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Outcomes of river rehabilitation on instream hydraulics and fish communities

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Michelle Anne Smith

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ABSTRACT

OUTCOMES OF RIVER REHABILITATION ON INSTREAM HYDRAULICS AND FISH COMMUNITIES

All fish species have specific habitat requirements, which differ according to life history and life stage. Where requirements are not met, or are inadequate for a particular species, the species will be locally absent or the population in poor condition and abundance. As a result of numerous anthropogenic stressors, river systems, and consequently fish physical habitat, have undergone considerable transformation, frequently resulting in homogenisation of the river channel, often to the detriment of the fish biota present. Following the emergence of the EC Water Framework Directive (WFD) and EC Habitats Directive (HD), there has been an increase in river rehabilitation schemes to ameliorate anthropogenic pressures of rivers and augment ecological status to meet specific obligations. However, despite the extensive implementation of river rehabilitation programmes little follow up monitoring, and dissemination of results takes place leaving a paucity of information on the outcomes of such schemes on in stream hydraulic conditions and fish community composition. Four river rehabilitation schemes on three UK Rivers were monitored to assess the effects of the schemes on instream hydraulic conditions and fish community composition and structure.

Variation of instream hydraulics was assessed before and after the rehabilitation to investigate the environmental outcomes of river rehabilitation schemes. Little significant change in hydraulic conditions was observed following river rehabilitation at all sites surveyed although a significant decrease in depth and a significant increase in flow velocity was observed at the most upstream site following weir removal from the River Dove at Dovedale.

Little change in fish species composition was observed following river rehabilitation at all sites.

Given the importance of physical habitat to fish, surveys were conducted on a meso-scale in heterogeneous rivers to assess differences in hydraulic conditions and fish species composition of different habitat types. Glides were generally deeper than riffles, with fish species composition dominated by >1+ brown trout whereas riffles were generally shallower than glides. The composition of the fish community in riffles generally contained a greater proportion of bullhead and 0+ brown trout than glides.

Due to the importance of fish as an indicator of ecological quality under WFD guidelines, it is imperative to understand the intricate linkages between fish species and hydraulic habitat. Habitat use of all species captured was investigated and despite similarity in the range of values measured, different preferences were shown by different species. The relationship between descriptors of fish community composition and measures of hydraulic habitat were also investigated and revealed that individual hydraulic parameters have little influence over fish community composition.

1 CHAPTER ONE: GENERAL INTRODUCTION

Rivers are subjected to many pressures, direct impacts of anthropogenic activity resulting in change to the river system. Pressures can be divided into three groups: physical, biological and chemical (Degerman *et al*., 2007), and are the results of a wide variety of anthropogenic demands on river systems. Pressures include the need for flood protection, water abstraction, waste disposal and navigation for recreational amenities, industry, urbanisation, agriculture and aquaculture (Cowx, 2002). Any physical pressure on the river system results in some alteration of the river flow and the physical condition of the river. Alterations to the channel frequently involve channelisation, removal of large woody debris, construction of instream structures (e.g. weirs) or general homogenisation of the channel (Harper *et al*., 1998a; Harper and Everard, 1998b; Wolfert, 2001; Ormerod, 2003; Schanze *et al*., 2004; Giller, 2005; Lepori *et al*., 2005; Schweizer *et al*., 2007b). These alterations modify the instream transport of water and sediment, adjusting the morphology and physical character of the river (Dunne & Leopold, 1978; Poff *et al*., 1997; Lucas and Marmulla, 2000; Lepori *et al*., 2005; FAO, 2008) and availability of instream habitat (Harper and Everard, 1998b; Schweizer, 2007b).

Fish have specific habitat requirements to fulfil basic daily needs, which vary according to species and life stage (Cowx *et al*., 1993; FAO, 2008). The abundance and diversity of fish species within a river reach are dependent upon the availability and diversity of physical habitat types (Cowx *et al*., 1993; FAO, 2008), with loss of physical habitat frequently attributed to reduced fish stocks (EIFAC, 1984; Hellawell, 1988; Cowx, 1994). Fish stocks may be impacted by loss of physical habitat for a particular species, or a specific life stage.

Attempts to mitigate pressures and impacts are becoming increasingly popular as pressure increases to ameliorate problems arising from the use and misuse of freshwater resources and habitats (Maddock, 1999; Walker *et al*., 2002; Giller, 2005; Mainstone and Holmes, 2010). River rehabilitation schemes frequently aim to ameliorate the impacts of flood mitigation schemes, or improve channel degradation (Walker *et al*., 2002). Conventional river rehabilitation has focused on enhancing fish production through habitat improvement structures (Gore *et al*., 1998); these strategies can potentially increase channel diversity increasing habitat availability and diversity (Bayley *et al*., 2000).

The development of a series of Directives and legislation, and changing social and environmental philosophies, have led to an increase in demand for river restoration across the European Union to mitigate negative impacts of physical modifications.

Significant developments include the Habitats Directive (HD) (‘Conservation of natural habitats and of wild fauna and flora’ 92/43/EEC) (May 1992) and the Water Framework Directive (WFD) (2000/60/EC) (December 2000). Both Directives require maintenance, enhancement or restoration of habitats as a legal requirement, recognising that physical structure and ecological functioning is the key to habitat conservation (Clarke *et al*., 2003). These legislations support implementation of measures, actions applying mechanisms to deal with a particular issue (Environment Agency, 2009a), to improve the quality and ecological status of all water bodies, encouraging rehabilitation of riverine habitats through the reinstatement of important fluvial processes (Clarke *et al*., 2003). Following acceptance of the WFD into UK policy there is a requirement for all water bodies not designated as artificial or heavily modified to achieve ‘good ecological status’ by 2027 (Harvey *et al*., 2008; Hatton-Ellis, 2008; Wolter, 2010), spurring an increase in river rehabilitation projects taking place in an attempt to improve river quality. Where rivers are designated as heavily modified, and cannot reach good ecological status without reducing capacity to meet anthropogenic demands (Hering *et al*., 2010), or where more than 75% of the length is urbanised (Environment Agency, 2011a), WFD guidelines require achievement of good ecological potential, the best possible status without compromising societal use of the water body, by 2015.

The WFD aims to set minimum ecological standards defining ecological quality based on comparison to a type-specific reference condition using various quality elements (Hatton-Ellis, 2008; Nardini *et al*., 2008). The WFD puts ecology at the base of management decisions (Hering *et al*., 2010), encompassing several biological quality elements (fish, macroinvertebrates, macrophytes and phytoplankton) to express ecological status and importantly, recognises hydromorphology as a key element of habitat quality (Harvey *et al*., 2008, Clarke *et al*., 2003, Newson, 2002, Newson and Large, 2006). This signifies a considerable change in river management, with emphasis placed on biological and physical associations and recognition that hydromorphology is a key factor in defining habitat quality (Harvey *et al*., 2008; Vaughn *et al*., 2009; Hering *et al*., 2010), acknowledging that ecology is protected through the correct management of hydrology and geomorphology (Environment Agency, 2011b).

Biological quality elements are used to place the water body in a class status, with numerous EU projects developing methods to standardise assessment (FAME: Schmutz *et al*., 2007; Wiser: Hering *et al*., 2012; Rebecca: van de Bund and Solimini, 2007). Hydromorphology supports biological elements (UK Tag, 2003) and is central to conservation as it provides a template on which all other ecological structures and functions are built (Vaughn *et al*., 2009). Although not used in classification, hydromorphological elements potentially affect the water body’s ability to achieve good ecological status (IMPRESS, 2003) with water body classification based on changes in hydromorphology with significant adverse effects (Weiss *et al*., 2008).

As fish (species composition, abundance, age structure) are an important biological indicator of ecological status, and are the biological component most sensitive to hydromorphology (Nardini *et al*., 2008), this research focuses on the response of fish to hydromorphological changes following river rehabilitation.

The link between hydromorphology and ecology has become a major issue in river research with a requirement to understand linkages to meet WFD targets (Vaughn *et al*., 2009). When planning river rehabilitation schemes, understanding the links between hydromorphology and ecology is of paramount importance owing to the provision of "physical habitat" for fish. Physical habitat emphasises the importance of understanding linkages between physical conditions and processes within the river channel and the habitat requirements of target fish species and has seen a growth in importance in river management (Newson, 2002). Detailed understanding of the complexities of fish-habitat interactions and the impacts of any instream works on in–channel physical conditions are integral to providing effective river rehabilitation. This is imperative with the increasing pressure to maintain and improve habitat under WFD and HD legislation and the increasing scope of physical instream works focused upon river rehabilitation. However, there remains a dearth of information on:

- Outcomes of river rehabilitation schemes on in-channel hydraulics
- Outcomes of river rehabilitation schemes on fish community dynamics
- Long term consequences of river rehabilitation schemes

Gaps in knowledge of the physical and biological outcomes of river rehabilitation exist due to a scarcity of follow up monitoring (Cowx, 1994; Gillilan *et al*., 2005; Wohl *et al*., 2005; FAO, 2008; Roni *et al*., 2008; Sawyer *et al*., 2009). River rehabilitation projects should aim to reinstate natural fluvial dynamics within the system (Stanford *et al*., 1996; Kemp *et al*., 1999; Lucas and Marmulla, 2000; FAO, 2008), but there is a lack of understanding of these processes and fish habitat requirements at different life stages (Swales, 1994a; Cowx and van Zyll de Jong 2004). As a consequence many rehabilitation attempts proved ineffective or failed the desired outcome (where an outcome was defined and monitoring was undertaken to determine such effects) (Cowx and van Zyll de Jong, 2004b). However, scientific research builds confidence in understanding hydromorphology and ecology (Environment Agency, 2011 c) and should be used to underpin future river rehabilitation plans; this is only possible with extensive monitoring and dissemination of results.

Most consequences of river rehabilitation projects are reported on Salmonid streams due to their manageability in terms of monitoring and reporting outcomes. Also there is most information available on the ecology and life history of Salmonid species due to their commercial and economic importance. However, there is little information available on the consequences of river rehabilitation measures on instream hydraulics, fish community structure and the interaction between hydraulic habitat and fish species. Where river rehabilitation projects are monitored, there is a tendency to monitor a specific species or hydrological element as opposed to taking a holistic approach as project objectives are frequently targeted at improving populations of a specific species.

1.1 Aims and objectives

This research aims to assess the results of river rehabilitation schemes on both instream hydraulics and fish species composition and assess the linkages between fish community composition, instream hydraulics and physical habitat. Primary objectives are:

- to review how river form and function provides physical habitat for fish
- to determine the physical outcomes of river rehabilitation in terms of instream hydraulics
- to determine the response of fish community composition to river rehabilitation
- to assess fish community structure and linkages with instream hydraulics and physical habitat

To meet the objectives, four river rehabilitation projects, with different objectives, implemented on three UK Rivers were monitored between 2010 and 2011. These were:

- Small-scale weir removal at two sites (Dovedale and Hartington) on the River Dove, Derbyshire
- Narrowing of an over-widened channel on Lowthorpe Beck, East Yorkshire
- Installation of gravels as artificial riffles on the River Stiffkey, North Norfolk

Following a review of the literature in Chapter 2, the research is divided into key topics, which are addressed in Chapters 4 to 6. Details of survey sites, data collection and analysis are presented in Chapter 3. Chapter 7 discusses findings from the investigation and suggests applications and future research themes.

Chapter two: reviews current knowledge of ecosystem functioning and the relationship between biological, specifically fish, and physical processes within the context of the EU Water Framework Directive with reference to pressures and impacts on river systems, the need for river rehabilitation programmes and the importance of monitoring such schemes.

Chapter three: provides a description of the survey and data analysis methods, details are given of the rivers surveyed, rehabilitation work carried out on each river and details of the specific sites monitored prior to and post river rehabilitation works.

Chapter four: assesses the result of physical river rehabilitation projects on hydraulic parameters within the river reach and investigates differences in hydraulic conditions in habitat patches of heterogeneous channels.

Chapter five: assesses the result of river rehabilitation schemes on fish community dynamics and assesses differences in fish communities in visually identified habitat patches in heterogeneous channels.

Chapter six: links descriptors of fish communities with environmental parameters and assesses the outputs of the Salmonid habitat model, HABSCORE, quantifying habitat availability for brown trout (*Salmo trutta* (L.)) prior to and post river rehabilitation. Habitat use of species captured is investigated and habitat preference is inferred.

Chapter seven: summarises the findings of Chapters 4 to 6 in context to the literature review provided in Chapter 2. Conclusions are drawn and areas for future study, implications for river rehabilitation projects and project management and monitoring are discussed.

This study is intended to provide information on biological (fish) and environmental (hydraulics and physical habitat) impacts of river rehabilitation schemes, unravelling some of the intricacies of fish-habitat interactions. It emphasises the importance of monitoring river rehabilitation projects and disseminating findings to aid future management. Findings will aid the design of future rehabilitation schemes as pressures increase to improve the physical and biological quality of river systems.

2 CHAPTER TWO: A REVIEW OF THE ROLE OF RIVER PROCESSES AND FISH PHYSICAL HABITAT IN RIVER REHABILITATION TO ACHIEVE E.U. WATER FRAMEWORK DIRECTIVES

Through naturally occurring hydrology and intricately intertwined flow and sediment processes, rivers provide physical habitat for fish at all life stages. In addition to a rivers ecological character, it play an important functional role to society providing services in the form of navigational pathways, means of food production, sources of water abstraction for industry, agriculture and to satisfy the demands of an ever expanding society, means of waste disposal, power generation and a recreational facility for angling, aquatic activities and aesthetic pleasure (Harper *et al*., 1998a; Harper and Everard, 1998b; Giller, 2005; Lepori *et al*., 2005; Schweizer, 2007b). It is as a result of these services to society that throughout history rivers have been severely modified through the construction of artificial dams, weirs and levees, channelisation and removal of in-channel physical features resulting in alteration of the natural flow paradigm and consequently the physical nature of the channel. It is widely documented that reduced habitat availability is a significant cause of reduced fish abundance and fish community composition (Hellawell, 1988; Cowx, 1994). This is of particular relevance to recent legislation across the European Union for the protection and improvement of riverine habitats, namely the Habitats Directive (HD) ('Conservation of natural habitats and of wild fauna and flora' 92/43/EEC), and the Water Framework Directive (2000/60/EC). The emergence of these legislative Directives has led to an increase in popularity of river rehabilitation schemes throughout Europe, particularly those involving in-channel physical restoration techniques due to recognition within the WFD that hydromorphology is a key supporting element to good ecological status and habitat quality (Clarke *et al*., 2003; Harvey *et al*., 2008). In many cases these rehabilitation schemes are not accompanied by the relevant and necessary monitoring schemes to allow evaluation of project success, thus there remains a paucity of knowledge on the consequences of channel rehabilitation for physical habitat structure and fish diversity and abundance (FAO, 2008).

The aims of this review are to explore the physical processes within river channels and their intricate association with fish life history strategies with a particular focus on salmonid species, namely brown trout (*Salmo trutta* (L.)).

- This review specifically addresses the questions:
- How do river processes create physical habitat for fish?
- What physical pressures are rivers subjected to?
- What measures are being implemented to improve in stream conditions?
- What is known about the outcomes of river rehabilitation?

2.1 River form and function

The flow of water and sediment through the river channel, coupled with local catchment conditions is the driving force of river morphology (Langbein and Leopold, 1964; Brookes, 1994; Harper and Everard, 1998b; Poff *et al.*, 1997; Stanford *et al*., 1996; Werrity, 1997), physical habitat (Bunn and Arthington, 2002) and river ecology (Poff *et al*, 1997; FAO, 2008). Channel morphology and habitat configuration are defined by river discharge and the quality, quantity and timing of bed and bank sediments entering the channel from the surrounding ecotone and upstream reaches (Petts, 1994; Zalewski *et al*., 1994; Kemp *et al*., 1999) through bank erosion and resuspension from the bed (Harper and Everard, 1998b). Along the course of a river, instream hydraulics are influenced by areas of deposition and storage (Petts, 1994) resulting in variation of depth, velocity and substrate composition (Gore, 1994; Acreman *et al*., 2005) causing a sequence of erosion and depositional features (Newson *et al*., 1998b). High-energy upstream areas are dominated by erosion with characteristic pool-riffle sequences and more sluggish, downstream areas are dominated by deposition (Welcomme, 1994). These processes are driven and maintained by the kinetic energy of running water and rely upon continuity and connectivity within the river system (Amoros and Bornette, 2002).

The natural hydrograph, expressed as mean daily discharge (Poff *et al*., 1997), demonstrates a range of flows over time (Figure 2.1) and the riverine system is dependent upon this variation. The hydrograph varies between rivers and river sections and is a result of geology and land use (Petts, 1985; Poff *et al*., 1997; Werritty, 1997; Knighton, 1998) varying over multiple timescales of hours, days, seasons, years and longer (Stanford *et al*., 1996; Poff *et al*., 1997). The whole range of discharges and flow events (Figure 2.1) are central to maintenance of the channel form (Poff *et al*., 1997; Knighton, 1998). Alterations in river flow during extreme events, such as individual floods, or changes to the sediment input (Gore, 1994) effect erosion and deposition processes and the way the river disperses energy, resulting ultimately in channel change, associated with stream power, sediment size and sediment load (Gordon *et al.*, 1994), although these changes are rarely instant and often result in a change or sequences of changes over time (Brookes, 1994). These high flows maintain channel configuration (Figure 2.1), other smaller peak flows are important for maintenance of in-channel features (e.g. pool-riffle sequence) which provide important habitat for aquatic species (Figure 2.1).

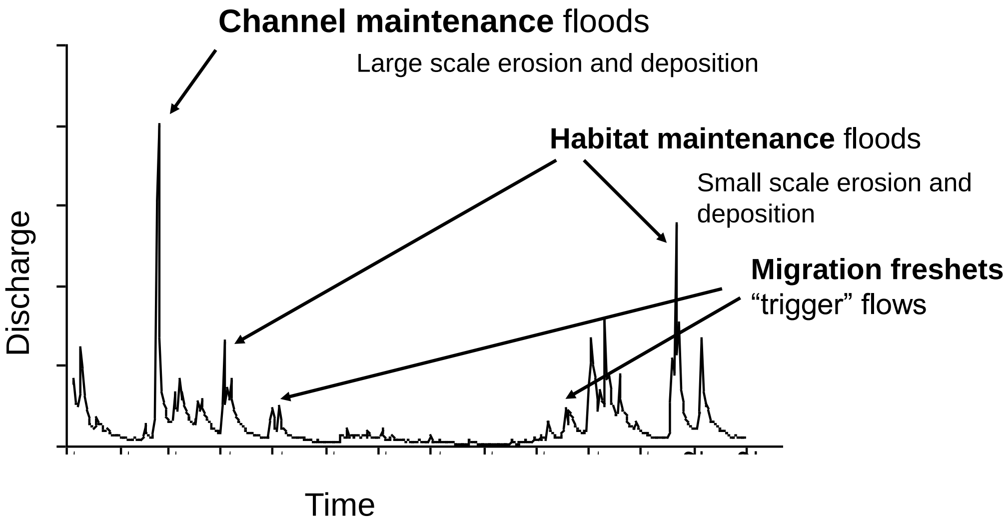


Figure 2.1: An example of a hydrograph showing natural flow variation for maintaining river channel form and function (Cowx, unpublished).

River flow processes have direct influence on fish species with specific features of the hydrograph acting as physiological cues to initiate key life events, such as migration or spawning (Bunn and Arthington, 2002; FAO, 2008). High flows in the autumn act as “triggers” to begin upstream movement in migratory species such as brown trout (Figure 2.1). Perhaps more subtly, but of equal importance, and the main focus of this review, river flow processes influence the configuration of biotic communities, particularly fish through the creation and maintenance of physical habitat (Newbury and Gabury, 1993; Cowx *et al*., 1993; FAO, 2008), which reflects the composition and variation of within channel physical and hydraulic conditions (Petts, 1994). This has a major influence over the composition and structure of the biotic community (Cowx et al., 1993; Maddock, 1999; Bunn and Arthington, 2002) with fish species showing adaptations to each habitat type (Schiemer, 2000; FAO, 2008).

Habitat is a common expression throughout the realm of ecology; with regards to freshwater fish it describes the area where a fish species can live although depending upon life phases a fish may migrate over a small local area or considerable distances (Cowx *et al*., 2004; Durance *et al*., 2006). Fish require a number of different functional habitats throughout their lifecycle (Schlosser, 1991; Cowx *et al*., 1993) to complete specific life history requirements (Figure 2.2), namely spawning, nursery areas for juveniles, feeding and protection from predators (FAO, 2008). It is frequently postulated that biodiversity increases with an increase in habitat diversity and availability (Cowx *et al*., 1993; Harper and Everard, 1998b). Each functional habitat unit must provide the appropriate habitat features for a fish species to thrive (Table 2.1), although continuity and connectivity between habitats is an essential component of availability and use (FAO, 2008).

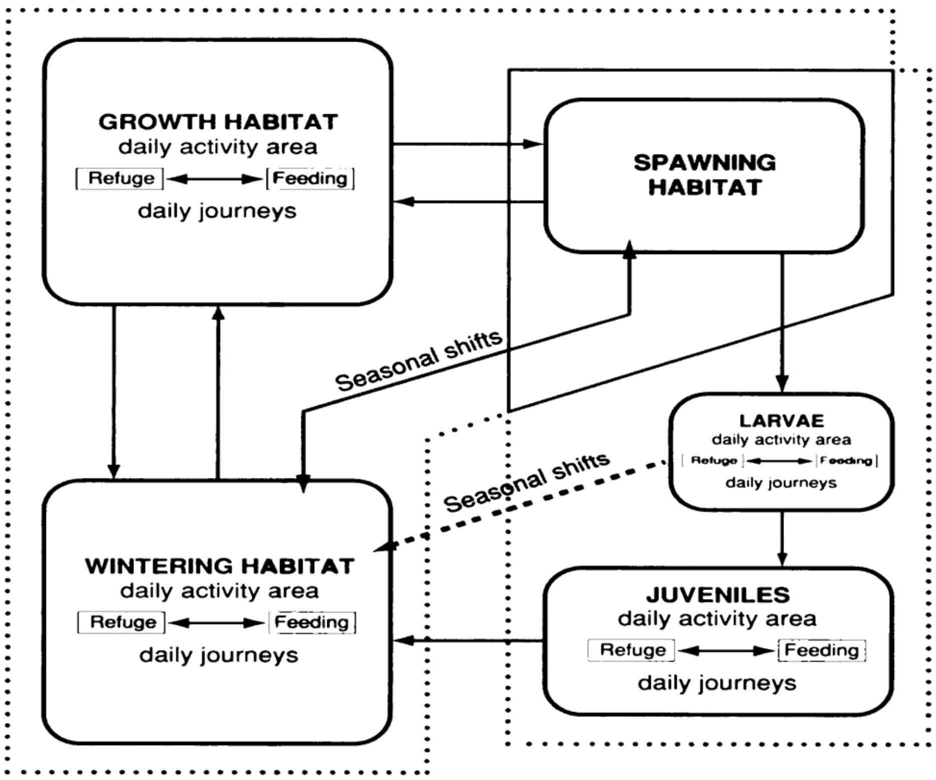


Figure 2.2: Functional Units in fish ecology (Cowx and Welcomme, 1998).

Physical habitat is accepted as a fundamental feature of riverine science (Clifford *et al.*, 2006). In-stream biodiversity is predominantly driven by abiotic factors (Stanford *et al.*, 1996), with habitat diversity and availability being the result of hydraulic forces within the channel (Cowx *et al.*, 1993; Jungwirth *et al.*, 1995; Harper and Everard, 1998b) that provide the variability of within-channel features including depth, width, substrate size and composition, and influence fish community structure (Cowx *et al.*, 1993; Poff *et al.*, 1997). The nature of the physical habitat alters along the river gradient (Huet, 1949; Cowx *et al.*, 1993) due to the influence of the flow regime and hence the physical structure of the channel. Various ecological concepts have developed to describe the alteration of physical condition along the river gradient. Huet's Zonation (1959) is used to describe the change in fish community structure with river gradient and width as fish species shift from those tolerant of faster flow conditions (trout, barbel, chub) in the headwaters to species with lower swimming abilities (bream, carp, tench) in the wider, more sluggish downstream areas (Cowx *et al.*, 1993). The River Continuum Concept states that along the physical gradient from source to mouth there is a concomitant change in the biotic community with alterations in hydraulics and geomorphology

(Vannote *et al*., 1980) and Stazner *et al*. (1988), developed the concept of 'hydraulic stream ecology' to describe the way in which species assemblages alter along the length of a river as a result of empirical changes in the stream hydraulics (Newson and Newson, 2000).

Table 2.1: Principal habitat features important to fish (Cowx and Welcomme, 1998).

Reproduction **Access to spawning areas** Provision of suitable depths and water velocity Absence of barriers to movement **Spawning** Suitable spawning substrate **Incubation of eggs** Stability of substrate Provision of adequate temperature and oxygen and water movement
Feeding and Growth **Availability of food organisms** Bankside and aquatic vegetation Substrata suitable for invertebrate production Supply of allochothounous organic material **Best use of energy for maintaining position and food gathering** Cover and shade e.g. rocks/tree trunks Diversity of flow type Pool-riffle sequences Aquatic and bank side vegetation Appropriate temperature range
Self Protection **From physical displacement by current** Shelter and visual isolation e.g. varied bed profile through: undercut banks, rocks, tree trunks, roots, accumulated debris, aquatic vegetation, weedy shallow marginal slacks (juvenile), including backwaters and lateral systems

The variation in channel structure, and thus the provision of physical habitat for fish, along the length of a river is ultimately linked to the transport of water and sediment through the channel. This highlights the importance of instream physical processes for aquatic biota, particularly for fish, which experience instream hydraulics as functional/physical habitat. Instream processes are controlled over a number of scales and hence it is important in river management a holistic approach is taken.

2.2 Scale

A holistic view of landscape ecology considers the hierarchical nature of the complete system (Newson and Newson, 2000), with rivers considered over the relevant spatial scales from the whole catchment network to the smallest spatial scale of a single substrate particle (Frissell *et al*., 1986). These features result from hydrological processes acting over a variety of spatial and temporal scales. Large-scale features are the result of infrequent, large-scale events (Lane and Richards, 1997 in Clarke *et al*., 2003). These features in turn determine the occurrence of smaller scale features such as the pool-riffle sequence, which are the result of more frequent small-scale events (Townsend and Hildrew, 1994; Biggs *et al*., 2005; Lane and Richards, 1997 in Clarke *et al*., 2003) (Figure 2.1). It is suggested that the hierarchical approach used to describe river channel morphology may also be used to explain habitat occurrence (Frissell *et al*., 1986; Harper and Everard, 1998). Fish also respond to system processes over a whole range of scales, undertaking short-term, local activities involving small-scale movements between habitats such as daily feeding to long-term, broad-scale seasonal migrations within and between catchments (Durance *et al*., 2006).

The use of prefixes such as meso- and micro- are becoming widely used to demonstrate habitat size, with a strong emphasis on physical structure (Mcoy and Bell, 1991 in Harper and Everard, 1998b). The largest habitat scale is the macro-scale, incorporating dynamics of the entire catchment. Microhabitat is the smallest scale describing the immediate area a fish is situated at any particular time (Cowx *et al*., 2004). Micro habitat availability is influenced by hydraulic features such as depth and flow velocity or substrate characteristics (Cowx and Welcomme, 1998; Petts, 1994) and their interactions at a precise time or location (Cowx *et al*., 2004, Stazner *et al.*, 1988), corresponding to an area of homogeneity of approximately 1 m in size (Frissell, 1986).

The intermediary scale is the meso-scale, which depicts instream units of relatively uniform flow and substrate types such as a pool or riffle, occurring at the interface of hydrological and geomorphological forces (Tickner *et al*., 2000; Wood *et al*., 1999). The meso-scale approach to habitat identification is perhaps the most useful due to its potential linkage between micro and macro habitat characteristics (Kershner and Snider, 1992 in Clarke *et al*., 2003). The meso-scale has increased in recognition following recent developments, including WFD requirements to define river 'quality' through a series of factors, defining a novel science, termed "ecohydraulics", linking hydraulic, hydrological and morphological processes with ecological aspects of riverine science, (Kemp *et al*., 2000; Newson, 2002; Harvey and Clifford, 2009; Lancaster and Downes, 2010; Rice *et al*., 2010).

The ecohydraulics of a channel are strongly influenced and organised by instream physical conditions (Newson, 2000). Habitat patches are not necessarily permanent features and may change in space and time over short to long time-scales (Death and Winterbourn, 1994 and Armitage and Pardo, 1995 in Pardo and Armitage, 1997; Brookes, 1994) and with discharge level (Hauer *et al*., 2009), as a result of naturally occurring processes over various temporal and spatial scales (Clarke *et al*., 2003).

Functional habitat is an expression commonly used in freshwater ecology to define a region of instream habitat characterised qualitatively by homogenous distribution of an abiotic physical factor (Harper *et al*, 1992 and 1998; Armitage and Pardo, 1995; Pardo and Armitage, 1997; Kemp *et al*., 2000), such as substrate composition (Harper *et al*., 1992 and 1998). Similarly, a popular concept is the 'physical biotope' (Harvey and Clifford, 2009), which is used to describe constituent parts of the channel bed with discrete hydraulic conditions (Newson and Newson, 2000). Physical biotopes depict the abiotic environment (Clifford *et al*., 2006) through the integration and simplification of interactions of hydraulic microhabitat variables, commonly depth, substrate and flow velocity (Padmore, 1997; Newson and Newson, 2000; Wadeson and Rowntree in Clifford *et al*., 2006); facilitating qualitative identification of meso-scale hydraulic conditions through dominant flow types (Newson and Newson, 2000). The natural variation of rivers from source to mouth leads to diversity of flow biotopes (Kemp *et al*., 1999) across both lateral and longitudinal profiles (Padmore, 1997), which are controlled by local geomorphology and hydraulics (Hill *et al*., 2008). Changes in geomorphology result in changes in physical habitat at a scale that fish experience (Wheaton *et al*., 2010). This suggests these habitat patches are a basic component of instream habitat allowing assessment of habitat availability and diversity (Padmore, 1998). Although the physical biotope and functional habitat approaches are rooted in different research origins, there is great potential for amalgamation of the two (Newson *et al*., 1998b), which appear mutually complementary (Newson *et al*., 1998b; Kemp *et al*., 1999, 2000; Harper *et al*., 2000; Harvey and Clifford, 2009) (Figure 2.3) providing a practical foundation for assessment of meso-scale habitat within rivers (Harvey and Clifford 2009). This approach suggests the possibility of exploring the temporal and spatial variability of instream physical habitat (Newson and Newson, 2000) offering a way of assisting amalgamation of hydraulic, geomorphological and biological expertise (Clifford *et al*., 2006). Thomson *et al.* (2001) approached meso-scale units by relating riverbed morphology and channel hydraulics by means of the Hydromorphic/Hydraulic Unit (HMU). These HMUs are areas identified by relatively uniform substrate and flow type (Thomson *et al*., 2001), as with flow biotopes. HMU size and location frequently correspond with that of the mesohabitat, particularly of adult fish (Parasiewicz, 2007a), suggesting synonymy between these approaches (Parasiewicz, 2001). A novel terminology, now well accepted in ecohydraulics describes these significant flow characteristics (Wadeson, 1994) (Table 2.2).

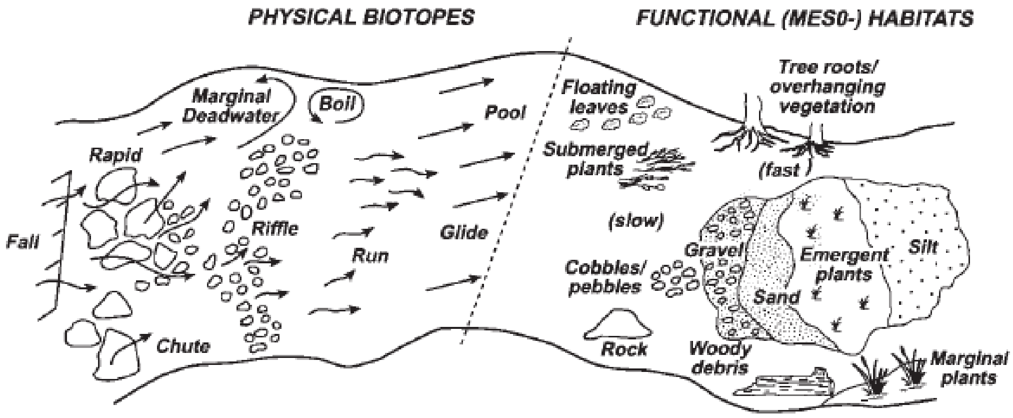


Figure 2.3: Physical biotopes and functional (meso) habitats. (From Newson and Newson, 2000).

Units of instream habitat and flow biotopes are determined through visual identification of the surface flow and characterised through measurement of hydraulic variables, most successfully Froude number (Padmore, 1997; Kemp *et al*., 2000), a dimensionless number describing the relationship between flow velocity and depth, leading to a union between hydraulic and physical biotopes (Newson *et al*., 1998b). This emphasises the practicality of biotopes as a standard instream component despite lack of biological justification (Newson and Newson, 2000), meaning, despite the recognition of a considerable association between physical habitat and biotic response, methods of geomorphological characterisation remain undeveloped (Orr *et al*., 2008). However, Schwartz and Herricks (2008) showed that fish community structure could be distinguished between mesohabitat types by feeding guild during the summer low-flow period.

Table 2.2: Definitions of hydromorphological units corresponding to mesohabitats described by dominant flow type (Parasiewicz, 2007a and Newson and Newson, 2000)

Biotope/HMU	Description	Flow type
Waterfall	Water falls vertically and without obstruction from a distinct feature, generally more than 1m high and often across the full channel width	Free fall
Spill-chute over areas of exposed bedrock.	Fast, smooth boundary turbulent flow over boulders or bedrock. Flow is in contact with the substrate, and exhibits upstream convergence and downstream divergence.	Chute
Cascade- chute flow over individual boulders	Stepped rapids with small waterfalls and very small pools behind boulders	
Cascade- at the downstream side of the boulder flow diverges or "breaks"	White-water tumbling waves with the crest facing in an upstream direction. Associated with "surging" flow.	Broken standing waves
Rapid	Higher gradient reaches with faster current velocity, coarser substrate, and more surface turbulence. Convex streambed shape.	
Riffle	Undular standing waves in which the crest faces upstream without breaking. Shallow stream reaches with moderate current velocity, some surface turbulence, and higher gradient. Convex streambed shape.	Unbroken standing waves
Run	Surface turbulence does not produce waves, but symmetrical ripples which move in a general downstream direction. Monotone stream channels with well-determined thalweg. Streambed is longitudinally flat and laterally concave.	Rippled
Boil	Secondary flow cells visible at the water surface by vertical "boils" or circular horizontal eddies	Upwelling
Glide	Flow in which relative roughness is sufficiently low that very little surface turbulence occurs. Very small turbulent flow cells are visible, reflections are distorted and surface "foam" moves in a downstream direction. A stick placed vertically into the flow creates an upstream facing "v". Moderately shallow channels with laminar flow, lacking pronounced turbulence. Flat streambed shape.	Smooth boundary turbulent
Pool-occupies the full channel width. Marginal dead water- does not occupy full channel width	Surface foam appears to be stationary and reflections are not distorted. A stick placed on the waters surface will remain still. Deep water impounded by a channel or partial channel obstruction. Slow. Convex streambed shape.	Scarcely perceptible flow
Backwater	Slack areas behind channel margins caused by eddies behind obstructions.	
Side arm	Channels around islands, smaller than half river width, frequently at different elevations to main channel.	

The survey of instream mesohabitat units has potential to complement habitat quality assessments such as River Habitat Survey (RHS) by facilitating assessment of biodiversity (Tickner *et al*., 2000). RHS is the standard riverine hydromorphology survey in the UK (Vaughan, 2010a); it relies on the measure of occurrence of instream morphological features to assess habitat status, comparing features present with those expected at an unmodified site (Raven *et al*., 1998; Raven *et al*., 2000; Walker et al., 2002). By gauging the presence of mesohabitats in terms of their biotic assemblage, this approach has potential to inform management of river reaches to maximise biodiversity (Tickner *et al*., 2000).

The WFD requires hydromorphological assessment of water bodies to enable better understanding of biological and chemical river quality data; recognising the importance of hydromorphological elements, along with chemical elements, in supporting biological quality to define ecological status. Newson and Newson (2000) suggest that habitat scale should be a focal point in the investigation and progression of river management tools. Meso-scale habitat assessment holds potential to provide a fast but efficient means of assessing habitat, and collating information from microscale studies to macroscale investigations (Kemp *et al*., 2000), although currently information at this scale is most deficient (Kemp *et al*, 2000). The mesohabitat/biotope approach to habitat assessment has three principal advantages; it is practical, allowing integration into existing assessment methods, e.g. RHS, it simplifies hydraulic conditions into "real world" units and facilitates union between multiple disciplines (Clifford *et al*., 2006).

The meso-scale provides an intermediary between the ranges of scales over which riverine processes occur and is practical in describing instream features such as the pool-riffle sequence. This approach is also useful in considering physical habitat for fish in terms of the functional units used by different species and life stages.

2.3 Fish species requirements

A range of factors influence the way in which freshwater fish select and use habitats, many of these factors are inter-linked and are related to the physical processes and hydraulics within the river channel (Figure 2.4). The presence or absence of a particular species at a site depends upon the life-history strategy of that species and the ability to survive in localised environmental conditions (Noble *et al*., 2007; Schmutz *et al*., 2007; Cowx *et al*., 2009; Durance *et al*., 2006; Kemp *et al*., 2011; Blanck *et al*., 2007; Barbour, 1991). Due to their economic importance and extensive distribution, many studies of fish-habitat relationships have focused on salmonid species (Kemp *et al*., 2011; Gosselin *et al*., 2012).

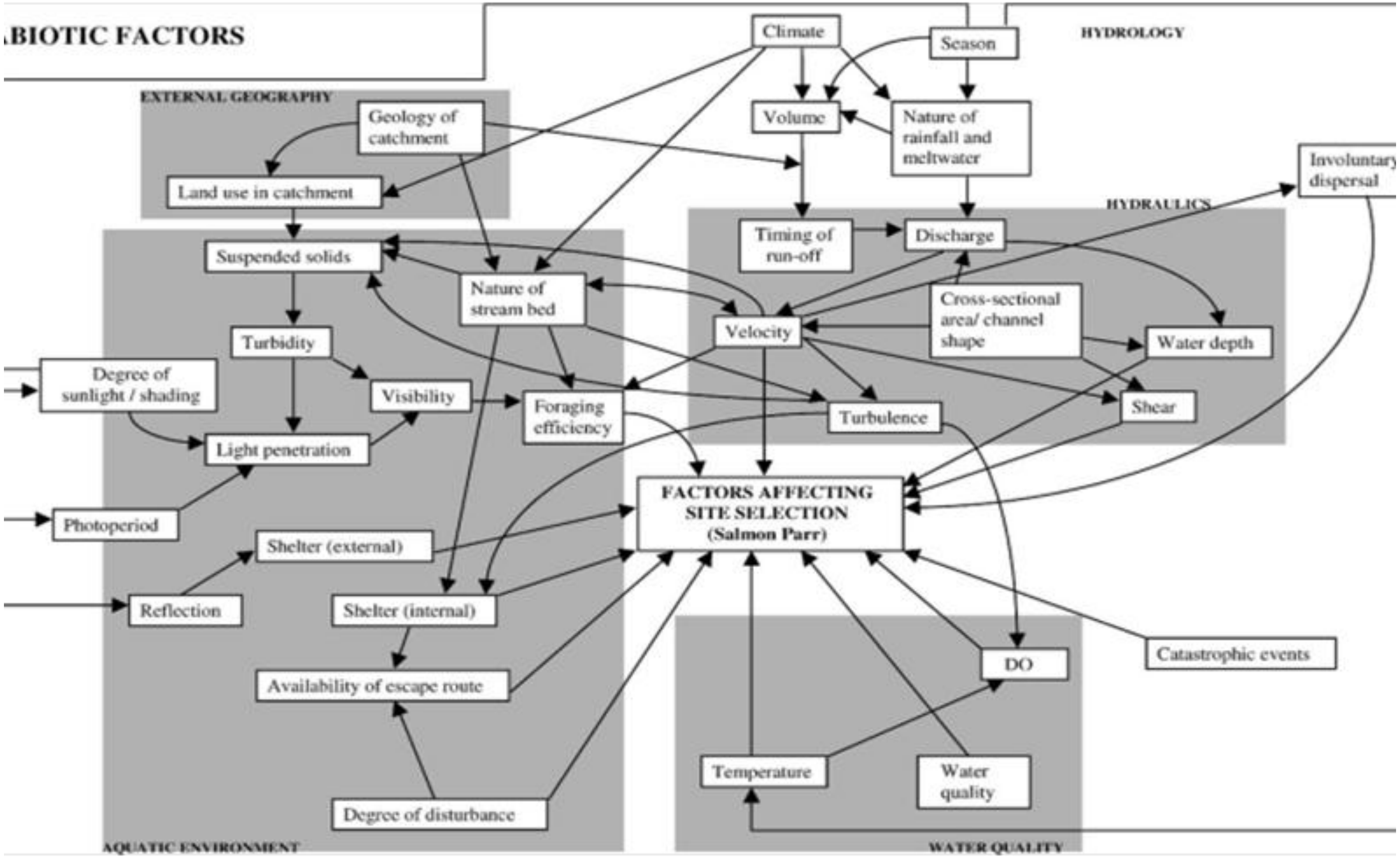


Figure 2.4: Complexity of abiotic interactions affecting salmon and trout parr (from Armstrong 2003)

2.3.1 Brown trout

Atlantic salmon (*Salmo salar* (L.)) and brown trout (*Salmo trutta* (L.)) are two of the most common and widespread salmonid species within the UK. The focus of this review will be on brown trout, which displays both migratory, anadromous strains, in which parr become smolts and migrate to sea before returning to natal streams as "sea trout"; and non-migratory, resident strains, which remain in fresh water throughout the lifecycle; within the genetic population both strains are often referred to only as "brown trout" (Crisp, 1996; Cowx *et al*., 2004).

Briefly, the lifecycle of brown trout (Figure 2.5) involves deposition of eggs within redds in flowing water with a gravel bed substrate. The eggs hatch into alevins, remaining in gravel until the yolk sack is almost exhausted when they leave the gravel and swallow an air bubble to accomplish neutral buoyancy, becoming fry capable of taking food externally. The fry then disperse to become parr and take up and defend territories (Crisp, 1996).

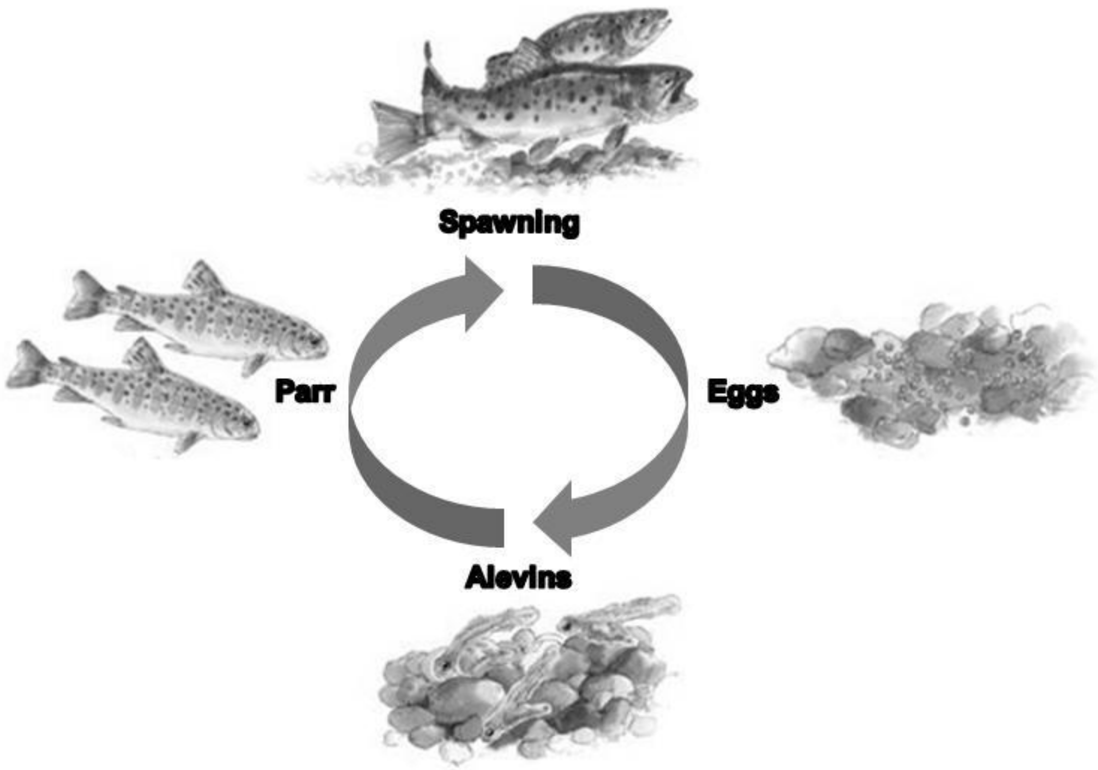


Figure 2.5: Diagram showing the lifecycle of Salmo. Trutta (Brown Trout) (after: http://www.wildtrout.org/content/trout-lifecycle on 18/07/2013).

The brown trout life cycle involves migrations between several habitats related to instream structures (Jungwirth *et al*., 2000). Solomon & Templeton (1976) (in Crisp, 1996) proposed that trout generally show five phases within their lifespan:

- Migration from hatching area to downstream nursery (0 to 6 months)
- Downstream migrations from nursery areas to adult “growth” habitats (6 to 15 months)
- Restricted movements of adults (15 months to spawning)
- Migration to upstream spawning habitat
- Post-spawning migration downstream

Throughout the lifecycle, brown trout use various habitats; successful use of these habitats depends upon appropriate environmental conditions being met. Abiotic factors such as temperature, water flow velocity, fluctuations in discharge and availability of cover (Figure 2.4) (Binns and Eiserman, 1979 in Armstrong, 2003) influence habitat availability and use, although it appears that depth, current, substrate and cover (Heggenes, 1990 in Armstrong *et al*., 2003) are most influential upon the abundance and distribution of brown trout (Armstrong *et al*., 2003).

It appears habitat selection alters with body size (Greenberg *et al*., 1996) and there is a correlation between depth and size with larger fish being found in deeper pool habitats and smaller fish in shallower habitats (Greenberg *et al*., 1996; Heggenes, 1996; Gosselin *et al*., 2012; Bohlin, 1977; Kennedy and Strange, 1982; Crisp, 1996 and Maki-Petays *et al*., 1997 in Armstrong *et al*., 2003). Use of water velocity also appears to be related to body size (Heggenes, 1996). In a study of mesohabitat use by brown trout, Gosselin *et al* (2012) found that most individuals were found in glides and runs, with adults more associated with runs than parr.

Spawning and inter-gravel stages

Brown trout show migratory behaviour when approaching readiness to spawn, generally migrating upstream to their natal river to spawn in their natal tributary. In Europe this is usually in autumn or early winter. Sea trout return from the sea to migrate upstream (Figure 2.5), whilst resident trout also migrate upstream (Crisp, 1996; Cowx *et al.*, 2004). This migration can only occur where migratory access is available and not restricted by instream obstacles such as weirs and dams (Cowx *et al*., 2004).

Survival of offspring depends upon site selection resulting in successful spawning, egg survival and hatching, site selection is influenced by a range of environmental cues, including gravel size, intra-gravel flow, stream velocity, cover and depth (Crisp, 2000 in Cowx *et al*., 2004). Redds are normally constructed in gravel substrates with high intragravel flows and low silt concentration (Hobbs, 1937; White, 1942; Stuart, 1953a in

Crisp, 1996), with eggs buried to depths of 5 to 25 cm (Crisp & Carling, 1989 in Crisp, 1996). It is suggested by McCart (1969 in Crisp, 1996) that a minimum water velocity is required at the spawning site to allow movement of bed material to be initiated allowing redd construction. Although brown trout show plasticity in spawning and nursery habitat requirements (Kemp *et al*., 2011), spawning sites are selected in areas of shallow flow approximately 15-45 cm in depth and 20-55 cm s^{-1} flow velocity (Louhi *et al*., 2008), although a review by Cowx *et al* (2004) found brown trout spawn in the depth range 6-91 cm and velocity range 10.8-81 cm s^{-1}. Suitable spawning grounds may differ between different rivers and catchments (Louhi *et al*., 2008) and redds are often constructed where there is a change in hydraulic head, typically where water is accelerating over gravel such as at the head of a riffle (Crisp, 1996).

Crisp (1996) suggested that spawning and incubation gravels ideally should contain less than 15% fines (particles of <1.0mm or <2.0mm diameter). The structure of the riverbed consists of an open structured large particle framework with a smaller particle matrix filling the spaces (Crisp, 2000 in Cowx *et al*., 2004). The build up of fines within the gravel can influence incubation success and emergence (Crisp, 1996; Kemp *et al*., 2011).

Brown trout generally select gravels with a mean diameter of 20-30 mm but can spawn effectively in gravels with a median grain size of 60 mm diameter (Crisp and Carling, 1989 in Crisp, 1996); Louhi (2008) found brown trout spawn in areas of pebble substrate between 16-64 mm. The availability of suitable spawning habitat can be a limiting factor for salmonid population productivity (Kondolf and Wolman, 1993 in Armstrong *et al*., 2003). A general guide to spawning site composition was suggested by Fluskey (1989) to be 10% cobbles (64 to 190 mm), 35% very coarse gravel (32 to 64 mm), 25% coarse gravel (16 to 32 mm), 20% medium gravel (8 to 16 mm and 10% fine gravel (4 to 8 mm) for river rehabilitation purposes (Crisp, 1996).

Cover also appears to be important in spawning site selection and Crisp (1996) suggested that salmonid species may require areas of deep, slow flowing water whilst waiting to spawn and between spawning events and as protection from predators. Witzel and MacCrimmon (1983) (in Armstrong *et al*., 2003) found that 84% of redds were recorded within 1.5 m of cover (logs/branches).

Fry and Parr requirements

Typically, 0+ brown trout remain in the location of hatching until the autumn when they undertake a primary dispersal period (Egglishaw and Shackley, 1977 in Cowx *et al*.,

2004) and relocate to deeper water with larger stones (Rimmer *et al*., 1984 in Cowx, 2004) to find winter habitats (Cowx *et al.*, 2004). This is a critical period and is the time at which the cohort size may be established (Cowx *et al.*, 2004). Brown trout require a great degree of in-channel physical habitat diversity, typically a pool-riffle habitat providing diversity in flow with a substrate combination of cobbles, boulders and coarse gravel (Cowx *et al.*, 2004).

Trout fry display preference for water velocities of 0-20 cm s^{-1} and use greater velocities as they grow (Bardonnet and Heland, 1994 in Armstrong *et al.*, 2003). The optimal depth for trout fry throughout all seasons appears to be 5-35 cm (Armstrong *et al.*, 2003) with trout less than 7 cm occupying areas of below 20-30 cm depth (Bohlin, 1977; Kennedy and Strange, 1982; Bardonnet and Heland, 1994 in Armstrong *et al.*, 2003), although Cowx *et al.* (2004) showed fry are present at a range of depths below 60 cm.

Trout parr show preference for cobble and boulder substrate (Crisp, 1996) and have been found in shallow riffles of velocity 20-50 cm s^{-1} (Heggenes, 1996; Armstrong *et al.*, 2003) trout parr have also been observed in areas of water over 27 cm in depth with velocities <28 cm s^{-1} (Crisp, 1996). In their review, Cowx *et al* (2004) found trout parr are capable of utilising depths between <5.1-300 cm and velocities of 0-65 cm s^{-1}.

It appears well accepted within the literature that habitat preferences change as size increases, within and between populations and at different seasons of the year (Cowx *et al.*, 2004).

Adults and spawning movements

Older, resident trout have similar habitat requirements to younger trout although the effects of age and size should be accounted for (Cowx *et al*., 2004). Trout tend to occur at sites with substrate of 8-128 mm diameter (Eklov *et al*., 1999 in Armstrong *et al*., 2003), although adult brown trout have been reported to select coarser substrate with a diameter over 128 mm whilst actively avoiding bedrock areas (Heggenes, 1996; Armstrong *et al*., 2003). Hegennes (1996) and Gosselin (2012) reported the use of habitat with fine grain substrate, although they are usually associated with coarser substrate as size increases. However, Heggenes (1996) reported this is of less influence than depth and water velocities during summer habitat use. Adult trout are associated with deeper, faster flowing glides, deep pools and undercut banks (Heggenes, 1996; Cowx *et al*., 2004), although adult trout gather at shallow, fast flowing gravel beds during the spawning season (Gosselin, 2012). Adult brown trout are reported to utilise habitat of depth between 9-305 cm and 0-142cm s^{-1} flow velocity (Cowx *et al*., 2004), whereas, when investigating mesohabitat use, Gosselin *et al.* (2012) found a strong association between run habitat and adult trout, whereas parr were associated more with glide habitat and preferred depths of 30-50 cm and velocities below 0.40 ms^{-1}.

Cover (boulders, overhung banks and deep pools) also has a substantial influence over adult trout habitat (Binns and Eiserman, 1979 in Armstrong *et al*., 2003). This may be due to protection from predation provided by areas of cover (Armstrong *et al*., 2003) and protection from direct sunlight (Crisp, 1996). Areas of cover are particularly important as rest areas during upstream migration to spawning sites (Crisp, 1996).

2.3.2 Community composition

The structural diversity of rivers is reflected in the fish communities present, and these communities are an important indicator of ecosystem integrity over a range of scales with the structure of the fish population providing information of overall conditions (Scheimer, 2000). This is of particular importance due to the prominence of fish as an indicator of biological quality under WFD requirements (Schmutz *et al*., 2007; Noble *et al*., 2007a). A range of methods has been developed to measure the status of the fish population including several indices of biotic integrity (IBIs), which are commonly based around ecological guilds (Schmutz *et al*., 2007; Noble *et al*., 2007a; Noble *et al*. 2007b). Noble *et al.* (2007b) suggested a spatially based IBI for UK Rivers and found that despite limitations in data, spatially based multi-metric indices are a feasible option for ecological assessment. Their analysis revealed that in salmonid dominated river reaches where brown trout prevail, other species are often present in synchrony, in particular *Cottus gobio* (L.) (bullhead), *Barbatula barbatula* (L.) (stone loach), *Anguilla anguilla* (L.) (eel), and all lamprey species (*Lampetra fluviatalis* (L.) (brook lamprey), *Lampetra planeri* (Bloch.) (river lamprey) and *Petromyzon marinus* (L.) (sea lamprey). It is therefore beneficial when considering habitat requirements of major species to also consider habitat requirements of other species within the community (Table 2.3), many of which require protection under the EC Habitats Directive (92/43/EEC).

2.3.3 Eel

The European eel (*Anguilla anguilla* (L.)) has widespread distribution living in many different habitat types (Laffaile *et al*., 2003). Eel populations are declining globally (Dekker, 2003; Davies *et al*., 2004; van Ginneken and Maes, 2005; Castoguay *et al*., 2007). Eels are protected under the EC Habitats Directive thus there is a requirement to protect, conserve and enhance the aquatic environment where eels spend part of their life cycle. Habitat restoration could be one option to enhance stocks providing it is based on a thorough understanding of eel-habitat relationships (Laffaille *et al*., 2003).

Table 2.3: Habitat requirements of brown trout and synchronous species (modified from Cowx, 2004)

Species	**Lifestage**	**Depth (cm)**	**Flow (cm/sec)**
Salmo trutta	Fry	<60	0-<30
	0+	<20-30	<10-50
	Juvenile	5-240	0-44
	Parr	<5.1-300	0-65
	Adult	9-305	0-142
	Spawning	6-91	10.8-81
Anguilla anguilla	Juvenile	<600	>10
Barbatula barbatula	Juvenile	0-20	Still, elevated
Cottus gobio	Juvenile	Shallow	Elevated
	Adult	>5-40	>40
	Spawning	>5	
Lampetra fluviatilis	Larvae	0-100	1-50
	Spawning	20-150	100-200
Lamptetra planeri	Larvae	<50	8-10
	Spawning	3-150	30-50
Petromyzon marinus	Larvae	0-220	0-17
	Spawning	13-170	30-200

The European eel lifecycle is long and comprises five key stages (Bertin, 1956 in Arai *et al.*, 2006). European eel spawns in the Sargasso Sea, the leptocephalus larvae drift on the Gulf Stream and North Atlantic current across the Atlantic Ocean (Arai, 2006). The leptocephali metamorphose into glass eels then enter fresh water as elvers to migrate upstream (Ibbotson *et al*., 2002; van Ginneken and Maes, 2005; Arai *et al*., 2006). Eels are well-distributed although access to river systems can be blocked by man-made, natural or water quality barriers (Davies *et al*., 2004).

They spend the rest of their lives, in the yellow stage, in rivers and estuaries (Ibbotson *et al*., 2002), until maturation into the silver stage. This can be up to 20 years, when they move downstream to the sea to migrate and spawn in the Sargasso Sea (Ibbotson *et al*., 2002; van Ginneken and Maes, 2005; Arai *et a*l., 2006). It has been suggested that male silver eels leave the European coast as early as August whereas female silver eels depart in September-October (Usui, 1991 in van Ginneken and Maes, 2005). Starkie (2003) stated that until recently it was accepted that European eel derived from a single, randomly-mating population; and if this is true, pressures on the population in one country can impact upon recruitment elsewhere.

Eels can be found in almost all freshwaters. In fresh water, they appear to prefer lowland lakes and rivers where there is plenty of cover, the water is still or slow-moving, and the bottom is muddy although they can be found in upland rivers and mountain streams. They can tolerate a wide range of temperature, pH and dissolved oxygen (Davies *et al*., 2004).

2.3.4 Stone loach

Stone loach (*Barbatulla barbatulla* (L.)) is a small benthic fish found in lentic and lotic habitats throughout Europe (MacKenzie and Greenberg, 1998). It can tolerate a wide range of environmental conditions (Smyly, 1955). Surface velocity appears to show little effect on stone loach due to its benthic nature (Smyly, 1955; Facey and Grossman, 1992 in MacKenzie and Greenberg, 1998), although Smyly (1955) observed it to be most abundant in areas of low flow. Copp (2004) stated stone loach generally prefers elevated water velocities. Smyly (1955) also stated little influence of channel substratum on stone loach distribution and found the species in areas of sand, gravel and mud, although it shows preference for gravel substrate (Prenda *et al*., 1997 in Copp and Vilizzi, 2004) with larger substrate used with increased development (Zweimuller, 1995 in Copp and Vilizzi, 2004). Cobbles also appear to be of elevated importance at some life stages and habitats such as riffles (Copp and Vilizzi, 2004). In field studies, stone loach of small size (<3cm) were found in areas of shallow depth and low flow velocity whereas larger fish were found in deeper areas of higher flow velocity (Zweimuller 1995 in MacKenzie and Grossman, 1998). The species is found in elevated numbers in streams with artificial riffles as river rehabilitation measures opposed to control stretches (Copp and Vilizzi, 2004), suggesting habitat preference varies within and between streams, between seasons and the diel cycle (Copp and Vilizzi, 2004), and with life stage.

