of FESWMS were due to mesh errors. Furthermore, there was often an increase in standard deviation values at the very lowest of eddy viscosity and Manning's roughness. This was seen at all sites apart from LU. This means that despite the increase in accuracy, there is more scatter in the relationship between the measured and predicted depths and velocities.

The effects of altering the model parameters was only looked at in terms of the distribution of depths and velocities at one of the sites for the purposes of expediency. It showed that the main areas of influence of altering the parameter sets were areas of low depth, and high velocity. It was seen that the potential influence of altering the parameter set were probably going to be confined to the velocity predictions rather than the depth predictions.

5.7.3. Assessing the primary impacts of altering the compensation flows

When the pre and post change discharges on the Rivelin were considered, it was seen that the post-change discharge produced higher depths and velocities than the pre-change discharge. The alterations in depth distributions were generally not very large, but greater differences were found in the velocity distributions (especially at higher velocities). This generally led to an increase in both 0+ and>0+(<20cm) trout habitat. Less change in the velocity and depth distributions was seen in the Loxley, due to the proportionally smaller change in discharge, with the area of greatest change being a decrease in higher velocities with the post-change discharge. In both cases mesh resolution had little impact on the pre and post change distributions of depths and velocities.

It was seen in Section 5.6 that both Rivelin sites had no habitat available in terms of good depth habitat for >0+(<20cm) trout. However, we know that trout within that age/size class live nearby, thanks to the fisheries surveys. This highlights the importance of less than perfect habitat to brown trout, and enhances the need for a number of fuzzy classes and the combination of depth and velocity, in order to properly assess habitat availability of organisms. To look at the distributions of depth and velocity in a given reach and relate them to known brown trout habitat preferences is interesting, but this means that the depths and velocities are treated independently. One of the areas with supposedly good velocities could have an inappropriate depth, or vice versa, it is therefore of interest to judge the habitat suitability of a point in terms of a combination its velocity and depth and this is what will be addressed in the next chapter.

5.7.4. Final conclusions

The importance of the two-dimensional approach in incorporating spatial variation has been highlighted by this chapter, especially in the Rivelin. The hydraulic models could be calibrated successfully, but must be conducted in each of the study reaches separately, which has implications for model transferability. Eddy viscosity was seen to be the most important model parameter. It appeared that most of the errors within the calibration of the model stemmed from the topographical survey. As both of the mesh resolutions used were of a higher resolution than the topographic survey and so using a more dense mesh brought no new topographical information. Because of this, the distributions of velocities and depths predicted by the hydraulic model were very similar for both mesh resolutions.

Chapter 6

Linking hydrodynamics to ecology: a fuzzy logic approach

6.1. Introduction

Chapters 3 and 4 highlighted the importance of hydraulics, hydraulic variability, spatial variation and reach scale morphology for invertebrate and fish populations respectively. Chapter 5 showed that each of these can be accounted for to some extent using a depth averaged two-dimensional (2D) hydraulic model. It is the purpose of this chapter to try and link the output of the 2D model to ecology. It was seen in the previous chapter that velocities and depths are crucial in determining theoretical habitat availability. By using cumulative frequency plots of velocity and depth, inferences about habitat availability could be made. Since the advent of hydraulic models in instream methodologies, a key challenge has been to directly link hydraulic predictions to ecology. Numerous methods of doing so have been developed, and each of them will be reviewed in Section 6.2.

This study uses the concept of fuzzy logic to link the hydraulic predictions produced by FESMWS to instream habitat availability. The methodology will be discussed in detail in section 6.3. Unlike the only previous use of fuzzy logic in instream habitat modelling (Jorde *et al.*, 2001; Schneider and Jorde, 2003) fuzzy rules are created for macroinvertebrates as well as brown trout.

One of the main themes running through this thesis has been one of variability and uncertainty in both ecological and modelling data. The results presented in this chapter are no different. Although the most interesting aspect of the output from this section will be comparing pre and post change habitat availability (and comparing that with the measured ecology), it is also important to assess the susceptibility of the modelling output to variations in model factors such as mesh resolution, and parameter sets.

Therefore, there are three aims for this chapter:

1) to assess the sensitivity of the fuzzy rules to decisions made during model development (such as mesh resolution and parameter sets);
2) to assess the pre-change habitat predictions for brown trout and macroinvertebrates; and
3) to compare the pre and post change habitat predictions for brown trout and macroinvertebrates.

The first section of this chapter will review the literature pertaining to the modelling of habitat based on hydraulic predictions. The second section will identify, justify and finally explain in detail the fuzzy approach taken in this chapter. The first of the results sections will address the sensitivity of the fuzzy rules to alterations in the predicted velocities and depths. The next two sections will address the predicted habitat available for both macroinvertebrate guilds and brown trout, in terms of the pre and post change discharges, whilst abstractly validating (or otherwise) the model with respect to the results presented in Chapters 3 and 4 respectively (Figure 1.2). A final chapter will discuss the results presented and provide links to the literature presented in Section 6.2.

6.2. Literature review

The previous chapter highlighted both the potential of 2D models in predicting the hydrodynamics of upland Millstone Grit rivers, and also the potential of that hydrodynamic information to be ecologically relevant. There have been a number of methods used to link hydraulic predictions to ecology in the form of habitat predictions. It is the purpose of this section to introduce and critique the various approaches taken.

6.2.1. Theory of the different approaches

6.2.1.1. Habitat Suitability Curves

Habitat Suitability Criteria (HSC) are generally used in habitat modelling to translate structural and hydraulic characteristics of streams into indices (Habitat Suitability Indices, HSIs) of habitat quality. This is the most commonly used approach, usually within the PHABSIM software. There are a number of categories of habitat suitability criteria, which are defined by using the type of information and data treatment used to generate the criteria. Criteria that are derived from personal experience and professional opinion or from negotiated definitions are defined as **Category I**. These criteria can be developed quickly and at minimal cost, compared to the more data-intensive approaches.

Category II criteria are based on frequency distributions of microhabitat attributes measured at locations used by the target species. These criteria are known as utilisation or habitat use functions because they represent the conditions that were being occupied by the target species when the observations were made (Thomas and Bovee, 1993). Perhaps the most controversial and significant source of error within habitat modelling lies in the selection of HSC that supposedly reflect the selection of microhabitat by a fish species in the stream under study (Thomas and Bovee, 1993). The benefit of category II criteria is that they are based on data, not on someone's opinion. However, error does exist within this method, in the form of the environmental availability bias, explained as follows by Manly *et al.* (1993). Even though a resource item is highly favoured by a species, it will not be used much if the resource is difficult to find. Conversely, less favoured resource items will be used in a larger proportion if they are the only ones available. In the context of microhabitat utilisation, this means that individuals will be forced to use sub-optimal conditions if optimal conditions are unavailable. By observing only the conditions used the most often in a given stream, an investigator could confuse optimal microhabitat with conditions that are merely tolerable (Manly *et al.*, 1993).

Category III criteria are designed to reduce the bias associated with environmental availability. These criteria are referred to as electivity or preference functions. Resource selection refers to the utilisation of resources disproportionate to their availability (Manly *et al.* 1993). For example, suppose that 10% of the stream mesohabitats occur as riffles, but 90% of the target species are found in riffles, such disproportionate use is interpreted as an active selection on the part of the target organism.

The use of category III habitat suitability curves has become the classic approach to quantifying habitat values. With field data available, the most common methodology to establish preference curves consists firstly of building histograms of the presence of fish as a function of the abiotic variable observed at the fish location (Leclerc, 2005). A second histogram representing the total number of presences and absences of fish with respect to local abiotic factors is also prepared (Leclerc, 2005). The latter histogram is considered to represent the availability of conditions. The preference curve is derived by dividing the first histogram by the second (class wise) and normalising the result by setting the maximum value to 1.0 (Leclerc, 2005). However, the variables are also treated independently in this method, so when calculating the composite suitability index, habitat preference curves can incur the same criticism as the simpler habitat suitability curve method (Leclerc, 2005).

As has been seen in Chapters 3 and 4, physical habitat is dependent on more than one variable (e.g. depth, velocity etc) and several suitability indices must be compared to define a composite suitability index (Vadas and Orth, 2001). Many researchers multiply the SIs for individual individual habitat variables to produce a composite HSI (Beecher *et al.*, 2002; Vadas and Orth, 2001). This method is based on the assumption that fish select each particular variable independently of other variables (Bovee, 1986), as multiplication of the individual SIs is analogous to multiplying assumed independent probabilities of different variables (Vadas and Orth, 2001). The HSI is thus calculated using the following equation:

$$HSI = SI_1 \times SI_2 \times \ldots\ldots\ldots \times SI_n \qquad (6.2.1)$$

A number of alternative methods have been suggested for calculating a composite HSI. The arithmetic-mean HSI is based on the assumption that good habitat conditions on one variable can compensate for poor conditions on another variable (Ahmadi-Nedushan *et al.*, 2006). In another approach, the lowest SI assumes that the most limiting factor determines the upper limit of habitat suitability and the possibility that high SI values cannot compensate for low SI values in other variables (Kormann *et al.*, 1994).

These habitat suitability curves are used to describe the adequacy of various combinations of whatever parameters are of interest to produce an estimate of the quantity and/or quality of habitat in terms of surface area. This metric is referred to as Weighted Usable Area (WUA). Once the composite suitability (Ci) has been determined, then the amount of WUA using all cells at this specific discharge is computed according to the following equation:

$$WUA = \sum_{i=1}^{n} Ai * Ci \qquad (6.2.2)$$

Where:

WUA = total Weighted Usable Area in stream at a specified discharge
Ci = composite suitability for cell i
Ai = vertical view area of cell i (bed area or volume)

In this expression the area and its composite suitability are taken into account linearly which means that a fairly large area of poor value could provide as much habitat as a limited area of excellent value (Leclerc, 2005). This algebraic behaviour has raised some criticism in the literature arguing that, in terms of biological productivity, equal amounts of WUA, even though they seem equivalent in quantity, will not produce the same biomass as a result (Leclerc, 2005). Mohardt and Mesick (1988) (amongst others) also point out that the weighted usable area, which results from the use of these curves, is an index and cannot be measured directly. As such it is difficult to validate habitat models which use WUA as the predictor of fish habitat. Indeed, it is very rare for habitat model predictions to be compared with actual ecological validation data.

Certain assumptions in the PHABSIM model may explain a potential lack of correlation between WUA and measured fish densities (Bourgeois *et al.,* 1996). The calculations of WUA also implicitly consider each habitat unit as biologically equivalent (Bovee, 1982). However, there is no reason to believe that habitat units as defined by PHABSIM are biologically equal and provide similar production rates unless each is an exact replica of the other unit (Mathur *et al.*, 1985). If each unit of WUA is biologically identical, a prediction of standing stock of fishes as a result of flow alteration may be possible (Mathur *et al.*, 1985). However, in the present form of the calculations in PHABSIM, several combinations of depth, velocity, and substrate can give the same amount of WUA, none of which may support a similar fish biomass (Mathur *et al.*, 1985).

Moir *et al.* (2005) found that the choice of HSI used was critical. They found that predictions of between-year and within-site habitat availability based on utilisation (Type II) and preference (Type III) HSIs developed on their study catchment in Scotland corresponded significantly with the use of the site by spawning fish. However, predictions based on utilisation HSIs developed in streams in southern England did not correspond significantly with patterns of site use. This indicates that the use of appropriate HSIs is vital.

Despite the intuitive appeal of using type III (preference) indices several authors argue for adoption of Type II (utilisation) indices (e.g. Lambert and Hanson, 1989; Moyle and Baltz, 1985; Parsons and Hubert, 1988). Preference indices are argued to be of greatest value in improving the transferability of HSIs between different rivers (e.g. Baker and Coon, 1997; Jowett, 2002; Thomas and Bovee, 1993). Although the problems of using *ex-situ* HSIs are well known, they are often used without assessment of transferability (e.g. using procedures such as described by Thomas and Bovee, 1993) or subsequent testing of model output (e.g. Armour and Taylor, 1991; Maddock *et al.*, 2001).

6.2.1.2. Multivariate approaches

Simple multiplication of the cell suitability (as occurs in standard HSIs) may not be correct as water depth and velocity are not independent of each other and they may not also be equally important. Therefore simple multiplication of individual suitability of these variables to produce composite values for each cell may introduce errors (Vismara *et al.*, 2001). Multivariate HSIs allow the interaction and relative importance of individual habitat variables to be incorporated into measures of composite suitability (Vismara *et al.*, 2001).

Multiple linear regression can be used to describe the relationship between the dependent variable (e.g. species abundance) and independent variables (e.g. abiotic predictors). Vismara *et al.* (2001) used a second order polynomial regression to model preference of brown trout (*Salmo trutta fario*) in an Italian river. The data was fitted by varying the order of the depth and velocity terms and adding or removing the interaction term (Vismara *et al.* 2001). Their results showed that bivariate HSCs can effectively add important information, and give a more realistic representation of the habitat selection of the target species.

As many habitat features are correlated with each other (Armstrong *et al.*, 2003), the use of multiple linear regression with typical habitat variables may lead to a problem called multicollinearity (Ahmadi-Nedushan *et al.*, 2006). This situation occurs when some of predictor variables are highly correlated (Afifi and Clark, 1996; Montogmery and Peck, 1992). When multicollinearity is present, the computed estimates of the regression coefficients are unstable and have large standard errors (Ahmadi-Nedushan *et al.*, 2006). The regression coefficients fluctuate when used across samples; and even a slight change in the data can result in different regression coefficients (Ahmadi-Nedushan *et al.*, 2006).

Multiple linear regression is used when the response variable (e.g. species abundance) is continuous, but is not an appropriate method for analysing dichotomous response variables, as is the case for data of presence-absence of species (Ahmadi-Nedushan *et al.*, 2006). An appropriate model in such circumstances is logistic regression, which can be used to analyse the relationship between a Bernoulli (or binary) response (suitable versus unsuitable) and explanatory environmental factors describing the quality of habitat (e.g. depth, velocity and substrate) (Ahmadi-Nedushan *et al.*, 2006). The model estimates the probability of a positive response occurring given a set of explanatory variables (e.g. Agresti, 1996)

Guay *et al.* (2000) developed a probabilistic habitat model by constructing a matrix containing information on the physical conditions noted where and when an Atlantic salmon parr was observed in the calibration section of the reach and on the physical conditions estimated where and when no fish were observed. This matrix was used by Guay *et al.* (2000) to develop a model to estimate the probability of observing a fish under given combinations of physical conditions. Guay *et al.* (2000) was the first attempt to use and validate the ability of a biological model based on logistic regression to predict fish distributions. They found that in their study, fish distribution was better predicted by a biological model based on a habitat probability index than with a standard habitat suitability index.

Some of the problems encountered by standard habitat preference curves, such as issues of assuming independence between parameters are avoided in the probabilistic approach. The probabilistic method uses a multivariate approach in which all variables are considered simultaneously and with no arbitrary correction. In addition, the polynomial approach used by Guay *et al.* (2000) to model the logistic equation and to assess habitat probability index may be more appropriate for accounting for the possibility that a below-average current speed, from a preference index perspective, may provide above-average habitat conditions when associated with specific combinations of depth and substrate composition. Hence, the mathematical structure that Guay *et al.* (2000) used to estimate habitat probability indices may allow a better representation of the statistical as well as biological interaction among the physical variables used to estimate habitat quality than that used to estimate habitat suitability indices.

The Tiffan *et al.* (2002) biological model was based upon a habitat probabilistic model. Guay *et al.* (2000) used a similar approach, but compared it with Habitat Suitability Index approaches. They found that the HPI produced better results than the HSI, and they noted that this may be because of the multivariate nature of the HPI approach in which predictions from the hydraulic models are considered simultaneously rather than independently, so dealing with a common criticism of HSC approaches.

The order (linear and/or quadratic) of the algebraic terms forming the polynomial can have some importance in the predictive power of the model (Leclerc, 2005). The linear term often establishes a certain proportionality between the abiotic variable considered and the probability of species presence, especially in the lower range of the variable (Leclerc, 2005). The addition of quadratic terms as a function of the same variables transforms the polynomial into a parabola which will damp the ever increasing or decreasing effects of the linear terms (Leclerc, 2005). This statement has a corollary regarding the range of values of explanatory variables considered by the measurement protocol (Leclerc, 2005). If this range if limited to a narrow domain of representitiveness, the linear component might be sufficient to reproduce accurately the fish distribution in this limited region but with low predictability outside this specific range (Leclerc, 2004).

6.2.1.3. Fuzzy logic

Uncertainties exist in ecological modelling. A large inherent uncertainty of ecological data results from the presence of random variables, incomplete or inaccurate data, use of approximation estimations instead of direct measurements (Ahmadi-Nedushan *et al.*, 2006). The use of the fuzzy approach may be a way around this. Compared to conventional methods, fuzzy logic allows for a better use of imprecise and uncertain measurements and vague expert knowledge in two ways: (i) the representation and handling of imprecise data as defined as fuzzy sets; and (ii) the representation and processing of vague expert knowledge in the form of linguistic rules with imprecise terms defined as fuzzy rules (Ahmadi-Nedushan *et al.*, 2006).

The fuzzy set theory (Zadeh, 1965) is an extension of classic set theory, and is built around the central concept of fuzzy sets or membership functions. Fuzzy set theory enables the processing of imprecise information by means of membership functions, in contrast to Boolean characteristic mappings (Zadeh, 1965). In conventional set-theory, mapping of a classical set only takes two values: one, when an element belongs to the set; and zero, when it does not. In fuzzy set theory, an element can belong to a fuzzy set with its membership degree varying from zero to one (Adriaenssens *et al.*, 2004).

Fuzzy logic has been used in instream habitat studies using fuzzy information available on the selective behaviour of target species (Jorde *et al.,* 2001; Schneider and Jorde, 2003). The fuzzy rule based method operates with combinations of qualitative and semi-quantitative criteria to which a suitability level is attributed according to specialists (Leclerc, 2005). The procedure starts by setting up checklists with possible combinations of relevant physical criteria and lets the specialists define in natural language whether habitat quality is low, medium or good for the proposed combinations. The remainder of the methodology is outlined in Figure 6.2.1. This method is implemented in the CASIMIR model (Computer Aided Simulation Model for Instream Flow Requirements). The process by which the analysis takes place was set out in detail in Schneider and Jorde (2003).

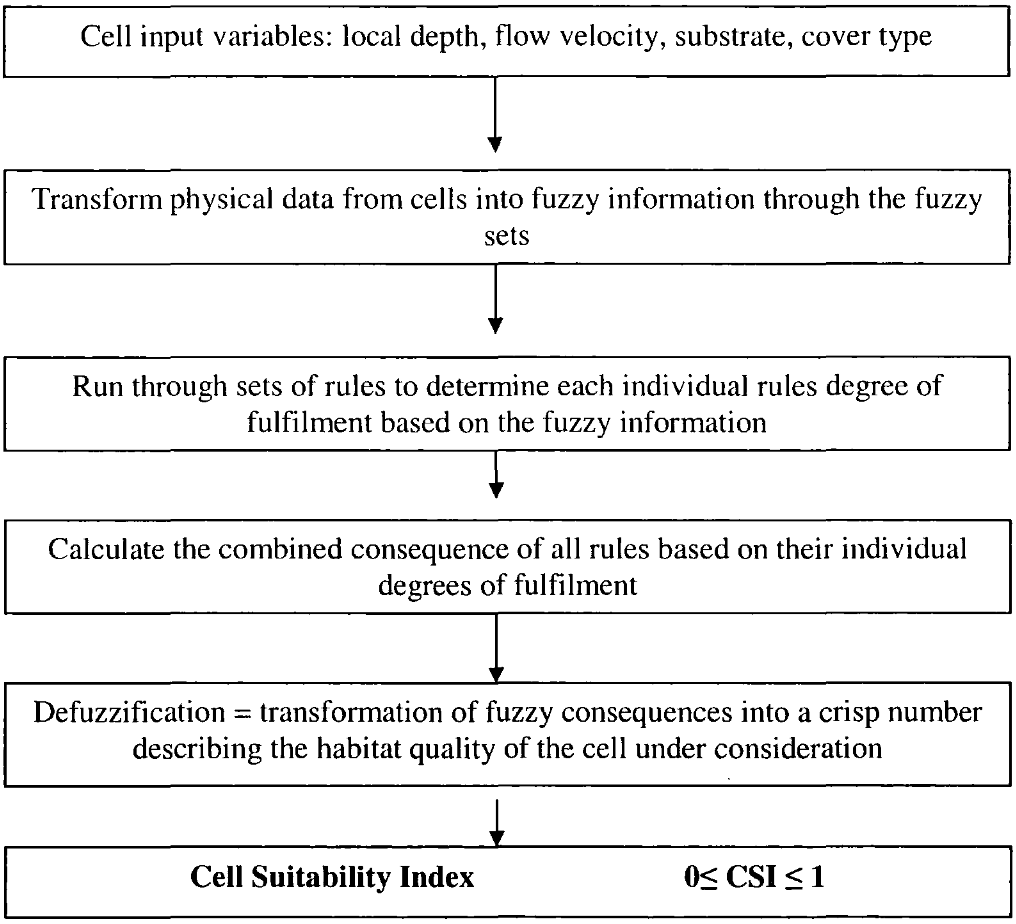


Figure 6.2.1.. Flow chart to show the basic methodology of the Fuzzy Logic method (From: Schneider and Jorde, 2003).

The potential advantages of this method were highlighted by Schneider and Jorde (2003) as: (i) knowledge about habitat requirements is usually qualitative, but can be numerically processed by the fuzzy rule based approach; (ii) fuzzy logic calculations consider multivariate effects but no independence of the parameters is required; (iii) fuzzy rules use combinations of physical parameters as the input variable; (iv) a comparatively small number of measured or observed data is needed; (v) new parameters can be added easily; and (vi) because of the fuzziness, the demands upon the accuracy of the hydraulic calculations or observed physical parameters are lower than in conventional approaches.

However, Schneider and Jorde (2003) did point out two potential limitations: (i) the number of fuzzy rules increase rapidly as more parameters are considered; and (ii) the rules are so close to human language that they can give the false impression that they can be easily defined by persons regarding themselves as experts.

6.2.1.4. Bioenergetics

Relationships between fish growth and environmental variables such as food intake and temperature have been known for many years (Elliott, 1976). The concept of a bioenergetic model for fish growth has been added by several researchers (e.g. Fausch, 1984; Hughes and Dill, 1990; Van Winkle *et al.*, 1998; Hayes *et al.*, 2000).

Van Winkle *et al.* (1998) created a tool for projecting flow and temperature effects on trout populations by linking the hydraulic component of the PHABSIM model to an individual-based hydraulic model. In the model, a trout is located each day in a cell within a habitat type with specified characteristics (i.e. velocity, depth, cover, density of other trout) as a means of simulating daily average flow-related effects on the energetics, movement, spawning and risks of mortality for that trout. As has been explained earlier, habitat suitability indices are required in PHABSIM applications to estimate wetted usable area (WUA). Several studies have demonstrated that the results of such applications are more sensitive to habitat suitability indices than representation of physical habitat (e.g. Mathur *et al.*, 1985; Orth, 1987; Armour and Taylor, 1991). The substitution of an individual-based habitat population model in place of habitat suitability indices increases realism by mechanistically representing the processes underlying reproduction (Van Winkle *et al.*, 1998). Van Winkle *et al.* (1998) found that predicted and observed lengths were in good agreement, but predicted and observed abundances were not.

6.2.2. The use of habitat models

6.2.2.1. Habitat Suitability Indices

The application of habitat suitability curves within the UK have seen them coupled with one dimensional hydraulic models under the guise of PHABSIM. The Institute of Hydrology carried out the first trials of PHABSIM in the UK (Bullock et *al.*, 1991) and further developed PHABSIM under the auspices of a research programme to determine ecologically acceptable flows (Dunbar *et al.*, 1996). Spence and Hickley (2000) outlined a number of unpublished uses of PHABSIM within the UK: the upper reaches of the river Severn (looking at minimum maintained flow); river Worfe (abstraction issues); Afon Vrynwy (pump-back scheme); and the River Kennet (groundwater abstraction).

The first application of PHABSIM in the UK for the assessment of an ecologically acceptable flow regime for a current operational water resources problem was by Johnson *et al.* (1993). They used the PHABSIM software to model the effects of groundwater abstraction on salmonid habitat availability, on the River Allen, Dorset, and their outputs suggested that the most sensitive life stage considered was the fry/juvenile life stage of trout.

Maddock *et al.* (2001) used PHABSIM (using habitat suitability curves) to investigate the reallocation of compensation flows in a much more complicated system than this investigation in the Upper Derwent catchment, Derbyshire. As their indicator species they used brown trout, grayling and the invertebrate families; Rhyacophilidae, Leuctridae, Chloroperlidae, and Heptageniidae. They found that a significant increase in trout numbers would occur if the flow were to be increased from 57 to 80 Ml/d on the river Derwent.

Gibbins and Acornley (2000) and subsequently Gibbins *et al.* (2001) used PHABSIM to assess the extent of redd stranding associated with various Kielder reservoir HEP releases.

They used PHABSIM to formulate a seasonally variable and hydroelectric power regime that limits impacts and increases suitable 0+ and spawning habitat.

Strevens *et al.* (1999) used PHABSIM, in conjunction with a groundwater model to assess the impacts of groundwater abstraction on the trout fishery of the River Piddle, Dorset, where the results indicated large habitat losses for juvenile brown trout in zones of reduced flow.

It can be seen from the above section that PHABSIM has been used to guide decision makers for a number of different problems. However, the PHABSIM approach (1D – HSC) does have a number of limitations which have been examined both in Chapter 5 and above. Two dimensional hydrodynamic models have not as yet been used in instream flow problems in the United Kingdom, but have been used in North America. For example, Leclerc *et al.* (1995;1996) used a finite element solution of the depth-averaged flow equations, including a wetting and drying treatment, with a Habitat Suitability Index (HSI) to explore habitat changes on the Moisie river, Quebec (Leclerc *et al.*, 1995) and the Ashuapmushuan River, Quebec (Leclerc *et al.*, 1996).

6.2.2.2. Multivariate Methods

Tiffan *et al.* (2002) quantified flow-dependent changes in subyearling autumn Chinook salmon (*Oncorhynchus tshawytscha*) rearing habitat in the Columbia River using a spatially explicit two dimensional model. With the flow changes being created by hydroelectric operations in an upstream reservoir. The biological model in Tiffan *et al.* (2002) study was based upon a habitat probabilistic model. Tiffan *et al.* (2002) used the depth averaged model to simulate flow depths and velocities at 36 steady state discharges. The biological model was based upon multivariate logistic regression in which the probability of subyearling fall Chinook Salmon presence was predicted from physical habitat parameters predicted by the model. The results showed that estimates of rearing habitat decreased as flows increased and that estimates of the area that fish could become stranded in initially rose, but then fell. However, they noted that the 16m^2 resolution adopted was too coarse to characterise adequately the needs of subyearling fish.

6.2.2.3. Fuzzy Logic

Fuzzy logic approaches have yet to be applied to modelling instream ecology in a water resource context. Jorde *et al.* (2001) used fuzzy logic in a habitat simulation model for fish habitat evaluation of several rivers on Switzerland and concluded that observed fish densities show a higher correlation with fuzzy based simulations than for those based on preference functions. Fuzzy logic remains unused in UK instream habitat studies.

6.3. Methods

6.3.1. Introduction

In this chapter, habitat modelling is conducted using a method which is based on fuzzy information of the selective behaviour of target species. Figure 6.3.1 shows that the fuzzy rules (red) are combined with the hydraulic output presented in Chapter 5 in order to produce habitat suitability maps. Fuzzy rules were determined for both invertebrates and brown trout. For the invertebrates, rules were created for functional feeding guilds (e.g. scraper, filterer, predator; after Cummins, 1973) rather than the usual approach of using invertebrate target species. Fuzzy rules were also created for four life stages of brown trout: spawning; nursery (0+); juvenile (>0+(<20cm)); and adult (>20cm).

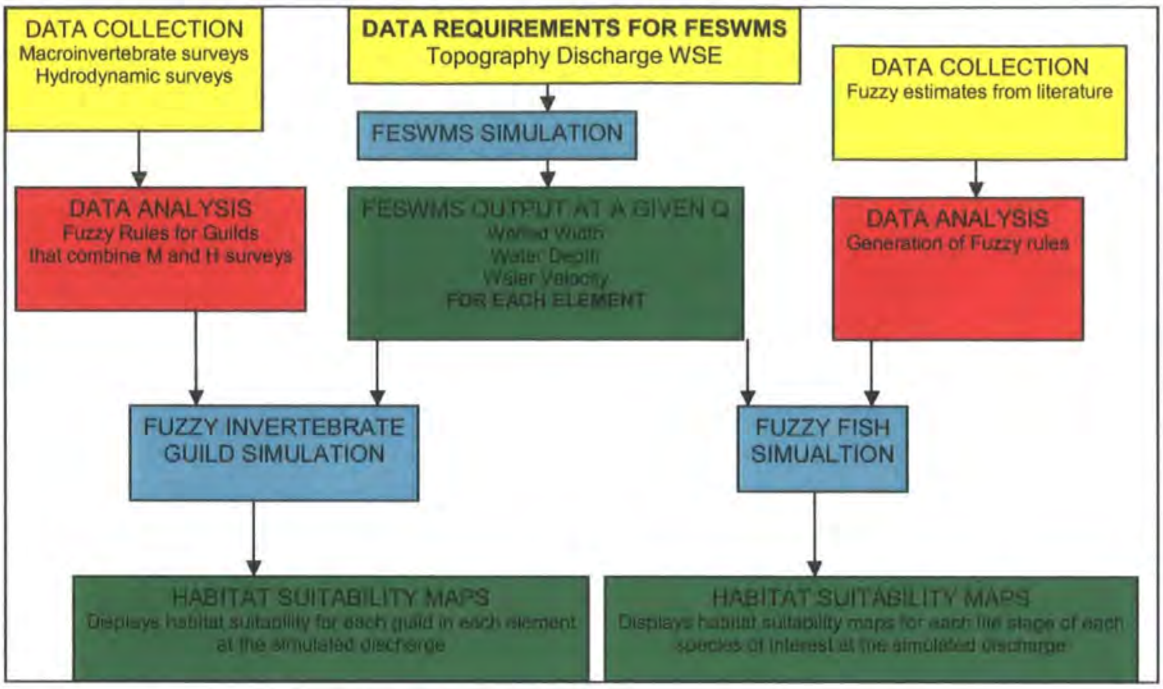


Figure 6.3.1. Figure to show the data flow through the modelling process. Q= discharge; WSE = water surface elevation.

This section begins with a justification of the modelling approach used in this study, in terms of both fuzzy logic and guild analysis. The second section outlines the fuzzy logic methodology in detail. The final section shows and discusses the fuzzy rules used in the habitat simulations.

6.3.2. Justification of modelling approach

6.3.2.1. Fuzzy approach

The review in Section 6.2 emphasises that there has been significant development of habitat modelling approaches using hydrodynamic models over the last 10 years. This progress aside, a number of difficult issues remain. Central to these difficulties is the fact that the hydrodynamic models need to be informed by ecological knowledge. Much of this is traditionally based upon field measurements (e.g. of depth and velocity) of where particular organisms have been observed at particular life stages (e.g. Heggenes, 1996; de Crispin de Billy and Usseglio-Polatera, 2002; Armstrong *et al.*, 2003). Typically, these measurements record a preferred organism preference (where they are living) and not a possible organism preference (where they could live) (Manly *et al.*, 1993). The latter is more difficult to estimate, not least because exposure to certain extreme conditions (where an organism could possibly live) means that certain conditions may only be suitable for a specific period of time, but also because designing experiments to evaluate possible organism habitat is almost impossible. Section 4.2 noted that habitat will only limit organism growth and survival if the densities of fish are sufficiently high relative to the size of the fish (Armstrong *et al.*, 2003). If ecological measurements are made when this is not the case, the organism-habitat relationships will partly reflect other processes and hence will be noisy with respect to what could be possible habitat.

Fuzzy models are designed for applicability to situations where only imprecise or even ambiguous information is available (Ross, 1995) and given the above observation about the noise that will be implicit in the ecological preference data, this is why fuzzy modelling has appeal for habitat modelling. Fuzzy models are particularly valuable for situations where the noise in our knowledge is non-random and not necessarily quantifiable: the situation is ambiguous (Ross, 1995) rather than uncertain (in the classical sense). Thus, fuzzy analysis commonly maps onto linguistic definitions (good, bad) rather than numerical definitions. The ecological uncertainty surrounding habitat preference, which is both methodological and substantive, means that developing fuzzy habitat models it is possible to explicitly retain the ambiguity that is implicit in habitat preferences knowledge.

This study has not collected the same point data of fish location and habitat parameters that most traditional instream habitat modelling studies collect, as a product of the ecological sampling structures chosen (see Sections 3.3 and 3.4). This data is traditionally used to create one of the data-driven habitat suitability curves, habitat preference curves or habitat probability curves. Univariate habitat suitability curves can be created by using a combination of existing data and expert opinion; however the more traditional (and advanced) preference curve and the probabilistic approach require this data. Rather, in this study, fuzzy rules are developed based on established (and generally extensive) ecological knowledge and the data reported in Chapters 3 and 4 are used for validation purposes. This reflects the fundamental principles of a fuzzy approach in which all forms of knowledge can be brought to a particular problem. It is also a part of assessing model transferability in which extensive fuzzy model calibration is not used as a requirement of model development.

6.3.2.2. Invertebrate guild approach

It has been rare over the past two decades that macroinvertebrate habitat considerations have been made (Gore *et al.*, 2001). As was seen in Section 6.2, the majority of instream habitat modelling has concentrated on target fish species rather than macroinvertebrates, and if invertebrates have been considered, it is usually in the form of two or three indicator species. These indicator species provide no information on community structure and/or potential brown trout food base. A decline in the food base and/or community structure has the potential of leading to further degradation of the target fish species for management, as well as less efficient energy processing throughout the ecosystem (Gore *et al.*, 2001). It is important then, that prediction of changes in macroinvertebrate habitat are considered in conjunction with analysis of instream flow requirements for target fish species.

Lamaroux and Souchon (2002) suggested and implemented the use of guilds, a well-known ecological concept that allows for the combination of a number of species and even of various life stages that have similar habitat requirements. Grouping criteria are generally based on the assumption what those species which share common feeding or reproduction strategies. For 21 species-specific size classes of fish, Lamaroux and Souchon (2002) developed four guilds, by clustering habitat preferences. Furthermore the habitat values of these four guilds of fish (pool, bank, riffle, and midstream) could be predicted using average characteristics such as discharge, width, depth and average particle size. Lamaroux and Souchon (2002) found that habitat values for different habitat guilds could be predicted using a number of average characteristics.

Although the Lamaroux and Souchon (2002) work above was conducted on fish guilds, there does appear to be potential, to split the macroinvertebrate population (containing a very large number of families) present in the rivers into guilds and to use the hydraulic model output to predict the habitat availability of invertebrate guilds in the channel (Table 3.3.5). This approach has not previously been used in the prediction of macroinvertebrate populations using hydraulically driven simulations. There are three reasons why the guild approach was taken: (i) the invertebrate communities of the Rivelin and Loxley are so diverse and abundant, that to model each species would be laborious or an indicator species would have to be selected; (ii) if the indicator species route were chosen, this approach would not provide the decision maker any information on potential changes in invertebrate community structure or community changes; and (iii) it will aid in the transferability of the model, as all species may not be present from one Millstone Grit stream to another. As all that would be needed in order to populate the guilds for a different river would be one or more Environment Agency standard kick samples, with the invertebrates populating the guilds altered accordingly.

6.3.3. Fuzzy-logic methodology

The approach to habitat modelling is restricted to the consideration of depth and velocity. As the approach is two-dimensional, this implicitly includes consideration of the wetted usable area, as nodes predicted as wet (depth>0) and/or dry as a function of model solution.

Both depth and velocity are interpreted into three classes each: poor, medium, and good; and habitat into six classes: unsuitable, very poor, poor, good, very good and excellent. Fuzzy subsets are then defined for depth (D_i) and Velocity (V_i) that define the grade of membership of each predicted depth (d) or velocity (v) of each of the i (poor, medium or good) subsets:

$$\begin{aligned}
D_p &= \{[d, \mu_{D_p}(d)];\, d \in D, \mu_{D_p}(d) \in [0,1] \\
D_m &= \{[d, \mu_{D_m}(d)];\, d \in D, \mu_{D_m}(d) \in [0,1] \\
D_g &= \{[d, \mu_{D_g}(d)];\, d \in D, \mu_{D_g}(d) \in [0,1] \\
V_p &= \{[v, \mu_{V_p}(v)];\, v \in V, \mu_{V_p}(v) \in [0,1] \\
V_m &= \{[v, \mu_{V_m}(v)];\, v \in V, \mu_{V_m}(v) \in [0,1] \\
V_g &= \{[v, \mu_{V_g}(v)];\, v \in V, \mu_{V_g}(v) \in [0,1]
\end{aligned} \tag{6.3.1}$$

where: p is poor, m is medium and g is good; and $\mu L_i(l)$ is the grade of membership of the predicted value l(d or v) in L_i(D_i or V_i), which equals one for at least one value of L for each *i*. In this scheme, when 0< $\mu L_i(l)$ <1, has a partial membership of L_i, and this is the sense in which the analysis is fuzzy, with *l* potentially being a partial member of more than one L_i. We then specify a fuzzy rule for Habitat (*Hk*) based on two premises (for D and V):

$$\text{If } D_i \otimes V_j \text{ then } H_k\text{, for } K \text{ values of } k \tag{6.3.2}$$

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

$$\mu_{Hk} = \mu_{Hk,Di(d)}\mu_{Hk,Vj(v)} \tag{6.3.3}$$

where, μ_{Hk} is the degree of fulfilment of habitat class *k*, as defined by each possible combination of *Di* and *Vj* (from (2)), given the predicted values of *d* and *v*. The nine rules that come from (2) could be used to provide nine habitat classes. However, a symmetrical habitat classification is used, weighting *D* and *V* equally in the determination of habitat suitability (Table 6.3.1). This can be made more sophisticated by changing the weightings to reflect the known importance of velocity and depth in contributing to a particular habitat class, possibly informed by field data or traditional habitat suitability analyses, or calibrated onto measured relationships between habitat and productivity for a specific reach or set of reaches.

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

Table 6.3.1. Table to show the symmetrical definition of habitat classes in relation to the rule set defined in Equation 6.3.2.

