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Identifying limitations of monitoring the success of river rehabilitation schemes for freshwater fish

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Natalie Vivee Angelopoulos

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M.Sc. (Hull 2008)

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ABSTRACT

IDENTIFYING LIMITATIONS OF MONITORING THE SUCCESS OF RIVER REHABILITATION SCHEMES FOR FRESHWATER FISH

Hydromorphological degradation impacts on habitat availability for biota in rivers. Degradation occurs as a consequence of multiple pressures driven by anthropogenic actions such as agriculture, urbanisation, industry, water supply, flood protection, navigation and transportation, fisheries and recreation. Rehabilitation of rivers is a tool that underpins the Water Framework Directive (WFD) by means of addressing river degradation by re-introducing habitat heterogeneity to impacted river systems. Many past and recent papers have highlighted a lack of information on the success of rehabilitation projects leading to a paucity of information and validation of the efficacy of such schemes. A review of 49 UK rehabilitation case studies found only 59% of these projects were monitored and worryingly only 24% of the 49 projects recorded any degree of success in the outcomes of the project. Clearly, there are limitations in monitoring the success of river rehabilitation schemes and it is this 'key' issue that is addressed in this thesis, to move from decisions based largely on subjective judgements to those supported by scientific evidence.

A literature review was carried out to identify best practise knowledge of the key steps for designing effective project planning for river rehabilitation and identifying existing limitations, focusing specifically on monitoring design and impact assessment. In addition to the literature review, a number of small scale rehabilitation case studies were monitored and evaluated to give a practical insight into the limitations and constraints that arise, when measuring projects success. Fish were selected as the chosen biological indicator and were monitored in addition to a number of habitat variables to identify rehabilitation success. The following array of case studies were evaluated, listed in order of complexity:

- Instream remediation of Driffield Beck, a small urban stream
- Channel narrowing of Lowthorpe Beck, a Yorkshire chalk stream
- Brash revetment to prevent bank erosion on the River Manifold
- Small weir removals on the River Dove
- Artificial riffle reinstatement on the River Stiffkey

As climate change pressures become more frequent on river systems, it also becomes an important driver for river rehabilitation mitigation and adaptation strategies. The EU Floods Directive and UK Flood and Water Management Act need to be integrated with the EU WFD and Habitats Directive, to work towards flood risk management (FRM) whilst still considering river heath. This approach is still in its early stages and there are no examples in the literature that report successful FRM that has incorporated river rehabilitation. Two case studies from an urban river setting on the River Don in Sheffield provide a practical insight into the limitations of monitoring and evaluating case studies where FRM is combined with river rehabilitation on the local habitat and fisheries.

Overall, monitoring was limited spatially and temporally for all case studies, this can be overcome by planning a suitable monitoring design before any rehabilitation takes place so long-term (pre and post), spatial monitoring can overcome natural variability. Furthermore, fish were a poor indicator for rehabilitation success over such a short monitoring timeframe; it is therefore advised that multiple biological quality elements are monitored to strengthen the evaluation of project success. Stocking of fish by the Environment Agency became a large hindrance when evaluating fisheries data for some of the case studies, because it masked changes that could have occurred as a result of rehabilitation. This is where the importance of communication between different stakeholders is vital, to reduce conflicting actions. Collaboration between FRM and conservation specialists is also necessary to achieve a win-win scenario and to endeavour to integrate these two conflicting objectives. There is much uncertainty when identifying rehabilitation success; current concepts in literature consider the application of endpoints and benchmarks against which to measure performance however, there are no definite criteria to date. Limitations, in monitoring and evaluating need to be overcome to establish appropriate targets for benchmarking and endpoints to reach project success. Furthermore, river restoration programme goals often only address problems on single rivers at a small scale and therefore have limited impact on catchment-scale processes. Potential benefits of implementing river rehabilitation and conservation at a catchment-scale are subsequently addressed.

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CHAPTER 1. INTRODUCTION

1.1. Degradation of rivers

Rivers are among the most impacted ecosystems worldwide (Malmquist & Rundle 2002). Growing human demands for food security, navigation and renewable, 'green' energy have resulted in rivers becoming heavily utilised as a result of direct and indirect human demand (Purseglove 1998). The most intensive period of river modification occurred between 1930 and 1980 as industry developed globally (Brookes *et al*. 1983) and is still ongoing to date. Activities such as the construction of dams and weirs, the straightening and deepening of river channels, the conversion of floodplains to agricultural land, flood-defence engineering, water abstraction, water transfer and pollution have caused heavy modification of rivers (Dynesius & Nilson 1994; Aarts *et al.* 2004; Vaughn *et al*. 2009). This degradation of aquatic systems has lead to a world wide effort to rehabilitate aquatic habitat for economic, social and environmental reasons (NRC 1992).

Hydromorphological degradation occurs as a consequence of multiple pressures driven by anthropogenic actions such as agriculture, urbanisation, industry, water supply, flood protection, navigation and transportation, fisheries and recreation (FORECASTER 2010). These drivers lead to pressures that alter the morphology of the river channel through, for example impoundments and channelization, alteration of riparian vegetation and instream habitats, increased sedimentation, embankments and loss of connectivity. These pressures may also influence water abstraction and flow regulation that will interfere with the hydrological regime by altering river discharge, interbasin flow transfer and may possibly lead to hydro-peaking in some instances. Direct and indirect anthropogenic actions can result in water quality issues that can lead to the degradation of a river through point and diffuse source pollution resulting in the introduction of toxic substance, eutrophication, organic and inorganic inputs. All degradation, whether it is resulting from water quality or morphological alteration, will impact on habitat availability for biota. Rehabilitation of rivers is a tool that has become widespread across the globe to address river degradation by re-introducing habitat heterogeneity to the river system to encourage higher ecological diversity. The terms 'river rehabilitation' and 'river restoration' have been debated over recent years and in many cases are used synonymously in broad terms to define any physical modification made to a river channel through geomorphological alteration. On the whole, the most frequent definition of 'river restoration' is to return the river to its pre-disturbed state, whereas 'river rehabilitation' is to improve river habitat to produce a positive outcome to benefit river health (Roni & Beechie 2013). Within the content of this thesis these are the preferred terms and used in this context.

Spatial and temporal heterogeneity across a range of scales are fundamental characteristics of aquatic systems (Frissel *et al*. 1986; Palmer & Poff 1997; Ward *et al.* 2001). Modification of rivers has resulted in the loss of habitat availability, creating unfavourable conditions for aquatic organisms and consequently reducing the heterogeneity of the ecosystem (Postel *et al.* 1996; Sala *et al.* 2000; Tilman *et al.* 2001). As a result, large numbers of fish species are threatened and fish productivity in most rivers has declined (Welcomme & Halls 2004; Huckstorf *et al.* 2008).

The pressure on freshwater systems is predicted to increase due to continuous human demands and expected climate change (Alcamo *et al*. 2007). It is therefore important to reduce these pressures by balancing human needs with the needs of rivers themselves, whilst incorporating the uncertainties faced by climate change (Petts 2008). Climate change becomes an important driver for river rehabilitation mitigation and adaptation strategies (Battarbee *et al.* 2008) by increasing ecological resilience through recovery of lost habitat form and function (Mainstone & Holmes 2010). Many negative pressures on rivers can also be mitigated through careful remediation planning and habitat management. However, numerous river rehabilitation projects tend to take an opportunistic approach through trial and error and it is bad practise to continue in this manner (Buijse *et al*. 2005). One key issue that needs to be addressed is moving from decisions based largely on subjective judgements to those supported by scientific evidence (Boon & Raven 2012). Current scientific understanding of river rehabilitation is generally poor (Vaughan *et al*. 2009) and because of this insufficient understanding of ecological processes there have been problems implementing the WFD (Boon & Raven 2012). The absence of knowledge is due to a lack of understanding of the design and implementation stage of project planning for rehabilitation schemes even though there are several, easily accessible river rehabilitation manuals (e.g. UK Ward *et al.* 1994 – The New Rivers and Wildlife Handbook; UK PRAGMO RRC (Hammond *et al*. 2011); Europe – Cowx & Welcome (1998) that provide detailed guidance in this area, few projects use this structured method. The 'benefit' or 'success' of rehabilitation projects are poorly documented and one of the main reasons for this can be narrowed down to the uncertainty of how to determine success at a local and catchment scale. Many past and recent papers have highlighted a lack of information on the success of rehabilitation projects leading to this paucity in data (Tarzwell 1937; Reeves *et al*. 1991; Roni *et al*. 2002; Bernhardt *et al.* 2005; FAO 2008; Roni *et al*. 2008). Setting benchmarks and end points that are linked to clearly defined project goals is the latest approach to help determine the measure of success. However, from the vast quantity of literature available on river rehabilitation, only a small number of papers address the issue of measuring success and an even smaller number address benchmarking and endpoints. This is almost certainly a consequence of poor project monitoring, evaluation and dissemination that should play a vital role in rehabilitation programmes to determine the effectiveness of rehabilitation actions to support the WFD (Roni 2005; Wolter 2010). A comprehensive review in the USA revealed only 10% of over 37,000 projects had been monitored up to 2005 (Bernhardt *et al*. 2005) and although awareness of the value of monitoring is growing, putting it in to practise is still challenging. Similarly, a review of 72 UK rehabilitation case studies through scientific publications and grey literature revealed 68% were monitored, but only 24% of these projects recorded rehabilitation success (Angelopoulos 2013 – unpublished data). Other findings from the UK review revealed 1% of projects failed, 50% had no information and 25% of findings were unclear. The majority of case studies classified as successful very rarely report how they have come to this conclusion, overall it is clear that there are difficulties when quantifying rehabilitation success. In general, this is a difficulty throughout all rehabilitation practises as there are no definite criteria to define endpoints and benchmarks against which to measure performance and with no exact criteria, establishing appropriate targets for rehabilitation activities appears challenging.

Freshwater river ecosystems are intrinsically linked and have a natural habitat continuum between river and landscape (May 2006). As a consequence, it is difficult to conserve a small reach of river by simply using rehabilitation practice at a local level. Therefore, another key issue to be addressed is the question of 'scale' and its significance in the way rivers function. The importance of scale in river conservation and management has grown over the past 20 years, advancing from Wards (1989) conference paper on the 'four dimensional nature of lotic ecosystems' (Boon & Raven 2012), right up to more recent advances in integrated catchment management (ICM) to support WFD. River rehabilitation programme goals often only address problems on single rivers at a small scale and therefore have limited impact on catchment-scale processes (Buijse *et al*. 2005; Eden & Tunstall 2006). While ICM has started to be applied within the UK, single, small scale rehabilitation exercises are still employed most frequently with no association to catchment plans at a larger scale. Consequently, there is still a requirement to understanding pressures at a catchment scale to advance from small scale river rehabilitation to large scale, ICM.

It has long been an ambition of ecologists to understand the links between river physical structure and ecosystem needs to advance from the limited information already assembled. It is important to understand how each system responds to natural and human-induced changes (Naiman *et al*. 1995; Bernhardt *et al*. 2005), including the fundamental relationships between fishes and their habitats (Fladung *et al.* 2003). River rehabilitation is a complex process and multi-disciplinary knowledge is needed to address primary causes of ecosystem degradation to enable a more holistic approach towards river rehabilitation (Kondolf *et al*. 2006; Palmer & Allan 2006; Roni *et al*. 2008; Beechie *et al*. 2010). It is imperative that river rehabilitation is not recognised as a simple reversal for hydromorphological degradation and must be used with caution. Underlying problems must be resolved (e.g. water quality issues) before any structural rehabilitation projects are implemented (Cowx & Welcomme 1998; FAO 2008). For this reason there is a need for further understanding of:

- key drivers of rehabilitation for rivers;
- hydro-geomorphological processes and river functioning synonymously;
- riverine habitat and ecosystem dynamics for aquatic biota synonymously.

1.1.1. *Key drivers of rehabilitation for rivers*

Over the last two decades there have been significant changes in environmental legislation and regulation across the world (Boon & Raven 2012). Nature conservation and in particular river rehabilitation are increasingly considered as part of a much wider framework of environmental policy and practise. There are a number of European directives in place to support the ecological health of rivers such as The Habitats Directive (HD (92/43/EEC)) and the Water Framework Directive (WFD (2000/60/EC)). The HD works towards maintaining biodiversity to a favourable conservation status to protect biodiversity through application of designated Special Areas of Conservation (SACs) and Special Protection Areas (SPAs). While the WFD endeavours to improve ecological functioning through rehabilitation and uses ecosystem health as the basis for decisions, which requires the characterisation of all water bodies according to five quality (from 1 – high status to 5 – bad status) classes. There are four Biological Quality Elements (BQE) involved in the monitoring of river health, fish, macroinvertebrates, macrophytes and phytoplankton (Schmutz *et al*. 2007). River Basin Management Plans (RBMPs) are a requirement of the WFD to reach good ecological status (GES) through the Programme of Measures (PoM) and signifies that human activities have only had a slight impact on the ecological characteristics of aquatic biota. In some cases, where a considerable amount of modification has occurred, the river channel is classified as heavily modified water bodies (HMWB) and means that a surface water body cannot reach GES and therefore has to aim for 'good ecological potential' (GEP), other water bodies such as canals are further classified as artificial also aim only for GEP. The RBMPs are to be updated every six years, and the second round of RBMPs are to be released in 2015. The WFD therefore aims to prevent deterioration in ecological status of rivers and has the potential to increase the number of rehabilitation schemes undertaken across Europe and to ensure it is maintained once achieved (Logan & Furze 2002; England *et al.* 2007). It is the Environment Agency, Welsh Rivers Authority and the Scottish Environment Protection Agency that have the responsibility for delivering WFD in UK. In addition, non-governmental organisations (NGOs) in the UK have been active in river conservation and management since before the 1990's, but their influence and involvement became more noticeable during this time. Notable amongst these are the Association of Rivers Trusts (ART) in England and Wales, the Rivers and Fisheries Trusts in Scotland (Boon & Raven 2012) and they are good examples of how organisational structure has evolved to specifically support river rehabilitation practise. The River Restoration Centre (RRC) and the European River Restoration Centre (ERRC) are further organisations that bridge the gap between scientists, practitioner and stakeholders by transferring existing knowledge and generating new information that can help guide river rehabilitation.

In addition there are several directives in place that oppose the WFD but are necessary to support river management from a socio-economic perspective. For example, the EU Floods Directive (FD) and the UK Flood and Water Management Act (FWMA). The Department for Environmental, Food and Rural Affairs (Defra) consultation document 'Making Space for Water' (Defra 2004) emphasises the need for a more holistic approach to flood risk management (FRM) that delivers the greatest environmental, social and economic benefits (England *et al*. 2007). Both the FD and FWMA recognise the need to include certain elements of the WFD to support this holistic approach and include river rehabilitation within sustainable flood risk management and water resource management (Mainstone & Holmes 2010). However, integrating river rehabilitation into FRM is still in its early stages and much more research is needed to identify best practise.

1.2. Conservation of inland waters

There has been an increase in demand for river rehabilitation to support nature conservation (Waal *et al*. 1998). Rehabilitation can be used to improve habitat diversity and connectivity of a river. Rehabilitation towards pristine conditions is idealistic but usually impractical (Cowx & Welcomme 1998) because irreversible changes are already present in catchment boundary conditions (Findlay & Taylor 2006). Consequently, the return of rivers to 'pristine' condition has instigated countless scientific debates to what rehabilitation targets should achieve (Haslam 1996; Schouten 1996; Dobson & Cariss 1999; Pretty *et al*. 2003) and it is argued that programmes should focus on creation of 'natural' features to improve the health of the ecosystem (Rhoads *et al*. 1999). It is, however, more resourceful to preserve inland waters that have not yet been degraded to maintain high ecological status (FAO 2008). The outcome of any future rehabilitation schemes must therefore first aim to identify the pressure(s) on the system and then work towards relieving the river of this pressure to what is seen as an achievable goal, such as GES or GEP. Once an achievable goal has been reached through river rehabilitation, it must be conserved to maintain river health.

River management has progressed from the belief that engineering was the key component of river rehabilitation, towards a more multidisciplinary approach that considers the importance of hydrological, physical, biological and physio-chemical factors (Figure 1.1) (Hooke 1999; Findlay & Taylor 2006; Mainstone & Holmes 2010)). When looking in detail at the scope of river rehabilitation it is essential that a hierarchical view is taken when identifying these four important ecological components for habitat integrity because their understanding can help alleviate impacts (Sear 1994; Beechie & Bolton 1999; Mainstone & Clarke 2008; Beechie *et al*. 2010). River rehabilitation, if applied well could be key to conserving inland waters by correcting underling processes that have lead to degradation, sequentially influencing habitat and biotic production (Roni 2005b) and enabling the river to become self-sustaining by imitating its natural process (Roni & Quinn 2001).

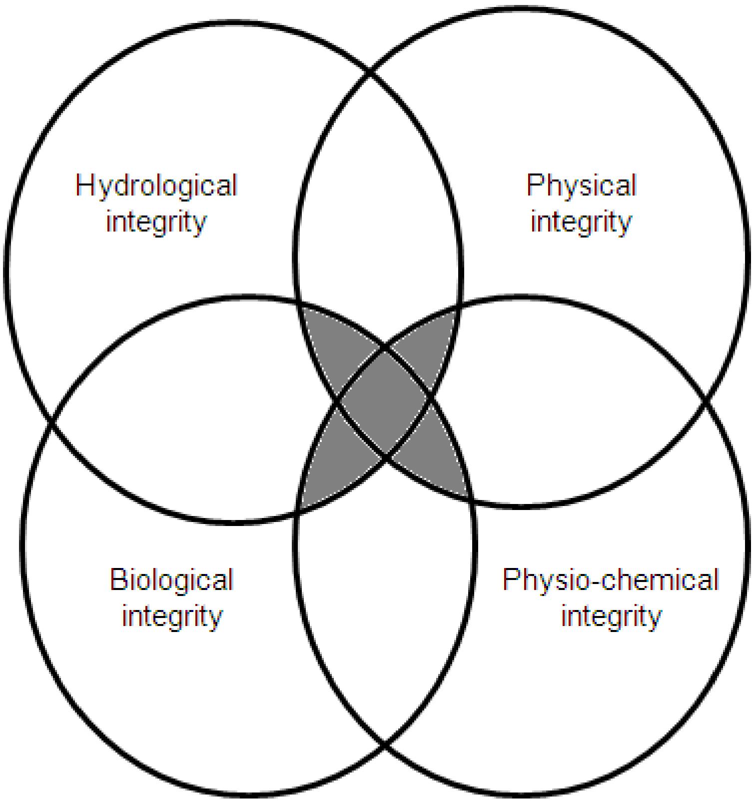


Figure 1.1 The four main ecological components that constitute river form and function (adapted from Mainstone & Holmes 2010)

Common techniques for enhancing and creating habitat range from large scale, physical modification, for example channel narrowing, re-meandering and re-profiling to create features such as pool-riffle and backwaters (Cowx & Welcomme 1998; RRC 1999; Pretty *et al*. 2003) to more small scale, instream habitat methods involving the placement of a variety of artificial and natural structures to recreate habitat diversity within a channel such as logs, wood boulder and gravel (Roni *et al*. 2008). Waal *et al*. (1998) believed rehabilitation schemes should be completed at a larger scale, but catchment rehabilitation is seldom undertaken mainly due to its expense, leaving small scale rehabilitation more affordable and practical, even if a little cosmetic at times (Jasperse 1998). Nevertheless, since freshwater river ecosystems are intrinsically linked and have a natural habitat continuum between river and landscape (May 2006) it is difficult to conserve a small reach of river by simply using rehabilitation practice at a local level. Fortunately, potential benefits of implementing river rehabilitation and conservation at a catchment-scale are being increasingly recognised as an essential component of future rehabilitation practise (Hodder *et al*. 2010), especially through the WFD aims to combine catchment scale understanding across a range of aquatic ecosystems to improve ecological status through RBMP.

1.2.1. Geomorphology of rivers & Ecosystem dynamics

The WFD recognises how ‘hydromorphological quality’ is the fundamental foundation towards rivers reaching GES and for this reason the EU States have drafted RBMPs with rehabilitation measures focusing on restoring river hydrology and morphology as an element that underpins ecological status on a whole basin approach (Jansson *et al*. 2005; Kondolf *et al*. 2006; Mika *et al*. 2008; Wohl *et al*. 2005; FAO 2008; Brierly *et al*. 2010). To reach WFD targets substantial investment in these measures is now required. The increasing ambition of ecologists and geomorphologists that work with rivers has helped enhance our knowledge to define and understand the processes that influence the pattern and character of river systems (Rosgen 1994). Flow variability and dynamics influence physical processes within a river across a wide range of spatial and temporal scales (Vannote *et al*. 1980; Junk *et al*. 1989; Poff & Ward 1990; Poff *et al*. 1997; Sparks 1995), creating complexity through a combination of depositional and erosional processes and in turn, influence the composition and structure of the aquatic community (Schumm 1977; Naiman *et al*. 1992; Poff & Allen 1995; Poff *et al*. 1997; Richter *et al*. 1997; Cowx & Welcomme 1998; Cowx *et al*. 2004; Pont *et al*. 2009). The interaction between flow regimes, the local geology and landform determine physical habitat for biota, specifically fish, such as substrates and their stability, distribution of flow types for example pool – riffle systems, vegetation and undercut banks (Frissel *et al*. 1986; Cobb *et al*. 1992; Newbury & Gaboury 1993; Beechie *et al*. 2010). The presence and quality of physical habitat can influence the presence and abundance of biota within a river section (Gorman & Karr 1978; Milner *et al*. 1985). Disrupting physical processes as a result of human pressures can lead to long term loss of habitat diversity within the river channel and can therefore have detrimental effects on biota (Beechie & Bolton 1999; Diana *et al*. 2006). Consequently, there are two key actions when identifying potential ecological benefits of river rehabilitation, to identify the change in habitat and to subsequently investigate how the habitat change has affected biota.

Fish utilise an extensive selection of habitats within a river system; many species show signs of distinct preferences (Pretty *et al.* 2003) for their daily and seasonal requirements for each of their life stage (Cowx *et al*. 2004). Understanding the habitat requirements of fish species through knowledge of their life history traits is fundamentally important; spawning, feeding, nursery and refuge habitats are the main functional units required for specific life stages as part of the life cycle of the species (Figure 1.2) (Bain & Stevenson 1999). Not only is the availability of each of these functional units important, but the connectivity between them is vital for a fish species to complete their life cycle (Cowx & Welcomme 1998).

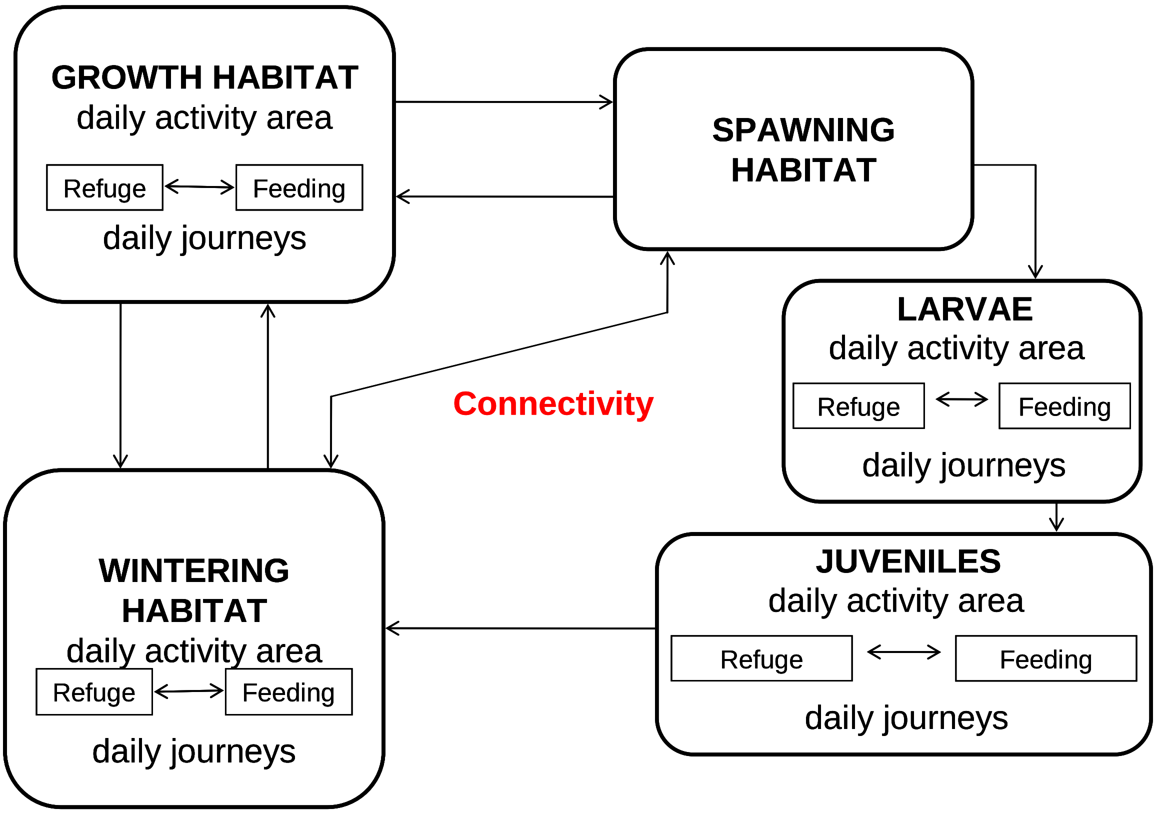


Figure 1.2. Functional units for fish (taken from Cowx & Welcomme 1998).

Globally, there is a need to understand ecological responses to changes in physical habitat as a result of anthropogenic pressures (Vaughan *et al*. 2009), loss of structural complexity and degradation of spawning, nursery and refuge habitats through river modification, can have implications for the fish fauna (Pretty *et al.* 2003). River rehabilitation is needed to address channel degradation by introducing diversity back in to the river channel by improving habitat for aquatic biota. To understand what 'best practise' measures are needed for successful river rehabilitation, knowledge of habitat suitability criteria for individual species is needed (FAO 2008). Habitat suitability criteria are based on the assumption that a species' preferred habitat is influenced by the most favourable conditions therefore, as the favourable condition decreases so will the species preference (Petts 2008). Life history studies from the literature can be used to produce habitat suitability criteria for individual fish species, but the viability of life history studies is directly related to the diversity and extent of natural habitats and related processes within a river basin (Cowx *et al*. 2004). Consequently, preference may be influenced by limitations in or absence of habitat. Some fish species can thrive under extreme conditions, including habitats that have been drastically degraded by anthropogenic causes, and it must be recognised that most species may only survive under sub-optimal habitat conditions and may therefore not fulfil important life stages such as breeding (Cowx *et al*. 2004). The relationship between fish community structure and the functional complexity of riverine habitat makes the use of functional ecological guilds (a group of species that exploit the same class of environmental resources in a similar way (Root 1967)) more suitable than the use of single species habitat preferences. Thus, it is imperative that habitat rehabilitation projects aim to benefit species on a whole ecosystem level by improving all the functional units used by the fish population at various life stages, because the risk of managing species on an individual level could lead to population decline (Nehlsen *et al*. 1991; Lichatowich *et al*. 1995; Reeves *et al*. 1995; Frissell *et al*. 1997; Cowx & Welcomme 1998; Beechie & Bolton 1999; Palmer *et al*. 2005). For example, rebuilding anadromous fish populations (e.g. salmon, lamprey) requires habitat rehabilitation that covers the entire watershed because their life cycle includes headwater spawning reaches, mid river spawning and rearing habitats, and estuarine rearing habitat (Beechie *et al*. 2010). Rehabilitation can be identified as being successful when the ecosystem contains sufficient biotic and abiotic resources to be self sufficient, sustaining itself both structurally and functionally (SER 2004).

River rehabilitation should seek to improve the ecological integrity of rivers, and the number of rehabilitation schemes in the UK increase each year in an attempt to return our rivers to good ecological status. As our knowledge of river rehabilitation increases over the years many of our questions remain unchanged and unsolved, such as how much rehabilitation do we need? And how do we know if our efforts are succeeding? It is the absence of adequate monitoring and evaluation that constrains the ability to assess the effectiveness of rehabilitation techniques, consequential to the outcome of rehabilitation being intangible and difficult to quantify (Eden & Tunstall 2006; FAO 2008). Presently, there are many single, small-scale efforts to measure rehabilitation impacts but these differ in their goals, are not integrated with one another and measure different factors, at different temporal and spatial scales, with different techniques (Roni *et al*. 2002). For this reason, this thesis gives a practical insight in to the limitations in monitoring design and evaluation of a number of different, small scale rehabilitation schemes that increase with complexity throughout chapters 3 and 4.

Monitoring and evaluation provides critical information on restoration project effectiveness, in addition the selection of assessment methods for habitat and biota is vital when determining how physical habitat and biota respond to different restoration techniques (Roni *et al*. 2013). As previously mentioned there are four BQEs involved in monitoring for WFD classification of river health (Schmutz *et al*. 2007), fish are the focal community used to monitor the effectiveness of the rehabilitation schemes within this thesis since they are an important indicator species that can be used to assess the ecological integrity of rivers (Karr 1991; Schiemer 2000). Measuring fish population response is the most direct way to understand outcomes of rehabilitation (White *et al*. 1999) additionally; it is also advantageous to focus on one sensitive or dominant species because they are more responsive to environmental change (Sedgwick 2006). Brown trout (*Salmo trutta* L.) were selected as an indicator species because they are categorised as a low tolerance/sensitive species (SNIFFER 2008) and will be most reactive to rehabilitation works. Furthermore, brown trout adults mature after 3 years (Maitland 2004) meaning a relatively short timeframe is needed to identify any changes in life history traits of one generation. Habitat data were collected and analysed through the application of HABSCORE, this habitat technique was determined by the focal species (brown trout) because HABSCORE is a standard method for measuring and evaluating salmonid stream habitat (Wyatt, Barnard & Lacey 1995).

1.3. Thesis Scope

The scope of this thesis works towards moving forward by identifying a number of key areas that need more focus:

- scientific understanding of river rehabilitation;
- project planning framework;
- implementation of monitoring, evaluation and dissemination;
- application of benchmarking and endpoints to measure success;
- integrated catchment management;
- intricacies of integrating river rehabilitation into flood risk management.

The objectives of this research are to:

- Review existing literature to identify best practise knowledge for adaptive management for river rehabilitation to develop environmentally and socially acceptable methods for a) effective rehabilitation of rivers and b) flood risk management whilst protecting aquatic biota.
- Provide managers and policy makers with a suitable understanding of the processes involved with project planning framework for river rehabilitation.
- Monitor and assess several, different small scale river rehabilitation projects to identify issues that need to be addressed.

The thesis is structured to address these objectives as follows:

Chapter 2 Gap analysis of current literature to examine the stages required in an adaptive framework for river rehabilitation such as goals and objectives, specific hypotheses, selecting a monitoring design and impact assessment, monitoring parameters, spatial and temporal replication and evaluation and knowledge transfer to ensure project success.

Chapter 3 Overall this chapter will identify what constrains rehabilitation project design and implementation by investigating a number of rehabilitation case studies. The case studies start at a basic level and build complexity with each completed scheme. In doing so, they show the limitations in monitoring and impact assessment design at a range of scales that enable patterns and rules to be teased out. Correspondingly, the specific objectives for each case study are presented at the start of each relevant section and these will vary according to their complexity. The case studies are:

- **3.3** instream remediation of Driffield Beck, a small urban stream;
- **3.4** channel narrowing of Lowthorpe Beck, a Yorkshire chalk stream;
- **3.5** brash revetment to prevent bank erosion on the River Manifold;
- **3.6** small weir removal on the River Dove;
- **3.7** artificial riffle reinstatement on the River Stiffkey.

Chapters 4 and 5 investigate the effectiveness of integrating river rehabilitation within flood risk management to provide the opportunity for an effective, sustainable framework essential to meet objectives under the EU Water Framework Directive (WFD) and Habitats Directive along with the EU Flood Directive and the UK Flood and Water Management Act. The investigation uses case studies from the River Don in Sheffield City Centre and a suburb of the city where flood mitigation works have been carried out.

Chapter 6 summarises the information gathered from all previous chapters and highlights the key components that are overlooked during river rehabilitation planning. It also suggests how to overcome these issues to advance current rehabilitation practise, to measure project success and therefore increase scientific knowledge of river rehabilitation. Furthermore there are recommendations for future study.

CHAPTER 2. ASSESSING THE IMPACT OF RIVER REHABILITATION SCHEMES – A MISSING DIMENSION OR UNNECESSARY PROCEDURE?

Many river rehabilitation projects take an opportunistic approach through trial and error and it is bad practise to continue in this manner (Buijse *et al*. 2005). One key issue that needs to be addressed is moving from decisions based largely on subjective judgements to those supported by scientific evidence (Boon & Raven 2012) and this is almost certainly a consequence of poor project monitoring, evaluation and dissemination that should play a vital role in rehabilitation programs to determine the effectiveness of rehabilitation actions to support the WFD (Roni 2005; Wolter 2010). The absence of knowledge is due to a lack of understanding of the design and implementation stage of project planning for rehabilitation schemes even though there are several; easily accessible river rehabilitation manuals that provide detailed guidance in this area, few projects use a structured method. There are clearly a number of constraints that need to be understood and overcome to achieve 'good practise' for river rehabilitation. This chapter provides an analysis of current literature to examine the stages required in an adaptive framework for successful river rehabilitation and subsequently identifies gaps in the literature that will be addressed throughout this thesis. Effective catchment management involves the complex interactions between social, economic and environmental systems when working towards river conservation through rehabilitation activities (Letcher & Giupponi 2005). Balancing a number of ecosystem services whilst supporting the ecological diversity of rivers is challenging. Cultural and supporting ecosystem services are inclined to support ecological heterogeneity, whereas provisioning and regulatory services are less likely (Van der Meulen *et al*. 2008; Table 2.1). Although investigating the value of ecosystem services was not in the scope of this study, a discussion of the difficulties that arises when attempting to incorporate habitat rehabilitation into flood risk management practise (regulating service) can be found in Chapters 4 and 5.

For a river to achieve ecological heterogeneity it needs to consist of a variety of habitats such as different flow dynamics (pool-riffle-run systems), depths, cover, substrate etc (Cowx & Welcomme 1998). Fish need all of these various habitats to fulfil each of their life stages such as growth, feeding and refuge as an adult, juvenile or larvae. Since ecological diversity and more specifically fish assemblage are directly related to the variety and extent of natural habitats within a river basin (Gorman & Karr 1978; Schiemer *et al.* 1991; Pearsons *et al*. 1992; Cowx & Welcomme 1998), fish were chosen as the biological quality element to be monitored during this thesis. The majority of rehabilitation schemes result in habitat manipulation, and for this reason it is imperative not only to monitor habitat change but also fish as response variables because they are ideal ecological indicators to monitor habitat diversity and rehabilitation effectiveness (Karr 1991; White *et al*. 1999; Schiemer 2000; Roni *et al*. 2005b). When studying species response it is important to monitor entire community structures, but at a single species level fisheries assessment should be completed through the study of fish population dynamics, density, age & recruitment (Beechie *et al*. 2003).

Table 2.1. Some ecosystem services present on rivers (Van der Meulen et al. *2008).*

Provisioning	Regulating	Cultural	Supporting
Food & goods Biomass for renewable energy Water supply Fish production Fibre & fuel Hydroelectric power Transportation	Nutrient removal Temperature regulation Caron sequestration Flood protection Groundwater recharge Pollution control Soil formation Pollination Nutrient cycling	Recreation Aesthetic values Education services	Habitat biodiversity

Rehabilitation methods provide a mechanism that should lead to long-term sustainable development where remedial action should focus on the underlying cause(s), with a primary objective (Table 2.2) of restoring the system to an acceptable state, ultimately leading to a self-sustaining resource (Cowx 1994). A well designed adaptive management project planning framework for river rehabilitation will reduce the uncertainty of management actions (Roni *et al*. 2005a) through the implementation of policies and application of a logical path that links rehabilitation goals (Table 2.2), watershed assessment, identification of rehabilitation needs, selection and prioritisation actions, design of projects, and development of a monitoring programme (Beechie *et al*. 2013; Figure 2.1, Table 2.3). More specifically, river health and fisheries status should be assessed and evaluated prior to establishing objectives at a catchment and local scale for river rehabilitation (Figure 2.1). An understanding of the pressures and impacts responsible for degradation will also enable suitable objectives to be established that address degradation issues and enable appropriate measures to be identified for rehabilitation (Figure 2.1). In some instances river health and fishery status may be satisfactory without any intervention through rehabilitation and therefore, objectives can be reset to sustainability.

Setting benchmarks and end points (Table 2.2) that are linked to clearly defined project goals is an effective way to measure project success (Table 2.2) and this should be included in the first stages of project planning. Once rehabilitation measures have been identified they need to be assessed for risk and uncertainty to confirm they are environmentally, socially and economically acceptable, if they are not satisfactory alternative solutions need to be established. Satisfactory rehabilitation measures need to be prioritised, once implemented post-monitoring is essential to evaluate river health and assess benefits. The framework should be transferable to individual rehabilitation projects by drawing on commonalities in objectives and techniques. An adaptive management framework (Figure 2.1) allows each of the stages of project management to be easily visualised and highlights where monitoring fits in to the framework. Selecting a suitable monitoring design, monitoring parameters with both spatial and temporal replication is essential for evaluation and knowledge transfer (Roni 2005; Beechie & Roni 2013) (Figure 2.1, Table 2.3).

Table 2.2. Definitions of key words for measuring project success.

Term	Description
Objective	Statement of specific and measurable outcomes
Goal	Statements of vision that defines project intent
Benchmark	A measurable target for restoring degraded sections of river within the same river or catchment
Endpoint	Endpoints are linked closely to project objectives to produce a target level of restoration, whether this is an ecological (to restore a level of function/species), social (ecosystem services) or physio-chemical (river morphology, water quality) endpoint.
Success	When objectives have been achieved to the standard required by the benchmark and endpoints

Effective management requires the collaboration between disciplines (e.g. hydrologist, biologist, ecologist, geologist, economist, sociologists, policy makers and the community) to distinguish between the social, economic and environmental requirements of rivers to allow accurate objectives to be set for river rehabilitation (Letcher & Giupponi 2005). Establishing objectives that relate to the functional aspect of the ecosystem is central to the development and applicability of a suitable monitoring strategy (Dewberry 1996 *cited in* Downs & Kondolf 2002; England *et al*. 2007) for successful river rehabilitation and should be one of the first steps within the framework. Objectives should work towards benefiting fish communities whilst enhancing our understanding of how communities respond to changes in physical habitat over time, taking into account the needs of individual fish species, size classes and guild structure, to recognise the 'missing' habitat and identify the habitat improvement measure needed. Designing a channel that will function naturally to meet rehabilitation goals is a complex process, monitoring and evaluation are put in place to identify rehabilitation project success, but how do we assess what is successful? Despite the improved knowledge of ecological, economic and social aspects of river rehabilitation (Postel & Richter 2003), there was still no agreement on what represents a successful rehabilitation project up till 2005 (Jansson *et al*. 2005). Over the past few years there has been much deliberation on how to determine project success and ideas have progressed towards setting benchmarks and end points that are linked to clearly defined project goals to help determine the measure of success within river rehabilitation (Bernhard *et al*. 2007; Roni & Beechie 2013). However, because this is a recent concept, few projects have incorporated benchmarking and endpoints in to their planning stage.

Benchmarking uses representative sites otherwise known as 'reference sites' on a river that have the required ecological status and are relatively undisturbed, this is then used as a target for restoring other degraded sections of river within the same river or catchment. This approach therefore uses appropriate undisturbed sites of the same river type (Rheinhardt *et al*. 1999), rather than attempt to create conditions unrelated to the original ones at the site of interest and is consequently more likely to result in long-term success (Choi 2004; Palmer *et al*. 2004; Suding *et al*. 2004; Woolsey *et al*. 2007). It is imperative that endpoints accompany benchmarking in the planning process to guarantee the prospect of measuring success because endpoints are feasible targets for river rehabilitation, especially as they do not need to be quantifiable. Endpoints are different to benchmarks as other demands on the river systems also have to be met and benchmarks can only function as a source of inspiration on which the development towards the endpoints is based (Buijse *et al*. 2005). River characteristics, pressures and rehabilitation measures are all factors that contribute to the individuality of each rehabilitation scheme. For this reason, benchmarks and endpoints play a vital role in the ability to measure river rehabilitation success because they use individual project objectives to determine project success.

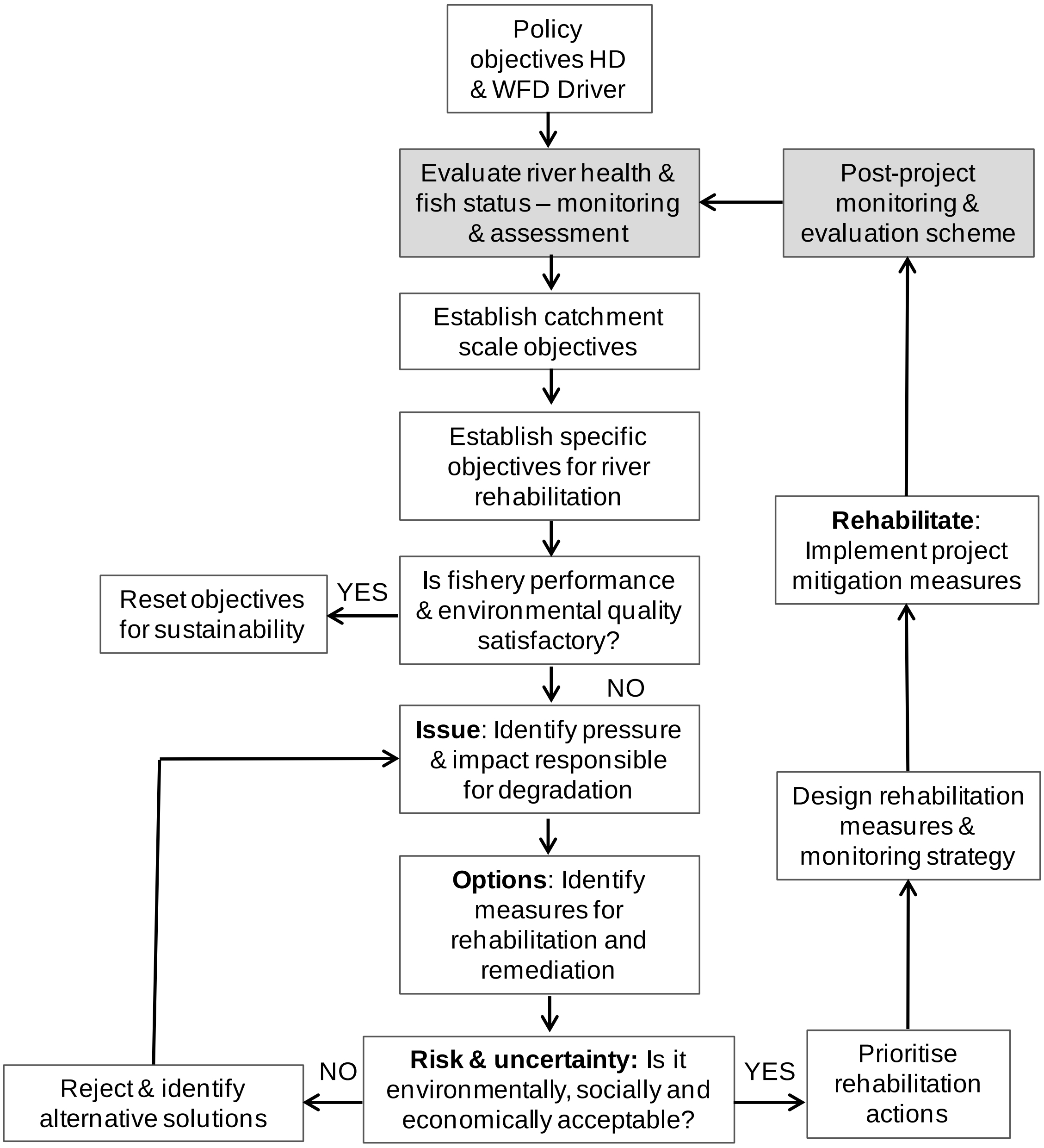


Figure 2.1. An adaptive management framework flow diagram for river rehabilitation project planning.

Goals and objectives of individual rehabilitation projects can vary because the particular ecological process of concern can differ between projects (Wohl *et al*. 2005), and given that benchmark standards cannot always be achieved, especially on urban rivers, endpoints will therefore assist in moving rehabilitation effort towards benchmark standards through application of the SMART approach to decide what is achievable and what is feasible (Hammond *et al*. 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).

Riverine ecosystems consist of a set of hierarchically nested physical, chemical and biological processes operating at widely varying space and timescales (Hermoso *et al*. 2011) and as a result large scale rehabilitation planning is critical. Unfortunately the majority of rehabilitation projects occur at a smaller scale with no consideration of the wider scale of degradation within rivers and often these small scale rehabilitation efforts can be more destructive than constructive (Frissell & Nawa 1992). The project planning of a rehabilitation scheme should also incorporate habitat unit (small scale) and reach (mid-scale), in addition to river basin (large scale) scales, when determining the scale of river degradation, selecting the type of rehabilitation action when monitoring the rivers biotic and abiotic response to rehabilitation work (Frissell & Ralph 1998; Roni *et al*. 2003; Roni 2005). Planning and implementation scales of river rehabilitation do not necessarily have to be the same, providing that the individual rehabilitation scheme is integrated at the whole catchment scale (Hermoso *et al*. 2011). The scale of monitoring should be decided in association with the target species or communities present determining their scale of response to physical change requires distinguishing between habitat, reach, or sector, and network scale effects and for fish in particular, their different life stages will need to be considered (FAO 2008). Smaller scale monitoring may be more achievable not only because it may be less problematic to carry out, but because it is likely to decrease natural variability in addition to sampling and observer error. However, it cannot always be transferred to a river basin scale because of uncertainty in the movement, survival and dynamics of biota (Roni 2005).

Table 2.3. Rehabilitation planning framework (Taken from Roni & Beechie 2013), identifying the stage where benchmarking and endpoints are require.

Programme component	Step	Product	Purpose
COMPREHENSIVE REHABILITATION PLAN	Define the rehabilitation goal	Clearly defined goal that specifies biological aims of rehabilitation	The rehabilitation goal guides the design of watershed assessment and choice of prioritisation criteria
	Benchmarking & endpoints	Clearly define benchmarks by use of reference sites, and end points	To measure the level of rehabilitation success
	Assess watershed process, habitat, and biota	Maps and tables indicating the causes of habitat change, which habitats have been most altered, which habitat losses most influence biological declines	Watershed assessment results are the basis for identifying rehabilitation needs, and inform the design of monitoring programs
	Identify rehabilitation needs	Maps and tables indicating which kinds of rehabilitation actions are needed where	Rehabilitation needs focus the selection of actions on those that are most important for achieving the biological aims of rehabilitation
	Select rehabilitation actions	Specific techniques that address identified needs and follow the process-based rehabilitation principles	Rehabilitation actions are selected to effectively address identified problems
	Prioritise rehabilitation actions	Prioritisation approach, and ranked rehabilitation actions or types of actions	Prioritisation should improve success and cost-effectiveness of rehabilitation efforts (i.e. actions are more focused on key problems)
PROJECT PROPOSAL	Design rehabilitation project and monitoring program	Project designs that accommodate driving processes and address high-priority rehabilitation needs	Produce rehabilitation actions that focus on effectiveness and sustainability
IMPLEMENTATION AND MONITORING	Implement rehabilitation and monitoring	Monitoring program that focuses on determining effectiveness of rehabilitation action types or suites of rehabilitation actions	Provide information needed to adjust rehabilitation plans or designs based on biological effectiveness of rehabilitation action

It is therefore essential that a fundamental understanding is needed in environmental science and the natural variability that accompanies it. Once these pressures have been identified, the scale of suitable measures for rehabilitation can be selected and evaluated through a risk assessment protocol. If proven to be economically and socially acceptable, rehabilitation measures can then be implemented. Post-project monitoring after implementation is essential to assess the success of rehabilitation works. Several attributes are needed to determine if and when successful rehabilitation is complete, a number of the following attributes should be considered (SER 2004):

- contain a community structure of species that occur in the reference ecosystem;
- consists of indigenous species to the greatest practicable extent;
- have a physical environment that is capable of sustaining reproducing populations of the necessary species;
- is suitably integrated into larger ecological matrix or landscape, with which it interacts through abiotic and biotic flows and exchanges;
- treated potential underlying threats to the health and integrity of the rehabilitated ecosystem;
- be self-sustaining to the same degree as its reference ecosystem.

It is difficult to have consistent measures for project success, but the use of objectives in the project framework allows managers to know when they have reached their goal. The feedback loop within an adaptive management framework (Figure 2.1) provides managers with the ability to account for uncertainty through evaluation of outcomes, and facilitate improved understanding of the efficacy of rehabilitation measures. This will enable all managers to adjust developments appropriate for the conditions and objectives (Bash & Ryan 2002; Wohl *et al*. 2005).

Developing a standardised methodology for river rehabilitation across a diverse array of river types is challenging because they cannot always be managed in the same way. The use of stream classification (grouping rivers with similar properties) is known to simplify rehabilitation strategies leading to a stream-type-dependent view (Bostelmann *et al*. 1998 *cited in* de Waal *et al*. 1998). However, it is not always plausible to have standardised methods for river rehabilitation. There are numerous manuals available to advise on river rehabilitation practices. Nevertheless, few manuals deal with the project appraisal side of rehabilitation practices, manuals that do address this are ‘Rehabilitation of Rivers’ by Cowx & Welcomme (1998), the RRC’s ‘Practical River Restoration Appraisal Guidance for Monitoring Options’ (PRAGMO) and ‘Stream & Watershed Restoration – A guide to restoring Riverine processes and habitats’ by Roni & Beechie (2013) which is a more up to date version of ‘Monitoring stream and watershed restoration’ by Roni (2005). They aim to assist practitioners involved in setting monitoring protocols as part of river rehabilitation project (Hammond *et al.* 2011).

2.1. Monitoring

Monitoring is imperative to all river rehabilitation project planning frameworks as it facilitates the evaluation of overall project effectiveness by assessing results (outcomes) against objectives. It is a vital stage in adaptive management as it influences the decisions made to continue, modify or discontinue management actions (Bash & Ryan 2002). Although the need for monitoring has been acknowledged in recent years (Roni & Beechie 2013) the majority of river rehabilitation schemes fail to assess outcomes and effectiveness, however, there are an increasing number of scientific publications in the peer reviewed literature relating to effectiveness, evaluation, and assessment and monitoring (Figure 2.2). In 2000, 43% of all publications in Web of Knowledge (June 2013) referenced the terms effectiveness, evaluation, assessment or monitoring, subsequently only small progress has been made to date (2006-44%; 2012-50%). This demonstrates progress in river rehabilitation science and management through the implementation of monitoring and evaluation (Ryder & Miller 2005; Wohl *et al*. 2005; Tompkins & Kondolf 2007; Woolsey *et al*. 2007). Nevertheless, there is still need for vast improvement as many projects are not monitored for example, of 37,099 projects listed in the U.S. National River Restoration Science Synthesis database, 20% had no project goals identified, only 58% reported project costs, and just 10% indicated any measure of assessment or monitoring (Bernhardt *et al*. 2005). The application of monitoring and evaluation should be promoted within river rehabilitation project planning as it will assist the EU Water Framework Directive's aim to ensure rivers reach good ecological status or potential by the year 2015.

There are a number of challenges and uncertainties to account for when attempting to understand the intricacies of how ecosystem networks respond to river rehabilitation. Challenges and uncertainties can be seen as a hindrance but can be overcome by increasing the efficiency of monitoring and evaluation through an adaptive management framework. Effective monitoring should follow a strategic listing of questions, such as what when and how should we monitor to identify the appropriate procedure/protocol for each individual rehabilitation projects.

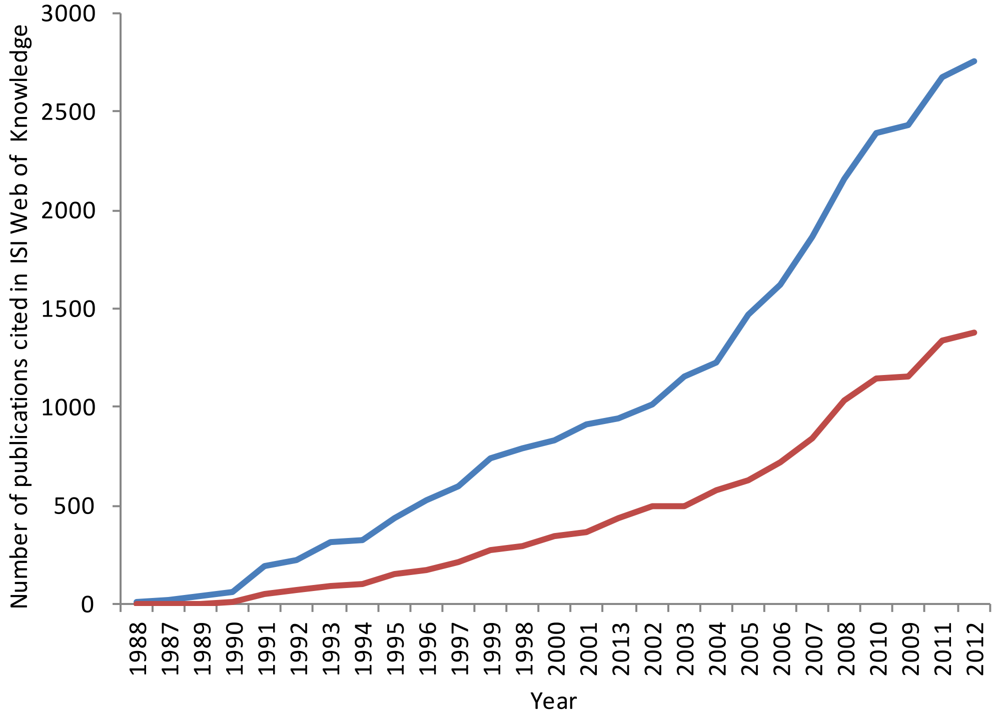


Figure 2.2. Publication trends for evaluating stream rehabilitation containing the key words Topic=(river* OR floodplain OR stream OR riparian) AND Topic=(restor* OR rehab* OR mitig* OR conserv*), Topic=(river* OR floodplain OR stream OR riparian) AND Topic=(restor* OR rehab* OR mitig* OR conserv*) AND Topic=(evaluat* OR effectiv* OR assess* OR monitor*)(Web of knowledge completed April 2013).

2.2 Importance of monitoring within a project framework

Monitoring and evaluation of rehabilitation schemes is a necessary process that should be included in all project planning frameworks because it determines the effectiveness of actions, and thus supports WFD requirements (Wolter 2010). Without well-designed monitoring and evaluation, adaptive management of rehabilitation ecology is implausible (Downs *et al*. 2002). The number of biological and multi-species metrics that can be used to measure and monitor aquatic ecosystem health has grown rapidly (Karr 1981; Karr 1991; Welcomme *et al*. 2006) and although they are vital contributing factors to successful monitoring, singularly they are insufficient in the assessment of river rehabilitation (Beechie *et al*. 2009). Selecting monitoring parameters should depend on the goals and objectives, definition of scale and selection of study design (FAO 2008). Fish are common BQEs used to monitor and evaluate instream rehabilitation (Roni *et al*. 2005b), use of these biotic monitoring techniques combined with physical habitat assessment can strengthen a rehabilitation scheme (England *et al*. 2007).

2.1.1. *Fish monitoring and habitat assessment methods*

The question of 'what' to monitor is challenging yet imperative, and ought to be used to standardise monitoring methods and make them transferable across rivers. The variable(s) to be measured during monitoring need to be identified prior to the start of survey work to help characterise the variability of the river system (Sedgwick 2006). The majority of rehabilitation schemes result in habitat manipulation, and for this reason it is imperative not only to monitor habitat change but also fish populations as response variables because they represent one of the best opportunities to test hypotheses within a river rehabilitation project (Roni *et al*. 2005b). Riverine fish are an important indicator group that can be used for assessing ecological integrity of rivers (Karr 1991; Schiemer 2000) because ecological diversity and more specifically fish assemblage are directly related to the variety and extent of natural habitats within a river basin; noting a river containing a complex habitat structure will have a diverse fish community (Gorman & Karr 1978; Schiemer *et al.* 1991; Pearsons *et al*. 1992; Cowx & Welcomme 1998). When studying species response it is important to monitor entire community structures, but at a single species level fisheries assessment should be completed through the study of fish population dynamics, density, age & recruitment (Beechie *et al*. 2003). Measuring population response is perhaps the most direct way to understand outcomes of rehabilitation (White *et al*. 1999). However, in some circumstances it may be advantageous or necessary to focus on one or several target or dominant species because they are more sensitive to environmental change (Sedgwick 2006). Indicator species can be used as the main interest of study and can be selected in two ways: firstly by using a sensitive species that will be the first to show a response to rehabilitation works, or secondly, by selecting the fish species that had poor status in the pre-monitoring assessment. This species can be used as an indicator to assess if rehabilitation work has benefited the fisheries status. Abiotic factors must also be considered within a monitoring framework to assess the possible cause of river deterioration or vice versa, basic habitat variables such as reach dimensions, substrate, flow and percentage cover should be collected regularly to assess the changes that have occurred to the morphology of the river through rehabilitation actions.

Sampling variability can occur during monitoring because of the variety of monitoring teams and agencies involved over long term data collections, resulting in potential inconsistencies in data collection that could further hinder data analysis (Archer *et al*. 2004). It is therefore imperative that monitoring methods are quantifiable, repeatable and consistent; this will enable them to be comparable across different locations by other monitoring teams to reduce sampling variability (Archer *et al*. 2004).

2.2.1 *Spatial & temporal replication*

The timeframe over which monitoring programmes are implemented should capture the natural range of behaviour of the river to show the timeframe over which geomorphological adjustments occur (Brierley *et al*. 2010). However, it is difficult to foresee the recovery time-scale for any rehabilitation project, especially those based around geomorphological modifications. When physical structures are installed in river channels to improve fish habitat, the adjustment process that occurs over time can sometimes be more harmful than good (Rosgen 1994). Ecological recovery time from this type of habitat modification depends on hydromorphological characteristics of the river (Brookes 1996; Sear *et al*. 1998) and how this further affects ecological process within the river; for this reason long-term monitoring is needed to enhance understanding (England *et al*. 2007). Few ecosystems have been studied comprehensively in terms of their abiotic parameters, species composition, community structure, functional attributes, and responses to natural disturbance (Clewell & Rigour 1997). Recognising when monitoring should take place is vital to increase the accuracy and understanding of the success level of each rehabilitation project. Both pre and post monitoring is essential within a river rehabilitation project planning framework. Pre-monitoring includes the collection of baseline data to assess the status of river health and fisheries health, and assist in the identification of river rehabilitation objectives (Kondolf & Downs 1996). Baseline data (or pre-monitoring data) can be used within river rehabilitation assessment to compare the status of habitat and fisheries of the river between pre and post monitoring of the rehabilitation works. Evaluating multiple control sites across a spatial scale will allow the level of success of rehabilitation projects to be measured by taking in to account patch dynamics (Clegwell & Rigour 1997) to give a comprehensive review of the biota local to that river. Post-monitoring is an essential phase that is needed to assess the success of rehabilitation works, and long-term, post-monitoring will provide a more valuable data source for evaluation purposes. It is not always easy to know the length of monitoring needed to identify project success, Kondolf & Micheli (1995) suggest it should cover at least 2 generations of the longest living species. Therefore, for trout this would be a minimum of 6 years, but for a species that takes longer to mature, for example chub (mature between 3-5 years) (Maitland 2004), it would be a minimum of 6 to 10 years of monitoring and this is rarely possible in real life scenarios.

Regular, long-term monitoring will account for natural variability in fish population dynamics as illustrated by Figure 2.3. The dots represent the occurrence of regular sampling of the metric in question, whilst the red line shows the variability between each measured metric over time. If sampling were to be sporadic, for example, to only sample the time demonstrated by the red dots (show a decline in the measured metric) or the green dots (show an incline in the measured metric), a contradictory view could be had. So, if fish were to be the metric collected over time, sampling that occurred only at the time represented by the red dots would show a decrease in fish numbers over time, however, sampling that occurred only at the time represented by the green dots would show an increase in fish numbers over time. If sampling was to occur more regularly, for example, at the time of both green and red dots, a more accurate observation could be made and in the case of Figure 2.3, the metric measured would be considered stable. Overall, Figure 2.3 is an exaggerated example to demonstrate that infrequent monitoring can give false results that can be overcome with frequent monitoring that will capture the variability within the data and will give a more accurate portrayal. A number of unmanageable factors (weather, predation, disease *etc*.) are known to depress populations even when the habitat can support a larger population

(Block *et al*. 2001). These factors can influence natural fluctuations within populations that only frequent monitoring can identify. Therefore, the timeframe and frequency of monitoring is completed is fundamental when it comes to overcoming the complex interactions within an ecosystem and understanding the lag time associated with rehabilitation activities (Beechie *et al*. 2000, 2005, 2009). Long-term monitoring of rehabilitation works is essential to ensure that the population has time to adjust to time-dependent changes (Block *et al*. 2001) so accurate evaluation of the rehabilitation scheme can be made, although short-term monitoring will not necessarily capture these time-dependant changes it is still necessary.

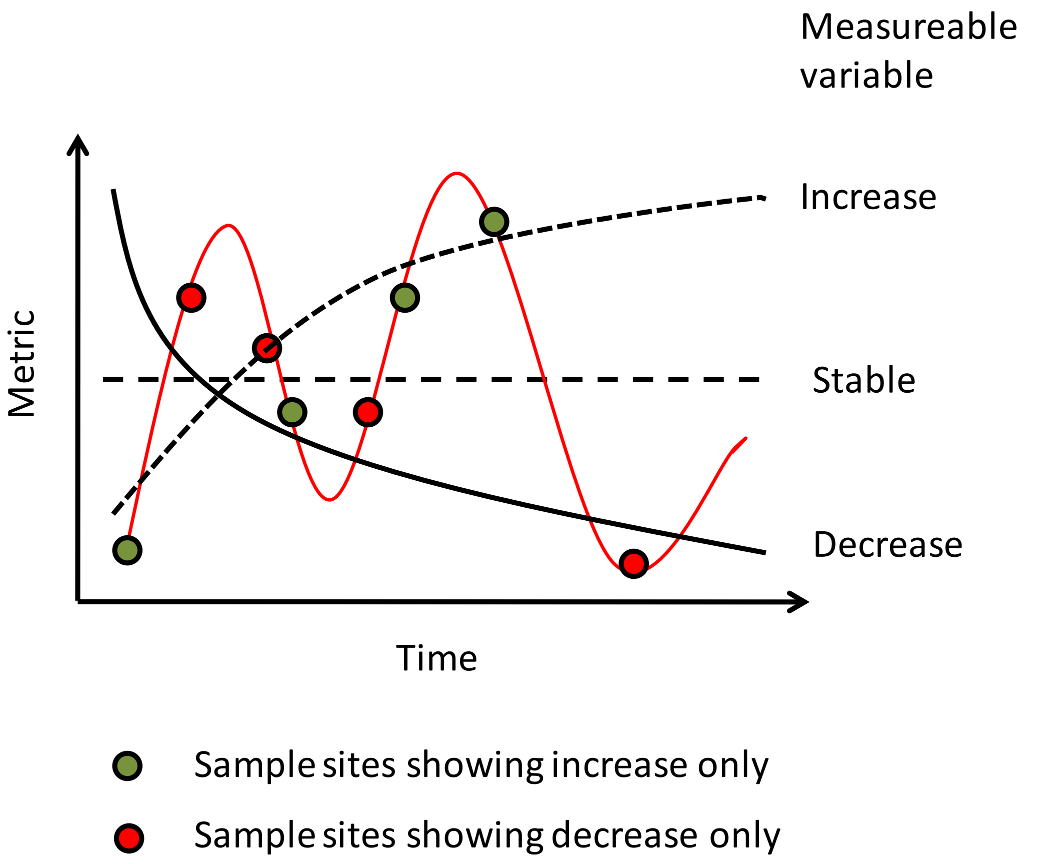


Figure 2.3. Conceptual graph demonstrating the importance of long term, regular monitoring.

Both spatial and temporal monitoring is required when monitoring rehabilitation schemes. Ecosystems exhibit natural fluctuations from patch dynamics that differs between sites (Clewell & Rigour 1997) and hence, a single control site does not necessarily represent the habitat across the whole river, and therefore, cannot accurately assess the efficiency of a rehabilitation scheme. Successful monitoring requires multiple control sites and frequent sampling of variables to overcome patch dynamics and to identify trends, thus improving understanding of the natural variability that occurs within a river ecosystem.

Control sites provide a basis of comparison between the rehabilitated area and the conditions before rehabilitation, accounting for natural variability and helping to differentiate between any ecological changes associated with seasonality or temporal population dynamics, and are essential for river rehabilitation monitoring (White & Walker 1997; Roni 2005; England *et al*. 2007). Nevertheless, on the whole most rehabilitation projects do not include these fundamental requirements in their experimental design (Minns *et al*. 1996). The National Research Council (NRC 1992) stated that '*one of the most effective ways to establish rehabilitation goals is to evaluate the success of stream rehabilitation by comparing biological communities in a disturbed reach to communities in a set of relatively undisturbed reference streams of the same order in the same eco-region*.' Reference and control sections should be selected with caution to ensure the geology (including gradient), hydrology, biology and scale of habitat modification are comparable with the rehabilitated reach (Roni 2005; Wyzga *et al*. 2009).

There is a dispute over the definition of control and reference sites amongst some scientists. Some believe they have the same meaning, others believe a control site is more specifically defined as being identical to the treatment site, with the exception being the treatment or rehabilitation action, whereas a reference often is defined as the ideal or pristine state, with conditions unaltered by human activities (Downs *et al*. 2002). Pairing a control or reference site to an impacted site within the same reach will significantly account for variability, albeit worth noting these paired sites will not be totally independent of each other because there may be upstream-downstream effects and fish movements (Roni *et al*. 2005b). Within a stream, control reaches generally should be located upstream from treatment reaches (Roni *et al*. 2005b), but there are always limitations when using control or references sites to assess the success of river rehabilitation as sites may not be accessible or may not be present (Clewell & Rieger 1997). In extreme cases a similar near-by river could provide a more comparable reach but the risk of including the effect of other variables is increased (Sedgwick 2006).

2.3 Planning an impact assessment

The rationale behind monitoring river rehabilitation is to detect differences in habitat characteristics that result from an impact (rehabilitation) and can be achieved by comparing the mean difference from the data to determine the overall significance of the differences. Within the content of this chapter the term impact is used to describe anthropogenic changes made to rivers, whether they are negative changes resulting from pressures, or changes due to rehabilitation (either positive or negative). Irrespective, they are both man-made changes and therefore defined by the word 'impact'. Unfortunately our ability to detect changes in habitat features is often masked by the natural heterogeneity of a river system. Understanding the magnitude of these changes and where sources of error might occur allows scientists and managers to design 'meaningful' monitoring studies that can account for this variability (Archer *et al*. 2004), otherwise known as impact assessments. '*The principle idea of an impact assessment, as opposed to a monitoring study, is not only to show a change has taken place but also to provide evidence, in statistical terms, that it is meaningful'* (Sedgwick

2006). In the previous section the importance of identifying project aims and objectives, setting specific hypotheses, identifying the parameters needed prior to any rehabilitation project and discussed the importance of monitoring design was established.

The effectiveness of management actions cannot be evaluated without ongoing monitoring and evaluation of outcomes (Brierley *et al*. 2010), and therefore it is essential that an impact assessment is suitably planned for each rehabilitation scheme. The design of appropriate monitoring and evaluation programmes for stream rehabilitation will differ by project type as well as by region, geomorphology, scale, and a variety of other factors (Roni *et al*. 2005c). The use of impact assessments within adaptive management enables suitable monitoring designs to be identified for different project types. The development of sampling designs has improved our ability to detect human impacts against natural variability of river ecosystems through the use of statistical analyses (Ellis & Schneider 1997). A variety of impact assessments techniques are available for detecting environmental impacts of rehabilitation project whose data collection methods differ spatially and temporally. These impact assessment techniques are Before/After (BA) contrasts at a single site, Before/After and Control/Impact (BACI) sampling sites and repeated BACI and post-treatment design (Ellis & Schneider 1997). Conquest (2000) believed that analysis of data from a BACI design in combination with professional expertise is acceptable but only where caution is used in data interpretation. However, restrictions in design criteria can cause complications in impact assessment that could prevent the outcome of meaningful evidence to identify the basis of the impact (Sedgwick 2006). The following section introduces the different types of impact assessments available for river rehabilitation monitoring, the methods stated are applied in Chapter 3 when analysing a variety of rehabilitation case studies.

2.3.1 Monitoring design

BA design

Impact assessment using a BA design is intended to focus monitor at the impact/treated site before and after rehabilitation (Green 1979), therefore it only measures the site of impact so it is generally replicated in time rather than space (Morrisey 1993; Roni 2005).

BACI design

Impact assessment using a BACI design consists of sampling before and after at the impacted site and also at a control site. This was first proposed by Green (1979) and further developed by Stewart-Oaten *et al*. (1986). The addition of a control(s) is intended to account for environmental variability and temporal trends found in both the control and treatment areas and therefore increase the ability to differentiate treatment effects from natural variability further strengthening the power of analysis (Smith *et al*.

1993; Roni *et al*. 2005a). An impact is therefore determined as an effect over and above that which can be attributed to temporal and spatial influences (Sedgwick 2006). Adding statistical power to the BACI design may isolate the treatment effects and can be achieved with multiple control sites (spatial replication) and long-term sampling (temporal replication) to overcome natural variability (Underwood 1994; Roni 2005). The design applies a 2 way factorial ANOVA, '*when the existence of a significant interaction term is taken as evidence of an impact effect, i.e. the existence of an effect above and beyond what would be expected from the mean differences in the before and after periods and the upstream/downstream locations*' (Sedgwick 2006).

Repeated BACI, or Paired BACI (BACIP) as it is also known, is a further modification from Green's (1979) BACI design. Hulbert (1984) identified problems in the original BACI design where treatments are not replicated or replicates are not statistically independent (otherwise known as pseudoreplication). This led to a new design, the before after control impact paired (BACIP) sampling design developed by Stewart-Oaten *et al*. (1986); Bernstein & Zalinski (1983) also contributed to the development of paired BACI designs. Stewart-Oaten's BACIP model differs from Green's model by explicitly including estimates of temporal variability, as well as spatial variability of replicates These BACIP designs sample each site several times, at random, prior to and then after the start of the potential disturbance (Ellis & Schneider 1997). A replicated BACI design potentially is the most powerful of all study designs because it includes replication in both space and time (Downes *et al*. 2002) and increase the applicability of results to other areas (Roni *et al*. 2005a). In the absence of true replication, the results of both BA and BACI study designs should be considered for case studies whose results will still contribute to our knowledge of physical and biological processes (Roni *et al*. 2005a). The BACIP design can be too demanding on time, recourses and cost to meet management objectives for river rehabilitation, it may be ideal for research, but is not practical in reality.

Post-treatment designs

Post-treatment designs have frequently been used for impact assessment of many river rehabilitation projects where the collection of pre-data has not been an option. Insufficient or no prior data from the impacted site may limit the scope of the design and reduce the efficiency of the analysis. In general, post-treatment designs tend to apply spatial replication more than temporal replication and therefore, data from the impacted site can be compared to a control/reference site, enabling a BACI design to still be generated but with a more limited capacity to detect temporal variation (Sedgwick 2006). Downes *et al*. (2002) believed it is still possible to learn from post-treatment designs even though there scope is limited; this is also accentuated by other studies (Hilborn & Walters 1981; Hicks *et al*. 1991). There are two main post-treatment designs and these are intensive post-treatment (IPT), where multiple years of data are collected at one or few paired control and treatment sites; and extensive post-treatment (EPT), where many paired treatment and control sites are sampled over a one to three year period (Hall *et al*. 1978; Hicks *et al*. 1991 *cited in* Roni *et al*. 2005a). The EPT design includes only spatial replication and is therefore most applicable at reach scale whereas, the IPT design comprises of both spatial and temporal replication and can be used as a BACI design without the pre-data, but it is still limited in its efficiency to identify change from a rehabilitation action (Roni *et al*. 2003; Roni *et al*. 2005a).

Resource calculation – target variance

Resource calculation is an essential procedure within an impact assessment for river rehabilitation project planning as it helps to determine how many years, locations or samples are required to isolate the environmental impact from natural variability that can occur at a temporal or spatial scale (Green 1989; Fairweather 1991; Faith *et al*. 1991; Osenberg *et al*. 1992; Cooper & Barmuta 1993; Osenberg *et al*. 1994; Zar 1999; Roni 2005). Resource equations usually use a small pilot study or routine monitoring programme to determine the level of future sampling however, in the context of this it was used to establish if sufficient sites were sampled to identify a change with a stated level of probability, where a reliable estimate will have a low variance. It is also necessary to consider the degree of change (target variance) that would have meaningful and relevant implications for the future integrity of the fishery; it is an important decision because it will determine the subsequent remedial actions (Sedgwick 2006). Primarily, it is not statistical but biological and involves estimating the magnitude of change through fish abundance that will have ecological or management relevance.