2.3.5 Bullhead

Bullhead (*Cottus gobio* (L.)) is included within the EC Habitats Directive (Carter *et al*., 2004) and is sensitive to habitat modification resulting in changes to the natural sediment and flow regime (Tomlinson and Perrow, 2003). Bullhead prefers lotic conditions, inhabiting rivers, streams and brooks (Carter *et al*., 2004) and tends to be more prolific in hard-water lowland streams than upland streams (Tomlinson and Perrow, 2003). Bullheads use a range of habitat depending on life stage; large, coarse substratum is essential for completing the lifecycle, particularly for breeding (Tomlinson and Perrow, 2003; Gosselin *et al*, 2010). During spawning, females lay eggs, which adhere to the underside of a stone, hatching after 20-30 days (Tomlinson and Perrow, 2003). Larvae have a yolk sack attached which is absorbed after approximately 10 days when larvae disperse (Maitland and Campbell, 1992 in Tomlinson and Perrow, 2003).

0+ fish prefer shallow, gravel riffles (Gubbels 1997; Prenda *et al*. 1997; Perrow *et al*. 1997; Punchard *et al*. 2000 in Tomlinson and Perrow, 2003), and Carter (2004) found 0+ bullhead preferred coarse substrate (gravel and cobbles) and generally shallower depths, but water velocity preference varied from high to very high according to season (Carter *et al*., 2004). Adult bullhead prefer areas of shelter provided by large woody debris, tree roots and large stones (Perrow *et al*., 1997 in Tomlinson and Perrow, 2003), although all age classes require slack water areas in high flows (Tomlinson and Perrow, 2003; Gosselin *et al*., 2010). Water depth appears not to be critical to bullhead as long as depths are greater than 5 cm with sufficient flow velocity (Tomlinson and Perrow, 2003). Carter (2004) states bullhead distribution is generally in shallow (10-20cm) water areas of elevated flow velocity (>10cm/s) with large grain substrate (gravel and cobble). Gosselin *et al*., 2010, confirmed these findings stating depth preferences vary between 0.10-0.30 m with velocities between 0 and 0.20 m s^{-1} although habitat use tends to differ between sites and studies. Cobble is considered as the key predictor of bullhead abundance (Knaepkins *et al*, 2002 in Gosselin *et al*, 2010).

2.3.6 Lamprey

Lamprey are Agnatha, a family of jawless fish, of which there are three species in the UK: river lamprey (*Lampetra fluviatalis* (L.), brook lamprey (*Lampetra planeri* (Bloch.)) and sea lamprey (*Petromyzon marinus*) (Gilvear *et al*., 2008). There is difficulty in distinguishing between river lamprey and brook lamprey (Gardiner, 2003 in Nunn *et al*., 2008) and both species are considered to be very closely related, or the same species (Maitland, 2003). Sea lamprey is the largest British lamprey species (Maitland, 2003).

Most species of lamprey display similar life cycles, with adults migrating upstream to spawning areas at sexual maturity (Gilvear *et al*., 2008), although brook lamprey are resident. Lamprey spawning areas are usually stony or gravelly patches in running water, usually as a pool breaks into a riffle, were eggs are laid in small nests (Jang and Lucas, 2005; Maitland, 2003; Gilvear *et al*., 2008; Lucas *et al*., 2009). Typical water depths at spawning sites are 0.2-1.5 m with flow velocity 0.5-1.5 m s^{-1} (Lucas *et al*., 2009).

After hatching, lamprey ammocoetes (larvae) (Nunn *et al*., 2008) move to burrow in areas of fine substrate with slow water flow (Maitland, 2003; Gilvear *et al*, 2008; Lucas *et al*., 2009), often at the edges of rivers away from the main current (Maitland, 2003). Water depths between 0.1-0.5 m, with flow velocities between 8-10 cm s^{-1} are suggested to be optimal (Maitland, 2003; Gilvear *et al*, 2008). Ammocoetes develop within silt beds for several years before metamorphosing into adults (Maitland, 2003; Gilvear *et al*., 2008). River and sea lampreys are anadromous, although some resident populations exist (Maitland *et al*., 1994; Renaud, 1997 in Jang and Lucas, 2005), migrating to the sea after metamorphosis (Maitland, 2003; Gilvear *et al.*, 2008).

Lamprey species are a protected species under the EC Habitats Directive due to their sensitivity to habitat modification (Jang and Lucas, 2005; Gilvear *et al*., 2008; Nunn *et al*., 2008; Lucas *et al*., 2009) therefore protection of habitat for all life stages, areas of silt and low flow velocity for ammocoetes and gravel patches for spawning adults, is of paramount importance.

2.4 Habitat assessment

The condition of biological communities is heavily influenced by the availability and composition of physical habitat (Barbour, 1991). In terms of assessing availability of physical habitat, it is defined as a discrete area within the channel with homogenous flow, depth and velocity characteristics. When assessing habitat availability and use, it is important to understand how these habitats change as a result of anthropogenic influence, how they are used by various species and how occurrence may be sampled (Rabeni, 2000). In the USA habitat quality is assessed in terms of ecological integrity (Harper *et al*., 2000), and throughout Europe methods are developing to assess ecological integrity based upon biological quality elements in accordance with WFD guidelines (FAME: Schmutz *et al*., 2007; Wiser: Hering *et al*., 2012; Rebecca: van de Bund and Solimini, 2007). There is general agreement that ecological integrity is concerned with the importance of physical habitat (flow velocity, depth, substrate) to the normal functioning of the aquatic ecosystem, whilst other factors (water quality and quantity) are affected by these physical conditions within, along and in the watershed of a stream (Bradley *et al*., 2012). It is therefore important to understand the importance of relationships between physical habitat conditions and instream biota and that this occurs over various spatial scales (Rabeni, 2000). Many major assessment protocols measure some of these influential factors often in the locale of the sampling site, however, due to the current lack of understanding of the ecological role of these physical variables approaches and evaluation shows extensive variation (Rabeni, 2000).

Habitat management is often undertaken on the site scale, this is also the scale at which functional relationships between habitat and fish can be established, as fish, and particularly salmonids can be sampled with some degree of accuracy in the physical habitat features where they are found (Milner *et al*., 1998). Theoretically, the significance of habitat availability in the abundance of salmonids suggests the possibility of deriving predictive relationships between habitat features and abundance (Milner *et al*., 1998). This is a useful management tool for predicting habitat availability and use and impacts of management actions (Barbour, 1991). There have been a number of attempts to produce methods of defining and quantifying salmonid habitat value including the Instream Flow Incremental Methodology (IFIM) (Bovee, 1982), and one of the most successful modelling systems for UK salmonids is HABSCORE (Armstrong *et al*., 2003), which is often employed to assess fisheries quality through habitat assessment emphasising the importance of physical channel structure on quality (Milner *et al*., 1998; Harper, 2000).

2.4.1 HABSCORE

HABSCORE is one of a series of fish-habitat models and is a system for measuring and evaluating in-stream salmonid habitat features; it is based on a series of empirical statistical models (which are both species and age specific) relating salmonid populations to observed habitat variables at a site and catchment scale (Barnard, 1999; Milner *et al*., 1998). It is based upon five salmonid species/age combinations (0+ salmon, >0+ salmon, 0+ trout, >0+ trout <20 cm, > 0+ trout >20 cm (Milner *et al*., 1998). HABSCORE allows the user to integrate data from more than one sampling occasion and allows the population estimate to be statistically reliable; this is advantageous due to the notorious variability of fish population data (Milner *et al*., 1998).

The HABSCORE model was developed through quantitative electric fishing surveys at 602 sites between 30-100 m in length in England and Wales between 1993 and 1994, to obtain estimates of maximum likelihood fish population estimates (Carle and Strub, 1978: Milner *et al*., 1998). All sites were steep channelled upland regions typical of salmonid habitat, measured less than 15 m in width and as far as was possible determined to be natural and not suffering environmental impacts (Milner *et al*., 1998). Stepwise multiple regression analysis was used to model population size against habitat variables, including the incidence of instream features (e.g. substrate-flow interactions) (Milner *et al*., 1998).

Data collection relies on completion of a series of forms relating to the catchment, site habitat (Table 2.4) and fish data for the site. Habitat assessment for HABSCORE is completed during normal low summer flows; droughts and floods should be avoided (Barnard, 1999). The collection of habitat data relies on three data sets:

- catchment variables mainly derived from 1:50,000 Ordnance Survey maps (Milner *et al*., 1998);
- a general site description;
- detailed instream habitat description for successive 10-m sections (Barnard, 1999) measuring a range of variables either directly or through allocation of scores (Milner *et al*., 1998).

Table 2.4: Habitat data collected for application of the HABSCORE model (Barnard, 1999)

Catchment Data	**General Site Data**	**Detailed Habitat Data**
Distance from tidal limit	National grid reference	Wetted width
Distance from principal source	Total percentage cover of reaches water surface	Equally spaced depth measurements to coincide with ¼, ½ and ¾ width.
Link number (the number of lower order tributaries)	Migratory access for salmonid species (any barriers are present)	Substrate in 10m section
Conductivity	Substrate embeddedness of whole reach	Flow in 10m section
Discharge code	Flow conditions in whole reach	Cover in 10m section
Catchment gradient	Upstream land use	
Site gradient	Potential impacts to the site	

Although not all variables are used in model application, superfluous information is useful in analysis of results (Barnard, 1999). There are two major outputs from HABSCORE:

- ***Habitat Quality Score*** (HQS): determines the expected average fish population density (n/100 m^{-2}) with 95% confidence limits representing the potential stock status of the site and habitat quality presented as fish abundance, this can by calculated using habitat data only (Milner *et al*., 1998).
- ***Habitat Utilization Index*** (HUI): measures the usage of available habitat. It measures the difference between the observed population and the expected population size (HQS) with confidence limits (Milner *et al*., 1998).

Variability of fish populations can be expressed as spatial variance due to site location and physiographical and environmental features at site and catchment scale, temporal variance due to the time of sampling and error variance, which can be due to site-specific trends, measurement inaccuracy or random factors across all sites. This variability influences the potential uses and limitations of habitat models, these models operate by explaining where possible variation of fish populations on a spatial scale (Milner *et al*., 1998).

HABSCORE has potential for use in environmental impact assessment (EIA) in separating variation due to natural habitat factors from the impact of external pressures. It is also useful in assessing the impacts of management practices by using HQS as a covariate to unravel complexities of habitat variation between sites (Milner *et al*., 1998). HABSCORE is also valuable in environmental and fish quality assessment, enabling fish abundance (HQS) targets to be set and performance (HUI) to be evaluated against these targets, where fishery survey data are lacking fish-carrying potential of the reach can be calculated (Milner *et al*., 1998).

Model application holds practical benefits in that they are useful in describing habitat quality on large scales with relatively small-scale and inexpensive data collection (Milner *et al*., 1998). However, care must be taken as fish-habitat relationships are complex and models provide only an empirical, statistical simplification of these intricate ecological processes (Crisp, 1996; Armstrong *et al*., 2003). Variables applied are expected to have an influence over ecological functioning and fish abundance, but there is strong correlation between variables therefore model variables may not be those that have the strongest influence over fish/ age group abundance (Armstrong *et al*., 2003).

In summary, fish have different habitat requirements according to species and life stage. If these requirements are not met, a species may be absent or, where present may be in low numbers. Various methods are developing to evaluate fish habitat availability and ecological integrity. Habitat availability and thus fish species abundance and composition is fundamentally linked to the physical processes operating with rivers and their catchments therefore, the numerous pressures influencing river systems can have a negative impact upon habitat availability and thus fish abundance and community composition.

2.5 Pressures

Fishes are among the World's most precious natural resources (Ormerod, 2003). Decline in status of inland fisheries can be closely correlated with habitat loss and degradation (Hellawell, 1988; Cowx, 1994; Aarts *et al*., 2004). Any pressure acting on a freshwater river system can have implications for instream habitat availability (Arthington *et al*., 2004; Schweizer *et al*., 2007a), thus composition, abundance and age structure of fish species present (Cowx *et al*., 1993). Habitat loss can be a press disturbance, a long-term disturbance causing persistent modification of species composition, or a pulse disturbance, causing an instant impact on fish densities, or can be both (Parasiewicz, 2007b). Rivers are subjected to numerous and often competing pressures (Palmer *et al*., 2010; Null and Lund, 2011). Pressures can be divided into three groups: physical, biological and chemical (Degerman *et al*., 2007); or according to Giller (2005) may be classified as four categories, ecosystem destruction, physical habitat alteration, water chemistry alteration and direct species additions or removals (Malmqvist and Rundle 2002 in Giller 2005). The focus of this overview is on physical pressures and consequences of physical habitat alterations.

Use of rivers to fulfil requirements of society frequently involves a range of physical alterations to the river channel (Harper *et al*., 1998a; Harper and Everard, 1998b; Wolfert, 2001; Ormerod, 2003; Schanze *et al*., 2004; Giller, 2005; Lepori *et al*., 2005; Schweizer, 2007b); resulting in changes to natural form and processes (Dunne and Leopold, 1978, Poff *et al*., 1997; Lucas and Marmulla, 2000; Lepori *et al*., 2005; FAO, 2008), either directly or indirectly (Wolfert, 2001). Other pressures are associated with water management activities which involve controlling and altering the hydrological cycle affecting both latitudinal and longitudinal connectivity and the quantity and dynamics of river flow; this impacts upon the aquatic ecosystem increasing homogenisation through reduced flow variability and increases habitat loss and fragmentation (Giller, 2005). Any alteration to the flow regime or the physical structure of the river channel results in altered sediment transport (Lucas and Marmulla, 2000; Dunne *et al.*, 1978; Poff *et al.*1997 with consequences for morphology and the physical nature of the river (Poff *et al.*, 1997), often resulting in simplification of the channel with loss of hydraulic and morphological variability (Ward *et al*., 2001 in Schweizer, 2007b). Consequently, habitat configuration is impacted (Harper and Everard, 1998b) and the continuity and connectivity between habitat patches is interrupted (Stanford *et al.*, 1996), with major impacts on fish populations; reduced recruitment is particularly pertinent (EIFAC, 1984). However, there are difficulties identifying specific causes of reduced fish populations, as negative impacts commonly operate in combination over various spatio-temporal scales (Ormerod, 2003).

Human demands on rivers often overlap, frequently with opposing requirements (Schanze *et al*., 2004) with pressures usually impacting upon more than one environmental factor (Ormerod *et al.,* 2010) often exacerbating already prevalent pressures. Rivers are especially vulnerable to a pressure anywhere within the catchment due to their linear nature (Malmqvist and Rundle 2002 in Giller, 2005). Each physical impact has its own set of physical and biological consequences, some of the most notable direct physical impacts on rivers are:

2.5.1 Channelisation and river engineering

Worldwide, many river systems have been regulated and channelised (Petts, 1989 in Schweizer, 2007b). In England and Wales, most rivers have been subjected to channelisation for flood defence, land drainage and navigation (Brookes, 1985a; Ebrahimnezad and Harper, 1997). Channelisation modifies the river channel resulting in loss of meander bends, increased depth, over-widening, restructuring of the banks with homogenous concrete or rip rap and removal of instream obstructions (Welcomme, 1994; Punchard *et al*., 2000) producing a monotonous channel with little habitat diversity. Straightening can also remove up to 50% of the channel length (Petts, 1984) increasing the gradient and escalating in-channel flow velocity (Brooker, 1985; Cowx *et al*., 1993; Punchard *et al*., 2000), reducing the potential for erosion and deposition (Petts, 1984) and increasing potential for substrate transportation, removing the characteristic pool-riffle system and creating uniform flow biotopes such as runs and glides (Punchard *et al*., 2000). Often, an armoured bed results from the selective removal of fine sediments by increased flow velocity (Petts, 1984) and the possibility of considerable deposition downstream increases, particularly if the channel has been widened (Brookes, 1988). The loss of structural diversity and simplified flow pattern resulting from channelisation is a major cause of habitat degradation in rivers (Giller, 2005) and has consequences for species presence and fish community composition (Petts, 1984; Mann, 1988; Swales, 1988 in Pretty *et al*., 2003; Welcomme, 1994).

2.5.2 Weirs and dams

Modifications of the river channel can involve transversally barring the channel with weirs and dams, altering discharge and channel form (Welcomme, 1994). Traditionally dams were built to create reservoirs for power plant cooling, aquaculture and mill operations (Tomsic *et al*., 2007). Many weirs, small-scale dams, have been constructed to increase water level behind the dam producing areas of more steady water volume, which has been suggested to be beneficial to fishes (Fjellheim and Raddum, 1996), and previously used as fish habitat enhancement providing cover structures and trapping sediment (Gore and Hamilton, 1996). Fjelheim, (1996) found that weir construction increased the density of brown trout older than 2 years, which they suggest is due to elevated temperatures and improved conditions for over-winter survival; although they suggest weir density should be carefully considered to maintain intermittent riffle sections for spawning and nursery areas.

However, fragmentation of the continuity of the river system is a key impact of dam and weir construction (Arnekleiv and Ronning, 2004; Tomsic *et al*., 2007). River continuity is a hydromorphological quality element under WFD guidelines, although sections of rivers close to weirs are presently excluded from WFD classification (Mueller *et al*., 2011). River fragmentation due to weirs and dams has consequences for fish directly, delaying or completely blocking both upstream and downstream journeys of migratory species, such as salmon, trout and eel (Bernacsek, 2001; Larinier, 2001; Poff and Hart, 2002; Baumgartner and Harris, 2007; de Leaniz, 2008; Thorstad *et al*., 2008; Fjeldstad *et al*., 2011). Delay or blocked migration can impact fisheries by preventing migratory species from reaching suitable resting or spawning areas, reducing reproductive success through spawning in unsuitable areas or delayed arrival at suitable spawning areas (Thorstad *et al*., 2008). Blocking of migrants can also cause crowding around weirs increasing predation by opportunistic predators, increasing mortality from angling and encouraging the spread of infectious disease (Baumgartner and Harris, 2007; de Leaniz, 2008). Indirectly, weirs impact upon fish through modification of the physical processes within the channel effecting flow velocity, depth and substrate distribution (Bernacsek, 2001; Tomsic *et al*., 2007). Upstream, water depth is often increased and flow velocity reduced causing a ponding effect; sediment becomes finer or covered with silt (Cowx *et a*l., 1993; Poff and Hart, 2002; Ashley *et al*., 2006; de Leaniz, 2008) as reduced velocity and presence of the obstruction interferes with or stops sediment transport along the longitudinal channel interfering with the rivers natural processes (Stanley and Doyle, 2003; Ligon *et al*., 1995; de Leaniz, 2008). These changes in geomorphological processes can lead to altered nutrient and energy flux (Poff and Hart, 2002), alteration of the thermal regime (Poff and Hart, 2002), storage of contaminants in trapped sediments (Bernacsek, 2001; Ashley *et al*., 2006), separation of the channel from the floodplain (Bernacsek, 2001; de Leaniz, 2008) and habitat alteration at a range of spatial scales (Poff and Hart, 2002) including loss or degradation of spawning sites and areas of fast flow for young salmonids (Poff and Hart, 2002; Fjeldstad *et al*., 2011). Downstream of the weir, higher flow velocities expose clean gravel substrates with a rejuvenated erosion zone produced for some distance downstream (Welcomme, 1994). Natural gravel recruitment can be limited reducing quality and extent of downstream spawning areas (Larinier, 2001; de Leaniz, 2008). Transformed habitat availability can result in altered abundance and composition of prey resulting in fish inhabiting these different habitat patches being forced to change their diet (Baumgartner and Harris, 2007), affecting food-chain length and predator-prey interactions (Power *et al*., 1996).

Impoundment can cause river habitat to shift from lotic to lentic causing potential problems for species requiring relatively fast flow conditions (Larinier, 2001), resulting in altered fish community composition through altered habitat availability (Jackson and Marmulla, 2001; de Leaniz, 2008) with river-adapted species being replaced over time (Jackson and Marmulla, 2001).

Mueller *et al*. (2011) hypothesised that upstream and downstream sides of weirs within one river differ in abiotic habitat characteristics, biodiversity and community composition. Their research in the Rivers Elbe, Rhine and Danube found that habitat characteristics differentiated strongly between upstream and downstream weir sides with water depth significantly higher and flow velocity significantly lower upstream than downstream, and much finer sediment composition upstream than downstream. Fish composition upstream and downstream of the weir was associated with different ecological requirements, which they suggested is related to changes in water depth, current speed and substrate composition. The differences upstream and downstream of the weir were often greater than differences between rivers (Mueller *et al.*, 2011).

Small weirs (<5m) have equally important impacts on hydrological processes and fish migrations and habitat availability, although these are often less well understood (Larinier, 2001; Hart *et al*., 2002; de Leaniz, 2008). Small weirs show many of the same characteristics as large dams, trapping sediment (Ashley *et al*., 2006), altering habitat availability, the life history of migratory species (Larinier, 2001; Hart *et al*., 2002; de Leaniz, 2008) and fish community structure (Baumgartner and Harris, 2007; de Leaniz, 2008). The collective effect of many small weirs can be particularly treacherous (Williams, 1998; Naughton *et al*., 2005 in de Leaniz, 2008; Thorstad *et al*., 2008) due to their extent (de Leaniz, 2008). However, weir height is a poor indicator of passability as this is much more dependent upon general hydraulic characteristics of the reach and fish size and type (Larinier, 2001; de Leaniz, 2008). Objects can be permanently impassable for all species, passable by specific species or individuals, or passable only at certain times of the year when flow conditions are permissive (Larinier, 2001).

2.5.3 Climate change

Although not a focus of this overview, a factor requiring consideration when managing any environmental or ecological issue is climate change. Climate change is having a major impact on alterations of the thermal regime and runoff, varying conditions from those typically experienced by species (Ormerod *et al*., 2010); with potentially major impacts upon riverine systems as the hydrological cycle and atmospheric temperature are intrinsically linked. An approximate increase in temperature of 0.76°C between 1906 and 2005, with a more rapid increase over the last five decades, has been linked with several changes in the hydrological cycle and hydrological systems with changes in volume, timing and intensity of precipitation events and evaporation impacting upon river flow (Bates *et al*., 2008). River discharge is an important feature of habitat availability and diversity for fish to complete important life stages (Xenopoulos *et al*., 2005), However, due to anthropogenic modifications of catchments impacting and altering flow regimes, it is difficult to conclude the impacts of climate change on river runoff and discharge (Bates *et al*., 2008), although alteration of river discharges due to climate change can alter the severity of other pressures acting upon rivers, often aggravating existing problems (Riley *et al*., 2009; Ormerod *et al*., 2010).

2.6 The Water Framework Directive

Under Water Framework Directive requirements there is a need to analyse pressures and impacts acting within each river basin district to assist in the development of monitoring programmes and develop programmes of measures, operational by 2012 (IMPRESS, 2003) to improve areas not identified as “good status” by 2015 (Logan and Furse, 2002). Each river basin assessment must contain an analysis of basin characteristics, a review of the impact of human activity on water quality status (IMPRESS, 2003; Schanze *et al*., 2004) and an economic analysis of water use (IMPRESS, 2003). It is during the assessment of status through identification of pressures and impacts that a clear understanding of interaction between habitat and biological factors is needed (Logan and Furse, 2002). The outcomes of these assessments are used to assemble River Basin Management Plans (RBMP)

(IMPRESS, 2003). The degradation of river ecosystems has in many cases resulted in the need for river rehabilitation (Sawyer *et al*., 2009) and the assembly of these RBMPs allows identification of where river rehabilitation works are required and the nature of the work. These plans will be reviewed and updated every six years as a cyclical process with further rounds of RBMP preparation planned for 2021 and 2027.

2.7 River rehabilitation

Attempts to mitigate pressures and impacts are becoming more and more widespread as demand increases to ameliorate problems arising from the use and misuse of freshwater resources and habitats (Maddock, 1999; Walker *et al*., 2002; Giller, 2005; Mainstone and Holmes, 2010). Through searching the literature, there is difficulty in finding estimates of the total cost of river rehabilitation. The majority of river rehabilitation work is carried out by River's Trusts and details of individual river rehabilitation projects are available through the National River Restoration Inventory (NRRI) through the River Restoration Centre (RRC) website. The Environment Agency (EA) also carries out river rehabilitation work, but information regarding these projects is not so easily accessible. Accessibility of information is being improved through collaborations between the RRC and EA such as the LIFE + RESTORE project which aims to collate all information into a freely available online inventory.

River rehabilitation schemes frequently aim to ameliorate the impacts of flood mitigation schemes, or improve channel degradation, particularly where societal support is available (Walker *et al*., 2002), but schemes often encounter obstacles as a result of societal demands requiring planning from all stakeholders (Schanze *et al*., 2004). According to Wheaton (2005), river rehabilitation is derived from nine background motives: ecosystem restoration, habitat restoration, flood control, floodplain reconnection, bank protection, sediment management, water quality, aesthetics and recreation, although it is difficult to characterise motives and put them into a hierarchy (Wheaton, 2005). However, regardless of motive or scale, the principal aim of river rehabilitation is to improve damaged systems into healthy ones, augmenting ecosystem services (Giller, 2005) most commonly through active river rehabilitation measures (Welcomme, 1994).

Various strategies can be applied at different scales depending on target fish or communities:

- Basin approach: aims to rehabilitate the river basin as a whole or rehabilitate representative ecosystems within the basin and the connections between them.

- Ecosystem approach: aims to restore the processes that create and maintain habitat.
- Species approach: concentrates on one or more species with particular economic or social value (FAO, 2008).

The decline in most fish species is often attributable to habitat loss or degradation (Sheldon, 1988) as physical habitat is considered to be a key factor in abundance and diversity of fish species providing water quality is sufficient (Gorman and Carr, 1978; Milner *et al*., 1985; Punchard *et al.,* 2000). Conventional river rehabilitation has focused on enhancing fish production through habitat improvement structures (Gore *et al*., 1998) and can be broadly separated into methods that:

- impound or modify river flow including current deflectors, dams and weirs and boulder placements;
- provide cover;
- improve spawning area (Swales, 1994b, Gore *et al*., 1998, Harper *et al*., 1999, Bayley *et al*., 2000).

These strategies can potentially increase channel diversity increasing habitat availability and diversity (Bayley *et al*., 2000). Physical habitat is of particular importance for fish as each fish species has specific requirements determining survival and abundance (Swales, 1994b) and river rehabilitation is designed with the assumption that preferred physical conditions can be produced and related to flow conditions to deliver habitat requirements (Gore *et al*., 1998). River rehabilitation measures targeting habitat improvement must therefore only be implemented based upon sound understanding of habitat requirements and utilisation of the target species (Swales, 1994a). However, it is difficult to predict the response of an ecosystem to rehabilitation (Adams *et al*., 2004) and failure to improve the fish assemblage following river rehabilitation may be the result of an inappropriate scheme for the river concerned.

The implementation of the WFD is considered to symbolise a significant shift in management concepts used on European rivers (Vaughan *et al*., 2009; Hering *et al*., 2010), placing greater weight on ecosystem functioning through amalgamation of biological and physical elements and processes requiring future management and restoration work to be centred around ecological and hydromorphological principles (Clarke *et al.* 2003); with recognition that hydromorphology is a key factor in defining habitat quality (Harvey *et al*., 2008). Under WFD guidance, “good ecological status” and "good ecological potential" are defined by chemical, physical, biological and morphological factors and a requirement that not only is good status achieved but that no deteriation occurs (Pollard and Huxham, 1998; Logan and Furse, 2002; IMPRESS, 2003; Clarke *et al*. 2003). Therefore, emphasis has been placed upon the relationship between river form and processes and associated ecology (Newson, 1998b) raising the challenge to integrate flow regime with channel geomorphology and freshwater ecology (Newson, 2002; Newson and Large, 2006); key features supporting these ecosystems (Clarke, *et al.* 2003). To meet WFD criteria, any proposal for river rehabilitation should derive from a detailed understanding of the ecology, hydrology, morphology and pressures acting upon a system, and should be customised to the target river (Stanford *et al*., 1996, Lucas and Marmulla, 2000; Ward *et al*., 2001; FAO, 2008). Attempts to restore the natural processes and dynamics of the river system should be made at a suitable scale within the catchment as well as focusing on mitigating the immediate pressure and impact (Harper *et al*., 1999; Kemp *et al*., 1999) aiming to allow a more natural morphology and ecology to exist (Kemp *et al*., 1999). Failed restoration attempts are often a consequence of poor geomorphological understanding (Moss, 1998).

Extensive river rehabilitation projects using various methods take place throughout the UK to improve physical and hydraulic characteristics and increase fish abundance and diversity to improve environmental value (Swales, 1994b; Holmes *et al*., 1998 in Pretty *et al*., 2003). Numerous habitat rehabilitation projects focus primarily on the construction of instream features, provision of fish passage and the restoration of channel form (Bayley *et al*., 2000); however, systems take time to respond to shifting conditions (Bayley *et al*., 2000; Hering *et al*., 2010). It is crucial that bed form design is produced, which also allows sustainable hydraulic functioning and the recreation of physical habitat for river rehabilitation must be based upon a thorough understanding of river form and process (Emery *et al*., 2003).

Hydromorphological restoration or enhancement can be undertaken 'passively' or 'actively' (Boon, 1992 in Malavoi, 2009). Passive methods of river rehabilitation aim to allow the natural hydrological processes erosion and deposition to restructure rivers slowly, naturally reinstating channel heterogeneity (Brookes, 1992; Pretty *et al*., 2003; Gillilan *et al*., 2005; Giller, 2005; Malavoi, 2009). Active methods are more dynamic and use specific measures to modify channel configuration and increase heterogeneity or increase variations in stream flow (Gillilan *et al*., 2005; Giller, 2005), particularly in sluggish streams with little sediment transport (Malavoi, 2009). Generally, techniques used in river rehabilitation attempt to restore natural features using physical instream methods such as channel narrowing, bank re-profiling and reinstating riverbed features

(Cowx and Welcomme, 1998) to stabilise substrate or modify flow conditions (de Jalon, 1995 in Harper, 1999; Gore *et al*., 1998). Often, these active methods of rehabilitation are required as natural recovery from channel modification may be limited, particularly in reaches where stream power is insufficient to transport sediment and form instream features (Pretty *et al*., 2003).

However rehabilitation is designed, planning should acknowledge that the river is one component of the whole catchment system; therefore, although most measures will be implemented on a local scale, consideration should be given to processes occurring at scales further up the hierarchy (Moss, 1998; FAO, 2008). Potential impacts of any river rehabilitation measure must be considered from a catchment perspective prior to implementation to prevent unfavourable impacts elsewhere in the system (Cowx, 1994). Some methods of active instream river rehabilitation, particularly those studied later in this thesis are described below.

2.8 Measures

2.8.1 Installation of gravels to the channel bed

The introduction of specific habitat features has become a popular method of river rehabilitation (Sear and Newson, 2004) to re-establish hydraulic geometry (Pasternak *et al*., 2008) mitigating the adverse effects of channelisation. One technique is the addition of gravels to the riverbed or installation of artificial riffles (Walther and Whiles, 2008). The pool-riffle sequence is considered to be a primary feature of mesoscale physical habitat and the key determinant of in-channel flow patterns (Emery *et al*., 2003). Installation of gravels and artificial riffles in river rehabilitation aims to improve physical parameters such as depth and flow velocity (Sawyer *et al*., 2009) and increase the diversity of aquatic habitat, particularly spawning areas (Barlaup *et al*., 2008).

Sear (2004), reporting on the hydraulic impact and performance of gravel installations on the River Waveney, Suffolk, showed installation of gravel beds increased the range of values of physical parameters used to define physical habitat in the reach, increasing the range of depth, flow velocity and substrate available within the rehabilitated reaches (Sear and Newson, 2004). Harper (1998) found that 3 years following installation of artificial riffles, the most shallow riffle installations diversified their physical environment, creating scour pools and new functional habitats within the shallow environment, and Schwarz (2007) found the total number of mesohabitats increased in the equivalent distance following construction of a new pool-riffle sequence, with a concurrent increase in fish abundance, diversity and biomass. In

Norwegian rivers, Barlaup (2008) concluded the addition of gravel leads to an increase in the availability of spawning areas and can be an important tool for habitat restoration, although their findings suggested it is important to monitor gravel installations for displacement of gravel downstream or sedimentation.

2.8.2 Weir removal

Removal of dams and weirs is becoming an increasingly popular method of river rehabilitation to improve ecological integrity of river systems, particularly in the USA (Poff and Hart, 2002; Bushaw-Newton *et al*., 2002; Orr *et al*., 2006; Tullos, 2009; Fjeldstad *et al*., 2011), through overturning negative impacts of weirs and dams including obstruction of migratory routes and alteration of riverine processes (Orr *et al*., 2006, de Leaniz, 2008). Outside of America, dam removal is relatively new, although it is fast becoming widespread within Europe (de Leaniz, 2008). Large dam removal is carried out for numerous reasons not least for structural and safety reasons, although dam decommissioning is becoming a useful management tool in river restoration (de Leaniz, 2008). It is seen as an ecologically friendly way to reinstate geomorphic and hydrological processes to river channels by returning the natural flow regime to the system, thereby increasing river ecosystem diversity (Tomsic *et al*., 2007) and reducing fragmentation. Many small weirs are also being removed and thus far the greatest proportion of removals are of those less than 5 m (Doyle *et al*., 2000).

Removal of barriers to fish migration is one of the most positive perceived impacts of weir and dam removal, although little published literature exists regarding changes in population size of migratory species following weir and dam removal (Stanley and Doyle, 2003).

Restoration of the hydraulic regime is another positive ecological benefit of dam removal, reinstating connectivity and geomorphological complexity (Gregory *et al*., 2002). The removal of even small dams and weirs has potential to restore the flow regime rapidly (Hart *et al*., 2002; Shafroth *et al*., 2002). However, one of the principal impacts of dam removal is the release of impounded sediment into downstream reaches (Bushaw-Newton *et al*, 2002; Stanley and Doyle, 2003) resulting in physical changes to the channel (Doyle *et al*., 2000) and potentially impacting aquatic biota (Rathburn and Wohl, 2001; Shafrot*h et al*., 2002), particularly salmonids (Gregory *et al*., 2002). Doyle (2000) stated that although the response of salmon to dam removal provides an example of a population with great potential to recover, an evaluation of potential impacts of the Elwha Dam, Washington revealed that suspended sediment loads could have major negative impacts for Salmon migrating upstream to spawn and should be scheduled to minimise such impacts. Released sediments may also reduce the ability of the channel to move water and sediment downstream (Rathburn and Wohl, 2001), and can fill or partially fill channels resulting in temporary avulsion (Shafroth *et al*., 2002). The sediment travels downstream in a sediment pulse (Shafroth *et al*., 2002) and it is difficult to predict how long it will take for sediment to be transported downstream; this can vary from months to years, and the impacts on channel configuration that will be generated along the way (Grant *et al*., 2001; Gregory *et al*., 2002; Rathburn and Wohl, 2001) These geomorphic adjustments are strongly dependent upon the sediment type and the ability of the river to transport sediment (Hart *et al*., 2002). Doyle (2000) reported that, where sediment movement has been recorded, findings consistently show immediate morphological changes downstream of the dam following removal and provides examples from several American rivers including Muskegon River, Michigan, Hudson River and Milwaukee River, Wisconsin. Problems associated with transport of stored sediment are exacerbated if contaminants have been stored in the sediments (Bushaw-Newton *et al*., 2002; Ashley *et al*., 2006).

In the former impoundment, increased water velocity and change in substrate composition can result from changes in channel geometry and mobilisation of sediment resulting from down cutting through reservoir sediment (Orr *et al*., 2006). Little published information exists regarding the environmental responses of dam removal, but Bushaw-Newton (2002) found that following removal of a 2 m dam on lower Manatawny Creek, South-eastern Pennsylvania in 2000, major channel changes occurred in the previously impounded reach and downstream reaches as a result of increased sediment transport and fishes shifted from lentic to lotic species. Large-scale sediment transport and habitat change negatively impacted some fish species downstream of the dam although this appears to be a temporary response (Bushaw-Newton *et al*., 2002). Orr *et al*., (2006) found that removal of two small dams on Boulder Creek, Wisconsin showed very minor disturbance to the reach-scale channel form, substrate composition varied little from background levels and although some sediment movement was observed, changes were localised and sediment moved quickly out of the studied area. Changes to channel form were reported to be limited to the formerly impounded upstream area and areas further upstream and downstream appeared relatively unchanged (Orr *et al*., 2006). Fjeldstad (2011) found that following removal of two concrete weirs in the River Nidelva in south-east Norway, there was an increase in spawning activity of Atlantic salmon in the restored reach the first season after weir removal which is attributed to changes in water depth and velocity to conditions more suitable for spawning, egg survival and juveniles. Removal of the weirs also indicated removal of barriers delaying and blocking migration with increased migration rates and the migration peak shifting to be one month earlier than pre-removal (Fjeldstad *et al*., 2011).

When assessing changes associated with weir and dam removal it is useful to assess spatial changes upstream, where changes can occur in velocity and geomorphological processes, downstream, where erosion and sediment accumulation can occur, and in free-flowing areas of the reach (Hart *et al*., 2002). It must be noted that ecological responses to dam removal may occur over a range of time scales, fish previously blocked by the obstacle may move upstream into the former impoundment within days of removal, other longer term changes may occur as a result of species adjusting to changes in channel structure although no studies are recorded over a period long enough to conclude rates of response of ecosystem components (Hart *et al*., 2002) and, as noted by Freeman *et al.* (2001), variability in flow between years is likely to have an influence on biotic assemblages in lotic systems through the recruitment of different species.

For dam and weir removal to be considered an effective river rehabilitation method, it is important to be able to predict the potential benefits of such schemes (Hart *et al*., 2002) although very few studies have examined the biological and hydrological effects (Doyle *et al*., 2000; Grant, 2001; Orr, 2006; Bushaw-Newton *et al*., 2002) leaving the long-term impacts of removal poorly understood (Bushaw-Newton *et al*., 2002; de Leaniz, 2008), where knowledge is available it is generally based upon large-scale dam removal (Doyle *et al*., 2000). Physical alterations can result from removal of even a small human-made dam (Gregory *et al*., 2002; Velinsky *et al*., 2006) with different outcomes likely, due to the large variation in size of dams and differences in geology, climate and size and urbanisation of river basins (Grant, 2001; Hart *et al*., 2002; Ashley *et al*., 2006; Velinsky *et a*l., 2006). It is suggested some ecosystem components may take up to 10 years to adjust to post-removal conditions (Bushaw-Newton *et al*., 2002). The possibility that a river may take a long time, or never fully recover from the impact of the dam also needs to be a consideration (de Leaniz, 2008). Long term monitoring, involving collection of baseline data prior to removal in a quantity sufficient to allow appraisal of system change, and data collection from a control site (Bushaw-Newton *et al*., 2002), is therefore essential to clarify how species persistence is influenced by hydrological variability (de Leaniz, 2008). The increase in removal of small dams provides an ideal platform from which scientists can begin to understand interactions between dam removal and ecological and hydrological consequences through monitoring and evaluation of impacts at different scales (Doyle *et al*., 2000; Grant, 2001; Orr *et al*., 2006). Unfortunately, to date it is impossible to draw conclusions about the effects of weir removals due to the paucity of project monitoring and the wide variety of outcomes where monitoring has taken place, also, monitoring often involves just one ecosystem component frequently relying on qualitative rather that quantitative processes with limited spatial and temporal replication (Hart *et al*., 2002).

2.8.3 Channel narrowing

Over-widening of river channels is a consequence of past drainage activity and the activity of grazing animals (Wild Trout Trust). An over widened channel reduces habitat diversity, and the increase in cross-sectional area reduces flow velocity and encourages the deposition of fine sediment (Wild Trout Trust). Narrowing of the river channel is an effective method of river rehabilitation to reduce the cross-sectional area of the channel and create habitat for aquatic and marginal biota. Channel narrowing increases local flow velocity, encouraging scouring and thus reduces sedimentation (Dennis *et al*., 2011). This can thus be beneficial to the availability of spawning areas for fish (Dennis *et al*., 2011). Channel narrowing should be undertaken using locally available materials and could use natural wood, brush would from tree cutting, coir fibre, non-biodegradable or 'hybrid' geotextiles (Wild Trout Trust; Dennis *et al*., 2011). Building causeways, mid-channel islands, groynes or redistributing instream gravel are further methods for channel narrowing (Wild Trout Trust; Dennis *et al*., 2011).

2.9 River rehabilitation goals

Many questions remain over defining a realistic goal for river rehabilitation projects (Haslam, 1996, Dobson and Cariss, 1999 in Pretty *et al*., 2003) with lack of definition of success remaining (Palmer *et al*., 2005; Giller, 2005). Common rehabilitation goals include restoring ecological, geomorphic and hydraulic processes (Sawyer *et al*., 2009) and recreating a stable and sustainable ecosystem, with ecological value (Gore *et al*., 1998). Haslam (1996) suggested the term river restoration implies intention of returning a degraded river to its original form, whereas enhancement suggests increasing the value, importance or attractiveness of the river, but does not imply full restoration. Contemporary theories, however, suggest it is unrealistic to return a river to its "natural" state thus objectives should be focussed on reducing adverse impacts and enhancing present condition (Swales, 1994b; Pretty *et al*., 2003; Bain and Meixler, 2008), reinstating natural processes and allowing these to restore habitat conditions (Swales, 1994b). With regard to ecological success, debate remains as to the realism of expecting to restore a river to a natural state or if the contemporary natural state can be realistically defined (Pretty *et al*. 2003). Giller (2005) questioned if the goal for rehabilitation schemes should be restoration of the whole pre-disturbed system and its processes, the recovery of lost or damaged species populations, or improvement of the connections between the restored system and other ecosystems.

On the other hand, Schanze *et al*. (2004) and Palmer *et al*. (2005) suggested the most successful river rehabilitation for contemporary society lies at the junction between stakeholder success, learning success and ecological success (Figure 2.6) allowing ecologically beneficial river rehabilitation actions, such as reinstating hydrological features whilst to some extent preserving their anthropogenic use and contributing to future knowledge and management practice.

Figure 2.6: A diagrammatic description of the most successful river rehabilitation projects modified from Palmer et al. (2005).

However the success of river rehabilitation is determined it is essential that monitoring and evaluation of project outcomes take place in order to influence future river rehabilitation plans and feed into adaptive management loops.

2.10 Monitoring and evaluation

Despite the recent increase in popularity of river rehabilitation schemes, there is little follow up information reporting on successes and failures of such schemes (Ormerod, 2004; Gillilan *et al*., 2005; Palmer and Allan, 2006; Woolsley *et al*., 2007; Harris, 2012) in particular with regards to fish populations (Cowx, 1994; FAO, 2008; Roni *et al*., 2008), leaving results of many projects unknown (Wohl *et al*., 2005 in Sawyer *et al*., 2009). Many published studies focus on biological response to river rehabilitation using invertebrates or plants as indicators (Embrahimnezhad and Harper, 1997; Biggs *et al*., 1998; Harper *et al*., 1998; Pretty *et al*., 2003), although benefits to fish are sometimes inferred (Pretty *et al*., 2003). Of the published results available, the majority are from USA, Canada and Western Europe (Roni *et al*., 2008). Monitoring, evaluation and adaptive management of rehabilitation projects is vital in determining the success of schemes (Schanze *et al*., 2004; Palmer and Allan, 2006; FAO, 2008; Woolsley *et al*., 2007; Brierley *et al.,* 2010; Hammond *et al*., 2011) and whilst large-scale projects are most likely to have detailed follow up studies, the more widespread, small-scale projects are all too frequently overlooked (Pretty *et al.,* 2003). Where results are published, potential bias towards publication of positive results may also be a concern (Roni *et al*., 2008).

Monitoring of a restoration project should be based upon sound data collection, wherever possible, both pre-implementation and post-implementation. This should include thorough design rationale, criteria for determining success and baseline surveys (Downs and Kondolf, 2002; Henry *et al*., 2002; Wheaton *et al*., 2004 in Sawyer *et al*., 2009; Hammond *et al*., 2011), at the site and downstream where necessary (Schanze *et al*., 2004; Gillilan *et al*., 2005; Downs and Kondolf, 2002; Palmer *et al*., 2005; Sawyer *et al*., 2009;) without which the results of implementation cannot be assessed (Frissell and Ralph, 1998 in Gillilan *et a*l., 2005; Brierley *et al*., 2010). Without this monitoring it is impossible to learn from successes and failures and develop suitable project controls (Gillilan *et al*., 2005), which will govern efficient schemes for future use (Schanze *et al*., 2004). A reference system could also be employed for simultaneous assessment between comparable systems, to establish changes due to restoration and those occurring as a result of natural hydrological processes (Henry *et al*, 2002; Hammond *et al*., 2011).

The effectiveness of a rehabilitation project can only be assessed using post-project appraisals and monitoring to drive adaptive management and link science with implementation (Downs and Kondolf, 2002; Schanze *et al*., 2004; Wohl *et al*., 2005; Sawyer *et al*., 2009). For monitoring to be successful it must be based around knowledge of scale-related processes, pressures experienced by the particular system, the nature of the monitored river (Johnson *et al*., 2007; Bunn *et al*., 2010) and be undertaken on both a short and long term basis. Haslam (1996) suggested a long-term data set should be collected over one, two, five and ten years. This is an imperative step in determining at what stage and to what degree the system has become self-maintaining and if the rehabilitation attempt may be considered successful (Henry *et al*., 2002), allowing the whole range of natural river adjustments to be captured (Brierley et al., 2010). In addition to long-term monitoring, Roni (2008) suggested there is need for broad-scale, watershed level monitoring and the development of consistent metrics to evaluate rehabilitation results. Holmes (1998) suggested there is a need for auditing work to be carried out, including an "audit trail", enabling information on the type of work undertaken to be retrieved and an "audit of implemented work" recording project successes and failures, although it has been suggested that rehabilitation work is now being audited through consistent and reputable literature databases reporting evaluated restoration projects regionally and nationally (Roni *et al*., 2002; Bernhardt *et al*., 2005; Bernhardt *et al*., 2007 in Sawyer *et al*., 2009) These results need to be disseminated openly to allow managers and practitioners to learn from previous successes and failures.

An important point to note is that, for river rehabilitation success to be evaluated it needs to be related to proposed objectives (Schanze *et al*., 2004) and should be related to the requirements of the target species (Cowx *et al*., 2009). This should be based upon a thorough understanding of species requirements and restoration of the natural processes that will maintain the development and protection of these characteristics (Gillilan *et al.,* 2005; Brierley *et al.,* 2010). Haslam (1996) expressed concern that there is a danger of a basic "restoration scheme" developing, which has the risk of becoming a "recommended river" with bland, insipid features. Without an understanding of linkages between physical habitat and species response, there is a risk of supposition that 'biodiversity' is simply equated to 'geodiversity' (Newson and Large, 2006).

2.11 Conclusions

The structure and organisation of river physical habitat is intrinsically linked to hydrological, geomorphological and hydraulic processes acting upon the river system. Fish have different habitat requirements depending on species and life stage; these habitats can be considered at the catchment scale through to the smallest micro-scale. Rivers provide many amenities to human populations and consequently receive a great deal of physical stress, interrupting natural physical processes. These pressures can have a significant impact upon physical habitat availability and can subsequently alter fish community composition. As a result of recent legislation (WFD, HD) there is increasing pressure to rectify impacts of these pressures and rehabilitate rivers to good ecological status. Many river rehabilitation programmes are taking place as a result of these requirements; although it is evident within the literature that very little monitoring of physical outcomes takes place. There is some evidence of monitoring of biotic response through invertebrate populations, but there is a significant lack of knowledge on the impacts of rehabilitation on fishes. It appears through the literature that there is potential to focus habitat availability on a mesoscale, which holds the potential to bridge the gap between catchment scale and microhabitat. There is currently little knowledge available at the mesoscale however; preliminary studies have shown mesoscale features to have discrete hydraulic conditions suggesting there is potential for investigation into biotic associations. Whilst the emergence of WFD and HD have increased awareness and drive to rehabilitate rivers taking into account natural physical processes there is a need to monitor these programmes to gather knowledge and inform future plans; and whilst quantifying habitat on a mesoscale appears to be in its infancy, there is potential for research developments that may guide a rapid means of habitat assessment.

Through reviewing the literature, gaps in knowledge and areas needing further research have been identified as:

- Information on hydraulic and biological outcomes of river rehabilitation
- The potential of focusing on habitat availability on a mesoscale
- Relationships between fish and instream hydraulics

The research in this thesis aims to contribute to the knowledge of biological and hydraulic outcomes of various river rehabilitation measures including small-scale weir removal, channel narrowing and installation of gravel riffles. Biological and hydraulic distinctiveness of instream habitat units is investigated to contribute to understanding of habitat on a meso-scale, and habitat use and relationships between biodiversity and physical habitat are also investigated to assess some of the complexities of fish-habitat interactions in context to river rehabilitation to meet WFD requirements.

3 CHAPTER THREE: GENERAL MATERIALS AND METHODS

Following implementation of the Water Framework Directive (2000/60/EC) (WFD), there has been an increase in the number of river rehabilitation projects implemented throughout the UK to improve ecological status of rivers to meet targets under WFD guidelines. However, despite the magnitude of schemes applied to UK Rivers, there is little monitoring and dissemination of hydraulic and biological outcomes of such schemes. This research aims to monitor and record both the hydraulic and biological (fish community composition) consequences of four river rehabilitation projects.

3.1 Sites selected

Four sites were selected on three UK Rivers targeted for rehabilitation activity (Figure 3.1, Table 3.8). Prior to site selection a number of site visits were undertaken to assess suitability of sites and rehabilitation schemes for monitoring. The criteria for site-selection were:

- A suitable depth for electric fishing and habitat surveys to be carried out through wading
- A suitable width to allow electric fishing with a single anode
- A project time-scale that would allow collection of biological and hydraulic data both pre- and post- implementation or, would allow collection of temporal biological and hydraulic data for pre-existing river rehabilitation projects

Preliminary monitoring was undertaken on several sites that were later rejected (River Kennet, Newbury; River Manifold, Staffordshire; Driffield Beck, East Yorkshire; River Don, Sheffield) as they failed to meet criteria to achieve the projects objectives. Projects were ultimately selected that allowed both hydraulic and biological (fish) outcomes of river rehabilitation to be monitored, and allowed the relationship between hydraulic conditions and fish species composition to be investigated.

Projects selected were:

- Removal of a small-scale weir in two separate stretches of the River Dove, Derbyshire
- Narrowing of an over widened channel at Lowthorpe Beck, East Yorkshire
- Installation of gravel riffles at River Stiffkey, North Norfolk

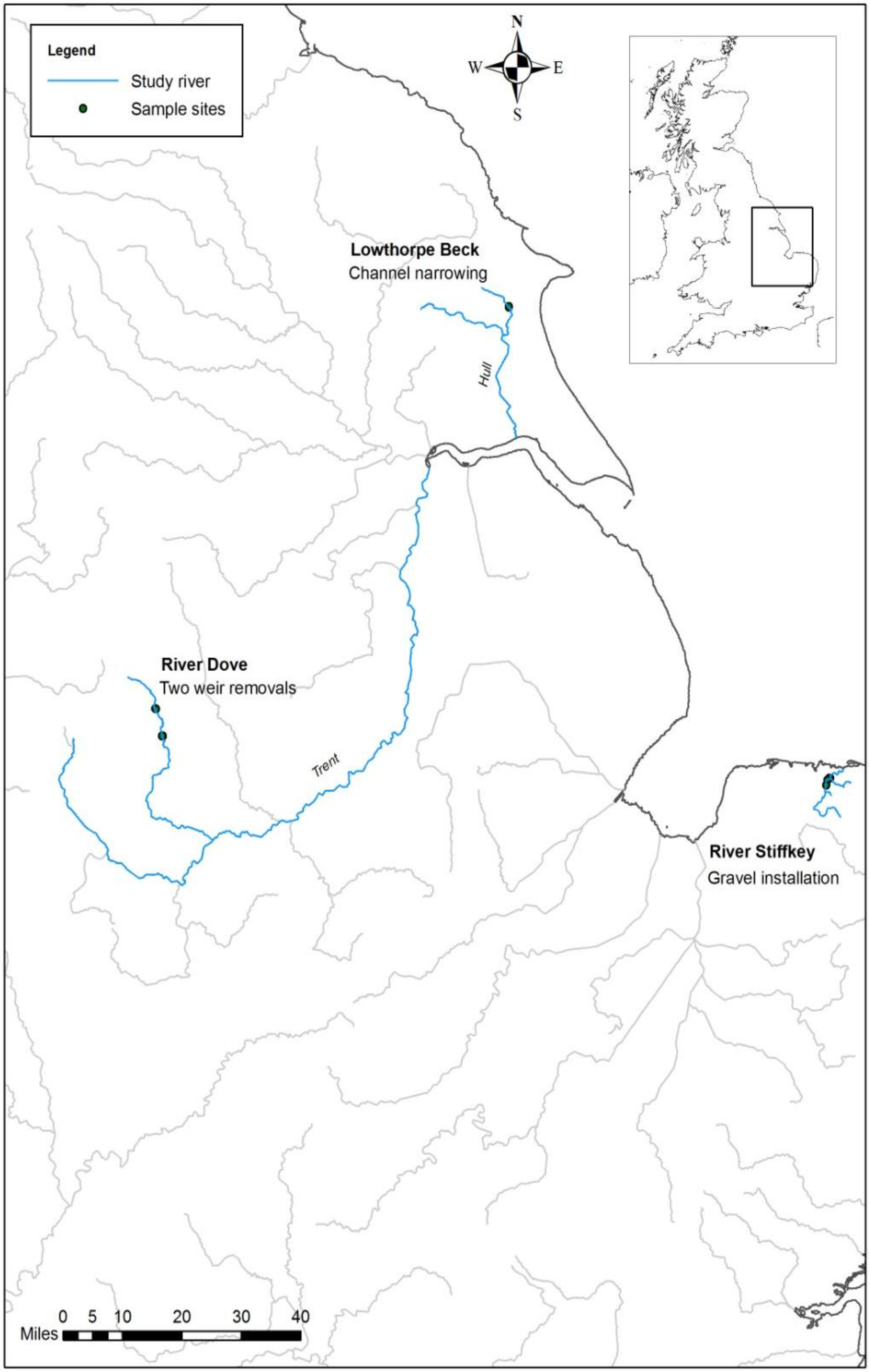


Figure 3.1: Location and description of surveyed river rehabilitation sites.

In each case river stretches were selected to encompass the rehabilitation project and the adjacent area. At Lowthorpe Beck, two further downstream stretches were monitored in addition to the river rehabilitation stretch.

River stretches for sampling were selected following site visits prior to surveys commencing and were chosen to comprise a representative range of HMUs. All river stretches were defined empirically in 50 or 100 m stretches (as these are standard distances over which to conduct electric fishing surveys). In heterogeneous channels (River Dove at Dovedale and River Stiffkey), survey stretches were chosen to encompass a number of the different HMUs present within the channel; these HMUs were then treated as individual sites (Table 3.8). In both heterogeneous channels surveyed these HMUs were riffles and glides and were identified visually using the descriptions given in Chapter 2. Sites were selected to begin and end with the start and finish of any particular HMU (riffle or glide). Where a survey stretch contained multiple HMUs (sites), the survey distance was increased to encompass all HMUs in their entirety, apart from where a HMU was a lengthy glide. If this were the case, or if the stretch contained homogenous glide habitat (River Dove at Hartington and Lowthorpe Beck) the glide was sectioned off to fit into the survey distance. Once selected, site locations were consistent throughout future sampling occasions.

3.2 Details of sites and river rehabilitation projects surveyed

3.2.1 Small-scale weir removal in the River Dove, Derbyshire

The River Dove rises on the western side of the Peak District National Park on the grit stone escarpment of Axe Edge, southwest of Buxton. It flows at or close to the limestone boundary for approximately 10 miles before it trenches into limestone close to Hartington before continuing through limestone at Dovedale (Monkhouse, 1960). The River Dove is composed of four main sections, the uppermost section flows through a broad valley of limestone hills on the east and grit stone or shale on the west, the second section flows below the village of Hartington where it becomes narrow through Beresford Dale and Wolfescote Dale. The third section through Dovedale flows for three to four miles through limestone crags and the fourth section below Thorpe Cloud is broad and open (Monkhouse, 1960). The River Dove flows through a primarily grassland catchment with a mean discharge of 1.931m^3s (Table 3.1), the long-term hydrograph (Figure 3.2) shows a dynamic river. All surveys were carried out during normal summer flow periods (Figure 3.3).

Table 3.1: Catchment characteristics of River Dove, Derbyshire data collected from National River Flow Archive, (http://www.ceh.ac.uk/data/nrfa/index.html accessed 26/01/2012)

Region	Catchment	Catchment Area	Mean Discharge	Relief	Land Cover	Geology
Midlands	Trent	83km^2	1.931m^{3}s	131.4mAOD-546.1mAOD	87.7% Grassland, 4.3% Woodland, 4.1% Arable, 0.9%Mountain/ Heath/Bog 0.4% Urban	100% Moderate permeability bedrock

The River Dove is within the Humber river basin. The ecological status of this basin is currently classified as poor and is expected to reach moderate by 2015 (Environment Agency, 2009b). The River Dove (from the source to the River Manifold) specifically is classified under WFD assessment as good ecological status with the objective of maintaining good ecological status by 2015. The River Dove is not designated as heavily modified. Fish as a biological quality element are currently classified as good with the target of maintaining good status by 2015, morphology is also currently classified as good with the target of maintaining good status by 2015 and hydrology is currently classified as high status with the target of maintaining high status by 2015.

Weirs are profuse throughout the River Dove, many were built and positioned systematically to produce feeding areas for trout and grayling and improve fishing for anglers, few of the weirs in the River Dove were used to power mills. Weirs can cause the impoundment of water and fine sediment behind the weir, interfering with the natural flow dynamic within the reach. Weirs can also present a barrier to fish migration. This study focuses on the consequences of small-scale weir removal as a method to mitigate physical modification of the channel resulting from small weirs. Two weirs were removed from the River Dove, as proposed by Trent Rivers Trust and The Wild Trout Trust, one at Dovedale and the other close to the village of Hartington.

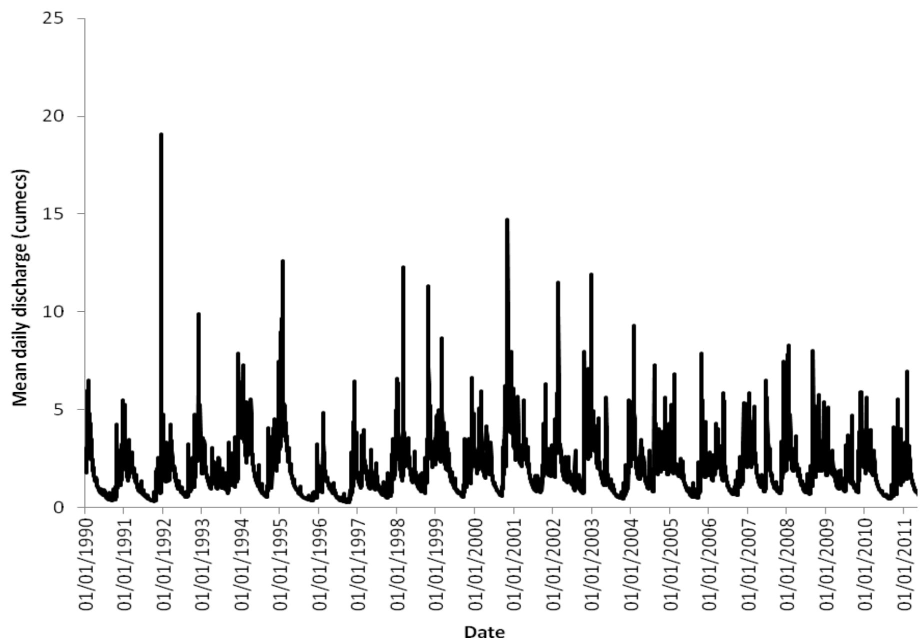


Figure 3.2: Long-term hydrograph of the River Dove. (Data source: Environment Agency).