The analysis so far provides nine outcomes which indicate the degree of fulfilment of each rule. If there was no fuzziness in the system, then there would only be a single outcome. As the level of fuzziness increases, so the number of outcomes increases to the maximum of nine. In order to provide a single habitat suitability index we defuzzify the analysis to produce a single 'crisp' number. Two numbers are produced. The first number is a habitat suitability index and accounts for the total habitat available within a given reach. The second number is the habitat suitability index weighted by the area of habitat so as to provide a measure of habitat quality.

6.3.4. Fuzzy rules used

Fuzzy rules were created for invertebrate guilds and brown trout as these were the main focus of Chapters 3 and 4, and therefore it will hopefully be possible to abstractly validate the predictions made by the habitat model to known ecological observations. It would be easy for decision makers to create fuzzy rules for the other fish species present in the study rivers (e.g. Bullheads; Brook lamprey) however, they have not been included in this chapter.

6.3.4.1. Invertebrate guilds

The guilds to which each family belong were shown in the invertebrate chapter (Table 3.3.5). In this case the guilds used in this analysis are populated by the invertebrates present in the river and not by every species found in Britain. This is an attempt to reduce the fuzziness of the rules and improve the accuracy of the predictions, and to make the predictions more comparable to the observed guild proportions in the nearby sampling sites. If there were too many families in each guild, the fuzzy rules may become too broad to be meaningful.

The flow velocity preferences used in the LIFE calculations (Extence *et al.*, 1999) are used to inform the velocity habitat preferences of family level invertebrates. After an extensive literature survey it was found that depth habitat preference data were more difficult to come by. Therefore, to inform the fuzzy rules for guild depth, the information shown on the Centre for Intelligent Environmental Systems (Cies) website was used (www.soc.staffs.ac.uk/research/groups/cies2/) (Table 6.3.2). Both of these sources have the common desirable characteristic in that they are generally available, and so help the transferability of the model, and as such the only difficult information needed to further populate the guilds is to decide which feeding guild a given family belongs

	LIFE Velocity preference (cm/s)	Min depth found at(cm)	Max depth found at (cm)	Mean Depth (cm)
FILTERER				
Ephemeridae	20-100	0.3	300	33
Hydropsychidae	20-100	0.3	266	24
Simuliidae	20-100	0.3	250	23
SCRAPER				
Goeridae	>100	1	230	25
Taeniopterygidae	20-100	1	166	21
Psychomyiidae	20-100	0.3	333.33	34
Elmidae	20-100	0.3	333.33	28
Ancylidae	20-100	0.3	300	28
Hydroptilidae	Slow and standing	1	333.33	35
Planorbiidae	Slow and standing	0.3	300	44
Lymnaeidae	Slow and standing	0.3	300	35
Physidae	Slow and standing	1	280	47
Valvatidae	Slow and standing	1	300	59
COLLECTOR-GATHERER				
Heptageniidae	>100	0.3	217	21
Baetidae	20-100	0.3	300	29
Ephemerellidae	20-100	1	280	26
Leptophlebiidae	20-100	0.3	216	25
Beraeridae	20-100	4	166.6	32
Hydrophilidae	Slow and standing	1	250	26
SHREDDER				
Gammaridae	20-100	0.3	333.3	31
Lepidostomatidae	20-100	1	183	24
Leuctridae	20-100	1	217	20
Sericostomatidae	20-100	0.3	166	22
Asellidae	Slow and standing	1	333.3	37
Leptoceridae	Slow and standing	1	333.3	37
Limnephilidae	Slow and standing	1	333.3	30
Nemouridae	Slow and standing	0.3	167	21
Tipulidae	Slow and standing	0.3	300	23
PREDATOR				
Chloroperlidae	>100	2	65.3	19
Perlodidae	>100	1	150	20
Rhyacophilidae	>100	1	280	21
Odontoceridae	>100	2	93	21
Pescicolidae	20-100	1	333.3	45
Polycentropodidae	Slow and standing	0.3	300	34
Dytiscidae	Slow and standing	0.3	300	35
Scirtidae	Slow and standing	1	165	20
Erpobdeliidae	Slow and standing	0.3	300	33
Glossiphonidae	Slow and standing	0.3	333.3	33
Planariidae	Slow and standing	0.3	333.3	32
Sialidae	Slow and standing	0.3	300	44

Table 6.3.2. Table to show the members of each of the feeding groups present in the Rivelin and Loxley and some habitat preference information.

The information in Table 6.3.2 is used to populate the fuzzy rules in Table 6.3.3. For example, for the filtering invertebrate guild, the LIFE velocity preferences for the families are all 20-100cm/s. Therefore, it can be said that poor habitat lies outside of this range, medium habitat lies at the edge of this range and good habitat exists in the middle of this range. One important feature of the structure of the fuzzy model used is that there are two niches available for good habitat. This is of vital importance when the scraper, shredder and predator guilds are considered. For each of these guilds, there are two distinct groups of invertebrates, each preferring vastly different flow velocities. However, within the fuzzy rules, twin-peaks can be created (red areas in the velocity rules), with good habitat found at within each of these ranges of velocities (Figure 6.3.2).

	Poor	**Medium**	**Good**
Filterer Velocity	<0.20m/s >1m/s	0.20-0.40m/s 0.80-1m/s	0.40-0.8m/s
Filterer Depth	<0.05m >2m	0.05-0.20m 0.40-2m	0.20-0.40m
Scraping Velocity	0.05-0.2m/s >1m/s	0.20-0.40m/s 0.8-1m/s	0-0.05m/s 0.40-0.80m/s
Scraping Depth	<0.05m >2m	0.05-0.2m 0.4-2m	0.2-0.4m
Collector gatherer Velocity	<0.1m/s >1m/s	0.1-0.4m/s 0.8-1m/s	0.40-0.80m/s
Collector gatherer Depth	<0.05m >1m	0.05-0.2m 0.3-1m	0.2-0.3m
Shredder Velocity	0.05-20m/s >1m/s	0.2-0.4m/s 0.8-1m/s	0-0.05m/s 0.40-0.80m/s
Shredder Depth	<0.1m >1m	0.1-0.2m 0.4-1m	0.2-0.4m
Predator Velocity	0.1-0.7m/s >1.5m/s	0.7-1m/s 1.3-1.5m/s	0-0.1m/s 1-1.3m/s
Predator Depth	<0.1m 0.5-1.5m	0.1-0.25m 0.35-0.50m	0.25-0.35m

Table 6.3.3. Table to show the fuzzy rules chosen for the invertebrate guilds based on the invertebrates present in the Rivelin and Loxley.

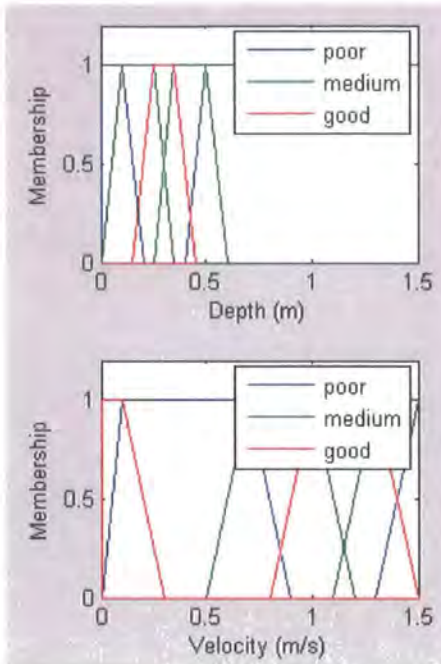


Figure 6.3.2. Figure to graphically show the fuzzy rules for the predator guild.

6.3.4.2. Brown trout

On the basis of available data (e.g. Heggenes, 1996; de Crispin de Billy and Usseglio-Polatera, 2002; Armstrong *et al.*, 2003), habitat have been mapped onto the set memberships shown in Table 6.3.4. A number of life stages of brown trout have been chosen. The conditions needed for spawning were first considered, as it is a vital component of the salmonid life cycle. The three subsequent age-size classes of brown trout were chosen as these were the main age-size classes investigated in Chapter 4, and as such will allow easier comparison with those findings.

As with the invertebrates, information from the literature was used to inform the fuzzy rules. There have been many studies on brown trout habitat requirements; good habitat was defined as habitat always used by the life stage of interest; medium habitat was defined as habitat sometimes used; and poor habitat defined as habitat rarely used. At all times ensuring agreement between the literature.

	Poor	Medium	Good
Spawning Velocity	<0.011m/s >0.80m/s	0.11-0.35m/s 0.50-0.80m/s	0.35-0.50m/s
Spawning Depth	<0.06m >0.82m	0.06-0.25m 0.40-0.82m	0.25-0.40m
Nursery (0+) Velocity	<0.05m/s >0.20m/s	0.05-0.15m/s	0.15-0.20m/s
Nursery (0+) Depth	<0.05m >0.35m	0.05-0.20m 0.30-0.35m	0.20-0.30m
Rearing (>0+(<20cm)) Velocity	<0.05m/s >0.70m/s	0.05-0.10m/s 0.40-0.70m/s	0.20-0.40m/s
Rearing (>0+(<20cm)) Depth	<0.05m >1.22m	0.05-0.50m 0.75-1.22m	0.50-0.75m
Adult (>20cm) Velocity	<0.05m/s >0.80m/s	0.5-0.10m/s 0.3-0.8m/s	0.1-0.3m/s
Adult (>20cm) Depth	<0.20m >1.5m	0.20-0.40m 1-1.5m	0.4-1m

Table 6.3.4. Table to show the fuzzy rules used for the brown trout habitat simulations.

6.3.4.3. Precision of the fuzzy rules

Figure 6.3.3 shows a visual example of the fuzzy rules for rearing (>0+(<20cm)) trout. The slope of the line is provided by a precision value within the model. The precision value for depth and velocity can be set independently. For this chapter the precision values have been set at 0.2m/s for velocity and 0.1m for depth. The precision value for velocity is higher because of the larger amounts of uncertainty (worse precision) in the hydraulic calibration results using the point velocities compared with the point depths. Table 6.3.5 shows the best calibration results from each of the meshes used. It is from this that the precision values have been chosen. The largest precision values for the point depth calibration is 7cm, whilst the largest precision values for the point depth validation is 22cm. Therefore, the use of precision values within the fuzzy model of 0.1m and 0.2m/s helps account for errors from the hydrodynamic model. However, it must be noted that imprecision will not only result from the hydraulic model but also imprecision within the habitat preferences of the organisms.

	Pre-change waters edge position	Post-change waters edge position	Point Depth	Point Velocity
RU-1	0.17±0.20m	0.09±0.14m	0.12±0.06m	-0.06±0.15m/s
RU-2	0.14±0.19m	0.05±0.11m	0.12±0.06m	-0.06±0.15m/s
RD-1	0.20±0.31m	0.17±0.23m	0±0.03m	0±0.22m/s
RD-2	0.25±0.33m	0.22±0.24m	0±0.03m	0±0.22m/s
LU-1	0.33±0.49m	0.14±0.20m	0.01±0.05m	-0.08±0.17m/s
LU-2	0.47±0.55m	0.17±0.25m	0±0.05m	-0.06±0.19m/s
LD-1	0.43±0.39m	0.49±0.53m	0.15±0.07m	-0.06±0.18m/s
LD-2	0.62±0.51m	0.73±0.58m	0.17±0.07m	-0.07±0.17m/s

Table 6.3.5. Table to show a summary of the 'best' calibration results.

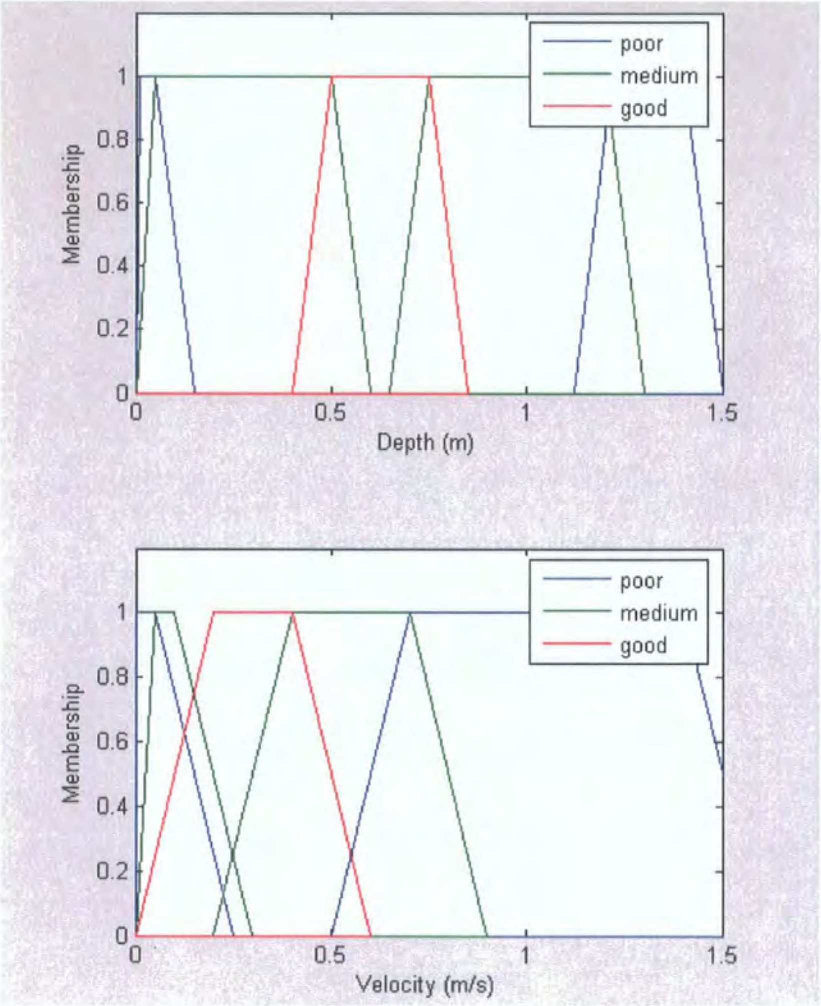


Figure 6.3.3. Figure to show the fuzzy rules for >0+ (<20cm) brown trout.

6.4. Assessing the sensitivity of the fuzzy rules

6.4.1. Sensitivity to uncertainties in simulated depth and velocity

This section assesses the sensitivity of the fuzzy rules to uncertainties in the simulated velocities and depths. This analysis uses the habitat data for each of the fuzzy rules created (for all age/size classes of brown trout and each of the invertebrate guilds). This is conducted by comparing the habitat predictions against the point depth and point velocity calibration simulations conducted for RU1. The depth and velocity accuracies are plotted against the eddy viscosity values used in this calibration as it was seen in Section 5.4 that this was the most effective model parameter. This investigation was conducted only for one mesh resolution as little difference could be seen between the hydraulic outputs from the two mesh resolutions shown in Chapter 5.

Figure 6.4.1 show that altering the eddy viscosity in the hydraulic model does influence the habitat predictions of the fuzzy model for each of the invertebrate guilds. Figure 6.4.2, shows that the habitat predictions for each of the life stages of brown trout are also sensitive to uncertainties in both depth and velocities. The predicted available habitat appears to decrease with decreases in eddy viscosity for each of the fuzzy rules. This is probably due to the reduction in wetted area caused by the decrease in eddy viscosity. On the basis of these results, it is possible to obtain an estimate of the magnitude of the HSI change required to distinguish a predicted compensation flow impact from uncertainty due to calibration. This will be done by relating the results presented here to any potential changes in predicted habitat presented later in this chapter.

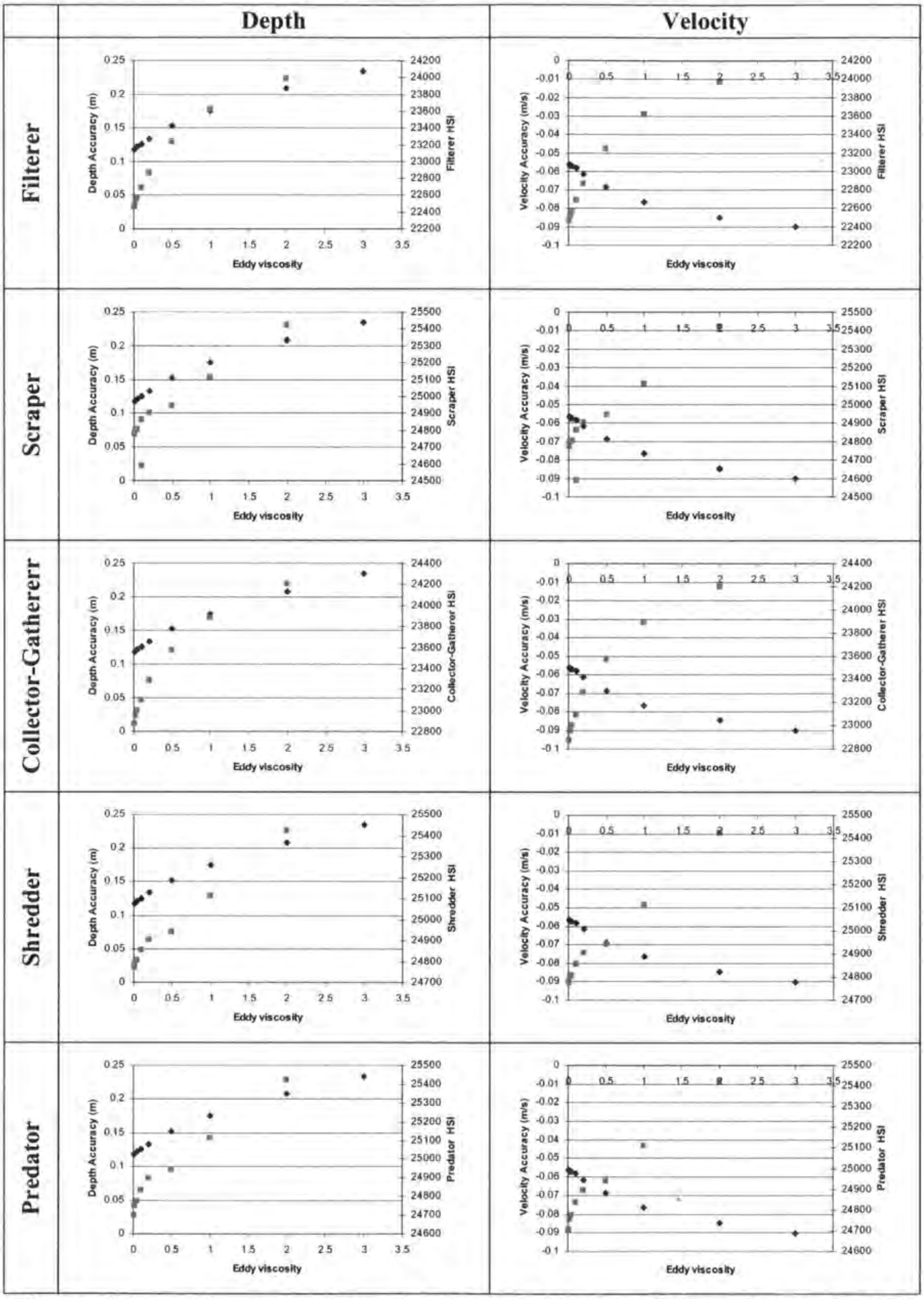


Figure 6.4.1. Figure to show the fuzzy habitat predictions for each of the macroinvertebrate guilds modelled at RU1 compared with the point depth and velocity accuracy calibration results. Red dots are the modelled habitat suitability index; blue dots are simulation accuracy.

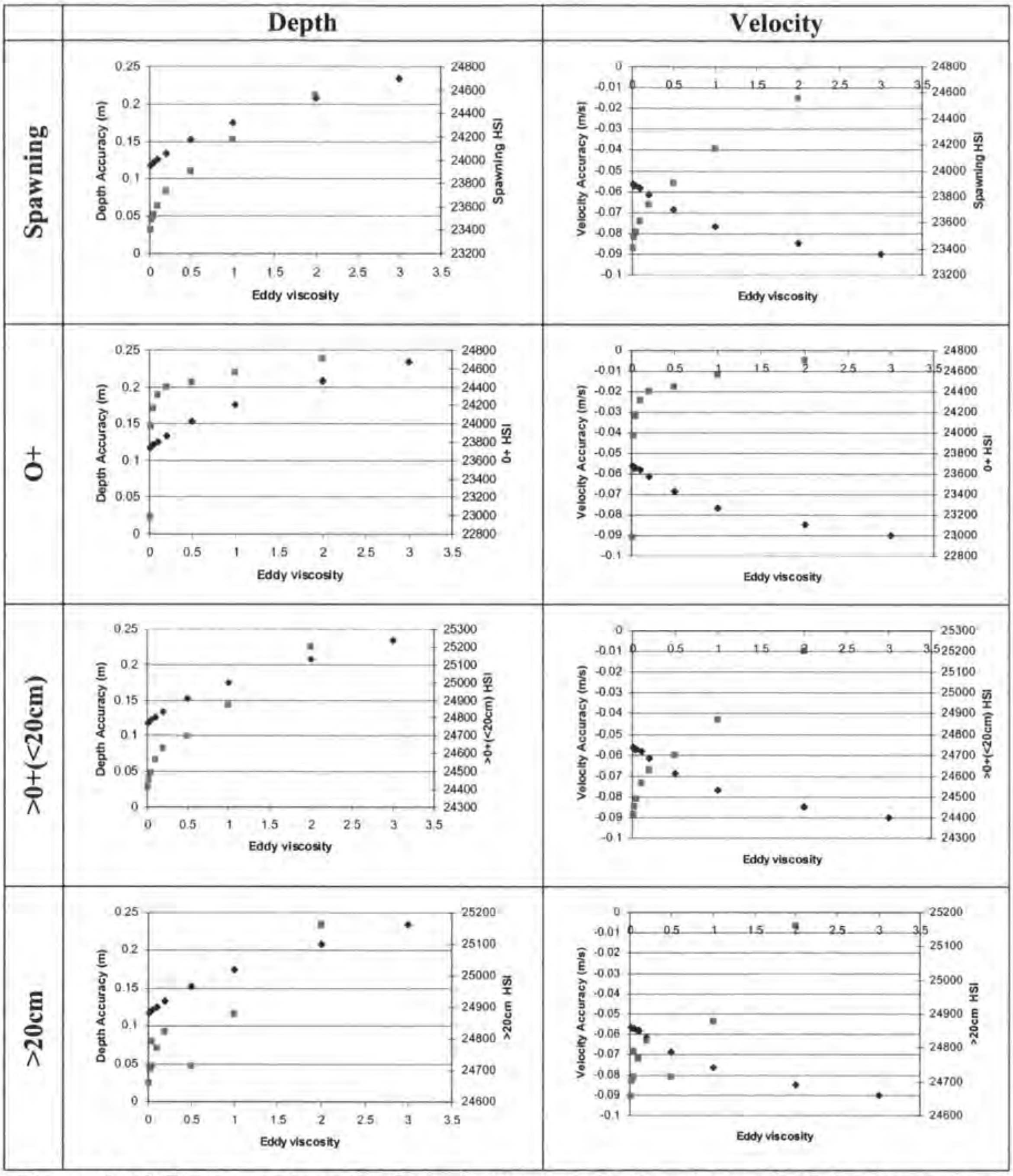
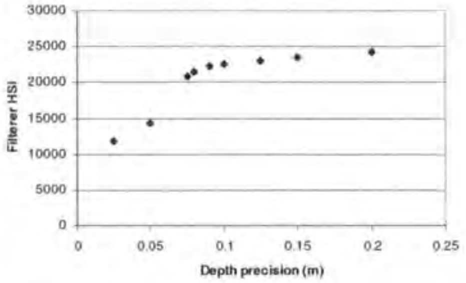
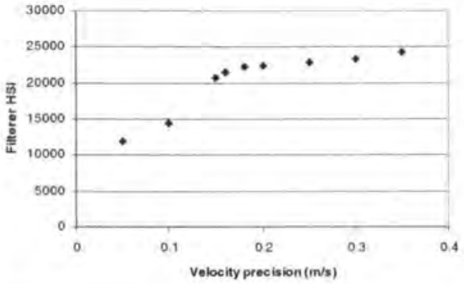
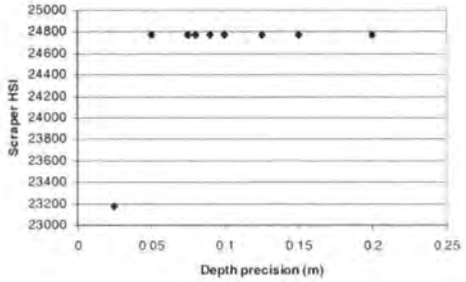
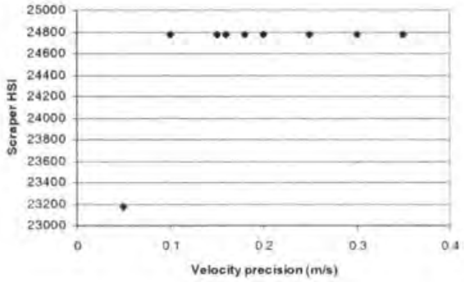
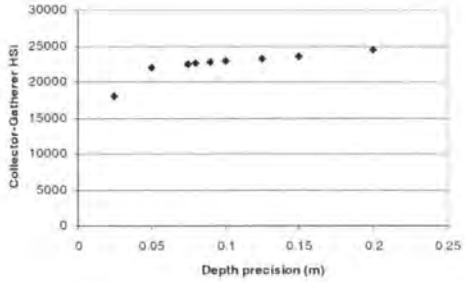
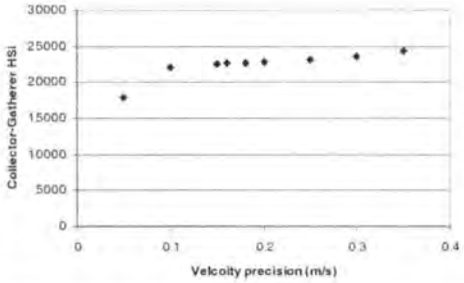
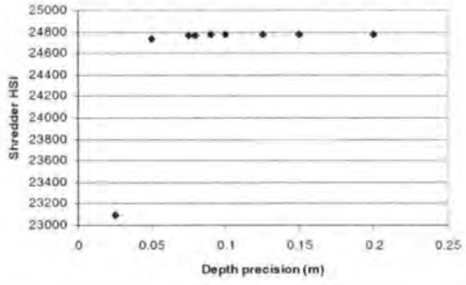
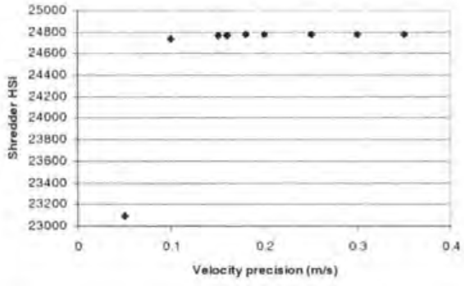


Figure 6.4.2 Figure to show the fuzzy habitat predictions for each of the brown trout age/size classes modelled at RU1 compared with the point depth and velocity accuracy calibration results. Red dots are the modelled habitat suitability index; blue dots are simulation accuracy.

6.4.2. Sensitivity to precision values

For the same simulation presented above, the potential impacts of altering the precision values within the fuzzy model were also investigated. As was explained in Section 6.3.4.3, the precision values control the transition from zero membership to perfect membership of a given class. It is therefore of interest to assess the potential impact of altering these precision values on habitat predictions. The values of the depth and velocity precision were varied within the simulation for the RU (0.25m resolution mesh) point velocity and depth calibration data, as to illustrate the effects of altering the precision values.

Figure 6.4.3 shows that the filterer guild are extremely sensitive to changes in the precision values of both depth and velocity, and much more so than due to hydraulic model uncertainties (Figure 6.4.1). This is followed by the collector-gatherer guild and then the predator guild. The shredder and scraper guilds are largely insensitive to velocity and depth precision except for the most precise values submitted. This implies that the distributions of velocities and depth predictions at this site are close to the suitable habitat limits of the filterer and collector gatherer guilds such as small changes in the depth and velocity precision cause large changes in the habitat suitability predictions. This implies that these guilds are most likely to be sensitive to changes in the flow regime.

<table>
<tr><th></th><th>Depth Precision</th><th>Velocity Precision</th></tr>
<tr><td>Filterer</td><td>
</td><td>
</td></tr>
<tr><td>Scraper</td><td>
</td><td>
</td></tr>
<tr><td>Collector-Gatherer</td><td>
</td><td>
</td></tr>
<tr><td>Shredder</td><td>
</td><td>
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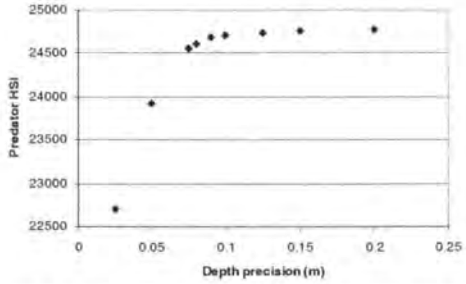
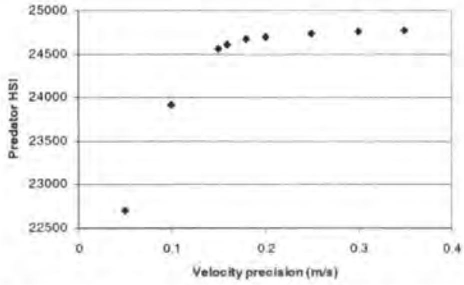
<tr><td>Predator</td><td>
</td><td>
</td></tr>
</table>

Figure 6.4.3. Figure to show the fuzzy habitat predictions for each of the macroinvertebrate guilds modelled at RU1 whilst altering the depth and velocity precision values within the fuzzy model.

The most affected brown trout life stage is 0+, which is affected much more than by the uncertainties propagated by the hydraulic model. The habitat predictions of >0+(<20cm) and >20cm brown trout were the next most influenced by the precision changes. Predictions of spawning habitat availability were largely unaffected until the smallest precision values used. This indicates that the distributions of suitable depths and velocities are close to the suitable habitat limits for 0+ brown trout, such that small changes in the velocity and precision values lead to large changes in habitat availability predictions.

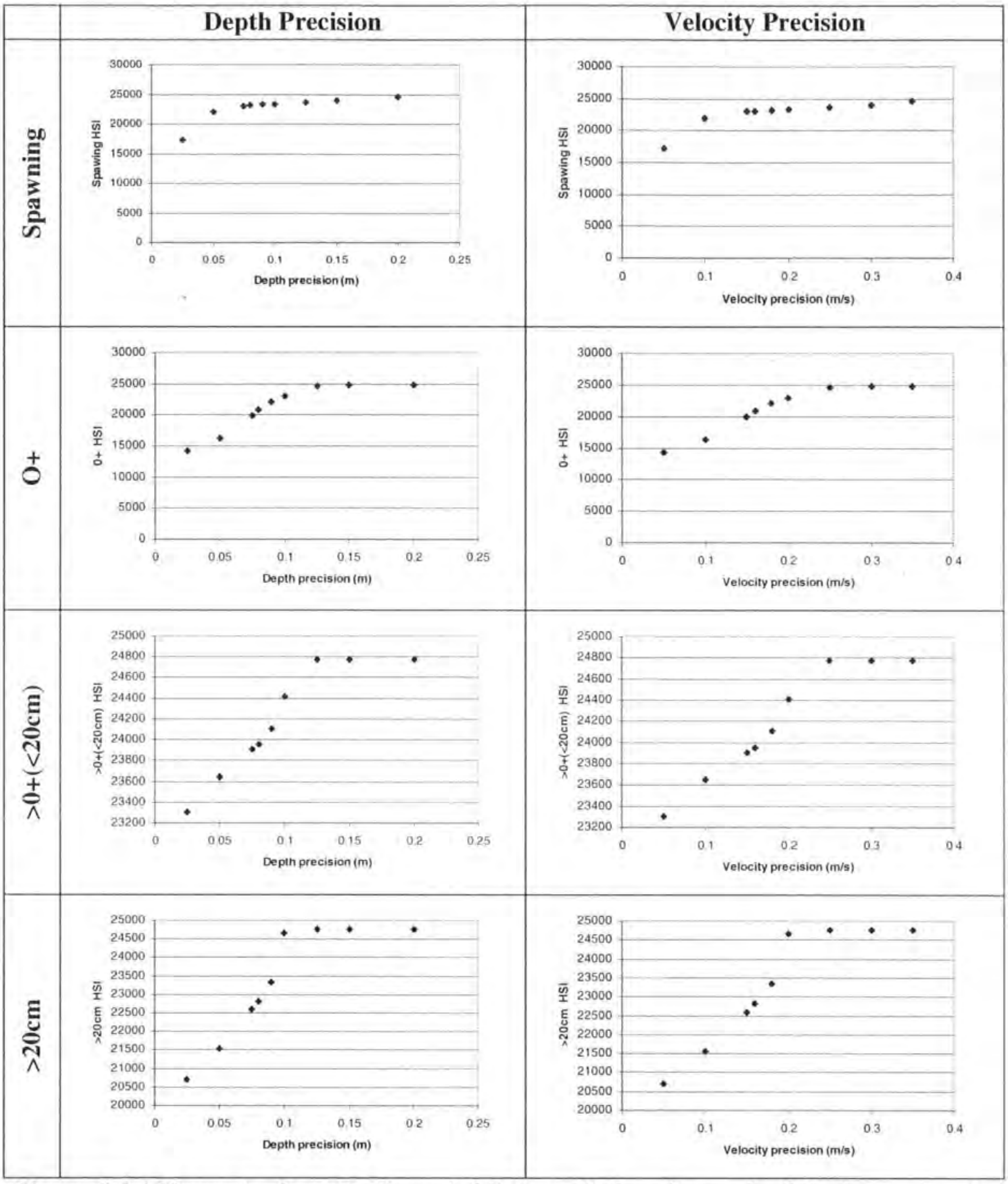


Figure 6.4.4 Figure to show the fuzzy habitat predictions for each of the brown trout age/size classes modelled at RU1 plotted against variations in the depth and velocity precision values.

6.4.3. Summary

This section has demonstrated that each of the fuzzy rules are sensitive to model uncertainty due to calibration and alterations in the precision values. In the example reach used, the most sensitive fuzzy rules were for the filterer guild and 0+ brown trout. On the basis of the results presented in Section 6.4.1, it will be possible to distinguish a predicted compensation flow impact from uncertainty due to mode calibration.

6.5. Macroinvertebrate habitat modelling

This section aims to present the results of the invertebrate guild fuzzy habitat modelling. The hydraulic simulations used as the basis for the fuzzy model in this and the subsequent section (Section 6.6) are those which had the nearest match to the parameter set which provided the best calibration with the point depth and velocity measurements (see Sections 5.4 and 5.6). At each site, the mesh with the best calibration results was used in order to aid the simplicity of presentation The first part of this section will address the habitat simulations for the pre-change conditions, and the second section will address the post-change simulations. In both sections the habitat predictions will be compared to the measured invertebrate guilds at the nearby sites, as a form of abstract validation.

6.5.1. Pre-change habitat predictions

Figure 6.5.1 to 6.5.5 show the proportions of feeding guilds found at the nearby macroinvertebrate sampling sites. This data will be used to abstractly validate the fuzzy guild model throughout this section as despite the habitat estimates provided by the fuzzy model not being ecologically meaningful, a comparison is interesting. The comparison is restricted to the sampling seasons where augmentation is at a minimum (Figures 2.10 and 2.11 in Chapter 2), namely summer and autumn 2003.

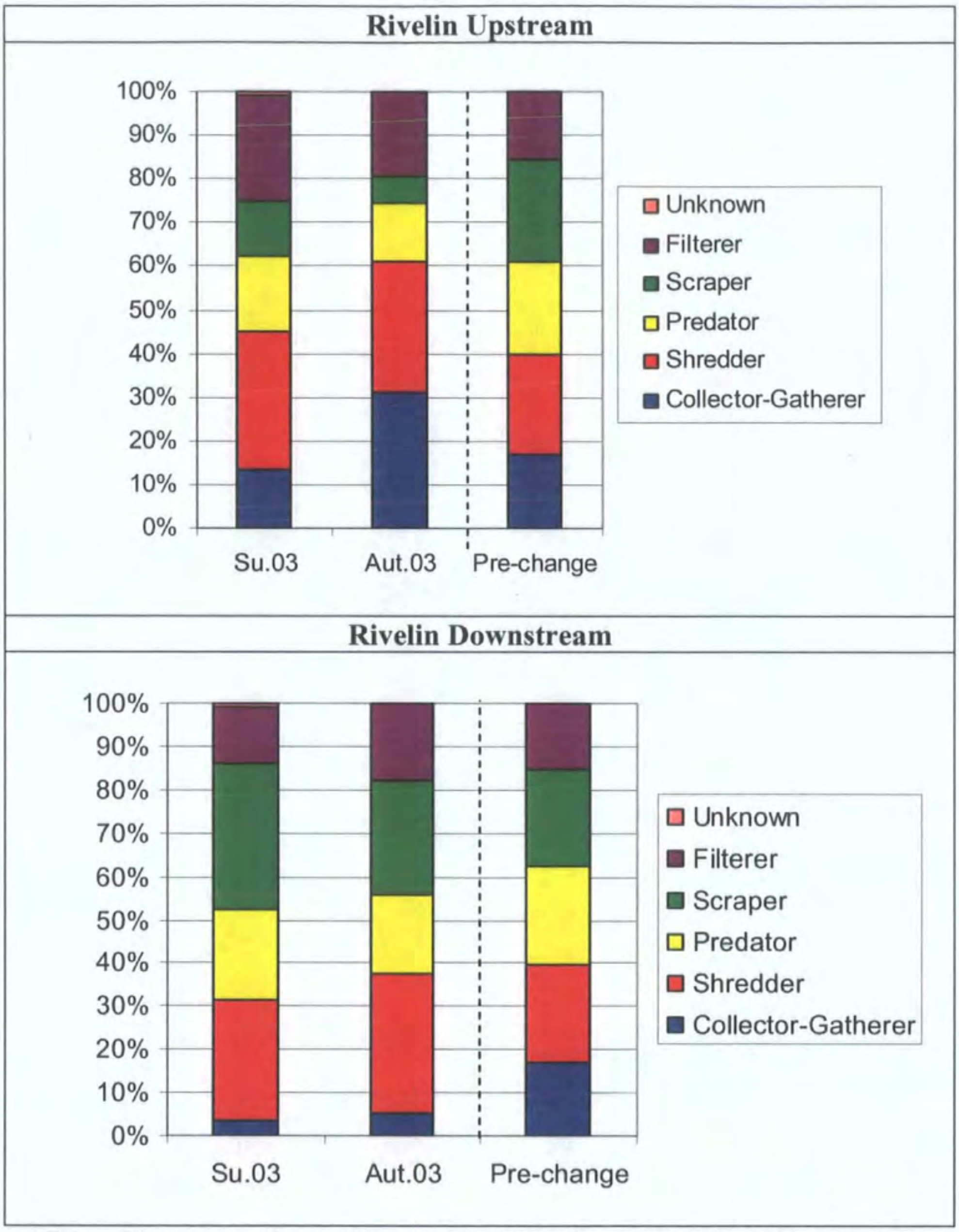
6.5.1.1. Rivelin

At RU (Table 6.5.1), the highest available habitat predictions were for scrapers, shredders and predators, but with lower predictions of the collector-gathering guild. Figure 6.5.1 however, shows that the collector-gatherer guild is important at RU especially in autumn 2003, and this is not reflected in the habitat predictions of fuzzy model. The fuzzy model also appears to under predict the filterer guild.