2.4 Importance of evaluation and transfer of knowledge

The availability of information gathered about successful rehabilitation programmes is limited because evaluation of the majority of rehabilitation measures is inadequate (Roni *et al*. 2005; Klein *et al*. 2007). Assessing rehabilitation success can be challenging, nevertheless, it is essential to advance our understanding of river rehabilitation. Incorporating monitoring and evaluation into a well defined project framework, where performance standards and monitoring protocols are outlined prior to project installation (Clewell & Rigour 1997) will help overcome the question of 'how much is enough?' Evaluating the effectiveness of rehabilitation projects by testing objectives against outcomes should provide evidence of success.

Reporting results for dissemination is an important step and will enable us to learn from success or failures, and thus improve river rehabilitation and management tools. Successful river rehabilitation requires effective communication amongst a number of specialist multidisciplinary groups to strengthen scientific knowledge in this area. Members of the public, in addition to planners, decision makers, politicians, farmers, practitioners, water managers, engineers to name a few, all need feedback and education about current information on river rehabilitation to support future actions (Wade *et al*. 1998).

This chapter demonstrates that there is good understanding of the design and implementation stage for successful project planning for rehabilitation schemes, guidance in this area is current and easily accessible through manuals previously mentioned. Nevertheless, few rehabilitation projects can declare if their actions are successful, it is also clear that only a limited number of projects apply suitable design and implementation. It is therefore apparent there are constraints that need to be understood and overcome to achieve ‘good practise’ for river rehabilitation. The subsequent chapters will attempt to identify what constrains rehabilitation project design and implementation by investigating a number of rehabilitation case studies. In particular, by focusing on limitations in monitoring design and to identify problems in the impact assessment stages to therefore identify how to overcome these issues and advance current rehabilitation practise and therefore, increase scientific knowledge of river rehabilitation.

CHAPTER 3. IDENTIFYING LIMITATIONS FOR REHABILITATION SUCCESS

3.1. Introduction

The previous section reviewed the important stages of a project planning framework and highlighted the need for a better understanding of monitoring and evaluation to determine project success. The subsequent chapter will attempt to identify limitations in monitoring design using practical examples of river rehabilitation that increase with complexity. It is imperative to have an understanding of the pressures responsible for degradation, how they degrade habitat, further impacting on biota to then identify suitable measures for remedial action. Firstly, this chapter overviews the numerous drivers, pressures, impacts and measures on river ecosystems and then gives a more detail account of the pressures and measures involved in the case studies provided.

Hydromorphological degradation occurs as a consequence of multiple pressures driven by anthropogenic actions such as agriculture, urbanization, industry, water supply, navigation (and transport in general), fisheries, recreation and flood protection (FORECASTER 2010). These drivers lead to pressures that alter the morphology of the river channel, such as impoundments and channelization, alteration of riparian vegetation and instream habitats, increased sedimentation, embankments and loss of connectivity (Cowx & Welcomme 1998). These drivers may also influence water abstraction and flow regulation that will interfere with the hydrological regime by altering river discharge, interbasin flow transfer and may possibly lead to hydro-peaking in some instances (Poff *et al*. 1997). Modification of rivers has resulted in the loss of habitat availability, creating unfavourable conditions for aquatic organisms and consequently reducing the heterogeneity of the ecosystem (Postel *et al.* 1996; Sala *et al.* 2000; Tilman *et al.* 2001). More specifically these modifications to aquatic habitats have detrimental effects on fish populations and their functional units. Rehabilitation of rivers is a tool implemented through measures to improve hydromorphological and ecological conditions to overcome pressures. The aim of river rehabilitation is to re-create functional habitats and connectivity between these habitats (Figure 1.2, Cowx & Welcomme 1998) and has become a widespread tool across the globe to address river degradation by re-introducing habitat heterogeneity to the river system to encourage higher ecological diversity (FAO 2008). The number of rehabilitation schemes in the UK increases each year in an attempt to return our rivers to good ecological status, and the Environment Agency invests approximately £10 million a year on rehabilitation works (Walker 2002), although this has increased as a result of the drive to meet WFD targets. The majority of techniques used in river rehabilitation attempt to rehabilitate natural features by using physical instream methods such as channel narrowing, bank re-profiling and reinstating riverbed features to improve rivers. Additionally, the loss of longitudinal connectivity can be overcome for some species by the installation of fish passage facilities or fish easements (Cowx & Welcomme 1998). More detailed examples of pressures and WFD hydromorphology mitigation measures can be found in Appendix 1, (modified from Royal Haskoning). The discussion below and following subsection take a few of the more common measures and examine their efficacy, through case studies, in relation to achieving improvement in ecological quality for fisheries.

There has been a sharp decline of salmonid populations in recent years (Roni & Quinn 2001) and in particular anadromous salmonids, due to four main factors: over harvesting of fish, impacts of aquaculture, impacts of migratory movements and reductions in the potential productivity of salmonid habitats that occur in rivers (Lichatowich 1999; Montgomery 2003) and it is the later that drives this chapter. Habitat modifications through river rehabilitation most frequently take place on salmonid rivers due to their commercial value, feasibility of operation and relatively low costs of measures. This chapter will specifically focus on brown trout, with the exception of 3.7 artificial riffle reinstatement case study on the River Stiffkey, where sea trout are also referred to. Trout are ideal indicator species when monitoring river rehabilitation because of their life history traits. Their short life span allows for monitoring of more than one generation in a short timeframe and their fast growth enables identification of rapid change.

Essential habitat for adult salmonid spawning and embryos in rivers includes cool, clear and well oxygenated gravel-bed streams with suitable substrate size, water velocity, water temperature, water depth and dissolved oxygen (Quinn 2005; Wheaton et al 2010). Substrate size will influence the suitability of redds (Kondolf & Wolman 1993), velocity and depth will influence access to spawning sites and energy expenditure (Quinn & Buck 2001; Quinn *et al*. 2001a, 2001b; Hinch *et al*. 2002; Ferreira *et al*. 2010). These essential salmonid spawning characteristics are regularly put under pressure and one of the main problems in the UK is the input of fine sediment from arable fields in to rivers, which has been shown to impact negatively on salmonid reproductive success (Alexander & Hansen 1986; Kondolf 2000; Ferreira *et al*. 2010). Negative effects include the clogging of gravels, increase turbidity of water and input of toxic chemicals, all of which affect salmonid spawning habitat by smothering eggs and reducing oxygen levels (Pawson 2008). The introduction of gravel riffles and flow deflectors as rehabilitation measures to overcome the degrading of salmonid spawning habitat have become progressively more common, for example, Harpers Brook (Harper *et al*. 1998). In some cases artificial riffles have been proven to enhance hydro-geomorphological characteristics such as depth, water velocity and substrata, all of which benefit river ecosystems, especially salmonid spawning habitats (Downs & Thorne 1998).

Although a vast amount of work has been completed on salmonid spawning preferences and the types of abiotic factors that define good conditions are now well known (Greig *et al*. 2007), little research has been carried out to determine the success of rehabilitated spawning habitats and their benefits for fish populations (Kondolf & Wolman 1993; Geist & Dauble 1998; Reeves *et al*. 1991; House 1996; Gortz 1998). As rehabilitation measures are becoming increasingly widespread, it is crucial that legislative support be translated into river rehabilitation techniques that benefit freshwater ecosystems (Downs & Kondolf 2002; Bernhardt *et al*. 2005). More attention needs to focus on the importance of rehabilitation monitoring (Wheaton *et al*. 2010) and consequently, part of this chapter will focus on assessing the impact of installing artificial riffles and how it affects the recruitment success of resident brown/sea trout.

Habitat diversity of many streams and rivers has been degraded over the years as a result of debris removal from the channel mainly to reduce flood risk (ART 2005). The importance of instream habitat has now been recognised as a vital component for functioning river ecosystems and therefore instream rehabilitation works have become a popular technique to re-introduce habitat diversity to rivers. The placement of instream structures to increase habitat diversity and increase fish production is one of the key rehabilitation methods used to date and has a long history (Tarzwell 1934, 1937, 1938; White 1996, 2002). It mainly involves methods such as placing large woody debris (LWD), boulders and other materials into the stream channel. This can provide (ART 2005):

- Cover and living space for young fish
- Stabilizes river bank and beds
- Creates diverse niche habitats (pools and undercuts)
- Collects leaf-litter/retains fine organic matter & provides a substrate for biological activities by microbial & invertebrate organisms thereby providing a food source for fish
- Can improve water quality / oxygen levels
- Stores carbon
- Increases the range of stream temperatures
- Increases sediment deposition and storage
- Provides a respite to fish during high flows

Instream habitat maintenance can benefit river channel processes, ecology and to a certain extent river morphology; nevertheless, these types of efforts will only enhance and will not rehabilitate because they do not overcome the underlying cause of degradation (Roni *et al*. 2005). The aim of the case study on artificial riffle reinstatement is to investigate instream habitat remediation and fisheries habitat surveys to assess if it is an appropriate technique to reintroduce habitat diversity.

Geomorphologically, bank erosion is a natural process and is a central component of meander formation, lateral channel migration and movement of sediment over a long time frame (Lawler *et al.* 1997). River banks erode through a combination of natural geomorphic processes (Thorne 1982) encouraged through flow hydraulic relations such as water volume and velocity fluctuations (Lawler 1992; Brierly & Murn 1997). Although bank erosion has been identified as a natural geomorphic process, the undercuts it produces result in the collapsing of the bank, contributing considerable amounts of material to the overall catchment sediment yields (Coldwell 1957; Imeson & Jungerius 1977; Grimshaw & Lewin 1980; Trimble 1993). In addition, anthropogenic pressures can cause bank erosion and is more evident over a shorter time frame, for example, from livestock overgrazing and uncontrolled cattle drinking causing poaching of banks (Couper & Maddock 2001). There are many reasons why bank protection practises take place, including to reduce the loss of land for land owners and to minimise the detrimental effects of sediment loading on freshwater ecosystems. Material from bank erosion will either be deposited locally on the stream bed or transported downstream to be deposited on bars, floodplains and the riverbed. Deposition of bank side sediment into the channel can cause water quality problems, and is harmful to invertebrate communities and can degrade fish spawning habitat (Soulsby *et al*. 2001). Sediments will smother gravels, reducing the flow of oxygenated water to fish eggs and can therefore reduce the survival rate of salmonid eggs (Chapman 1988; Kondolf 2000). Several measures can be applied to reduce bank erosion, for example, rivers can be narrowed by creating a berm using faggots and can be filled in with excess channel silt to construct a new margin, returning the width of river to its previous state, increasing flow velocities and depth (FAO 2008). Preventing bank erosion as a result of livestock can be easily resolved by fencing to exclude cattle from river banks; this reduces the direct effect cattle have on bank stability. It will also enable vegetation to establish, strengthening the bank. There are also several bio-engineering techniques available to reduce bank side erosion, such as ‘soft revetments’ to reduce soil erodability by resisting tension and increasing cohesion by the reinforcement of the bank through rooting systems (Vidal 1969; Bull 1997; Abernethy & Rutherford 2001; Simon & Collision 2002). In addition bankside vegetation, willow cuttings, brash bundles, conifer tops, pinning logs or trees and branches can be used to redefine channel characteristics, improve bank stabilisation and contribute to the storage of sediment and habitat for aquatic biota (Jones *et al*. 2010). These techniques are becoming popular in river reengineering in the UK (Anstead & Boar 2010), especially as eroding banks threaten floodplain structures and agricultural land (Couper & Maddock 2001). It is important to recognize that to rehabilitate habitat diversity the processes (e.g. channel bed erosion and deposition) that create and maintain river channels need to be understood (Kondolf 2000b; Wohl *et al*. 2005; Kondolf *et al*. 2006). For example, can be drawbacks with the application of bio-engineering techniques because they can affect river processes downstream, especially by transferring the erosion problem to a downstream location. Sections 3.4 and 3.5 provide case studies to investigate if channel narrowing and soft revetments, respectively, are suitable techniques to provide suitable habitat for fish.

River continuum is fundamental for freshwater ecosystem function because many freshwater species rely on the longitudinal and lateral connectivity of a river to complete each life stage. Loss of longitudinal and lateral connectivity through the construction of barriers can lead to isolation of suitable habitat, isolation of populations, failed recruitment and local extinction (Cowx & Welcomme 1998; Bunn & Arthington

2002; Fjeldstad *et al*. 2011). Fragmentation of river systems are found in the majority of large rivers in the world (Nilsson *et al*. 2005) and can also be found in smaller river systems. Literature available on the affects of barriers and barrier removal on larger rivers is plentiful, but documentation on smaller obstacles such as weirs (those that do not cause a permanent barrier to fish migration) and their removal on small rivers is less common (Garcia de Leaniz 2008). Physical barriers obstruct the free movement of all fish species both upstream and downstream limiting their range to essential feeding, breeding and refuge sites and therefore affecting species composition and population structure of resident fish populations (Cowx & Welcomme 1998; Bain & Stevenson 1999). Several studies have shown that small weirs (<5 m) can also act as a barrier across a water course regardless of their height because their passability depends upon the hydraulic characteristics, water temperature, river flow and fish species attempting to migrate (Larnier 2001; Bunn & Arthington 2002). Small physical barriers can cause delays in migration or complete obstruction, and this will depend on the timing of migration and swimming capabilities of the target species (Northcote 1998). Small weirs can hinder the movement of a proportion of the population that are weak swimmers or young life-stages and can therefore impact on populations by increasing mortality and predation and decreasing egg production (O'Hanley & Tomberlin 2005). They can also impact on stream habitat, flow, temperature and sediment regimes (Bain & Stevenson 1999; Larinier 2001; Hart *et al.* 2002). The modification of flow caused by such weirs can also alter the structure of communities and function of river ecosystems (Baumgartner 2007), preventing gravel recruitment along the river, leading to a reduction in quality and extension of downstream gravel spawning areas (Kondolf 2000, 2001). There is a requirement through the WFD to identify the impact of structural barriers on aquatic ecosystems and to implement strategies of mitigation. Although the impacts that larger barriers have on river fragmentation and fish movement are well known, they are complicated and costly to mitigate. Consequently, it could be beneficial, and cheaper, to include the removal of smaller barriers as a mitigation measure. Although smaller barriers may have less of an effect on habitat diversity and fish movement, there is still proof that their removal alleviates pressure on fish movement (Garcia de Leaniz 2008). Weir removal can, however, have physical, biological and social implications that need to be taken into account in the planning process (Doyle *et al.* 2000; Heinz Center 2002; Hart *et al.* 2002). These include:

- stability of riparian margins;
- sediment and gravel transport;
- flood risk;
- potential transport of toxic sediments;
- reduction of stream width upstream of the weir;
- other changes in the river channel;
- social and cultural issues.

Not all of the above issues will occur for every small weir removal as may be the case for large scale weir removal but the possibilities should always be kept in mind. Stability of riparian margins and flood risk are implications that will be dramatically reduced in the case of a small weir removal, but sediment and gravel transport, the potential remobilisation of toxic sediments, reduction of stream width or the lowering of the water column and social and culture issues may all occur to some degree. Sediment and gravel built up behind the weir will be transported downstream after its removal, but to what extent this will impact on downstream habitat will depend on the composition and levels of fine sediment trapped behind a weir prior to its removal (Roni *et al*. 2008). Fine sediments can cause serious degradation of spawning habitat and fine sediment contribution greater than 15-30 percent of total substrate volume will be detrimental to the survival of eggs and embryos of salmonid species (Bunn & Arthington 2002). There are several studies that have investigated the impact of small barriers, but they are mainly concerned with fish migrations (Lucas & Frear 1997; Ovidio & Phillipart 2002) or population isolation (Morita & Yokota 2002; Meldgaard *et al*. 2003). Some studies have found that weir removal has had little effect on rivers and in some cases they have returned to pre-impoundment conditions (Ashley *et al*. 2006; Velinsky *et al*. 2006). However, not all weirs are likely to respond in the same manner (Levin & Tolimieri 2001), in particular small weir removals. Studies assessing the impact of small obstacles on fish community structure are few and mainly in France and the United States (Cumming 2004; Gillette *et al*. 2005; Poulet 2007). Small scale weir removals are common practise and are more abundant than larger weir removals but are typically not monitored (Purcell *et al*. 2002). For this reason, the case study will assess the effects of a small weir removal on fish population structure in the River Dove, UK to improve current knowledge that will assist towards future management of small weir removals.

While the theory, and to some extent the scientific knowledge, of ecological rehabilitation has developed rapidly over the past 20 years (Higgs 2003; van Andel & Aronson 2006; Falk *et al*. 2006), there is still a need to ensure that it can be put into practise, and this can be done by continually underpinning rehabilitation with science (Hobbs 2007). Current scientific understanding of river rehabilitation is generally poor (Vaughan *et al*. 2009), many uncertainties still remain and there is limited understanding of how river systems and catchments respond to rehabilitation (Szaro *et al*. 1998; Downs & Kondolf 2002; Gillilan *et al.* 2005; Jansson *et al*. 2005). Many past and recent papers have highlighted a lack of information on the success of rehabilitation projects and consequently, there are many calls for further research through monitoring and evaluation to improve knowledge in this area (Tarzwell 1937; Reeves *et al*. 1991; Brookes & Shields 1996; Ward *et al*. 2001; Downs & Kondolf 2002; Roni *et al*. 2002; Bernhardt *et al.* 2005; Roni *et al*. 2008). Although there is a steady increase of rehabilitation projects each year, the absence of adequate monitoring and evaluation limits the measure of project success (Possingham 2012). As a consequence there are a number of constraints that need to be understood and overcome, to enable monitoring design theory to be put in to practise for assessing river rehabilitation. The following case studies will attempt to identify what constrains rehabilitation project design and implementation by investigating the outcomes of a range of relatively small scale projects on fish communities and habitat. The order of case studies are discussed as each increases in complexity:

3.3 Instream remediation of Driffield Beck;

3.4 Channel narrowing of Lowthorpe Beck;

3.5 Brash revetment to prevent bank erosion on the River Manifold;

3.6 Small weir removal on the River Dove;

3.7 Artificial riffle reinstatement on the River Stiffkey.

The aim of case studies is to introduce 'real life' river rehabilitation scenarios to identify 'real life' limitations to river rehabilitation monitoring design and to recognise how to overcome these issues, whilst concurrently increasing scientific understanding of river rehabilitation. Most rehabilitation projects are completed within the year that they are proposed, especially if they are small scale projects and therefore this tends to limit any base line data to 1 year pre-rehabilitation work, demonstrated by the case studies presented in this chapter. The monitoring design for each case study varied depending on the number of years fisheries data were collected/available for control and impact sites, pre and post rehabilitation. Driffield Beck and Lowthorpe Beck were both limited because suitable control sites were not identified and therefore, before and after data could only be considered for a monitoring design (Table 3.1). Unfortunately, because trout numbers were either low or not present, a BA monitoring design could not be performed. Both the Manifold and Stiffkey case studies were already in progress and no suitable fisheries baseline data were available (Table 3.1). Therefore, a post-treatment monitoring design was performed, with 2 years' fisheries data collected at the rehabilitated and control sites. The river Dove case study was the only study where a full BACI monitoring design could be performed because fisheries data were collected before and after rehabilitation and the impact and control sites (Table 3.1).

Stocking is present in all river systems, with the exception of the River Stiffkey, and was a further constraint when attempting to assess project success. This was only established half way through the project and therefore, could not be overcome. However, it is an important factor to consider regarding rehabilitation of trout streams, especially as it can hinder identification of rehabilitation success because it affects the ability to detect change in wild populations. As a result, habitat monitoring plays a key role when identifying rehabilitation change.

*Table 3.1. Case study monitoring design (*N/A monitoring design cannot be applied where there is insufficient years on monitoring or limited number of fish caught)*

Case study	Control/ Impact	Before	After		Site number	Monitoring design
		1 year	1 year	2 year		
Driffield	IMPACT				2	N/A*
	CONTROL					
Lowthorpe	IMPACT				1	N/A*
	CONTROL				2	
Manifold	IMPACT				4	Post-treatment
	CONTROL				6	
Dove	IMPACT				4	BACI
	CONTROL				6	
Stiffkey	IMPACT				3	Post-treatment
	CONTROL				2	

Specific objectives are:

- To determine if a variety of river rehabilitation techniques improve habitat for fish, specifically brown trout;
- To determine if river rehabilitation success can be determined through a variety of monitoring designs and impact assessments.

3.2. Materials and methods

River rehabilitation occurs to increase the health status of rivers and because of this it is important to monitor the outcome of these schemes to identify their success from a biological and morphological approach. Fish were the main biological focus, specifically brown trout, in addition to instream habitat variables, to endeavour to identify river rehabilitation success for the five case studies previously mentioned. Constraints in rehabilitation monitoring design are also of interest to recognize why the majority of rehabilitation case studies do not report their success levels. The material and methods subsequently discussed include all generic materials and methods applied throughout this chapter; however, because there were site specific issues with data availability (Table 3.1) methods applied differ between case studies.

3.2.1. Fisheries and habitat survey methods

Quantitative survey sites (estimates of absolute abundance based on a three-catch removal method (see Zippin (1956) and Carle and Strub (1979)) were isolated by upstream and downstream stop-nets to ensure no escape from, or migration into, the sample area to allow an estimate of numbers of fish are present to be derived. Semi-quantitative surveys (involving a single fishing) used natural obstacles, such as small waterfalls, cascades and shallow riffles, to act as barriers to fish movements out of the survey area; population estimates were subsequently derived from calibration of the fishing efficiency. The quantitative and semi-quantitative electric fishing strategies involved three operatives (one anode operator and two netsmen) fishing in an upstream direction, with a fourth operator on the bank supervising safe operation of the electric fishing equipment. A 2kVA generator with an Electracatch control box producing a 220 V PDC output was employed. Fish were caught in dip nets by operatives positioned either side of the anode operator during each fishing exercise; the process was repeated for each run of the three-catch removal method with catches kept separate for data collection. Following each survey, fish were identified to species, measured (fork length, mm) and a sample of scales removed for ageing purposes (using the appropriate Environment Agency Management System (Britton 2003)), before returning the fish live to the river. After each electric fishing survey, habitat data were collected at each site and recorded on standard HABSCORE forms used by the Environment Agency. HABSCORE is a system for measuring and evaluating stream salmonid habitat features based on empirical statistical models, the abundance of 0+ and >1+ brown trout were estimated for each site to relate fisheries data to habitat quality score (Wyatt, Barnard & Lacey 1995) (see Section 3.2.3).

3.2.2. Data analysis

Several methods of analysis were selected in an attempt to identify a change in fish and habitat data resulting from rehabilitation actions, their methods and justifications are as follows:

Density estimates and abundance categories

Density estimates were used to assess the status of the focal species brown trout fish populations according to the matrix procedure adopted by the Environment Agency Fisheries Classification Scheme (EA-FCS; Table 3.2). The EA-FCS was developed to allow comparison of juvenile salmonid monitoring data with a juvenile database derived from over 600 survey sites in England and Wales (Mainstone *et al.* 1994). The classification of salmonid populations is based on a grading scale (A–F) and provides an indication of the status of salmonid populations in study rivers. The EA-FCS grading scheme is translated as follows: Grade A (excellent), Grade B (good), Grade C (fair or average), Grade D (fair/poor), Grade E (poor) and Grade F (fishless). The population density grades for the EA-FCS are detailed in Table 3.2. This grading system enabled the comparison of river status before and after river rehabilitation, to assess rehabilitation success using brown trout density as an indication.

Estimates of abundance of 0+ and ≥1+ brown trout were derived by the three-catch removal method, specifically the Maximum Likelihood Method (Carle & Strub 1978). In all cases the population density at each site was expressed as numbers/100m^2. The efficiency of sampling effort or probability of capture (P) was calculated from the Maximum Likelihood Methods and was used to calibrate the survey gear. Density estimates of 0+ and ≥1+ fish/100m^2 at semi-quantitative sites were derived from the method of gear calibration. This uses the probability of capture (P) derived from quantitative survey sites to derive relative density (N/100m^2) as: $N = ((C / P) / A)*100$, where *C* is the total number of fish caught in the single run and *A* is the sampling area (Cowx 1996). Density estimates for brown trout were compared between sites from the rivers sampled and used in the derivation of HABSCORE outputs.

Table 3.2. Salmonid abundance (N/100m^2) classifications used in the Environment Agency Fisheries Classification Scheme (EA-FCS).

	Abundance classification					
Species group	A	B	C	D	E	F
0+ brown trout	≥38.0	17.0-37.9	8.0-16.9	3.0-7.9	0.1-2.9	0
≥1+ brown trout	≥21.0	12.0-20.9	5.0-11.9	2.0-4.9	0.1-1.9	0
0+ salmon	≥86.0	45.0-85.9	23.0-44.9	9.0-22.9	0.1-8.9	0
≥1+ salmon	≥19.0	10.0-18.9	5.0-9.9	3.0-4.9	0.1-2.9	0

Derivation of density estimates of other species was not possible as catches in the second and third runs were often greater than the first run, which contradicts one of the main assumptions of depletion sampling that the population is reduced on each sampling run. For example, depletion sampling for bullhead (*Cottus gobio* L.) needs to be species specific and intensive because of their behaviour, causing them to become immobilized *in situ* underneath stones and making them difficult to detect by survey operators and thus the recommendation for species-targeted surveys (Cowx & Harvey 2003). FCS2 would be the preferred method to use within this study, but it is currently not possible for use outside the Environment Agency.

European Fish Index

The European Fish Index (EFI+) is a multi-metric index based on a predictive model that derives reference conditions from abiotic environmental characteristic of individual sites and quantifies the derivation between the predicted fish assemblage. The EFI+ database is based on approximately 30,000 fish assemblage surveys covering more than 14,000 sites from 2,700 rivers in 15 European eco-regions. It includes information about fish assemblage, environmental characteristics and human pressures. The database also includes a comprehensive list of European freshwater fish species assigned to functional guilds according to their ecological characteristics. All of this information was used to calculate metrics for the EFI+ index & classes (Table 3.3). The purpose of the index is to evaluate the ecological status of sites at the European scale and because of this the index is comparable between eco-regions, river types and different local environments (EFI+ CONSORTIUM 2009).

The EFI+ method requires three types of essential data (Full input data & categories can be found in Appendix 3):

1) Data from single-pass electric fishing catches to calculate the assessment metrics. Individuals from all species have to be measures (total length in mm) to compute the observed values of metric. Results should be recorded as number of individuals caught per species, including the numbers in two size classes determined by a 150 mm threshold.
2) Data describing environmental conditions at the site scale or at the river segment scale as well as the sampling method.
3) Data describing fishing method.

The EFI+ indicies and class boundaries for salmonids can be found in Table 3.3. The indicies vary between 0 and 1. For each metric an undisturbed site would have an index value close to 0.80, and a highly disturbed site a value lower than the 25% quantile of the index distribution for undisturbed sites (EFI+ CONSORTIUM 2009). A grading system is used in conjunction to the index to classify a river as undisturbed to disturbed (Class 1-5; Table 3.3) .

Table 3.3. European ecological class boundaries for salmonid rivers.

	Salmonid index
Class 1	0.911-1
Class 2	0.755-0.911
Class 3	0.503-0.755
Class 4	0.252-0.503
Class 5	0-0.218

Limitation of the EFI+

Several restrictions are present with the EFI+ that can limit its application (EFI+ CONSORTIUM 2009):

- Sampling location
- Environment
- Sampling method applied
- Low species richness
- Number of fish caught

The index cannot be applied if the sampling location has been undertaken in lateral water bodies, if upstream lakes are present at the sample location or if environmental condition represent winter dry period. EFI+ should also be used with caution when the sampling method applied was by boat. These limitations were absent from all sample sites present within this thesis, however, low species richness and number of fish caught were limitations that were found in some of the sample sites within this thesis. The fish index is unsuitable if species richness is limited to one species, in most cases this relates to European head waters where brown trout are the only fish present. Therefore, the only case where such species composition based metric could react is when the response to a disturbance is an increase of species richness, for example, river rehabilitation. Also, limitations in the use of the index arise when few specimens are caught, the software still allows the calculation of the indicies, but the results must be considered with caution. These limitations are considered in the relevant chapters.

Length distributions

Length distributions were constructed for brown trout where sufficient numbers of fish were caught, and these were compared between sites in the rivers sampled. The methodology involved assigning fish lengths of a particular species into 10-mm size classes to determine the total number of fish in each age size class. 10-mm size class is sufficient to see clear definition between the different age groups. Length distributions, supported by ageing of scales from selected length groups, were used to separate 0+ fish from older age groups. Scales from each fish were examined under a microfiche projector and the fish aged by counting the number of annuli, taking care to note any false checks. More than one scale was examined to ensure correct interpretation of the annuli and a quality control procedure of 20 random fish was used to ensure accuracy. The total scale radius and scale radius to each annuli were measured from the nucleus to the scale edge.

3.2.3. HABSCORE data collection and outputs

HABSCORE is a system for measuring and evaluating stream salmonid habitat features based on empirical statistical models relating the population size of five (0+ salmon, >0+ salmon, 0+ trout, >0+ trout (<20cm), >0+ trout (>20cm)) salmonid species/age combinations (Wyatt, Barnard & Lacey 1995). Using the information from three HABSCORE questionnaires, the software produces a series of outputs, which includes estimates of the expected populations (the Habitat Quality Score, HQS) and the degree of habitat utilisation (the Habitat Utilisation Index, HUI), for each of the salmonid species/age combinations (Wyatt, Barnard & Lacey 1995).

To collect information for HABSCORE analysis a questionnaire on the habitat found at each site was completed following each fisheries survey. The methodology of habitat data collection and completion of the relevant form (HABform), along with completion of river catchment information (MAPform) and fisheries information (FISHform) are documented by Barnard & Wyatt (1995).

Data from the three completed forms (HABform, MAPform and FISHform) at each site were entered into the HABSCORE for Windows program and the outputs described below were produced for trout populations (definitions from Wyatt, Barnard & Lacey (1995)). Note, HABSCORE uses density estimates as the input variable, thus it is possible to use the densities derived from the calibrated gear method to calculate the HUI and HQS scores thus increasing the HABSCORE coverage.

Habitat Quality Score (HQS)

The HQS value is a measure of the habitat quality expressed as the expected long-term mean density of fish (in numbers per $100m^2$). The HQS is derived from habitat and catchment features, and assumes that neither water quality nor recruitment are limiting the populations. The HQS is used as an indicator of the potential of the site, against which the observed size of populations may be compared.

HQS lower and upper confidence limits

These are the lower and upper 90% confidence limits for the HQS, in numbers/$100m^2$. The confidence limits given should enclose the average observed density for a site on 90% of occasions. The probability of getting an observed average density lower than the lower confidence limit by chance alone is therefore 5%.

Habitat Utilisation Index (HUI)

The HUI is a measure of the extent to which the habitat is utilised by salmonids. It is based on the difference between the 'observed' density and that which would be expected under 'pristine' conditions (i.e. the HQS). When the 'observed' density and the HQS are identical, the HUI takes the value of one. HUI values less than one will occur when the observed densities are less than expected and HUI values greater than one will occur when observed densities are higher than expected.

HUI lower and upper confidence limits

These are the upper and lower 90% confidence limits for the HUI, expressed as a proportion. An upper HUI confidence interval <1 indicates that the observed population was significantly less than would be expected under pristine conditions. Conversely, a lower HUI confidence interval >1 indicates that the observed population was significantly higher than would normally be expected under pristine conditions.

3.2.4. Impact assessment

Environmental monitoring is required when the objectives of a monitoring programme require the actual effects to be determined. In terms of an impact assessment, the key objective is to detect a change in a given variable, such as fish populations. Monitoring a given variable allows detailed scientific information to be gained and provides not only information about the current status of the environment but also feedback about the actual environmental impacts of a project, such as river rehabilitation. The following impact assessment was applied to detect differences in 0+ and >1+ brown trout fish densities that resulted from an impact (rehabilitation) and this was achieved by comparing the mean difference to determine the overall significance of the differences (Sedgwick 2006). Fisheries data were tested for normality (Kolmogorov-Smirnov test, Skewness and Kurtosis) and homogeneity of variance (Levene test), as a consequence of a small data set there was a positive skew and variances were significantly different, therefore fish density data were pre-treated by using $\log_n$ transformation (ln+1) before further analysis.

BACI Design:

1. Mean density was calculated for 0+ and ≥1+ age groups of trout for all case study sites.

2. The target variance* (a change of 5 fish/100m²) for 0+ and ≥1+ age groups of trout was calculated:

Equation 3.1 *(mean density of 5 fish/100m² before/(Ø*SQRT(2))^2*

Ø is a given value relating to the associated degrees of freedom determined by: (number of control sites + number of impact sites) - 2 .

**The preliminary decision is biological and involves estimating the magnitude of change in fish abundance that can have ecological and management relevance. A change of 5 fish/100m² was chosen to detect a change in population density between the Environment Agency's FCS grading.*

3. The actual variance (Vx (Sedgwick, 2006); Equation 3.2) of the full BACI quadrant for 0+ and ≥1+ age groups of trout was calculated.

Equation 3.2 *V(x) = (Vytr)*(1/(mB*nT)+1/(mA*nT)+1/(mB*nC)+1/(mAnC))*

Vytr = Residual variance (Error Mean Square (EMS) of a two factor ANOVA without replication)

mA = No of occasions after the event (years)

mB = No of occasions before the event (years)

nT = No of test (i.e. impact) sites

nC = No of control (i.e. control) sites

4. The actual variance was compared to the target variance to identify if there were sufficient data to allow a significant impact to be detected.

5. If the actual variance was greater than the target variance a statistically significant change of 5 fish could not be identified. In this instance a resource calculation was performed, i.e. the number of years and sites in Equation 3.2 was increased to establish how many sites and years of data would be required to derive statistically robust outputs.

6. If the actual variance was less than the target variance a statistically significant change of 5 fish **could be** identified and the impact assessment was performed (Equation 3.3). The impact is calculated from the differences in mean abundance derived from the BACI design. This is defined as:

(Change in impact (or test) area) – (Change in control area).

Equation 3.3 *Impact (**x**) = (y AT - y BT) – (y AC - y BC)*

y BT is the mean abundance, over all sites and times, **before** the event, in the **test** area.

y AT is the mean abundance, over all sites and times, **after** the event, in the **test** area.

y BC is the mean abundance, over all sites and times, **before** the event, in the control area.

y AC is the mean abundance, over all sites and times, **after** the event, in the control area.

Post-treatment Design:

Where fisheries data is only present post-rehabilitation.

Stages 1 & 2 above are followed.

1. The variance is then calculated between control and impact sites (*V(Z)* (Sedgwick, 2006) Equation 3.4) of a post-treatment design for 0+ and >1+ age groups of trout:

Equation 3.4

$$V(Z) = \frac{VysI + VrI/m}{nI} + \frac{VysC + VrC/m}{nC}$$

Where:

$VysI$ = Spatial variance of impact site

$VysC$ = Spatial variance of control site

VrI = Measurement error variance of impact site*

VrC = Measurement error variance of control site*

m = Number of years

nI = Number of impact sites

nC = Number of control sites

**Error Mean Square (EMS) of a two factor ANOVA without replication)*

2. The actual variance was compared to the target variance to identify if there were sufficient data to allow a significant impact to be detected.

3. If the actual variance was greater than the target variance a statistically significant change of 5 fish **could not be** identified. In this instance a resource calculation was performed, i.e. the number of years and sites in Equation 3.4 was increased to establish how many sites and years of data would be required to derive statistically robust outputs.

4. If the actual variance was less than the target variance a statistically significant change of 5 fish **could be** identified and the impact assessment was performed (Equation 3.5). The impact is calculated from the differences in mean abundance derived from the control and impact site. This is defined as:

Equation 3.5 x = *(impact area) – (control area).*

i.e. Impact (**x**) = yI – yC

y I is the mean abundance, for all impact sites after the event

yC is the mean abundance, for all control sites after the event

t-test

An independent t-test was then performed to determine the significance of difference for 0+ and >1+ brown trout density between samples (e.g before, after, control & impact) and determined whether a significant difference occurred after rehabilitation (an increase or decrease in brown tout density), by calculating confidence limits of the change (p<0.05) and therefore, expressing more clearly the outcome of the analysis (Sedgwick 2006).

Equation 3.6 $$t = \frac{x}{SEofdifference}$$ *for nI + nC -2df*

x = yI – yC

SE = standard error of differences

nI = number of impact sites

nC = number of control sites

Effect size was then calculated from the t-statistic as a guideline to assess how meaningful the effect is:

Equation 3.7 $$r = \sqrt{\frac{t^2}{t^2 + df}}$$

Where:

r = 0.10 (small effect)

r = 0.30 (medium effect)

r = 0.50 (large effect)

3.3. Casestudy introduction

The following 5 case studies case studies are investigate in an attempt to identifying limitations in project success. Each case study deals with a number of pressures that impact of the ecological grading of the river (Table 3.4). Lowthorpe Beck and the River Stiffkey are categorised as HMWB through the WFDs RBMP, therefore, these rivers need only reach GEP, while it is expected that Driffield Beck and the Rivers Manifold & Dove reach GES (Table 3.4). The case study measures (Table3.4) attempt to overcome pressures at a local scale, with the exception of the River Stiffkey, where gravel augmentation is introduced along the whole stretch of river.

Table 3.4. WFD classification of case study rivers

River	HMWB	Overall status	Target status	Land use	Pressure	Case study measures
Driffield	No	Poor	GES - 2015	Urbanisation	Morphological alteration: - riparian vegetation - channelization - instream habitat	In-stream habitat works: - bed re-profile - instream vegetation
Lowthorpe	Yes	Poor	GEP - 2027	Agriculture	Morphological alteration: - sedimentation - riparian vegetation - instream habitat	Morphology planform: - narrowing of channel
Manifold	No	Good	GES - 2015	Agriculture	River fragmentation: - barrier to upstream & downstream migration Morphological alteration: - erosion - sedimentation	River continuity: - removal of weir Reduce erosion & sedimentation: - brash revetment
Dove	No	Good	GES - 2015	Agriculture	Fragmentation: - barrier to upstream & downstream migration	River continuity: - removal of weir
Stiffkey	Yes	Poor	GEP - 2027	Agriculture	Flood protection: - restriction of lateral migration Morphological alteration: - riparian vegetation - sedimentation	Instream habitat works: - gravel augmentation

3.4. Instream remediation of Driffield Beck

3.4.1. Introduction

Driffield Beck is a meandering chalk stream located in the River Hull basin, East Riding of Yorkshire and is fed by many small chalk streams (Figure 3.1). Two sections, adjacent to each other were surveyed: Water Forlones and Cattle Market (Table 3.4). Both are located in Driffield town centre where the main pressure is a loss of instream habitat diversity resulting from the straightening and narrowing of the beck with concreted walls and a compacted gravel river bed (Figure 3.2a & 3.3a). There is also no instream or bank side vegetation. Driffield Beck is categorised as 'poor' status through the WFD water body classification and has a target of GES by the year 2015 (Table 3.4), as a result The East Yorkshire Chalk Rivers Trust (EYCRT) along with the Environment Agency and Yorkshire Wildlife Trust sought to resolve these problems by increasing habitat diversity within the beck through small instream habitat remedial measures. The compacted gravel bed was raked and substrate loosened at both sites, large bits of concrete and bricks were removed and the remaining gravel was positioned to create a meandering stream. The Beck is too shallow to create a pool-riffle system. Pre-planted coir mattresses were secured to the river bed to reinforce the new meandering profile and to also provide cover and habitat for fish and other aquatic organisms in Driffield Beck (Figures 3.2b & 3.3b). All of these measures were put in place to overcome the pressures acting on Driffield Beck.

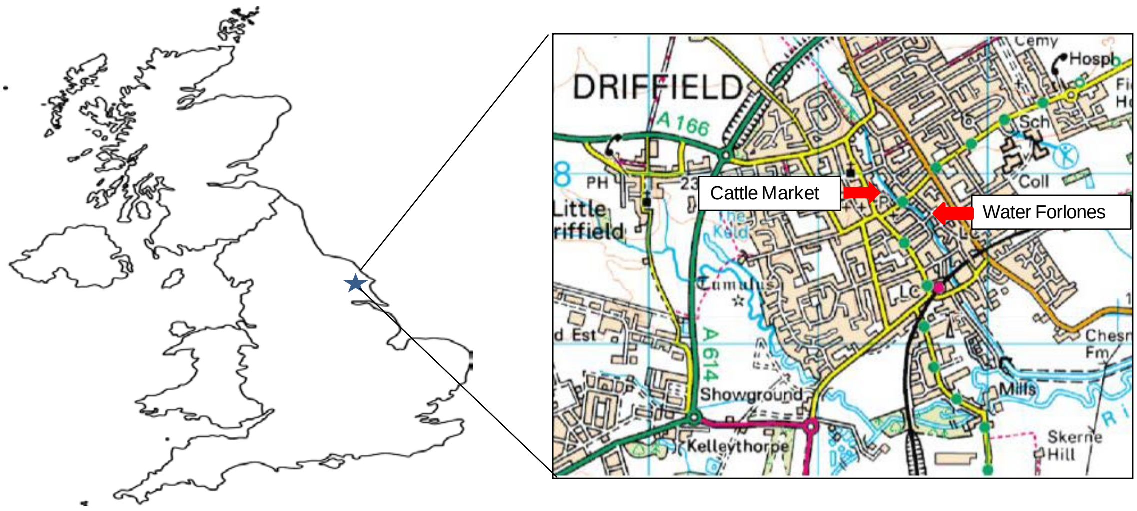


Figure 3.1. Location of instream habitat remediation at Driffield Beck (Source: Ordnance Survey).

Figure 3.2. Water Forlones at Driffield Beck a) before instream habitat remediation and b) post instream habitat rehabilitation.

Figure 3.3. Cattle Market at Driffield Beck a) before instream habitat remediation and b) post instream habitat rehabilitation.

Aims & objectives

The aim of this study was to determine if instream habitat remediation is an appropriate technique to reintroduce habitat diversity to a constrained channelized river that cannot be morphologically modified. Furthermore, this study determines if river rehabilitation success can be determined through application of a basic monitoring design.

Specific objectives were to compare suitable brown trout habitat availability and brown trout density and age structure prior to and following instream remedial action within the study reach. Fish community structure was also evaluated before and after stream remedial action. Furthermore, to test and establish a monitoring programme that will help assess the direct effect of rehabilitation schemes on fish communities.

3.4.2. Methods

Fish and habitat surveys

Fisheries surveys were carried out at the study site on 19 July 2010 (prior to instream habitat maintenance works) and 12 July 2011, post rehabilitation. Control sites were not surveyed on the Beck due to its limited size and restricted access, furthermore, Environment Agency annual fisheries data could not be used as suitable controls as other representative rivers were not surveyed annually (Table 3.5). The focus of the study was primarily on brown trout, but bullhead were also present in the Beck. After each electric fishing survey, habitat and environmental data were collected at each site in the format used for HABSCORE (Section 3.2.3).

Table 3.5. Fisheries survey site details in Driffield Beck July 2009 and July 2010

Site identifier/NGR	River Name	Survey date	Length/mean width/area	Survey method and gear
Water Forlones TA02745782	Driffield Beck – Town Centre	19/07/10	50 m/5.3 m 265 m2	Generator. Quantitative.
Water Forlones TA02745782	Driffield Beck – Town Centre	12/07/11	49 m/3.1 m 151.9 m2	Generator. Quantitative.
Cattle Market TA02745785	Driffield Beck – Town Centre	19/07/10	50 m/4.2 m 210 m2	Generator. Quantitative.
Cattle Market TA02745785	Driffield Beck – Town Centre	12/07/11	50 m/2.6 m 130 m2	Generator. Quantitative.

3.4.3. Results

HABSCORE analysis

Raw depth data from HABSCORE was used to quantify some of the changes in the channel before and after river rehabilitation (Figure 3.4 & 3.5), the habitat quality score (HQS Table 3.8) taken from the HABSCORE output, can also be used to quantify habitat improvement for brown trout at Driffield Beck. At Water Forlones there were only slight variations in depths when comparing 2010 to 2011, depths did not exceed 20cm. The HQS increased after habitat rehabilitation work for 0+, but decreased for >1+ (<20cm) and >1+ (>20cm) (Table 3.8), most probably because suitable depths (>20cm) are missing from the stretch. At Cattle Market there were only slight variations in depths when comparing 2010 to 2011, depths did not exceed 20cm in 2010, whereas habitat rehabilitation increased depths to >20cm in some areas. After habitat improvement works, the HQS increased for 0+, >1+ (<20cm) and >1+ (>20cm) (Table 3.8), most probably because depths over >20cm were present which are more suitable for larger fish.

HABSCORE outputs for the two sites on Driffield Beck revealed variations in the observed densities, predicted densities and habitat utilisation by trout (Table 3.5). Observed densities of 0+ trout at Driffield Beck in 2010 were lower than predicted by the Habitat Quality Score (HQS) at Water Forlones and Cattle Market, suggesting poorer populations than expected, however, HUI upper CLs were only <1 in 2011 at both sites and therefore only significantly lower in the post works survey (Table 3.5).

Observed densities of >1+ trout (<20 cm) were lower than predicted by the HQS at Water Forlones and Cattle Market in both 2010 and 2011, but they were only significantly lower at Cattle Market in 2011 (Table 3.5).

Observed densities of >1+ trout (>20 cm) in 2010 were lower than predicted by the HQS at Water Forlones and Cattle Market in 2010 and 2011, indicating poorer populations than expected but the HUI upper CLs were >1, suggesting populations at this site were not significantly lower (Table 3.5).

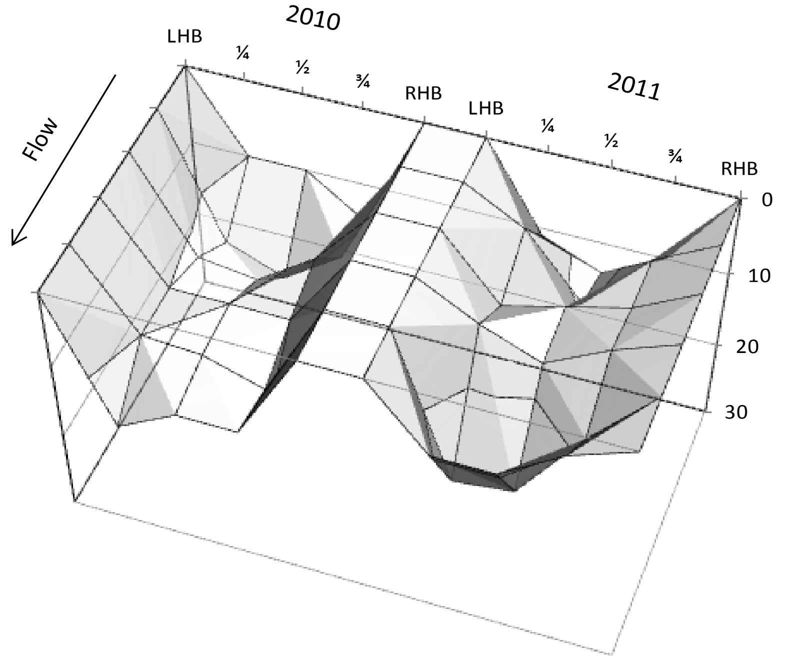


Figure 3.4. Changes in channel depth recorded during HABSCORE at Water Forlones.

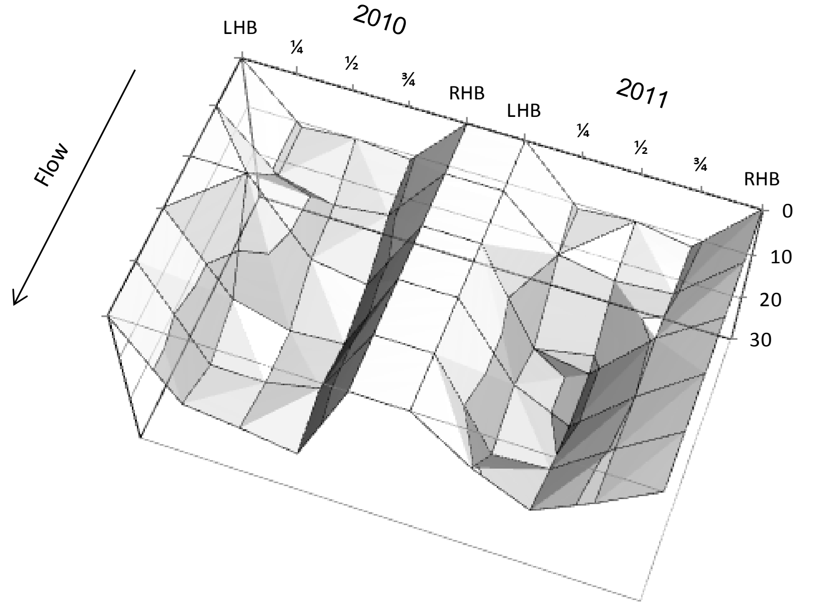


Figure 3.5. Changes in channel depth recorded during HABSCORE at Cattle Market.

Status of fish population on Driffield Beck

July 2010 Driffield Beck fisheries survey found similar species composition between both sites, with brown trout and bullhead the only species caught (Table 3.6). In July 2011, species composition was also similar between sites with brown trout and bullhead, but 3-spined stickleback (*Gasterosteus aculeatus* L.) were also present (Table 3.3). The number of brown trout caught was noticeably fewer in 2011 than 2010, but numbers of bullhead were noticeably greater in 2011 than 2010 (Table 3.6). 3-spined stickleback were only present in 2011 with considerable numbers caught (Table 3.6).

Table 3.6. Number of fish of different species captured at two sites at Driffield Beck in July 2010 & 2011.

Species	Water Forlones		Cattle Market	
	2010	2011	2010	2011
Brown trout	11	0	12	7
Bullhead	2	63	6	54
3 Spined stickleback	0	12	0	33

Density estimates

In 2010 0+ trout dominated catches at Water Forlones and Cattle Market as there were no >1+ trout caught, as a result >1+ trout were classified as Class F (fishless) whilst 0+ trout were Class D (fair) at both sites (Table 3.7). In 2011 0+ trout dominated catches at Water Forlones and Cattle Market as there were no >1+ trout caught, as a result >1+ trout were classified as Class F (fishless) whilst 0+ trout were Class F (fishless) at Water Forlones and Class E (poor) at Cattle Market (Table 3.7).

Table 3.7. Total population estimate (N), population density (D) (± 95% C.L.) and abundance classification of trout derived from fisheries surveys on Driffield Beck in July 2010 & 2011 (density of fish given as numbers per 100m^2). Details of derivation of estimates are provided in the text.

Date	Site	Total Population		Population Density		Abundance Class	
		0+	≥1+	0+	≥1+	0+	≥1+
2010	Water forlones	12±4	0	4.53±6.21	0	D	F
	Cattle market	12±1	0	5.71±0.63	0	D	F
2011	Water forlones	0	0	0	0	F	F
	Cattle market	1±0	0	0.77±0	0	E	F

Length frequency distribution

In 2010, 0+ brown trout captured at Water Forlones were in the size range 71-87 mm while no ≥1+ individuals were captured (Table 3.8). Brown trout were not captured at

Water Forlones in 2011 fisheries surveys (Table 3.8). In 2010 0+ brown trout captured at Cattle Market were in the size range 67-79 mm while no >1+ trout were captured. In the 2011 only one 65-mm 0+ trout was caught at Cattle Market; again no >1+ trout were present.

3.4.4. Discussion

In stream habitat remediation works were undertaken at two sites on Driffield Beck to improve habitat diversity within a restricted channel in an urban setting. **One of the main aims of this study was to assess if habitat remediation works to re-meander a constrained channel can be used as an appropriate technique to introduce suitable habitat back in to Driffield Beck, specifically for fish**. In addition, from a qualitative view the instream habitat remediation works added only a small amount of diversity back into both the Cattle Market and Water Forlones sections on Driffield Beck. The bed substrate vastly improved from being flat and embedded, to a mobile gravel bed. The pre-planted coir mattresses supported the meandering profile created from the instream habitat works and also provided a small amount of cover for fish and showed signs of supporting an invertebrate community. The flow was low at both sites when sampled in 2011 so it was difficult to see how the in stream habitat works would respond to higher flows, and it remains too early to determine if the instream habitat works will be successful long term. Overall the HQS (density) showed that the instream habitat works improved habitat for 0+ trout at both sites between pre and post works (Table 3.5), but the fisheries did not respond to occupy this increased habitat availability. This suggests that brown trout recruitment was limited even though 0+ trout were present at all sites with the exception of Water Forlones in 2011. This would usually indicate recruitment into the population, however, the absence of >1+ trout at all of the sites in 2010 and 2011 suggests poor survival rate to older age groups and that brown trout present were possibly stocked. Further investigation into the Environment Agency's stocking data indicates that most of trout caught were almost certainly stocked because stocking of 0+ trout has taken place several times a year, for many years on Driffield Beck. For example, 3380 brown trout were stocked in 2009, 1450 in 2010 and 2775 in 2011. Thus it is likely the 0+ individuals caught in the surveys were stocked and are merely surviving and not thriving. There are presumably additional pressures on Driffield Beck that hinder trout from having a self-sustaining population and stocking should not be seen as a measure to overcome these problems (Cowx & Welcomme 1998). Rather the additional pressures on the Driffield system need to be identified and addressed to rehabilitate a self-recruiting trout population. In particular, attention needs to be directed to siltation problems of potential spawning gravels, improved access to spawning areas and water quality issue that may be impacting survival and reproduction.

Table 3.8. HABSCORE outputs for 0+, >1+ (<20 cm) and >1+ (>20 cm) across all sites at Driffield Beck for 2010 and 2011 catches. (Note: Shaded area represents sites where the observed population was significantly higher (HUI lower CL column) or lower (HUI upper CL column) than would be expected under pristine conditions)

Site Code	Age category	Observed number	Observed density	HQS (density)	HQS lower CL	HQS upper CL	HUI	HUI lower CL	HUI upper CL
Water Forlones	0+								
	2010	12	4.40	21.84	5.53	86.2	0.20	0.03	1.40
	2011	0	0	49.56	12.72	193.08	0.01	0.00	0.09
	>1+ (<20 cm)								
	2010	0	0	1.84	0.42	7.96	0.20	0.03	1.21
	2011	0	0	0.29	0.30	2.81	0.40	0.13	1.21
	>1+ (>20 cm)								
	2010	0	0	3.42	0.80	14.66	0.20	0.03	1.18
	2011	0	0	1.48	0.48	4.54	0.45	0.15	1.39
Cattle market	0+								
	2010	12	5.59	26.92	6.89	105.25	0.21	0.02	1.74
	2011	1	0.76	55.46	14.50	212.11	0.01	0.00	0.09
	>1+ (<20 cm)								
	2010	0	0	2.28	0.53	9.81	0.20	0.03	1.23
	2011	0	0	5.56	1.31	23.55	0.14	0.02	0.82
	>1+ (>20 cm)								
	2010	0	0	1.16	0.39	3.51	0.40	0.13	1.21
	2011	0	0	1.77	0.59	5.34	0.43	0.14	1.30

When comparing 2010 (pre) to 2011 (post) fisheries data at both sites, species composition increased from 2 species (brown trout and bullhead) to 3 species (brown trout, bullhead and 3-spined stickleback). In addition, brown tout numbers were lower post instream habitat works at both sites suggesting the rehabilitation had negative effects on this species, conversely, bullhead numbers were higher post instream habitat works at both sites suggesting habitat was created for this species. Nevertheless, there is inadequate monitoring to conclude with confidence that the reduction in brown trout numbers, and the presence of 3-spined stickleback and increase of bullhead numbers in 2011 were due to instream habitat works.

Furthermore, another aim of this study was to determine if river rehabilitation success can be established through application of a basic monitoring design. The surveys clearly highlighted the limitations in the monitoring programme to detect if a change had occurred resulting from the in stream habitat works at Water Forlones and Cattle Market at Driffield Beck. Both spatial and temporal monitoring is required at both sites on Driffield Beck to overcome natural variability and identify possible changes as a result of the instream habitat works by providing a basis for comparison between the rehabilitated area and the conditions before rehabilitation. Unfortunately Environment Agency fisheries data could not be used as suitable control data for this study as sites from Driffield Beck and other representative rivers were not surveyed annually. This limitation in monitoring data caused further restrictions in monitoring design criteria and therefore an impact assessment could not be performed.

Recommendations

Overall, there is little evidence from this study that suggests in stream habitat remediation works have benefited trout populations in Driffield Beck in the short term. There seems to be several underlying pressures on Driffield Beck that need to be addressed before rehabilitation measures should be considered. **It is recommended that full spatial & temporal assessment of fish populations and habitat surveys from reaches that represent the whole stretch of Driffield Beck are undertaken to get a overview of the status and pressures on the river**. This will also identify areas of Driffield Beck that have self-recruiting fish populations and suitable habitat and thus can be used as guidance when determining the most beneficial rehabilitation measures for the future.

The effectiveness of management actions cannot be evaluated without ongoing monitoring and evaluation of outcomes (Brierley *et al*. 2010). A before-after study design was considered the most appropriate for this case study but it failed to provide adequate response to detect and cause effect. This was because the surveys were replicated in time rather than space (Morrisey 1993; Roni 2005) and without spatial replication there is assumption need for extensive temporal replication that was not possible (Conquest 2000). This raises a key issue in the length of time for pre and post monitoring in scenarios where no control sites are available. **It is recommended that fish populations are monitored temporally and spatially to identify if instream habitat works benefit fish populations, such as in the Driffield Beck, through application of a before-after-control-impact assessment** to strengthen the statistical power of analysis (Smith *et al*. 1993; Roni *et al*. 2005a). Control sites should be introduced, whether from the Beck itself or representative rivers in the catchment, to overcome natural variability. However where this is not possible, as in the Driffield Beck, surveys should be carried out at a frequency both before and after sufficient to account for sampling variability.

There is an obvious lack of recruitment to the adult life stages in the brown trout populations in Driffield Beck, and this needs to be addressed. **It is therefore recommended that the habitat preferences for each life stage of the brown trout are revisited and additional 'missing' habitat(s) identify and incorporated into future rehabilitation measure**.

3.5. Channel narrowing of Lowthorpe Beck

3.5.1. Introduction

Lowthorpe Beck is a meandering chalk river situated in the River Hull basin; the upper reaches have a strong flow becoming slower in the lower reaches. Lowthorpe Beck at Harpham (Figure 3.6) experiences pressures such as over widening, which contributes to the large quantity of sediment present in the channel, this further reduces river depth and flow velocities resulting in a loss of habitat diversity. The Beck is currently classified as 'poor' status by the WFD water body classification and has a target of GEP by the year 2027 (Table 3.4). As a result The East Yorkshire Chalk Rivers Trust (EYCRT) carried out channel narrowing on as a rehabilitation measure to narrow a 60-m, over wide stretch that was heavily silted, with no marginal habitat (Figure 3.7). A berm was created on the right hand bend of the river using of faggots; the silt removed from the channel was used to fill behind the berm to help establish the new margin. Upstream of the project site was a low timber weir, this has been removed following the re-sectioning work enabling the river to return to a more natural state. Natural plant re-colonisation will be encouraged on this bank as seeds from many of the local species will be contained in the silt.

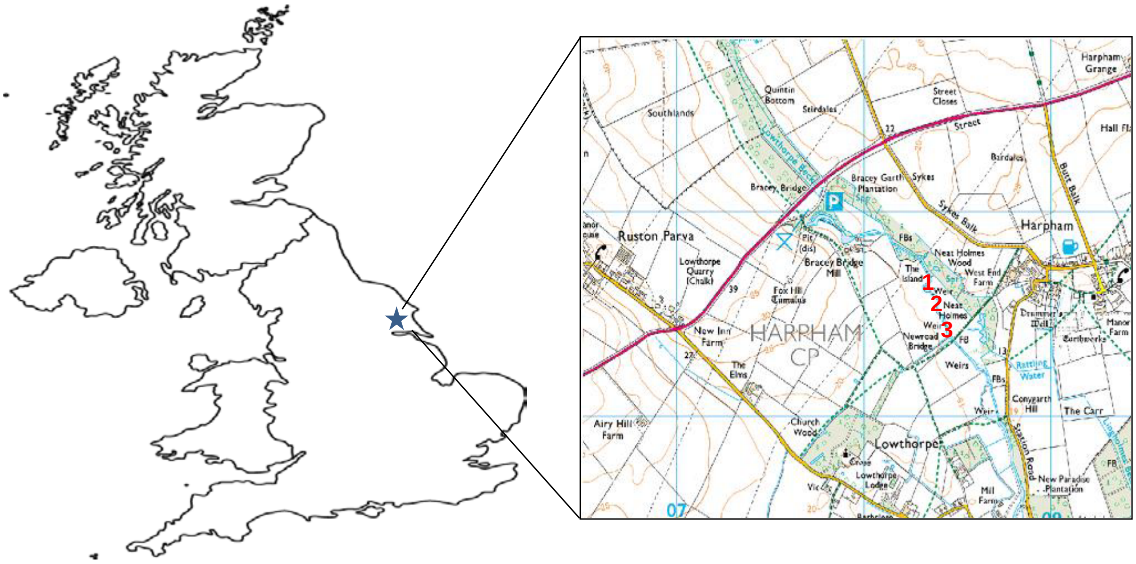


Figure 3.6. Location of Harpham 1-3 on Lowthorpe Beck (Source: Ordnance Survey).

Figure 3.7. Channel narrowing on Lowthorpe Beck a) immediately after works February 2011 b) established margin July 2011.

Aims & objectives

The aim of this study was to determine if channel narrowing at Harpham, Lowthorpe Beck is an appropriate rehabilitation technique to improve fisheries habitat, specifically for brown trout. Furthermore, this study determines if river rehabilitation success can be determined through application of a basic monitoring design. Specific objectives were to compare suitable brown trout habitat availability and brown trout density and age structure prior to and following channel narrowing within the study reach. Fish community structure was also evaluated before and after stream remedial action. Furthermore, to test and establish a monitoring programme that will help assess the direct effect of rehabilitation schemes on fish communities.

3.5.2. Methods

Fish and habitat surveys

Fisheries surveys were carried out on three reaches (Figure 3.6; Table 3.9) on Lowthorpe Beck on the 23rd July 2010 (prior to channel narrowing) and on the 12th July 2011 (post channel narrowing) using quantitative electric fishing (Section 3.1.1). The three sites were Harpham 1 (H1), which was used as a control, Harpham 2 (H2) located immediately downstream of Harpham 3 (H3) which was assessed for downstream effects from channel narrowing (Figure 3.6). H3 was the impact site where channel narrowing occurred (Figure 3.7). The focus of this study was primarily brown trout, but other species may also be present in fewer numbers according to historical Environment Agency reports. After each electric fishing survey, habitat and environmental data were collected at each site in the format used for HABSCORE (Section 3.1.3).

Table 3.9. Fisheries survey site at Harpham, Lowthorpe Beck July 2010 and July 2011.

Site identifier/NGR	River Name	Survey date	Length/mean width/area	Survey method and gear
Harpham 1 TA 083 615	Lowthorpe Beck	23/07/10 12/07/11	54 m/7 m 377 m2 52 m/8.4 m 337 m2	Generator. Quantitative.
Harpham 2 TA 083 616	Lowthorpe Beck	23/07/10 12/07/11	45 m/6.5 m 295.1 m2 50 m/7.58 m 379 m2	Generator. Quantitative.
Harpham 3 TA 083 617	Lowthorpe Beck	23/07/10 12/07/11	37 m/7.9 m 293.8 m2 35 m/7.2 m 252.7 m2	Generator. Quantitative.

3.5.3. Results

HABSCORE analysis

Raw depth data from HABSCORE was used to quantify some of the changes in the channel before and after channel narrowing at Harpham 3 (Figure 3.8), the habitat quality score (HQS Table 3.12) taken from the HABSCORE output, can also be used to quantify habitat improvement for brown trout at Harpham 3. Channel narrowing decreased variability of depths and overall, increased channel depth. The absence of a variety of depths was portrayed by the decrease in HQS between 2010 and 2011, for 0+, >1+ (<20cm) and >1+ (>20cm) (Table 3.12).

HABSCORE outputs for the sites on Lowthorpe Beck revealed variations in the observed densities, predicted densities and habitat utilisation by trout (Table 3.12). Observed densities of 0+ trout in Lowthorpe Beck were lower than predicted by the

Habitat Quality Score (HQS) at all three sites in 2010 and 2011, suggesting poorer populations. HUI upper CLs were only >1 at H2 in 2010 and H3 in 2011, therefore observed populations for 0+ trout at all other sites were significantly lower than expected (Table 3.9). Observed densities of >1+ trout (<20 cm) were higher than predicted by the Habitat Quality Score (HQS) at all sites except H1 2011, but they were only significantly higher in H1 and H2 in 2010 (Table 3.12). H1 trout densities for >1+ (<20 cm) captured in 2011 were lower than expected, but not significantly lower because the HUI upper is >1 (Table 3.9). Observed densities of >1+ (>20 cm) were higher than predicted by the Habitat Quality Score (HQS) at all sites except H1 in 2011 and H2 in 2010, however, they were only significantly higher at H1 in 2010, H2 in 2011 and H3 in 2011 (Table 3.12). H1 in 2011 and H2 in 2010 have lower than expected densities, but not significantly lower as the HUI upper CL is >1 (Table 3.12).

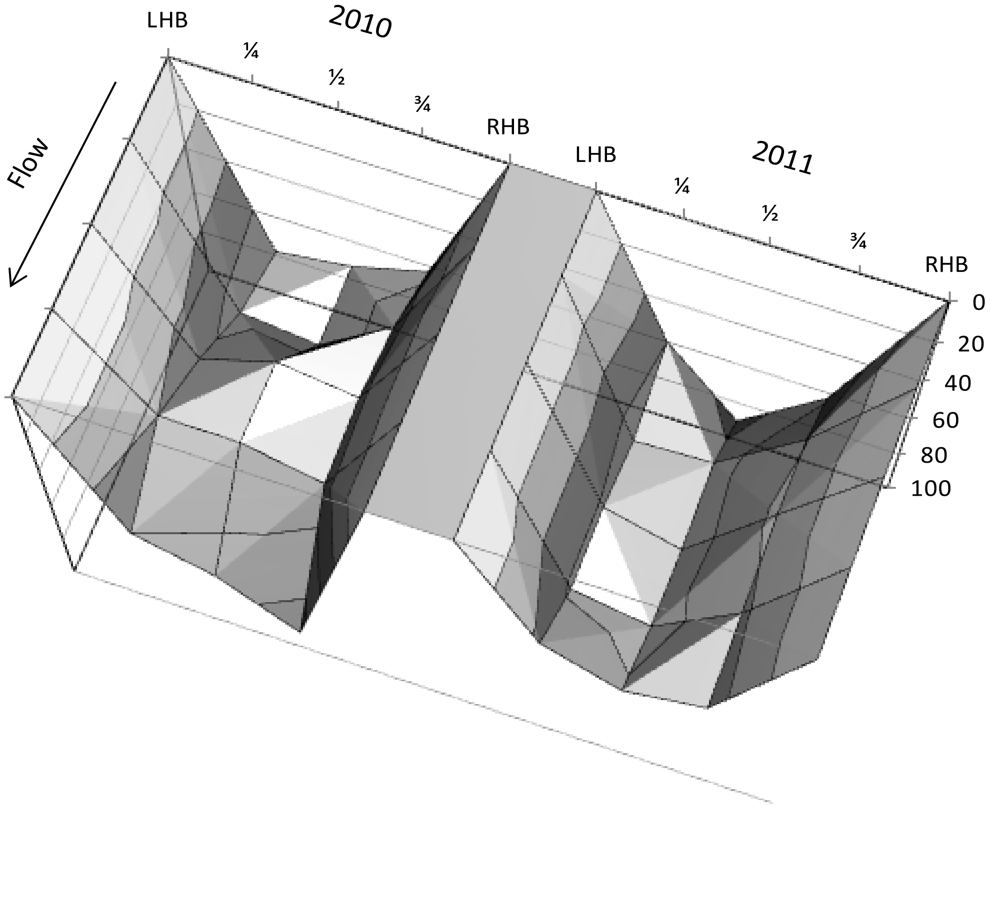


Figure 3.8. Changes in channel depth recorded during HABSCORE at Harpham 3.

Status of fish population on Lowthorpe Beck

Fish species composition was similar across all three sites on the Lowthorpe Beck in July 2010. Brown trout and bullhead were present at all sites; brown trout was the dominant species whereas bullhead was present in low numbers. Lamprey and other species such as 3-spined and 9-spined stickleback (*Pungitius pungitius* L.) were present, but not at all sites (Table 3.10). In July 2011 species composition differed between the three sites (Table 3.10). Brown trout was the dominant species and found at all sites, whereas bullheads were present at H2 and H3 but in low numbers; lamprey and 3-spined stickleback were only present at H3 (Table 3.10).

Species composition differed at each of the three sites between 2010 and 2011 (Table 3.10). A higher number of species was found in H1 and H2 in 2010 than 2011, but the numbers of the other species in 2010 was only represented by a few individuals (Table 3.10). The number of species found at H3 was higher in 2011 than 2010, but like the other sites this was only a few individuals of the other species, 3-spined stickleback (Table 3.10).

Table 3.10. Number of fish of different species captured at three sites at Lowthorpe Beck in July 2010 & 2011.

Species	Harpham 1		Harpham 2		Harpham 3	
	2010	2011	2010	2011	2010	2011
Brown trout	35	4	44	17	13	22
Bullhead	1	0	1	2	6	9
3 spined stickleback	2	0	1	0	0	3
9 spined	0	0	1	0	0	0
Lamprey	0	0	0	0	1	12
Total Species	3	1	4	2	3	4

Density estimates

In 2010, >1+ trout dominated catches at all three sites surveyed on Lowthorpe Beck; as a result >1+ trout were classified as Class C (fair/average) at all sites (Table 3.11, Figure 3.9). In 2010, 0+ trout were only captured at H2 and e classified as Class E (poor), whereas H1 and H3 were classified as Class F (fishless) (Table 3.11). In 2011 there were no 0+ trout captured at any of the three sites and classified as Class F (fishless), where as the abundance class varied for >1+ trout in 2011 catches, H1 was classified as Class E (poor), H2 t Class D (fair/poor) and Harpham 3 Class C (fair/average) (Table 3.11, Figure 3.9).

Table 3.11.Total population estimate (N), population density (D) (± 95% C.L.) and abundance classification of trout derived from fisheries surveys on Lowthorpe Beck in July 2010 & 2011 (density of fish given as numbers per 100m^2). Details of derivation of estimates are provided in the text.

Date	Site	Total Population		Population Density		Abundance Class	
		0+	≥1+	0+	≥1+	0+	≥1+
2010	Harpham 1	0	35±0	0	9.28±0.17	F	C
	Harpham 2	3±1	16±0	1.02±0.46	5.42±0.19	E	C
	Harpham 3	0	17±4	0	5.78±2.43	F	C
2011	Harpham 1	0	4±0	0	0.92±0.10	F	E
	Harpham 2	0	17±1	0	4.49±0.32	F	D
	Harpham 3	0	24±3	0	9.49±2.23	F	C

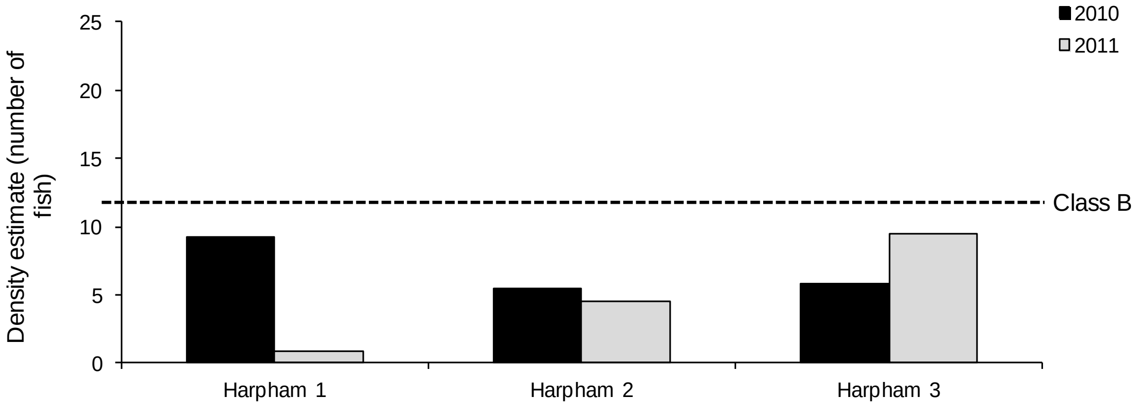


Figure 3.9. Density estimates of >1+ trout at Harpham 1 - 3 in 2010 and 2011, Class B and above indicates good population density.

European Fish Index Calculations

The EFI + database calculate a deterioration in ecological class boundaries at the impacted site (H3), in 2010 Fish Index Cass was 2 (Fish Index 0.81) and in 2011 Fish Index Class was 3 (Fish Index 0.71). A deterioration was also found at the control site H2, in 2010 Fish Index Cass was 2 (Fish Index 0.81) and in 2011 Fish Index Class was 3 (Fish Index 0.61). However, an increase in ecological class was found at H1, in 2010 Fish Index Cass was 4 (Fish Index 0.5) and in 2011 Fish Index Class was 3 (Fish Index 0.64).

Length Frequency

0+ brown trout were absent from H1 in 2010 and 2011. In 2010, >1+ trout caught were in the size range 150-370 mm and in the 2011 surveys between 160-295 mm (Figure 3.10). The oldest trout caught at H1 was 3+ in 2010 and 2+ in 2011.

