THE UNIVERSITY OF HULL

The influence of flow management and habitat improvement works on fish communities in Yorkshire rivers

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by

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ABSTRACT

Many of the rivers in the UK are heavily modified by channelisation, impoundment (dams and weirs and off-river storages), land drainage and flood defence. These modifications have reduced the natural variability of flow and habitat diversity and in turn rivers are failing to meet Water Framework Directive (WFD: 2000/60/EEC) targets. Mitigation measures such as modifying reservoir flow releases and habitat improvement works are carried out to remediate the potential impacts of river development. This thesis examines the effectiveness of modified reservoir flow releases and habitat improvement works in Yorkshire rivers using brown trout (*Salmo trutta* L.) as the indicator of change.

The importance of natural flow regimes and how reservoirs and flood defence works have had negative impacts on fish populations was reviewed. The current UK guidance around managing reservoir releases and reducing flood risk was reviewed with regards to what measures are in place to mitigate their impacts and what biological responses are expected. One of the main conclusion was that to meet WFD targets, monitoring is required to investigate the effectiveness of activities aimed at improving rivers to inform management decisions and ensure activities are efficient and cost effective.

The long term effects of introducing seasonal compensation flows and a single freshet were examined by comparing differences in the hydrological regime and monitoring brown trout populations downstream of water storage reservoirs in Yorkshire. Hydrological parameters were not significantly different following the introduction of the revised reservoir release programme and brown trout populations were found to be variable throughout the years studied, and any changes in population characteristics could not be attributed to the new regime and further changes to the reservoir releases maybe required.

Manual radio tracking was used to obtain a detailed knowledge of the movements and distribution of adult brown trout downstream of two water storage reservoirs in Yorkshire following the introduction of single freshet releases (November 2012) to stimulate upstream migration. Brown trout occupied small home ranges and a single freshet release did not result in long distance upstream migration possibly because the releases were not performed at the appropriate time of year or the magnitude was inadequate to promote migration. The number of releases was increased to one each in the months of October, November, and December 2013 but still did not result in long distance upstream migration. It was suggested that the freshet releases which lasted only 8 hr, provided brown trout with little opportunity to move a reasonable distance. Further changes to the reservoir releases may be made to meet the flow profile recommended by UKTAG for autumn and winter flow elevations to support spawning migrations.

A monitoring programme was designed to detect changes in brown trout population following habitat improvement works. Baseline surveys carried out as part of this programme found brown trout to be present at low densities and exhibit slow growth rates, which was attributed to lack of suitable habitat, particularly spawning and juvenile riffle habitats, lack of deeper pooled areas for larger brown trout and lack of available cover. It was recommended any habitat improvement works should therefore improve flow, habitat and sediment issues.

A further study compared brown trout population and habitat parameters at Malin Bridge on the River Don pre and post flood defence and subsequent habitat improvement works, the latter designed to mitigate adverse effects of flood defence works. The flood defence works provided very little habitat diversity and cover for larger brown trout, instream channel features were added to improve habitat. Following the improvement works brown trout populations returned to densities and composition found prior to flood defence works, indicating impacts associated with flood defence works can be reduced when incorporating habitat improvement works into flood risk management.

1 GENERAL INTRODUCTION

Rivers are hugely important to humans and wildlife providing a range of ecosystem services (Cowx & Portocarrareo, 2011; TEEB, 2011 (Figure 1.1)). Rivers support wildlife including invertebrates, plants, birds and mammals, which all rely on water and have interconnecting complex relationships forming a river ecosystem. Fish depend on many different types of habitat within a river for refuge, spawning, feeding and nursery areas (Copp & Peňáz, 1988; Copp, 1989; Junk *et al*., 1989; Cowx & Welcomme, 1998; Aarts *et al*., 2004). Unfortunately, due to human pressures on rivers there are conflicting needs between the use of rivers for human purposes and the ecological requirements of plants and animals (Martin-Ortega *et al*., 2015).

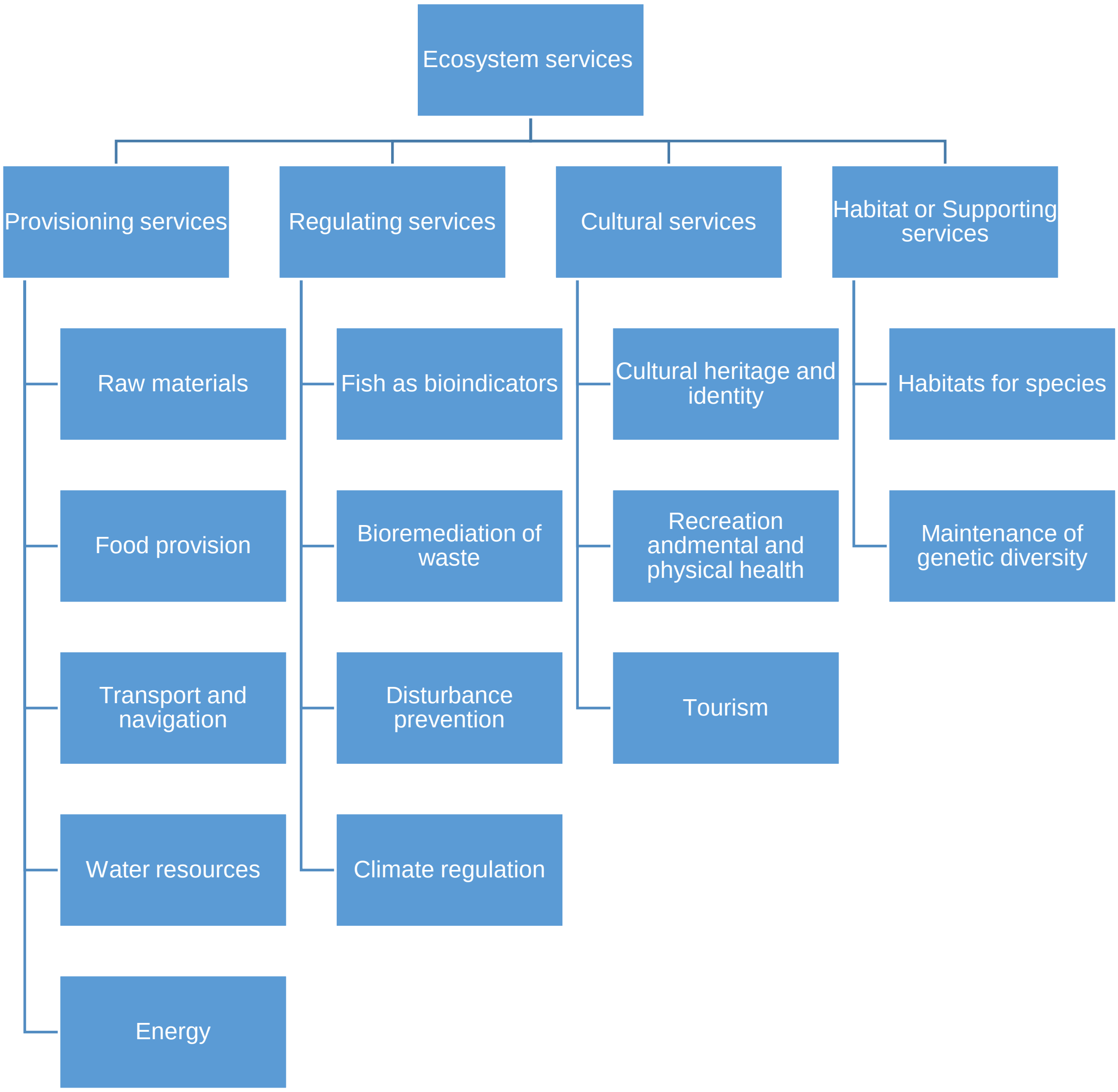


Figure 1.1. Ecosystem categories and types relevant to humans and freshwater fish conservation (Adapted from Cowx & Portocarrareo, 2011 and TEEB, 2011).

For more than 6000 years, humans have used and modified rivers (Nienhuis & Leuven, 2001), and settlements around rivers have been established to provide services including water, transportation of goods, food and power. However, some of these services can no longer be used due to the degraded state of the rivers in urban and industrial settings (Grimm *et al*., 2008; Everard & Moggridge, 2012). In 2007 the United Nations Population Fund (UNFPA) stated that half of the world's human population lived in urban areas, and this is expected to rise. There has been little consideration for the environment during urban development (Baron *et al*., 2002). Historical modifications include; channelization, impoundment (dams and weirs and off-river storages), land drainage, hydropower and flood defence (Dynesius & Nilson 1994; Bunn & Arthington, 2002; Aarts *et al*., 2004; Batalla *et al*., 2004; Murchie *et al*., 2008; Vaughn *et al*., 2009; Warner, 2012). Rivers have been canalised for navigation (Nienhuis & Leuven, 2001), to allow boats to import and export goods from settlements. Water has been controlled to provide energy for mills with the use of weirs and sluice gates (Nienhuis & Leuven, 2001). Reservoirs have been constructed for a variety of human purposes including flood control, irrigation diversions, sediment control, industrial supply, public supply, hydropower and recreational uses including fishing and boating (Brandt, 2000; Chapman, 1992). The hydrological regime downstream of these structures can be drastically changed and in the UK, around 90% of rivers are regulated affecting the natural flow regime (Acreman *et al*., 2009). Industries have been built on floodplain land, consequently destroying and fragmenting habitats, with waste from these industries polluting rivers (Nienhuis & Leuven, 2001). Flooding has been a persistent issue in the UK, which has led to management practices, that disconnecti rivers from floodplains, and reduce habitat quality and availability for fish (Junk & Wantzen, 2004). These modifications have reduced habitat availability for aquatic organisms, and restrict lateral and longitudinal migration of fish, which are important for spawning and refuge. As a result, a large number of fish species are threatened and fish productivity in most rivers has declined (Welcomme, 2001; Halls & Welcomme, 2004; Huckstorf *et al*., 2008). After many years of modifying rivers, rehabilitation and mitigation measures are required to alleviate the impacts created by humans with the aim to conserve biodiversity and ecosystem services (FAO, 2008).

Globally, attempts have been made to mitigate the hydrological impacts caused by reservoirs by modifying reservoir flow releases (Petts, 2009, Olden & Naiman 2010; Rolls *et al*., 2013). Modifying reservoir flow releases can vary depending on the reservoir's capability and use and understanding the implications of flow releases on downstream fish populations is vital for environmental flow management (Mims & Olden, 2012; Poff & Schmidt 2016). River rehabilitation through habitat improvement works can remediate the potential impacts as a result of river engineering, for example, flood defence works (Cowx & Welcomme, 1998). These methods are used in the UK and are driven by the Water Framework Directive (WFD: 2000/60/EEC), legislation responsible for sustainable water management across Europe that aims to improve the ecological status of aquatic ecosystems through restoration and/or maintenance (Acreman *et al*., 2009). Delivering river rehabilitation projects has grown in popularity over the years but very few have sufficient pre- and post-monitoring of fish populations to determine success of the works (Verdonschot *et al*., 2013). Yet this is important to help understand whether the types of works carried out have been effective and if any ecological improvement has been seen to meet WFD targets (Bernhardt & Palmer, 2011).

In Yorkshire, located in the Northern England, Yorkshire Water Services and the Environment Agency (EA) have been exploring a number of rehabilitation measures to mitigate the impacts created by reservoirs and flood defence works. In the Yorkshire Water region, rivers have been impounded for many decades, with the vast majority constructed between 1800 and 1950, to provide power to mills. Yorkshire Water Services currently operate more than 120 impounding reservoirs for water supply in a catchment area of approximately 11900 km^2. The introduction of the WFD has meant that water companies like Yorkshire Water Services have to invest considerable resources into research surrounding water management (Bowles *et al*., 2012). Understanding the effectiveness of theses management measures is important, not only to protect fish species and improve river habitats but to ensure that Yorkshire Waters resources are efficiently managed. Flooding in the UK has always been a feature of rivers and is considered one of the UK's most damaging and costly natural hazards (Brown & Damery, 2002; Stevens *et al*., 2014). The EA are responsible for managing flooding from rivers by carrying out river maintenance and river flood defences works to help reduce the risk of flooding. Since the introduction of the WFD, new approaches to flood risk management are being developed to be more sympathetic to natural process and aimed at having environmental, social, economic and flood alleviation benefits (Wharton & Gilvear, 2007). In Yorkshire, brown trout (*Salmo trutta*, L.) are a common species throughout the region, making them a strong model for biological monitoring and they are frequently used to assess the biological response to river rehabilitation techniques (Roni *et al*., 2005; Pont *et al*., 2006). Brown trout are more mobile than macroinvertebrates and other fish species such as bullhead (*Cottus gobio* (L.)), with the ability to move large distances making them an ideal species to study when investigating connectivity. Studying brown trout at these sites can be applied into a wider global context and the findings from this study can be transferred and applied.

The thesis is divided into two sections, the first two data Chapters (3 & 4) aim to investigate Yorkshire Waters current reservoir release regime and understand the effectiveness of these releases by monitoring long term brown trout population trends and individual brown trout behaviour. Knowledge gained would provide guidance and support to Yorkshire Water Services on future water management decision. The second section (Chapters 5 & 6) investigates habitat rehabilitation works in Yorkshire. Chapter 5 looks at rehabilitation as an alternative measure to environmental flow management and rehabilitation methods in Chapter 6 are carried out as a response to flood risk management work. These chapters aim to highlight the importance of habitat rehabilitation through monitoring brown trout population and the need for robust study designs.

Chapter 2 reviews the importance of the natural flow regimes for fish and associated impacts of flow modifications with reference to reservoir releases, flood defence works and how mitigation and rehabilitation measures can alleviate impacts.

Chapter 3 examine the long-term effects of modifying reservoir flow releases on brown trout downstream of water storage reservoirs, with specific reference to introducing seasonally variable compensation flows and freshet releases. The results will then be used to identify the effectiveness of flow modification to improve the status of brown trout populations downstream of reservoirs.

Chapter 4 examines movements of brown trout in response to freshet releases from two impounding water storage reservoirs to determine the appropriate flow building blocks required to encourage spawning migrations.

Chapter 5 provides a framework to monitor fish population change in response to habitat modification works as an alternative to modifying reservoir flow releases.

Chapter 6 compares brown trout populations and habitat parameters to pre and post flood defence and habitat improvement works in the rivers Lovely and Rivelin at Malin Bridge in Sheffield.

Chapter 7: Integrates the knowledge gained from Chapters 2 to 6 and provides recommendations highlighting the importance of monitoring and adaptive management.

2 IMPORTANCE OF THE NATURAL FLOW REGIME FOR FISH

2.1 INTRODUCTION

Fish communities and populations rely on a variety of habitats and processes within a river basin. Certain habitats are required throughout the various stages of a fishes life history (Cowx & Welcomme, 1998; Cowx *et al*., 2004). The river ecosystem is driven by many physical factors, including temperature, oxygen, light, concentration of suspended sediment, and flow/river discharge (Acremen *et al*., 2009). The natural flow regime is considered to be an important driver for river and floodplain wetland ecosystems (Bunn & Arthington, 2002; Enders *et al*., 2009). Maintaining the variability of the natural flow and considering all aspects (magnitude, frequency, duration, timing and predictability) of the regime is vital for rehabilitating and sustaining habitats for fish (Galat & Lipkin, 2000; Lytle & Poff, 2004). Providing suitable flow characteristics is essential for water resource management and contributes to improving water bodies to a Good Ecological Potential by 2027 (WFD: 2000/60/EEC).

This chapter provides a concise literature review of the importance of the natural flow regime for fish, associated impacts of flow modifications with reference to reservoir releases, flood alleviation and how mitigation and rehabilitation measures can alleviate impacts.

2.2 THE NATURAL FLOW REGIME

The natural flow regime is the term used to describe the range and variation of flow (Richter *et al*., 1996; Poff *et al*., 1997). Natural flow regimes vary globally, and are influenced by the climate (temperature and rainfall) and catchment run off (land use, topography, geology), and not affected by dams, weirs, abstraction and river management (Poff & Zimmerman, 2010). Long term variation in flow forms the physical habitat over a large area such as catchments and sub-catchments. Short term hydrological events influence small scale physical habitat in rivers and river reaches. Long and short term variations in hydrological events influence food and habitat availability for fish (Kennard *et al*., 2007). There are several elements to a natural flow regime include timing, predictability/continuity, rate of change, amplitude/magnitude (including extreme high and low flows), frequency and duration of flow events (Welcomme & Halls 2001; Enders, 2009). Figure 2.2 provides an example of a natural flow regime with a number (frequency) of high and low flows (magnitude), occurring at different times of the year (timing), each flow event occurring for different periods of time (duration), usually correlated to periods of high or low rainfall (predictability). Every river has a specific natural flow regime with an associated biotic community (Naiman *et al*., 2008); as a result fish have evolved to cope with the natural variation of flows, including spawning and feeding behaviour (Bunn & Arthington, 2002).

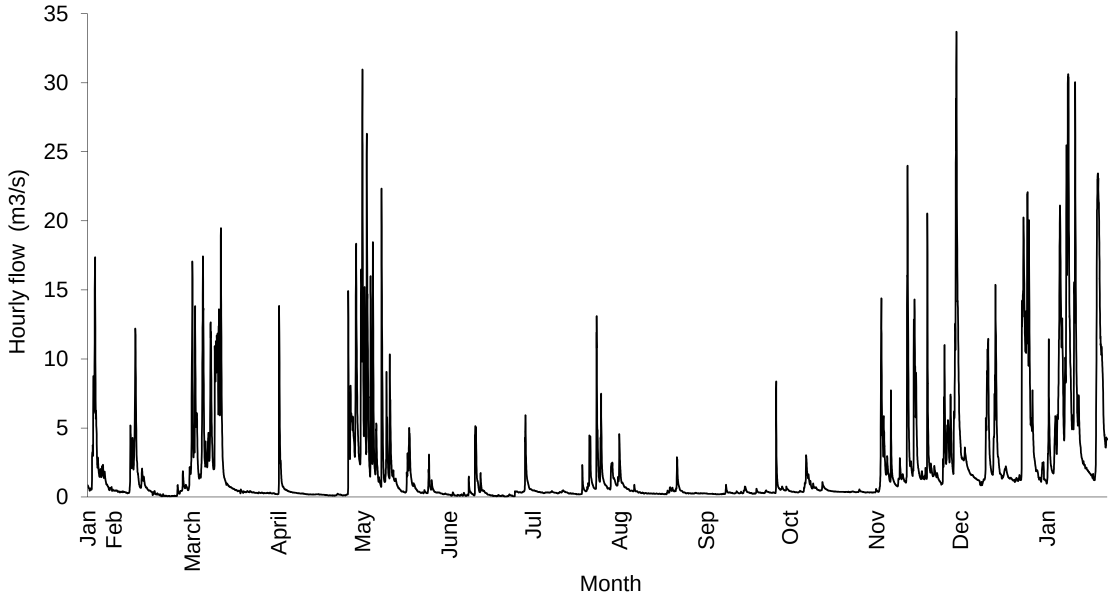


Figure 2.1. Example of a natural flow regime, demonstrating the variation of flow throughout the year (Keltneyburn, Scotland 2004).

2.2.1 Timing

River fish species rely on environmental cues for various aspects of their life stages including reproduction, survival and feeding (Humphries *et al*., 1999; Forsythe *et al*., 2012). Timing of flow events are important to coincide with the physiological readiness to spawn by fish; flow requirements are species specific with certain fish species relying on a particular flow event (Mann, 1996; Lucas & Baras, 2001; Nykänen *et al*., 2004). The importance in timing of high flow events can increase connectivity and habitat availability during spawning migrations, allowing fish to ascend natural and manmade barriers (Gibson *et al*., 2005; Bradley *et al*., 2012). Downstream spawning migrations are undertaken by tupong (Pseudaphritis urvillii) (Crook *et al*., 2010), Australian grayling (Prototroctes maraena) (Koster & Dawson, 2009) in response to within-channel flow peaks. Evidence of high water flows have been associated with stimulating salmonid smolt migration (Hvidsten *et al*., 1995; Aldvén *et al*., 2015). Aldvén *et al*. (2015) found that the peak of migration of brown trout and Atlantic salmon (*Salmo salar* L.) coincided with increases in discharge in the Himleån, a river on the west coast of Sweden. This highlights the importance of flow conditions and how they act as cues for migration and maturation. Some species have evolved to start the maturation of their spawn during their migration while others are able to postpone the maturation process until water levels begin to rise. In the Mekong, cyprinid species are sensitive to the timing of flood events, upstream migration from their feeding habitats to spawning sites occurs before or at the same time and the rising limb of a flood (Welcomme & Halls, 2001). River flow is an environmental variable influencing the timing of upstream migration of salmonids in rivers and small streams. For example, in South West Norway the number of ascending brown trout per day was correlated with mean monthly water discharge in late summer and early autumn (August–October) in the River Imsa and relatively high water flow stimulated the river ascent early in the season (Jonsson & Jonsson, 2002). Timing flow events can have negative effects on fish, for example during the incubation period, brown trout eggs have been reported to be washout by high flow events and, in extreme low flow, egg desiccation (Spina, 2001; Zorn & Nuhfer, 2007). Timing of emergence of brown trout larvae from gravels and hydrological conditions at the time can influence the year class strength, with this is referred to as the critical period (Lobón-Cerviá 2004; Daufresne *et al*., 2005). Swimming capabilities in newly emerged trout are very limited and during periods of high flow, brown trout become displaced and washed downstream (Daufresne *et al*., 2005). As brown trout develop and mature their repertoire of behavioural responses increases, making them less susceptible to sudden changes in flow (Ayllón *et al*., 2009).

2.2.2 Continuity/predictability

The continuity or the predictability of a flow event is the correlation between a flow event and the annual cycle. Some flow events are linked with other environmental indicators, for example if there was a period of heavy rainfall it would be predicted that there would be a period of high flow (Naiman *et al*., 2008). Life history composition of fish assemblages are associated with predictability of flow (Mims & Olden, 2012). When the predictability is altered such as a decrease in high flow events, this is correlated with a decrease in richness, diversity and abundance of fish species (Enders, 2009), in such events spawning opportunities could be potentially reduced in salmonids and there could be a higher risk of redds (a series of nets created by females (Jonsson & Jonsson, 2011)) becoming stranded (Johnson *et al*., 1994; Gibbins & Acornley, 2000).