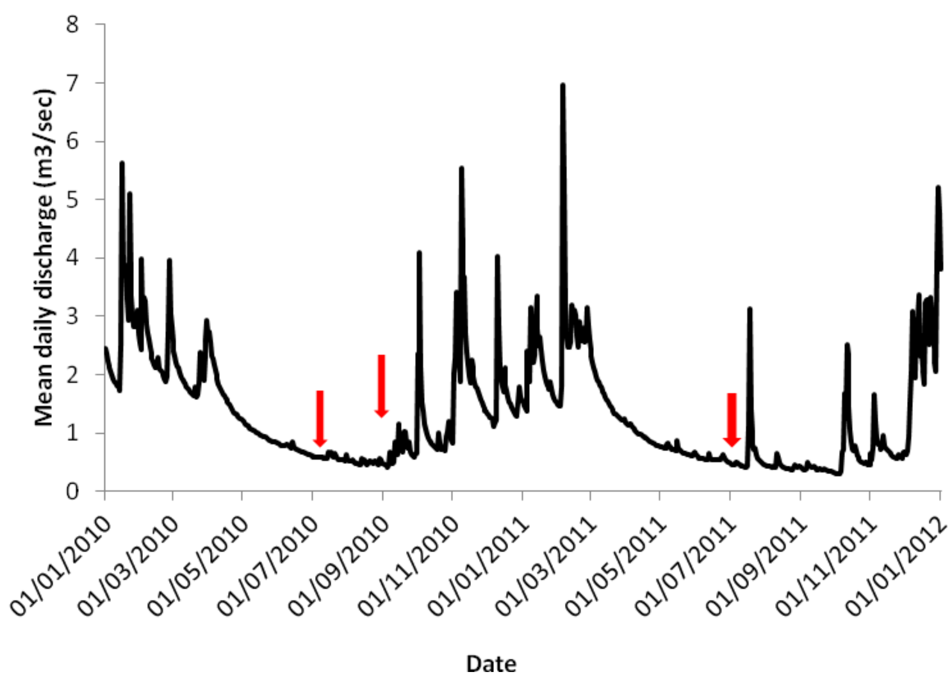


Figure 3.3: Hydrograph of the River Dove throughout the survey period. Arrows indicate survey dates (Data source: Environment Agency).

Both weirs were small rock weirs, projecting approximately 1 m from the streambed. At both sites weir removal was undertaken by removing the boulders of which the weirs were comprised.

Specific objectives for the removal of these weirs were:

- To increase hydrological connectivity
- To improve connectivity for fish migration
- To increase hydraulic diversity
- To increase habitat diversity and availability, particularly access to upstream spawning grounds for brown trout

Removal of a small-weir at Dovedale on the River Dove

The rock weir was removed from the River Dove at Dovedale (Figure 3.4) (Plates 3.1 and 3.2) in July 2010. Electric fishing and habitat surveys, as detailed in section 3.3, were carried out on a 200-m stretch of the river, split into two individual reaches of 100 m (Table 3.8), one upstream of the weir and one downstream of the weir (Figure 3.5).

Baseline fisheries and habitat data (see section 3.3) were collected in July 2010, the day before the weir was removed. Habitat data were collected the day after weir removal in July 2010. Further fisheries and habitat data were collected in September 2010, approximately eight weeks post-removal and in July 2011, one year post-removal (Table 3.8).

Dovedale sample site details

Site 1: was the most downstream site beginning 100 m downstream of the weir and extending 10 m upstream. On all survey dates this site was classified by visual observation as a riffle with shallow, turbulent flow and mainly large boulder and cobble substrate. Canopy cover and riparian vegetation were present on all survey occasions.

Site 2: began immediately upstream of Site 1 and extended upstream to the weir. This site was classified by visual observation as a glide with a steady even flow apart from a small area of turbulence immediately downstream of the weir. The substratum was primarily gravels and cobbles with some silt and there was both riparian vegetation and canopy cover at the site.

Site 3: was located upstream, beginning at the weir and extending 40 m upstream. As this site was located immediately upstream of the weir there was some impoundment of water by the weir. The water was deep and the flow was slow, silt was the dominant substrate. The site was classified through visual observation as a glide (Plate 3.3). There was canopy cover and some riparian vegetation at this site. Following weir removal, the impoundment was less obvious at the site although the water remained deep and slow flowing (Plate 3.4).

Site 4: was the most upstream site, beginning immediately upstream of Site 3 and extending 50 m upstream. The channel split into two parts at the upstream extent of Site 4 but the focus of the surveys was on the main channel. This whole site was classified through visual observation as a riffle. Site 4 was dominated by a gravel substratum, the water was shallow with a steady flow, both riparian vegetation and canopy cover were present. Following weir removal, this site appeared very different visually, the depth was lower and the gravel substratum more exposed (Plates 3.5 and 3.6).

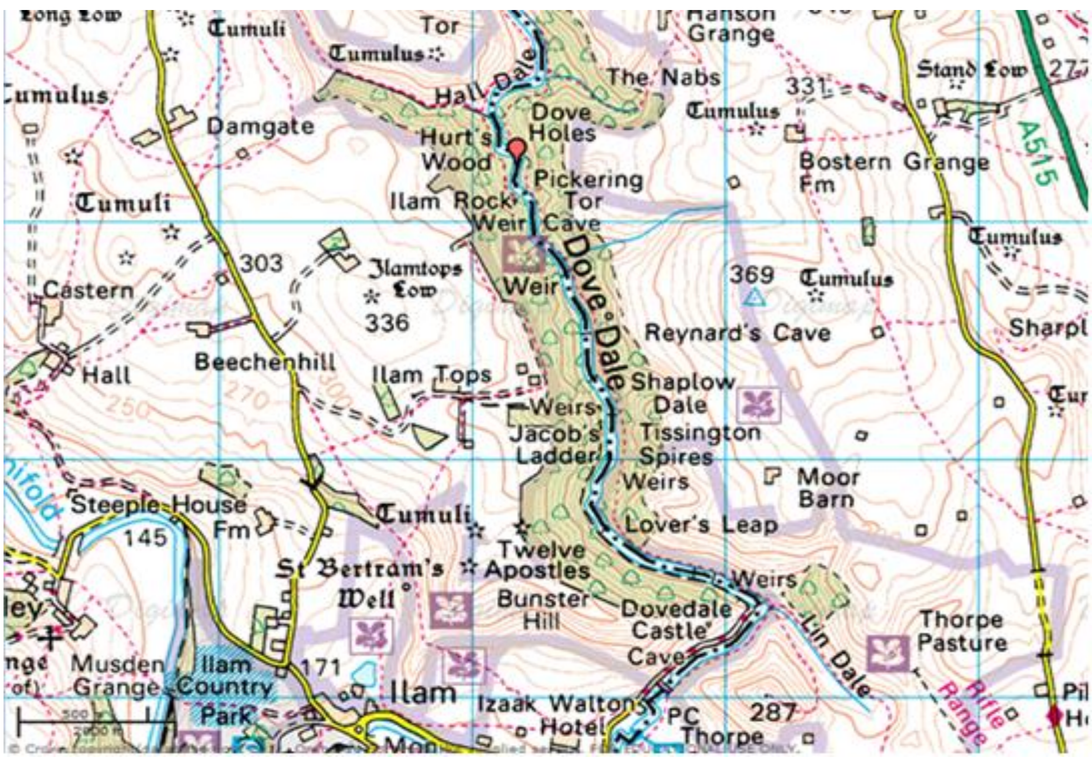


Figure 3.4: Location of weir removal site on River Dove, Dovedale. Red markers indicate the location of the weir. Map created and modified using Digimap, January 2012.

Plate 3.1: Weir in the River Dove at Dovedale prior to removal, July 2010

Plate 3.2: Weir site in the River Dove at Dovedale post removal, July 2011

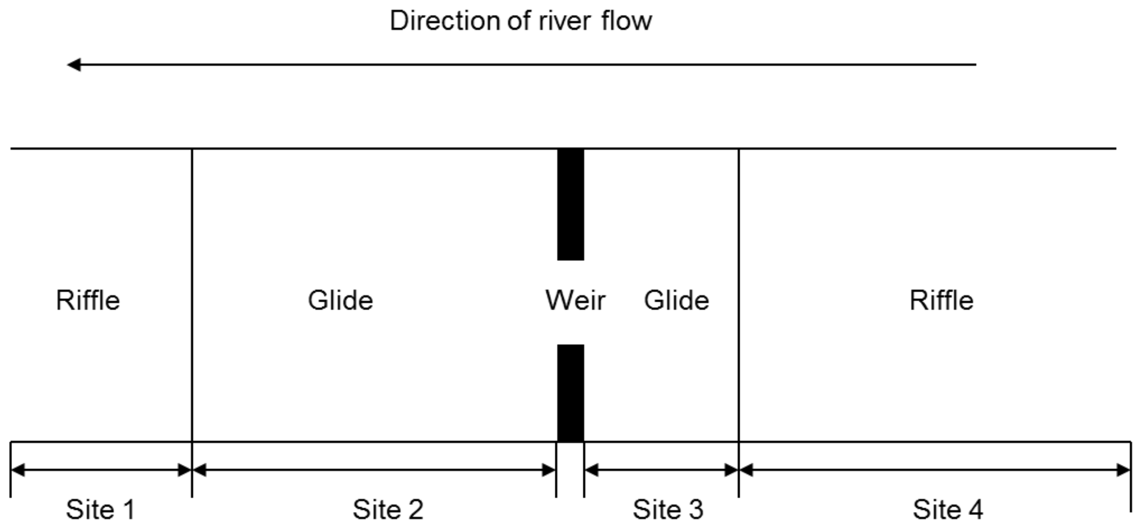


Figure 3.5: Spatial arrangement of sampled sites at River Dove, Dovedale (not to scale).

Sites codes will be used to refer to site number and rehabilitation state in subsequent chapters (Table 3.2).

Table 3.2: Site codes used to detail site number, HMU type and survey date.

Site detail	HMU	State	Date
1a	Riffle	Pre	July 2010
1b	Riffle	Post	September 2010
1c	Riffle	Post +1 year	July 2011
2a	Glide	Pre	July 2010
2b	Glide	Post	September 2010
2c	Glide	Post +1 year	July 2011
3a	Glide	Pre	July 2010
3b	Glide	Post	September 2010
3c	Glide	Post +1 year	July 2011
4a	Riffle	Pre	July 2010
4b	Riffle	Post	September 2010
4c	Riffle	Post +1 year	July 2011

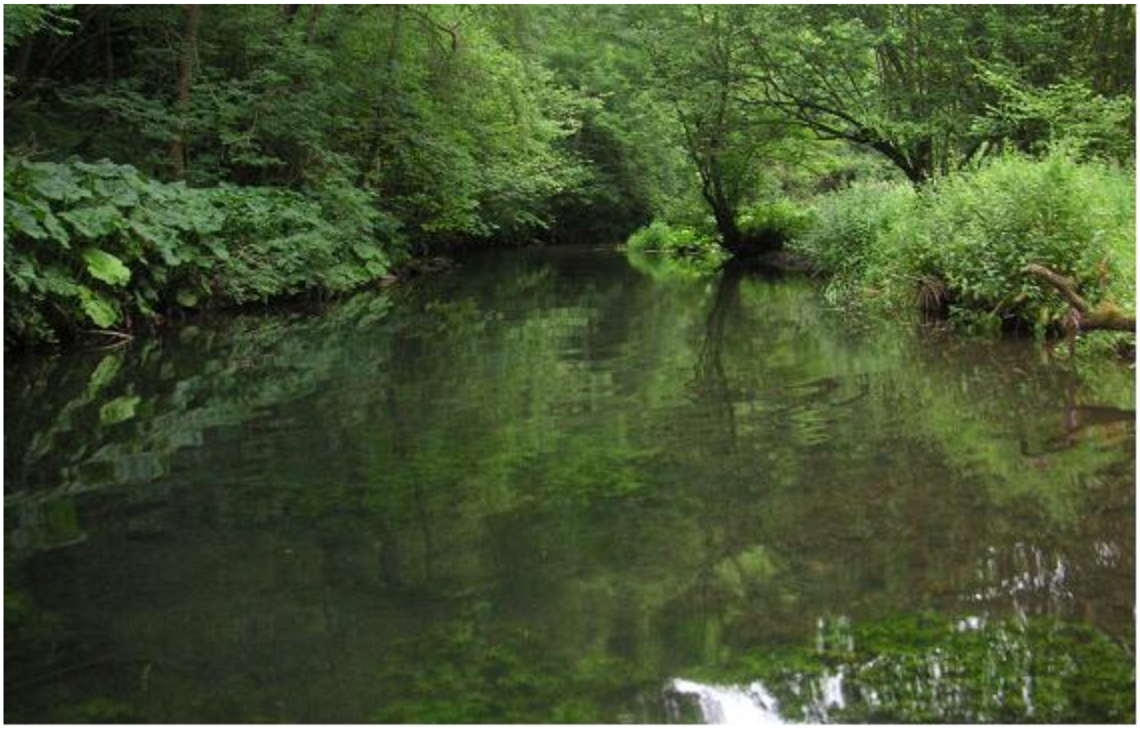


Plate 3.3: Site DD3 glide prior to weir removal July 2010

Plate 3.4: Site DD3 glide post weir removal July 2011

Plate 3.5: Site DD4 riffle pre removal July 2010.

Plate 3.6: Site DD4 riffle post removal July 2011

Removal of a small-weir at Hartington on the River Dove

A further small rock weir (approximately 1 m from the streambed) (Plates 3.7 and 3.8) was removed from the River Dove upstream of Dovedale close to the village of Hartington (Figure 3.6) in September 2010.

Plate 3.7: Weir in the River Dove at Hartington prior to removal, photograph taken April 2010.

Plate 3.8: Weir site in the River Dove at Hartington post removal, photograph taken July 2011.

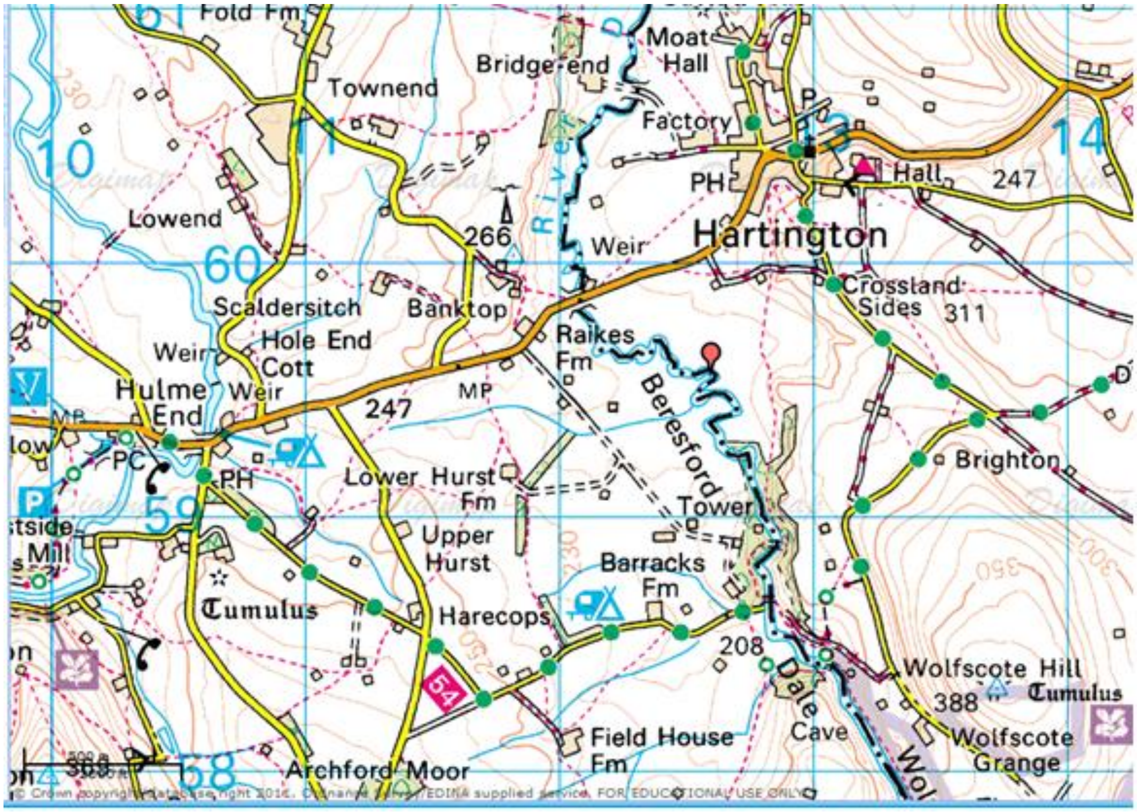


Figure 3.6: Location of weir removal site on River Dove, Hartington. The red marker indicates weir location. Map created and modified using Digimap, January 2012.

Electric fishing and habitat surveys were carried out, as detailed in section 3.3, on a 200-m stretch divided into two 100-m sections, one upstream of the weir, the other downstream of the weir (Table 3.8) (Figure 3.7). Electric fishing surveys and habitat surveys (see section 3.3) were carried out prior to weir removal in July 2010. In this case the weir was removed gradually to avoid any adverse impacts from releasing stored sediments. Further electric fishing and habitat surveys were carried out in September 2010, as weir removal was almost complete and a further survey post-removal in July 2011 (Table 3.8).

Hartington sample site details

Site 1: was the most downstream site beginning 100 m downstream of the weir. The habitat was classified through visual observation as a glide throughout the site. The flow was steady and deep and the channel was incised with raised banks. There was some canopy cover present at the site and the dominant substrate was silt.

Site HD2: began immediately upstream of the weir and stretched for 100 m upstream. The flow was deep and steady and classified through visual observation as a glide with uniform habitat, incised banks and silt-dominated substrate. There was some canopy cover and riparian vegetation at this site.

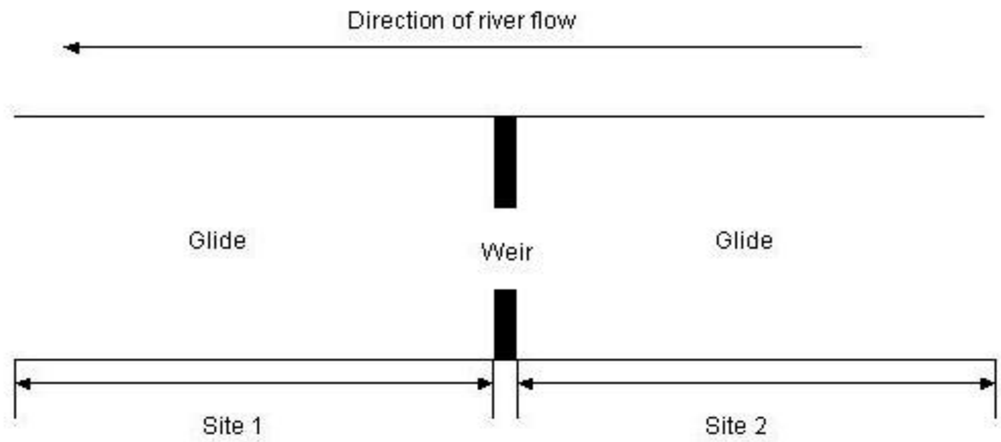


Figure 3.7: Spatial arrangement of surveys sites at River Dove, Hartington (not to scale).

Sites codes will be used to refer to site number and rehabilitation state in subsequent chapters (Table 3.3).

Table 3.3: Site codes used to detail site number, HMU type and survey date.

Site detail	HMU	State	Date
HD1a	Glide	Pre	July 2010
HD1b	Glide	Post	September 2010
HD1c	Glide	Post +1 year	July 2011
HD2a	Glide	Pre	July 2010
HD2b	Glide	Post	September 2010
HD2c	Glide	Post +1 year	July 2011
HD3a	Glide	Pre	July 2010
HD3b	Glide	Post	September 2010
HD3c	Glide	Post +1 year	July 2011
HD4a	Glide	Pre	July 2010
HD4b	Glide	Post	September 2010
HD4c	Glide	Post +1 year	July 2011

Channel-narrowing at Lowthorpe Beck (Foston Beck) at Harpham, East Yorkshire

Lowthorpe Beck is a stretch of Foston Beck located at Harpham, north of the town of Driffield in East Yorkshire. Foston Beck is one of two main tributaries making up the River Hull Headwaters, although the river name changes as it flows through different parishes along its length. The River Hull Headwaters are the most northerly chalk stream system in Britain, located to the east of the Yorkshire Wolds, discharging into the sea at the Humber Estuary.

Between Harpham, Elmswell and Kirkburn, to the confluence between the West Beck and Frodingham Beck at Emmotland, the river has been designated a Site of Special Scientific Interest (SSSI) (Figure 3.8) (Rayner and Covington, 2009). Although some areas of the SSSI have remained unmodified, the majority of the River Hull Headwaters has been heavily modified over time for a range of different purposes including land drainage, flood defence, water supply, fish farming and navigation (Rayner and Covington, 2009).

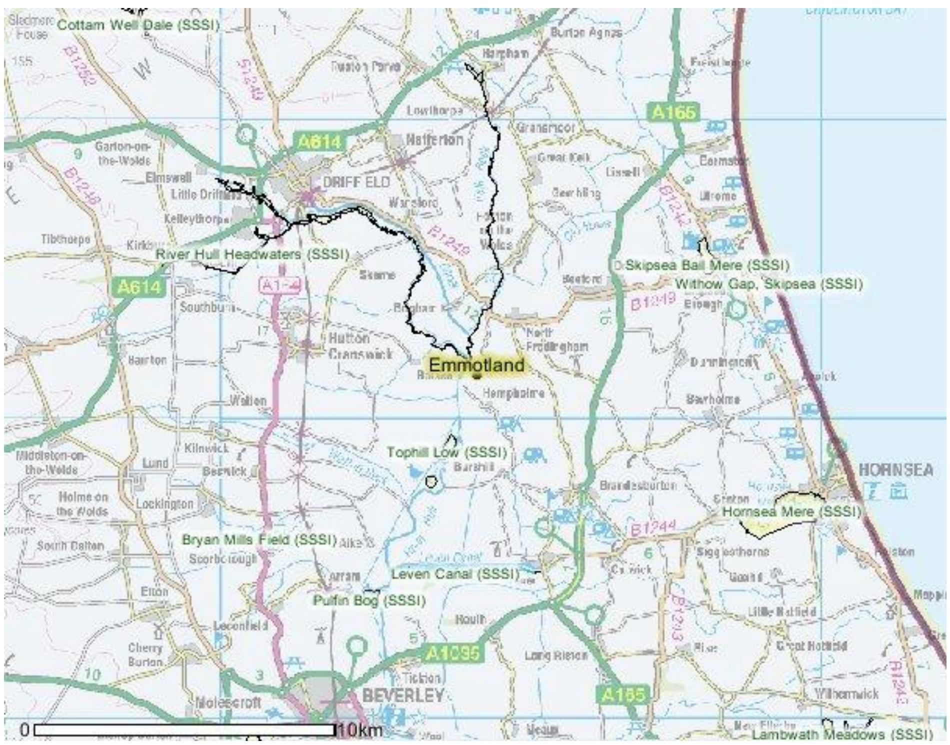


Figure 3.8: River Hull Headwaters SSSI

(Source: http://www.natureonthemap.naturalengland.org.uk/map.aspx?m=sssi accessed on 22.08.12).

The river flows through chalk, soils are light textured and free-draining and localised land use is dominated by agriculture, predominantly arable (Gale and Rutter, 2006) (Table 3.4). The long-term hydrograph (Figure 3.9) shows a fairly steady flow with peaks during winter months and a mean discharge of 0.667 m^3s (Table 3.4). All surveys were carried out during summer low flows (Figure 3.10)

Table 3.4: Catchment characteristics of Lowthorpe Beck, data collected from National River Flow Archive (http://www.ceh.ac.uk/data/nrfa/index.html accessed 26/01/2012)

Region	Catchment	Catchment Area	Mean Discharge	Relief	Land Cover	Geology
North East	Hull	57.2km^2	0.667m^3s	6.4mAOD-162.9mAOD	78.2% Arable, 16.8% Grassland, 1.7% Woodland, 0.4% Urban	100% High permeability bedrock

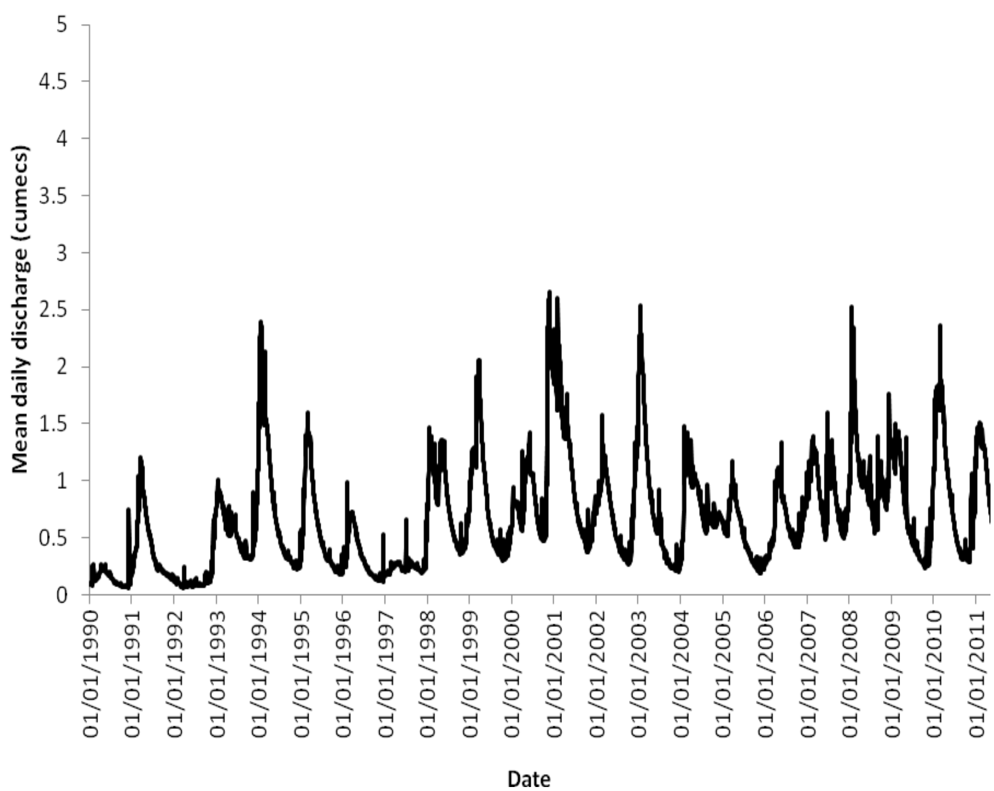


Figure 3.9: Hydrograph of Foston Beck at Foston Mill throughout 1990-2011 (Data source: Environment Agency).

The River Hull Headwaters are within the Humber river basin district, which, is currently classified as poor ecological status and is expected to attain moderate ecological status by 2015 (Environment Agency, 2009b). Lowthorpe Beck is currently classified as poor ecological status with the target of attaining good ecological status by 2027. Lowthorpe beck is designated as heavily modified. Fish as a biological indicator are classified as poor status with the target of achieving good status by 2015. Hydrology is currently classified as moderate with the target of maintaining moderate status.

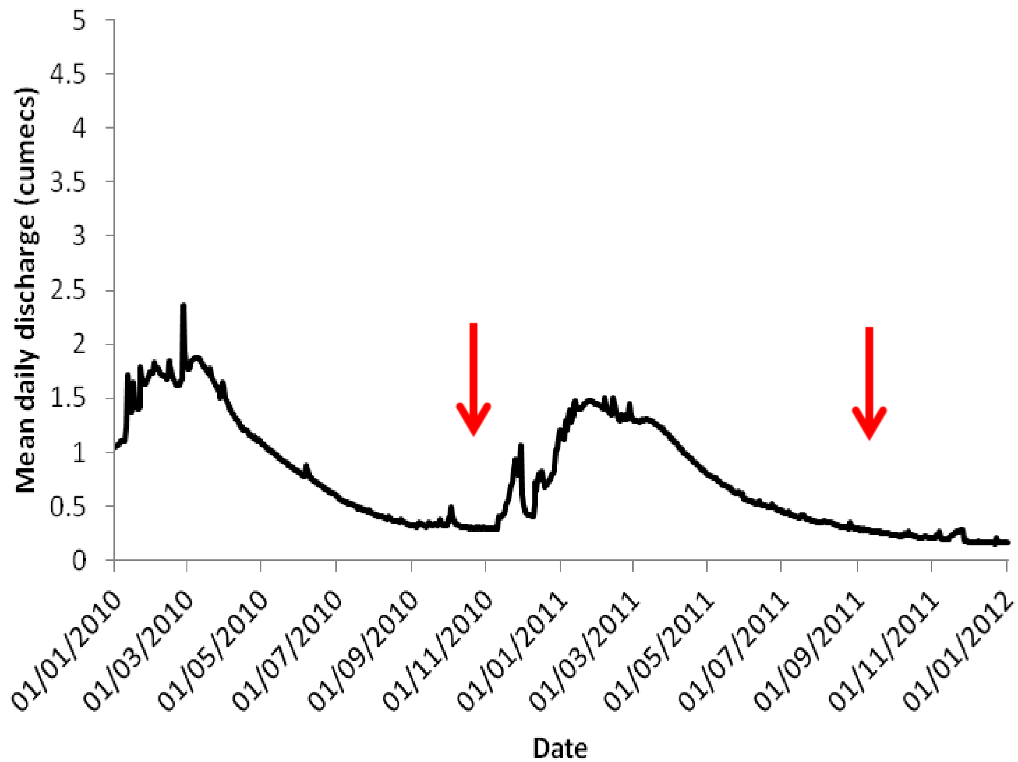


Figure 3.10: Hydrograph of Foston Beck at Foston Mill throughout the survey period. Arrows indicate survey dates (Data source: Environment Agency).

Channel narrowing at Lowthorpe beck aims to address the following pressures and impacts suffered by the River Hull Headwaters:

Accumulation of fine sediment: sedimentation is largely caused by localised land management resulting in large amounts of fine sediments being deposited into the River Hull system either as diffuse supply from land run-off or point supply from field drains and tributaries. Discharges from two local fish farms, absence of buffer zones and livestock poaching riverbanks also increase the input of fine sediments to the watercourse. As a result fine sediment and silt can be observed on the riverbed, particularly in modified areas where river flow is sluggish or impounded (Rayner and Covington, 2009).

Channel modification: sections of the River Hull Headwaters have been modified, straightened and dredged to increase channel capacity for land draining, milling activities, flood defence and navigation, resulting in over deepening, steep banks, disconnection from the flood plain and absence of flow diversity and lack of habitat diversity Rayner and Covington, 2009).

River rehabilitation at Lowthorpe Beck

The East Yorkshire Chalk Rivers Trust carried out river rehabilitation work on a small stretch of Lowthorpe Beck at Harpham (Figure 3.11). The study-site suffered from a very steep bank profile, which limited the habitat available for wetland plant species to colonise in the stream margins. The site also suffered from silt deposition due to “over-engineering” (Alan Mullinger *pers. Comm*.). An over-widened reach of Lowthorpe Beck was identified and measures were undertaken to narrow the channel (Plate 3.9, 3.10 and 3.11). A berm was created on the right hand riverbank using alder cut from a nearby field where it was growing out into the SSSI and damaging the plant community. Alder poles (approximately 20-40 m in diameter and up to 4 m in length) were placed between a double line of stakes and wired down then covered with a coir geotextile to limit silt escape (Plate 3.10). The enclosed area was then back-filled with silt from the streambed (Alan Mullinger *pers. Comm*.).

Specific objectives of the channel narrowing at Lowthorpe Beck were:

- To increase flow velocity through the area to encourage transportation of fine sediment
- To increase availability of cover and bank side habitat

Electric fishing and habitat surveys were carried out, as detailed in section 3.3, on three stretches of Lowthorpe Beck (Figures 3.11 and 3.12) in July 2010, prior to works commencing in July 2010 and post-completion in July 2011. Stretches were selected to represent the site where works took place, a stretch immediately downstream of works and a stretch further downstream.

Plate 3.9: Site LB3 prior to channel narrowing works. Photograph taken May 2010.

Plate 3.10: Coir geotextile used to narrow the channel at Lowthorpe Beck at Harpham, East Yorkshire. Photograph taken February 2011.

Plate 3.11: Site LB3 post channel narrowing works. Photograph taken July 2011.

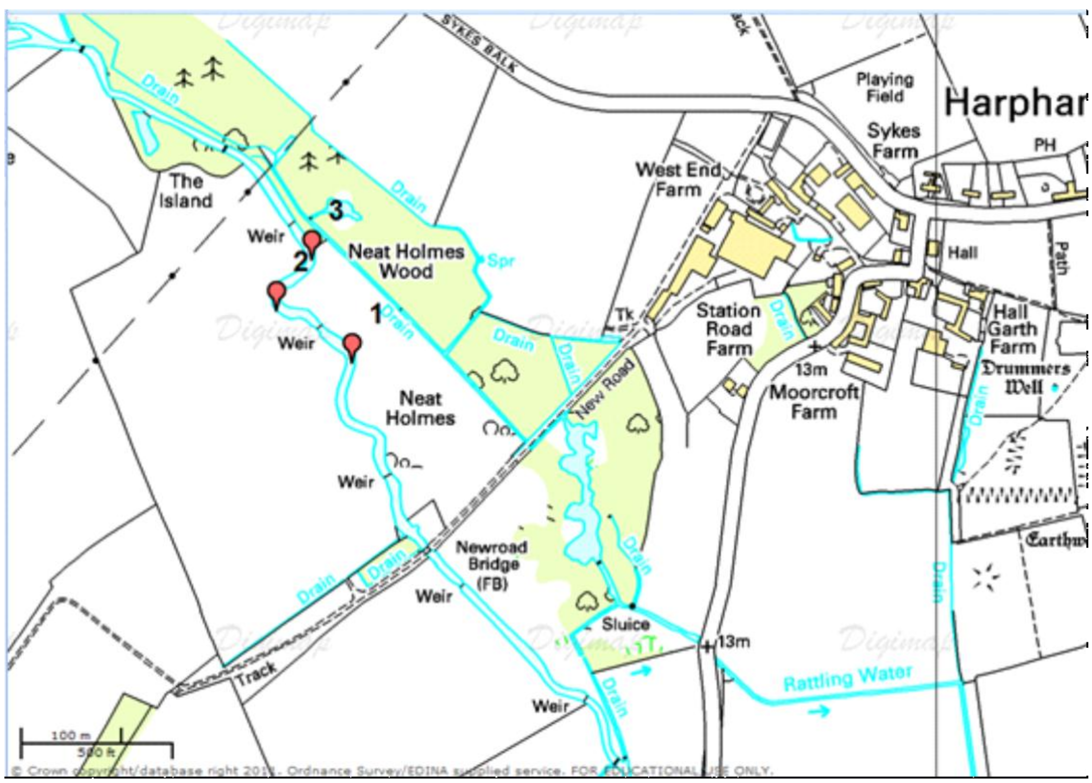


Figure 3.11: Location of survey sites on Lowthorpe beck at Harpham. Red markers indicate site location; Site three is the site of channel narrowing. Map created and modified using Digimap, January 2012.

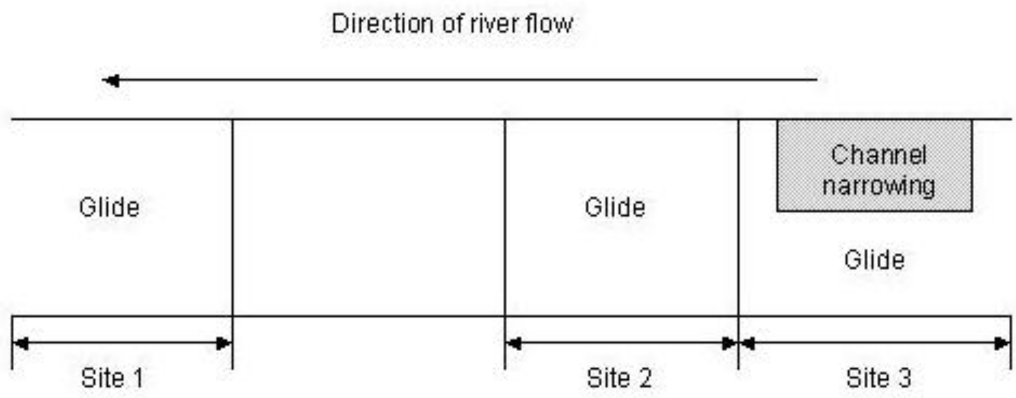


Figure 3.12: Spatial arrangement of surveyed sites at Lowthorpe Beck, Harpham (not to scale).

Sample site details

Site 1: was the most downstream site located 150 m from the channel narrowing works. The site measured 50 m in length and was classified through visual observation as a glide. The flow was deep and steady and the substratum dominated by silt. Canopy cover and riparian vegetation were absent but instream emergent vegetation, *Hippuris vulgaris* (mare tail), *Sparganium* (bur-reed) and *Berula erecta* (lesser-water parsnip) was abundant.

Site 2: was a further 50 m upstream and immediately downstream of the narrowed site. The flow was uniform, deep and steady with a silt-dominated substratum. The site measured 50 m in length and was classified through visual observation as a glide. Canopy cover and riparian vegetation were absent but instream emergent vegetation (as Site 1) was abundant.

Site 3: was the narrowed site. The site was characterised through visual observation as a glide (Plate 3.9 and 3.10) into a deep pool. The deep pool is omitted from later analysis, as it was too deep to survey effectively. Substrate throughout the reach was dominated by silt and the flow was deep and steady. Some instream emergent vegetation (species as above) was present, but canopy cover was lacking despite the presence of riparian trees on the right hand bank.

Sites codes will be used to refer to site number and rehabilitation state in subsequent chapters (Table 3.5).

Table 3.5: Site codes used to detail site number, HMU type and survey date.

Site detail	HMU	State	Date
1a	Glide	Pre	July 2010
1b	Glide	Post	July 2011
2a	Glide	Pre	July 2010
2b	Glide	Post	July 2011
3a	Glide	Pre	July 2010
3b	Glide	Post	July 2011

3.2.2 Installation of gravels in the River Stiffkey, North Norfolk

The River Stiffkey is the most northern river in Norfolk and one of only three Norfolk rivers discharging into the sea. The source of the River Stiffkey is near the village of Croxton; the river flows north for 18 miles to discharge into the North Sea to the East of Wells (Turner, 1954). The River Stiffkey's headwaters flow through wooded hills of glacial debris over chalk and then over low-lying former washlands before entering the sea. Localised land use in the upper regions is woodland and arable, and in the middle and lower reaches is grazing meadow (Pawson, 2008) (Table 3.6).

Table 3.6: Catchment characteristics of River Stiffkey, North Norfolk data collected from National River Flow Archive, (http://www.ceh.ac.uk/data/nrfa/index.html accessed 26/01/2012)

Region	Catchment	Catchment Area	Mean Discharge	Relief	Land Cover	Geology
Anglian	Norfolk rivers group	$87.8km^2$	$0.584m^3s$	5.3mAOD-97.1mAOD	77.9% Arable, 11.7% Grassland, 8.6% Woodland, 0.6% Urban	100% High permeability bedrock

The long-term hydrograph (Figure 3.13) shows a fairly steady hydrograph with some peak flows in the winters of 1993, 2004 and 2007. The hydrograph has a mean discharge of 9.584 m^3s (Table 3.9). All surveys were undertaken during normal summer flows (Figure 3.14).

The River Stiffkey is in the Anglian region, which, is currently classified as poor ecological status, and is expected to achieve moderate ecological status by 2015 (Environment

Agency, 2009a). The River Stiffkey specifically is designated as heavily modified and is currently classified as poor ecological status with the target of achieving good ecological potential by 2027. Fish as a biological quality element are classified as good with the target of remaining good by 2015.

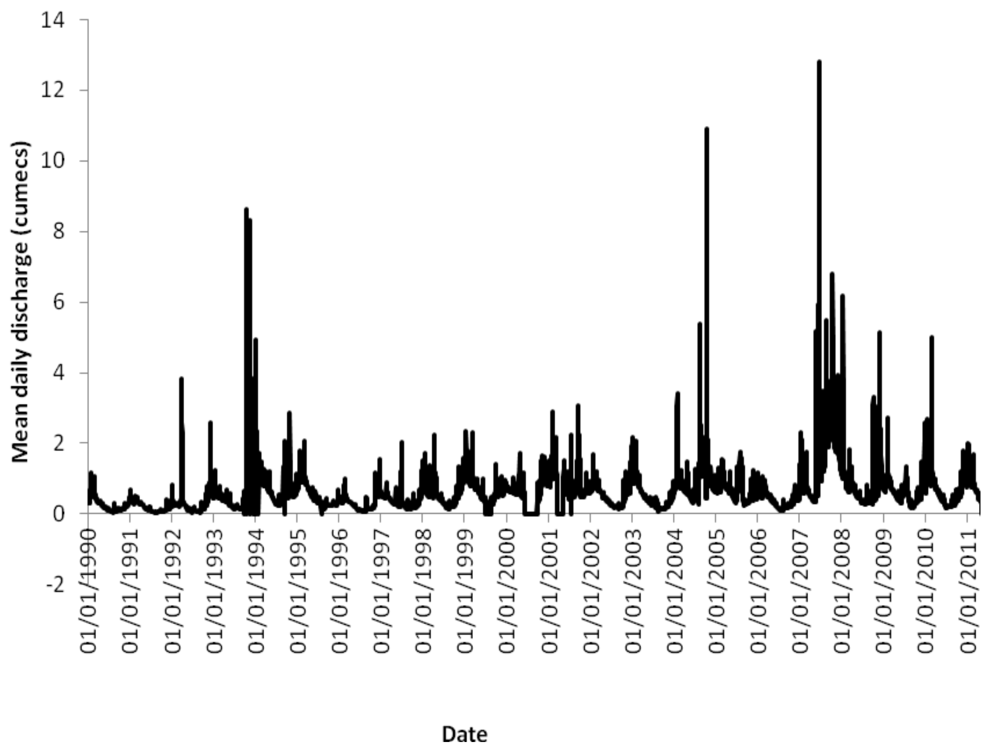


Figure 3.13: Hydrograph of the River Stiffkey, 1990-2011 (Data source: Environment Agency).

The River Stiffkey has suffered physical modification and has been impacted by channelisation, drainage, and clearance of the floodplain and flow regulation by mills, weirs and sluices. Widening and dredging of the river for flood defence during the 1970s and 1980s has led to lowering of the riverbed and loss of the pool-riffle sequence and bank side wetland habitat, and increased siltation (Pawson, 2008) resulting in reduced habitat diversity.

Above the village of Wighton, the River Stiffkey has good spawning and juvenile habitat for brown trout, but below Wighton the river is over-widened and lacking suitable spawning habitat for resident brown trout and sea trout, as bed substrate is dominated by silt and sand (Pawson, 2008, Gill *et al*., 2009). No stocking is recorded as taking place at the River

Stiffkey and there is concern that lack of spawning areas limits recruitment (Pawson, 2008; Gill *et al*., 2009).

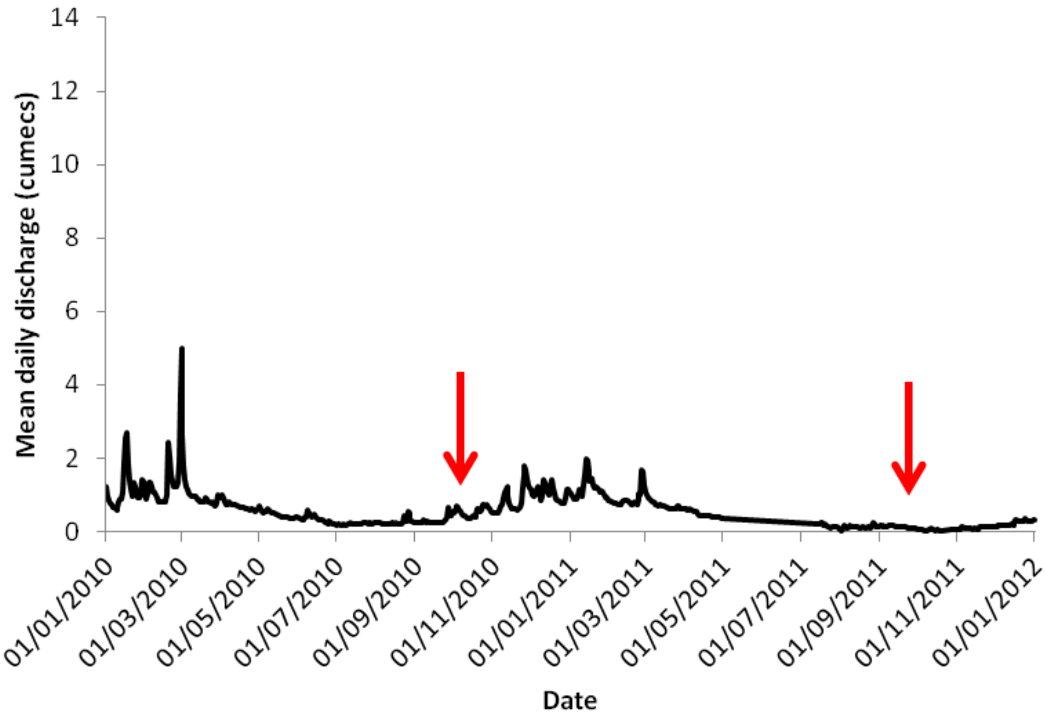


Figure 3.14: Hydrograph of the River Stiffkey throughout the survey period. Arrows indicate survey dates.

Gravel installation works in the River Stiffkey

River rehabilitation works in the River Stiffkey were initiated by the Wild Trout Trust (WTT) (Pawson, 2008) with the primary objective of introducing spawning gravels to increase spawning habitat availability within various river sections (Gill *et al*., 2009). Further management works aimed to create berms, install wooden flow deflectors and install fences to restrict livestock access to improve stability in the riparian zone (Gill *et al*., 2009,). These works aimed to provide shallow areas with increased flow velocity and gravel substrate (riffles), thus increasing spawning habitat availability for trout (Anon).

A gravel installation project was completed at River Stiffkey between 2002-2003 (Gill *et al*., 2009; Anon) when three riffles totalling 85 m in length were installed at Wighton under Environment Agency (EA) Land Drainage Consent Reference AE/2002/00331 (Anon). The installation of these structures was reported to be successful and to support spawning activity each winter since their installation (Anon). Further installation of gravels to the River Stiffkey between Grove Farm, Wighton and Swan's Nest Plantation, downstream of Warham in 2008 aimed to build on this success and further increase the availability of spawning habitat (Anon).

Specific objectives of the gravel installations in the River Stiffkey were:

- To increase the availability of spawning habitat for sea trout and resident brown trout
- To increase habitat diversity and availability
- To increase channel heterogeneity

Surveys were conducted in September 2010 and 2011 and were used to assess variation in hydraulic parameters and fish community composition between HMUs and between survey years.

Sample site details

The fishing and habitat surveys, as detailed in section 3.3 were carried out on four stretches of the River Stiffkey (Table 3.8) (Figure 3.15), each representing a stretch that had a gravel installation, only stretches encompassing a gravel installation were selected for surveying.

Within each stretch individual HMUs were identified through visual observation, these were treated as individual sites (Figure 3.16).

Stretch 1: was the most downstream stretch approximately 20 m upstream of the confluence between the River Stiffkey and Binham stream. It measured approximately 90 m in length (Plate 3.12).

Site 1: was the most downstream glide, located immediately downstream of a 2008 gravel installation. This site had a deep, steady flow and the substrate was dominated by silt.

Site 2: was the most downstream gravel installation. The gravels were installed in 2008. This site had shallow, turbulent flow and the substrate was primarily cobble and gravel.

Site 3: was a glide directly upstream of Site 1. This site had a deep, steady flow and the substrate was dominated by silt.

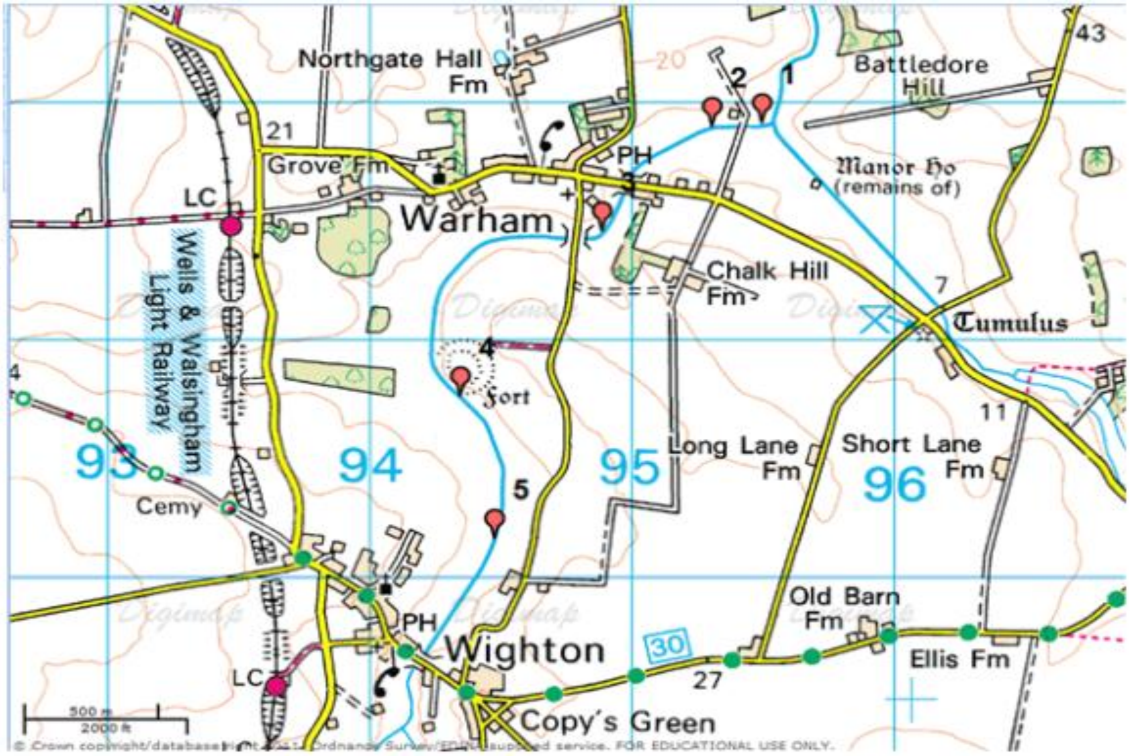


Figure 3.15: Location of survey sites on River Stiffkey. Red circles indicate site location; site four shows the only area of naturally occurring gravels in the reach. Map created and modified using Digimap, January 2012.

Stretch 2: was upstream of Stretch 1, it measured approximately 90 m in length (Plate 3.13).

Site 4: was a 2008 gravel installation. This site had a shallow, turbulent flow and the substrate was mainly cobble and gravel.

Site 5: was a glide directly upstream of Site 4. This site had a deep steady flow and the substrate was dominated by silt.

Stretch 3: was upstream of Stretch 2 located downstream of the road bridge at Warham, it measured approximately 50 m in length (Plate 3.14).

Site 6: was a 2008 gravel installation. This site had a shallow turbulent flow and the substrate was primarily cobble and gravel.

Site 7: was a glide immediately upstream of Site 6. This site had a deep, steady flow and the substrate was dominated by silt.

Stretch 4: was the most upstream stretch, downstream of the village of Wighton, it measured approximately 80 m in length (Plate 3.15).

Site 8: was a 2003 gravel installation. This site had a shallow turbulent flow and the substrate was primarily cobble and gravel.

Site 9: was a glide located between S8 and S10. This site had a deep, steady flow and the substrate was dominated by silt.

Site 10: was a 2003 gravel installation and was the most upstream site surveyed. This site had a shallow turbulent flow and the substrate was primarily cobble and gravel.

At all sites local land use was dominated by agriculture, canopy cover was absent but riparian vegetation was abundant.

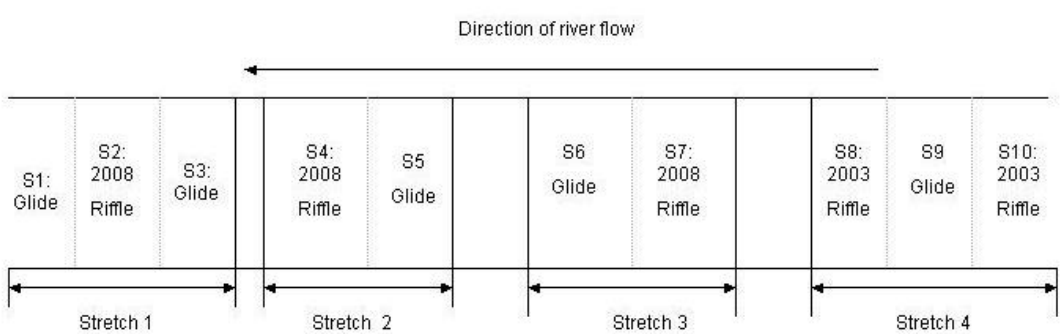


Figure 3.16: Schematic representation of spatial location of surveyed sites at River Stiffkey, North Norfolk (not to scale)

Sites codes will be used to refer to site number and rehabilitation state in subsequent chapters (Table 3.7).

Table 3.7: Site codes used to detail site number, HMU type and survey date.

Site detail	HMU	Date
1a	Glide	September 2010
1b	Glide	September 2011
2a	2008 Riffle	September 2010
2b	2008 Riffle	September 2011
3a	Glide	September 2010
3b	Glide	September 2011
4a	2008 Riffle	September 2010
4b	2008 Riffle	September 2011
5a	Glide	September 2010
5b	Glide	September 2011
6a	2008 Riffle	September 2010
6b	2008 Riffle	September 2011
7a	Glide	September 2010
7b	Glide	September 2011
S8a	2003 Riffle	September 2010
8b	2003 Riffle	September 2011
9a	Glide	September 2010
9b	Glide	September 2011
10a	2003 Riffle	September 2010
10b	Riffle	September 2011

3.3 General Methodology

Wherever possible, sites were surveyed for baseline data prior to rehabilitation works. The exception was the River Stiffkey, North Norfolk, where river rehabilitation works had been completed over two-phases in 2003 and 2008. In this case, sites of gravel installation were monitored according to their date of installation and data were collected in 2010 and 2011, to assess any temporal change.

Electric fishing surveys were used to collect data on the number and composition of fish species present at each site. This method was used, as it is efficient at ensuring as many fish as possible are captured to give a representation of the fish present within the survey reach. Sampling took place at various dates between July and early October in 2010 and 2011 (Table 3.8) to increase the likelihood of catching 0+ trout during

Plate 3.12: The River Stiffkey Stretch 1 September 2010

Plate 3.13: The River Stiffkey Stretch 2 September 2010

Plate 3.14: The River Stiffkey Stretch 3 September 2010

Plate 3.15: The River Stiffkey Stretch 4 September 2010

Table 3.8: Details of Survey Rivers, Sites and dates.

River name	River rehabilitation work	Number of sites monitored	Site number	NGR	HMU type	Spatial location	Date of pre survey	Date of post survey	Date of further post survey
Dove, Dovedale	Small-scale Weir Removal	4	DD1	SK144 526	Riffle	Begins 100m downstream weir, extending 10m upstream	01/07/2010	21/07/2010	06/07/2011
			DD2	SK144 526	Glide	Immediately upstream of Site 1, extending upstream to weir	01/07/2010	21/07/2010	06/07/2011
			DD3	SK143 526	Glide	Begins at weir and extends 40m upstream	01/07/2010	22/07/2010	07/07/2011
			DD4	SK143 526	Riffle	Immediately upstream of Site 3, extending 50 m upstream	01/07/2010	22/07/2010	07/07/2011

River name	River rehabilitation work	Number of sites monitored	Site number	NGR	HMU type	Spatial location	Date of pre survey	Date of post survey	Date of further post survey
Dove, Hartington	Small-scale Weir Removal	2	HD1	SK124 596	Glide	100 m downstream - weir	27/07/2010	28/09/2010	27/07/2011
			HD2	SK125 595	Glide	Weir to 100 m upstream	28/07/2010	29/09/2010	27/07/2011
Lowthorpe Beck, Harpham, East Yorkshire	Channel Narrowing	3	LB1	TA083 615	Glide	100 m downstream of channel narrowing	23/07/2010	12/07/2011	n/a
			LB2	TA083 616	Glide	50 m downstream of channel narrowing	23/07/2010	12/07/2011	n/a
			LB3	TA083 617	Glide	Site encompasses channel narrowing work	23/07/2010	12/07/2011	n/a

River name	River rehabilitation work	Number of sites monitored	Site number	NGR	HMU type	Spatial location	Date of pre survey	Date of post survey	Date of further post survey
Stiffkey, North Norfolk	Installation of gravels to the stream bed	10	S1	TF954419	Glide	Most downstream site	13/ 09/2010	27/09/2010	n/a
			S2	TF954419	Riffle	Upstream of S1, 2008 gravel	13/ 09/2010	27/09/2010	n/a
			S3	TF954419	Glide	Upstream of S2	13/ 09/2010	27/09/2010	n/a
			S4	TF953419	Riffle	Downstream of S5, 2008 gravel	14/09/2010	27/09/2010	n/a
			S5	TF953419	Glide	Upstream of S4	14/09/2010	27/09/2010	n/a
			S6	TF948414	Riffle	Downstream of S7, 2008 gravel	16/09/2010	28/09/2010	n/a
			S7	TF948414	Glide	Upstream of S6	16/09/2010	28/09/2010	n/a
			S8	TF944401	Riffle	Downstream of S9, 2003 gravel	15/09/2010	28/09/2010	n/a
			S9	TF944401	Glide	Between S8 and S10	15/09/2010	28/09/2010	n/a
			S10	TF944401	Riffle	Upstream S9, 2003 gravel	15/09/2010	28/09/2010	n/a

the electric fishing process, as it is generally accepted that fish are more easily caught as body length increases (Zalewski and Cowx, 1990).

Wherever possible, surveys were conducted on the same dates each year to maintain consistency in data collection.

Quantitative electric fishing was carried out using the same method for each survey: estimates of absolute abundance based on a three-catch removal method (Carle and Strub, 1978). Three operatives (one anode operator and two netsmen) fished the stretch wading in an upstream direction. A fourth operator was located on the bank supervising safe operation of the electric fishing equipment. The equipment comprised a 2kVA generator powering an Electracatch control box producing a 220 V DC output. Each survey stretch was isolated by the positioning of stop nets at the top and bottom of the stretch to prevent fish migrating into or out of the survey area. Each stretch was fished three times with the catch from each being kept separate. During the fishing exercise, as many fish as possible were caught in dip nets by the operatives positioned either side of the anode.

Prior to commencement of the fishing exercise individual HMUs (riffle and glide) within the stretch were identified. During the first run of the fishing exercise operatives observed the start and end of each HMU and placed fish captured from each unit into representative buckets positioned on the bank. Although this is not standard methodology, employing this method allowed details of the fish captured in each HMU type to be recorded. Stop nets were not used to isolate individual HMUs as it was assumed the assembly of stop nets would disturb fish location. This method was employed during the first run only and subsequent runs were fished continuously throughout as it was assumed that the fish would be disturbed and the location of capture would no longer represent fish location thus only first run data were used in analysis. Triple-run data was used to calculate total population density in the whole reach for application of the HABSCORE model (see later section 3.3.1).