One 0+ brown trout (105 mm long) was captured at H2 in 2010 and no 0+ trout was captured in 2011. In the 2010 surveys the >1+ trout captured were in the size range 151-341 mm and in the 2011 surveys between 160 and 330 mm (Figure 3.11). The oldest trout caught at H2 was 2+ in both 2010 and 2011. No 0+ trout were captured at H3 in 2010 or 2011. In the 2010 surveys >1+ trout were captured in the size range 157-479 mm and in the 2011 surveys >1+ trout were between 164 and 479 mm (Figure 3.12). The oldest trout caught at H3 in both 2010 and 2011 was 4+.

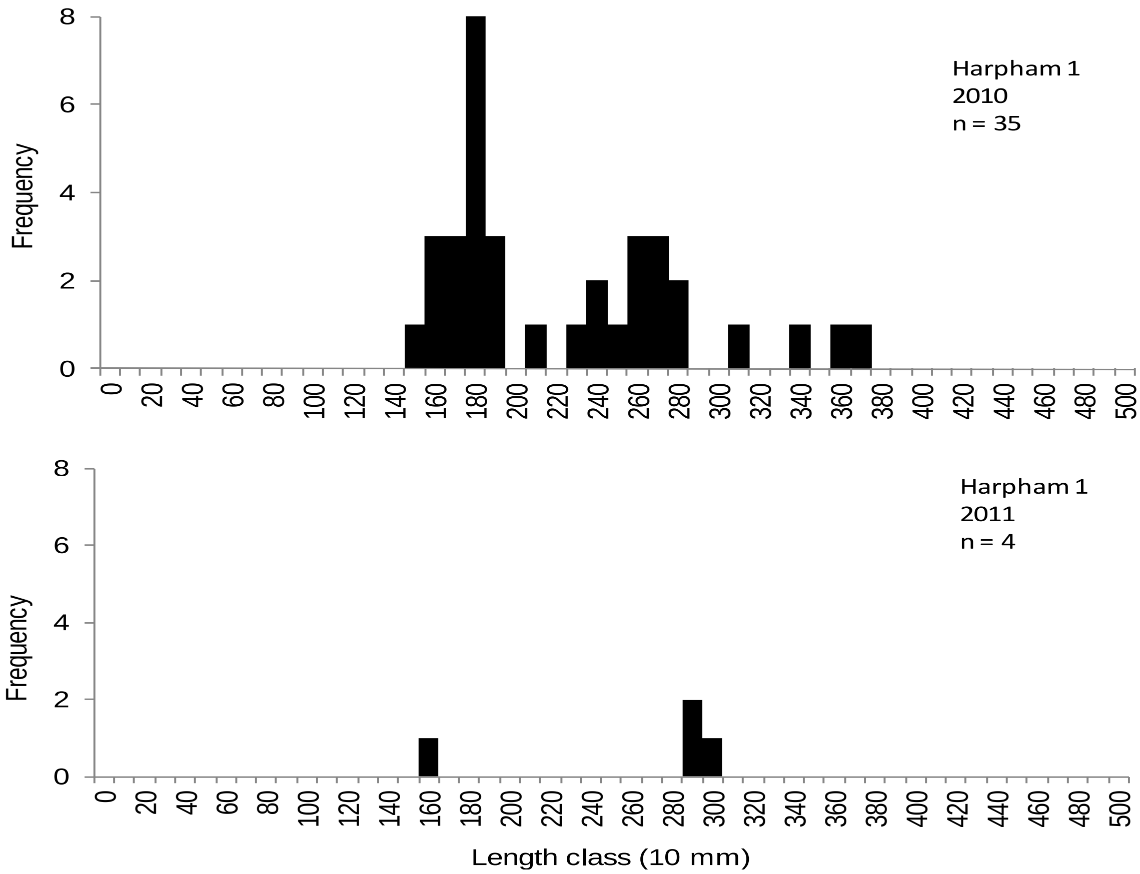


Figure 3.10. Length distributions of brown trout at Harpham 1 in 2010 and 2011.

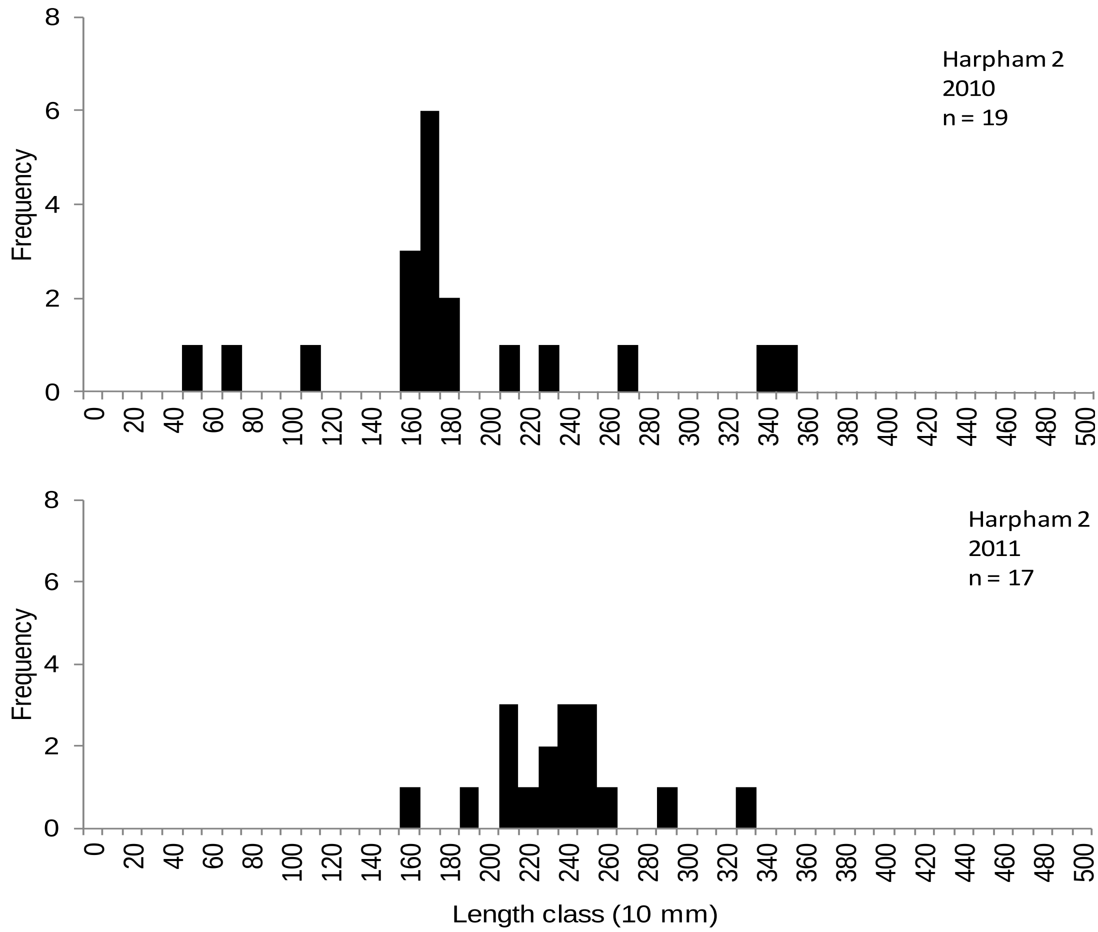


Figure 3.11. Length distributions of brown trout at Harpham 2 in 2010 and 2011.

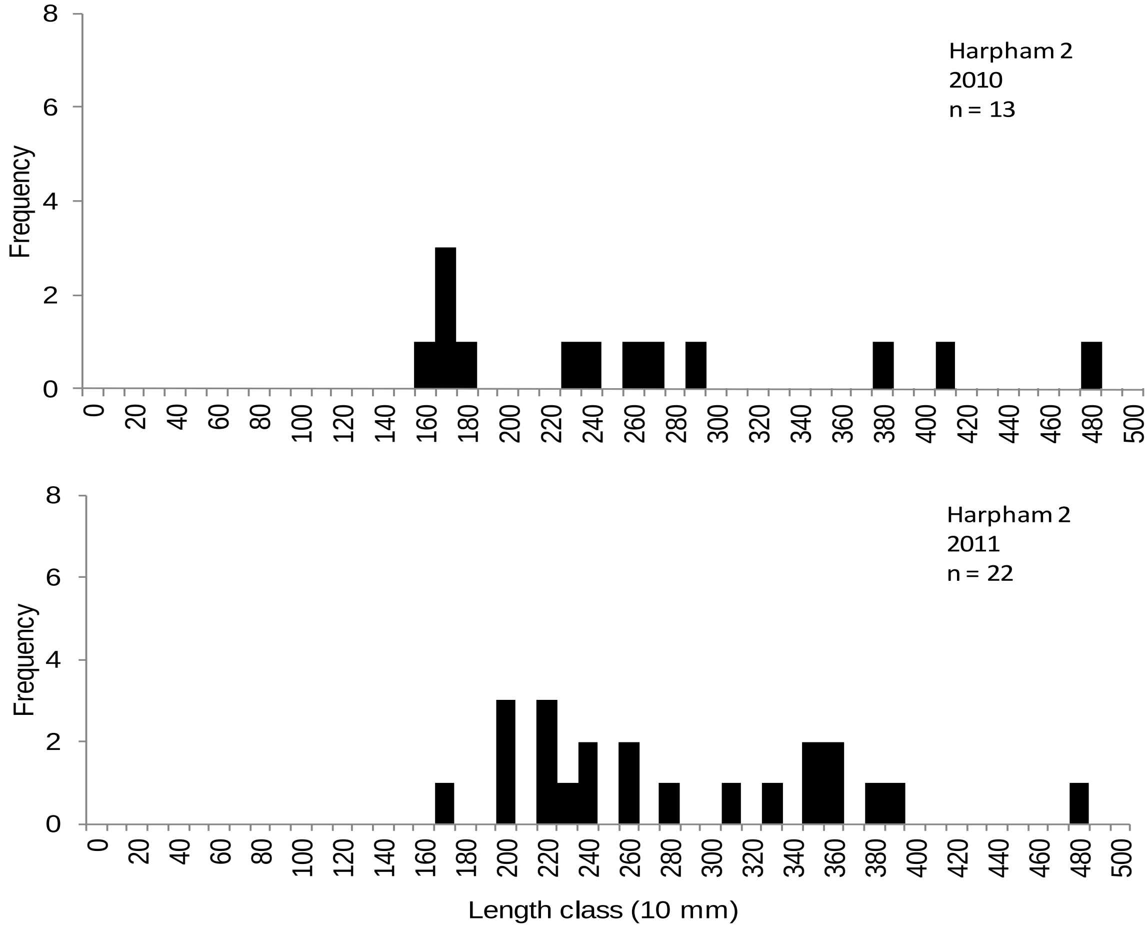


Figure 3.12. Length distributions of brown trout at Harpham 3 in 2010 and 2011.

Table 3.12. HABSCORE outputs for 0+, >1+ (<20 cm) & >1+ (>20 cm) across all sites on Lowthorpe Beck for 2010 and 2011 catches. (Note: Shaded area represents sites where the observed population was significantly higher (HUI lower CL column) or lower (HUI upper CL column) than would be expected under pristine conditions)

Site Code	Age category	Observed number	Observed density	HQS (density)	HQS lower CL	HQS upper CL	HUI	HUI lower CL	HUI upper CL
Harpham 1	0+								
	2010	0	0	2.57	0.64	10.38	0.10	0.01	0.70
	2011	0	0	2.85	0.71	11.44	0.08	0.01	0.56
	>1+ (<20 cm)								
	2010	18	4.62	0.25	0.05	1.16	18.35	2.88	1.17
	2011	1	0.23	0.50	0.11	2.29	0.45	0.07	2.84
	>1+ (>20 cm)								
	2010	17	4.37	1.39	0.45	4.29	3.14	1.02	9.67
	2011	3	0.68	0.82	0.27	2.49	0.84	0.27	2.57
Harpham 2	0+								
	2010	3	1	3.06	0.76	12.30	0.33	0.05	2.35
	2011	0	0	1.99	0.49	8.04	0.11	0.02	0.77
	>1+ (<20 cm)								
	2010	11	3.66	0.37	0.08	1.68	9.85	1.57	61.79
	2011	2	0.44	0.33	0.07	1.56	1.33	0.20	9.04
	>1+ (>20 cm)								
	2010	5	1.66	1.75	0.56	5.50	0.95	0.30	2.99
	2011	15	3.29	0.92	0.30	2.82	3.56	1.16	10.90
Harpham 3	0+								
	2010	0	0	3.31	0.83	13.23	0.10	0.01	0.70
	2011	0	0	2.73	0.68	10.93	0.15	0.02	1.03
	>1+ (<20 cm)								
	2010	5	1.67	0.35	0.08	1.60	4.73	0.74	30.14
	2011	2	0.81	0.30	0.06	1.39	2.73	0.41	19.32

	>1+ (>20 cm)								
	2010	10	3.35	1.17	0.38	3.60	2.86	0.88	9.28
	2011	20	8.08	0.80	0.26	2.47	10.06	3.25	31.22

3.5.4. Discussion

Channel narrowing was carried out at Harpham 3 on the Lowthorpe Beck to improve habitat diversity within an over widened channel. **The main objective of this study was to assess if channel narrowing can be used as an appropriate technique to introduce suitable habitat back in to Lowthorpe Beck, specifically for fish**. Channel narrowing reduced the river width by 1.5 m and increased depth and flow, and natural re-colonization of the berm strengthened the bank and introduced cover for fish, therefore, changing the habitat diversity in this section. An increase in flow would usually improve the river bed by removing fine sediments, but Lowthorpe Beck experiences high volumes of sediment input and as a result the entire river bed is covered in sediment. In some places sediment was over 50 cm deep. Consequently, there were no obvious improvements observed relative to sediment reduction at H3 at Lowthorpe Beck in 2011. 0+ trout were absent at all sites in 2010 and 2011 with the exception of H2 in 2010 where only 3 individuals were caught. This suggests an absence or limited recruitment within the reach and possibly reflects poor quality gravel beds for spawning in the accessible vicinity. Indeed the fishery appears to be maintained by regular stocking and this would mask any bottleneck in recruitment. Overall the HQS (density) showed that channel narrowing reduced habitat diversity for 0+ trout at H3 (Table 3.12), something that was also supported by actual fisheries data that showed observed densities were lower than HQS in 2011 as opposed to the situation in 2010 prior to rehabilitation. The same trends were found for H1 and H2 where no habitat modifications were made. This may be because all three sites were in the same proximity suggesting that a change in habitat within the reach could have modified habitat quality for brown trout at all sites, especially as H1 and H2 are downstream of H3. There were good numbers of >1+ fish in 2010, but fish in the size group <20 cm were absent suggesting there is suitable habitat for larger fish. This was supported by the HQS score (density) (Table 3.12). However, in 2011 there were low numbers of >1+ fish, suggesting that habitat conditions resulting from the channel narrowing are not suitable for all >1+ trout (Table 3.12), nevertheless, this is most probably a response to the disturbance of the channel modification works and may take several years to reset. Overall, it appears that recruitment in Lowthorpe Beck is weak and the trout stocks are supported almost exclusively by stocking, Environment Agency data indicate that brown trout are stocked into Lowthorpe Beck annually and this usually occurs in the first week of July with numbers within the range of 105 to 625 and lengths range between 280 mm to 330 mm. Unfortunately, 2010 and 2011 fisheries surveys took place the week after stocking and therefore do not necessarily give a true representation of the local brown trout population, Environment Agency stocking data were only received after the surveys took place and was therefore, too late to overcome. However, the use of stocking only highlights that there must be additional pressures on Lowthorpe Beck that suppress trout from having a self-sustaining population and stocking should not be seen as a measure to overcome these problems (Cowx & Welcomme 1998).

When comparing 2010 (pre) and 2011 (post) fisheries data at the impacted site (H3), species composition increased from 3 species (brown trout, bullhead and lamprey) to 4 species (brown trout, bullhead, lamprey and 3-spined stickleback). Albeit, the EFI detected deterioration in ecological class boundaries and this was most likely because of the limited fisheries data used for the EFI analysis. For all three sites at Lowthorpe beck, low species richness and number of fish caught were a limitation and therefore, the outcome of this index needs to be used with caution.

Comparison of the impact site (H3), the control site (H1) and the site immediately downstream of the impact site (H2), found no major differences in species composition, although diversity in species composition was due to the presence of low numbers of other species (Table 3.10). Overall brown trout was the dominant species and present at every site, although brown trout numbers were lower in 2011 than 2010 at H3, but this was the same at all sites and thus could be the result of a poor year of natural recruitment, typical of that found in fish populations (Crisp 1993). The results suggest that further monitoring will be required to tease out the effects of natural recruitment from the outcomes of the rehabilitation works. However, given that the control sites also showed the same trends it is suspected that there was no immediate benefit of the rehabilitation works. Lampreys were present in low numbers at H3, and this is important because it is a designated conservation species in the UK. Consequently protection of the species is paramount in any instream river activities and this needs to be considered when doing further rehabilitation works on Lowthorpe Beck. This is particularly pertinent where the aim is to reduce the silt build up which is the critical habitat for lamprey ammocetes, brook and river lamprey (Maitland 2003). Plans must therefore be put in place where some areas of silt are maintained to ensure the maintenance of this species.

The absence of 0+ trout but the presence of >1+ trout implies that there may be problems with suitable brown trout spawning habitat. Essential spawning habitat in rivers includes cool, clear and well oxygenated gravel-bed streams with suitable substrate size, water velocity, water temperature, water depth and dissolved oxygen (Chapman 1988; Quinn 2005; Wheaton *et al*. 2010). Sediment input from the surrounding land is a clear pressure on Lowthorpe Beck and is a common problem on many UK rivers. It reduces the quality of spawning habitat for trout by clogging the gravels and therefore has a negative impact of reproductive success (Alexander & Hansen 1986; Kondolf 2000; Heywood & Walling 2007; Ferreira *et al*. 2010). The introduction of gravel riffles and flow deflectors as rehabilitation measures to overcome the degradation of salmonid spawning habitat have become progressively more common, for example, Harpers Brook (Harper *et al*. 1998) and may also be a possible solution for the further rehabilitation of Lowthorpe Beck. Such artificial riffles have been proven to enhance hydro-geomorphological characteristics such as depth, water velocity and substrates, all of which benefit river ecosystems, especially salmonid spawning habitats (Downs & Thorne 1998).

Recommendations

Overall, there is little evidence that suggests channel narrowing has benefited trout populations in Lowthorpe Beck. There seems to be several underlying pressures on Lowthorpe Beck that need to be identified before being addressed with rehabilitation measures. **It is proposed that there is need for spatial appraisal of the status of the fish populations and habitat surveys from reaches that represent the whole of Lowthorpe Beck to improve understanding of the pressures on this HMWB and target future rehabilitation actions more precisely to reach GEP by 2027**. This will also enable areas of Lowthorpe Beck that have self-recruiting fish populations and suitable habitat to be identified and used as guidance when determining the most beneficial rehabilitation measures for the future**. If such actions are targeted and enable the brown trout to recruit naturally, it will remove the need for stocking and the fishery could potentially be developed into a wild trout fishery that would attract more specialist anglers and allow integration into initiatives such as the wild trout passport scheme.**

The effectiveness of management actions cannot be evaluated without ongoing monitoring and evaluation of outcomes (Brierley *et al*. 2010). Unfortunately an impact assessment could not be used in this study due to a lack of both temporal monitoring. Currently a before-after study design seems more achievable for this case study however, it is generally replicate in time rather than space (Morrisey 1993; Roni 2005) and without replication there is no statistical assumption (Conquest 2000). **Therefore, I recommend that fish populations are monitored temporally and spatially to identify if channel narrowing will benefit fish population in Lowthorpe Beck through application of a before-after-control-impact assessment** to strengthen the statistical power of analysis (Smith *et al*. 1993; Roni *et al*. 2005a). Control sites should be introduced, whether from the Beck itself or representative rivers in the catchment, to overcome natural variability.

3.6. Brash revetment to prevent bank erosion on the River Manifold

3.6.1. Introduction

The River Manifold is a meandering, spate river located in the Peak District and is a sister stream to the River Dove. Two sites were surveyed on the stretch of river between Longnor and Hulme End, Froghall and Ludwell Farm (Figure 3.14). The majority of land either side of this section of the River Manifold is used for agricultural purposes, cattle grazing is specifically one of the main drivers of river degradation in this area and thus, results in bank erosion that reduces available land to farmers and also increases sediment input into the river (Figure 3.15a). Although the overall status of the River Manifold is 'good' (Table 3.4), it is still essential to preserve this high status by continually identifying areas for conservation. Therefore, this erosion pressure driven by agriculture was overcome with channel maintenance measures, Trent Rivers Trust have chosen ‘brash revetment’ as the method for bank protection (Figure 3.15b). This includes bundles of twiggy brash tied into faggots and secured along the base of the bank to protect it from flow and to slow down the erosion process (ART 2005).

Brash will further benefit the river bank forming a more secure bank over time as sediment accumulate within the twiggy bundles.

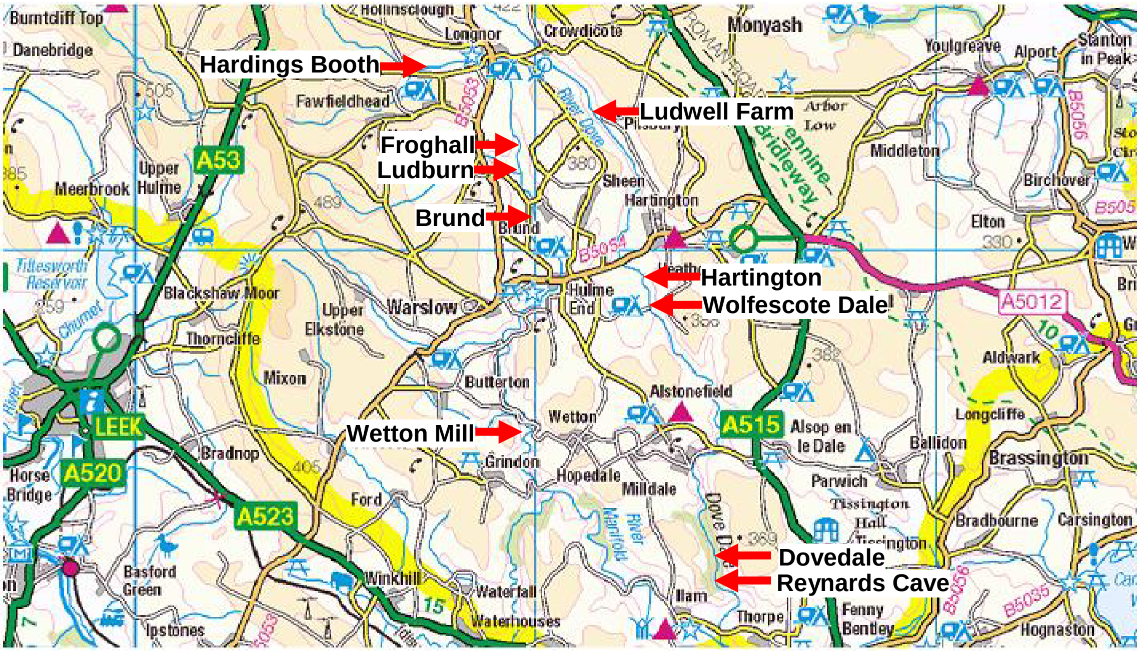


Figure 3.13. Location of sample sites for the River Dove and Manifold (Source: Ordnance Survey).

Figure 3.14. River Manifold a) bank erosion b) brash revetment as a measure to reduce bank erosion.

Aims and objectives

The aim of this study was to determine if brash revetment for bank protection has an effect on fisheries habitat, specifically for brown trout, through application of a post-treatment impact assessment. Specific objectives were to compare suitable brown trout habitat availability, fish community structure and more specifically brown trout density and age structure pre and post brash revetment works at Froghall and post brash revetment at Ludwell Farm. Furthermore, additional representative control sites were included to test and establish a monitoring programme that will help assess the direct effect of river maintenance schemes on fish communities.

3.6.2. Methods

Fish and habitat surveys

Two sites (Froghall b and Ludburn b) were studied to determine the effects of brash revetment on the local fish community (Figure 3.14; Table 3.13). Surveys were carried out at Froghall fisheries in September 2009 (pre), 2010 (post) and 2011 (post) to assess how fish populations respond to brash revetment works. Surveys were also carried out at Ludburn fisheries in September 2010 (post) and 2011 (post) for the same purpose. Fisheries and habitat data were also collected from six control sites, three on the River Dove (Reynard's Cave, Ludwell Farm & Wolfescote Dale) and three on the River Manifold (Froghall a, Ludburn a, Hardings Booth, Brund and Wetton Mill) in 2010 and 2011 (Figure 3.13; Table 3.13), to overcome natural variability by applying spatial replication for the impact assessment.

3.6.3. Results

HABSCORE analysis

HABSCORE outputs for trout at Froghall and Ludburn on the River Manifold revealed variations in the observed densities, predicted densities and habitat utilisation (Table 3.18). Observed densities of 0+ trout in the River Manifold in 2010 were lower than predicted by the Habitat Quality Score (HQS) at all sites suggesting poorer populations than expected. HUI upper CLs were >1 for Froghall a and Froghall b, therefore observed populations were not significantly lower than expected, however HUI upper CLs were <1 for Ludburn a and Ludburn b, therefore observed populations were significantly lower than expected (Table 3.18). Observed densities of 0+ trout in the River Manifold in 2011 were lower than predicted by the Habitat Quality Score (HQS) at all sites suggesting poorer populations than expected, the HUI upper CLs were <1 for all Froghall sites suggesting observed populations were significantly lower than expected (Table 3.18).

Observed densities of >1+ trout (<20 cm) in the River Manifold in 2010 were lower than predicted by the Habitat Quality Score (HQS) at all sites except Froghall b, suggesting poorer populations than expected. However, HUI upper CLs were >1 for Froghall, Ludburn a and Ludburn b, therefore observed populations were not significantly lower than expected (Table 3.18). Observed densities at Froghall b in 2010 were higher than expected but HUI lower CL was <1 and therefore observed population were not significantly higher (Table 3.18). Observed densities of >1+ (<20 cm) trout in the River Manifold in 2011 were also lower than expected with the exception of Froghall b, however, observed populations were only significantly lower at Ludburn a, because HUI upper CLs were <1 (Table 3.18). Observed densities at Froghall b were higher than expected, but not significantly as HUI lower CL was <1 (Table 3.18).

Observed densities of >1+ trout (>20 cm) in the River Manifold in 2010 were higher than predicted by the Habitat Quality Score (HQS) at all sites, suggesting better populations than expected, however, only significantly higher at Froghall a, where HUI lower CL was >1 (Table 3.18). Observed densities of >1+ trout (>20 cm) in the River Manifold in 2011 were higher than expected at Froghall a and Froghall b, but not significantly because HUI lower CL was <1. Densities at Ludburn a and Ludburn b in 2011 were of lower than expected, but not significantly because HUI upper CL were >1 (Table 3.18).

Status of fish populations at Froghall and Ludburn on the River Manifold

Species composition at Froghall was similar across the three survey years with six species present in 2009 and five in 2010 and 2011 (Table 3.14). Catches at Froghall were dominated by bullhead in each year; brown trout, minnow (*Phoxinus phoxinus* (L.)) and stoneloach (*Barbatula barbatula* (L.)) were also present in good numbers. Brown trout and minnow numbers dropped between pre (2009) and post (2010 and 2011) surveys, whereas bullhead and stoneloach numbers increased (Table 3.14). Grayling (*Thymallus thymallus* (L.)) were caught in low numbers in 2009 but was absent in 2010 and 2011. Lamprey were caught in low numbers in 2009 and 2011 but good numbers were caught in 2010 (Table 3.14). Species composition at Froghall b was also similar across the three survey years although an increase in species present was evident from 4 species in 2009 to five in 2010 and six in 2011 (Table 3.14). Catches again were dominated by bullhead, but with good numbers of brown trout and stoneloach in all years (Table 3.14). Brown trout numbers increased from pre (2009) to immediate post (2010) surveys but fell in the later post (2011) survey (Table 3.14). Stoneloach numbers decreased from pre (2009) to immediate post (2010) surveys but increased again in the later post (2011) survey (Table 3.14). Grayling were not present in 2009 and 2010, but were present in low numbers in 2011; lampreys were present in all years but in low numbers (Table 3.14).

Table 3.13. Fisheries survey site details for the River Manifold and the River Dove summer 2010 and 2011.

Site identifier/NGR	River Name	Survey date	Length/mean width/area	Survey method and gear
Froghall a SK 093 638	Manifold	23/09/2009 23/09/2010 21/09/2011	87 m/4.6 m/ 398m2 87 m/4.6 m/ 398m2 80 m/4.5 m/ 360m2	Generator Quantitative
Froghall b SK 093 641	Manifold	23/09/2009 23/09/2010 21/09/2011	50 m/7.2 m/ 360m2 97 m/8.1 m/ 785m2 67 m/5.2 m/ 346m2	Generator Quantitative
Ludburn a SK 095 626	Manifold	24/09/2010 21/09/2011	74 m/4.5 m/ 333m2 47 m/4.4 m/ 205m2	Generator Quantitative
Ludburn b SK 095 628	Manifold	24/09/2010 21/09/2011	67 m/5.3 m/ 352m2 50 m/5.4 m/ 271m2	Generator Quantitative
Hardings Booth SK 069 644	Manifold	15/04/2010 22/09/2011	95 m/3 m/ 285m2 60 m/3.4 m/ 204m2	Generator Semi-Quantitative
Brund SK 099 612	Manifold	07/04/2010 23/09/2011	105 m/6 m/ 630m2 78 m/7.9 m/ 614m2	Generator Quantitative
Wetton Mill SK 095 560	Manifold	13/04/2010 23/09/2011	100 m/12 m/ 1200m2 69 m/11 m/ 750m2	Generator Semi-Quantitative
Wolfescote Dale SK 131 583	Dove	07/04/2010 22/09/2011	180 m/10 m/ 1800m2 64 m/7.2 m/ 460m2	Generator Semi-Quantitative
Ludwell Farm SK 117 630	Dove	07/04/2010 22/09/2011	170 m/4 m/ 680m2 60 m/4.2 m/ 252m2	Generator Semi-Quantitative
Reynard's Cave SK 144 525	Dove	15/05/2010 06/07/2011	100 m/12.4 m/ 1240m2 105 m/12.3 m/ 1292m2	Generator Quantitative

Species composition at Ludburn a was the same four species in 2010 and 2011 catches (Table 3.14). Bullhead dominated in both survey years and stoneloach were the next most abundant species (Table 3.14). Brown trout and lamprey were both present in 2010 and 2011, but in low numbers (Table 3.14). Abundance of all species increased between 2010 and 2011 catches with the exception of brown trout (Table 3.14). Species composition at Ludburn b was the same five species in 2010 and 2011 catches, with a total of 5 species (Table 3.14). Bullhead dominated catches; minnow, brown trout, stoneloach and lamprey were also present, with better numbers in 2010 than 2011, with the exception of stoneloach (Table 3.14).

Density estimates

Density estimates and abundance classifications of brown trout varied in catches between sites in 2009, 2010 and 2011 (Tables 3.15, 3.16 and 3.17). 0+ trout at Froghall a and Froghall b in 2009 were classified as fair/average (Class C) while ≥1+ trout were classified as good (Class B) at Froghall a and fair (class D) at Froghall b (Table 3.15). In 2010, 0+ trout were classified as fair/average (Class C) at Froghall b, fair (Class D) at Froghall a and poor (Class E) at both Ludburn a and b (Table 3.16). ≥1+ trout were classified as good (Class B) at Froghall a, fair/average (Class C) at

Froghall b and Ludburn b and fair (Class D) at Ludburn a (Table 3.16; Figure 3.14). In 2011, 0+ trout densities were poor (Class E) at Froghall b and Ludburn b, and were absent (Class F) from Froghall a and Ludburn a (Table 3.14; Figure 3.15). ≥1+ brown trout were classified as good (Class B) at Froghall a, fair (Class D) at Froghall b and Ludburn b, and poor (Class E) at Ludburn a (Table 3.17; Figure 3.16).

Table 3.14. Number of fish of different species captured at Froghall and Ludburn on the River Manifold in September 2009, 2010 & 2011.

Species	Froghall a			Froghall b			Ludburn a		Ludburn b	
	2009	2010	2011	2009	2010	2011	2010	2011	2010	2011
Brown trout	70	62	22	47	68	32	11	1	37	10
Bullhead	172	203	209	219	297	233	150	189	120	116
Stoneloach	27	51	77	65	57	75	19	26	20	26
Minnow	77	49	19	0	11	11	0	0	64	16
Grayling	2	0	0	0	0	3	0	0	0	0
Lamprey	8	41	5	7	7	10	2	5	180	33
Total Species	6	5	5	4	5	6	4	4	5	5

Table 3.15. Total population estimate (N), population density (D) (± 95% C.L.) and abundance classification of trout derived from fisheries surveys at Froghall, River Manifold in September 2009 (density of fish given as numbers per 100m^2). Details of derivation of estimates are provided in the text.

Site No	Total Population (N)		Population Density (D)		Abundance Class	
	0+	≥1+	0+	≥1+	0+	≥1+
Froghall a	55±7	69±18	13.8±3.4	17.3±8.9	C	B
Froghall b	38±1	8±1	10.6±0.5	2.2±0.16	C	D

Table 3.16. Total population estimate (N), population density (D) (± 95% C.L.) and abundance classification of trout derived from fisheries surveys at Froghall and Ludburn, River Manifold in September 2010 (density of fish given as numbers per 100m^2). Details of derivation of estimates are provided in the text.

Site No	Total Population (N)		Population Density (D)		Abundance Class	
	0+	≥1+	0+	≥1+	0+	≥1+
Froghall a	12±1	53±2	3.49±0.4	15.41±1.26	D	B
Froghall b	38±1	33±4	10.98±0.62	19.54±2.06	C	C
Ludburn a	1±0	10±0	0.30±0	2.98±0.21	E	D
Ludburn b	6±1	32±2	1.71±0.07	9.14±1.14	E	C

Table 3.17. Total population estimate (N), population density (D) (± 95% C.L.) and abundance classification of trout derived from fisheries surveys Froghall and Ludburn,

River Manifold in September 2011 (density of fish given as numbers per 100m²). Details of derivation of estimates are provided in the text.

Site No	Total Population (N)		Population Density (D)		Abundance Class	
	0+	≥1+	0+	≥1+	0+	≥1+
Froghall a	0±0	55±13	0±0	16.77±8	F	B
Froghall b	4±1	15±3	1.3±0.51	4.89±1.67	E	D
Ludburn a	0±0	1±0	0±0	0.33±0	F	E
Ludburn b	2±0	8±0	0.74±0	2.96±0.19	E	D

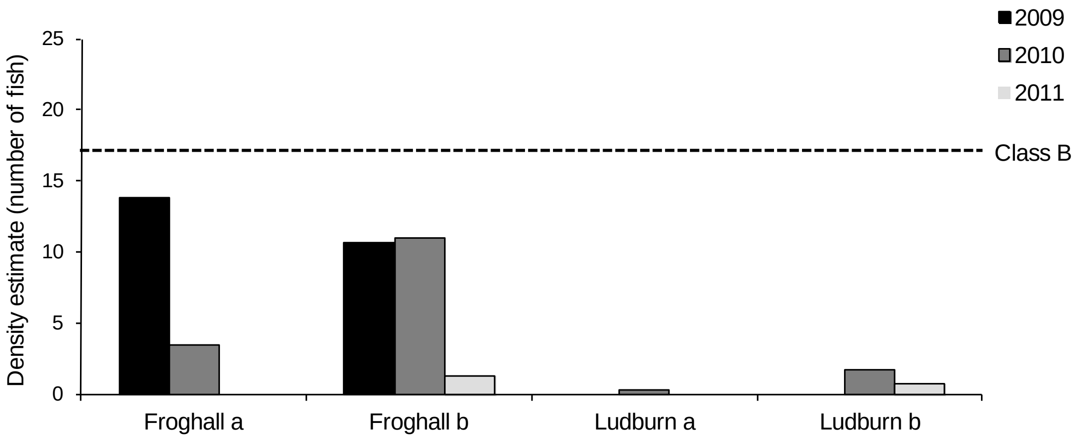


Figure 3.15. Density estimates of 0+ trout on the Manifold in 2009, 2010 and 2011, Class B and above indicates good population density.

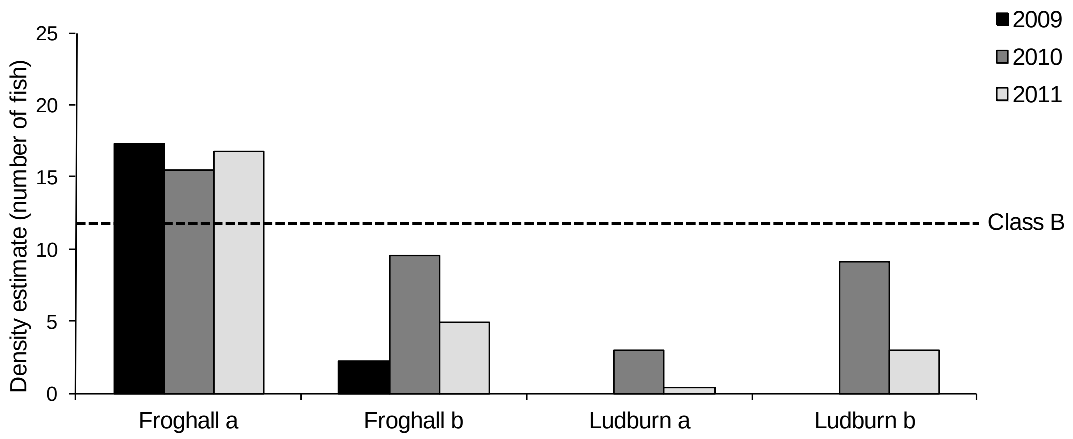


Figure 3.16. Density estimates of >1+ trout on the Manifold 2009, 2010 and 2011, Class B and above indicates good population density.

European Fish Index Calculations

There was no change in ecological class boundaries after brash revetment at both Froghall b and Ludburn b. The EFI + database calculate a Fish Index Cass 2 for 2010

(Fish Index 0.89 for Froghall & Fish Index 0.84 Ludburn b) and 2011 (Fish Index 0.91 for Froghall & Fish Index 0.85 Ludburn b) at both sites.

Length frequency

0+ brown trout captured at Froghall a in 2009 were in the size range 44-111 mm while ≥1+ individuals were in the size range 125-500 mm. In 2010 0+ trout captured were in the size range 69–91 mm while ≥1+ individuals were in the size range 121-415 mm. In 2011 no 0+ trout were captured, but ≥1+ individuals were between 132 and 395 mm long (Figure 3.17). 0+ brown trout captured at Froghall b in 2009 were in the size range 66-115 mm while ≥1+ individuals were in the size range 120-228 mm. In 2010 0+ trout were captured in the size range 58–115 mm while ≥1+ individuals were in the size range 122-489 mm. In 2011, 0+ trout captured were in the size range 70–119 mm and ≥1+ individuals between 131 and 260 mm long (Figure 3.18).

Only one 0+ brown trout (77 mm) was captured at Ludburn a in 2010, while ≥1+ individuals captured were in the size range 152-293 mm. In 2011 no 0+ trout and only one ≥1+ trout (166 mm) were caught (Figure 3.19). 0+ brown trout captured at Ludburn b in 2010 were in the size range 72-85 mm while ≥1+ individuals were in the size range 117-408 mm. In 2011 0+ trout captured were in the size range 93 – 105 mm and ≥1+ trout between 138 and 414 mm long (Figure 3.20).

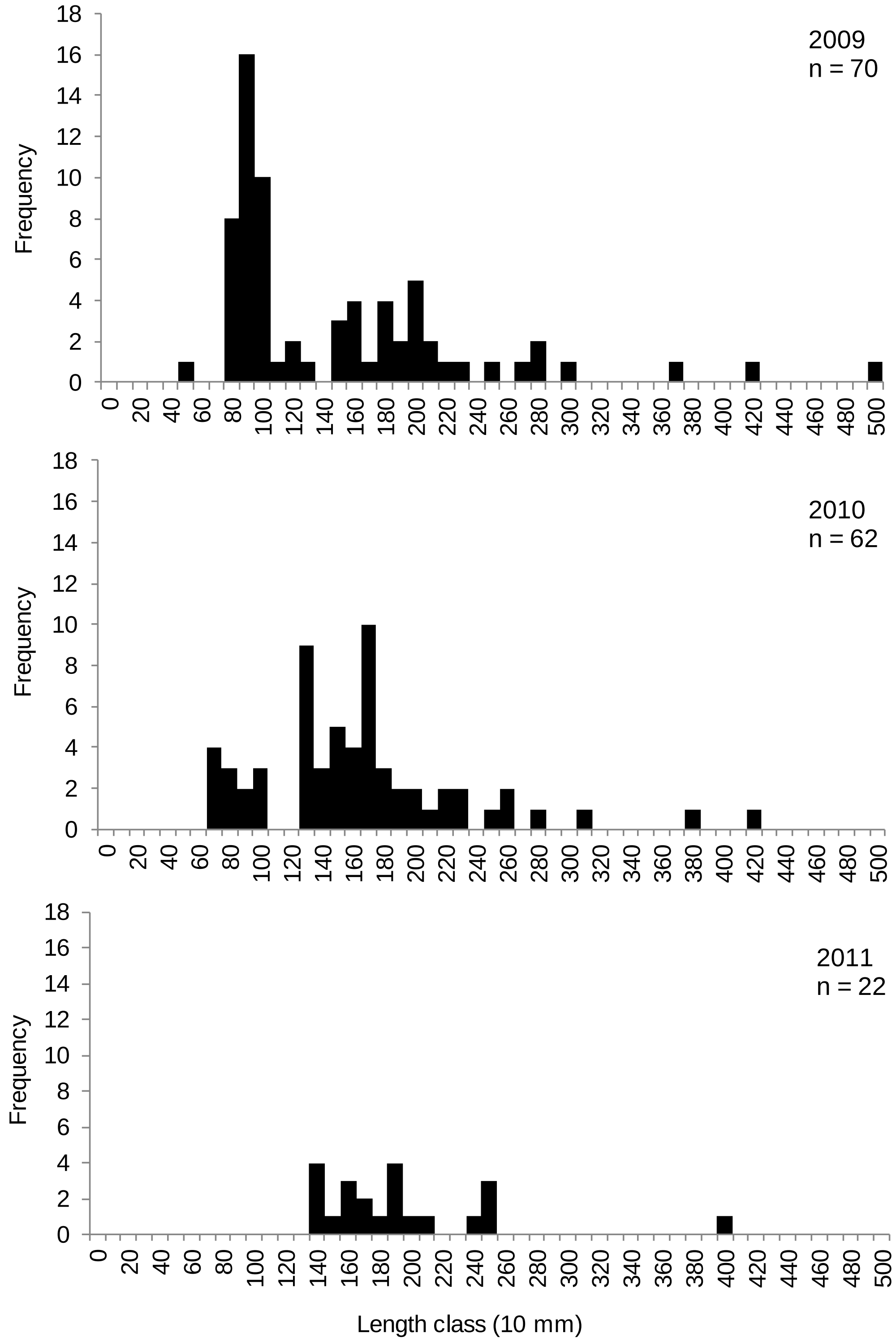


Figure 3.17. Length distributions of brown trout at Froghall a, River Manifold.

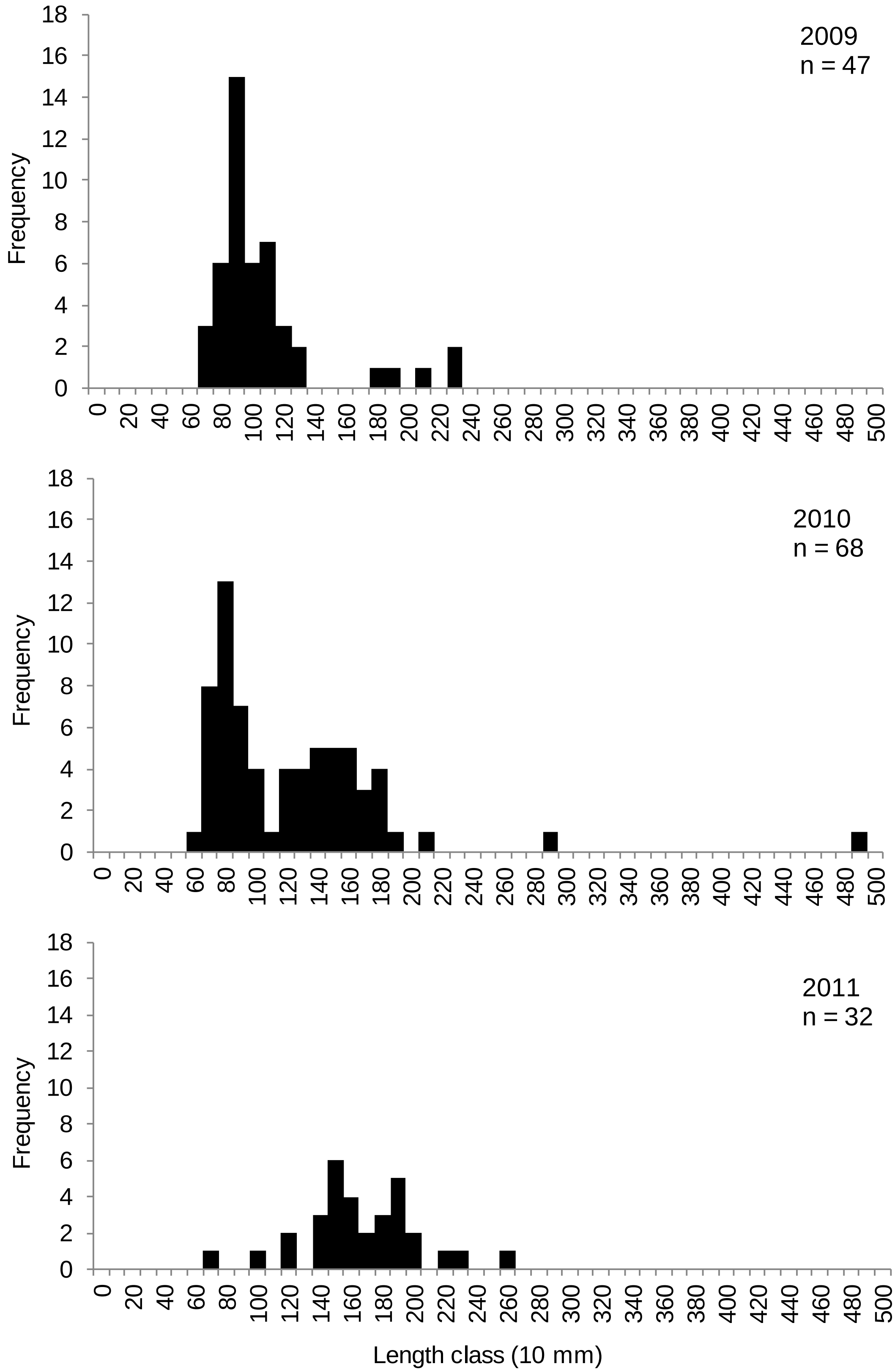


Figure 3.18. Length distributions of brown trout at Froghall b, River Manifold.

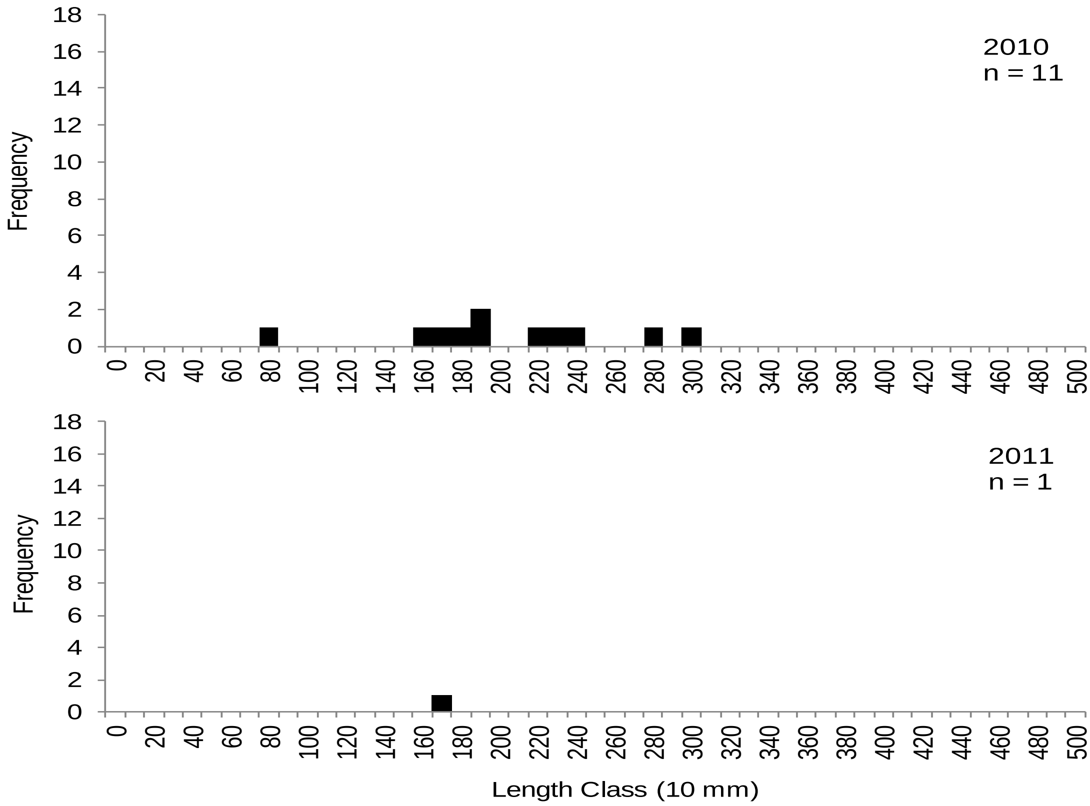


Figure 3.19. Length distributions of brown trout at Ludburn a, River Manifold.

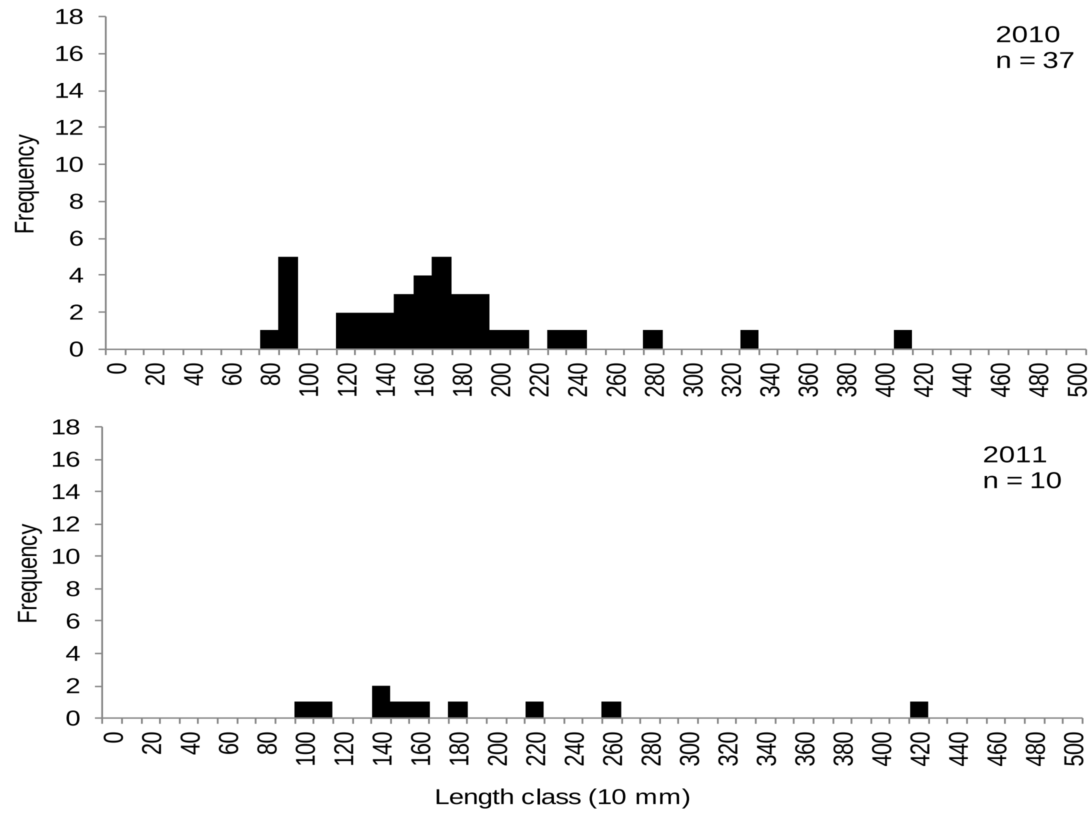


Figure 3.20. Length distributions of brown trout at Ludburn b, River Manifold.

Table 3.18. HABSCORE outputs for 0+, >1+ (<20 cm) **,** *>1+ (>20 cm) trout at four sites on the River Manifold for 2010 and 2011 catches. (Note: Shaded area represents sites where the observed population was significantly higher (HUI lower CL column) or lower (HUI upper CL column) than would be expected under pristine conditions).*

Site Code	Age category	Observed number	Observed density	HQS (density)	HQS lower CL	HQS upper CL	HUI	HUI lower CL	HUI upper CL
Froghall a	0+								
	2010	12	3.21	16.65	4.5	61.62	0.19	0.03	1.27
	2011	0	0	43.43	11.67	161.62	0.01	0.00	0.05
	>1+ (<20 cm)								
	2010	40	10.71	16.24	3.74	70.52	0.66	0.11	4.01
	2011	37	11.13	12.99	3.06	55.16	0.86	0.14	5.30
	>1+ (>20 cm)								
	2010	12	3.21	0.56	0.18	1.73	5.69	1.85	17.47
	2011	7	2.11	0.90	0.30	2.71	2.35	0.74	7.41
Froghall b	0+								
	2010	33	9.73	26.39	7.18	97.04	0.37	0.06	2.41
	2011	2	0.61	18.79	5.06	69.71	0.03	0.00	0.21
	>1+ (<20 cm)								
	2010	35	10.32	8.61	1.95	37.99	1.20	0.19	7.44
	2011	31	9.41	9.38	2.18	40.33	1.00	0.16	6.19
	>1+ (>20 cm)								
	2010	3	0.88	0.36	0.11	0.00	2.48	0.80	7.75
	2011	4	1.21	0.81	0.27	2.45	1.49	0.49	4.51
Ludburn a	0+								
	2010	1	0.31	12.35	3.26	46.80	0.03	0.00	0.17
	2011	0	0	27.11	7.36	99.77	0.02	0.00	0.12
	>1+ (<20 cm)								
	2010	5	1.57	2.21	0.49	9.93	0.71	0.11	4.47
	2011	0	0	6.10	1.46	25.46	0.08	0.01	0.48

	>1+ (>20 cm) 2010 2011	 5 1	 1.57 0.5	 0.66 0.92	 0.21 0.30	 2.04 2.78	 2.38 0.54	 0.77 0.18	 7.37 1.64

Site Code	Age category	Observed number	Observed density	HQS (density)	HQS lower CL	HQS upper CL	HUI	HUI lower CL	HUI upper CL
Ludburn b	0+ 2010 2011	 6 2	 1.82 0.74	 18.75 23.42	 4.99 6.36	 70.36 86.21	 0.10 0.30	 0.01 0.00	 0.64 0.21
	>1+ (<20 cm) 2010 2011	 26 5	 7.89 1.85	 10.70 6.60	 2.48 1.58	 46.05 27.58	 0.74 0.28	 0.12 0.05	 4.47 1.65
	>1+ (>20 cm) 2010 2011	 6 3	 1.82 1.11	 1.06 1.30	 0.35 0.42	 3.25 3.98	 1.72 0.86	 0.56 0.28	 5.28 2.64

Impact assessment and resource calculation

A post-treatment design (Section 3.2.4, Equation 3.2) was used to assess the effects of brash revetment measures on 0+ and >1+ brown trout populations in the River Manifold at Froghall and Ludburn (Table 3.1a). The impact assessment was assessed both in terms of confidence limits of the mean difference in 0+ and >1+ brown trout density and by means of a *t*-test. The mean change in 0+ brown trout density at Froghall, between the impact sites (M = -1.50, SE = 0.04) and the control sites (M = 0.37, SE = 0.51), t (1) = 4.015, was 0.58 ± 2.26 fish per 100 m², the 95% confidence limits (-1.67 – 2.84) encloses zero and therefore the difference was not significant p= 0.16 (p > 0.05, r = 0.94). The mean change in >1+ brown trout density at Froghall, between the impact sites (M = -0.46, SE = 0.63) and the control sites (M = 0.23, SE = 0.15), t (1) = 0.88, was 0.78 ± 3.20 fish per 100 m², the 95% confidence limits (-2.42 – 3.99) encloses zero and therefore the difference was not significant p= 0.54 (p > 0.05, r = 0.4381). The standard errors for the mean difference in 0+ brown trout at impact and control sites was large and therefore a large amount of variability between the means of different samples. The standard errors for the mean difference in >1+ at impact sites was large and therefore a large amount of variability between the means of different samples, however, standard error was low at the control sites.

The mean change in 0+ brown trout density at Ludburn, between the impact sites (M = -0.35, SE = 0.09) and the control sites (M = 0.37, SE = 0.51), t (1) = 1.71, was 0.27 ± 2.11 fish per 100 m², the 95% confidence limits (-1.84 – 2.38) encloses zero and therefore the difference was not significant p= 0.34 (p > 0.05, r = 0.75). The mean change in >1+ brown trout density at Froghall, between the impact sites (M = -0.91, SE = 0.00) and the control sites (M = 0.23, SE = 0.15), t (1) = 7.83, was 0.31 ± 3.60 fish per 100 m², the 95% confidence limits (-3.29 – 3.91) encloses zero and therefore the difference was not significant p= 0.08 (p > 0.05, r = 0.98). The standard errors for the mean difference in 0+ brown trout at impact and control sites was large and therefore a large amount of variability between the means of different samples. The standard errors for the mean difference in >1+ at impact sites was large and therefore a large amount of variability between the means of different samples, however, standard error was low at the control sites.

Sampling size was too small to overcome variability between the mean density of 0+ and >1+ brown trout at the impact sites and for 0+ brown trout at control sites. Only two impact sites were sampled across two years at Froghall and Ludburn. A larger number of samples, across a large number of years would reduce standard error and reduce variability between the means. The resource calculation (Section 3.2.4, Equation 3.4) for Froghall determined that the variance in the actual data for 0+ trout (0.35) was not low enough for a change of 5 fish 100 m2 to be detected statistically, but the target variance (0.27) can be reached by monitoring two year pre and two years post brash modification with four control and four impact sites. The resource calculation identified the variance in actual data for >1+ trout (0.05) was low enough to identify a change of 5 fish 100 m2 to be detected statistically. The resource calculation for Ludburn determined that the variance in the actual data for 0+ trout (0.21) was low enough to identify a change of 5 fish 100 m2 to be detected statistically. The resource calculation identified the variance in actual data for >1+ trout (0.28) was not low enough for a change of 5 fish 100 m2 to be detected statistically, therefore the target variance (0.27) can be reached by monitoring two year pre and two years post brash modification with three control and three impact sites.

3.6.4. Discussion

The River Manifold is a watercourse that has natural meandering processes in conjunction with the expected erosional processes on the outer bank and sediment deposition on the inner bank. This facilitates alternate bar and pool topography and enables the meandering channel to migrate downstream (Duan *et al*. 1999). The concept of instream maintenance works to protect farmer's land, such as soft revetments, is used to reduce the occurrence of erosion and loss of land, and is becoming a more frequent tool to control this natural process. Annual stocking of ≥1+ (>20cm) brown trout, primarily for angling, by the Environment Agency on the River Manifold is a regular occurrence and consequently there is the possibility these stocking events have influenced the 2010 and 2011 fish catch data for both Froghall and Ludwell, further influencing the true representation of the local brown trout population. The HABSCORE outputs indicate that the HQS is lower than the actual observed density for ≥1+ (>20 cm) trout for the majority of sites, suggesting the habitat will not support a high number of trout of this size (Table 3.18). The HQS score for 0+ trout indicates that the habitat is especially suitable for 0+ trout at all sites, but actual observed density of 0+ trout at all sites was extremely low. Subsequently, the stocking of ≥1+ trout and the low number of 0+ trout only highlights that there must be additional pressures on the River Manifold that suppress trout from having a self-sustaining population (Table 3.18). The missing habitat (feeding, breeding or refuge) needs to be identified and rehabilitated for both 0+ and ≥1+ trout to enhance the population and encourage it to become self sustaining.

Overall the output of the fisheries surveys on the River Manifold in 2009, 2010 and 2011 established that there were no direct benefits of the soft revetment on the status of the fish population at Froghall or Ludburn. This was also supported by the EFI for each site, were there was no change in ecological class boundary after brash revetment. At sites a and b at Froghall the dominant species remained the same before and after brash revetment work suggesting it did not significantly impact on the fish community. At sites a and b at Ludburn the dominant species remained the same for the two years after brash revetment work suggesting it did not significantly impact on the fish community. The abundance classification of 0+ trout at both Froghall sites was similar before and after brash revetment work, therefore there was no direct effect on 0+ trout numbers. Although 0+ trout numbers declined in 2011 this reduction could not be related directly to the brash revetment. The abundance classification of >1+ trout did not significantly change after the brash revetment at both Froghall sites. At both Ludburn sites there was a decline in the abundance of both 0+ and >1+ trout between 2010 and 2011 possibly in response to the brash revetment work. However, because there is an overall reduction in trout numbers at this site in 2011, there is the possibility it resulted from natural variability. Nevertheless, even though the abundance classification did vary before and after brash revetment at some sites and for some ages classes, the post treatment design did not identified a significant decrease. There is still the possibility that the decrease at some sites was a response to the brash revetment, but this is unlikely and cannot be definitively stated since natural variability could have also been interacting on the trout population. The control sites used in the post treatment analysis enabled some spatial variability to be overcome, but the monitoring design was limited temporally (two years post) for both Frog Hall and Ludburn, further monitoring for an additional number of years to overcome this temporal variability is recommended. The duration of monitoring needed to detect a change is particularly important in any type of monitoring study design to overcome natural variability and identify a possible change as a result of the impact, in this case brash revetment.

Sediment loading from erosion further upstream can have detrimental effects on freshwater ecosystems and can reduce the availability of suitable spawning grounds for trout by smothering gravels. Erosion from agricultural land is also known for causing water quality problems, and these pressures could be limiting the trout population on the Manifold, especially 0+ trout. The low number of grayling captured suggests that their population is also weak and further studies need to be done to identify underlying pressures that impact on this species.

Soft revetment appears to be a cost effective solution to prevent bank erosion; it also has great aesthetic appeal and can provide habitat for fish, but it is necessary to consider the implications the revetment work may have on the surrounding environment. Overall, this study identified no relation between brash revetment and positive or negative effects on the fish community that can be identified through short term monitoring**. It is thus recommended that long term monitoring is implemented in studies of this nature and a minimum 2 years pre and 2 years post, with 4 control and 4 impacted sites are surveyed to account for natural variability on fish population abundance**. Rivers are very complex systems and can be sensitive to change; a common effect once a bank is protected is for the point of erosion to be transferred to the next available point downstream (ART 2005). For this reason the main cause of the erosion should be investigated prior to any rehabilitation work and other methods for erosion control should be considered before bank protection. There is the concept that revetment work to reduce erosion is a rehabilitation technique, however, although it attempts to maintain the river channel, it actually hinders the natural processes and is therefore a contradictory technique working more towards land management than river rehabilitation.

Recommendations

Knowledge of the effects of brash revetment on fish communities is important, especially as literature is limited in this area. **As a consequence, further research is needed and with guidance from the resource calculation it is recommended that a minimum of two year pre and two year post monitoring is need with four impact and four control sites to detect with statistical confidence a change in 0+ trout and for >1+ trout a minimum of two year pre and two years post monitoring is needed with three control and three impact sites**.

Sediment loading from upstream erosion can have detrimental effects on freshwater ecosystems by reducing the availability of suitable spawning grounds for trout and other lihophilic species. **A more detailed assessment of how the fine sediment affects suitable spawning grounds is needed on the River Manifold to assess if this is the limiting factor for a self-sustaining brown trout population**.

Often livestock activity is an additional pressure on river bank erosion and can be prevented by stopping their access to the watercourse by fencing (ART 2005). Although live stock activity was not extensive at the sampled areas, it was found, but further **fencing will assist in reducing the speed of erosion by reducing this pressure**. Alternatively there is always the ‘do nothing' approach since the erosion of a water course is a natural process; to interfere could cause subsequent knock on effects downstream and impact on future hydro-geomorphological processes within the River Manifold.

3.7. Effect of small weir removals on freshwater rivers

3.7.1. Introduction

The River Dove is located in the Peak District with one bank in Derbyshire and one in Staffordshire and runs off the southern end of the Pennines. Fragmentation is a common pressures on this river resulting from a large number of weirs, systematically placed throughout the 30 K stretch and hinder the up and down stream movement of fish. Although the Dove is classified as 'good' status by the WFDs water body classification, it is still essential to either maintain this level of integrity or aim to reach 'high' status. For this reason, Trent Rivers Trust and Wild Trout Trust proposed the removal of two rock weir on the River Dove, to increase continuity and diversity in to the river by the removal of these barriers. Hartington is the location of the first weir (Figure 3.22, Table 3.19); here the river is meandering, narrow and deep with a silt substrate. The second weir removal is located at Dovedale (Figure 3.23, Table 3.19), where the river is wider, less meandering and generally shallower with boulder substrate. Many of the weirs in Dovedale are systematically positioned and were built to produce pool-areas to increase the feeding ground for brown trout and grayling, to improve fishing for anglers, only a few of the weirs on the Dove were used to power mills. Hartington and Dovedale weir removals took place in July 2010 by taking away the large boulders of which they were comprised (Figure 3.22 & 3.23).

Aims and objectives

The aim of this study was to determine if weir removal at two sites on the River Dove was an appropriate rehabilitation technique to improve fisheries habitat, specifically for brown trout. Furthermore, this study determines if river rehabilitation success can be determined through application of a BACI monitoring design. Specific objectives were to compare suitable brown trout habitat availability, fish community structure and more specifically brown trout density and age structure pre and post weir removal at Hartington and Dovedale on the River Dove. Furthermore, additional representative control sites were included to test and establish a monitoring programme that will help assess the direct effect of river maintenance schemes on fish communities.

3.7.2. Methods

Fish and habitat surveys

In total 2 sites (Hartington and Dovedale) were surveyed to determine the effects of weir removal on the local fish community. Fisheries surveys were completed July 2010 (pre), September 2010 immediately after weir removal and September 2011 one year after weir removal, to assess how weir removal affected local fish populations at these sites (Table 3.19). Fisheries and habitat data were also collected from six control sites, three on the River Dove (Reynard’s Cave, Ludwell Farm & Wolfescote Dale) and three on the River Manifold (Hardings Booth, Brund and Wetton Mill) in 2010 and 2011, to apply spatial replication for the impact assessment and therefore overcome variability (Table 3.19). The focus of this study was primarily brown trout, but other species may also be present according to historical Environment Agency reports. After each electric fishing survey, habitat and environmental data were collected at each site in the format used for HABSCORE (Section 3.1.3).

Figure 3.21.. Hartington a) before b) after weir removal.

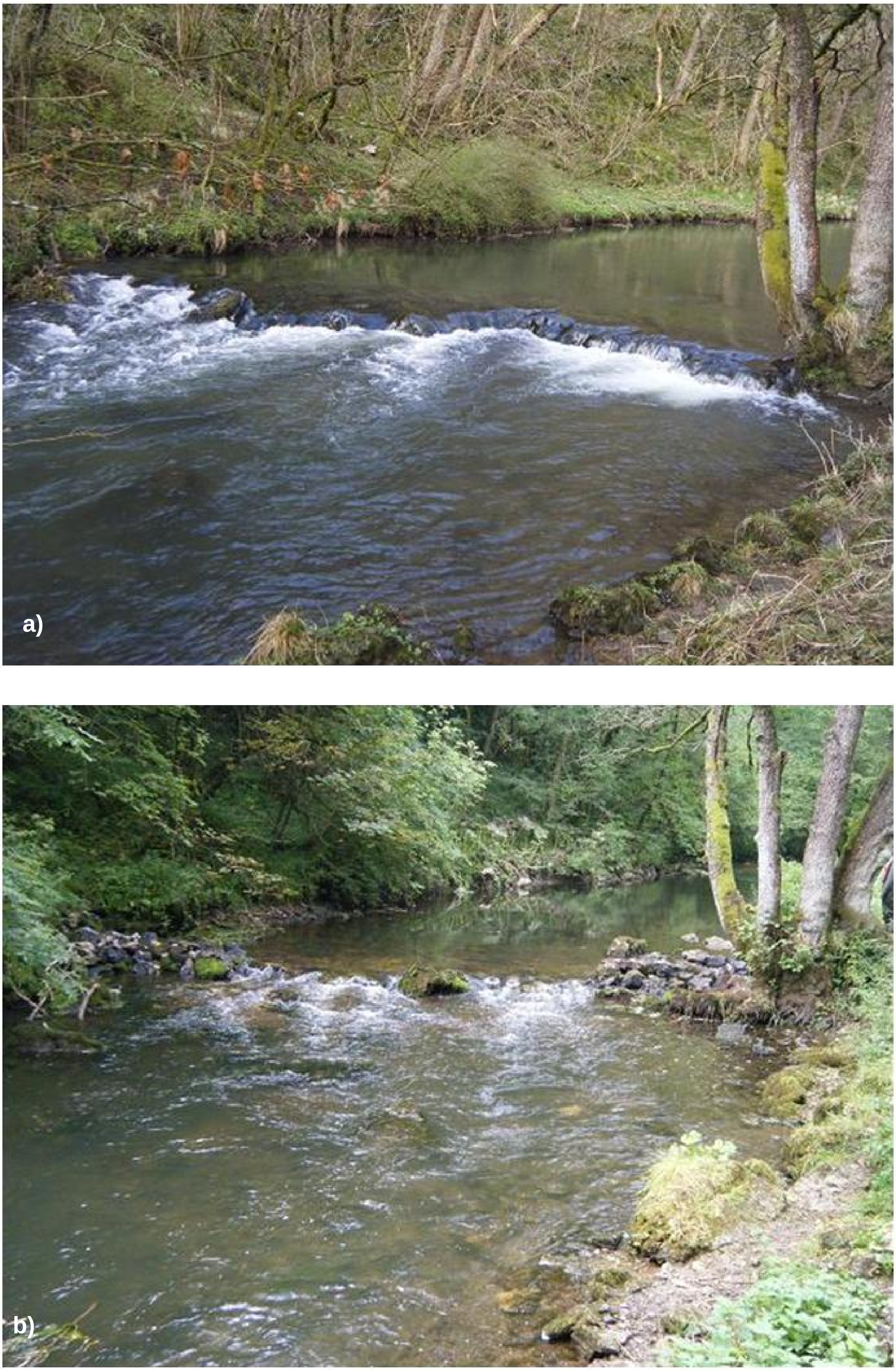


Figure 3.22. Dovedale a) before b) after weir removal.

Table 3.19. Fisheries survey site details for the River Manifold and the River Dove summer 2010 and 2011.

Site identifier/NGR	River Name	Survey date	Length/mean width/area	Survey method and gear
Hartington u/s SK 125 595	Dove	27/07/2010 28/09/2010 27/07/2011	100 m/7 m/ 705 m^2 100 m/7 m/ 705 m^2 94 m/7 m/ 610 m^2	Generator Quantitative
Hartington d/s SK 124 596	Dove	27/07/2010 28/09/2010 27/07/2011	84 m/7 m/ 585 m^2 100 m/7 m/ 705 m^2 100 m/6 m/ 664 m^2	Generator Quantitative
Dovedale u/s SK 143 526	Dove	01/07/2010 08/09/2010 06/07/2011	80 m/8 m/ 664 m^2 80 m/8 m/ 664 m^2 100 m/9 m/ 860 m^2	Generator Quantitative
Dovedale d/s SK 144 526	Dove	01/07/2010 08/09/2010 06/07/2011	89 m/8 m/ 710 m^2 89 m/8 m/ 710 m^2 95 m/8 m/ 741 m^2	Generator Quantitative
Hardings Booth SK 069 644	Manifold	15/04/2010 22/09/2011	95 m/3 m/ 285 m^2 60 m/3 m/ 204 m	Generator Semi-Quantitative
Brund SK 099 612	Manifold	07/04/2010 23/09/2011	105 m/6 m/ 63 m^2 78 m/7 m/ 614 m^2	Generator Quantitative
Wetton Mill SK 095 560	Manifold	13/04/2010 23/09/2011	100 m/12 m/ 1200 m^2 69 m/11 m/ 750 m^2	Generator Semi-Quantitative
Wolfescote Dale SK 131 583	Dove	07/04/2010 22/09/2011	180 m/10 m/ 1800 m^2 64 m/7 m/ 460 m^2	Generator Semi-Quantitative
Ludwell Farm SK 117 630	Dove	07/04/2010 22/09/2011	170 m/4 m/ 680 m^2 60 m/4 m/ 252 m^2	Generator Semi-Quantitative
Reynard's Cave SK 144 525	Dove	15/05/2010 06/07/2011	100 m/12 m/ 1240 m^2 105 m/12 m/ 1292 m^2	Generator. Quantitative

3.7.3. Results

HABSCORE analysis; Hartington

Raw depth data from HABSCORE was used to quantify some of the changes in the channel before and after channel narrowing at Hartington (Figure 3.24), the habitat quality score (HQS Table 3.24) taken from the HABSCORE output, can also be used to quantify habitat improvement for brown trout at Hartington. The removal of Hartington weir resulted in a noticeable reduction the depth of the weir pools both upstream and downstream of the weir (Figure 3.24). HQS upstream of the weir increased after the removal of the weir for 0+, >1+ (<20cm) and >1+ (>20cm) (Table 3.24). The reduction in depth may have resulted in change in flow types that are more suitable for all life stages of brown trout. HQS downstream increased for 0+ and >1+ (<20cm) after weir removal, but decreased for >1+ (>20cm). This was possibly because larger trout that prefer deep pools.