2.2.3 Rapidity of change

The rapidity or rate of change can be referred to as flashiness; this term is used to describe how quickly a river responds from the time of rainfall within a catchment to the time of a flood or high flow event. The size of the catchment and the land uses influence the rapidity of change, for example a larger catchment will be less flashy than a smaller catchment and a catchment that is predominantly urban will be flashier than an agricultural area (Konrad & Booth, 2005). A greater rapidity of change could result in downstream displacement, more probably of larval and juvenile fish, due to their reduced development and swimming ability. The rapidity of flow decreasing after a high flow event could potentially leave fish stranded, increasing fish mortality (Jonsson & Jonsson, 2011).

2.2.4 Amplitude/ magnitude

The amplitude is the difference between lowest water level and the maximum water level during a flow event. Natural low-flow periods are important and provide the conditions (e.g. concentrated prey, warm temperatures) that stimulate spawning and support recruitment of many species of fish (Humphries *et al*., 1999; King *et al*., 2003). High flows events are important for maintaining habitats, flushing fine sediments from gravel, improving spawning substrate and survival rate of eggs and larval life stages related to living within the gravel substrate (e.g. brown trout alevins) (Robison, 2007; Jonsson & Jonsson, 2011), higher flows can increase connectivity allowing fish to ascend natural and manmade barriers (Acreman *et al*., 2009; Bradley *et al*., 2012). As mentioned previously in Section 2.2.1, higher flows are important for the migration and movement of fish.For example, Aldvén *et al*. (2015) found that the peak of migration of brown trout and salmon coincided with increases in discharge in the Himleån, a river on the west coast of Sweden. When the amplitude is extreme (higher or lower) compared to the norm it can have detrimental effects. For example, when flow events are higher than expected this could potentially wash out individual fish or affect behaviour and habitat use (Vehanen *et al*., 2000). Vehanen *et al*. (2000), during a controlled flume experiment, found an increased flow, increased displacement of brown trout especially after first exposer to high flow, there was a higher proportion of juvenile brown trout displaced during winter. The reason for this is that swimming capabilities are reduced during cold periods (Heggenes & Traaen, 1988). High flow events have been documented to flatten brown trout redds and cause an influx of fine sediment, reducing horizontal pumping flows and permeability of redds, ultimately affecting the water exchange (Greig *et al*., 2005; Schindler Wildhaber *et al*., 2014). An extreme decrease in amplitude could reduce spawning movements and even expose redds (Johnson *et al*., 1994; Gibbins & Acornley, 2000; Enders, 2009). In the study carried out by Gibbins & Acornley (2000), Atlantic salmon redds became stranded following a period of reduced flow, this has also been reported for other species such as brown trout (Spina, 2001; Zorn & Nuhfer, 2007).

2.2.5 Duration

Duration is the period of time associated with a specific flow event (Welcomme & Halls, 2001). An increased period of a flow event can provide more opportunities for feeding and spawning movements, increasing growth and survival (Welcomme, 1985). Equally periods of extended extreme flow events could have negative effects on fish populations.

All aspects of the natural flow regime are important, changes to the natural flow regime can alter the hydrological and geomorphological status in rivers and wetlands. These together form the physical environment and habitat that supports aquatic organisms, changes may prevent fish from reproducing and completing their life cycles. Consequences when modifying the natural flow components are expanded upon in Section 2.3.

2.3 MODIFICATION OF NATURAL FLOW REGIMES

The variation of a natural flow regime is an important factor in river ecosystem functioning, maintaining biological diversity (Chapter 2.2). Changes to the natural flow regime can reduce biological diversity and ecological functions in aquatic ecosystems (Poff *et al*., 1997; Robertson *et al*., 2001). Anthropogenic activities have altered flows in rivers and streams, managing these activities is essential for balancing human and wildlife needs for water, this becomes increasing important when factoring in the uncertainties of climate change (Gibson *et al*., 2005; Petts, 2009).

2.3.1 Reservoirs

Reservoirs are water bodies constructed or modified for a variety of human purposes including flood control, irrigation diversion, sediment control, industrial supply, public supply, hydropower and recreational uses including fishing and boating (Chapman, 1992; Brandt, 2000). Olden *et al*. (2014) provided one of the first studies assessing the global success of large scale flow experiments (FEs). Out of the 113 FEs studied, the primary purpose of dams were to provide power (>40%) and secondly water supply (30%); other purposes included flood control (<20%), recreation (<10%), navigation (<5%) and prevention of saltwater intrusion (<5%). As a result, the hydrological regimes downstream of these structures are drastically changed, and the natural flow variability is reduced. For example Petts (1984), reported that reservoirs can reduce seasonal flow variability, alter the timing of annual extreme flows and reduce the mean annual discharge by 80%. Fitzhugh & Vogel (2011) found a 25% reduction in the mean annual flood and Graf (2006), found that in some cases annual peak discharges can be reduced up to 90%.Adjusting the natural flow regime poses one of the greatest risks to the aquatic community; leading to changes in sediment deposition, nutrient loading, energy input and biota downstream of reservoirs (Ligon *et al*., 1995; Richter *et al*., 2003; Acreman *et al*., 2009; Vörösmarty *et al*., 2010). The ICUN (2013) reported that dam construction is one of the major causes for freshwater species extinction. Ligon *et al*., (1995) highlighted a case study where the construction of dams resulted in the reduction of peak flows on the River McKenzie. This resulted in the protection of farmlands and towns from flooding but the reduction in peak flows stabilised the channel and prevented the creation of mid-channel bars. Braided channels disappeared and areas of spawning gravels were lost which had a negative effect on chinook salmon (*Oncorhynchus tshawytscha* (W.) populations, with the average salmon population halving between 1969 and 1986. The regulated flow from Keilder Reservoir to the Tyne and North Tyne reduced the availability of suitable spawning habitat of Atlantic salmon by approximately one third to what it would be under optimal discharge (Gibbins & Acornley, 2000).

Temperature regimes have been influenced by the modification of flows. Olden and Naiman (2010), reported that the construction of Faming George Dam decreased late spring and summer (May – August) temperatures 17^{o}C to 5.7^{o}C, whereas the winter temperatures (December – March) increased from 0.7^{o}C to 5.4^{o}C. Magnitude of minimum and maximum water temperatures for comparable durations to pre dam constructions increased and decreased respectively. Frequency and duration of low and high water temperatures were significantly lower.

Behavioural responses to contrasting hydrological regimes have been reported, Allyon *et al*. (2014), found that the brown trout behaviour differed between highly variable flow regimes and stable flow regimes. Brown trout that were from rivers with highly variable flow and more frequent, longer duration and higher magnitude flow events were more likely to hold positions in high velocity habitats, whereas brown trout found in more stable environments were more likely to selected covered habitats, reducing biological interactions.

Reservoirs regulate and slow the flow in rivers downstream of them, have shifted the fish community composition from lotic species that are associated with high flow environments to lentic species that occupy low flow environments, this change of taxanomic groups puts endemic species at risk of extinction (Poff *et al*., 1997; Marion *et al*., 2012).

2.3.2 Flood Defence

Floods are recognised as high flow events that exceed bank full level and are an important part of the hydrological regime and provide many ecological benefits (Bayley 1995; Poff *et al*., 1997). Floods maintain ecosystem processes, for example, transportation and cycling of nutrients, sediment and organisms, and create (riffle and pool formation) and maintain habitats and channel structure (Poff *et al*., 1997; Williams *et al*., 2015). Floods are important for maintaining wetlands, Robertson *et al*. (2001), reported that spring flooding was vitally important for wetland macrophytes, to maintain species richness. Flooding acts as a trigger to enable fish to spawn in floodplain habitats, Górski *et al*. (2010), found that rheophilic species were timing spawning in response to floods and releasing eggs in the floodplain.

Historically western European rivers floodplain forests dominated the floodplains, in upper and lower reaches. Raised bogs were frequently in the upper reaches and peat-bogs, freshwater and brackish marshes in the lower reaches (Nienhuis & Leuven, 2001). The whole catchment was able to store a large amount of water and even in periods of prolonged rainfall the release of water was slow. Deforestation, land cultivation, river regulation, urbanisation, increase in human population has led to a dramatic change in fluctuations of water level. A combination of changes in land use and the perception that climate change is increasing the frequency of flood events around the world (Nienhuis & Leuven, 2001, Palmer *et al*., 2009).

Despite many environmental benefits, flooding can been seen as a disadvantage, especially in urbanised areas, causing a vast amount of damage to humans' and their livelihoods (Walsh *et al*., 2005) (Chapter 6.1). Flooding can cause damage to private homes, agricultural, commercial and industrial stocks and facilities, infrastructure (e.g. roads, railways, bridges and ports, energy and water supply lines, and telecommunications), public facilities (e.g. hospitals, schools) and natural resources and the environment. Other indirect impact arise due to the disruption of goods and services (and therefore economic activity) (Kundzewicz *et al*., 2014).

Due to the potential detrimental impacts of flooding many modifications have been made to rivers to prevent flooding which have had long term negative change on the riverine environment (Sparks *et al*., 1998). Impoundment, channel realignment and instream engineering works for flood defence purposes can alter the depth, velocity, substrate, flow and flow variation (Petts, 1984; Brookes, 1988). Many rivers have been channelized for navigation purposes and regulated by weirs and sluices for water resource control and flood defence causing habitat fragmentation (Nienhuis & Leuven, 2001). Canalisation, involves straightening and widening of a rivers course. Canalised rivers can support fish communities and have even been found to support similar species diversity, richness, abundance and biomass of fish (Stammler *et al*., 2008).

Despite this flood defence works restrict fish to the main channel, preventing lateral movements into very important habitats that act as refuge from high flows, spawning and nursery habitats for larval fish (Cowx & Welcomme, 1998; Bolland *et al*., 2008). Canalised rivers are uniform and they lack habitat diversity and fail to support various life stages of fish, Millidine *et al*., (2012), found that although the canalised rivers studied supported fish fry populations, the low flows during winter and lack of coarser substrate made conditions unfavourable for brown trout and salmon parr. Following the growing concerns about the effects of engineering works on the health of the rivers in England, flood management approach shifted from flood defence to flood risk management in the 1990 and 2000s due to the increase of flood events causing significant economic damage especially in the summer of 2007 and winter floods of 2013/2014 in England (Krieger, 2013; Thorne, 2014).

Table 1.1. Impact of reservoirs on downstream ecosystems (modified from Richter *et al*., 1998 and Acreman *et al.,* 2000).

First order impacts	Immediate abiotic effects that occur simultaneously with dam closure and influence the transfer of energy, and material, into and within the downstream river and connected ecosystems
1. Hydrology 2. Water quality 3. Sediment load	• reduced flow due to increased evaporation from the reservoir and/or abstractions • reduced variability of flow except where operational procedures result in fluctuations that occur at non-natural rates (e.g. pulse releases for hydropower production) • reduced magnitude, duration and frequency of flooding • thermal regulation (Olden and Naiman, 2010) • salinisation due to increased evaporation • alteration of dissolved oxygen and nitrogen content • alteration of pH • alteration of nutrient, hydrogen sulphide, manganese and iron concentrations (controlled largely by whether or not the reservoir is stratified and level of release) • reduction in load because of sedimentation within the reservoir • changes to river turbidity
Second order impacts	abiotic and biotic changes in upstream and downstream ecosystem structure and primary production, which result from the modification of first order impacts by local conditions and depend upon the characteristics of the river prior to dam closure (e.g. changes in channel and floodplain morphology, changes in plankton, macrophytes and periphyton). These changes may take place over many years.
1. Geomorphological Changes to channels, Floodplains and deltas 2. Plankton 3. Attached algae 4. Aquatic macrophytes 5. Riparian vegetation	• degradation where reduced sediment load results in increased erosion • aggradation where reduced flows result in increased sedimentation • populations increase where dams mitigate floods, regulate temperatures, reduce turbidity and reduce downstream effluent dilution • populations increase where dams enhance low flows, reduce flood magnitude and frequency, reduce turbidities, regulate thermal regime and reduce substrate erosion • rooted plant populations may increase where dams reduce flooding and substrate erosion and enhance deposition of fine nutrient-rich sediment • floating plant populations may increase where dams reduce high discharges so that channels are not flushed • species dependent on flood pulses (e.g. riparian forest trees) may be adversely affected as consequence of flood mitigation • reduction in silt deposition and nutrient replenishment on floodplains results in reduced soil fertility • Soil moisture for plants
Third order impacts	long-term, biotic, changes resulting from the integrated effect of all the first and second order changes, including the impact on species close to the top of the food chain (e.g. changes in invertebrate communities and fish, birds and mammals). Complex interactions may take place over many years before a new "ecological equilibrium" is achieved.
1. Invertebrates 2. Fish 3. Birds and mammals	• marked changes in macroinvertebrate distribution and abundance (often a decrease in diversity) occur as a consequence of changes in flow regime and physicochemical conditions (e.g. temperature, turbidity and dissolved oxygen). • marked changes in fish populations occur as a consequence of blockage of migration routes and changes in flow regime (specific spawning cues), physicochemical conditions (e.g. temperature, turbidity and dissolved oxygen), primary production and channel morphology. • changes in bird and mammal populations arise as a consequence of changes in floodplain habitat and fragmentation of the river corridor

2.4 CURRENT UK GUIDANCE OF FLOW REGULATION AND FLOOD RISK MANAGEMENT

Humans rely on rivers for many reasons (Figure 1.1), as a result these have caused many impacts upon the river systems for example, managed land for agriculture has increased in the last century due to population increase combined with the use of heavy machinery and toxic pesticides, impacts upon rivers as a result of changes to agricultural practices have led to increased levels of phosphates and increased levels of sedimentation, reducing water quality for aquatic life (Pretty *et al*., 2003). Urbanisation has resulted in many impacts on the rivers affecting the quality, quantity, discharge and thermal regimes, habitat availability lateral and longitudinal connectivity (Verdonschot *et al*., 2013).

In the UK, the Habitats Directive (HD: 92/43/EEC), the Water Framework Directive (WFD: 2000/60/EEC) and EU Floods Directive (FD: 2007/ 60/ EC) are three of the European directives that were created to set objectives for water protection for the future. The HD was adopted in 1992 with the aim to promote the maintenance of biodiversity; Member States were required to maintain/restore natural habitats and wild species to a favourable conservation status. Member States introduced robust protection for habitats and species of European importance, whilst taking into account economic, social and cultural requirements. The UK approach to the HD resulted in; Biodiversity Action Plans listing a number of priority species and habitats, for example brown/sea trout and chalk rivers.

The WFD is responsible for sustainable water management and aims to improve the chemical and ecological status of aquatic ecosystems through restoration and/or maintenance (Acreman *et al*., 2009). The WFD requires all Member States to establish River Basin Management Plans (RBMPs) and Programme of Measures (PoM), which set out measures to improve water in rivers, lakes, estuaries, coasts and in groundwater. All surface waters are expected to achieve at least a Good Ecological Status (GES) when possible. In some cases where a great deal of modification has occurred (e.g. Reservoirs), these are known has Heavily Modified Waterbodies (HMWB) and as a result are only expected to achieve a Good Ecological Potential (GEP). Ecological classification is defied by a number of quantifiable elements including biological elements (fish, invertebrates and macrophytes) and hydromorphological elements (channel morphology, channel planform and lateral connectivity) (Table 2.2). Water bodies subject to major hydrological impacts are usually have Poor or Bad statuses (Table 2.2), emphasising the importance of mitigation measures to improve river flows (UKTAG, 2014).The original aim was for these rivers to achieve GES or GEP by 2015, in 2009 the agreed target was to ensure at least 32% of waters in England were to achieve a GES or GEP, in 2013 25% of surface water bodies were at a GES/GEP or better (EA, 2013). It was apparent that it was going to take longer to attain the goal (2015) than originally anticipated, the revised deadline was extended to 2027. The extension allows for a longer period to significantly reduces the costs and/or provide more time for technically challenging objectives.

Table 2.2. Ecological classes for waterbodies in the Water Framework Directive (Taken and modified from Acreman & Ferguson, 2010).

Status Class	Description	Actions
High	Undisturbed, pristine waterbody, with natural flow regime, water quality and biology equal to reference conditions. Healthy stands of water plants, dominated by several different underwater species. Abundant plants at the water's edge, emerging from the water. Water clear, except in flood. Diverse common and rare insects, amphibians, reptiles, fish and birds	Maintain as high
Good	Water quality and biology deviate only slightly from reference conditions. Predominantly natural species. Healthy stands of water plants, dominated by several different species that lie in the water close to the surface. Abundant plants at the water's edge, emerging from the water. Water clear, except in flood. Common and occasional rare species of insects, amphibians, reptiles, fish and birds present	Maintain as good
Moderate	Moderate deviations in biology and water quality from reference conditions. Luxuriant growth of plants, but mainly of a single species; growth may almost block the channel. Emergent plant growth at the banks is present but not very extensive. Water can be turbid with a green or brown tinge, particularly during spring. Only widespread and common species of insects, amphibians, reptiles, fish and birds present. Some wetland species showing signs of stress. Some invasion by terrestrial species	Measures needed to improve status to good
Poor	Major deviation in biology from reference conditions. Significant pollution, a few sickly looking plants covered in green slime or with long trailing fronds of blanket weed (filamentous algae). Emergent plants at the water's edge either very sparse or absent. A few sickly looking plants within a dominant single species. Very turbid, green or brown coloured water for much of the summer. Few species of fish, invertebrates present	As above
Bad	Heavily polluted with very few or no natural animals present. No plants visible at all. Either bare bottom sediments or a covering of green or brown slime on the bottom. Very turbid, green or brown coloured water for most of the summer	As above

The FD was commissioned in 2006 with the aim to reduce and manage the risks that floods pose to human health, the environment, cultural heritage and the economy. All Member States were required to carry out an initial assessment by 2011 to identify the flood risk in each of the river basins and coastal areas that are at risk of flooding. Flood risk maps were to be drawn up by 2013 and establish flood risk management plans focusing on prevention, protection and preparedness by 2015. Flood risk management is an important part of the RBMPs, and the FD is coordinated by the WFD. Flood risk management considers; flood insurance; flood risk communication; environmental policies for example preserving wetlands and aims at “making space for water/rivers”, with the understanding that flooding cannot be fully prevented (Krieger, 2013). This shift in the way floods are managed aims to restore floodplains whilst reducing the conflict between flood defence and conservation. Flood and Water Management Act 2010 (UK Government, 2010), introduced a new investment system for England and Wales so that the cost of flood alleviation schemes were not funded by the government alone, this approach was more likely to enable more local choice and encourage innovative cost effective options where society plays a greater role. More recently the Water Act 2014 introduced safeguarding the movement of fish through regulated waters (Water Act, 2014).

The UK government must meet the requirements of WFD and part of this is by ensuring river flow/rivers are rehabilitated to improve populations. Currently in the UK a group of experts and specialist (UK Technical Advisory Group (UKTAG)) make technical recommendations to the UK government on implementing the WFD. Many UK rivers are heavily modified and are failing to meet a GEP, efforts are being made to mitigate and rehabilitate these rivers to improve the ecological status to meet WFD targets by 2027.

2.5 RIVER REHABILITATION AND MITIGATION MEASURES WITH SPECIFIC REFERENCE TO RESERVOIRS AND FLOOD DEFENCE

After many years of modifying rivers, rehabilitation and mitigation measures are required to elevate the impacts created by humans with the aim to conserve biodiversity and ecosystem services (FAO, 2008). Rehabilitation refers to the partial return to a pre-disturbance structure or function, river reaches are expected to see improvements or enhancements from its previous state (Wharton & Gilvear, 2007), rehabilitation often requires a single, sometimes major, expenditure, in many cases followed by smaller scale maintenance activities (FAO, 2008). Mitigation seeks to balance the impacts of an ongoing use of the river system, the use is usually deemed more valuable (socially and economically) than the fisheries and the fish, mitigation often involves an ongoing cost (FAO, 2008). Restoration is a commonly used phrase which implies a complete structural and functioning river back to its pre disturbed state (Wharton & Gilvear, 2007). The latter phrase is unrealistic, with reference to rivers that have undergone many years of river management it is highly unlikely these rivers will ever be restored fully (Wharton & Gilvear, 2007). There are many rehabilitation techniques and mitigation measures to improve habitats that have been degraded from human activities (FAO, 2008).

2.5.1 Reservoir mitigation measures

To address the problems generated by reservoirs, dam removal would appear to be the obvious answer, many issues involved with dam removal makes it difficult to do. Dam removal can be expensive and the ecological benefits are hard to predict (Wan *et al*., 2015), dramatic changes in the river hydraulics and channel morphology in turn affecting other ecosystem components (Doyle *et al*., 2003). Sediment accumulation can influence dam removal decisions, if the quality of the sediment is poor and potentially toxic, there is a risk that the contaminant in the sediment could be washed downstream once dam removal has taken place affecting human and the ecosystem health (Shuman, 1995). Enhanced movement of sediment and sediment aggregation could potentially burry organisms (Bushaw-Newton *et al*., 2002). Renöfält *et al*., (2013) found that dam removal in 2008 drastically declined invertebrate density from a mean value of 248 (2007) to 70 (2008) individuals per sample, samples taken in 2011 (three years on) showed a partial recovery of 198 individuals per sample, however taxonomic richness continued to decline following the dam removal.

When removal is not possible, rehabilitating a rivers natural flow can be facilitated by controlled dam releases and improve a number of processes (Roni *et al*., 2013). Using available water, water managers are challenged to meet a range of objectives and efforts have been made to simulate the natural flow regime by modifying the reservoir outflow, commonly referred to as ‘environmental flows’. Environmental flows are defined as ‘the quantity, timing, duration, frequency and quality of water flows required to sustain freshwater , estuarine and near shore ecosystems and the human livelihoods and well-being that depend of them’ (Acreman & Ferguson, 2010; Poff & Schmidt, 2016). WFD Environmental Flow Assessments (EFA) are used by scientists to inform and prioritise certain aspects of the natural flow regime that will most likely benefit downstream ecology (King *et al*., 2003, Richter *et al*., 2006). EFA indicate the specific temporal characteristics and the quantity of flow required to maintain the downstream river ecosystem and maintaining specific riverine features (Arthington *et al*., 1992; Tharme & King, 1998). The outputs from an EFA comprise of one or more potential modified hydrological regimes for the river, the environmental flow requirement(s) (EFRs) or environmental water allocation(s), each regime is linked to an objective with a goal for the future condition (Arthington *et al*., 2004).