At RD, the best habitat predictions again appear to be for the scraper, shredder and predator guilds (Table 6.5.1), and these three guilds dominate the invertebrate structure in two pre-change sampling seasons of interest (Figure 6.5.1). These periods are when the discharge of the river may have been closer to the modelled discharge than for any of the other sampling seasons. The habitat predictions for the filterer and collector gatherer guilds were lower than for the other guilds, and this is also matched by the observed guild proportions (Figure 6.5.1).

	Total Collector G Habitat	**Weighted Collector G Habitat**	**Total shredder habitat**	**Weighted shredder habitat**	**Total predator habitat**	**Weighted predator habitat**	**Total Scraper Habitat**	**Weighted Scraper Habitat**	**Total Filterer Habitat**	**Weighted Filterer Habitat**
RU pre change	12111	72.39	16661	99.58	15394	92.01	16731	100	11239	67.17
RD pre change	19516	76.47	25520	100	25520	100	25520	100	17083	66.94

Table 6.5.1. Table to show the pre-change habitat predictions for the macroinvertebrate guilds at both Rivelin modelling sites for the optimal mesh and optimal parameter set.



Rivelin Upstream

Rivelin Downstream

Figure 6.5.1. Figure to show the proportion of measured guilds in summer and autumn of 2003 against the proportions of pre-change predicted HSI at the two Rivelin modelling and associated invertebrate sites.

6.5.1.2. Loxley

At LU, Table 6.5.2 shows that there was marginally higher habitat availability for the shredder and scraper guilds. Figure 6.5.2 shows that at the nearby invertebrates sampling site, there is a large proportion of shredding invertebrates present, but very few scraping invertebrates. The other dominant measured guild was the filterer guild which did not have the highest habitat prediction. What is striking about Figure 6.5.2 is the within year variability in the measured feeding guilds. This perhaps indicates that other factors are controlling the invertebrate populations within the Loxley (a possibility alluded to in Chapter 3).

The LD site has a similar pattern of predicted habitat availability with the scraper and shredder guilds having the highest predicted habitat (Table 6.5.2), but Figure 6.5.1 shows that the proportions of predicted habitat were similar. The habitat predictions for the shredder guild probably slightly higher than for any other guilds and this is reflected by the relatively high proportions of shredding invertebrates at the nearby invertebrate site (Figure 6.5.1). Despite the lower within year variability at the downstream Loxley site, it is difficult to see a correlation between predicted and observed guild proportions.

	Total collector Gatherer Habitat	**Weighted collector gatherer Habitat**	**Total Shredder habitat**	**Weighted shredder habitat**	**Total predator habitat**	**Weighted Predator habitat**	**Total Scraper Habitat**	**Weighted Scraper Habitat**	**Total Filterer Habitat**	**Weighted Filterer Habitat**
LU pre change	37106	92..76	39982	99.95	37183	92.95	40003	100	37481	93.67
LD pre change	33481	94.7	35346	99.97	33882	95.83	35356	100	33063	93.51

Table 6.5.2. Table to show the pre-change habitat predictions for the macroinvertebrate guilds at both Loxley modelling sites. For the optimal mesh and optimal parameter set.

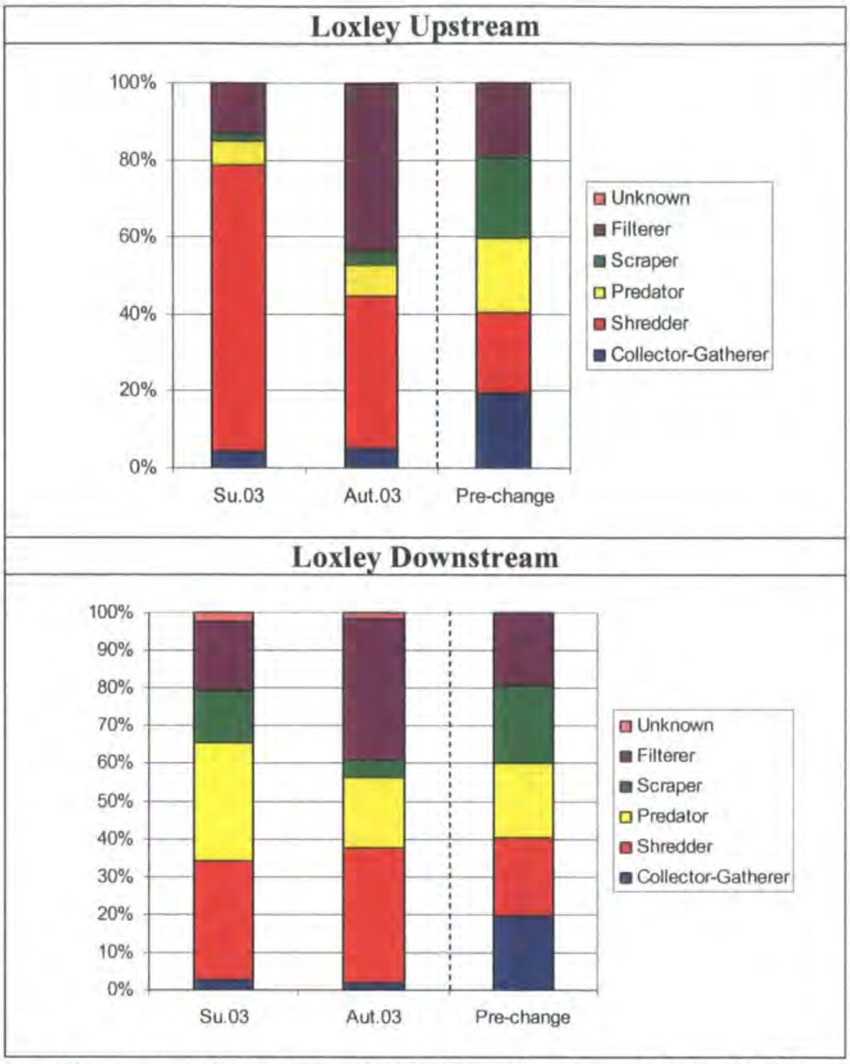


Figure 6.5.2. Figure to show the proportion of measured guilds in summer and autumn of 2003 against the proportions of pre-change predicted HSI at the two Loxley modelling and associated invertebrate sites.

6.5.1.3. Summary

At the upstream site on the Rivelin there appeared to be little correlation between the measured invertebrate guilds and the available habitat predictions. At the downstream site on the Rivelin the guild proportions were dominated by scraper shredder and predator for both the measured and predicted. There appeared to be a greater detachment of the Loxley habitat predictions and the proportions of guilds present in the river (with the Loxley not being hydraulically limited?). The shredder and scraper guilds have the highest habitat predictions of any of the guilds, and whilst the shredder guild is important in the measured invertebrate populations there are smaller proportions of the scraper guild predicted. This perhaps indicates a detachment of environmental variables and the invertebrate population in the Loxley, a possibility that was explored in Chapter 3.

6.5.2. Post-change habitat predictions

6.5.2.1. Rivelin upstream (RU)

Table 6.5.3 shows that there was an increase in predicted habitat for each of the invertebrate guilds simulated at RU, with the largest increase found for filterers and collector-gatherers, but Figure 6.5.3 shows that this increase is proportionally very small. This suggests that altering the compensation flows may not impact adversely on any aspect of the macroinvertebrate population at RU in the case that hydraulics are limiting. The amount of change in predicted habitat availability for each of the guilds is greater than the amount of change seen due to model calibration uncertainty in Figure 6.4.1.

This is an important piece of information for decision makers to be in possession of. It is also apparent from the findings presented in Chapter 3 that there was no large change in the Rivelin macroinvertebrates as a consequence of altering the compensation flows (Figure 6.5.3). Furthermore, Figure 6.5.3 appears to show that there was no systematic change in the actual proportions of feeding guilds at the nearby sampling site, which reflects the fuzzy predictions of the feeding guilds. Figure 6.5.3 shows that the amount of change predicted by the model appears inconsequential compared to natural variations.

	Total collector gatherer Habitat	Weighted collector gatherer Habitat	Total Shredder Habitat	Weighted Shredder Habitat	Total Predator habitat	Weighted Predator Habitat	Total Scraper Habitat	Weighted Scraper Habitat	Total Filterer Habitat	Weighted Filterer Habitat
RU pre change	12111	72.39	16661	99.58	15394	92.01	16731	100	11239	67.17
RU post change	16057	81.94	19427	99.13	18150	96.62	19464	99.32	15576	79.48
Percent Change	32.6%		16.6%		17.9%		16.3%		38.0%	

Table 6.5.3. Table to show the pre-and post change habitat predictions for the macroinvertebrate guilds at RU. For the optimal mesh and optimal parameter set.

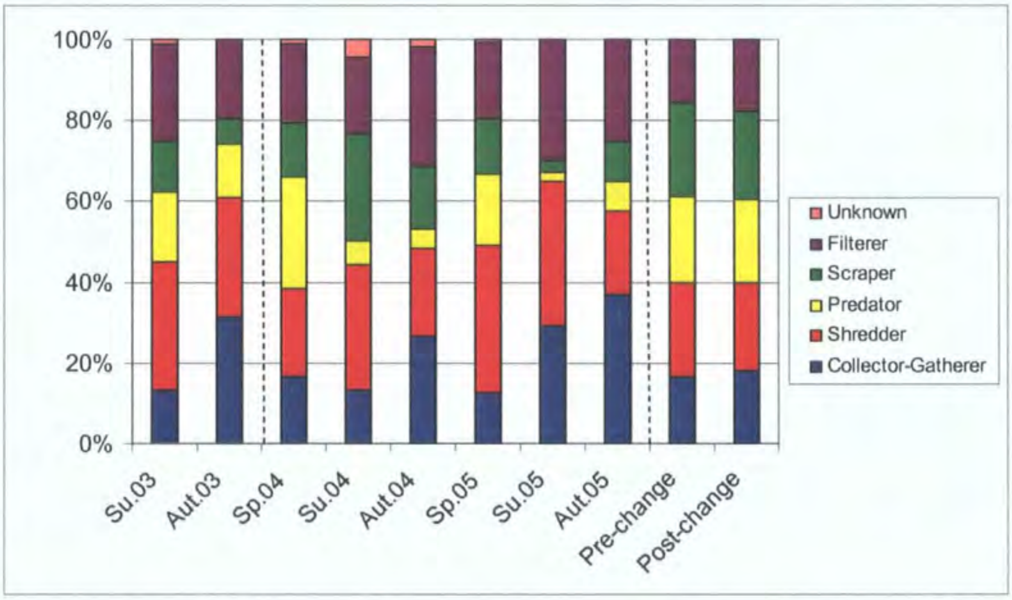


Figure 6.5.3. Figure to show the proportions of each feeding guild found at the RU macroinvertebrate sampling site from summer 2003 to autumn 2005; combined with the proportions of HIS from the pre and post change habitat predictions.

6.5.2.2. Rivelin downstream (RD)

At RD an increase in predicted habitat was seen for each of the invertebrate guilds apart from the predator guild, with the largest increases again seen in the habitat predictions for the filterer and collector-gatherer guilds. The predicted changes in available habitat for the collector-gatherer and filterer guilds are higher than changes due to model uncertainty quantified in Figure 6.4.1. But the changes to the remaining guilds are indistinguishable from the model uncertainties. Figure 6.5.4 shows that the changes in predicted available habitat are very small compared to the variations in the proportions of guilds at the nearby sampling site.

	Total collector gatherer Habitat	Weighted collector gatherer Habitat	Total Shredder Habitat	Weighted Shredder Habitat	Total Predator habitat	Weighted Predator Habitat	Total Scraper Habitat	Weighted Scraper Habitat	Total Filterer Habitat	Weighted Filterer Habitat
RD pre change	19516	76.47	25520	100	25520	100	25520	100	17083	66.94
RD post change	22123	84.78	26095	100	25371	97.22	26096	100	20957	80.31
Percent Change	13.4%		2.3%		-0.6%		2.3%		22.7%	

Table 6.5.4. Table to show the pre-and post change habitat predictions for the macroinvertebrate guilds at RD. For the optimal mesh and optimal parameter set.

When the predictions are abstractly validated, Figure 6.5.4 shows that there is an increase in the proportions of filtering invertebrates present at the invertebrate site, with little or no change to be observed in the collector gatherer guild. There does however, appear to be a reduction in the measured proportion of predator invertebrates found at RD, apart from in spring 2005, which agrees with the predictions of the RD simulations.

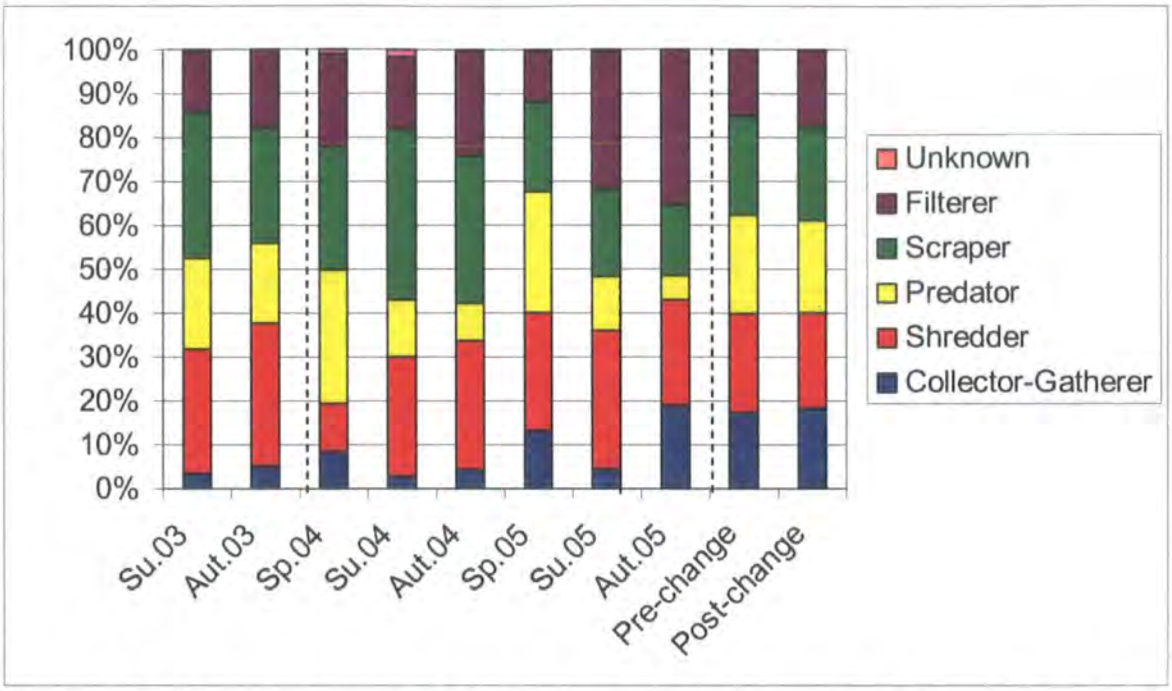


Figure 6.5.4. Figure to show the proportions of each feeding guild found at the RD macroinvertebrate sampling site from summer 2003 to autumn 2005; combined with the proportions of HIS from the pre and post change habitat predictions.

6.5.2.3. Loxley upstream (LU)

Table 6.5.5 shows that there is a decrease in predicted overall habitat availability and a decrease in habitat quality for each of the feeding guilds under the post-change flows. However, this decrease is so small that it is within the potential changes to habitat predictions caused by model calibration uncertainty (Figure 6.4.1). Figure 6.5.5 shows that this decrease does not appear to affect the proportions of predicted available habitat at all.

	Total Collector G Habitat	Weighted Collector G Habitat	Total shredder habitat	Weighted shredder habitat	Total Predator habitat	Weighted Predator Habitat	Total Scraper Habitat	Weighted Scraper Habitat	Total Filterer Habitat	Weighted Filterer Habitat
LU pre change	37106	92.76	39982	99.95	37183	92.95	40003	100	37481	93.67
LU post change	36160	91.57	39457	99.92	36278	91.87	39488	100	36502	92.44
Percent Change	-2.5%	N/A	-1.3%	N/A	-2.4%	N/A	-1.3%	N/A	-2.6%	N/A

Table 6.5.5. Table to show the pre-and post change habitat predictions for the macroinvertebrate guilds at LU. For the optimal mesh and optimal parameter set.

There is a lack of change in the observed proportions of feeding guilds at the nearby macroinvertebrate sampling site. It may be that if the habitat has decreased similarly for each of the feeding guilds their proportions may not change. The shredder guild remains dominant in the post-change invertebrate communities. This is despite the shredder guild having habitat predictions of a similar size to the other guilds and so it may be that factors other than velocity and depth are controlling invertebrate populations at this site.

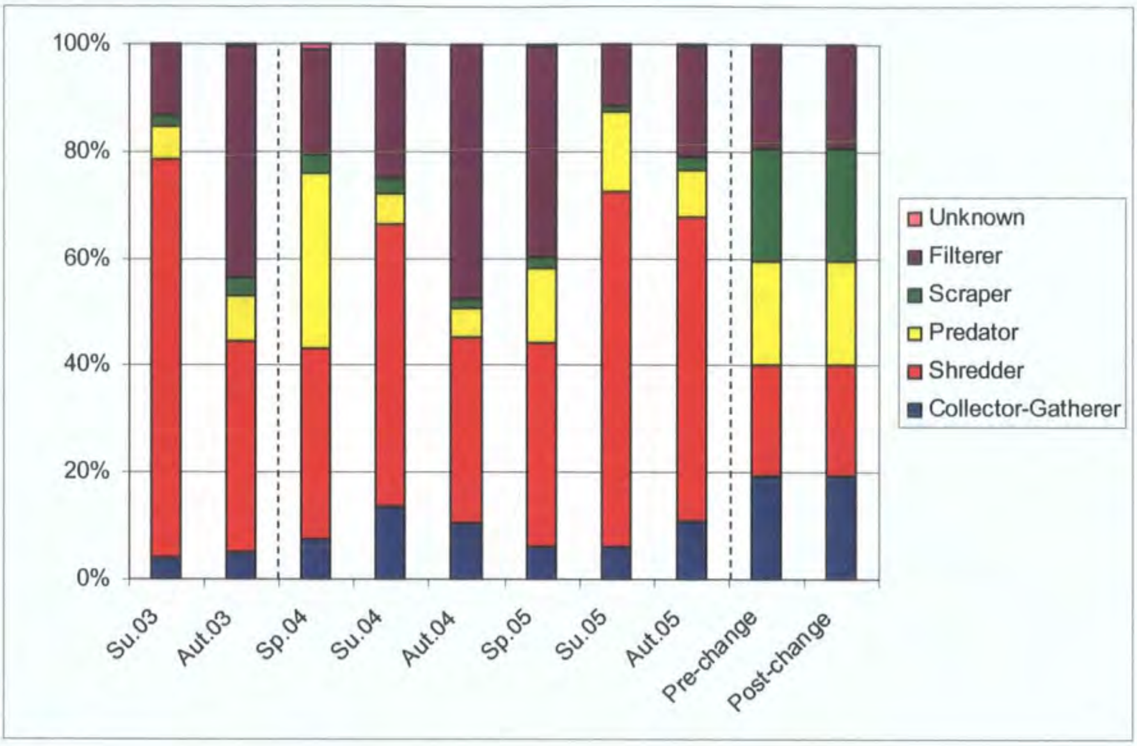


Figure 6.5.5. Figure to show the proportions of each feeding guild found at the LU macroinvertebrate sampling site from summer 2003 to autumn 2005; combined with the proportions of HIS from the pre and post change habitat predictions.

6.5.2.4. Loxley downstream (LD)

Table 6.5.6 shows that there are again reductions in predicted habitat for each guild apart from the predator guild, however, these changes are small and are within the potential changes caused by uncertainty in model calibration (Figure 6.4.1). Again, these reductions are not substantial, and do not alter the proportions of predicted available habitat for any of the guilds (Figure 6.5.6).

	Total Collector G Habitat	Weighted Collector G Habitat	Total shredder habitat	Weighted shredder habitat	Total Predator habitat	Weighted Predator Habitat	Total Scraper Habitat	Weighted Scraper Habitat	Total Filterer Habitat	Weighted Filterer Habitat
LD pre change	33481	94.7	35346	99.97	33882	95.83	35356	100	33063	93.51
LD post change	32769	94.00	34857	99.99	34205	98.12	34859	100	32236	92.48
Percent Change	-2.1%		-1.4%		1.0%		-1.4%		-2.5%	

Table 6.5.6. Table to show the pre-and post change habitat predictions for the macroinvertebrate guilds at LD. For the optimal mesh and optimal parameter set.

Despite the increase in predicted predator habitat, Figure 6.5.6 shows no systematic increase in the predator guild in the observed guild proportions. The shredder guild remains dominant, as in the upstream site, despite the slight decrease in total habitat availability. The lack of alteration in the proportions of available habitat predicted (Figure 6.5.6) may mean that the within-year variation may override any habitat changes.

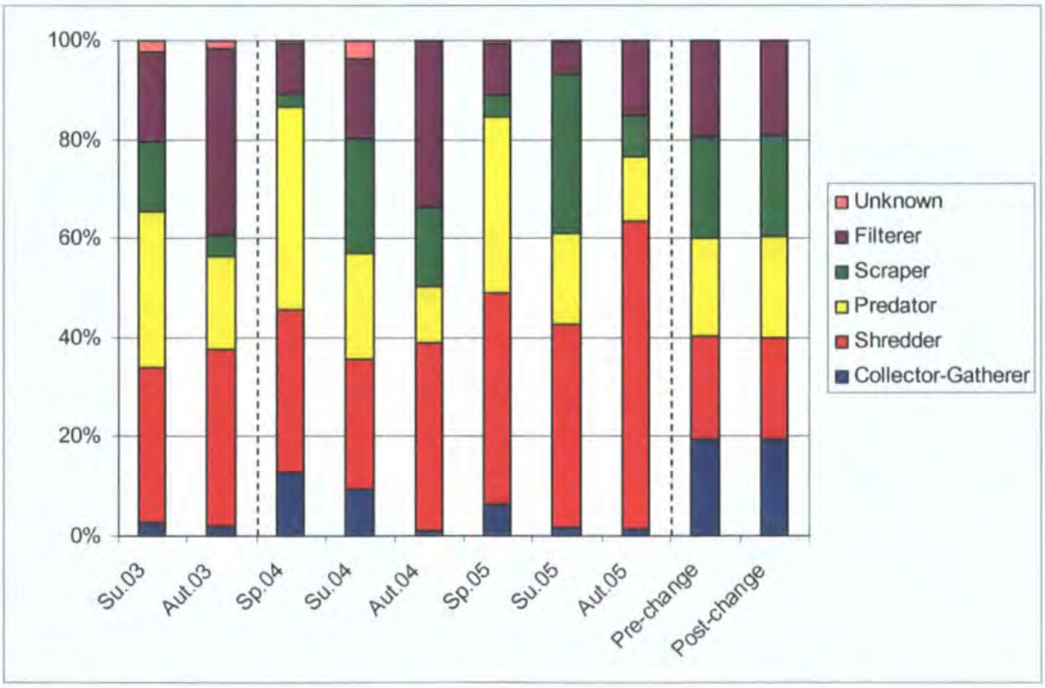


Figure 6.5.6. Figure to show the proportions of each feeding guild found at the LD macroinvertebrate sampling site from summer 2003 to autumn 2005; combined with the proportions of HSI from the pre and post change habitat predictions.

6.5.2.5. Summary

In the Rivelin there was an increase in predicted available habitat for each of the fuzzy guilds (apart from a very small decrease in predators at RD). The greatest predicted increase was for the filterer and collector-gatherer invertebrates at both of the Rivelin sites. It was however, difficult to relate these changes in habitat predictions to the observed invertebrate populations. But the predicted changes to the Rivelin habitat were generally greater than changes which could have occurred due to model uncertainty during calibration (Figure 6.4.1). There were much smaller predicted habitat changes in the Loxley which cannot be distinguished from potential changes caused by model calibration, with a decrease in available habitat predicted for each invertebrate guild (apart from predator at LD. There was no systematic change in the observed feeding guilds at either of the Loxley sites.

6.5.3. Summary

In the pre-change habitat predictions for the Rivelin, there appeared to be some associations between the habitat predictions and the proportions measured invertebrates at the nearby sites (especially at the downstream site). In the Loxley, there was much less agreement between the habitat availability predictions and the observed proportions of guilds. A detachment of invertebrates and environmental variables in the Loxley was suggested in Chapter 3, and this may be further evidence of this or there may be other limiting factors. The post-change habitat availability predictions appeared not to match the observed invertebrate guilds on either of the rivers. This is surprising given the large amount of change in the Rivelin habitat predictions. This could perhaps indicate that two years post-change data is insufficient to capture any significant change in the macroinvertebrate guilds.

6.6. Brown trout habitat modelling

It is the purpose of this section to examine the output from modelling the habitat of four life stages of brown trout which are of interest. It was seen in Section 6.4 that the fuzzy rules for brown trout are sensitive to alterations in velocities and depths. The simulations used in this section are the ones which have the closest parameter sets to those which provided the best calibration results with the point depth and velocity validation (see Sections 5.4 and 5.6). The first section will examine the pre-change modelling output, whilst the second section will compare the pre and post change habitat predictions. In both cases the habitat predictions will be related back to the trout data explored in Chapter 4 as a form of abstract validation.

6.6.1. Pre-change modelling

The purpose of this section is twofold: firstly, to assess the model performance with respect to the pre-change trout data; and secondly to provide a baseline from which any changes in habitat predictions can be assessed.

6.6.1.1. Rivelin

The habitat predictions show that the most pre-change available habitat for both sites in the Rivelin is for 0+ trout, whilst the spawning habitat is worse than the habitat predictions for live trout. Table 6.6.1 also shows that RD has slightly better habitat quality for the older trout, this is despite the same discharge simulated in both meshes.

	Total Spawning Habitat	**Weighted Spawning Habitat**	**Total 0+ Habitat**	**Weighted 0+ Habitat**	**Total >0+(<20cm) Habitat**	**Weighted 0+(<20cm) Habitat**	**Total >20cm Habitat**	**Weighted >20cm Habitat**
RU pre change	13735	82.09	15826	94.59	15990	95.57	15354	91.77
RD pre change	21997	86.24	25507	100	24842	97.39	25261	98.99

Table 6.6.1. Table to show the pre change habitat predictions for the different life stages of brown trout at both Rivelin modelling sites. For the optimal mesh and optimal parameter set.

Figure 6.6.1 shows the pre-change HABSCORE Habitat Quality Score (HQS) values for 0+ trout were higher than for any other age-size class, and this was also seen in the fuzzy habitat predictions. For information, RU is proximal to fishing site 1, whilst RD is proximal to site 7. It is seen that at fishing site one, the HQS of >0+(<20cm) appears to be higher than that of 0+ trout, but this is not replicated in the fuzzy habitat predictions. But in general, along the Rivelin, the 0+ habitat is better than that available for the older age/size classes. This is replicated in the fuzzy habitat predictions for both modelling sites. But the comparison of fuzzy HSI predictions is difficult as the values are not biologically meaningful in the same way as predicted HQS densities.

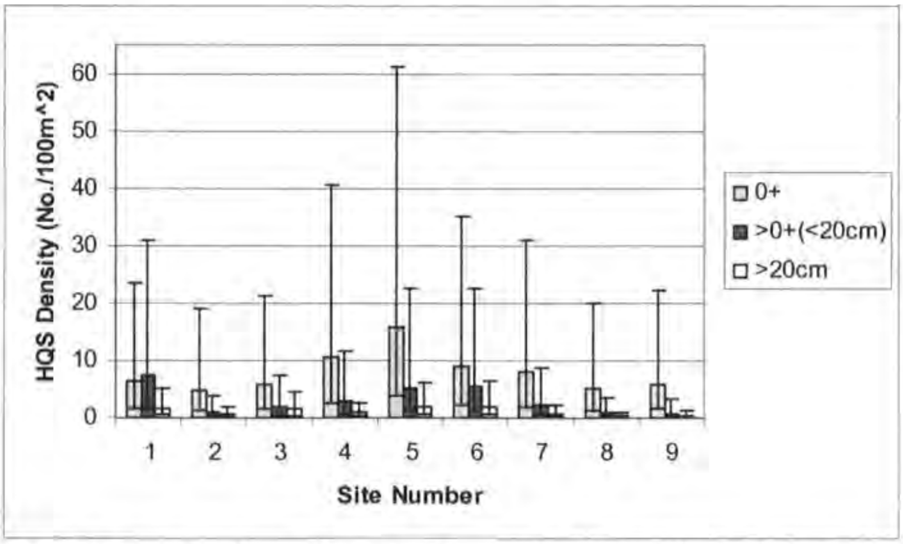


Figure 6.6.1. Figure to show the pre-change HABSCORE Habitat Quality Score (HQS) at each of the Rivelin fishing sites.

For the pre-change trout densities (Figure 6.6.2), the densities of >0+(<20cm) trout were generally higher than any of the other age/size classes, and this is not what was expected either from the fuzzy logic habitat modelling or HABSCORE. This phenomenon was discussed in detail in Chapter 4 and a number of reasons for this were discussed, with the results suggesting that >0+(<20cm) brown trout are habitat limited within the Rivelin. Both the fuzzy habitat HSI predictions and the HQS results suggest a good available habitat for 0+ brown trout, but the observed densities do not reflect that. This may be due to a lack of spawning habitat, a problem which is alluded to by the fuzzy habitat predictions.

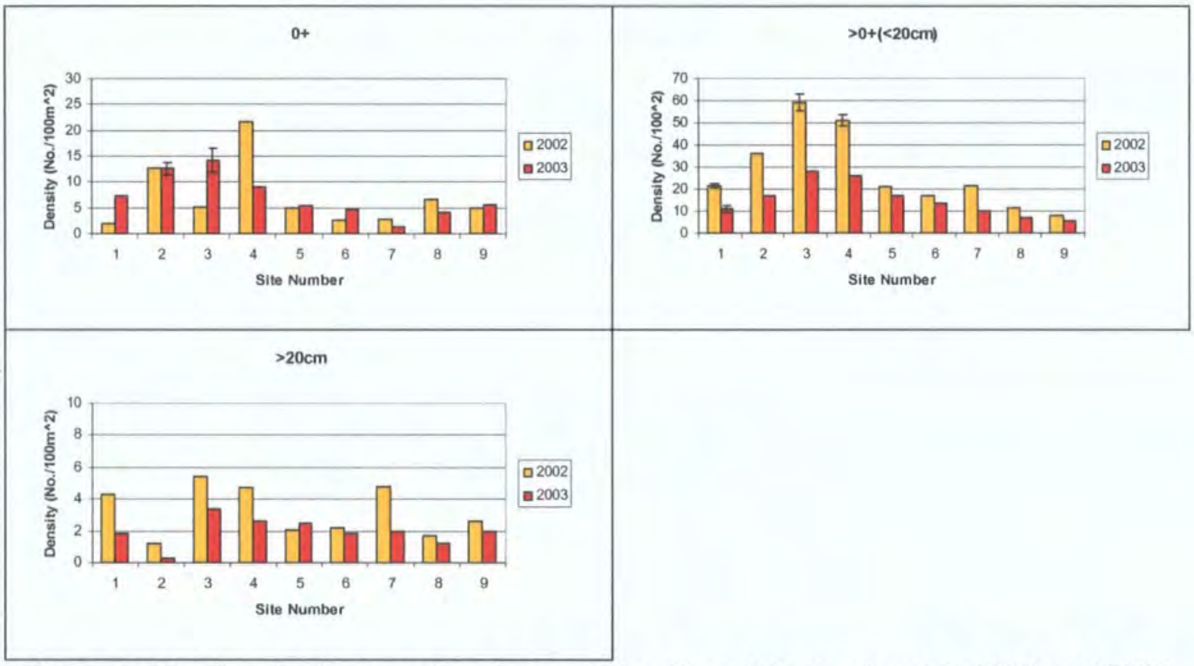


Figure 6.6.2. Graphs to show the observed densities of brown trout in 2002 and 2003 for the three main age/size classes for the Rivelin. Site one is the uppermost and site 9 the lowest. Error bars showing the Carle-Strub variance. Scales vary to aid visualisation.

6.6.1.2. Loxley

Table 6.6.2 shows that the best habitat predictions at LU were found for the two older age/size classes of trout (>0+(<20cm) and >20cm). Spawning again had the lowest habitat predictions. At LD, the habitat predictions for each life stage were similar, apart from the habitat predictions for spawning which were lower compared to the other life stages simulated at that site. In terms of habitat quality, LU had better habitat quality for older trout, whilst LD has better habitat quality predictions for younger trout and spawning.

	Total Spawning Habitat	**Weighted Spawning Habitat**	**Total 0+ Habitat**	**Weighted 0+ Habitat**	**Total >0+(<20cm) Habitat**	**Weighted 0+(<20cm) Habitat**	**Total >20cm Habitat**	**Weighted >20cm Habitat**
LU pre change	36966	92.41	37799	94.49	39783	99.45	39884	99.67
LD pre change	34200	96.73	35293	99.82	35065	99.18	35256	97.72

Table 6.6.2. Table to show the pre change habitat predictions for the different life stages of brown trout at both Loxley modelling sites. For the optimal mesh and optimal parameter set.

Interestingly, Figure 6.6.3 does not replicate the fuzzy habitat predictions. The pre-change HQS values for the Loxley show the highest HQS values for 0+ trout with lower densities for the older age/size classes, whilst the fuzzy predictions showed a higher habitat availability for older trout at LU, and similar habitat availability for all age/size classes at LD. This could be due to the fact that habitat availability as predicted by the fuzzy model cannot be directly linked to brown trout density (as was discussed in the literature review).

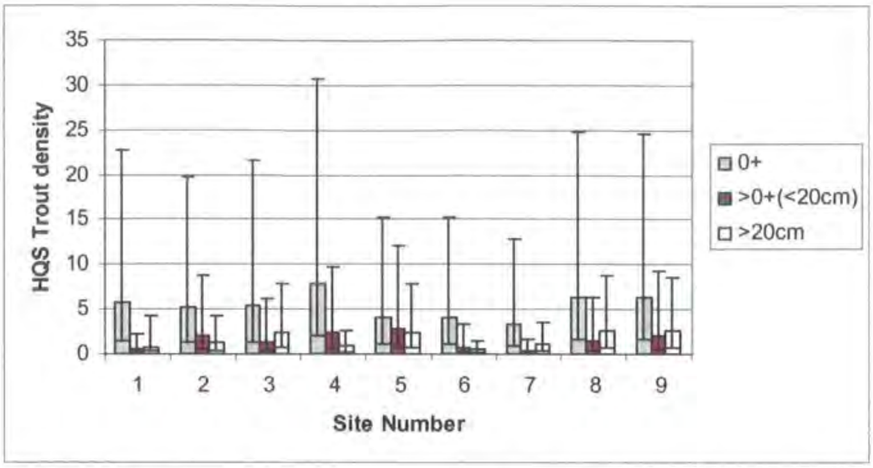


Figure 6.6.3. Figure to show the pre-change HABSCORE HQS scores on the Loxley.

Figure 6.6.4 shows that generally the highest observed densities of brown trout were found in the >0+(<20cm) age/size class. This corresponds with the high fuzzy habitat predictions for that age/size class at LU. The observed densities of >20cm are lower than for either of the other age/size classes, and this is despite the fuzzy model predicting more available habitat for this age/size class than for any other. Fewer older brown trout can be supported by a river due to their greater size, so despite good habitat, the >20cm trout on the Loxley may be density dependent, or the populations may be controlled by other factors.

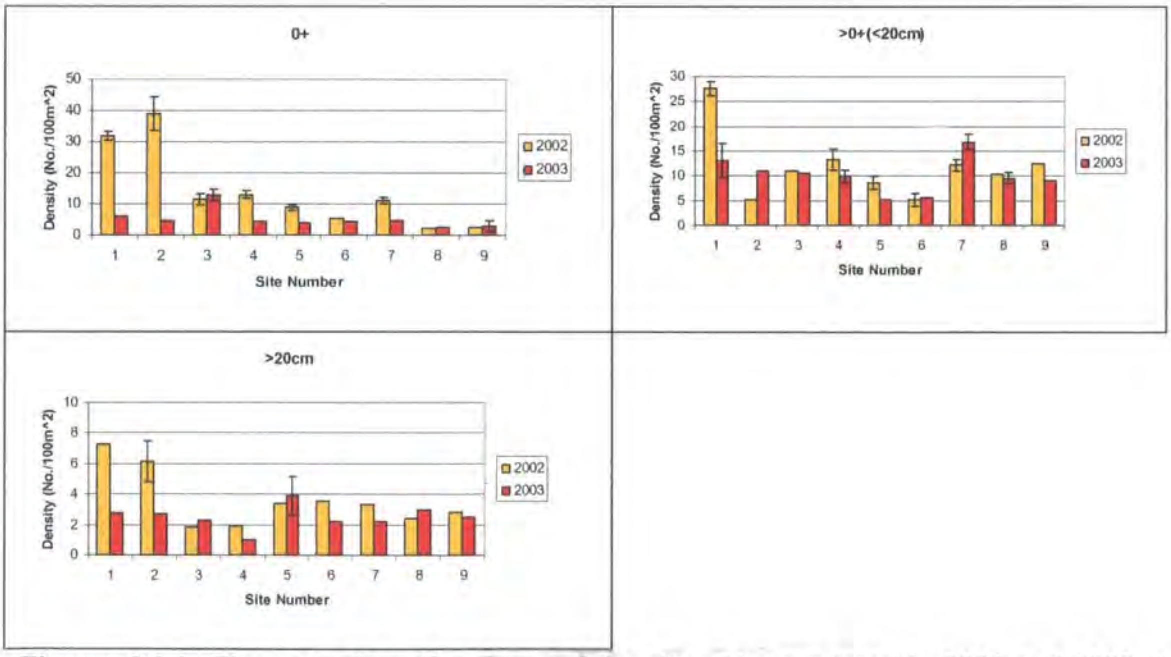


Figure 6.6.4. Figure to show the observed density of brown trout in 2002 and 2003 at each site on the Loxley for three age/size classes. Scales vary to aid visualisation.