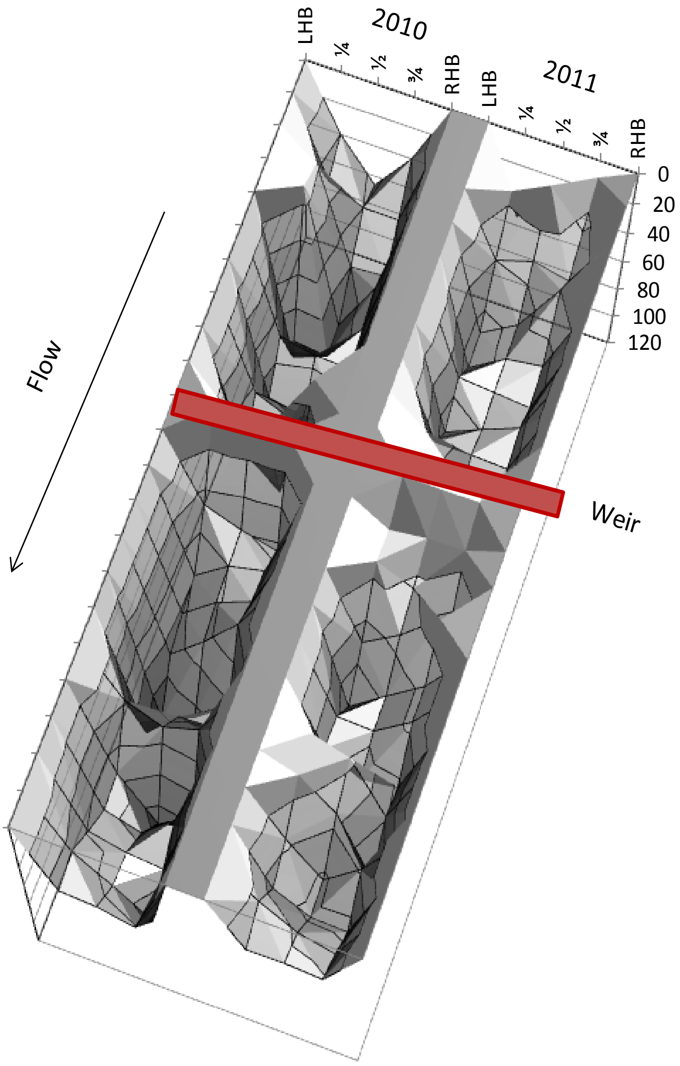


Figure 3.23. Changes in channel depth recorded during HABSCORE at Hartington.

Observed densities of 0+ trout at Hartington in 2010 were lower than predicted by the Habitat Quality Score (HQS) both downstream and upstream of the weir, suggesting poorer populations than expected. HUI upper CLs were >1 for Hartington d/s and Hartington u/s, therefore observed populations were not significantly lower than expected (Table 3.24). Densities of 0+ trout at Hartington in 2011 were also lower than predicted by the HQS both downstream and upstream of the weir suggesting poorer populations than expected, but not significantly lower than expected (Table 3.24). Observed densities of >1+ (<20 cm) trout at Hartington in 2010 were higher than predicted by the Habitat Quality Score (HQS) at both Hartington d/s and Hartington u/s suggesting better populations than expected. HUI lower CL was >1 for Hartington d/s and therefore densities were significantly higher at this site in 2010 (Table 3.24). Observed densities of >1+ (<20 cm) trout at Hartington in 2011 were higher than predicted by the Habitat Quality Score (HQS) at both Hartington d/s and Hartington u/s suggesting better populations than expected. However, HUI lower CL was <1 for both sites and therefore densities were not significantly higher in 2011 (Table 3.24). Observed densities of >1+ (>20 cm) trout at Hartington in 2010 were higher than predicted by the Habitat Quality Score (HQS) at both Hartington d/s and Hartington u/s suggesting better populations than expected. HUI lower CL was only >1 for Hartington u/s and therefore densities were only significantly higher at this site in 2010 (Table 3.24). Observed densities of >1+ (<20 cm) trout at Hartington in 2011 were higher than predicted by the Habitat Quality Score (HQS) at both Hartington d/s and Hartington u/s suggesting better populations than expected. However, HUI lower CL was <1 for both sites and therefore densities were not significantly higher in 2011 (Table 3.24).

HABSCORE analysis; Dovedale

The removal of Dovedale weir resulted in a noticeable reduction in depth of the both the upstream and downstream sections of the weir (Figure 3.25). HQS upstream of the weir slightly decreased after the removal of the weir for 0+, >1+ (<20cm) and >1+ (>20cm) (Table 3.24). The HQS downstream remained similar values 0+, >1+ (<20cm) and >1+ (>20cm) (Table 3.24).

Observed densities of 0+ trout at Dovedale in 2010 were higher than predicted by the Habitat Quality Score (HQS) at Dovedale d/s and Dovedale u/s suggesting a better population than expected. HUI lower CLs were <1 for both sites and therefore, observed populations were not significantly higher than expected (Table 3.24). Observed densities of 0+ trout at Dovedale d/s in 2011 were slightly lower than predicted by HQS, however the HUI upper CL was >1 and therefore densities were not significantly lower (Table 3.24). Densities of 0+ trout at Dovedale u/s in 2011 were higher than predicted by HQS, however, HUI lower CLs is <1 and therefore not significantly higher than predicted (Table 3.24). Observed densities of >1+ (<20 cm) trout at Dovedale in 2010 were higher than predicted by the Habitat Quality Score (HQS) at Dovedale d/s and Dovedale u/s suggesting a better population than expected. HUI lower CLs were <1 for both sites and therefore, observed populations were not significantly higher than expected (Table 3.24). Observed densities of >1+ (<20 cm) trout at Dovedale in 2011 were higher than predicted by the Habitat Quality Score (HQS) at Dovedale d/s and Dovedale u/s suggesting a better population than expected. HUI lower CLs were <1 for both sites and therefore, observed populations were not significantly higher than expected (Table 3.24). Observed densities of >1+ (>20 cm) trout at Dovedale in 2010 were higher than predicted by the Habitat Quality Score (HQS) at Dovedale d/s and Dovedale u/s suggesting a better population than expected. HUI lower CLs were >1 for both sites and therefore, observed populations were significantly higher than expected (Table 3.24). Observed densities of >1+ (>20 cm) trout at Dovedale in 2011 were higher than predicted by the Habitat Quality Score (HQS) at Dovedale d/s and Dovedale u/s suggesting a better population than expected. HUI lower CLs were <1 for both sites and therefore, observed populations were not significantly higher than expected (Table 3.24).

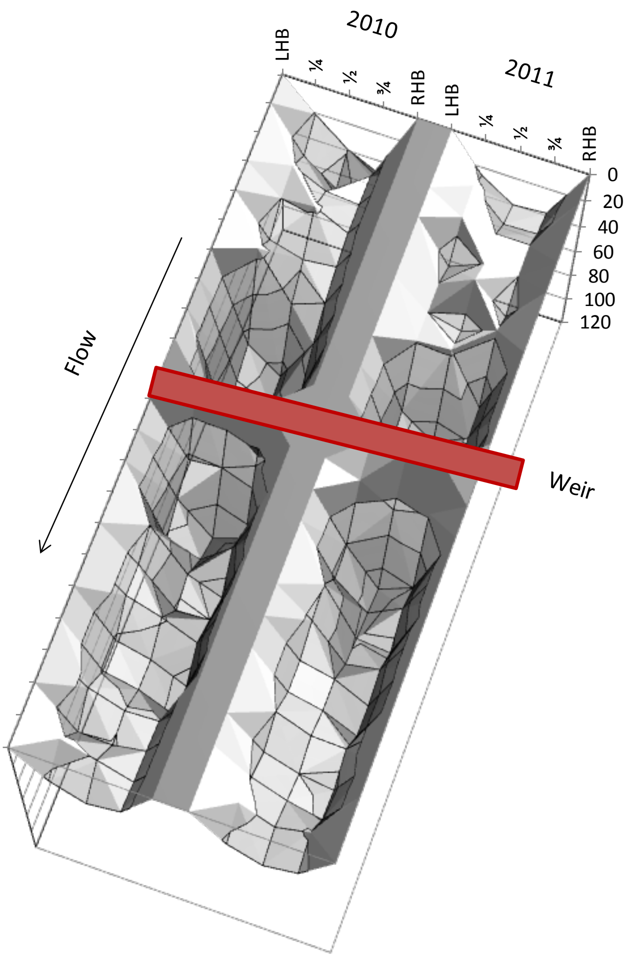


Figure 3.24. Changes in channel depth recorded during HABSCORE at Dovedale.

Functional habitat pre and post weir removal

Weir removals at both Hartington and Dovedale did not seem to influence the geomorphology of the river channel in the surrounding area, but in the immediate area they created shallow riffles, improving habitat through pool riffle enhancement within the river. Weir removal did not alter the substrate at either site; at Hartington silt was the dominant substrate whilst boulders were the dominant substrate at Dovedale in both 2010 and 2011 surveys.

Status of fish populations at Hartington and Dovedale on the River Dove in 2010 and 2011.

Species composition up and downstream at Hartington comprised four species (brown trout, bullhead, grayling and lamprey) in 2010 (pre), 2010 (post) and 2011 (Table 3.20). Brown trout dominated in the downstream section at Hartington in 2010 (pre), 2010 (post) and 2011 surveys followed by bullhead dominant in 2010 (pre) and 2010 (post) but grayling was the second most important species in the 2011 survey (Table 3.20). The number of brown trout and bullhead caught after weir removal was lower than numbers caught before the weir removal. Brown trout and bullhead numbers caught declined further in the 2011 survey (Table 3.20). Grayling numbers were similar in the downstream reach at Hartington pre and post removal in 2010 but increased in 2011 (Table 3.20). Brown trout dominated catches in the upstream section at Hartington in the 2010 (pre), 2010 (post) and 2011 surveys followed by lamprey in 2010 (pre) and 2010 (post) but grayling in 2011 surveys (Table 3.20). The number of brown trout caught in 2010 after the weir removal was lower prior to the removal, but numbers of brown trout caught were higher in 2011 than either 2010 survey (Table 3.20). Grayling numbers were lower after the weir removal in 2010 than before removal but much higher in 2011 (Table 3.20).

Species composition at the downstream site in Dovedale was similar across the three survey samples (Table 3.20). Bullhead was dominant at the downstream site at Dovedale in 2010 (pre) and 2011, but brown trout dominated in the post weir removal survey in 2010 (Table 3.20). Low numbers of grayling were present in both the pre- and post weir removal surveys in 2010 but were absent in 2011; lamprey were only present in pre survey in 2010 (Table 3.20). Species composition at the upstream site Dovedale was similar in all surveys, with brown trout dominant in the pre weir removal survey in 2010, but bullhead dominated in the post 2010 and 2011 surveys followed by brown trout; low numbers of grayling were also present at all three surveys (Table 3.20). Lamprey was present in low numbers at the upstream site in Dovedale in both 2010 surveys but were absent from 2011 catches; one minnow was caught in the 2010 (post) survey, but they were absent in other surveys (Table 3.20).

Density estimates

The density and abundance of 0+ and >1+ brown trout in the Dove at Hartington in July 2010 before the weir removal were similar in reaches both downstream and upstream of the weir; they were classified as poor (Class E) and fair/average (Class C), respectively (Table 3.21; Figure 3.26). In September 2010 catches, directly after the Hartington weir removal abundance classifications of 0+ and >1+ trout were similar to the pre removal estimates from the July 2010 surveys (Table 3.22; Figure 3.26). In the July 2011 surveys at Hartington, one year after the weir removal, 0+ and >1+ brown trout densities downstream of where the weir previously was located were the same as catches in the previous year (Table 3.23; Figure 3.26), however, catches of >1+ trout upstream of where the weir was located were classified as fair/average (Class C), the same as July and September 2010, whereas the abundance class of 0+ trout increased to fair (Class D) (Table 3.22; Figure 3.26).

Density estimates and abundance classification of 0+ brown trout catches in July 2010, before the weir removal in Dovedale were Class D downstream of the weir and Class C upstream; abundance of >1+ trout was good (Class B) both downstream and upstream of the weir (Table 3.21; Figure 3.26). In September 2010 catches, directly after Dovedale weir removal abundance classifications of 0+ were the same as during the pre July 2010 surveys before the weir removal >1+ trout abundance decreased from good (Class B) to fair/average (Class C) (Table 3.22; Figure 3.26). In July 2011, one year after the weir removal, 0+ trout classification for downstream and upstream sites was the same as in the previous year, whereas >1+ classification remained at Class C downstream of where the previous weir was located but abundance of >1+ trout upstream increased to Class B (Table 3.23; Figure 3.26).

Table 3.20. Number of fish of different species captured at Hartington and Dovedale in the River Dove in July 2010 & 2011.

Species	Hartington d/s			Hartington u/s			Dovedale d/s			Dovedale u/s		
	2010 (pre)	2010 (post)	2011	2010 (pre)	2010 (post)	2011	2010 (pre)	2010 (post)	2011	2010 (pre)	2010 (post)	2011
Brown trout	78	90	49	53	44	65	127	115	83	223	175	109
Bullhead	24	14	10	8	9	17	173	107	104	64	373	152
Grayling	6	6	23	7	2	35	1	2	0	4	4	3
Lamprey	21	1	14	61	13	3	15	0	0	5	2	0
Minnow	0	0	0	0	0	0	0	0	0	0	1	0
Total Species	4	4	4	4	4	4	4	3	2	4	5	3

Table 3.21. Total population estimate (N), population density (D) (± 95% C.L.) and abundance classification of trout derived from fisheries surveys at Hartington and Dovedale before weir removals July 2010 (density of fish given as numbers per 100m²). Details of derivation of estimates are provided in the text.

Site No	Total Population		Population Density		Abundance Class	
	0+	≥1+	0+	≥1+	0+	≥1+
Hartington d/s	2±0	76±1	0.29±0	10.86±0.18	E	C
Hartington u/s	4±2	49±2	0.68±0.26	8.33±0.33	E	C
Dovedale d/s	25±11	112±6	3.51±1.49	15.73±0.89	D	B
Dovedale u/s	106±10	126±3	16.56±1.56	19.69±0.45	C	B

Table 3.22. Total population estimate (N), population density (D) (± 95% C.L.) and abundance classification of trout derived from fisheries surveys at Hartington and Dovedale after weir removals September 2010 (density of fish given as numbers per 100m²). Details of derivation of estimates are provided in the text.

Site No	Total Population		Population Density		Abundance Class	
	0+	≥1+	0+	≥1+	0+	≥1+
Hartington d/s	11±0	82±5	1.56±0.06	11.6±0.77	E	C
Hartington u/s	5±0	40±3	0.88±0.06	7.02±0.58	E	C
Dovedale d/s	45±5	79±11	5.77±0.6	10.13±1.36	D	C
Dovedale u/s	113±49	105±2	11.41±4.97	10.61±0.18	C	C

Table 3.23. Total population estimate (N), population density (D) (± 95% C.L.) and abundance classification of trout derived from fisheries surveys Hartington and Dovedale after weir removals July 2011 (density of fish given as numbers per 100m²). Details of derivation of estimates are provided in the text.

Site No	Total Population		Population Density		Abundance Class	
	0+	≥1+	0+	≥1+	0+	≥1+
Hartington d/s	8±0	42±3	1.33±0.08	6.98±0.50	E	C
Hartington u/s	18±4	48±1	2.73±0.64	7.27±0.17	D	C
Dovedale d/s	28±70	59±9	3.89±0.93	8.19±0.51	D	C
Dovedale u/s	53±3	59±4	12.33±0.78	13.72±0.99	C	B

European Fish Index Calculations

The EFI + database calculate an increase in ecological class boundaries at Hartington u/s, in 2010 Fish Index Cass was 3 (Fish Index 0.73) and in 2011 Fish Index Class was 2 (Fish Index 0.84). Ecological class boundaries were unchanged (Fish Index Cass 2) when comparing before and after weir removal at Hartington d/s (2010 Fish Index 0.80 & 2011 Fish Index 0.82), Dovedale d/s (2010 Fish Index 0.87 & 2011 Fish Index 0.87), and Dovedale u/s (2010 Fish Index 0.86 & 2011 Fish Index 0.87).

Length Frequency; Hartington

Only two 0+ brown trout were caught downstream of Hartington weir in the 2010 survey before the weir removal, both at 82 mm; >1+ trout were caught in the size range 107–

395 mm (Figure 3.27). In the 2010 post weir removal surveys, 0+ trout were captured in the size range 72–107 mm, where as >1+ trout were caught between 122 and 414 mm long. In 2011 0+ trout were caught in the size range 51-75 mm, while >1+ trout between 103 and 387 mm long (Figure 3.27).

0+ brown trout caught upstream of Hartington weir in 2010 before the weir removal were in the size range 59-83 mm and >1+ trout in the size range 110–405 mm (Figure 3.28). In 2010 upstream surveys, post weir removal, 0+ trout captured were in the size range 75–104 mm, where as >1+ trout were between 127 and 410 mm long. In the 2011 surveys 0+ trout caught upstream were in the size range 54-98 mm, and >1+ trout between 115 and 405 mm long (Figure 3.28).

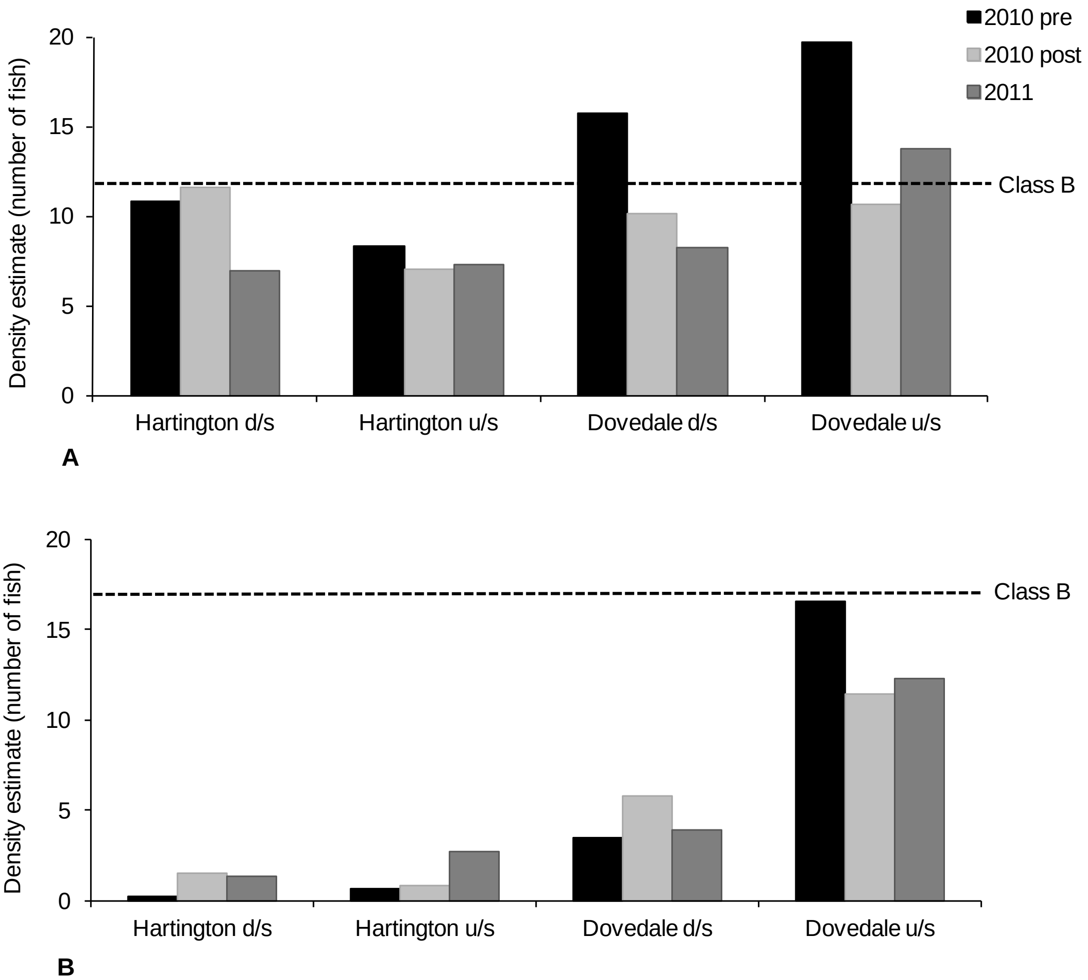


Figure 3.25. Density estimates of >1+ trout (A) and 0+ trout (B) downstream and upstream of Hartington and Dovedale weirs, on the River Dove before weir removal (2010 pre), directly after weir removal (2010 post) and one year after weir removal (2011), Class B and above indicates good population density.

Length Frequency; Dovedale

Only two 0+ brown trout (41 and 73 mm) were caught downstream of Dovedale weir in the 2010 fisheries surveys, before the weir removal, >1+ trout were in the size range 95–353 mm (Figure 3.29). In 2010 surveys, post weir removal 0+ trout captured were in the size range 55–118 mm, where as >1+ trout were between 124 and 355 mm long. In 2011, 0+ trout were in the size range 51-795 mm, where as >1+ trout caught were between 106 and 321 mm long (Figure 3.29). 0+ brown trout upstream of Dovedale weir in the 2010 fisheries surveys before weir removal were between 34 and 77 mm long, >1+ trout in the size range 93–387 mm (Figure 3.30). In the 2010 upstream surveys post weir removal, 0+ trout captured were in the size range 38–118 mm, where as >1+ trout were between 120 and 390 mm long. In the 2011 surveys 0+ trout caught upstream were in the size range 40-86 mm, where as >1+ trout were between 107 and 365 mm long (Figure 3.30).

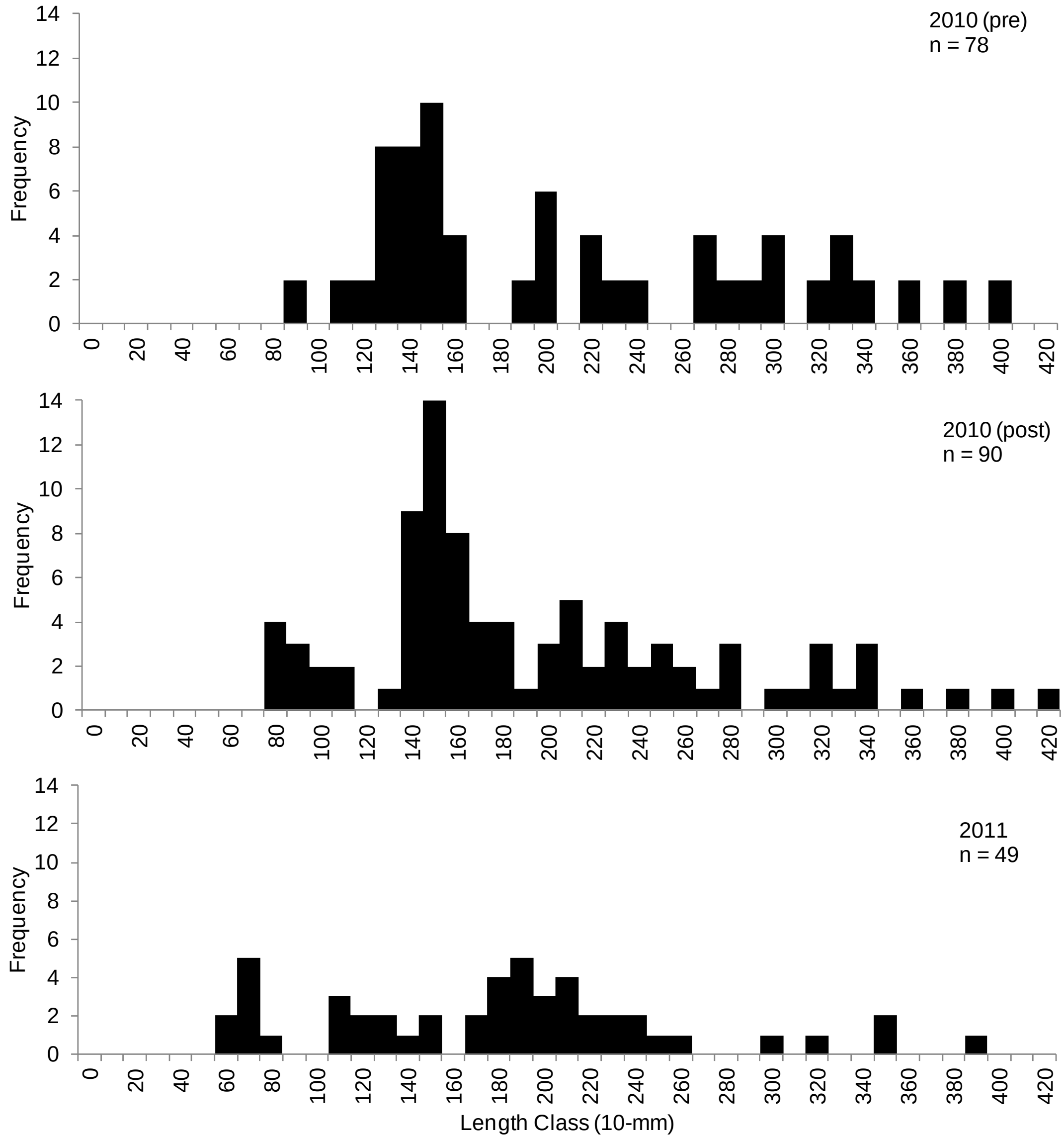


Figure 3.26. Length distribution of brown trout at downstream of Hartington weir, river Dove, in 2010 (pre weir removal), 2010 (post weir removal) and 2011 (one year after weir removal).

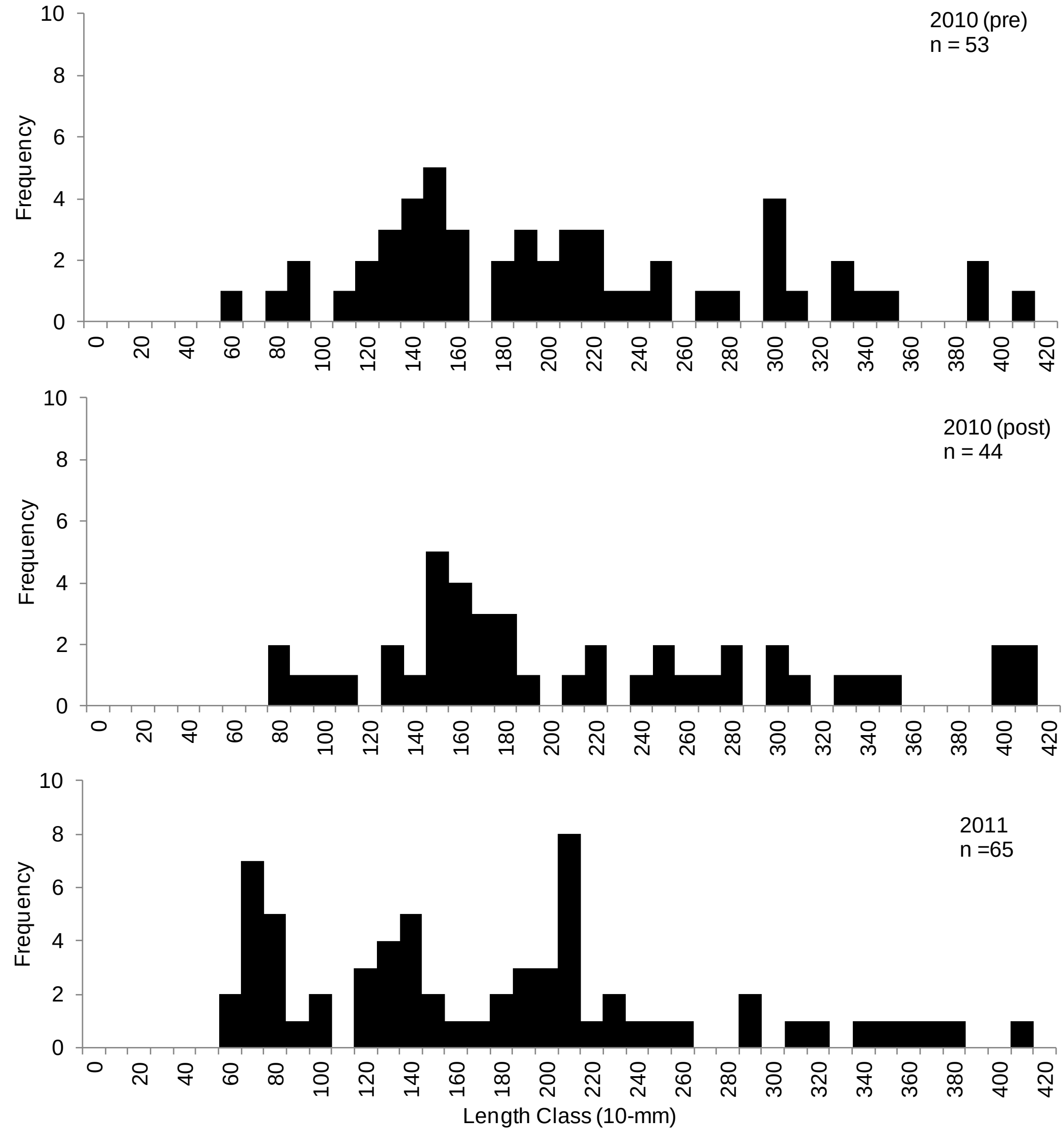


Figure 3.27. Length distribution of brown trout at upstream of Hartington weir, river Dove, in 2010 (pre weir removal), 2010 (post weir removal) and 2011 (one year after weir removal).

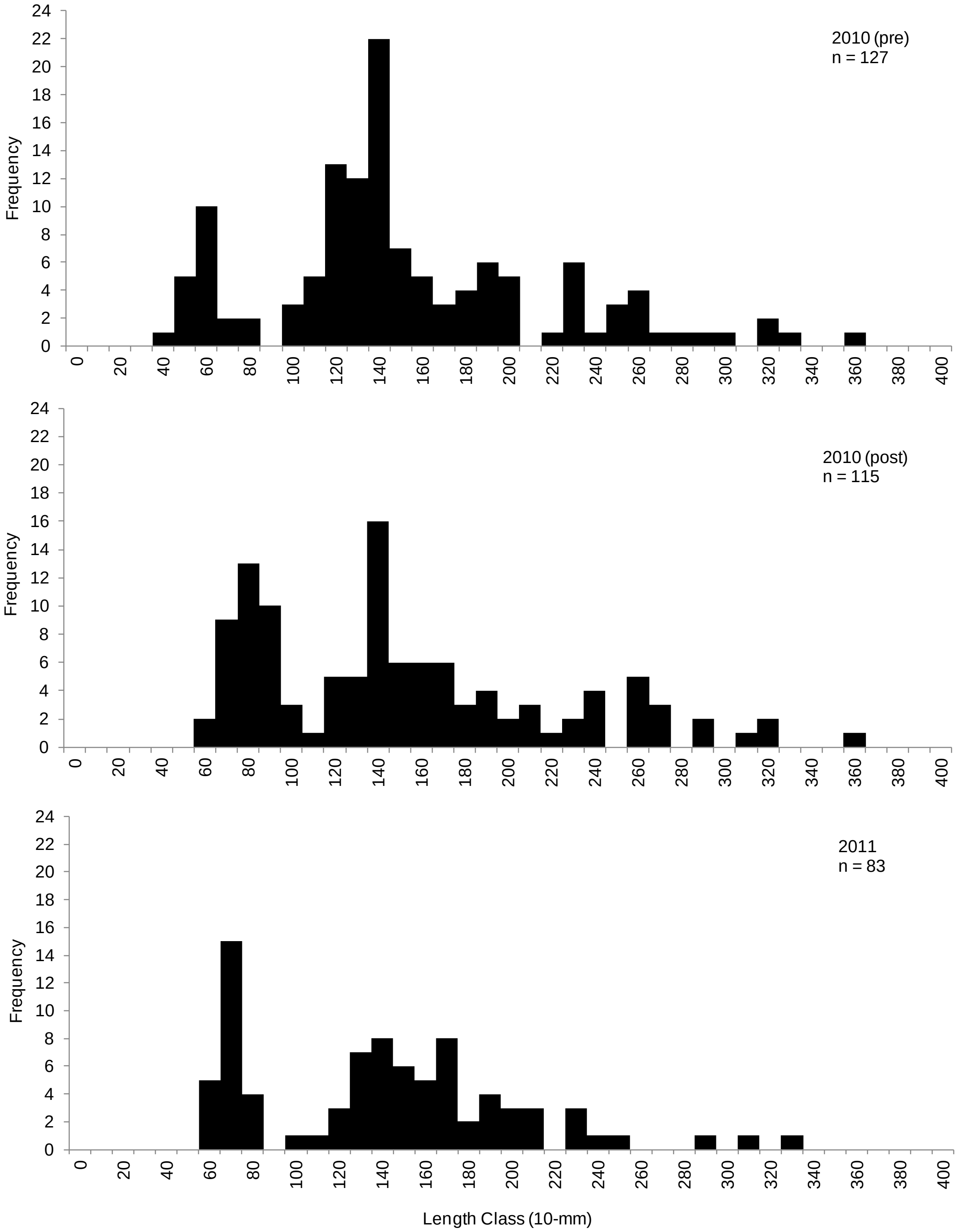


Figure 3.28. Length distribution of brown trout at downstream of Dovedale weir, river Dove, in 2010 (pre weir removal), 2010 (post weir removal) and 2011 (one year after weir removal).

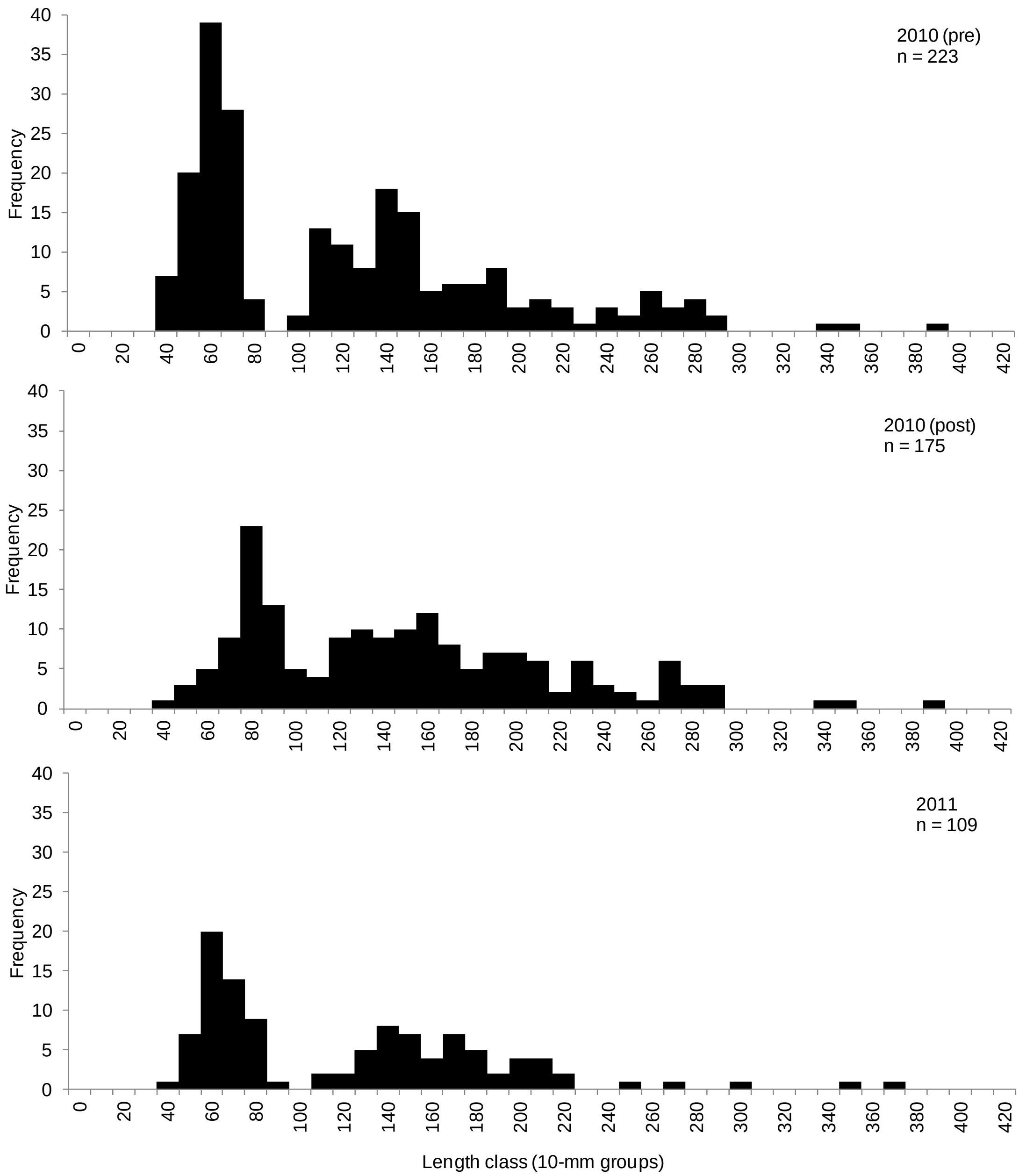


Figure 3.29. Length distribution of brown trout at upstream of Dovedale weir, river Dove, in 2010 (pre weir removal), 2010 (post weir removal) and 2011 (one year after weir removal).

Impact assessment and resource calculation

A BACI design (Section 3.2.4, Equation 3.3) was used to assess the effects of small weir removal on 0+ and >1+ brown trout in the River Dove at Hartington and Dovedale. The impact assessment was assessed both in terms of confidence limits of the mean difference in 0+ and >1+ brown trout density and by means of a t-test. At Hartington the mean change in 0+ brown trout density between the impact sites (M = 0.42, SE = 0.34) and the control sites (M = 0.68, SE = 0.28), t (3) = 0.47, was -0.36 ± 1.52 fish per 100 m^2, the 95% confidence limits (-1.16 – 1.88) encloses zero and the difference was not significant p= 0.67 (p > 0.05), r = 0.26). At Hartington the mean change in >1+ brown trout density between the impact sites (M = -0.40, SE = 0.22) and the control sites (M = 0.70, SE = 0.28), t (3) = 4.85, was -0.12 ± 1.50 fish per 100 m², the 95% confidence limits (-1.38 – 1.62) encloses zero and the difference was significant p= 0.02 ($p > 0.05$), r = 0.94). At Dovedale the mean change in 0+ brown trout density between the impact sites (M = -0.23, SE = 0.33) and the control sites (M = 0.68, SE = 0.28), t (3) = 1.68, was -0.19 ± 1.51 fish per 100 m², the 95% confidence limits (-1.32 – 1.7) encloses zero and the difference was not significant p= 0.20 ($p > 0.05$, r = 0.70). At Dovedale the mean change in >1+ brown trout density between the impact sites (M = -0.74, SE = 0.09) and the control sites (M = 0.70, SE = 0.28), t (3) = 3.88, was 0.09 ± 1.47 fish per 100 m², the 95% confidence limits (-1.38 – 1.56) encloses zero and the difference was significant p= 0.03 ($p > 0.05$, r = 0.91). A resource calculation cannot be calculated because there is insufficient temporal data and therefore temporal variance cannot be estimated.

The standard errors for the mean difference in 0+ and >1+ brown trout at impact sites was large and therefore a large amount of variability between the means of different samples. The standard errors for the mean difference in 0+ and >1+ at control sites was low and therefore a low amount of variability between the means of different samples. There were 4 sample for the impact sites and 6 samples for the control, it is more than likely the large number of sample sites that resulted in low variance between the means of the control site. Overall, sampling size was too small to overcome variability between the mean density of 0+ and >1+ brown trout at the impact sites and for 0+ brown trout at control sites, a larger number of samples, across a large number of years would reduce standard error and reduce variability between the means.

3.7.4. Discussion

It is known that fragmentation of rivers by weirs can restrict flow and reduce biotic integrity by obstructing the movement of fish (Dynesius & Nilsson 1994; McLaughlin *et al*. 2006; Harford & McLaughlin 2007; Alexandre & Almeida 2009). This study demonstrates that the removal of small obstacles at both Hartington and Dovedale altered the habitat immediately upstream of the weir by reducing the water depth, but only at a local scale. The removal of the weir itself created a riffle, increasing flow diversity at a meso-scale within the channel and was a much needed enhancement at Hartington where the flow and habitat were very uniform. Various other studies have found that small physical obstacles cause changes in habitat at a local scale (Hagglund & Sjoberg 1999; Dodd *et al*. 2003; Santucci *et al.* 2005; Poulet 2007; Alexandre & Almeida 2009) and therefore, it would be expected that the removal of these structures will only benefit at a local scale, although if they are a serious.

The outputs of the fisheries surveys on the River Dove in 2010 and 2011 provide an insight into the status of the fish populations following the two weir removals. At Hartington species composition was the same at all sites before and after weir removal, the EFI calculated an increase in ecological boundary at Hartington u/s after the removal of the weir, where no change was noted at Hartington d/s. Fish composition varied across all sites before and after weir removal at Dovedale, but only as a result of less dominant species, suggesting that the weir removals did not significantly improve the fish community, this was supported by no change in the ecological boundary calculated by the EFI. This may be because the majority of native fish within an ecosystem will adapt to natural fluctuations within the environment (Gehrke & Harris 2001) and can therefore adapt to small, local changes resulting from small weir removals.

When assessing total numbers of the dominant species brown trout at Dovedale before and after weir removal, there was a small decrease of brown trout numbers in the downstream site whereas there was a large decrease in the upstream site. When assessing age structure and population density of brown trout at Dovedale there was a decreased in 0+ and ≥1+ individuals after the weir removal, with the exception of >1+(>20 cm) in the downstream section. However, the decrease in fish abundance was not enough change the abundance classification, except for >1+ trout in the downstream section that changed from B to C class (Table 3.21, 3.22 & 3.23). This suggests that trout population were not enhanced by the weir removal. When assessing total numbers of brown trout at Hartington before and after weir removal, there was a small decrease of brown trout numbers in both the upstream and downstream sites. When assessing age structure and population density of brown trout at Hartington there was a decreased in ≥1+ individuals and a small increase in 0+ fish after the weir removal at both sites. However, there were no changes in the abundance classification at the Dovedale sites except for >1+ trout in the downstream section that reduce to class C from B. Overall, there were no obvious improvement in brown trout numbers, supported through the BACI design. The control sites used in the BACI analysis enabled some spatial variability to be overcome, but the monitoring design was limited temporally (one year pre and one year post) and it is recommended that further monitoring for an additional number of years is undertaken to overcome temporal variability. The duration of monitoring needed to detect a change is particularly important in any type of monitoring study design, to overcome natural variability and identify a possible change as a result of the impact, in this case weir removals. Unfortunately, a resources calculation could not be undertaken to work out

Table 3.24. HABSCORE outputs for 0+ trout at four sites on the River Dove for 2010 and 2011 catches. (Note: Shaded area represents sites where the observed population was significantly higher (HUI lower CL column) or lower (HUI upper CL column) than would be expected under pristine conditions).

Site Code	Age category	Observed number	Observed density	HQS (density)	HQS lower CL	HQS upper CL	HUI	HUI lower CL	HUI upper CL
Hartington d/s	0+								
	2010	3	0.43	1.18	0.31	4.50	0.37	0.05	2.46
	2011	8	1.32	2.02	0.53	7.67	0.66	0.10	4.38
	>1+ (<20 cm)								
	2010	55	7.93	0.99	0.21	4.76	7.99	1.21	52.72
	2011	24	3.97	1.43	0.32	6.42	2.78	0.45	17.38
	>1+ (>20 cm)								
	2010	34	4.90	2.32	0.75	7.12	2.11	0.69	6.50
	2011	17	2.81	1.59	0.52	4.87	1.76	0.58	5.41
Hartington u/s	0+								
	2010	4	0.74	1.09	0.28	4.28	0.68	0.10	4.76
	2011	17	2.49	2.87	0.76	10.91	0.87	0.13	5.80
	>1+ (<20 cm)								
	2010	25	4.62	1.08	0.22	5.36	4.26	0.63	28.85
	2011	24	3.52	1.72	0.38	7.81	2.04	0.32	12.85
	>1+ (>20 cm)								
	2010	24	4.43	1.12	0.36	3.47	3.97	1.27	12.40
	2011	24	3.52	2.01	0.66	6.17	4.79	0.57	5.36
Dovedale d/s	0+								
	2010	35	5.07	3.83	1.01	14.48	1.32	0.20	8.98
	2011	28	3.75	3.76	0.99	14.18	1.00	0.15	6.69
	>1+ (<20 cm)								
	2010	86	12.45	2.68	0.61	11.79	4.64	0.76	28.52
	2011	48	6.43	3.41	0.79	14.75	1.89	0.31	11.44

	>1+ (>20 cm) 2010 2011	 23 48	 3.33 6.43	 0.70 3.41	 0.22 0.79	 2.16 14.75	 4.79 1.89	 1.54 0.31	 14.89 11.44

Site Code	Age category	Observed number	Observed density	HQS (density)	HQS lower CL	HQS upper CL	HUI	HUI lower CL	HUI upper CL
Dovedale u/s	0+ 2010 2011	 117 53	 16.42 6.70	 3.57 3.92	 0.92 1.03	 13.75 14.86	 4.60 1.71	 0.68 0.26	 31.12 11.42
	>1+ (<20 cm) 2010 2011	 100 49	 14.04 6.19	 2.41 2.34	 0.54 0.55	 10.76 9.93	 5.82 2.65	 0.94 0.44	 36.14 15.83
	>1+ (>20 cm) 2010 2011	 31 10	 4.35 1.26	 0.95 0.72	 0.31 0.23	 2.89 2.23	 4.57 1.75	 1.51 0.57	 13.88 5.44

the duration and intensity of monitoring needed to overcome natural variability because of insufficient temporal data.

Annual stocking of ≥1+ brown trout, primarily for angling, by the Environment Agency on the River Dove is a regular occurrence and consequently there is the possibility these stockings have influenced the 2010 and 2011 fish catch data for both Dovedale and Hartington, further influencing the true representation of the local brown trout population. The HABSCORE outputs indicate that the HQS is lower than the actual observed density for ≥1+ (<20 cm) and ≥1+ (>20 cm) trout, suggesting the habitat will not support a high number of trout. Subsequently, the event of stocking ≥1+ trout and the low number of 0+ trout only highlights that there must be additional pressures on the River Dove that suppress trout from having a self-sustaining population. The bottleneck in habitat limiting recruitment (feeding, breeding or refuge) needs to be identified and remediated for both 0+ and ≥1+ trout to enhance the population and encourage it to be self sustaining. In addition, the river suffers from water quality problems in the upstream reaches above Hartington, as a result of dairy farm effluent and until this is solved stocking should not be seen as a measure to overcome this problem. Furthermore habitat restoration will be ineffective if water quality issues are not primarily addressed.

When considering small physical obstacles, there is some discrepancy in opinions to overcome the importance of their removals, especially as they do not always cause permanent obstruction to longitudinal movement. Hartington and Dovedale weirs do not always act as barriers, nevertheless it is cumulative effects of the high number of weirs on the River Dove that potentially cause problems difficulties with fish migration, further increasing pressure on the life-cycle of fish species and consequently the local structure patterns of its fish assemblages (Welcomme *et al*. 2006). The removal of the majority of these weirs would reduce fragmentation of the river system, enabling most fish to complete each life stage that requires longitudinal and lateral connectivity within a river. This study recognises that small weir removal will only modify the river at a local scale, thus is an ideal rehabilitation solution to overcome small physical obstructions, with little impact on the surrounding river channel. Conversely, it is important to realise that the permeability of many small weirs to fish, especially boulder weirs, under different flow conditions will not always act as an obstruction, many of them have been found to contribute to habitat diversity within the channel (Hvidsten & Johnsen 1992; Linlokken 1997). It is imperative that a weir is assessed in terms of permeability and that this is evaluated in addition to the habitat diversity it may, or may not contribute to the ecosystem before its removal is determined. Catchment scale planning within any rehabilitation is essential but with weir removals it is a vital characteristic that should not be overlooked (Fagan 2002; Vaughan *et al*. 2009).

Recommendations

Knowledge of the effects of small obstacles on fish population fragmentation within a catchment is important to understand. This should be a primary action to assess if their removal will benefit fish communities and other freshwater biota. The removal of small physical obstacles is becoming a regular occurrence especially as it meets the requirements of the WFD; however, literature on their success is limited. **As a consequence, further research is needed to understand habitat and fish response to the removal of small obstacles as a restoration measure.**

Overall, this study suggests that weir removal is a suitable rehabilitation method to aid longitudinal connectivity for fish. However, there is a need for further temporal monitoring to be confident with this conclusion and for that reason **it is recommended that additional post monitoring is necessary to strengthen the impact assessment, to overcome natural variability** and enable a resource calculation to be calculated for to determine the scale of this monitoring. Further investigation into the effects of pollution upstream of Hartington is needed since other underlying problems cannot be addressed while there is a water quality problem.

3.8. Artificial riffle as a rehabilitation technique to improve spawning habitat for brown/sea trout

3.8.1. Introduction

The River Stiffkey is a chalk stream located in North Norfolk (Figure 3.31). In recent years the production of anadromous sea trout has been almost non-existent within the River Stiffkey. This loss of habitat, in particular spawning habitat, result of decades of changes in faming and land management practices and flood prevention practise, especially channel deepening and straightening. Change in land use and agricultural practise has increased the level of silt entering rivers and there is concern that in some parts of England and Wales increased sediment loads in water courses may be having a significant impact on trout stocks and are therefore understood to be the main factors constraining trout production on the River Stiffkey (Pawson 2008), especially as the water quality has been proven to be good to excellent (Pawson 2008). There have been no records of stocking since 1995 so the fishery evidently relies on natural production, but natural trout reproduction is probably limited as a consequence of poor spawning habitat (Pawson 2008).

The principle river rehabilitation technique that has taken place on the River Stiffkey is gravel reinstatement, in an attempt to reintroduce suitable riffle spawning habitat for brown trout and to attract sea trout (Figure 3.32) at four locations within various river sections identified by the Wild Trout Trust (WTT). Deflectors have also been used in conjunction with gravel augmentation to accelerate the flow, creating localized scouring that de-silts the river bed and creates and maintains pools (Downs & Thorne 1998). The instalment of riffles is part of larger, network scale rehabilitation of sea trout habitats throughout Norfolk and it intends to improve sustainable management of fish species moving between the North Sea and freshwater systems. The project works directly with the Environment Agency and the Wild Trout Trust (WTT) and is integral to The Interreg IVB North Sea Region Programme run under the auspices of the River Trust.

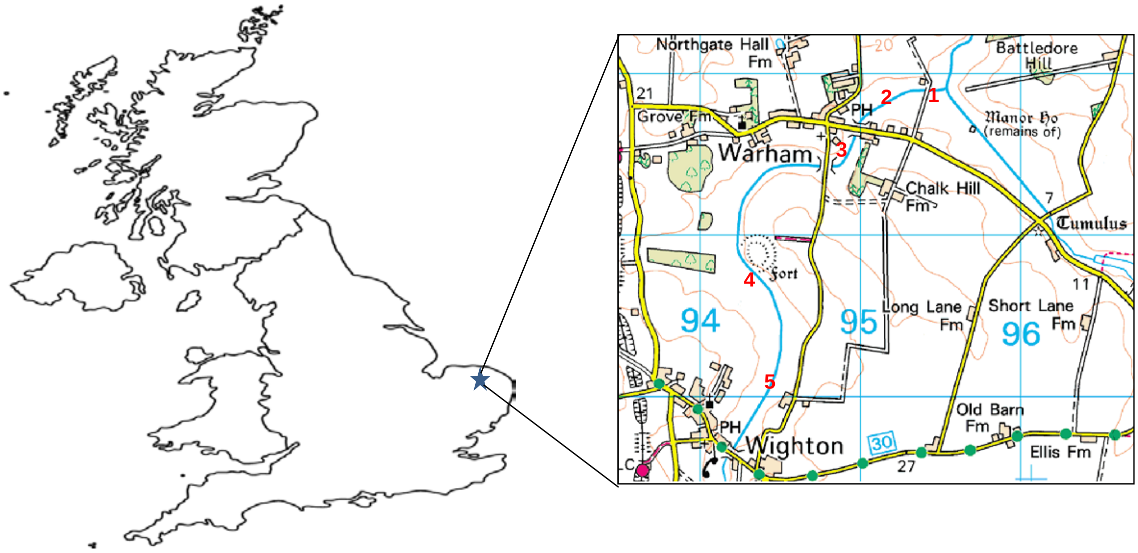


Figure 3.30. Location of sample sites for the River Stiffkey (Source: Ordnance Survey).

Figure 3.31. The River Stiffkey a) before and b) after channel narrowing and gravel introduction.

Aims and objectives

The aim of this study was to determine if gravel reinstatement for riffle enhancement has an effect on fisheries habitat, specifically for brown trout, through application of a post-treatment impact assessment. Specific objectives were to compare suitable brown trout habitat availability, fish community structure and more specifically brown trout density and age structure post gravel reinstatement works at 3 sites on the River Stiffkey. Furthermore, additional representative control sites were included to test and establish a monitoring programme that will help assess the direct effect of gravel reinstatement on fish communities.

3.8.2. Methods

Fish and habitat surveys

In total 5 sites were surveyed in 2010 and 2011 (Figure 3.31). Gravels were installed at River Stiffkey (RS) sites 1-3 in 2008, RS4 contained a natural riffle and RS5 contained a gravel that was introduced in 2002 and is now established (Table 3.25). This study focuses on the reproductive success of the 3 gravel (RS1-RS3) augmentations in the River Stiffkey using the natural riffles (RS4) and the established riffles of 2002 (RS5) as control sites and applies spatial replication to overcome the absence of pre monitoring. The analysis and interpretation of the data from the survey will aim to inform on the status of fish populations and habitat at the River Stiffkey.

3.8.3. Results

HABSCORE analysis

HABSCORE outputs for the sites on the River Stiffkey revealed variations in the observed densities, predicted densities and habitat utilisation by trout (Table 3.29). Observed densities of 0+ trout in the River Stiffkey in 2010 were lower than predicted by the Habitat Quality Score (HQS) at RS2 and RS3, suggesting poorer populations than expected however, HUI upper CLs were >1 and therefore observed populations were not significantly lower than expected (Table 3.29). Observed densities of 0+ trout were higher than predicted by the HQS at RS1, RS4 and RS5 suggesting better populations than expected, but the HUI lower CLs were <1 suggesting observed populations were not significantly higher than would be expected (Table 3.29). Observed densities of 0+ trout in the River Stiffkey in 2011 were lower than predicted across all sites (RS1-RS5), but only significantly lower at RS4 where the HUI upper CL was <1 (Table 3.29).

Observed densities of >1+ trout (<20 cm) in 2010 were higher than predicted by the HQS at all sites (RS1-RS5), but they were only significantly higher at RS1 where the HUI lower CL was >1 (Table 3.29). Observed densities of >1+ trout (<20 cm) in 2011 were lower than predicted by the HQS at RS4 suggesting poorer populations than expected however, HUI upper CLs was >1 showing observed populations at RS4 were not significantly lower (Table 3.29). Observed densities of >1+ trout (<20 cm) in 2011 were higher than predicted by the HQS at all other sites (RS1-RS3 & RS5) suggesting better populations than expected, but HUI lower CLs were <1 suggesting observed populations were not significantly higher (Table 3.29).

Observed densities of >1+ trout (>20 cm) in 2010 were lower than predicted by the HQS at RS3 and RS5, indicating poorer populations than expected however, HUI upper CLs were only <1 at RS5, suggesting populations at this site were significantly lower (Table 3.29). Observed densities of >1+ trout (>20 cm) in 2010 were higher than expected by the HQS at RS1, RS2 & RS4 indicating better population than expected, but HUI lower CLs were >1 suggesting they were not significantly higher (Table 3.29). Observed densities of >1+ trout (>20 cm) in 2011 were lower than predicted by the HQS at RS5, however it was not significantly lower because the HUI upper CL was >1 (Table 3.29). Observed densities of >1+ trout (>20 cm) in 2011 were higher than predicted by the HQS at RS1-RS4 indicating better populations than expected, but HUI lower CLs were <1 suggesting observed populations were not significantly higher (Table 3.29).

Functional habitat of introduced gravels

Riffles were found at each site and their dimensions were estimated by measuring the length of the gravel. Riffle length varied between sites, RS1, RS2 and RS5 and the latter had the longest riffle length of 20 m, RS3 had a riffle length of 10 m and RS4 contained natural riffles across the whole length of the surveyed section, which was 50 m. All riffles, with the exception of the natural riffle (RS4), were constructed by the placement of cobbles to support the morphology of the introduced pool-riffle system. At RS5, riffles were installed in 2002 and are still shallower than the main river channel and dominated by cobbles, suggesting it is well established. RS1-RS3 riffles were installed in 2008, and habitat surveys in 2010 and 2011 confirmed the riffles were still dominated by cobbles, suggesting these riffle instalments will also establish themselves in the future, but further monitoring is needed to confirm this assertion. Depth profile of each newly reinstated riffle suggest a small variation in depths, however, there were no major changes and each riffle has kept its pool-riffle form (Figure 3.33). There are a small number of natural riffles in the River Stiffkey in the shallower sections of the river (e.g. RS4). However, in the sections of the Stiffkey where riffles have been introduced there are no natural riffles and therefore, it can be assumed that the deepest sections along the river between the riffle introductions is representative of the depths of the whole stretch before riffle installation and not necessarily due to pool scouring after riffle installation. Also, because there were no pre-assessments of habitat before riffle instalment, it can be assumed that the habitat present was the same as the habitat found in the runs, such as deep channelized areas with a sand-silt substrate and a large number of submerged and emergent plants.

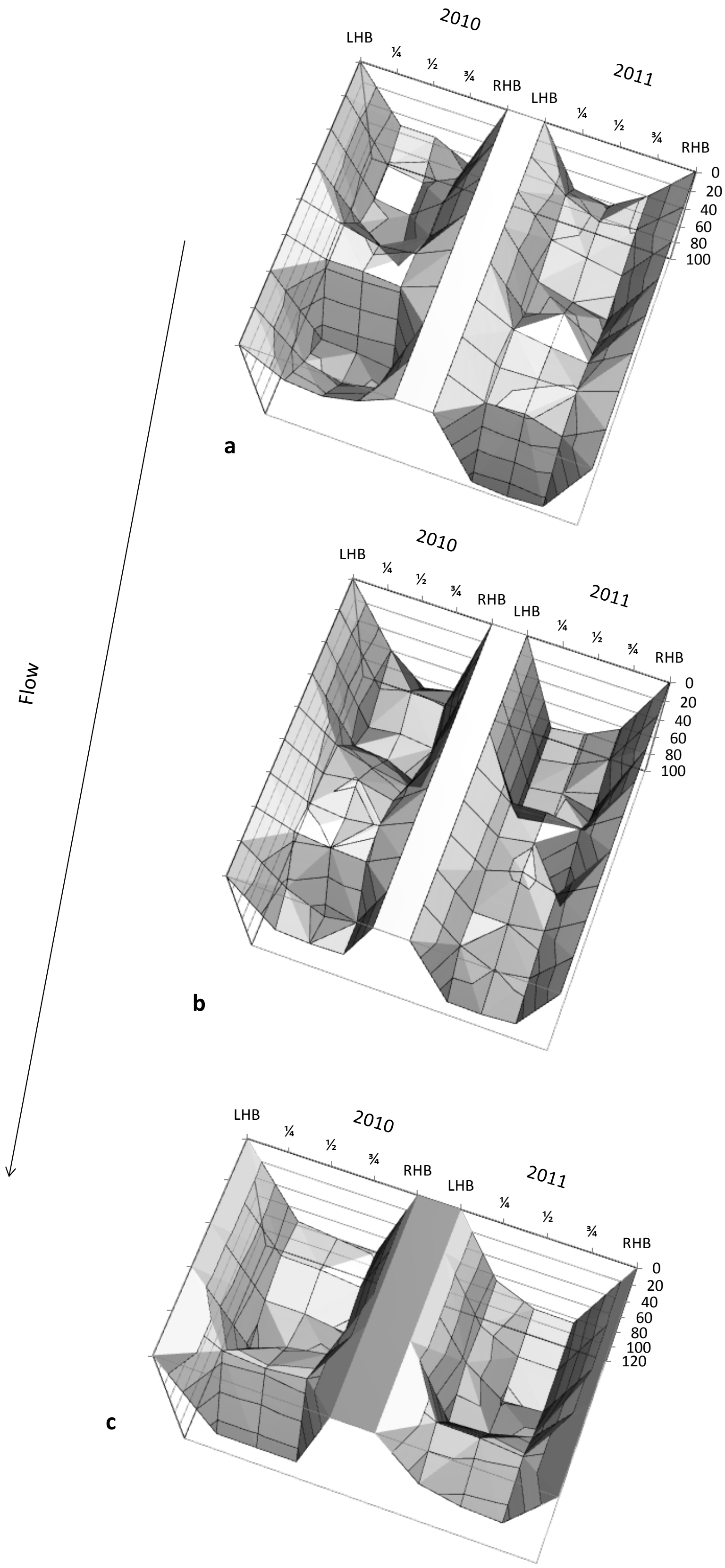


Figure 3.32. Changes in channel depth between 2010 and 2011 for the new gravel reinstatements a – Stiffkey 1, b – Stiffkey 2 & c – Stiffkey 3.

Table 3.25. Fisheries survey site details at the River Stiffkey in September 2010 & 2011.

River name NGR	Site name	Survey date	Length/mean width/area	Survey method and gear
Stiffkey TF 954 419	Site 1 – Downstream of private road	13/09/10 27/09/11	79 m/4.4 m/344.8 m^2 78.6 m/4.4 m/344.3 m^2	Generator. Quantitative.
Stiffkey TF 953 419	Site 2 – Upstream of private road	14/09/10 27/09/11	87 m/3.9 m/340.8 m^2 87 m/3.9 m/340.8 m^2	Generator. Quantitative.
Stiffkey TF 948 414	Site 3 – Downstream of Warham road bridge	16/09/10 28/09/11	50 m/2.92 m/146 m^2 50 m/2.92 m/146 m^2	Generator. Quantitative.
Stiffkey TF 942 409	Site 4 – Natural riffles, near the fort	15/09/10 28/09/11	40 m/4.1 m/163 m^2 40 m/4.1 m/163 m^2	Generator. Quantitative.
Stiffkey TF 944 401	Site 5 – Older 2003 riffles, upstream of fort	15/09/10 28/09/11	80 m/3 m/242 m^2 80 m/3 m/242 m^2	Generator. Quantitative.

Status of fish populations at RS 1-5 at the River Stiffkey in September 2010 and 2011

Species composition in the River Stiffkey in September 2010 was similar across all sites (Table 3.26). Catches at RS1-RS5 were dominated by brown trout, bullhead, eel (Anguilla Anguilla L.) and lamprey with lesser numbers of stoneloach, 3-spined stickleback and flounder (Table 3.26). Species composition was also similar across all sites in September 2011 (Table 3.26) but on this occasion dominated by bullhead and brown trout, with lesser numbers of eel and lamprey, stoneloach, 3-spined stickleback and flounder (Table 3.26). Brown trout, eel and stoneloach abundances were fewer in 2011 than in 2010, whereas bullheads were more abundant in 2011 than 2010 with the exception of RS4 (Table 3.26). There were fewer lamprey in 2011 than 2010, with the exception of RS2 (Table 3.26). Three spined stickleback were more abundant in 2010 than 2011 at RS2 and RS3 but the opposite trend was found, at RS1 and RS5; no difference in abundance was found at RS4 between years (Table 3.26). No adult sea trout were captured during the 2010 or 2011 surveys.

Table 3.26. Number of fish of different species captured at one site on the River Stiffkey in September 2010 & 2011.

Species	Site 1		Site 2		Site 3		Site 4		Site 5	
	2010	2011	2010	2011	2010	2011	2010	2011	2010	2011
Brown trout	69	40	55	32	21	11	46	8	59	18
Bullhead	188	428	268	270	56	111	40	37	46	112
Eel	76	23	21	11	16	2	10	0	11	2
Lamprey	48	26	25	28	2	0	7	5	5	1

Stoneloach	35	29	24	17	4	1	1	0	3	0
3 Spined stickleback	28	39	36	21	17	6	5	5	1	10
Flounder	0	0	0	0	1	1	0	0	0	0

Density estimates

Density estimates and abundance classifications of brown trout varied between sites in 2010 (Table 3.27), ≥1+ brown trout dominated populations at RS1-RS3 while 0+ brown trout dominated populations at RS4-RS5. At RS1-RS3, 0+ trout were classified as fair (class D) to poor (Class E), but as good (Class B) at RS4 and fair (Class C) at RS5 (Table 3.28; Figure 3.34). ≥1+ trout were classified between good (Class B) and fair/average (Class C) across most sites (Table 3.27; Figure 3.35).

In 2011, density estimates and abundance classifications of brown trout (Table 3.28) also varied between sites: ≥1+ brown trout dominated populations at RS1-RS5. 0+ trout were classified as fair (Class D) to poor (Class E) at RS1-RS5 (Figure 3.34) whereas the ≥1+ trout were classified between fair/poor (Class D) and fair/average (Class C) (Table 3.28; Figure 3.35).

Table 3.27. Total population estimate (N), population density (D) (± 95% C.L.) and abundance classification of trout derived from fisheries surveys in River Stiffkey in September 2010 (density of fish given as numbers per 100m^2). Details of derivation of estimates are provided in the text.

Site No	Total Population		Population Density		Abundance Class	
	0+	≥1+	0+	≥1+	0+	≥1+
1	17±4	53±1	4.94±1.17	15.41±0.29	D	B
2	9±1	47±1	2.65±0.14	13.82±0.29	E	B
3	8±1	12±1	5.48±0.98	8.22±0.60	D	C
4	32±4	15±1	19.63±2.25	9.20±0.48	B	C
5	37±9	32±2	11.16±3.68	13.22±0.70	C	B

Table 3.28. Total population estimate (N), population density (D) (± 95% C.L.) and abundance classification of trout derived from fisheries surveys in River Stiffkey in September 2011 (density of fish given as numbers per 100m^2). Details of derivation of estimates are provided in the text.

Site No	Total Population		Population Density		Abundance Class	
	0+	≥1+	0+	≥1+	0+	≥1+
1	5±1	35±2	1.45±0.32	10.17±0.44	E	C
2	5±1	27±1	1.47±0.48	7.92 ± 0.36	E	C
3	5±0	6±1	3.42±0.00	4.11±0.48	D	D
4	1±1	7±1	0.61±0.43	4.29±0.37	E	D
5	7±1	11±1	2.89±0.40	4.55±0.32	E	D

European Fish Index Calculations

The EFI + database calculate an increase in ecological class boundaries at all recent riffle reinstatements (S1–3), Fish Index Cass increased from Class 3 in 2010 (Fish Index S1=0.82, S2=0.86 & S3=0.85) to Class 2 in 2011 (Fish Index S1=0.92, S2=0.0.92 & S3=0.92). There was no change in ecological class boundaries (Class 2) for the established riffles (S4-5), between 2010 (Fish Index S4=0.89 & S5=0.86) and 2011 (Fish Index S4=0.88 & S5=0.81).

Length frequency

The size distribution of 0+ brown trout at the various sites in 2010 were all in the same approximate size range 80-115 mm while ≥1+ individuals were between approximately 130 and 320 mm long (Figure 3.36). The differences found in size structure related to the presence of older age groups; for example the largest trout caught at RS1 was 2+ (274 mm long) but the oldest fish caught at RS2 and RS3 were 3+ (315 mm and 326 mm long respectively) (Figure 3.36).

In 2011 0+ brown trout were captured in the size range 88-120 mm while ≥1+ individuals were captured in the size range 125-315 mm (Figure 3.37); again the differences related to the age of the oldest fish, with large 3+ individuals found at RS2 and RS3 (273 and 315 mm, respectively) (Figure 3.37).

The presence of 0+ trout at RS1-RS5 in 2010 and 2011 indicated natural recruitment, but numbers of 0+ trout were lower in 2011 than 2010. The presence of good densities of ≥1+ brown trout across all sites in 2010 indicated good survival of brown trout from recruitment in 2009; however densities of ≥1+ brown trout across all 5 sites in 2011 were lower than 2010 suggesting poorer recruitment in 2010. Brown trout >200 mm were present at RS1-RS5 in 2010 and 2011, but as expected, at lower densities than the younger age classes (Figures 3.36-3.37).

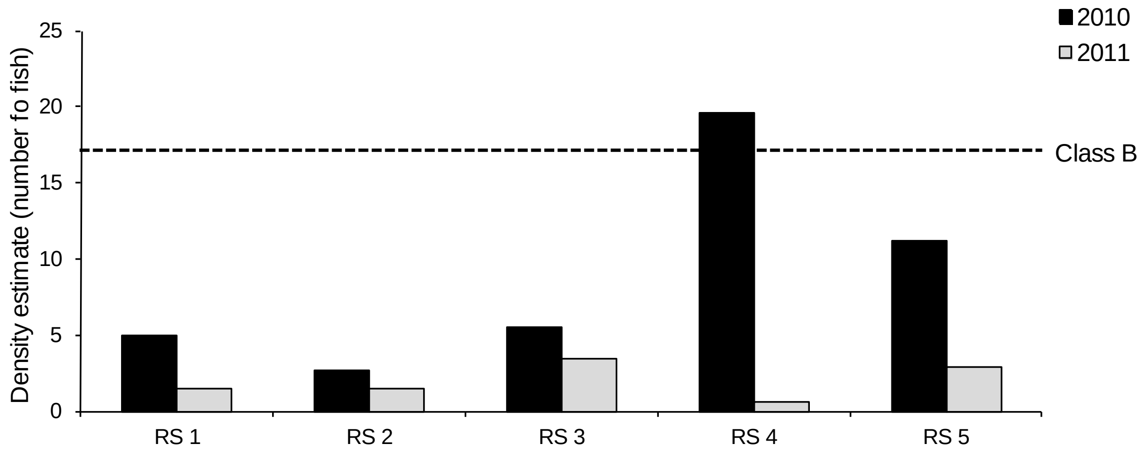


Figure 3.33. Density estimates of 0+ trout at RS1-RS5 in 2010 and 2011, Class B and above indicates good population density.

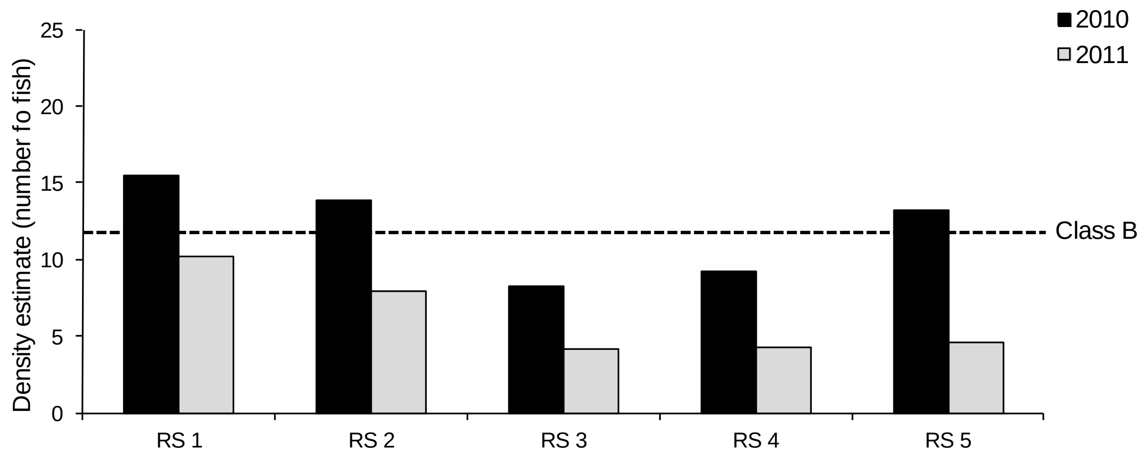


Figure 3.34. Density estimates of >1+ trout at RS1-RS5 in 2010 and 2011, Class B and above indicates good population density.