Environmental flows are required to sustain rivers, efforts have been made to further understand the ways in which humans can use water resources while maintaining downstream ecosystems. Large scale flow experiments have become more common since the first published experiment in the 1960s, such experiments have been conducted around the world but more commonly in the United States, Australia and South Africa (Olden, 2014). There are a number of methodologies applied and have been classified into four categories by Tharme (1996, 2003); (1) hydrological, (2) hydraulic rating, (3) habitat simulation and (4) holistic methodologies (Arthington *et al*., 2004).

Hydrological: The most basic of methods using historical naturalised hydrological monthly or average daily flow data to recommend environmental flows, usually a proportion of flow or minimal flow for example Q_{95} - the flow equalled or exceeded 95 percent of the time (expanded upon in Chapter 4).

Hydraulic rating: This method measures changes in hydraulic variables (maximum depth or wetted perimeter) across a single flow-limited river cross-sections (riffles), environmental flows are calculated from plotting discharge against hydraulic variable(s). Curve breakpoints indicate at what point a decrease in discharge significantly reduces habitat quality.

Habitat simulation: This method expands on the above hydraulic habitat-discharge relationship and includes modelled analysis of quantity and suitability of the physical river habitat for the target biota. This more complex approach takes multiple measurements from a river reach using depth, velocity, substratum composition and cover data, and models the changes in physical microhabitat using hydraulic programs. The most well-known habitat simulation-modelling package is PHABSIM, incorporating the Instream Flow Incremental Methodology (IFIM).

Holistic: There are many holistic methodologies which have a broader approach to environmental flows. Sparks (1995) advised to estimate the natural flow regime that originally supported a range of species, rather than focusing on a few aspects of the flow regime to meet the requirements for only one or two species. Holistic methodologies consist of a range of components including, geomorphology, hydraulic habitat, water quality, riparian and aquatic vegetation, macroinvertebrates, fish and other vertebrates with some dependency upon the river/riparian ecosystem (i.e. amphibians, reptiles, birds, mammals). Examples of holistic methods include Building Block Methodology (BBM) (King *et al*., 2000), Expert Panel Assessment Method (EPAM) (Swales & Harris, 1995), Scientific Panel Assessment Method (SPAM) (Thoms *et al*. 1996; Cottingham *et al*., 2002), Benchmarking Methodology (Brizga *et al*., 2001) and Environmental Flow Management Plan Downstream Response to Imposed Flow Transformations (King *et al*., 2003).

In the UK, current flow guidance provided by the UKTAG is developed by taking recommendations from scientists, literature, similar case studies in other countries and uses a building block approach. The BBM is commonly referred to and implemented when designing release programs for reservoirs. The BBM comprises several methods, it is intended to guide, organise data and knowledge to provide the necessary output. The output is a modified flow regime that is applied to achieve a desired future condition for a particular river (Tharme & King, 1998). A holistic approach considering many aspects including social river resources, groundwater storage, hydrology, hydraulics, geomorphology, water quality, vegetation, aquatic invertebrates and fish. There are four main blocks part of the BBM (Figure 2.3); dry-season (minimum compensation) (Section 2.5.1.1) and wet-season flows (seasonal compensation) (Section 2.5.1.1), small pulses of higher flow (freshets) (Section 2.5.1.2) and wet-season floods (channel and habitat maintenance flows) (Section 2.5.1.3).

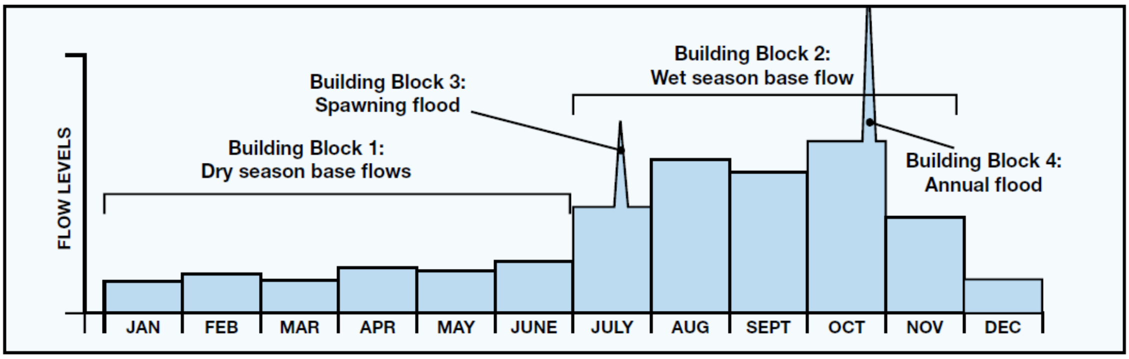


Figure 2.2. Flow building blocks, basic elements of flow variability that are considered to be ecologically important used in the Building Block Methodology (taken from O'Keeffe and Le Quesne, 2009).

2.5.1.1 Minimum compensation and seasonal compensation flows

In the UK, 70% of reservoirs release a minimum compensation flow; minimum compensation flows are a constant discharge release, typically set at 20% of the average daily flow or 95th percentile (Q_{95}) (Bradley *et al*., 2012) (Figure 2.3). Compensation flows were originally intended to provide a constant supply of water to mill owners, in 1963 the Water Resource Act, required River Authorities to set minimal acceptable flows to protect downstream river ecosystems (Dunbar *et al*., 2008). Minimum compensation flows, commonly referred to as base flows, were recognised for protecting fish species, allowing fish to able to migrate access a range of habitats during periods of low water level, and more crucially maintain a continuously wetted habitat to ensure rivers don't become dry (UKTAG, 2013). Despite the importance of minimum compensation flows, alone they do not meet the flow requirements for ecosystems downstream of reservoirs, mimicking natural flow variation is far more complex (Petts, 2009). Gibbins & Acornley (2000), reported the compensation flow from Kielder Reservoir only provided one third of the spawning habitat that would be available if optimum discharge was applied. Some reservoir release programs in the UK have varied seasonal flows, minimum compensation flows in the winter periods are increased to simulate naturally higher water levels in the wetter months (Acreman *et al*., 2009). Techniques were developed from using unregulated reaches to estimate seasonal variation in flow (NERC 1985), with the aim to improve the density and diversity of invertebrates and aid salmonid migration without extreme reductions in water resources (Ward & Stanford, 1987). Seasonal flows have also been found to be important for the dispersal, germination, growth and riparian community composition (Greet *et al*., 2011),

2.5.1.2 Freshes/ Freshets

Freshes or freshets are short periods of higher flow (Figure 2.3); spring freshets aim to support fish migration, either as from river to sea migrations for salmonid smolts, shad and sea lamprey or from sea to river migrations to spawning habitats (Acreman *et al*., 2009; Bradley *et al*., 2012). Summer freshets aim to flush away accumulated fine sediment which will help to clean spawning gravels for fish, flush plant debris and remove any plant or animal species that thrive under stable flow conditions. Freshets can provide greater opportunities for fish to pass partial barriers and access new habitats that would not have been able to be passed under lower flow conditions (Bradley *et al*., 2012).

2.5.1.3 Flood flows and/or channel/habitat maintenance flows

Channel and habitat maintenance flows are focused on influencing the physical structure of the stream, creating and maintaining stream habitat and morphology (Robison, 2007) (Figure 2.3). These higher flood flows scour the bed maintaining stream habitats such as pool and riffle habitats, clear gravels by flushing fines, which in turn aims to improve spawning and macro invertebrate habitat and in addition clearing riparian vegetation and moving large woody debris (Robison, 2007).

2.5.2 Biological responses to reservoir mitigation measures

Where mitigation measures of the natural flow regime have been carried out, significant ecological improvements have been found (Postel & Richter, 2003). For example freshets were released from Clanwilliam Dam at the appropriate time (October–January), freshets were released to increase spawning success of Clanwilliam yellowfish (*Barbus capensis* (Smith, 1841)), but the water temperature had to be a steady 19ºC to increase spawning success. Additional epilimnetic (the layer of water above the thermocline) releases were suggested to be released after the freshet releases to maintain the water temperature for successful embryo and larval development (King, 1998). Flow experiments have improved conditions for endangered species including redfin minnow (*Pseudobarbus asper* (Boulenger, 1911)) in South Africa (Cambray, 1991) and cui-ui sucker (*Chasmistes cujus* (Cope, 1883)) in the United States (Rood *et al*., 2005). Flushing flows were implemented in the first case study to decrease the salinity of pool habitats to initiate spawning and in the second case high spring flows were released to promote reproduction Rood *et al*., 2005. In some instances there are unexpected results when changes to the flow regime are made, for example flow pulses were introduced to promote spawning migration of short nose sturgeon (*Acipenser brevirostrum* (Lesueur,1818)) and other fishes in the Savannah River. Wrona *et al*., (2007) found that during radio telemetry monitoring, short nose sturgeon responded to a pulse in March but instead of moving upstream to spawning grounds they moved downstream. It was suggested that the water temperature from the flow pulse was colder than if it were to be a natural pulse event. It was suggested that future pulses should be timed to coincide with natural high flow events to minimize the temperature fluctuation. Thorstad *et al*., (2005) found that with the aid of freshet releases the number of weirs passed per hour was higher during two freshets and the distance moved was higher during a single freshet by Atlantic salmon in the River Mandalseva.

There have been case studies where the introduction of freshets have had no real influencing effect that could be detected, Reinfields *et al*., (2013) found that similar numbers of Australian bass (*Macquaria novemaculeata* (Steindachner,1866)) performed post-spawning upstream return migrations during flow pulses and during regulated base flow conditions. Bradford *et al*., (2011) found that flow experiments on Bridge River only increased salmon abundance where channels were previously dry, prior to the change in flows in channels where the baseline flows maintained a wetted area there was no change salmonid abundance. Thorstad *et al*., (2005) found that in the River Orkla only 4% of the Atlantic salmon migrated upstream during a freshet.

Unfortunately in many cases flow release programs are carried out but are never monitored, for example on the River Daleelva (Alfredsen *et al*., 2012). A building block type approach was developed for the River Daleelva, a heavily regulated river used for hydropower generation in western Norway. The aim was to develop a flexible three level (low, medium and high flow scenarios) environmental flow regime to meet the needs for key focus species the Atlantic salmon and how it could be applied to current Norwegian legislation (Alfredsen *et al*., 2012). Unfortunately this case study is one of many cases which have implemented flow rehabilitation methods but failed to monitor them as a result of funding and timing restraints. Sabaton *et al*., (2008) highlighted the number of experiments that were never published or possibly unpublishable, those studies that were carried out over short periods of time failed to provide enough spatial or temporal information on fish populations.

Internationally incorporating a range of flows, including daily monthly and seasonal flows, is ecologically valuable but sometimes difficult to implement, for example; it can be expensive especially if structural changes need to be made, there are few automatic flow gauges and manually operating sluices on a daily basis would be impractical and ultimately ensuring there is enough water to meet human requirements (e.g. irrigation, consumption and hydropower) (Stalnaker *et al*., 1996; Maddock *et al*., 2001; Dyson *et al*., 2003). Ways to overcome this issue include installing automatic sluices or install instream habitat improvement structures and carry out river rehabilitation methods (Maddock *et al*., 2001). In narrow channels it is suggested weirs, deflectors, boulders and large woody debris should be installed to make the most of low flows and increase velocity (Maddock *et al*., 2001; Whiteway *et al*., 2010).

2.5.3 Flood defence mitigation measures

To mitigate the effects of flood defence works, rehabilitation techniques can be carried out to improve instream channel habitats and processes, techniques include; installation of log weirs, boulder structures, channel re-profiling by remaindering rivers that were previously channelized and creation of floodplain habitats, further examples and intended habitat rehabilitation objectives can be found in Table 2.3 (Roni *et al*., 2013). Instream structures like log weirs and boulder clusters can change physical habitats, increasing pool frequency, pool depth, habitat complexity, spawning gravels, sediment and can diversify and increase localised flows (e.g. Negishi and Richardson 2003; Binns, 2004; Pierce *et al*., 2013). Techniques have to be selected carefully to ensure the risk of flooding is not increased.

Table 2.3. Common habitat rehabilitation techniques to improve instream channel habitats and processes (modified from Roni *et al*., 2013).

Objective	Structure/ technique	Considerations
Habitat improvement for fish including, pools, riffles, spawning habitat, cover and complexity.	• Log or boulder structures	• Size and orientation should be comparable natural wood and boulder accumulations. • Consider the durability or longevity of the structure • Maintenance • intensity of treatment (number and complexity of structures per kilometre)
	• Engineered logjams	
	• Plant vegetation or other structures to provide cover	• Maintenance to reduce flood risk and enable access for boats
	• Gravel addition (spawning habitat)	• Risk of gravels becoming clogged up with finer sediment
	• Remeandering	• Consider potential of the site
	• Excavation of new floodplain habitats	• Water quality and temperature of the newly created habitats
Improve water quality, hydrology and sediment load	• Remove roads or improve road surfaces by reducing impermeable surfaces, stabilising or resurfacing	• Complete road removal includes removing fill, reprofiling the slope, and replanting or seeding so the footprint of the road is no longer visible and the natural contours of the land are restored (Beechie *et al*., 2005) • Maintenance
	• Modify agriculture practices (e.g. Crop rotation, reduce crop row spacing, conservation tillage and buffer strips)	• Site specific and choose best practice to suit land owner
	• Increase instream flows and/or flood flows	• Understanding that the quantity and timing of flows is critical for sustaining key physical and ecological processes
	• Reconnect sediment sources	• Similar approaches to lateral connectivity
Improve longitudinal and lateral connectivity for fish migration and sediment transport	• Dam removal or partial removal	• Stream channel morphology • Hydrology • Sediment properties and transport • Habitat • Downstream biotic community • Cost • Time, drawdown of water level which could take years • Would require additional sediment removal, channel reconstruction and riparian planting

Table 2.3 (Continued). Common habitat rehabilitation techniques to improve instream channel habitats and processes (modified from Roni *et al.*, 2013).

Objective	Structure/ technique	Considerations
Improve longitudinal and lateral connectivity for fish migration and sediment transport	• Fish passage	Fish pass design should consider • Dam, weir, or barrier height and width • River flow timing, magnitude, and velocity • Bypass or ladder length, slope, substrate • Attraction flows • Species of interest • Maintenance
	• Levee setback or removal	• Consider other improvement works for a more rapid recovery of morphology.
	• Reconnection from main channel to floodplains	• Water quality from previous land use • Elevation differences between the mainstream and relict channels
Restore riparian zone, vegetation and processes, improve bank stability and instream conditions, increase or decrease shade	• Planting of trees and vegetation	• Lag time between treatment and changes in instream conditions • Increase flooding risk
	• Removing invasive species	• labour intensive • Using herbicides can harm other native plants and animals and could contaminate nearby streams
	• Thinning or removal of shrubs	• Reduce shading and cover for fish
	• Fencing and grazing reduction	

2.5.4 Biological response to habitat improvement works whilst considering flood risk

Responses to habitat improvement work have been varied and it has been speculated that many rehabilitation projects have not been published as a result of a lack of funding, limited manpower, difficulty with sampling and quantifying the biological response and problems isolating impacts caused by either habitat improvement works or natural variations (Baldigo *et al*., 2010).

In 2013, Lorenz *et al*. investigated 36 rehabilitation projects and found that removing bank and bed fixations, installing woody debris or re-establishing a more natural river profile by connecting the main channel to backwaters and improving longitudinal connectivity by removing weirs increased the abundance of adult and young of the year (0+) fishes (stone loach (*Barbatula barbatula* (L.)), gudgeon (*Gobio gobio* (L.)), grayling (*Thymallus thymallus* (L.)) and brown trout). Pierce *et al*. (2013) found that trout abundance increased as a result of several projects involving instream flow enhancement methods. It was speculated that where habitat improvement works have occurred, this created new habitats to forage in and as a result reduced competition; dominant fish vacated habitats and were re-occupied by subdominant fish in turn increasing fish abundance. In addition, it was found that manmade boulder structures attracted more brown trout, compared to streams that had not undergone habitat improvements (Shuler *et al*., 1994). Branco *et al*. (2013), found that boulders increased the weighted usable area for Iberian barbel (*Luciobarbus bocagei* (Steindachner, 1864). Baldigo *et al*. (2010) had mostly positive results following habitat improvement works (increase bank stability, reduce bed and bank erosion and maintain water quality), with community richness, diversity and total biomass increasing in four of the six reaches, whereas there was no positive improvement in fish assemblages in the other two reaches.

There are many rehabilitation techniques that have not been completely assessed and often not met desired ecological goals (Roni *et al*., 2013; Schwartz& Herricks, 2007). Pretty *et al*. (2003) found that installing artificial riffles and flow deflectors had little influence on improving fish abundance, species richness and diversity. Possible reasons highlighted included poor water quality, restricted access, the structures failed to improve physical habitat for fish and techniques used were inappropriate for location. Muotka & Syrjänen (2007) found that densities of 0+ trout were lower following instream works, although when comparing it to other unmodified reaches there was no significant difference in 0+ trout densities, highlighting the importance for reference reaches. Another stream in this study found a weak positive relationship between the instream works and trout density, it was concluded that there was a lack of suitable pools ideal wintering habitat. Steward *et al*., (2009) carried out a study looking at the effectiveness of instream channel structures on restoring salmonid populations, it was concluded that instream structures may attract fish but not restore fish populations.

The long-term impact of instream channel structures have appeared to be a temporary success when improving fish habitat, over the years and subject to high flow events the structures decay and with added channel adjustments have meant many structures over time have become ineffective (Thompson, 2002). Thompson (2002) highlighted that the most successful and stable structures overtime were log structures, including log dams and deflectors. Success of instream structures are dependent on structure type, materials and design. It is essential when choosing an appropriate rehabilitation technique that there are clear objectives, considering the desired timeframe and scale (Roni *et al*., 2008). For example connectivity, sediment and hydrology are mostly associated with rehabilitating watershed and reach scale processes and localized scale rehabilitation is associated with habitat improvement techniques (Roni *et al*., 2008). With many techniques occasional maintenance is required sometimes more often depending on which technique, this is necessary to ensure the benefits of the rehabilitation continue.

Rehabilitation techniques are site specific and can be more difficult to integrate into urban river systems. It is far more limiting, complex, and expensive to rehabilitate rivers in urbanized areas due to human infrastructure including buildings, roads and sewage lines (Bernhardt & Palmer, 2007). A real challenge, especially in urban areas, is to ensure that the water quality is sufficient to support aquatic life, as rehabilitation efforts would otherwise be meaningless if the water quality was poor (Pretty *et al*., 2003; Shurker *et al*., 2015). Rehabilitation in urbanised areas currently focuses on two main aspects; restoring channel form and channel stability. Another challenge when carrying out rehabilitation works in urbanised areas is considering flood risk.

In terms of river management flood control and nature conservation were in the past considered to conflict one another (Downs & Thorne, 2000), in the UK it is the role of the Environment Agency (EA) to ensure land drainage and flood defence requirements are met alongside conservation strategies. The aim when delivering rehabilitation projects is to provide improved physical habitats that are sustainable, flood defence that meets statutory standards, and legitimate needs of landowners and river users (Downs & Thorne, 2000). In urban areas where rehabilitation works are highly visible, Caruso & Downs (2007), found that individuals and organisations had contrasting views about what is aesthetically pleasing and what provides adequate ecological improvements with particular reference to trees. Throughout the project Caruso & Downs (2007), found that concerns around flood risk which meant higher levels of communication with locals, highlighting that urban river rehabilitation comprises of social, political and environmental factors.

Further research is required to understand the ecological, geomorphic and hydrologic aspects in urbanised areas and their response to rehabilitation techniques. This will inform decisions and management plans for the future.

2.6 STUDY SPECIES

Fish populations have a high economic and conservation value and are commonly used to monitor rehabilitation activities (Radford *et al*., 2004). Salmonids are in particular a common study species because of their response to a wide range of pressures (chemical pollution, flow regulation, physical habitat modification and habitat fragmentation) and a good ecological indicator (Roni *et al*., 2005; Pont *et al*., 2006). The main study species throughout thesis is brown trout, they are native to the UK, occupying a range of habitats including brooks, rivers, lakes, estuaries and coastal sea. They are a UK Biodiversity Action Plan (UKBAP) priority species (Malcolm *et al*., 2010) as they have one of the most diverse life histories of all fish (Jonsson, 1989). There are two forms of brown trout; anadromous and potamodromous. Anadromous brown trout more commonly known as sea trout; they hatch in freshwater and migrate to sea to feed and return to freshwater to spawn. Potamodromous brown trout or resident brown trout spend the whole of their lives in freshwater performing only small migrations to feeding, nursery and spawning areas (Jonsson & Jonsson, 2011). Anadromous sea trout and potamodromous resident brown trout are commonly regarded as constituting a single species (Campbell, 1977).

Both forms of brown trout perform spawning migrations from September through to as late as January typically associated in chalk streams when temperatures fall between 10 and 12 ˚c (Ovidio *et al*., 1998). Spawning migrations are associated with moderate and small flow increases, with a maximum ascent occurring between 7.5 and 10 m^3s^{-1}, elevated flows provide the opportunity to migrate, possibly to overcome barriers, movements have been found to occur more commonly at night to reduce predation risk, lower flows during this period could also leave brown trout at risk to predation and potentially delay spawning (Ovidio *et al.,* 1998; Heggenes *et al.,* 1999). Female brown trout excavate nests called redds to lay their eggs, this occurs in areas of gravel substrate (8 – 128 mm) (Armstrong *et al*., 2003), where the eggs are submerged by the gravel after laying. Larger females are able to move coarser gravels and spawn in areas where gravel substrate is much larger and these are usually buried much deeper than smaller females (Fleming, 1996). It is important that redds are laid in areas with fast flows to ensure that the incubating eggs receive good intra-gravel flows. Several males compete to gain access to a single nesting female, the dominant male occupies the closes position to the female and divides his time between courting and preventing other males approaching her (Jonsson & Jonsson, 2011).