6.6.1.3. Summary

This section showed how difficult it is to relate available habitat predictions to actual fisheries data. This is because available habitat predictions represent only one set of habitat controls and because of deviations from habitat limitation it is difficult to compare against measures of trout density (which are life-stage specific). It was observed that the 0+ age/size class had the highest habitat prediction on the Rivelin, whilst on the Loxley, the highest habitat predictions were for >0+(<20cm) and >20cm brown trout.

6.6.2. Post-change modelling

6.6.2.1. Rivelin

Table 6.6.3 also shows that a large increase in spawning habitat available under the post-change regime compared to the pre-change regime. There is an increase in total 0+ habitat availability within this study reach. The magnitude of the increase in predicted habitat of around 3000 are much higher than those changes in habitat predictions which could be due to uncertainties in model calibration shown in Figure 6.4.2. There was also an increase in total >0+(<20cm) trout habitat, which is interesting as in the 2005 brown trout population there was a faster growth rate than seen in either of the pre-change years. This could be due to habitat improvement. This increase in >0+(<20cm) trout habitat was also seen when the velocity and depth were considered independently in Chapter 5. There is also an increase in >20cm trout habitat. Thus, it appears that the new flow regimes are positive for the available habitat for brown trout in the Rivelin.

	Total Spawning Habitat	**Weighted Spawning Habitat**	**Total 0+ Habitat**	**Weighted 0+ Habitat**	**Total >0+(<20cm) Habitat**	**Weighted >0+(<20cm) Habitat**	**Total >20cm habitat**	**Weighted >20cm habitat**
RU pre change	13735	82.09	15826	94.59	15990	95.57	15354	91.77
RU post change	17063	87.07	18738	95.62	18834	96.11	18065	92.18
Percent change	24.2%		18.4%		14.7%		17.7%	

Table 6.6.3. Table to show the output of the brown trout habitat simulations for the best parameter sets at RU. For the optimal mesh and optimal parameter set.

Table 6.6.4 shows that at RD, there is an increase in habitat availability for each of the life stages simulated. The predicted increase in habitat availability for the >0+(<20cm) and >20cm life stages is smaller than for the other life stages. When compared to the changes in habitat arising form model calibration uncertainty (Figure 6.4.2), it is probable that the increase in habitat predicted here is indistinguishable. The increase in predicted habitat availability was particularly large for spawning. As predicted in Chapter 5, there is an increase in the amount of habitat available for >0+(<20cm) at the lower Rivelin site, which again may have been an influence on the increased growth rates for older trout in 2005 compared to the pre-change trout populations.

	Total Spawning Habitat	**Weighted Spawning Habitat**	**Total 0+ Habitat**	**Weighted 0+ Habitat**	**Total >0+(<20cm) Habitat**	**Weighted 0+(<20cm) Habitat**	**Total >20cm Habitat**	**Weighted >20cm Habitat**
RD pre change	21997	86.24	25507	100	24842	97.39	25261	98.99
RD post change	23752	91.03	25657	98.34	25619	98.19	25668	98.36
Percent Change	7.8%		0.6%		3.1%		1.6%	

Table 6.6.4. Table to show the output of the brown trout habitat simulations for the best parameter sets at RD. For the optimal mesh and optimal parameter set.

Figure 6.6.5 shows that HABSCORE predicted a decrease in 0+ habitat in terms of HQS, with no real change in the >0+(<20cm) trout habitat. It is interesting the fuzzy model did predict an increase in >0+(<20cm) trout habitat (especially at RU) as did the hydraulic parameters considered alone. The HABSCORE HQS suggested an increase in habitat for >20cm trout, which was also predicted by the fuzzy habitat model at RU.

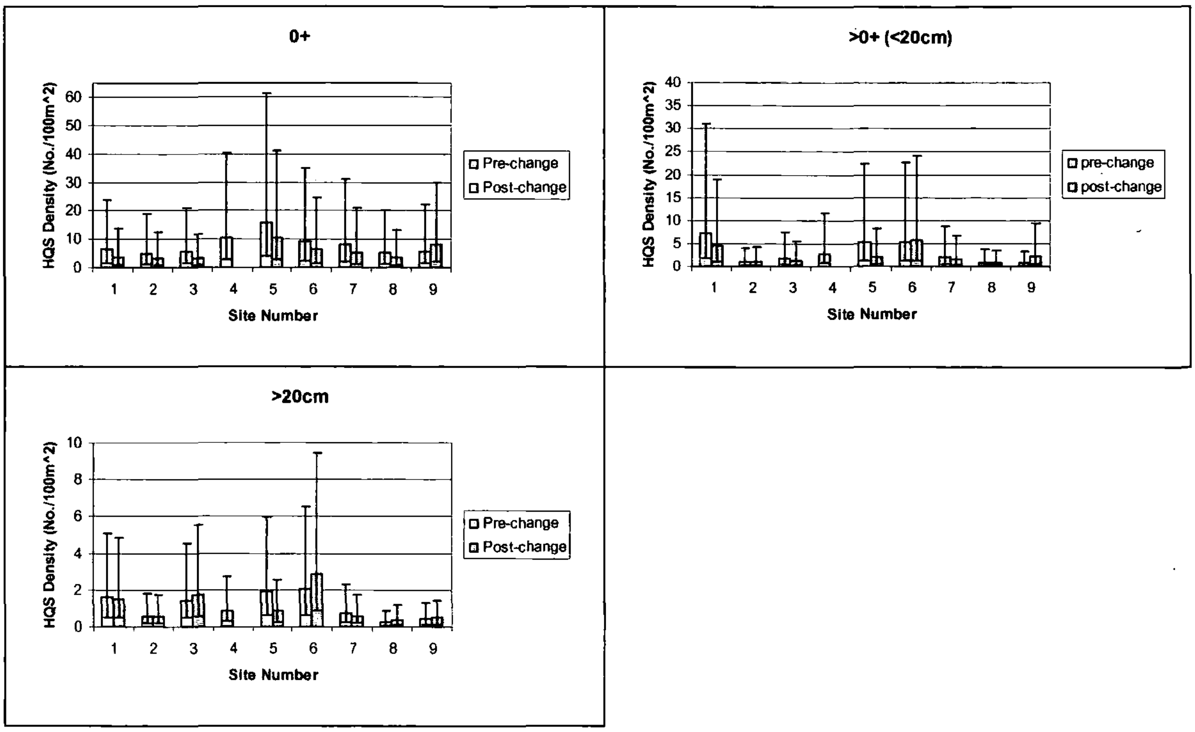
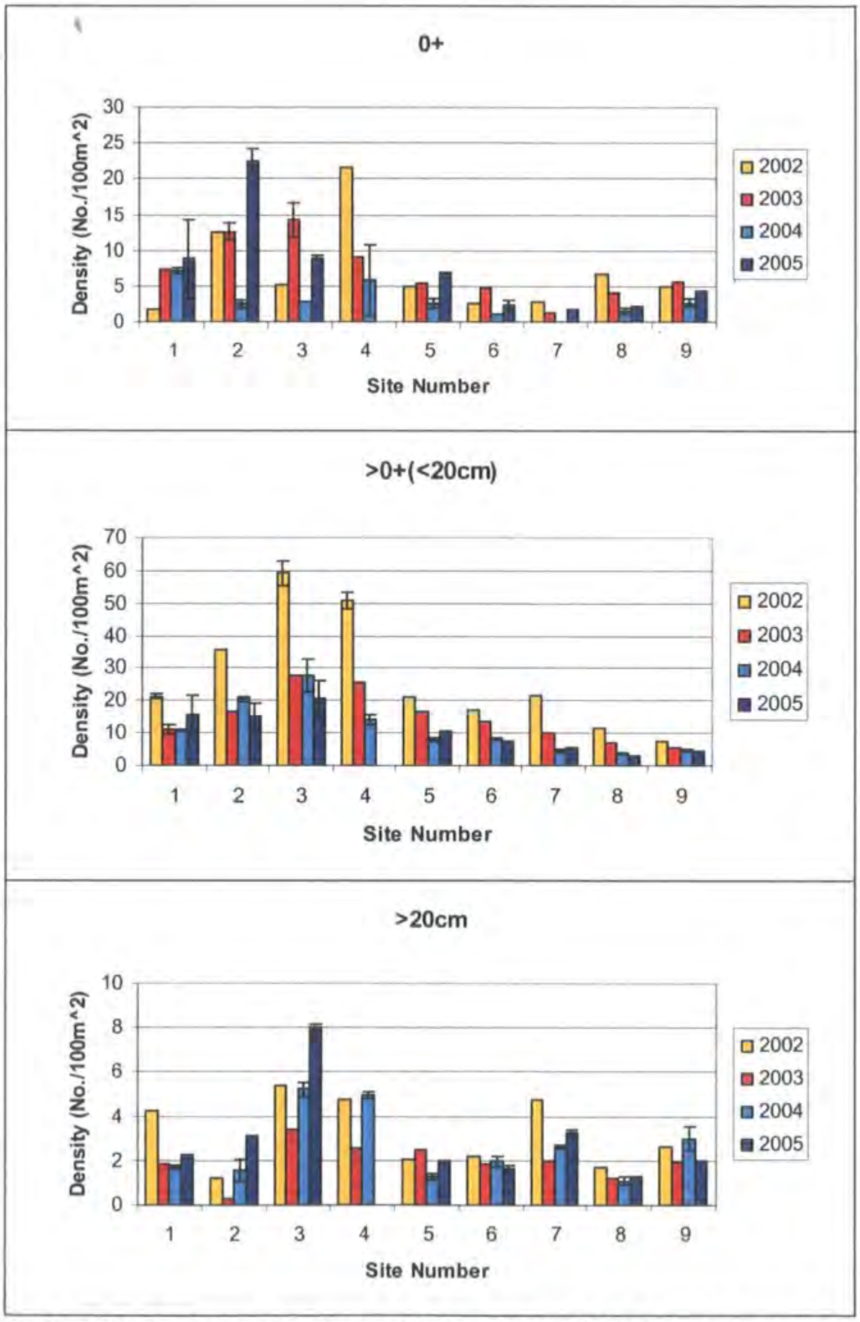


Figure 6.6.5. Figure to show the pre and post change HQS scores for each site on the Rivelin.

Figure 6.6.6 shows that there was no systematic change in the observed brown trout density across each of the sampling seasons in the Rivelin. There was perhaps a slight decrease in 0+ density and >0+(<20cm) density in the post change samples in the lower reaches of the Rivelin. At both of the Rivelin modelling sites, there was an increase in the habitat availability for each life stage simulated. but there was no corresponding increase in density. The >0+(<20cm) brown trout appeared to be habitat limited in the results presented in Chapter 4. It is interesting therefore that this habitat limited population did not show a direct increase in density following the increase in habitat availability afforded by the increase in compensation flows. This indicates that other factors may be controlling the >0+(<20cm) trout populations as well as physical habitat.

Both sites displayed an increase in the habitat available for spawning, but no increase in the density of 0+ trout was seen in 2005, which was the only 0+ cohort spawned under the new flow regime. This highlights the difficulty in interpreting results over such a short temporal period, as two years of the new flow regimes may not be sufficient time for the fish populations to respond to a change in baseflow, and we should trust the model output in that the new flow regime creates better habitat for the older trout. The increased habitat availability for >0+(<20cm) trout may have been evidenced in the statistically faster growth rates seen in the older trout in the 2004-2005 growth period compared with the previous growth periods (Chapter 4).


0+

>0+(<20cm)

>20cm

Figure 6.6.6. Graphs to show the observed densities of the brown trout populations in the Rivelin for each of the fisheries sampling seasons.

6.6.2.2. Loxley

Table 6.6.5 shows that there is a decrease in habitat predictions for each of the brown trout life stages. However, this decrease in predicted habitat availability is very small, and of similar size to that propagated by model calibration uncertainty (see Figure 6.4.2), and as a consequence, indistinguishable. At LU, these decreases are not very large compared to the changes seen in the Rivelin, and for >0+(<20cm) and >20cm trout the weighted habitat available increases (Table 6.6.5). There appears to be an increase in the quality of habitat for the older trout under the post-change regime at this site.

	Total Spawning Habitat	**Weighted Spawning Habitat**	**Total 0+ Habitat**	**Weighted 0+ Habitat**	**Total >0+(<20cm) Habitat**	**Weighted 0+(<20cm) Habitat**	**Total >20cm habitat**	**Weighted >20cm habitat**
LU pre change	36966	92.41	37799	94.49	39783	99.45	39884	99.67
LU post change	36449	92.3	36820	93.24	39306	99.54	39402	99.78
Percent Change	-1.4%		-2.6%		-1.2%		-1.2%	

Table 6.6.5. Table to show the output of the brown trout habitat simulations for the best parameter sets at LU. For the optimal mesh and optimal parameter set.

Table 6.6.6 shows that there appears to be a decrease (albeit a very small one) for each of the life stages of brown trout simulated at LD. Again, the decreases in predicted habitat are indistinguishable form potential changes in habitat predictions due to model calibration uncertainty (Figure 6.4.2). The quality of habitat increases for the >0+(<20cm) trout under the post-change flow regime.

	Total Spawning Habitat	**Weighted Spawning Habitat**	**Total 0+ Habitat**	**Weighted 0+ Habitat**	**Total >0+(<20cm) Habitat**	**Weighted 0+(<20cm) Habitat**	**Total >20cm Habitat**	**Weighted >20cm Habitat**
LD pre change	34200	96.73	35293	99.82	35065	99.18	35256	99.72
LD post change	33613	96.43	34794	99.81	34649	99.4	34668	99.45
Percent Change	-7.1%		-1.4%		-1.2%		-1.7%	

Table 6.6.6. Table to show the output of the brown trout habitat simulations for the best parameter sets at LD. For the optimal mesh and optimal parameter set.

The HABSCORE HQS values shown in Figure 6.6.7 display no consistent changes in brown trout habitat along the Loxley when the pre and post change values are compared. These findings are mirrored by the fuzzy modelling results which show a very small decrease in habitat for each of the life stages simulated, with no major alterations in the proportioning of habitat predictions.

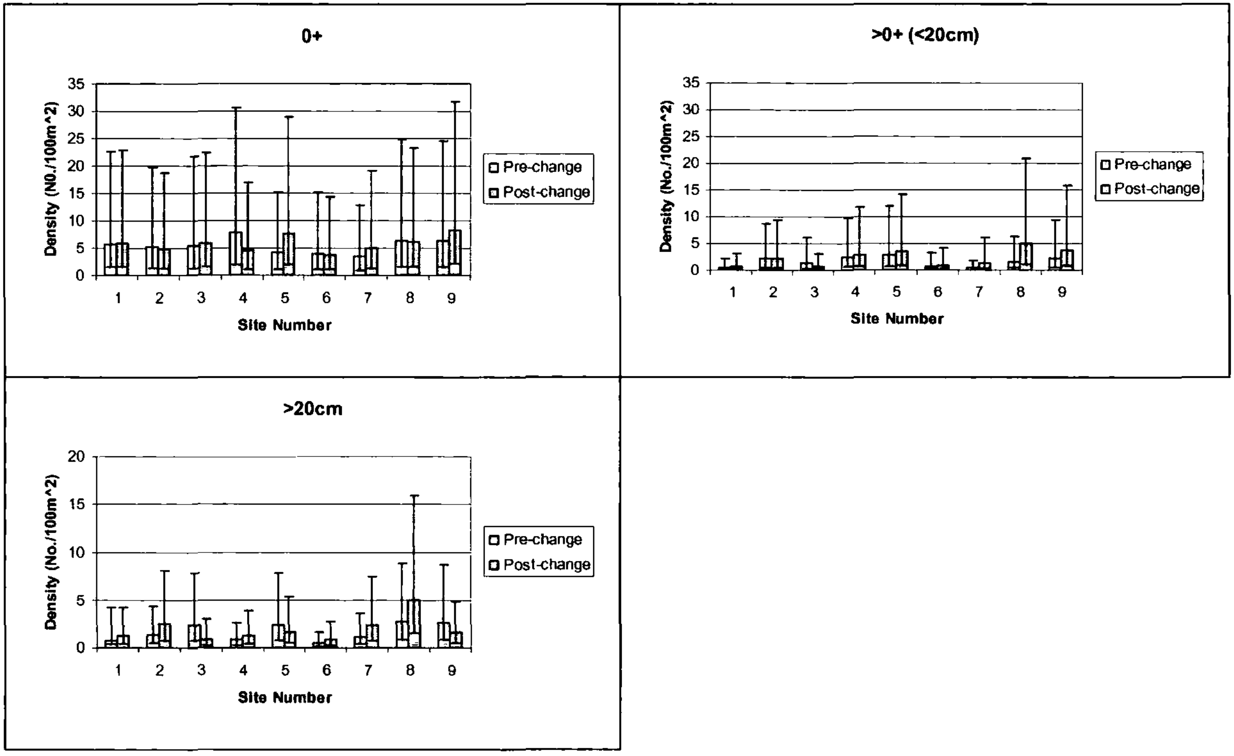


Figure 6.6.7. Figure to show the pre and post change HQS scores for each site on the Loxley.

Figure 6.6.8 shows that there were no systematic changes in the observed densities of trout found in the Loxley. This is also what you would expect with the very smaller predicted changes in brown trout habitat availability via the fuzzy model. No changes were observed in the growth rates of the brown trout captured in the Loxley, unlike in the Rivelin, so perhaps the fuzzy model is picking up on a lack of habitat change for brown trout in the Loxley, or any potential habitat changes in the Loxley are lost in the hydraulic model calibration uncertainty.

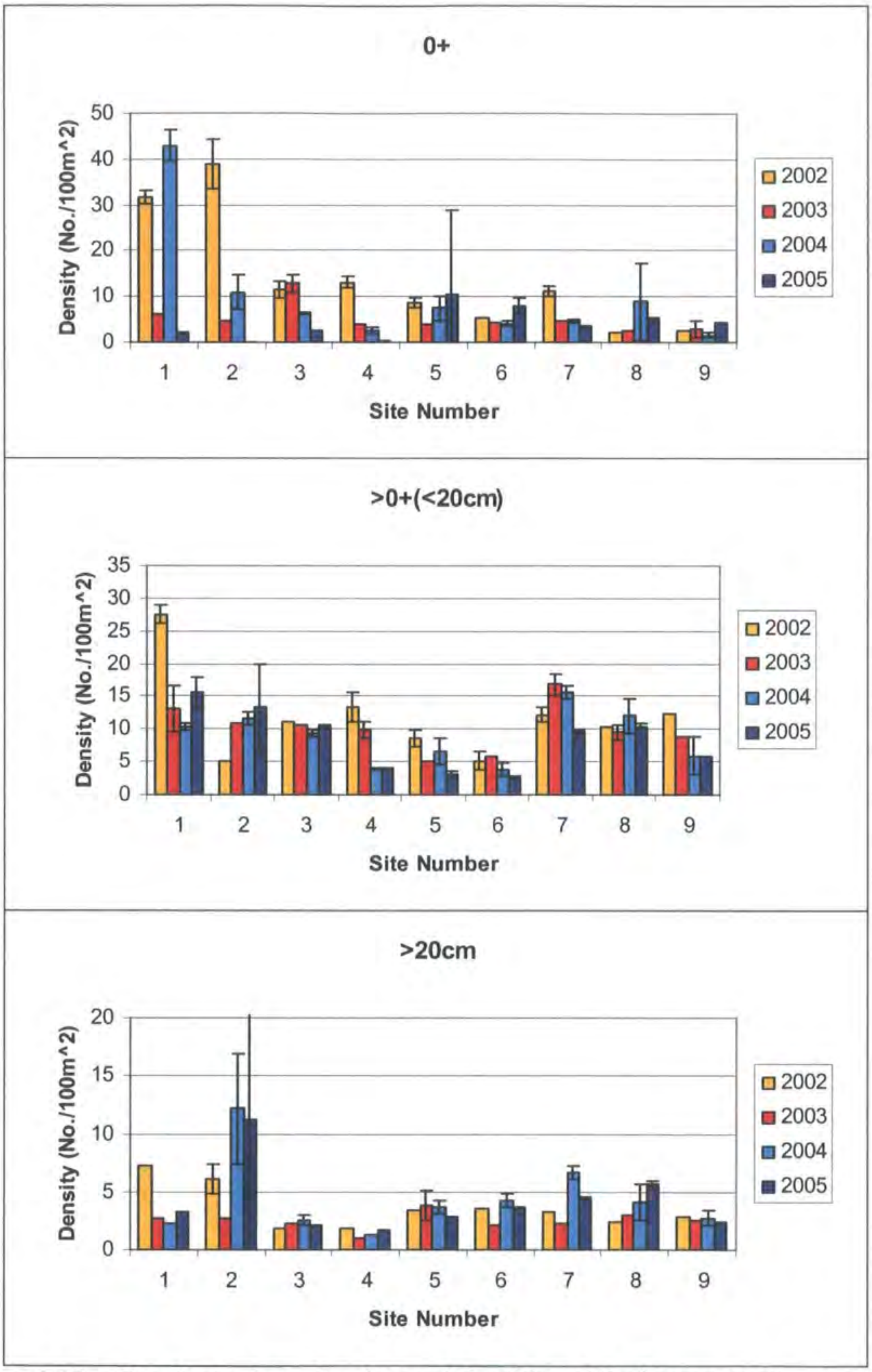


Figure 6.6.8. . Graphs to show the observed densities of the brown trout populations in the Loxley for each of the fisheries sampling seasons.

6.6.2.3. Summary

An increase in predicted habitat for each life stage of brown trout was seen in the Rivelin, whilst there were decreases for each life-stage in the Loxley. However, the increases in habitat created by the change in compensation flows were much larger than the decreases observed in the Loxley. The habitat increases predicted at RU can be thought of as being greater than any potential changes caused by model calibration uncertainty; whilst this is not the case at the downstream site on the Rivelin, or at either site on the Loxley. No evidence for these changes in available habitat could be seen in the observed densities of brown trout in either the Rivelin or the Loxley. The increase in predicted available habitat for older trout in the Rivelin was perhaps evidenced by an increased growth rate for older trout in the observed populations in the Rivelin.

6.6.3. Summary

This section has shown that it was difficult to validate the pre-change habitat availability simulations. This is because the (abstract) validation data used within this section are measures of brown trout density. However, as was explained in the literature review and methodology, the value of HSI is not a biologically meaningful result. So, to compare brown trout density and a value of HSI is difficult, but it was more interesting to compare changes in these density metrics with changes in predicted habitat. It was seen that there was little systematic change to either the HQS for observed density of brown trout when the pre and post change values were compared (as was seen in Chapter 4). There were much larger increases in predicted habitat availability for brown trout in the Rivelin (which only appeared separated from model uncertainty at RU), than there were decreases in the Loxley. The increased habitat availability in the Rivelin may have been evidenced by the higher growth rates in post-change older trout in the Rivelin.

6.7. Discussion and conclusions

6.7.1. Introduction

In many instream habitat modelling studies, the habitat suitability curves or habitat probability curves are populated by habitat preference data. This, in itself presents an issue in terms of whether this is recording preferred organism preference (where they are living) and not a possible organism preference (where they could live) (e.g. Manly *et al.*, 1993). A more practical issue for this study was the lack of site specific habitat use for brown trout in particular. This precluded the use of Category III HSCs and also probabilistic methods much as those used by Guay *et al.* (2000) and Tiffan *et al.* (2002).

A criticism of traditional habitat suitability curve approaches is that they treat velocity and depth independently. However, it is intuitive that both macroinvertebrates and trout will 'choose' their habitat based upon a combination of velocity and depth (and perhaps other factors such as substrate) so as to treat these variables as independent is erroneous. In the UK, most of the instream habitat modelling conducted has used PHABSIM, which used a one-dimensional hydraulic model and category III habitat suitability curves which treat velocity and depth independently (e.g. Maddock *et al.*, 2001; Gibbins and Acornley, 2000). As well as incorporating the uncertainty inherent in ecology nicely, a fuzzy logic approach does not treat velocity and depth independently. This coupled with the need for the model to be transferable to catchments where habitat preference data may not be available made the use of fuzzy logic an attractive option. This discussion first assesses the performance of the fuzzy model, then assesses the implications of the predictions for the Rivelin and Loxley.

6.7.2. Assessing model performance

It was seen in section 6.4.1 that the predictions of both invertebrate guild and brown trout habitat availability are sensitive to uncertainties depth and velocity distributions. It was of importance to establish the sensitivity of each of the fuzzy rules a priori in relation to variations in depth and velocity because then any change in fuzzy habitat predictions can be attributed to changes in effective predicted hydraulics and if there is no (or very small) change in predicted habitat, this can be attributed to a lack of effective change to the hydraulic predictions. This section also allowed us to distinguish predicted compensation flow impacts from uncertainty due to model calibration. This is vital, as it is of interest to know whether changes in the habitat predictions are due to the changing hydraulics of the streams or due to errors within the model calibration.

Section 6.4.2 assessed the sensitivity of each of the fuzzy rules to variation in the depth and velocity precision values used with respect to one model simulation. It was seen that the filterer, predator and collector-gatherer guilds were most influenced by variations in the precision values. All life stages of brown trout were influenced similarly apart from the spawning life stage. Following this, one set of precision values were used for the remaining habitat predictions. The precision values used were considered carefully, as it was not wanted to make the precision values so high as to overly fuzzify the model, but too low precision values would have lead to an almost univariate approach. The precision values were matched to the precision values from the hydraulic model calibration, so as to try and incorporate this uncertainty to some extent.

The use of guilds for modelling the macroinvertebrates is unique and because of which was more open to failure. The rationale for using guilds instead of target species was outlined in Section 6.3. So, one of the points that this discussion must focus on is the performance of the guilds on two levels:

1) are the fuzzy rules imposed on the guilds sensitive to changes in depth and velocity?; and
2) can fuzzy habitat predictions be abstractly validated to the observed guild proportions within the study rivers.

On the Loxley, the highest habitat predictions in the pre-change scenarios were for the shredder and scraper guilds. Analysis of the proportions the invertebrate guilds found at the nearby sampling sites showed that the most important feeding guild was the shredder guild, with the scraper guild being largely insignificant. This suggests a lack of correlation between the observed and predicted guilds in the Loxley. This could be because of three reasons:

1) the fuzzy rules imposed for each of the guilds are incorrect and so are unable to produce meaningful ecological results;
2) the fuzzy rules are too fuzzy (or the precision values used are too great), in which case the extra hydraulic modelling detail provided by the 2-D model will be negated by over-fuzziness of the guild rules;
3) there is no major change in the guild habitat availability in either of the Loxley sites; and
4) the invertebrates in the system are not hydraulically limited.

It is perhaps more likely that the third option is correct, as it was seen in Chapter 3 that there was little or no change to the invertebrate community in the Loxley when the post-change invertebrates were compared to the pre-change invertebrates. Figures 6.5.4 and 6.5.5 in this chapter illustrated that there were little or no perceptible changes in the observed guild proportions at either of the Loxley sites. The scraper guild has very similar habitat availability compared to each of the other guilds in the pre-change simulations, but the very low abundance of this guild at either of the sampling sites may be further indication of a detachment of the invertebrate communities and environmental variables within the Loxley. This possibility was discussed at length in Chapter 3.

This viewpoint is further enhanced by the fact that there appeared to be some relationship between the predictions of guild habitat availability in the downstream site of the Rivelin and the observed data. At RD, the importance of scrapers, predators and shredders in the model output matched the proportions in the pre-change observed invertebrate data. What it does further indicate is that the guild fuzzy rules were able to create habitat predictions which are sensitive to changes in depth and velocity, and which perhaps correspond to the real-world.

There were large increases in habitat availability in the Rivelin for each of the feeding guilds (apart from predator) in the post-change simulations, which were seen to be higher than possible changes due to model uncertainty. But there were no corresponding changes to the measured proportions of feeding guilds at the nearby invertebrate site. The change in predicted habitat was smaller on the Loxley and could not be distinguished from potential changes from model calibration uncertainties. It could be that the alteration to the compensation flows in the Loxley are so proportionally small compared to those in the Rivelin that it creates no change to the habitat available for macroinvertebrates. The lack of corresponding change to the larger changes in the Rivelin could be due to the small timescale of this study. The two-year pre and post sampling strategy has already been highlighted as a limitation of this work (Chapters 3 and 4). It may take longer than two years for the predicted increases in habitat on the Rivelin to be converted into changes in the functional feeding guilds, that is, if the guilds are habitat limited.

The fuzzy habitat modelling of the different age/size classes of brown trout did appear sensitive to alterations in depths and velocities (as evidenced by Section 6.4). In both the Rivelin and Loxley, it was difficult to see any patterns in the habitat prediction replicated in the pre-change observed densities of brown trout, as for the pre-change predictions in the Rivelin, the highest habitat availability was for the 0+ age/size class, but the highest trout densities were observed in the >0+(<20cm) age/size class, and a similar pattern emerging on the Loxley. When comparing the pre and post change habitat predictions, the Rivelin increased in habitat availability and the Loxley decreased, but no systematic changes in trout density were seen. There could be for a number of reasons for this detachment of fuzzy habitat prediction and observed trout densities:

1) the fuzzy rules are incorrect;
2) the trout populations of the Rivelin and Loxley are not habitat limited; and
3) two years is not enough time for a change in baseflow to be transferred into a change in trout density.

The predicted changes in habitat between the pre and post change simulations were much larger at the Rivelin upstream site compared with the remaining site. The change in compensation flow led to an increase in predicted habitat on the Rivelin and a decrease in predicted habitat on the Loxley. The increases in habitat availability were not followed by a change in trout density in the Rivelin. It was argued in Chapter 4 that the >0+(<20cm) trout population were habitat limited. It may therefore be expected that if that age/size class were habitat limited that it should be the one to respond to the increase in available habitat. This was not observed in terms of trout densities in the Rivelin. However, it was seen that older trout in the Rivelin do appear to grow faster under the post-change regime compared to the pre-change regime. Although growth rate is usually dependent on food availability (Milner *et al.*, 2003), Gibson (1993) reported that generally growth rate decreases with increasing density. So maybe the extra habitat in the Rivelin is manifest in a slight decrease in density and hence an increase in growth rate. A post-change sampling period of two years may be insufficient to detect changes in trout density, but the increase in available habitat may have been manifest in the growth rates. The percentage predicted habitat change on the Loxley were so small (and indistinguishable from model uncertainty) that it may not influence the trout populations at all, or if it does, it may take longer than two years for it to do so.

One potential limitation of this study is the focus on flow velocity and depth as the two major determinants of habitat suitability. It was outlined in Chapter 4 that substrate is of great importance to brown trout populations especially during spawning (e.g. Frost and Brown, 1967; Raleigh *et al.*, 1986) and as older trout (e.g. Heggenes, 1988; Greenberg *et al.*, 1996). But it is unlikely that the composition of the substrate will change with the small changes in compensation flows explored here. However, in the desire to keep the simplicity of the modelling approach it was chosen to ignore substrate as it is usually a function of the bulk flow properties (i.e. depth and velocity).

A further limitation is the use of steady state discharges. It was chosen to use steady state discharges as they are the aspect of the compensation flow which are controlled by the regulator directly. However, the importance of flow variability with respect to both macroinvertebrates and trout was highlighted in the literature reviews (e.g. Clausen and Biggs, 1997; Lobon-Cervia and Rincon, 2004), and in the results presented in Chapters 3 and 4. It would have been interesting to run a hydrograph through the hydraulic model and assess the habitat availability in that way. This would have been especially interesting in the spawning period (October), and the output compared to the following years 0+ trout densities.

The use of a HSI as the crisp number in this study, precluded a direct comparison of the Rivelin and Loxley or study sites (due to differences in site mesh area) and so enters a much larger debate about the use of metrics such a total available habitat and / or weighted usable area in habitat modelling studies. Mohardt and Mesick (1988) point out that the weighted usable area, which results from the use of habitat suitability curves is an index and cannot be measured directly and as such it is difficult to validate these models. This was seen in this work. This is a problem that fuzzy logic models do not overcome either. This is unfortunate, and because of this the model validation in this chapter consisted of abstract comparisons with nearby ecological sampling sites.

A different approach is the use of bioenergetic or object-orientated modelling such as that highlighted in the literature review (e.g. Van Winkle *et al.*, 1998) are a method of avoiding the HSI issue, but such a modelling approach was out of the scope of this study. Such an endeavour would also require hydrological time series two-dimensional or greater modelling, which requires a much greater computational effort. This is perhaps the next step in water resource modelling as it has a number of advantages, the first of which is that it is able to be directly validated given correct ecological sampling. Second, the incorporation of a time series hydrograph into the 2D model allows the incorporation of temporal flow variability into the modelling process (an ecological factor highlighted in both Chapters 3 and 4). This was something which was lacking in this chapter, as due to computing constraints, only steady state simulations could be performed.

Also highlighted in Chapters 3 and 4 was the importance of the interaction of reach scale morphology with discharge and spatial variability in these upland Millstone Grit catchments. This spatial variability in captured by virtue of combining the 2D hydrodynamic model and the fuzzy habitat model. This is demonstrated in Figures 6.7.1 to 6.7.4 which shows the habitat suitability map for the shredder invertebrate guild for each modelling reach for both the pre and post-change discharges. Figures 6.7.1 and 6.7.2 show in particular the patchy nature of habitat for the shredder guild, due to the influence of reach scale morphology and boulders within the Rivelin. This is also evident to a slightly lesser extent in the Loxley plots.

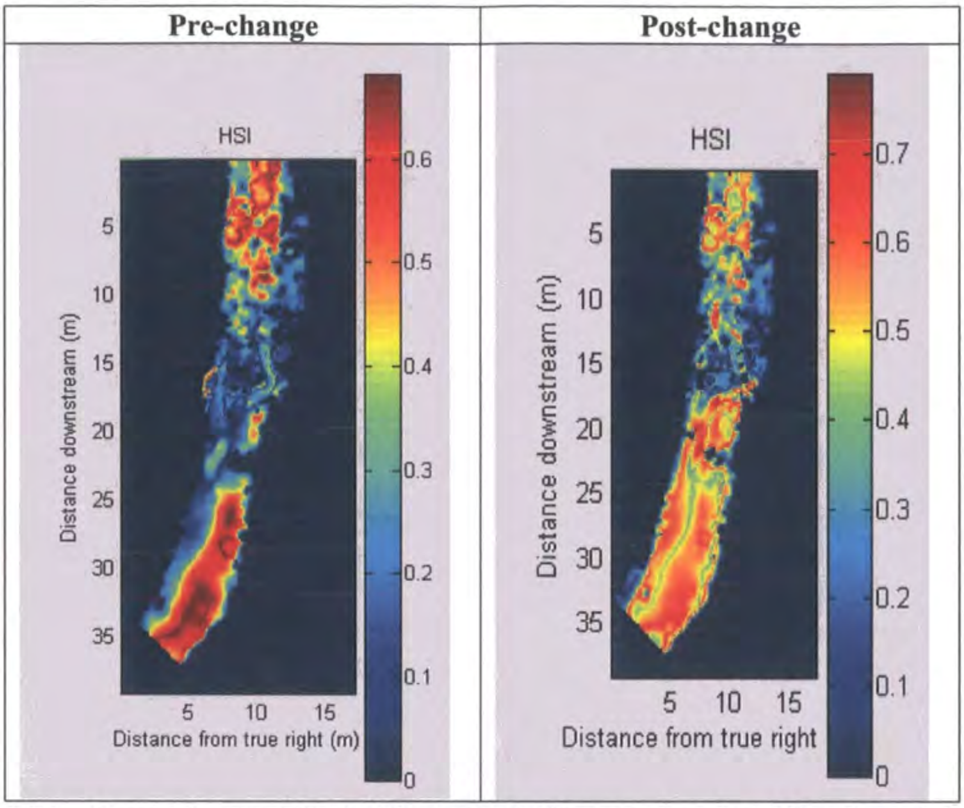


Figure 6.7.1. Figure to show the habitat suitability map for shredding invertebrates at RU under both the pre-change compensation flows.

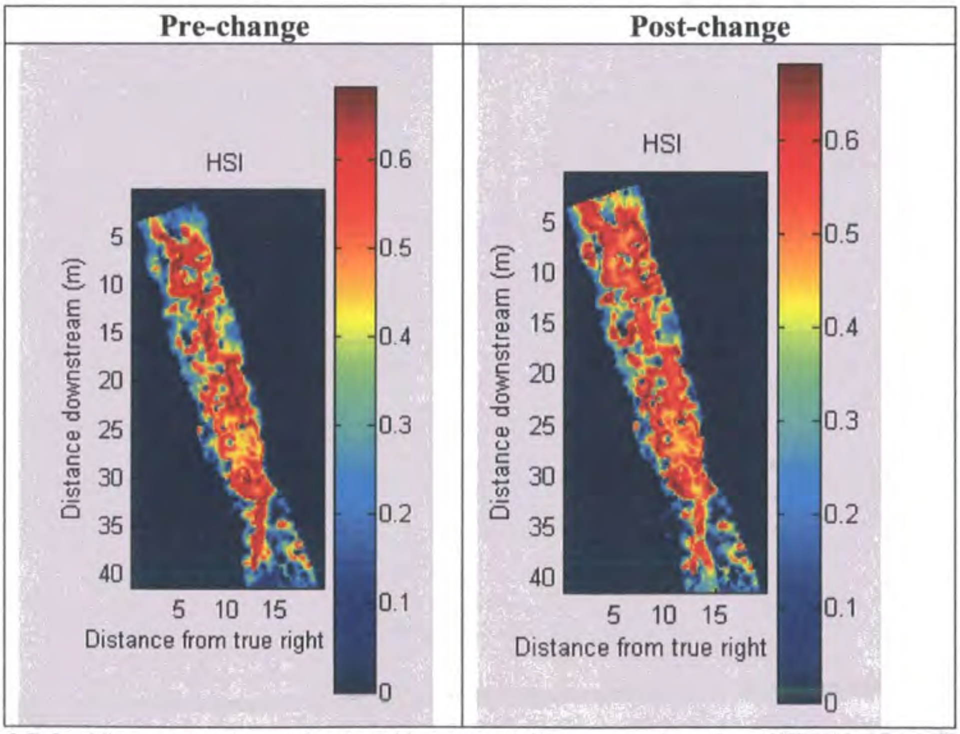


Figure 6.7.2. Figure to show the habitat suitability map for shredding invertebrates at RD under both the pre-change compensation flows.