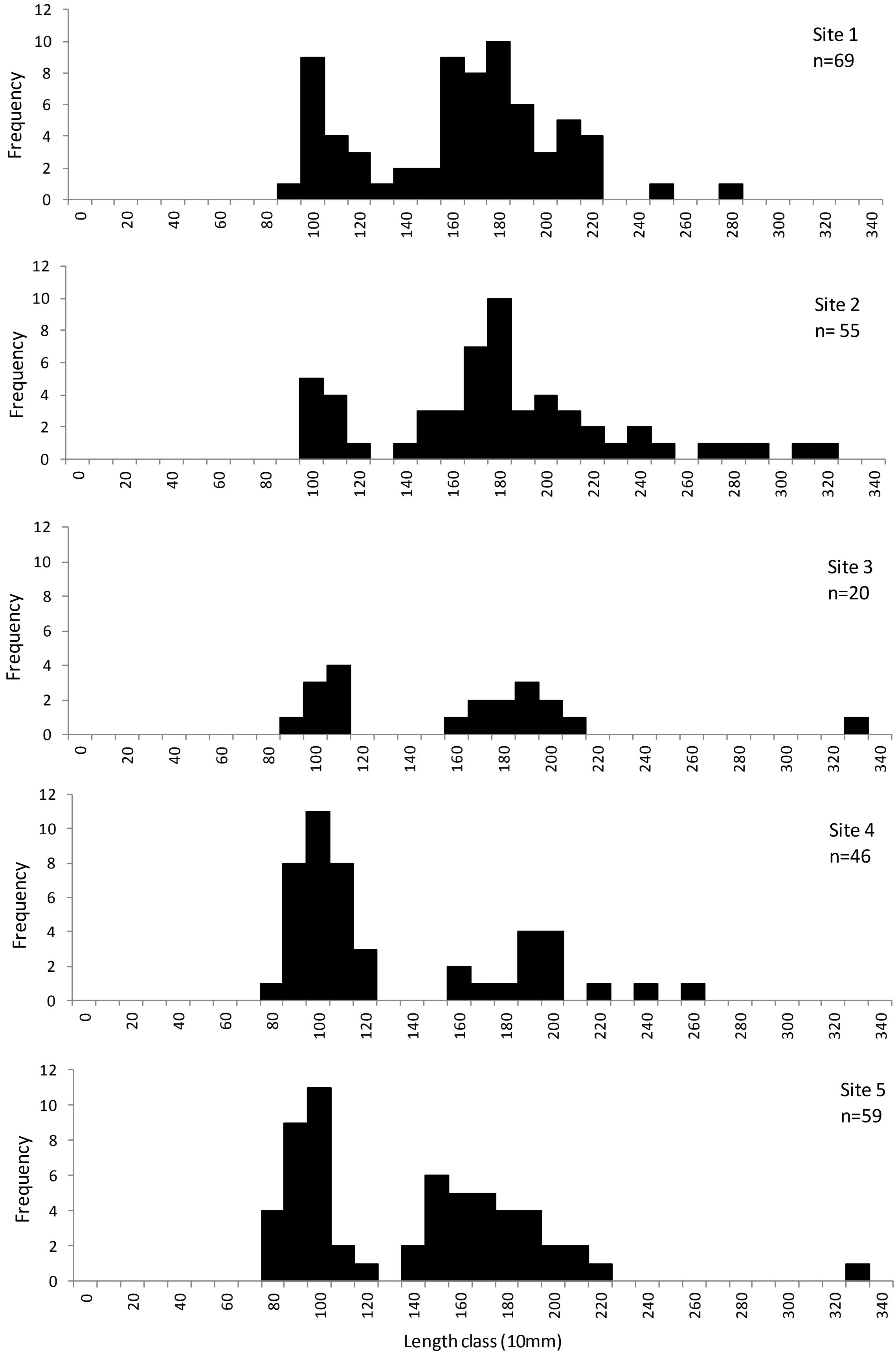


Figure 3.35. Length distributions of brown trout in River Stiffkey at all sites in 2010.

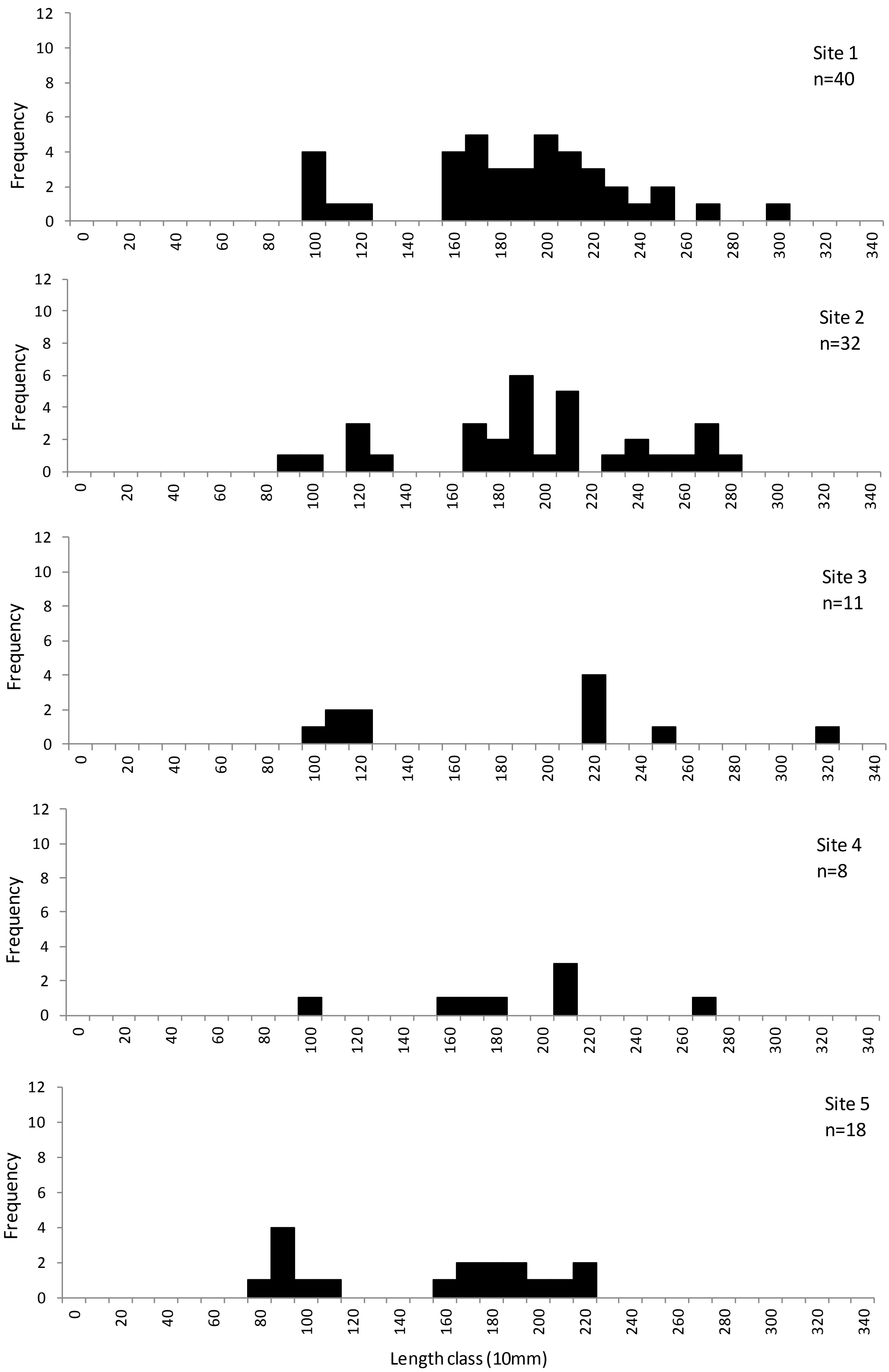


Figure 3.36. Length distributions of brown trout in River Stiffkey at all sites in 2011.

Eels were present at all 5 sites in 2010 and all sites except RS4 in 2011 (Figure 3.38 – 3.39). In 2010 at RS1 Eels were captured in the size ranges 30-470 mm at RS1 in 2010, 20-370 mm in RS2, 15-545 mm in RS3, 170-350 mm in RS4 and 10-155 in RS5 (Figure 3.38). In 2011 the size ranges were 123-538 mm at RS1, 120-513 mm at RS2, only two eel were caught (160 mm and 300 mm) at RS3 and again only two eel were captured (190 mm and 230 mm) at RS5 (Figure 3.39).

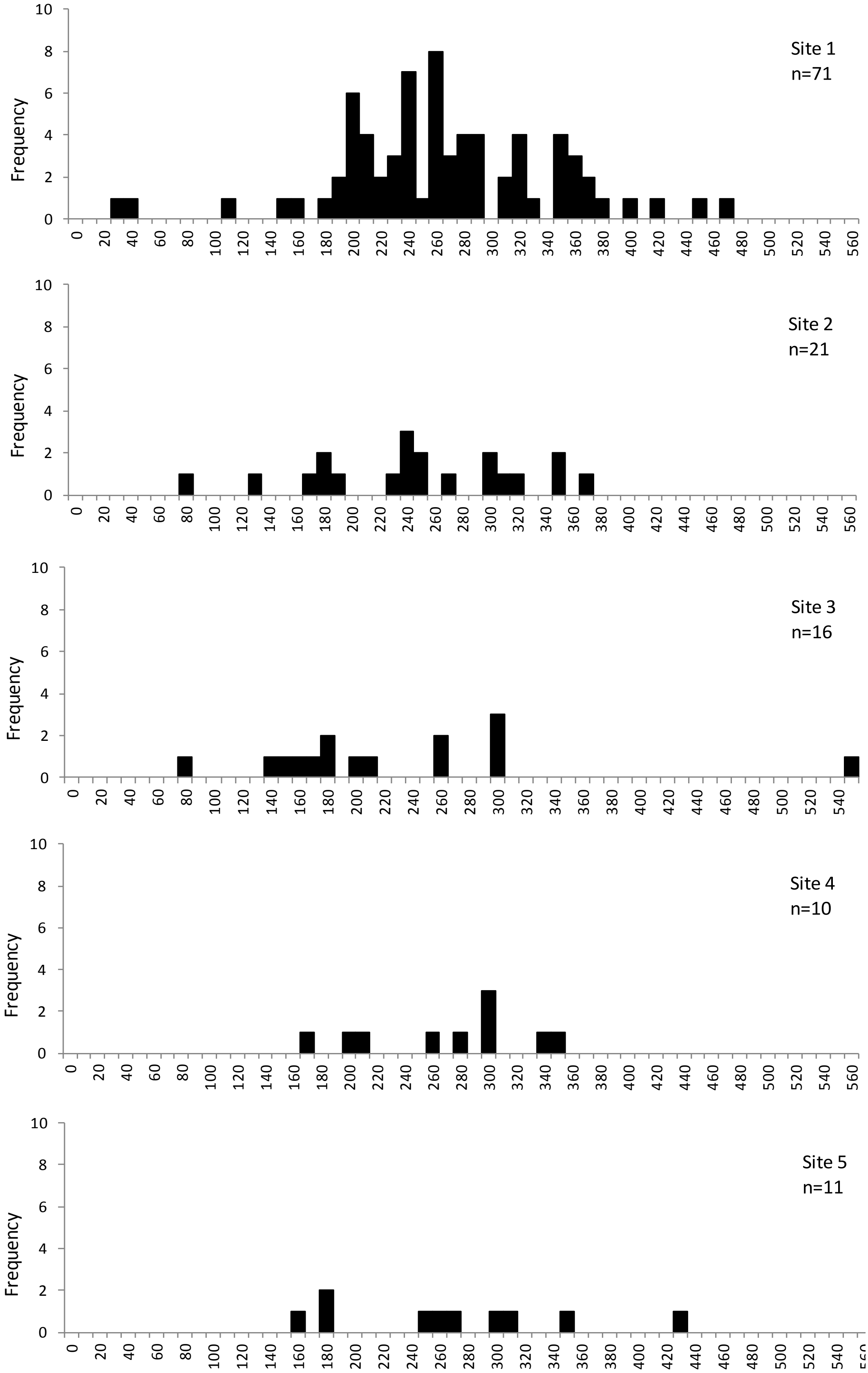


Figure 3.37. Length distributions of eel in River Stiffkey at all sites in 2010.

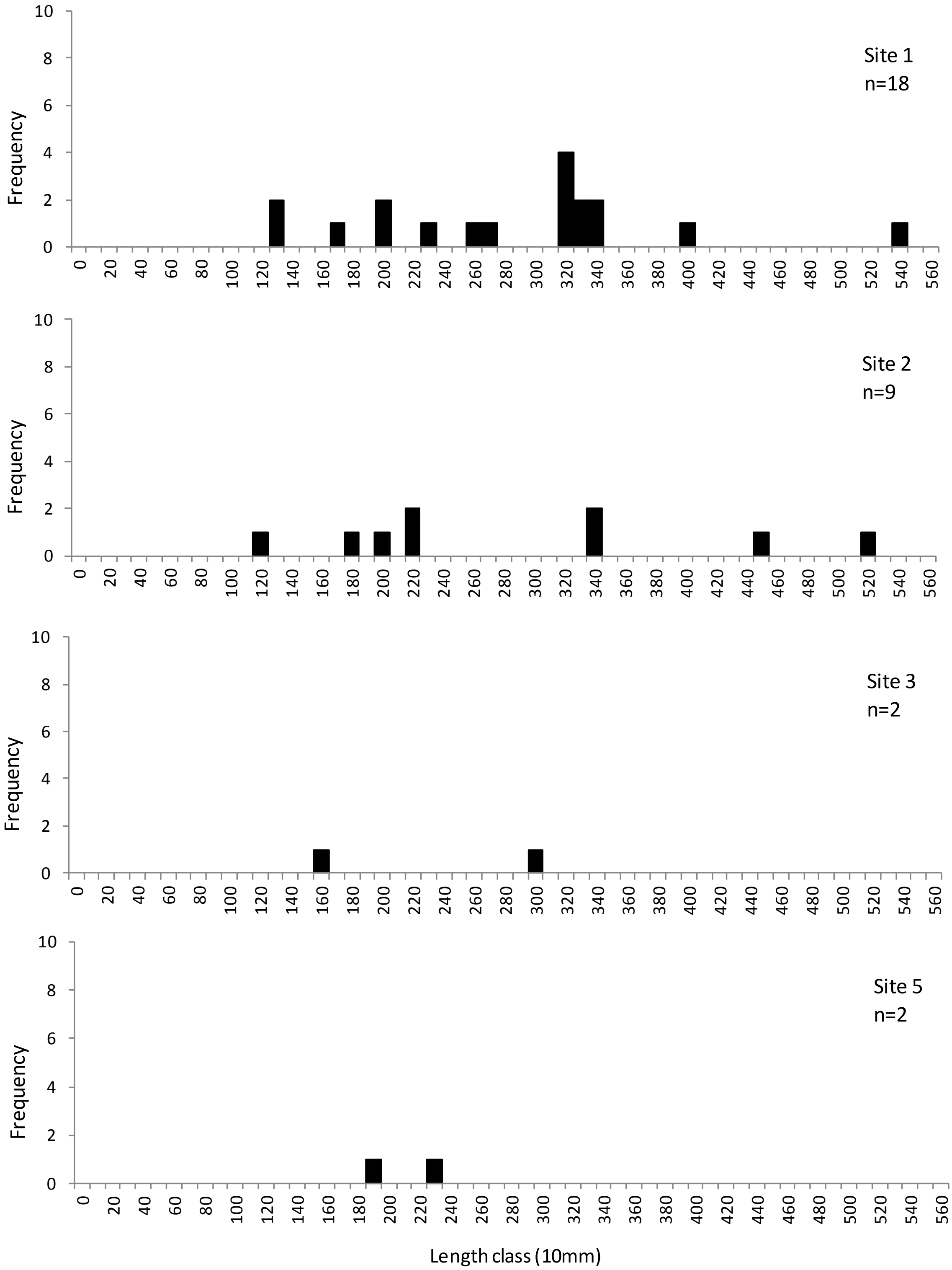


Figure 3.38. Length distributions of eel in River Stiffkey at all sites in 2011.

Impact assessment and resource calculation

A post-treatment design (Section 3.2.4, Equation 3.2) was used to assess the response of 0+ and >1+ brown trout to the introduction of artificial riffles in the River Stiffkey. The impact assessment was assessed both in terms of confidence limits of the mean difference in 0+ and >1+ brown trout density and by means of a t-test. The mean change in 0+ brown trout density between the impact sites (M = 0.65, SE = 0.09) and the control sites (M = 0.80, SE = 0.25), t (3) = 0.54, was -1.69 ± 2.17 fish per 100 m^2, the 95% confidence limits (-0.69 – 0.98) encloses zero and therefore the difference was not significant p= 0.63 ($p > 0.05$, r = 0.30). The mean change in >1+ brown trout density between the impact sites (M = 11.60, SE = 1.56) and the control sites (M = 7.82, SE = 2.13), t (3) = -2.84, was -2.75 ± 3.66 fish per 100 m^2, the 95% confidence limits (-8.03 – 0.46) encloses zero and therefore the difference was not significant p= 0.07 ($p > 0.05$, r = 0.85). The standard errors for the mean difference in 0+ and >1+ brown trout at impact and control sites was low and therefore a low amount of variability between the means of different samples. The resource calculation (Section 3.2.4, Equation 3.5) determined that the variance in the actual data for 0+ trout (0.03) and >1+ trout (0.081) was low enough to identify a change of 5 fish per 100 m^2 to be statistically detected (actual variance must be below the target variance (0.16)). **Therefore, there were a sufficient number of years and sites to detect changes in fish populations on the River Stiffkey.**

3.8.4. Discussion

Rehabilitation works have been carried out on the River Stiffkey to improve spawning habitat availability for trout through the introduction artificial gravels within various river sections. The main objective of this study was to see if gravel introduction is an appropriate rehabilitation technique to re-introduce suitable habitat into the River Stiffkey, specifically for brown and sea trout. The RS5 riffle that was installed to the River Stiffkey in 2002 shows signs of being established within the original river bed, with a pool-riffle system still in position and the majority of gravels still in place. The 2008 installation of riffles at RS1-RS3 are still in place and are also integrated into the original river bed. The establishment of these riffles, especially those of 2002, suggest that this rehabilitation technique could possibly be considered as a long term method for future projects to achieve improved spawning bed habitat, on the assumption they are not degraded by siltation or other perturbation.

Table 3.29. HABSCORE outputs for 0+, >1+ (<20 cm) and >1+ (>20 cm) across all sites on the River Stiffkey for 2010 and 2011 catches. (Note: Shaded area represents sites where the observed population was significantly higher (HUI lower CL column) or lower (HUI upper CL column) than would be expected under pristine conditions).

Site Code	Age category	Observed number	Observed density	HQS (density)	HQS lower CL	HQS upper CL	HUI	HUI lower CL	HUI upper CL
Site 1	0+								
	2010	17	5.04	2.51	0.63	10.09	2.00	0.29	14.04
	2011	5	1.38	4.12	1.04	16.36	0.33	0.05	2.32
	>1+ (<20 cm)								
	2010	42	12.44	1.33	0.31	5.73	9.33	1.55	56.13
	2011	21	5.78	1.99	0.46	8.59	2.91	0.48	17.58
	>1+ (>20 cm)								
	2010	11	3.26	1.09	0.35	3.38	3.00	0.96	9.36
	2011	14	3.85	1.40	0.46	4.33	2.75	0.86	8.48
Site 2	0+								
	2010	9	2.70	3.19	0.80	12.81	0.85	0.12	5.87
	2011	2	0.55	3.75	0.93	15.01	0.15	0.02	1.06
	>1+ (<20 cm)								
	2010	33	9.90	2.69	0.63	11.51	3.63	0.61	22.06
	2011	15	4.11	1.72	0.40	7.41	2.39	0.40	14.49
	>1+ (>20 cm)								
	2010	14	4.20	3.78	1.19	12.02	1.11	0.35	3.53
	2011	15	4.11	2.45	0.78	7.71	1.68	0.53	5.28
Site 3	0+								
	2010	8	5.50	5.53	1.40	21.92	0.99	0.14	6.88
	2011	2	0.96	3.05	0.77	12.12	0.32	0.05	2.18
	>1+ (<20 cm)								
	2010	10	6.87	3.92	0.88	17.42	1.75	0.28	10.86
	2011	3	1.45	1.32	0.30	5.71	1.10	0.18	6.65

	>1+ (>20 cm) 2010 2011	 2 6	 1.37 2.89	 3.44 2.83	 1.07 0.90	 11.03 8.88	 0.40 1.02	 0.12 0.32	 1.28 3.23

Site Code	Age category	Observed number	Observed density	HQS (density)	HQS lower CL	HQS upper CL	HUI	HUI lower CL	HUI upper CL
Site 4	0+ 2010 2011	 32 0	 19.75 0	 5.07 5.23	 1.29 1.33	 19.97 20.52	 3.89 0.11	 0.57 0.02	 26.71 0.73
	>1+ (<20 cm) 2010 2011	 11 4	 6.79 2.22	 3.93 4.25	 0.92 0.97	 16.84 18.50	 1.73 0.52	 0.29 0.09	 10.42 3.22
	>1+ (>20 cm) 2010 2011	 4 4	 2.47 2.22	 1.45 1.91	 0.47 0.62	 4.45 5.84	 1.70 1.17	 0.56 0.38	 5.21 3.58
Site 5	0+ 2010 2011	 32 6	 13.14 1.61	 4.69 6.52	 1.17 2.66	 18.83 25.62	 2.80 0.25	 0.40 0.04	 19.74 1.70
	>1+ (<20 cm) 2010 2011	 28 9	 11.50 2.42	 4.21 2.08	 0.97 0.48	 18.30 8.88	 2.73 1.16	 0.45 0.19	 16.64 7.00
	>1+ (>20 cm) 2010 2011	 4 3	 1.64 0.81	 7.50 2.18	 2.28 0.72	 24.63 6.65	 0.22 0.37	 0.07 0.12	 0.72 1.13

The functional use of the introduced riffles is also extremely important to ensure suitable habitat is provided for brown and sea trout. Siltation is a major cause of loss of ecological integrity and good habitat for spawning and juvenile rearing of brown/sea tout and is a significant impact in many Anglian rivers where changes in farming practices have clearly influenced siltation rate (Pawson 2008). Sear *et al*. (2006) identified this to be a main factor limiting trout production in other Anglian rivers such as the Galven and Nar, and it is believed to be the same for the Stiffkey (Pawson 2008). As a consequence fine sediment may smother gravels and thus reduce effective habitat improvement (Downs & Thorne 1998). Although sediment was not directly monitored for this study, observations suggest that the riffle instalments reduced the amount of sediment local to the riffle; however, the deeper pooled areas contained high volumes of silt, showing similar characteristics to the unmodified parts of the River Stiffkey.

It can be concluded that the River Stiffkey's trout population is reproductively self sustaining because there is no record of stocking so the fishery relies on natural production (Pawson 2008). Existing fish data (for the years 1988, 1992, 1995, 2000, 2007 and 2008) can be used as a baseline of fish presence, densities and biomass for eight sites along the River Stiffkey (Figure 3.36) (Pawson 2008). Fish species present in the previous surveys were eel, brown trout, 3-spined stickleback, 9-spined stickleback, stone loach, gudgeon (*Gobio gobio* (L.)), bullhead, brook lamprey, river lamprey and flounder (*Platichthys flesus* (L.)). The fish community has been dominated by eels, with brown trout the second most abundant in terms of biomass (Pawson 2008). Comparison of species previously caught with those in this study, found all species were present with the exception of 9-spine stickleback and gudgeon, which were absent from 2010 and 2011 survey data. The 2010 and 2011 survey data confirmed that eel and brown trout are still the dominant species, together with bullhead. When considering species richness, the EFI identified an increase in ecological class boundaries at all recent riffle reinstatements (S1-3), increasing from Class 3 to 2, whereas the ecological boundary for the riffles already established remained unchanged. Although the riffle reinstatement work was mainly aimed at brown/sea trout, it is clear that their introduction has also benefited many other species present. This is most certainly consequence of the improved flow type to pool-riffle and the enhanced river bed substrate from sediment to gravel.

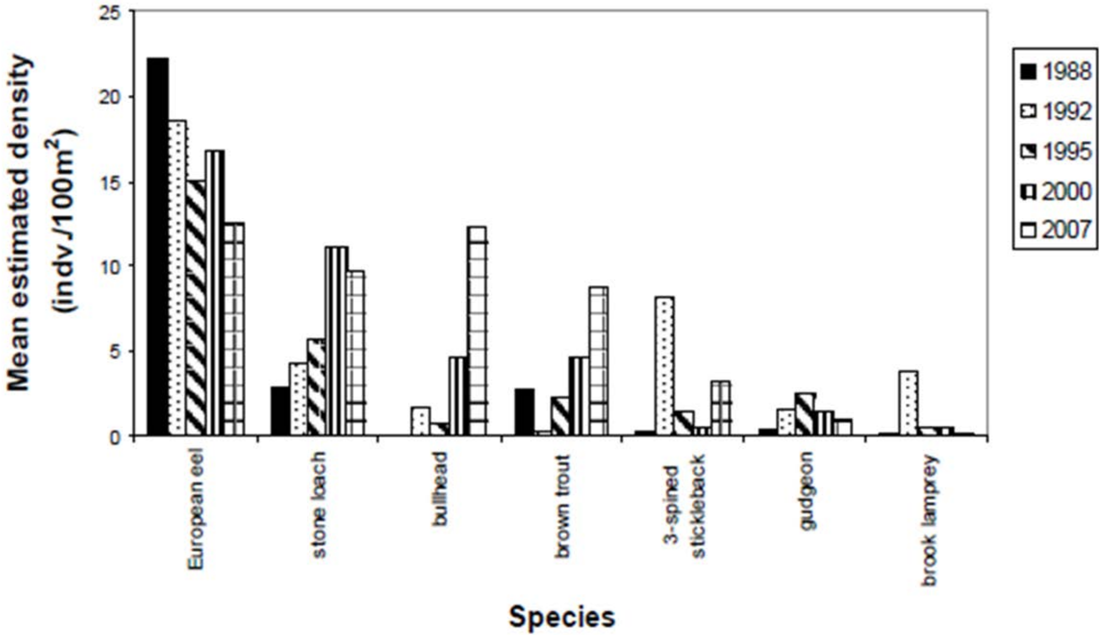


Figure 3.39. Mean estimated density of principal fish species found in the River Stiffkey 1988 to 2007 (taken from Pawson 2008).

The outputs of the fisheries and habitat surveys on the River Stiffkey in 2010 and 2011 provide an assessment of the status of the fish populations following the introduction of spawning gravels. Overall the presence of 0+ individuals and good numbers of ≥1+ brown trout for both years indicate there is suitable spawning habitat for brown trout within the area for brown trout. Numbers of 0+ and ≥1+ brown trout at each site were lower in 2011 than in 2010 suggesting poor recruitment in 2011 and generally a poor year for 1+ brown trout. Brown trout numbers were lower at every site in 2011 than in 2010, suggesting that the reduction is caused by natural variability influencing habitat availability and not a response to gravel reinstatements. This is because trout numbers were also low at RS4 (natural riffles) and RS5 (established gravels). The large number of eels found across a range of reaches suggests the River Stiffkey has suitable habitat for this species and also benefits from good longitudinal access up and down stream. The number of eels was low across all sites in 2011 compared with 2010 and therefore suggests poor survival or dispersion in 2011, possibly a response to natural variability. Bullheads were present at all sites on the River Stiffkey in 2010 and 2011, with evidence of juveniles and mature spawning stock suggesting that habitat conditions are suitable for this species in the river reach studied. Sea trout were not present in the 2010 or 2011 this can suggest a lack of favoured habitat still remains an issue in the River Stiffkey, especially as water quality and longitudinal connectivity are proven not to hinder their life cycles. Sea trout spawn October to December, because sampling took place in September of each year this may have been too early to document any returning sea trout.

The duration of monitoring needed to detect a change is particularly important in any type of monitoring study design to overcome natural variability and identify a possible change as a result of the intervention, in this case gravel reinstatement. A post-treatment monitoring design was applied to analyse the success of gravel introductions on fish populations, in particular brown/sea trout. This was completed by spatial replication because the monitoring design was limited to only post-data collection. Although there was a noticeable reduction in trout caught between 2010 and 2011 across all sites, the post-treatment design did not identify a significant difference between 0+ and >1+ trout densities when comparing the grouped means of the impacted sites (RS1-RS3) to the control sites (RS4 & RS5). The resource calculation supported the post-treatment design used and approved there were a sufficient number of years and sites to detect a change in fish population (with 5 fish per 100 m² precision). Most work on salmonids has suggested that more than 10 years (5 before and 5 after) are needed to detect significant changes in fish abundance, unless the magnitude of change is very large (>threefold; Bisson *et al*. 1997; Roni *et al*. 2003). With this in mind, and although the resource calculation supports the post-treatment design used, further monitoring would strengthen the impact assessment, especially as there are not sufficient pre-data. Replication can help overcome limitations such as no pre-data; unfortunately it is usually avoided in most monitoring designs due to cost and logistics, even though it is recognised as a critical measure to reduce variability (Roni *et al*. 2005b).

Recommendations

There is already a relatively large amount of information assembled on brown/sea trout habitat for juvenile and adult stages, especially information on spawning preferences (Armstrong *et al*. 2003). In recent years this knowledge has been used to enhance habitat rehabilitation techniques to benefit aquatic ecosystems, in particular for fish. This, however, does not necessarily mean fish will respond well to remediation measures, after all they are still artificial measures and only a limited number of short term studies have taken place on riffle reinstatement (Moreau 1984; Crispin *et al.* 1993; Iversen *et al.* 1993; House 1996; Gortz 1998). **Consequently, further research is needed to understand fish response to change in habitat through rehabilitation measures, in particular sea/brown trout response to riffle reinstatement.**

Overall, there is little evidence from this study that suggests adding artificial riffles or flow deflectors will substantially improve the conservation value of the fish assemblage in the River Stiffkey, in particular brown/sea trout densities of both 0+ and >1+ trout. **Specific recommendations for the River Stiffkey would be to monitor the population response for at least another 3 years to overcome possible natural variability in population dynamics**.

This study focused on gravel reinstatements as a single rehabilitation technique and consequently **it is recommended that the habitat preferences for each life stage of brown and sea trout are revisited and additional 'missing' habitat(s) identified and addressed with the appropriate rehabilitation measures**. Sediment loading is an ongoing pressure on the River Stiffkey and therefore this pressure needs to be addressed in order for any future rehabilitation work to benefit long term. There is a requirement to manage sediment loading through erosion control from agricultural pressures, fencing would provide a barrier between cattle and the river bank, therefore reducing erosion and buffer zones would reduce excess sediment entering the river from land. A thorough project planning framework should be applied with sufficient monitoring by means of an impact assessment to strengthen analysis and to advance our knowledge on rehabilitation projects.

3.9. General discussion

This chapter gives a practical insight in to the process of monitoring and limitations in evaluating a number of ‘real life’ rehabilitation schemes and demonstrates the challenges that need to be overcome to collect meaningful results to measure project success. The overall, major constraint during the evaluation of case studies in this thesis was the lack of involvement I was able to have in the decision process of the planning of the rehabiliation schemes. All rehabilitation work had either already took place or was in the early stages. For example, take Figure 2.1, the adaptive management framework flow diagram that demonstrates all of the important stages that should be considered, and followed during a rehabilitation project. Many of these stages had not been considered by those accountable for each of the rehabilitation schemes presented in this chapter. Evaluating river health and fish status is the first stage to consider before considering river rehabilitation, this had been over looked in all 5 case studies resulting in either no pre-fisheries data (Manifold and Stiffkey) or a limited 1 years pre-fisheries data (Driffield Beck, Lowthorpe Beck and Dovedale weir removal). The lack of sufficient pre-monitoring data limited the evaluation of all case studies. However, this was overcome with control sites, where possible, meaning a number of impact assessments were trialled (BACI for Dovedale weir removal and Post-treatment design for the Manifold brash revetment and the Stiffkey gravel reinstatement). Unfortunately, they were still limited by sample size because the timeframe of the PhD allowed for only two years of sampling data to be collected. These limitations of sample size and time frame for monitoring are also found in management practise due to timeframes and budgets of projects.

Although fish are regularly reported as good indicator for change, especially for evaluation of river rehabilitation success, this was not the case for the 5 case studies presented in this chapter. Impact assessments were difficult to apply on small sample sizes. A large sample size is needed to reduce variability between the mean densities and therefore, I recommend monitoring is considered before rehabiliation commences, enabling a good sample size for pre-data. Control sites were applied to overcome absent pre-monitoring, but in most cases the variability between mean density of 0+ and >1+ was large. Further investigation found this could be due to a difference in habitat characteristics of the chosen control sites. A good example of this is the River Stiffkey case study, where both impact and control sites were on the same river and the characteristics of this river were similar on the whole stretch. Although a small sample size (2 control and 3 impact sites), little variation was found between the mean density of 0+ and >1+ brown trout. Whereas, for the Manifold and Dove case studies, a larger number control sites (6) were used across different rivers in the same catchment, with similar characteristics, in an attempt to produce a robust sample size. However, this resulted in high variability of brown trout densities between sites. This highlights the importance of selecting suitable control sites. Long-term spatial monitoring is essential to evaluated river rehabilitation success when using fish as indicators of this success. Overall, the application of HABSCORE for each of the case studies was a sufficient method to identify river rehabilitation improvements of habitat for trout populations, although there was insufficient fisheries data short term, this can be overcome by using the HQS as an indicator of habitat improvement after river rehabilitation. While acknowledging that resources will often be limited for monitoring data, this can be overcome by suitable project planning and monitoring design and does not necessarily have to be costly and time consuming, incorporating the Environment Agency's annual fisheries monitoring and habitat (HABSCORE) data can strengthen statistical analysis for impact assessment and resource calculations.

Freshwater river ecosystems are intrinsically linked and have a natural habitat continuum between river and landscape (May 2006), because of this it is difficult to conserve a reach of river by simply using rehabilitation practice at a local level. However, river rehabilitation program goals often only address problems on single rivers at a small scale and therefore have limited impact on catchment-scale processes (Buijse *et al*. 2005; Eden & Tunstall 2006). Small scale river rehabilitation is most frequently employed because it is cheap, easy to apply and quick to accomplish and because of this it is important to understand how it can be integrated in to catchment scale rehabilitation. The rehabilitation measures investigated in this chapter were localised and should be part of a catchment scale planning approach to recover fish populations to improve ecological status for WFD, with the exception of the River Stiffkey that is part of a larger scale study for sea trout in Norfolk rivers.

The application of the EFI is a practical method when assessing the status of rivers at a European scale because it enables ecological status to be assessed on a community level. It is the only fish index that has been successfully used at a European scale, thus allowing comparisons of river health and more specifically river rehabilitation outcomes, between European countries, further supporting the WFD (EFI+ CONSORTIUM 2009). Nevertheless, such a complex database does not come without limitations (Sampling location, environment, sampling method applied, low species richness & number for fish caught). Low species richness and low numbers of fish prevented the use of EFI for Driffield Beck, whilst EFI was applied with caution for the Hartington case study because numbers of fish were low. This highlights the need for adequate sampling to assess the abundance and structure of fish assemblage and the population structure of the species caught. In cases of low density, expert judgement will have to be satisfactory to assess the ecological status of the river.

CHAPTER 4. FLOOD RISK MANAGEMENT OF AN URBAN RIVER – RIVERINE MITIGATION METHODS AND IMPLICATIONS FOR FISHERIES

4.1. Introduction

Flooding in rivers is a natural occurrence that can have devastating effects on people, property, infrastructure and economy. Climate change predictions indicate the likely increase in the risk of flooding from rivers emphasising the importance of effective flood risk management. Combining mitigation and enhancement measures to improve the status of river with flood risk management provides the opportunity for an effective, sustainable framework that is vital to meet obligations under the EU WFD and HD. This chapter examines the complexities that arise when trying to incorporate river rehabilitation within flood risk management and the issues that should be addressed to provide win-win scenarios.

Flooding in an urban environment is usually caused by excessive rainfall and inefficient drainage systems; urbanisation has accelerated the transport of water, pollutants and sediment from urban areas into rivers and their typically constrained nature is inadequate to cope with the increased flow volumes being experienced, especially during intense rainfall events being experienced in recent years (Leopold 1968; Finkenbine *et al* 2000; Andjelkovic 2001; Paul & Meyer 2001; Walsh *et al.* 2001). Economic damages caused by urban floods are particularly high (Munich 2005) because of continuing urbanization and population growth, in addition to the increase in the magnitude and frequency of floods mentioned (COM 2006; WMO/GWP 2008; Bruin & Borrows 2006). Urban rivers have been channelized, deepened and/or widened over many years for navigation and flood control, enlarging river embankments and reducing the complexity within the channel. This enables a larger volume of water to move through the channel at a faster rate, reducing the effects of flooding on property and infrastructure. The lateral movement of a natural floodplain is an essential process within river form and function (Andjelkovic 2001), and provides additional habitat and feeding grounds for fish species. Urban developments are frequently situated on floodplain areas and therefore lateral flood movement (and flooding) is not practical, especially as the channel is restricted by vertical embankments and surrounded by industry. However, if channel capacity is exceeded flooding will occur (WMO/GWP 2008). The reduction in river complexity as a direct consequence of channelization decreases the heterogeneity of the ecosystem by deepening and widening the river, removing meanders and eliminating instream riparian habitat features (Cowx & Welcomme 1998). Flood mitigation actions on an already channelized river will increase the pressure on river ecosystem functioning (McCarthy 1985; Pretty *et al*. 2003) resulting in further homogeneity of the system by additional reduction of riparian vegetation and nutrient dynamics, to permit rapid clearance of water from the floodplain. This therefore has negative effects on aquatic biota, in particular fish species, by diminishing refuges for feeding and breeding of fish species (Brookes 1985; Smith, Harper & Barham 1990; Wilcock & Essery 1991; Hodgson & O'Hara 1994; Cowx & Welcomme 1998).

4.1.1.Policy

Approaches towards flooding have changed in recent years from flood protection to flood risk management (Klijn *et al*. 2008; Manojlovic & Pasche 2008; Vinet 2008; Hecker *et al*. 2008; Twigger-Ross *et al*. 2009; Mostert & Junier 2009). Past flood management methodologies were inclined to support economic impacts more than environmental and social impacts (Andjelkovic 2001), although more recent views have advanced to a multidisciplinary approach. Flood risk management (FRM) measures and actions are dependent on political support through legislation (Bruin & Borrows 2006), such as the European Floods Directive and the UK Flood and Water Management Act (2010).

*European Floods Directive (FD)**:*** The EU Floods Directive (Directive 2007/60/EC) came in to force in 2007 and aims to reduce and manage the risks that floods pose to human health, the environment, cultural heritage and economic activity. It requires:

> *'Member States to assess if all water courses and coast lines are at risk from flooding, to map the flood extent and assets and humans at risk in these areas and to take adequate and coordinated measures to reduce this flood risk' (EC 2011).*

UK Flood and Water Management Act (FWMA): The Flood and Water Management Act (2010) is specifically for the UK and requires the development of a National flood and Coastal Erosion Risk Management Strategy Defra (2004).

> *'It provides comprehensive flood risk management for people homes and businesses, helps safeguard community groups from unaffordable rises in surface water drainage charges and protects water supplies to the consumer' (COM 2006).*

The FWMA requires the Environment Agency and other local flood authorities to develop strategies for risk management. Additional terms for the FWMA brought into force July 2011 require:

> *'risk management authorities to act consistently with the national strategy in carrying out their flood and coastal erosion risk management functions' (Defra 2004).*

4.1.2. Integrating flood mitigation directives with conservation and rehabilitation

To apply EU directives that have conflicting intentions can be challenging. For example, any flood mitigation plans carried out under the EU FD or UK FWMA should also incorporate the WFD and HD as they are drivers towards 'sustainable' flood risk management solutions that aim to achieve 'good ecological potential' of urban rivers. However challenging flood risk management can be, environmental conservation should never be overlooked.

Approximately 5.2 million properties in England (or 1 in six) and 360,000 properties in Wales are at risk of flooding (Environment Agency 2007). Catchment Flood Management Plans (CFMPs) support both the EU FD and the UK FWMA by giving an overview of the flood risk across each river catchment and establish local management policies that alleviate downstream flood risk through a number of ways such as reduce channel maintenance in different river reaches, and active channel and floodplain rehabilitation in others (Environment Agency 2007). The EU FD and the FWMA are supported through European programmes such as Room for the River, Making Space for Water, Living Rivers, and Environmental River Enhancement. In England the "Making Space for Water" strategy put in place by Defra in 2004 emphasises the need for a more holistic approach to flood management that delivers the greatest environmental, social and economic benefits (England *et al*. 2007). The connection between these factors and how they intrinsically link is vital, especially for urban river rehabilitation management (Figure 4.1). These plans have the potential to work in coordination with river rehabilitation planning processes by providing a sound evaluation of flood risk constraints and opportunities for river rehabilitation (Mainstone & Holmes 2010).

Although there are EU and UK policies in place to support flood risk management, they are still difficult to implement at the local level where there are many stakeholders involved in the decision process. Ecological integrity is usually compromised for FRM and it is only in more recent years that management methods have encouraged river rehabilitation to be incorporated in to management plans. FRM should take account of sustainability and the natural environment (Bruin & Borrows 2006), and combining riverine mitigation and enhancement measures with flood risk management provides the opportunity for an effective, sustainable framework essential to meet objectives under the EU WFD and HD. The Environment Agency has a ministerial Flood Defence High Level Target to "*Ensure no net loss to habitats covered by Biodiversity Action Plans and seek opportunities for environmental enhancements*" (EA 2007: Policy Number: 606_06) and therefore, the Environment Agency and its partners promote a more environmentally sound approach to sustainable river management that underpins the WFD (Environment Agency 2007).

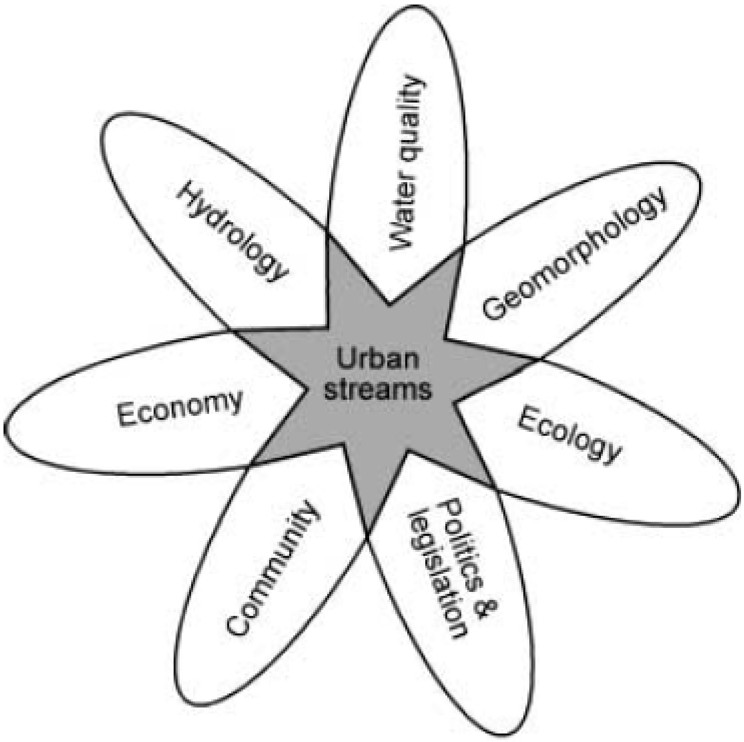


Figure 4.1. A conceptual illustration of the different factors that combine to affect urban stream rehabilitation management (taken from Findlay & Taylor 2006).

However, there are a number of challenges and uncertainties to account for when attempting to understand the intricacy of how ecosystem networks respond to river rehabilitation, so linking instream-habitat rehabilitation to flood risk management can be extremely complex as they work, at first sight, towards conflicting results. For example, the is the main regulator and developer of in-stream structures in England and Wales and from a flood risk management point of view the Environment Agency tends to oppose the use of in-stream structures as they often result in unfavourable flood risks consequences In-stream structures will only be approved if the following apply (Environment Agency 2007):

- “where it is demonstrated to our satisfaction that there are overriding social or economic reasons in line with our sustainable development remit and there is no reasonable alternative.”
- “where we agree that the ecological effects and impact on flood risk will be insignificant or will be adequately compensated.”
- “where the Agency accepts that the watercourse or floodplain is in unfavourable ecological condition and rehabilitation or enhancement using in-stream structures is justified and acceptable to Flood Risk Management.”

- "where there are important strategic or operational reasons to monitor or regulate river levels and flows and alternative methods are not viable."

However, it is also the responsibility of the Environment Agency to conserve, protect and enhance the environment, to identify and reduce potential impacts associated with flood mitigation works and propose measures for mitigation (Environment Agency 2010). Unfortunately, the rehabilitation of urban stream channels is highly constrained (Bernhardt & Palmer 2007) and their reinstatement to pre-disturbed conditions is an impractical vision since there are constant external pressures on this type of river system. For this reason the WFD classification of river health is to aim for '*good ecological potential*' for Heavily Modified Water Bodies (HMWB) to ensure River Basin Management Plans (RBMP) do not avoid these more complex systems. It is therefore important to recognise that the majority of urban rivers that cannot be returned to '*near natural*', non-impacted rivers and therefore it is essential that suitable habitat enhancement techniques are incorporated into urban river management to improve the ecological functioning through habitat heterogeneity and more specifically to benefit the fisheries (Sheehan & Rasmussen 1999; Findlay & Taylor 2006). Social, political and environmental factors are imperative for good management practise of urban river rehabilitation and it is their combination that determines the overall success of urban river rehabilitation (Findlay & Taylor 2006). Ultimately, suitable flood risk management enables the assessment of how rivers will respond morphologically and ecologically to climate, land use change and river management (Sear & Arnell 2006).

4.1.3. Aims and objectives

The aim of this chapter was to determine the effects of flood mitigation works that incorporate rehabilitation practice on fish communities. This was achieved by comparing species diversity and habitat characteristics pre and post flood mitigation & rehabilitation works to identify if a change occurred. Fish are a valuable economic resource and are of public concern; because of this the Environment Agency proposed a fisheries survey was conducted to assess how the flood works impacted on the fishery. In addition, fish were the preferred BQE monitored, to identify change after the flood mitigation and rehabilitation works because their taxonomy, ecological requirements and life history traits are better known than any other species. Species have specific habitat requirements and thus exhibit predictable responses to human induced habitat alterations; therefore depressed growth and recruitment are easily assessed and reflect stress. The longevity of many fish species enables assessment to be sensitive to disturbance over relatively long time scales (FAME CONSORTIUM (2004). The following chapter will provide an insight in to the subsequent flood mitigation measures and methods and the importance of integrating river rehabilitation techniques into flood risk management.

4.2. River Don Catchment

The River Don, South Yorkshire, rises from the Pennines and flows east through the Don Valley, via Penistone, Sheffield, Rotherham, Mexborough, Conisbrough, Doncaster and Stainforth, to its confluence with the River Ouse near Goole where the two rivers discharge into the Humber Estuary (Amisah & Cowx 2000). The main tributaries of the Don are the rivers Loxley, Rivelin, Sheaf, Dearne and Rother (Environment Agency 1997). The River Don is a recovering river following decades of pollution resulting from agricultural drainage, industrial revolution, urban development, population growth (Firth 1996, 1997), inadequate treatment of sewage effluent and past mining activities resulting in considerable degradation to the health of the river. Poor river health inevitably resulted in damage to the fish stocks of the catchment over the years (Firth 1996), consequently resulting in a high proportion of the river length being fishless into the mid-1980s (Firth 1997). However, improvements to the River Don have occurred over time due to a number of changes such as the upgrading of sewage treatment works, improving ammonia levels and the biological oxygen demand and the decline in the steel industry has reduced the discharge of metals in to the river. More recent years have realised the importance of ecosystem conservation which has helped benefit the heath of the River Don by improving water quality and the general habitat, and therefore improving the status of the fisheries (Firth 1997; Amisah & Cowx 2000).

The City of Sheffield, UK, is located within the River Don catchment and is classified as 'high risk' with respect to flooding (Environment Agency 2010). The city experienced severe flooding in June 2007 when the River Don exceeded the capacity of its river channel and inundated large areas of Sheffield City following prolonged and heavy rain in the Don catchment, where almost 100 mm fell in just 24 hours on the 25 June (Environment Agency 2007). As a result of this unusual event approximately 1,200 homes and 1,000 businesses were flooded (Environment Agency 2011, www.environment-agency.gov.uk/research/library/publications/40547.aspx). The flooding was magnified by large mature trees that had fallen and washed downstream and were identified by the Environment Agency as obstacles, contributing to the build up of silt and flood debris resulting in the back up of flood water and trees that remained standing collected flood debris causing further obstruction within the channel. Shoal islands (created by the deposition of silts and gravels) had been deposited immediately downstream of the weirs where trees and other vegetation have colonised, reducing the cross section of the river channel. The expansion of the islands formed serious obstructions, reducing the area available to flood water and therefore increasing the risk of localised flooding (Environment Agency 2010).

4.3. Malin Bridge case study

4.3.1. Introduction

As climate change pressures become more frequent on river systems, it also becomes an important driver for river rehabilitation mitigation and adaptation strategies. The EU

Floods Directive and UK Flood and Water Management Act need to be integrated with the EU WFD and Habitats Directive, to work towards flood risk management (FRM) whilst still considering river heath. This approach is still in its early stages and there are no examples in the literature that report successful FRM that has incorporated river rehabilitation. The following case study at Malin Bridge gives insight into flood & rehabilitation work on a smaller river.

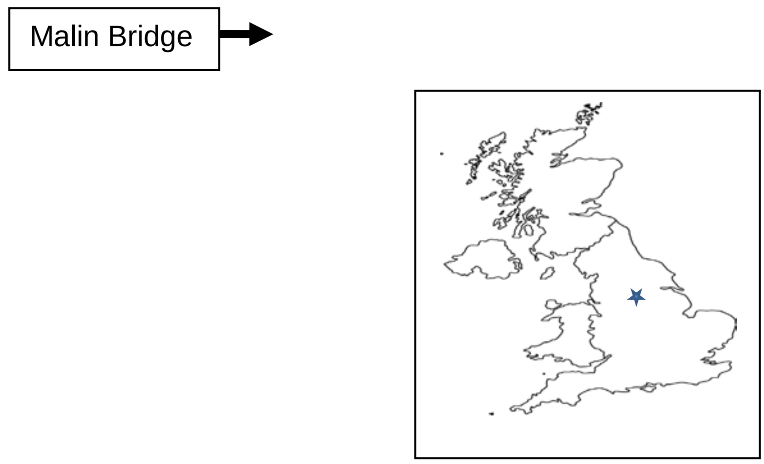


Figure 4.2. Location of sample sites at Malin Bridge (Source: Ordnance Survey).

4.3.2. Background

Malin Bridge is on the suburbs of Sheffield and is where the Rivers Rivelin and Loxley meet (Figure 4.2). The Environment Agency completed flood mitigation works in 2009 to reduce the risk of flooding to the area after the 2007 Sheffield floods, by shoal and tree removal (Figure 4.3). The works were driven by the FD and the WFD and therefore, river rehabilitation works were also included in the project planning. Unfortunately, during the flood mitigation works contractors removed an excessive amount of shoal and trees from the site, reducing habitat diversity within the river resulting in the need for further remedial works than was primarily intended. In addition to the excessive removal of instream habitat, two weirs were also uncovered, Rivelin Mill and Burgon/Ball, one of which was believed to be passable before the shoal removal. They are now both classified as being impassable to most fish at low flow. In 2010 and 2011 the channel was re-profiled, a rock riffle installed and instream boulders added as remediation actions, to try and introduce diversity to the river channel (Figure 4.3). Large boulders were used to frame the rock riffle, with smaller material being used to infill between boulders; the boulders were large enough that they did not move under high flows. Natural re-colonisation of vegetation was the method chosen for the re-profiled channel and this will be maintained in the future.

Figure 4.3. Malin Bridge a) before flood defence work (2009), b) after flood defence work (2009) and c) after channel re-profiling (2011), riffle instalment and boulder placement.

4.3.3. Aims and objectives

The aim of this study was to determine the effects of flood mitigation and river rehabilitation works on the fish community at Malin Bridge, specifically brown trout, through application of a BACI assessment. Specific objectives were to compare suitable brown trout habitat availability, fish community structure and more specifically brown trout density and age structure pre and post flood mitigation and river rehabilitation works at the River Loxley and River Rivilin. Furthermore, additional representative control sites were included to test and establish a monitoring programme that will help assess the direct effect of river maintenance schemes on fish communities.

4.3.4. Fish survey methodology

Fisheries surveys were carried out at two sites, Rivelin and Loxley on 3 July 2009 (prior to flood defence works), 21 July 2010 (following flood defence works) and 21 July 2011 (following rehabilitation works) using quantitative electric fishing (Section 3.1.1) (Table 4.1). Three control sites, Hospital, Allotment and Rowel Lane were surveyed by the Environment Agency in 2009 and 2010.

Table 4.1. Fisheries survey site details in the Rivers Loxley and Rivelin in July 2009, July 2010 and July 2011

River / NGR	Site Name	Survey date	Length/mean width/area	Survey method and gear
Loxley SK32578932	River Loxley left hand channel between A6101 and B6076 bridges	03/07/09 21/07/10 21/07/11	47 m/6.6 m/310.2 m^2 39 m/11.8 m/460.2 m^2 41 m/8.5 m/349 m^2	Generator. Quantitative.
Rivelin SK32578932	River Rivelin right hand channel between A6101 bridge and small weir	03/07/09 21/07/10 21/07/11	67 m/6.5 m/435.5 m^2 58 m/6.3 m/365.4 m^2 58 m/6.0 m/348 m^2	Generator. Quantitative.
Rivelin SK29708727	Hospital	2009 2010	Area: 350	Generator. Quantitative.
Rivelin SK32108830	Allotment	2009 2010	Area: 350	Generator. Quantitative.
Loxley SK29918952	Rowel Lane	2009 2010	Area: 420	Generator. Quantitative.

The focus of the study was primarily brown trout (*Salmo trutta* L.) and bullheads (*Cottus gobio* L.), but lamprey may be present in the study reach (probably brook lamprey, *Lampetra planeri*, because of barriers to anadromous lamprey species migration in the Don catchment) as they are found in the upper reaches of the River Rivelin (Harvey & Cowx 2004). Lampreys are protected under European designation, thus it was recommended by the Environment Agency that the presence/absence of lampreys was assessed during the fish surveys (Section 3.1.1). After each electric fishing survey, habitat and environmental data were collected at each site in the format used for HABSCORE (Section 3.1.3).

4.4. Results

4.4.1. HABSCORE

Raw depth data from HABSCORE was used to quantify some of the changes in the channel before and after river rehabilitation (Figure 4.4), the habitat quality score (HQS Table 4.4) taken from the HABSCORE output, can also be used to quantify habitat improvement for brown trout at Rivers Loxley and Rivelin. There were slight variations in depths when comparing 2010 to 2011 Loxley data, with a noticeable reduction in depth of the weir pool (furthest up stream transect; Figure 4.4). The River Loxley's HQS increased between 2010 flood works and 2011 habitat rehabilitation work, for 0+, >1+ (<20cm) and >1+ (>20cm) (Table 4.4). There were noticeable variations in depths between 2010 and 2011 for the Rivelin (Figure 4.4). In 2010, the depth was mainly shallow (0-20cm) across the sampled area, with deep sections (40-60cm) upstream at the weir pool, whereas in 2011, changes in depth were variable (0-60cm) with more frequent deep pools (Figure 4.4). The HQS increased between for 0+ and >1+ (<20cm) trout between 2010 and 2011, however, there was a small decrease in HQS for >1+ (>20cm).

HABSCORE outputs for the sites on the Rivers Loxley and Rivelin revealed variations in the observed densities, predicted densities and habitat utilisation by trout (Tables 4.4). In the River Loxley at Malinbridge the observed densities of 0+ trout and >0+ trout (< 200mm) were higher in all years than predicted from the Habitat Quality Score (HQS), suggesting better populations than expected; the HUI lower CL was >1 for 0+ trout in 2010 and >0+ trout (< 200mm) in 2009 and 2010 indicating the populations were significantly higher than predicted (Table 4.4). The observed density of >0+ trout (> 200mm) was lower than predicted (HQS) in 2009 and 2011, suggesting poorer populations than expected but in 2010 was higher than predicted; the observed populations were not significantly lower or higher than predicted (Table 4.4). In the River Rivelin at Malinbridge in 2009 the observed density of 0+ trout was lower than predicted from the Habitat Quality Score (HQS), indicating poorer populations than expected, the HUI upper CL was >1 therefore the observed population was not significantly lower than expected (Table 4.4). In 2010 and 2011 the observed densities of 0+ trout were higher than predicted (HQS), suggesting better populations than expected; the HUI lower CL in 2010 was >1 therefore the observed population was significantly higher than expected in this year (Table 4.4). The observed densities of >0+ trout (< 200mm) and >0+ trout (> 200mm) in all years were higher than predicted (HQS), suggesting better populations than expected; the HUI lower C.L. was >1 for >0+ trout (< 200mm) in 2010 and 2011 therefore the observed populations were significantly higher than expected (Table 4.4).

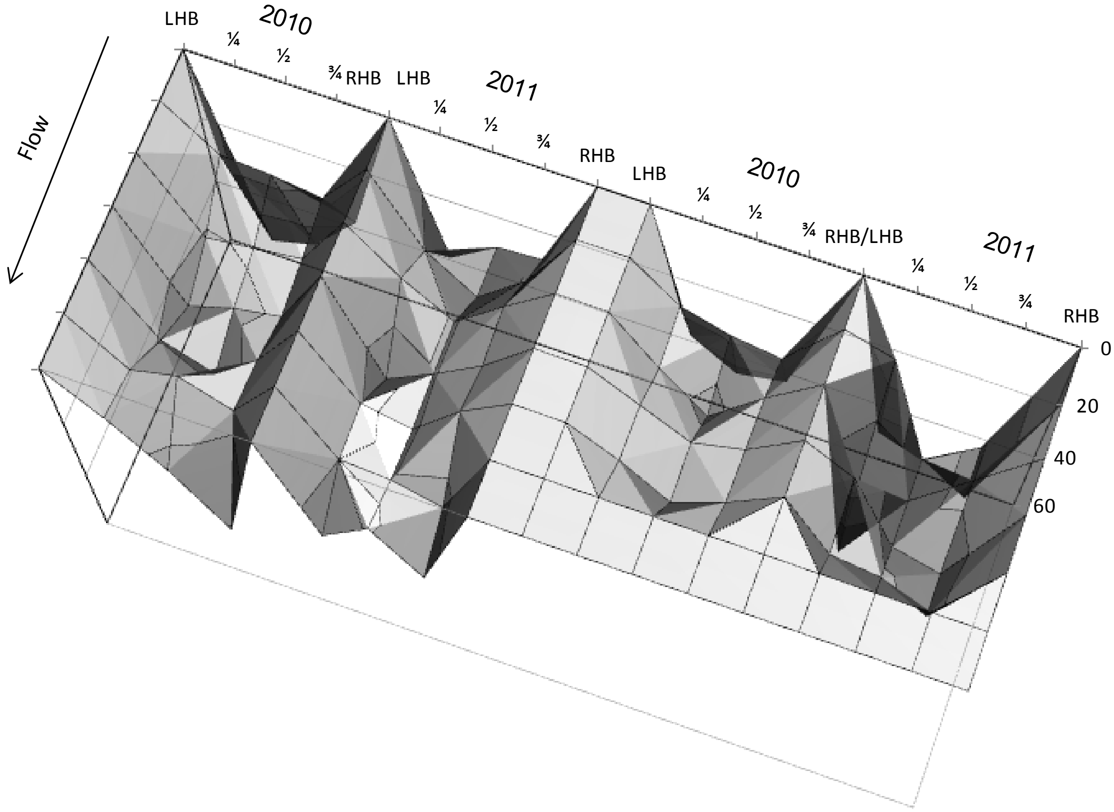


Figure 4.4. Changes in channel depth recorded during HABSCORE.

4.4.2. Status of fish populations

Densities of brown trout were similar in the rivers Loxley and Rivelin in surveys in 2009 (Figure 4.5, Table 4.3). Brown trout ≥1+ dominated catches in both rivers in 2009 and the presence of good densities of ≥1+ brown trout indicated good survival of brown trout from recruitment in 2008. Recruitment of brown trout occurred in both rivers in 2009 as indicated by the presence of 0+ fish (Figure 4.5, Table 4.3). Brown trout >200 mm were present in both rivers in 2009, but at lower densities than the smaller age classes (Figures 4.5, Table 4.3). 0+ brown trout populations in the rivers Loxley and Rivelin in 2009 were classified as fair/poor (class D), while ≥1+ brown trout populations were good (class B) in the River Loxley and fair (class C) in the River Rivelin (Table 5.3). Overall in 2009 the reach at Malinbridge contained fair/poor (class D) 0+ brown trout populations and good (class B) ≥1+ brown trout populations (Table 4.3) and was considered an important area for brown trout spawning and recruitment.

Densities of brown trout varied in the rivers Loxley and Rivelin in surveys in 2010 contrasting markedly with 2009 findings (Figure 4.5, Table 4.3). In both rivers, 0+ brown trout dominated catches in 2010 with the greatest densities in the River Rivelin, but densities in both rivers were considerably greater than recorded in 2009 (Figure 4.5, Table 4.3). ≥1+ brown trout densities were higher in the River Rivelin in 2010 than 2009, but ≥1+ brown trout densities were lower in the River Loxley in 2010 than in 2009 (Figure 4.5, Table 4.3). 0+ brown trout populations in the rivers Loxley and Rivelin in 2010 were classified good (class B) and excellent (class A), respectively. In both cases an improvement in 0+ abundance was found compared with 2009 (Table 4.3). ≥1+ brown trout populations in 2010 were classified fair (class C) in the River Loxley and good (class B) in the River Rivelin (Table 5.3). Overall in 2010 the reach at Malinbridge contained good (class B) 0+ brown trout populations and fair (class C) ≥1+ brown trout populations (Table 4.3) and was considered an important area for brown trout spawning and recruitment.

Densities of brown trout in the rivers Loxley and Rivelin were lower in 2011 than 2010 but higher than in 2009 (Figure 4.5, Table 4.2). In both rivers ≥1+ brown trout dominated catches in 2011 with the River Rivelin having the greatest densities of ≥1+ brown trout; ≥1+ brown trout densities were higher in 2011 than in 2009 or 2010 (Figure 4.5, Table 4.2). ≥1+ brown trout populations in 2011 were classified as good (class B) in the River Loxley and excellent (class A) in the River Rivelin (Table 4.3). 0+ brown trout densities were higher in the River Rivelin than the River Loxley and were lower than in 2010 but higher than in 2009 in both rivers (Figure 4.5, Table 4.2). 0+ brown trout populations in the rivers Loxley and Rivelin in 2011 were classified as fair/poor (class D) and fair (class C), respectively (Table 5.3). Overall in 2011 the reach at Malinbridge contained fair/poor (class D) 0+ brown trout populations and excellent (class A) ≥1+ brown trout populations (Table 4.3) and was considered an important area for brown trout spawning and recruitment.

4.4.3. Length frequency distributions

Length distributions of brown trout in the River Loxley in 2009 revealed 0+ individuals in the size range 39-52 mm, with ≥1+ individuals in the size range 90-314 mm (Figure 4.6). In the River Rivelin in 2009 0+ brown trout caught were in the size range 46-64 mm, with ≥1+ individuals in the size range 97-281 mm (Figure 4.7). Overall the reach at Malinbridge in 2009 contained 0+ brown trout in the size range 39-64 mm, with ≥1+ individuals in the size range 90-314 mm (Figure 4.8).

European Fish Index Calculations

The EFI + database calculate no change in ecological class boundaries (Class 1) between 2010’s flood mitigation and 2011’s river rehabilitation at both Rivelin (Fish Index for 2010=1.00 and 2011=1.00) and Loxley (Fish Index for 2010=1.00 and 2011=0.99).

Table 4.2. Total population estimate (N), population density (D) and probability of capture (P) (± 95% C.L. at quantitative sites) of trout derived from fisheries surveys in Rivers Loxley and Rivelin at Malinbridge in July 2009, 2010 and 2011 (density of fish given as numbers per 100m^2). Details of derivation of estimates are provided in the text. At quantitative sites (*) the probability of capture did not differ significantly between runs ($\chi^2 < \chi^2_{0.95}$ (3.84)). N/A = not applicable.

Site	Total Population (N)		Population density (D)		Probability of capture (P) (χ^2 value)	
No.	0+	≥1+	0+	≥1+	0+	≥1+
Loxley 2009	10±3	39±6	3.22±1.02	12.57±2.12	*0.47±0.28 (3.07)	*0.54±0.19 (1.42)
Loxley 2010	121±18	25±1	26.29±3.98	5.43±0.23	*0.54±0.11 (0.16)	*0.78±0.13 (0.14)
Loxley 2011	21±2	49±1	6.03±0.61	14.06±0.38	*0.68±0.20 (0.72)	*0.79±0.09 (0.16)
Rivelin 2009	14±5	52±4	3.21±1.15	11.94±1.07	*0.42±0.29 (0.81)	*0.64±0.15 (0.11)
Rivelin 2010	168±10	67±3	46.03±2.73	18.36±0.72	*0.61±0.09 (2.46)	*0.74±0.10 (0.14)
Rivelin 2011	32±3	104±4	9.20±0.91	29.89±1.15	0.66±0.17 (5.27)	*0.72±0.10 (0.24)
Loxley/Rivelin combined 2009	27±9	92±8	3.62±1.32	12.34±1.12	*0.38±0.26 (1.33)	*0.58±0.12 (0.38)
Loxley/Rivelin combined 2010	290±15	92±3	35.14±1.91	11.15±0.36	*0.57±0.07 (1.00)	*0.75±0.09 (0.23)
Loxley/Rivelin combined 2011	54±5	153±4	7.75±0.66	21.97±0.59	*0.64±0.15 (2.45)	*0.75±0.08 (0.36)

Length distributions of brown trout in the River Loxley in 2010 revealed 0+ individuals in the size range 35-75 mm, with ≥1+ individuals in the size range 97-284 mm (Figure 4.6). In the River Rivelin in 2010 0+ brown trout caught were in the size range 43-80 mm, with ≥1+ individuals in the size range 105-297 mm (Figure 4.7). Overall the reach at Malinbridge in 2010 contained 0+ brown trout in the size range 35-80 mm, with ≥1+ individuals in the size range 97-297 mm (Figure 4.8).

Length distributions of brown trout in the River Loxley in 2011 revealed 0+ individuals in the size range 48-59 mm, with ≥1+ individuals in the size range 97-275 mm (Figure 4.6). In the River Rivelin in 2011 0+ brown trout caught were in the size range 48-77 mm, with ≥1+ individuals in the size range 103-249 mm (Figure 4.7). Overall the reach at Malinbridge in 2011 contained 0+ brown trout in the size range 48-77 mm, with ≥1+ individuals in the size range 97-275 mm (Figure 4.8). Size distribution did not vary between sites or years suggesting that the flood works and rehabilitation works did not impact on the population structure of brown trout.

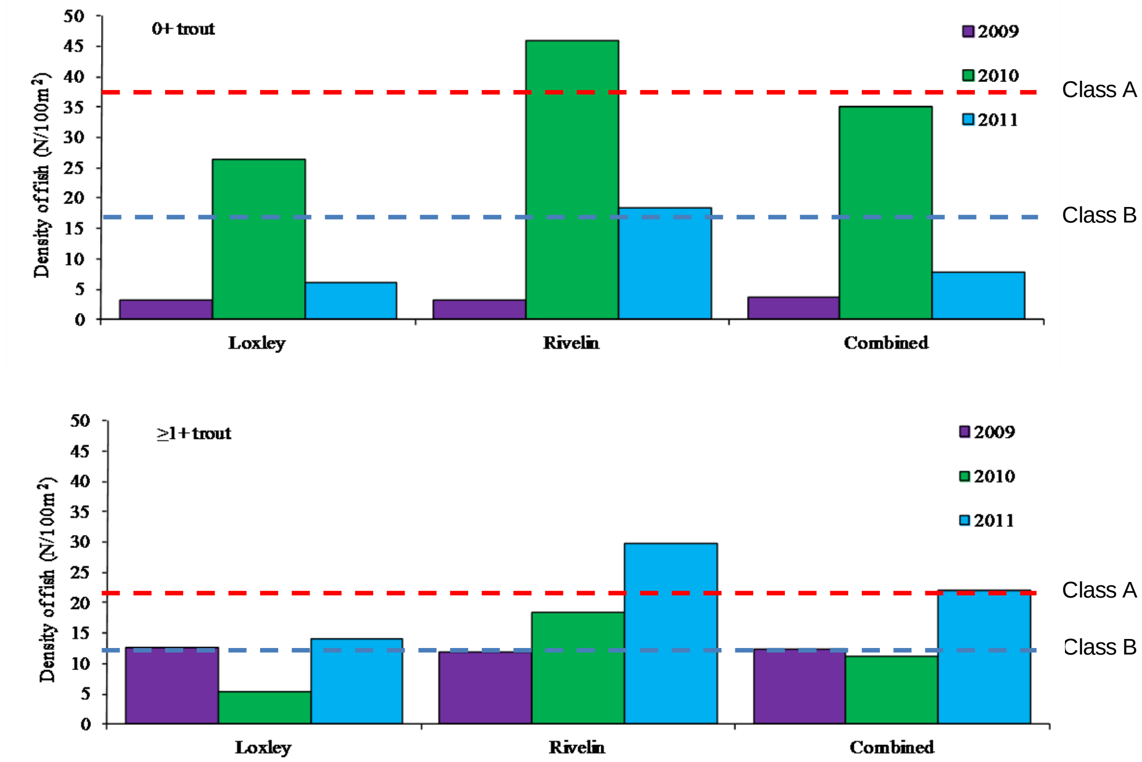


Figure 4.5. Density estimates of trout in Rivers Loxley and Rivelin between 2009 and 2011, Class A indicates excellent population density and Class B indicates good population density.

Table 4.3. Abundance classifications of trout in Rivers Loxley and Rivelin at Malinbridge, in July 2009, 2010 and 2011, derived from comparison of densities with the EA-FCS.

Site No.	Age group	2009	2010	2011
Loxley	0+ trout	D	B	D
	≥1+ trout	B	C	B
Rivelin	0+ trout	D	A	C
	≥1+ trout	C	B	A
Loxley/Rivelin combined	0+ trout	D	B	D
	≥1+ trout	B	C	A

Large numbers of bullheads were caught in the Rivers Loxley and Rivelin in 2009, indicating the importance of the reach at Malinbridge for this species. In the River Loxley bullheads caught were in the size range 40-81 mm while in the River Rivelin bullheads caught were in the size range 21-89 mm (Figures 4.9-4.11). Bullheads spawn between February and June and fractional reproduction is common in productive lowland rivers (Fox 1978). Generally, bullheads attain a length of 40–50 mm after their first year, 60 mm after their second and 70–90 mm after their third (Maitland & Campbell 1992). Therefore at Malinbridge in 2009 the majority of bullheads captured were ≥1+ individuals and the low number of 0+ bullheads (two fish of 21 and 26 mm) would suggest that spawning/hatching of bullheads had not fully occurred when surveys were carried out in 2009. Indeed, surveying for bullheads is recommended from August to October to ensure that 0+ individuals are captured if spawning occurs late in the summer (Cowx & Harvey 2003). In 2010 there were again high numbers of bullheads caught in the rivers Loxley and Rivelin indicating the importance of the reach for the species (Figures 4.9-4.11). In both rivers there was evidence of good numbers of 0+ bullheads probably due to the surveys in 2010 being carried out 18 days later than in 2009, and also possibly indicating earlier hatching/spawning of bullheads in 2010 compared with 2009. In 2011, bullheads were caught in high numbers in the River Rivelin and in lesser numbers in the River Loxley, with 0+ bullheads only caught in the River Rivelin (Figures 4.9-4.11).

One stoneloach (*Barbatula barbatulus* (L.)) of 104 mm was caught in the River Loxley in 2009 and four (Total Length (TL) 75, 90, 117 and 120 mm) were caught in the River Riverlin. Three perch (*Perca fluviatilis* L.) (Fork Length (FL) 140, 140 and 153 mm) and one tench (*Tinca tinca* (L.)) (TL 150 mm) were also caught in the River Rivelin in 2009. In 2010, five stoneloach (TL 82, 88, 105, 112 and 132 mm) were captured in the River Loxley but they were absent in the River Rivelin stoneloach. Three perch (FL 140, 140 and 178 mm) were caught in the River Loxley in 2010 while three perch (FL 137, 148 and 157 mm) and one mirror carp (FL109 mm) were caught in the River Rivelin. In 2011, two stoneloach (TL 66 and 116 mm) were captured in the River Loxley while no other species apart from brown trout and bullheads were found in the River Rivelin.

During fisheries surveys in 2009, 2010 and 2011, little suitable juvenile lamprey habitat was found in either river and no juvenile lampreys were captured or observed, however they are historically known to be present at Malinbridge (C. Essery pers. comm.) and in the River Rivelin upstream of Malinbridge (pers. obs.).