Incubation of eggs and hatching occurs from October to April, maintaining water levels over redds (spawning nest) is critical to avoid high mortalities (Malcolm *et al.,* 2012). Within this time the embryos develop during the winter and alevins hatch in the following spring, alevins dwell in the gravels after hatching still carrying their yolk sack and approximately 1 month after alevins emerge from the gravels as fry. Fry emerge from the gravels between April and May, with their yolks sacks almost completely absorbed, they start to feed on epibenthic and drifting arthropods (Jonsson & Jonsson, 2011). Steady flows are required, high flows could displace fry due to the reduced swimming capabilities and low flows cold potentially restrict food supply (Acreman & Ferguson, 2010; Jonsson & Jonsson, 2011). For up to three years juvenile fish remain in the river, brown trout at this point will either migrate to sea as smolts in the spring or mature into resident brown trout. As juveniles develop they become less vulnerable to fluctuations in flow due to increased swimming abilities to withstand higher flow events, one exception is during periods of drought this can have a negative effect on resident brown trout (UKTAG, 2013). Resident brown trout are called parr for the first year and then become adult brown trout, they establish territories where they can feed on drifting invertebrates, seeking occasional shelter from predators under an undercut bank, rock or tree roots. Brown trout smolts mature to become sea trout and will spawn in rivers every year since maturation (Jonsson & Jonsson, 2011).

Resident brown trout are the primary study species throughout the thesis, because of their sensitivity to river flows and their frequently recognised as good indicators of river health (Acreman, 2001) and human impacts through many centuries have put pressure on this species. The current state of many of UK water bodies are not reaching a good ecological potential, it is essential that we try to further our understanding around the response of brown trout to mitigating and rehabilitation measures.

3. INFLUENCE OF MODIFIED RESERVOIR FLOW RELEASES ON BROWN TROUT POPULATIONS IN DOWNSTREAM RIVER REACHES

3.1 INTRODUCTION

The construction of dams to create reservoirs for human purposes (Chapter 2.3.1) has had detrimental effects on natural river processes and the aquatic life that depend on river systems for their life history, which include free flowing river habitats, obstruction of fish from migration and reduction in water quality in rivers downstream of reservoirs (Jager & Smith, 2008). Flow regimes have been seriously altered and typically have reduced flow variability coupled with a reduction in extreme high and low flow events (Petts, 1984, Nilsson *et al*., 2005, Marren *et al*., 2014). Flow characteristics downstream of reservoirs are influenced by reservoir features (e.g. spillway shape), reservoir function (e.g. hydropower, water supply) and inflow dynamics (Petts, 1984). Further influences on flow downstream of reservoirs include local climate and weather conditions, for example an increase in rainfall can rapidly fill reservoirs to storage capacity and cause excess water to spill over the reservoir outfall (Higgs & Petts, 1988) increasing flow in the receiving watercourse. The impact on the downstream ecology, with specific reference to fish, identified that reservoirs were a migration barrier and alterations in flow leads to declines in fish populations (Burt & Mundie, 1986; Dynesius & Nilsson, 1994, Wollebaek *et al*., 2011) (Chapter 2.3.1). As a result, operators worldwide have been obliged to maintain and manage water releases from reservoirs to attempt to alleviate problems downstream of the impounding reservoir (Acreman *et al*., 2009; Petts 2009; Olden & Naiman, 2010). Releases of water from reservoirs that are intended to mitigate the impacts are referred to as “environmental flows”. The most basic requirement is to provide rivers downstream of a reservoir with a minimal flow to supply enough water throughout the year to maintain river ecology; these can be referred to as compensation flows (Chapter 2.5.1.1).

Following the introduction of the European Water Framework Directive (WFD) in 2000, all members, including the UK, were required to improve the ecological status of rivers to a Good Ecological Status or for Heavily Modified Water Bodies (HMWB), such as impounded rivers, to achieve a Good Ecological Potential (GEP) by 2027 (Chapter 2.4). The UK government has a responsibility to meet the requirements set by the WFD, and therefore the Environment Agency (EA) and Yorkshire Water Services have been redesigning and experimenting with reservoir releases in an attempt to ensure river reaches downstream of reservoirs are able to meet GEP. The WFD 48 project (Acreman *et al*., 2006), focused upon setting flows to meet environmental requirements and concluded that the conventional mode of operating reservoirs, with a minimum compensation flow (Figure 3.1), would mean rivers would not achieve GES or GEP because natural variability of river flow is essential for a healthy river ecosystem (Figure 3.1).

Yorkshire Water Services have introduced seasonal compensation and freshet releases to several rivers across the Yorkshire region and these flows are intended to mimic natural flow variation. Seasonal compensation releases are designed to mimic periods of low and high flow, for example a constant lower flow is released from reservoirs during a period when flow would be naturally lower (summer) and reservoir flow releases are increased in periods where larger amounts of rainfall and increased flow (winter) are expected (Figure 3.1). Periods of low flow are necessary to provide stable flows for post emergence of salmonids with no rapid increases or decreases and higher seasonal flows ensure distribution for spawning and dispersal between a ranges of habitats and maintain sufficient water flow through gravels for brown trout eggs and alevins developing in redds to survive (Cowx *et al.,* 2004). Freshet releases are designed to mimic periods of heavy rainfall events when rivers levels and flow would be high, the aim is to aid the migration of fish species, allowing them to ascend natural barriers and access new and/or spawning habitats (Hannahford & Acreman, 2007; UKTAG, 2013).

There are some possible consequences to modified flow regimes, for example, low flows in the summer could reduce the dispersal of young of year brown trout, reduce the wetted area, increase water temperatures, reduce dissolved oxygen, reduce water quality and potentially reduce brown trout densities (Bunn & Arthington, 2002; UKTAG, 2013). A reduced summer flow could increase sedimentation, although elevated flows during the summer could help to maintain clean the gravels for the winter period. Periods of high flow have been documented to displace 0+ salmonids, impacting population densities. The risk of displacement is when fry emerge from gravels, this is referred to as the critical period and the strength of the year class has been correlated with hydraulic conditions and timing of emergence (Petterson, 1982; Lobón-Cerviá 2004; Daufresne *et al*., 2005). Peterson (1982) found that following a series of winter freshets, juvenile coho salmon (*Oncorhynchus kisutch* (W.)) were found as far as 33 km downstream from their residential habitats.

The importance of a varied flow regime and maintaining all aspects of a natural flow regime (magnitude, timing, duration and frequency) (Chapter 2.2) has been widely published, with most of the literature being conceptual and modelled, very few providing empirical evidence (Armstrong & Nislow, 2012, Arthington, 2015). For example, Enders *et al.* (2009) described the life history patterns of Atlantic salmon and presented a detailed hypothetical managed flow regime that provides varying hydrological flows throughout the year ideal for Atlantic salmon. Although this is relevant, the effectiveness of improving Atlantic salmon populations as a result of the regime was not assessed. The lack of scientific evidence provides little indication to determine precisely which of these elements are vital (Acreman *et al*., 2006). The current guidance on environmental flows are mainly based on available knowledge and expert opinion and are recommended to be fully tested to conclude whether applying current guidance has any meaningful influence on the protection of rivers or whether it is a precautionary option (Acreman *et al*., 2006).

As mentioned in Chapter 2.6, resident brown trout are the primary study species for this study. They are sensitive to river flows and one of the most common species within the study sites, making them ideal for biological monitoring of the response to modifying flow regimes from water storage reservoirs (Acreman, 2001). Modified flows are intended to mitigate the impacts created by reservoirs (Chapter 2.5. 1) and in the UK they are designed to meet requirements of fish species including brown trout, with the intention to improve the overall ecological status of the river.

The aim of this chapter is to examine the long-term (eight years following flow change) effects of modifying reservoir flow releases on the downstream brown trout populations, with specific reference to introducing seasonally variable compensation flows and freshet releases.

Specific objectives: -

- To analyse the hydrological regime in the River Holme catchment and assess the regime prior to and following a change in the flow releases from upstream reservoirs;
- To identify temporal fluctuations and spatial variations in brown trout population density and population structure in the rivers before and after changes in flow release programme;
- To identify annual differences in growth including before and after changes in flow release programme;
- To determine the habitat suitability and usage prior to and following flow change using HABSCORE.

Outputs will determine the effectiveness of the introducing seasonal compensation flows and single freshet release on brown trout populations.

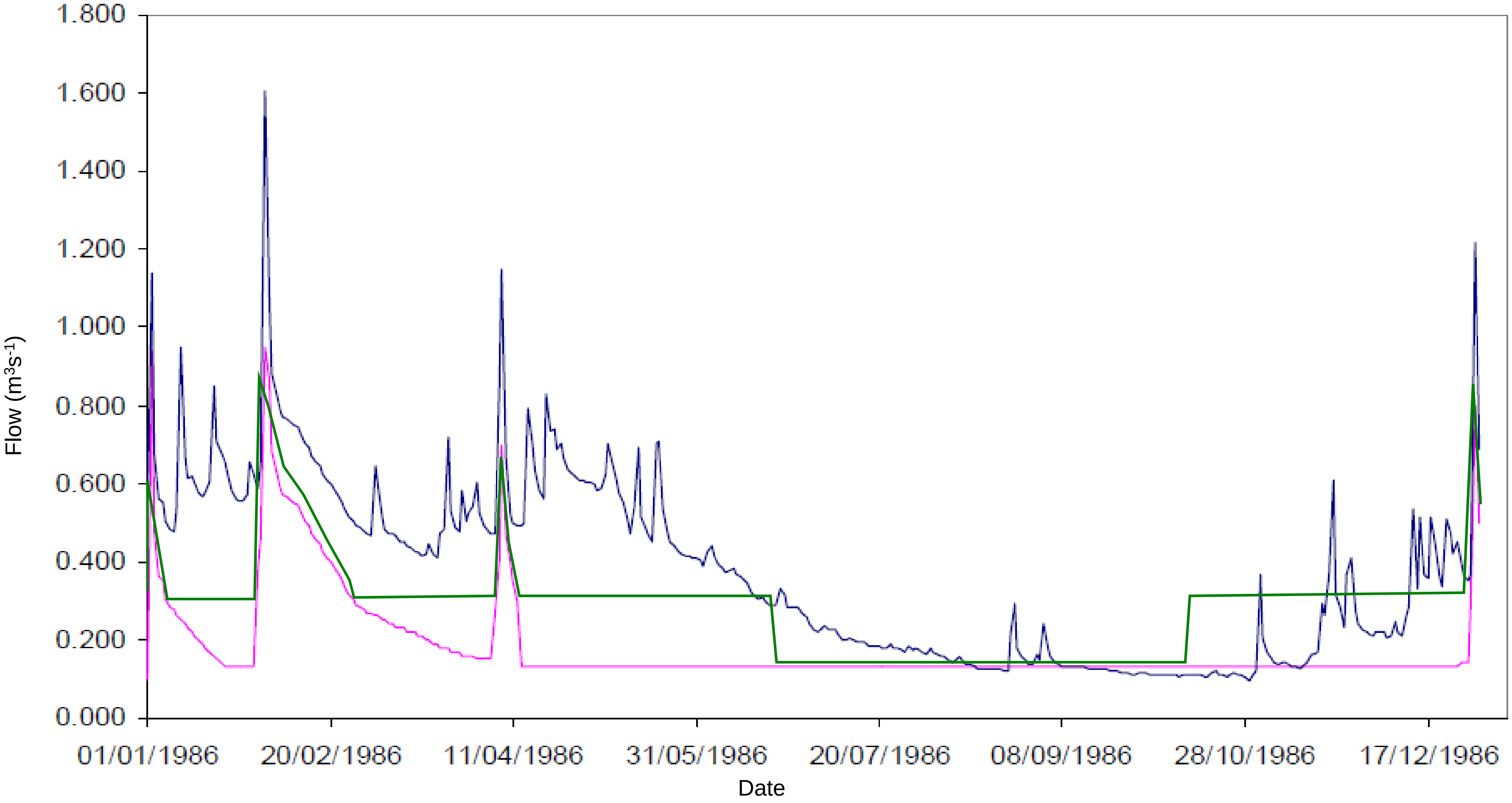


Figure 3.1. Examples of river flow regimes: natural (blue), impacted by an impoundment with a minimum compensation flow (pink) and impacted by an impoundment with a seasonal compensation flow (green) taken and adapted from Acreman *et al*., 2006.

3.2 METHODOLOGY

3.2.1 Study area

The reservoirs used for investigating the influence of modified flow releases were Holme Styes, Digley and Brownhill. All three reservoirs are located in or on the edge of the Peak District national park in West Yorkshire (Figure 3.2). The outflow from Digley and Brownhill reservoirs are received by the River Holme and from Holme Styes Reservoir the River Ribble. Six sites were sampled for fish on the River Holme and four on the River Ribble (Figure 3.2) prior to (2003 and 2002) and following (2004-2009 and 2012-2013) modified flow releases. Sites selected were based on EA data collection in the first two years of this study, for consistency these sites were continued to be sampled. For the same reason, this is why no reference sites were included within the study.

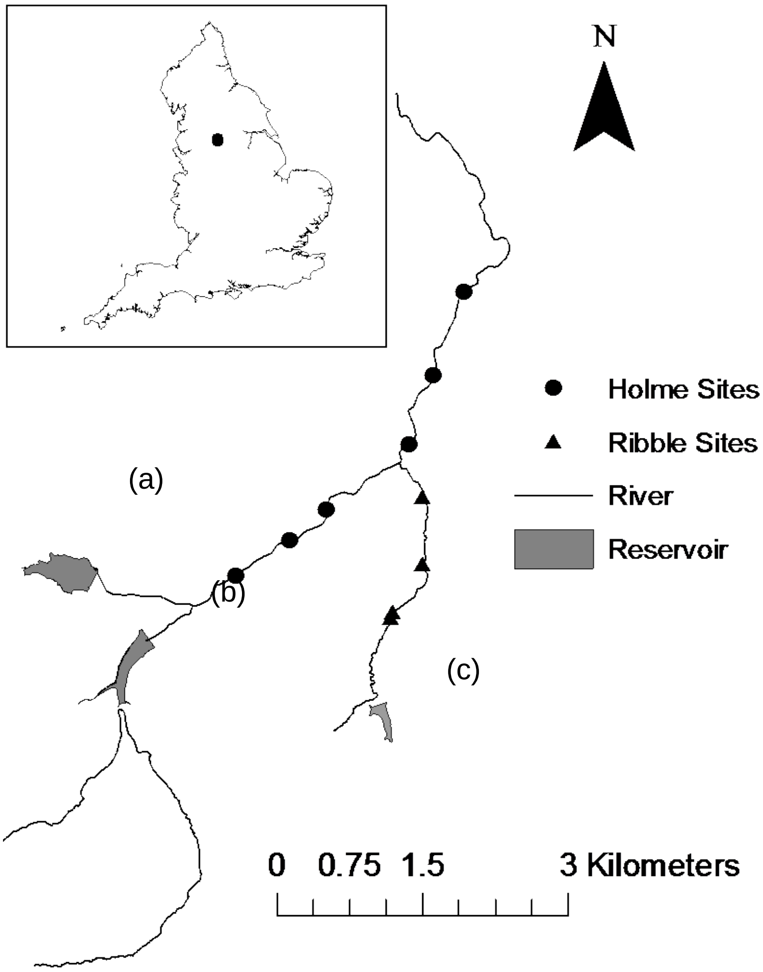


Figure 3.2. A map of England showing the location of the study, and a more detailed map showing the locations of the three study reservoirs ((a) Digley, (b) Brownhill and (c) Home Styes) and River Holme and River Ribble sample sites.

The original compensation flows for the three reservoirs were established in the 1980s. Compensation releases were calculated and implemented by Yorkshire Water Services using values from inflow measurements (flows coming into the reservoir that exit under a natural system) from 1920 to 2011. The original compensation release from Holme Styes Reservoir was 0.89 ML day^{-1}, 6.56 ML day^{-1} from Digley Reservoir and 6.82 ML day^{-1} from Brownhill Reservoir (Table 3.1). Plans were made to introduce seasonal compensation flows and freshet flows in 2004 (Table 3.1). The revised regime for Holme Styes Reservoir increased the minimum compensation flow between January and September to 1.98 ML day^{-1}, with a period of higher winter compensation flow between October and December set at 4ML day^{-1} and a single freshet release in November (Table 3.1). The revised minimum compensation flow at Digley Reservoir was lower (5.10 ML day^{-1}) than the old regime (6.56 ML day^{-1}), the winter compensation flows was set at the same value as the original minimum compensation value (6.56 ML day^{-1}) with a single freshet release in November (Table 3.1). The revised minimum compensation flow at Brownhill Reservoir was lower (5.40 ML day^{-}1) than the old regime (6.82 ML day^{-1}), the winter compensation flows was set at the same value as the original minimum compensation value (6.82 ML day^{-1}) with a single freshet release in November (Table 3.1). The rivers Holme and Ribble are regulated, despite this there is some natural gather form the catchment, which to some extent provides variation in flow (see Figure 3.3). Queens Mill, located on the River Holme (NGR: SE14229 15754), was the nearest flow gauge downstream of Holme Styes Reservoir (13.7 km), Digley (14.5 km) and Brownhill reservoirs (14.0 km). Due to the location of Queens Mill being several kilometres downstream of the study reaches, data include catchment gather of rainfall but represents an indication of flows in the Holme.

Table 3.1. Original and revised flow release programme from Holme Styes, Digley and Brownhill reservoirs into the rivers Holme and Ribble.

Reservoir	Original Compensation Release	Old Regime	Revised Regime		River	Start Date of Revised Regime
Holme Styes	Operated as a continuous discharge of 0.89 ML day^{-1} by agreement with the EA (since the 1980's).	0.89 ML day^{-1}	Jan – Sep	1.98 ML day^{-1}	Ribble	01/04/04
			Oct – Dec	4 ML day^{-1} (+ Nov freshet flow)		
Digley	Operated as a continuous discharge of 6.56 ML day^{-1} by agreement with the EA (since the 1980's).	6.56 ML day^{-1}	Jan – Sep	5.10 ML day^{-1}	Holme	01/04/04
			Oct – Dec	6.56 ML day^{-1} (+Nov freshet flow)		
Brownhill	Operated as a continuous discharge of 6.82 ML day^{-1} by agreement with the EA (since the 1980's).	6.82 ML day^{-1}	Jan – Sep	5.40 ML day^{-1}	Holme	01/04/04
			Oct – Dec	6.82 ML day^{-1} (+Nov freshet flow)		

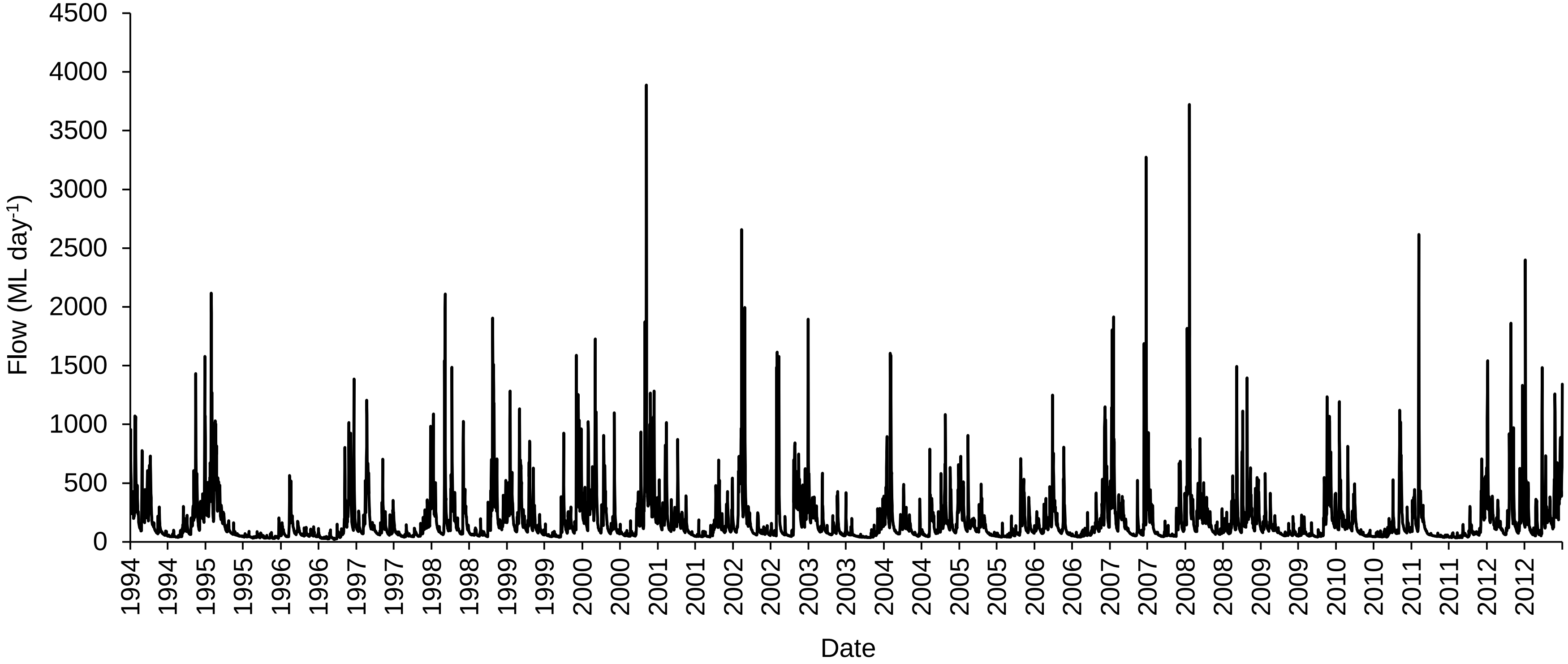


Figure 3.3. Mean daily discharge (ML day^{-1}) for the duration of the study Queen's Mill 1994 – 2014.