Following each survey, individual fish were identified to species level, fork length (mm) was measured and scale samples were removed for ageing purposes (using the appropriate Environment Agency Management System (Britton, 2003), fish were then returned to the river.

On completion of electric fishing surveys, habitat surveys were conducted to give a detailed description of the environmental characteristics influencing fish captured in the reach. On every occasion this was carried out after electric fishing surveys were completed to minimise pre-survey disturbance to the reach. Collection of physical habitat data was based around the completion of the HABform (see Appendix 1), a pre-prepared form used for capturing environmental data used in the salmonid habitat model HABSCORE. This method of data collection was chosen because HABSCORE is a widely accepted, standard method of habitat data collection used in salmonid streams. Additional significant habitat variables and methods of measurement were decided upon by reviewing key texts (Gordon *et al*., 1994 and Bain and Stevenson, 1999).

In the first instance, a description of the survey stretch was recorded using a basic reconnaissance method. The length of each survey stretch was measured; the dominant land use on both sides of the channel recorded and percentage of riparian cover visually estimated and recorded on the HABform (see Appendix 1). The number and type of HMUs within the reach were identified visually, using the descriptions in Chapter 2, measured, and length (m) and width (m) recorded on pre-prepared forms. The presence of vegetation was also recorded as canopy (shading by trees), riparian (growing on the river banks) and instream (submerged and emergent).

Working in an upstream direction from the downstream extent of the reach, a lateral transect of habitat variables was measured every 10 m. For each transect, the channel width (m) was measured using a tape measure. Depth (m) measurements were taken using a depth pole (readings taken of downstream side to minimise disruption by breaking water) at three points along each transect to represent ¼, ½, ¾ channel width from left to right facing upstream. Although not part of the HABform or HABSCORE model, flow velocity (m/sec) at 60% depth (average column velocity) was recorded (Valeport model 801 electromagnetic flow metre, 10 second average (m/sec) was employed in surveys prior to September 2010, after this date a Valeport propeller metre was used) concurrently to depth measurements.

Photographs of the reach were taken to allow future site recognition and for permanent visual records of the survey reach.

3.3.1 Data processing

Following data collection, data from electric fishing surveys were processed using the following methods.

Scales were read under a low powered Microfische. The number of annuli; irregular, closely spaced rings, cutting over previously laid down rings on an annular basis; were counted and measurements taken of the distance from the focus of the scale to each annulus and the total scale radius (Bagenal and Tesch, 1978). A quality control procedure was followed, as described in Musk *et al.* (2006) to minimise error. Ten percent of scales were checked by a secondary reader and in the event of a discrepancy; the scale was reviewed until consensus was reached. When the primary reader found difficulty in ageing a second opinion was sought outside of the quality control procedure.

Age data was used to assign brown trout into age categories (0+, >1+). For analysis of community composition (Chapter 5), 0+ and >1+ brown trout were considered as separate species due to the importance of young-of-the-year in community composition and as an indicator of recruitment to the brown trout population. The densities of both 0+ and >1+ brown trout were calculated using the equation:

$$\text{Density} = \frac{\text{number of individuals captured}}{\text{site area}} \times 100$$

Additionally, density estimates of 0+, ≥1+ (<20 cm) and ≥1+ (>20 cm) brown trout /100 m^2 for use within the HABSCORE model (Chapter 6) were derived from absolute abundance estimates determined from the Carle and Strub (1978) removal method. In all cases the population density at each site was expressed as N/100 m^2.

3.3.2 Data analysis

Data were analysed using Microsoft excel, SPSS version 19, PRIMER-E (Plymouth Routines in Multivariate Ecological research) version 6, BRODGAR and HABSCORE for windows. Data analysis was approached using different methods to meet the project objectives stated in Chapter 1. Each chapter relates to one of the key objectives.

Chapter 4

Data were analysed to determine the physical outcomes of river rehabilitation in terms of instream hydraulics. Differences between instream hydraulics and physical conditions were investigated within sites before and after river rehabilitation; and in heterogeneous channels differences in hydraulic conditions between HMUs were investigated before and after river rehabilitation. In the case of the River Stiffkey, differences were investigated between survey dates.

Following data collection, Froude number was calculated for each point where depth (m) and flow velocity (m/sec) was measured using the equation:

$$Fr = V/(gD)1/2$$

Where *Fr* = Froude number, *V*=velocity (m/s), g = acceleration due to gravity (9.8 m/s^2), *D* = depth (m)

Froude number was calculated because it is a potentially useful variable for describing habitat for aquatic organisms (Moir *et al*., 2002) and has previously been found to be a key factor in discriminating between hydraulic biotopes (Kemp *et al*., 2000).

Mean depth (m), mean flow velocity (m/sec) (at 60% depth), mean Froude number and mean width (m) were calculated for each site on all survey occasions, as these are the basic foundation of fish habitat assessment (Gordon *et al*., 1994, Bain and Stevenson, 1999), using Microsoft Excel. Means plots of these values were plotted using SPSS version 19 to assess change in hydraulic variables following river rehabilitation, or between years at the River Stiffkey.

Standard deviation of depth and flow velocity was also calculated in Microsoft Excel, these were calculated to assess the degree of variation from the mean of depth and flow velocity within a site.

In order to test for statistically significant differences in hydraulic variables at each site following river rehabilitation, or between years at the River Stiffkey, raw data sets were tested for normality using a Kolmogorov-Smirnov test and a Levene's test was applied to test homogeneity of variance. All normally distributed data were subjected to Analysis of Variance (one-way ANOVA); a non-parametric alternative (Mann-Whitney U or Kruskall-Wallis test) was applied where data were not evenly distributed or homogeneity of variance was not equal. Where appropriate the least significant difference (LSD) *post hoc* test was applied.

In heterogeneous channels (River Dove at Dovedale and River Stiffkey), differences in measured and derived hydraulic variables were assessed between different HMUs in each river stretch at each river rehabilitation stage. Following the Kolmogorov-Smirnov test for normality and the Levene's test for homogeneity, one-way ANOVA was used on normally distributed data. Where data was not normally distributed, a non-parametric alternative (Mann-Whitney U or Kruskall-Wallis) test was applied. Where appropriate the least significant difference (LSD) *post hoc* test was applied. All data analysis was carried out in SPSS version 19.

Mean depth (m), mean flow velocity (m/sec), standard deviation of depth, standard deviation of flow velocity, mean width (m) and Froude number were used to construct similarity matrices using Euclidean distance to investigate differences between sites according to river rehabilitation state and HMU type. These were displayed as Multi Dimensional Scaling (MDS) plots using PRIMER (Plymouth Routines in Multivariate Ecological research) version 6. Clusters were overlaid to show similarity between groups of sites.

Principal component analysis (PCA) was carried out on the above hydraulic variables to investigate relationships between variables. Site numbers and HMU type were overlaid on the PCA plot to assess the strength of relationship between each site and HMU type with hydraulic variables. PCA analysis was carried out in PRIMER, version 6.

Chapter 5

Data were analysed to determine the response of fish community composition to river rehabilitation. Fish community composition was investigated within sites before and after river rehabilitation. In heterogeneous channels, fish community composition between HMUs was investigated before and after river rehabilitation. Data from the River Stiffkey was used to investigate differences in fish community composition within and between sites according to survey date.

Analysis relating to fish community composition at each site was conducted on first run data only as it was assumed fish would be disturbed after the first run and location of capture may no longer represent habitat occupancy. General analytical methods were applied to each sample as described below.

For each site on each sampling occasion, the number of fish captured was summed and the percentage composition of each species calculated and displayed as stacked histograms of species composition.

Fish community structure at each site on each sampling occasion was described using measures of species diversity, evenness of species composition, total fish density, species richness, population density of 0+ brown trout and population density of >1+ brown trout.

Species diversity was calculated using the Shannon-Weiner diversity index (*H'*), which is the most widely used measure of diversity in aquatic ecology. This was calculated using the equation:

$$\boldsymbol{H' = -\Sigma\, P_1 \ln P_1}$$

Where $\boldsymbol{P_i}$ = the proportion of individuals of species in the community.

Evenness was calculated using Pielou's measure of evenness (*J*) (Marshall and Elliott, 1997):

$$\boldsymbol{J = H'/H'_{max}}$$

Where H= Shannon-Weiner diversity index and H'_{max} = the maximum value of H'.

Species richness is the sum of the total number of different species collected in the sample and density was calculated as the number of individuals (total or specific species) captured over $100m^2$, as described above in section 3.3.

Graphs of diversity, evenness, richness and density at each site at each stage of river rehabilitation were plotted using Microsoft Excel.

To examine similarities in catch composition within and between sites according to habitat type and rehabilitation stage, Bray-Curtis similarity matrices, which calculate the compositional dissimilarity of sites based upon counts at each site, were calculated using percentage species composition at each site at each rehabilitation state and presented as multi-dimensional scaling (MDS) plots. Analysis was carried out in PRIMER-E (Plymouth Routines in Multivariate ecological Research) version 6.

Chapter 6

Data were analysed to assess fish community structure and linkages with in stream hydraulics and physical habitat. Habitat availability for and use by brown trout was investigated at each river stretch using the HABSCORE model. The influence of hydraulic and physical habitat variables, and habitat use by each species was investigated and the relationship between hydraulic parameters and descriptors of community composition were investigated.

Density data for brown trout (0+, >1+ <20 cm and >1+ >20 cm) were processed through the HABSCORE model to obtain habitat quality scores (HQS) and habitat utilisation scores (HUI) for each site at each stage of river rehabilitation. Observed habitat use (expressed as density) was compared with the expected density under pristine conditions (HQS) and upper and lower HUI confidence intervals were used to determine if differences were significant.

The composition of fish species was calculated as a percentage of the total catch as detailed above. Canonical correspondence analysis (CCA) was carried out in BRODGAR version 2.7.2 to assess the influence of each of the environmental variables depth (m), standard deviation of depth, flow velocity (m/sec) (at 60% depth), standard deviation of flow velocity, width (m) and Froude number on each of the fish species captured. The percentage abundance of minor species minnow (*Phoxinus phoxinus*), and nine-spined stickleback (*Pungitius pungitius*) were removed from analysis as they were captured in very low abundance at one site only throughout surveys.

Relationships between fish community descriptors (Shannon-Weiner diversity index, evenness of species distribution, total species richness, total fish density, density of 0+ brown trout and density of >1+ brown trout as detailed above) and environmental parameters (mean flow velocity (m/sec) (at 60% channel depth), mean depth (m), mean width (m) and Froude number) were tested for normality using a Kolmogorov-Smirnov test. Not all data were normally distributed therefore non-parametric Spearman rank correlation test was used to investigate correlations between environmental parameters and fish community descriptors. Data analysis was conducted in SPSS version 19.

Data from first-run electric fishing was used to quantify the number of fish of each species captured in each site. Habitat parameters, mean flow velocity (m/sec), mean depth (m) and Froude number were calculated for each habitat unit as detailed in section 3.3. Percentage frequency histograms were constructed to display the range of habitat conditions utilised and infer habitat preferences of the species captured. Histograms were created in Microsoft Excel.

4 CHAPTER FOUR: DETERMINING THE PHYSICAL OUTCOMES OF RIVER REHABILITATION IN TERMS OF INSTREAM HYDRAULICS

4.1 Results

4.1.1 Small-scale weir removal from the River Dove at Dovedale

Differences in hydraulic variables within sites following weir removal from the River Dove at Dovedale

Following removal of the small rock weir from the River Dove at Dovedale, the depth in the upstream reaches visually appeared to be lower, specifically at Site 4b and Site 4c (Plate 3.4). Measurements of mean water depth in Sites 3a, 3b and 3c and 4a, 4b and 4c (Table 4.1, Figure 4.1) proved the depth did decrease in the upstream sites following weir removal. However, depth was significantly lower only in Site 4. The statistically significant change in depth was between Site 4a and Site 4c (Table 4.1). Depth did not change significantly in downstream sites (Sites 1a, 1b and 1c and 2a, 2b and 2c) (Table 4.1).

After weir removal, flow velocity at Site 4 (4b and 4c) increased significantly (Table 4.2, Figure 4.2), between Sites 4a and 4c and 4b and 4c, respectively. Flow velocity at Sites 2 and 3 also increased significantly following weir removal (Table 4.2, Figure 4.2), between 2a and 2c and 2b and 2c; and 3a and 3c and 3b and 3c, respectively.

There was a significant increase in Froude number at Site 4 (Table 4.3, Figure 4.3), between Sites 4a and 4c and 4b and 4c, respectively. Froude number also increased significantly at Sites 2 and 3. The increase was between Sites 2a and 2c and 2b and 2c; and 3a and 3c, respectively (Table 4.3).

No statistically significant change in channel width was recorded following weir removal at all sites (Table 4.4, Figure 4.4).

Table 4.1: Differences in mean depth recorded at each site in the River Dove at Dovedale (values highlighted in yellow indicate statistical significance).

Site	Depth (m)			Statistical test and significance	Post *hoc.* tests	
	Mean	SD	*n*			
DD1a	0.26	0.08	6	ANOVA:		
DD1b	0.34	0.15	6	F = 1.487,	n/a	
DD1c	0.24	0.08	6	$p = 0.257$		
DD2a	0.48	0.15	30	ANOVA:		
DD2b	0.49	0.14	27	F = 2.304,	n/a	
DD2c	0.40	0.17	27	$p = 0.106$		
DD3a	0.74	0.26	9	ANOVA:		
DD3b	0.55	0.21	12	F = 2.613,	n/a	
DD3c	0.49	0.26	9	$p = 0.092$		
DD4a	0.39	0.78	21	ANOVA:	4a/4b	0.097
DD4b	0.29	0.14	21	F = 4.083,	4a/4c	0.006
DD4c	0.23	0.15	24	$p = 0.022$	4b/4c	0.273

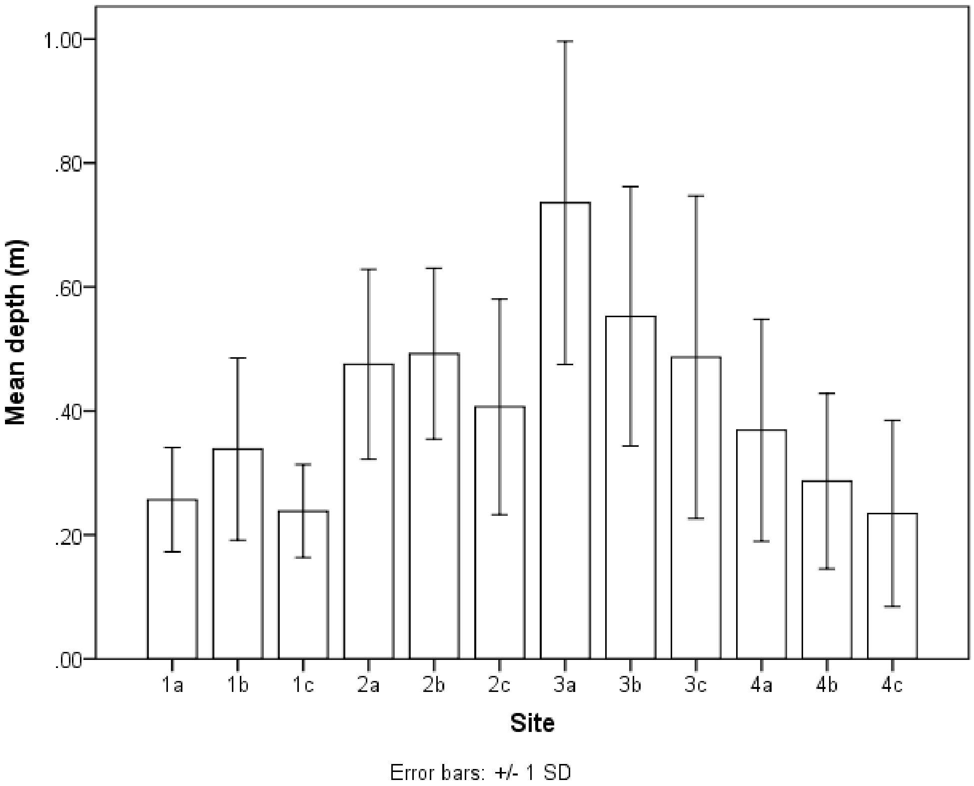


Figure 4.1: Mean depth at each site in the River Dove at Dovedale.

Table 4.2: Differences in mean flow velocity recorded at each site in the River Dove at Dovedale (values highlighted in yellow indicate statistical significance).

Site	Flow velocity (m/sec)			Statistical test and significance	Post *hoc.* tests	
	Mean	SD	*n*			
DD1a	0.36	0.36	4	Kruskall – Wallis: p = 0.344	n/a	
DD1b	0.69	0.26	5			
DD1c	3.48	4.03	6			
DD2a	0.37	0.18	21	Kruskall - Wallis: *p* = 0.000	2a/2b	0.480
DD2b	0.35	0.24	26		2a/2c	0.000
DD2c	2.1	1.68	26		2b/2c	0.000
DD3a	0.10	0.09	7	Kruskall – Wallis: p = 0.007	3a/3b	0.076
DD3b	0.29	0.27	12		3a/3c	0.013
DD3c	1.87	1.72	9		3b/3c	0.016
DD4a	0.33	0.27	20	Kruskall – Wallis: p = 0.006	4a/4b	0.092
DD4b	0.46	0.23	16		4a/4c	0.006
DD4c	2.54	2.15	14		4b/4c	0.011

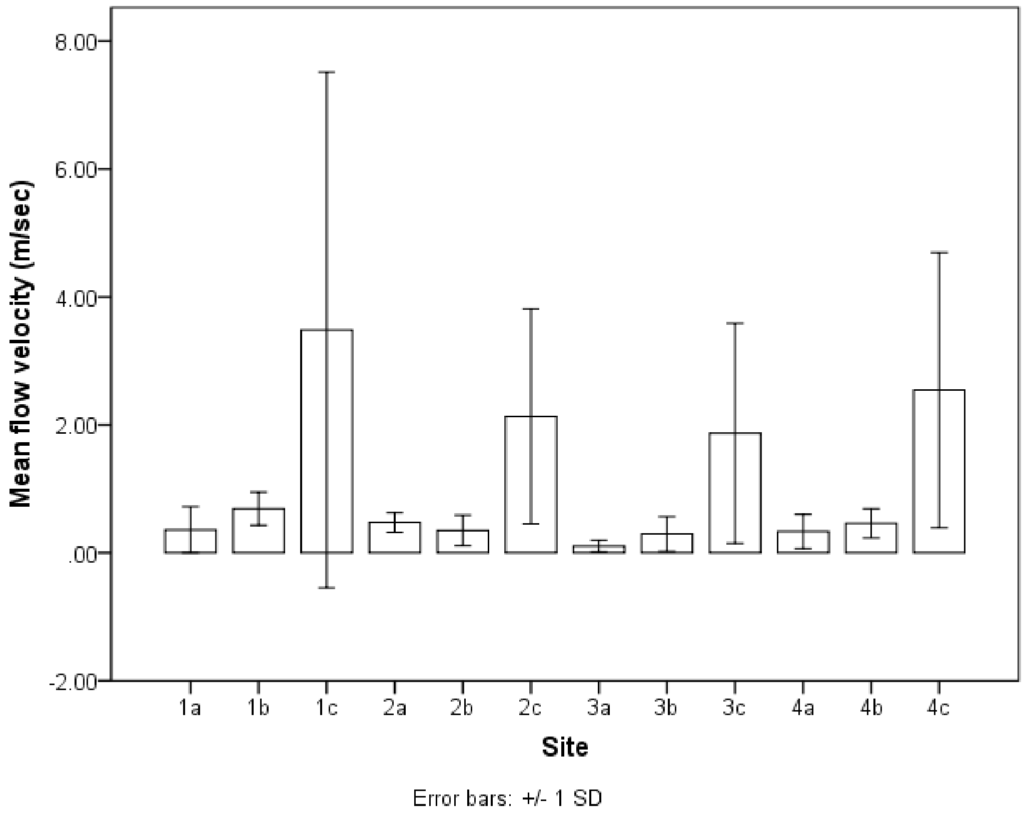


Figure 4.2: Mean flow velocity at each site in the River Dove at Dovedale.

Table 4.3: Differences in mean Froude number recorded at each site in the River Dove at Dovedale (values highlighted in yellow indicate statistical significance).

Site	**Froude number**			**Statistical test and significance**	**Post *hoc.* tests**	
	Mean	SD	*n*			
DD1a	0.066631	0.05	4	Kruskall – Wallis: p = 0.284	n/a	
DD1b	0.090088	0.029289	5			
DD1c	0.640799	0.741281	6			
DD2a	0.047681	0.028641	22	Kruskall - Wallis: *p* = 0.000	2a/2b	0.222
DD2b	0.045307	0.05352	26		2a/2c	0.000
DD2c	0.353735	0.406292	26		2b/2c	0.000
DD3a	0.009787	0.009852	7	Kruskall - Wallis: *p* = 0.031	3a/3b	0.176
DD3b	0.049628	0.068983	12		3a/3c	0.017
DD3c	0.393725	0.235692	9		3b/3c	0.076
DD4a	0.064849	0.06464	20	Kruskall - Wallis: *p* = 0.009	4a/4b	0.278
DD4b	0.076645	0.038714	9		4a/4c	0.005
DD4c	0.705546	0.692023	14		4b/4c	0.032

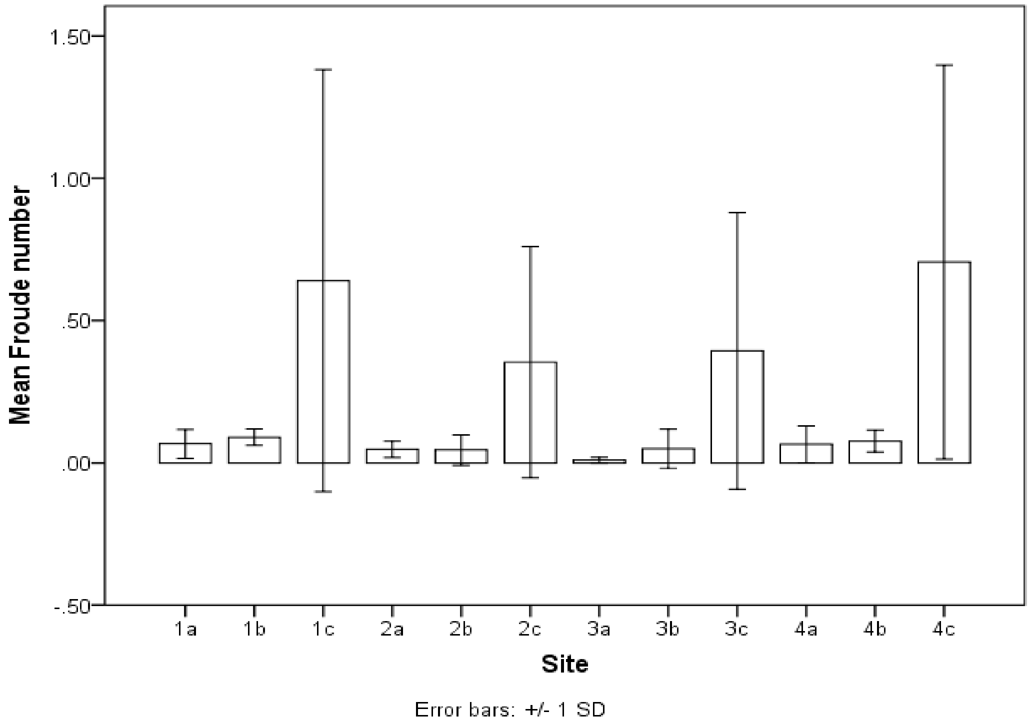


Figure 4.3: Mean flow velocity at each site in the River Dove at Dovedale.

Table 4.4: Differences in mean width recorded at each site in the River Dove at Dovedale (values highlighted in yellow indicate statistical significance).

Site	Width			Statistical test and significance	Post *hoc.* tests
	Mean	SD	*n*		
DD1a **DD1b** **DD1c**	8 8.7 7.8	0.21 0.14 0.35	2 2 2	ANOVA: F = 7.921, p = 0.064	n/a
DD2a **DD2b** **DD2c**	7 8 8	1.6 1.4 1.21	10 9 9	ANOVA: F = 0.143, p = 0.868	n/a
DD3a **DD3b** **DD3c**	8.3 8.7 8	0.58 0.62 1.63	3 4 3	ANOVA: F = 0.234, p = 0.798	n/a
DD4a **DD4b** **DD4c**	8.4 9 8.4	3.13 3.53 1.73	7 7 7	ANOVA: F = 0.107, p = 0.899	n/a

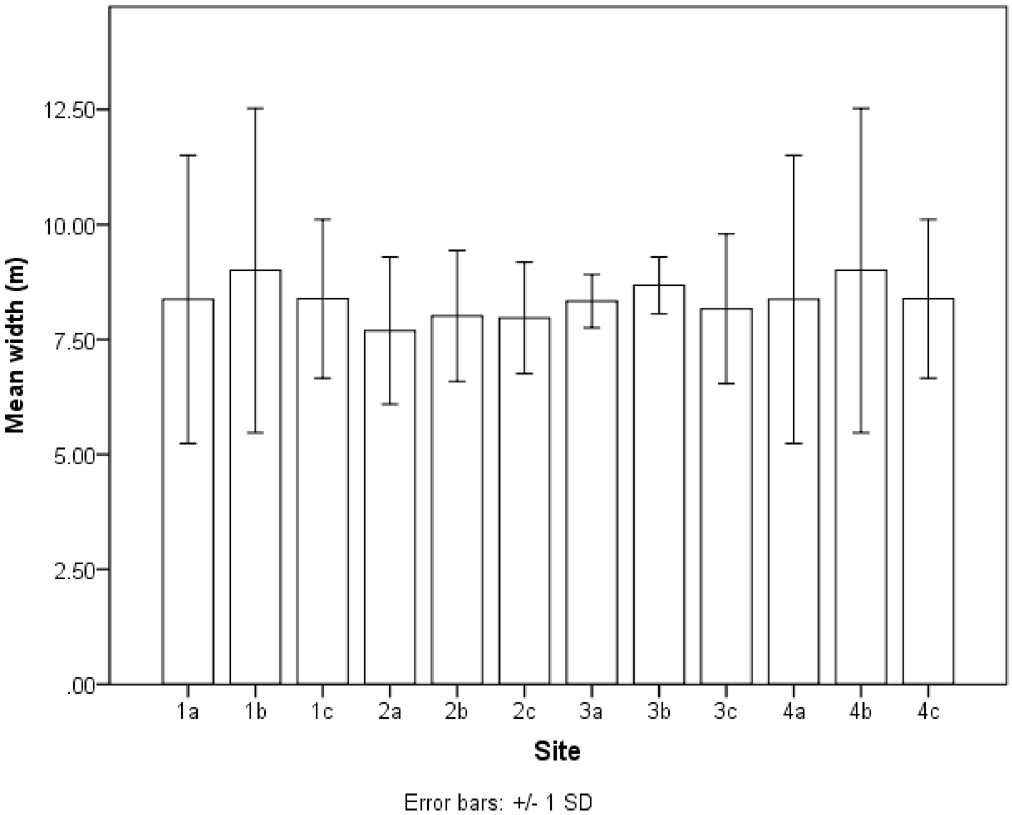


Figure 4.4: Mean width at each site in the River Dove at Dovedale.

Differences in hydraulic variables between HMUs following weir removal from the River Dove at Dovedale

Downstream of the weir, depth was significantly greater in Site 2 than Site 1 during all surveys. Upstream of the weir, depth was significantly higher in Site 3 than Site 4 during all surveys (Table 4.5).

Following weir removal, flow velocity was significantly higher in Site 1b than Site 2b (Table 4.6). Before weir removal, flow velocity was significantly higher in Site 4a than Site 3a (Table 4.6). At all other stages, there was no significant difference in flow velocity between sites.

There was no significant difference in Froude number between sites downstream of the weir during any of the surveys. Upstream of the weir, Froude number was significantly different between Site 3a and Site 4a. Following weir removal, there was no statistically significant difference in Froude number between sites upstream of the weir (Table 4.7).

There was no significant difference in width between sites (Table 4.8) during all surveys.

Table 4.5: Differences in mean depth between HMUs in the River Dove at Dovedale (values highlighted in yellow indicate statistical significance).

State	Depth (m) at site		Statistical test and significance
	DD1	**DD2**	
a	0.26	0.48	ANOVA: F = 11.351, *p* = 0.002
b	0.34	0.49	ANOVA: F = 5.963, *p* = 0.021
c	0.24	0.40	Mann-Whitney U: *p* = 0.025
	DD3	**DD4**	
a	0.74	0.39	ANOVA: F = 20.037, *p* = 0.000
b	0.55	0.29	ANOVA: F = 18.944, *p* = 0.000
c	0.49	0.23	Mann Whitney U: *p*= 0.010

Table 4.6: Differences in mean flow velocity between HMUs in the River Dove at Dovedale (values highlighted in yellow indicate statistical significance).

State	Flow velocity (m/sec) at site		Statistical test and significance
	DD1	**DD2**	
a	0.36	0.37	ANOVA: F = 0.004, *p* = 0.950
b	0.69	0.35	ANOVA: F = 8.318, *p* = 0.007
c	3.48	2.1	Mann Whitney U: *p*=0.865
	DD3	**DD4**	
a	0.10	0.33	Mann Whitney U: *p*= 0.036
b	0.29	0.46	Mann Whitney U: *p*=0.088
c	1.87	2.54	ANOVA: F = 0.627, *p* = 0.437

Table 4.7: Differences in mean Froude number between HMUs in the River Dove at Dovedale (values highlighted in yellow indicate statistical significance).

State	Froude number at site		Statistical test and significance
	DD1	**DD2**	
a	0.066631	0.047681	ANOVA: F = 1.169, *p* = 0.290
b	0.090088	0.045307	ANOVA: F = 3.250, *p* = 0.082
c	0.640799	0.353735	Mann Whitney U: *p*=0.717
	DD3	**DD4**	
a	0.009787	0.06464	Mann Whitney U: *p*=0.013
b	0.049628	0.038714	ANOVA: F = 1.109, *p* = 0.306
c	0.393725	0.692023	ANOVA: F = 1.379, *p* = 0.253

Table 4.8: Differences in mean width between HMUs in the River Dove at Dovedale (values highlighted in yellow indicate statistical significance).

State	Width (m) at site		Statistical test and significance
	DD1	**DD2**	
a	8	7	ANOVA: F = 0.049, *p* = 0.830
b	8.7	8	ANOVA: F = 0.430, *p* = 0.529
c	7.8	8	ANOVA: F = 0.058 *p* = 0.815
	DD3	**DD4**	
a	8.3	8.4	ANOVA: F = 0 *p* = 0.984
b	8.7	9	ANOVA: F = 0.032 *p* = 0.862
c	8	8.4	ANOVA: F = 0.035 *p* = 0.857

Similarity between sites in the River Dove at Dovedale

All sites labelled a and b (2010) were similar (Euc. 3.5). Sites 1b, 3b, 4a and 4b were similar (Euc. 2.5) and Sites 2a and 2b were very similar (Euc. 1.5). Sites 1a and 3a were least similar to other sites. Sites 2c, 3c and and 4c were similar (Euc. 3.5). Sites 2c and 3c were very similar. Site 1c was least similar to other sites (Figure 4.5). Sites 1a, 1b and 4b were negatively associated with mean depth and standard deviation of depth. Sites 2a, 2b, 3a, 3b and 4a were positively associated with mean depth and standard deviation of depth. Sites 1c, 2c, 3 c and 4c were positively associated with high mean flow velocity, standard deviation of flow velocity and Froude number (Figure 4.6).

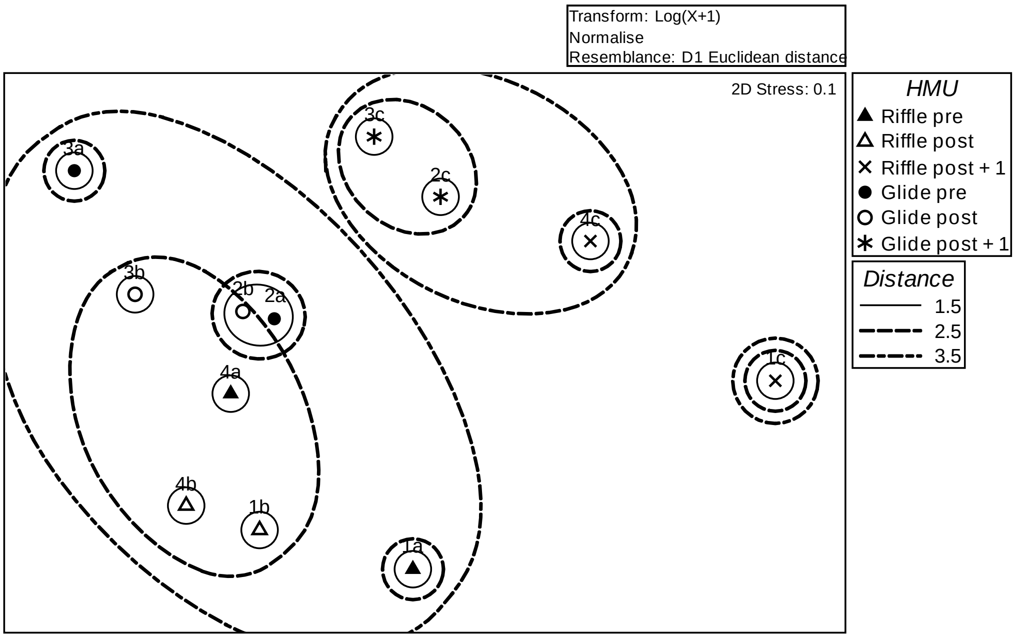


Figure 4.5: MDS plot of similarity between sites in the River Dove at Dovedale.

4.1.2 Small-scale weir removal from the River Dove at Hartington

Depth was significantly lower after weir removal at both downstream (Site 1b and 1c) and upstream (Site 2b and 2c) sites (Table 4.9, Figure 4.8). Downstream, depth was significantly lower in Site 1b than 1a and Site 1c than 1b. Upstream; depth was significantly lower in Site 1b than 1a and 1c than 1a (Table 4.9, Figure 4.7).

Flow velocity downstream (Site 1) was significantly higher following weir removal. Flow velocity was significantly higher in Site 1b than 1a, 1c than 1a and 1c than 1b (Table 4.10, Figure 4.8). There was no significant change in flow velocity following weir removal at Site 2 (Table 4.10, Figure 4.8).

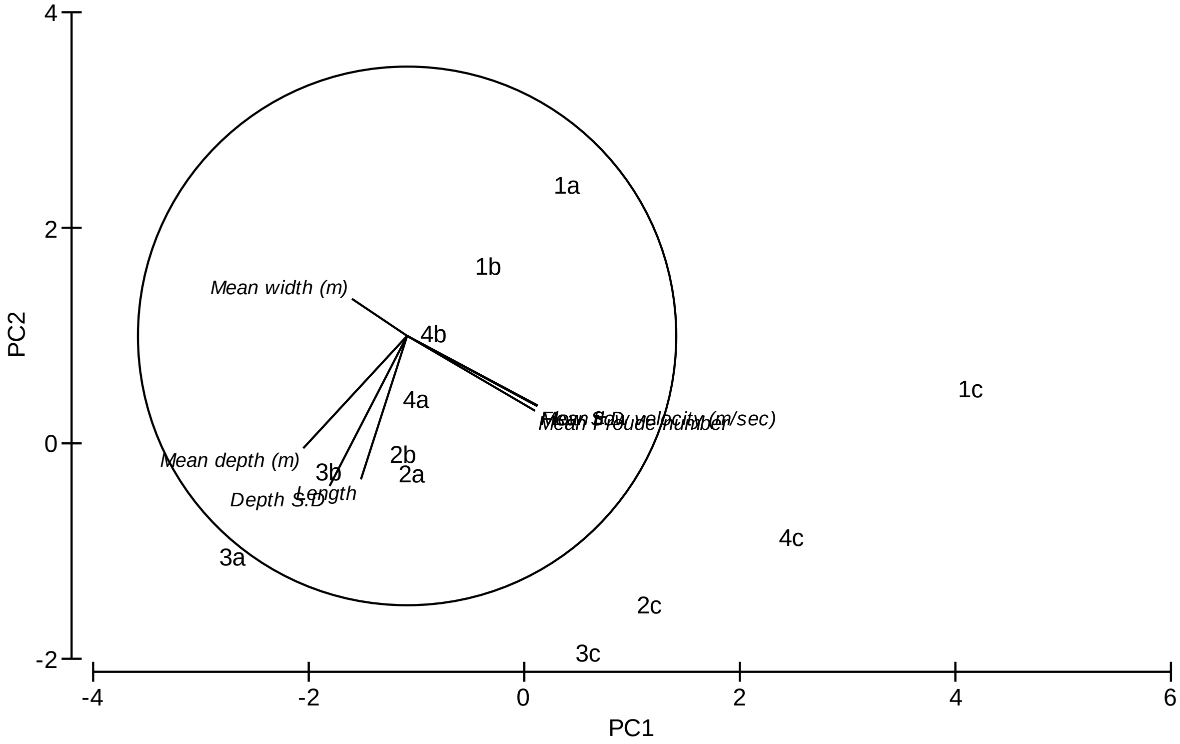


Figure 4.6: PCA plot of hydraulic variables within sites in the River Dove at Dovedale.

Froude number was significantly higher downstream (Site 1) following weir removal. Froude number was significantly higher in Site 1b than 1a, 1c than 1a and 1c than 1b (Table 4.11, Figure 4.19). There was no statistically significant change in Froude number following weir removal at Site 2.

There was no statistically significant change in width following weir removal (Table 4.12, Figure 4.10).

Table 4.9: Differences in mean depth recorded at each site in the River Dove at Hartington (values highlighted in yellow indicate statistical significance)..

Site	Depth (m)			Statistical test and significance	Post *hoc*. tests	
	Mean	SD	*n*			
D1a	0.70	0.21	36	ANOVA: F = 22.415, *p* = 0.000	1a/1b	0.740
D1b	0.68	0.04	18		1a/1c	0.000
D1c	0.41	0.22	36		1b/1c	0.000
D2a	0.71	0.26	26	ANOVA: F = 9.019, *p* = 0.000	1a/1b	0.026
D2b	0.56	0.2	30		1a/1c	0.000
D2c	0.45	0.25	39		1b/1c	0.056

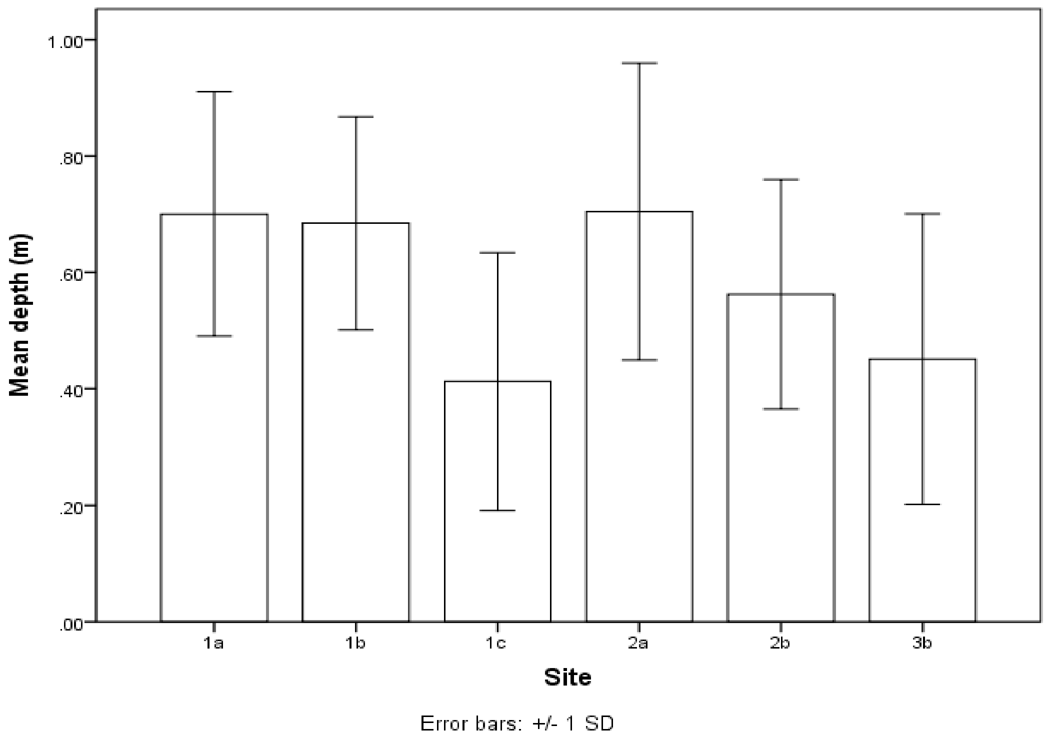


Figure 4.7: Mean depth at each site in the River Dove at Hartington.

Table 4.10: Differences in mean flow velocity recorded at each site in the River Dove at Hartington (values highlighted in yellow indicate statistical significance).

Site	Flow velocity (m/sec)			Statistical test and significance	Post *hoc*. tests	
	Mean	SD	*n*			
D1a	0.04	0.03	34	Kruskall – Wallis: p = 0.000	1a/1b	0.003
D1b	0.07	0.04	18		1a/1c	0.002
D1c	0.51	0.42	28		1b/1c	0.001
D2a	0.03	0.02	20	Kruskall – Wallis: p = 0.285	n/a	
D2b	0.11	0.16	18			
D2c	0.38	0.46	30			

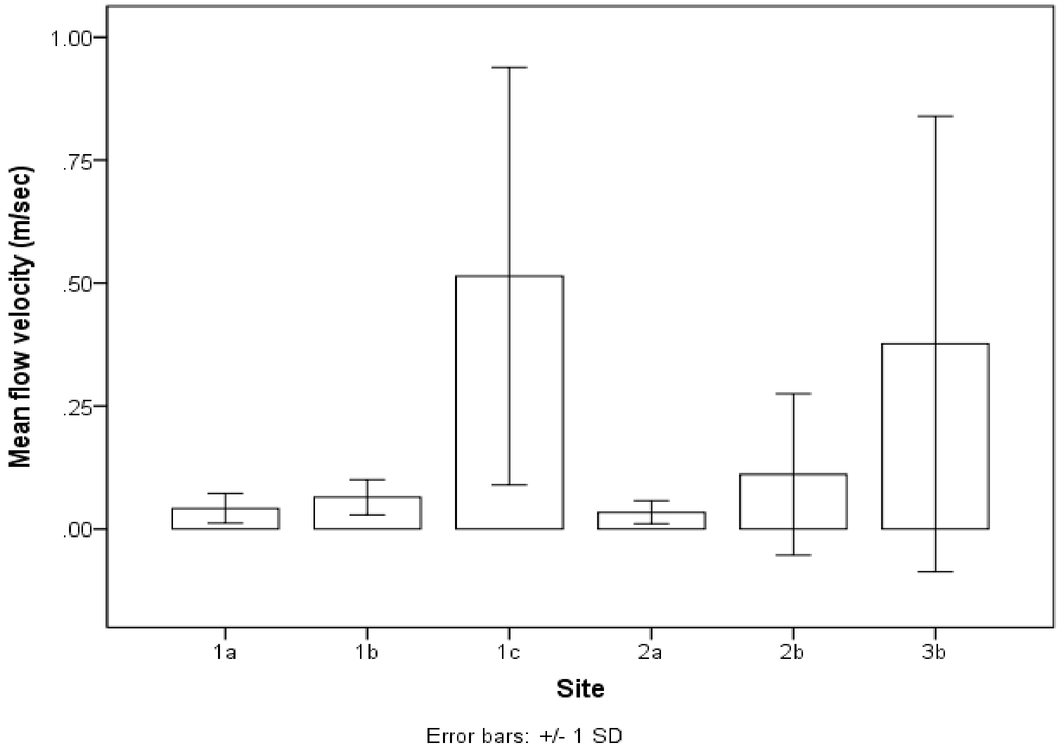


Figure 4.8: Mean flow velocity at each site in the River Dove at Hartington

Table 4.11: Differences in mean Froude number recorded at each site in the River Dove at Hartington (values highlighted in yellow indicate statistical significance).

Site	Froude number			Statistical test and significance	Post *hoc*. tests	
	Mean	SD				
D1a	0.003567	0.003114	34	Kruskall – Wallis:	1a/1b	0.023
D1b	0.005368	0.003324	18	*p* = 0.001	1a/1c	0.004
D1c	0.059348	0.054861	26		1b/1c	0.002
D2a	0.003832	0.00417	20	Kruskall – Wallis:		
D2b	0.011636	0.017143	18	*p* = 0.455	n/a	
D2c	0.048814	0.070528	29			

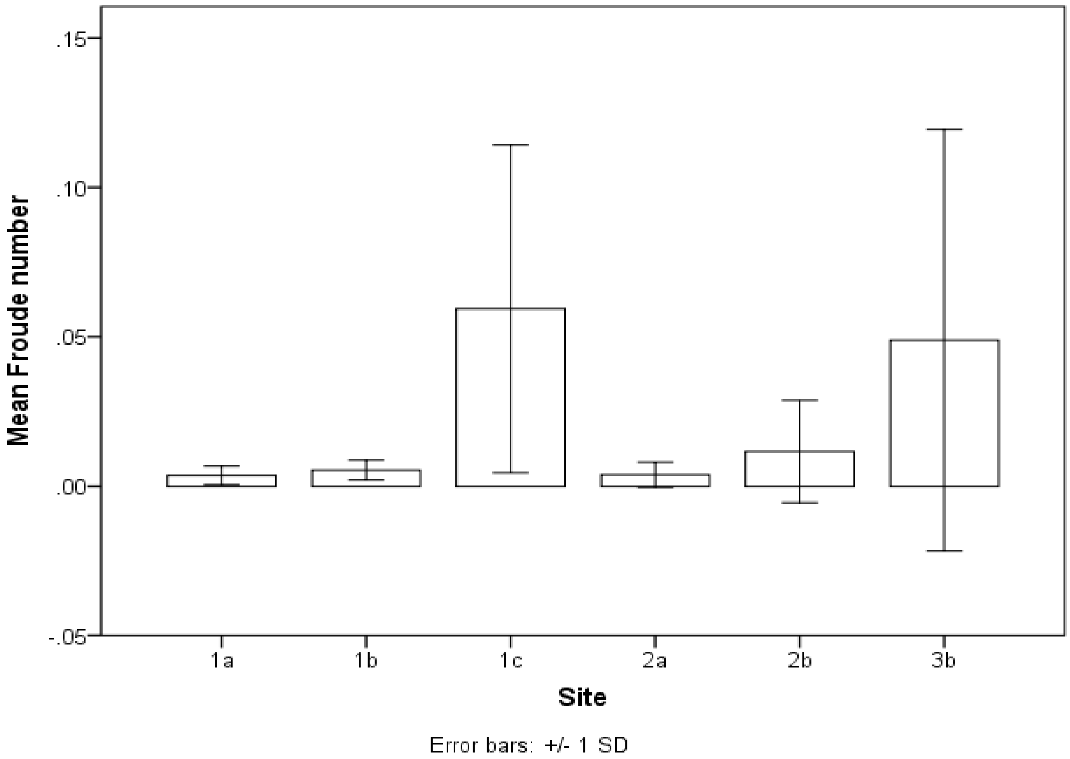


Figure 4.9: Mean Froude numberat each site in the River Dove at Hartington.

Table 4.12: Differences in mean width recorded at each site in the River Dove at Hartington.

Site	**Width (m)**			**Statistical test and significance**	**Post *hoc*. tests**
	Mean	SD	*n*		
D1a	6.9	1.41	12	ANOVA: F = 0.249, *p* = 0.781	n/a
D1b	7	1.38	12		
D1c	6.6	1.29	12		
D2a	7	1.07	10	ANOVA: F = 2.507, *p* = 0.098	n/a
D2b	7.3	1.53	10		
D2c	6.1	1.48	13		

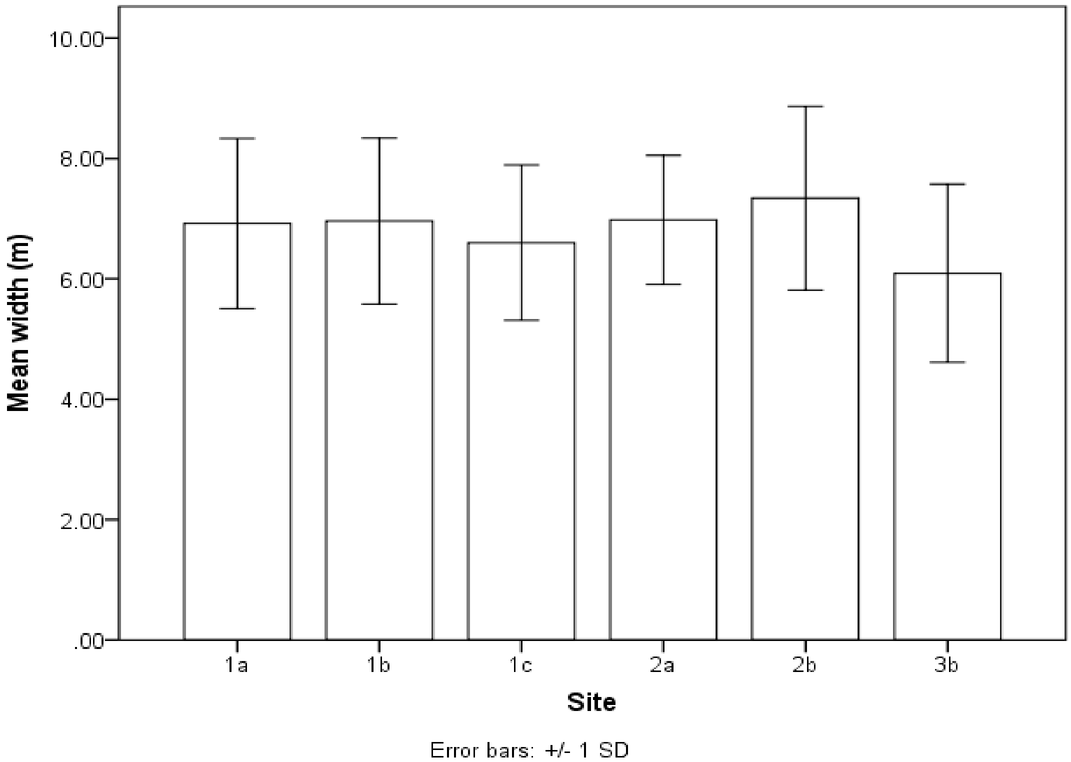


Figure 4.10: Mean width at each site in the River Dove at Hartington.

Similarity between sites in the River Dove at Hartington

Sites 1a, 1b, 2a and 2b were similar (Euc. 3). Within this cluster, Sites 1a and 2a were similar (Euc. 2) and Sites 1b and 2b were similar (Euc. 2). Sites 1c and 2c were not similar to other sites but were similar to one another (Euc. 3) (Figure 4.11). Sites 1a, 1b and 2a were positively associated with high mean depths and negatively with flow velocity and Froude number. Following weir removal, Site 2b was positively associated with mean width. Site 1c was positively associated with Froude number and mean flow velocity and was less similar to Site 2c, which was positively associated with high standard deviation of depth (Figure 4.12).

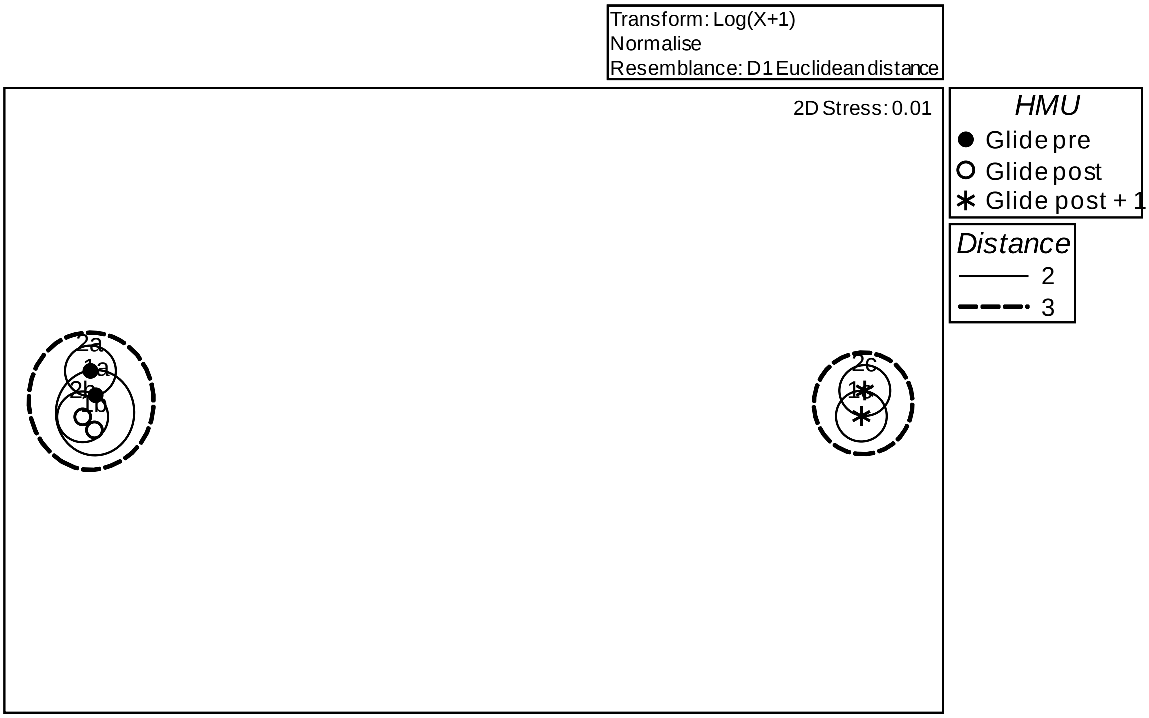


Figure 4.11: Similarity of sites in the River Dove at Hartington.

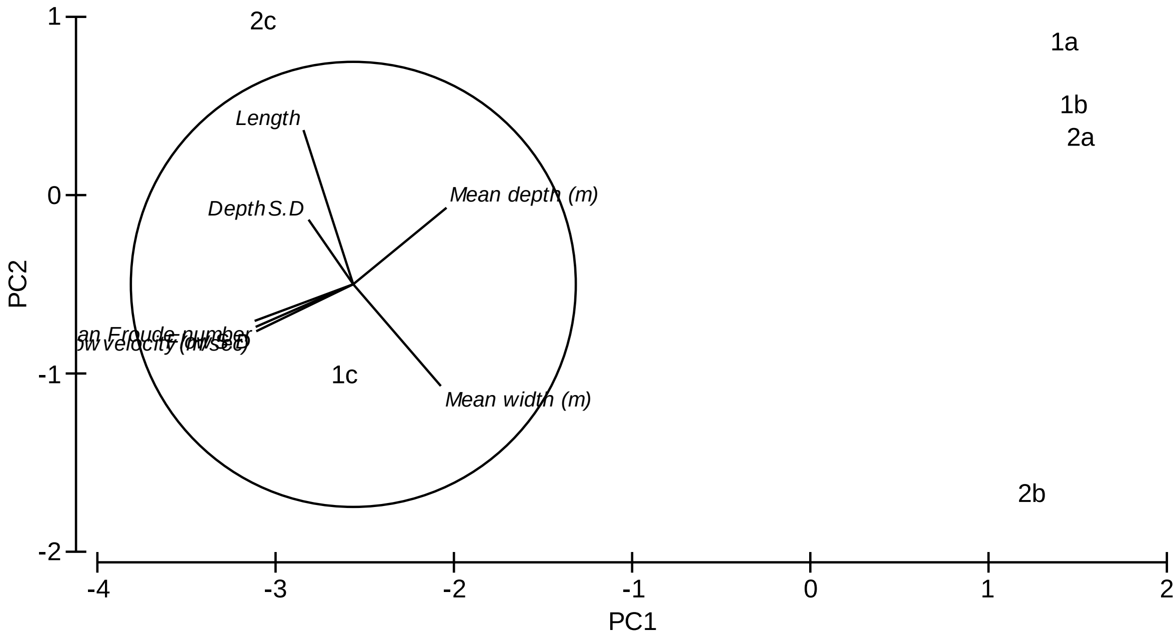


Figure 4.12: PCA plot of hydraulic variables within sites in the River Dove at Hartington

4.1.3 Channel narrowing at Lowthorpe Beck, East Yorkshire

Following channel narrowing at Lowthorpe Beck there was no statistically significant change in depth (Table 4.13, Figure 4.13) and flow velocity (Table 4.14, Figure 4.14).

Froude number at Site 3 increased significantly after channel narrowing (Table 4.15, Figure 4.15) but showed no significant change at other sites. Width at Site 2b was significantly higher than at Site 2a but there was no significant change in width at other sites (Table 4.16, Figure 4.16).

Table 4.13: Differences in mean depth recorded at each site in Lowthorpe Beck, East Yorkshire (values highlighted in yellow indicate statistical significance)..

Site	Depth (m)			Statistical test and significance
	Mean	SD	*n*	
LB1a **LB1b**	0.74 0.65	0.13 0.73	21 18	ANOVA: F= 2.786, *p*= 0.104
LB2a **LB2b**	0.74 0.74	0.13 0.13	18 18	ANOVA: F= 1.238, *p*= 0.274
LB3a **LB3b**	0.61 0.6	0.18 0.15	15 15	ANOVA: F=0.041, *p*= 0.841

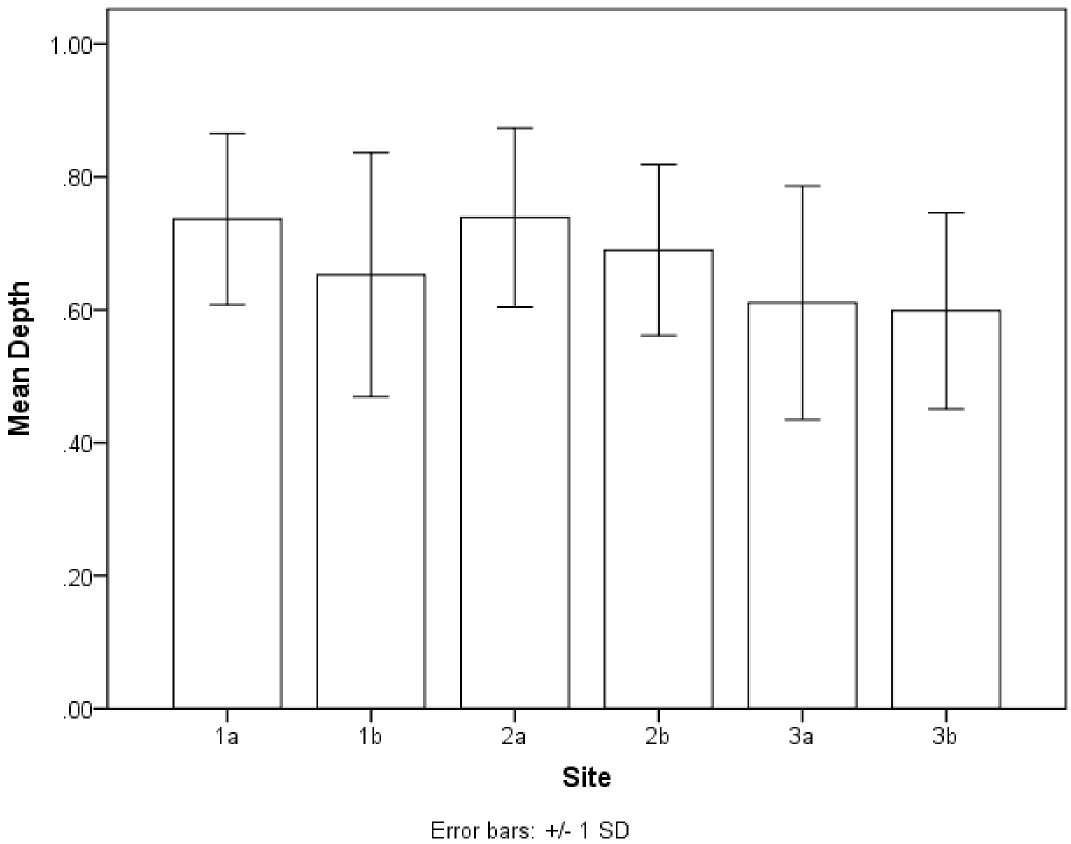


Figure 4.13: Mean depth at each site in Lowthorpe Beck, East Yorkshire.