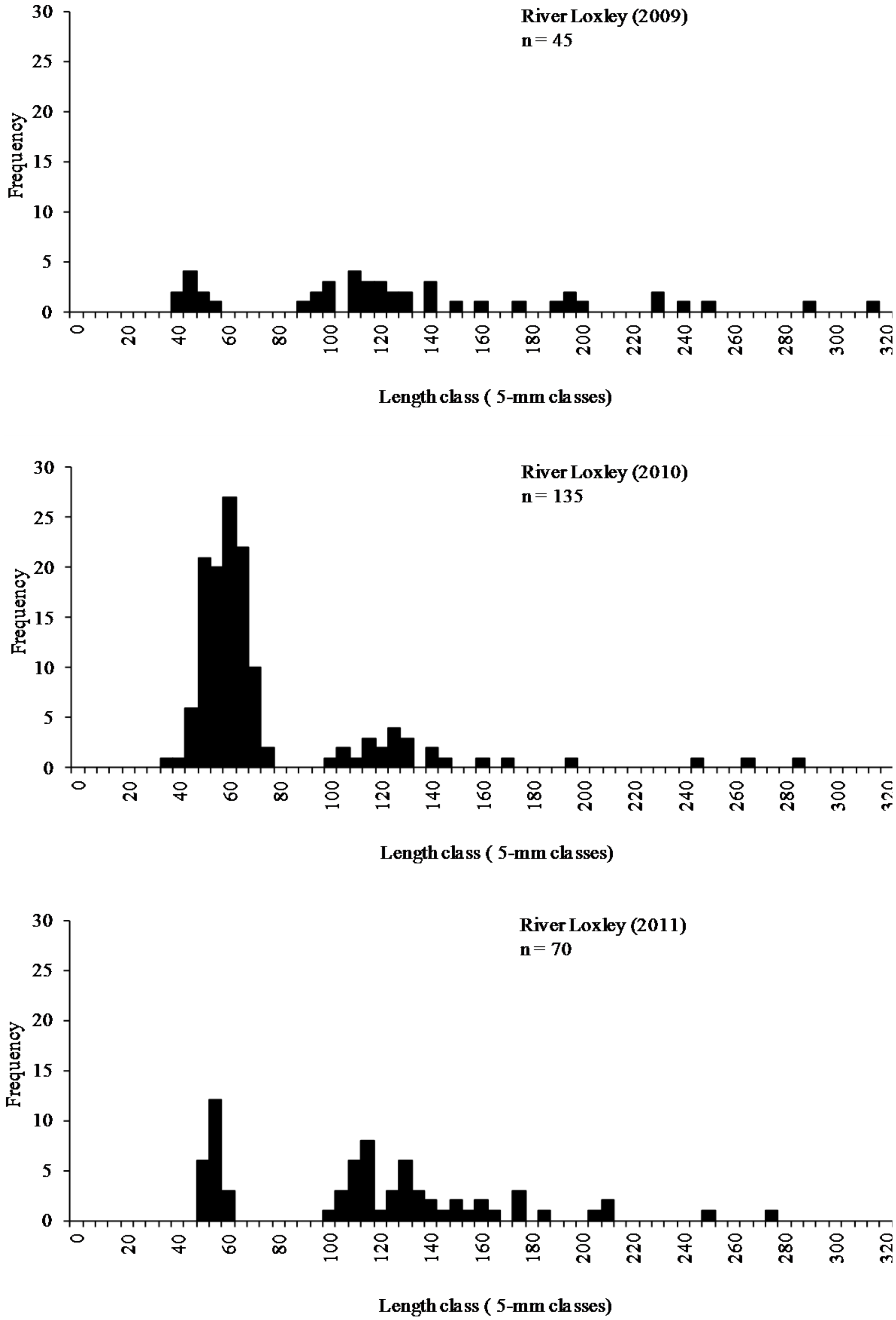


Figure 4.6. Length distributions of brown trout in the River Loxley, Malinbridge in July 2009, 2010 and 2011

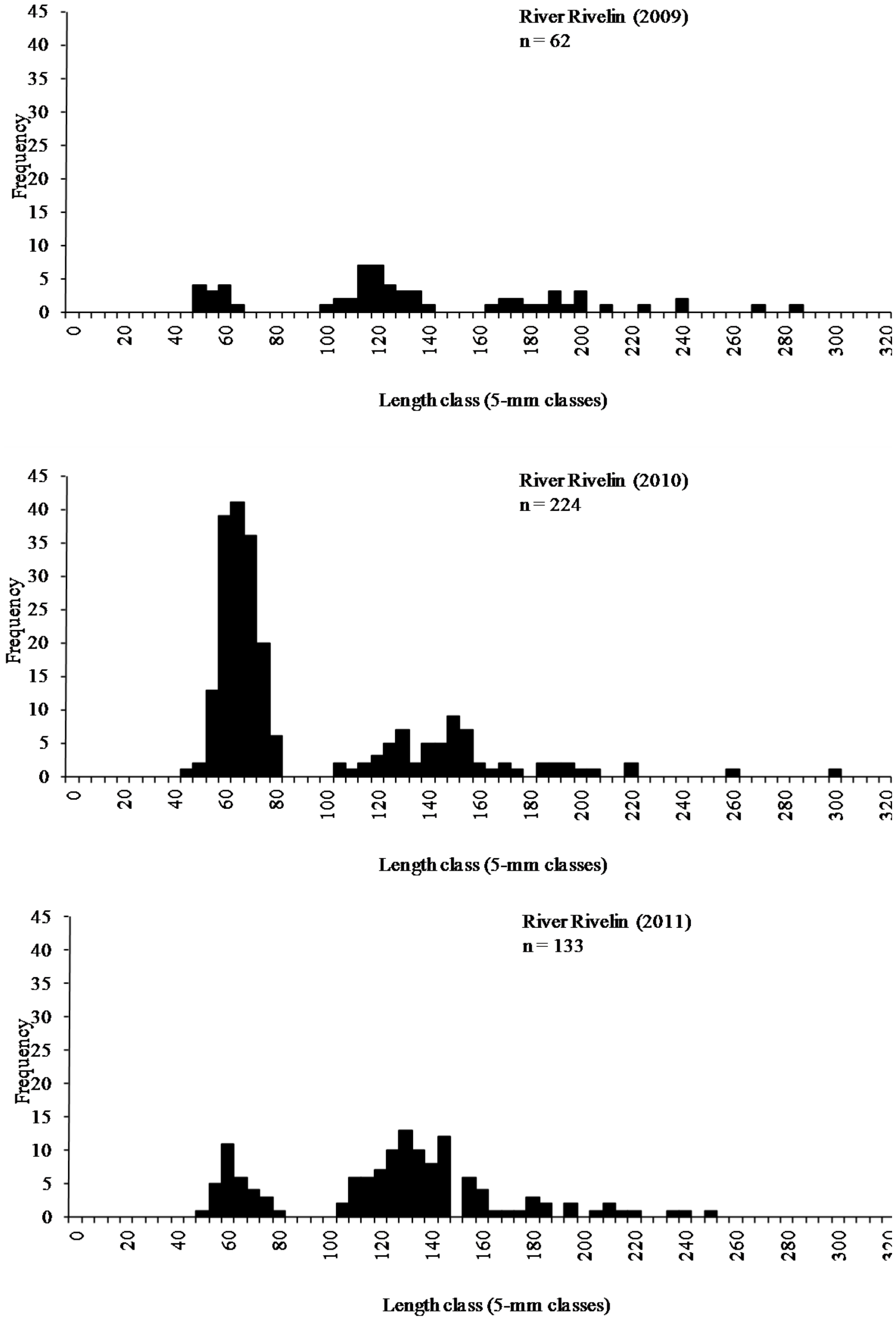
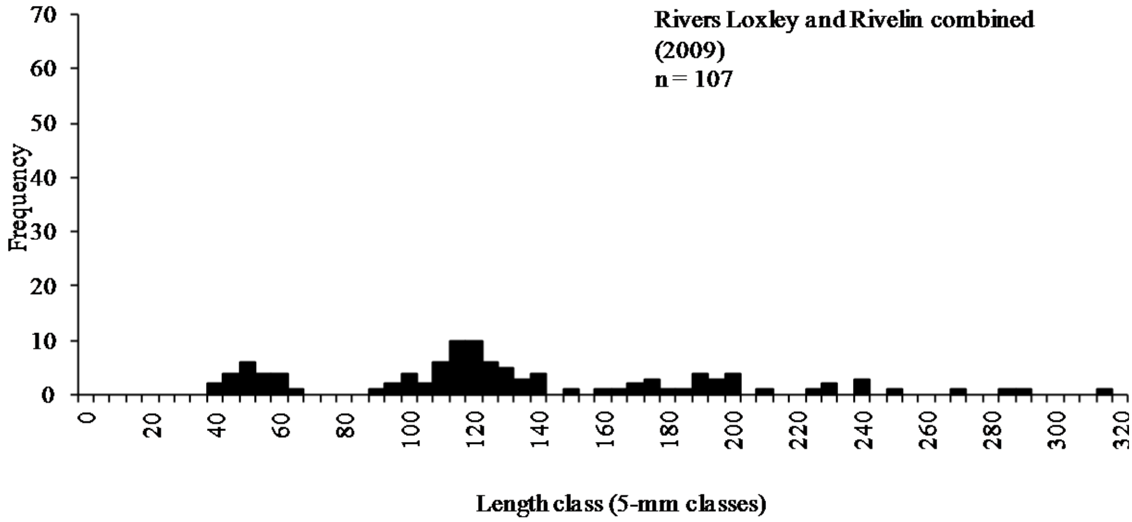
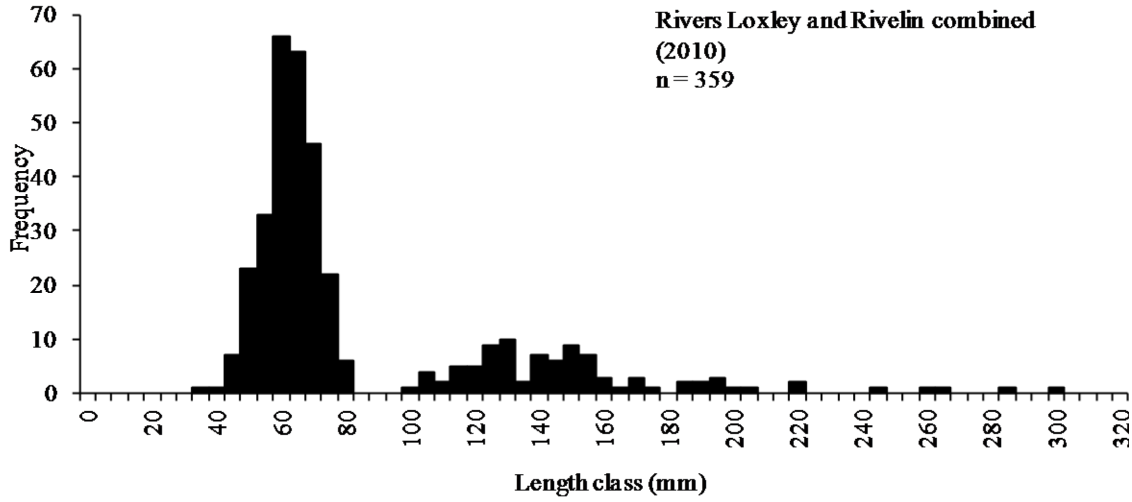


Figure 4.7. Length distributions of brown trout in the River Rivelin, Malinbridge in July 2009, 2010 and 2011





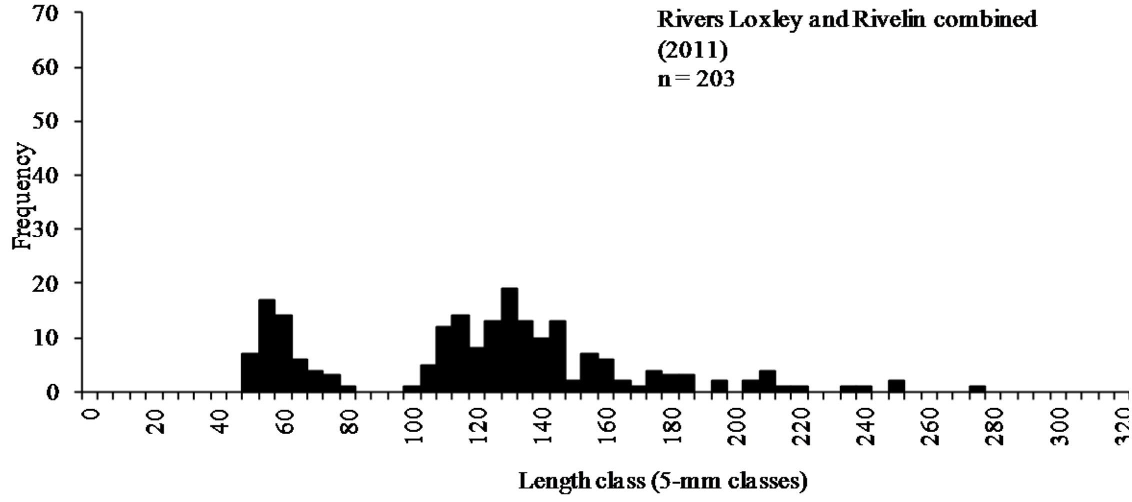


Figure 4.8. Length distributions of brown trout combined in the Rivers Loxley and Rivelin, Malinbridge in July 2009, 2010 and 2011

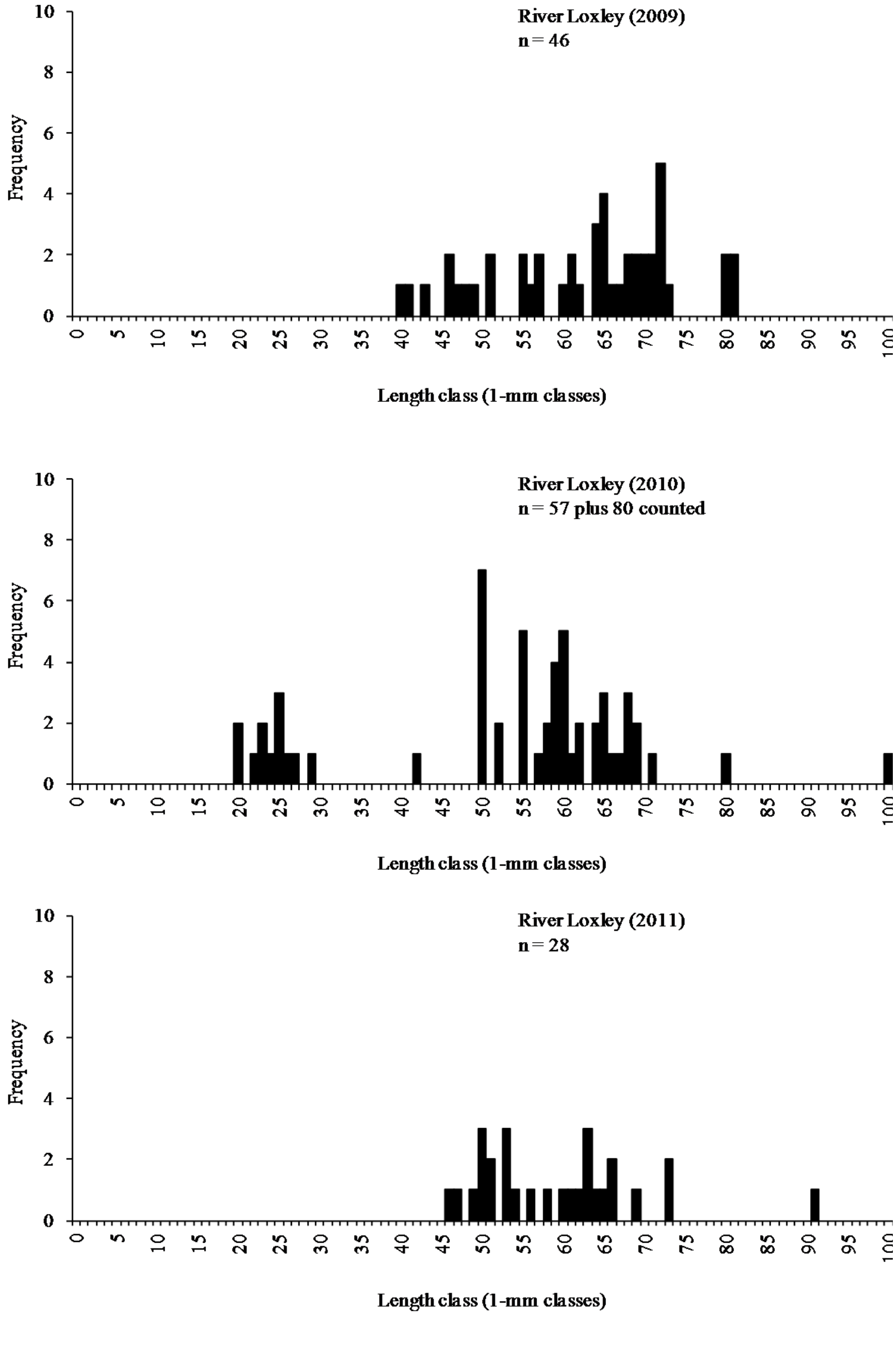
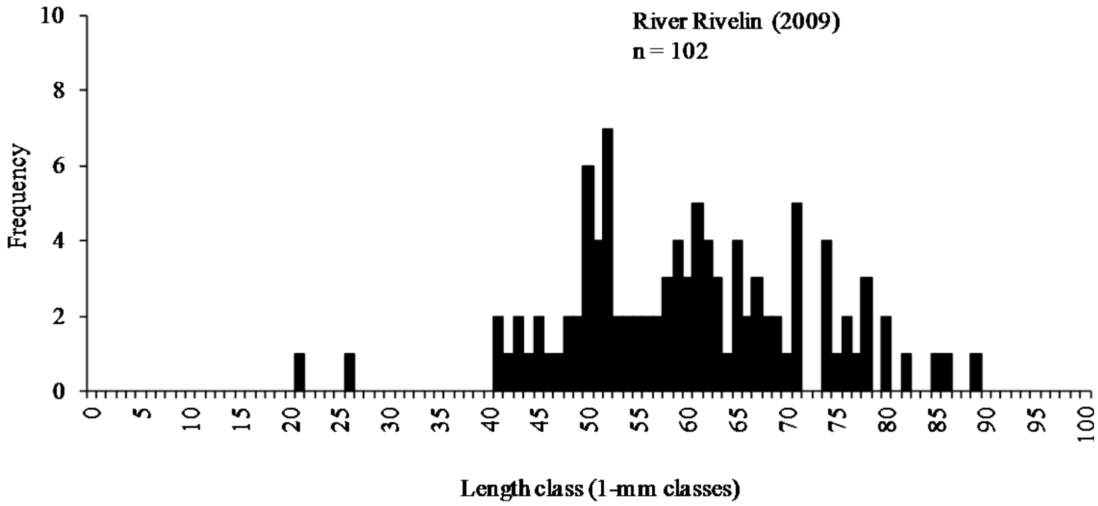
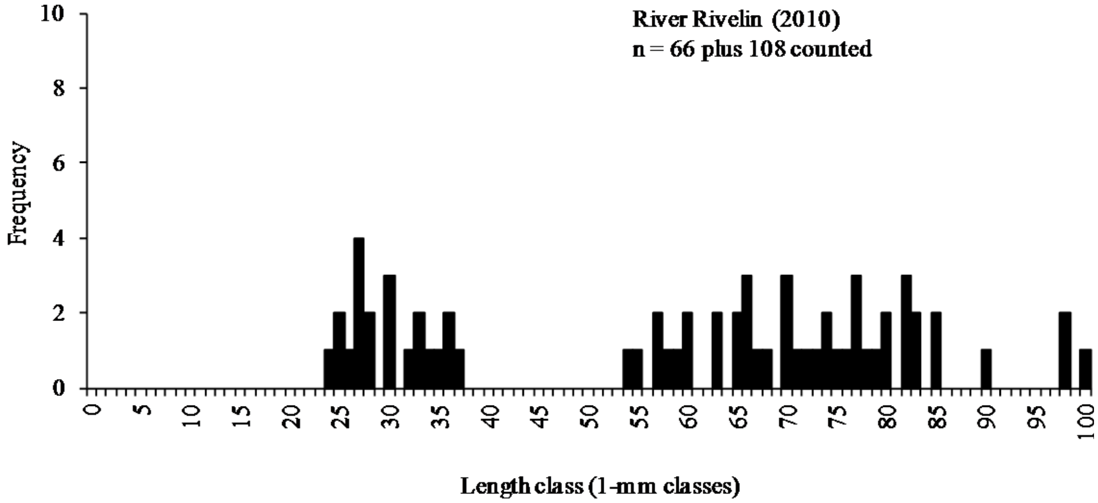


Figure 4.9. Length distributions of bullheads in the River Loxley, Malinbridge in July 2009, 2010 and 2011





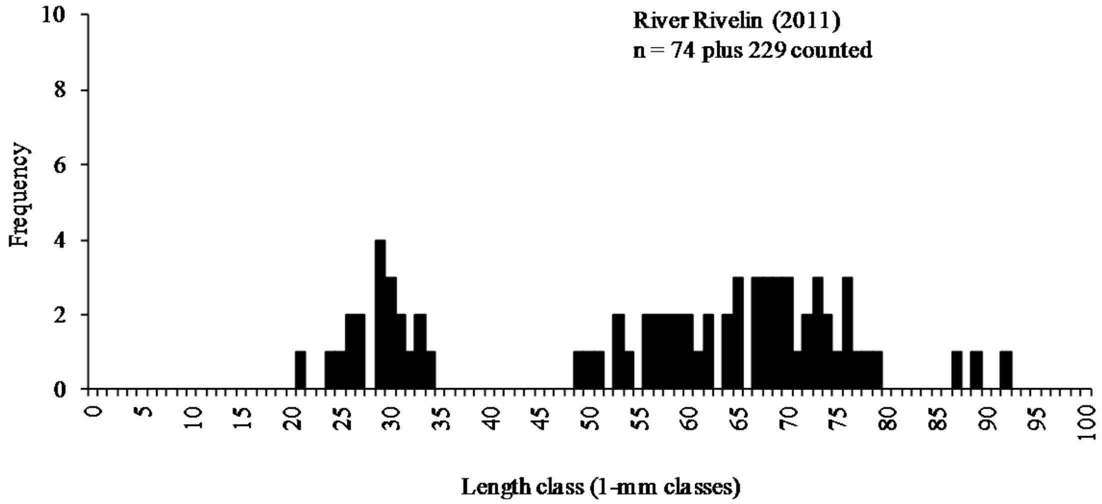


Figure 4.10. Length distributions of bullheads in the River Rivelin, Malinbridge in July 2009, 2010 and 2011

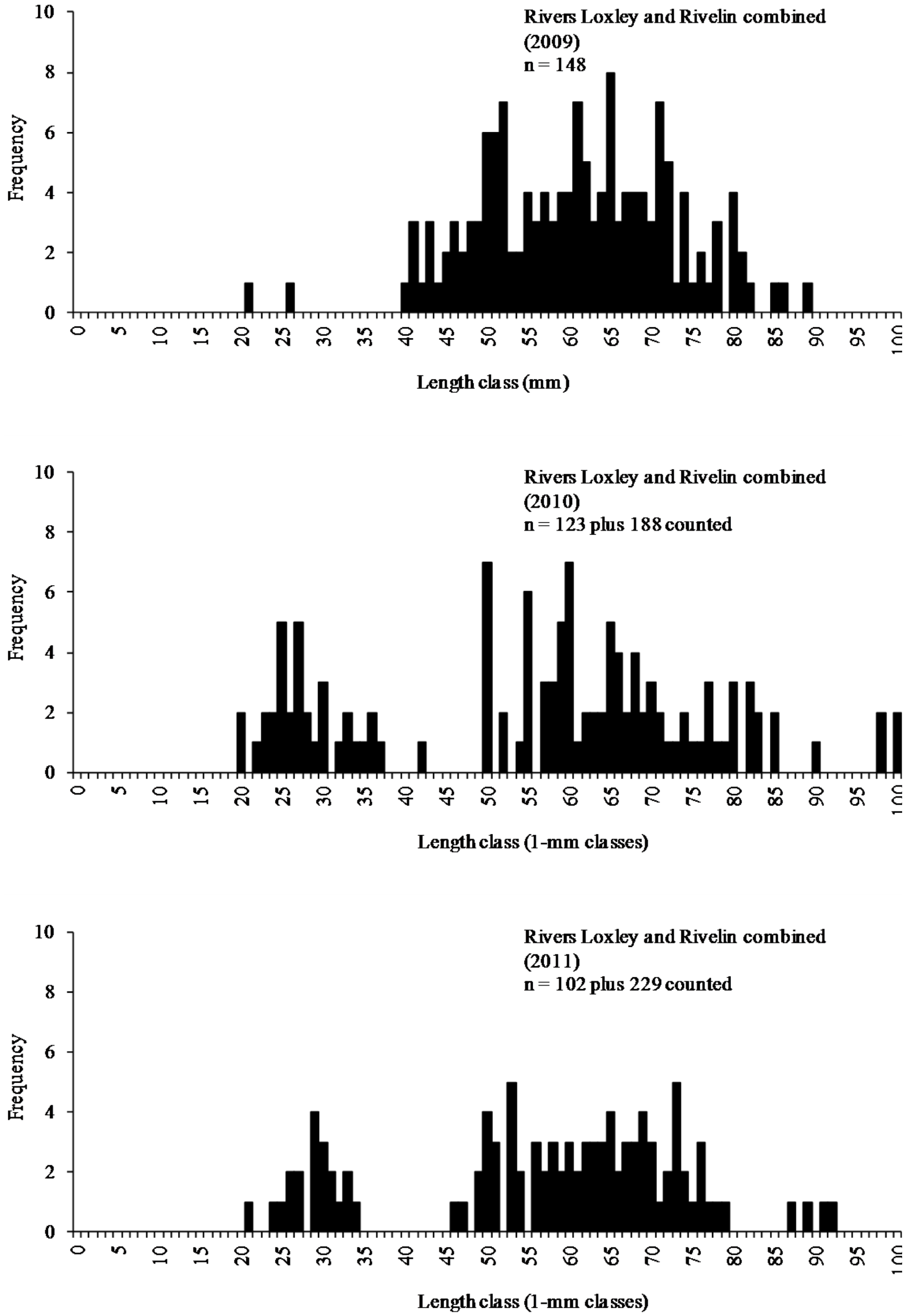


Figure 4.11. Length distributions of bullheads combined in the Rivers Loxley and Rivelin, Malinbridge in July 2009, 2010 and 2011

Table 4.4. HABSCORE outputs for the River Loxley at Malinbridge. (Note: Shaded area represents sites where the observed population was significantly higher (HUI lower CL column) or lower (HUI upper CL column) than would be expected under pristine conditions).

Age/size group / fisheries data group	Observed number	Observed density	HQS (density)	HQS lower CL	HQS upper CL	HUI	HUI lower CL	HUI upper CL
0+ trout								
2009	10.0	1.60	1.08	0.25	4.18	1.48	0.22	10.21
2010	121.0	25.49	2.18	0.57	8.38	11.69	1.73	79.02
2011	21.0	6.31	4.93	1.31	18.56	1.28	0.19	8.51
>1+ trout (<20 cm)								
2009	33.0	5.28	0.32	0.07	1.48	16.54	2.54	106.76
2010	22.0	4.63	0.26	0.06	1.19	17.61	2.81	110.34
2011	44.0	13.22	2.96	0.69	12.66	1.54	0.49	4.84
>1+ trout (>20 cm)								
2009	6.0	0.96	1.56	0.45	5.37	0.62	0.18	2.14
2010	3.0	0.63	0.50	0.16	1.62	1.26	0.39	4.06
2011	5.0	1.50	1.54	0.49	4.84	0.97	0.31	3.06

Table 4.5. HABSCORE outputs for the River Rivelin at Malinbridge. (Note: Shaded area represents sites where the observed population was significantly higher (HUI lower CL column) or lower (HUI upper CL column) than would be expected under pristine conditions).

Age/size group / fisheries data group	Observed number	Observed density	HQS (density)	HQS lower CL	HQS upper CL	HUI	HUI lower CL	HUI upper CL
0+ trout								
2009	14.0	3.34	6.03	1.61	22.62	0.55	0.08	3.75
2010	168.0	47.10	4.99	1.29	19.32	9.44	1.39	63.89
2011	32.0	9.96	8.37	2.20	31.82	1.19	0.18	7.96
>1+ trout (<20 cm)								
2009	46.0	10.99	3.90	0.92	16.58	2.81	0.47	16.81
2010	60.0	16.82	0.62	0.14	2.77	27.09	4.37	168.09
2011	96.0	29.88	2.55	0.61	10.71	11.71	1.98	69.17
>1+ trout (>20 cm)								
2009	6.0	1.43	1.25	0.40	3.89	1.15	0.37	3.59
2010	6.0	1.68	0.86	0.28	2.70	1.95	0.62	6.10
2011	8.0	2.49	0.82	0.26	2.53	3.05	0.99	9.44

4.4.4. Impact assessment and resource calculation

A BACI design (Section 3.2.4, Equation 3.3) was used to assess the effects of flood mitigation works on 0+ and >1+ brown trout populations at Malin Bridge. The impact assessment was assessed both in terms of confidence limits of the mean difference in 0+ and >1+ brown trout density and by means of a t-test. The mean change in 0+ brown trout density between the impact sites (M = 2.14, SE = 0.27) and the control sites (M = -0.19, SE = 0.64), t (1) = -2.57, was 3.89 ± 2.16 fish per 100 m^2, and the 95% confidence limits (1.73 – 6.05) does not enclose zero, however, this was not significant (p= *0.24 (p* > 0.05), r = 0.93). The mean change in >1+ brown trout density between the impact sites (M = 0.99, SE = 0.30) and the control sites (M = -0.38, SE = 0.42), t (1) = 1.93, was 1.31 ± 2.10 fish per 100 m^2, and the 95% confidence limits (-0.79 – 3.41) enclosed zero and therefore the difference was not significant (p= 0.30 (p > 0.05), r = 0.89). The standard errors for the mean difference in 0+ and >1+ brown trout at impact sites was low and therefore there was a low amount of variability between the means of different samples. The standard errors for the mean difference in 0+ and >1+ at the control sites was large and therefore a large amount of variability between the means of different samples. A resource calculation cannot be performed because there was insufficient temporal data and therefore temporal variance cannot be estimated. Temporal data collection was limited before the PhD commenced and was further limited to the timeframe of the PhD.

4.5. Discussion

Ecological integrity is usually compromised for flood risk management and it is only in more recent years that management methods have encouraged river rehabilitation to be incorporated into management plans. The Environment Agency has a ministerial Flood Defence High Level Target to "*Ensure no net loss to habitats covered by Biodiversity Action Plans and seek opportunities for environmental enhancements*" (Environment Agency 2007: Policy Number: 606_06) and therefore, the Environment Agency and its partners' promote a more environmentally sound approach to sustainable river management that underpins the WFD (Environment Agency 2007). Unfortunately, the rehabilitation of urban stream channels are highly constrained (Bernhardt & Palmer 2007) and their reinstatement to pre-disturbed conditions is an impractical vision since there are constant external pressures on this type of river system. For this reason the WFD classification of river health is to aim for '*good ecological potential*' for Heavily Modified Water Bodies (HMWB) to ensure RBMP do not avoid these more complex systems.

The output of the fisheries and habitat surveys at Malinbridge in 2009 provided an indication of the baseline status of fish populations in the Rivers Loxley and Rivelin prior to the flood defence works in 2010 and the further rehabilitation works in 2011.

When assessing ecological status of the Rivers Rivelin and Loxley, at a European scale, the EFI identified no change before or after flood mitigation and restoration works. However, when looking in more detail at the focal species, brown trout, small changes were found. Brown trout populations in 2009 were overall classed as fair (class C), similar to other sites in the Loxley and Rivelin catchments. The presence of 0+ individuals and good numbers of ≥1+ brown trout indicated the reach was an important spawning and nursery area for brown trout. HABSCORE in the River Loxley in 2009 revealed marginally higher densities of 0+ trout than predicted; >1+ trout (<200 mm) were significantly higher than predicted indicating that the populations were generally greater than would be expected. HABSCORE in the River Rivelin in 2009 revealed 0+ trout densities were lower than predicted, although not significantly, indicating recruitment potential was not achieved relative to the habitat conditions at the site. >1+ trout (<200 mm) and >1+ trout (>200 mm) densities were higher than predicted, this was significant ad therefore indicates that the populations were generally greater than would be expected. Bullheads were found in high numbers at Malinbridge in 2009 and the reach was considered to be an important area for this species.

Flood defence works were carried out in winter 2009/spring 2010 resulting in considerable alteration to the habitat structure, flow regime and also uncovering a small weir on the River Loxley. Flood mitigation resulted in the removal of shoal material, instream and bankside vegetation, therefore reducing habitat diversity in the channel. Fisheries surveys in July 2010 following completion of the flood defence works allowed comparison of data with findings from surveys in 2009. 0+ brown trout dominated catches in 2010 in both the Rivers Loxley and Rivelin, with densities considerably greater than in 2009. Abundance classifications for both rivers combined indicated an improvement from fair/poor (class D) in 2009 to good (class B) in 2010. These data suggest the flood defence works have improved the reach for juvenile trout, possibly a result of creating a shallow habitat more favourable to small trout (which may have expedited easier capture of fish). However, the removal of cover for small trout (e.g. overhanging trees, large boulders) may make these fish more susceptible to predation. Additionally the conclusion should be treated with caution as 2010 appeared to be a good year for trout recruitment in the Loxley and Rivelin (pers. comm. D. Smallwood), which may have masked any impact of the flood defence works. By contrast, ≥1+ brown trout abundance classifications for both rivers combined indicated a decline from good (class B) in 2009 to fair (class C) in 2010. These data suggest the flood defence works have caused deterioration in quality of habitat for larger trout, probably a result of the shallowing of the river bed meaning limited deeper pools for larger trout. It is likely that larger trout in the river will have moved out of the study reach and used suitable habitat downstream and upstream of the reach. HABSCORE in the Rivers Loxley and Rivelin in 2010 revealed some variation in the actual and predicted densities of trout. In the River Loxley and River Rivelin, HABSCORE revealed significantly higher densities of 0+ trout and ≥1+ trout (<200 mm) than predicted, indicating that the populations were generally greater than would be expected; ≥1+ trout (>200 mm) densities were lower than predicted based on the habitat. This suggests that the populations of trout <200 mm (i.e. 0+ and 1+ trout) were higher than expected from the habitat, but this is surprising since the flood mitigation works at Malin Bridge resulted in the removal of shoal, instream and bank side vegetation, therefore reducing habitat diversity in the channel (Cowx & Welcomme 1998).

Following the flood defence works the Environment Agency initiated a series of rehabilitation works in 2010/2011 to improve habitat by channel re-profiling and installation of instream boulders. The Environment Agency also planned to install fish passes on three weirs in autumn/winter 2011. Fisheries surveys in July 2011 following completion of the rehabilitation works allowed comparison of data with findings from surveys in 2009 and 2010. ≥1+ brown trout dominated catches in both the rivers Loxley and Rivelin in 2011, at densities greater than in 2009 and 2010. Abundance classifications for both rivers combined were good (class B) in 2009, fair (class C) in 2010 and excellent (class A) in 2011. These data suggest the rehabilitation works may have improved the quality of habitat for larger trout, probably a result of the deepening of the river bed in places meaning an increase in deeper pools for larger brown trout. However, the higher densities of ≥1+ brown trout may also be a result of good survival of the large number of 0+ fish found in 2010. 0+ brown trout densities were lower in both rivers in 2011 compared with 2010, but were higher than found in 2009 prior to the flood defence/rehabilitation works. Abundance classifications of 0+ trout for both rivers combined were fair/poor (class D) in 2009, good (class B) in 2010 and fair/poor (class D) in 2011. 0+ brown trout populations in 2011 were similar to those found in 2009 and represent a return to the more typical densities encountered in the more diverse habitat present in these years. HABSCORE in the rivers Loxley and Rivelin in 2011 revealed some variation in the actual and predicted densities of trout., HABSCORE revealed higher densities of 0+ trout and ≥1+ trout (<200 mm) in the River Loxley and River Rivelin than predicted, indicating that the populations were generally greater than would be expected. This suggests that the populations of trout <200 mm (i.e. 0+ and 1+ trout) were higher than expected compared with the habitat. ≥1+ trout (>200 mm) densities were higher than predicted in the River Rivelin and marginally lower than predicted in the River Loxley in 2011.

Overall, based on one year’s data the flood defence works appeared to have improved habitat for 0+ brown trout and caused a deterioration of habitat for ≥1+ brown trout. The flood defence works changed the habitat in the reach from a diverse pool/riffle sequence with overhanging vegetation and variable substrate to a reach of uniform shallow depth, substrate and flow. Aquatic and riparian vegetation are vital for healthy and sustainable watercourse ecosystem, even more so on an urban river system where there is little or no variation in physical structure (Cowx & Welcomme 1998). The potential impacts of channel clearance work have been found to remove key habitat features causing a direct impact on fish populations (Downs & Thorne 1998), but this does not seem to be the case in the Malin Bridge post-flood defence surveys.

Brown trout require diverse habitats throughout their life cycle with shallow riffle areas for use by juveniles (0+ fish) and deeper glides and pools for larger individuals (Cowx & Welcomme 1998); this reduces the potential for competition for food and habitat within species. It is this suitable habitat that the Malin Bridge rehabilitation works endeavoured to create. Following the rehabilitation works, 0+ brown trout populations decreased (BACI identified the decrease was not significant) but were at the level found prior to flood defence works and hence reflected the more variable habitat found at the site in 2009. ≥1+ brown trout populations increased (BACI assessment indicated that this was not significant) in 2011 compared with previous years, probably due to a combination of the good recruitment in 2010 and more diverse habitat provided. Vegetation is a main habitat feature for fish and provides cover along with many other benefits (Cowx & Welcomme 1998; FAO 2008). The absence of this habitat may have contributed to the reduction in 0+ trout numbers following flood defence work. The absence of vegetation from Malin Bridge does not seem to have impacted on ≥1+ brown trout populations; this may be because the habitat created by the new boulders provides suitable cover for the larger trout that would usually have been provided by vegetation cover. Observations during the 2011 fish surveys identified deep, scoured areas behind the large boulders where large trout were captured illustrating their importance as a rehabilitation technique, findings also supported by Downs & Thorne (1998). Nevertheless, 0+ trout were not found to be using the boulders and therefore, there is certainly a need for further rehabilitation to provide cover for smaller fish. In most cases riparian rehabilitation efforts through replanting would provide necessary habitat for all fish life stages, especially 0+ trout (Cowx & Welcomme 1998; Bernhardt & Palmer 2007), but this is not the most suitable rehabilitation technique to integrate into urban flood mitigation planning. The Environment Agency considered replanting the re-profiled banks at Malin Bridge, but to manage natural re-growth of vegetation on site was the preferred option. Further to replanting or management of natural re-growth, the control of invasive species is vital at modified sites such as Malin Bridge because invasive plants such as Himalayan balsam and Japanese knotweed are opportunistic and out compete native plants. In 2011, observations identified a large quantity of Himalayan balsam present at the site along with a lower number of native species. Control measures for the removal of Himalayan balsam should aim to prevent flowering and are best carried out before June. Chemical control can be carried out but will be likely to kill other plants in the area, thus cutting and removal of the plant roots on a regular basis across a 3 year period will likely be most effective (Environment Agency 2010). Maintaining the re-growth of native plants is an important action in river rehabilitation to reduce the cost and time attempting to eradicate invasive species; this process seems to have elapsed throughout the habitat modification procedure at Malin Bridge. It is essential the maintenance of native bank side vegetation is built into the long term goals of any rehabilitation project framework.

Bullheads were found in high numbers at Malinbridge in 2009, 2010 and 2011 and the reach was considered an important area for this species; there was no obvious impact of flood defence or rehabilitation works on bullhead populations.

The findings within this study should be treated with caution as only one year’s data were collected post-rehabilitation. Future monitoring at Malin Bridge will enable the functionality of this rehabilitation action to be assessed and increase our knowledge of how to incorporate river rehabilitation successfully into the planning stage to work towards to FD and WFD collectively. The BACI analysis, through use of control sites enabled some spatial variability to be overcome, but the monitoring design was limited temporally (two years pre and one year post) and it is recommended further monitoring for an additional number of years is carried out to overcome temporal variability. The duration of monitoring needed to detect a change is particularly important in any type of monitoring study design, to overcome natural variability and identify a possible change as a result of the impact. Unfortunately, a resources calculation could not be performed to work out the duration and intensity of monitoring needed to overcome natural variability because of insufficient temporal data.

4.5.1. Recommendations

Malin Bridge flood works are proposed as part of an ongoing programme to alleviate flooding issues in the Sheffield area and it should be highlighted that it is a work in progress. Although the present data do not necessarily demonstrate that flood mitigation and rehabilitation methods at Malin Bridge had negative effects on the extant fish population, the rehabilitation techniques applied did not overly prove beneficial to the fish populations either. It is essential that long term data are collected to identify trends and overcome natural variability. **It is recommended that monitoring continues in all sections following flood defence works to identify any changes in fish populations in response to habitat modifications and therefore increase our knowledge of flood risk management strategies that incorporate rehabilitation practise.**

Furthermore, the Environment Agency plan to construct fish passes on the three weirs at Malinbridge in 2013, and additional assessment through mark recapture of fish should assess fish movement upstream of these weirs. The assessment of these fish passes will contribute to the Environment Agency's '*fish pass easement'* project that proposes to open up the longitudinal connectivity of the River Loxley and River Rivelin.

4.6 SHEFFIELD FLOOD MITIGATION & RESTORATION WORKS

Following the June 2007 floods, the Environment Agency planned and managed flood mitigation work to reduce the flood risk in the Sheffield area by removing obstructions within the River Don channel at four sites (Nursery Street, Blonk Street, Effingham Street and Brightside) (Figure 4.12). The following chapter will provide an insight in to the subsequent flood mitigation measures and methods and the importance of integrating river rehabilitation techniques into flood risk management, with fish populations being a focal point.

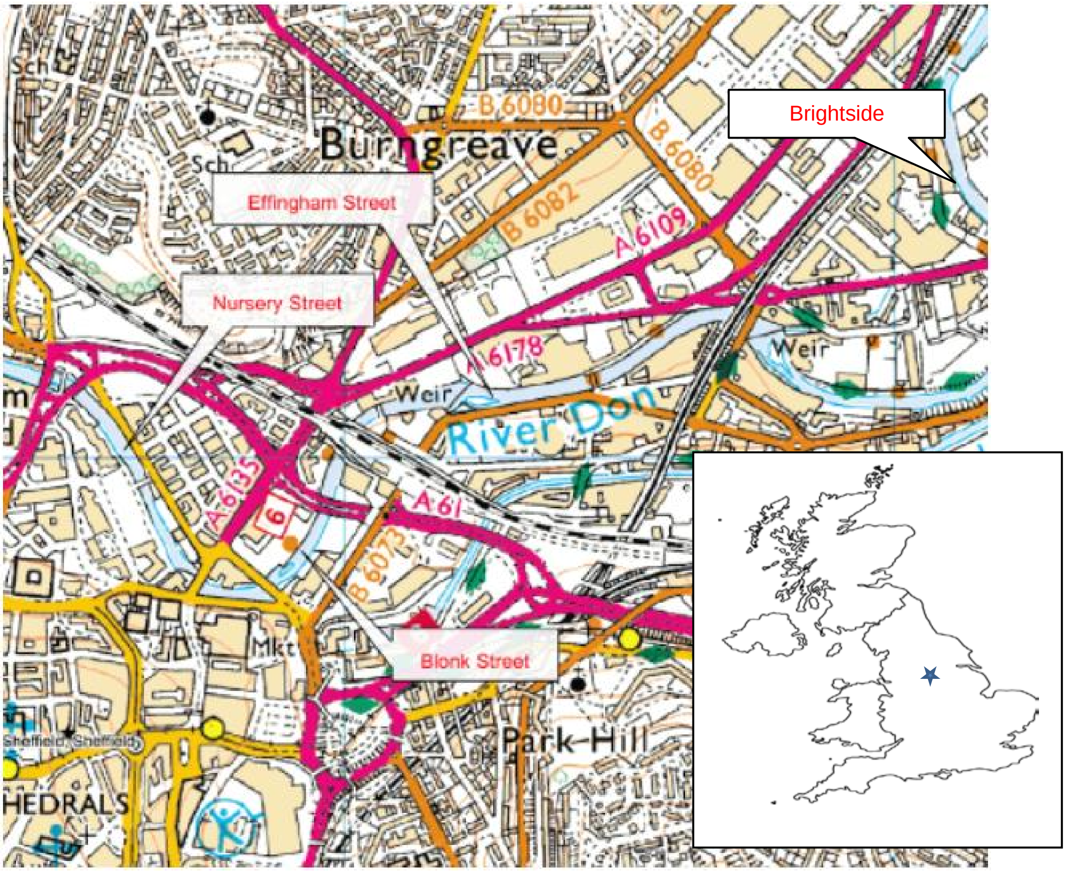


Figure 4.12. Location of flood works and river rehabilitation on the River Don, Sheffield (Source: Ordnance Survey).

In the ongoing Sheffield flood risk management scheme the Environment Agency has incorporated a variety of rehabilitation techniques to be used across the four sites (Nursery Street, Blonk Street, Effingham Street and Brightside) on the River Don, these include the replanting of appropriate native species and in-instream features such as triangular flow deflectors, midstream boulder clusters, coarse woody debris and rock riffles to ensure a diversity of flow types within the channel (Environment Agency 2010).

Triangular flow deflectors were installed at three sites (Nursery Street, Blonk Street and Effingham Street) on the River Don to provide a variety of flow types and habitats similar to those created by natural features (Figure 4.13).

Figure 4.13. A triangular 'flow deflector' at Nursery Street, River Don (source: Environment Agency 2009).

The flow deflectors installed at all sites project out as far as the outside edge of the existing berm, as this depicts the natural low flow width as the river tries to narrow through sedimentation at the channel margins. The flow deflectors are spaced approximately 50 m apart and extend 2.0 m to 2.5 m from the bank and consist of pre-established coir roll/matting and coarse woody debris. They were planted with similar species to those that were previously present to give a more natural appearance and enhance biodiversity, providing cover for fish and food for aquatic invertebrates. Further technical details of the flow deflector can be found in the Environment Agency (2009) report.

The River Don has many weirs interrupting the natural sediment transfer system and as a consequence in-channel sediment features are lacking, in particular pool-riffle sequences. The Environment Agency reinstated such features by introducing rock riffles in the hope to increase the ecological diversity of specific reaches. The main benefits are (Environment Agency 2009):

- To create or deepen pools and help aid fish ascend;

- To collect and hold fish spawning gravel above a structure;
- To increase downs stream oxygen levels;
- To encourage gravel bar formation for spawning below a structure;
- To trap fine sediments to prevent their movement.
-

However, it is important to ensure the rock riffles do not impact on fish migration by acting as barriers to upstream and downstream movement (Environment Agency 2009).

4.5.2. Nursery Street

Flood mitigation and rehabilitation works took place at Nursery Street in March 2010 by the removal of trees and shrubs along the right hand bank. The foot print of the side shoal along the right hand bank was not directly modified, but alterations were made due to the introduction of triangular flow deflectors that were built along the shoal; these extended out to the edge of the shoal (Figure 4.14).

A rock riffle was also introduced at Nursery Street downstream of the arched pedestrian bridge was to initiate a diverse flow pattern within the section. The rock riffle was strategically placed to ensure it did not sit above water level to therefore minimise effects on water levels from a flood risk management point of view. The midstream portion was cleared to create a feature intermediate between a riffle and two opposing rock deflectors (Figure 4.15) (Environment Agency 2009).

4.5.3. Blonk Street

In March 2010, trees and shrubs were removed at Blonk Street along the left hand bank and the island upstream, the side shoal along the left bank was reduced as little as possible, and flow deflectors were built along this shoal (Figure 4.16).

4.5.4. Effingham Street

In March 2010 trees and shrubs were removed at Effingham Street along both banks and from the island directly below Effingham weir (Figure 4.17). A flow deflector was introduced along the left hand bank and extended out as far as the shoal.

4.5.5. Brightside

In March 2011 tree and shrub removal took place at Brightside reducing the cover across the channel (Figure 4.18). This is the only works that have occurred at this section to date.

Figure 4.14. Photograph of Nursery Street before (a) and after (b) flood mitigation and rehabilitation works.

Figure 4.15. Rock riffle (a) at Nursery Street, River Don, downstream of the arched pedestrian bridge.

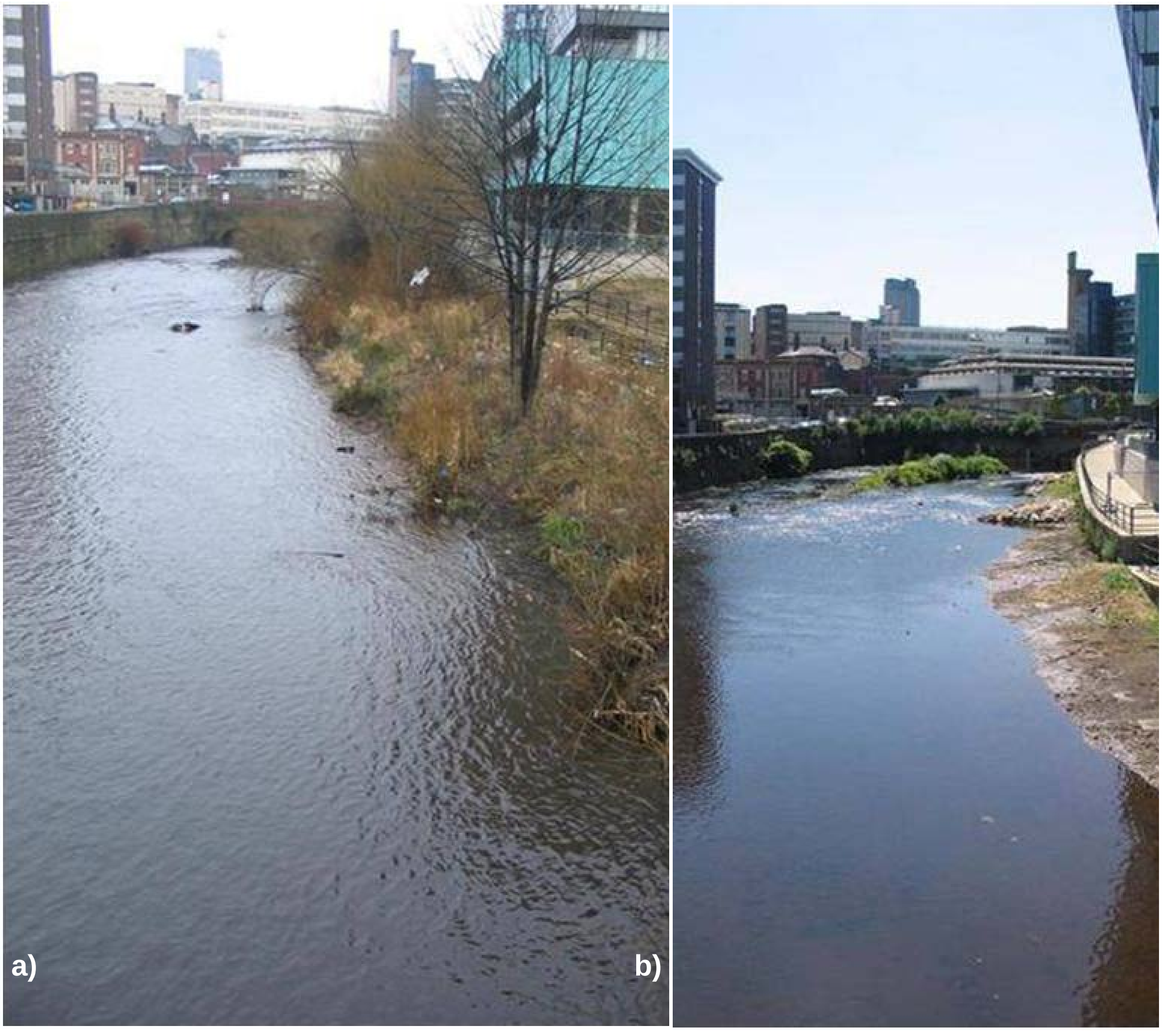


Figure 4.16. Photograph of Blonk Street before (a) and after (b) flood mitigation and rehabilitation works.

Figure 4.17. Photograph of Effingham Street before (a) and after (b) flood mitigation and rehabilitation works.

Figure 4.18. Photograph of Brightside before (a) and after (b) channel clearance works took place.

4.6. Materials and methods

4.6.1. Fisheries survey methodology

Fisheries surveys were carried out at Nursery Street, Blonk Street, Effingham Street and Brightside's (Table 4.6) in February 2010, August 2010 and August 2011 to assess the impact of the flood mitigation and habitat enhancement works on the River Don. Club Mill Weir, a control site up stream was also surveyed. Summer fisheries surveys for each year were intended to be done in the same month but due to adverse weather conditions (high river levels) they were pushed back therefore there is dissimilarity between some of the sampling dates. The surveys followed the same methodology outlined in Section 3.1.1, except quantitative surveys (triple runs) using stop-nets were not possible for the four survey reaches due to the large widths and deep areas that were encountered. Therefore, fish populations were assessed by semi-quantitative electric fishing (Section 3.1.1), surveying long reaches of the river in each section, ensuring the full variety of habitats (e.g. pools, shallower and deeper glides and riffles) were sampled. Surveying in each reach was via a combination of boat-based electric fishing in deep areas, and wading electric fishing in shallow riffle areas.

*Table 4.6. Fisheries survey site details in the River Don, Sheffield in February and August 2010 & 2011 (*sections with flood defence works completed 2010, ** sections with flood defence works completed 2011, ***control).*

Site Name	NGR	Survey date	Survey length/mean width	Survey method and gear
1. Nursery street section*	SK 356879	19/02/10 11/08/10 30/07/11	231 m/20.2 m 250 m/21.6 m 250 m/21.6 m	Generator. Semi-quantitative.
2. Blonk street section*	SK 359878	02/02/10 23/08/10 30/07/11	450 m/28.1 m 475 m/27.5 m 475 m/27.5 m	Generator. Semi-quantitative.
3. Effingham street section*	SK 364881	03/02/10 11/08/10 26/07/11	340 m/23.5 m 355 m/22.3 m 355 m/22.3 m	Generator. Semi-quantitative.
4. Brightside section**	SK 386902	04/02/10 12/08/10 28/09/11	247 m/18.0 m 255 m/17.5 m 255 m/17.5 m	Generator. Semi-quantitative.
5. Club Mill Weir***	SK 342889	19/02/10 11/08/10 30/07/11	230 m/15.0 m 217 m/15.3 m 217 m/15.3 m	Generator. Semi-quantitative.

After each electric fishing survey, habitat and environmental data were collected at each site in the format used by the Scottish Fisheries Co-ordination Centre's survey methods (SFCC 2007). Although the SFCC (2007) habitat survey provides a method for recording detailed information relating to salmonid habitat, it was also a practical method to use as a guide for collecting habitat data at 50m stretch intervals as it included assessment of flow, substrate, vegetation and cover. Each site surveyed was split into 50-m intersects were data were collected for percentage composition of substrate, percentage coverage of instream vegetation, substrate stability, percentage of different flow characteristics and percentage canopy cover as follows SFCC (2007):

Substrate: Record the percentages of each substrate type in the survey stretch wetted area:

- HO - High organic: Very fine organic matter
- SI - Silt: Fine, sticky, mostly inorganic material, individual particles invisible
- SA - Sand: Fine, inorganic particles, < 2mm diameter, individual particles visible

- GR - Gravel: Inorganic particles 2-16mm diameter
- PE - Pebble: Inorganic particles 16-64mm diameter
- CO - Cobble: Inorganic particles 64-256mm diameter
- BO - Boulder: Inorganic particles >256mm diameter
- BE - Bedrock: Continuous rock surface
- OB - Obscured: Roots, wood, sheets of iron, barrels etc. that obscure the river bed
- and cannot physically be moved for inspection
-

Instream vegetation; Record the percentage of the bed in the visible survey stretch which is covered by instream vegetation (i.e. within the *bed visible* area). Include ALL types of vegetation (including algae) in this category providing the vegetation serves as cover for fish. A thin layer of algae/mosses that just cover the surface of rocks does not count as cover in this case.

Substrate: Record the degree of substrate stability and compactness using one of the two variables Stable / Unstable - All stream beds are to some extent unstable. This variable is used to identify stretches where stream mobility is extreme and where one might expect the entire bed to move during floods.

Flow percentages: Record the percentages in the survey stretch wetted area of each flow type:

- SM - Still marginal < 10cm deep, water still or eddying
- DP - Deep pool > 30 cm deep, water flow slow, eddying
- SP - Shallow pool < 30cm deep, water flow slow, eddying
- DG - Deep glide > =30 cm deep, water flow moderate/fast
- SG - Shallow glide < 30 cm deep, water flow moderate/fast
- RU - Run water flow fast, unbroken standing waves at surface; water flow is silent
- RI - Riffle water flow fast, broken standing waves at surface; water flow is audible
- TO - Torrent white water, chaotic and turbulent flow, water flow is noisy, difficult to distinguish substrate

Canopy cover: Estimate the percentage of the survey stretch wetted area covered by overhanging branches from trees or shrubs.

4.7. Data analysis

Length distribution

Length frequency distributions were constructed for fish where sufficient numbers were caught and these were compared before and after flood and restoration works at each site to identify the number of individuals in 10mm length categories (Section 3.1.2).

Bray-Curtis similarity

Bray-Curtis similarity measure (Bray & Curtis 1957) was developed specifically for species abundance data and can also be used on relative abundance data. In this chapter I applied it to untransformed relative abundance data of each fish species (to overcome the difference in length of each sample site) and ordinate it using non-metric multidimensional scaling (MDS). PRIMER was the chosen statistical package to perform Bray Curtis and MDS to investigate similarities (or dissimilarity) in species composition, for each of the impacted sites (Blonk Street, Nursery Street, Effingham Street, & Brightside) and the control site (Club Mill Weir), before and after flood mitigation & rehabilitation. It was the chosen similarity measure because it is hardly influenced by rare species with low abundance (Gauch 1982), and therefore, analysis could be performed using all fish data present in Table 4.7.

The Bray-Curtis similarity index (C_z) represents the overall similarity between each pair of samples, taking the abundance of all species into consideration, and is calculated as:

Equation 4.1

$$C_z = \frac{2W}{(a+b)}$$

where *W* is the sum of the lesser percent abundance value of each species common to the catches at two sites (including tied values), and *a* and *b* are the sums of the percent abundances of species in the catches at site *a* and *b*, respectively. The index ranges from 0 (no species in common) to 1 (identical samples), and a similarity profile test (SIMPROF) was used to ascertain whether clusters of sites were significantly similar with one another (Clarke & Warwick 2001).

Canonical correspondence analysis

Canonical correspondence analysis (CCA) provides an efficient procedure for the direct analysis of combined species and environmental data sets. It applies multiple regressions to select a linear combination of environmental variables that explains the maximum amount of variation in the species scores on each axis of the site ordination (Waite 2000). The distribution of species and sample points jointly represent the dominant ecological relationship in so far as they be explained by the explanatory variables, therefore giving an immediate understanding of how the independent variable (environmental factors) are related to the dependent variables (species abundance) (McGarigal *et al*. 2000). Environmental factors are shown as axes indicated by arrows pointing in the direction of maximum change of the variable across the diagram. The length of the arrow marking a variable is proportional to the rate of change of the variable along the direction indicated by the arrow. Environmental variables with long arrows are most strongly correlated with the ordination axis and thus with the pattern of community variation described by the ordination.

For this reason CCA was performed on the Sheffield data, with the outcome as a tri-plot so samples, species and environmental data could be correlated. Not all habitat data was used for the CCA environmental data set, only seven significant habitat variables (boulder, deep glide, silt, shallow glide, gravel, cobble & riffle) that are key for fish habitat preference were chosen from data collected by the SFCC method. A limitation of CCA is that is it sensitive to rare (low number) of species, this was overcome by eliminating those species and only selecting the top 12 dominant species.

4.8. Results

4.8.1. Status of fish populations

Fish populations at Nursery Street, Blonk Street, Effingham Street and Brightside sections were assessed in February 2010 prior to flood defence works. Surveys in August 2010 and 2011 were aimed at assessing the status fish populations following the flood defence works and the impact of the works. Flood defence works did not take place at Brightside until spring 2011, therefore August 2010 data collected from Brightside represents pre-works fisheries status and October 2011 status of the fish population after flood mitigation works (Table 4.6).

The total number of fish species caught at Nursery Street, Blonk Street, Effingham Street and Brightside varied between sites and survey time (Table 4.7). At Nursery Street fewer fish species were caught in August 2010 and August 2011 than February 2010, while at Blonk Street, Effingham Street and Brightside the number of species caught in August 2010 and August 2011 was greater than in February 2010. At Nursery Street, Blonk Street and Effingham Street the total number of fish caught was greater in August 2010 than both February 2010 and August 2011. Total number of fish caught at Brightside, however, was greater in October 2011 than February 2010 and August

2010 (Table 4.7). Catches in the Nursery Street section in February 2010, August 2010 and August 2011 were dominated by grayling and brown trout, with greater numbers of most species found in August 2010 than February 2010 and August 2011 (Figure 4.19). This pattern was also found at Effingham Street with a dominance of grayling and good numbers of brown trout and generally greater numbers of other species in both August 2010 and 2011 surveys compared with February 2010 (Table 4.7, Figure 4.19). Grayling was the dominant species at Blonk Street at each survey, but with good numbers of both brown trout and dace (*Leuciscus leuciscus* (L.)); roach (*Rutilus rutilus* (L.)) numbers were higher August 2010 than February 2010 and August 2011 (Figure 4.19). Catches at Brightside in February 2010 were dominated by grayling and bullhead in contrast to August 2010 and October 2011 where catches were dominated by dace and minnow, with good numbers of grayling, roach and chub (*Leuciscus cephalus* (L.)). More fish of most species were found in August 2010 and October 2011 compared with February 2010 (Table 4.7, Figure 4.19). Brown trout, gudgeon, minnow and grayling were the most dominant species caught at Club Mill Weir in both August 2010 and October 2011, however, fish numbers were relatively higher in the August 2010 surveys than the October 2011 surveys (Table 4.7, Figure 4.19). Surveys in August at all four sites caught greater numbers of dace, perch (*Perca fluviestilis* L.) and roach; five barbel (*Barbus barbus* (L.)) were caught at Nursery Street in August 2010 and six barbel at Brightside in August 2010 and 12 in October 2011 (Table 4.7).

European Fish Index Calculations

The EFI + database identified an increase in ecological class boundary after flood mitigation and rehabilitation works at Brightside, increasing from Class 3 (Fish Index = 0.58) in 2010 to Class 2 (Fish Index = 0.76) in 2011. There was no change in ecological class boundaries for Club Mill Weir (Class 2; Fish Index for 2010=0.78 and 2011=0.79) and Nursery Street (Class 3; Fish Index for 2010=0.58 and 2011=0.72). A decrease in ecological class boundary was identified between 2010 (Class 2) and 2011 (Class 3) at Effingham (Fish Index for 2010=0.76 and 2011=0.74) & Blonk Street (Fish Index for 2010=0.86 and 2011=0.75).

4.8.2. Fish species length distribution

Nursery Street

Length distributions of grayling in the Nursery Street section in February 2010 revealed one year old fish in the size range 110-145 mm, and ≥2 year old individuals in the size range 170-290 mm (Figure 4.20); the oldest grayling captured was 4 years old. In August 2010, 0+ grayling captured were in the size range 70-100 mm indicating recruitment into the population, and ≥1+ individuals in the size range 160-300 mm (Figure 4.20); the oldest grayling captured was 2+ years old. In August 2011, 0+ grayling captured were in the size range 78-94 mm indicating recruitment in to the population and ≥1+ individuals in the size range 185-268 mm (Figure 4.20). One year old brown trout captured in the Nursery Street section in February 2010 were in the size range 100-140 mm, while ≥2 year old individuals were in the size range 160-380 mm (Figure 4.21); the oldest trout captured was 4 years old. Only one 0+ brown trout was captured in the Nursery Street section in August 2010, while ≥1+ brown trout captured were in the size range 120-470 mm and the oldest trout captured were aged 4+ (Figure 4.21). In August 2011 only two 0+ brown trout (71 mm and 91 mm long) were captured at Nursery Street, while brown trout ≥1+ were captured in the size range 185-268 mm (Figure 4.21). Only one roach and four perch were caught in the Nursery Street section in February 2010. In August 2010, 0+ roach caught were in the size range 20-30 mm indicating recruitment, with older individuals in the size range 130-220 mm (Figure 4.22). Perch caught were in the size range 200-280 mm with no evidence of recruitment (0+ individuals <100 mm) (Figure 4.23). In August 2011 only one 0+ roach (43 mm long) was captured, with older individuals in the size range 148-198 mm (Figure 4.22). A small number of perch (length 215-269 mm) were caught in the August 2011 surveys but there was no evidence of recruitment (Figure 4.22). Other species were caught in too few numbers in the Nursery Street section to warrant length distribution analysis (Table 4.7).

Blonk Street

Length distributions of grayling in the Blonk Street section in February 2010 revealed one year old fish in the size range 110-170 mm, and ≥2 year old individuals in the size range 200-330 mm (Figure 4.24); the oldest grayling captured was 4 years old. In August 2010, 0+ grayling captured were in the size range 80-120 mm indicating recruitment into the population, and ≥1+ individuals in the size range 180-300 mm (Figure 4.24); the oldest grayling captured was 3+ years old. In August 2011, 0+ grayling captured were in the size range 71-106 mm indicating recruitment, ≥1+ individuals were present in the size range 101-297 mm (Figure 4.24). One year old brown trout captured in the Blonk Street section in February 2010 were in the size range 90-130 mm, while ≥2 year old individuals were in the size range 160-430 mm (Figure 4.25); the oldest trout captured was 4 years old. In August 2010, 0+ brown trout captured were in the size range 70-110 mm indicating recruitment into the population, while ≥1+ individuals captured were in the size range 130-440 mm (Figure 4.25); the oldest brown trout captured were 3+ years old. In August 2011 only one 0+ brown trout (81 mm long) was captured; ≥1+ individuals were captured in the size range 125-420 mm (Figure 4.25). In February 2010, roach caught were in the size range 140-220 mm (age range 5-7 years) and dace in the size range 200-290 mm (age range 5-7 years) (Figure 4.25 & 4.26). Surveys in August 2010 revealed evidence of roach recruitment (0+ individuals in the size range 20-30 mm) and ≥1+ individuals in the size range 50-270 mm (Figure 4.26); dace caught were in the size range 70-240 mm with no evidence of recruitment (0+ individuals) at the site (Figure 4.27). In August 2011 no 0+ roach were captured indicating poor recruitment, but ≥1+ individuals were captured in the size range 171-243 mm (Figure 4.26). Dace captured were in the size range 78-242 mm with no evidence of recruitment (Figure 4.27). Other species were caught in too few numbers in the Blonk Street section to warrant length distribution analysis (Table 4.7).

*Table 4.7. Number of fish of different species captured in four sections of the River Don in February 2010 (prior to flood defence works) and August 2010 (following flood defence works). *Brightside flood defence took place spring 2011.*

	Number of fish caught													
	Nursery Street			Blonk Street			Effingham Street			Brightside			Club Mill Weir	
Species	Feb-10	Aug-10	Aug-11	Feb-10	Aug-10	Aug-11	Feb-10	Aug-10	Aug-11	Feb-10	Aug-10	Oct-11	Aug-10	Oct-11
Grayling (*Thymallus thymallus* L.)	89	180	32	81	151	108	69	222	125	20	88	74	48	33
Brownt trout (*Salmo trutta* L.)	26	51	21	35	41	57	14	97	74	11	20	36	97	85
Dace (*Leuciscus leuciscus* (L.))	5	10	4	19	46	53	0	3	0	9	195	179	0	3
Perch (*Perca fluvietilis* L.)	4	19	5	0	12	2	0	3	3	2	8	6	22	1
Roach (*Rutilus rutilus* (L.))	1	32	16	11	44	17	0	6	3	4	88	27	1	0
Pike (*Esox lucius* L.)	1	0	0	1	0	0	0	0	0	0	0	0	0	0
Gudgeon (*Gobio gobio* (L.))	4	3	22	1	9	0	0	0	10	2	20	42	59	13
Minnow (*Phoxinus phoxinus* (L.))	28	42	14	1	23	16	1	12	31	11	128	187	50	39
Stoneloach (*Barbatula barbatula* (L.))	1	2	0	0	2	0	1	8	0	0	5	5	7	6
Lampetra	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Bullhead (*Cottus gobio* L.)	5	6	3	18	65	22	0	43	20	16	12	11	27	5
3-spined stickleback (*Gasterosteus aculeatus* L.)	4	3	0	0	0	0	0	0	0	0	13	26	4	1
Chub (*Leuciscus cephalus* (L.))	0	0	0	1	3	0	0	15	0	3	51	43	1	0
Barbel (*Barbus barbus* (L.))	0	5	0	0	0	0	0	0	0	0	6	12	0	0
Common bream (*Abramis brama* (L.))	0	0	0	0	1	6	0	0	0	0	3	0	0	0
Rudd (*Scardinius erythropthalmus* (L.))	0	0	0	0	1	0	0	1	0	0	0	0	0	0
Ruff (Gymnocephalus cernuus (L.))	0	0	2	0	0	1	0	0	0	0	0	2	1	0
Roach/Bream hybrid	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Number of species	12	11	9	9	12	10	4	10	7	9	13	13	11	9
Total number of fish	169	353	119	168	398	283	85	410	266	78	637	650	317	186

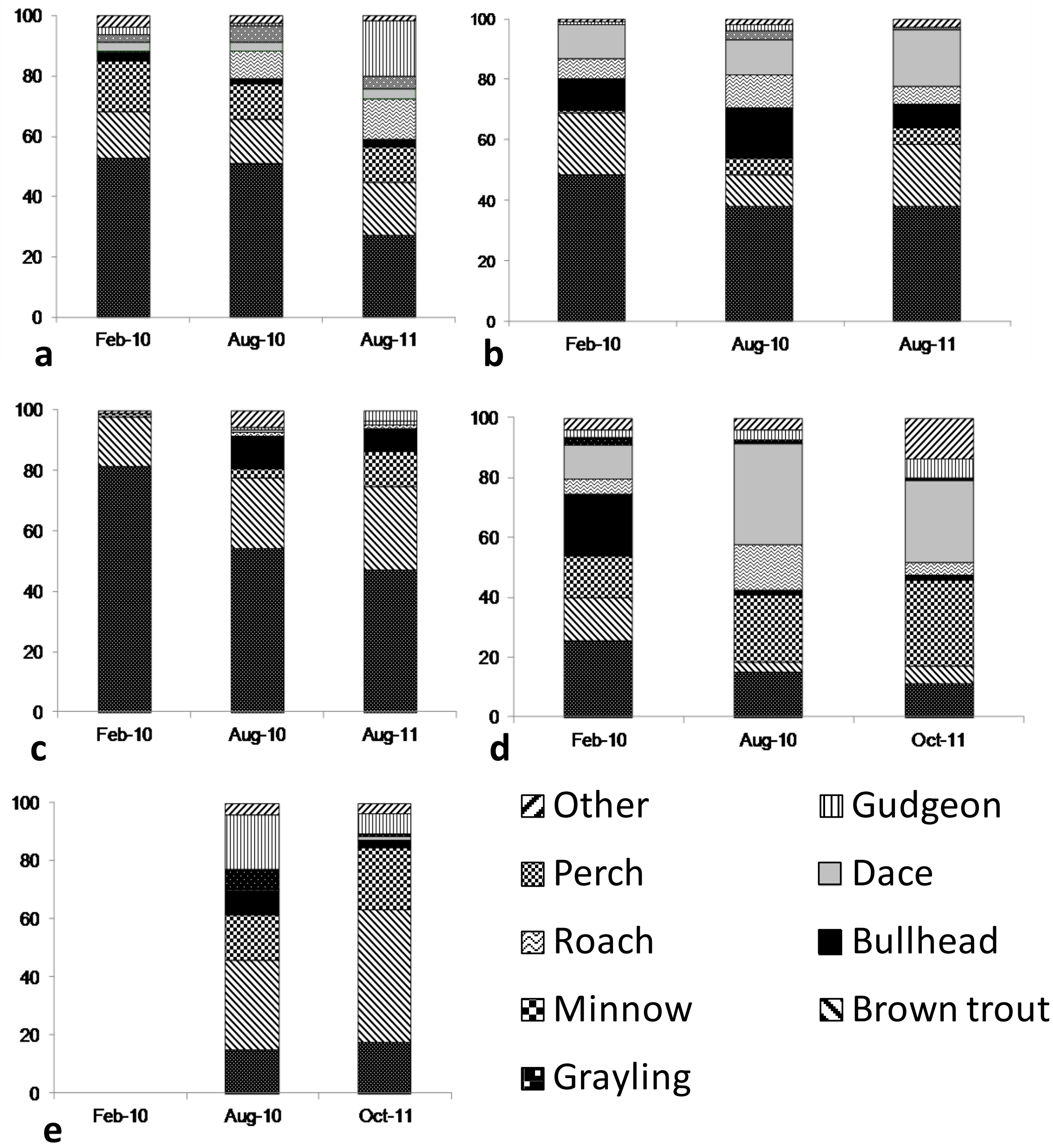


Figure 4.19. Percentage abundance of fish species captured at a) Nursery Street, b) Blonk Street, c) Effingham Street, d) Brightside and e) Club Mill Weir.