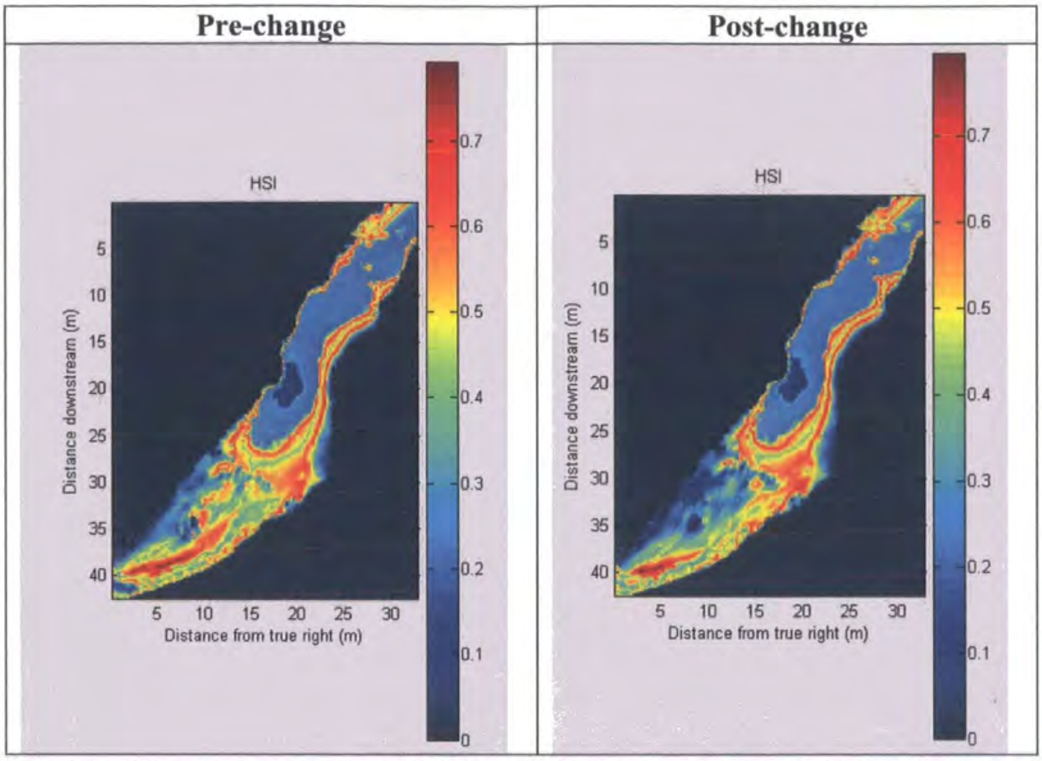


Figure 6.7.3. Figure to show the habitat suitability map for shredding invertebrates at LU under both the pre-change compensation flows.

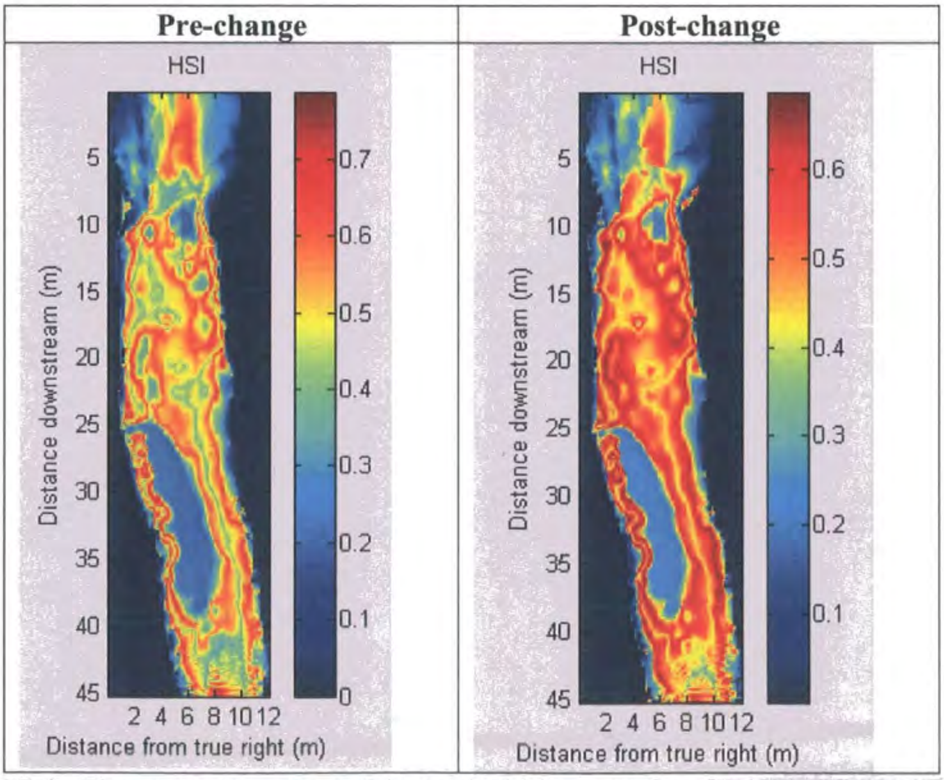


Figure 6.7.4. Figure to show the habitat suitability map for shredding invertebrates at LD under both the pre-change compensation flows.

6.7.3. The modelled impact of compensation flows on the Rivelin and Loxley

Fuzzy logic has not been used in a water resources in the UK to date. The first section of this discussion has hopefully argued the benefits and effectiveness of this approach, whilst observing some of the limitations of the approach taken.

On the Rivelin, there were predicted increases in each of the invertebrate guilds in the post-change simulations compared with the pre-change simulations. The largest increases were seen for the filterer and collector-gatherer guilds. At both sites on the Rivelin, these increases were larger than changes which could have been due to model calibration uncertainty. The Loxley suffered a small decrease in each of the invertebrate feeding guilds which was indistinguishable from potential changes from errors in model calibration.

As with the invertebrates, the predicted habitat available for each life stage of brown trout increased in the post-change simulations in the Rivelin, and there was a smaller decrease in predicted trout habitat on the Loxley. The changes habitat availability were only distinguishable from potential changes due to model calibration uncertainty at RU.

6.7.4. Final conclusions

The work presented above has shown the potential of 2D modelling coupled with a fuzzy logic habitat model as a tool for aiding decision makers in compensation flow studies (for brown trout in particular). The design of the modelling approach was such that the discharges simulated were directly controllable by the regulatory body. It has been seen throughout this thesis that all aspects of the system are influenced by hydrological variability and perhaps other factors. By using this modelling approach, the results are not contaminated by flow variability, and as a consequence the approach can be used to assess the specific changes advocated. However, the uncertainties propagated by the calibration of the hydraulic model must be considered at all times when interpreting the results. Most importantly this section has used information presented in Chapters 3, 4 to link with the modelling results presented in chapter 5 (Figure 1.2).

Chapter 7

Synthesis and conclusions

7.1. Introduction

The aim of this thesis was to use both field and modelling techniques to assess the ecological implications of altering compensation flow regimes in upland Millstone Grit streams. This has been done by using three broad strands of enquiry: macroinvertebrate sampling; fisheries sampling; and modelling (Figure 1.2). Throughout this thesis, I have referred to a modified Petts (1984) diagram, which summarises the potential impacts of steady state compensation flows (Figure 7.1). It has been shown in this thesis that changes in the compensation flows have impacted upon the wetted perimeter, water depth and flow velocities present within each of the rivers. It has also been shown that both the existing compensation flow regimes and the alteration of the compensation flow regimes had an impact on the ecology of the systems.

However, the work has shown that variability in space and time and over various scales plays a key role in these upland Millstone Grit catchments, and complicates the interpretation of Figure 7.1. Variability within the study catchments comes in two forms: flow variability in time and geomorphological variability in space. The flow variability interacts with the geomorphological variability to produce the physical habitat available for invertebrates and fish. Both of these types of variability act at different scales, with flow variability seen to be influential in both the longer term (e.g. droughts) and shorter term (e.g. spates provided by reservoir overtopping and tributary inputs). The spatial variability inherent within the systems can also be seen to act over a number of scales: (i) tributary-scale; (ii) reach-scale; (iii) boulder scale; and (iv) micro-scale. The response of both macroinvertebrates and fish to changes in compensation flow are likely to be much more complex than implied in Figure 7.1.

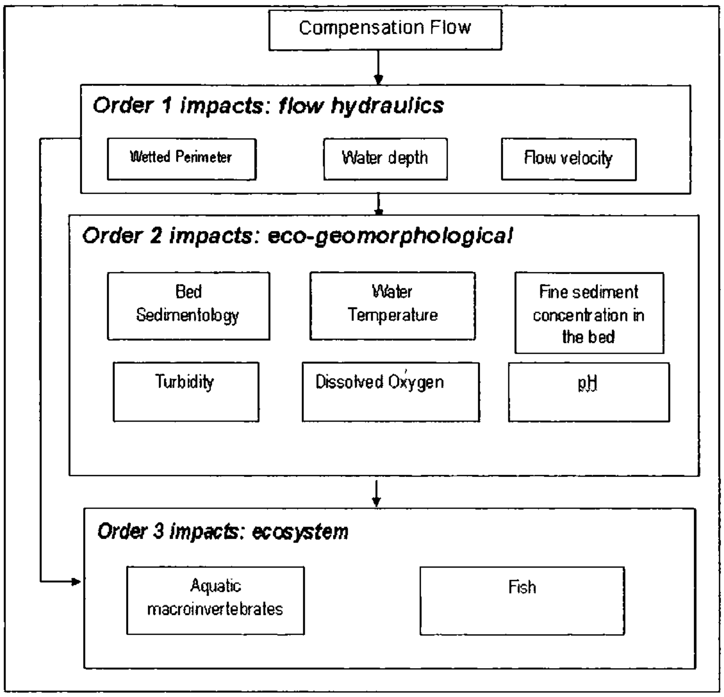


Figure 7.1. Flow diagram to show the potential impacts of altering the compensation flow in the rivers. Adapted from Petts (1984).

A key question which also needs to be addressed in this section is whether the ecological methodologies were correct and/or adequate for the purpose. What is also of interest is to assess the extent to which the model dealt with this variability and the extent to which it links with the ecological work. This chapter has three main sections. The first section presents the substantive findings from the thesis and links them to the existing literature. This section itself has three sub-sections, with long term flow variability addressed first, short-term variability addressed second and finally the results with respect to spatial variability are presented. The second main section is a critical review of the methods used in this work; and the final section analyses the opportunity for future work.

7.2. Substantive findings

7.2.1. Long term flow variability

The potential impacts of long term flow variability such as drought on the Rivelin and Loxley was shown for invertebrates by using a combination of the Environment Agency kick sample data (covering the 1996 drought period) and a dry summer/autumn in 2003 which was included in the Surber sampling. Although droughts may not impact on these regulated rivers to the same extent as in unregulated rivers as a result of the compensation releases, they will have an impact in terms of reducing overtopping and tributary inputs into the river. It is also these periods when the compensation flows are the only flow present in the river, and so the impacts of these periods on invertebrate and fisheries communities is of particular interest. By encompassing two of those periods in the invertebrate sampling, it was possible to characterise drought impacts on the Rivelin and Loxley.

Chapter 3 showed that when the Rivelin and Loxley drought samples were compared using the ANOSIM routine, there is a statistically significant difference between the 1996 samples and samples taken in any other year. However, no difference was found between the two periods in the samples taken upstream of the reservoirs. When the important family lists for the drought and non-drought samples were compared it showed that the differences were driven by changes to slow flow loving low abundance families within the community and not the core species. The low-rainfall period of summer/autumn 2003 showed that the important species within the Loxley were those that preferred slower flows during that period. Little other change was observed in the 2003 low-rainfall period. This indicates that there is good refugia from drought impacts for the most important species within the rivers, as observed in the literature by Cowx *et al.* (1984) and Wood and Petts (1999).

Unfortunately, the fisheries record is only four years long (2002-2005). In this time only one dry period was covered (summer/autumn 2003). The fisheries populations displayed an interesting response to the low flows present at the end of 2003. This was manifest in the populations sampled in 2004, with lower densities of 0+ trout found in 2004 than for any of the sampling seasons in the Rivelin, but not in the Loxley. On the Rivelin, there were some statistically significant differences between 2004 and other sampling seasons in terms of 0+ trout density, but there were none on the Loxley. It was also seen that the growth rate of trout in the 2003-2004 period was slower than that for the other sampling periods on both the Rivelin and Loxley for trout growing to one year old. This is an indication that growth rates may be the best way in this study to ascertain potential impacts of the compensation flows. It has been shown that growth of salmon in freshwater were poor for year classes affected by the summer droughts of 1976, 1983 and 1984 (Elliott, 1985; 1993b) but a detailed assessment of the affects of drought on growth rates has yet to be made. The results from this study perhaps indicate that even in regulated systems, the influence of low-rainfall periods are manifest in the form of slower growth rates of brown trout when compensation flows are too low.

The potential usefulness of calculating brown trout growth rates is also illustrated when the growth rates for the trout in the Rivelin appeared (and was statistically in some cases especially for the older trout) to be faster in the 2004-2005 period than for any of the preceding growth periods. The impact of small alterations in discharge such as the ones trialled in this study could be manifest first in changes to trout growth rate rather than density. This must be linked in some way to an increase in food availability or energy expended in obtaining food, as Milner *et al.* (2003) observe that growth rate represents the individuals success in acquiring energy through food.

The modelling approach taken was to use a 2D depth averaged hydraulic model to model the steady state discharges for the pre and post change compensation flows for both the Rivelin and Loxley. Steady state simulations were conducted because Yorkshire Water are only able to change the compensation releases from the Rivelin and Loxley reservoirs on a steady state basis. It can therefore be thought that these simulations form a baseline habitat for the Rivelin and Loxley, and it is these baseline conditions which will persist in dry spells. It was seen through the invertebrates and fish sections that the results are contaminated by the variability inherent within the systems. The advantage of the modelling approach taken in this work is that the results are not contaminated by the flow variability and so can be used to assess the specific changes advocated. It is also impossible to predict the levels of augmentation which would occur post-change.

Chapter 5 showed that increasing the compensation flows in the Rivelin led to an increase in both depths and velocities, with greater increases in velocities. A much smaller reduction in both depth and velocities was observed in the Loxley. Chapter 6 used a fuzzy logic model to link the hydrodynamic predictions to ecology. The fuzzy logic modelling predicted increases in each of the invertebrate feeding guilds in the post-change simulations compared to the pre-change simulations. The largest increases were observed in the filterer and collector-gatherer guilds. The model also predicted an increase in brown trout habitat availability in the Rivelin upstream site for each life-stage modelled. Any changes in habitat availability predicted in the Loxley, could not be distinguished from errors quantified in the model calibration.

7.2.2. Shorter term flow variability

Short term flow variations provided by the rainfall events and reservoir overtopping were tracked by using rainfall data and discharge data provided by the Environment Agency; and reservoir overtopping data provided by Yorkshire Water. The hydrological analysis presented in Chapter 2 highlighted the importance of flow augmentation to the hydrology of both the Rivelin and Loxley. It was seen that the Rivelin is subject to more overtopping events than the Loxley due to the smaller size of its reservoir, and its discharge is subject to more variation than that of the Loxley. When the compensation flows were changed, the Rivelin was subject to proportionally fewer overtopping events. When the cumulative frequency curves comparing the pre and post change discharge were compared, the flow duration curve of the Rivelin flattened out to resemble that of the Loxley, thus indicating a decrease in flow augmentation. This is of interest because the importance of flow augmentation (or lack of) was evidenced in Section 7.2.1, through the changes in invertebrate populations in the drought periods.

It would obviously have been impossible to sample the invertebrates so frequently as to take account of individual spate events, but samples were taken in spring (May), summer (August) and autumn (November) in order to incorporate a temporal element into the sampling. A more frequent sampling procedure may have been desirable (Table 7.3) but would have been impossible to implement, in order to include the spatial element. However, it was possible in Chapter 2, to characterise each of the macroinvertebrate samples in terms of the recent hydrological properties of the system (28 day rainfall depth; discharge on day of sample; time since last overtopping event). This allows a qualitative assessment of the effects of these parameters on the invertebrate population. A further examination of the environmental conditions at the time of invertebrate sampling was conducted in the form of a suite of environmental variables sampled at the same location as the Surber samples, the usefulness of such will be examined in section 7.2.3.

The greater flow variability on the Rivelin compared with the Rivelin was seen to have a number of impacts on the invertebrate community under the pre-change conditions (Table 7.1). The higher density of invertebrates in the Loxley (which has less flow variability when compared to Rivelin; and with flow variability acting as a proxy for disturbance) agrees with the findings of Englund, (1991); Palmer *et al.* (2005); Doeg *et al.* (1989); Jowett and Duncan, (1990); and Lake *et al.* (1989). The greater number of families in the more flow variable (more disturbed) Rivelin agrees with the findings of McCabe and Gotelli (2000).

Rivelin contained more families	Loxley contained fewer families
Rivelin fewer individuals per sample	Loxley more individuals per sample
Rivelin more diverse	Loxley less diverse
Rivelin invertebrates more related to measured environmental variables	Loxley invertebrates less related to measured environmental variables
Rivelin contains more fast flow loving invertebrates	Loxley contains fewer fast flow loving invertebrates
Important species are more variable in the Rivelin	Important species are stable in the Loxley

Table 7.1. Table to summarise the pre-change invertebrate results.

The amount of between season variation of the environmental variables measured with each Surber sample decreased in the Rivelin following the change in compensation flows but increased in the Loxley. Two reasons were hypothesised for this on the Rivelin: (i) the post-change increased discharge on the Rivelin homogenises the environmental variables to a greater extent than the pre-change variables; and (ii) the increased discharge released from the compensation flow reservoir led to a decrease in overtopping (as evidenced by Table 2.5), and hence this would lead to a decrease in flow variability. On the Loxley, the decreased discharge could have led to the variability provided by both overtopping and rainfall events having a greater impact on the measured environmental variables.

Results for the post-change samples (Table 3.4.54) showed that the correlation between the macroinvertebrate population and the measured environmental variables decreased at the upstream Rivelin site and increased at the upstream Loxley site. As discussed above, there was a decrease in augmentation in the Rivelin after the change in compensation flows. This, coupled with increased buffering from the higher compensation flows, may have increased the detachment of the measured environmental variables and the macroinvertebrate community in the Rivelin. There was no increase in overtopping in the Loxley (Table 2.5), but the lower post-change compensation flows in the Loxley may mean that any overtopping has a greater impact than any pre-change overtopping, perhaps leading to the better correlation between measured environmental variables and the macroinvertebrate communities. Through the above discussion it is apparent that both the long and short term scales of flow variability influence the macroinvertebrate population of both rivers.

The fisheries surveys were conducted on a yearly basis and so were not of a frequent enough nature to account for spate flows. However, it is probably not the individual spate flows which determine the available habitat for fish. This is because fish have longer life cycles than that of most macroinvertebrates. The timing of higher autumn flows is important, in terms of prompting brown trout to spawn (e.g. Ladle, 2002). The drought of summer/autumn 2003 would have meant that such elevated/spate flows would not have occurred in the Rivelin and Loxley. It was seen that 2004 had the lowest density of 0+ trout in the Rivelin of any of the sampling seasons, thus perhaps indicating that the low flows in autumn 2003 led to reduced spawning or egg/ alevin survival than in previous years. A decrease in 0+ trout density was not observed in the Loxley in 2004, which could have been buffered by the larger compensation flow release.

The modelling in this thesis was conducted using steady state simulations for the pre and post change compensation flows only (see Section 7.2.1). Judging from the discussion above with regard to the two scales of hydrological variability to run the model retrospectively for entire hydrological years would have been especially interesting as it is the immediately preceding habitat for invertebrates and the whole year which are of interest for trout. It was seen in Chapter 2 that both of the rivers spend very little time at baseflow (especially in the post-change scenarios) and so it became difficult to relate the fuzzy habitat predictions to observed invertebrate and brown trout populations. Using the fuzzy model it would have been possible to produce a time series of habitat availability for the age/size class of interest throughout the year, had the time been available to run the simulations.

It would have been especially interesting to run the model for the hydrological time series through the spawning period preceding each of the fisheries surveys in order to assess the spawning habitat for a given spawning period. This would allow us to compare the available spawning habitat, with the 0+ trout habitat populations the following year. This would allow us to establish whether the stream was limited by a lack of spawning habitat or whether there are other reasons for the low 0+ HUI values. In October/November, the flows are not going to be near to the baseflow due to tributary inputs (hence making the steady state predictions of spawning habitat inappropriate). The fuzzy model predicted large increases in spawning habitat availability at the upstream Rivelin modelling site under the post-change compensation regime. However, a large increase in 0+ trout density was not observed in 2005.

7.2.3. The conditioning influence of spatial variability

This thesis has shown that the impacts of compensation flow on instream ecology will depend on the location of the sampling site within the river. There was a large amount of within river variation in both the Rivelin and Loxley, with the upstream and downstream Surber sampling sites being statistically significantly different in both the pre and post change invertebrate communities. In the pre-change invertebrate communities, RU was the site subject to the greatest amount of effective flow variation and had the most variable invertebrate community, evidenced in Table 7.1. The LU site, characterised by probably the least variable flow, had the highest density of invertebrates (Table 7.1). This again highlights the importance of flow variability in driving the within river differences. The fact the upstream sites are the sites where the impact of impoundment was felt concurs with the serial discontinuity concept. However, with the sampling design herein, it is not possible to conclude how far downstream the impact of the impoundment was felt.

River-scale variation in brown trout habitat predictions were observed in the Rivelin, where Habitat Quality Scores (HQS) for each of the trout age/size classes appeared to be higher in the middle reaches of the Rivelin (sites 4, 5, 6). No consistent pattern was observed in the Loxley HQS scores. Observed densities of trout were generally higher in the upper reaches of the Rivelin compared to the lower reaches, in both the pre and post-change samples. There was also little consistent within river variation of trout populations in the Loxley.

There were few statistically significant differences between the central and marginal samples in either the pre or post change samples in terms of the invertebrate community. Interestingly, there were also few statistically significant differences between the central and marginal sampled in terms of the measured environmental variables. This reflects the influence of reach scale morphology on these upland Millstone Grit streams creating complex flow patterns. Therefore, this spatial heterogeneity provided by reach-scale morphology must be considered when designing invertebrate sampling strategies for the investigation of compensation flows in Millstone Grit streams.

The impact of reach-scale morphology was noted clearly in the fisheries populations. The HABSCORE Habitat Quality Score (HQS) showed that there appeared to be no difference between the Rivelin and Loxley in terms of the predicted densities of brown trout of any age/size class. This is despite the large difference in compensation flows between the two rivers. Table 4.21 showed that when the observed densities of brown trout were statistically compared, there was only one statistically significant difference (the Rivelin having a greater density of >0+(<20cm) trout (in 2002)). This indicates that despite the greater discharge, the Loxley is unable to support a greater density of brown trout. This could be because of external factors impacting upon the fisheries populations, but if that were the case, high HQS values and lower density values would be expected. The reason why the Rivelin can support a similar density of trout as the Loxley despite the much lower discharge could be due to the interaction of reach-scale morphology and discharge. The reach scale morphology of the river can change the depth (e.g. Kennedy and Strange, 1982); and substrate (e.g. Greenberg *et al.*, 1995). However, it was seen that the growth rates for older (two and three years old) Loxley trout were generally faster than for the Rivelin. So again, it may be growth rates of the trout may be the aspect of fisheries ecology where the impact of the compensation flows are most evident.

Further evidence for the importance of reach-scale morphology was presented in the predictions of brown trout habitat availability presented in Chapter 6. There was a large predicted increase in brown trout habitat availability under post change conditions at the upstream Rivelin site but not at the downstream Rivelin site, despite the same change in discharge. As the two main input conditions into the two-dimensional model were discharge and topography; this variation in habitat predictions must be due to the differing reach scale topography of the two sites. Again, this highlights the importance of reach-scale morphology in determining brown trout habitat availability.

The discussion above shows the that we have knowledge of the importance of tributary and reach scale variation with respect to compensation flows, but the studies herein have not quantified the influence/importance of boulder and micro-scale morphology. This is one of the limitations of this study in that, firstly, the influence of boulder and micro-scale variability has not been quantified and secondly, how this variability affects ecology with respect to compensation flows has also not been quantified. These points will be addressed in greater detail in Section 7.3.

7.2.4. A new conceptual framework?

Throughout this thesis, Figure 1.1 has been referred to as a framework for visualising the impact of compensation flows. However, the twin spectres of flow variability and spatial variability have complicated this framework. Therefore, Figure 7.2 is presented as an attempt to synthesise this discussion graphically. The catchment itself is firstly conditioned by climatic and land use drivers. The left hand side of the diagram displays the hierarchical nature of spatial variability within the system, feeding into the reach-scale study sites used herein. The compensation flows themselves are conditioned, firstly by overtopping events, and then further downstream, tributary inputs and abstraction. A feedback exists between the compensation flows and the amount of overtopping (as, depending on reservoir size, an increase in compensation flows could decrease the amount of overtopping and vice versa);

and there is a co-variation between tributary inputs and overtopping events. Any changes to ecology caused because of the alteration in compensation flows will also be superimposed upon the movement of brown trout and invertebrates within the systems. The movement of adult brown trout is illustrated by the green lines in the diagram, showing the fact that they are probably able to move both upstream and downstream. The red lines illustrate 0+ trout and invertebrate movements; in which they can easily move downstream, but move upstream with more difficulty.

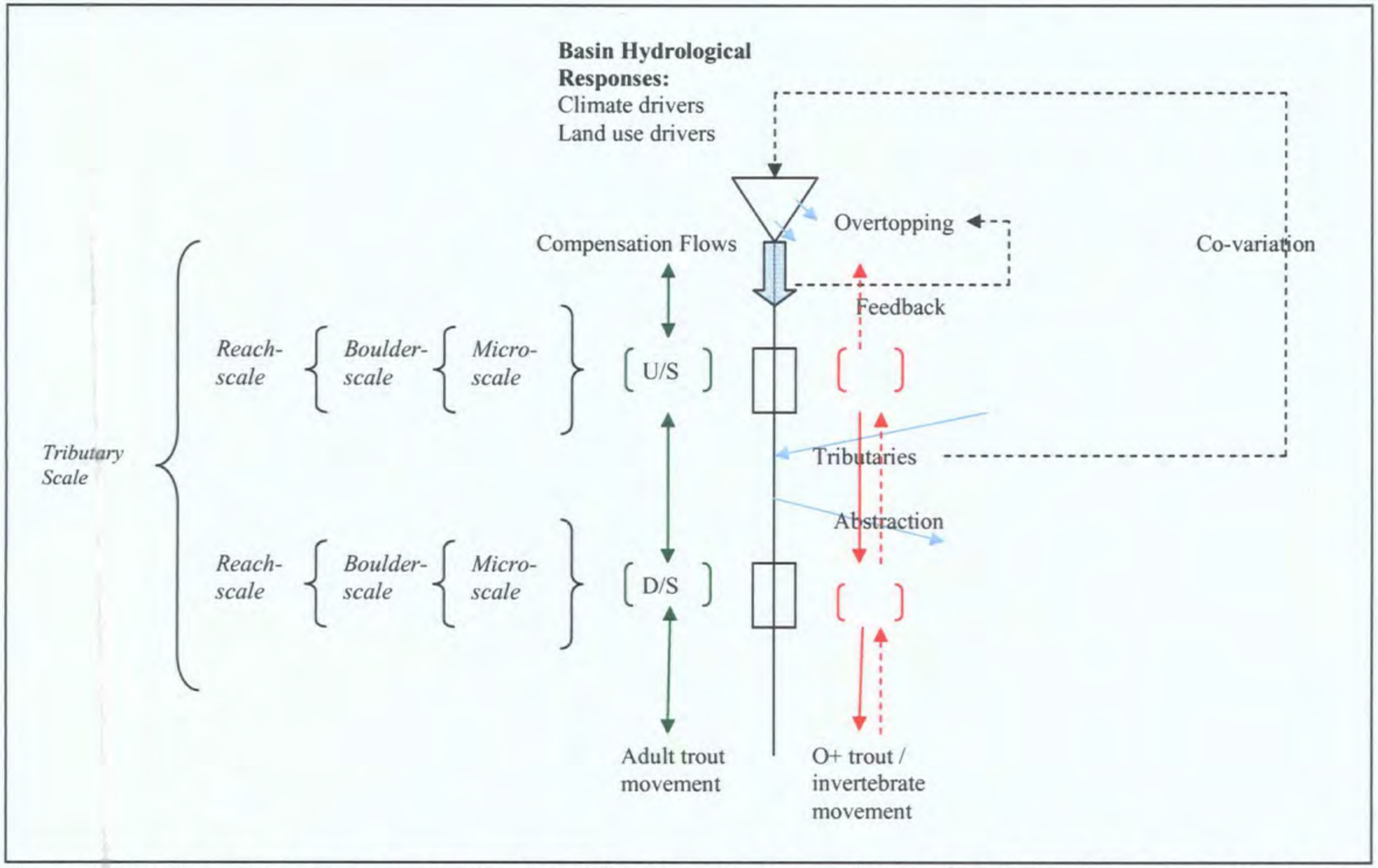


Figure 7.2. Flow chart to conceptualise the different scales types of variation derived in a single upland Millstone Grit stream. Green lines show adult brown trout movement and blue lines show 0+ trout and invertebrate movement.

7.3. Critical review of methods

The short length of data record in this study (only four years invertebrate and fisheries data) has been highlighted in Chapters 3, 4 and 6 as a potential limitation to this study and in Table 7.2. This is because it leaves two years pre-change data with which to characterise the system before the change in compensation flow and only two years to assess any potential changes to the ecology of the systems. This post-change assessment of the fisheries populations was made more difficult by the fact that the 0+ fish captured in the first post-change survey (2004) were spawned under the pre-change flow regime, so only one cohort of 0+ brown trout were captured which had spent the entirety of their life cycle under the post-change conditions. However, due to the project timescale and funds, this was the best data available.

Within-site variability was accounted for in the Environment Agency kick samples by having a distribution of sampling sites along each of the study rivers. Included in this are sampling sites situated upstream of the reservoir. However, the nature of the rivers are so different upstream of the reservoirs that it was impossible to establish how far down the influence of the impoundment was felt in order to investigate the Serial Discontinuity Concept (Ward and Stanford, 1983a, 1983b). Only two Surber sites were used for each of the Rivelin and Loxley, and one on the Hipper, because of the more intense sampling effort involved in taking ten Surber samples. Within river-variability was accounted for in the Rivelin and Loxley by situating one of the samples just downstream of the compensation reservoir and the second further downstream. The ten replicates were needed so that statistical tests could be conducted.

Within-river variability was accounted for to some extent in the fisheries sampling procedure by the placement of nine sites along each of the rivers (Figure 4.3.1). Nine sites provided a coverage of 450m in rivers of several kilometres long. As such, the spatial coverage of the study may be considered poor. Greater spatial coverage could have been incorporated into the fisheries sampling procedure by using single-pass fisheries surveys but this would have compromised the accuracy of the density estimates (Table 7.3).

Within river variation was accounted for in the modelling by using two modelling sites on each river. The reach-scale approach was a necessary concession when using the two-dimensional hydraulic model because computational constraints meant that running simulations for longer stretches of river was impossible. If a one-dimensional model were used (as by: Johnson *et al.*, 1993; Maddock *et al.*, 2001; Gibbins and Acornley, 2000; Gibbins *et al.*, 2001; Strevens *et al.*, 1999), each of the rivers could have been simulated in their entirety. This would have provided the perfect measure of modelled within river variability. However, a 1-D approach would still have needed a very detailed data collection phase and placed only slightly reduced computational demands.

In order to account for reach scale morphology at the invertebrate sampling sites, the Surber replicates were taken at 10m intervals. This meant that the length of each of the sampling sites was 40m, and the first cross section was located in such a place so that this sampling span included both a pool and a riffle. These ecological units are very important to invertebrates (e.g. Angradi, 1996; Brown and Brussock, 1991; Grubaugh *et al.*, 1996; McCulloch, 1986; Thorp, 1992; Wohl, 1992). Further reach scale morphology was accounted for by the sampling of invertebrates in both central and marginal areas of the channel as one of the key impacts of reach scale morphology will be the distribution of flow within the given reach.

METHOD	RATIONALE	BENEFITS	DISADVANTAGES	IMPLICATIONS
Experimental Design				
Paired catchment design	Simple way to alter the flows. Nice experimental designs possible.	Statistical comparisons possible.	The study was only really limited to these two rivers, thus ignoring the larger spatial scale.	The potential impact of altering compensation flows should be studied on other rivers.
Hipper as a control	Had to have a control.	Allowed some sort of comparison of regulated and unregulated MG catchments.	Geographically too far away. Not enough sampling sites; especially for fisheries.	Either use a control catchment properly or not at all.
Two years pre and post change	The maximum timescale available over the study.	Allowed a characterisation of the impacts and an analysis of impacts of changing the flows.	Two years probably not enough to observe any changes over the natural variability.	The data set needs extending to be sure of any implications of altering compensation flows.
Invertebrates				
Kick sampling	It is the EA's standard method for sampling invertebrates in MG rivers.	Was useful for longer scale historical context.	The lack of within season replication made statistical analyses difficult. Perhaps underused in the post-change context. Data format unhelpful.	It is the EA's standard method for sampling invertebrates in MG rivers – and so their viability compensation flow studies must be established.
Surber sampling	Used so as to sample discrete areas of riverbed.	Good for trying to link env and inv – but proving causality was very difficult. Good for investigating the differences between C and M samples.	The invertebrates sampled depended greatly on positioning of net in the field. Small sampling area. Relatively large temporal gap.	Could perhaps be used to investigate the ecological meaning of flow biotopes or functional habitats?
Univariate	Needed to investigate the usefulness of such measures in comp. flow studies as they are widely used in the literature.	Showed that univariate methods are limited.	Displayed few patterns. Shannon couldn't discriminate between seasons when multivariate analyses could do.	Cannot be used in isolation.
LIFE index	It is logical to assess the flow preferences of invertebrates in these streams as the flow velocity of water may change with a change in compensation flows.	Gave a good indication of the flow velocity preferences of invertebrates.	Underused.	Lots of LIFE data to be calculated from all the Surber samples.
Multivariate	Used because it uses community structure as a basis for statistical analyses rather than a metric.	Allowed lots of pertinent data analysis methods to be used.		

Table 7.2. Table to show a critical review of methods for the experimental design and the invertebrates.

Fisheries	RATIONALE	BENEFITS	DISADVANTGES	IMPLICATIONS
50m stop-netted reach surveys	To gather information on the population density and growth rates of BT (and other species).	Allowed an incorporation of detailed population information and spatial scale. Good for comparing with HABSCORE.	Time consuming technique – only 21 sites possible. 4 years not enough data.	Consider single-pass fisheries surveys as this would allow a greater spatial area to be covered in future studies.
HABSCORE	To investigate whether compensation flow impacts can be deciphered using HABSCORE.	Provided a useful measure of available physical habitat and use.	Expensive to collect.	
Dahl-lea back calculation	Used to extend growth rate data-sets.	Dahl-lea works well with low population densities.		Alternative back-calculation methods could be used in different studies.
Growth rate analysis	To give a measure of how easily BT were finding food.	This was perhaps the one indicator in fisheries where changes were observed. Quicker response time than BT densities?	Some variation in the growth rates within cohorts.	Growth rate data should be collected in future compensation flow studies.
Hydrodynamic modelling				
EDM surveying	Used to create a DEM of the four study reaches.	Accurate to around 1cm. Could be used despite the heavily wooded nature of the sites.	Very time consuming.	Other techniques such as GPS or photogrammetry may be used on other catchments.
FESWMS	FESWMS was chosen because it was finite element and was cheap to obtain. Had good wetting and drying representation.	Was able to incorporate the inherent spatial variability. Errors generally mesh generated.	Very time consuming. Didn't incorporate temporal variability due to computational limitations.	Perhaps a 1-D model would have sufficed – but that would still have taken a long time to set up.
Waters edge validation	Interested in assessing the model accuracy in predicting wetted area – something which could change with a change in compensation flows.	Allowed assessment of the models accuracy in predicting wetted perimeter.	Perhaps the field survey was not detailed enough (only about 15cm).	Such validation is vital in compensation flow studies using two- and three dimensional models.
Point depth and velocity validation	Used to see how effective the model was a predicting point velocity and depths.	Good for estimating both accuracy and precision.	Only 60 per modelling reach.	Needs to be collected and used.

Table 7.3. A critical review of the methods used for the fisheries and hydrodynamic modelling chapters.

Fuzzy modelling				
Invertebrate Guild fuzzy rules	Was used in an effort to avoid indicator species for invertebrates.	Would have allowed the model output to include a measure of community structure.	Didn't appear to work. Either the concept is fundamentally wrong or the guilds are wrong.	Do not use in the current form.
Brown trout life stage fuzzy rules	Used to correspond with the HABSCORE and observed density data.	Seemed to work.	Only used for the 4 life-stages.	Can be used in the current form; as the fuzzy rules are transferable.
Fuzzy logic	Used to incorporate the natural uncertainty in these systems; and because of the lack of habitat preference data.	Quite a good approach given the data constraints.	Experimental.	Needs development. Could include measures of habitat choice – e.g. older BT use of boulders.
'Abstract' validation	Needed to try and validate the model in some way.	Gave an idea of the performance of the model.	Was difficult to compare habitat availability predictions with nearby habitat use.	Perhaps in the future it would be best to model habitat use rather than availability (e.g. Bioenergetic modelling).

Table 7.4. A critical review of the methods used for the fuzzy modelling chapter.

The use of the replicates allowed the sampling of different flow types (e.g. riffle, pool etc) and so this allowed the assessment of the importance of flow type as a measured environmental variable. Analysis showed that flow type was the most important measured environmental variable. However, the sampling structure was designed to compare central and marginal samples statistically rather than flow types (Table 7.2), and so variations in community structure between the different flow types could not be examined statistically.

The fisheries surveys were conducted on 50m reaches with stop nets placed at either end of the reach. This allowed the direct assessment of the number of fish within a given reach. The reach morphology was not assessed directly, but HABSCORE surveys were conducted in 2003 and 2005. This allowed a direct assessment of the observed density of trout and reach-scale habitat availability at each of the fisheries sites. As was touched on before, the use of triple-catch data fisheries surveys limited the spatial coverage of the study.