Table 4.14: Differences in mean flow velocity recorded at each site in Lowthorpe Beck, East Yorkshire (values highlighted in yellow indicate statistical significance)..

Site	Flow velocity (m/sec)			Statistical test and significance
	Mean	SD	*n*	
LB1a **LB1b**	0.11 0.43	0.09 0.73	21 18	Mann Whitney U: *p*= 0.124
LB2a **LB2b**	0.08 0.50	0.06 0.71	18 18	Mann Whitney U: *p*= 0.543
LB3a **LB3b**	0.08 0.18	0.05 0.35	14 15	Mann Whitney U: *p*= 0.068

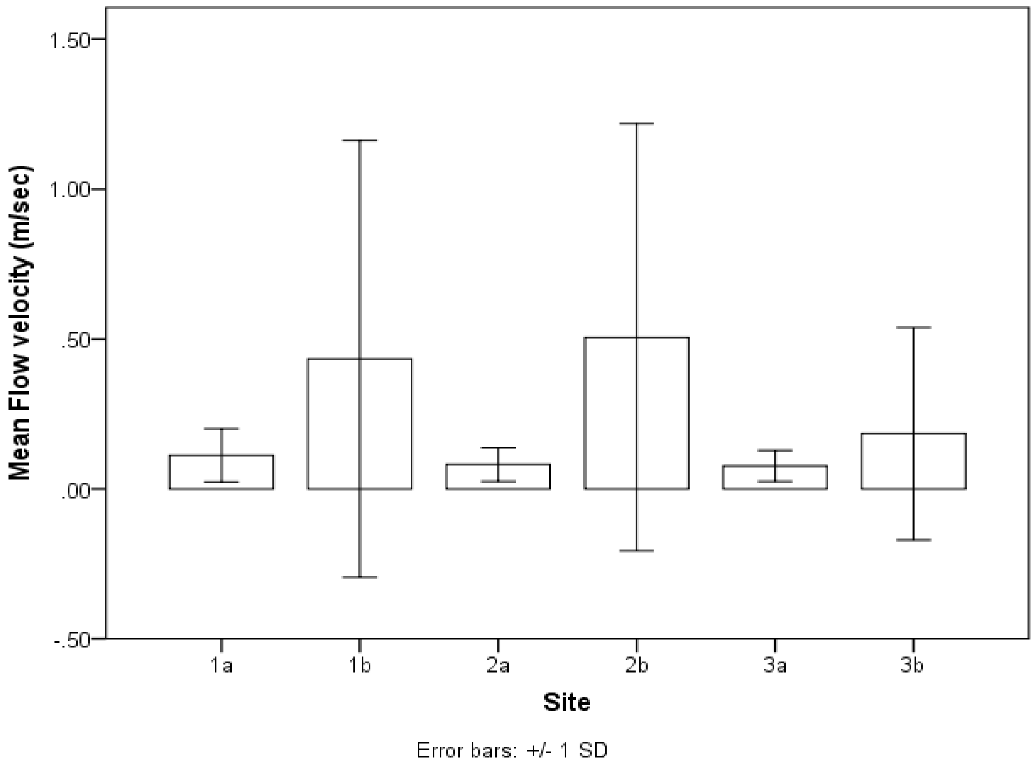


Figure 4.14: Mean flow velocity at each site in Lowthorpe Beck, East Yorkshire.

Table 4.15: Differences in mean Froude number recorded at each site in Lowthorpe Beck, East Yorkshire (values highlighted in yellow indicate statistical significance). .

Site	Froude number			Statistical test and significance
	Mean	SD	*n*	
LB1a **LB1b**	0.008262 0.051932	0.00701 0.102549	21 18	Mann Whitney U: *p*= 0.131
LB2a **LB2b**	0.005827 0.041102	0.004116 0.063945	18 18	Mann Whitney U: *p*= 0.676
LB3a **LB3b**	0.007037 0.014023	0.004525 0.027086	14 15	Mann Whitney U: *p*=0.029

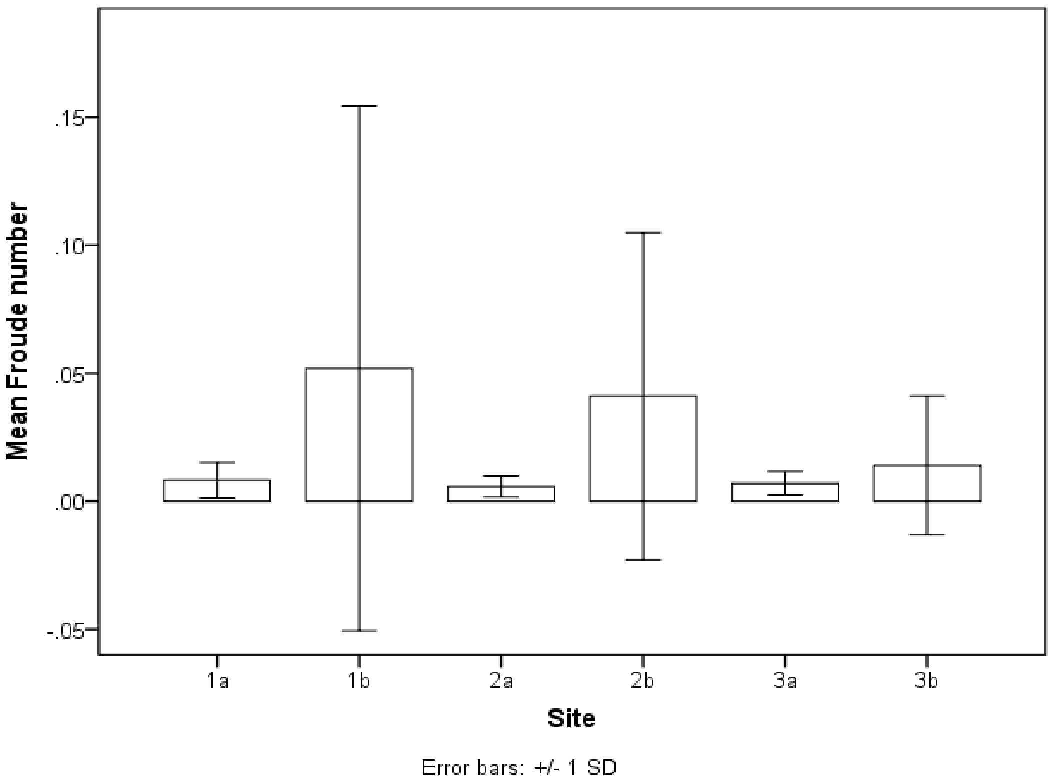


Figure 4.15: Mean Froude number at each site in Lowthorpe Beck, East Yorkshire.

Table 4.16: Differences in mean width recorded at each site in Lowthorpe Beck, East Yorkshire (values highlighted in yellow indicate statistical significance).

Site	**Width (m)**			**Statistical test and significance**
	Mean	SD	*n*	
LB1a **LB1b**	7.1 8.4	0.58 1.72	7 6	Mann Whitney U: *p* = 0.085
LB2a **LB2b**	6.5 7.6	0.6 0.47	6 6	ANOVA: F= 12.275, *p*= 0.006
LB3a **LB3b**	7.9 7.2	1.52 0.82	5 5	ANOVA: F= 0.775 *p*=0.404

Similarity between sites in Lowthorpe Beck, East Yorkshire

Sites 1a and 2a were similar (Euc. 2). Sites 1b and 2b were not clustered together, so were not similar to one another. Sites 3a and 3b were clustered together indicating similarity (Euc. 3) (Figure 4.17). Sites 1a and 2a were defined by high mean depths. Sites 1b and 2b were associated with high mean flow velocities and Froude numbers. Site 3a was associated with a high percentage of silt abundance and Site 3b with the standard deviation of depth (Figure 4.18).

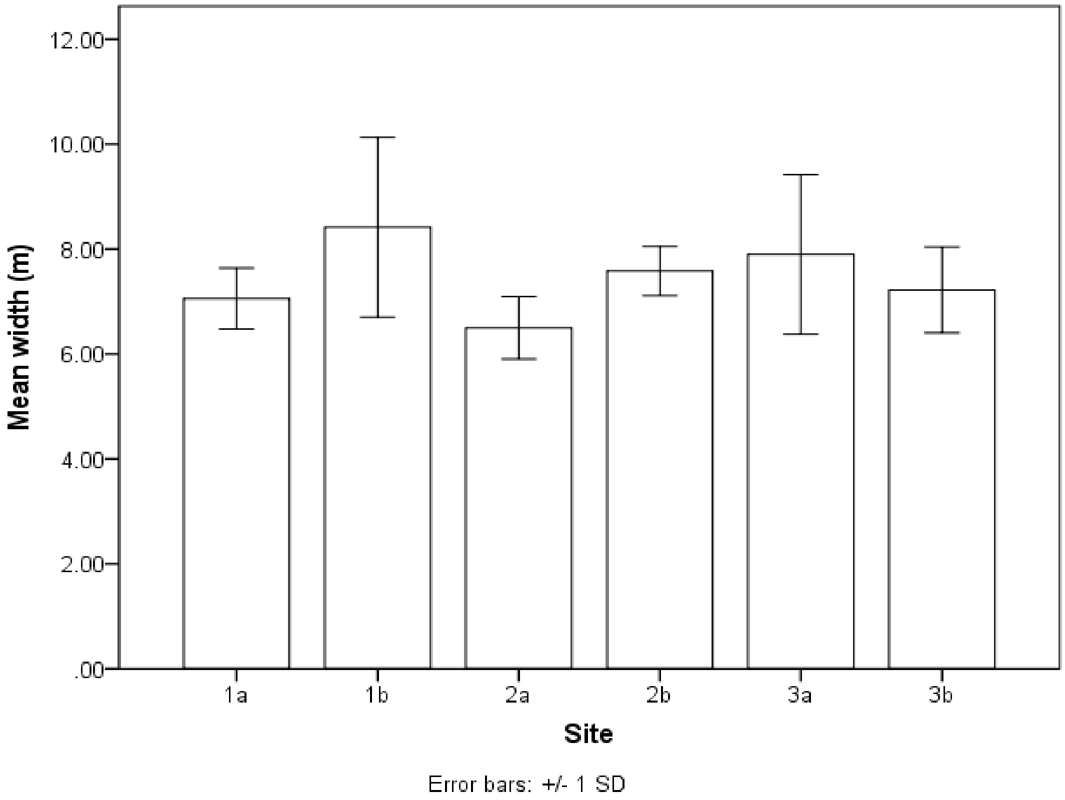


Figure 4.16: Mean width at each site in Lowthorpe Beck, East Yorkshire.

4.1.4 Installation of gravel riffles in the River Stiffkey, North Norfolk

Depth was significantly higher at Site 1b than 1a, 7b than 7a, Site 8b than Site 8a and S10b than Site 10a (Table 4.17, Figure 4.19). At all other sites, there was no significant change in depth between surveys.

Flow velocity was significantly higher at Site 2b than Site 2a, Site 5b than Site 5a and Site 10b than Site 10a (Table 4.18, Figure 4.20). At all other sites there was no significant change in flow velocity between surveys.

Froude number was significantly higher at Site 2b than Site 2a, Site 5b than Site 5a and Site 10b than Site 10a (Table 4.19, Figure 4.21). At all other sites there was no significant change in Froude number between surveys.

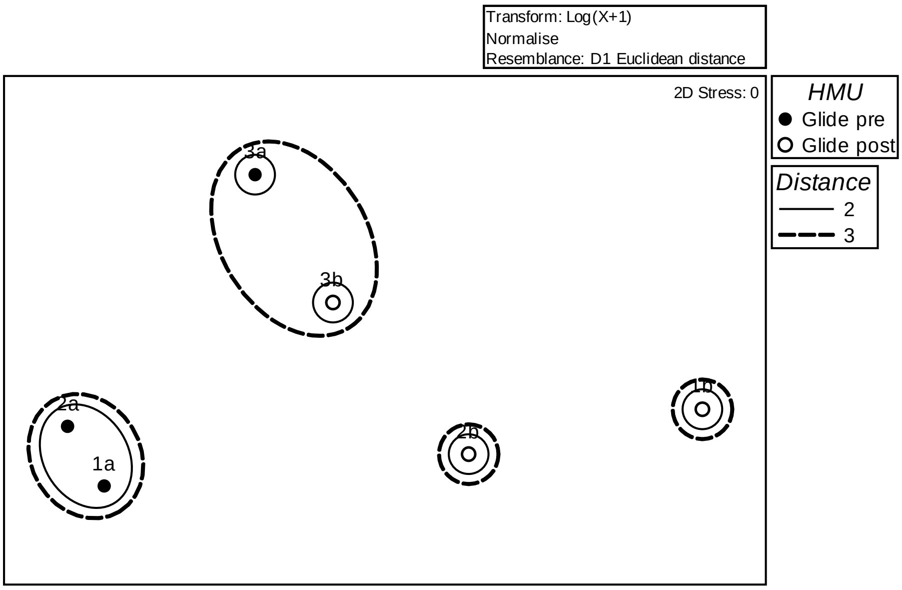


Figure 4.17: Similarity of sites in Lowthorpe Beck, East Yorkshire.

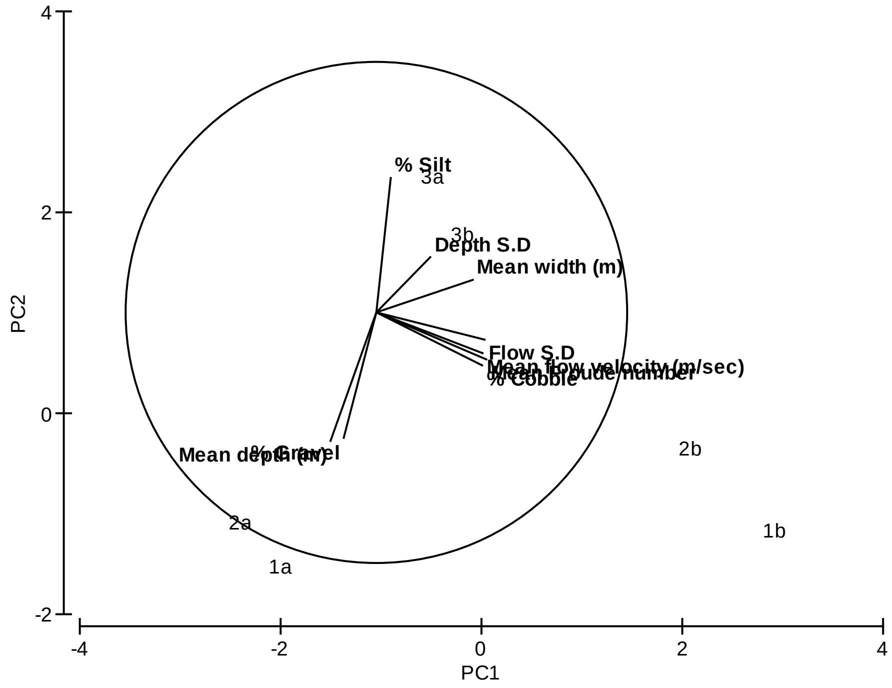


Figure 4.18: PCA plot of hydraulic variables within sites in Lowthorpe Beck, East Yorkshire.

Site 3b was significantly narrower than Site 3a, Site 7b was significantly wider than Site 7a and Site 9b was significantly wider than Site 9a (Table 4.20, Figure 4.22). At all other sites there was no statistically significant change in channel width between surveys.

Differences in hydraulic variables between HMUs after gravel installation in the River Stiffkey, North Norfolk

In stretch 1, depth was significantly higher in Sites 1b and 3b than Site 2b. In stretch 2, depth was significantly higher in Site 5a than Site 4a and in Site 5b than Site 4b. In stretch 3 depth was significantly higher in Site 7a than 6a and in stretch 4 depth was significantly higher in Site 9a than Site 8a, in Site 10a than Site 8a, in Site 9b than Site 8b and in Site 8b than Site 10b (Table 4.21).

At stretch 1, there was a significant difference in flow velocity between Site 1b and Site 2b and Site 2b and Site 3b. At stretch 4, there was a significant difference in flow velocity between Site 8a and Site 9a and between Site 8a and Site 10a. There was also a statistically significant difference in flow velocity between Site 8b and Site 9b, between Site 8b and Site 10b and between Site 9b and Site 10b (Table 4.22).

Froude number was significantly higher in Site 2b than Site 1b in stretch 1. In stretch 2, Froude number was significantly higher in Site 4a than Site 5a. At stretch 4, Froude number was significantly higher in Site 8a than Site 9a and in Site 8a than Site 10a. Froude number was also significantly higher in Site 8b than Site 9b and Site 10b than Site 9b. There were no other significant differences in Froude number between HMUs within survey stretches in the River Stiffkey (Table 4.23).

Table 4.17: Differences in mean depth recorded at each site in the River Stiffkey, North Norfolk (values highlighted in yellow indicate statistical significance). .

Site	Depth			Statistical test and significance
	Mean	SD	*n*	
S1a	0.49	0.24	9	Mann Whitney U:
S1b	0.9	0.02	3	*p* = 0.012
S2a	0.33	0.20	9	Mann Whitney U:
S2b	0.23	0.05	9	*p* = 0.723
S3a	0.51	0.16	9	ANOVA: F = 0.692,
S3b	0.46	0.11	12	*p*= 0.416
S4a	0.37	0.18	18	Mann Whitney U:
S4b	0.25	0.05	9	*p* = 0.291
S5a	0.74	0.10	12	ANOVA: F = 2.543,
S5b	0.80	0.09	12	*p*= 0.125
S6a	0.61	0.31	6	Mann Whitney U:
S6b	0.47	0.19	6	*p* = 0.259
S7a	0.74	0.08	12	ANOVA: F = 6.278,
S7b	0.85	0.13	12	*p*= 0.020
S8a	0.26	0.04	9	Mann Whitney U:
S8b	0.41	0.14	9	*p* = 0.001
S9a	0.76	0.08	12	Mann Whitney U:
S9b	0.8	0.16	15	*p* = 0.241
S10a	0.29	0.06	3	ANOVA: F = 7.732,
S10b	0.39	0.05	6	*p*= 0.027

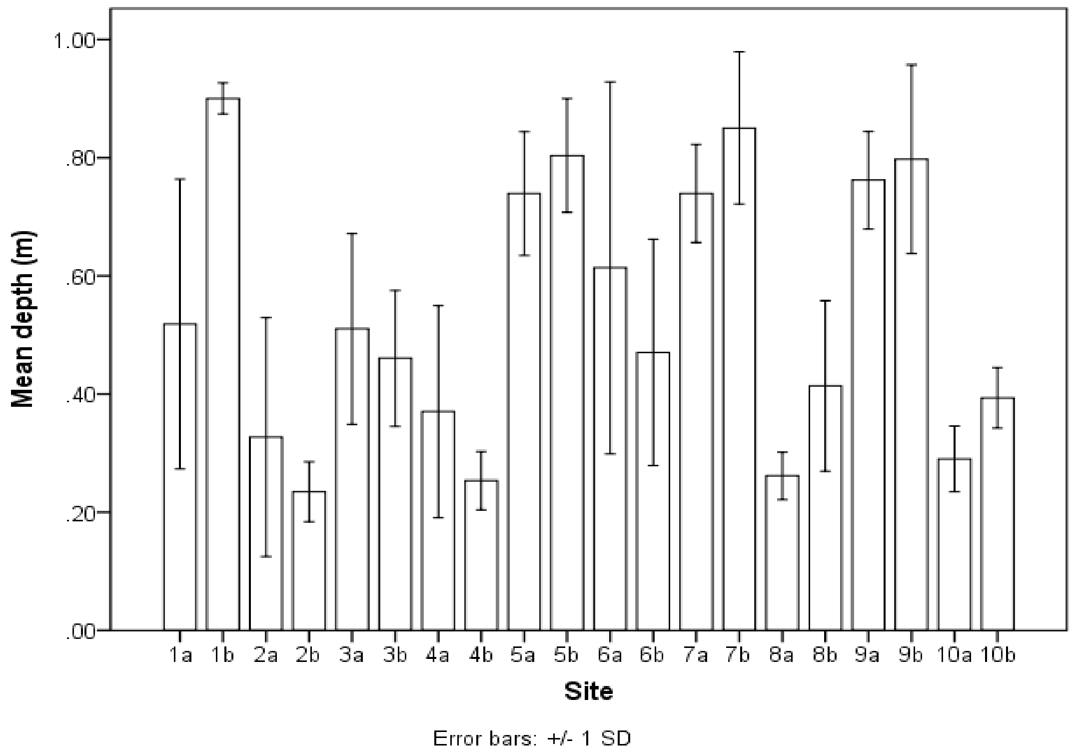


Figure 4.19: Mean depth at each site in the River Stiffkey, North Norfolk.

Table 4.18: Differences in mean flow velocity recorded at each site in the River Stiffkey, North Norfolk (values highlighted in yellow indicate statistical significance). .

Site	Flow velocity			Statistical test and significance
	Mean	SD	*n*	
S1a	0.30	0.19	9	Mann Whitney U:
S1b	0.63	0.61	3	p = 0.308
S2a	0.42	0.33	6	ANOVA: F = 31.695,
S2b	4.63	1.79	7	*p* = 0.000
S3a	0.21	0.15	9	Mann Whitney U:
S3b	0.45	0.47	10	p = 0.623
S4a	0.38	0.25	17	Mann Whitney U:
S4b	1.89	2.16	8	p = 1.000
S5a	0.37	0.34	11	ANOVA: F = 14.969,
S5b	1.02	0.4	8	*p* = 0.001
S6a	0.46	0.46	6	ANOVA: F = 0.354,
S6b	0.68	0.72	4	*p* = 0.568
S7a	0.17	0.09	12	Mann Whitney U:
S7b	0.56	0.63	8	*p* = 0.786
S8a	0.39	0.08	9	Mann Whitney U:
S8b	1.39	1.15	8	*p* = 0.335
S9a	0.18	0.07	12	Mann Whitney U:
S9b	0.54	0.67	12	*p* = 1.000
S10a	0.16	0.05	3	Mann Whitney U:
S10b	2.55	0.66	4	*p* = 0.032

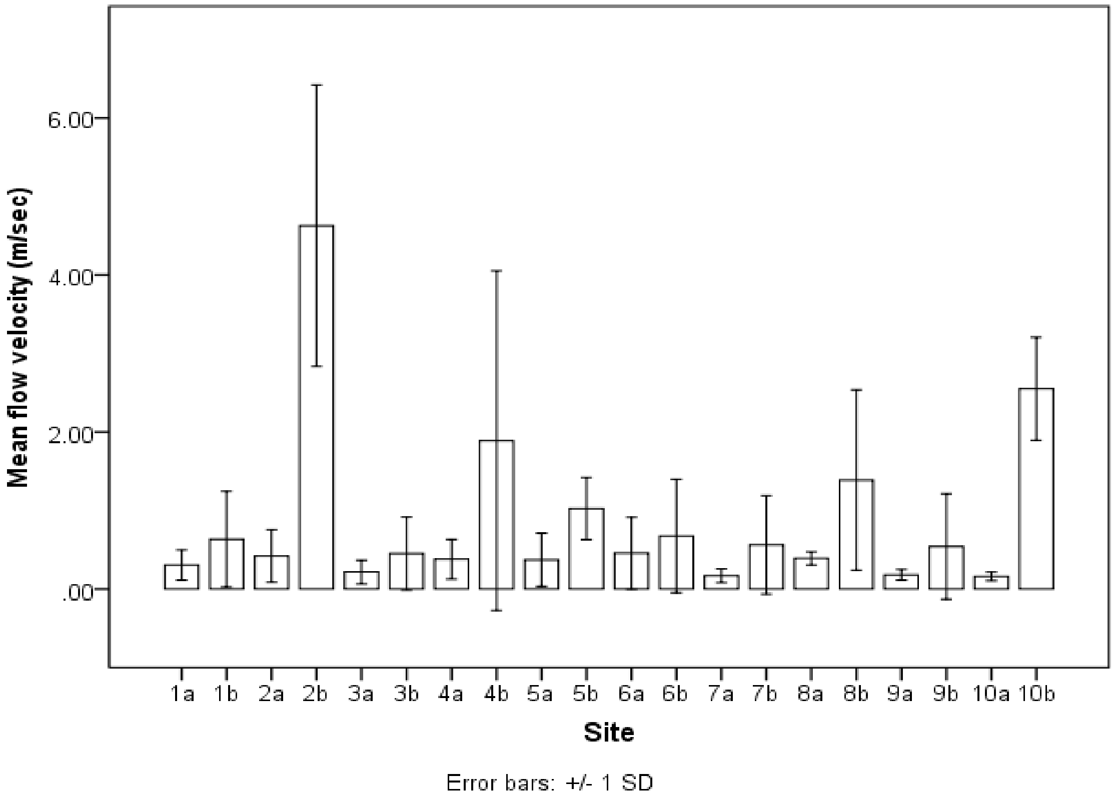


Figure 4.20: Mean flow velocity at each site in the River Stiffkey, North Norfolk

Table 4.19: Differences in mean Froude number recorded at each site in the River Stiffkey, North Norfolk (values highlighted in yellow indicate statistical significance).

Site	Froude number			Statistical test and significance
	Mean	SD	*n*	
S1a	0.046869	0.038203	9	ANOVA: F = 0.191,
S1b	0.035917	0.03514	3	*p* = 0.672
S2a	0.077553	0.088329	6	ANOVA: F = 30.630,
S2b	1.123977	0.453041	7	*p* = 0.000
S3a	0.030031	0.035287	9	Mann Whitney U:
S3b	0.053126	0.055739	10	*p* = 0.744
S4a	0.079136	0.071053	17	Mann Whitney U:
S4b	0.403647	0.485762	8	*p* = 0.683
S5a	0.009389	0.005711	11	Mann Whitney U:
S5b	0.068455	0.031012	8	*p* = 0.000
S6a	0.068093	0.07218	6	ANOVA: F = 0.007,
S6b	0.071726	0.052096	4	*p* = 0.934
S7a	0.01144	0.005747	12	Mann Whitney U:
S7b	0.0306	0.034567	8	*p* = 0.786
S8a	0.077962	0.023282	9	Mann Whitney U:
S8b	0.202309	0.170658	8	*p* = 0.386
S9a	0.040438	0.041093	12	ANOVA: F = 0.268,
S9b	0.031879	0.039855	12	*p* = 0.0.610
S10a	0.027581	0.004249	3	ANOVA: F = 43.915,
S10b	0.315007	0.073232	4	*p* = 0.001

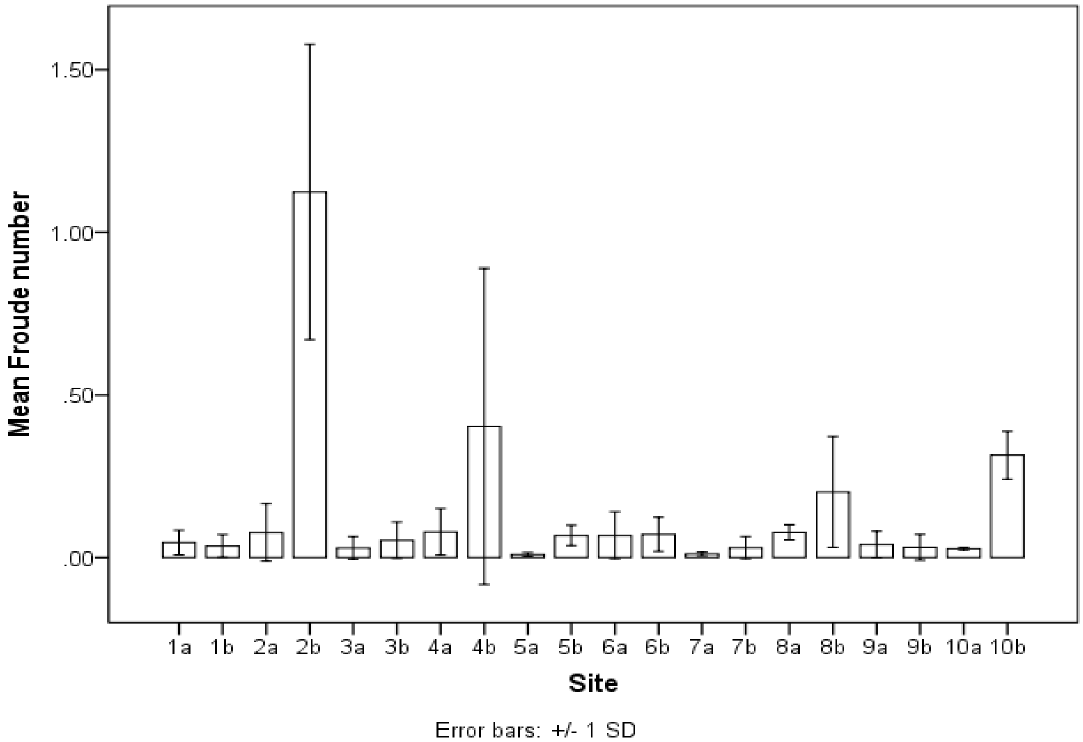


Figure 4.21: Mean Froude number at each site in the River Stiffkey, North Norfolk.

Table 4.20: Differences in mean width recorded at each site in the River Stiffkey, North Norfolk (values highlighted in yellow indicate statistical significance).

Site	Width			Statistical test and significance
	Mean	SD	*n*	
S1a	3.6	0.67	3	Mann Whitney U:
S1b	4.2	n/a	1	*p* = 0.655
S2a	3.7	0.94	3	ANOVA: F= 0.284,
S2b	3.4	0.53	3	*p*=0.622
S3a	5.7	0.61	3	ANOVA: F= 7.084,
S3b	4.9	0.3	4	*p*=0.045
S4a	3.55	0.77	6	ANOVA: F= 0.940,
S4b	3.1	0.17	3	*p*=0.365
S5a	4.6	0.88	4	Mann Whitney U:
S5b	4.05	0.1	4	*p*= 0.166
S6a	3.4	0.28	2	Mann Whitney U:
S6b	3	n/a	2	*p*= 0.102
S7a	2.9	0.66	4	ANOVA: F= 12.417,
S7b	4.1	0.27	4	*p*=0.012
S8a	3.5	0.65	3	ANOVA: F= 1.003,
S8b	4.1	0.81	3	*p*=0.373
S9a	2.6	0.22	4	ANOVA: F= 23.830,
S9b	3.8	0.45	5	*p*=0.002
S10a	3.2	n/a	1	Mann Whitney U:
S10b	3.8	0.07	2	*p*= 0.221

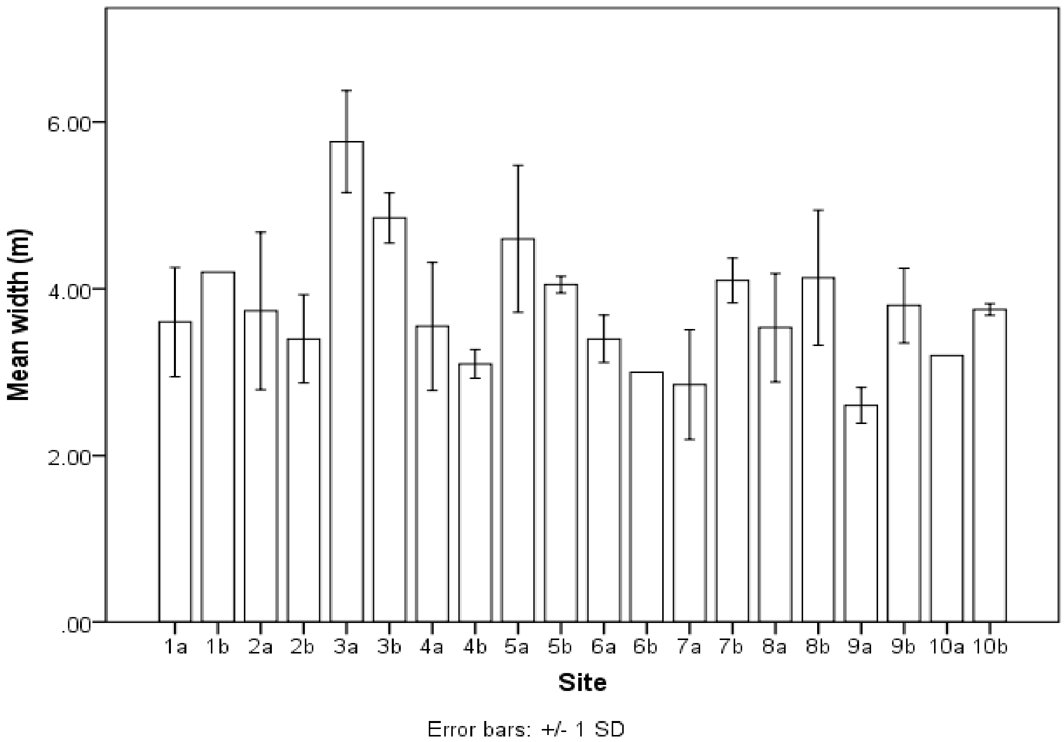


Figure 4.22: Mean width at each site in the River Stiffkey, North Norfolk.

At stretch 1, width was significantly higher in Site 3a than Site 1a and Site 3a than Site 2a. At stretch 2, width was significantly higher in Site 5b than Site 4b. Width was significantly higher in Site 7b than Site 6b in stretch 3. There were no other significant differences in width between HMUs in survey stretches of the River Stiffkey (Table 4.24).

Similarity between sites in the River Stiffkey, North Norfolk

Apart from Sites 2b, 4b, 6a, 9b and 10b, all sites were broadly similar in hydraulic variables (Euc. 4) (Figure 4.23). Of sites not within the main cluster, Sites 6a and 9b were similar (Euc. 3), Sites 4b and 10b were also similar (Euc. 3) and shared broad similarity with 2b (Euc. 4) (Figure 4.23). Within the main cluster of sites, Site 1b was least similar to other sites. Sites 5a, 5b and 7b were similar (Euc. 2) and shared similarity with 3a (Euc. 3). Sites 7a and 9a were similar (Euc. 3). Sites 1a, 2a and 4a were very similar (Euc. 2) and shared similarity (Euc. 3) with 3b, 6b, 8a, 8b and 10a (Figure 4.23).

All sites apart from 2b, 4b and 10b, which were associated with flow velocity, Froude number and percentage composition of cobble, were associated with the mean and standard deviation of depth and the percentage composition of silt (Figure 4.24).

Table 4.21: Differences in mean depth between HMUs in the River Stiffkey, North Norfolk (values highlighted in yellow indicate statistical significance).

State	**Depth (m) at site**			**Statistical test and significance**	**Post *hoc.* tests**	
	Stretch 1					
	S1	**S2**	**S3**			
a	0.49	0.33	0.51	ANOVA: F = 2.179, *P*=0.135	-	
b	0.9	0.23	0.46	ANOVA: F =64.054 *P*=0.000	S1/S2 S1/S3 S2/S2	0.000 0.000 0.000
	Stretch 2					
	S4	**S5**				
a	0.37	0.74	-	Mann Whitney U: *p* = 0.000	-	
b	0.25	0.80	-	ANOVA: F = 262.821, *P*=0.000	-	
	Stretch 3					
	S6	**S7**				
a	0.61	0.74	-	ANOVA: F = 25.254, *P*=0.000	-	
b	0.47	0.85	-	Mann Whitney U: *p* = 1.000	-	
	Stretch 4					
	S8	**S9**	**S10**			
a	0.26	0.76	0.29	ANOVA: F = 162.537 *P*=0.000	S8/S9 S8/S10 S9/S10	0.000 0.000 0.525
b	0.41	0.80	0.39	ANOVA: F = 29.008 *P*=0.000	S8/S9 S8/S10 S9/S10	0.000 0.000 0.790

Table 4.22: Differences in mean flow velocity between HMUs in the River Stiffkey, North Norfolk (values highlighted in yellow indicate statistical significance).

State	**Flow velocity (m/sec) at site**			**Statistical test and significance**	**Post *hoc.* tests**	
	Stretch 1					
	S1	**S2**	**S3**			
a	0.30	0.42	0.21	Kruskall-Wallis: P = 0.463	-	-
b	0.63	4.63	0.45	ANOVA: F = 30.098, *p* = 0.000	S1/S2 S1/S3 S2/S3	0.000 0.810 0.000
	Stretch 2					
	S4	**S5**				
a	0.38	0.37		Mann Whitney U: *p*= 1.000	-	
b	1.89	1.02		Mann Whitney U: *p*= 0.469	-	
	Stretch 3					
	S6	**S7**				
a	0.46	0.17		Mann Whitney U: *p*= 0.687	-	
b	0.68	0.56		ANOVA: F = 1.227, *p* = 0.294	-	
	Stretch 4					
	S8	**S9**	**S10**			
a	0.39	0.18	0.16	ANOVA: F = 19.464, *p*= 0.000	S8/S9 S8/S10 S9/S10	0.000 0.001 0.853
b	1.39	0.54	2.55	ANOVA: F = 8.631 *p*= 0.002	S8/S9 S8/S10 S9/S10	0.043 0.001 0.038

Table 4.23: Differences in mean flow velocity between HMUs in the River Stiffkey, North Norfolk (values highlighted in yellow indicate statistical significance).

State	**Froude number at site**			**Statistical test and significance**	**Post *hoc.* tests**	
	Stretch 1					
	S1	**S2**	**S3**			
a	0.046869	0.077553	0.030031	Kruskall-Wallis: *p* = 0.426	-	-
b	0.035917	1.123977	0.053126	Kruskall-Wallis: *p* = 0.001	S1/S2 S1/S3 S2/S3	0.017 0.865 0.001
	Stretch 2					
	S4	**S5**				
a	0.079136	0.009389	-	Mann Whitney U: *p*= 0.000	-	
b	0.403647	0.068455	-	Mann Whitney U: *p*= 0.916	-	
	Stretch 3					
	S6	**S7**				
a	0.068093	0.01144	-	Mann Whitney U: *p*= 0.639	-	
b	0.071726	0.0306	-	Mann Whitney U: *p*= 0.142	-	
	Stretch 4					
	S8	**S9**	**S10**			
a	0.077962	0.040438	0.027581	Kruskall – Wallis: *P* = 0.000	S8/S9 S8/S10 S9/S10	0.033 0.013 0.386
b	0.202309	0.031879	0.315007	Kruskall – Wallis: *P* = 0.006	S8/S9 S8/S10 S9/S10	0.033 0.308 0.003

Table 4.24: Differences in mean width between HMUs in the River Stiffkey, North Norfolk (values highlighted in yellow indicate statistical significance).

State	Width (m) at site			Statistical test and significance	Post *hoc.* tests	
	Stretch 1					
	S1	**S2**	**S3**			
a	3.6	3.7	5.7	ANOVA: F = 0.835 *P* = 0.021	S1/S2 S1/S3 S2/S3	0.835 0.016 0.012
b	4.2	3.4	4.9	ANOVA: F = 10.858 *P* = 0.015	S1/S2 S1/S3 S2/S3	-
	Stretch 2					
	S4	**S5**				
a	3.6	4.6	-	ANOVA: F = 3.998 *P* = 0.081	-	
b	3.1	4.1	-	ANOVA: F = 85.952 *P* = 0.000	-	
	Stretch 3					
	S6	**S7**				
a	3.4	2.9		ANOVA: F = 1.178 *P* = 0.339	-	
b	3	4.1		ANOVA: F = 29.333 *P* = 0.006	-	
	Stretch 4					
	S8	**S9**	**S10**			
a	3.5	2.6	3.2	ANOVA: F = 3.872 *P* = 0.096	-	
b	4.1	3.8	3.8	ANOVA: F = 0.427 *P* = 0.669	-	

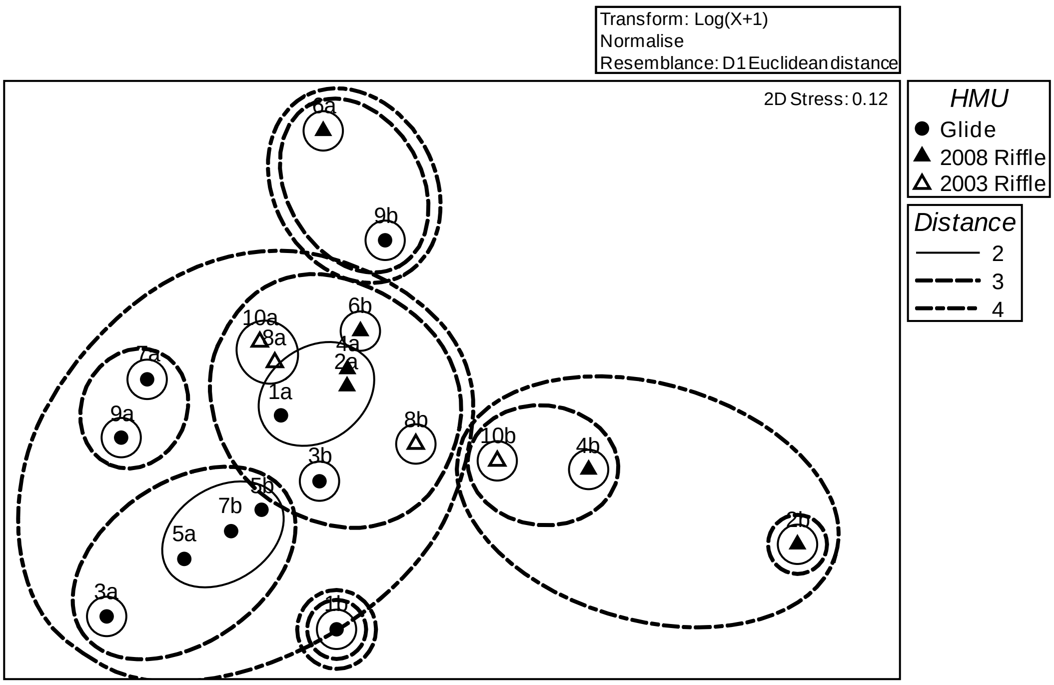


Figure 4.23: Similarity of sites in River Stiffkey, North Norfolk

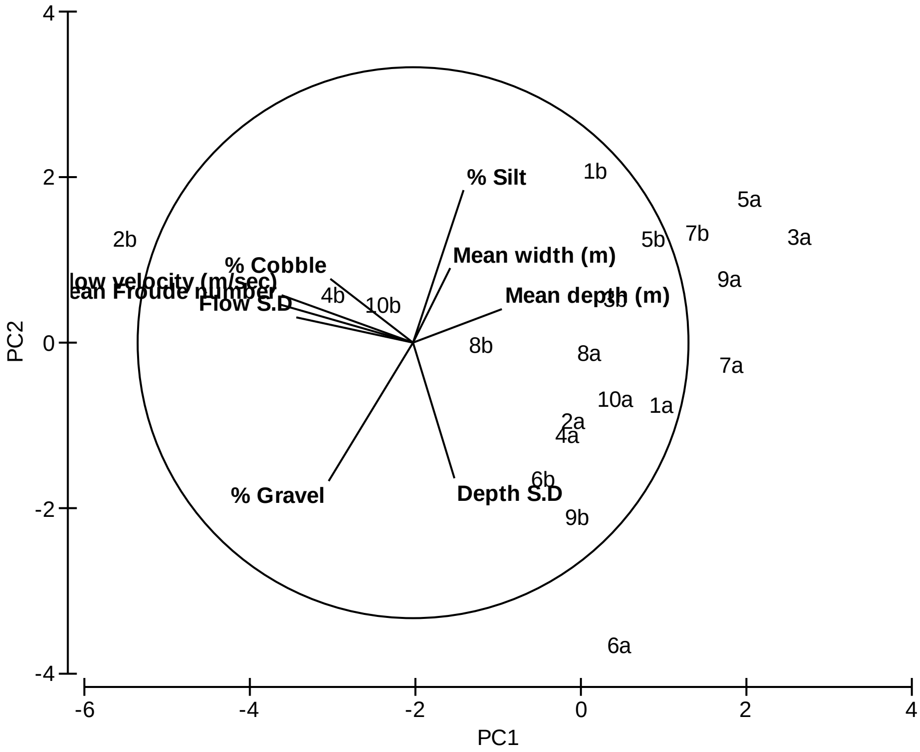


Figure 4.24: PCA plot of hydraulic variables within sites in River Stiffkey, North Norfolk

4.2 Discussion

Four river rehabilitation projects, including weir removal, channel narrowing and gravel installation; on three UK Rivers were studied for changes in hydraulic conditions following implementation.

Weir removal from the River Dove

The river channel at Dovedale was heterogeneous, displaying both glide and riffle habitats. Following removal of a small weir from the River Dove at Dovedale, the channel immediately upstream of the weir looked shallower and the upstream riffle looked more apparent. Little change was observed in the rest of the reach.

Both downstream and upstream of the weir, glide sites were significantly deeper than riffle sites during all surveys. Downstream of the weir, flow velocity was significantly higher in the riffle than the glide prior to weir removal. Upstream of the weir, flow velocity was significantly higher in the riffle than the glide before weir removal only. Following removal there was no significant difference in flow velocity between the riffle and glide upstream. However, significant changes in hydraulic conditions were recorded in the riffle (Site 4) following weir removal. In this riffle (Site 4), depth decreased significantly following weir removal and flow velocity and Froude number increased significantly suggesting that the most change took place at this upstream site following weir removal. Weirs increase the water level behind the barrier producing an area of steady water volume (Fjellheim and Raddum, 1996), the construction of the barrier can alter the physical processes within the channel modifying flow velocity, depth and substrate distribution (Bernacsek, 2001; Tomsic *et al*., 2007) resulting in increased water depth and reduced flow velocity upstream (Cowx *et a*l., 1993; Poff and Hart, 2002; Ashley *et al*., 2006; de Leaniz, 2008). Research by Mueller *et al*. (2011) found that abiotic characteristics upstream and downstream of weirs could differ so significantly, differences between upstream and downstream were often greater than between rivers. Following removal of the weir, flow velocity and Froude number increased at Sites 2 and 3 and ordination analysis indicated these sites increased in similarity suggesting the impoundment was removed and water was able to flow freely through the reach and connectivity had been restored.

The reinstatement in connectivity and removal of the impoundment causing increased flow velocity and Froude number in the previously impounded site (Site 3) could explain why there was no significant difference between the glide and riffle in the upstream reach following weir removal. These findings suggest abiotic conditions in the upstream reach increased in similarity and heterogeneity was reduced. This was confirmed by the ordination analysis, which showed Sites 1a, and 3a were least similar to other sites indicating there was more variation in hydraulic conditions between sites before the weir was removed and Sites 2c and 3c were more similar following weir removal suggesting the impoundment was removed and more uniform glide habitat was present. The water depth in the impoundment did decrease following weir removal, although not significantly. The reduced difference between HMUs following weir removal suggests that the increased similarity between Sites 3 and 4 is possibly attributable to the increased flow velocity following weir removal.

All 2010 sites were clustered in the MDS plots and 2011 sites were clustered, this suggests there was a difference in instream hydraulics between years, this may be due to differences in discharge between years. The Q-values were similar between survey dates, Q93 in 2010 and Q98 in 2011, respectively. However, as there was no significant difference in the wetted width of sites between surveys, it is possible that differences in hydraulic conditions are a result of weir removal rather than the annual hydrograph.

Following weir removal from the River Dove at Hartington, little effect on the physical nature of the channel was observed however, depth was significantly lower upstream and downstream after weir removal than before. Ordination analysis also confirmed that Sites 1a, 1b, 2a and 2b, all surveyed in 2010 were separated from Sites 1c and 2c. The similarity between Site 1a and 2a and 1b and 2b suggested there was some change in sites between surveys but there was little difference between sites upstream and downstream of the weir both before and after weir removal. This contrasts with findings by Mueller *et al*. (2011) who found that habitat characteristics differentiated strongly between upstream and downstream weir sides with water depth significantly higher and flow velocity significantly lower upstream than downstream. This indicates that the small size of the weir pre-removal (approximately 1 m from the streambed) resulted in negligible impacts upon the abiotic characteristics within the channel.

Site 1c and 2c were however less similar to one another than Sites 1a and 1b and 2a and 2b suggesting that one year after weir removal upstream and downstream sites were less similar to one another in the homogenous channel at Hartington. Site 2c was positively associated with standard deviation of depth suggesting there was a change in depth upstream following weir removal and variation of depth within the channel upstream increased. The annual hydrograph influences the instream hydraulics, however there was little difference in Q-values between survey years, Q94 in 2010 and Q98 in 2011, respectively. Given there was no significant difference in wetted width at sites between years, it is possible the changes in hydraulics are attributable to weir removal opposed to differences in the annual hydrograph.

Although little evidence is available detailing the outcomes of weir removal, a study by Orr *et al*. (2006) reported decreased depth and increased flow velocity within the former impoundment. Findings from the weir removal at Hartington on the River Dove contrast with those of Orr *et al*. (2006) as a significant increase in mean flow velocity and Froude number was detected in the downstream site whilst no significant changes were recorded upstream. As it is well documented that weirs impound water upstream increasing water depth and slowing flow velocity (Cowx *et a*l., 1993; Fjellheim and Raddum, 1996; Poff and Hart, 2002; Ashley *et al*., 2006; de Leaniz, 2008) it would be expected that removal of the weir would concur with findings by Orr *et al*. (2006) and cause significant changes upstream. In this study, there may have been little change in the impounded area following weir removal as the weir was of such a small size (approximately 1 m from the stream bead), and Hart (2002) proposed smaller weirs have negligable impacts upon the river system.

Weir removal summary

Little significant impact of weir removal was observed at both sites. The consequences of weir removal appeared to be different according to whether the channel was heterogeneous or homogenous. Removal of the weir from the heterogeneous channel at Dovedale appeared to reduce instream heterogeneity by increasing the similarity between HMUs. This suggests the weir pre-removal may have been encouraging habitat diversity; however, the river channel at Dovedale displays a range of habitats suggesting the loss of some similarity may not necessarily be a negative impact. Reinstatement of connectivity within the channel appears to be the most positive impact of weir removal in the River Dove at Dovedale. The River Dove at Hartington was a homogeneous channel, following weir removal, little significant impact was observed although some changes in hydraulics downstream were recorded. It appears that, similar to at Dovedale, the reinstatement of connectivity within the channel at Hartington may be the most positive consequence of weir removal. Fragmentation of the river system is a key impact of dam and weir construction (Arnekleiv and Ronning, 2004; Tomsic *et al*., 2007) therefore removal is a key step in reintroducing continuity.

The impacts of small weirs (<5m), although often less well understood than large weirs show equally important impacts on hydrological processes (Larinier, 2001; Hart *et al*., 2002; de Leaniz, 2008) and the combined effect of many small weirs can be particularly treacherous (Williams, 1998; Naughton *et al*., 2005 in de Leaniz, 2008; Thorstad *et al*., 2008) due to their number and distribution (de Leaniz, 2008). In a river such as the River Dove where small weirs are prolific, the removal of several small weirs may go some way to increase connectivity through the system. However, in terms of increasing hydraulic complexity and habitat diversity it must be questioned whether the removal of two small weirs is enough to make a significant impact.

It must also be considered, what are the long-term impacts of weir removal, and will instream hydraulic variability continue to increase? Further monitoring is required to determine if weir removal is successful in reinstating hydrological connectivity and hydraulic complexity and, as hydrological systems are controlled over a variety of scales, monitoring must record the impacts of large-scale flood events, which have the potential to have most impact upon channel configuration. The release of stored sediment and contaminants trapped within sediment (Bernacsek, 2001; Ashley *et al*., 2006) and their movement downstream following weir removal must also be considered (Bushaw-Newton et al., 2002; Stanley and Doyle, 2003). It is therefore recommended that in addition to recording the instream hydraulic consequences of weir removal, a programme of monitoring measuring substrate and water quality also be carried out.

Channel narrowing at Lowthorpe Beck, East Yorkshire

Following narrowing of the channel at Lowthorpe Beck, few significant impacts were observed. It would be expected that the channel would be significantly narrower at Site 3 as this is where the work took place but this was not the case. This may be because the berm created was a half moon shape and as transects were taken at 10m intervals, averaging of values may negate any significant change in width. It is likely that the channel is only significantly narrower at the widest point of the berm. Froude number did however increase significantly at Site 3 after channel narrowing and this may be a result of the flow of water being constricted through the site as a result of the channel narrowing. Mean channel width increased significantly at Site 2 after channel narrowing, this would concur with water flow being constricted through Site 3 and opening out at Site 2.

The aim of the work was to increase flow velocity to remove silt build up within the channel. Flow velocity however did not increase significantly at any site. It is therefore likely that there was little influence on the silt deposition within the channel. The lack of significant impact of the channel narrowing may be due to the size of the berm created or due to the location of work undertaken.

It appears that channel narrowing has had little impact on instream hydraulics at Lowthorpe Beck, however, this could be due to the assessment method used and it is recommended that future monitoring be continued to determine if there is any significant change over time. An accurate method of measuring substrate composition also needs to be applied concurrently to determine if there is any significant change in bed composition following channel narrowing.

Installation of gravels at River Stiffkey, North Norfolk

There were differences in hydraulic variables within sites between surveys regardless of whether the site represented riffle or glide habitat. Within river stretches there were differences in the hydraulic variables within sites according to which HMU they were classified as. However, significant differences between HMUs within river stretches were not consistent at sites suggesting between year differences in discharge influence how distinctive HMUs are within sites. As would be expected, there was a general pattern that wherever significant differences between sites (HMUs) were detected within survey stretches depth was generally higher in glides than riffles and flow velocity and Froude number were generally higher in riffles than glides. This suggests that the presence of the gravel installations as artificial riffles within the channel promote hydraulic complexity. Pasternak *et al*. (2008) stated gravel installation is used to restore hydraulic geometry, aiming to increase diversity of instream flow conditions (Barlaup *et al*., 2008) and Sear and Newson (2004) reported a wider range of values of physical parameters, depth, velocity and substrate conditions, following gravel installations on the River Waveney, Suffolk. As there was no pre-treatment baseline data available for the River Stiffkey it is not possible to state whether hydraulic complexity was increased following installation of the gravel riffles but it can be stated that there was hydraulic complexity within the reach where gravel installations were present. However, the absence of distinctiveness between riffles of different ages suggests there was little adjustment in physical conditions over time and installation of the gravels alone altered instream hydraulics, with differences possibly related to the annual hydrograph. It is possible the Q-values on survey dates influenced the hydraulic conditions within and between sites and although there was little difference in the Q-value, Q94 in 2010 and Q99 in 2011, respectively, wetted width was significantly different at Sites 3, 7 and 9 between years suggesting the annual hydrograph influenced instream conditions. A future programme of monitoring will clarify whether there is any change in hydraulic conditions within the reach over time. However, it must be noted that baseline monitoring should precede future river rehabilitation projects wherever possible.

4.3 Conclusions

Little impact of river rehabilitation was observed on instream hydraulics although there were some significant changes at individual sites. Determining the success of a project depends upon the original objectives of the project and what desired outcomes were anticipated. For example, one of the objectives of the weir removal projects in the River Dove was to reinstate connectivity. This may therefore be deemed successful for this objective as the barrier was removed and connectivity was reinstated, however, this was only successful within this immediate reach as weirs are so profuse within the River Dove. It must therefore be questioned whether it is enough to consider a small increase in connectivity within a small river stretch successful river rehabilitation. To maximise the benefits of any river rehabilitation project, the work must be considered from a whole catchment perspective and undertaken where maximum benefit could be achieved.

The lack of impact on instream hydraulics in this study may be because a location was selected that did not allow maximum benefit to be achieved, or it may be that the system takes time to adjust to the works undertaken and further future monitoring may reveal changes. Data in this study were collected over a relatively short timescale in

2010 and 2011, with only one baseline data set collected prior to river rehabilitation works (apart from at the River Stiffkey where it was not possible to collect baseline data). The natural hydrograph varies over time (Poff *et al*., 1997) associated with timescales of hours, days, seasons, years and longer (Stanford *et al*., 1996; Poff *et al*., 1997). Flow rating curves indicate the Q-values were different between survey dates indicating changes in hydraulic variables recorded at sites could be related to differences in discharge. However, a significant change in wetted width with increased Q-value was recorded only at the River Stiffkey meaning it is possible changes in hydraulic variables at other rivers (Rive Dove and Lowthorpe Beck) were related to the river rehabilitation project undertaken.

Furthermore, data were collected over a limited temporal period between July and September in both years, representing summer conditions. Hydrological systems operate over a variety of timescales with large scale events resulting in greatest channel change, therefore further future monitoring must be undertaken to assess the consequences of large scale events such as large floods.

The meso-scale approach used in this research depicts units of flow and substrate such as the pool-riffle (Tickner *et al*., 2000). However, these units are not permanent features and alter over temporal and spatial scales (Brookes, 1994; Pardo and Armitage, 1997; Clarke *et al*., 2003; Hauer *et al*., 2009). Data collected in this study were analysed over a meso-scale, providing mean hydraulic values for instream units. Although these methods have proved positive in detecting changes in hydraulic conditions within and between instream units, they do not allow detection of subtle, small-scale changes in hydraulic conditions. Further studies could be developed to detect small-scale changes in hydraulics by using fixed-point measurements with recording of co-ordinates to assess point-scale changes, and also to detect migration of instream units within the reach. This would also assist in bridging gaps between the micro and meso scales to give a holistic view of system hydraulics. It is also important that substrate composition is measured accurately. Changes to the amount of water or sediment for transportation through the river channel results in changes to channel morphology (Gore, 1994; Gordon, *et al*. 1994; Brookes, 1994; Harper and Everard, 1998b; Stanford *et al*., 1996; Poff *et al*, 1997). All instream hydraulics are related to sediment transport which creates the mosaic of instream features within the river channel. These features are an important determinant of the hydromorphological quality of a river, which is in turn an important aspect of good ecological status under Water Framework Directive requirements. Methods could be improved by measuring grain-size at fixed points through surveys to detect small-scale changes not detectable visually.

5 CHAPTER 5: DETERMINING THE RESPONSE OF FISH COMMUNITY COMPOSITION TO RIVER REHABILITATION

5.1 Results

5.1.1 Small-scale weir removal at Dovedale on the River Dove

The fish communities at Site 1a and Site 1c were both dominated by bullhead but, Site 1b was dominated by 0+ brown trout. The composition of fish species was relatively constant at Sites 2a, 2b and 2c with bullhead and >1+ brown trout most abundant. Sites 3a and 3b had the greatest diversity of fish species with fish species composition including 0+ brown trout, >1+ brown trout, bullhead, >1+ grayling and lamprey ammocoete; >1+ brown trout were dominant. The fish community at Site 3c was composed of 0+ brown trout, >1+ brown trout, bullhead and >1+grayling; lamprey ammocoete were absent. The fish community at Site 4a was dominated by 0+ brown trout but >1+ brown trout and bullhead were also present. Fish communities at Sites 4b and 4c were dominated by bullhead; 0+ and >1+ brown trout were also present (Figure 5. 1).