3.2.2 Fish survey methodology

Fisheries surveys on the rivers Holme and Ribble were carried out by the EA in September 2002 and again in 2003 prior to the introduction of seasonal compensation and freshet flows in 2004 (Table 3.1). Thereafter, surveys were carried out by Hull International Fisheries Institute (HIFI) (2004-2009 and 2012-2013: MT personally involved in the last two survey years). Yorkshire Water did not require any monitoring in 2010 and 2011, therefore urveys were not carried out . Surveys in 2012 and 2013 were a baseline for a new flow trial planned for October 2013 that was not within the scope of this study. Surveys were carried out in September to ensure 0+ brown trout were sufficiently large enough to be susceptible to capture by electrofishing (>50mm). Undertaking surveys in September therefore aims to provide a more accurate reflection of the fish populations and how successful the recruitment (the number of young fish in a year class at some arbitrary point in time (Kjesbu *et al*., 2016)) is within the rivers.

The three-catch removal method (Carle & Strub, 1978) to obtain absolute estimates of abundance was adopted for all fisheries surveys. The fishing strategy involved three persons (one anode operator and two operators capturing fish) fishing in an upstream direction over a set length of river of 50m between stop nets, and a fourth person on the bank supervising safe operation of the electric fishing equipment. Fishing equipment consisted of a 2kVA generator powering an Easyfisher (EFU-1) or Electracatch control box producing a 240 V DC output. The two individuals netting (dip nets) caught as many fish as possible whilst wading either side of the anode operator. Fish caught in each run were kept separately for subsequent data collection. Following the three runs, brown trout were measured to the nearest mm (folk length) and a small number of scales were removed for ageing analysis from brown trout greater than 50 mm in length; anything smaller does not have well defined readable scales and can be assumed to be 0+.

3.2.3 HABSCORE data collection

HABSCORE data were collected at each site in each year of survey (Figure 3.2). HABSCORE measures and evaluates stream salmonid habitat features based on empirical statistical models relating the population size of five salmonid species/age combinations (Wyatt *et al.,* 1995). Three HABSCORE questionnaires (HABform, MAPform and FISHform) were used to produce several outputs, including estimates of the expected populations (the Habitat Quality Score, HQS) and the degree of habitat utilisation (the Habitat Utilisation Index, HUI), for each of five salmonid species/age combinations (Wyatt *et al.,* 1995). The first questionnaire (HABform) was completed at each site following electric fishing surveys, and involved recording key site habitat information. Habitat data collection commenced at the downstream limit of the survey site, where the wetted width (m) of the river was measured and depth (cm) measurements were taken at three equally spaced intervals corresponding to ¼, ½ and ¾ along the measured wetted channel width. Measurements were recorded in 10 m sections of each survey site. In each 10 m cross-section substrate composition (bedrock/artificial, boulders, cobbles, gravel/coarse sand, fine sand and compacted clay) and flow type (cascade/torrential, turbulent/broken deep, turbulent/broken shallow, glide/run deep, glide/run shallow, slack deep and slack shallow) were recorded based on the percentage of five abundance categories; dominant (≥ 50%), frequent (≥ 20% & ≤ 50%), common (≥ 5% & ≤ 20%), scarce (> 0% & < 5%)and absent (0 %). If the last section of river was <15 m the exact length was noted and treated as one section, if the length was >15 m it was split into another two sections with the first being a 10 m section and the last section the remaining length; for example 16 m would be treated as a 10 m section and a 6 m section. Information collected was used evaluation of salmonid stream habitat features.

The two further questionnaires (MAPform and FISHform) were completed for each site (Barnard & Wyatt, 1995). MAPform was completed by collection of relevant information from OS Maps (1:50000) and River Water Quality Maps (1:25000). FISHform was completed by recording of fisheries statistics of the five salmonid species/age combinations derived from the fisheries surveys. Although HABSCORE allows predication of suitability for salmon this was not undertaken as salmon are absent from the Holme catchment.

Data from the three completed questionnaires (HABform, MAPform and FISHform) at each site were entered into the HABSCORE for Windows program and a number of outputs (Section 3.2.4.5) were produced for brown trout populations.

3.2.4 Data analysis

3.2.4.1 Hydrological flow analysis

Flow data from the River Holme at Queens Mill gauging station, obtained from the Environment Agency, was assessed using Indicators of Hydrological Alteration (IHA) version 7.1 (The Nature Conservancy, 2009). The software was used to assess the potential difference in hydrological conditions pre (1994 – 2003) and post (2004 – 2012) flow modification (introduction of seasonal compensation flow and freshet flows). Pre flow modification data dates back further than fisheries data to provide a comparable period of time to the post flow modification data. Fisheries data were not gathered pre 2002, consequently fisheries data analysis is restricted to years analysed for this study.

The IHA method calculates a total of 67 statistical parameters, these are divided into to two groups, the IHA parameters and the Environmental Flow Component (EFC) parameters. IHA calculates 33 parameters divided into five main components: monthly magnitude, annual magnitude and duration of extreme water conditions, timing of annual extreme water conditions, frequency and duration of high and low pulses, and the rate and frequency of change in water conditions. There are 34 parameters for five different types of Environment Flow Components (EFCs): extreme low flows, low flows, high flow pulses, small floods, and large floods. (The Nature Conservancy, 2009). Non-parametric analysis was used to determine significant differences in IHA parameters to compare pre (1994 – 2003) and post (2004 – 2012) flow modification. When comparing pre- to post-impact flow magnitudes, IHA parameter groups #1 (months) and #2 (1,3,7,30 & 90 minimum and maximum, number of zero days and base flow) were used instead of EFCs.

Range of Variability Approach (RVA) was used to analyse the change between the two time periods, and the analysis uses the pre-data variation of IHA parameter values as a reference for defining the extent to which the pre flow regimes have been altered. In an RVA analysis, the full range of pre-flow modification data for each parameter is divided into three different categories. Non parametric RVA analysis places data into three categories of equal size, 17 percentiles from the median. The lowest category contains all values less than or equal to the 33rd percentile; the middle category contains all values falling in the range of the 34th to 67th percentiles; and the highest category contains all values greater than the 67th percentile. (The Nature and Conservancy, 2009). Firstly, the expected frequency is calculated, the frequency with which the "post-flow modification" values of the IHA parameters should fall within each of the three categories. The same is repeated for the observed frequency of which the “post-flow modification" annual values of IHA parameters of which actually fell within each category. This expected frequency is equal to the number of values in the category during the pre-flow modification period multiplied by the ratio of post- flow modification years to pre-flow modification years. Finally, a Hydrologic Alteration (HA) factor is calculated for each of the three categories as:

(*observed frequency – expected frequency*) / *expected frequency*

A positive HA value means that the frequency of values in the category has increased from the pre-flow modification to the post-flow modification period (with a maximum value of infinity), while a negative value means that the frequency of values has decreased (with a minimum value -1). For the purpose of this study, a change (positive or negative) in frequency of each category could be either good or bad (ecologically) depending on which category has changed and the time which it occurred. This is subjective and any changes detected will be expanded upon in the discussion.

3.2.4.2 *Density estimates of brown trout*

Density estimates of 0+ and >0+ fish per 100 m^2 at quantitative sites were derived from absolute abundance estimates determined from three-catch removal method (Carle & Strub, 1978). These data were used to assess the status of the fish populations according to the Environment Agency Fisheries Classification Scheme (Section 3.2.4.3).

Statistical analysis was performed on density estimates to compare mean density before and after flow changes. Data were checked for homogeneity of variance using a Levene's test. If the variances were equal a paired two sample assuming equal variances an ANOVA was performed, if variances were not equal a non-parametric equivalent test was used (Kruskal-Wallis). Pre density estimates (2002 and 2003) were compared to two post density estimate categories, post short term (2004-2009) and post long term (2012 and 2013).

3.2.4.3 *Classification of population estimates*

Density estimates calculated from fisheries surveys were compared to the Environment Agency Fisheries Classification Scheme (EA-FCS). This enables the first two years of data collection and outputs by the EA to be compared to subsequent data collected, but also using this grading scheme allows for comparison to national trends and puts densities into a context of how good or bad they are. 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.

Table 3.2. 0+ and >0+ brown trout abundance (N/100 m^2) classifications used in the Environment Agency Fisheries Classification Scheme (EA-FCS), colours are assigned for clarity in subsequent data analysis.

	Abundance classification					
Species group	A	B	C	D	E	F
0+ brown trout	≥38.00	17.00-37.99	8.00-16.99	3.00-7.99	0.10-2.99	0.00
>0+ brown trout	≥21.00	12.00-20.99	5.00-11.99	2.00-4.99	0.10-1.99	0.00

3.2.4.4 Length at age determination of growth rate

Calculation of growth rates of brown trout was facilitated by the collection of scale samples from a representative number of fish (three brown trout from every 10-mm size class) (Britton, 2003). Determination of the age and growth of fish is an important tool in the assessment of fish population dynamics (Bagenal, 1978, Rifflart *et al*., 2006). For example, fast growth rates within a fish population could indicate that habitat and environmental conditions are very good, whereas slow growth rates could indicate poor habitat and environmental conditions (Ayllon, 2014; Giller & Greenberg, 2015).The age and growth of brown trout at each site and each year were determined by the interpretation of annual growth checks (annuli) that appear on the scales of fish (Bagenal & Tesch, 1978). These are formed during periods of faster and little or no growth, with the latter generally occurring during the winter months in temperate regions.

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. The total scale radius and scale radius to each annuli were measured from the nucleus to the scale edge. Analysis of the data involved assessment of the relationship between the length of the fish, scale radius to annuli and total scale radius (Dahl-Lea method, Francis (1990)):

$$Li = (Si/Sc) \times Lc \qquad \text{(Equation 1)}$$

where, Li is the length (mm) at year 1, Si is scale radius at length Li, Lc the length at capture and Sc the scale radius at capture.

For each brown trout, the length at age was back-calculated from the scale radius to each annuli using Equation 1. This calculation was repeated for each fish and the mean length for each age from all fish in the population was calculated. Data were then tabulated and displayed graphically for length at age one and two in the rivers Holme and Ribble. Sample sizes on occasions were too small to allow statistical comparison for fish length at age three between individual sites. For this reason, sites were combined for each river and a *t*-test was performed to see if there were any differences in length at age one and age two brown trout between pre (2001-2003) and post (2004-2006) flow change years. Analysis of Variance (ANOVA) was used to compare the length at age one and two growth between years.

3.2.4.5 HABSCORE analysis and ouputs

Data from HABform, MAPform and FISHform at each site were entered into the HABSCORE for Windows program. Habitat data were divided into pre (2003 (2002 and 2003 fish density data)), short term post (2006 (2004- 2006 fish density data)) and long term post (2013 (2007-2009 and 2012 and 2013 fish density data)) flow modifications. Data for at least two years were used to account for temporal variation, the following outputs were produced for brown trout population categories; 0+, >0+ (<20 cm) and >0+ (>20 cm) (definitions from Wyatt *et al.,* 1995):

Habitat Quality Score (HQS)

The HQS value is a measure of the habitat quality expressed as the expected long-term average 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 / 100 m^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.

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.

Log$_e$ HUI

This is the natural logarithm of the HUI. Negative values will represent an observed population less than that which would be expected given the habitat. The data were tabulated from each site and interpreted in relation to the fish population data.

3.3 RESULTS

3.3.1 Hydrological flow parameters

Throughout the study period (1994-2012), flows on the River Holme varied and all five EFC catogories were classified at some point (Figure 3.4). Most frequently classified were low flows (n= 4650), followed by high flow pulse (n= 1328) (Figure 3.4). Extreme low flows(n= 607), small floods (n= 316) and large floods (n= 39) were less frequently classified. Comparing pre and post median monthly flows, seven out of the twelve months fell within the RVA boundaries (Figure 3.5). May, August and September fell just outside the boundaries but less so than Febuary and March (Figure 3.5). Post median monthly flows were higher in January, April, August, September and December in comparison to pre median monthly flows and were lower in February, March, May, June and November. Median values were most comparable between pre and post flow changes in the months of July and October (Figure 3.5).

The IHA analysis revealed that the degree of alteration for each of the hydrological indicators were not significantly different for all tested parameters. Parameters that were most different but not significantly included median flow values in August (P= 0.11), September (P= 0.13) and base flow index (minimum compensation flows) (P= 0.12) with increases in flow in post years. Flow exceedance curves between the two time periods are comparable, the only difference in the curve is that pre changed flows had slightly lower flows (Figure 3.5).

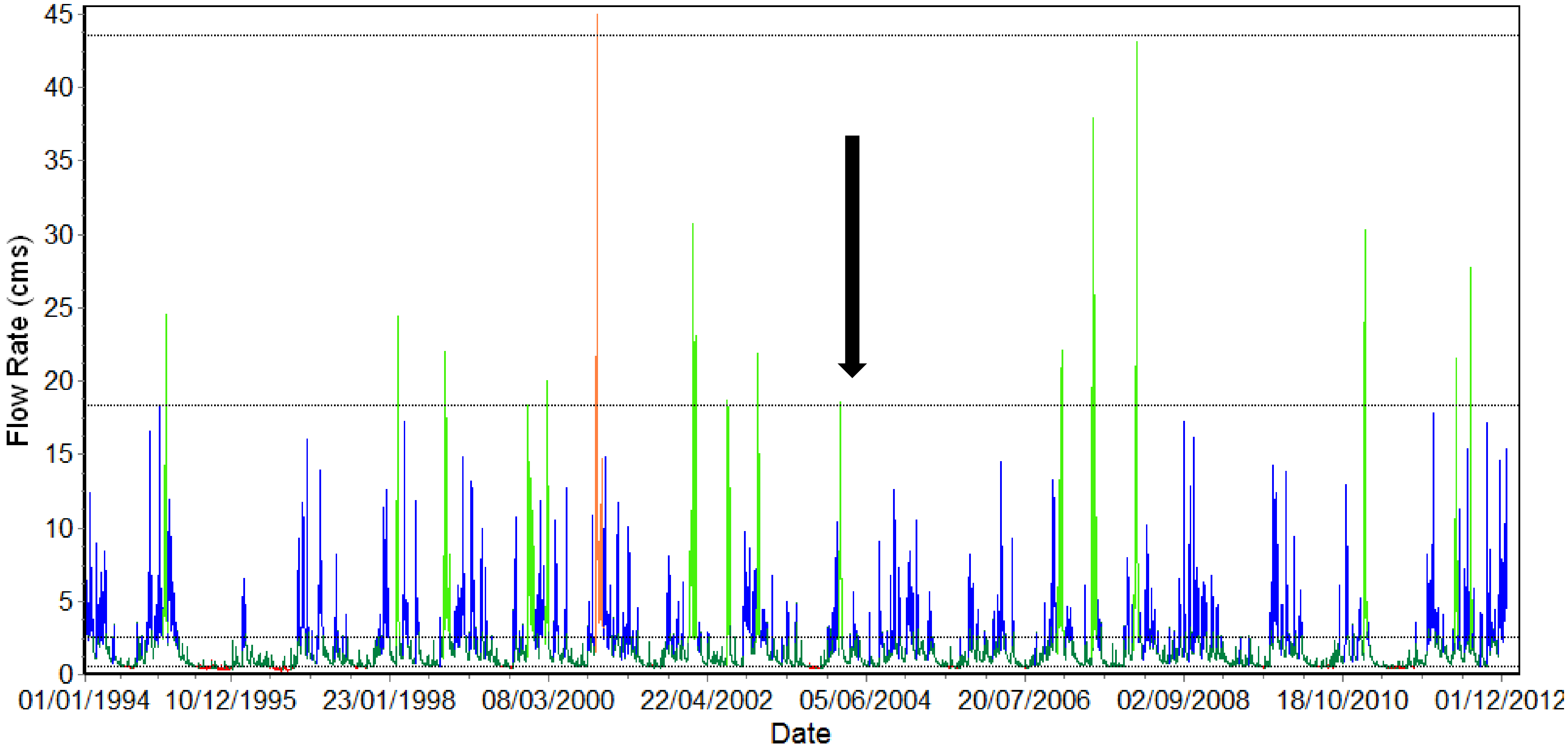


Figure 3.4. Queen's Mill Environmental Flow Components; extreme low flows (red), low flows (dark green), high flow pulse (blue), small floods (light green) and large floods (orange) (1994-2012), extreme low flow threshold, high flow perceived threshold, small flood peak flow threshold and large flood peak flow threshold (Black dashed lines in ascending order). Arrow indicates the introduction of the revised flow release programme.

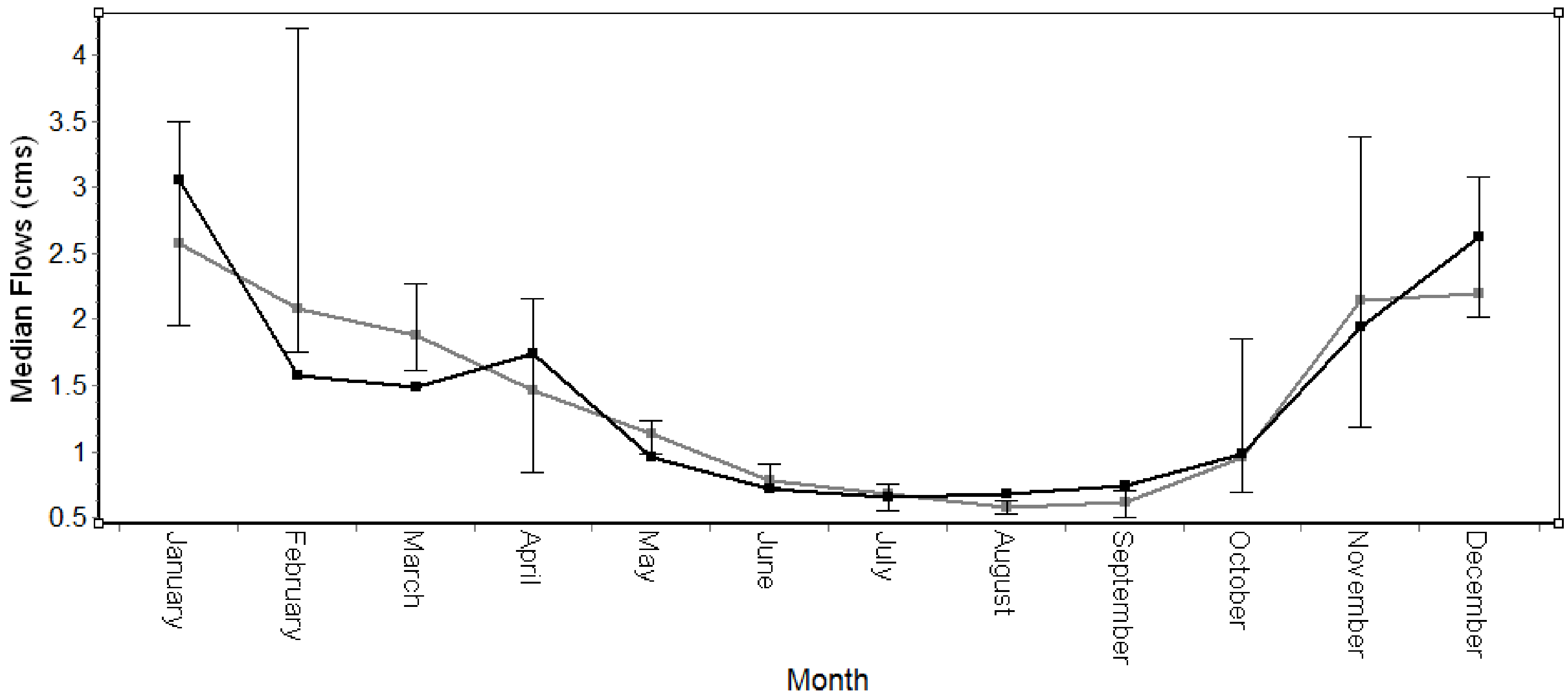


Figure 3.5. Monthly median flows pre (grey (1994-2003)) and post (black (2004-2012)) flow changes, data from Queen's Mill flow gauge, including Range of Variability Approach (RVA) boundaries.

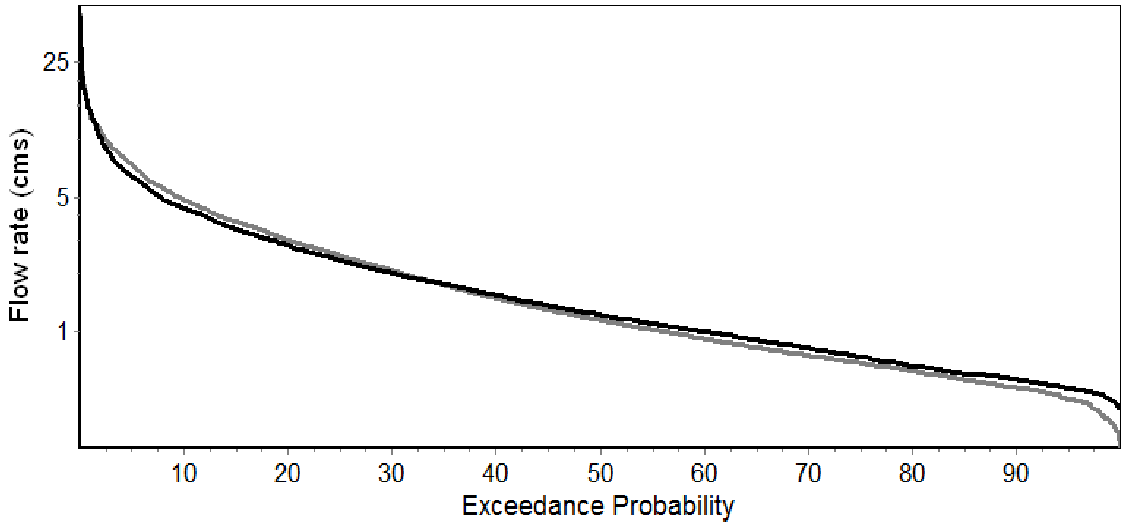


Figure 3.6. Flow exceedance curve of pre (grey (1994-2003)) and post (black (2004-2012)) flow changes, data from Queen's Mill flow gauge.