By virtue of its two-dimensional nature, the modelling process includes the reach scale morphology of the modelling sites (e.g. Leclerc *et al.*, 1995; 1996; Ghanem *et al.* 1996). By surveying the major morphological features such as the top of banks, bottom of banks and bars, the basic reach morphology of the study reaches is captured. This information is further enhanced by the surveying of lateral cross-sections at one metre intervals along the reach, thus capturing both the longitudinal and lateral morphology of the study site.

The extent of surveying necessary in order to get the reach-scale topography is such that it limits the transferability of the model (Table 7.3). Each reach needed around 10 days work in order to gain the necessary information. This, when combined with the expert knowledge, computational power and time needed to run the two-dimensional model, limits the ease of use of the fuzzy logic and two-dimensional approach. With the Water Framework Directive, there may be more demand for a physically based model such as this to be used in contentious water resource situations, however the large resource demands of this approach must be noted. However, even if a one-dimensional model were chosen, and the entire river length simulated, similar issues surrounding the time taken for data collection, expert model users would exist. However, with a one-dimensional model, the demands on computational power are smaller.

The 2D model provides depths and depth averaged velocities at each node on the simulation mesh. The impact of reach scale morphology was seen in each of the study sites in terms of flow velocity (Figure 5.7.1). The results of the fuzzy model display how important this reach scale morphology is in determining the habitat availability. For example, Figures 7.3 and 7.4 show the habitat availability for spawning and adult brown trout at RU, under both the pre and post-change conditions. It can be seen that in the middle of the reach there is a transition from a riffle to a pool, and the definition of this transition and what it means for the respective habitats is prominent. It has been observed in the literature that riffles (or the tails of pools) are good areas for brown trout spawning (Burner, 1951; Stuart, 1953; Hoopes, 1972). It is encouraging that this observation is replicated in the spatial pattern of habitat predicted by the fuzzy model.

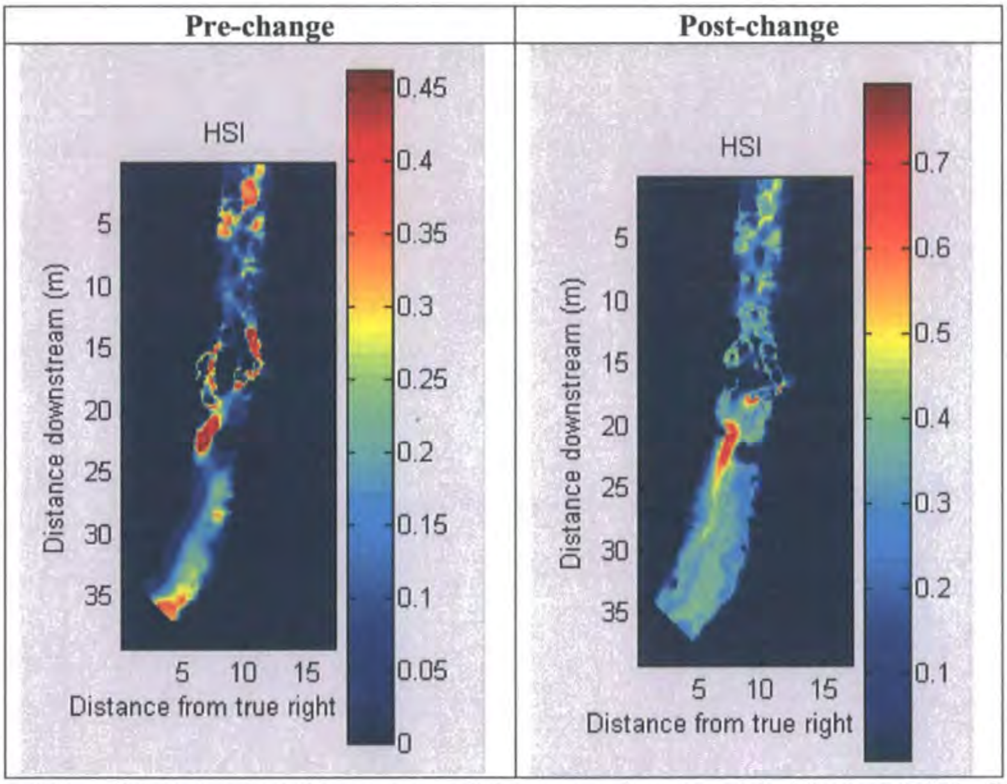


Figure 7.3. Habitat suitability maps of the spawning trout habitat at RU1, for the pre- and post-change compensation flows. Scales vary.

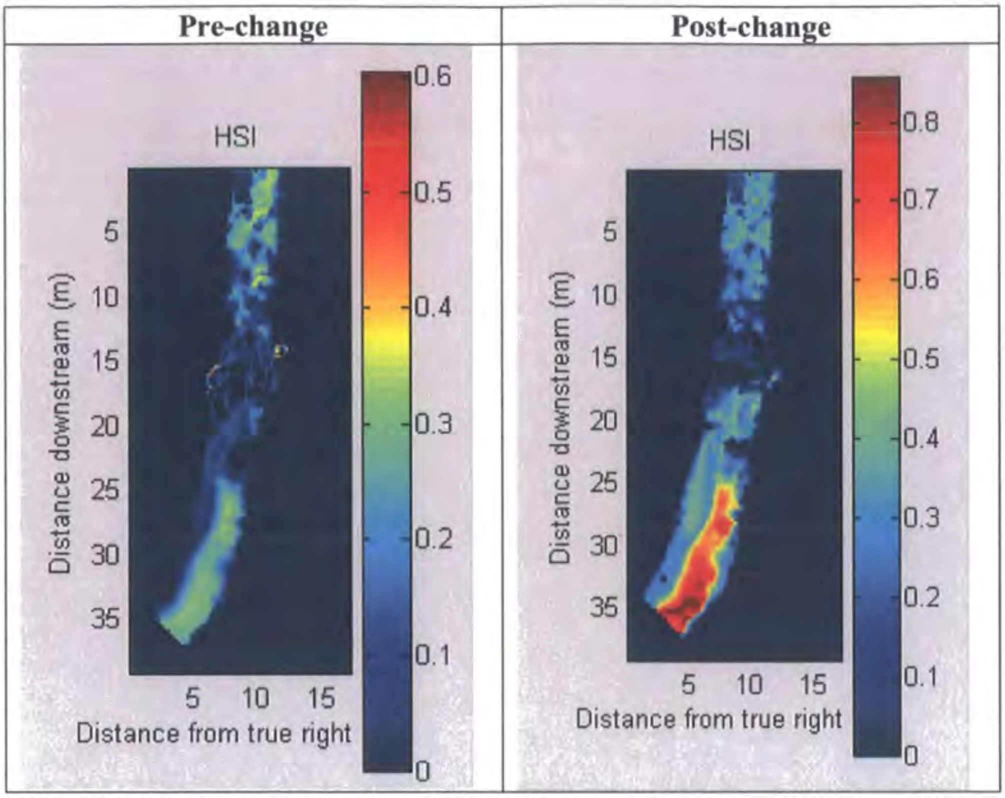


Figure 7.4. Habitat suitability maps of the adult trout habitat at RU1, for the pre- and post-change compensation flows. Scales vary.

Boulders provide flow variability of the scale of around 1-2m^2 (Figure 7.2), and the surface area of the Surber sampler itself is 0.09m^2, and so it samples within the sphere of influence of the boulder. This is one of the advantages of using Surber samplers in studies such as this, as due to the localised nature of the area sampled, the environmental variables for that area can be measured and may be thought of as being relatively consistent. In this case the impact or otherwise of a boulder will probably be represented by the measured environmental variables.

The fisheries sampling programme was designed in such a way that it did not sample boulder habitat per se, but boulder influence will exist within the reaches. However, the number of boulders within a reach is assessed as part of habitat within the HABSCORE program. Boulders are very important for adult trout feeding, as adult trout can tend to rest in the area of slow flow behind the boulder and wait for drifting invertebrates to drift past in the faster flow past the sides of the boulder (Greenberg *et al.*, 1996). This habitat provides extra value over purely the habitat quality of the depth and velocity combined, as it 'costs' the trout very little energy to be near an extremely good food source.

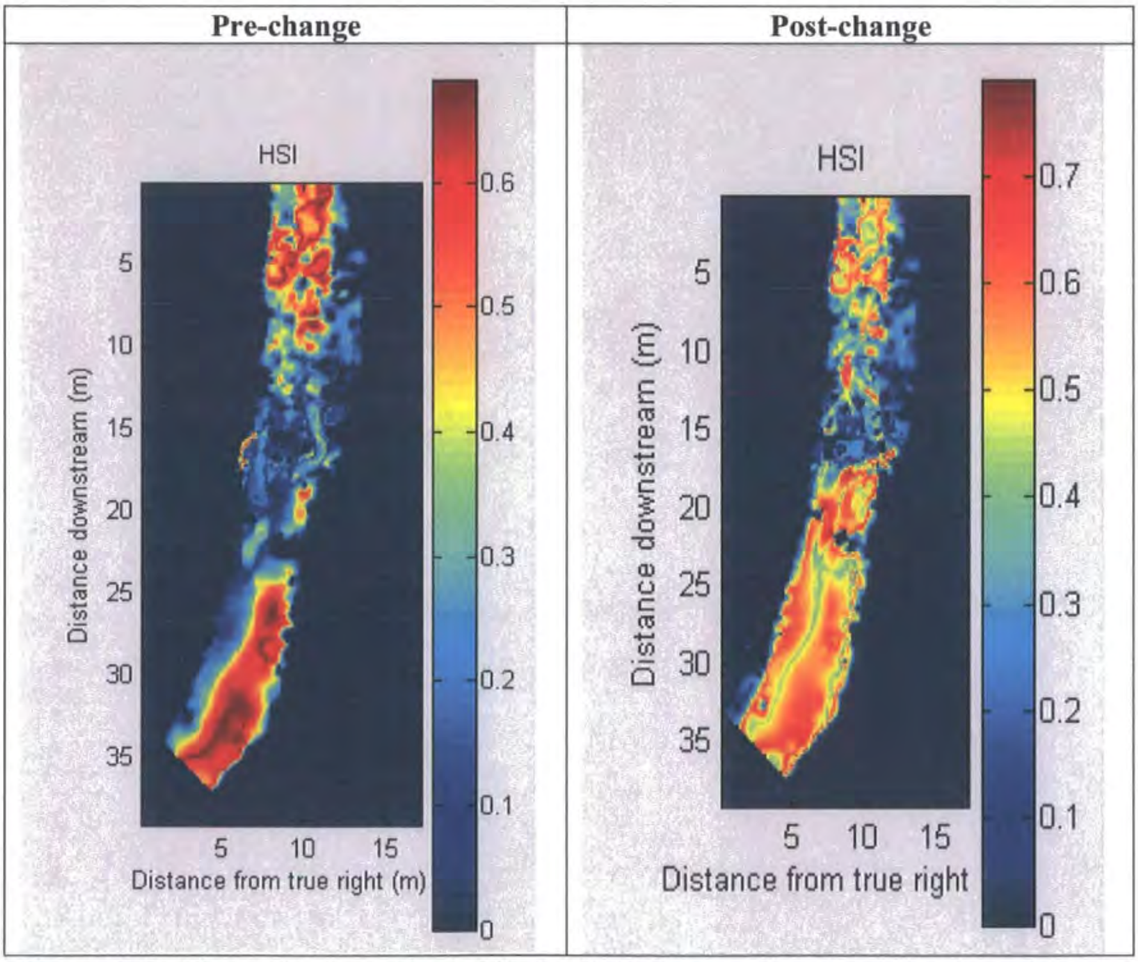
Boulders were included in the model, by surveying every boulder in each modelled reach during the surveying process. During the kriging process which was conducted in order to standardise the mesh spacing, the boulders were put in the process as breaklines, which ensured their presence in the mesh. This should have allowed the spatial complexity of the mesh to be maintained.

Chapter 5 discussed the effectiveness of the surveying conducted in accounting for sufficient spatial variation. It was observed that the precision values when validating the point velocities were much worse than for the point depths, and this could be due to a lack of spatial variation included in the modelling meshes. However, Figure 7.3 above shows the importance of boulders are captured to some extent within the RU reach (and all of the reaches as seen in Figures 5.3.5 to 5.3.7), with the boulders in the centre of the reach cause flow separation and alterations of the habitat available for two age/size classes of brown trout. The importance of boulders within modelled reaches with respect to modelling fish habitat has been highlighted by Crowder and Diplas (2000a, 2000b, 2002).

As stated above, the areas behind boulders provide good habitat for older brown trout, not only in terms of the slower velocity, but because of the proximity of drifting invertebrates, meaning the energy expended / feeding ratio is very good (Milner *et al.*, 2003). Unfortunately, the fuzzy model has not been modified to take account of this added value of habitat. It should be possible in future versions of the fuzzy model to add value to habitat which is slow flowing near some very fast flowing habitat, as that type of habitat is very good for trout feeding/resting.

Boulder-scale morphology is also especially important, in terms of influencing the spatial habitat availability for the invertebrate feeding guilds. As an example, the HSI map for the shredder guild at the upstream sampling site on the Rivelin is presented (Figure 7.5). It shows that the boulders create patches of good and inferior habitat, especially in the riffle areas in the upper part of the reach. When there is less boulder interference with the flow in the pool at the bottom end of the site, the shredder habitat becomes a lot less patchy.

Overall, the use of guilds in both the invertebrate work and the modelling appeared limited (Table 7.4). In terms of the invertebrate analysis the guild analysis was used so that a measure of functional community structure was included in the analysis. This provided some interesting insights into the proportions of feeding guilds present at the various sites, however, little systematic change could be observed in the guilds. The use of guilds in the fuzzy modelling was less successful. There was little or no correlation between the proportions of guilds observed and the predicted habitat of the guilds. A number of reasons were hypothesised for this: i) the fuzzy rules were incorrect; and ii) the concept of using invertebrate guilds instead of key invertebrate species is fundamentally flawed. Therefore, of the fuzzy logic model is to be used again to predict invertebrate habitat, new fuzzy rules must be devised; or a safer, key species approach must be taken.



Pre-change

Post-change

Figure 7.5. Figure to show the habitat suitability map for shredding invertebrates at RU under both the pre-change compensation flows.

The Surber sampler had a surface area of $0.09m^2$ and despite that relatively small size there is still a lot of variation of substrate and velocity within that area. These micro-scale topographic variations are very important to macroinvertebrates, and in turn create small scale variations in velocity. Lancaster (1999) noted that the small scale variations in velocity are very important to the distribution of macroinvertebrates. Averaging velocity over both depth and an area of $0.09m^2$ in the field could lead to errors. First, it is probably not the depth averaged velocities which are of importance to macroinvertebrates (e.g. Lancaster, 1999), as it is the near bed velocities which impact directly on the invertebrates which live on the surface or indeed, beneath the surface. However, it is very difficult to measure the near bed velocity. When combined with the need for the field methodology to be repeatable and efficient it was decided to use depth averaged velocity. Using depth averaged velocity is also compatible with the model predictions. Errors may also be propagated by spatially averaging velocity across the Surber sampling area, as due to the varied substrate, large velocity gradients of velocities may exist.

One of the consequences of ignoring the micro-scale impacts on invertebrates may have been the poor correlation between the measured environmental variables and the invertebrate communities when using the BIOENV routine within PRIMER. However, this poor correlation could also be due to a genuine detachment of environmental variables and invertebrates. This confirms one of the major issues with this work in that it is difficult to determine to what extent the environmental variables (and depth and velocity in particular) impact upon ecology, and therefore linking any ecological changes which occurred post-compensation flow change to that change in compensation flows. This difficulty in assigning causation between flow and biota has been observed previously (e.g. Kinsolving and Bain, 1993). Another reason for the poor correlation between the macroinvertebrate communities observed and the measured environmental variables could be the family approach taken to the multivariate data analysis. If the species level data were used, perhaps a closer association could be found, but the reasons for using a family approach are clear. Furthermore, biotic interactions were ignored throughout this thesis.

The choice of a 2-D model by the very definition means that the velocities are depth averaged. As discussed above, perhaps the most important aspect of flow velocity for macroinvertebrates is the near-bed velocities. A 2-D model provides no direct information on these near bed velocities. However, perhaps in the future, if a 3D scheme could be used, the information provided on near-bed velocities would be valuable ecology information for macroinvertebrates. This will also apply to bottom-dwelling fish species such as Bullhead (*Cottus gobio*). However, three-dimensional models are much more computationally and time intensive than two and one dimensional models; and so the benefit gained from using the three-dimensional model must be weighed against the increased costs.

Much micro-scale topographic information was lost via 2 mechanisms in the modelling process. The first one was that the lateral cross-sections were surveyed to beaks of slope at an accuracy of around 15cm, and so any features smaller than that will be ignored. Furthermore, the 1m spacing of the lateral cross sections ensures that further micro-scale information will be lost in between the cross-sections. However, there is only so much surveying that this cost-effective, and to capture the topographic complexity of the study sites fully would have taken too long. Micro-scale features are also lost during the kriging process, as even if micro-topographic features were included in the survey, if they are of a scale of <25cm or <15cm depending on the mesh, this information will be lost. The effectiveness or otherwise of the surveying conducted for this study was discussed in Chapter 6. The effects of micro-topographical variation was also evident in the validation of the point velocities; with Lane *et al.* (1995; 1999) and Nicholas and Smith (1999) highlighting the effect of micro-topographic variation on the depth averaged validation readings. This could be one of the reasons for the poor accuracy of the 2D model in predicting the point velocity calibration data.

The micro topography in terms of substrate is not accounted for in the fuzzy rules, as the two major determinants of habitat suitability used were depth and velocity. This is a potential limitation of the study as substrate has been found to be of importance to brown trout especially during spawning (e.g. Frost and Brown, 1967; Raleigh *et al.*, 1986) and for adult trout (Heggenes, 1988; and Greenberg *et al.*, 1996). If substrate had been used in this study it may have added a 'proxy' representation of micro topography of relevance to brown trout, and as such may have been a useful habitat determinant.

As with many other problems and issues associated with studies such as this one, it is a question of scale and variability, and the relative importance and costs of studying them. In terms of modelling, variability was accounted for in terms of the river-scale, reach-scale and boulder scale, but micro-scale flow variability processes were ignored. For the macroinvertebrates it was easier to contextualise and highlight the importance of the flow variability, and all scales of spatial scale were accounted for (to some extent) apart from micro-scale topography (which may be of importance). It was seen that the fisheries populations were more impacted upon by the longer-term flow variability such as low flow periods, but less affected by spate flows, with the river and reach scale were accounted for.

7.4. Recommendations for future work

The spatial discontinuity concept (SDC) (Ward and Stanford, 1983a; 1983b) proposes that dams act as pervasive agents of disturbance acting discontinuities within the river continuum. The only empirical evaluation of the SDC in the UK is contained in Armitage and Blackburn (1990) on the River Tees. They found that the distance before the master variable (Chrironomids) re-set was only 0.5km. This distance is greater than the distance to the upstream sampling site, but less than the distance to the downstream sampling site. Therefore, if the monitoring were to be continuing, it may be worth considering only extending the data set at the two upstream sites, so as to cut the costs of the fieldwork by around half, but at the same time continue to assess the impacts of the compensation flow changes, the whole influence of which are unlikely to have been covered by two years data.

One modelling site was situated near the upstream invertebrate sampling site and the other near the downstream invertebrate sampling site and by virtue of the frequency of the fisheries sampling sites, relatively close to a fisheries sampling site. One of the problems with using a fuzzy logic approach and its HSI measure, there is no way to directly compare the habitat availability of the different modelling reaches. This is because the HSI presented is a function of area, and because the different study reaches have different widths and lengths, it is impossible to compare the modelling reaches directly. In the future, if a bioenergetic modelling approach were adopted (e.g. Fausch, 1984; Hughes and Dill, 1990; Van Winkle *et al.*, 1998; Hayes *et al.*, 2000) it would be valid to compare the predictions of habitat use with the observed ecology within the systems.

It appears that it doesn't matter where in the river reach the samples are taken from, as long as a variety of habitats are sampled. It would have been interesting to take Surber samples and replicates in the four different flow types, to allow statistical comparison of the different flow types, which was not possible with the existing sampling strategy. It has been seen that flow type may be a useful biotope in instream ecology. 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 is said to provide 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). Rempel *et al.* (2000) showed the importance of deep- slow flowing areas, and Pardo and Armitage (1997) associated invertebrate groups and mesohabitats linked to substrate.

7.5. Final conclusions

With a very simple paired catchment experimental and modelling set-up, this thesis aimed to investigate the ecological impacts of redesigning compensation flow releases. Despite the very simple design it was very difficult to detect these impacts. This detection was complicated by two main sources of variation which act over a number of scales (Figure 7.2):

1) flow variability; and
2) spatial variability.

The importance of flow variability was first highlighted in Chapter 2, where it was seen that the smaller size of the Rivelin compensation flow reservoir, led to more overtopping events than from the Loxley compensation reservoir. The importance of this flow variability was seen in the macroinvertebrate results, with the two upstream sites displaying a reaction to the different flow regimes. The Loxley appeared homogenised by the much larger compensation flow; which may also have served to buffer the instream ecology against any overtopping events which occur. Very little change was observed in the macroinvertebrate populations after the change in compensation flows. This is despite the very large (proportionally) change in compensation flow on the Rivelin. This lack of change could be due to a detachment of the invertebrate community from the environmental variables or due to the interaction of reach scale geomorphology and discharge creating patchy habitats within the rivers (Figure 7.5), meaning that changes in habitat are not linearly equated to increases in discharge.

The importance of the interaction of reach scale morphology and discharge was also seen in the fisheries work conducted. The densities of brown trout observed in the Rivelin were very similar to those observed in the Loxley, this is despite the much lower compensation flow releases into the Rivelin. The analyses also showed that the >0+(<20cm) life stage appeared to be habitat limited in the pre-change populations in both rivers. The 2003-2004 dry period was seen to have an effect on the 0+ trout populations within both of the rivers. When the compensation flows were changed, the growth rate of older brown trout in the Rivelin increased, but as with the invertebrates, establishing causality between the environmental change and the fisheries populations is difficult.

The two-dimensional modelling first showed that a two-dimensional model can be used to simulate the hydrodynamics of upland Millstone Grit catchments such as the Rivelin and Loxley. When combined with a fuzzy-logic based habitat model, available habitat predictions were made for a number of life stages of brown trout and functional feeding groups of macroinvertebrates. The discharges simulated were those directly controllable by the regulatory body. By using this modelling approach, the results were not contaminated by flow variability (as discovered to be of importance in Chapters 3 and 4), and as a consequence the approach can be used to assess the specific changes advocated.

This study has therefore shown that flow variability is vitally important to compensation flow rivers. This flow variability, combined with the geomorphology of the systems determines the impact of altering the compensation flows.

Appendix A

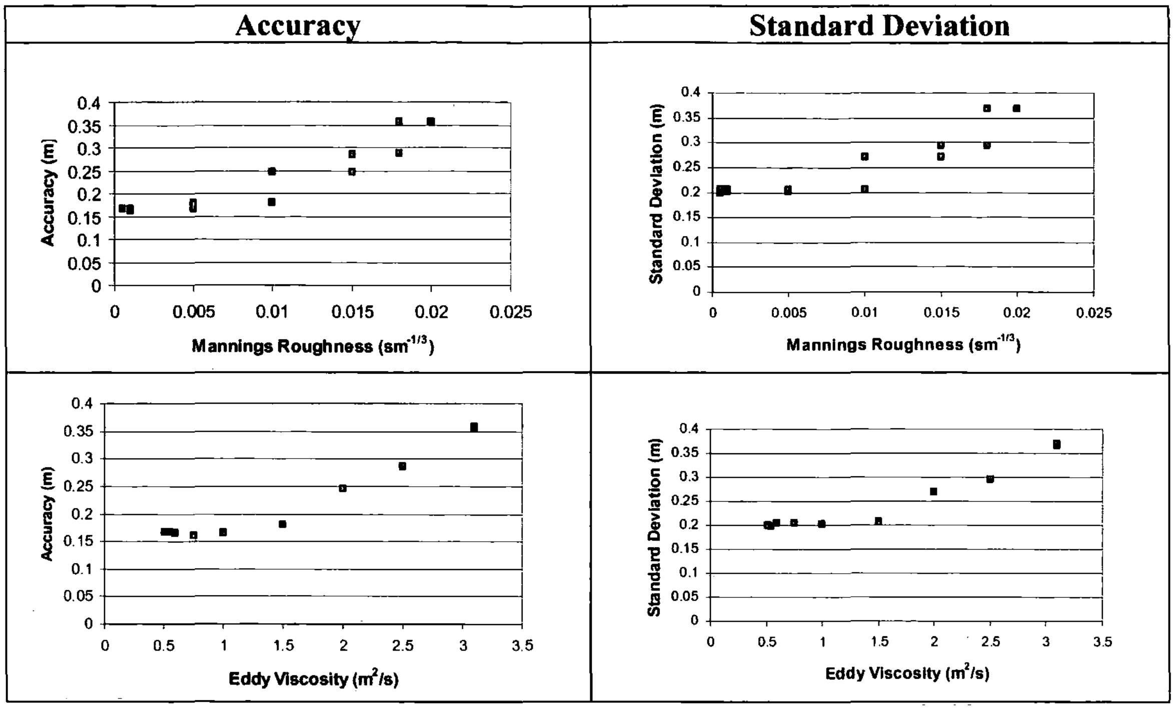


Figure 5,4.1. Figure to show the waters edge calibration results for RU-1 for the pre-change waters edge.

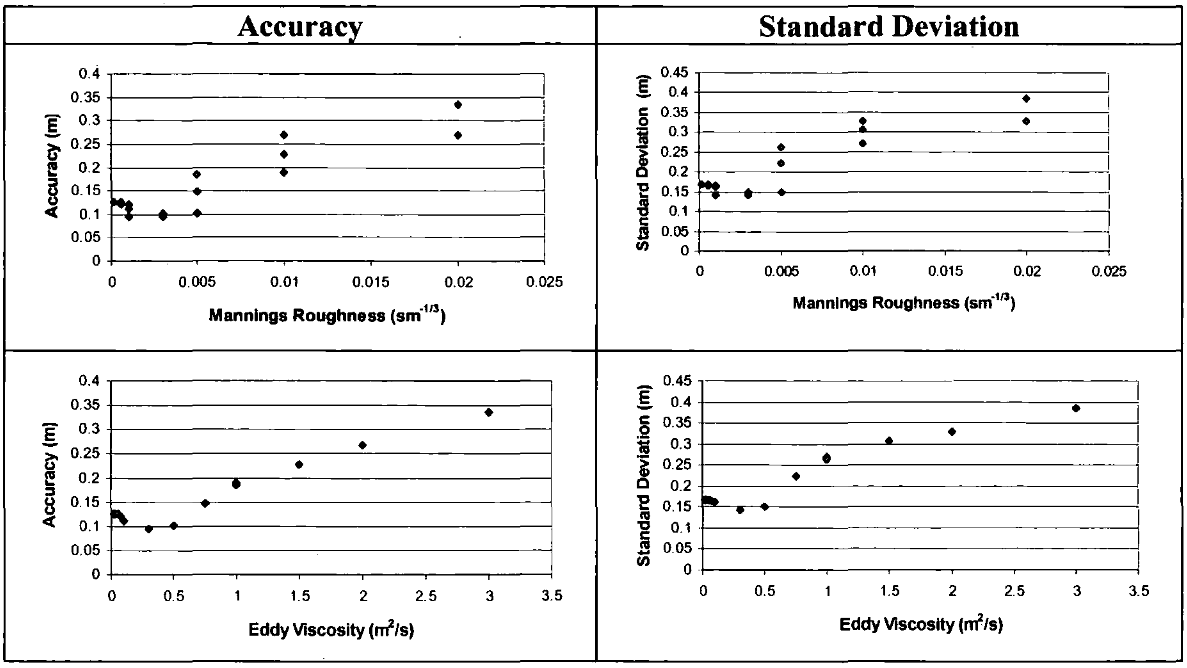


Figure 5.4.2. Figure to show the waters edge calibration results for RU-1 for the post-change waters edge.

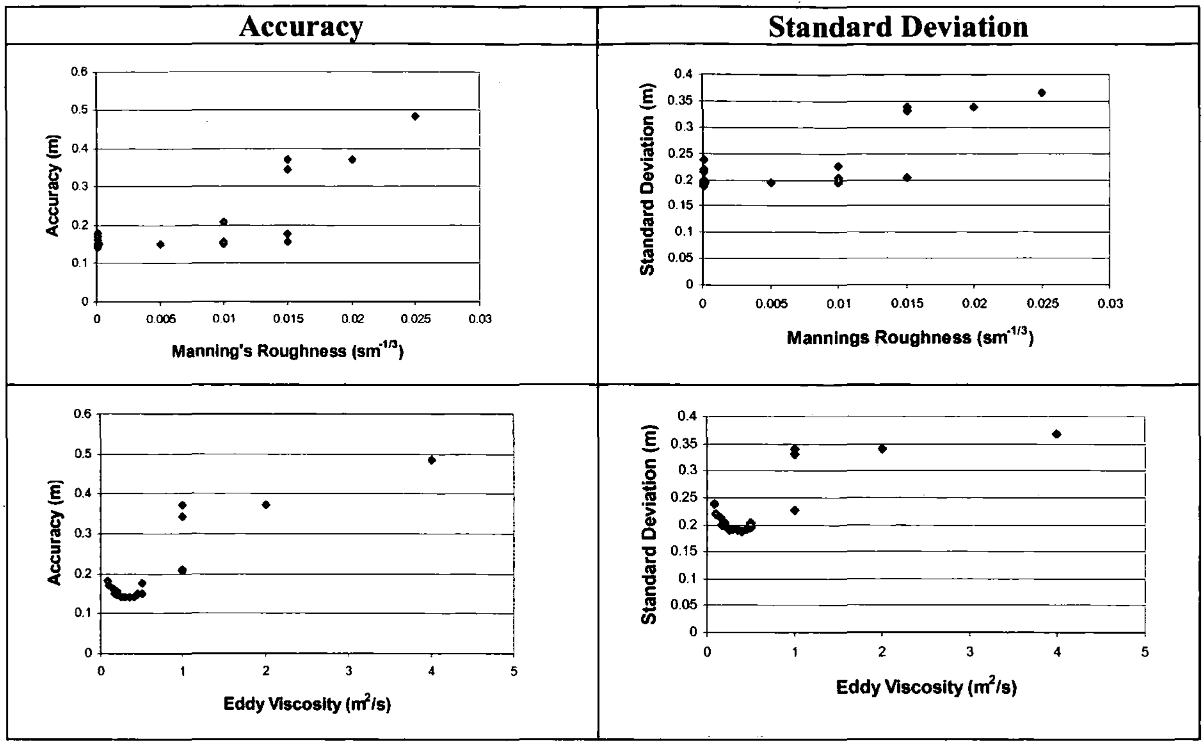


Figure 5.4.3. Figure to show the waters edge calibration results for the pre-change waters edge validation at RU-2.

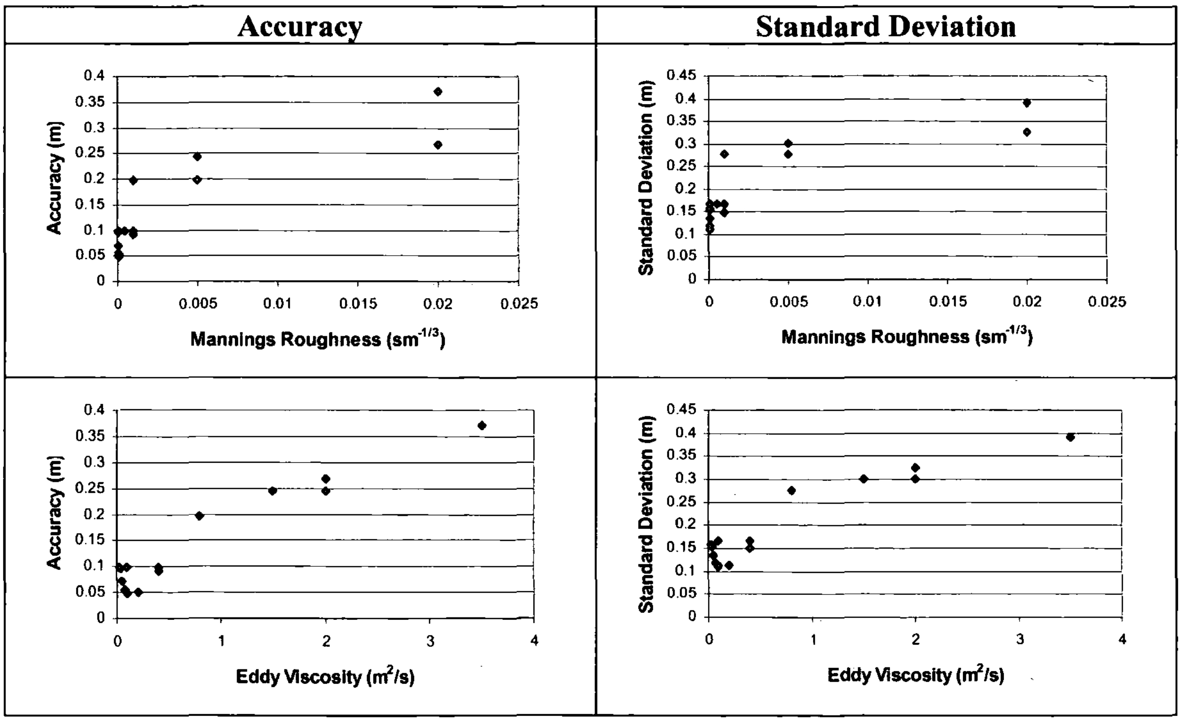


Figure 5.4.4. Figure to show the waters edge calibration results for the post-change waters edge validation at RU-2.

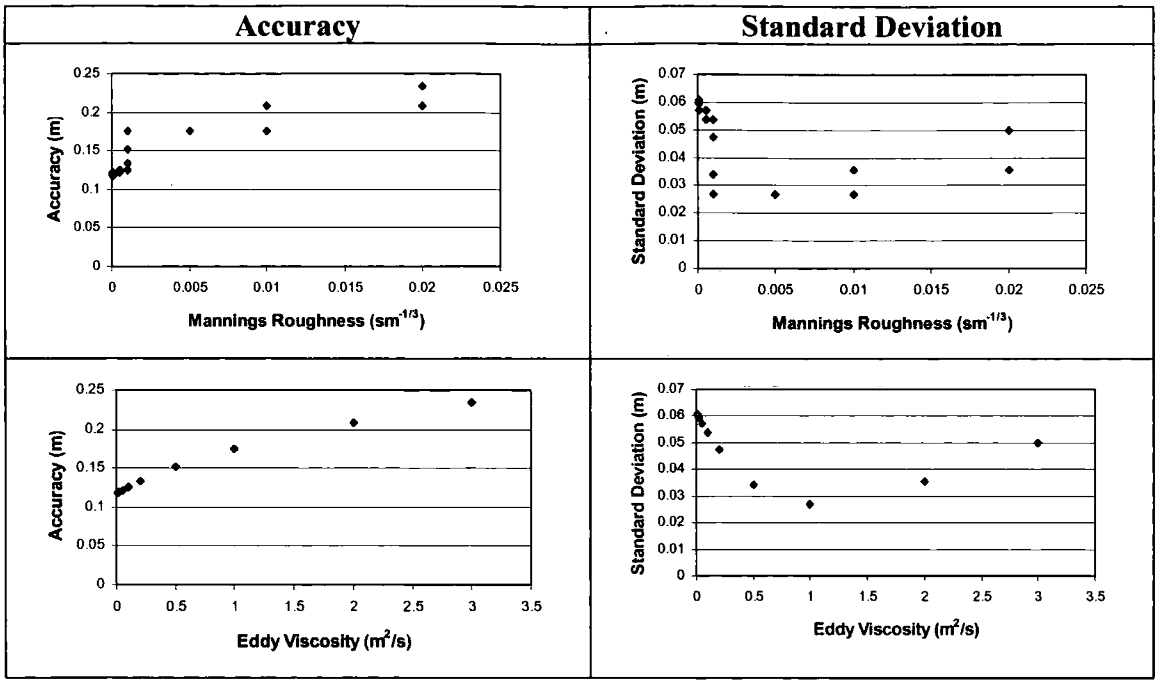


Figure 5.4.5. Figure to show the accuracy and standard deviation of the point depth calibration data compared with the simulation RU-1.

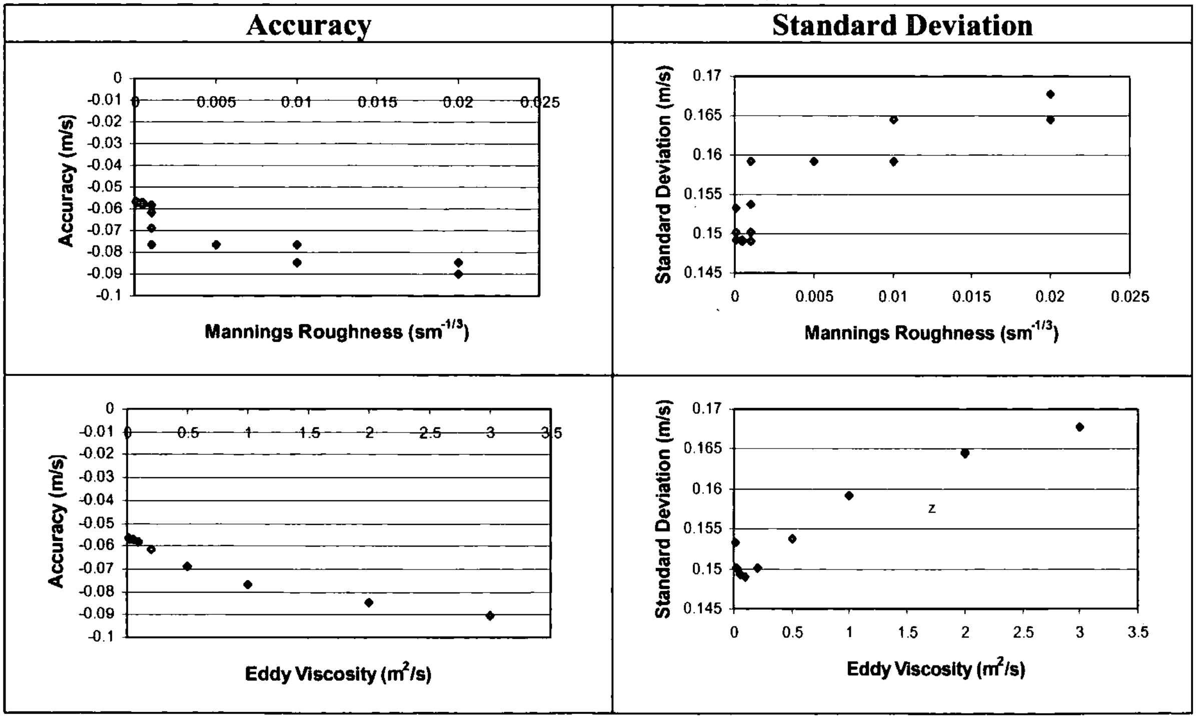


Figure 5.4.6. Figure to show the accuracy and standard deviation of the point velocity calibration data compared with the simulation for RU-1.