The Shannon Weiner diversity index was higher at Site 1b than at Site 1a but was lowest at 1c. At Site 2, the Shannon Weiner diversity index was highest at 2b and was lowest at 2c. Site 3 had the highest diversity index of all sites, during all surveys. The Shannon-Weiner diversity index was relatively constant at Sites 3a and 3b but was lowest at Site 3c. Site 4 had the lowest Shannon-Weiner diversity index of all sites during all surveys. Diversity was highest at Site 4a (Figure 5.2).

The evenness of species distribution decreased between Site 1a, 1b and 1c. At Site 2, the evenness of species distribution increased between Sites 2a, 2b and 2c, but remained lower than at Site 1. Species evenness was lowest at Site 3 during all surveys, than at all other sites although evenness increased between 3a, 3b and 3c. Evenness was highest at Site 4a and decreased at 4b and 4c (Figure 5.2).

At Site 1, total fish density was lower at Site 1b than Site 1a but was highest at Site 1c. Total fish density decreased between Site 2a, 2b and 2c. At Site 3, total fish density was highest at 3a and was lowest at Site 3c. Total fish density was highest at Site 4a and decreased at Site 4b and 4c (Figure 5.3).

Species richness decreased between Sites 1a, 1b and 1c. At Site 2, species richness increased between Sites 2a, 2b and 2c. Species richness increased between Sites 3a, 3b and 3c. At Site 4, species richness decreased between Sites 4a, 4b and 4c (Figure 5.3).

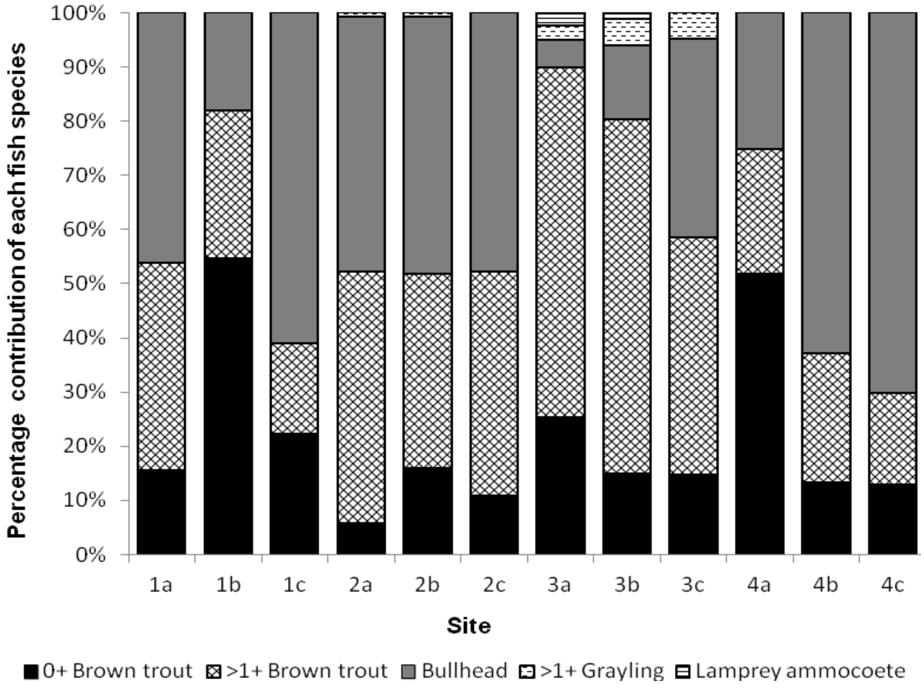


Figure 5.1: Percentage species composition at Dovedale before, after and one year after weir removal weir removal.

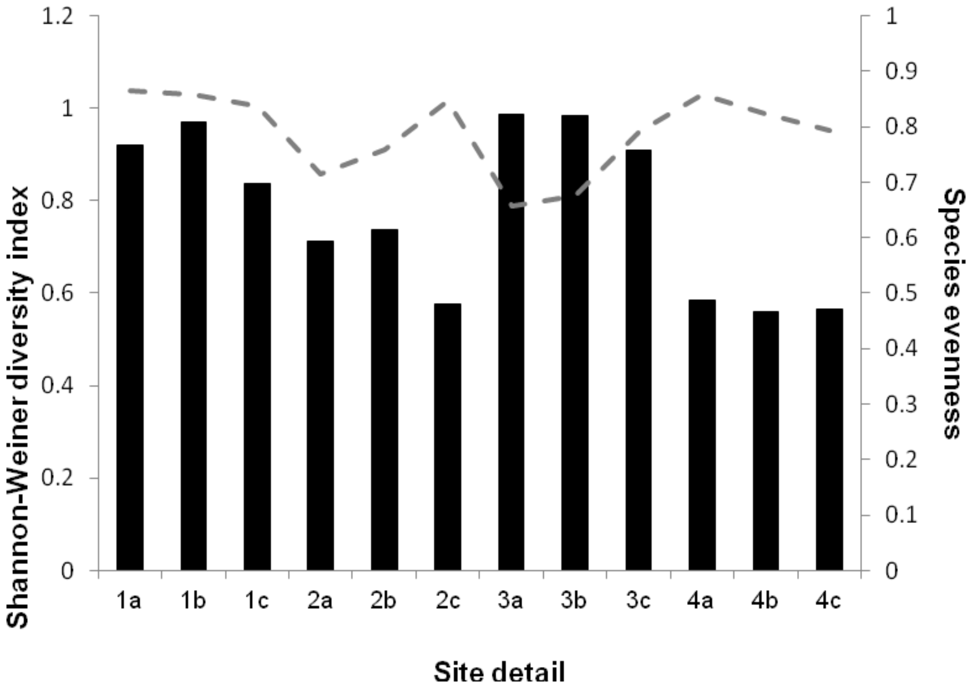


Figure 5.2: Shannon Weiner diversity index (black bars) and species evenness (broken line) at Dovedale before, after and one year after weir removal.

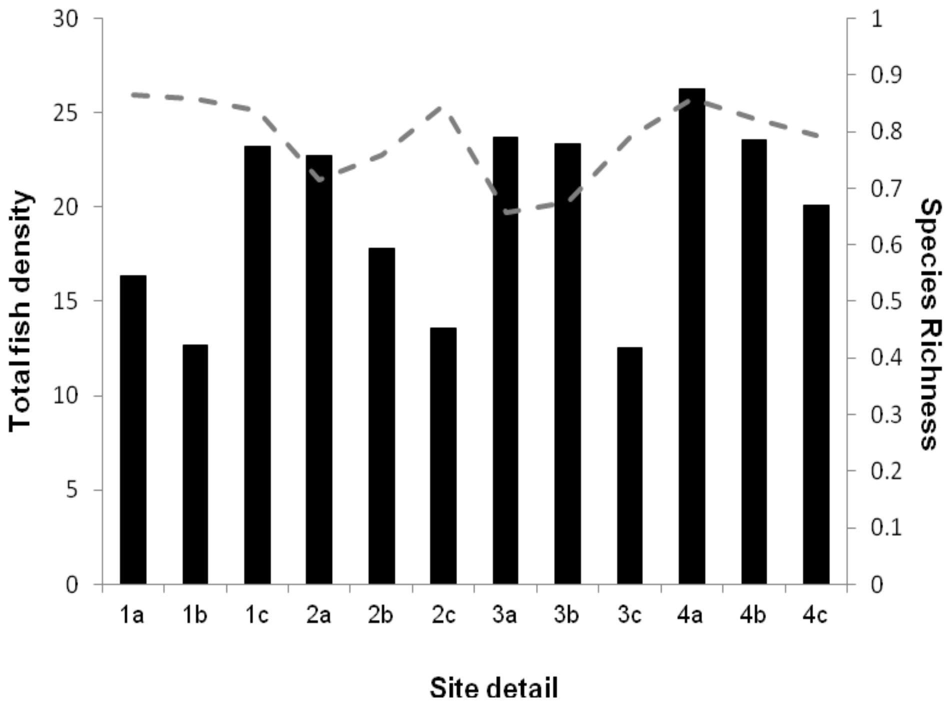


Figure 5.3: Total fish density (black bars) and species richness (broken line) at Dovedale before, after and one year after weir removal.

Differences in fish community composition between HMUs in the River Dove at Dovedale

In the heterogeneous channel at Dovedale, the fish community at riffle sites (Sites 1 and 4) generally had a greater proportion of 0+ brown trout and bullhead whereas glide sites (Sites 2 and 3) were dominated by >1+ brown trout.

Downstream of the weir, the Shannon-Weiner diversity index was higher in Site 1 than in Site 2 on all survey dates. Evenness was lower in Site 2 than Site 1 apart from one year after weir removal when evenness at Site 2c was higher than evenness at Site 1c (Figure 5.2).

Total fish density was lower in Site 1a than Site 2a and Site 2b than Site 1b but was higher in Site 1c than Site 2c. Species richness was also higher in Site 1a than Site 2a and Site 1b than Site 2b but was higher in Site 2c than Site 1c (Figure 5.3).

Upstream of the weir, the Shannon-Weiner diversity index was higher in Site 3 than Site 4 at all stages. Evenness was higher in Site 4 than Site 3 at all stages. Total fish density was higher in Site 4 than Site 3 at all stages and species richness was higher in Site 4 than Site 3 at all stages (Figure 5.3).

All sites were 40% similar in fish species composition. Sites 3a and 3b were least similar to other sites but were highly similar to one another (80%). Sites 1b and 4a were also highly similar (80%) to one another but less similar to other sites. All other sites were 60% similar in fish species composition. Sites 1c, 4b and 4c were highly similar (80%) to one another and Sites 1a, 2a, 2b, 2c and 3b were also highly similar (80%) to one another (Figure 5.4).

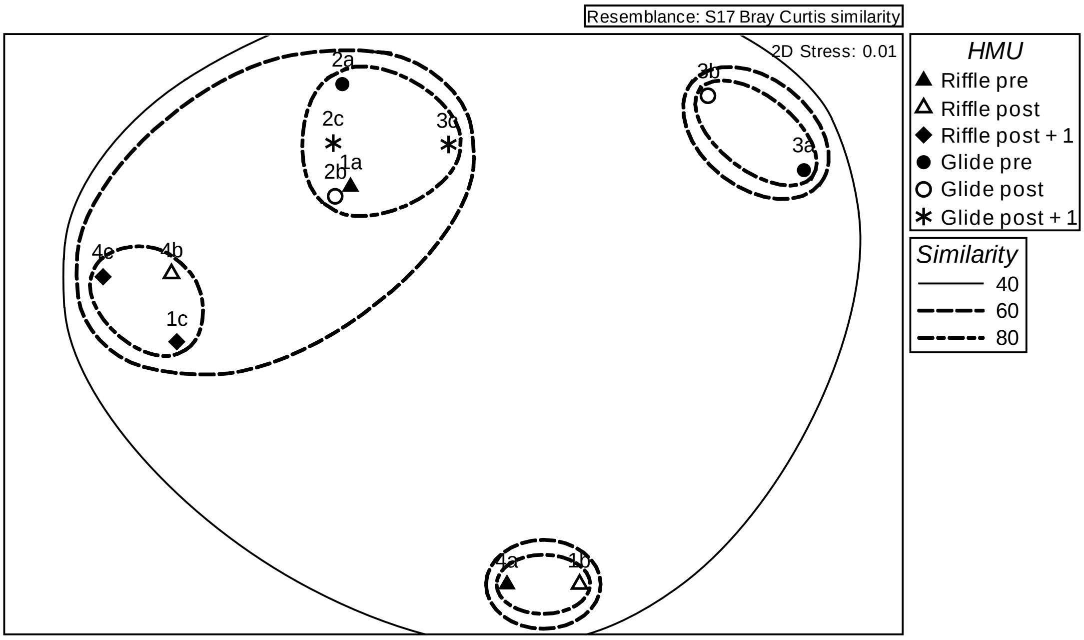


Figure 5.4: MDS plot of similarity of fish species between sites.

5.1.2 Small scale weir removal from the River Dove at Hartington

The fish community at Site 1 was dominated by >1+ brown trout. Site 1a was composed of 0+ brown trout, >1+ brown trout and bullhead. The fish communities at Sites 1b and 1c contained a higher percentage of 0+ brown trout, these sites also contained >1+ grayling. Site 1c contained 0+ grayling but this was a result of a stocking event and is consequently not used in subsequent analysis (Figure 5.5).

The fish community at Site 2 was dominated by >1+ brown trout. Site 2a was composed of 0+ brown trout, >1+ brown trout, bullhead, >1+ grayling and lamprey transformer. Site 2b was composed of 0+ brown trout, >1+ brown trout, bullhead, >1+ grayling and lamprey ammocoete. Site 2c was composed of 0+ brown trout, >1+ brown trout, bullhead, 0+ grayling and lamprey ammocoete. The 0+ grayling were the result of a stocking event so are not included in subsequent analysis (Figure 5.5).

The Shannon-Weiner diversity index increased between Site 1a and 1b and decreased between 1b and 1c, although 1c was higher than 1a. The Shannon-weiner diversity index was higher at Site 2 than at Site 1 at all surveys. Diversity increased between 2a and 2b and decreased between 2b and 2c. Diversity was lowest at Site 2c (Figure 5.6).

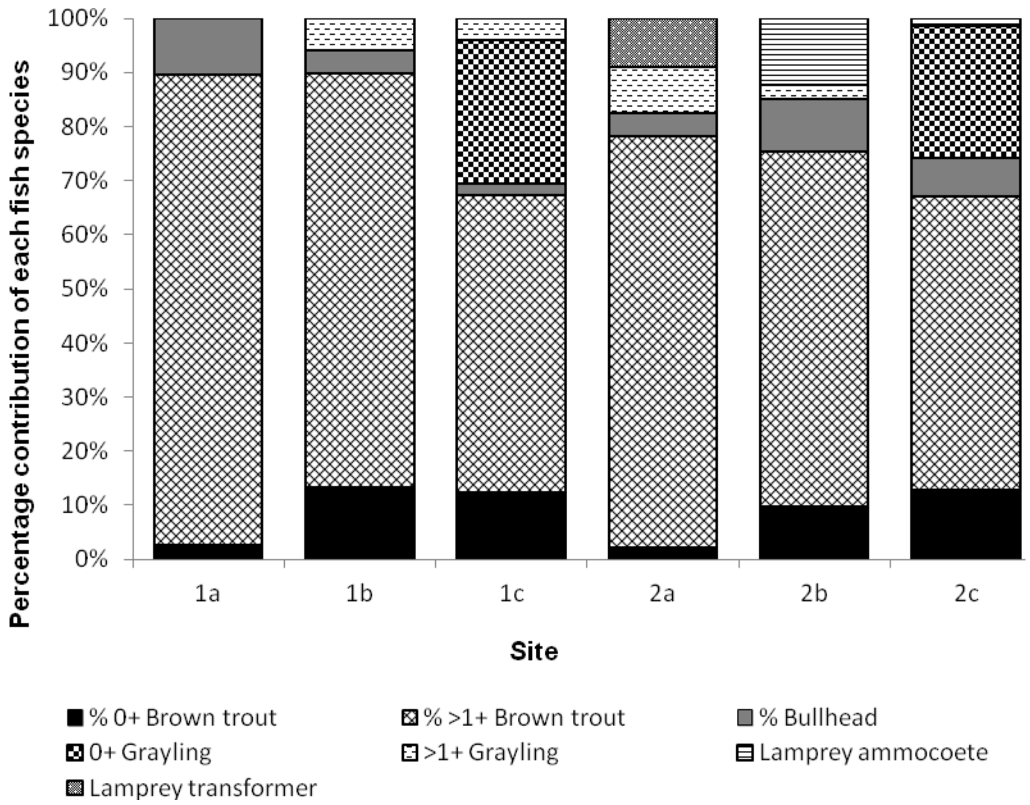


Figure 5.5: Percentage species composition at Hartington before, after and one year after weir removal.

Species evenness decreased between Site 1a and 1b but increased between Site 1b and 1c. Evenness was higher at Site 2 than Site 1 at all surveys. Evenness increased between Sites 2a, 2b and 2c (Figure 5.6).

Total fish density decreased between 1a and 1b but increased between 1b and 1c. At Site 2, total fish density increased between Site 2a, 2b and 2c (Figure 5.7).

Species richness increased between 1a and 1b then remained constant between 1b and 1c. Species richness was higher at Site 2 than at Site 1. Richness increased between Site 2a and 2b and decreased between 2b and 2c (Figure 5.7).

All sites were 60% similar in fish species composition. Site 2b was least similar to all other sites. Sites 1c and 2c were 80% similar to one another but less similar to all other sites. Sites 1a, 1b and 2a were highly similar (80%) to one another (Figure 5.8).

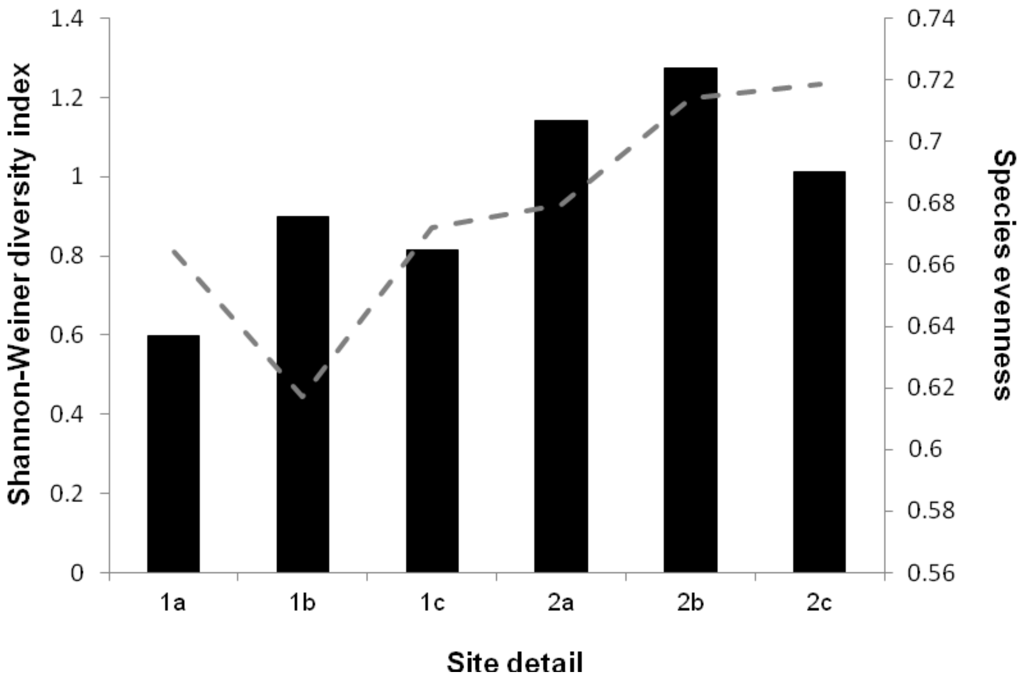


Figure 5.6: Shannon Weiner diversity index (black bars) and species evenness (broken line) at Hartington before, after and one year after weir removal.

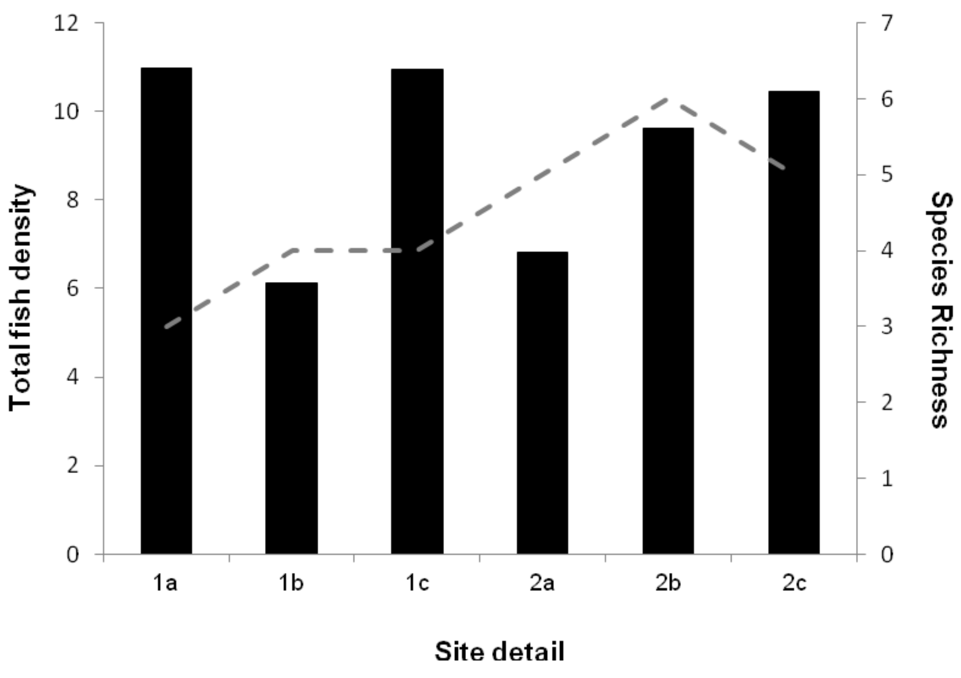


Figure 5.7: Total fish density (black bars) and species richness (broken line) at Hartington before, after and one year after weir removal.

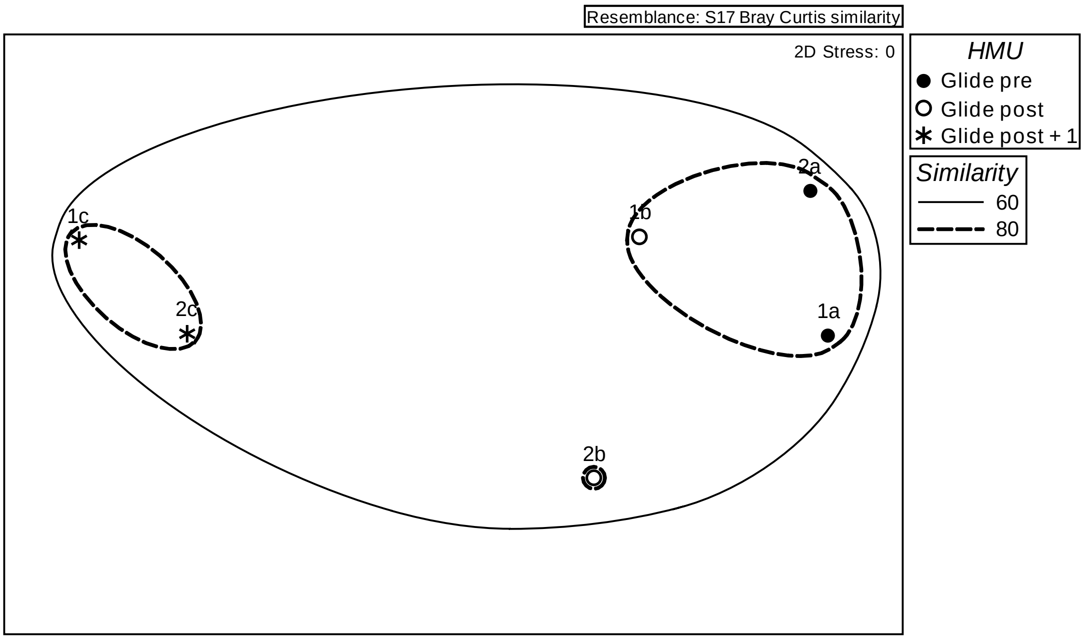


Figure 5.8: MDS plot of similarity of fish species between sites.

5.1.3 Channel narrowing in Lowthorpe Beck at Harpham

The fish communities at all sites were dominated by >1+ brown trout. Site 1a consisted of >1+ brown trout, bullhead and 3-spined stickleback. The fish community at Site 1b contained only >1+ brown trout. At Site 2a the fish community consisted of 0+ brown trout, >1+ brown trout, bullhead, 3-spined stickleback and lamprey ammocoete. The fish community at Site 2b was comprised of >1+ brown trout and bullhead. Site 3a consisted of 0+ brown trout, >1+ brown trout, bullhead and lamprey ammocoete. Site 3b consisted of >1+ brown trout, bullhead, 3-spined stickleback and lamprey ammocoete (Figure 5.9).

Shannon-Weiner diversity index was lower at Site 1b than Site 1a, and at Site 2b than 2a. Diversity was higher at Site 3b than 3a (Figure 5.10).

Species evenness was lower at Site 1b than Site 1a. At Site 2b, species evenness was higher than at Site 2a. Species evenness was higher at Site 3b than at Site 3a (Figure 5.10).

Total fish density decreased between Sites 1a and 1b, between Sites 2a and 2b and between Sites 3a and 3b (Figure 5.11).

Species richness decreased between Site 1a and 1b and between Site 2a and Site 2b. Species richness was constant between Sites 3a and 3b (Figure 5.11).

All sites were similar (60%) in fish species composition. Site 3b was least similar to other sites. All other sites were highly similar (80%) to one another (Figure 5.12).

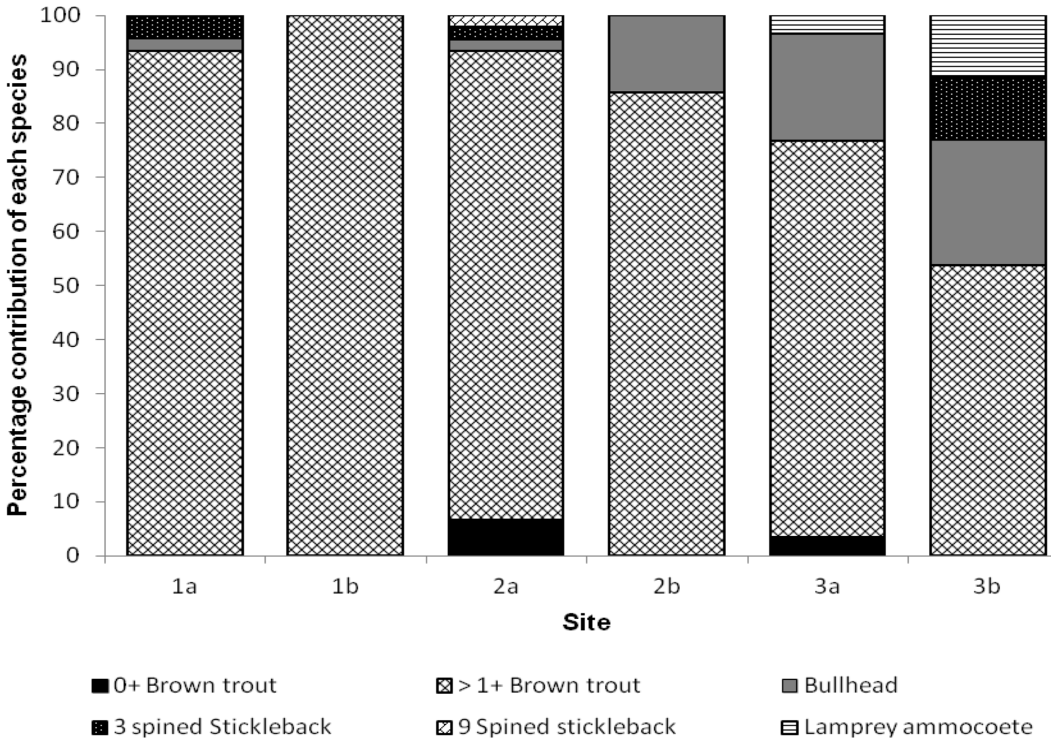


Figure 5.9: Percentage species composition in Lowthorpe Beck before and after channel narrowing.

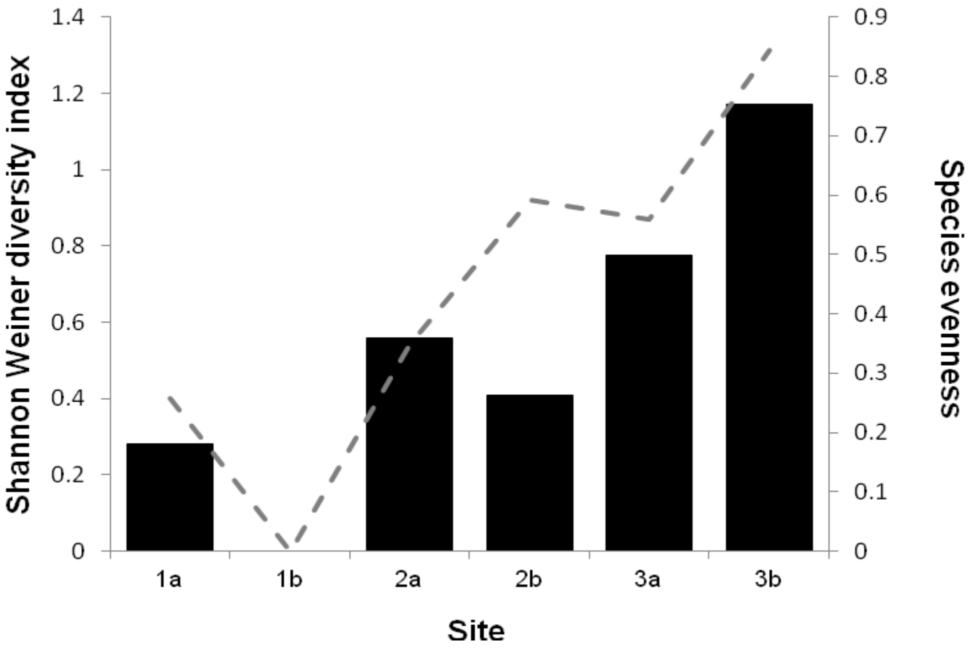


Figure 5.10: Shannon Weiner diversity index (black bars) and species evenness (broken line) at Lowthorpe Beck before and after channel narrowing.

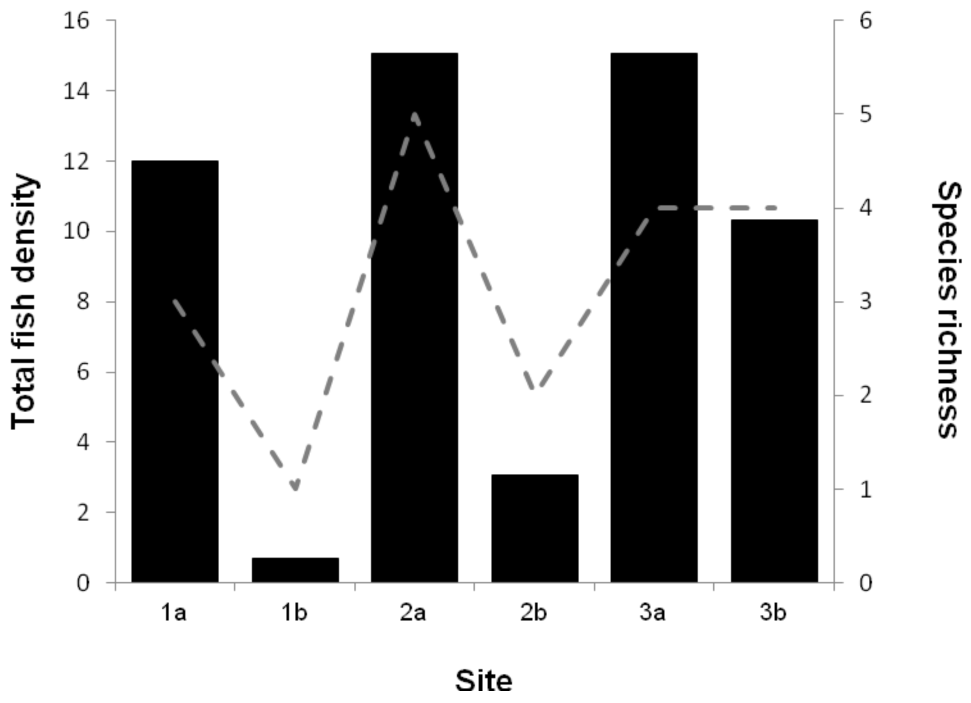


Figure 5.11: Total fish density (black bars) and species richness (broken line) at Lowthorpe Beck before and after channel narrowing.

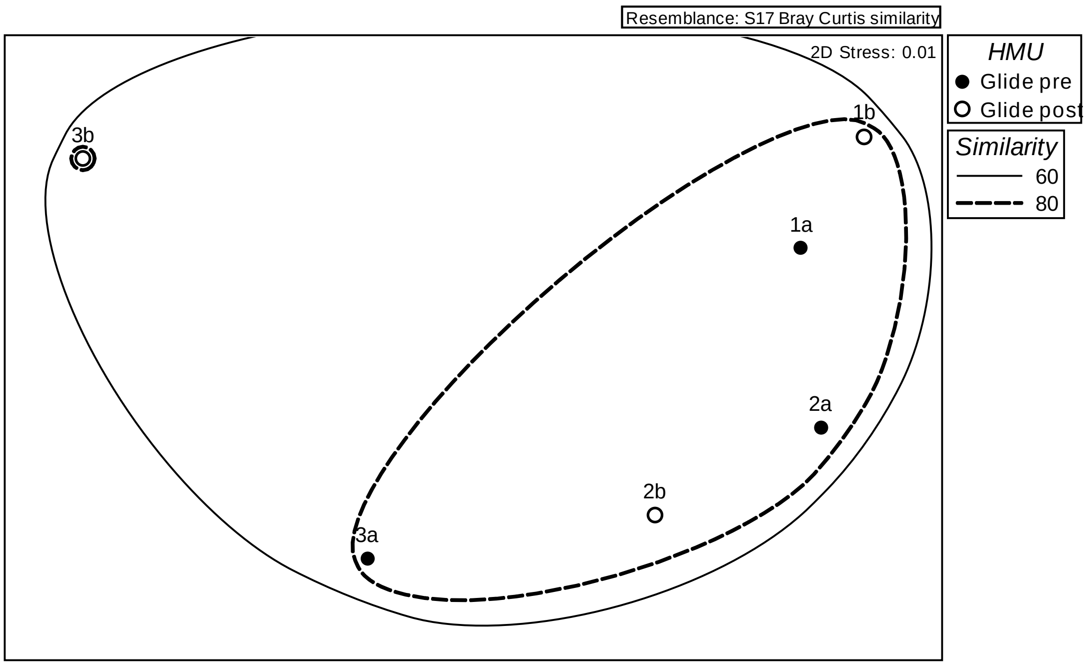


Figure 5.12: MDS plot of similarity of fish species between sites.

5.1.4 Installation of gravel riffles in the River Stiffkey, North Norfolk

The fish community at Site 1a was composed of 0+ brown trout, >1+ brown trout, bullhead, 3-spined stickleback, eel, stone loach, lamprey ammocoete and lamprey transformer. The fish community at Site 1b was composed of >1+ brown trout and bullhead. The fish community at Site 2a was dominated by bullhead with 0+ brown trout, >1+ brown trout, 3-spined stickleback, eel, stone loach, lamprey ammocoete and lamprey transformer also present. Site 2b was also dominated by bullhead with >1+ brown trout, 3-spined stickleback, eel and stone loach also present (Figure 5.13).

At Site 3, the fish community composition was composed of 0+ brown trout, >1+ brown trout, 3-spined stickleback, eel, stone loach, lamprey ammocoete and lamprey transformer. At 3b, the fish community was dominated by bullhead with 0+ brown trout, >1+ brown trout, 3-spined stickleback, eel, stone loach, lamprey ammocoete and lamprey transformer also present. The fish community composition at Site 4 was also dominated by bullhead. At Site 4a 0+ brown trout, >1+ brown trout, 3-spined stickleback, eel, stone loach, lamprey ammocoete and lamprey transformer were also present. At Site 4b, >1+ brown trout, 3-spined stickleback, eel and stone loach were also present. The greatest proportion of the fish community at Site 5a was >1+ brown trout; 0+ brown trout, bullhead, 3-spined stickleback, eel, stone loach and lamprey transformer were also present. At Site 5b, the greatest proportion of the fish community was composed of >1+ brown trout and bullhead; 3-spined stickleback, eel, stone loach, lamprey ammocoete and lamprey transformer were also present. Sites 6a and 6b were dominated by bullhead. At both Site 6a and 6b, 0+ brown trout, 3-spined stickleback and eel were also present. Stone loach were present at Site 6a but absent from Site 6b and >1+ brown trout were absent from 6a but were present at Site 6b. The fish community at Site 7a was dominated by >1+ brown trout; eel and lamprey transformer were also present. At Site 7b the greatest proportion of the fish catch was comprised of >1+ brown trout and bullhead; 0+ brown trout and 3-spined stickleback were also present. The fish community at Site 8a comprised of 0+ brown trout, >1+ brown trout and lamprey transformers but the catch was dominated by bullhead. The fish community at Site 8b was dominated by bullhead, 0+ brown trout, >1+ brown trout and 3-spined stickleback were also present but in small proportions. At Sites 9a and 9b, the fish community was dominated by >1+ brown trout. At both sites bullhead were also present. At Site 9a, 0+ brown trout and eel were present although these species were absent at Site 9b. The fish community at Site 10a was dominated by 0+ brown trout, >1+ brown trout; bullhead and eel were also present. At Site 10b, >1+ brown trout were absent, as were eel. Bullhead dominated the fish community at Site 10b, 3-spined stickleback were also present (Figure 5.13).

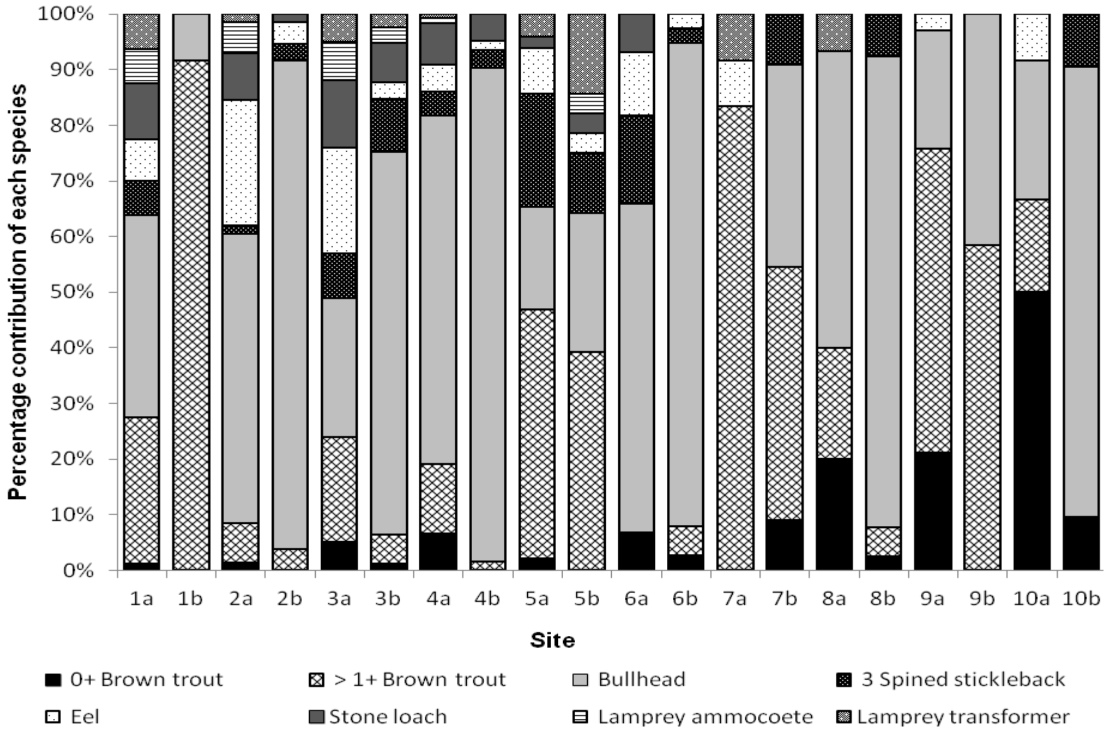


Figure 5.13: Percentage species composition at the River Stiffkey in 2010 and 2011.

The Shannon-Weiner diversity index was lower between Site 1b and 1a. At Site 2, Shannon-Weiner diversity index was lower at Site 2b than Site 2a. Shannon-Weiner diversity index was lower at Site 3b than Site 3a. At Site 4, diversity was lower at Site 4b than Site 4a. Diversity increased between 5b and 5a. At Site 6, diversity decreased between Site 6a and Site 6b. Diversity was higher at Site 7b than Site 7a. At Site 8, diversity was lower at Site 8b than at Site 8a. Diversity was lower at Site 9b than at Site 9a. At Site 10, diversity was lower at Site 10b than at Site 10a (Figure 5.14).

Evenness decreased between Site 1a and Site 1b. At Site 2, evenness decreased between Site 2a and 2b. At Site 3, evenness decreased between Site 3a and Site 3b. Evenness also decreased at Site 4 between Site 4a and Site 4b. Evenness decreased between Site 6a and 6b. At Site 7, evenness increased between 7a and 7b. Evenness decreased between Site 8a and 8b. Evenness increased between Site 9a and 9b. Evenness decreased between Site 10a and Site 10b (Figure 5.14).

Total fish density decreased between Site 1a and 1b. At Site 2, total fish density increased between Site 2a and Site 2b. Total fish density increased between Site 3a and Site 3b and also between Site 4a and Site 4b. At Site 5, total fish density decreased between Site 5a and Site 5b. Total fish density increased between Site 6a and Site 6b. At Site 7, total fish density decreased between Site 7a and 7b. Total fish density increased between Site 8a and Site 8b. At Site 9, total fish density decreased between Site 9a and Site 9b. Total fish density remained relatively constant at Site 10 (Figure 5.15).

Species richness decreased between Site 1a and 1b. At Site 2, species richness also decreased between Site 2a and 2b. Species richness remained constant between 3a and 3b. Species richness decreased between Sites 4a and 4b. Species richness remained constant between Site 5a and Site 5b and between Sites 6a and 6b. At Site 7, species richness increased between Site 7a and Site 7b. Species richness remained constant at Site 8 between Site 8a and 8b. At Site 9, species richness decreased between Site 9a and Site 9b and at Site 10, species richness decreased between Site 10a and Site 10b (Figure 5.15).

Differences in fish community composition between HMUs in the River Stiffkey, North Norfolk

The fish communities in riffles in 2010 were dominated by bullhead apart from Site 10a. In 2011 the fish communities in riffles were also dominated by bullhead. The fish communities in glides had a greater proportion of brown trout than riffles in both years although bullhead were more abundant in glides in 2011 than 2010. Most glide sites also had a greater diversity of species than riffles apart from at Stetch 4 in 2010 (Figure 5.13).

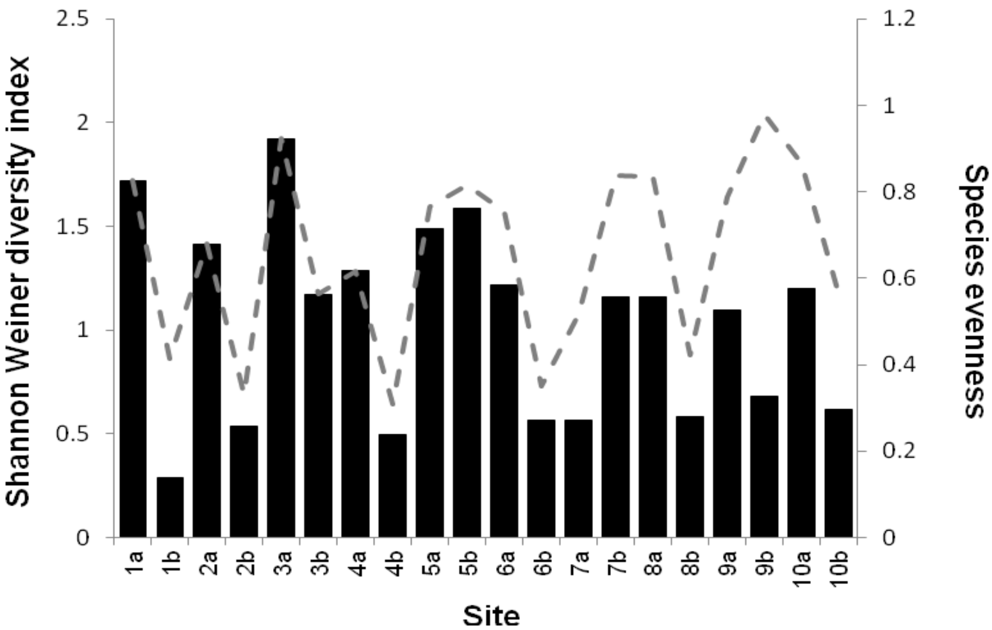


Figure 5.14: Shannon Weiner diversity index (black bars) and species evenness (broken line) at in the River Stiffkey in 2010 and 2011.

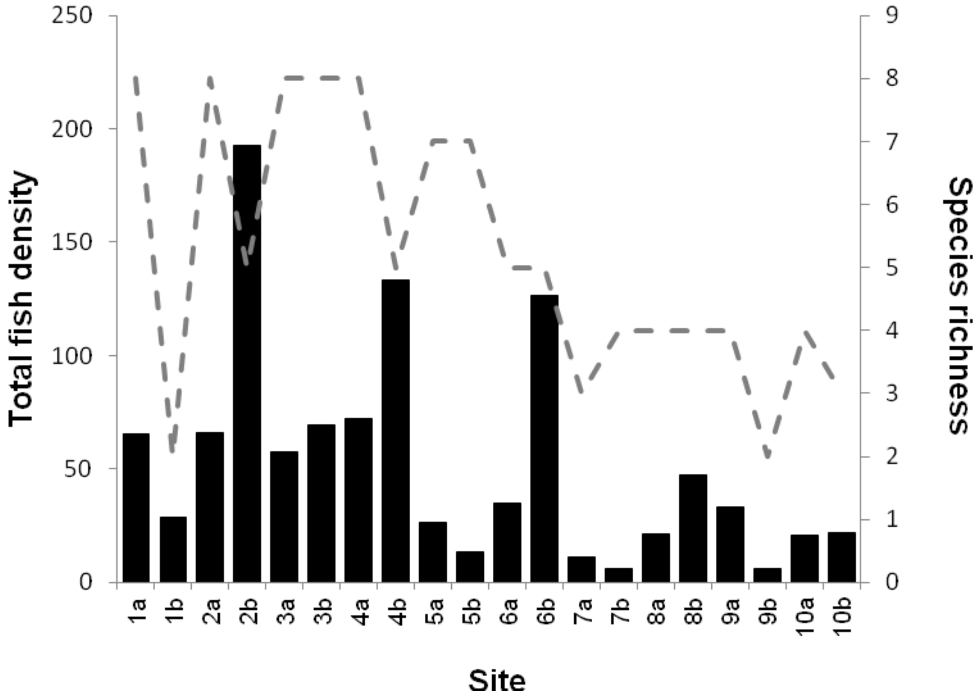


Figure 5.15: Total fish density (black bars) and species richness (broken line) in the River Stiffkey in 2010 and 2011.

In stretch 1, the Shannon-Weiner diversity index was higher in Site 1a than Site 2a but was higher in Site 3b than Sites 1b and 2b. In stretch 2, diversity was highest in Site 5 than Site 4 during all surveys. In stretch 3, Site 6a diversity was higher in Site 6a than Site 7a but was higher in Site 7b than Site 6b and in stretch 4, diversity was higher in Sites 8a, 9a and 10a than Sites 8b, 9b and 10b (Figure 5.14).

In stretch 1, evenness was highest in Site 3 during all surveys. In stretch 2, evenness was higher in Site 5 than Site 4 during all surveys. In stretch 3, evenness was higher in Site 6a than Site 7a but was higher in Site 7b than Site 6b. In stretch 4, evenness was highest in Site 9 during all surveys (Figure 5.14).

Total fish density was reasonably constant between Sites 1a, 2a and 3a in stretch 1 but was higher in Site 2b than Sites 1b and 3b. In stretch 2, total fish density was highest in Site 4 than Site 5 during all stages and in stretch 3, total fish density was highest in Site 6 than Site 7 during all surveys. In stretch 4, total fish density was highest in Site 9a than Site 8 and Site 10a but total fish density was highest in Site 8b than Site 9b and Site 10b (Figure 5.15).

In stretch 1, species richness was highest in Sites 1a, 2a, 3a and 3b. In stretch 2, species richness was highest in Site 4a than Site 5a and Site 4b than Site 5b. In stretch

3, species richness was higher in Site 6 than Site 7 during all surveys. Species richness was higher in Sites 8a, 8b, 9a and 10a than Sites 9b and 10b (Figure 5.15).

All sites were 20% similar in fish species composition. Sites 7b and 9b were highly similar (80%) and were similar (40%) to Sites 5a, 5b and 9a; these Sites were also similar (60%) to Sites 1b and 7a which were highly similar (80%) to one another.

Sites 1a and 3a were highly similar in fish species composition (80%), and were similar to Site 2a. Sites 8a and 10a were similar to one another (60%). Sites 3b, 4a and 6a were highly similar (80%) and were similar (60%) to Sites 2b, 6b, 8b and 10b which were highly similar (80%) to one another (Figure 5.16).

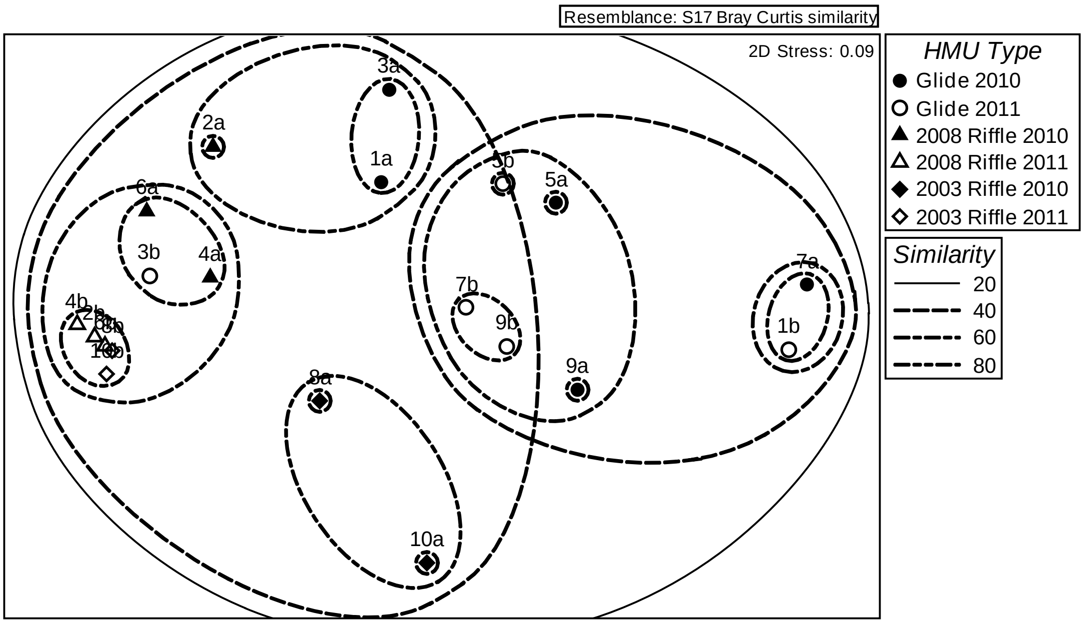


Figure 5.16: MDS plot of similarity of fish species between sites.

5.2 Discussion

Despite the increase in river rehabilitation projects throughout the UK, little monitoring of the outcomes on the fish community composition is reported. River rehabilitation projects are undertaken for a number of reasons. Following promulgation of the Water Framework Directive (WFD) into European legislation, there has been an increase in the number of river rehabilitation projects implemented to improve the physical structure and functioning of the river system. Fish are an important biological indicator of ecological status under WFD requirements. It is therefore important to have an understanding of the impact of a river rehabilitation scheme on the fish community composition within the target river to measure any improvement in the fish community or at least ensure there is no deterioration.

In this study, the impacts on fish community composition of four river rehabilitation schemes on three UK Rivers were monitored.

Weir removal from the River Dove at Dovedale

Little information exists in the literature regarding the consequences of weir removal (Doyle *et al*., 2000; Grant, 2001; Bushaw-Newton *et al*., 2002; Orr *et al*., 2006). It is well documented that weirs and dams can have implications for the fish community due to the barrier affect created by transversally blocking the river channel (Welcomme, 1994) negatively impacting continuity for migratory species (Bernacsek, 2001; Larinier, 2001; Poff and Hart, 2002; Baumgartner and Harris, 2007; de Leaniz, 2008; Thorstad *et al*., 2008; Fjeldstad *et al*., 2011). Alteration of physical habitat conditions may also be a consequence of dam and weir construction, with an impoundment created upstream increasing water depth, decreasing flow velocity and encouraging deposition of fine sediment (Cowx *et a*l., 1993; Poff and Hart, 2002; Ashley *et al*., 2006; de Leaniz, 2008). Alteration of habitat availability may result in altered fish community composition (Jackson and Marmulla, 2001; de Leaniz, 2008). It is perceived that removal of dams and weirs may restore the natural hydrological regime (Tomsic *et al*., 2007) reinstating geomorphic, thus habitat complexity and restoring connectivity for migratory species (Stanley and Doyle, 2003).

Following weir removal from the River Dove at Dovedale, there was little change in the fish species composition at all sites. Prior to weir removal Site 3a had the greatest species richness but one year after weir removal, Site 3c lost the presence of lamprey ammocoetes. Site 4a was dominated by 0+ brown trout but Sites 4b and 4c were dominated by bullhead suggesting some change in fish species composition upstream. That said, historical fisheries data from the Environment Agency (see Appendix 3) from a site downstream of the weir removal indicated that species composition at a downstream site was similar to the site surveyed and although the community was composed of the same species each year proportions of each species within the community varied. Bullheads were not recorded at the Environment Agency site downstream but were prolific in the sites surveyed for weir removal. This is likely to be due to the riffle areas present in these sites, which may not have been present at the Environment Agency site, as bullheads prefer shallow fast flowing water (see section 2.3.5).

The fish community composition between sites was fairly similar although the fish communities in glides were dominated by 0+ brown trout and the fish communities in riffles were dominated by bullhead. This would be expected given the habitat preferences of the species (see section 2.3.5). All sites surveyed before and after weir removal were very similar in fish species composition. Sites 3a and 3b were less similar to other sites but became more similar to other sites one year after weir removal. Sites 1a and 4b were also less similar to other sites but became more similar to other sites one year after weir removal. The River Dove at Dovedale was a heterogeneous channel; the impounded glide (Site 3) and upstream riffle (Site 4) had different hydraulic characteristics but became more similar following weir removal (Chapter 4), which may explain why the fish community composition became more similar.

Descriptors of community composition fluctuated between surveys. Shannon-Weiner diversity index decreased at Site 3, this was likely to be due to the lack of lamprey ammocoetes recorded post weir-removal. Evenness fluctuated between sites following weir removal however, as demonstrated by historical data from the Environment Agency this is likely to be due to the natural variation in numbers of individual species within the community. Total fish density decreased at Sites 2, 3 and 4 and fluctuated at Site 1. This may be due to the disturbance created by removing the weir causing fish to move from the area or simply due to the variation between years in the number of individuals within the population.

In conclusion, it appears the removal of a small weir from the River Dove at Dovedale has had little impact upon the fish community. Future monitoring will reveal if further changes in fish community composition occur over time.

Removal of a small weir from the River Dove at Hartington

During all surveys, the River Dove at Hartington was dominated by >1+ brown trout. Following weir removal the proportion of 0+ brown trout within the population increased at both Site 1 and Site 2. Grayling were also present at Site 1b and Site 1c following weir removal, although they were absent at Site 1a before. They also were absent from Site 2c one year after weir removal despite being present at Sites 2a and 2b. This may be due to weir removal facilitating the movement of fish into and out of the area or due to the natural variability displayed by fish populations between years. Unfortunately, no historical fisheries data was available for this site however; data from one survey close to this site in 2003 was available from the Environment Agency (Appendix 3). This showed the composition of the fish community was very similar to that recorded in weir removal surveys in 2010 and 2011, which suggests weir removal had very little impact upon fish community composition and the proportion of individual species within the community was probably influenced by the natural variability between years.

All sites were similar to one another in fish species composition. Sites 1c and 2c were most similar to one another and least similar to other sites suggesting there was some difference in fish community composition in 2011 than 2010; however this is likely to be the influence of the proportions of individual species within the population due to inter-annual variation. This is most likely as; both sites are similar in fish species composition suggesting this represents the community composition within the river stretch with little longitudinal effect of site location, and little difference between upstream and downstream sides of the weir. A long-term baseline data set would allow a firmer conclusion to be drawn regarding inter-annual variation. Descriptors of fish community composition fluctuated between surveys but no notable change was observed. Therefore it may be concluded that the removal of the small weir from the River Dove at Hartington had little impact upon fish community composition.

Weir Removal summary

Two small weirs (approximately 1 m from the streambed) were removed from the River Dove, one at Dovedale and the other close to the village of Hartington to increase connectivity through barrier removal and increase access to salmonid spawning areas. Little impact on fish community composition was observed at both sites. It is likely that the fish community composition in the weir-removal stretches doesn't alter much as weirs are so profuse within the system. It depends upon what impact the other weirs have on the system and what affect this has on the ability of fish to reach these upstream survey stretches. It is also important to note that the abundance and diversity of fish within the stretches depends upon other factors, which must be considered including water quality and availability of spawning areas elsewhere in the system. As it is not known whether fish were able to move over the weirs before removal, in future studies tracking fish movements before removal would be useful.

Channel narrowing at Lowthorpe Beck, East Yorkshire

Channel over-widening can have negative impacts for fish communities through the alteration of instream physical habitat. An over-widened channel suffers from homogeneity with pool-riffle sequences removed, increasing the presence of uniform glide habitat. Lowthorpe Beck at Harpham suffered from over-widening and displayed very uniform habitat conditions. Local agricultural land-use exacerbated the habitat uniformity through the input of fine sediment resulting in uniform flow conditions and fine silt substrate. Such conditions reduce the diversity of physical habitat for fish and remove spawning areas for Salmonid species reducing recruitment. During all surveys, and at all sites >1+ brown trout dominated the fish community. Sites 2a and 3a had 0+ brown trout present before channel narrowing but not after channel narrowing. This suggests that there may be a lack of natural recruitment within Lowthorpe Beck, which may be due to factors such as lack of suitable spawning areas elsewhere in the system. Following channel narrowing, minor species such as sticklebacks were not present in Site 1b and 2b although they were present before. Site 3a and 3b maintained the richness of species following channel narrowing, although the proportions in which species were recorded altered. Historical fisheries data from the Environment Agency (Appendix 3) indicated that the species present in the system were the same in previous surveys in 2004 and 2010, although the proportion of each species within the community varied between years. Following channel narrowing, the Shannon-Weiner diversity index decreased at all sites apart from Site 3. Total fish density also decreased at all sites. Lowthorpe Beck is used as a recreational fishery and is stocked annually (Table 5.1), stocking is often used as a management tool to maintain or enhance the fish population (Cowx and welcome, 1998). However, anglers have reported poor showing of stocked fish and heavy predation in 2011 (Alan Mullinger, *pers. comms*.); this may account for the low number of fish captured. The consistent capture of adult brown trout at Site 3 may be explained by the presence of the deep pool which provides suitable habitat for large brown trout in the form of deep slow flowing areas (Crisp, 1996; Heggenes, 1996; Armstrong *et al*., 2003; Cowx *et al*., 2004) that may also provide shelter and protection from predation (Armstrong *et al*., 2003). The consistent capture of other species at Site 3 may also be attributable to the cover provided, offering protection from predation.