Effingham Street

Length distributions of grayling in the Effingham Street section in February 2010 revealed one-year old fish in the size range 120-160 mm, with ≥2 year old individuals in the size range 180-320 mm (Figure 4.28); the oldest grayling captured was 4 years old. In August 2010, 0+ grayling captured were in the size range 70-120 mm indicating recruitment into the population and ≥1+ individuals in the size range 170-300 mm (Figure 4.28); the oldest grayling captured was 3+ years old. August 2011 surveys captured grayling of 0+ at 80-161 mm, a sign of recruitment; ≥1+ individuals capture were 171-310 mm (Figure 4.28). One individual brown trout of 104 mm (aged 1) was captured in the Effingham Street section while ≥2 year old individuals were in the size range 190-410 mm (Figure 4.29); the oldest trout captured was 4 years old. In August 2010, two 0+ brown trout (70 and 90 mm) were captured indicating minimal recruitment into the population, while ≥1+ individuals were captured in the size range 130-400 mm (Figure 4.29); the oldest brown trout captured were 3+ years old. August 2011 surveys showed signs of recruitment with individual 0+ brown trout in the size range 67-83 mm; ≥1+ individuals were present at 141-412 mm (Figure 4.29). Other species were caught in too few numbers in the Effingham Street section to warrant length distribution analysis (Table 4.7).

Brightside

Length distributions of grayling in the Brightside section in February 2010 revealed one year old fish in the size range 120-170 mm, and ≥2 year old individuals in the size range 210-320 mm (Figure 4.30); the oldest grayling captured was 4 years old. In August, 2010, 0+ grayling captured were in the size range 60-120 mm indicating recruitment into the population, and ≥1+ individuals in the size range 170-320 mm (Figure 4.30); the oldest grayling captured was 2+ years old. In October 2011, 0+ grayling captured were in the size range 80-130 mm and ≥1+ individuals in the size range 164-298 mm (Figure 4.30). In February 2010, one brown trout of 107 mm (aged 1) was captured in the Brightside section while ≥2 year old individuals were in the size range 240-380 mm (Figure 4.31); the oldest trout captured was 4 years old. In August 2010, 0+ brown trout were absent from the Brightside section while ≥1+ individuals captured were in the size range 170-340 mm (Figure 4.31); the oldest brown trout captured was 3+ years old. In October 2011 0+ brown trout captured were in the size range 78-106 mm and ≥1+ individuals in the size range 128-410 mm. Length distributions of roach in the Brightside section in February 2010 revealed individuals in the size range 140-220 mm while in August 2010 roach were caught in the size range 20-320 mm, with an age distribution of 1+ to 14+ and in October 2011 in the size range 54 – 260 mm; individuals <30 mm were aged as 0+ indicating recruitment in the section (Figure 4.31). One year old dace in the size range 50-60 mm were captured in February 2010 with small numbers in the size range 170-220 mm. In August 2010 dace captured were in the size range 30-260 mm, with individuals <40 mm aged as 0+ indicating recruitment in the section (Figure 4.32), and in October 2011 dace were captured between 11-234 mm. Three chub in the size range 330-430 mm, with an age distribution of 9+ to 13+ were captured in the Brightside section in February 2010 (Figure 4.34) while in August 2010 chub captured were in the size range 140-470 mm and in October 2011 in the size range 172-438. There was no evidence of chub recruitment in either year (0+ individuals <40 mm). Other species were caught in too few numbers in the Brightside section to warrant length distribution analysis (Table 4.7).

Club Mill Weir

Length distributions of grayling at Club Mill Weir in August 2010 revealed 0+fish in the size range 68-93 mm, and ≥1 year old individuals in the size range 179-309 mm (Figure 4.35). In October 2011, 0+ grayling captured were in the size range 79-130 mm indicating recruitment into the population and ≥1+ individuals in the size range 164-298 mm (Figure 4.36). In August 2010, there was no 0+ brown trout present at Club Mill Weir indicating poor recruitment, while ≥1 year old individuals were in the size range 137-252 mm (Figure 4.36). In October 2011, two 0+ brown trout (78 and 87 mm) were captured suggesting poor recruitment, while ≥1+ individuals captured were in the size range 105-410 mm (Figure 4.36). Other species were caught in too few numbers in the Brightside section to warrant length distribution analysis (Table 4.7).

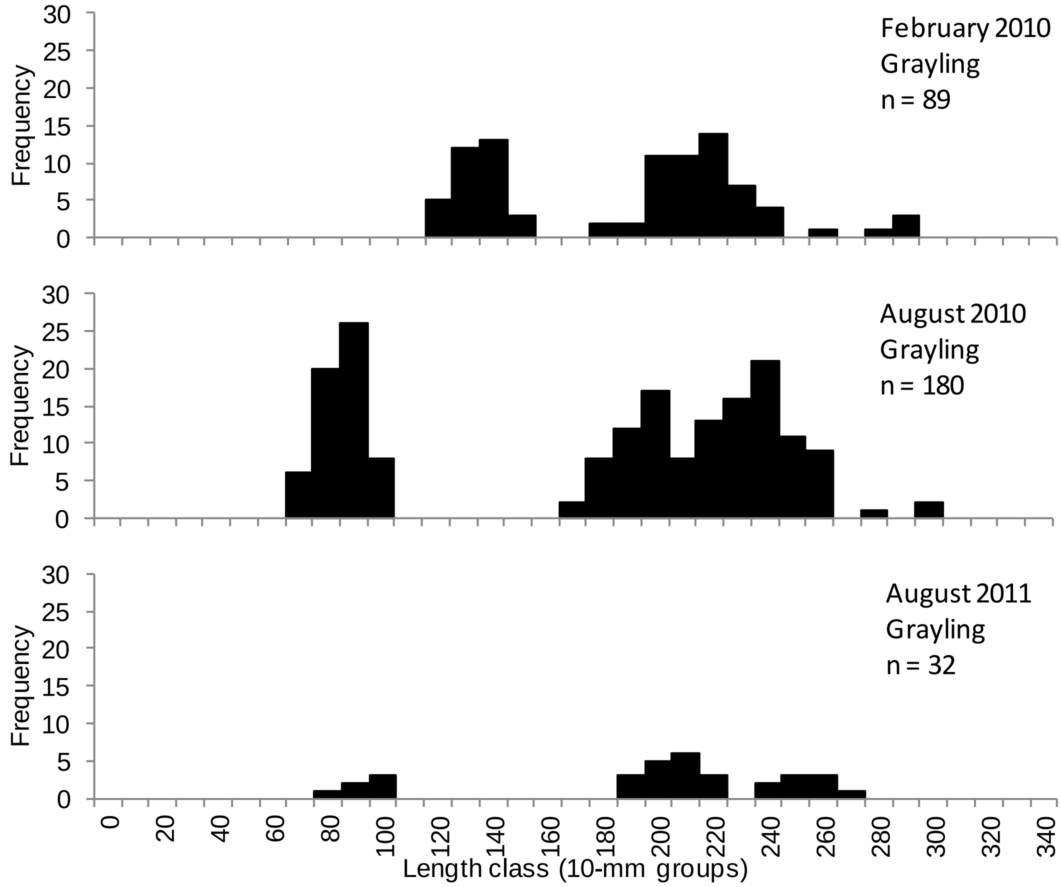


Figure 4.20. Length distributions of grayling at Nursery Street.

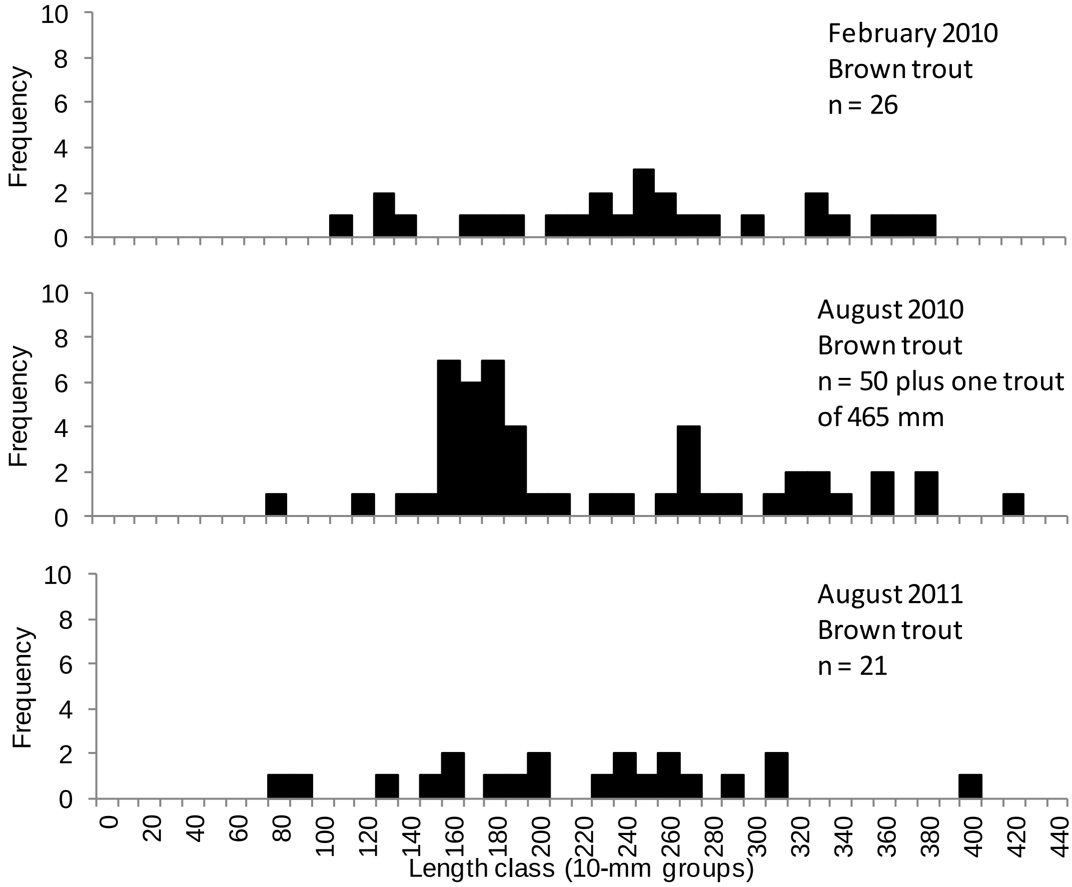


Figure 4.21. Length distributions of brown trout at Nursery Street.

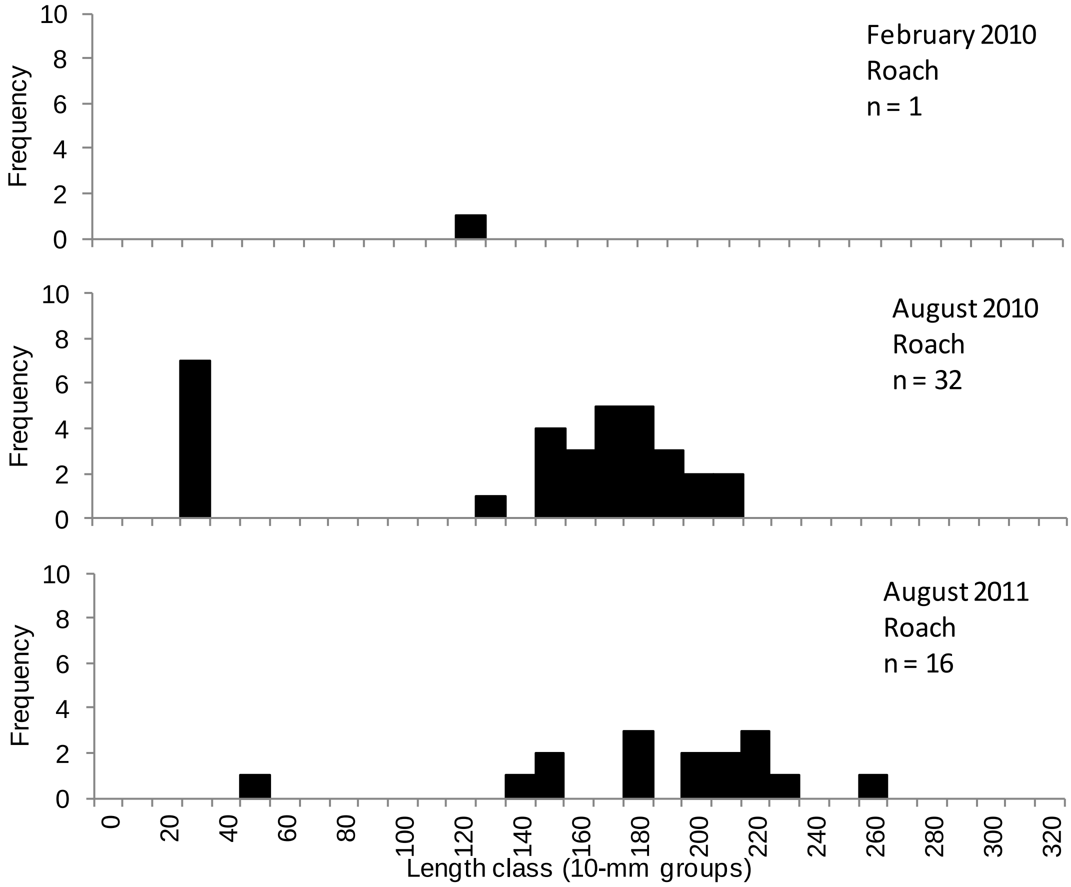


Figure 4.22. Length distributions of roach at Nursery Street.

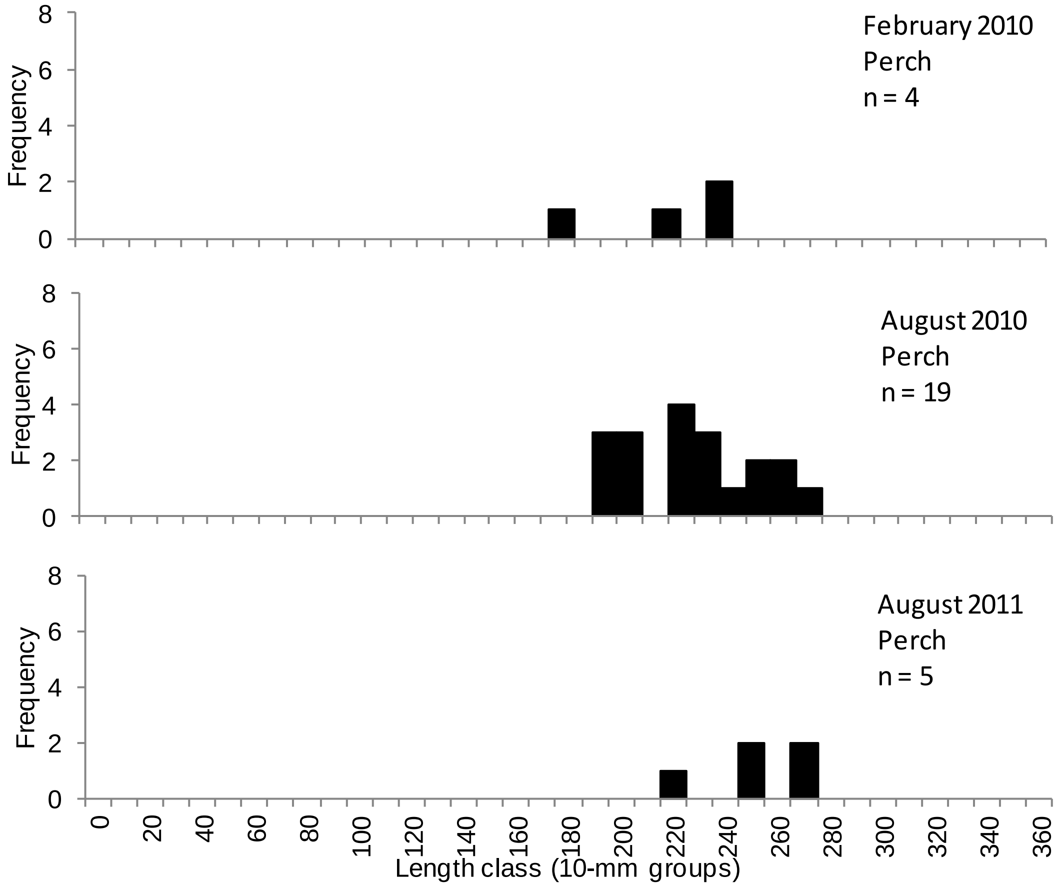


Figure 4.23. Length distributions of perch at Nursery Street.

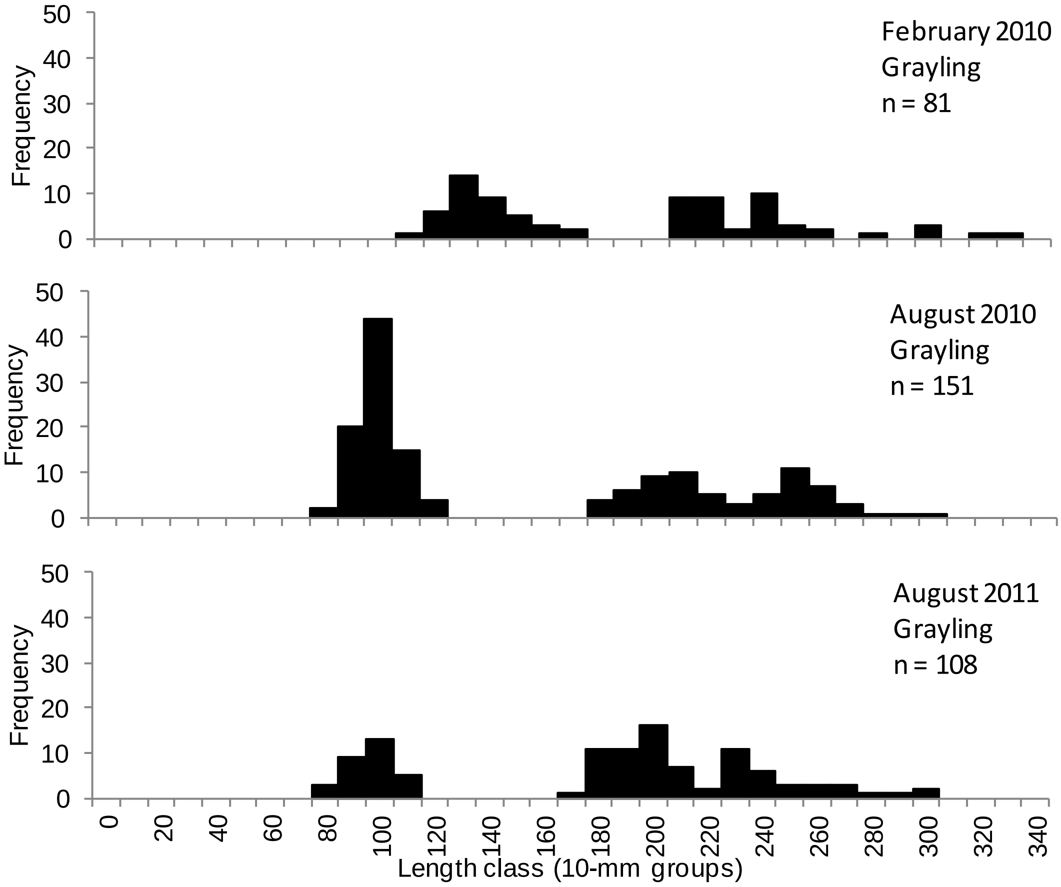


Figure 4.24. Length distributions of grayling at Blonk Street.

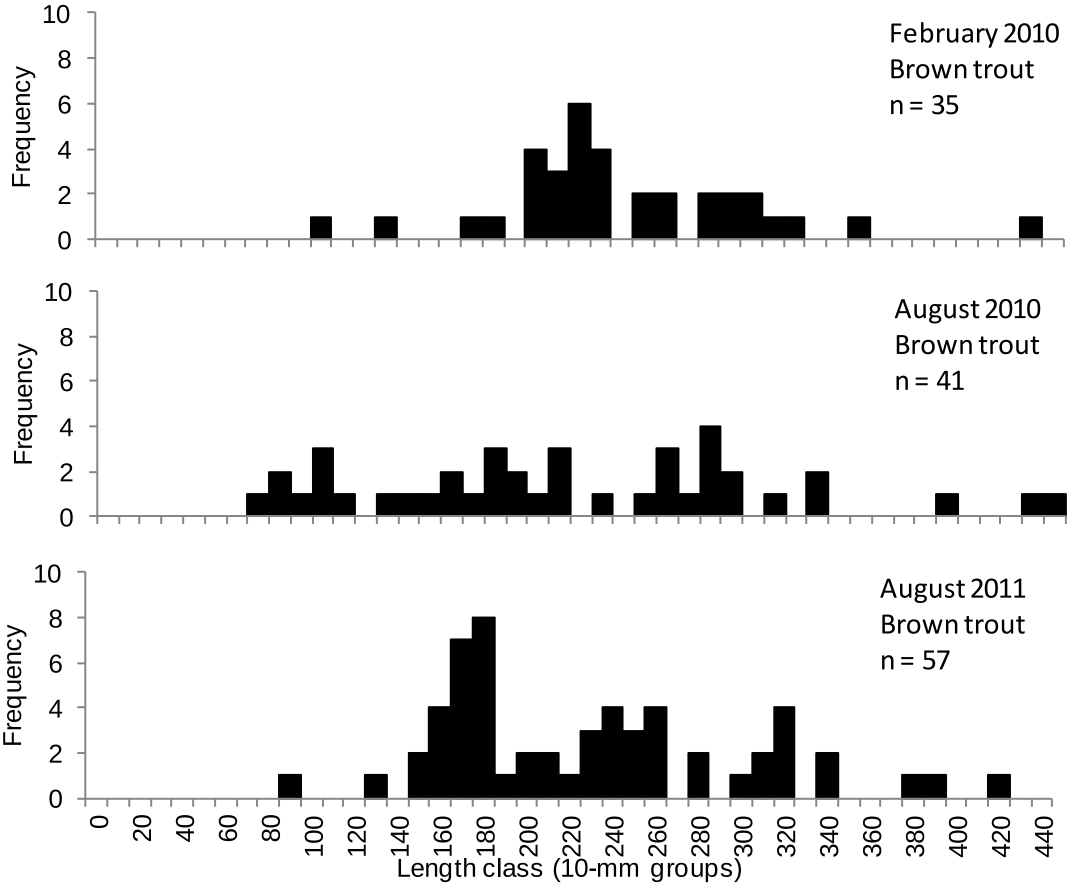


Figure 4.25. Length distributions of brown trout at Blonk Street.

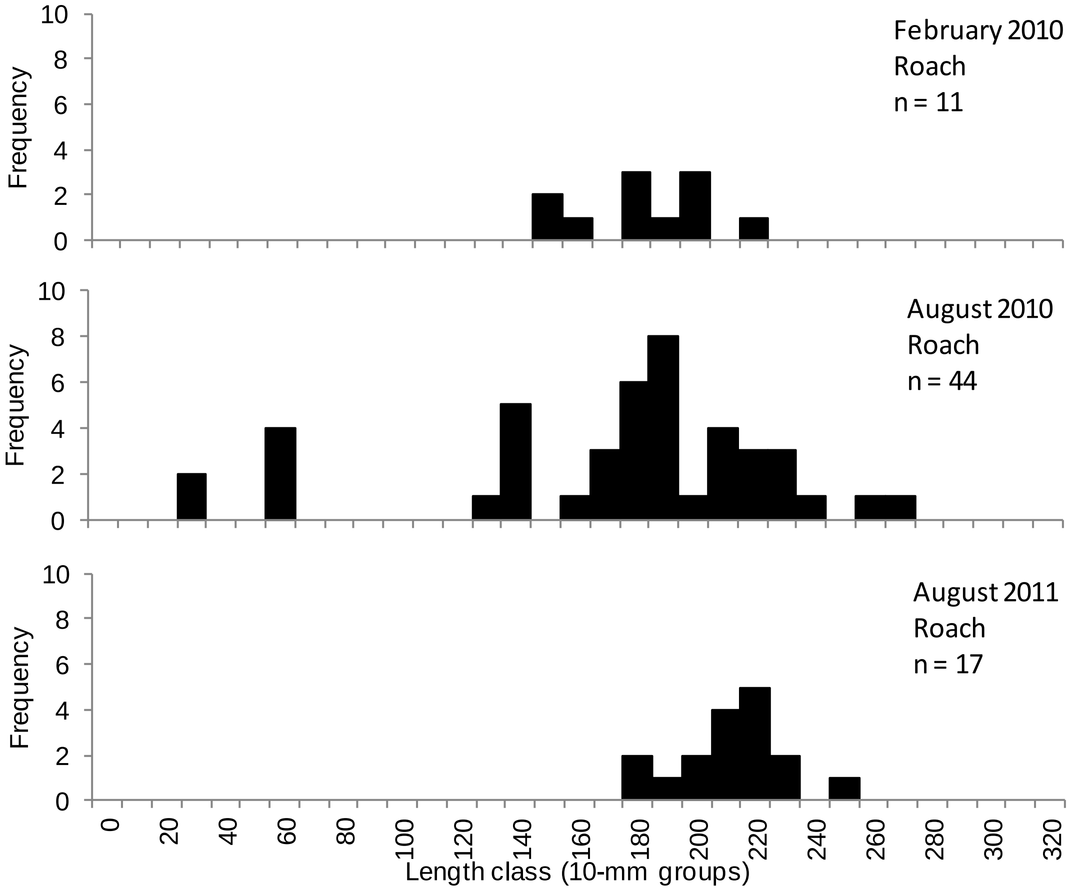


Figure 4.26. Length distributions of roach at Blonk Street.

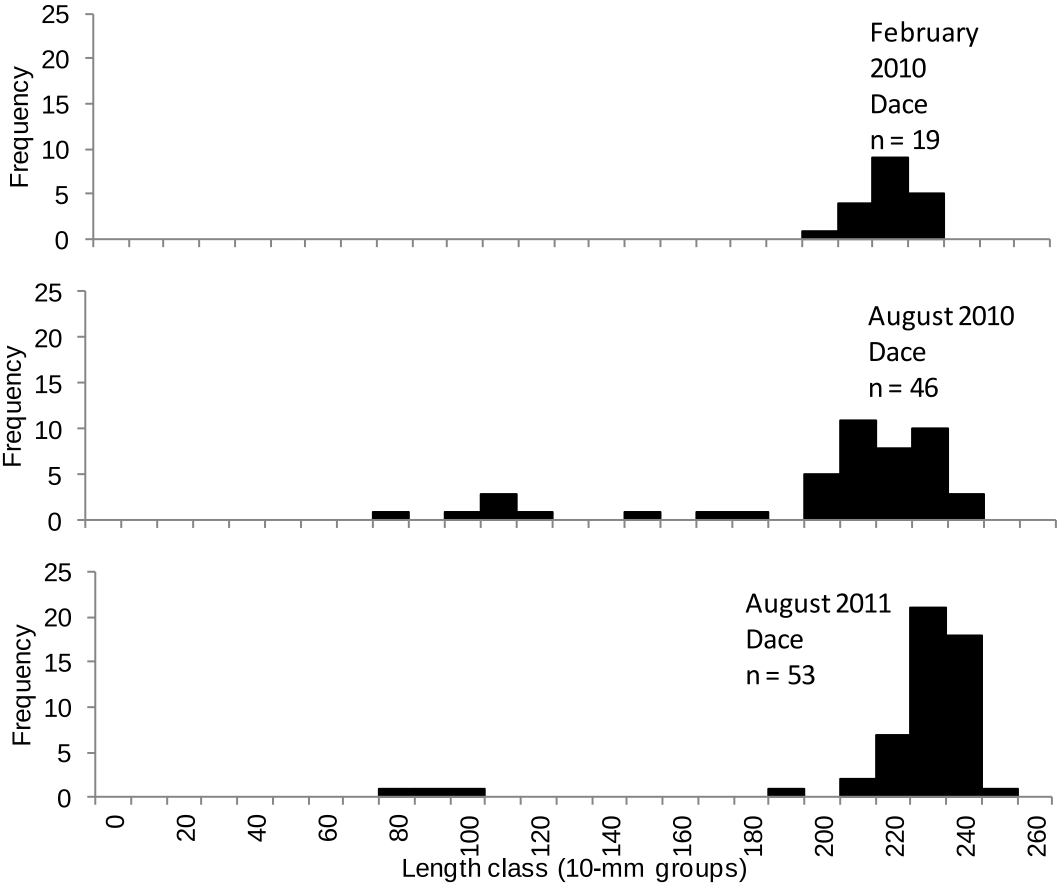


Figure 4.27. Length distributions of dace at Blonk Street.

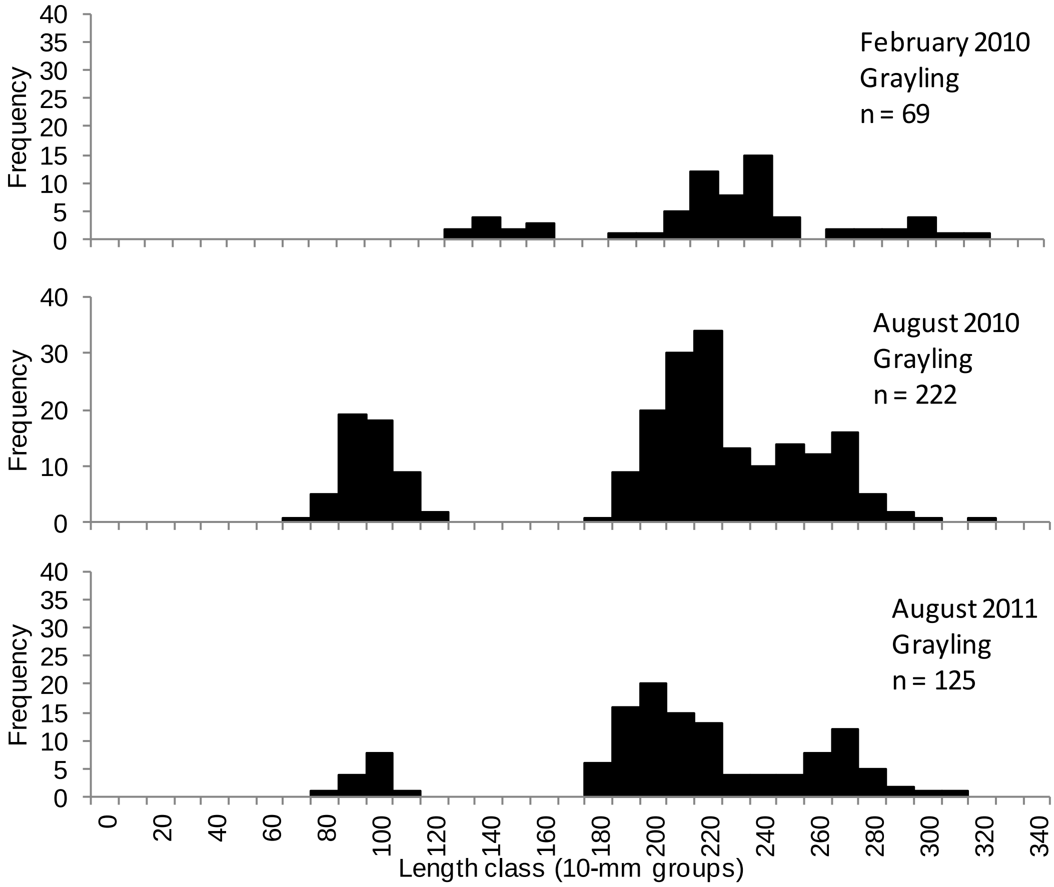


Figure 4.28. Length distributions of grayling at Effingham Street.

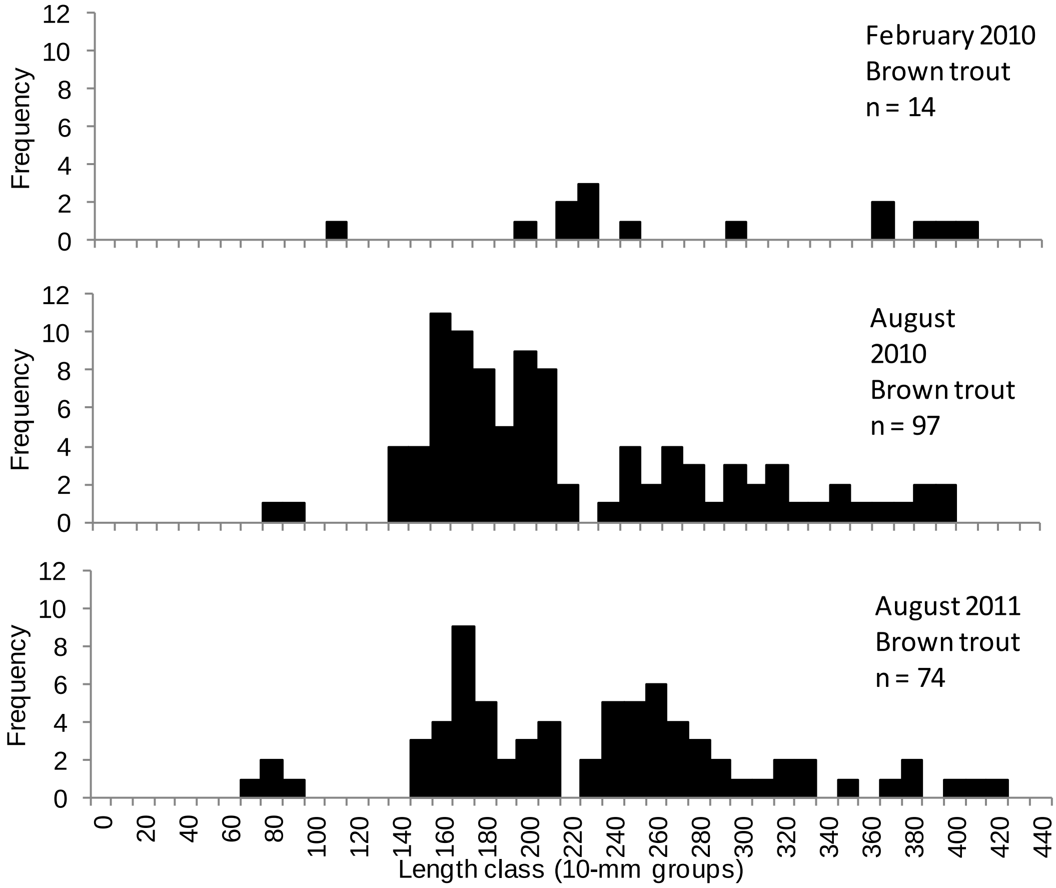


Figure 4.29. Length distributions of brown trout at Effingham Street.

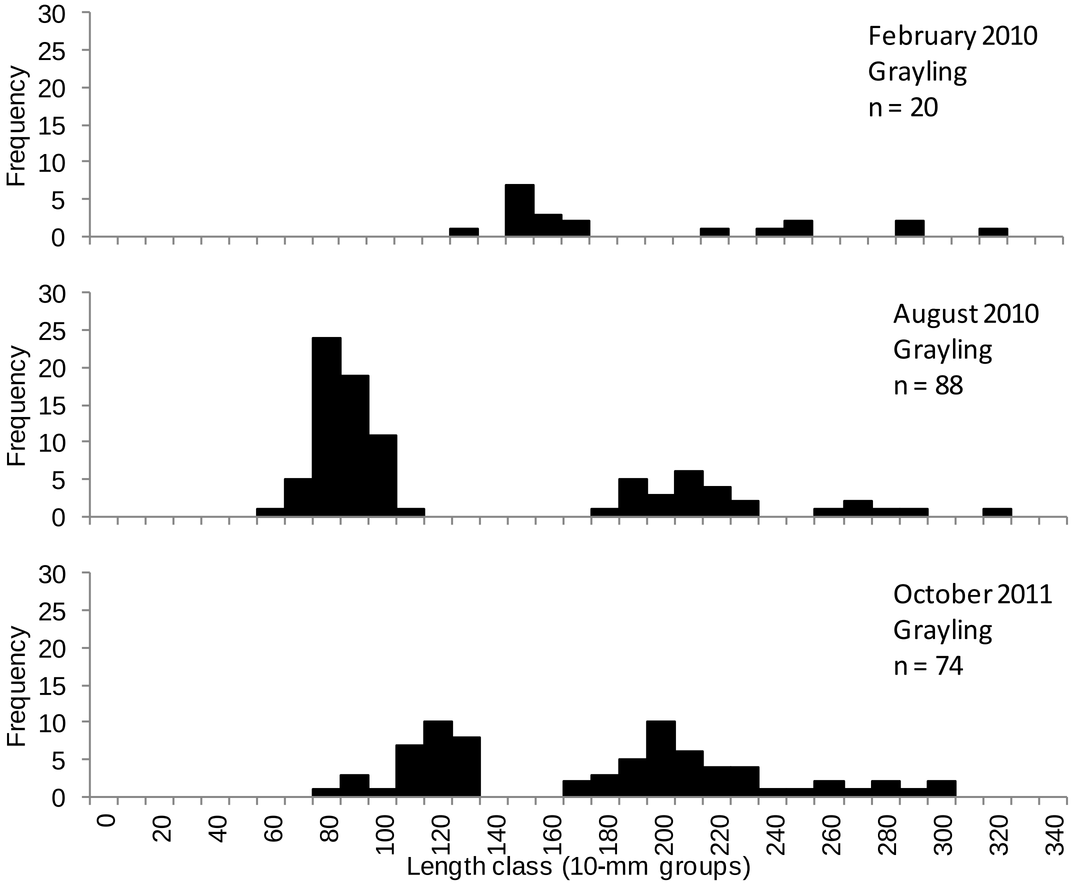


Figure 4.30. Length distributions of grayling at Brightside.

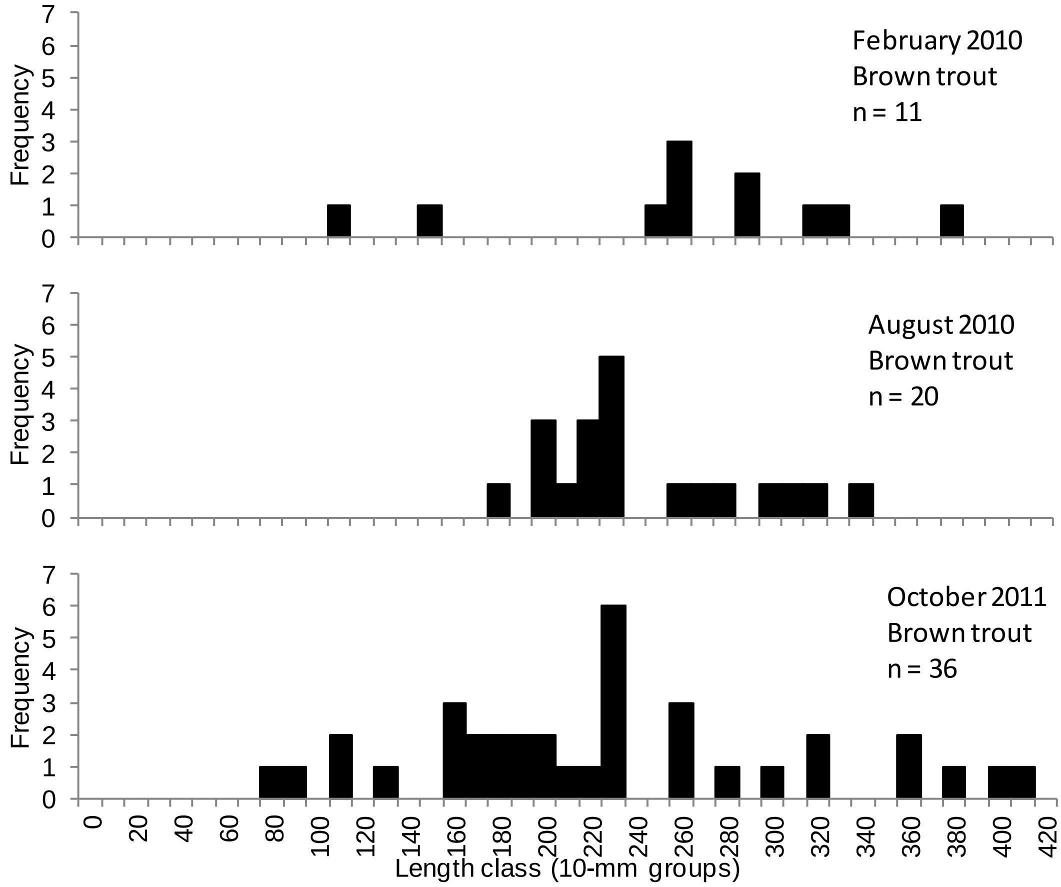


Figure 4.31. Length distributions of brown trout at Brightside.

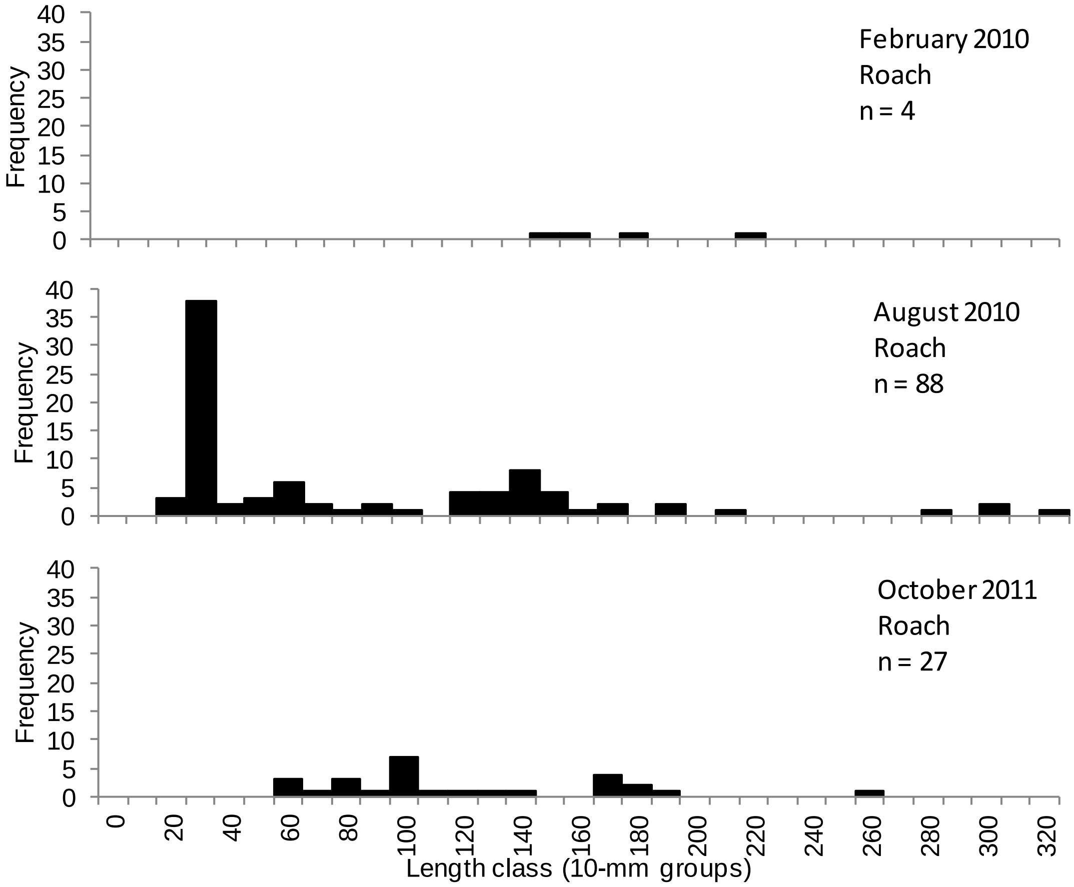


Figure 4.32. Length distributions of roach at Brightside.

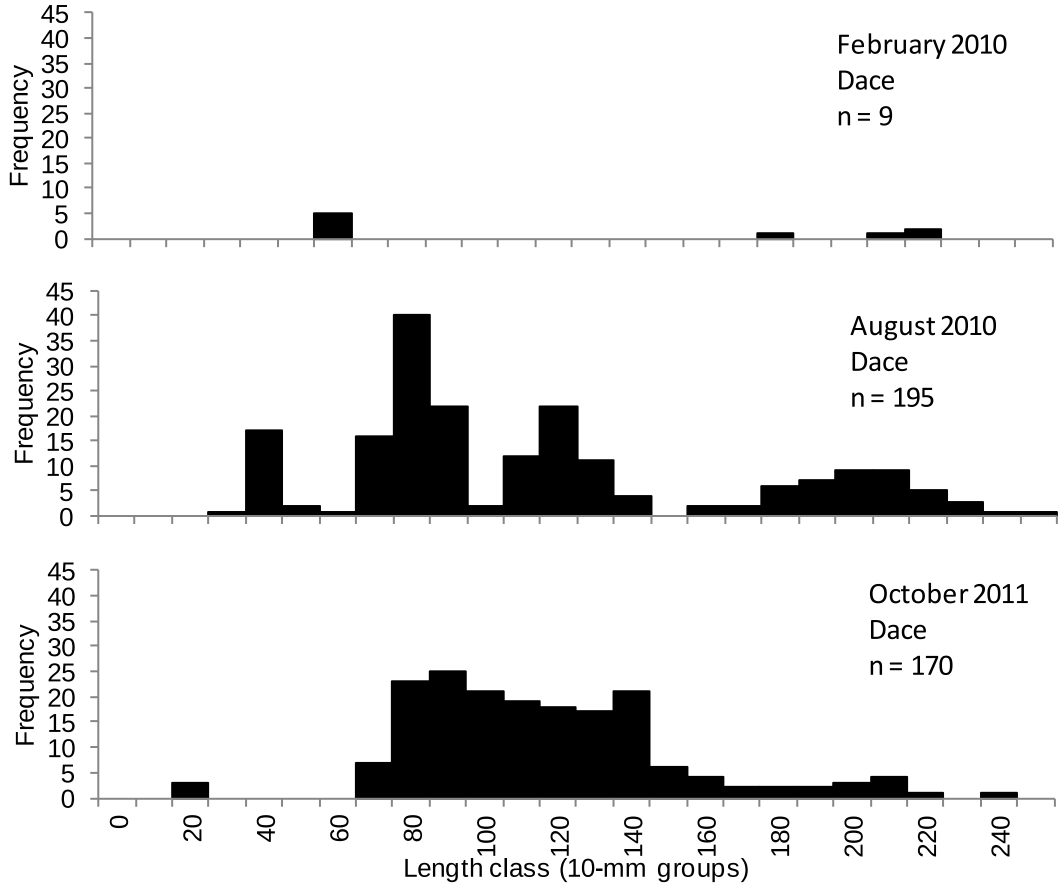


Figure 4.33. Length distributions of dace at Brightside.

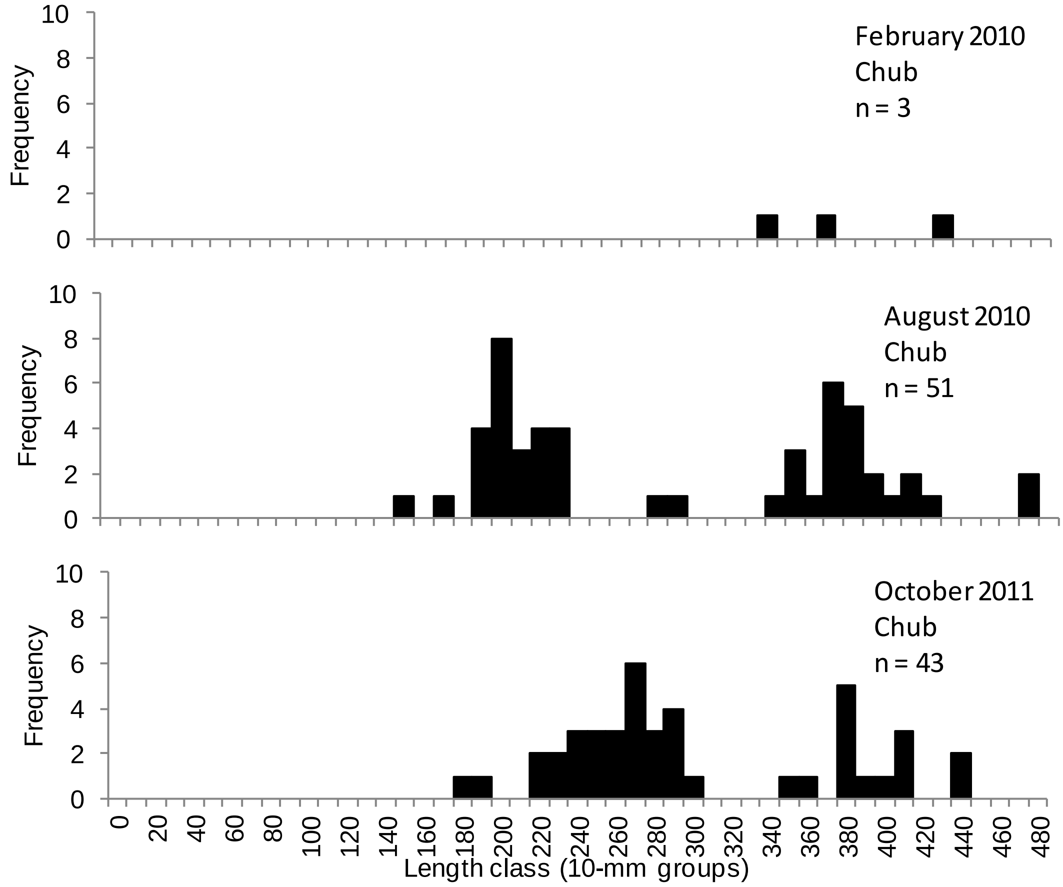


Figure 4.34. Length distributions of chub at Brightside.

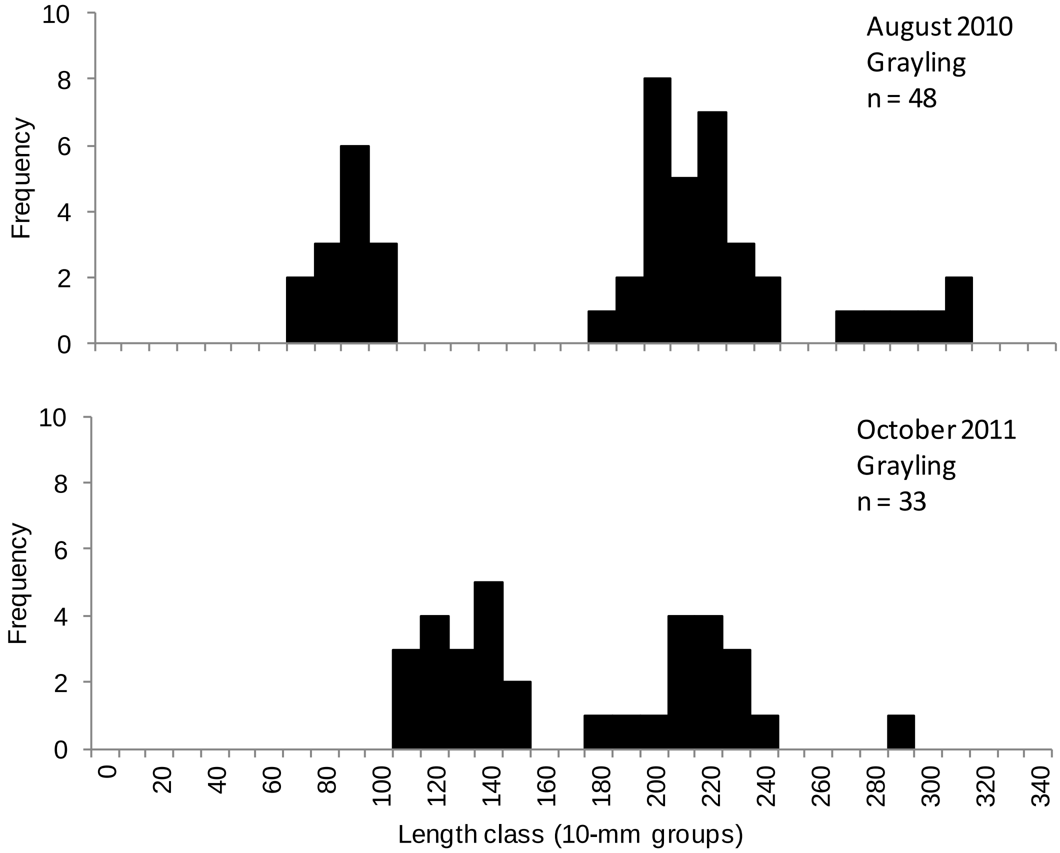


Figure 4.35. Length distributions of grayling at Club Mill Weir.

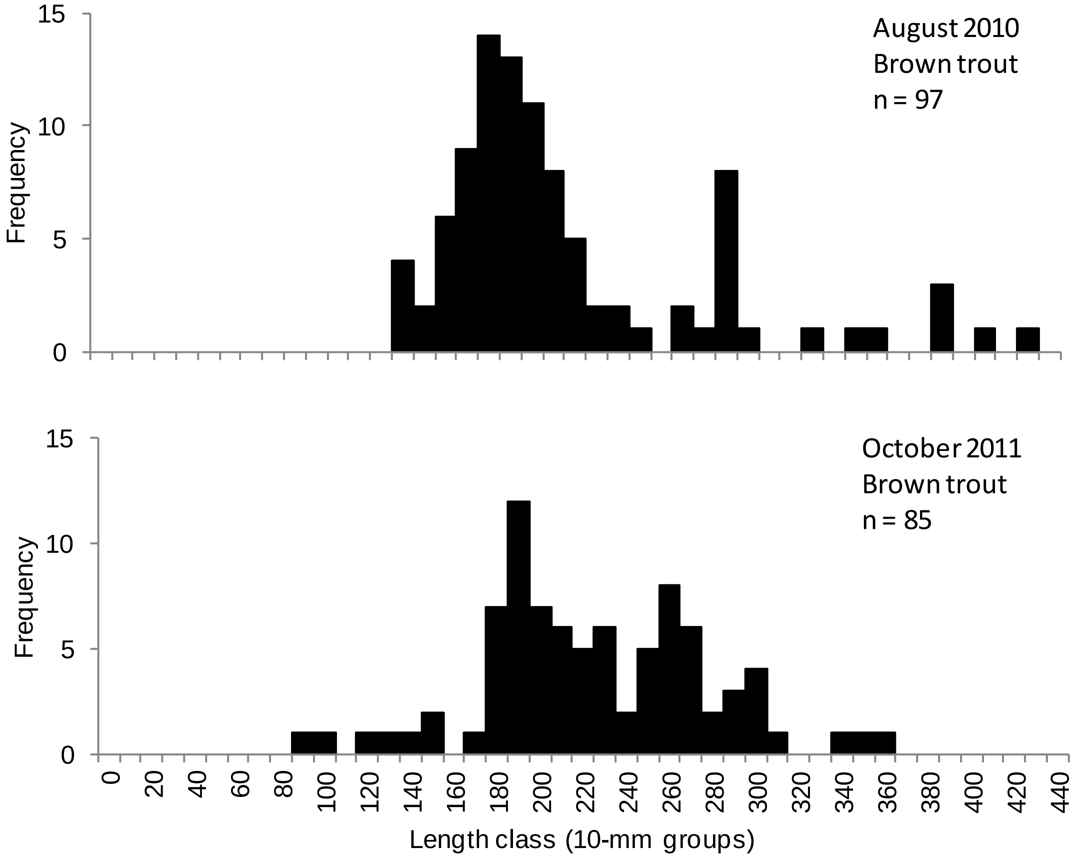


Figure 4.36. Length distributions of brown trout at Club Mill Weir.

4.8.3. MDS and cluster analysis

A Bray Curtis similarity matrix was produced on untransformed fish relative abundance data (to overcome the difference in length of each sample site) to determine similarity patterns between samples. Further analysis using MDS (Section 3.2.2) and cluster analysis identified a high similarity in fish relative abundance between years (2010 compared with 2011) at all sites: Blonk Street - 80.2%, Brightside - 82.4%; Club Mill - 74.5%; Effingham - 83.0% (Figure 4.18), but the MDS discriminated these two samples (Figure 4.37). However, the similarity between Nursery Street in 2010 and 2011 was relatively (56.0%), this relationship was also supported by the MDS where Nursery Street 2010 and 2011 have the largest distance between them (Figure 4.37). It is apparent from the catch data that it is the high number of Grayling captured at Nursery Street in August 2010 that separates it from August 2011 and furthermore links, by similarity, to Effingham Street August 2010 catch data. Nursery Street 2011 has a low number of grayling and is linked, by similarity, to Club Mill Weir 2010 and 2011 where grayling are also present in lower numbers, therefore establishing that it is grayling that cause a reduction in similarity between Nursery Street 2010 and Nursery Street 2011 (Table 4.7). A similar pattern can also be observed with gudgeon (Table 4.7). A High number of minnow present at both Brightside 2010 and 2011 catches reduce the sites similarity to the other 4 sites (Figure 4.27).

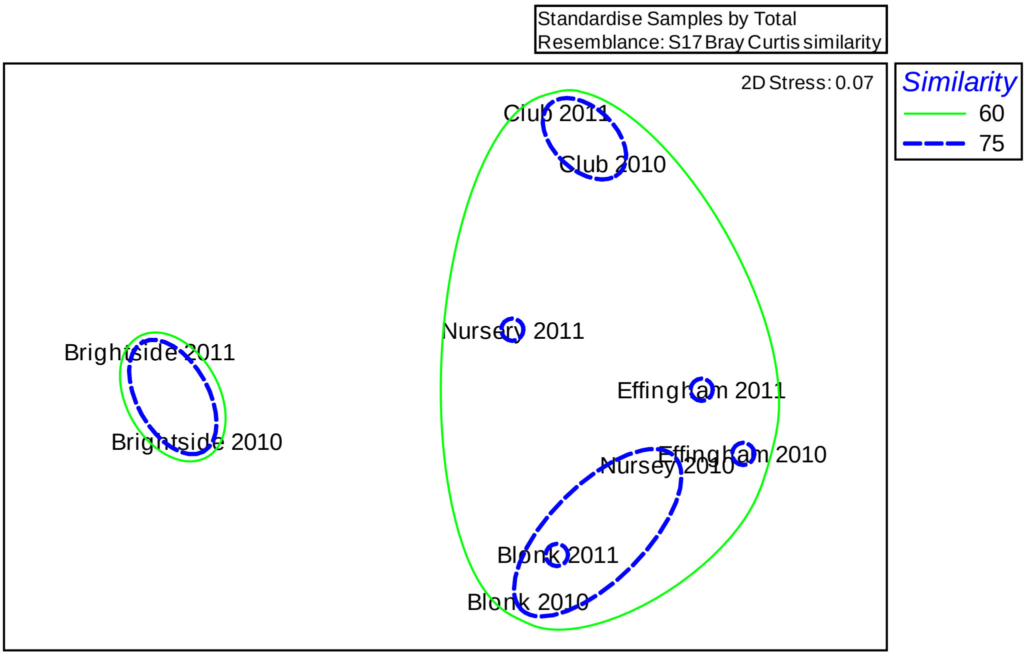


Figure 4.37. MDS showing the similarity of fish catches between sites.

Canonical correspondence analysis (CCA) can additionally elucidate the low similarity between Nursery Street 2010 and 2011 surveys (Figure 4.38). Nursery Street 2010 was dominated by shallow glide and a silt substrate of which grayling were positively correlated; conversely Nursery Street 2011 was dominated by a deep glide and boulder substrate of which grayling were negatively correlated against, nevertheless, brown trout, perch, gudgeon and ruff (*Gymnocephalus cemuus* (L.)) were all positively correlated (Figure 4.38). The CCA can further explain the small divergence that is shown in the MDS (Figure 4.37) between Effingham 2010 and 2011 (Figure 4.39), both grayling and bullhead make up the dominant species at both sites so subsequently it is the difference in dominant substrate at each site that results in the un-similarity between sites (Figure 4.38). Additionally, for Brightside, Blonk and Club Mill Weir the CCA (Figure 4.39) supports the cluster analysis (Figure 4.38) and MDS (Figure 4.37) previously calculated resulting in positive correlation between the two sampling years.

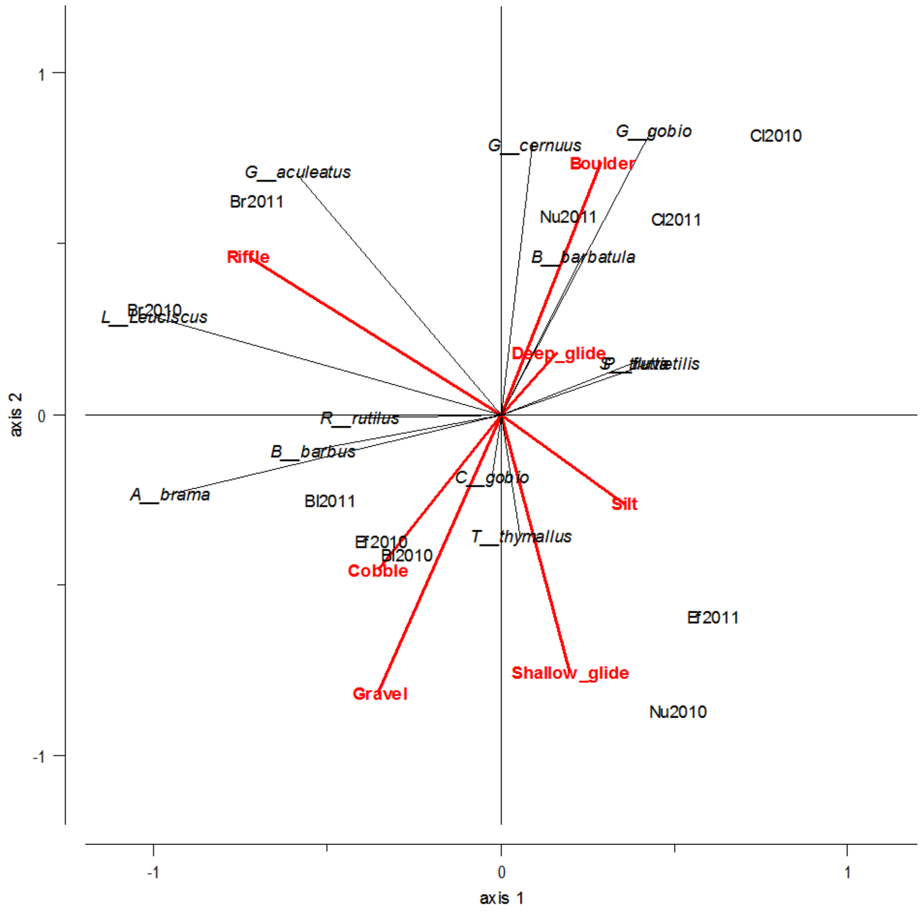


Figure 4.38. Triplot of the canonical correspondence analysis of 12 dominant fish species, 7 significant environmental variables across 5 sites on the River Don, Sheffield.

4.9. Discussion

Flood mitigation actions regularly result in the removal of shoal, instream and bank side vegetation; this will therefore reduce habitat diversity in the channel for all fish life stages. Although the E.A have a responsibility to protect people and infrastructure from flooding (EU Floods Directive and UK Flood and Water Management Act) they also have the responsibility to protect and conserve river ecosystems (EU HD and WFD) and therefore, river rehabilitation needs to be incorporated within their flood risk management plans. In an attempt to reduce flood risk to the city of Sheffield, river rehabilitation was also incorporated in to flood risk management plans, however the functionality of the rehabilitation is questionable. The E.A made a conscious effort not to remove any shoal material that was in stream (under water level) to reduce the disturbance within the channel however, they did remove the majority of bank side vegetation, vegetation on islands and over hanging trees at each site with the exception of the control site, Club Mill Weir. Aquatic and riparian vegetation are vital for healthy and sustainable watercourse ecosystem, even more so on an urban river system where there is little or no variation in physical structure (Cowx & Welcomme 1998). The main features of vegetation for fish are to provide shelter from predators through leaves and trunks, enable variations in water temperature, good food source by encouraging invertebrates and suitable spawning habitat for a selection of fish (Cowx & Welcomme 1998; FAO 2008). The potential impacts of channel clearance work will remove key habitat features causing a direct impact on fish populations. Boulder clusters and flow deflectors are known to enhance habitat conditions in straightened rivers by introducing diversity and flow deflectors additionally reinstating meanders (Downs & Thorne 1998). Boulder clusters and flow deflectors were introduced as a rehabilitation measure at Nursery Street, Blonk Street and Effingham to try and introduce diversity back into the river, however, due to caution from the E.A flood risk management team only small rehabilitation actions were taken. Pretty *et al*. (2003) found little evidence that the addition of flow deflectors improve the conservation value of the fish assemblage; nevertheless, there are plans to further extend these flow deflectors in summer 2012. Future monitoring of these sections will enable the functionality of this rehabilitation action to be assessed.

There are weir pools, shallow riffle and vegetated island areas in each of the study reaches, these areas are critical to the survival of grayling, brown trout, dace, roach, chub and barbel and any damage to these important spawning and nursery areas could have long lasting effects on the populations (Maitland 2004). The fast flowing areas downstream of the weirs and at riffles are important refuges for all fish species during summer periods (personal observation) when lower flows and elevated temperatures may force these species into the well oxygenated weir pools/riffles (Cowx *et al*. 2004). Flood mitigation works have removed all vegetation from islands to enable a large volume of water to pass through the channel, not only has this removed the majority of essential habitat features for fish but it may also change the stability of the island which could further result in a change in the morphology of these islands over time, to assess this change and impact on key habitat features, monitoring over a long time period is needed.

The species composition in the study sections of the River Don reflects its status as a recovering river following heavy degradation, with a dominance of pollution-intolerant species, such as grayling and brown trout, the latter a UK Biodiversity Action Plan priority species. Grayling and brown trout are considered important natural resources and their enhancement is critical to the EA national strategy to conserve and improve wild stocks of trout, sea trout and grayling whilst enhancing the environment for all types of fisheries for these species in England and Wales (Environment Agency 2003). Grayling and brown trout favour clean, pollution-free water conditions for spawning and growth, and their dominance at most sites indicates the improvement in water quality in the River Don.

The February 2010 fisheries surveys provided an indication of the baseline status of fish populations prior to flood defence works at Nursery Street, Blonk Street, Effingham Street and Brightside. The outputs of the fisheries surveys in August 2010 and August 2011 includes the assessment of fish populations following flood defence works and comparison with pre-works data, with the exception of Brightside where flood defence works took place in spring 2010. Surveys carried out at Nursery Street, Blonk Street, Effingham Street and Brightside in February 2010 revealed that other species were caught in generally low numbers, suggesting reaches were more suited to grayling and trout than coarse fish species (Harvey *et al.* 2010). However, Harvey *et al.* (2010) considered that this conclusion be treated with caution as the time of sampling may have resulted in an under-representation of other species in catches. Sampling coarse fish populations is usually carried out on large, deeper rivers between June (following spawning of most coarse fish species) and September, when fish are more active and often higher in the water column. Surveys post-September can result in poor catches of fish due to fish moving to deeper, inaccessible areas over-winter; fish also become less active in the colder months and hence are more difficult to capture with electric fishing. Subsequent surveys at Nursery Street, Blonk Street, Effingham Street in August 2010 and 2011; and at Brightside August 2010 and October 2010 revealed not only greater numbers of grayling and brown trout but greater numbers of coarse fish species such as roach, dace, perch, chub and barbel. The greater number of fish and similar or higher species diversity in the Nursery Street, Blonk Street and Effingham sections in August 2010 and 2011 surveys, compared to February 2010 suggest the flood defence works have had no obvious impact on the fish populations in the study reaches. When comparing August 2010 surveys with August/October 2011 surveys there is a general reduction in fish species numbers across the sites, there is the possibility that this could be a lag time of fish response to the flood mitigation works, however, as it is a general trend across all sites, including the control site Club Mill Weir, the probable verdict is fish response to natural variability. Nevertheless, the EFI + database identified an increase in ecological class boundary after flood mitigation and rehabilitation works at Brightside, no change for Club Mill Weir and Nursery Street, and a decrease in ecological class boundary at Effingham and Blonk Street. The conclusion should be treated with caution as this is based on two year’s survey data for assessing the potential impact of the flood defence works, and fish populations are prone to natural variation that can only be overcome by further monitoring. Therefore, it is recommended that annual monitoring of these fisheries at the same sample sites is carried out for a number of years after the flood defence works to detect and enable mitigation of any potential or unforeseeable impact.

4.9.1. Importance of the availability for fish life stages within an urban river system

The biological integrity of fish populations and communities are directly related to the variety and extent of natural habitats and related processes within a river basin (Cowx *et al*. 2004). Fish community structure and diversity, and resilience to disturbance, may therefore be related to habitat complexity (Gorman & Karr 1978; Schiemer *et al.* 1991; Pearsons *et al*. 1992) that is influenced by the geomorphologic and hydrological processes of a river. As a result, the loss of structural complexity and degradation of spawning, nursery and refuge habitats through river modification can have implications for the fish fauna (Pretty *et al.* 2003). Habitat suitability criteria is based on the assumption that a species preferred habitat is influenced by the most favourable conditions therefore, as the favourable condition decreases so will the species preference (Petts 2008).

Flood mitigation works can have an effect on fish species composition by influencing individual life stages as a result of channel modification altering their feeding breeding and refuge habitats. Fish exhibit diverse forms of reproduction such as different spawning behaviour and using diverse spawning habitats (Noble *et al*. 2007) and young stages represent critical periods in the life cycle of all fish species (Mann 1996; Copp 1997a; Cowx & Welcomme 1998). Salmonid embryos require a variety of habitat features, such as dissolved oxygen, water temperature, velocity and substrate size (Chapman 1988; Quinn 2005) to increase their survival rate. Channel clearance works, especially the removal of shoal material can have devastating effects on embryos if the work is carried out at the wrong time of year. Removing shoal habitat will remove suitable salmonid spawning ground and during critical months disturb any embryos already in place. Minnow and barbel also use gravel as suitable spawning habitat and therefore its removal will also impact on their recruitment. Channel clearance works can increase mortality rates by the release of silt in to the water column, having devastating affects downstream by covering gravels and possibly smothering embryos and salmonid larvae. The availability of nursery areas surrounding the spawning gravel is critical for the survival of fry (Armstrong *et al.* 2003). Juvenile and adult salmonids require large boulders, undercut banks, tree roots and submerged or overhanging vegetation for cover; these are imperative features determining salmonid abundance (deVore & White 1978; Crisp 1996). Unfortunately channel clearance resulting from flood mitigation methods may result in the removal of these habitats. The River Don brown trout populations were mainly dominated by ≥1+ fish. Low numbers of 0+ brown trout at Nursery Street 2010 and 2011, Blonk Street 2010 and 2011, Effingham Street 2010 and 2011, Brightside 2011, Club Mill Weir 2011 and the absence of 0+ trout at Brightside 2010 and Club Mill Weir 2010 indicate exceptionally poor recruitment for this species. This suggests there is poor availability of spawning habitat or poor suitable nursery habitat for juvenile trout at all five sites and because this is critical for the survival for 0+ trout (Armstrong *et al.* 2003), its absence could further result in a low survival rate to 1+ or greater. The low number of 0+ trout at Club Mill Weir (control site) suggests the flood mitigation and rehabilitation works may not have impacted on 0+ trout recruitment but there may already be pressures on this system that affect this age class of trout.

Grayling also rely on suitable gravels for spawning (Maitland 2004) and the absence or disturbance of this gravel, resulting from instream flood mitigation works could have detrimental effects on grayling survival. However, on the River Don the population structure of grayling was generally balanced, as indicated by the presence of juveniles and a mature adult stock. Evidence of spawning of grayling (i.e. 0+ fish), was found in all five study reaches indicating successful recruitment and the presence of suitable spawning substrate for these species. Numbers of 0+ grayling were lower in 2011 surveys when compared to 2010, even though mature grayling numbers were plentiful in 2010 surveys, therefore, suggesting a possible disturbance to suitable grayling spawning habitat or the response to natural variation found across all sites.

The majority of cyprinid species use submerged plants as suitable spawning substrata. The removal of instream channel vegetation for flood mitigation may reduce the recruitment of cyprinid species, however, they are inclined to be more versatile than salmonids when utilising other substrata for spawning purposes (Cowx *et al*. 2004). Most cyprinid species spawn during May to July (Mann 1996), so it is essential this is taken in to consideration when organising flood alleviation works. Most cyprinid larvae select habitats where the flow velocity is well below the critical level such as riparian habitats that also provide shelter, vital for juvenile and adult cyprinids to refuge (Cowx *et al.* 2004). The River Don roach population structure at Nursery Street, Blonk Street and Brightside appeared unstable. Low numbers of 0+ roach at Nursery Street and Blonk Street indicating poor recruitment in 2010 and 2011 surveys. The presence of a good number of mature roach in 2010 at these sites confirm it is not the cause of this reduction in 0+ roach and therefore, the reduction could be a result of unsuitable spawning and nursery habitat not supporting this species. Suitable spawning and refuge habitat such as weed in shallow water are missing from the Nursery Street as a result of flood defence work (Maitland 2004). At Brightside there was a high number of 0+ roach in 2010 but a low number of >1+ roach and in 2011 surveys poor numbers of both 0+ and >1+ roach illustrating how unbalanced the population is, because this influences a variety of age classes it is evident that vital habitats are missing and need to be identified to conserve future roach populations in the River Don in the Sheffield area. Dace were present across the majority of sites but in low numbers, at Blonk Street there were no 0+ individuals captured in 2010 or 2011 surveys suggesting the absence of suitable spawning habitat for Dace, but the present of >1+ Dace suggests suitable habitat for mature fish. Conversely, 0+ dace were captured at Brightside in both 2010 and 2011 surveys but in low numbers however, >1+ dace were caught in good numbers at both years suggesting suitable habitat for mature dace. Overall, 0+ chub were absent from all five sites on the River Don before and after flood mitigations works suggesting a long-term problem with suitable spawning habitat for chub. Mature chub were present in low numbers across several sites with the exception of Brightside where good numbers of mature chub were present; suggesting suitable habitat for the life stage of this species.