3.3.2 Brown trout population density trends

2.6.1.1 Spatial and annual variation in brown trout densities in the River Holme

Considerable differences were found in the densities of 0+ and >0+ brown trout in the River Holme between sites (D1 – D6) and years (2002-2009 and 2012-2013) (Table 3.3 and 4.4).

0+ trout were captured each year indicating recruitment of brown trout throughout the study, but the densities were highly variable (Table 3.3). On a site by site basis 0+ brown trout densities varied considerably, e.g. from 6.83 fish per 100 m^2 (D2) to 34.26 fish per 100 m^2 (D4) in 2003. If applied to the EAFCS that ranges from class D to B. Site by site variations changed throughout the years with no one single site being consistently poorer than the rest. Variation in densities were found at individual sites between study years (Table 3.3). For example, 0+ brown trout densities at D3 ranged from 0.7 (2004; class E) to 41.4 (2005; class A) fish per 100 m^2.

>0+ trout dominated catches at all study sites in the River Holme in most years, except in 2005 (all sites except D5), 2003 (3 sites; D3, D4 and D6) and 2008 (1 site; D3) and 2009 (1 site; D1). On a site by site basis, >0+ brown trout densities varied considerably, e.g. from 0.36 fish per 100 m^2 (D1) to 30.83 fish per 100 m^2 (D5) in 2009; if applied to the EAFCS that ranges from class E to A. Variations at each site between years were also found, e.g. >0+ brown trout at D1 were classified as good (class B = 2002, 2005-2008) but in 2009 the classification had dropped to poor (class E). Over the years D6

was on average one of the best sites for >0+ brown trout populations classified as excellent (class A = 2002, 2006, 2007 and 2009) but in 200 was classified as poor (class D - 2.06 fish per 100 m^2).

3.3.2.2 Spatial and annual variation in brown trout densities in the River Ribble

0+ brown trout dominated catches at R30 and R32 (except in 2002, 2007 and 2012), and occasionally at R31 (2004, 2008, 2012 and 2013) and R33 (2003, 2005, 2009 and 2013). 0+ brown trout were caught at all study sites in all study years, except R33 in 2002 (class F) (Table 3.3). When 0+ brown trout were caught there was noticeable site by site variation, e.g. in 2007 densities of 0+ brown trout ranged from 2.7 fish per 100 m^2 (R33) to 69.12 fish per 100 m^2 (R30) (Table 3.3); if applied to the EAFCS that ranges from class E to A. Variations at each site between years were also found, e.g. at site R33 0+ brown trout populations were classified as good (class B = 2003, 2005, 2009 and 2013) but in 2002 no 0+ brown trout were caught (class F) and at R32 0+ brown trout populations were classified as excellent (class A = 2003, 2005, 2006, 2008, 2009) but were poor (class E) in 2002 (Table 3.3). This demonstrates the annual variation in brown trout population densities between years at individual sites.

The status of >0+ brown trout populations in the River Ribble contrasted with the River Holme, and were only dominant in the River Ribble at two sites; R31 (except in 2004, 2008, 2012 and 2013) and R33 (except in 2003, 2005, 2009 and 2013) (Table 3.3 and 4.4). Variations between sites were found, e.g. at site R30 >0+ brown trout population densities were poor (class D) in 2012 but were excellent (class A) at site R33 in 2012. Although there were variations between sites, the overall mean density of >0+ brown trout was similar at all sites (R33 -18.5 fish per 100 m^2, R31 - 18.9 fish per 100 m^2, R32 - 19.5 fish per 100 m^2 and R30 - 22.51 fish per 100 m^2) (Table 3.4). Annual variations were found at individual sites, e.g. R30 were poor (class D) in 2012 but excellent (class A) in 2004-2009. Other sites ranged from class A-C throughout the study period (Table 3.4).

3.3.2.3 Pre and post flow change variation in brown trout densities in the River Holme

Sites (D1-D6) on the River Holme were combined to compare brown trout population densities pre and post flow change (Figure 3.6). Mean 0+ brown trout densities pre flow change (2002 & 2003) in the River Holme were different between years, in 2002 the mean density was 4.1 fish per 100 m^2 (class D) and in 2003 four times higher (17.5 fish per 100 m^2 (class B)) (Figure 3.6). 0+ brown trout population densities post flow change varied from 4.4 fish per 100 m^2 in 2004 (class D) to 35.5 per 100 m^2 in 2005 (class B); densities thereafter fell to 12.5 fish per 100 m^2 (class C) and were similar in 2008 (10.3 fish per 100 m^2) and 2009 (11.0 fish per 100 m^2) (Figure 3.6). Mean 0+ brown trout densities were lower (class D) in 2007 (7.7 fish per 100 m^2), 2012 (6.5 fish per 100 m^2) and 2013 (7.7 fish per 100 m^2). Overall, 0+ brown trout densities in the River Holme since flow modification (1 April 2004) were comparable with densities before flow modification (Figure 3.6).

Mean >0+ brown trout densities on the River Holme pre flow change (2002 & 2003) were slightly higher in 2002 (16.7 fish per 100 m^2) (class B) than in 2003 (11.4 fish per 100 m^2) (class C) (Figure 3.6). On the whole, post flow change did not appear to influence >0+ brown trout densities: mean >0+ brown trout densities were lowest in 2008 (7.6 fish per 100 m^2 (class C)) and highest in 2006 (20.9 fish per 100 m^2 (class B) (Figure 3.6).

There was no significant difference in 0+ and >0+ brown trout densities between the periods pre, and short and long term post the introduction of seasonal compensation and freshet flows in 2004 (K-W, n = 60, X^2 = 1.21, *d.f.* = 2, P = 0.5 and ANOVA: n = 60, F = 2.3, *d.f.* = 2, P = 0.1).

3.3.2.4 Pre and post flow change variation in brown trout densities in the River Ribble

Sites (R30-R33) on the River Ribble were combined to compare brown trout population densities pre and post flow change (Figure 3.6). Mean 0+ brown trout densities pre flow change (2002 & 2003) in the River Ribble were extremely different between years: in 2002 the mean density was 0.8 fish per 100 m^2 (class E) but in 2003 population density was nearly 35 times higher (27.6 fish per 100 m^2 (class B)) (Figure 3.6). Post flow change 0+ brown trout population densities ranged from 26.2 fish per 100 m^2 (class B) in 2012 to 51.1 fish per 100 m^2 (class A) in 2005. Despite 2002 being an unusually poor year for 0+ brown trout, densities of 0+ brown trout in the River Ribble were not significantly different between pre, short and long term post flow change (ANOVA: n = 38, F = 1.736, *d.f.* = 2, P = 0.191).

Mean >0+ brown trout population densities on the River Ribble pre flow change (2002 & 2003) were higher in 2002 (17.4 fish per 100 m^2) (class B) than in 2003 (6.8 fish per 100 m^2) (class C) (Figure 3.6). Post flow change >0+ brown trout population densities were good (class B) in 2012 (12.9 fish per 100 m^2) and 2013 (16.9 fish per 100 m^2) and excellent (class A) in the short term post years (2004-2009). There was a significant difference between >0+ brown trout densities before and after modification of the seasonal compensation and freshet flows on the River Ribble (ANOVA: n = 38, F = 3.746, *d.f.* = 2, P = 0.034). Densities of >0+ brown trout were not significantly higher in the post short term than densities before the flow change. The long term post >0+ brown trout densities were not significantly different to the pre and short term post densities.

Table 3.3. Densities (± 95% C.L at quantitative sites) of 0+ brown trout at all sites in the rivers Holme and Ribble, 2002-2009 and 2012- 2013. Colours represent EA-FCS grading scheme.

A (excellent)	B (good)	C (average)	D (fair/poor)	E (poor)	F (fishless)

River Name	Site	0+ brown trout 2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	Mean Pre	Mean Post
River Holme	D1	2.84	10.64	1.50	21.72	3.23	1.79	2.15	9.68	-	-	3.69	1.98	6.74	5.72
		±0.2	±2.2	±0.3	±0.9	±0.0	±0.3	±0.0	±0.2			±0.1	±0.2		
River Holme	D2	1.09	6.83	10.63	18.39	15.27	2.02	20.75	3.46	-	-	3.19	7.48	3.96	10.15
		±0.1	±0.7	±0.2	±1.5	±1.1	±0.1	±0.3	±0.0			±0.1	±0.3		
River Holme	D3	4.77	6.96	0.71	41.43	28.85	8.70	1.98	4.74	-	-	5.89	3.41	5.87	11.96
		±0.1	±1.1	±0.0	±0.7	±1.1	±0.1	±0.0	±0.4			±0.4	±0.6		
River Holme	D4	5.61	34.26	5.56	43.79	5.90	11.80	9.51	9.18	-	-	5.82	12.12	19.94	12.96
		±0.8	±5.3	±0.2	±2.1	±0.2	±0.3	±0.9	±0.8			±0.2	±1.4		
River Holme	D5	5.04	24.04	2.27	42.39	4.67	4.36	14.02	11.84	-	-	7.50	14.86	14.54	12.74
		±0.1	±0.5	±0.5	±12.1	±0.2	±0.6	±2.1	±2.6			±0.4	±1.2		
River Holme	D6	4.97	22.52	5.94	45.17	17.07	17.77	13.24	27.18	-	-	12.86	6.05	13.75	18.6
		±0.7	±4.9	±0.7	±2.3	±0.8	±0.8	±0.6	±2.2			±0.9	±0.3		
River Ribble	R30			48.3	111.76	57.35	69.12	91.18	67.65	-	-	50.00	74.80	-	71.27
				±2.5	±6.7	±1.4	±0.9	±1.6	±3.6			±0.8	±1.6		
River Ribble	R31	0.71	7.86	35.21	6.34	6.45	12.90	20.16	11.29	-	-	27.99	41.50	4.29	20.23
		±0.0	±1.3	±1.8	±0.2	±0.2	±0.9	±4.6	±0.1			±0.4	±3.4		
River Ribble	R32	1.74	48.70	30.47	67.97	43.14	23.53	41.18	85.29	-	-	13.72	34.76	25.22	42.51
		±0.0	±2.5	±3.2	±4.4	±4.6	±1.1	±0.6	±4.0			±0.4	±1.1		
River Ribble	R33	0.00	26.26	9.86	18.31	12.97	2.70	4.86	23.78	-	-	13.40	30.82	13.13	14.59
		±0.0	±2.7	±1.3	±0.8	±0.4	±0.0	±0.0	±0.3			±0.4	±0.9		

Table 3.4. Densities (± 95% C.L at quantitative sites) of >0+ brown trout at all sites in the rivers Holme and Ribble, 2002-2009 and 2012- 2013. Colours represent EA-FCS grading scheme.

A (excellent)	B (good)	C (average)	D (fair/poor)	E (poor)	F (fishless)

River Name	Site	>0+ brown trout													
		2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	Mean Pre	Mean Post
River Holme	D1	13.11	10.64	11.97	13.15	20.41	18.96	12.88	0.36			9.39	11.07	11.88	12.27
		±0.3	±0.2	±0.0	±0.2	±0.8	±0.7	±0.8	±0.1			±0.7	±0.3		
River Holme	D2	9.28	9.33	11.77	15.79	15.25	23.65	13.53	19.01			9.30	8.55	9.31	14.61
		±0.3	±0.4	±0.3	±0.7	±0.4	±0.5	±0.9	±0.5			±0.2	±0.2		
River Holme	D3	13.90	5.10	10.36	9.69	19.00	15.46	5.97	20.57			3.31	4.46	9.50	11.10
		±0.5	±1.1	±0.4	±1.3	±0.5	±0.3	±0.2	±0.4			±1.3	±0.1		
River Holme	D4	24.31	20.21	23.88	15.06	19.05	23.65	11.43	12.06			12.94	14.92	22.26	16.26
		±0.5	±1.0	±0.3	±0.4	±1.7	±0.7	±1.0	±0.4			±0.8	±0.7		
River Holme	D5	41.14	49.54	30.72	61.20	51.66	31.75	19.60	30.83			23.31	14.34	45.34	32.93
		±1.6	±3.7	±3.0	±10.3	±0.7	±1.2	±0.5	0.6			±0.8	±0.8		
River Holme	D6	36.42	14.60	2.06	13.90	27.87	27.17	15.33	27.57			15.83	15.28	25.51	18.13
		±0.5	±1.0	±0.4	±0.6	±0.8	±0.8	±1.2	±0.6			±0.8	±1.4		
River Ribble	R30	-	-	27.90	23.57	45.57	39.67	32.40	41.20			3.91	10.89		28.14
				±0.1	±0.8	±0.9	±0.3	±1.1	±1.1			±0.0	±0.6		
River Ribble	R31	28.54	10.73	16.20	25.30	20.20	22.60	12.10	19.40			8.87	25.53	19.64	18.78
		±0.2	±0.2	±0.3	±0.4	±0.7	±0.6	±0.1	±0.7			±0.5	±0.5		
River Ribble	R32	6.97	9.54	21.06	21.14	25.50	26.48	23.48	33.30			15.78	11.83	8.26	22.32
		±0.0	±0.3	±0.8	±0.2	±4.3	±0.4	±0.0	±0.7			±0.4	±0.1		
River Ribble	R33	21.83	11.78	16.92	15.92	22.20	22.70	14.02	16.74			22.90	19.50	16.81	18.86
		±0.3	±0.5	±0.2	±0.3	±0.4	±0.6	±0.2	±0.0			±0.6	0.3		

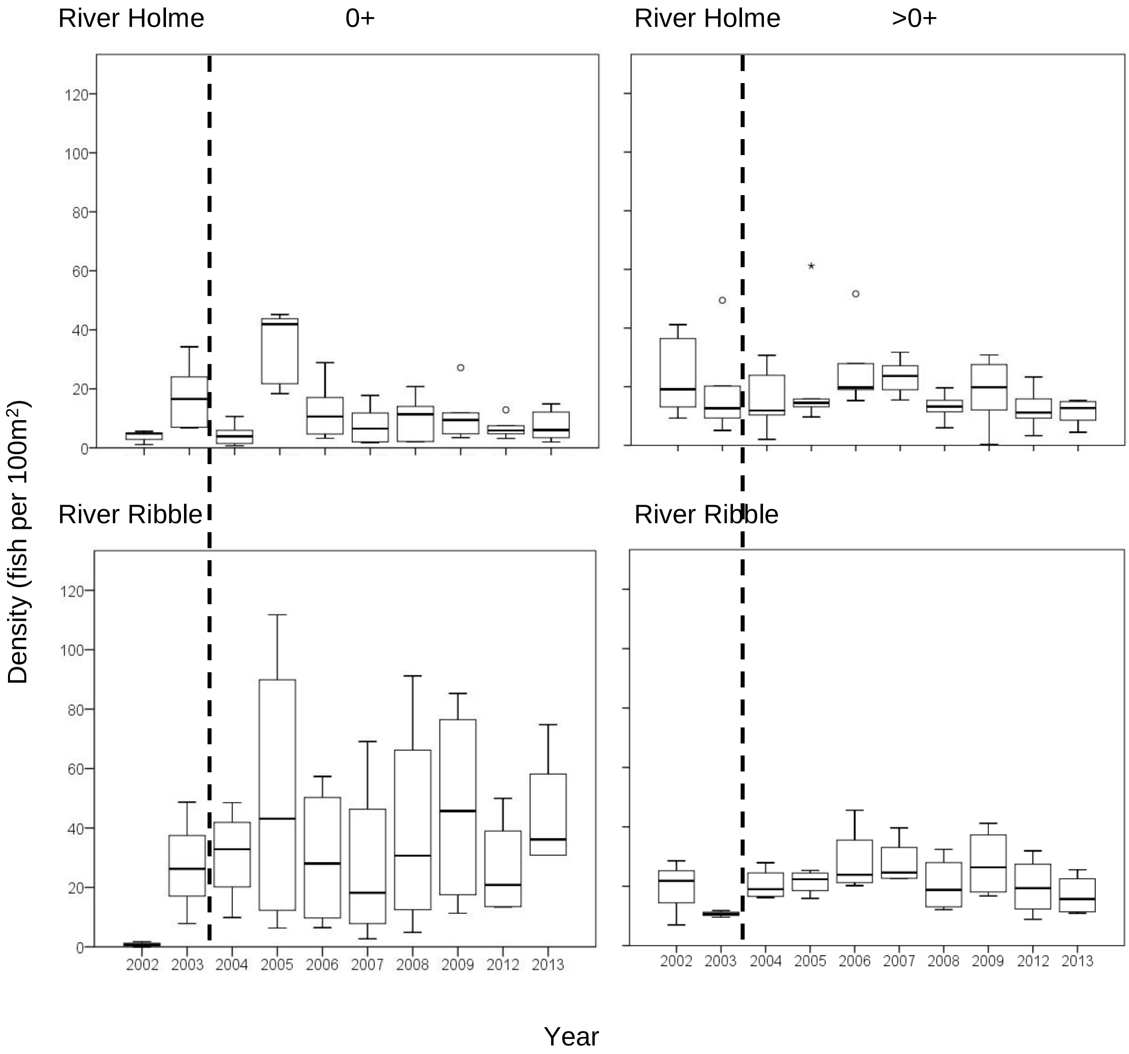


Figure 3.7. Density (all sites) of 0+ (left) and >0+ (right) brown trout in the rivers Holme and Ribble, 2002-2009 & 2012-2013. Densities in the River Ribble are based on surveys at three sites in 2002 and 2003 and four sites thereafter, dash line indicates introduction of the revised flow release programme.

2.6.1.2 Growth

Back-calculated length of brown trout at age 1 and 2 were calculated and yearly averages were compared in the rivers Holme and Ribble (Figure 3.7). Figure 3.7 shows a steady decline in brown trout growth for length at age 1 and 2 over the whole study in the rivers Holme and Ribble.

Back-calculated brown trout lengths at age 1 on average ranged from 65 mm in 2009 and 80 mm in 1998 (Figure 3.7a) in the River Holme. Brown trout lengths at age 1 were not significantly different between pre (2001-2003) and post (2004-2006) flow modifications on the River Holme (Pre= 73.5 ± 17.8 mm Post= 74.3 ± 16.8 mm; *t*-test, *t*

= -0.644, *d.f.* = 896, P = 0.520). Back-calculated brown trout lengths at age two on average ranged from 143 mm in 2001 to 163 mm in 2003 in the River Holme (Figure 3.7b). On average, brown trout lengths at age 2 were not significantly different between pre flow (155.2 ± 25.2 mm) post flow change (154.2 ± 26.8 mm; t-test, t = 0.187, *d.f.* = 470, P = 0.852).

Brown trout length at age 1 in the River Ribble ranged from 89 mm in 2002 to 60 mm in 2010 (Figure 3.7c). There was no significant difference in mean lengths at age 1 in the Ribble between pre and post flow change (Pre= 76.4 ± 21.3 mm Post= 77.1 ± 15.3 mm) (t-test, t = -0.306, *d.f.* = 208, P = 0.760). The range of back calculated length at age 2 on the River Ribble was between 120 mm in 2011 and 230 mm in 2002, this was possibly due to stocking and a small sample size, therefore averages were greater(Figure 3.7d). When grouped, no significant differences were found in length at age 2 in pre flow and post flow change periods (Pre= 159.2 ± 38.1 mm Post= 146.1 ± 18.5 mm) (t-test, t = 2.004, *d.f.* = 56, P = 0.050).

Analysis of Variance (ANOVA) results comparing length at age one and two of brown trout between years (1998-2013) on the rivers Holme and Ribble suggest that brown trout growth rates are similar between 1998 to 2007 but differ from the years 2008 to 2013 (Appendix Table 9.1).

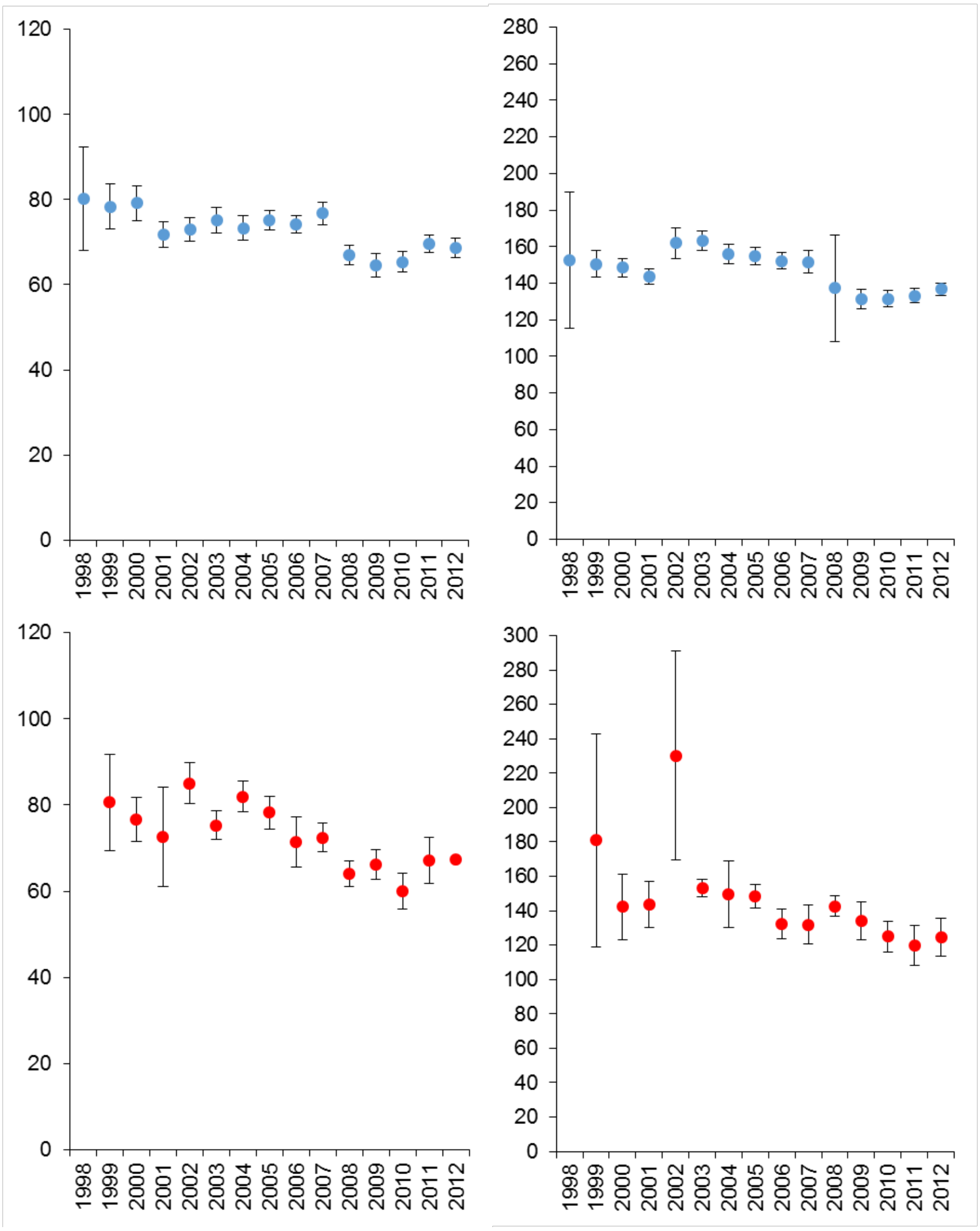


Figure 3.8. Back-calculated length (mm ± 95% confidence limits.) of brown trout at age 1 (a) and age 2 (b) in the River Holme (Blue) and age 1 (c) and age 2 (d) in River Ribble (Red).