It is difficult to draw a conclusion as to the influence of channel narrowing on the fish community, but considering the continuity of species captured at Site 3 and the composition of species recorded in historical data from the Environment Agency, and taking into account the reporting of predation in this area, it appears channel narrowing has had little impact upon fish community composition in Lowthorpe Beck. Continuing a monitoring programme at this site is recommended to detect changes attributable to channel narrowing as opposed to inter-annual variability or increased predation rates.

Table 5.1: Recent stocking records for Foston/Lowthorpe Beck (Data from Environment Agency)

Date	Number of Brown trout stocked
8/5-3/7/2008	435
7/5-2/7/2009	105
6/5-1/7/2010	625
5/5-7/7/2011	615

Gravel Installation on River Stiffkey

Channel uniformity can result in a lack of habitat diversity and a loss of spawning areas, particularly for salmonid species. Gravels were installed in the River Stiffkey over 2 phases in 2003 and 2008 to create artificial riffles to increase the availability of spawning habitat for migratory sea trout and resident brown trout. Surveys were undertaken in 2010 and 2011 to assess the impact of between year variations on the composition of the fish community.

There was an increase in the number of bullheads captured at all sites apart from Site 1 and Site 5 between surveys in 2010 and 2011. At all sites, 0+ brown trout, eels and stone loach decreased in number, at sites where captured previously between surveys in 2010 and 2011. Lamprey ammocoetes and transformers also decreased at all sites where captured previously apart from at Site 5 where there was an increase in lamprey transformers. Historical fisheries data from the Environment Agency at a site central to 2010 and 2011 survey sites (Appendix 3) indicated that despite the composition of the species within the fish community remaining fairly similar through the years, the proportions in which individual species within the community are captured varies between years. It is likely this is due to natural inter-annual variability of the populations and thus indicates that the variation observed in fish community structure between 2010 and 2011 is likely to be a result of inter-annual variability opposed to a consequence of gravel installation. Descriptors of community composition also fluctuate between sites and between surveys in 2010 and 2011, which is also likely to be a result of natural variability. As there was no baseline data pre-installation available, it is not possible to say if there was a change in fish community composition at each site following gravel installation. The installation of gravel riffles was intended to increase the spawning area available for sea trout and resident brown trout however there is no evidence to suggest recruitment has increased in the reach. Further studies focusing specifically on the population dynamics of brown trout in the survey stretches would be useful to provide evidence to this end. That said, although the composition of the fish community was similar at all sites, the fish communities in riffles had a greater proportion of bullhead and 0+brown trout whereas the fish communities in glides were dominated by >1+ brown trout. This would be expected given the habitat preferences of these species (see section 2.3.5) and indicates that the gravel riffles contribute to the maintenance of biodiversity within the reach.

5.3 Conclusions

River rehabilitation appears to have had little impact upon the composition of the fish community at all sites surveyed as historical data; where available, indicated there was inter-annual variation in community composition in the reaches before river rehabilitation took place. Some variation in fish community composition was observed between sites (HMUs) in heterogeneous channels. Glides were generally dominated by >1+ brown trout whereas bullhead and 0+ brown trout made up a larger proportion of the community in riffles. In most cases the river rehabilitation schemes monitored did not have objectives associated specifically with fish community composition therefore it is not appropriate to comment on the success of the project with regards to the composition of the fish community. It is however, essential to understand the outcomes of river rehabilitation projects on fish community composition given the importance of fish as a biological indicator under WFD requirements. If project design is based upon an understanding of outcomes for fish community composition, projects may be implemented that are not only beneficial to the physical state of the river channel but are also favourable to the composition of the fish community. It is recommended that river rehabilitation aim to maintain or improve habitat diversity to accommodate a number of different fish species and life stages. Catchment management must be used to identify where the most appropriate places are to undertake river rehabilitation in order to benefit the fish community, if that is the intention. Otherwise it is important that the requirements of the fish community are understood in order that no deterioration occurs.

These surveys were carried out over a time period of two years, monitoring should continue over a number of years to collect sound long-term data sets to allow more definitive conclusions to be drawn. It is recommended monitoring is done over the period of time to cover 2 generations of the longest living species (Kondolf and Micheli, 1995), which in these stream dominated by trout would be a minimum of 6 years (Kondolf and Micheli, 1995). Surveys were also, conducted to sample all species in the river and were not designed for a specific species. Therefore, species requiring specific survey methods, such as Lamprey (Cowx *et al*., 2009), were not targeted directly. It is therefore recommended that future monitoring and research programmes involve surveys targeting specific species. It must be noted that fish are influenced by a number of factors additional to the hydraulic consequences of instream river rehabilitation such as the presence of cover, water temperature, migratory barriers and water quality. All factors are also intrinsically linked to the physical quality of the stream. These factors should also be considered and monitored alongside fish populations and hydraulic conditions in future studies

6 CHAPTER SIX: ASSESSING FISH COMMUNITY STRUCTURE AND LINKAGES WITH INSTREAM HYDRAULICS AND PHYSICAL HABITAT.

6.1 Results

6.1.1 Habitat availability for all life stages of brown trout

Weir removal from the River Dove at Dovedale

At Stretch 1 habitat availability for 0+ and >1+ brown trout <20 cm decreased following weir removal but increased a year later. Habitat availability for >1+ brown trout >20 cm at Stretch 1 increased following weir removal. At Stretch 2, habitat availability for 0+ brown trout increased following weir removal but decreased a year later. Habitat availability for >1+ brown trout <20 cm decreased following weir removal but increased a year later. Habitat availability for >1+ brown trout >20 cm decreased at Stretch 2 following weir removal. (Table 6.1)

At Stretch 1a the HUI for >1+ brown trout >20 cm was higher than the HQS; lower confidence limits were higher than 1 indicating habitat use was greater than habitat availability. At Stretch 2a, HUI for >1+ brown trout >20 cm was higher than the HQS and lower confidence limits were greater than 1 indicating HUI was significantly higher than HQS. At all other stages there were no significant differences between HUI and HQS for all brown trout life stages (Table 6.1).

Weir removal from the River Dove at Hartington

At Site 1, habitat availability for 0+ brown trout increased following weir removal, as did habitat availability for >1+ brown trout less than 20 cm. Habitat availability for >1+ brown trout >20 cm decreased at Site 1 following weir removal. At Site 2, habitat availability for 0+ brown trout decreased following weir removal then increased a year after weir removal. Habitat availability for >1+ brown trout <20 cm at Site 2 increased following weir removal then decreased a year later. Following weir removal, habitat availability for >1+ brown trout >20 cm increased (Table 6.2).

At Site 1a the HUI for >1+ brown trout <20 cm was higher than the HQS; lower confidence limits were higher than 1 indicating habitat use was greater than habitat availability. At Site 2a, HUI for >1+ brown trout >20 cm was higher than the HQS and lower confidence limits were greater than 1 indicating HUI was significantly higher than HQS. At all other stages there were no significant differences between HUI and HQS for all brown trout life stages (Table 6.2).

Channel narrowing at Harpham, Lowthorpe Beck, East Yorkshire

Habitat availability for 0+ brown trout at Site 1 (Table 6.3) increased following channel narrowing. At Sites 2 and 3, habitat availability for 0+ brown trout decreased following channel narrowing. All sites decreased in the habitat available for >1+ brown trout of <20cm and >20cm following channel narrowing (Table 6.3).

At Site 1a the HUI for 0+ brown trout was significantly lower than the HQS; upper confidence limits were less than 1 indicating the HUI was significantly lower than the HQS. The HUI for >1+ brown trout <20 cm and >20 cm was higher than the HQS, lower confidence limits were > 1 indicating the HUI was significantly higher than the HQS (Table 6.3).

At Site 1b the HUI for 0+ brown trout was lower than the HQS; upper confidence limits were less than 1 indicating that HUI was significantly lower than HQS (Table 6.3).

At Site 2a the HUI for >1+ brown trout <20 cm was higher than the HQS, lower confidence limits were higher than 1 indicating HUI was significantly higher than the HQS (Table 6.3).

At Site 2b the HUI for 0+ brown trout was lower than the HQS; upper confidence limits were less than 1 indicating HUI was significantly lower than HQS. The HUI for >1+ brown trout >20 cm was higher than the HQS, lower confidence limits were higher than 1 indicating the HUI was significantly higher than the HQS (Table 6.3).

At Site 3a the HUI for 0+ brown trout was lower than the HQS; upper confidence limits were lower than 1 indicating the HUI was significantly lower than the HQS (Table 6.3).

At Site 3b the HUI for >1+ brown trout >20 cm was higher than the HQS, lower confidence limits were >1 indicating the HUI was significantly higher than the HQS (Table 6.3).

At all other sites and for all other life stages, there were no significant differences between HUI and HQS (Table 6.3).

Gravel installation at the River Stiffkey, North Norfolk

Habitat availability for 0+ brown trout (Table 6.4) increased between survey years in all stretches apart from Site 3. The amount of habitat available for >1+ brown trout of <20cm decreased in all stretches apart from site one, the most downstream site. Habitat availability also decreased for >1+ brown trout >20cm in all stretches apart from Site 1, the most downstream site (Table 6.4).

At stretch 1a the HUI for brown trout <20 cm was higher than the HQS, lower confidence limits were higher than 1 indicating that the HUI was significantly higher than the HQS (Table 6.4).

At Stretch 4a the HUI for >1+ brown trout >20 cm was lower than the HQS, upper confidence limits were less than 1 indicating the HUI was significantly lower than the HQS (Table 6.4).

6.1.2 Influence of environmental parameters on fish species captured

>1+ brown trout, grayling, lamprey ammocoetes and lamprey transformers were positively related to high mean depths. Grayling and >1+ brown trout were positively related to low standard deviation of depth and negatively related to flow velocity, standard deviation of flow velocity and Froude number. 0+ brown trout were positively related to mean width. Bullheads were positively associated with flow velocity, standard deviation of flow velocity and Froude number. Stone loach was negatively associated with mean width (Figure 6.1).

6.1.3 Relationships between descriptors of community composition and environmental parameters

Few significant relationships were detected between descriptors of fish community composition (diversity, evenness, richness, total fish density, density of 0+ brown trout and density of >1+ brown trout) and environmental variables. Negative correlations were detected between mean depth and total fish density, mean depth and density of 0+ brown trout, mean flow velocity and density of >1+ brown trout, mean width and species richness and mean width and total fish density. Positive correlations were detected between mean width and density of >1+ brown trout andFroude number and total fish density (Table 6.5).

Table 6.1: HABSCORE outputs at River Dove, Dovedale (values highlighted in yellow show habitat use is significantly higher than predicted under pristine conditions, values highlighted in blue indicate habitat use is significantly lower than predicted under pristine conditions).

Stretch number		Observed number	Observed density	HQS (density)	HQS lower CL	HQS upper CL	HUI	HUI lower CL	HUI upper CL
1	**a**								
	0+	35	5.07	3.83	1.01	14.48	1.32	0.20	8.98
	>1+ (<20 cm)	86	12.45	2.68	0.61	11.79	4.64	0.76	28.52
	>1+ (>20 cm)	23	3.33	0.70	0.22	2.16	4.79	1.54	14.89
	b								
	0+	37	4.51	3.62	0.94	13.84	1.25	0.19	8.39
	>1+ (<20 cm)	61	7.44	2.46	0.57	10.60	3.02	0.50	18.24
	>1+ (>20 cm)	24	2.93	1.60	0.51	4.97	1.83	0.59	5.72
	c								
	0+	28	3.75	3.76	0.99	14.18	1.00	0.15	6.69
	>1+ (<20 cm)	48	6.43	3.41	0.79	14.75	1.89	0.31	11.44
	>1+ (>20 cm)	48	6.43	3.41	0.79	14.75	1.89	0.31	11.44
2	**a**								
	0+	117	16.42	3.57	0.92	13.75	4.60	0.68	31.12
	>1+ (<20 cm)	100	14.04	2.41	0.54	10.76	5.82	0.94	36.14
	>1+ (>20 cm)	31	4.35	0.95	0.31	2.89	4.57	1.51	13.88
	b								
	0+	119	12.07	4.60	1.21	17.50	2.62	0.39	17.53
	>1+ (<20 cm)	73	7.40	2.08	0.49	8.75	3.57	0.60	21.16
	>1+ (>20 cm)	35	3.55	0.80	0.26	2.45	4.46	1.45	13.73
	c								
	0+	53	6.70	3.92	1.03	14.86	1.71	0.26	11.42
	>1+ (<20 cm)	49	6.19	2.34	0.55	9.93	2.65	0.44	15.83
	>1+ (>20 cm)	10	1.26	0.72	0.23	2.23	1.75	0.57	5.44

Table 6.2: HABSCORE outputs from River Dove at Hartington (Values highlighted in yellow indicate habitat use is significantly higher than predicted, values highlighted in blue indicate habitat use is significantly lower than predicted).

Site number		Observed number	Observed density	HQS (density)	HQS lower CL	HQS upper CL	HUI	HUI lower CL	HUI upper CL
1	**a**								
	0+	3	0.43	1.18	0.31	4.50	0.37	0.05	2.46
	>1+ (<20 cm)	55	7.93	0.99	0.21	4.76	7.99	1.21	52.72
	>1+ (>20 cm)	34	4.90	2.32	0.75	7.12	2.11	0.69	6.50
	b								
	0+	11	1.62	1.21	0.32	4.64	1.34	0.20	9.01
	>1+ (<20 cm)	46	6.78	1.17	0.25	5.46	5.81	0.90	37.66
	>1+ (>20 cm)	36	5.31	1.73	0.57	5.26	3.06	1.01	9.30
	c								
	0+	8	1.32	2.02	0.53	7.67	0.66	0.10	4.38
	>1+ (<20 cm)	24	3.97	1.43	0.32	6.42	2.78	0.45	17.38
	>1+ (>20 cm)	17	2.81	1.59	0.52	4.87	1.76	0.58	5.41
2	**a**								
	0+	4	0.74	1.09	0.28	4.28	0.68	0.10	4.76
	>1+ (<20 cm)	25	4.62	1.08	0.22	5.36	4.26	0.63	28.85
	>1+ (>20 cm)	24	4.43	1.12	0.36	3.47	3.97	1.27	12.40
	b								
	0+	4	0.68	0.97	0.25	3.74	0.70	0.10	4.75
	>1+ (<20 cm)	28	4.79	4.51	0.32	7.15	3.17	0.48	20.99
	>1+ (>20 cm)	16	2.74	1.55	0.51	4.68	1.77	0.58	5.35
	c								
	0+	17	2.49	2.87	0.76	10.91	0.87	0.13	5.80
	>1+ (<20 cm)	24	3.52	1.72	0.38	7.81	2.04	0.32	12.85
	>1+ (>20 cm)	24	3.52	2.01	0.66	6.17	1.75	0.57	5.36

Table 6.3: HABSCORE outputs for Lowthorpe Beck, East Yorkshire (values highlighted in yellow indicate habitat use is significantly higher than predicted under pristine conditions, values highlighted in blue indicate habitat use is significantly lower than predicted under pristine conditions).

Site number		Obs. Population size	Obs. density	HQS (density)	HQS Lower C.I	HQS Upper C.I	HUI	HUI Lower C.I	HUI Upper C.I
1	**a**								
	0+ Bt	0	0	2.57	0.64	10.38	0.10	0.01	0.70
	<20cm Bt	18	4.62	0.25	0.05	1.16	18.35	2.88	117.00
	>20cm Bt	17	4.37	1.39	0.45	4.29	3.14	1.02	9.67
	b								
	0+ Bt	0	0	2.85	0.71	11.44	0.08	0.01	0.56
	<20cm Bt	1	0.23	0.50	0.11	2.29	0.45	0.07	2.84
	>20cm Bt	3	0.68	0.82	0.27	2.49	0.84	0.27	2.57
2	**a**								
	0+ Bt	3	1	3.06	0.76	12.30	0.33	0.05	2.35
	<20cm Bt	11	3.66	0.37	0.08	1.68	9.85	1.57	61.79
	>20cm Bt	5	1.66	1.75	0.56	5.50	0.95	0.30	2.99
	b								
	0+ Bt	0	0	1.99	0.49	8.04	0.11	0.02	0.77
	<20cm Bt	2	0.44	0.33	0.07	1.56	1.33	0.20	9.04
	>20cm Bt	15	3.29	0.92	0.30	2.82	3.56	1.16	10.90
3	**a**								
	0+ Bt	0	0	3.31	0.83	13.23	0.10	0.01	0.70
	<20cm Bt	5	1.67	0.35	0.08	1.60	4.73	0.74	30.14
	>20cm Bt	10	3.35	1.17	0.38	3.60	2.86	0.88	9.28
	b								
	0+ Bt	0	0	2.73	0.68	10.93	0.15	0.02	1.03
	<20cm Bt	2	0.81	0.30	0.06	1.39	2.73	0.41	18.32
	>20cm Bt	20	8.08	0.80	0.26	2.47	10.06	3.25	31.22

Table 6.4: HABSCORE outputs for River Stiffkey (values highlighted in yellow indicate habitat use is significantly higher than predicted under pristine conditions, values highlighted in blue indicate habitat use is significantly lower than predicted under pristine conditions).

Stretch number		Obs. Population size	Obs. density	HQS (density)	HQS Lower C.I	HQS Upper C.I	HUI	HUI Lower C.I	HUI Upper C.I
1	**a**								
	0+ Bt	17	5.04	2.51	0.63	10.09	2	0.29	14.04
	<20cm Bt	42	12.44	1.33	0.31	5.73	9.33	1.55	56.13
	>20cm Bt	11	3.26	1.09	0.35	3.38	3	0.96	9.36
	b								
	0+ Bt	5	1.38	4.12	1.04	16.36	0.33	0.05	2.32
	<20cm Bt	21	5.78	1.99	0.46	8.59	2.91	0.48	17.58
	>20cm Bt	14	3.85	1.40	0.46	4.33	2.75	0.89	8.48
2	**a**								
	0+ Bt	9	2.70	3.19	0.80	12.81	0.85	0.12	5.87
	<20cm Bt	33	9.90	2.69	0.63	11.51	3.68	0.61	22.06
	>20cm Bt	14	4.20	3.78	1.19	12.02	1.11	0.35	3.53
	b								
	0+ Bt	2	0.55	3.75	0.93	15.01	0.15	0.02	1.06
	<20cm Bt	15	4.11	1.72	0.40	7.41	2.39	0.40	14.49
	>20cm Bt	15	4.11	2.45	0.78	7.71	1.68	0.53	5.28
3	**a**								
	0+ Bt	8	5.50	5.53	1.40	21.92	0.99	0.14	6.88
	<20cm Bt	10	6.87	3.92	0.88	17.42	1.75	0.28	10.86
	>20cm Bt	2	1.37	3.44	1.07	11.03	0.40	0.12	1.28
	b								
	0+ Bt	2	0.96	3.05	0.77	12.12	0.32	0.05	2.18
	<20cm Bt	3	1.45	1.32	0.30	5.71	1.10	0.18	6.65
	>20cm Bt	6	2.89	2.83	0.90	8.88	1.02	0.32	3.23
4	**a**								
	0+ Bt	32	13.14	4.69	1.17	18.83	2.80	0.40	19.74
	<20cm Bt	28	11.5	4.21	0.97	18.3	2.73	0.45	16.64
	>20cm Bt	4	1.64	7.5	2.28	24.63	0.22	0.07	0.72
	b								
	0+ Bt	6	1.61	6.52	1.66	25.62	0.25	0.04	1.7
	<20cm Bt	9	2.42	2.08	0.48	8.88	1.16	0.19	7
	>20cm Bt	3	0.81	2.18	0.72	6.65	0.37	0.12	1.13

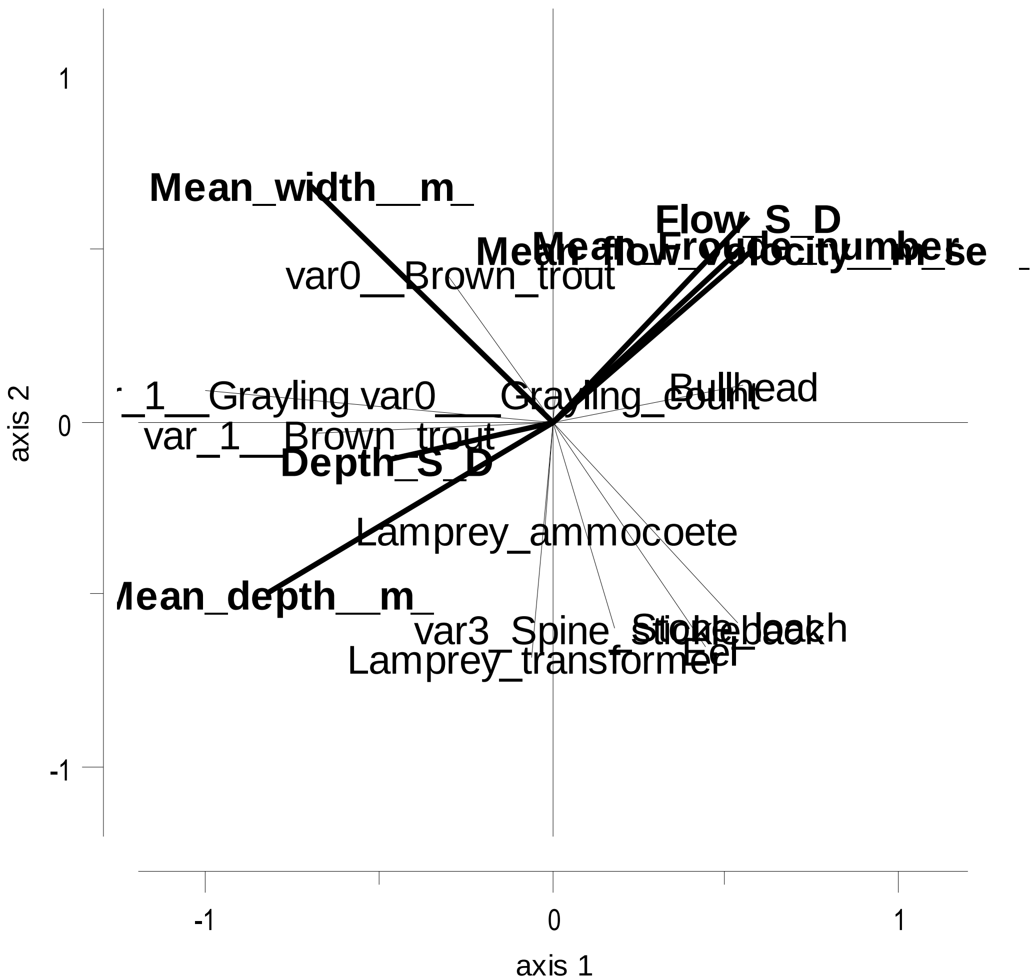


Figure 6.1: CCA plot showing association of fish species with environmental parameters at all survey sites

6.1.4 Habitat Use

Both 0+ and >1+ brown trout used a similar range of depths but results indicated 0+ brown trout preferred a shallower range of depths (0.4-0.5m) (Figure 6.2a) to >1+ brown trout (0.8m) (Figure 6.3a). Bullhead, lamprey ammocoetes, lamprey transformers, eel and stone loach were recorded in a similar depth range to 0+ and >1+ brown trout (Table 6.6) but results indicated lamprey ammocoetes (Figure 6.6a), lamprey transformers (Figure 6.7a), eel (Figure 6.8a) and stone loach (Figure 6.9a) showed preference for the mid-range recorded (0.4-0.6m), and bullhead for the lower depth range (0.3-0.5m) (Figure 6.4a).

Grayling were recorded in a narrower depth range (0.5-0.8m) than other species and results indicated preference for higher depths (0.7-0.8m) (Figure 6.5a).

0+ brown trout were recorded in a wider range of flow velocities than >1+ brown trout (Table 6.6). Results indicated 0+ brown trout show preference for flow velocities of 0.4m/sec (Figure 6.2b) but >1+ brown trout prefer a lower range of flow velocities (0.1-0.4 m/sec) (Figure 6.3b). Bullhead, lamprey transformer, eel and stone loach were recorded in a similar range of flow velocities (0.1-4.7 m/sec) (Table 6.6). Results indicated lamprey transformers (Figure 6.7b), eel (Figure 6.8b) and stone loach (Figure 6.9b) show preference for similar velocities (range 0.2-0.6 m/sec) but bullhead show preference for higher velocities (0.4-0.5 m/sec) (Figure 6.3b). Results indicated lamprey ammocoetes showed preference for similar velocities to lamprey transformers (Table 6.6) but were found in a narrower range (0.1-1.1 m/sec) (Figure 6.6b).

Results indicated grayling showed preference for a flow velocity of 0.1 m/sec although grayling were recorded in the range 0.1-0.9 m/sec (Figure 6.5b).

0+ and >1+ brown trout were recorded in a similar range of Froude numbers (Table 6.6). Results indicated 0+ brown trout showed preference for a higher range of Froude number (0.01-0.06) (Figure 6.2c) than >1+ brown trout (0.01) (Figure 6.3c).

Bullhead, lamprey transformer, eel and stone loach were recorded in the same range of Froude numbers (Table 6.6) but results indicated bullhead showed preference for a higher range (0.05-0.08) (Figure 6.4c) than lamprey transformer (Figure 6.7c), eel (Figure 6.8c) and stone loach (0.03-0.08) (Figure 6.9c). Lamprey ammocoetes were recorded in a lower range of Froude numbers (0.01-0.07) (Figure 6.6c) and results indicated a preference for 0.04-0.06.

Grayling were recorded in a wide range of Froude numbers (0.01-0.4) although results indicated preference for very low Froude numbers (0.01) (Figure 6.5c).

6.2 Discussion

Following promulgation of legislation concerning the protection and maintenance of running water habitats there has been an increase in river rehabilitation projects taking place throughout the UK and Europe. There has been a shift in the way in which river management and rehabilitation is approached, with greater emphasis placed on the structure and function of running water habitats (Clarke *et al*., 2003; Harvey *et al*., 2008), principally the interaction of river hydromorphology with ecology. Under Water Framework Directive guidelines good ecological status is measured using numerous quality indicators including fish as a biological quality indicator. All riverine processes are intrinsically linked and it is these processes, principally the flow of water and sediment through the river channel, that create the physical habitat template upon which fish can live.

The success of a river rehabilitation project depends upon what objectives were set out to be achieved and the motivation for the river rehabilitation. Project success can only be determined through ongoing monitoring and evaluation of outcomes set against clearly defined objectives (Kondolf and Micheli, 1995; Downs and Kondolf, 2002; Wohl, 2005; Bernhardt, 2007; England, 2007; Brierley, 2010; Hammond, 2011). That said, regardless of the project objectives, it is important that any instream physical river rehabilitation activity does not have a detrimental effect upon the fish community. It is therefore important to understand how abiotic factors influence fish species presence in order to improve fish populations and species diversity, or at least prevent deterioration.

As instream river rehabilitation alters the physical structure of the river channel this has some impact on the riverine processes and the structural organisation of the channel thus the amount of physical habitat available within the channel section. It is therefore important to understand the habitat requirements of species and how habitat availability will impact upon the species. Salmonid species are the most well studied and understood in terms of habitat requirements and models exist to determine the amount of habitat available within a stream section based upon this knowledge. One such model is HABSCORE (see section 2.4.1), which was used in this research to define habitat availability before and after river rehabilitation at several sites.

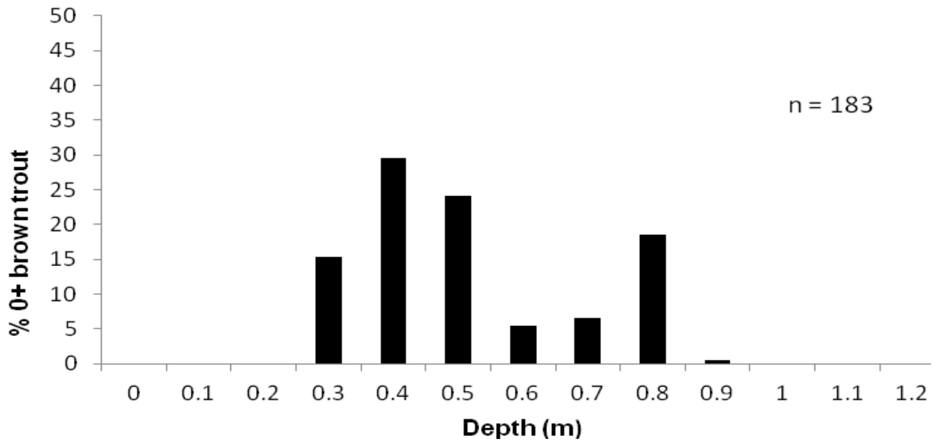
Following the removal of a small weir from the River Dove at Dovedale, there was little change in habitat availability for 0+ and >1+ brown trout <20 cm downstream of the weir however, habitat availability for >1+ brown trout >20 cm increased following weir removal. Upstream of the weir, there was little change in habitat availability for 0+ and >1+ brown trout <20 cm although habitat availability for >1+ brown trout >20 cm decreased. The River Dove through Dovedale is a heterogeneous channel with numerous weirs distributed throughout its length.

Table 6.5: Relationships between descriptors of community composition and environmental variables. Significant positive correlations are highlighted in yellow and significant negative correlations are highlighted in blue.

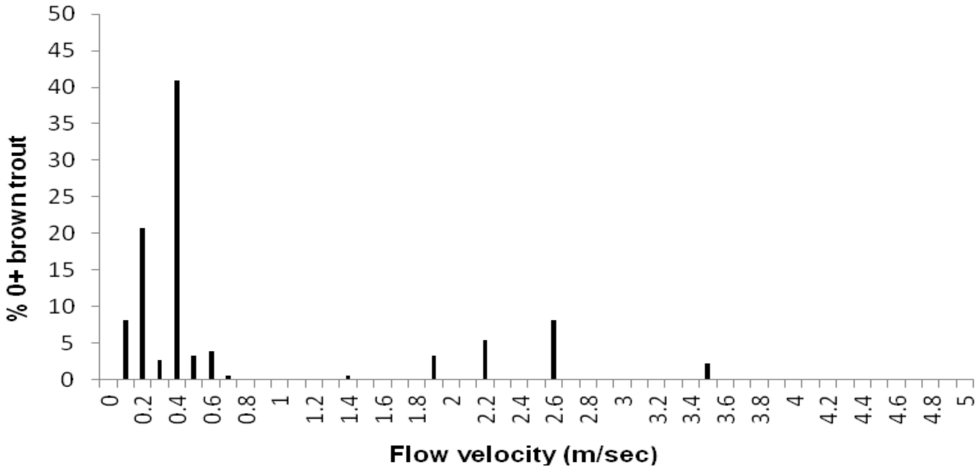
	Mean depth (m)	Mean flow velocity (m/sec)	Mean width (m)	Mean Froude number
Diversity	0.022 p = 0.892	-0.246 p =0.126	-0.220 p = 0.172	-0.118 p = 0.468
Evenness	-0.117 p = 0.472	0.000 p = 0.998	0.125 p = 0.444	0.064 p = 0.697
Richness	-0.108 p = 0.507	-0.096 p = 0.557	-0.317 p =0.046	0.047 p = 0.771
Total fish density	-0.432 p = 0.005	0.283 p = 0.076	-0.449 p =0.004	0.463 p = 0.003
0+ brown trout density	-0.459 p = 0.003	0.005 p = 0.978	0.041 p = 0.801	0.295 p = 0.065
>1+ brown trout density	-0.128 p = 0.432	-0.481 p =0.002	0.331 p = 0.037	-0.274 p = 0.087

Table 6.6: Habitat use by fish species captured in the rivers surveyed.

Species	Depth (m)		Flow velocity (m/sec)		Froude number	
	Range	Preferred	Range	Preferred	Range	Preferred
0+ brown trout	0.3-0.8	0.4-0.5	0.1-3.5	0.4	0.01-0.8	0.01-0.06
>1+ brown trout	0.3-1	0.8	0.1-2.6	0.1-0.4	0.01-0.71	0.01
Bullhead	0.3-1	0.3-0.5	0.1-4.7	0.4-0.5	0.01-1.13	0.05-0.08
Grayling	0.5-0.8	0.7-0.8	0.1-0.9	0.1	0.01-0.4	0.01
Lamprey ammocoete	0.4-0.9	0.5	0.1-1.1	0.3-0.6	0.01-0.07	0.04-0.06
Lamprey transformer	0.3-0.9	0.4-0.6	0.2-4.7	0.2-0.4	0.01-1.13	0.03-0.08
Eel	0.3-0.9	0.4-0.6	0.2-4.7	0.2-0.4	0.01-1.13	0.03-0.08
Stone loach	0.3-0.9	0.4-0.5	0.3-4.7	0.3-0.5	0.01-1.13	0.04-0.07



a)



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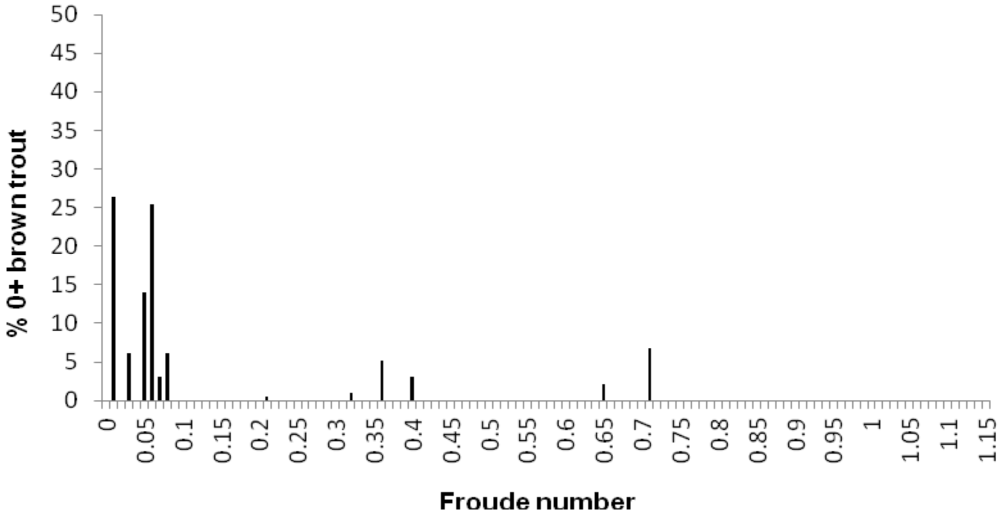
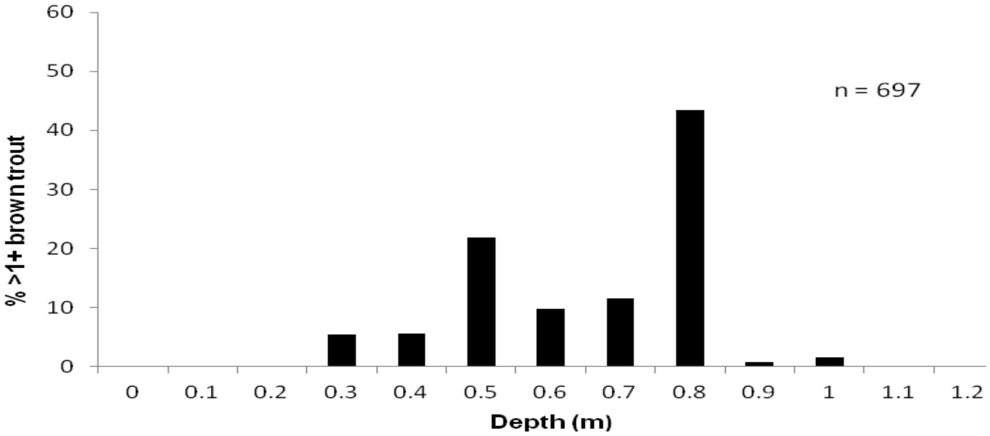


c)

Figure 6.2: Percentage of each habitat a) depth, b) flow velocity and c) Froude number used by 0+ brown trout throughout all survey sites.



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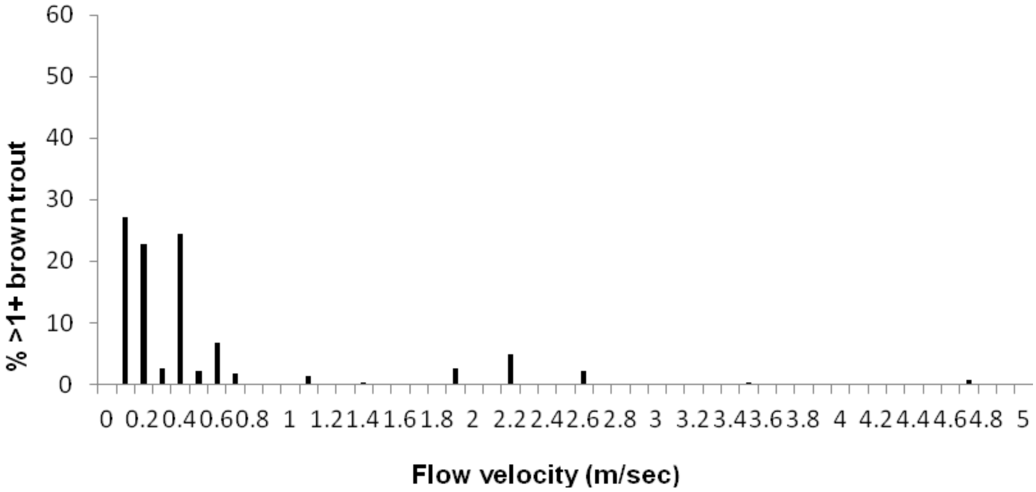

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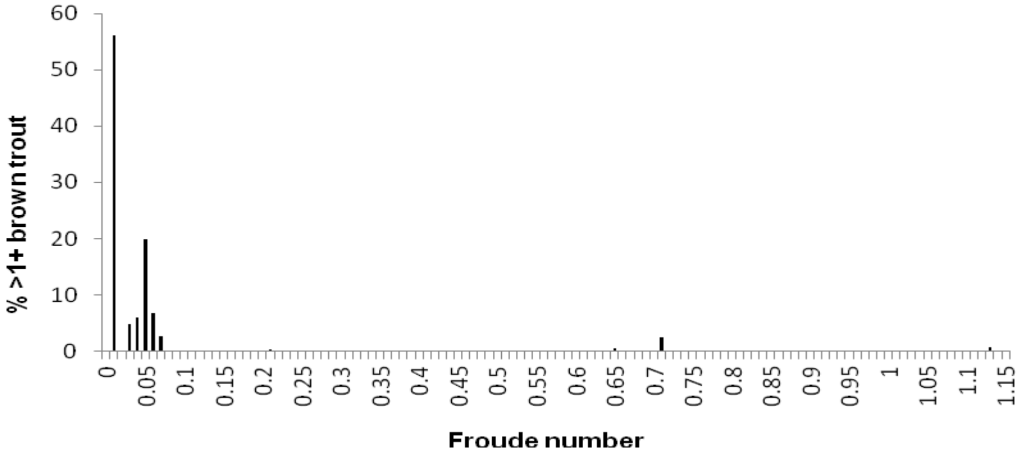
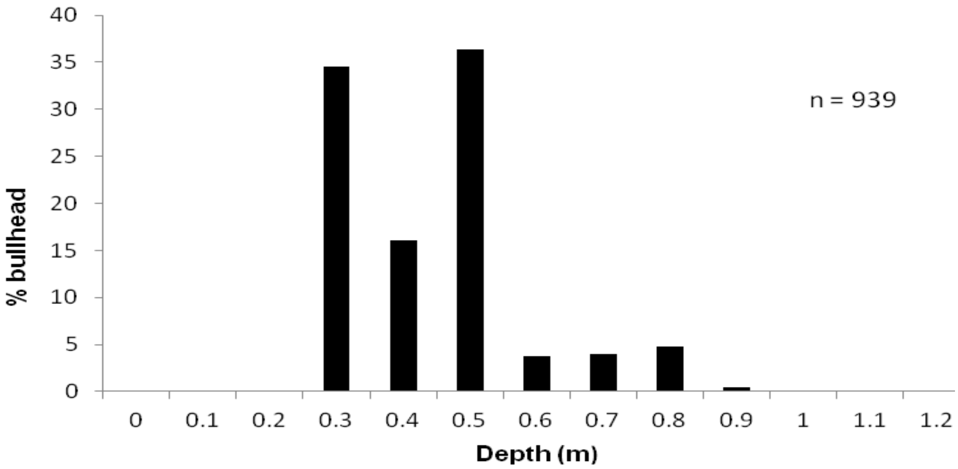


c)

Figure 6.3: Percentage of each habitat a)depth b) flow velocity and c) Froude number used by >1+ brown trout throughout all survey sites.



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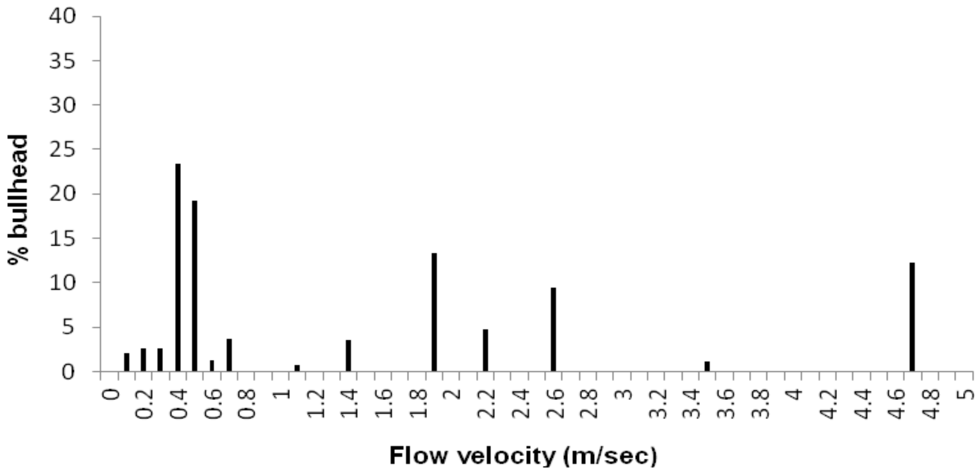

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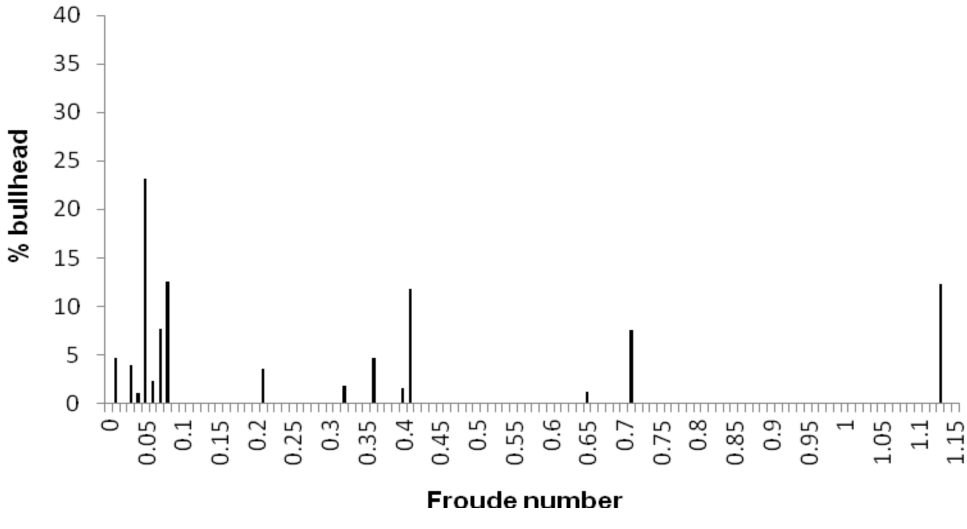
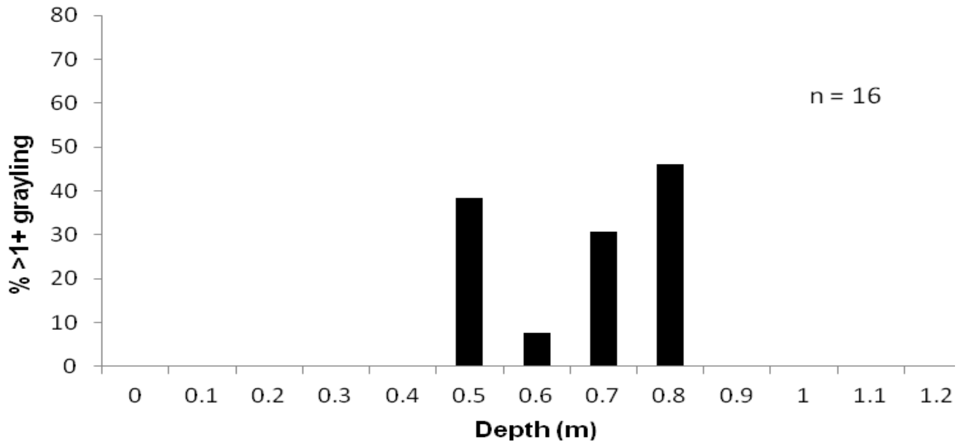


c)

Figure 6.4: Percentage of each habitat a) depth, b) flow velcoity and c) Froude number used by bullhead throughout all survey sites.



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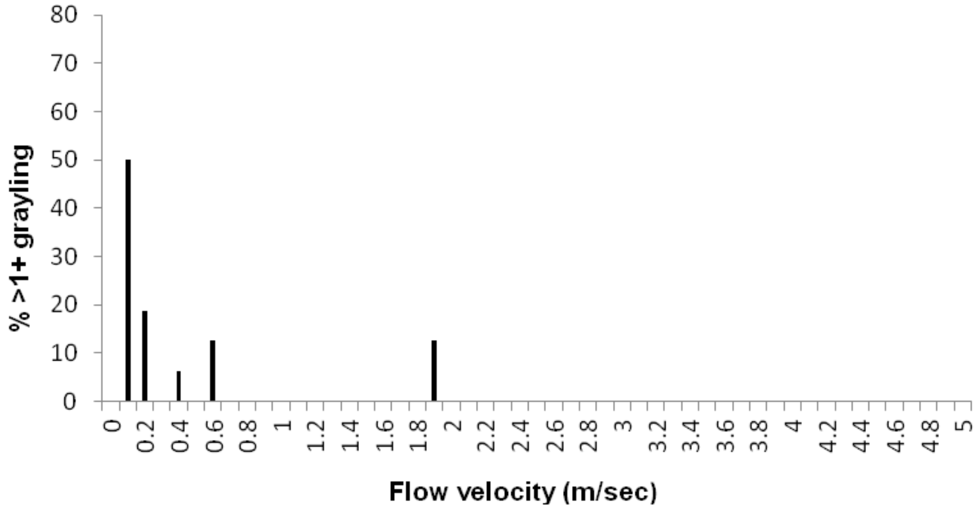

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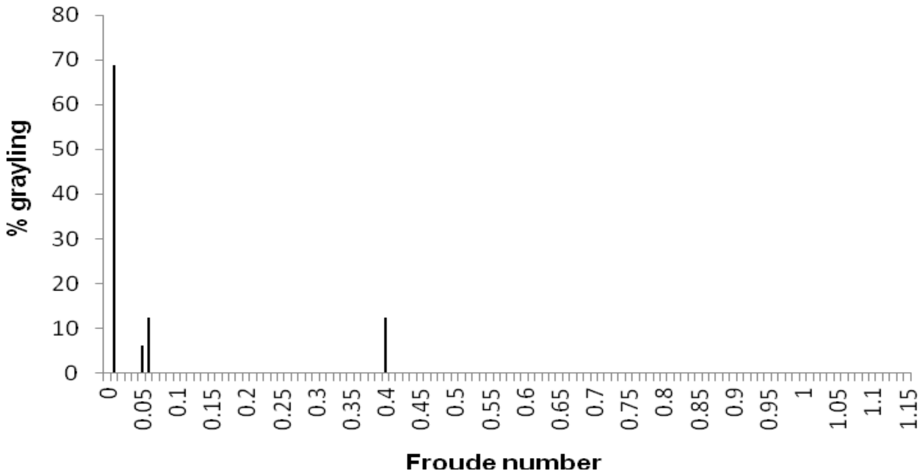
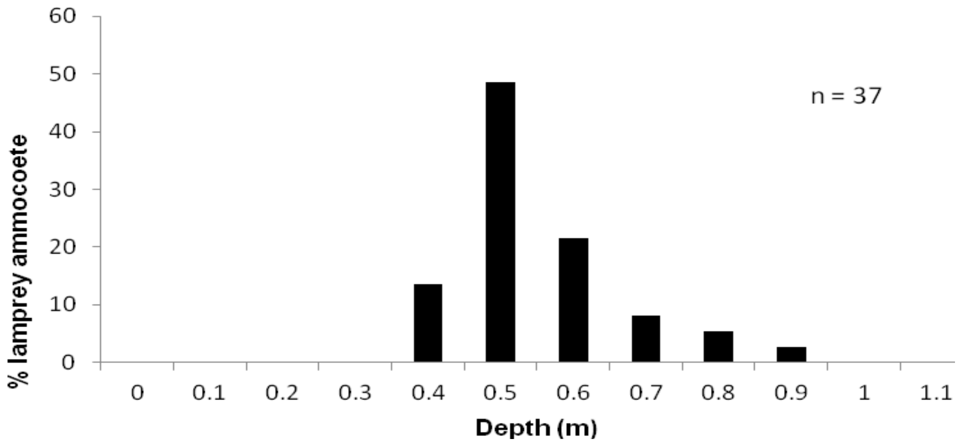
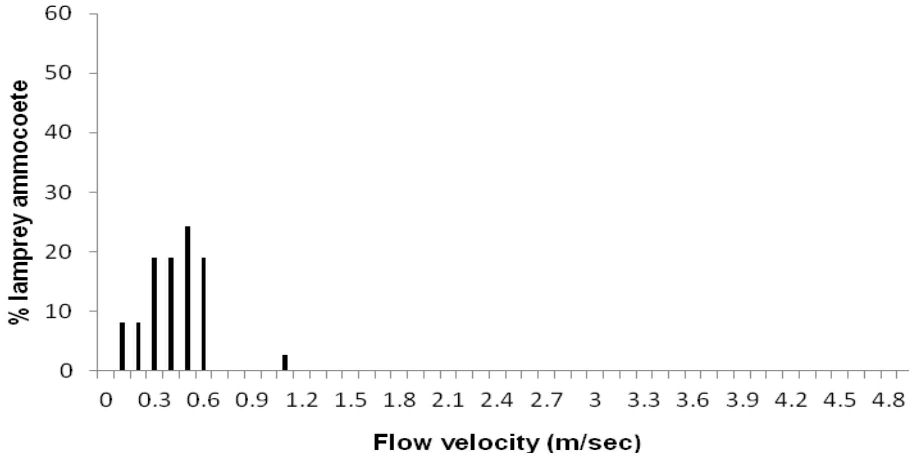


c)

Figure 6.5: Percentage of each habitat a) depth, b) flow velcoity and c) Froude number used by grayling throughout all survey sites.



a)



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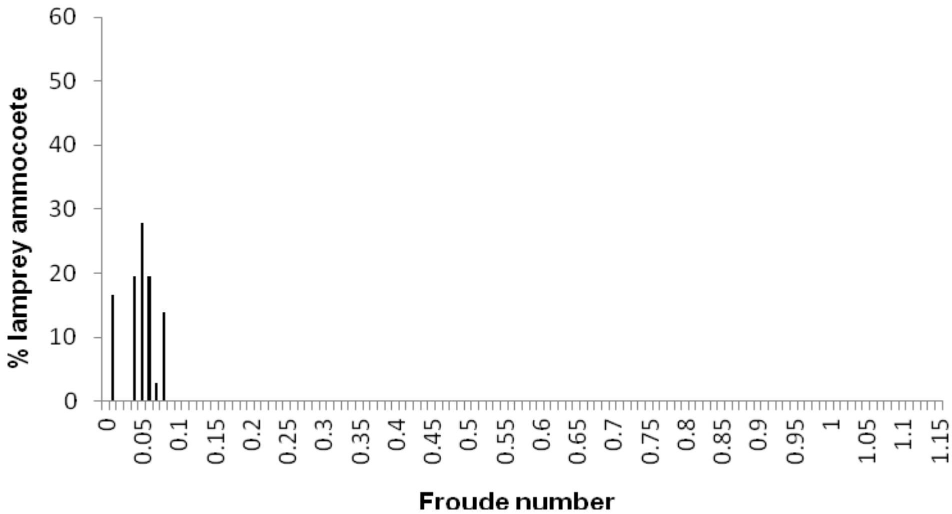
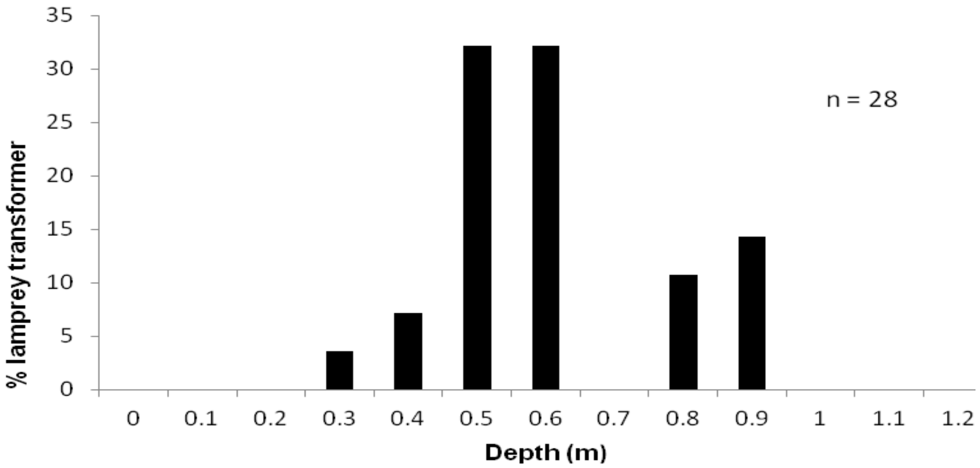


c)

Figure 6.6: Percentage of each habitat a) depth, b) flow velocity and c) Froude number used by lamprey ammocoete throughout all survey sites.