Overall, the output of the fisheries surveys in August 2010 and August/October 2011 identified no obvious impact of flood defence works in the Nursery Street, Blonk Street, Effingham Street and Brightside sections. **The decrease in fish numbers across all sites between August 2010 and August/October 2011 surveys was a general trend across all sites, including the control site and therefore indicates that this decrease was highly influenced by natural variability within the system and not a result of flood defence works**. It is recommended that monitoring continues in all sections following flood defence works to identify any further changes in fish populations in response to habitat modifications. There are many additional pressures on the urban rivers system of the River Don, for instance, urbanisation, channelisation and loss of connectivity to name a few. Numerous pressures in addition to the natural variability complicate the process to isolate the impact of the flood mitigation and rehabilitation works on the local fish population. Long term monitoring is essential to attempt to overcome these extra pressures and natural variability, ideally long term pre-monitoring is needed to get a true baseline however, this was not possible for this project and can only be advised for future projects.

4.10. Rehabilitation in an urban river

River rehabilitation is a complex practice; furthermore, urban river rehabilitation is even more so because of additional urbanised pressures on the system. Findlay and Taylor (2006) reported that there are many benefits to rehabilitate urban river systems although they cannot be returned to a natural non-impacted condition, ecological functioning can still be improved. The River Don at Sheffield is restricted both laterally and longitudinally; the restriction comes from high walled banks to protect the city from flood and the systematic weirs from past industry.

There is evidence of migrations amongst large bodied fish species (Baras & Lucas 2001), and often these migrations are to annual spawning grounds. Loss of spawning habitat due to river engineering is also a major constraint on fish populations in UK rivers and damage to key spawning areas in the River Don could seriously impact on the future status of the fishery. The River Dons considerable impoundment of the river through weirs makes migration extremely difficult; hence damage to any spawning grounds within individual sections could have serious consequences to the fish populations within a section. These restrictions to fish migration remove essential key habitats from the river, for fish such as breeding, feeding and refuge locations (Winemiller & Jepsen 1998; Welcomme *et al.* 2006). The scale of fish migration can range from tens of metres (resident fish, e.g. brown trout and bullheads) to tens or hundreds of kilometres (potamodromous migration, e.g. lake or river resident brown trout, barbel), or even to thousands of kilometres (diadromous migration, i.e. sea trout, salmon, eel) (Cowx *et al*. 2004). The ideal rehabilitation solution is to work at a catchment scale, to removal all barriers and open up the longitudinal connectivity of the river however, this is not always practical within urban rivers, especially the River Don. Fish passes are a method used to overcome connectivity restrictions resulting from weirs and are gradually being introduced on the river Don however; they are not suitable for all fish species. The ability of fish to navigate weirs in the River Don is unknown, but it is anticipated that fish may be constrained within impounded sections so must complete their spawning cycle within a limited river length. This can have serious implications on the sustainability of fish populations as it can result in competition for spawning areas. Therefore, rehabilitation of the river section between each weir is an alternative option to introduce suitable habitat for all life stages of fish present. Species that migrate laterally on and off floodplains in response to seasonal changes cannot do so in such urbanised rivers (Welcomme *et al.* 2006), it is suggested that slow moving side waters are included in urban river rehabilitation where flood plain connection is not practical.

One individual *Lampetra* ammocoete was captured in the Nursery street section in February 2010 probably arising from downstream movement from the River Rivelin where self sustaining populations exist (personal observation). *Lampetra* ammocoetes cannot be identified to species level in the field but the species is likely to be brook lamprey, (*Lampetra planeri*) because of barriers to anadromous lamprey species migration in the Don catchment. Brook lampreys are listed in annexes IIa and Va of the Habitats Directive, Appendix III of the Bern Convention, and in the Long List Species in the UK Biodiversity Action Plan (BAP), consequently protection of the species is paramount in any instream riverine activities.

From a conservation perspective flood mitigation is a destructive act that can only negatively affect river ecosystems, however when considering social and economic factors in the 'bigger picture', FRM becomes a very necessary procedure especially in current climate change circumstances. The dilemma is how to apply flood risk management while still conserving river ecosystem, perhaps integrating the needs for both can benefit future urban river rehabilitation from its existing approach? Most river rehabilitation techniques are not feasible within large urban rivers (Francis & Hoggart 2008) and present FRM will now have to guarantee future investment in to river rehabilitation to support the WFDs requirements, with this in mind maybe it can be used to our advantage to improve future knowledge of integrating flood mitigation and river rehabilitation.

4.11. Planning Monitoring & Assessment

The Environment Agency recognizes the importance of a well designed framework and have incorporated adaptive management in to the Sheffield plans to overcome challenges and uncertainties within flood risk management and to minimise potential environmental impacts. A well designed adaptive flood risk management plan for an urban river is important and will reduce the uncertainty of management actions (Roni *et al*. 2005a) accounted for through the implementation of policies and sampling strategies. It promotes an on-going process that should involve a sequence of steps and feedback loops designed to cover the planning, action, monitoring and evaluation components of a project framework (Smith *et al*. 1998). Urban flood risk management planning should start with the assessment of existing pressures on the river system and subsequently followed by the assessment of present and future flood risks (WMO/GWP 2008) whilst taking account of economic, social and environmental aspects. Objectives are an essential part of project planning and are vital when it comes to flood risk management as they help focus the project toward required end results and allows managers to know when they have reached their goal by testing objectives against results. Evaluating the effectiveness of flood risk management actions through monitoring is imperative, pre-monitoring and assessment evaluates the current river health and fish status and is crucial to isolate the impact of flood mitigation and rehabilitation works from existing pressures and natural variability, post-monitoring after implementation is necessary to assess the success of flood risk management actions or rehabilitation works (Section 2.2).

Reporting results for dissemination is an important step and will enable us to learn from success or failures, and thus improve flood mitigation and river rehabilitation management tools. When incorporating river rehabilitation such as in-stream structures in to flood risk management, adaptive management is fundamental to ensure there will be no adverse effects on flood risk, the river bank, river flows or existing structures (Environment Agency 2010). The application of monitoring and evaluation is slowly being promoted in various areas of FRM and river rehabilitation to assists the EU WFD aim to ensure rivers reach good ecological status or potential by the year 2027.

4.12. Future works

The River Don flood works are proposed as part of an ongoing programme in order to alleviate flooding issues in Sheffield and it should be highlighted that it is a work in progress. Although present data does not necessarily demonstrate that flood mitigation methods on the River Don had negative effects on the present fish population, the rehabilitation techniques applied did not prove to benefit fish populations either. It is essential that long term data is collected to identify trends and overcome natural variability. **It is recommended that monitoring continues in all sections following flood defence works to identify any changes in fish populations over time in response to habitat modifications and therefore increase our knowledge of flood risk management strategies that incorporate rehabilitation practise.**

CHAPTER 5. DISCUSSION

5.1. Limitations in monitoring river rehabilitation success

Current scientific understanding of river rehabilitation is generally poor (Vaughan *et al*. 2009), many uncertainties still arise and there is still limited understanding of how river systems and catchments respond to rehabilitation (Szaro *et al*. 1998; Downs & Kondolf 2002; Gillilan *et al.* 2005; Jansson *et al*. 2005). Many past and recent papers have highlighted a lack of information on the success of rehabilitation projects and consequently, there are many calls for further research through monitoring and evaluation to improve knowledge in this area (Tarzwell 1937; Reeves *et al*. 1991; Brookes & Shields 1996; Edgar *et al*. 2001; Ward *et al*. 2001; Downs & Kondolf 2002; Roni *et al*. 2002; Bernhardt *et al.* 2005; Roni *et al*. 2008). Although there is a steady increase of rehabilitation projects each year, there is still insufficient capability to assess the effectiveness of rehabilitation techniques and this results from inadequate monitoring and evaluation (Eden & Tunstall 2006; FAO 2008). The previous chapters identify limitations within the monitoring and evaluating stage of river rehabilitation by giving a practical insight in to a number of different rehabilitation schemes and demonstrate the difficulties in collecting meaningful results to measure project success.

The main outcome of this study identified temporal and spatial monitoring to be a limiting factor when assessing rehabilitation success. There was a general decrease in fish numbers across all case study sites between 2010 and 2011, including control sites and therefore, indicate that the decrease was highly influenced by natural variability within each system and not necessarily a result of the rehabilitation works. Evaluating river health and fish status is the first stage to consider before considering river rehabilitation, this had been over looked by river managers in all 5 case studies resulting in either no pre-fisheries data (Manifold and Stiffkey) or a limited 1 years pre-fisheries data (Driffield Beck, Lowthorpe Beck and Dovedale weir removal). The lack of sufficient pre-monitoring data limited the evaluation of all case studies. However, this was overcome with control sites where possible, meaning a number of impact assessments were trialled (BACI for Dovedale weir removal and Post-treatment design for the Manifold brash revetment and the Stiffkey gravel reinstatement). Unfortunately, they were still limited by sample size because the timeframe of the PhD, and allowed for only two years of sampling data to be collected. These limitations of sample size and time frame for monitoring are also found in management practise due to timeframes and budgets of constrains.

Fish as biological indicators, coupled with a lack of temporal monitoring further limited the ability to identify river rehabilitation success. Although fish are regularly reported as good indicator for change, especially for evaluation of river rehabilitation success, this was not the case for all 7 case studies presented in this thesis. Impact assessments were difficult to apply on small sample sizes. A large sample size is needed to reduce variability between the mean densities and therefore, I recommend monitoring is considered before rehabilitation commences, enabling a good sample size for pre-data. Control sites were applied to overcome absent pre-monitoring, but in most cases the variability between mean density of 0+ and >1+ was large. Further investigation found this could be due to a difference in habitat characteristics of the chosen control sites. A good example of this is the River Stiffkey case study, where both impact and control sites were on the same river and the characteristics of this river were similar on the whole stretch. Although a small sample size (2 control and 3 impact sites), little variation was found between the mean density of 0+ and >1+ brown trout. Whereas, for the Manifold and Dove case studies, a larger number control sites (6) were used across different rivers in the same catchment, with similar characteristics, in an attempt to produce a robust sample size. However, this resulted in high variability of brown trout densities between sites. This highlights the importance of selecting suitable control sites. Long-term spatial monitoring is essential to evaluated river rehabilitation success when using fish as indicators of this success.

It is therefore recommend that regular long term monitoring in addition to controls sites will account for natural variability in fish populations, the number of years needed to detect a change can be calculate by application of a resource equation. However, this involves a lot of time and effort to be invested into the planning of a monitoring design, often resulting in a pilot study to determine the level of future sampling for the resource equation, before any rehabilitation takes place. A resource calculation will identify which monitoring design (BI, BACI or Post-treatment Design, Section 2.3) is most feasible depending on the number of control and impact sites, in addition to the possible number of years pre and post monitoring. While acknowledging that resources will often be limited for monitoring, this can be overcome by suitable project planning. Monitoring does not necessarily have to be costly and time consuming. For example, incorporating the Environment Agency's annual fisheries monitoring and habitat (HABSCORE) data can strengthen statistical analysis for impact assessment and resource calculations. Unfortunately Environment Agency fisheries data were limited for some of the case studies within this thesis and therefore could not be used as suitable control data. This illustrates the necessity for pre-planning all stages of river rehabilitation to identify what information is available before mitigation work has begun.

In addition to fish being a poor indicator of success over a short time frame, stocking of fish by the Environment Agency became an unforeseen constraint that further hindered the ability to identifying project success. Stocking of fish is a popular management decision for rivers where fish numbers are low, especially sections where angling takes place. Pressures on river systems tend to reduce self-sustaining fish populations and stocking is usually used as a quick fix measure to overcome multiple existing pressures (Cowx & Welcomme 1998).Stocking should not be seen as a measure to overcome pressures because it masks change or bottleneck in recruitment that could occur as a result of river rehabilitation or existing pressures, limiting the ability to use fish as biological indicators for success. Stocking was present in all river systems, with the exception of the River Stiffkey and was only established half way through the project and could therefore, not be overcome. Stocking at Driffield Beck and Lowthorpe Beck are good examples to prove that it will not overcome exiting pressures and constraints, here the fish merely surviving and not thrive. If annual stocking continues then it is almost impossible to relate fish population change to river rehabilitation, even with long term monitoring. To overcome this constraint, stocking of fish would have to stop, or at least temporarily stopped while enough temporal monitoring took place on the current fish population. If stocking was still to occur then rehabilitation success would have to be based on monitoring multiple BQE and habitat variables. Additional BQE, such as invertebrate or macrophytes sampling would strengthen project evaluation in response to rehabilitation.

The application of the European Fish Index + is a common assessment method for rivers at a European scale because it enables ecological status to be assessed on a community level. It is the only fish index that has been successfully used at a European scale, thus allowing comparisons of river health and more specifically river rehabilitation outcomes, between European countries, further supporting the WFD (EFI+ CONSORTIUM 2009). Nevertheless, such a complex database does not come without limitations (Sampling location, environment, sampling method applied, low species richness & number for fish caught). Low species richness and low numbers of fish prevented the use of EFI for Driffield Beck, whilst EFI was applied with caution for the Hartington case study because numbers of fish were low. This highlights the need for adequate sampling to assess the abundance and structure of fish assemblage and the population structure of the species caught. In cases of low density, expert judgement will have to be satisfactory to assess the ecological status of the river. Monitoring habitat change through the application of HABSCORE for each of the case studies was a sufficient method to identify river rehabilitation improvements of habitat for trout populations, although there was insufficient fisheries data short term, this can be overcome by using the HQS as an indicator of habitat improvement after river rehabilitation.

It is a challenging prospect for all projects to include such detailed monitoring and further encourage future development in this area. This can partially be overcome by selecting only a number of projects to perform detailed, long term monitoring (both pre and post) and preferably those considered at a catchment scale as this is the direction future rehabilitation activities should be heading. This approach is also supported by Buijse *et al*. (2005) and Palmer & Allan (2005). Conversely, Bernhardt *et al*. (2007) established that only one of more than 300 interviewees mentioned that a scientific paper significantly informed the design and implementation of a project; it is also apparent that scientific research is unlikely to make its way quickly into rehabilitation practice (Shields *et al*. 2003). This demonstrates the importance of continued monitoring and evaluation for all projects, even if at a smaller level of detail, and that the information is disseminated rapidly and in a format that is accessible. However, care must be taken because 'one size does not fit all' and only gaining information on a few studies may not be transferable to all studies. Rehabilitation projects use a variety of techniques, on a variety of different rivers making comparison of different techniques difficult (Roni *et al*. 2008). Nevertheless, it is essential that each rehabilitation project is monitored to ensure it has reached its aim and objectives to rehabilitate and create habitat for specific species, to ensure that good status is accomplish and maintained (Kershner 1997; England *et al*. 2007). It is therefore essential that a sufficient amount of information is gathered from all projects and disseminated not only through scientific literature but through reports, websites (e.g. RESTORE WIKI) and stakeholder workshops. Furthermore, monitoring and analysis should not be too complicated; it needs to be practical and easily applied to common management practise. If individual rehabilitation projects prove effective at reaching their ecological goals, the probability of additional funding for monitoring will be higher (Bernhardt *et al*. 2007).

5.2. Benchmarking, endpoints and success

River rehabilitation is progressively more common in water management in the UK, but there is still limited information for evaluating the success of these efforts; this is also common in the United States and across Europe (Bernhard *et al.* 2007). Designing a channel that will function naturally to meet rehabilitation goals is a complex process, monitoring and evaluation are put in place to identify rehabilitation project success, but how do we assess what is successful? Freshwater ecosystems are dynamic systems with continuous movement; it is important to keep this in mind whilst considering 'when' to rehabilitate as they cannot be rehabilitated to a static state (Hobbs *et al*. 2011). It is also important to consider how to determine success at a local and catchment scale.

Despite improved knowledge of ecological, economic and social aspects of river rehabilitation (Postel & Richter 2003), there is still no agreement on what represents a successful rehabilitation project (Jansson *et al*. 2005). This is especially true as the judgment of rehabilitation success can vary between stakeholders, particularly if they are from different disciplines and it is likely that this will play an important role in what individual considers as success in the context of a conservation project (Howe & Milner-Gulland 2012a). This was identified by Jones (2012) where individuals from different disciplines scored rehabilitation project differently. Projects labelled rehabilitation successes should not be assumed to be ecological successes; many projects are classed as rehabilitation when no ecological aspects have been considered in the planning, such as protecting infrastructure and re-building parks that gain economic and social success. For example, Sutcliffe Park, River Quaggy – Chinbrook Meadows and River Pool Linear Park Enhancement are all rehabilitation case studies from a social perspective, to protect against flood mitigation and to look aesthetically pleasing to the public. However, they do not consider river processes or biota (RESTORE WIKI www.restorerivers.eu). Palmer *et al*. (2005) illustrated the most effective river rehabilitation projects lie at the intersection of the three primary axes of success: 1) stakeholder success reflects human satisfaction with rehabilitation outcome, 2) ecological success reflects advances in scientific knowledge; and 3) learning success and management practises that will benefit future rehabilitation action (Figure 5.1).

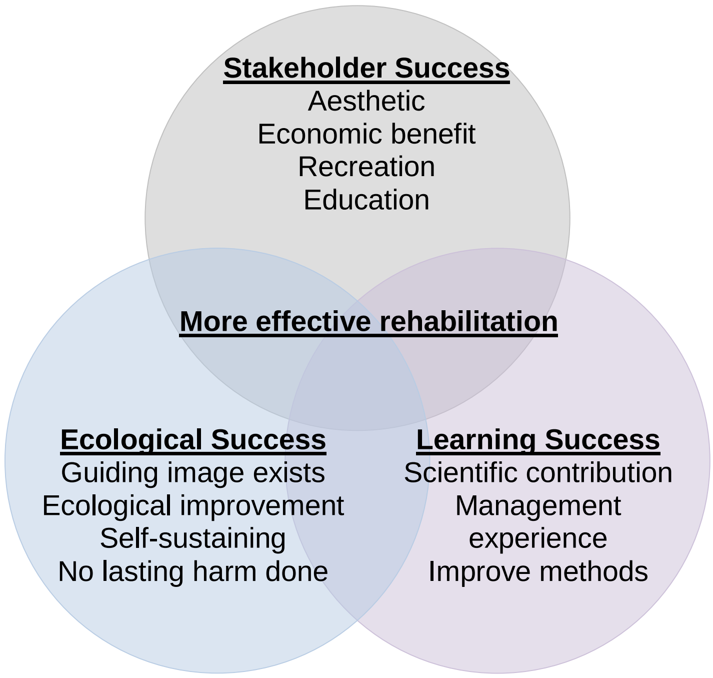


Figure 5.1. The most effective river rehabilitation projects lie at the intersection of the three primary axes of success (from Palmer et al. 2005).

Setting benchmarks and end points that are linked to defined project goals is the latest approach to help determine the measure of success (Roni *et al*. 2013) especially when goals are linked to objective success criteria to guide the process and the likelihood of achieving the end result (Bernhard *et al*. 2007). River rehabilitation requires several areas of knowledge such as ecology, hydrology and engineering (Doyle *et al*. 1999). If goals are relating to composition, structure, function and so on, it is difficult to know what measures should be used to quantify the success (Hobbs & Harris 2001). The meaning of 'success' will change depending on the type of project, the condition of the river health and the ecosystem services it supplies. For example, areas of HMWBs need only reach good ecological potential and therefore will have different benchmarks, endpoint and measures of success to achieve. It may be more achievable to reach a level of success when the goal is to rehabilitate a certain level of function/species rather than to attempt complete rehabilitation (Lockwood & Pimm 1999) and therefore realistic goals are essential for progress towards success (Hobbs & Harris 2001; Hobbs 2007).

The concept of increasing habitat heterogeneity to increase biodiversity through rehabilitation has been a long-standing approach (Jungwirth *et al*. 1995; Kondolf & Micheli 1995; Palmer *et al*. 1997; Kemp *et al.* 1999). Nevertheless this is not always the smartest approach. Introducing the design of benchmarking and endpoints into the planning stages will only strengthen rehabilitation practise as it steers away from ambiguous proposals towards a more definite ideal of the required ecosystem in a specific segment of river. However, from the vast quantity of rehabilitation literature available, only a small number of papers address the issue of measuring success and an even smaller number address benchmarking and endpoints. It seems there are no definite criteria to define endpoints and benchmarks against which to measure performance and with no exact criteria, establishing appropriate targets for rehabilitation activities appears challenging.

Benchmarking uses representative sites otherwise known as 'reference sites' on rivers that have the required ecological status and are relatively undisturbed. This is then used as a target for restoring other degraded sections of river within the same river or catchment. This approach therefore uses appropriate undisturbed sites of the same river type (Rheinhardt *et al*. 1999), rather than attempt to create conditions unrelated to the original ones at the site of interest and is consequently more likely to result in long-term success (Choi 2004; Palmer *et al*. 2004; Suding *et al*. 2004; Woolsey *et al*. 2007). It is imperative that endpoints accompany benchmarking in the planning process to guarantee the prospect of measuring success because endpoints are feasible targets for river rehabilitation, especially as they do not need to be quantifiable. It is important to note that endpoints are different to benchmarks, this is because other demands on the river systems also have to be met and references can only function as a source of inspiration on which the development towards the endpoints is based (Buijse *et al*. 2005). Given that benchmark standards cannot always be achieved, especially on urban rivers, endpoints will therefore assist in moving rehabilitation effort towards benchmark standards through application of the SMART approach (Chapter 2) to decide what is achievable and what is feasible. It is important to recognise what is the minimum acceptable achievement level of rehabilitation and what is the desirable level to have as a target end point that is still underneath the benchmark level, but still aims for WFD status targets. Subsequently, what can be compromised for this desired level, will it be cost, ecosystem services or ecological aspects?

Albeit, applying benchmarking to increase the accuracy and success of rehabilitation appears in theory to be an uncomplicated method, it increases the level of intricacy that rehabilitation needs to be applied. This is because natural instream habitats consist of complex multidimensional arrays of morphological conditions (substrate, woody debris, hydraulic patterns) along with the complex life structures and habitat guilds of the biota (Colwell & Futuyma 1971; Statzner *et al*. 1988; Strange 1999) and the environmental conditions (velocity, depth, temperature) and resources (food, space) on which they depend, all of which need to be incorporated in to river rehabilitation. As a result, river rehabilitation practise is prevented from moving forward as the reoccurring problem of how to rehabilitate such a complex systems is revisited and the only way to move forwards is to identify project success of which benchmarking and end points will play a vital role in future rehabilitation management. It will enable us to identify trends, successful techniques and compare actual performance with planned outcomes.

The use of benchmarking and endpoints in existing projects is minimal, and because of this it is difficult to find standardised criteria to define benchmarking and endpoints to measure success and determine appropriate targets for rehabilitation activities. Palmer *et al*. (2005) proposed 5 criteria to measure ecological success of river rehabilitation; however, further research in this area is still needed to understand if they are constructive:

1. the existence of a 'guiding image' to influence a dynamic endpoint that is identified a priori and guides the rehabilitation;
2. the ecosystems are improved and the ecological conditions of the river are measurably enhanced;
3. the adaptive capacity is increased so that the river ecosystem is more self-sustaining than before the rehabilitation;
4. no lasting harm is done by the rehabilitation;
5. some level of pre- and post-project assessment is conducted and the information shared.

Tangible, attainable and scientifically sound endpoints need to be identified to direct and focus efforts on the planning process and the definition of endpoints is necessary to develop prognostic tools that identify the geomorphological and ecological consequences of rehabilitation measures and their respective spatial and temporal scales (Buijse *et al*. 2005).

When considering spatial and temporal scale of a river system it emphasises that rivers are a continuous state, they are dynamic and forever changing, therefore it is important to make sure endpoints are understood and used in the correct manner. Part of the complexity of judging successful ecological rehabilitation at a spatial scale is deciding when the process is 'complete' (Jansson *et al*. 2005). Hughes *et al*. (2008) have an alternative idea towards rehabilitation that differs from the need for endpoints, to the requirement of 'open-ended' rehabilitation that would encourage natural processes dictate ecological outcomes rather than attempting to steer them to fit a pre-selected reference system. It is easy to appreciate that open-ended rehabilitation would overcome the uncertainty of an ever changing ecosystem; however, it will not advance river rehabilitation from where it is to date. The open-ended concept is not suitable for river rehabilitation management because it will produce the same practical issues, such as how to frame the goals for the project, and how to monitor and evaluate change of which Hughes *et al*. (2011) later identified. Perhaps it is a good suggestion that the open-ended concept promotes the need to assess long term outcomes, especially as system shifts in environmental processes are to be expected, and this highlights the importance of long term monitoring.

5.3. Catchment planning rehabilitation

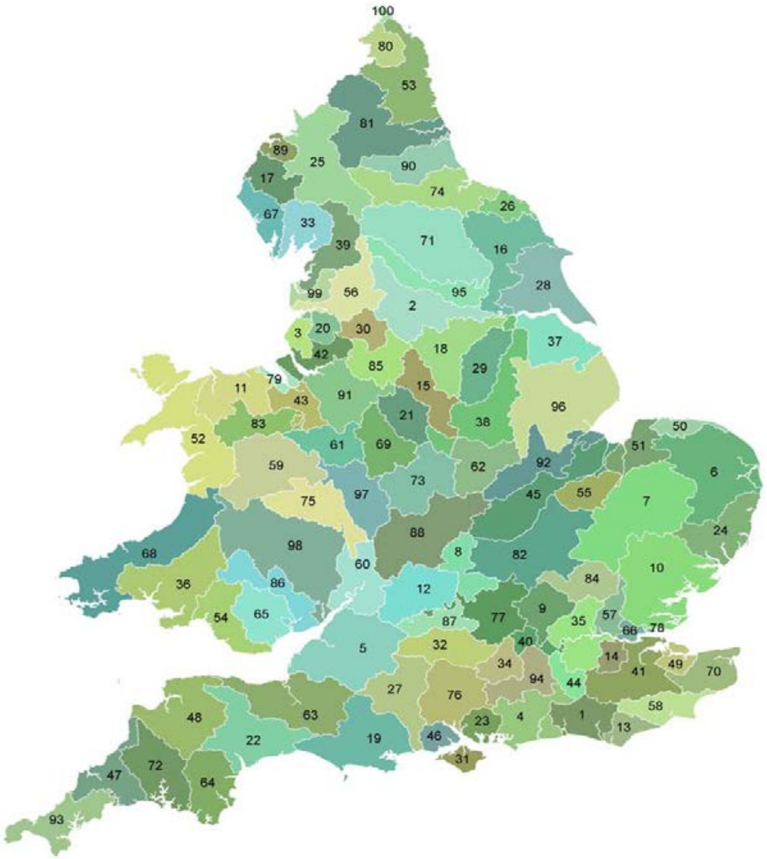
The concept of returning a river to a pristine or pre-existing state using mitigation measures to overcome degradation is unrealistic and dated, especially due to the irreversible changes in catchment boundary conditions (e.g. impervious surface area, hydrology, vegetation cover (Findlay & Taylor 2000)). Broad-scale processes and interactions between adjoining ecosystems will add further complexity, because impacts in one place may be the result of events or management decisions elsewhere (Findlay & Taylor 2000; Hobbs 2002). River rehabilitation programme goals often only address problems on single rivers at a small scale and therefore have limited impact on catchment-scale processes (Buijse *et al*. 2005; Eden & Tunstall 2006). While all case studies evaluated in this study were small scale rehabilitation projects, it is important to mention how they should be fitting into the bigger picture of catchment planning.

Freshwater river ecosystems are intrinsically linked and have a natural habitat continuum between river and landscape (May 2006), because of this it is difficult to conserve a small reach of river by simply using rehabilitation practice at a local level. Fortunately potential benefits of implementing river rehabilitation and conservation at a catchment-scale are being increasingly recognised as an essential component of future rehabilitation practise (Hodder *et al*. 2010), especially through the WFD aims to combine catchment scale understanding across a range of aquatic ecosystems to improve ecological status within specific river basins. Although the development of catchment scale management has started to be applied within the UK, single, small scale rehabilitation exercises are still employed most frequently with no association to catchment plans at a larger scale. Small scale rehabilitation is most frequently employed because it is cheap, easy to apply and is quick to accomplish. Consequently, it is therefore important to understand how to apply small scale rehabilitation to benefit at a larger scale and to integrate this approach at a catchment scale. Overall there is a need for more large scale catchment programmes where river basin wide assessment will enable prioritisation of rehabilitation sites (Buijse *et al*. 2005) and in some instances assessment will identify large pressures where rehabilitation at small scale, single reaches may not be an appropriate approach (Palmer *et al*. 2005). Catchment planning will require long-term planning over a number of years adapted over time that should be accustomed to changes to ensure the best rehabilitation methods are being applied at all times.

River Basin Management Plans (RBMPs) are a requirement of the WFD to reach sustainable catchment river rehabilitation to meet WFD objectives through the Programme of Measures (PoM) and further support catchment planning. RBMPs identify pressures and remedial actions at a river basin level and demonstrate what actions need to be taken to address pressures and how the actions will make a difference to the local environment. The RBMPs are to be updated every six year, with the second round of RBMPs are to be released 2015. The WFD therefore aims to prevent further deterioration of rivers and has the potential to increase the number of rehabilitation schemes undertaken across Europe, to achieve good ecological status (GES) and to ensure it is maintained once achieved.

In England and Wales there are 100 catchments (Figure 5.2a) and therefore it is important that catchment based management becomes a more used approach, especially by the Environment Agency. In April 2011, pilot studies were introduced to support RBMPs. Ten catchments (Figure 5.2a & Table 5.1) were selected as pilots with a view to apply effective elements to other catchments from 2013. In addition, a further 15 pilots (Figure 5.2b & Table 5.1) were established in January 2012 and these were hosted by external organisations such as rivers trusts and local water services. All pilot studies are to run until the end of 2012 and the most successful parts of the approaches will form the basis of a longer-term approach to promote good practise (Environment Agency 2012a). In addition to the 25 catchment pilots, the Environment Agency also include 41 wider initiatives in other catchments lead by organisations other than the Environment Agency (Figure 5.2b & Table 5.1). The catchment pilots are intended to protect river wildlife and habitats by:

- encouraging participation to improve the water environment;
- delivering a range of environmental benefits for the community;
- developing a shared understanding of the catchment priorities;
- making sure participants feel the pilot has made a difference in what can be achieved (Environment Agency 2012a).



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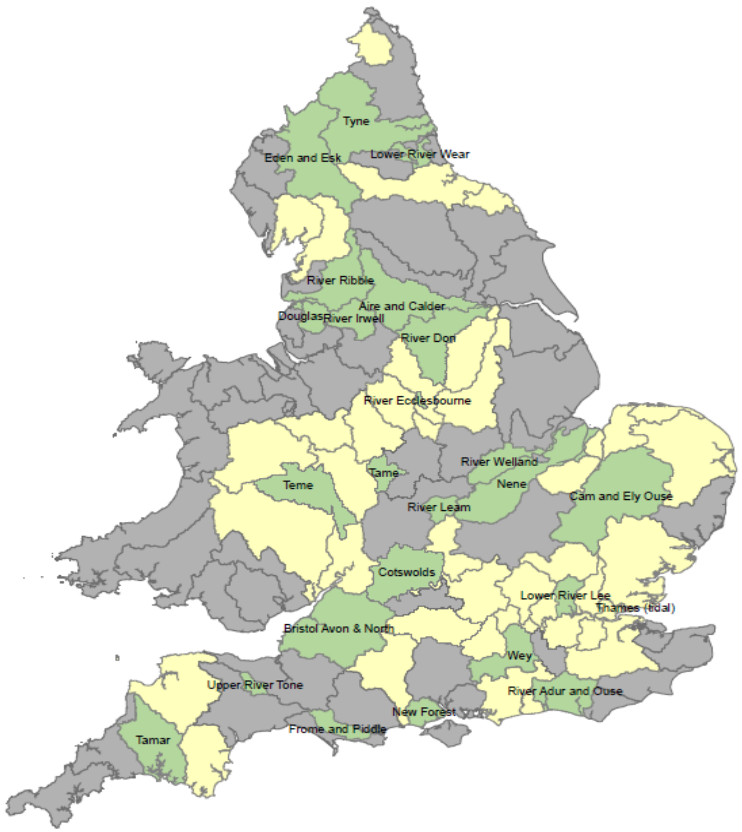

b

Catchment where a Pilot will be evaluated by Defra

Wider Catchment Initiatives

Figure 5.2. a) WFD management catchments (Table 5.1), b) a catchment based approach (2012a).

Table 5.1. Key to Water Framework Directive Management Catchments, catchment hosted by environment agency, catchment hosted by other organisations (Environment Agency 2012a).

ID	CATCHMENT NAME	RIVER BASIN DISTRICT	ID	CATCHMENT NAME	RIVER BASIN DISTRICT
1	Adur & Ouse	South East	51	North West	Norfolk Anglian
2	Aire and Calder	Humber	52	North West Wales	Western Wales
3	Alt/Crossens	North West	53	Northumberland Rivers	Northumbria
4	Arun & Western Streams	South East	54	Ogmore to Tawe	Western Wales
5	Bristol Avon & North Somerset Streams	Severn	55	Old Bedford	Anglian
6	Broadland Rivers	Anglian	56	Ribble	North West
7	Cam and Ely Ouse (inc South Level)	Anglian	57	Roding, Beam & Ingrebourne	Thames
8	Cherwell	Thames	58	Rother	South East
9	Colne	Thames	59	Severn Uplands	Severn
10	Combined Essex	Anglian	60	Severn Vale	Severn
11	Conwy and Clwyd	Western Wales	61	Shropshire Middle Severn	Severn
12	Cotswolds	Thames	62	Soar	Humber
13	Cuckmere & Pevensey Levels	South East	63	South & West Somerset	South West
14	Darent	Thames	64	South Devon	South West
15	Derbyshire	Derwent Humber	65	South East Valleys	Severn
16	Derwent (Humber)	Humber	66	South Essex Thames	Thames
17	Derwent (NW)	North West	67	South West Lakes	North West
18	Don and Rother	Humber	68	South West Wales	Western Wales
19	Dorset	South West	69	Staffordshire Trent Valley	Humber
20	Douglas	North West	70	Stour	South East
21	Dove	Humber	71	Swale, Ure, Nidd & Upper Ouse	Humber
22	East Devon	South West	72	Tamar	South West

23	East Hampshire	South East	73	Tame Anker and Mease	Humber
24	East Suffolk	Anglian	74	Tees	Northumbria
25	Eden and Esk Solway	Tweed	75	Teme	Severn
26	Esk and Coast	Humber	76	Test & Itchen	South East
27	Hampshire Avon	South West	77	Thame and South Chilterns	Thames
28	Hull and East Riding	Humber	78	Thames (tidal)	Thames
29	Idle & Torne	Humber	79	Tidal Dee	Dee
30	Irwell	North West	80	Till	Solway Tweed
31	Isle of Wight	South East	81	Tyne	Northumbria
32	Kennet and Pang	Thames	82	Upper and Bedford Ouse	Anglian
33	Kent/Leven	North West	83	Upper Dee	Dee
34	Loddon	Thames	84	Upper Lee	Thames
35	London	Thames	85	Upper Mersey	North West
36	Loughor to Taf	Western Wales	86	Usk	Severn
37	Louth Grimsby and Ancholme	Humber	87	Vale of White Horse	Thames
38	Lower Trent & Erewash	Humber	88	Warwickshire Avon	Severn
39	Lune	North West	89	Waver_Wampool	Solway Tweed
40	Maidenhead to Sunbury	Thames	90	Wear	Northumbria
41	Medway	Thames	91	Weaver/Gowy	North West
42	Mersey Estuary	North West	92	Welland	Anglian
43	Middle Dee	Dee	93	West Cornwall and the Fal	South West
44	Mole	Thames	94	Wey	Thames
45	Nene	Anglian	95	Wharfe and Lower Ouse	Humber
46	New Forest	South East	96	Witham	Anglian
47	North Cornwall, Seaton, Looe and Fowey	South West	97	Worcestershire Middle Severn	Severn
48	North Devon	South West	98	Wye	Severn
49	North Kent	Thames	99	Wyre	North West
50	North Norfolk	Anglian	100	Tweed	Solway Tweed

On World Water Day, 22 March 2011, Richard Benyon, Minister for Natural Environment and Fisheries, announced that these pilots should: *'Provide a clear understanding of the issues in the catchment, involve local communities in decision-making by sharing evidence, listening to their ideas, working out priorities for action and seeking to deliver integrated actions that address local issues in a cost effective way and protect local resources.'*

Lessons learned from the pilots managed by the Environment Agency are collated in the Catchment Pilots Lessons Portfolio that was produced October 2012, further to this document there is also a document to verify how the Environment Agency will support and assist others (Environment Agency 2012a).

The Catchment Restoration Fund (CRF) was created by Defra to support catchment level rehabilitation to work towards meeting the objectives of the WFDs good ecological status or good ecological potential. It has a £28 million fund for rehabilitation projects in the UK between 2012 and 2015. The Environment Agency is administering the CRF to support third sector groups to bring forward projects that are planned at the catchment level and will:

- restore natural features in and around water courses;
- reduce the impact of man-made structures on biota in watercourses;
- reduce the impact of diffuse pollution that arises from rural and urban land use.

The advantage of the CRF is that rehabilitation projects will be correctly planned at a catchment scale, projects are reviewed by technical experts in the Environment Agency, Natural England and the RRC and therefore the well planned projects will be approved. By the end of May 2012 131 application for over £54 million were received, only 42 projects were approved at the combine value of £24.5 million but this will still produce habitat improvements for over 300 water bodies. The CRF works towards well managed river rehabilitation as it encourages those parties involved to prepare a proposal for rehabilitation at a catchment scale that will meet the aims and objective proposed. It also recognises the importance of monitoring and evaluation of projects to ensure project success and lessons learnt for future progress, in addition to cost-effectiveness. The importance of river catchment scale management has been acknowledged in recent years and there are many schemes in place to support this as previously discussed. It is reasonable to suggest that even though river catchment management and specifically catchment scale rehabilitation is in the early stages of development, progress appears to be in the right direction.

5.4. Urban river rehabilitation

Previously urban river rehabilitation from an ecological perspective was given little consideration in comparison to social and economic aspects, but in recent years as the concept of 'sustainable development' has progressed all three aspects are now incorporated (Findlay & Taylor 2000). River rehabilitation projects have become more widespread across the UK in an attempt to reduce the loss of fish habitat through the mitigation of degradation that has resulted from anthropogenic disturbance. Rivers in urban areas tend to be the most degraded and large amounts of money are spent towards restoring these (Bernhardt & Palmer 2007). The goal of urban river rehabilitation should attempt to increase habitat diversity by restoring the ecological structure and function of the system, and to re-establish the natural temporal and spatial variation in these ecological attributes rather than stable conditions (Palmer *et al*. 2005). However, urban rivers are complex systems constrained by a number of existing pressures that cannot be overcome so easily, such as poor water quality, urbanisation, industry, navigation/transport, water regulation and barriers, more recent pressures involve flood protection and hydropower (Booth & Jackson 1997; Kemp & Spotila 1997; Schleiger 2000; Wang *et al*. 2000; Fitzpatrick *et al*. 2004; Blakely & Harding 2005; Brown *et al*. 2005; Schwartz & Herricks 2007). In England 94% of rivers have been modified (Brookes & Shields 1996) and in England and Wales HMWBs make up just over half of the total number of water bodies (Table 5.2, Figure 5.3; Environment Agency 2012b), illustrating the importance for high-quality urban river rehabilitation practise to be put into place. The WFD allows a water body to be identified as HMWB, where the balance of rehabilitation and socio-economic needs means good ecological potential. A large component of urban river rehabilitation is ecosystem services and because of this it is essential to integrate science and social science for river rehabilitation. This can be done by increasing interest in `stakeholder participation' (Healey 1998; Selman 1996; Sunley 1999). This is already in process through the SMURF Project (Sustainable Management of Urban Rivers and Floodplains, see www.smurf-project.info) and aims to develop more effective methods for involving the public and evaluating their responses.

Table 5.2. Water categories for HMWB of England and Wales (Environment Agency 2012b).

Water category	Number of HMWB	Total number of water bodies	% of number
Rivers	2826	5868	48.2
Lakes	266	432	61.6
Transitional	122	134	91.0
Coastal	77	99	77.8
Total HMWB	3291	6533	50.3

Available habitat is often seen as a limiting factor for urban stream health (Moses & Morris 1998) and in many rehabilitation schemes the focus is on returning habitat characteristics to the system in the hope that ecological health will improve (Rosgen 1994; Morris & Moses 1999; Brierley & Fryirs 2000). In some instances, urban river rehabilitation efforts attempt to reverse decades of physical degradation through reshaping the channel, manipulating habitat heterogeneity and replanting riparian vegetation to return the stream ecosystem towards non-urban 'reference' conditions (Bernhardt & Palmer 2007). However, this approach is not practical for urban river rehabilitation because many of the pressures cannot be removed and therefore, using a non-urban reference condition will not produce successful, meaningful results. The 'river continuum concept' addresses the longitudinal linkages within rivers (Vannote *et al*. 1980), while the 'flood pulse concept' integrates the lateral river–floodplain connections. Urban rivers undergo stress from both longitudinal and lateral restriction resulting from constrained banks and barriers caused by weirs. If the two main processes of rivers are missing from urban systems, how do we successfully rehabilitate them? It is here where urban river rehabilitation should be integrated within broader catchment management strategies to understand constraints operating on the whole catchment and identify those that will hinder success. For instance, longitudinal connectivity can be overcome by fish easement methods so upstream and downstream migration can occur, but unfortunately fish easements are not suitable for all fish, especially smaller cyprinid species. In addition, the majority of urban rivers are embanked which result in further constraints by denying fish lateral movement on to the floodplain, further restricting suitable habitat for each life stage of a fish. Therefore, it is necessary to ensure each section of river between barriers has suitable spawning, breeding, feeding and refuge sites for fish (Cowx & Welcomme 1998) to overcome the impacts from urban pressures.

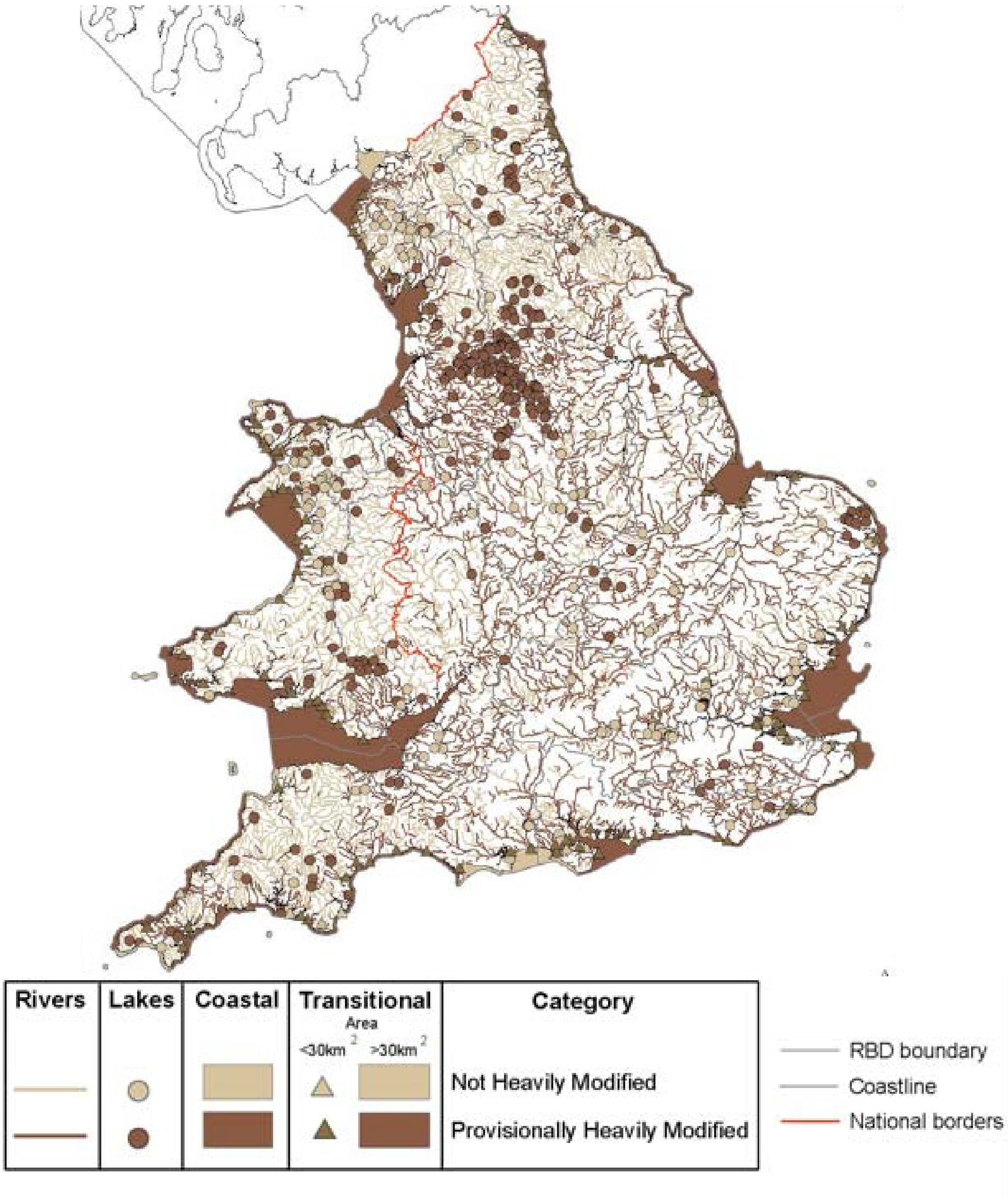

Figure 5.3. HMWB of England and Wales (Environment Agency 2012b)

There has been limited monitoring and evaluation on urban river rehabilitation to determine its success and identify techniques that are suitable for reaching WFD GEP (Bernhardt *et al*. 2005; Palmer *et al*. 2005). Site selection and project design in urban settings should be guided by a fundamental understanding of the operating constraints that may preclude success. An example to learn from is the urban river rehabilitation that took place on the River Skerne, Darlington where the site was restricted by utility pipes which constrained the desired rehabilitation (Eden *et al*. 1999) therefore, giving the appearance of an urban park rather than an active floodplain (Eden & Tunstall 2006). More detailed ecological, geomorphic and hydrologic research and evaluation of un-rehabilitated and rehabilitated urban streams is necessary for guiding the critical decisions about when rehabilitation can have a positive impact on urban stream ecosystems (Bernhardt & Palmer 2007). Management plans need to recognise the range of potential limiting factors such as constrained by available land, urban infrastructure, political pressures and a lack of technical knowledge to set realistic goals (Nilsson *et al*. 2003; Niezgoda & Johnson 2005). Therefore negotiation must be made between the ideal rehabilitation and the practical rehabilitation that is obtainable given the circumstances.

Climate change is a current subject matter that influences rehabilitation practise. In a recent editorial search for climate change papers in Aquatic Conservation, Ormerod (2009) found that 44 references were cited and 43 of these were published in 2000 or later. Its rapid development as an integral part of aquatic conservation has quickly advanced knowledge to confirm that rivers will be profoundly affected by climate change during this century (Boon & Raven 2012). Ecological impacts on biota are likely to be direct and indirect: direct effects will be on life cycles and growth rates from rising temperatures or displacement of animals and plants through higher flows and indirect effects such as the flow-related dilution of pollutants or transport sediments (Ormerod 2009) to indirect impacts such as flood risk management. The more recent pressure of flood risk management on urban rivers is in response to climate change that increases the frequency of flooding (Defra 2007). Provided that river rehabilitation is integrated in to flood risk management actions there is the possibility to reach a win-win situation. However, this integration of EU FD and the UK FWMA with the EU WFD and HD is still in its early stages. More advances are needed in this area of urban rehabilitation to identify an ideal approach.

5.5. Conclusion and recommendations

One key conclusion to be addressed is the need to progress from decisions based largely on subjective judgements to those supported by scientific evidence. The absence of knowledge is due to a lack of understanding of the design and implementation stage of project planning for rehabilitation schemes. Monitoring is often missing from the design stage for river rehabilitation planning even though it is a necessary component that enables project evaluation. **As a consequence, monitoring and evaluation should be compulsory for all river rehabilitation project planning and should be enforced through a regulatory body such as the Environment Agency in England and Wales and further supported by guidelines of '*best practise*' for river rehabilitation**. Monitoring can be achieved by data collection of one or several of the WFD's four BQE depending on what the aims of the project are. It is essential that an impact assessment is designed to not only show a change has taken place but also to provide evidence, in statistical terms, that it is meaningful. A variety of impact assessments techniques are available to detect environmental change for rehabilitation project whose data collection methods differ spatially and temporally. A replicated BACI design is the most powerful design because it includes replication in both space and time and this is recommended. A resource calculation can be applied to determine how many years pre and post monitoring is required to isolate the environmental impact from natural variability. ‘A Guide to Restoring Riverine Processes and Habitats’ by Roni and Beechie (2013) provides advanced river rehabilitation management to this end. It can be used as a guide as it considers all aspects of river rehabilitation planning for successful rehabilitation, with step by step guides and examples to direct the user.

Through the course of this study it has become apparent that there is paucity in data for rehabilitation projects that measure success. This is mainly attributable to a lack of understanding of how to measure success. **A requirement for the future is to define benchmarking and endpoints and to create a protocol to guide users (such as the Environment Agency and Rivers Trusts) to set realistic, quantifiable criteria for river rehabilitation.** This can be prepared by developing a database of good examples that have followed an ideal project framework process, to distinguish the number of successful projects of which to gather information of key features that influenced the project success.

As climate change pressures become more frequent on river systems, it also becomes an important driver for river rehabilitation mitigation and adaptation strategies. **The EU FD and UK FWMA need to be integrated with the EU WFD and HD, to work towards FRM whilst still considering river heath**. This approach is still in its early stages and there are no examples in the literature that report successful FRM that has incorporated river rehabilitation. This is almost certainly due to a lack of suitable project design and implementation. **Much more data needs to be collated on the outcomes of this type of rehabilitation project by means of suitable monitoring and evaluation through use of benchmarking and endpoints**. In most instances FRM is applied to rivers in urban areas to protect industry and housing from flooding. It is subsequently important to understand that urban rivers are complex systems constrained by a number of existing pressures in addition to FRM measures in progress. It is therefore suggested that the benchmarks used are practical, for instance they need to be from a similar urban section of river where the removal of those pressures acting on the system are restricted; using a non-urban reference condition will not produce successful results. **Furthermore, collaboration between FRM and conservation specialists is necessary to achieve a win-win scenario and to endeavour to integrate these two conflicting directives**. Regular meetings and workshops with both the flood risk and conservation management teams should bridge the gap between the different outlooks on projects.

River rehabilitation programme goals often only address problems on single rivers at a small scale and therefore have limited impact on catchment-scale processes. Fortunately potential benefits of implementing river rehabilitation and conservation at a catchment-scale are being increasingly recognised as an essential component of future rehabilitation practise, especially through the WFD. Catchment scale management is now being applied in the UK through RBMPs, catchment pilot projects that are additionally supported by wider initiatives in other catchments, and the CRF. To keep moving forward with catchment management there is a need to increase knowledge and therefore, **there is a need for detailed, long-term monitoring and evaluation of rehabilitation projects applied within large scale rehabilitation catchment programmes.** The findings can then be applied to other catchment management programmes and support decision making towards the 2^{nd} round of RBMP in 2015. **The next stage is to therefore bridge the gap between scientists, practitioners and stakeholders by transferring information on new findings through workshops and in the near future produce an up to date guide for effective river rehabilitation at a catchment scale.**

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APPENDIX 1. WFD HYDROMORPHOLOGY MITIGATION MEASURES: GROUPED BY PRESSURE (ROYAL HASKONING)

Pressure	Sector	Specific Measure
Working with physical form and function	Water resources, Agriculture, Industry and Infrastructure	Adopt strategic options and policies promoting natural recovery Use of engineering techniques to assist natural recovery Introduce minimal flow limits Introduce compensatory flows (not just at low flow levels) Regulate abstraction and discharge Reduce need for abstraction Water efficiency planning (domestic, business, industry, agriculture) Update policy and process guidance to take account of hydromorphology Improve understanding of responses to hydromorphological pressures Trial existing mitigation measures Develop and trial new mitigation measures Hydrological monitoring Morphological monitoring Hydrological appraisal Education on use of guidance Education on identifying opportunities for delivering mitigation measures Educate landowners on sensitive farming management practises Education and awareness raising of impacts on navigation
Bank reinforcement	Agriculture, FRM, Industry and Infrastructure, Navigation	Removal of hard engineering structure (e.g. naturalisation) Managed retreat Narrow over-wide channels Create low flow channels in over-widened/over-deepened channels Reconnect and restore historic aquatic habitats Bank re-profiling (rehabilitation) Adopt strategic options and policies promoting natural recovery Replace existing structures with new structural designs to minimise impact Replace hard defence with soft engineering Use soft engineering techniques Create compensation habitats Creation of shallow margin in front of hard defence Update policy and process guidance to take account of hydromorphology Limit further development of the bank zone Improve understanding of responses to hydromorphological pressures Trial existing mitigation measures Develop and trail new mitigation measures Morphological monitoring Morphological appraisal
Channel	FRM,	Removal of hard engineering structures (e.g.

alteration	Industry and Infrastructure, Navigation	naturalisation) Recreate a sinuous river channel (re-meandering) Narrow over-wide channels Create low flow channels in over-widened/over-deepened channels Reconnect and restore historic aquatic habitats Recreation of gravel bars and riffles using permanent and/or temporary bed structures (increase morphological diversity) River bed raising or lowering (regarding) Adopt strategic options and policies promoting natural recovery Use of engineering techniques to assist natural recovery Replace existing structures with new structural design to minimise impact Replace hard defence with soft engineering Modify existing structures Use soft engineering techniques Cessation of maintenance Strategic placement of dredging material (e.g. creation of shallow water zones or gravel bars) Create compensation habitats Update policy and process guidance to take account of Hydromorphology Improve understanding of responses to hydromorphological pressures Trial existing mitigation measures Develop and trail new mitigation measures Hydrological monitoring Hydrological appraisal Morphological appraisal Education on identifying opportunities for delivering mitigation measures Educate landowners on sensitive management practises
Flood embankments	Agriculture, FRM	Managed realignments of flood defences Adopt strategic options and polices promoting natural recovery Use of engineering techniques to assist natural recovery Construct breach or spillways Education on use of guidance Education on identifying opportunities for delivering mitigation measures
Floodplain development	Planning, Industry and Infrastructure, Urban Development	Managed realignment of flood defences Adopt strategic options and policies promoting natural recovery Use of engineering techniques to assist natural recovery Implementation of SUDS Introduce riparian vegetation Limit further development of the bank near shore zone Avoid or limit development in the flood plain Education on use of guidance

		Education on identifying opportunities for delivering mitigation measures
Flow regulation	Water Resources, Agriculture, FRM, Industry and Infrastructure, Navigation	Removal of hard engineering structures (e.g. naturalisation) Adopt strategic options and policies promoting natural recovery Use of engineering techniques to assist natural recovery Replacing existing structures with new structural design to minimise impact hydromorphological impact Replace hard defences with soft engineering Modify existing structures Construct breach or spillways Reinstate natural outfall level Install fish pass Use soft engineering techniques Change operational regime of weirs and locks Update policy and process guidance to take account of hydromorphology Regulate of in-channel structures Improve understanding of responses to hydromorphological pressures Develop and trail new mitigation measures Hydrological monitoring Morphological monitoring Hydrological appraisal Morphological appraisal Education on use of guidance Education on identifying opportunities for delivering mitigation measures
In-Channel Structures	Water Resources, Agriculture, FRM, Industry and Infrastructure, Navigation, Urban Development	Removal of hard engineering structures (e.g. naturalisation) Recreate a sinuous river channel (re-meandering) Narrow over-wide channels Create low flow channels in over-widened/over-deepened channels Reconnect and restore historic aquatic habitats Recreation of gravel bars and riffles using permanent and/or temporary bed structures (increase morphological diversity) Adopt strategic options and policies promoting natural recovery Use of engineering techniques to assist natural recovery Replace existing structures with new structural design to minimise impact Reinstate natural outfall level Install fish pass Use soft engineering techniques Regulation of in channel structures Cessation of maintenance Create reed fringes Create compensation habitat Creation of shallow margin in front of hard defence Update policy and process guidance to take account of

		hydromorphology Improve understanding of responses to hydromorphological pressures Hydrological monitoring Morphological monitoring Hydrological appraisal Morphological appraisal Education on use of guidance Education on identifying opportunities for delivering mitigation measures
Landuse management practices	Agriculture, Planning, Urban Development	Adopt strategic options and policies promoting natural recovery Removal of stock Reduce stock densities Reduce grazing time Introduce stock proof fences Improve river crossing for livestock Establish relocate feed and water troughs to reduce erosion Cultivate land for crop establishment in spring rather than autumn Adopt minimal cultivation systems Cultivate and drill across slope Leave autumn seed bed rough Loosen compacted soil layers' Establish in-field sediment buffer strips Cease maintenance of field drainage systems Re-site gateways away from high-risk areas Implementation of SUDS Establish and maintain artificial (constructed) wetlands for use as sediment traps Update policy and process guidance to take account of hydromorphology Educate landowners on sensitive management practices Hydrological monitoring Morphological monitoring Hydrological appraisal Morphological appraisal Education on use of guidance Education on identifying opportunities for delivering mitigation measures
Navigation	Navigation	Removal of hard engineering structures (e.g. naturalisation) Adopt strategic options and policies promoting natural recovery Use of engineering techniques to assist natural recovery Replace existing structure with new structural designs to minimise impact hydromorphological impact Use soft engineering techniques Modify existing structures Replace hard defences with soft engineering Develop/review appropriate dredging strategy Develop/review appropriate vegetation management

		plans Retain marginal vegetation Change operational regime of weirs and locks Create reed fringes Create compensation habitats Creation of shallow margins in front of hard defence Update policy and process guidance to take account of hydromorphology Regulation of in-channel structures Encourage reduction of boat wash impacts through traffic management sensitive areas Limit number of mooring permits available Restrict speed Lateral zoning to concentrate boats within central channel Avoid or prevent mooring in sensitive areas Designing mooring for ecological benefits Encourage use of environmentally friendly vessel design Develop and trial new mitigation measures Hydrological monitoring Hydrological appraisal Education on use of guidance Education on identifying opportunities for delivering mitigation measures Education and awareness raising impacts of navigation
Reclamation	Agriculture, Urban Development	Replenishment of mobile sediments Adopt strategic options and policies promoting natural recovery Use of engineering techniques to assist natural recovery Strategic placement of dredged material Create reed fringes Update policy and process guidance to take account of hydromorphology Limit further development of the bank zone Hydrological monitoring Morphological monitoring Hydrological appraisal Morphological appraisal Education on use of guidance Education on identifying opportunities for delivery mitigation measures
Sediment management	Agriculture, FRM, Coastal Defence	River bed raising or lowering Replenish mobile sediments Adopt strategic options and policies promoting natural recovery Use of engineering techniques to assist natural recovery Install silt and gravel traps Create reed fringes Update policy and process of guidance to take account of hydromorphology Improve understanding or responses to hydromorphological pressures

		Develop and trial new mitigation measures Hydrological monitoring Hydrological appraisal Education on guidance of tools Education on identifying opportunities for delivering mitigation measures Education and awareness raising of impacts of navigation
Vegetation management	Agriculture, FRM, Navigation	Adopt strategic options and policies promoting natural recovery Use of engineering techniques to assist natural recovery Develop/review appropriate vegetation management plans Change techniques to mage and minimise disturbance to hydromorphology Control or eradicate invasive species Introduce riparian vegetation Create compensation habitats Update policy and process guidance to take account of hydromorphology Improve understanding of responses to hydromorphological pressures Education on use of guidance Educate landowners on sensitive management practises Education and awareness raising of impact of navigation

APPENDIX 2 – EFI+ DATABASE INFORMATION

Site Code

Code given to each sampling site by user (could be country abbreviation + users own code of the site, e.g. DE0001).

Type: String, 15 positions, first two letters always capital.

Longitude

Longitude in decimal degrees, projection WGS 84.

Type: Numeric.

Latitude

Latitude in decimal degrees, projection WGS 84.

Type: Numeric.

Day

Type: Numeric values 1-31.

Month

Type: Numeric values 1-12.

Year

Type Numeric.

Country

Name of country (should be in English).

River Name

National name of the river (for transboundary, small rivers, the name of country where it confluences).

Type: String of chars.

Site Name

Location name e.g. indicating a nearby town or village.

Type: String of chars.

Altitude

The altitude of the site in metres above average sea level.

Type: Numeric.

Ecoregions

Ecoregions according to Illies.

id	Value	id	Value	id	Value
1	Iberian Peninsula	2	Pyrenees	3	Italy
4	Alps	8	Western Highlands	9	Central Highlands
10	The Carpathians	11	Hungarian Lowlands	12	Pontic Province
13	Western Plains	14	Central Plains	15	Baltic Province
16	Eastern Plains	18	Great Britain	20	Borealic Uplands
22	Fenno-Scandian Shield				

Mediterranean type

id	Value	id	Value
1	No	2	Yes

River Region

To define the River Region use the table below.

id	Value	id	Value	id	Value
1	Adriatic Sea (continental coast)	2	Adriatic Sea (peninsular coast)	3	Baltic Sea (continental coast)
4	Baltic Sea (peninsular coast)	5	Bay of Biscay (French coast)	6	Bay of Biscay (Spanish coast)
7	Bristol Channel	8	Danube	9	Douro
10	Ebro	11	Elbe	12	English Channel (continental coast)
13	English Channel (insular coast)	14	Garonne	15	Great Ouse
16	Guadalquivir	17	Guadiana	18	Gulf of Finland
19	Gulf of Riga	20	Irish Sea	21	Kattegat
22	Loire	23	Mediterranean Sea (French coast)	24	Mediterranean Sea (Spanish coast)

25	Medway	26	Mersey	27	Meuse
28	Nemunas	29	North Atlantic Ocean	30	North Sea (continental coast)
31	North Sea (insular coast)	32	Odra	33	Rhine
34	Rhone	35	Seine	36	Severn
37	Skagerrak	38	Tagus	39	Tees
40	Thames	41	Trent	42	Tyne
43	Tyrrhenian Sea	44	Wear	45	Weser
46	Wisla	47	Yorkshire Ouse		

Method

Definition, how electric fishing was carried out.

id	Value	id	Value	id	Value	id	Value
1	NoData	2	Boat	3	Wading	4	Mixed

Fished Area

Area of the section that has been sampled (sampled length * sampled width) given **in m²**.

Type: Numeric.

Wetted Width

Wetted width **in metres** is normally calculated as the average of several transects across the stream.
The wetted width is measured during fish sampling (performed mainly in autumn during low flow conditions).

Type: Numeric.

Flow Regime

id	Value	id	Value	id	Value	id	Value	id	Value
1	Permanent	2	Summer dry	3	Winter dry	4	Intermittent	5	NoData

Natural Lake Upstream

Normal flow pattern for the river. Divided into four classes:

id	Value	id	Value	id	Value
1	NoData	2	Yes	3	No

Geomorphology

Information in 5 categories to be selected:

id	Value	id	Value	id	Value	id	Value	id	Value	id	Value
1	Naturally constraint no mob	2	Braided	3	Sinuous	4	Meand regular	5	Meand tortous	6	NoData

Flood Plain

If the river has a former floodplain: Proportion of connected floodplain still remaining, in the following categories:

id	Value	id	Value	id	Value
1	Yes	2	No	3	NoData

Water Source

The source of the river water, assigned to the following three classes:

id	Value	id	Value	id	Value	id	Value	id	Value
1	NoData	2	Glacial	3	Nival	4	Pluvial	5	Groundwater

Upstream Drainage Area

Drainage area upstream of the site in km².

Type: Numeric.

Distance From Source

Distance from source in kilometres to the sampling site measured along the river. In the case of multiple sources, measurement shall be made to the most distant upstream source:
Data source: maps, preferably 1:25 000.

Type: Numeric.

River Slope

Slope of streambed along stream expressed **as per mill (m/km).** The slope is the drop of altitude divided by stream segment length.
The stream segment should be as close as possible to 1 km for small streams, 5

km for intermediate streams and 10 km for large streams. Data source: maps with scale 1:50 000 or 1:100 000.

Type: Numeric.

Air Temperature Mean

Average annual air temperature measured for at least 10 years. Given **in degrees Celsius (°C).** Data source: nearby measuring site, interpolated data.

Type: Numeric.

Air Temperature Mean January

Average January air temperature, given **in degrees Celsius (°C).** Data source: nearby measuring site, interpolated data.

Type: Numeric.

Air Temperature Mean July

Average July air temperature, given **in degrees Celsius (°C).** Data source: nearby measuring site, interpolated data.

Type: Numeric.

Sediment Size

Naturally dominant sediment information in the following categories:

id	Value	id	Value	id	Value	id	Value	id	Value	id	Value
1	Organic	2	Silt	3	Sand	4	Gravel/Pebble/Cobble	5	Boulder/Rock	6	NoData

Sampling Location

Where the sampling site is situated in relation to the river, in the following categories:

id	Value	id	Value	id	Value	id	Value
1	Main channel	2	Backwaters	3	Mixed	4	NoData

Species Name

Scientific name of species, according to the following list:

id	Value	id	Value	id	Value

1	Abramis ballerus	2	Abramis bjoerkna	3	Abramis brama
4	Abramis sapa	5	Achondrostoma arcasii	6	Achondrostoma occidentale
7	Achondrostoma oligolepis	8	Acipenser baeri	9	Acipenser gueldenstaedtii
10	Acipenser naccarii	11	Acipenser nudiventris	12	Acipenser oxyrinchus
13	Acipenser ruthenus	14	Acipenser stellatus	15	Acipenser sturio
16	Alburnoides bipunctatus	17	Alburnus albidus	18	Alburnus alburnus
19	Alburnus alburnus alborella	20	Alosa agone	21	Alosa alosa
22	Alosa fallax	23	Alosa macedonica	24	Alosa immaculata
25	Ameiurus melas	26	Ameiurus nebulosus	27	Ameiurus punctatus
28	Anaecypris hispanica	29	Anguilla anguilla	30	Aphanius fasciatus
31	Aphanius iberus	32	Aristichthys nobilis	33	Aspius aspius
34	Atherina boyeri	35	Atherina presbyter	36	Barbatula barbatula
37	Barbatula bureschi	38	Barbaus carpaticus	39	Barbus albanicus
40	Barbus barbus	41	Barbus bocagei	42	Barbus caninus
43	Barbus comizo	44	Barbus cyclolepis	45	Barbus euboicus
46	Barbus graecus	47	Barbus graellsii	48	Barbus guiraonis
49	Barbus haasi	50	Barbus meridionalis	51	Barbus microcephalus
52	Barbus peloponnesius	53	Barbus petenyi	54	Barbus plebejus
55	Barbus prespensis	56	Barbus sclateri	57	Barbus tyberinus
58	Benthophiloides brauneri	59	Benthophilus stellatus	60	Carassius auratus
61	Carassius carassius	62	Carassius gibelio	63	Chalcalburnus chalcoides

64	Chelon labrosus	65	Chondrostoma arrigonis	66	Chondrostoma genei
67	Chondrostoma miegii	68	Chondrostoma nasus	69	Chondrostoma soetta
70	Chondrostoma toxostoma	71	Chondrostoma turiense	72	Clarias gariepinus
73	Clupeonella cultriventris	74	Cobitis calderoni	75	Cobitis elongata
76	Cobitis elongatoides	77	Cobitis hellenica	78	Cobitis megaspila
79	Cobitis meridionalis	80	Cobitis paludica	81	Cobitis taenia
82	Cobitis vettonica	83	Coregonus albula	84	Coregonus autumnalis
85	Coregonus lavaretus	86	Coregonus maraena	87	Coregonus oxyrinchus
88	Coregonus peled	89	Coregonus pidschian	90	Cottus gobio
91	Cottus koshewniko	92	Cottus petiti	93	Cottus poecilopus
94	Ctenopharyngodon idella	95	Cyprinus carpio	96	Dicentrarchus labrax
97	Economidichthys pygmaeus	98	Economidichthys trichonis	99	Esox lucius
100	Eudontomyzon danfordi	101	Eudontomyzon mariae	102	Eudontomyzon vladykovi
103	Eupallasella perenurus	104	Fundulus heteroclitus	105	Gambusia affinis
106	Gambusia holbrooki	107	Gasterosteus aculeatus	108	Gasterosteus crenobiontus
109	Gasterosteus gymnurus	111	Gobio gobio	112	Gobio kesslerii
113	Gobio lozanoi	114	Gobio uranoscopus	115	Gymnocephalus baloni
116	Gymnocephalus cernuus	117	Gymnocephalus schraetser	118	Hemichromis fasciatus
119	Australoheros facetus	120	Hucho hucho	121	Huso huso

122	Hypophthalmichthys molitrix	123	Iberochondrostoma almacai	124	Iberochondrostoma lemmingii
125	Iberochondrostoma lusitanicum	126	Iberocypris palaciosi	127	Knipowitschia cameliae
128	Knipowitschia caucasica	129	Knipowitschia panizzae	130	Knipowitschia punctatissima
131	Knipowitschia thessala	132	Ladigesocypris ghigii	133	Lampetra fluviatilis
134	Lampetra planeri	135	Lepomis gibbosus	136	Lethenteron camtschaticum
137	Lethenteron zanandreai	138	Leucaspius delineatus	139	Leuciscus borysthenicus
140	Leuciscus cephalus	141	Leuciscus idus	142	Leuciscus keadicus
143	Leuciscus leuciscus	144	Leuciscus lucumonis	145	Leuciscus muticellus
146	Leuciscus pleurobipunctatus	147	Leuciscus souffia	148	Leuciscus svallize
149	Liza aurata	150	Liza ramada	151	Liza saliens
152	Lota lota	153	Micropterus salmoides	154	Misgurnus anguillicaudatus
155	Misgurnus fossilis	156	Mugil cephalus	157	Mylopharyngodon piceus
158	Neogobius fluviatilis	159	Neogobius gymnotrachelus	160	Neogobius kessleri
161	Neogobius melanostomus	162	Neogobius syrman	163	Oncorhynchus gorbuscha
164	Oncorhynchus kisutch	165	Oncorhynchus mykiss	166	Oncorhynchus tschawytscha
167	Oreochromis niloticus	168	Osmerus eperlanus	169	Pachychilon pictum
170	Padogobius bonelli	171	Padogobius martensii	172	Padogobius nigricans
173	Pelecus cultratus	174	Perca fluviatilis	175	Perccottus glenii

176	Petromyzon marinus	177	Phoxinus phoxinus	178	Pimephales promelas
179	Platichthys flesus	180	Pleuronectes platessa	181	Poecilia reticulata
182	Polyodon spathula	183	Pomatoschistus microps	184	Pomatoschistus minutus
185	Proterorhinus marmoratus	186	Pseudochondrostoma duriense	187	Pseudochondrostoma polylepis
188	Pseudochondrostoma willkommii	189	Pseudophoxinus beoticus	190	Pseudophoxinus stymphalicus
191	Pseudorasbora parva	192	Pungitius hellenicus	193	Pungitius pungitius
194	Rhodeus amarus	195	Romanichthys valsanicola	196	Romanogobio antipai
197	Romanogobio banaticus	198	Romanogobio belingi	199	Romanogobio vladykovi
200	Rutilus aula	201	Rutilus frisii	202	Rutilus heckelii
203	Rutilus pigus	204	Rutilus rubilio	205	Rutilus rutilus
206	Rutilus ylikiensis	207	Sabanejewia aurata	208	Sabanejewia balcanica
209	Sabanejewia bulgarica	210	Sabanejewia larvata	211	Sabanejewia romanica
212	Salaria fluviatilis	213	Salmo salar	214	Salmo trutta fario
215	Salmo trutta lacustris	216	Salmo trutta macrostigma	217	Salmo trutta trutta
218	Salmo trutta marmoratus	219	Salvelinus alpinus	220	Salvelinus fontinalis
221	Salvelinus namaycush	222	Salvelinus umbla	223	Sander lucioperca
224	Sander volgensis	225	Scardinius acarnanicus	226	Scardinius erythrophthalmus
227	Scardinius graecus	228	Scardinius racovitzai	229	Silurus aristotelis

230	Silurus glanis	231	Sparus aurata	232	Squalius alburnoides
233	Squalius aradensis	234	Squalius carolitertii	235	Squalius malacitanus
236	Squalius pyrenaicus	237	Squalius torgalensis	238	Syngnathus abaster
239	Syngnathus typhle	240	Thymallus thymallus	241	Tinca tinca
242	Triglopsis quadricornis	243	Tropidophoxinellus hellenicus	244	Tropidophoxinellus spartiaticus
245	Umbra krameri	246	Umbra pygmaea	247	Valencia hispanica
248	Valencia letourneuxi	249	Vimba vimba	250	Zingel asper
251	Zingel streber	252	Zingel zingel	253	Zosterisessor ophiocephalus

Total Number Run 1

All caught individuals (incl. 0+) of the species in run 1.

Type: Numeric.

Number Below 150 mm

Number of individuals with total length <=150mm for a given species for the first run of sampling.

Type: Numeric.

Number Over 150 mm

Number of individuals with total length > 150mm for a given species for the first run of sampling.

Type: Numeric.