3.3.3 HABSCORE

Habitat features were recorded at all sites surveyed on the Holme and Ribble in 2003, 2006 and 2013 to assess the potential brown trout habitat and the level of usage (see

Section 3.2.4.5) for definitions of the outputs). HABSCORE outputs were used to identify variation in the observed densities, predicted densities and habitat utilisation by 0+, >0+ (<20 cm) and >0+ (>20 cm) brown trout, pre and post flow change in 2004, with assessment over the short term (2006) and long term (2013).

3.3.3.1 Habitat usage of brown trout in the River Holme pre flow change, post short term and post long term flow change

Observed densities of 0+ brown trout in 2002 and 2003 pre flow change were lower than predicted by the Habitat Quality Score (HQS) at two sites in the River Holme (D1 and D2), and higher at the other site, but none were significantly different (Table 3.5). In the short term post flow change period, fish data from 2004-2006 were compared with habitat data for 2006, two years after the flow modification. Observed densities of 0+ brown trout were lower at all sites on the rivers Holme and Ribble, but not significantly lower, than predicted (Table 3.5). In 2013, long term observed densities of 0+ brown trout were lower than predicted by the Habitat Quality Score (HQS) at four sites in the River Holme (D1-D4) and higher at the other two (D5 and D6); none were significantly different.

Densities of >0+ brown trout (<20 cm) prior to the flow modification in 2004 were significantly higher at all sites in the River Holme. Short term post flow change >0+ brown trout (<20 cm) densities were higher at all sites, and significantly at four sites (D2, D4-D6). Long term post flow change > 0+ brown trout (<20 cm) densities were higher than predicted at all sites but not significantly (Table 3.5).

Significantly higher densities of >0+ brown trout (> 20 cm) than predicted from the HQS were found at five sites in the River Holme (D1, D2, D4-D6) pre flow change (Table 3.5).Densities of >0+ brown trout (> 20 cm) following flow modification in the short term were higher at all of the River Holme sites and significantly so at sites D4-D6 (Table 3.5). The same results were found in the long term post flow change; populations of >0+ brown trout (> 20 cm) were higher at all sites on the River Holme at all sites and significantly higher at D4-D6 (Table 3.5).

3.3.3.2 Habitat usage of brown trout in the River Ribble pre flow change, post short term and post long term flow change

Pre flow change observed densities of 0+ brown trout in the River Ribble were lower than predicted by the Habitat Quality Score (HQS), suggesting lower populations than expected; in all cases 0+ brown trout densities were not significantly lower than expected (Table 3.5). In the short term post flow change 2004-2006 fish data were compared to 2006 habitat data, two years after the flow modification and observed densities of 0+ brown trout were lower at all sites on the River Ribble, but not significantly lower than predicted (Table 3.5). Long term post flow change observed densities of 0+ brown trout were lower than predicted by the Habitat Quality Score (HQS) at two sites in the River Ribble (R32 and R33), but neither were significantly lower (Table 3.5).

Table 3.5. Relationship between observed densities and those predicted by HABSCORE HQS for 0+ trout, >0+ trout (<20 cm) and >0+ trout (>20 cm) in the rivers Holme and Ribble. Habitat data were divided into pre (2003 (2002 and 2003 fish density data)), short term post (2006 (2004- 2006 fish density data)) and long term post (2013 (2007-2009 and 2012 and 2013 fish density data)) flow change. +/- = density higher/lower than predicted. Shaded area represents sites where the observed population was significantly higher (blue) or lower (red) than would be expected under pristine conditions. NS denotes fish and HABSCORE not surveyed (Adapted from Bolland *et al*., 2011).

River	Site	0+			>0+ <20 cm			>0+ >20 cm		
		Pre	Short Post	Long Post	Pre	Short Post	Long Post	Pre	Short Post	Long Post
River Holme	D1	+	-	-	+	+	+	+	+	+
	D2	-	-	-	+	+	+	+	+	+
	D3	-	-	-	+	+	+	+	+	+
	D4	+	-	-	+	+	+	+	+	+
	D5	+	-	+	+	+	+	+	+	+
	D6	+	-	+	+	+	+	+	+	+
River Ribble	R30	NS	-	+	NS	+	+	NS	-	-
	R31	-	-	+	+	+	+	+	-	-
	R32	-	-	-	+	+	-	-	+	-
	R33	-	-	-	+	+	+	+	+	-

Pre flow change observed densities of >0+ brown trout (<20 cm) in the River Ribble were higher at all sites and significantly so at R31 and R33. In the short term post flow change 0+ brown trout (<20 cm) at all sites were higher than predicted but not significantly. Three of the four sites on the River Ribble (R30, R31 and R33) had higher 0+ brown trout (<20 cm) densities (Table 3.5), the reminding site R32 had a density lower than predicted by the HQS, but not significantly.

Observed densities of >0+ brown trout (> 20 cm) pre flow change in 2004 at R32 in the River Ribble were lower, but not significantly, than predicted from the HQS (Table 3.5). R31 and R33 were higher than predicted but not significantly. In the short term post flow change, >0+ brown trout (> 20 cm) were lower than predicted at R30 and R31 but higher at R32 and significantly higher at R33. River Ribble >0+ brown trout (> 20 cm) densities at all sites were lower than predicted by the HQS, and were significantly lower at sites R31 and R32 (Table 3.5).

3.4 DISCUSSION

3.4.1 Overview

The modified compensation flow regimes on the rivers Holme and Ribble resulted in seasonally adjusted flows being released from Holme Styes, Digley and Brownhill reservoirs. The introduction of seasonal compensation and freshet flows appears to have had no effect on variability in long term hydrological trends, fluctuations in fish population density estimates and relationships between observed brown trout densities and that predicted by the available habitat quality (HABSCORE HQS). Any differences observed were more than likely attributable to natural variability, drivers of this include, extreme climate evets, disease, predation and competition (Jonsson & Jonsson, 2011).

3.4.2 Hydrological trends

Flow is one of the main influences on the functioning of aquatic ecosystems (Lytle & Poff, 2004). Differences in the flow may effect physical, biological and chemical processes, including sediment transport, aquatic species emergence and nutrient flux (Jonsson & Jonsson, 2011; Deitch & Kondolf, 2012).

Despite being a regulated river, it was apparent from the hydrology of the River Holme was highly variable, suggesting that the river flow is influenced by the catchment, for example after rainfall events surface water runs off into the main river channel contributing to the high variation in flow in the River Holme. Freeze (1972) described four different components of total runoff: i) precipitation, the lesser of contributors, is when rain falls directly into the channel; ii) overland flow where runoff does not filtrate the substrate; iii) interflow where water filtrates into the soil and moves through the upper layers of the soil to the river; and iv) groundwater flow which takes longer to reach the river channel as it percolates deep through the soil. These components should be taken into consideration together with rainfall events, catchment size, geology and topography, to gain greater understanding of the catchment influence on the flow in the rivers Holme and Ribble. The location of the River Holme means that the river is fairly responsive to rainfall events and has a short lag time as a result of surrounding valleys, especially in the winter period when the soils are saturated. In addition when reservoirs are full, usually in the winter period, excess water will exit the reservoir into the receiving river, further contributing to the flow variation in the River Holme. When flow gauge data from Queens's Mill was analysed using IHA software, the outputs revealed that the overall flow varied throughout the 18 years examined and all ecological flow components were observed. The main characteristics of flow were dominated by low flows with frequent high flow pulses. The range of hydrological events experienced in the River Holme is critical to maintaining fish habitats and providing hydrological cues for salmonids (Jonsson & Jonsson, 2011).

The seasonal compensation flow had little influence on the hydrology. This was supported by the IHA analysis comparing hydrological data from 9 years pre and 8 years post compensation flow change. Monthly median flows and flow exceedance curves were comparable between pre and post flow changes. It is likely that the changes made to the reservoir flow releases were not that extreme, especially from Digley and Brownhill reservoirs. This could explain why there were no detectable changes in hydrological flow downstream at Queen's Mill. Digley and Brownhill reservoirs summer compensation flow decreased by around 1.4 ML day^{-1} and winter compensation flows from both reservoirs were the same as the original compensation flow apart from an additional freshet flow in November. The flow release programme from Holme Styes Reservoir changed the most in comparison to the other two reservoirs. The summer compensation flow at Holme Styes Reservoir more than doubled the daily amount from 0.89 ML day^{-1} to 1.98 ML day^{-1} and in the winter it was 4.5 times higher (4 ML day^{-1}) plus a freshet flow in November. Although there were no differences detected in the hydrology at Queen's Mill, it could be possible that there were hydrological differences directly downstream of the reservoirs.

3.4.3 Brown trout population trends

The introduction of seasonal compensation flows from Holmestyes, Digley and Brownhill reservoirs and the freshet releases were intended to improve the ecological status and biological classification of the river Holme and Ribble. Potential effects on brown trout were considered prior to modification. An increase in winter flow in the River Ribble from Holmestyes Reservoir could improve spawning habitat for brown trout by preventing silt accumulation in gravels and provide sufficient water flow through gravels for brown trout egg and alevin development in redds (Cowx *et al.,* 2004; Haywood & Walling, 2006; Sear *et al*., 2008), potentially increasing brown trout densities. The introduction of a freshet event from Holme Styes, Digley and Brownhill reservoirs could cause 0+ brown trout to be displaced by the high flows (Acreman & Ferguson, 2010; Jonsson & Jonsson, 2011), potentially having a negative effect on brown trout recruitment, but this was not found. A reduction in summer flow from Brownhill and Digley reservoirs could have several negative effects including, reduced dispersal of juvenile fish, increased sedimentation, increase in water temperature, reduced wetted area and reduced dissolved oxygen levels, but again no impact was found.

Throughout the study period brown trout densities varied between years, with typically younger life stages had greater variability in densities, Sabaton *et al*. (2008) found that

6 years after the flow was increase 0+ brown trout biomass varied nearly twice as much as adult brown trout. 0+ densities are more variable due to how sensitive they are to any environmental changes and mortality is very high during the early life stages. Swimming capabilities of newly emerged trout are very limited and during periods of high flow brown trout are displaced and washed downstream. As brown trout develop and become adults their repertoire of behavioural responses increases, making them less susceptible to sudden changes in flow (Ayllón *et al*., 2009). Time lagged environmental effects have been highlighted by Daufesne *et al*. (2015). For example the influences on >0+ trout populations can be seen when 0+ populations in previous years have been poor. In 2007 there was a period of high rainfall in the summer which caused vast amounts of flooding throughout the UK. It appeared that 0+ brown trout densities in that year, particularly in the River Holme, were negatively affected and the following year (2008) >0+ brown trout populations were lower than previous years.

Nicola *et al*. (2009) found that the timing, magnitude, duration of extreme water conditions was significantly related to the survival rates of cohorts. In 2005, 0+ brown trout populations were greatest in the rivers Holme and Ribble when the hydrology consisted of periods of low flows and high pulse flows, but no small floods. Small floods were absent in 2008 and 2009, thus based on similar flow patterns to 2005 0+ brown trout recruitment would be expected to be similar in these years. In 2009 the mean 0+ brown trout populations in the River Ribble were highest but this was not the case for the River Holme where he highest densities were in 2005, again reiterating the sensitivity and inability of 0+ brown trout being able to survive flood events.

Overall there was no differences in 0+ and >0+ brown trout densities pre and post flow changes on the rivers Holme and Ribble. Similar results were found by Poff & Zimmerman (2010), after literature review was performed looking at the aquatic response to flow alterations, it was specifically found that the alteration in flow magnitude (including the change in base flow) had no influence on fish abundance following flow modifications. It is possible the reason why significant differences were not found is because there were no significant differences found in the IHA analysis. Other studies have seen increases in fish abundances where there were highly contrasting pre and post flows, for example; following the introduction of environmental flows, Bradford *et al.* (2011), found increases in salmon populations in previously dry streams and only a marginal increase in salmon abundance in previously flowing reaches. And Jowett & Biggs (2006) found that brown trout and rainbow trout (*Oncorhynchus mykiss* (Walbaum, 1792)) abundances were at the highest ever recorded following the introduction of a seasonally variable compensation flow regime in the River Waiau in late 1997, from 450 $m^3 s^{-1}$ (pre 1997) to

12 m^3 s^{-1} in winter and 16 m^3 s^{-1} in summer (post 1997). In this study the minimum compensation flow was the hydrological category that had been influenced the most by the flow change and the number of extreme low flow events were reduced post flow change. Although the minimum compensation and number of extreme flows were not significantly different, the benefits of reducing extreme low flows could have helped maintain brown trout populations over the years, for example; reducing predation and density-dependent effects and maintaining habitat connectivity (Bradley *et al*., 2012).

Joett & Biggs (2006) highlighted how difficult it is to support the importance of flow variability for ecosystem maintenance when there is a lack of published biological evidence and when similar fish communities persist in rivers where flow variability are very different. Brown trout are highly adaptable and exhibit phenotypic plasticity (Klemetsen *et al*., 2003, Jonsson & Jonsson, 2011), which allows the species to change morphological, physiological, behavioural and ecological traits to adapt to changes in environmental conditions such as feeding opportunities, water temperature and current velocity. The ability to adapt can be detected by changes in growth, development rates, reproduction and survival (Stearns, 1992). In this study, growth analysis revealed annual variation of brown trout growth in the rivers Holme and Ribble, but there were no significant difference between pre and post flow change years. Growth is highly influenced by annual mean temperature and densities of juvenile salmonids (Jensen *et al*., 2000). In this study temperature variations from year to year were not recorded, but could be one of the contributors to variations in growth from year to year.

3.4.4 Habitat trends

HABSCORE is a useful tool to assess salmonid populations and whether densities are above or below what is expected based on the habitat available. In this instance HABSCORE was used to assess whether brown trout populations in the rivers Holme and Ribble were above or below expected before and after flow modifications.

On the River Holme, >0+ brown trout populations at many of the sites were significantly higher than predicted, whilst 0+ densities were lower than expected especially following the flow modifications. 0+ densities were on the whole lower than expected especially in the short term but slight improvements were seen at two sites in the long term. In the long term densities of >0+ (<20cm) brown trout were less than pervious possibly reflecting the poorer 0+ densities in the short term period. It also appeared that HABSCORE outputs were less variable for brown trout populations located further downstream.

HABSCORE outputs for 0+ brown trout populations in the River Ribble were, overall, poorer than predicted but for two sites in the long term there were improvements to the densities of brown trout. > 0+ (<20 cm) were within the confidence limits expected under pristine conditions predicted by HABSCORE but significantly so at two of the sites before flow modifications. For >0+ (>20cm) in the long term, trout populations were lower than predicted and significantly so at two of the four sites.

Overall, HABSCORE outputs suggest that flow modifications in the rivers Holme and Ribble have had no discernible impact on brown trout populations relative to the habitat quality at each site. For example, 0+ brown trout in rivers Holme and Ribble were consistently within the confidence limits expected under pristine conditions, and densities of >0+ (<20 cm) and >0+ (>20 cm) brown trout were often higher than predicted before and after the flow modification. Although HABSCORE revealed some negative outputs following the flow change, statistical outputs comparing brown trout density before and after were not significantly different. However, in the short term there was a significant improvement in >0+ brown trout densities on the River Ribble.

3.5 SUMMARY

Despite the rivers Holme and Ribble being HMWB, the hydrology of the rivers appear to be varied and maintaining all aspects of the natural flow regime. It is possible that the natural variation in the flow is sufficient to maintain brown trout populations in the River Holme. This is not to say that the flow changes have not had some form of physical and ecological benefit to the rivers Holme and Ribble. The work carried out highlights that the pre 2004 compensation flow provided similar hydraulic conditions for brown trout as post 2004 seasonal compensation and single freshet event. Recommendations for future study designs and further changes to the reservoir release programme to improve habitat and flow conditions for brown trout in the rivers Holme and Ribble are discussed further in Chapter 7.2.

4 INFLUENCE OF RESERVOIR FRESHET RELEASES ON INDIVIDUAL BROWN TROUT BEHAVIOUR

4.1 INTRODUCTION

All elements of the natural flow regime, including low, medium and flood flows, are considered to be ecologically important drivers for river and floodplain wetland ecosystems and functioning (Poff *et al*., 1997; Hart & Finelli, 1999; Bunn & Arthington 2002; Enders *et al*., 2009; Armstrong & Nislow, 2012), and fish communities rely on this variety of habitats and processes within a river basin (Cowx & Welcomme, 1998; Cowx *et al*., 2004). For example, elevated river levels and floods enable fish to migrate upstream to access and exploit suitable spawning habitat required for the completion of their life-cycles. However, the majority of rivers in the UK and the developed world are impounded in some way, often by water storage reservoirs for potable supply, flood control and hydropower (Vörösmarty *et al*., 2010; Gillespie *et al*., 2015).

Impoundments alter the characteristics of the natural hydrograph and deprive rivers downstream of ecologically important flood flows and sediments (Ligon *et al*., 1995; Richter *et al*., 2003; Acreman *et al*., 2009; Vörösmarty *et al*., 2010) (Chapter 2.3.1). Maintaining all aspects (magnitude, frequency, duration, timing and predictability) of the natural flow regime downstream of reservoirs is vital for maintaining healthy river ecosystems and sustaining habitats for fish (Galat & Lipkin, 2000; Lytle & Poff, 2004; Armstrong & Nislow, 2012) (Chapter 2.5.1). Adjusting the reservoir outflow to improve environmental quality during defined periods, referred to as 'environmental flows', is essential for water resource management to achieve Good Ecological Potential (GEP) and comply with the European Union Water Framework Directive (WFD; 2000/60/EEC) by 2027. Good ecological potential is the ecological quality that can be achieved in the affected water bodies without significant adverse impacts to the benefits provided by the uses, or a significant adverse impact on the wider environment.

Yorkshire Water Services were funded under the Asset Management Planning Cycle 5 (AMP5) to investigate compliance with GEP under the WFD and trialled nine different reservoir outflow mitigation actions to enhance the specific evidence base required for measures in mitigation (MM) 5, establish an appropriate baseline flow regime, and achieve GEP. One of the nine flow trials, Ciii, is to perform enhanced trials activity at sites currently with a freshet release programme. Freshets are included in the Building Block Methodology (BBM) (King, 2000), one of the best known approaches to managing environmental flows from impoundments (Chapter 2.5.1). Freshet releases are intended to simulate a natural high flow event to maintain spawning gravels and trigger fish migrations; especially spawning migrations (Chapter 2.5.1.2). Many studies have investigated reservoir releases and acknowledged the impacts on biota downstream of these structures and highlighted the importance of sustainable water management (Richter *et al*., 2003; Batalla *et al*., 2004; Vörösmarty *et al*., 2010). Few studies have investigated the impact of freshets on resident adult brown trout movements in response to modified flow releases from reservoirs. Research is more focused on anadromous salmonid spawning migrations (e.g. Hawkins and Smith, 1986; Hawkins, 1989; Laughton, 1991; Smith *et al*., 1994; Aprahamian *et al.,* 1998; Solomon *et al.,* 1999; Archer *et al*., 2008) from the sea into fresh water. Knowledge about the life history and migratory behaviour of resident brown trout is well studied (Chapter 2.6), but further studies are required to develop a practical and robust environmental flow guidance for the water industry to ensure they are mitigating the impacts of regulated flows from reservoirs on fish communities including brown trout populations. Brown trout are a model species to study environmental flows, due to their ability to make large distance spawning migrations, unlike other sedentary fish species (bullhead) and macro-invertebrates. They are a BAP species and efforts made to understand flow requirements for brown trout are vitally important.

The aim of the study was to examine, using radio telemetry, movements of brown trout in response to freshet releases from two impounding water storage reservoirs, Brownhill and Digley, near Holmfirth, northern England and determine the appropriate flow building blocks required to encourage spawning migrations at a particular site.

Specific objectives: -

- To identify the general movements of individual brown during autumn and winter, providing a baseline for comparison to the following objective;
- To establish whether the introduction of freshet releases help to promote spawning migration of brown trout by comparing movements between freshet days and control days and control reaches;
- To elucidate the importance of, different frequencies, magnitudes and durations of freshet releases on brown trout movements and how the current flow release programme compared to other studies and UKTAG guidance; and
- To identify any variation in temperature during the study and during freshet releases that could influence behavioural movements in brown trout.

The findings will inform the development and production of robust and workable UK water industry guidance on appropriate mitigation measures for reservoirs controlled by water companies.

4.2 METHODOLOGY

4.2.1 Study area

The study was performed on Marsden Clough downstream of Digley Reservoir, Ramsden Clough downstream of Brownhill Reservoir and the River Holme downstream of the Ramsden Clough and Marsden Clough confluence, near Holmfirth in West Yorkshire, UK (Figure 4.1).