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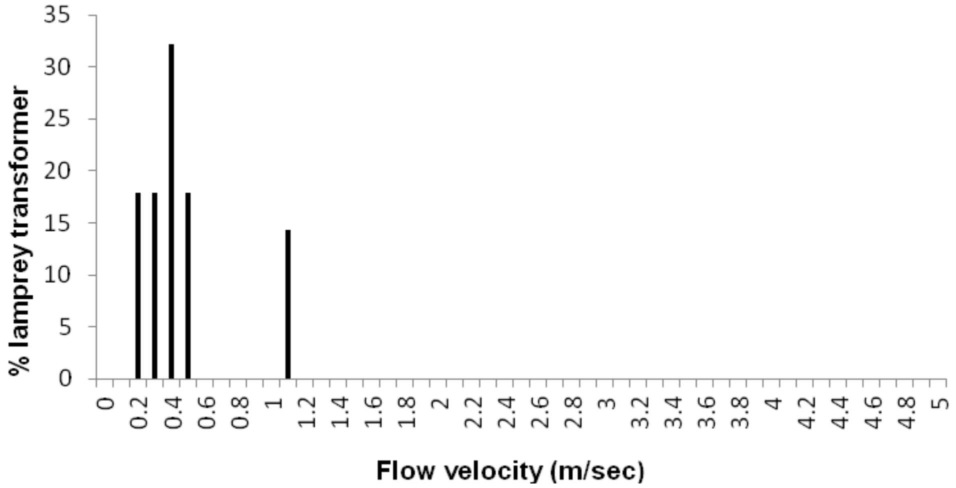

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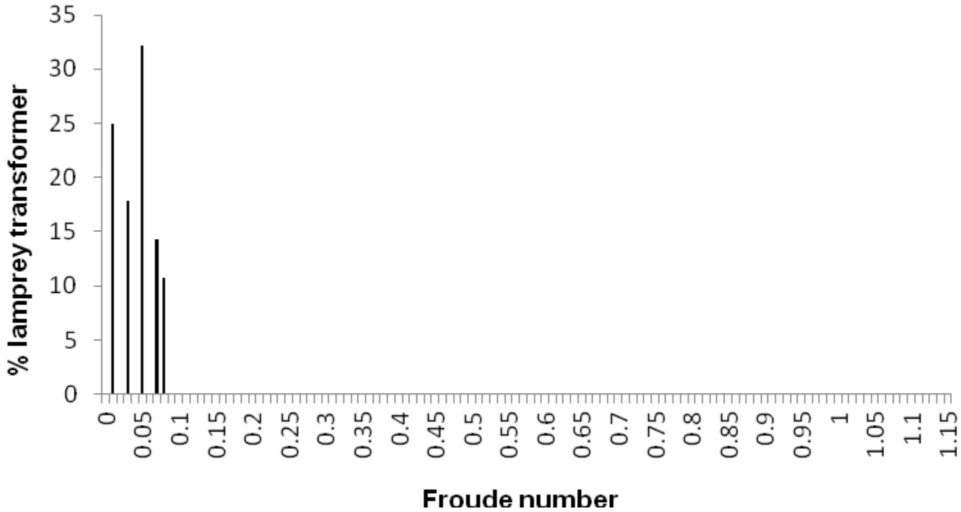
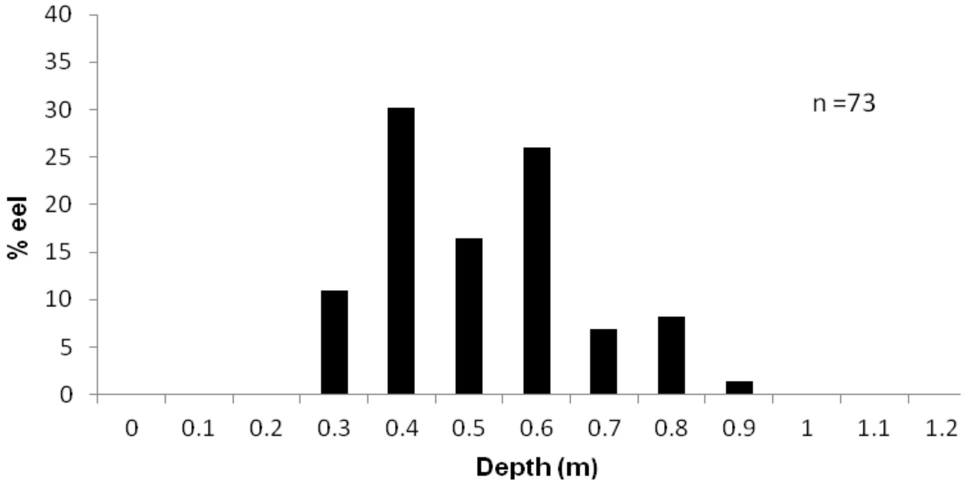
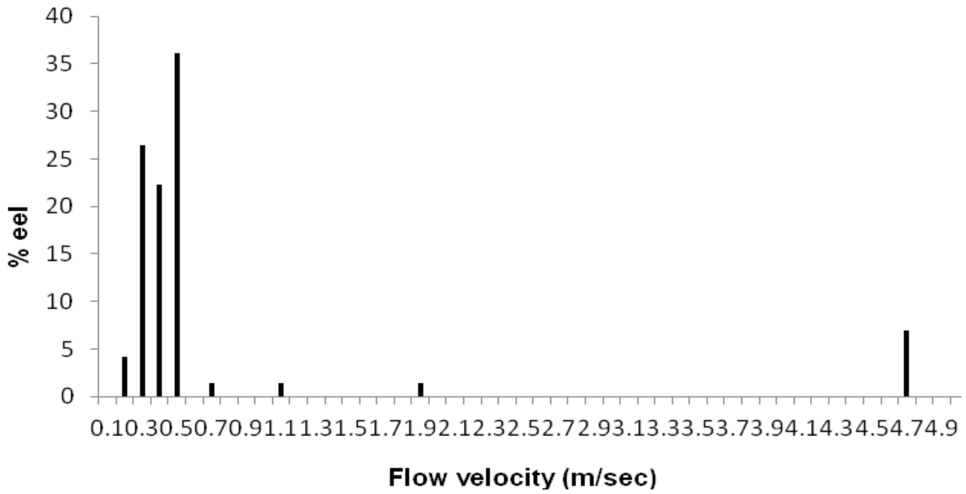


c)

Figure 6.7: Percentage of each habitat a) depth, b) flow velcoity and c) Froude number used by lamprey transformer throughout all survey sites.



a)



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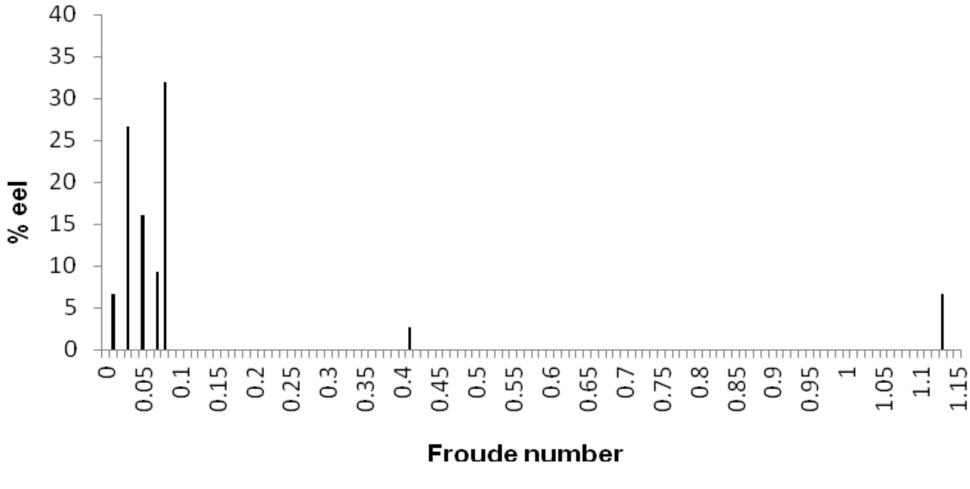
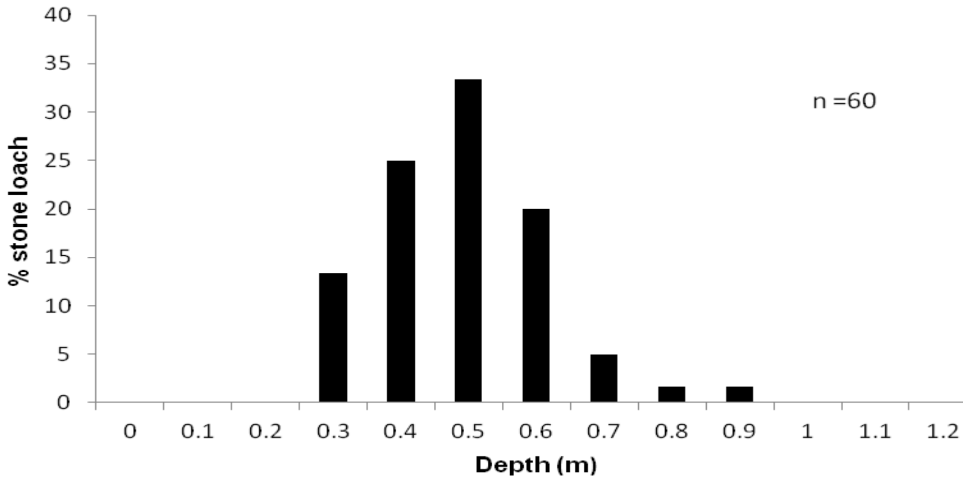
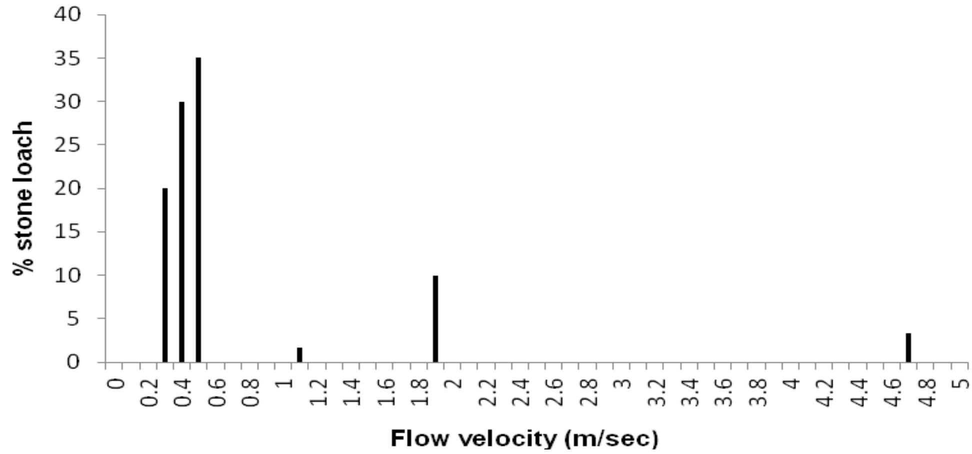


c)

Figure 6.8: Percentage of each habitat a) depth, b) flow velocity and c) Froude number used by eel throughout all survey sites.



a)



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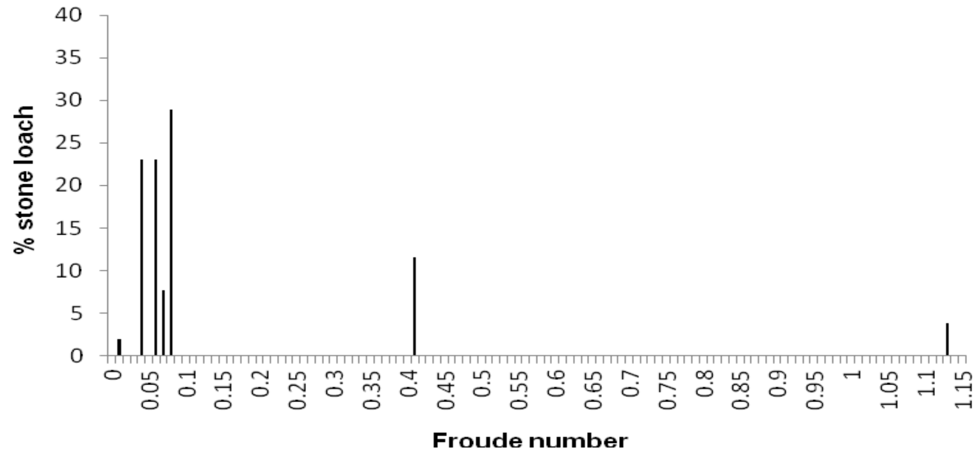

c)

Figure 6.9: Percentage of each habitat a) depth, b) flow velocity and c) Froude number used by stone loach throughout all survey sites.

The results of HABSCORE analysis suggest that removal of the small weir presents little benefit to the availability of habitat for all life stages of brown trout. This could be because the heterogeneity of the channel provided a diversity of habitat pre-removal and the small size of the weir (approximately 1 m from the stream bed) pre-removal meant it had little effect on habitat availability. With regards to habitat use by brown trout in the River Dove at Dovedale, habitat use was significantly higher than predicted (by HQS) by >1+ brown trout > 20 cm both downstream and upstream of the weir before removal. During all other surveys and for all other life stages of brown trout there was no significant difference between habitat use and predicted habitat use suggesting the sites were fulfilling their fish holding potential both before and after weir removal and weir removal has had little impact upon habitat use.

The River Dove at Hartington was a homogenous channel. Following weir removal, habitat availability for 0+ and >1+ brown trout <20 cm increased downstream following weir removal however, habitat availability for >1+ brown trout >20 cm decreased following weir removal. Upstream, weir removal had little impact on the habitat availability for 0+ brown trout and >1+ brown trout <20 cm but habitat availability for >1+ brown trout >20 cm increased. It appears weir removal had a positive impact on habitat availability for 0+ and >1+ brown trout <20 cm downstream and a positive impact for habitat availability for >1 + brown trout >20 cm upstream. However, following weir removal a large proportion of woody debris was recorded in the river. Woody debris may provide cover, an important habitat requirement for brown trout, and it is therefore difficult to distinguish whether the increase in habitat availability is a result of weir removal or the increase in in stream woody debris. In terms of habitat use, habitat use by >1+ brown trout <20 cm was significantly higher than predicted downstream and habitat use by >1+ brown trout >20 cm was significantly higher upstream, both before the weir was removed and after. During all other surveys and for all other life stages there was no significant difference between habitat availability and habitat use suggesting the sites were fulfilling their fish holding potential and weir removal had little impact upon habitat use.

At Lowthorpe beck in East Yorkshire habitat availability for >1+ brown trout <20 cm and >20 cm decreased at all sites. Habitat availability for 0+ brown trout increased at Site 1 but decreased at all other sites. This showed channel narrowing did not have a positive impact on habitat availability for brown trout. In terms of habitat use, habitat use by 0+ brown trout was significantly lower than predicted at Site 1a, 1b, 2b and 3a suggesting there was a lack of recruitment in the river, which may be due to a lack of spawning areas elsewhere within the system. At Site 1a, habitat use by >1+ brown trout <20 cm and >1+ brown trout >20 cm was significantly higher than predicted which was likely to be due to stocking as stocking was recorded in this river (Table 5.1). However, at site 2a habitat use was significantly lower than predicted for >1+ brown trout <20 cm.

Following channel narrowing, habitat use by >1+ brown trout >20 cm was significantly lower than predicted at Site 2b. Stocking was recorded to take place in the river in 2011 but predation of stocked fish by an otter was recorded by the fishing club (*pers comms* Alan Mullinger). At site 3b >1+ brown trout >20 cm habitat use was significantly higher than predicted but all fish were in a deep pool (personal observation) this may be due to this area providing refuge in the deep water from predation, as predation by an otter had been recorded by members of the fishing club (*pers comms* Alan Mullinger).

In the River Stiffkey, between years, habitat availability for 0+ brown trout increased at all stretches apart from Stretch 3. Habitat availability for >1+ brown trout <20 cm and >20 cm decreased between years at all stretches apart from Stretch 1. This indicates that either the gravel riffles became more established over time, increasing habitat availability for 0+ fish or inter-annual variation of conditions impacted upon the habitat availability between years. It is difficult to be conclusive as to the effect of the gravel installation as no data were recorded prior to installation. Results however do suggest that as habitat availability increased for 0+ brown trout, it decreased for older life stages. At Stretch 1a habitat use by >1+ brown trout < 20cm was significantly higher than predicted but at Site 4a habitat use by >1+ brown trout >20cm was significantly lower than predicted. All other sites showed no significant differences between HUI and HQS suggesting the River Stiffkey had a self-sustaining population of trout without stocking and was fulfilling its fish holding potential.

All species have specific requirements to fulfil their life history strategies. Whilst these requirements vary between species, a number of different species are frequently found in the same rivers making use of different feeding and habitat niches. Noble *et al.* (2007) found in their studies of spatially based IBIs, that a number of species were complementary to brown trout in salmonid based streams. These species are important to the composition of the community and in most cases were captured alongside brown trout in the rivers surveyed. The habitat requirements of these species may be less well known than brown trout, but it is important to understand the links between physical habitat and community composition particularly in relation to habitat alteration and river rehabilitation, and most importantly as many of these sympatric species are designated protected. It is important to identify how abiotic parameters influence the structure of the community and if any particular parameter has a greater influence than another so this may be a consideration for any future river modification.

Findings from the literature (see section 2.3) indicate bullhead (Cowx *et al*., 2004; Carter *et al*., 2004; Gosselin *et al*., 2010) and 0+ brown trout (Crisp, 1996; Heggenes, 1996; Armstrong *et al*., 2003) use shallow gravel areas of elevated flow velocity. As body size increases, brown trout use areas of deeper water and slower flow velocity (Armstrong *et al*., 2003; Cowx *et al*., 2004). Typically, lampreys, both transformers and ammocoetes, also use areas of low flow velocity and high depth with high silt content (Maitland, 2003; Gilvear *et al*., 2008). Findings from this research concur with the literature with CCA analysis indicating >1+ brown trout, grayling and lamprey ammocoetes and transformers were generally found in areas with high mean depth, low standard deviation of depth, low mean flow velocity, low Froude number and low standard deviation of flow velocity whereas bullhead were generally found in areas of high mean flow velocity, high standard deviation of flow velocity and high Froude number.

All species captured in this investigation were found within the same range of depths apart from grayling, which used a narrower range of higher mean depths. This was because of the limited number of sample sites constrained by the number of rivers sampled. Results did however indicate that within the range of depths used, species showed preference for a specific range of depths. Results indicated that lamprey ammocoetes and transformers, stone loach and 0+ brown trout showed preference for similar depths. Bullhead on the other hand showed preference for depths within the shallower range and >1+ brown trout and grayling showed preference for depths within the deeper range. These findings confirm information previously cited within the literature (see section 2.3).

Lamprey ammocoetes and transformers, stone loach and eel were found in similar preferred flow velocity ranges. Bullhead and 0+ brown trout showed preference for the same range of flow velocities. Grayling and >1+ brown trout showed preference for slow flow velocities which confirmed findings previously cited in the literature. Bullhead showed preference for higher Froude numbers, which would be expected due to their preference for low depths and high flow velocities. 0+ brown trout, lamprey ammocoetes and transformers, eel, stone loach, >1+ brown trout and grayling showed reference for low Froude numbers which would also be expected due to their preference for high depths and low flow velocities.

Findings of this research confirmed habitat use of many species previously recorded in the literature. Habitat use was collected over three rivers, two chalk streams and one alluvial river; therefore, findings are limited in application as they reflect a limited number of habitats, which influence the environmental conditions available to fish species. Surveys were also undertaken over a limited time period to correspond with river rehabilitation programmes and therefore reflect habitat use under conditions suffering disturbance due to works. It is also well documented that natural variability of in-stream conditions occurs within and between years and as a result, habitat availability and therefore use may vary. Thus it is important that monitoring is continued over a longer term to be more conclusive. That said, results suggest that whilst species have specific habitat requirements, they are able to occupy a wide range of hydraulic conditions, although numbers are greater in areas matching their preferred habitat requirements. This indicates that in order to maintain the entire fish community river rehabilitation projects should be planned to provide outcomes that benefit all species within the community. The overlapping habitat preferences of many species suggest that where conditions are appropriate for one species, conditions may also be suitable for other species. River rehabilitation projects have many different background motives but where a project is ecologically focussed it is often based upon improving habitat for Salmonid species, particularly enhancing spawning area for brown trout or salmon. If areas following river rehabilitation provide suitable habitat for brown trout, it appears suitable habitat will also be available to other complimentary species. For example, the overlaps in habitat use between eel, stone loach, bullhead and 0+ brown trout.

Few individual hydraulic parameters had any significant correlation with the structure of the fish community in the rivers sampled. Depth was negatively correlated with the density of 0+ brown trout, which would be expected given the habitat preference of 0+ brown trout. Mean flow velocity was negatively correlated with the density of >1+ brown trout which would also be expected given the habitat preference of >1+ brown trout for deep slow flowing water (see section 3.3.1).

Mean depth and mean width were negatively correlated with total fish density, which was likely to be due to the area of water available to hold fish, as mean width was positively correlated to the total density of >1+ brown trout. Mean width was also negatively correlated with species richness, this however may be due to the number of rivers sampled as the River Stiffkey, which was narrow had the highest species richness with some species (eel, stone loach) found only in this river, out of those surveyed.

Froude number was positively correlated with total fish density. This is likely to be due to the number of bullheads, which were prolific in surveys, particularly in 2011, and show preference for shallow water with higher flow velocities.

Results of this research suggest that whatever the background motives for river rehabilitation, consideration must also be given to the fish community present within the target river and habitat should be maintained or improved for all fish within the community. Most river rehabilitation activity focuses upon creating certain structures or channel forms that are perceived to provide good habitat (Beechie, 2010) however, this research suggests that fish are able to populate an area where the conditions are appropriate whether this is within a specific structure or not and the importance lies in having a diversity of habitat available, however this is created and maintained. It is unlikely river rehabilitation measures will achieve their desired outcomes where knowledge of the catchment processes are not known and not accounted for in project design (Wohl, 2005; Brierley, 2010), meaning an inter-disciplinary approach to river rehabilitation is useful (Roni and Beechie, 2013).

It must however be noted that fish respond to a number of factors in addition to instream hydraulics, and fish distribution is also likely to be limited by these factors which include water quality, water temperature, river connectivity, cover, food availability and predation.

This research was conducted between July and October and therefore represents summer habitat availability and use. Fish require different habitats at different life stages to fulfil specific requirements, therefore habitat use varies and the findings here may only be considered representative of summer habitat use. Further research should be conducted to observe habitat availability and use throughout the year and tracking studies would be useful for this.

Habitat use data were collected on a meso-scale using methods detailed in Chapter 2. Values of hydraulic variables were calculated by averaging transect data collected in each meso-habitat type therefore, although data represent average hydraulic conditions in a hydraulic unit, recordings were not made at the location each individual fish was captured. Therefore, whilst findings give a general indication of fish-habitat preferences further investigation, including point-abundance sampling would be required to confirm species-specific habitat preference.

Due to the number of streams surveyed, and the focus on small, wadeable Salmonid streams, caution must be applied if comparing results with other rivers. Also, a limited number of some species, such as eel and grayling were captured, for example the presence of eels only at River Stiffkey, which is a small chalk stream, dominated by brown trout, where surveys were close to the point of discharge into the sea. This gives limited representation of habitat use. It must, however, be noted that despite the limitations provided by the small range of area surveyed and limited number of individuals captured, general findings concur with those previously cited in the literature. It must also be noted that surveys were conducted to capture a representative sample of all species present and were therefore not targeted to specific species. It is recommended that future work involve species-specific surveys, especially for lampreys.

6.3 Conclusions

The four river rehabilitation projects monitored in this study showed little influence upon the habitat availability and use by brown trout. The river rehabilitation projects monitored were not implemented to specifically increase habitat availability for brown trout however, due to the importance of fish as a biological indicator of ecological status under WFD requirements, and the commercial value of brown trout it is important to understand the outcomes of river rehabilitation projects on the habitat availability and use of this species. Other sympatric species are also often found in trout-dominated river stretches and it is important to understand the habitat requirements of these species in order to promote biodiversity within river stretches. This research indicated that different species show preference for different habitat conditions. This has implications for the design of river rehabilitation projects as it is important that project implementation at least has no deleterious effect on the fish community, and wherever possible is beneficial to the fish community. There is a risk associated with river rehabilitation projects that work is directed at achieving a “target river” with a specific set of in-channel conditions, but this research highlights the need to protect and maintain habitat diversity to protect biodiversity. Therefore any river rehabilitation programme must consider the needs of the entire fish community through habitat diversity rather than attempting to manufacture specific hydraulic conditions.

7 CHAPTER 7: GENERAL DISCUSSION

This study was intended to provide information on biological (fish) and environmental (hydraulics) impacts of river rehabilitation schemes. This information is important as the number of river rehabilitation projects taking place is increasing following promulgation of legislative directives such as the Habitats Directive (HD) and the Water Framework Directive (WFD). Findings are intended to reveal the outcomes of several small-scale river rehabilitation projects to aid the design of future river rehabilitation schemes as pressure increases to improve the physical and biological quality of river systems.

The WFD uses several quality elements to define ecological status including biological, chemical and physical quality elements. Implementation of the WFD has seen a change in the way in which rivers are managed with increased focus placed upon physical conditions and processes. Many projects are thus focused on improving the hydromorphological quality of rivers, as, although hydromorphology is not a quality element used to define ecological status, it is an important supporting element upon which ecological status is based. This however has implications for aquatic biota; particularly fish as hydromorphology influences fish species diversity and abundance both directly through riverine connectivity and indirectly through creation of the habitat mosaic upon which species depend. This research aimed to assess the results of river rehabilitation schemes on both instream hydraulics and fish species composition and assess the linkages between fish community composition and instream hydraulics. Primary objectives were:

- to review how river form and function provides physical habitat for fish
- to determine the physical outcomes of river rehabilitation in terms of instream hydraulics
- to determine the response of fish community composition to river rehabilitation
- to assess fish community structure and linkages with instream hydraulics

Monitoring of a river rehabilitation procedure aims to determine whether a scheme is working as planned based on the measurement of specific parameters (England, 2007). It is imperative monitoring is undertaken and outcomes, positive or negative, reported in order to inform future river rehabilitation projects through adaptive management (Downs and Kondolf, 2002). This research aimed to meet the above objectives by monitoring the outcomes of four river rehabilitation projects implemented on three UK Rivers, each with different objectives (detailed below).

These were:

- Small-scale weir removal at two sites (Dovedale and Hartington) on the River Dove, Derbyshire

  - To increase hydrological connectivity
  - To improve connectivity for fish migration
  - To increase hydraulic diversity
  - To increase habitat diversity and availability, particularly access to upstream spawning grounds for brown trout

- Narrowing of an over-widened channel on Lowthorpe Beck, East Yorkshire

  - To increase flow velocity through the area to encourage transportation of fine sediment
  - To increase availability of cover and bank side habitat

- Installation of gravels as artificial riffles on the River Stiffkey, North Norfolk

  - To increase the availability of spawning habitat for sea trout and resident brown trout
  - To increase habitat diversity and availability
  - To increase channel heterogeneity

7.1 Weir removal from the River Dove at Dovedale

The River Dove at Dovedale was a heterogeneous channel with glide and riffle habitats. Before removal of the weir there were significant differences in depth and flow velocity between riffle and glide habitats. However, despite a significant increase in flow velocity and Froude number and a significant decrease in depth at the most upstream riffle (Site 4) following weir removal, there was an increase in similarity between HMUs following weir removal suggesting the habitat became more similar with only depth being significantly different between HMUs. This is likely to be due to the removal of the impounded glide allowing water to move freely through the reach, increasing flow velocity through barrier removal as flow velocity and Froude number also increased at Sites 2 and 3 following weir removal.

Following weir removal from the River Dove at Dovedale, there was little change in the fish species composition at all sites. Site 3 became more similar in fish species composition to other sites following weir removal. This concurs with results in Chapter 4, which suggest, the hydraulic conditions within this site also became more similar to other sites. The increased similarity of the fish composition at this site to other sites could be the result of decreased water depth creating less favourable conditions for species which prefer deep slow flowing water. That said historical data from a downstream site collected by the Environment Agency (Appendix 3) indicated the species richness within this section of the River Dove was fairly constant with only some between year variations in the percentage contribution of each species to the total community. This indicates that it is more likely that the changes in community composition were a result of natural variation and that weir removal had little impact upon fish community composition.

In terms of measuring the success of the weir removal from the River Dove at Dovedale it appeared the removal of the weir was successful in reinstating hydrological connectivity through the immediate river reach, this should in turn mean that connectivity for fish migration was also reinstated in this immediate river reach as the barrier was removed, although there was no change in the fish species composition to confirm this.

It is difficult to assess whether hydraulic complexity was increased following weir removal and what influence this had on habitat availability and diversity. There was a change in hydraulic conditions in the upstream riffle (Site 4) but sites increased in similarity following weir removal suggesting hydraulic complexity was not increased and in Chapter 6, HABSCORE analysis revealed there was little influence of weir removal on the habitat availability for brown trout.

7.2 Weir removal from the River Dove at Hartington

Prior to weir removal at Hartington on the River Dove, there was little difference between hydraulic conditions in upstream and downstream sides of the weir but one year after weir removal there was less similarity between upstream and downstream sides suggesting hydraulic diversity increased. This was likely to be due to the significant increase in flow velocity and Froude number downstream following weir removal.

All sites were similar in fish species composition regardless of survey date. Historical data from the Environment Agency (Appendix 3) collected at a downstream site indicated a similar composition of species within the community suggesting there was little influence of weir removal on the fish community composition in the River Dove at

Hartington. There was a small increase in the abundance of 0+ brown trout in the reach following weir removal; this may have been due to increased accessibility through the reach resulting from weir removal. It was however more likely to be a result of natural variability in the fish population as there was no increase in the spawning area available within the reach. In terms of the success of the weir removal from the River Dove at Hartington it is possible that the objective to reinstate hydrological connectivity, and thus improve passability for migratory species has been achieved, as the water is able to flow freely through the channel within the immediate reach. There was an increase in variation of depth within the upstream reach following weir removal and ordination analysis revealed upstream and downstream sites were less similar following removal suggesting increased hydraulic diversity. In Chapter 6, HABSCORE analysis revealed habitat availability for brown trout did not show a particular increase following weir removal.

7.3 Channel narrowing at Lowthorpe Beck, East Yorkshire

There was little significant change within the channel at Lowthorpe Beck following narrowing. Although Froude number increased significantly at sites following channel narrowing, there was generally very little impact on instream hydraulics. Therefore it may be concluded that channel narrowing was not successful in achieving the objective of increasing flow velocity to encourage the movement of fine sediment.

There was no objective regarding the desired outcome of channel narrowing on the fish community within the Lowthorpe Beck system however, it is important to monitor the impact of any river rehabilitation project on the fish species present, as they are an important biological indicator of ecological status under WFD requirements. It is difficult to conclude the impact of channel narrowing on the fish community composition in the survey stretch due to the suspected predation reported by the fishing club. Nonetheless, as there was very little change in the fish community composition in Site 3, it is possible that channel narrowing had little impact upon fish community composition.

7.4 Gravel installation at the River Stiffkey

During surveys of the rehabilitated stretch of the River Stiffkey, there were differences in hydraulic variables (depth, flow velocity, Froude number) within sites (HMUs) between years and between sites (HMUs) within years suggesting there was hydraulic diversity within the river reach.

The fish community composition was similar in all sites (HMUs) but the community in glides was generally dominated by >1+ brown trout whereas the fish community in riffles had a greater proportion of bullhead and 0+ brown trout. This would be expected given the respective habitat preferences of the given species (see section 2.3). The fish community composition varied within sites between survey years but this was likely to be due to natural between year variability. This was confirmed by historical data from the Environment Agency (Appendix 3), which indicated that the species present within the reach were reasonably constant between years but the proportion each species contributed to the community composition varied annually.

In terms of determining the success of gravel installations in the River Stiffkey at achieving the desired objectives it is difficult to be conclusive due to the lack of baseline data. From post rehabilitation monitoring it appears that there are gravel areas available within the rehabilitated river reach with areas of significantly higher flow velocity than their respective glides suggesting, these areas have potential to provide spawning habitat for sea trout and resident brown trout although there was no evidence to suggest there has been an increase in brown trout recruitment. It also appears that there was hydraulic diversity within the reach as there were significant differences in hydraulic variables between riffles and glides although without baseline data it was not possible to conclude whether this was the result of gravel installation.

7.5 The influence of hydraulic conditions on fish community composition

Some river stretches increased in the amount of habitat available for a particular life stage of brown trout (River Dove at Dovedale Site 1 >1+ brown trout >20 cm, River Dove at Hartington Site 1 0+ brown trout and >1+ brown trout <20 cm and Site 2 >1+ brown trout >20 cm, Lowthorpe Beck Site 1 0+ brown trout and all stretches of the River Stiffkey apart from Stretch 3 0+ brown trout). Other river stretches decreased in the amount of habitat available for a particular life stage of brown trout (River Dove at Dovedale Site 2 >1+ brown trout >20 cm, River Dove at Hartington >1+ brown trout >20 cm, Lowthorpe Beck all sites >1+ brown trout <20 cm and >20 cm and River Stiffkey all stretches apart from Stretch 1 >1+ <20 cm and >20 cm). There was however no remarkable change in habitat availability following any of the river rehabilitation projects or between survey years.

Investigations into habitat use by species within the communities concurred with previous citations within the literature (see section 2.3) with bullhead captured in shallow areas with higher flow velocity and brown trout preferring greater depths and slower flow velocities as body size increased. Lamprey ammocoetes and transformers were also captured in deep, slow flowing areas. Despite different species having preference for different hydraulic habitat conditions, few hydraulic variables had any significant correlation with the structure of the fish community. Depth, width and flow velocity were related to the densities of 0+ and >1+ brown trout, as would be expected given the habitat preferences of these species. These variables were also related to the total fish density in the given stretch, which is possibly due to the dominance of brown trout within the communities sampled.

These findings are important as it is widely accepted that different fish species and life stages have different requirements (Cowx *et al*., 1993; FAO, 2008). As river rehabilitation increases following acceptance of the WFD and HD river management is changing as focus is increasingly placed upon the role of hydrology and physical processes in supporting ecology (Vaughan *et al*., 2009). Many rehabilitation efforts focus on installing specific static structures to create hydraulic and habitat diversity as perceived good ecological status (Beechie, 2010). However, the increase in river rehabilitation means it is more important than ever to understand the link between fish and physical habitat. The findings of this research indicate the lack of significant linkages between fish community composition and hydraulics suggests that specific hydraulic conditions are not imperative to the fish community and it is more important that a range of habitat is available to support all species and lifestages in the community. Findings also suggest these river rehabilitation procedures had little impact on both the hydraulics and fish communities. It is however, only through monitoring these projects that outcomes are discovered and can be used to create a feedback loop.

7.6 The importance of monitoring

Despite short term monitoring indicating little change, this information is important as it provides details of the outcomes of various river rehabilitation projects, which can be used to inform future river rehabilitation plans through feedback and adaptive management. Adaptive management, although frequently missing from river rehabilitation procedures, is the process of monitoring outcomes of river rehabilitation and using this knowledge to inform on best practice for future schemes. It is defined by Halbert and Lee (1991) as

“ . . . an innovative technique that treats management programs as experiments. Rather than assuming that we understand the system that we are attempting to manage, adaptive management allows management to proceed in the face of uncertainty. Adaptive management uses each step of a management program as an information-gathering exercise whose results are then used to modify or design the next stage in the management program. In adaptive management, there is a direct feedback between science and management such that policy decisions can make use of the best available scientific information in all stages in its development.” (Downs and Kondolf, 2002).

Adaptive management supports a continual process and amalgamation of planning, acting, monitoring and evaluating so the application of all knowledge can lead to a management plan with maximum confidence and minimum risk (Downs and Kondolf, 2002; Hammond, 2011). It is thus important that monitoring of a river rehabilitation project be an integral part of the project process to be able to indicate success although in the past, project monitoring has tended to be a little *ad hoc* and not noticeably linked to project design (Hammond, 2011). The implementation of river rehabilitation projects should be based upon adaptive programmes with stepwise implementations, based upon multi-disciplinary planning and evaluation schemes and underpinned with a strong scientific background, with outcomes of current river rehabilitation schemes used to inform and influence future river rehabilitation (Buijse, 2005). However, repeatedly monitoring programmes are not as complex as the rehabilitation activity itself (Jansson, 2005).

Although conclusions have been drawn from the results, due to the short-term monitoring involved in this study it is difficult to determine whether each river rehabilitation project was successful and to provide any firm conclusions as to the outcomes of the river rehabilitation projects monitored (channel narrowing, small-weir removal and gravel installation). The strength of this study lies in the collection of simultaneous hydraulic and biological data sets which can be used to enhance knowledge of consequences of river rehabilitation as monitoring, where applied often relates to only one quality element. It is generally recognised that continued development of the scientific basis for restoration is essential if current and planned restoration efforts are to be successful (Hobbs, 2007) and project success can only be determined through ongoing monitoring and evaluation of outcomes (Brierley, 2010). That said it is only possible to determine success when clearly defined objectives are set against which to measure outcomes (Downs and Kondolf, 2002; Wohl, 2005; Bernhardt, 2007; England, 2007). Before a project is implemented, success criteria should be stated as specific objectives against which outcomes can be compared

(Downs and Kondolf, 2002). However, often clear descriptions of the aims and objectives of river rehabilitation are missing (Buijse, 2005) and goals are not linked to the success criteria of the objective (Bernhardt, 2007). Shortcomings of adaptive management are often associated with lack of clearly stated aims and project objectives, poor monitoring and recording of outcomes and unsuitable collection, processing and storage of pre-implementation data against which to compare outcomes (Brierley, 2010). Thus, many projects are difficult to monitor due to the lack of clearly stated objectives (Hammond, 2011). It is the case with this study that although some objectives of the work were stated, these objectives were rather general and intangible, lacking defined and quantifiable criteria against which to measure success. Therefore, despite monitoring being carried out and significant differences in hydraulic variables being detected, it was difficult to compare results against objectives to define success.

Hammond et al. (2011) suggest that during the planning stage overall project objectives should be set which will give clear focus as to the intended project deliverables and identify what monitoring needs to be undertaken; they also suggest monitoring objectives should be set which ensure monitoring programmes are designed to answer specific questions related to the project objectives. They propose a SMART approach to objective setting, which allows both the project and concomitant monitoring to be clearly defined in terms of expected outcomes, quantifiable, achievable and realistic in terms of a specified time frame (Hammond, 2011).

• **S**pecific (concrete, detailed, well defined),

• **M**easurable (quantity, comparison),

• **A**chievable (feasible, actionable),

• **R**ealistic (considering resources), and

• **T**ime-Bound (a defined time line)

Intrinsic to all monitoring, knowledge gathering and objective setting is the need for adequate baseline data as it is impossible to measure changes without knowledge of the current status (England, 2007). Baseline data should be used to develop objectives, encompassing the main environmental concerns, specific problems in the target reach and a catchment context (Downs and Kondolf, 2002; Brierley, 2010) as the failure of river rehabilitation projects is frequently linked the following common problems:

- Not addressing the root cause of habitat or water quality degradation;
- Not recognising upstream processes or downstream barriers to connectivity;
- Inappropriate uses of common techniques (one size fits all); an inconsistent (or complete lack of an) approach for sequencing or prioritising projects;
- Poor or improper project design;
- Failure to get adequate support from public and private organizations; and
- Inadequate monitoring to determine project effectiveness (Roni and Beechie, 2013).

Progression in the implementation of management schemes can only transpire through the application of ecohydrological principles within their landscape and evolutionary framework (Brierley, 2010). It is unlikely river rehabilitation measures will achieve their desired outcomes where knowledge of the catchment processes are not known and not accounted for in project design (Wohl, 2005; Brierley, 2010). A river's current physical state must be understood in terms of its adjustment to human induced disturbance and consequent deviation from a natural state, setting this in context to catchment scale processes allows assessment of off-site influences with the potential to cause change or limit recovery potential (Brierley, 2010) meaning an inter-disciplinary approach to river rehabilitation is useful (Roni and Beechie, 2013). It is now becoming widely accepted that river rehabilitation is more likely to be successful in its proposed objectives where there is consideration for catchment wide processes (Buijse, 2005; Wohl, 2005; Beechie, 2008) as understanding of underlying catchment processes and pressures, and comprehensive baseline data increases potential to set realistic targets of what river rehabilitation is achievable (Brierley, 2010). An important aspect of the WFD is the formulation of River Basin Management Plans (RBMPs), which promote the management of rivers based upon the spatial catchment area of the river as a natural geographical and hydrological unit as opposed to according to administrative boundaries.

The projects monitored in this study showed few changes in hydraulic variables and even fewer changes in the composition of the fish community. The findings from the monitoring of the weir removal from the River Dove at Hartington contrasted with findings from a previous study by 0rr *et al*. (2006) where a significant decrease in depth and a significant increase in flow velocity was detected in the impoundment following weir removal. It may be that this river rehabilitation project did not achieved the desired outcomes, as it was not implemented at a location that is suitable to achieve these outcomes. This may also be the case for the channel-narrowing project at Lowthorpe Beck, which failed to meet the objective to increase flow velocity. Whilst it appears the River Dove weir removal projects were successful in achieving the objective of increasing hydrological, thus migratory connectivity within the reach, this is the case only within the immediate reach and consideration should be given to whether the removal of two small weirs is enough to make a significant difference in a system where weirs are so prolific.

It is also important that monitoring continues over an extended period of time, as conclusions drawn here are from only one year post-implementation. Whilst it is under most circumstances unrealistic to continue monitoring over long periods of time, such as 20 years, any long-term study will contribute to knowledge (England, 2007) and thus adaptive management. Generally, it is assumed post-project monitoring should be undertaken over a period of three years as this is the period over which costings have been calculated but individual project objectives and river characteristics may influence the period over which monitoring should take place (Hammond, 2011). England (2007) suggests ecological restoration is dependent upon physical processes which may be slower to respond therefore monitoring should occur over a time frame, which encompasses a full range of the natural behaviour of the river, and the time over which geomorphological adjustments occur (Brierley, 2010; Hammond, 2011). With respect to monitoring the response of fish within the system, Kondolf and Micheli (1995) recommend monitoring the population of the longest living species within the system over two generations, which for trout would mean a minimum of 6 years. Haslam (1996) however, suggests monitoring should take place over 2, 5 and 10 years.

It should be noted that not only is it important for monitoring to take place, the effective dissemination of results is fundamental to future project design (England, 2007) and it appears this is heading in the right direction with tools such as the RESTORE and REFORM Wiki pages providing easily accessible information for anyone involved in river restoration.

7.7 Conclusion and Recommendations

The study revealed few significant changes in hydraulic variables and fish community composition following river rehabilitation at all sites. Projects were monitored over only two consecutive years given the limited time-scale of the project therefore this conclusion is the result of short term monitoring and tells of the short term outcomes of the projects studied. It would be beneficial to **continue monitoring of the outcomes of all river rehabilitation projects included in this study on both hydraulics and fish community composition**. This would allow a greater understanding of the positive and negative impacts of such schemes and would increase the capacity to draw conclusions as regards the impact of the projects in the long-term. **It is recommended monitoring should continue over a minimum of 3 years**. That said it is difficult to determine a projects success without distinct objectives and criteria against which to compare the outcomes. **Specific, tangible and measurable objectives should be built into project design before implementation of a river rehabilitation project. These objectives must be set based upon a thorough understanding of the physical processes and pressures influencing the entire catchment** this would ensure schemes are implemented in locations with maximum potential to benefit the hydromorphological quality of the system and the fish community. **Specific, quantifiable criteria against which success can be measured should also be set.**

This study concentrated on assessing the effects of several river rehabilitation projects on instream hydraulics and fish community composition before and after project implementation and between years at the River Stiffkey. **It would be beneficial wherever possible to monitor a control river of similar environmental and ecological conditions concurrently**. This would enable comparison of results and would increase confidence in determining if results are a consequence of river rehabilitation or inter-annual variability by distinguishing between background changes and those attributable to the rehabilitation project.

This study revealed there were few significant relationships between specific hydraulic variables and fish community dynamics suggesting rather than attempting to achieve specific hydraulic conditions **river rehabilitation projects should encourage diversity of in-stream physical conditions**. However, this study focused specifically on the hydraulic variables depth, flow velocity and Froude number. As instream hydraulics are intrinsically linked to the flow of water and sediment through the river channel it is also important to note the effects of river rehabilitation on the physical structure of the channel. Data in this study were collected over a meso scale; **it would be useful to measure instream hydraulics at fixed points and record co-ordinates to determine point-scale changes over time. It would also be useful to measure substrate composition using grain size count at fixed points** to determine the effect of river rehabilitation on physical conditions. It must also be noted that despite the apparent lack of linkages between specific hydraulic variables and fish community composition, **fish are influenced by many other factors including water quality, food availability, predation and availability of cover, and although these factors were beyond the scope of this study, it would be useful to incorporate monitoring of these factors into future monitoring programmes**. This study also gave insight only into summer habitat use. **It would be useful to investigate habitat**

use at different times within the year and it is suggested fish tracking studies may be useful to meet this end.

Perhaps most importantly, this study highlighted that the outcomes of river rehabilitation projects may not necessarily be what were expected and can differ from outcomes reported in other studies, as well as differing between stretches of the same river. **It is therefore essential that the results of monitoring of all river rehabilitation projects be disseminated as widely as possible**. The availability of results regarding the physical and biological impacts of river rehabilitation schemes will provide river managers, the academic community and recreational river users with information on the successes and failures of river rehabilitation schemes, which will enhance understanding and facilitate future river rehabilitation plans. Without accessibility of results it is impossible to learn through experience and to begin to draw conclusions regarding precise impacts of specific river rehabilitation activities.

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APPENDIX 1. HABFORM

HABSCORE for Windows v1.1 : HAB*form*   **Site habitat record**
NB - this form is double sided

Site identification

Site code		Catchment	
Site name		NGR	
River name		Survey date	

Riparian shading of the site

What percentage of the water surface of the site is overhung by riparian vegetation? Estimate this percentage, for the three vegetation classes indicated, to the nearest 5%.

Deciduous trees & shrubs		Coniferous trees		Herbaceous vegetation	

Migratory access

What is the accessibility of the site ?

	Salmon	Sea trout
Always accessible		
Sometimes accessible		
Never accessible		

Substrate embededness

What is the degree of substrate embededness throughout the site? Tick one box.

High		Medium		Low	

Flow conditions

Briefly describe the prevailing flow conditions (as observed at the time of the HABSCORE survey) in the space provided below.

....

Upstream land-use considerations

What is the principal land-use immediately upstream of the site? Tick appropriate box(es).

Moor / heathland		Coniferous woodland		Deciduous woodland	
Rough pasture		Urban development		Other	
Improved pasture		Industrial land		....	
Arable land		Tips / waste		....	

Potential impacts

Are there likely to be any impacts at the site from the following sources? Tick appropriate box(es).

pH effects		Stocking		Other	
Migration barriers		Habitat modification		....	
River engineering		Low flows		....	
Pollution		Flow regulation			

Width and depth profile at bottom stop net

Record widths to the nearest 0.1m and depths to the nearest 1.0cm.

Channel width	
Depth at ¼ channel width	
Depth at ½ channel width	
Depth at ¾ channel width	

Section dimensions

Record section lengths and widths to the nearest 0.1m and depths to the nearest 1.0cm.

Section length										
Section width										
Depth at ¼ channel width										
Depth at ½ channel width										
Depth at ¾ channel width										

Substrate

Absent	Scarce	Common	Frequent	Dominant
0%	>0% & <5%	≥5% & <20%	≥20% & <50%	≥50%
A	***S***	***C***	***F***	***D***

What percentage of the stream bed area in each section is composed of the following substrate types? Enter *A*, *S*, *C*, *F* or *D* as appropriate (see above table).

Substrate category										
Bedrock / artificial										
Boulders ***>25.6 cm***										
Cobbles ***6.4-25.6 cm***										
Gravel / coarse sand ***0.2-6.4 cm***										
Fine sand / silt ***<0.2 cm***										
Compacted clay										

Flow

What percentage of the water surface area in each section is composed of the following flow types? Enter *A*, *S*, *C*, *F* or *D* as appropriate.

Flow category										
Cascade / torrential										
Turbulent / broken ***deep***										
Turbulent / broken ***shallow***										
Glide / run ***deep***										
Glide / run ***shallow***										
Slack ***deep***										
Slack ***shallow***										

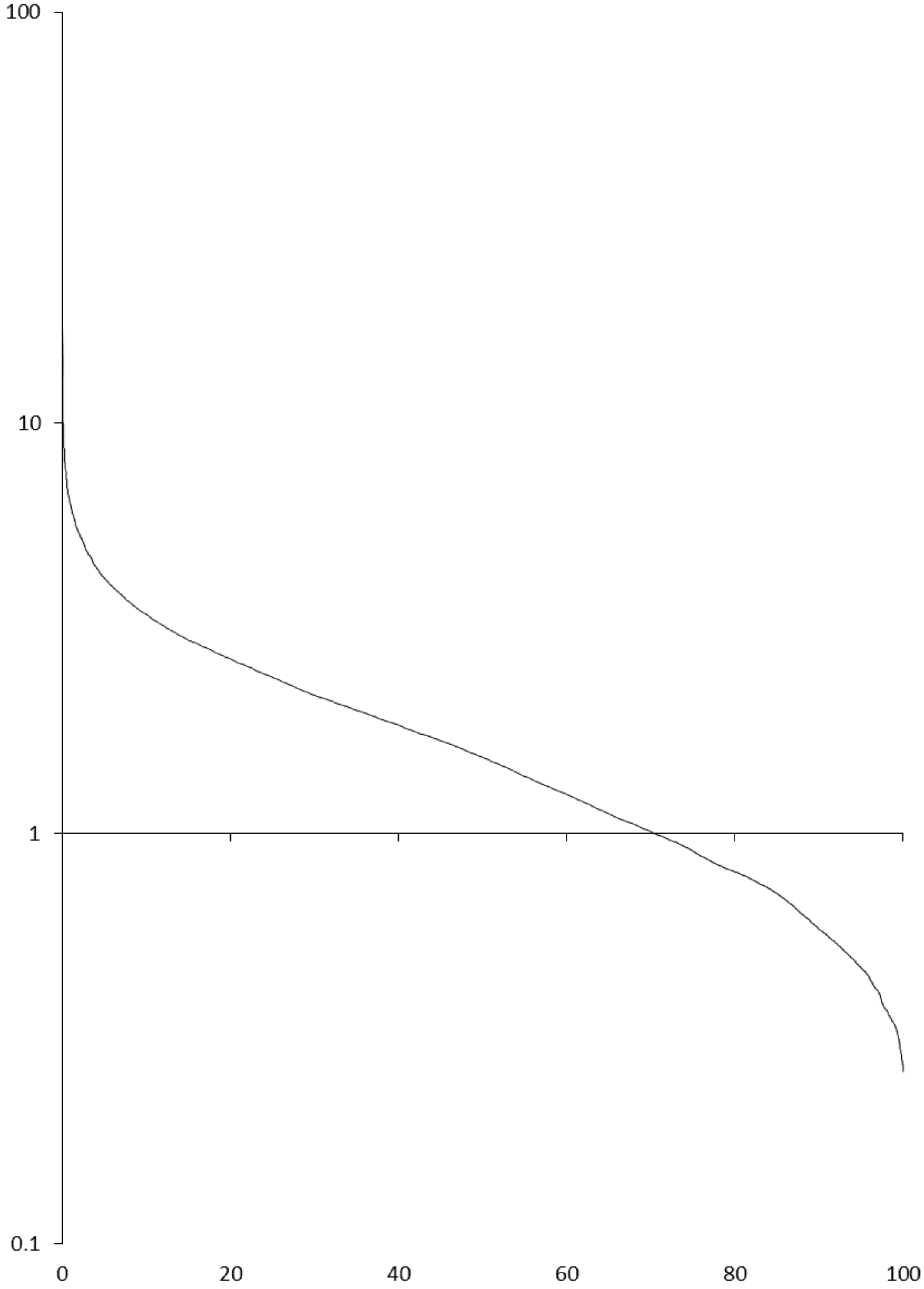
Sources of cover for >10cm trout

What percentage of the stream bed area in each section could provide cover (for a >10cm trout) in the form of *submerged overhang*, or *overhang within 0.5m of the water surface*? Indicate the abundance of cover within the various categories which are listed below. For 'submerged vegetation' include all macrophytes, mosses and algae which are providing cover. Estimate as 0, 1, 2, 3, 4, 5, 10, 15, 20, 25, ... 100%.

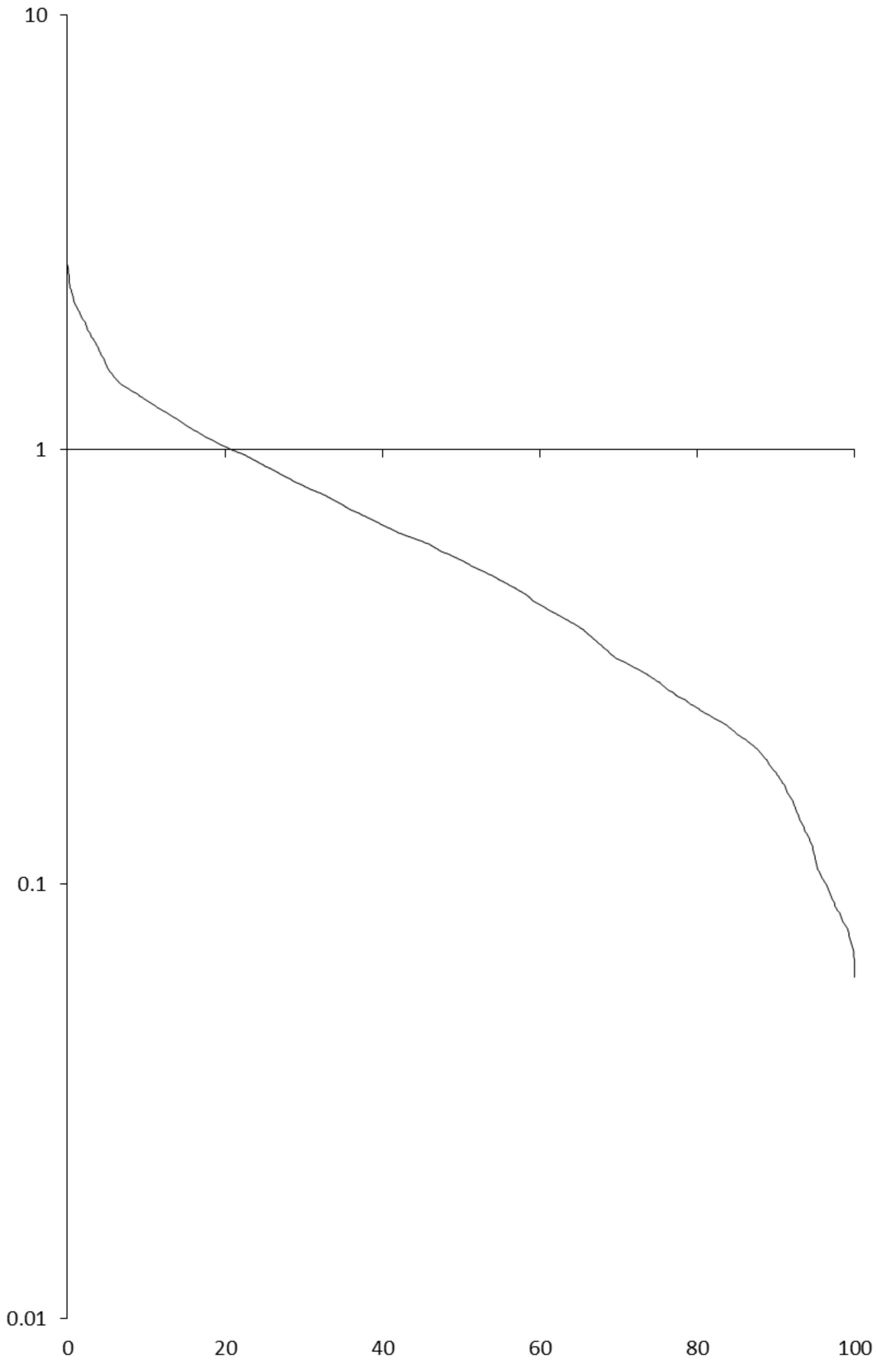
Source of cover										
Submerged vegetation										
Boulders, cobbles, etc.										
Tree root systems										
Branches and logs										
Undercut banks										
Other submerged cover										
Overhang ***within 0.5m***										
Area of deep water										

APPENDIX 2. FLOW DURATION CURVES (1990-2011) (DATA SOURCE: ENVIRONMENT AGENCY)

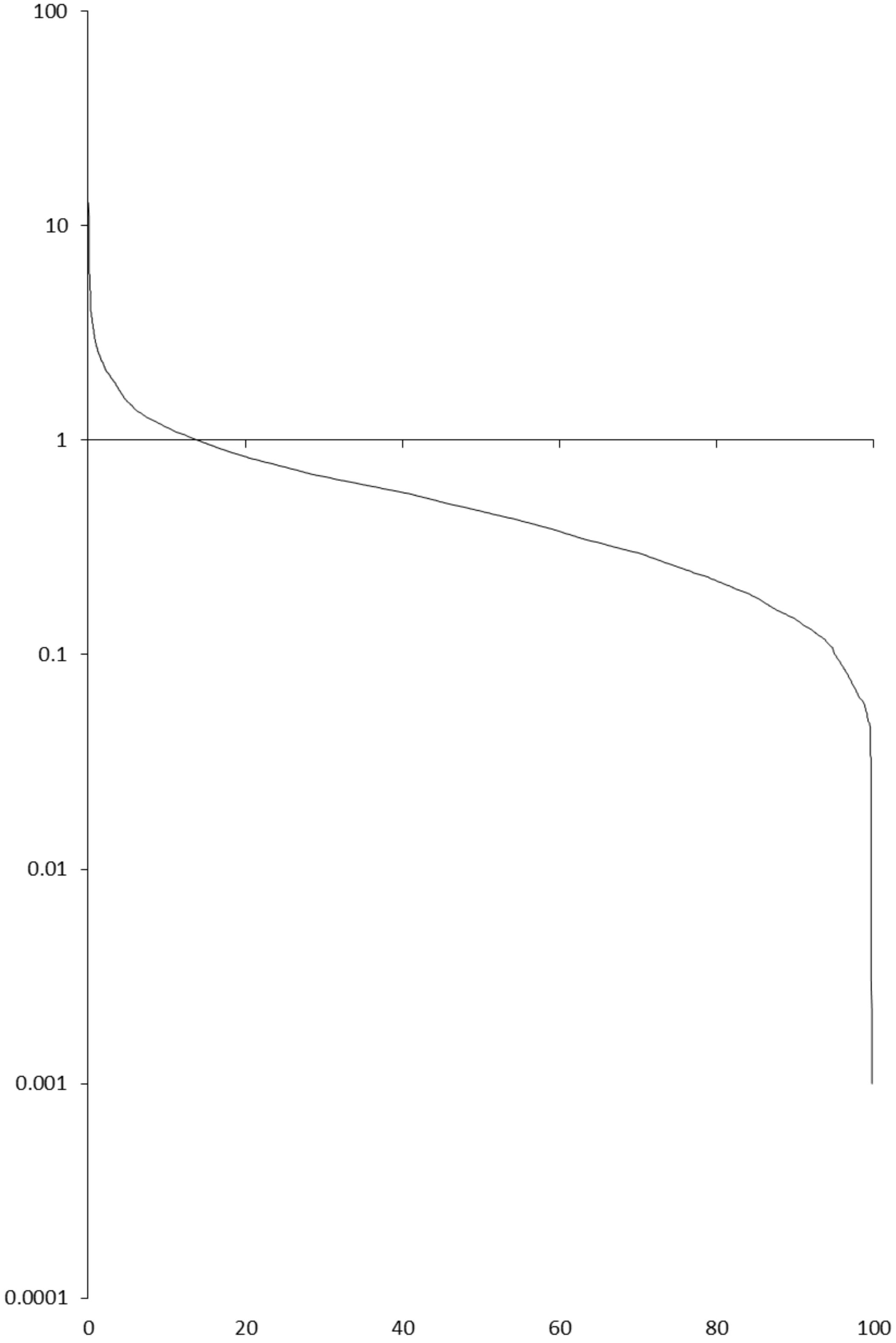
The River Dove



Lowthorpe Beck



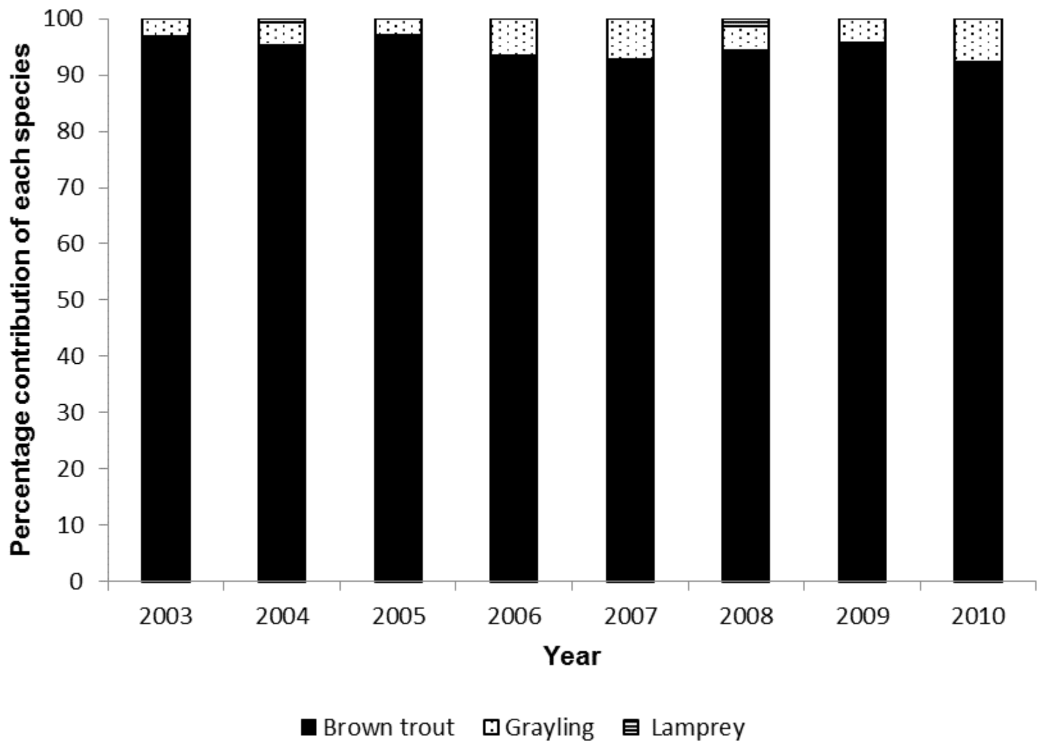
The River Stiffkey



APPENDIX 3. HISTORICAL FISH CATCH DATA FROM ENVIRONMENT AGENCY RECORDS

River Dove at Reynards Cave downstream of Dovedale (SK1450052500-SK1440052600)

	2003	2004	2005	2006	2007	2008	2009	2010
Brown / sea trout [*Salmo trutta*]	122	144	133	155	129	135	132	132
Grayling [*Thymallus thymallus*]	4	6	4	11	10	6	6	11
Bullhead [*Cottus gobio*]								
Brook lamprey [*Lampetra planeri*]		1				2		
Minnow [*Phoxinus phoxinus*]								



River Dove at Hartington downstream road bridge at Hartington (SK1221059630-SK1213059760)

Species	2003
Brown / sea trout [*Salmo trutta*]	16
Grayling [*Thymallus thymallus*]	1
Bullhead [*Cottus gobio*]	121
Brook lamprey [*Lampetra planeri*]	72

Lowthorpe Beck at Bracey Bridge (TA0773562100)

Species	2004	2010
3-Spine Stickleback	3	
Brown trout	27	6
Bullhead		1-9

River Stiffkey at Warham (TF9470041400)

Year	3-spined stickleback	Lamprey	Brown / sea trout	Bullhead	European eel	Flounder	Gudgeon	Rudd	Stone loach
1988	1	2	18	0	387	0	0	0	28
1992	50	12	0	21	381	0	59	0	76
1995	0	0	20	0	38	5	4	0	7
2000	1	0	6	51	208	1	2	0	15
2007	5	0	38	21	67	2	1	0	55
2008	5	0	42	89	22	0	0	0	43
2009	32	19	72	106	9	3	0	5	28
2011	4	3	82	77	65	0	0	0	10