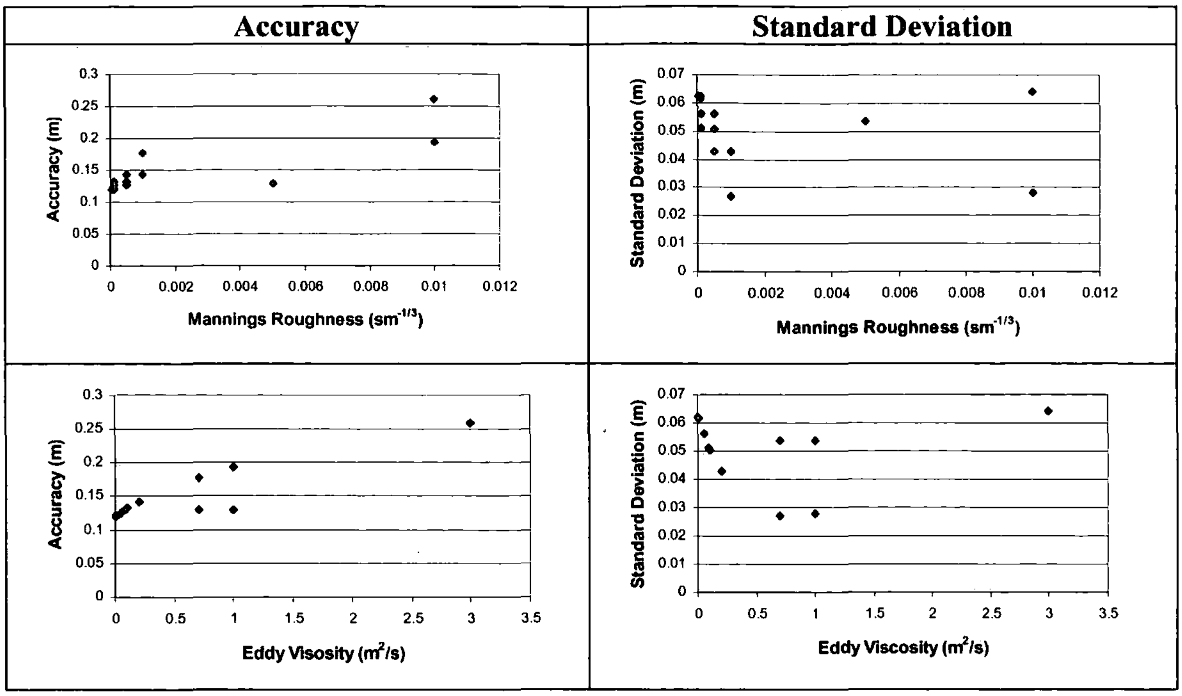


Figure 5.4.7. Figure to show the accuracy and standard deviation of the point depth calibration data compared with the simulation for RU-2.

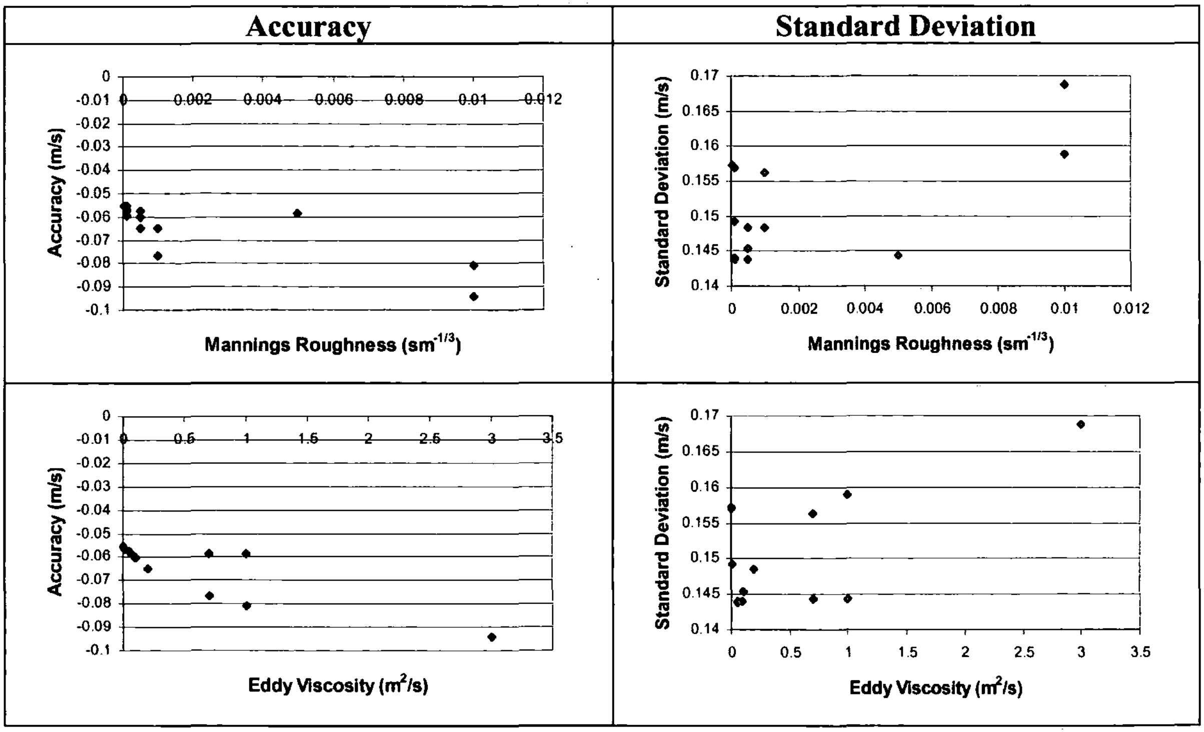


Figure 5.4.8. Figure to show the accuracy and standard deviation of the point velocity calibration data compared with the simulation for RU-2.

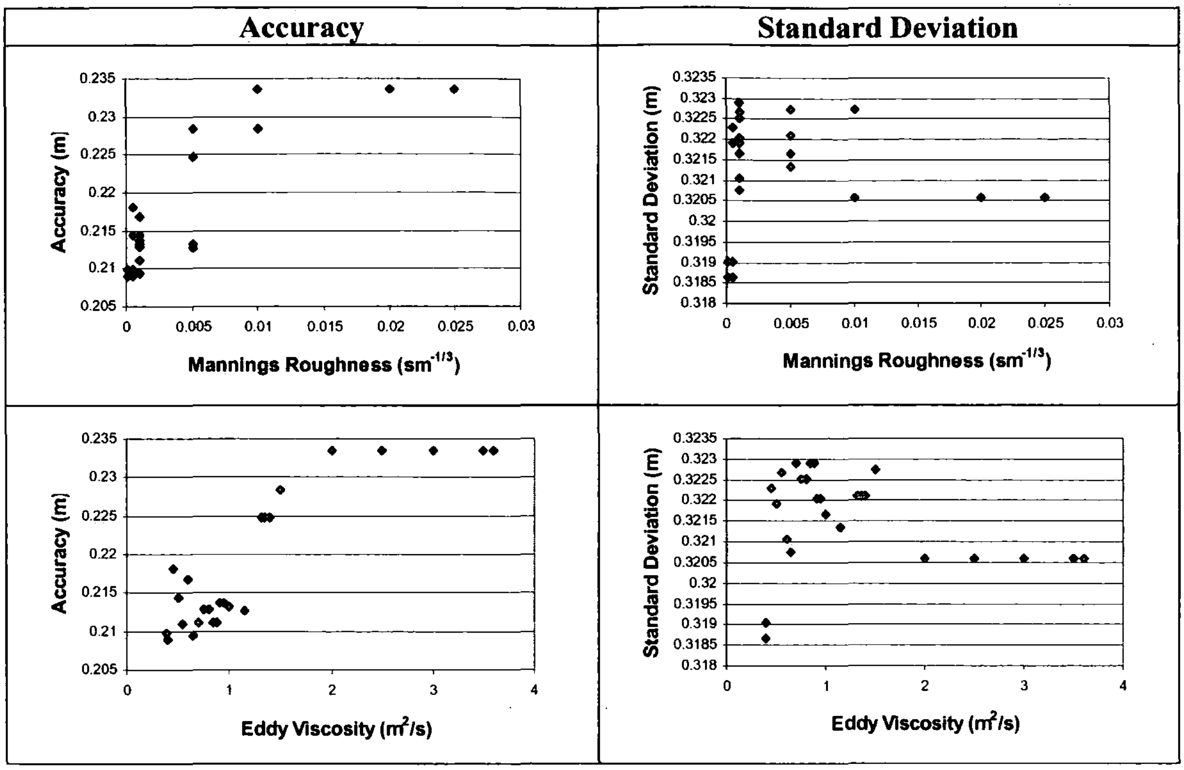


Figure 5.4.9. Figure to show the pre change waters edge calibration results for RD-1.

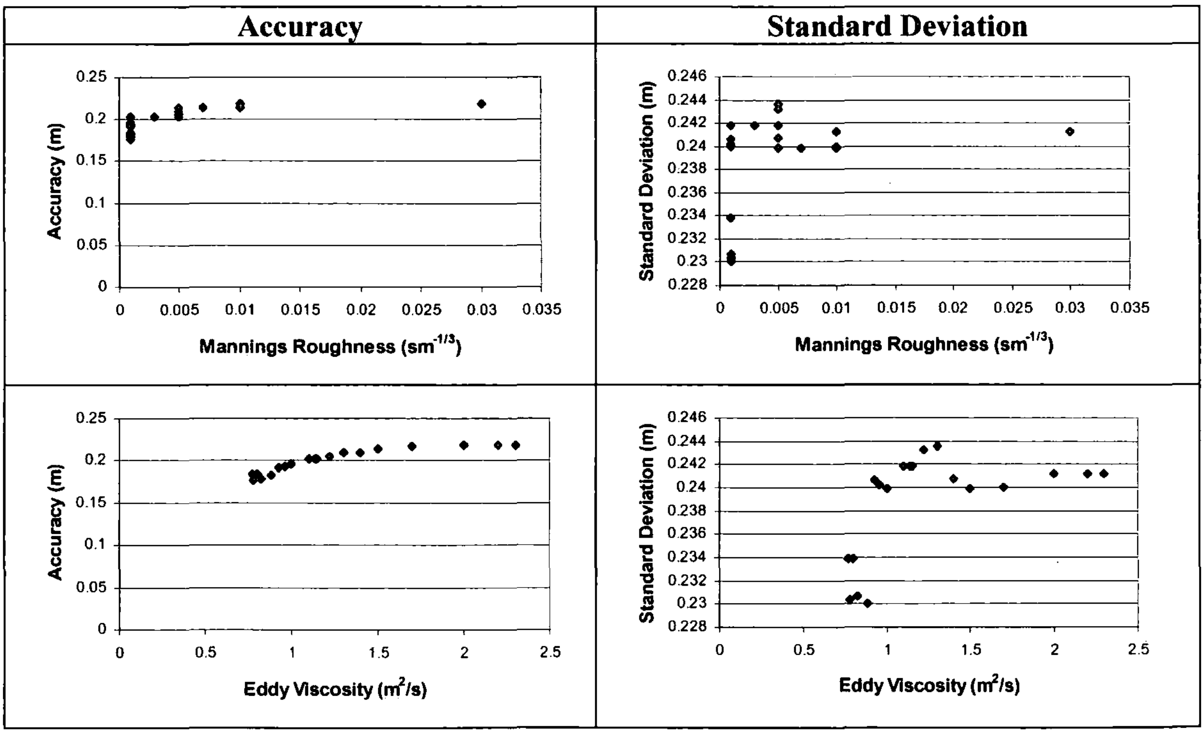


Figure 5.4.10. Figure to show the post change waters edge calibration results for RD-1.

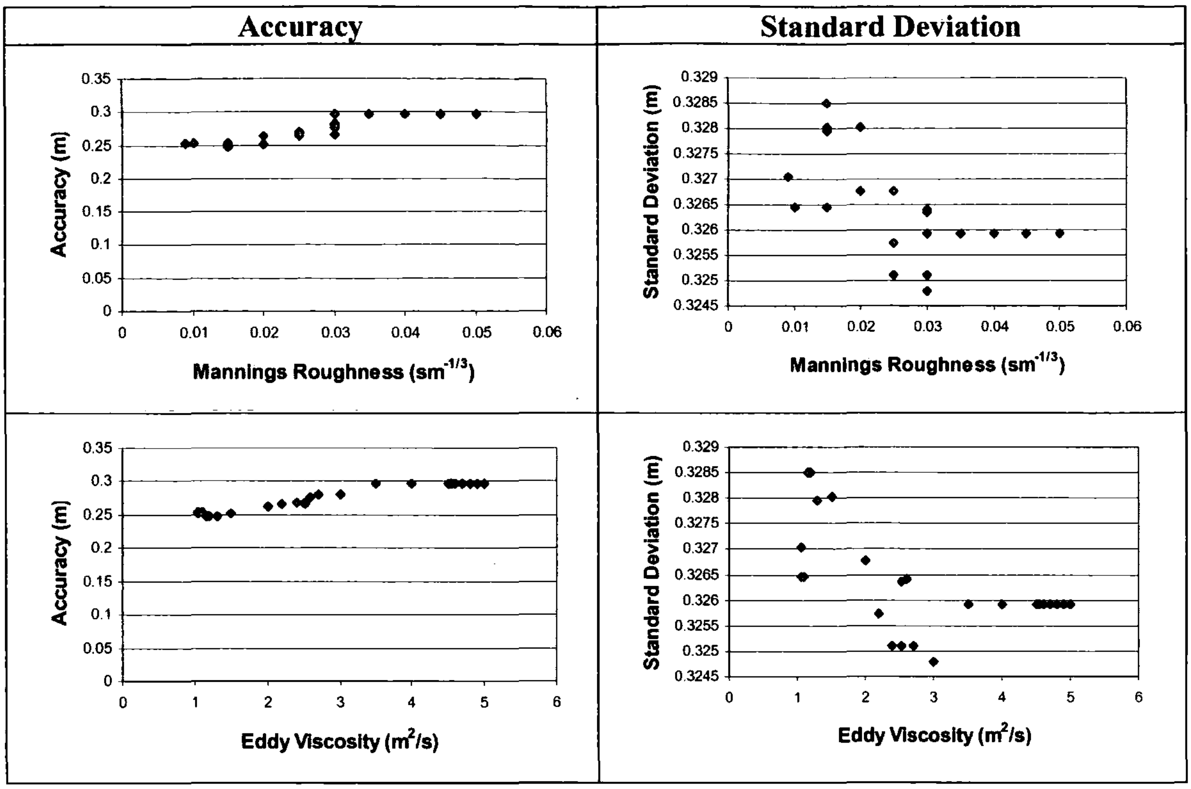


Figure 5.4.11. Figure to show the pre change waters edge calibration results for RD-2.

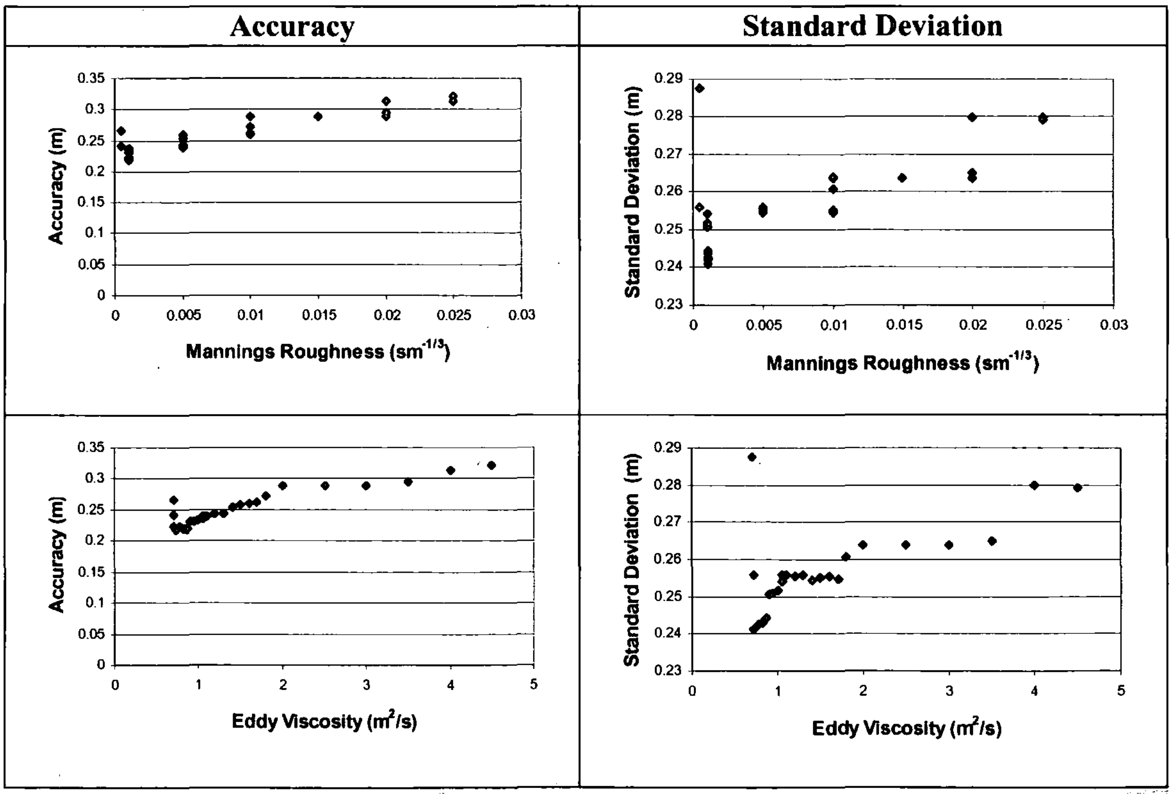


Figure 5.4.12. Figure to show the post change waters edge calibration results for RD-2.

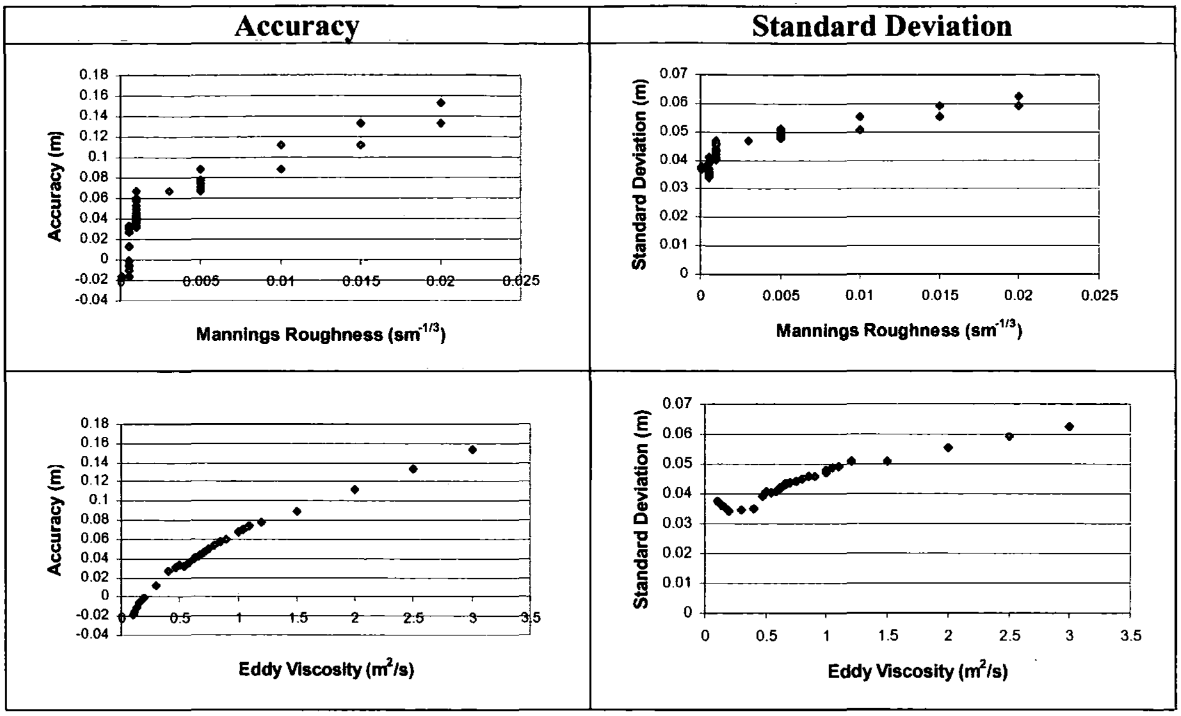


Figure 5.4.13. Figure to show the accuracy and standard deviation of the point depth calibration data compared with the simulation for RD-1.

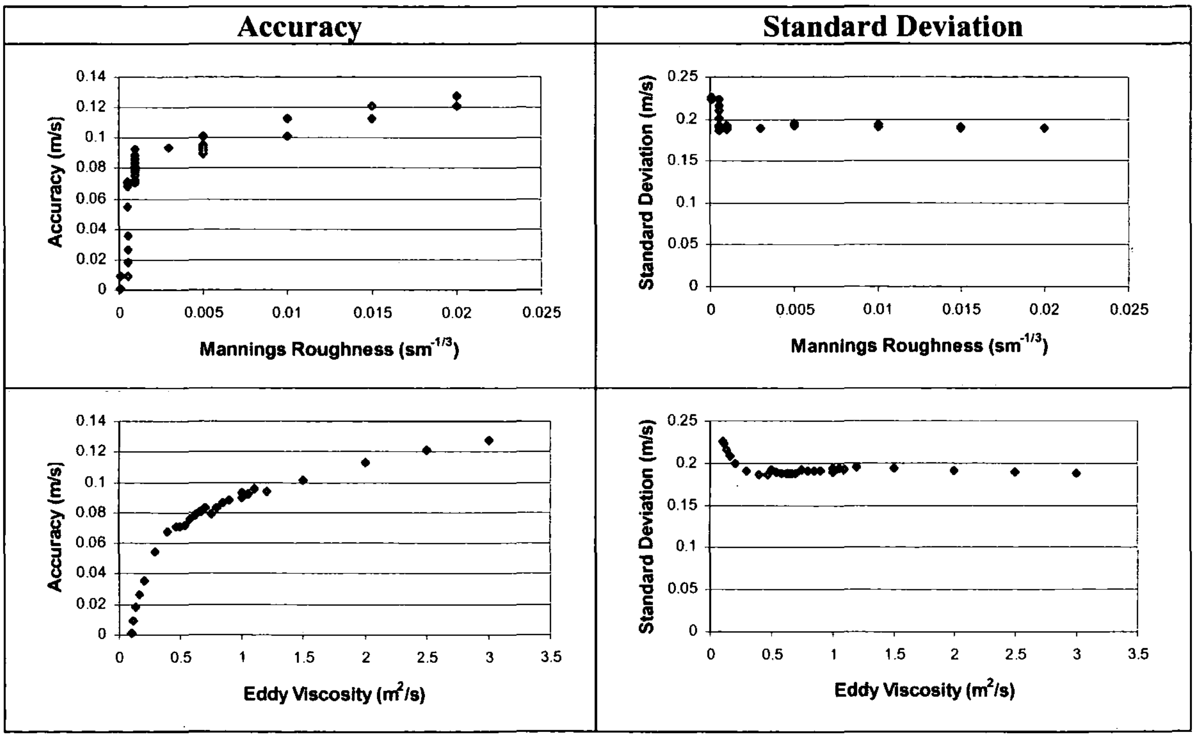


Figure 5.4.14. Figure to show the accuracy and standard deviation of the point velocity calibration data compared with the simulation for RD-1.

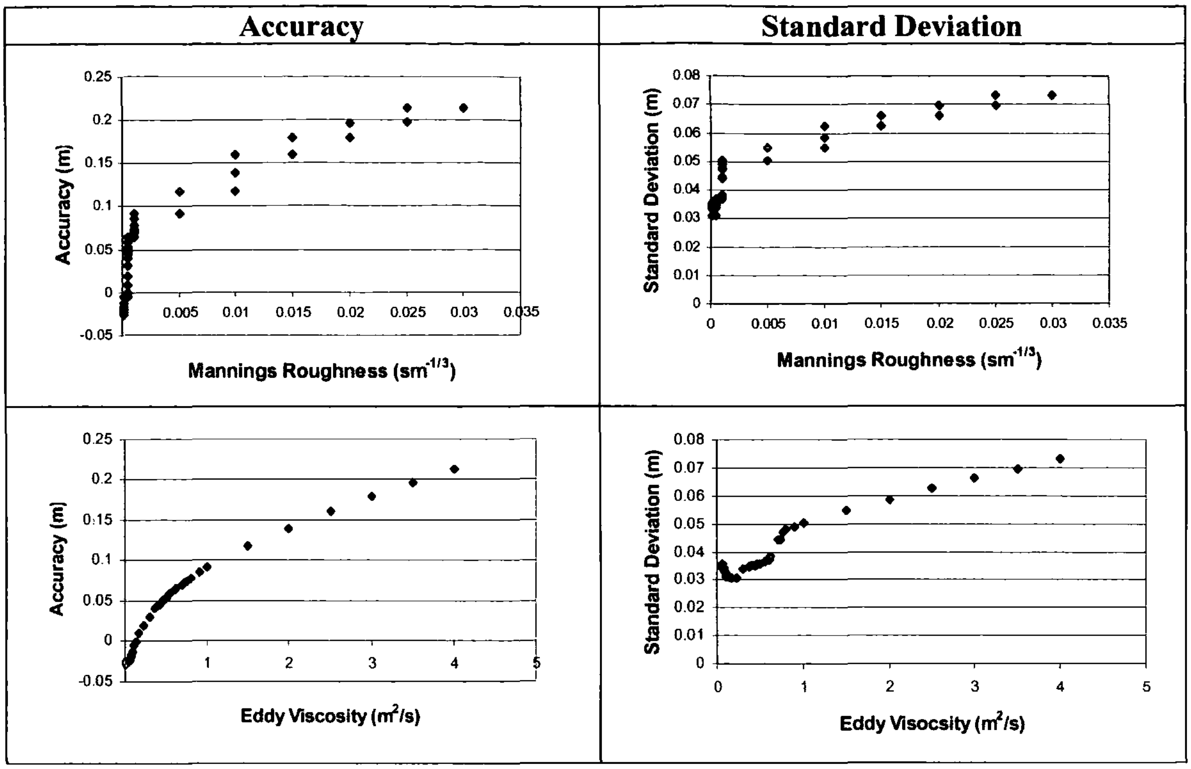


Figure 5.4.15. Figure to show the accuracy and standard deviation of the point depth calibration data compared with the simulation for RD-2.

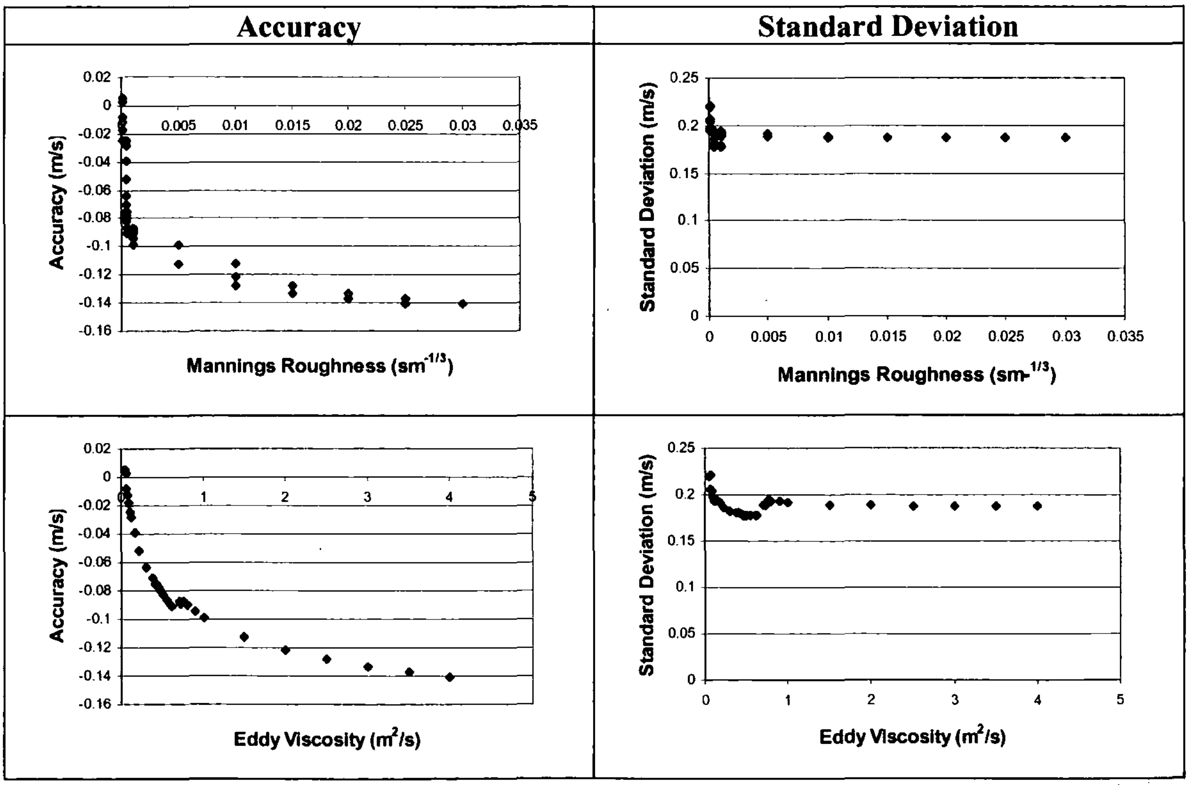


Figure 5.4.16. Figure to show the accuracy and standard deviation of the point velocity calibration data compared with the simulation for RD-2.

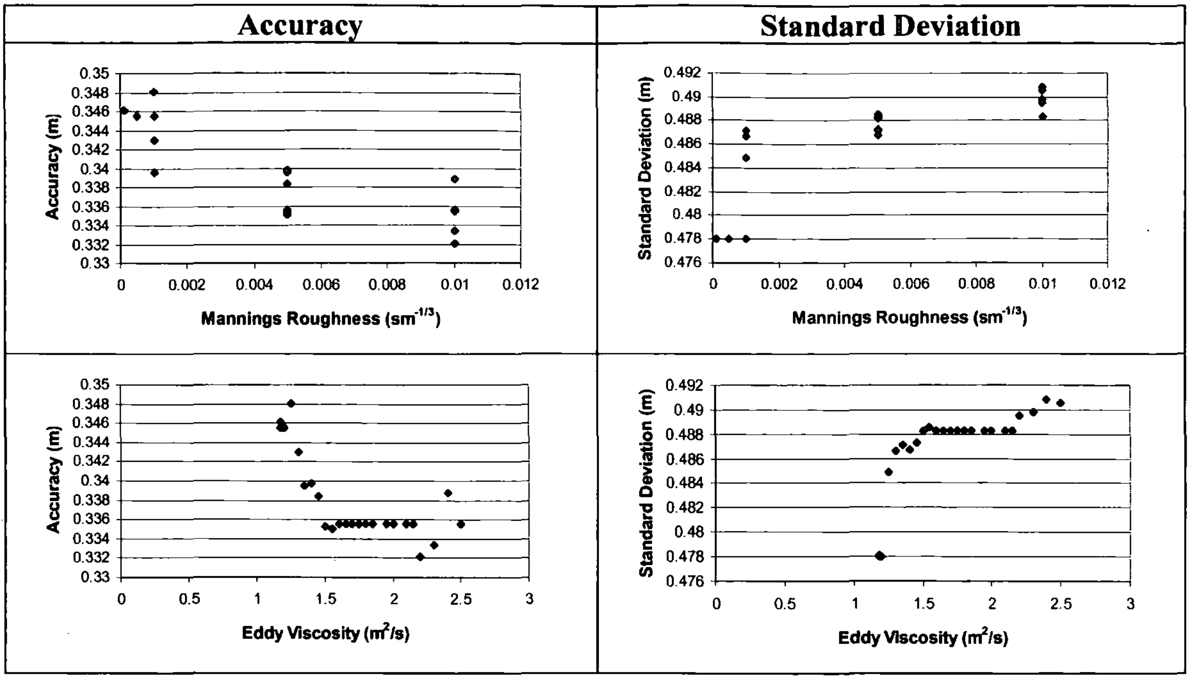


Figure 5.4.17. Figure to show the pre change waters edge calibration results for LU-1.

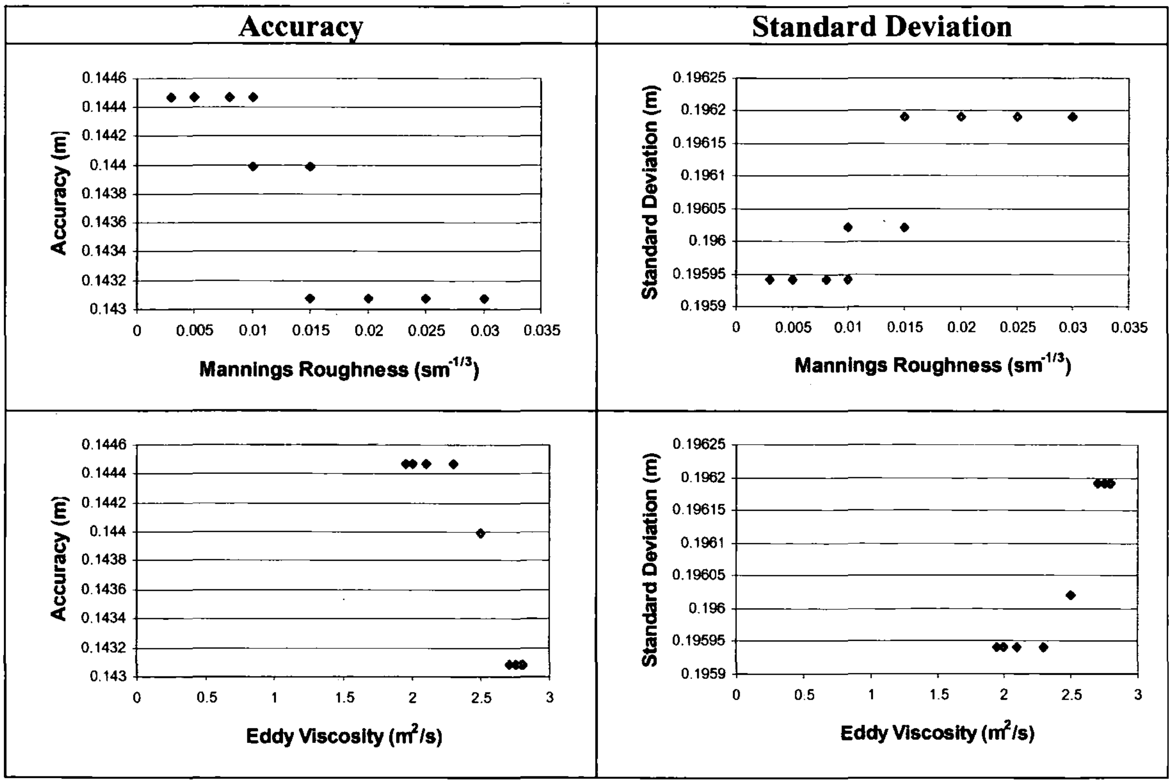
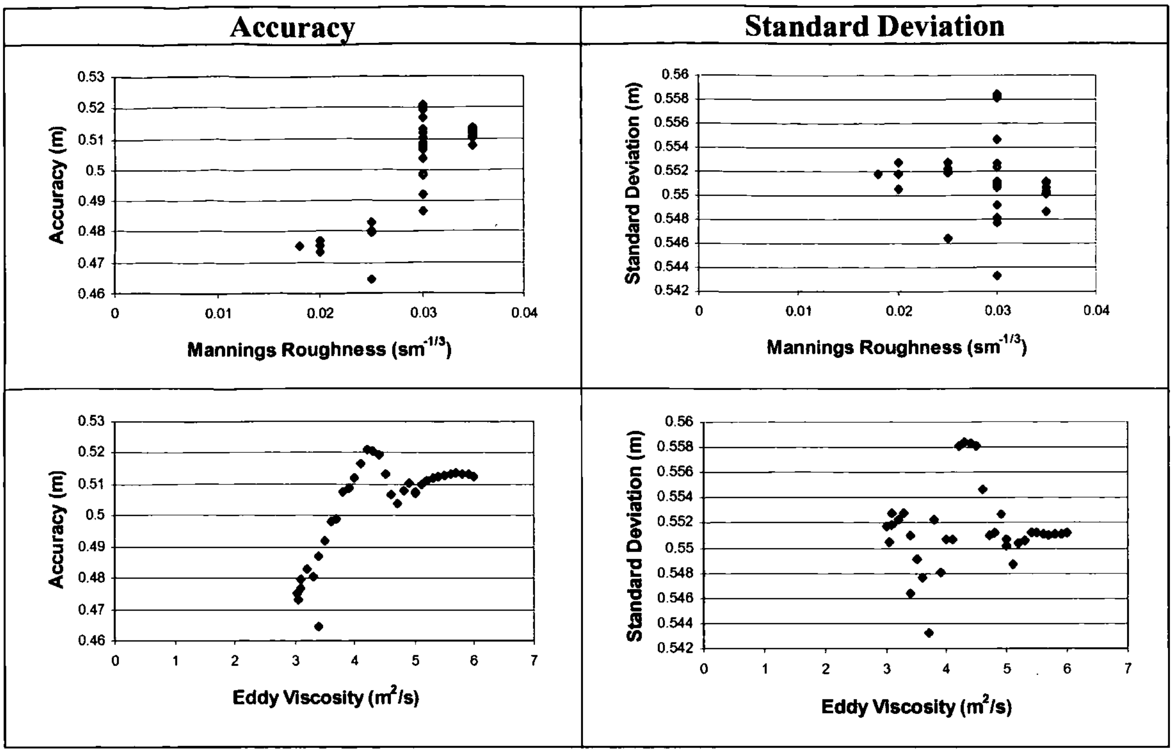
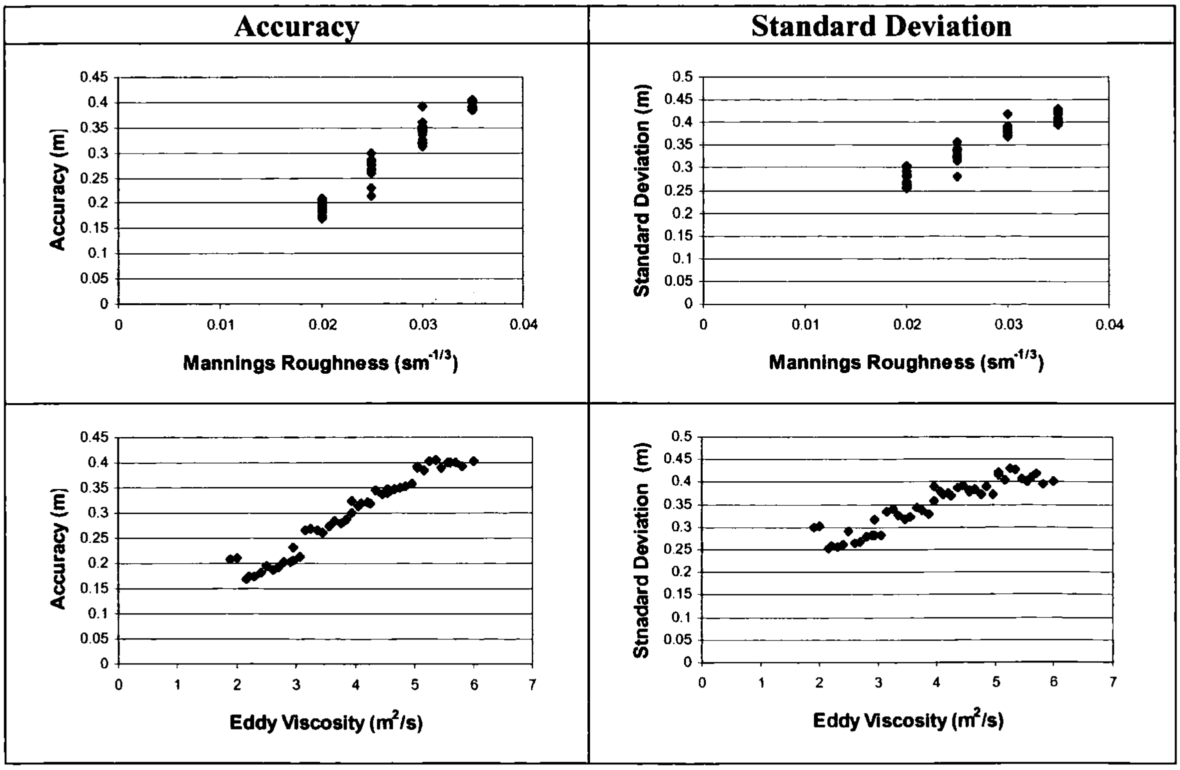


Figure 5.4.18. Figure to show the post change waters edge calibration results for LU-1.


Accuracy

Standard Deviation

Figure 5.4.19. Figure to show the pre change waters edge calibration results for LU-2.


Accuracy

Standard Deviation

Figure 5.4.20. Figure to show the post change waters edge calibration results for LU-2.

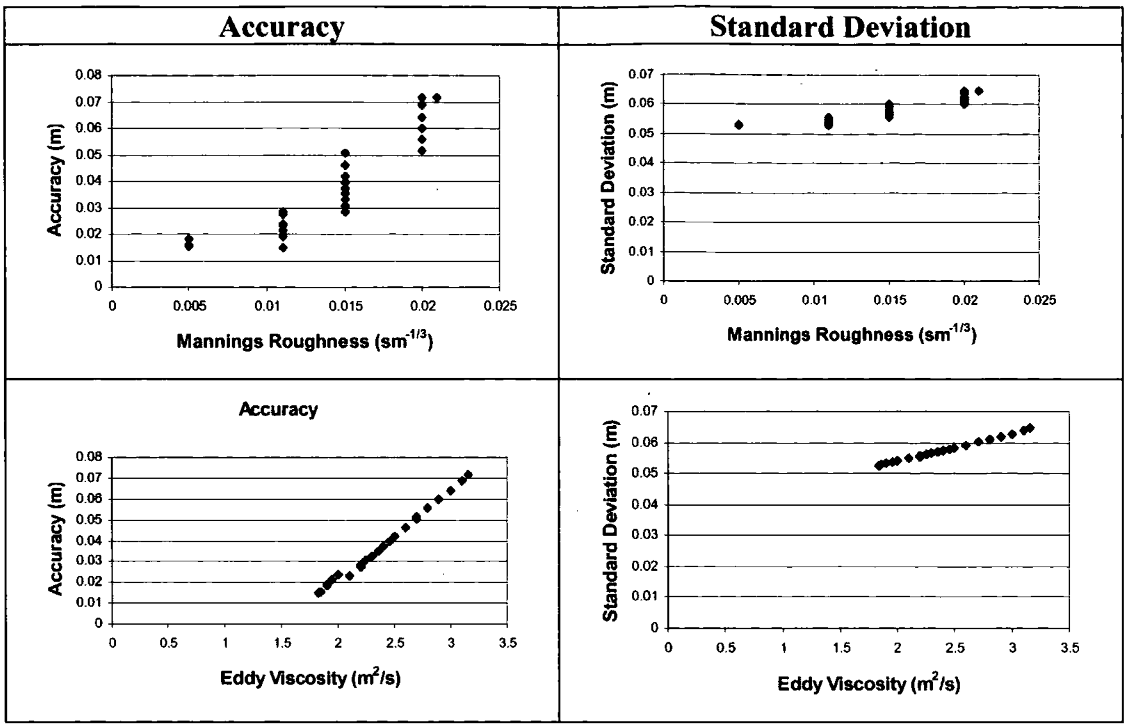


Figure 5.4.21. Figure to show the accuracy and standard deviation of the point depth calibration data compared with the simulation for LU-1.

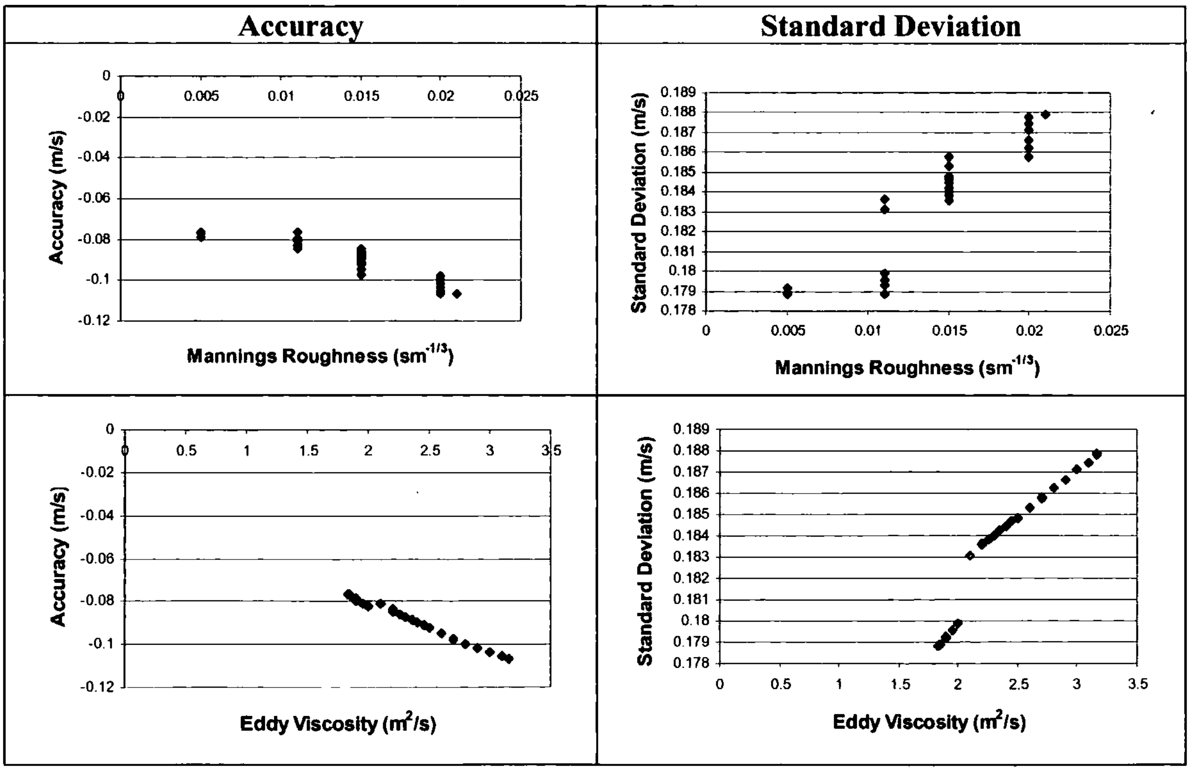


Figure 5.4.22. Figure to show the accuracy and standard deviation of the point velocity calibration data compared with the simulation for LU-1.

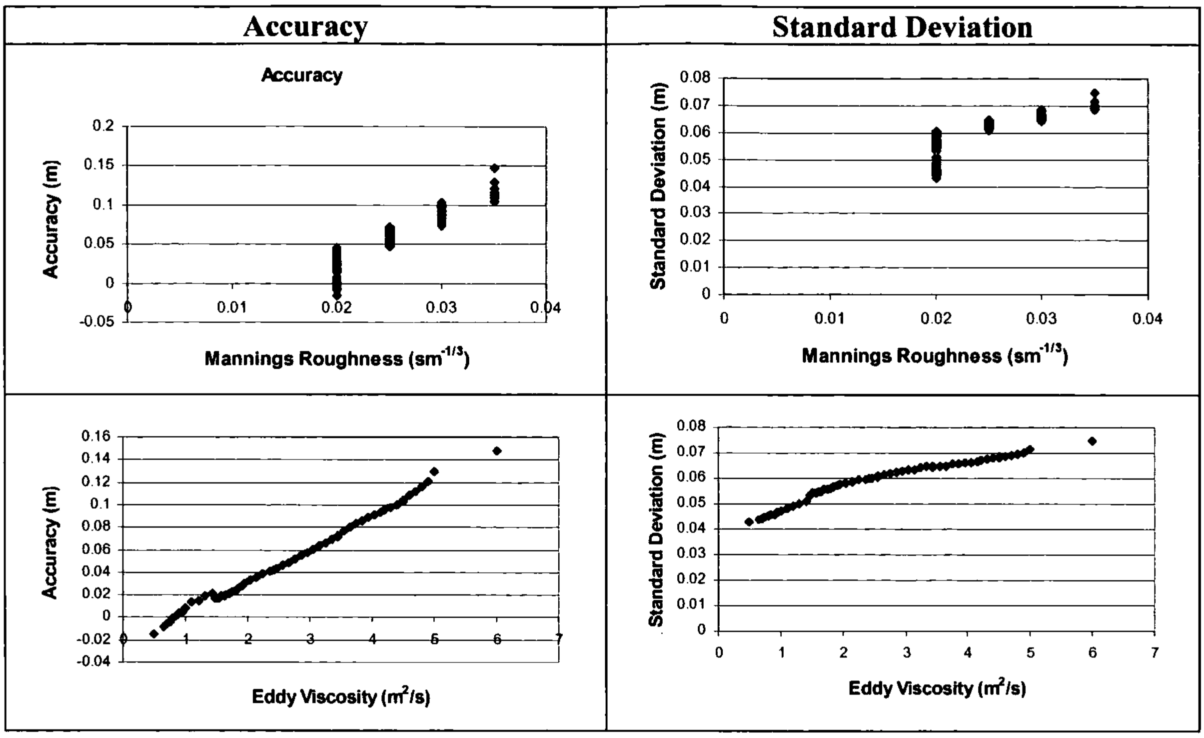


Figure 5.4.23. Figure to show the accuracy and standard deviation of the point depth calibration data compared with the simulation for LU-2.

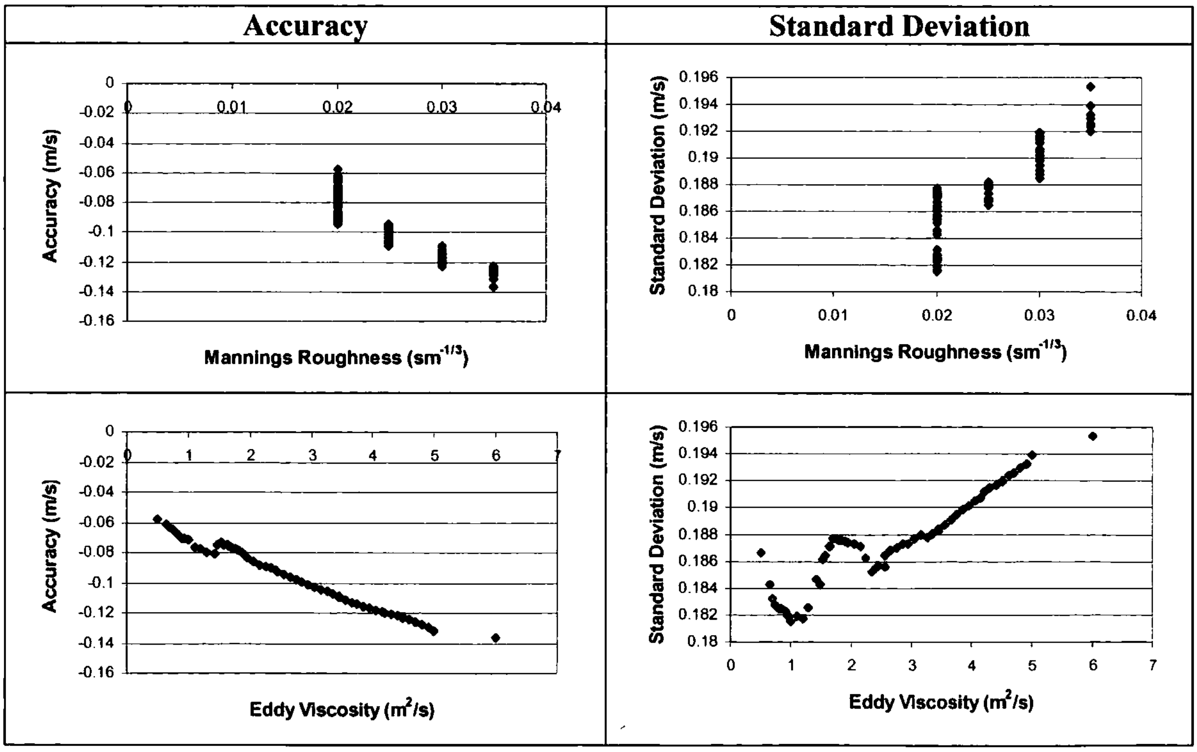


Figure 5.4.24. Figure to show the accuracy and standard deviation of the point velocity calibration data compared with the simulation for LU-2.

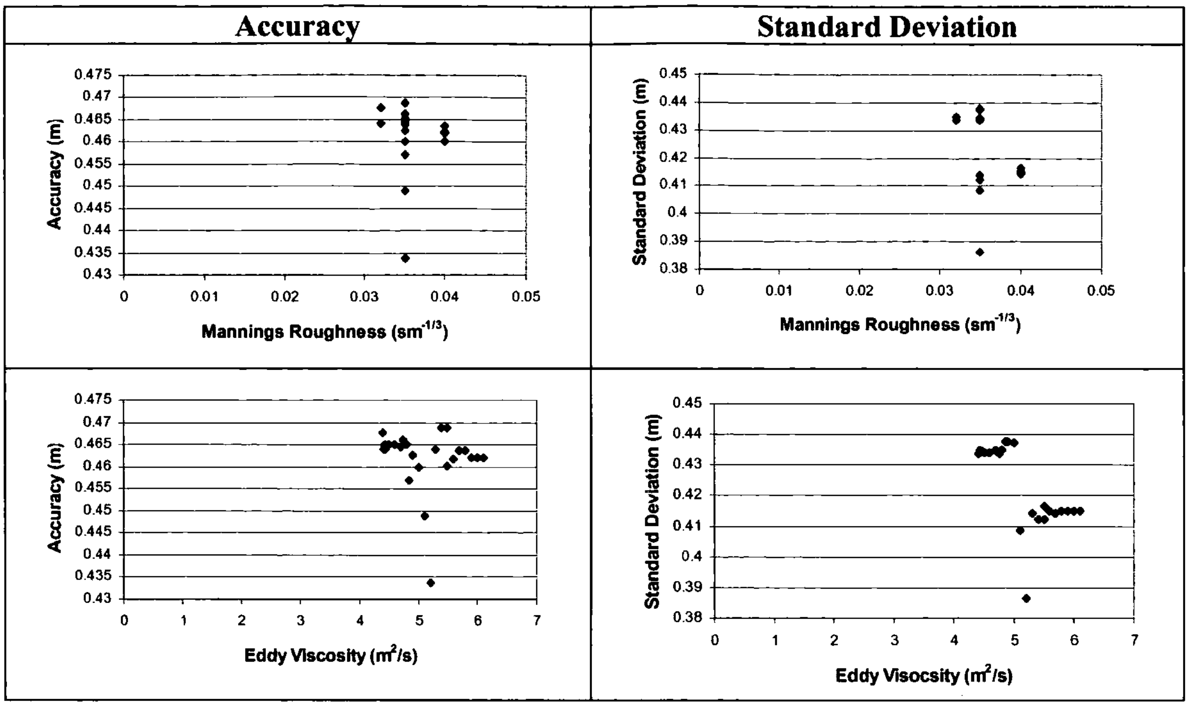


Figure 5.4.25. Figure to show the pre change waters edge calibration results for LD-1.

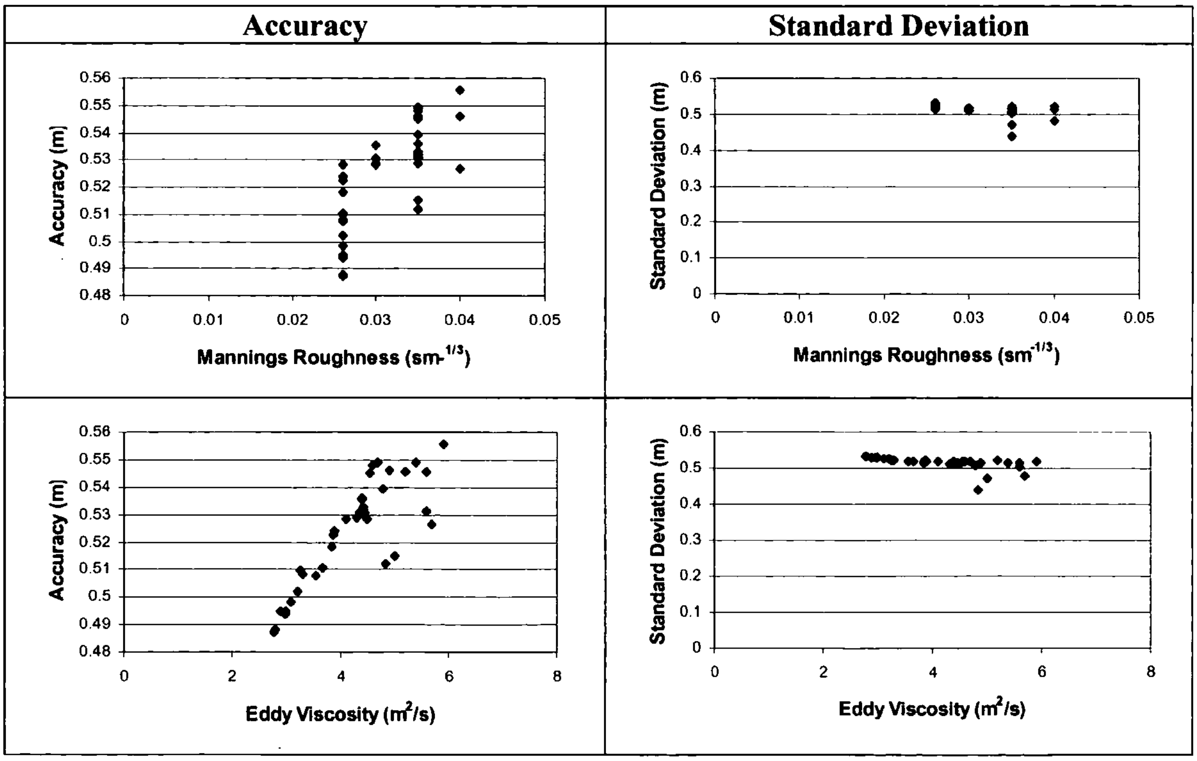


Figure 5.4.26. Figure to show the post change waters edge calibration results for LD-1.

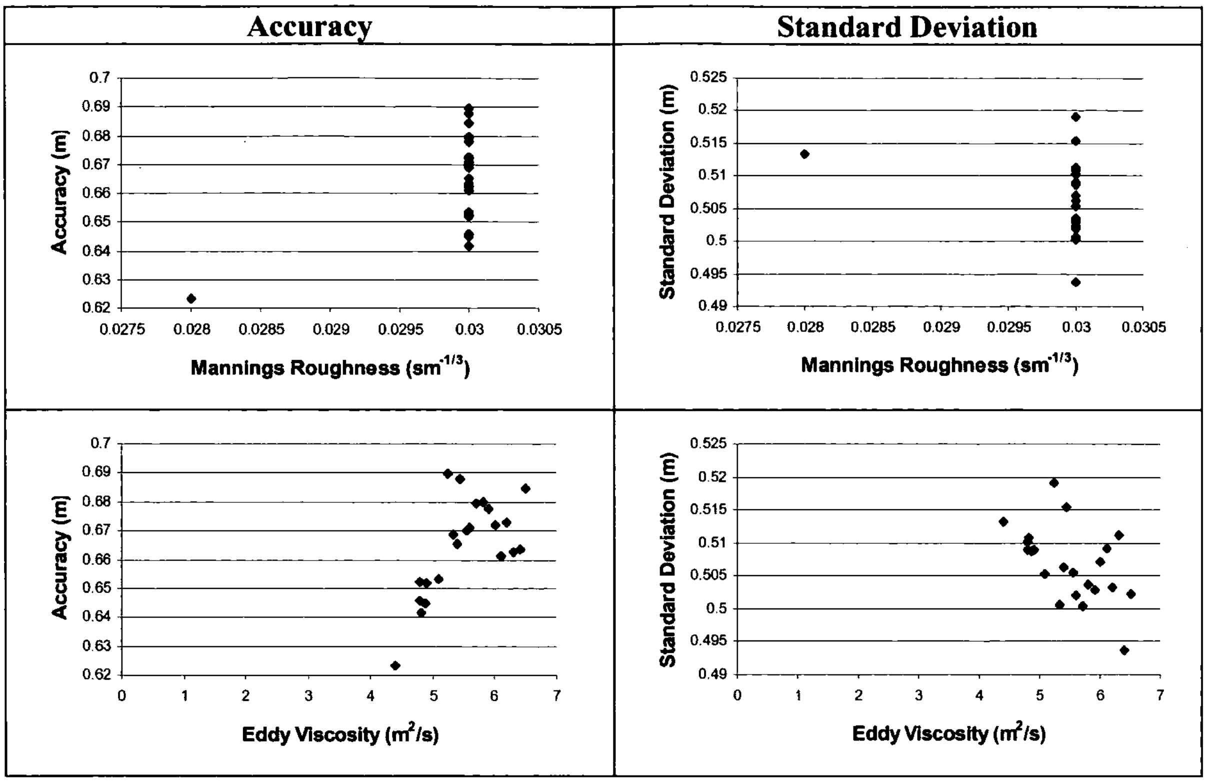


Figure 5.4.27. Figure to show the pre change waters edge calibration results for LD-2.

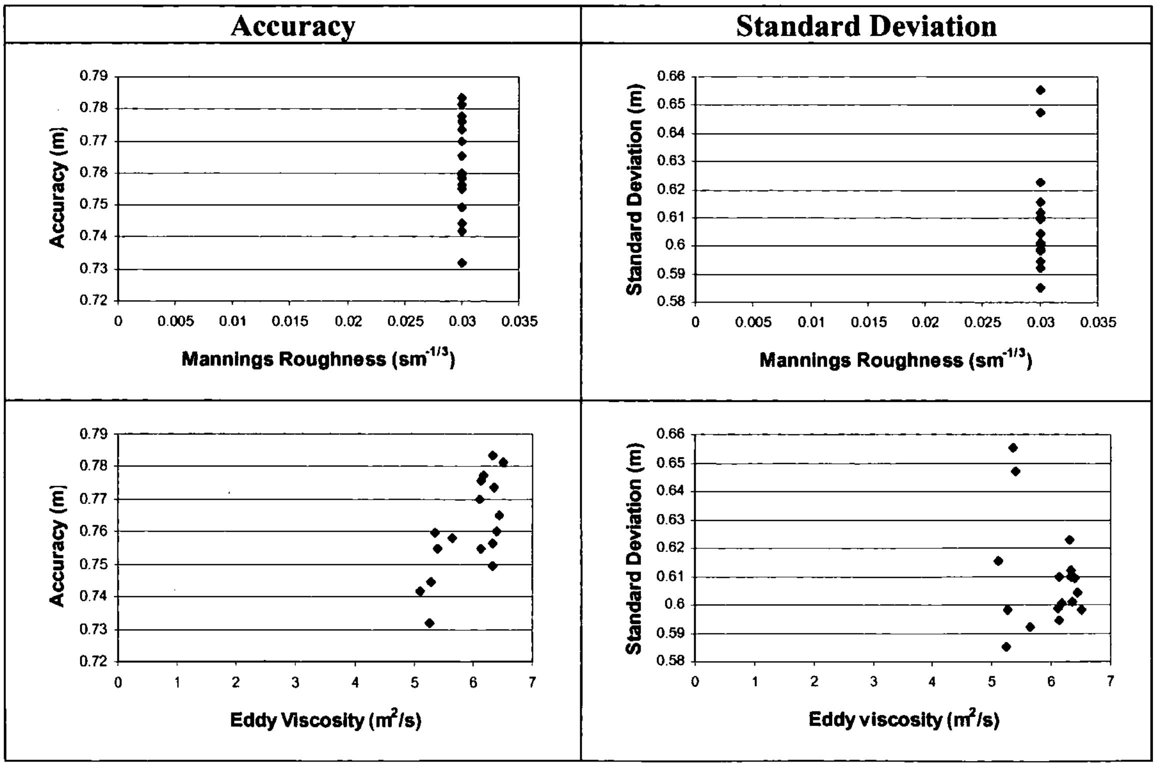


Figure 5.4.28. Figure to show the post change waters edge calibration results for LD-2.

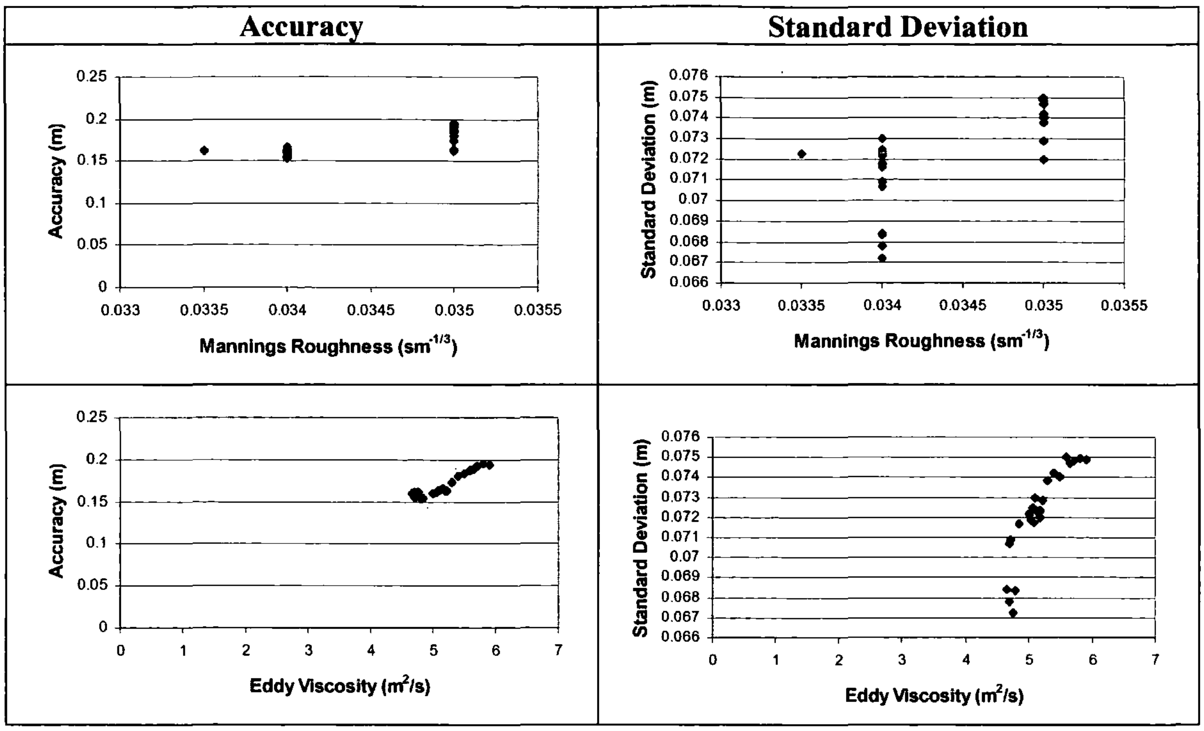


Figure 5.4.29. Figure to show the accuracy and standard deviation of the point depth calibration data compared with the simulation for LD-1.

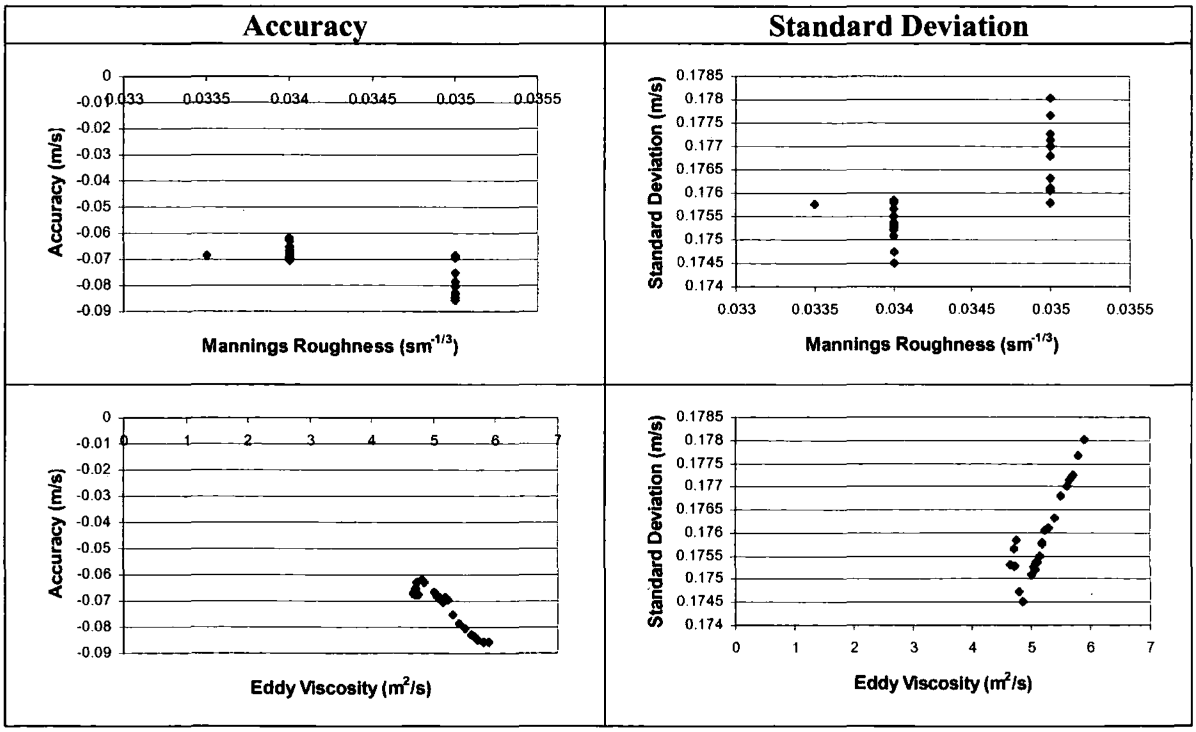


Figure 5.4.30. Figure to show the accuracy and standard deviation of the point velocity calibration data compared with the simulation for LD-1.

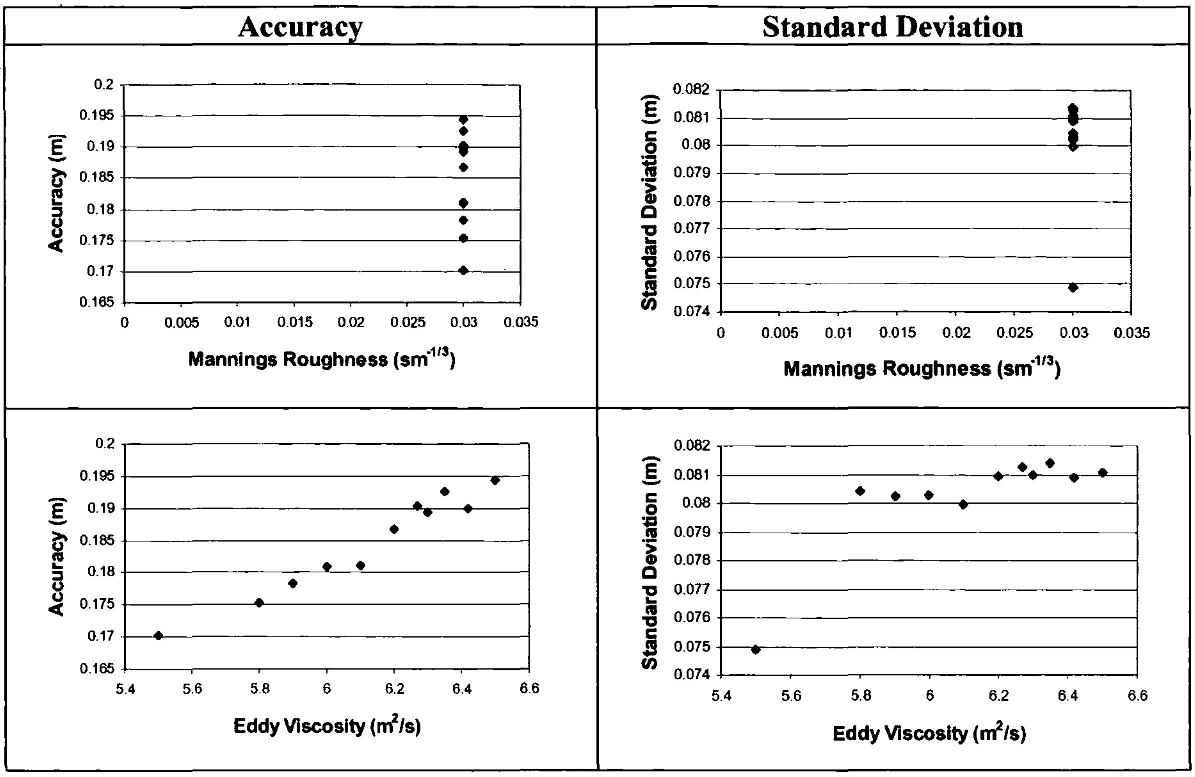


Figure 5.4.31. Figure to show the accuracy and standard deviation of the point depth calibration data compared with the simulation for LD-2.

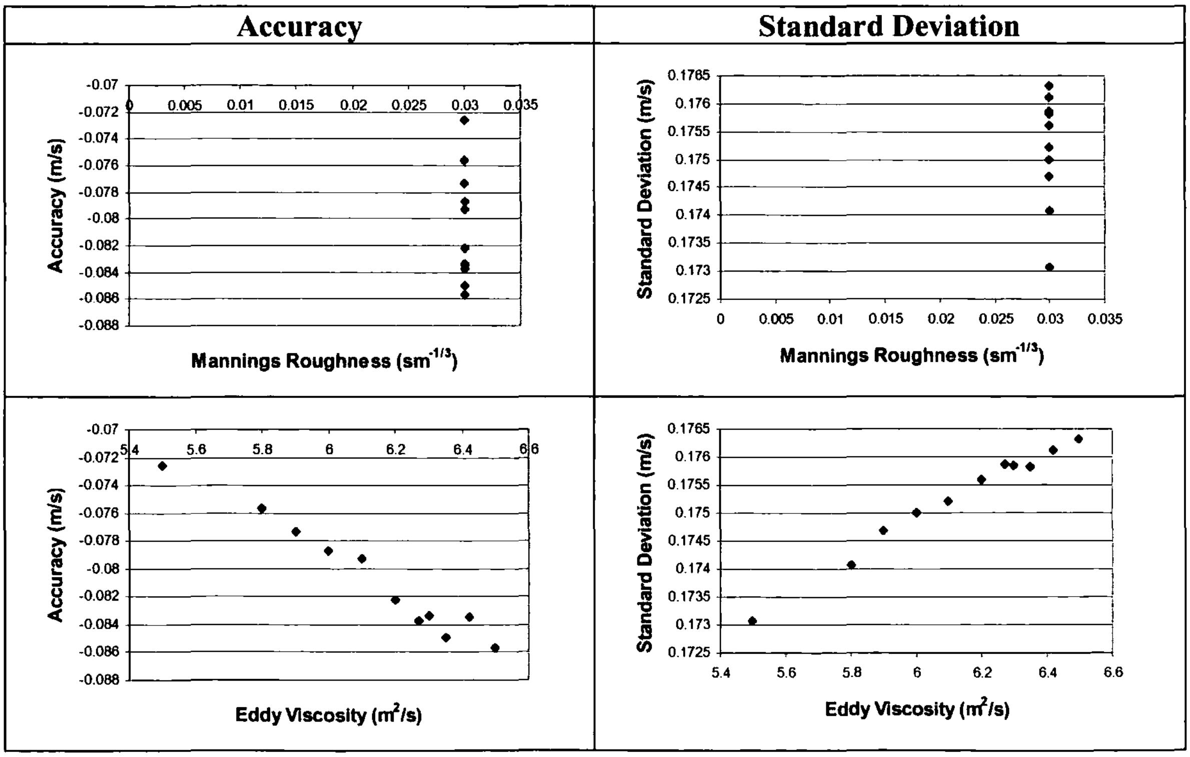


Figure 5.4.32. Figure to show the accuracy and standard deviation of the point velocity calibration data compared with the simulation for LD-2.

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