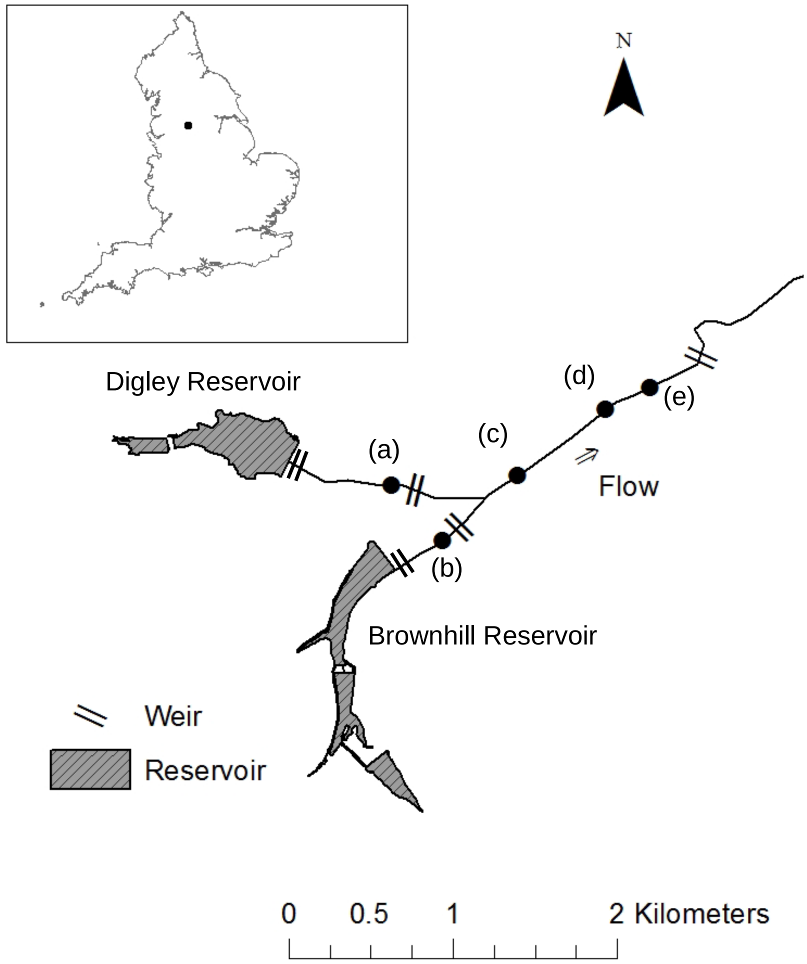


Figure 4.1. A map of England showing the location of the study, and a more detailed map showing the locations of (a) Marsden Clough (downstream of Digley Reservoir), (b) Ramsden Clough (downstream of Brownhill Reservoir), (c) Co-op Lane (d) Mill Pond and (e) Old Mill study reaches on the River Holme.

The frequency, magnitude and duration of freshet releases were altered between the two study years (Figure 4.2). A single small-magnitude (17.3 x Qn95; 69.0 ML/d) with an 8 hour duration freshet was released from Brownhill Reservoir on 15 November 2012; this was an identical flow profile to freshets released annually since 2004. Small- (17.3 x Qn95; 69.0 ML/d) and large-magnitude (122.4 x Qn95; 465.0 ML/d) freshet lasting 8 and

10 hours were released from Brownhill and Digley Reservoirs, respectively, on 16 and 17 October, 11 and 19 November and 11 and 12 December 2013.

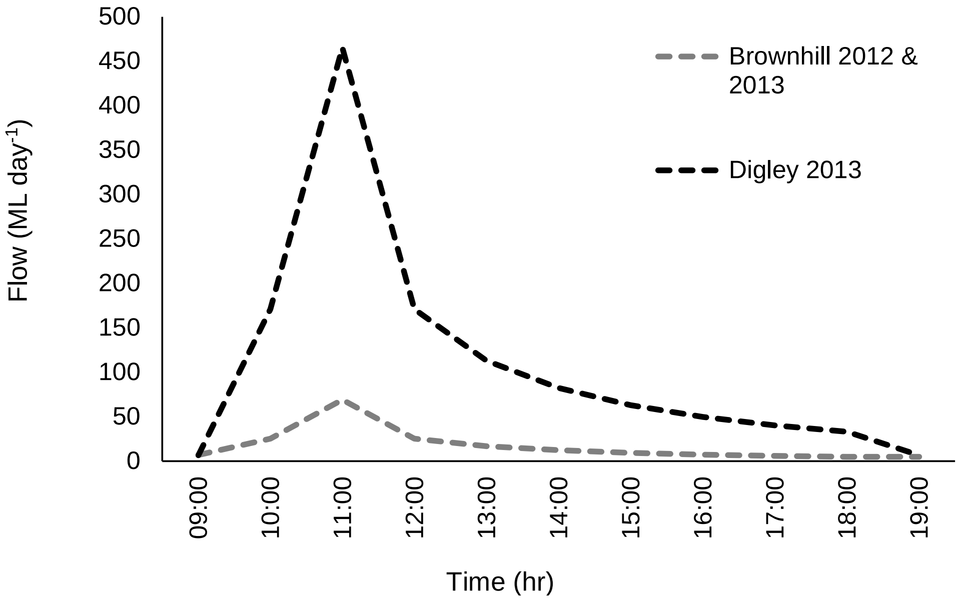


Figure 4.2. Flow profile (ML day^{-1}) of freshet releases from Brownhill and Digley reservoirs in 2012 and 2013.

Flow (m^3 s^{-1}) was measured at 15-min intervals at the Queens Mill gauging weir (SE 14229 15754) during the two tracking periods (mean daily flow = 2012 = 3.23 m^3 s^{-1}± 0.08, October 2013-Feburary 2014 = 1.74 m^3 s^{-1}± 0.004; Figure 4.3). Compensation releases were calculated and implemented by Yorkshire Water Services by using values from inflow measurements (flows coming into the reservoir that exit under a natural system) from 1920 to 2011. Daily Brownhill and Digley reservoir compensation releases (ML day^{-1}) and weekly overtopping data were logged by Yorkshire Water Services (Figure 4.4). It is important to highlight that Brownhill Reservoir compensation data does not include overtopping spill flows; compensation flows are measured in a separate chamber fed by a compensation pipe. Digley Reservoir compensation flows include overtopping spill flows as they exit via the same gauging weir. The flow exceedance curve (ML day^{-1}) was calculated for Digley and Brownhill reservoir compensation releases and flow at Queens Mill (Figure 4.5). Flow exceedance curves illustrate the relationship between the frequency and magnitude of river flow (Vogel & Fennessey, 1995). Q_{95} values can be calculated from the flow exceedance curves and are the average flow for any one day expected to be greater for 95 days in any 100 days in a regulated river, based on compensation flow data or nearest flow gage (ML day^{-1}). These values are a way of standardising the data allowing comparisons to be made between years and rivers. Water released during freshet releases is supplied via bottom draw-off valves at both Digley and Brownhill reservoirs, water temperature was recorded at 15-

min intervals in Ramsden Clough, Marsden Clough and Co-op Lane in 2012 using temperature loggers (Tinytalk, Orion Instruments, Chichester). In 2013/14 water temperature in Ramsden Clough and Marsden Clough was recorded every minute but only every 15 min at Co-op Lane.

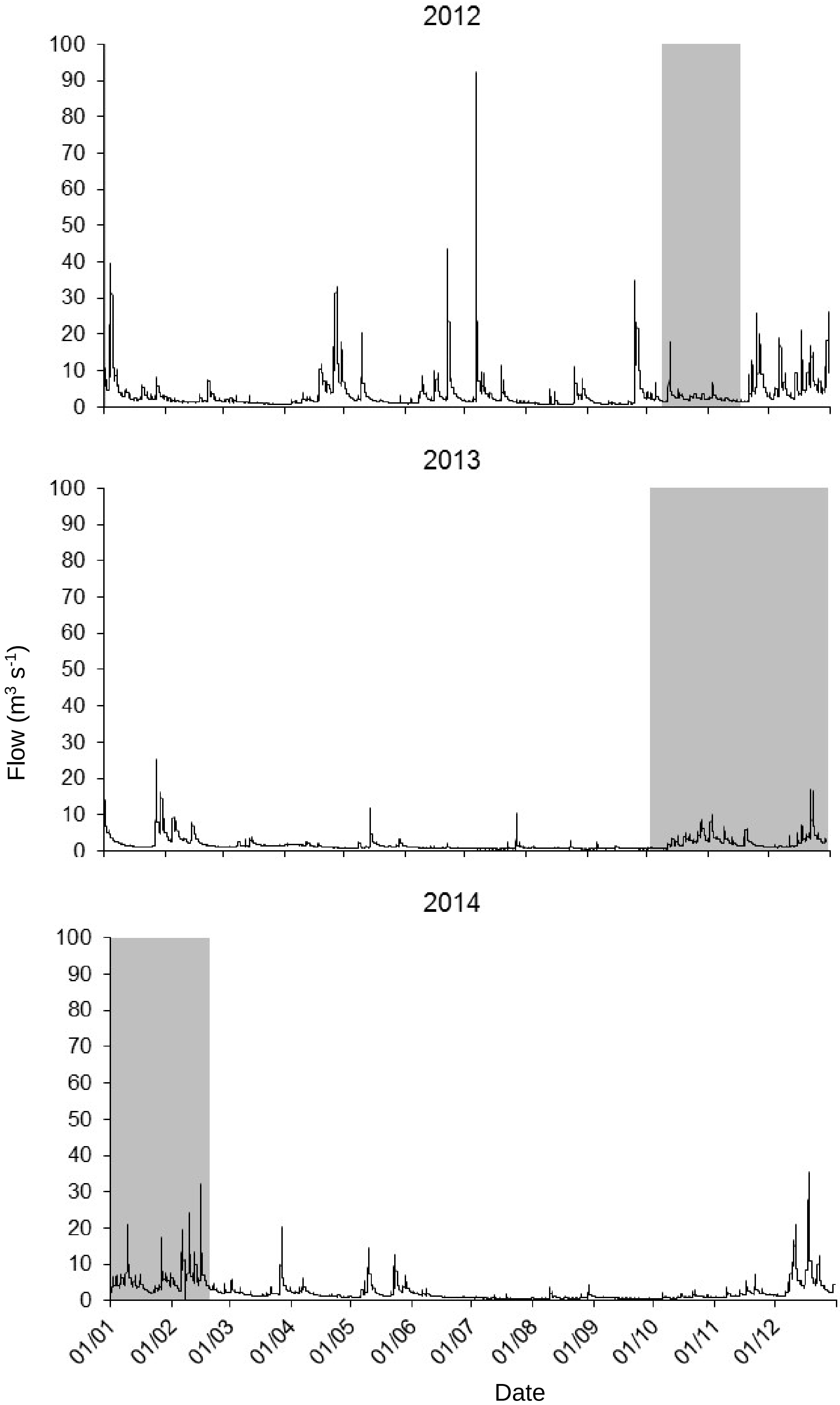


Figure 4.3. Mean daily flow ($m^3 s^{-1}$) (black line) of the River Holme at Queens Mill and period of tracking study (grey) in 2012, 2013 and 2014.

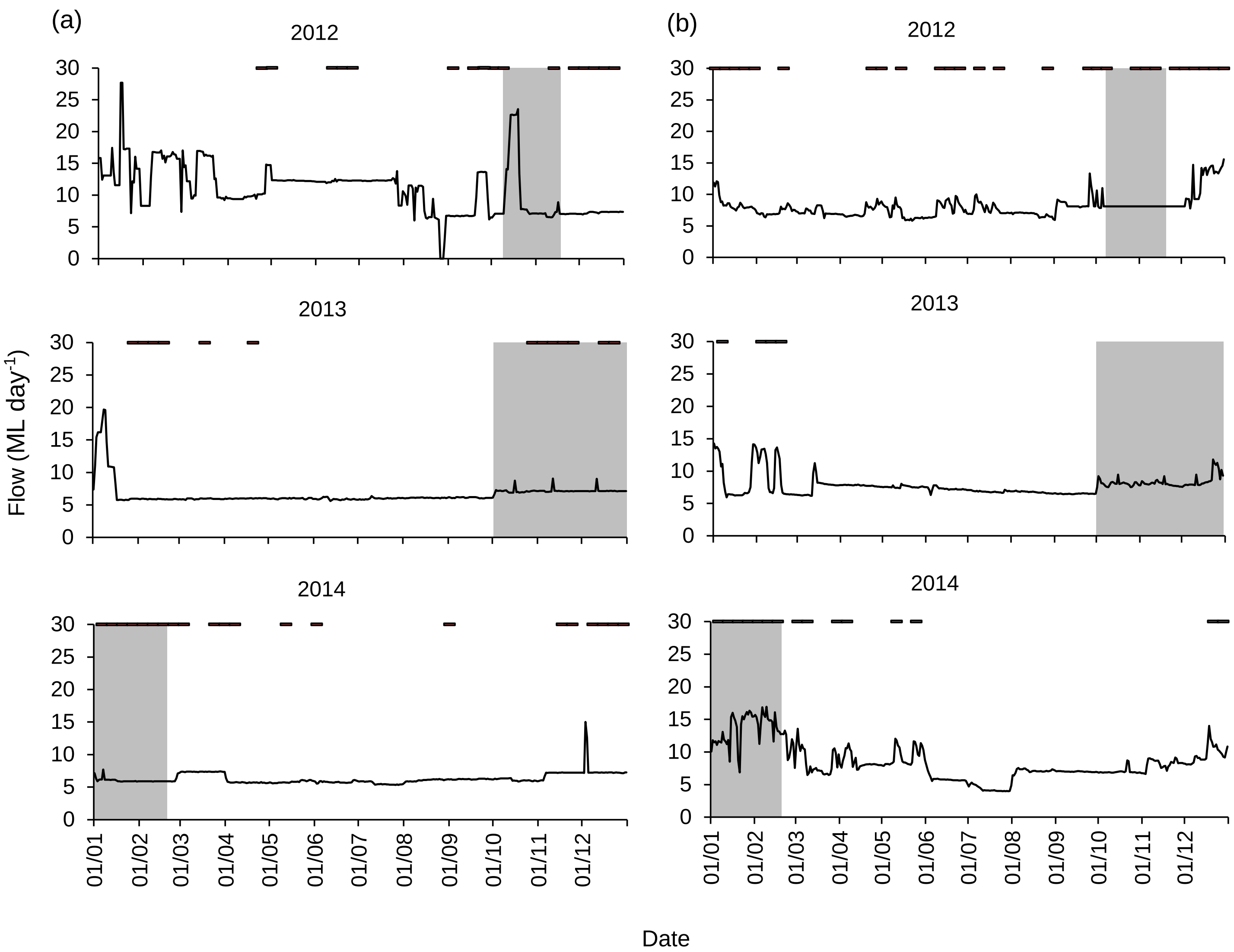


Figure 4.4. (a) Brownhill and (b) Digley reservoir compensation releases (ML day^{-1}) in 2012, 2013 and 2014, including overtopping events (black dashed line) and period tracked (grey).

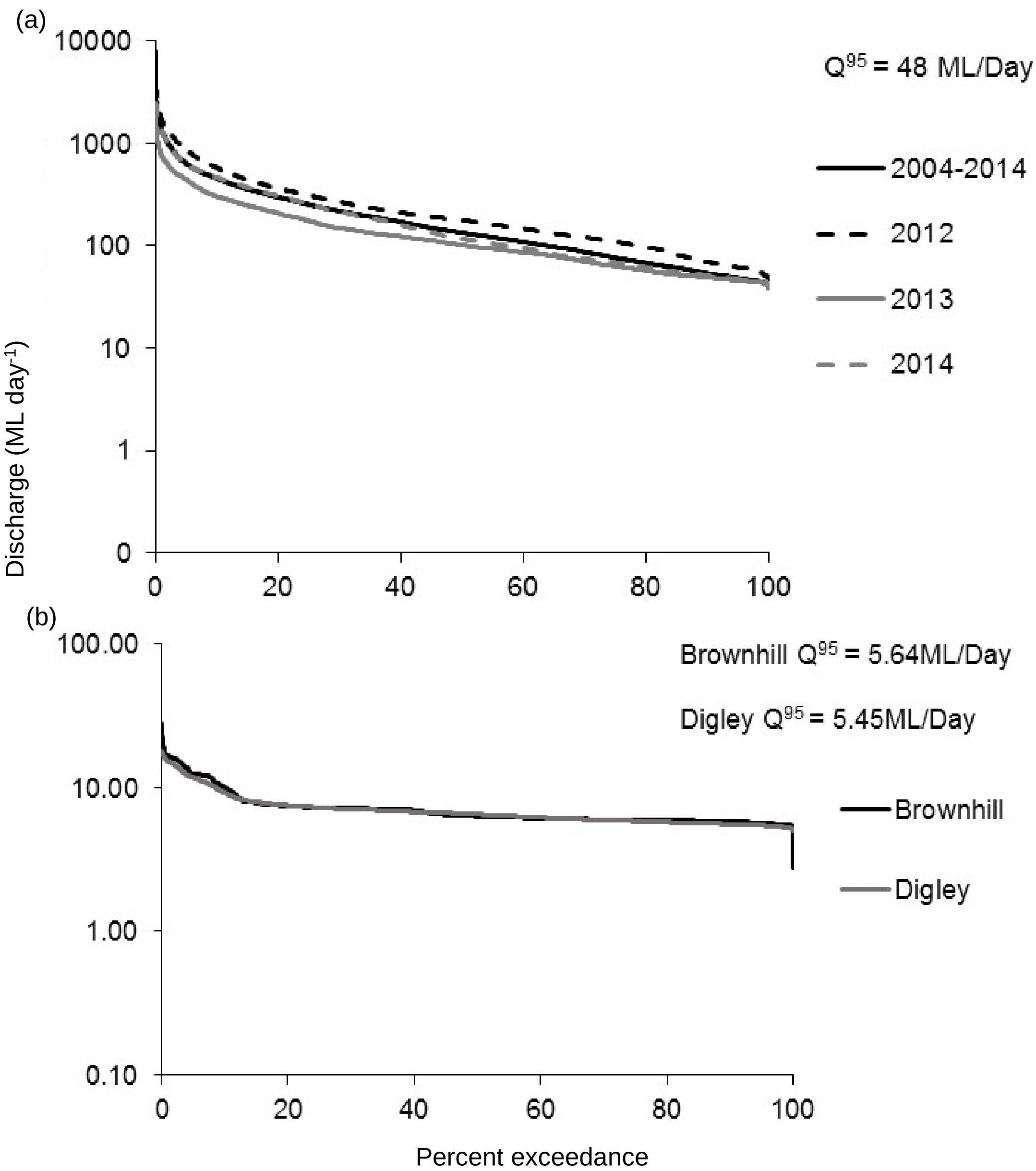


Figure 4.5. Flow exceedance curve for (a) Queens Mill flow gauge in 2012, 2013, 2014 and 2004-2014 and (b) Brownhill (2008 removed) and Digley reservoir compensation releases 2004-2014.

4.2.2 Sampling and tagging procedure

Brown trout were obtained from each study reach in 2012 and 2013/2014 for tagging using electric fishing (Table 4.1), which involved one anode operator and two netsman wading in an upstream direction, with a fourth operator supervising safe operation of the electric fishing equipment. In the first year of study a 2kVA generator was used to power an Electracatch control box producing a 200 V DC, 50Hz, 20% PDC output and 0.3 amps. In the second year of study back pack electric fishing equipment was used (Electracatch 24 V DC input, 200-400 V, 100 W, 50 Hz Pulsed DC).

Prior to tagging in the field, brown trout were anaesthetised using buffered tricaine methanesulphonate (MS-222, 0.1 g L-1). Body mass (g) and fork length (mm) were recorded (Table 4.). Fish were placed ventral side up in a clean V-shaped foam support, fishes gills were irrigated (diluted dose of anaesthetic) during the tagging procedure which lasted between 3-4 minutes. Radio transmitters were sterilised with diluted iodine solution and rinsed with distilled water prior to use.

The radio transmitters used in 2012 (type PIP, 20 x 10 x 5 mm, ~15 cm long, 0.1 mm diameter whip antenna, potted in medical grade silicone, 0.96 g weight in air; Biotrack, Wareham, UK.) had an expected life of 56 days (Figure 4.6). In the 2013-2014 study, the radio transmitters used (type Crystal controlled 2- stage, 15 x 7 x 4 mm, ~12 cm long, 0.1 mm diameter whip antenna, potted in medical grade silicone, 0.9 g weight in air; Advanced telemetry systems, USA) had an expected life of 135 days. Different tags were used in 2013-2014 because the battery life expectancy was longer than the ones previously used in 2012, this allowed for a longer tracking period in second study period. Prior to surgery, the unique frequency (between 173.000 and 173.999 MHz, with a nominal spacing of 10 kHz) of each tag was verified using a hand-operated receiver. An 8-10-mm long, ventro-lateral incision was made anterior to the muscle bed of the pelvic fins and the whip antenna was run via the incision in the body cavity to the exterior, posterior to the pelvic fins using a shielded needle. The transmitter was then inserted into the body cavity. The incision was closed with an absorbable suture. During the tagging procedure it was noted that the majority of tagged brown trout appeared to be male due to the milt they were producing. After the tagging procedure each brown trout was placed in a holding tank to recover. Only when the fish were fully recovered from the anaesthetic were they released back into the river at the approximate site of capture (Table 4.1). All fish were treated in compliance with the UK Animals (Scientific Procedures) Act 1986 Home Office licence number PPL 60/4400), all tagging was carried out by HIFI.

Figure 4.6. Picture of a radio tagged brown trout with a visual of the whip antenna

4.2.3 Radio tracking of brown trout

Throughout the study radio tagged brown trout were located from the bank (avoiding disturbing the brown trout) using a hand-operated receiver (Sika model, Biotrack, Wareham, UK) and a three-element Yagi antenna. The location of individual brown trout was determined to within 1 metre by triangulation. Once located, the position of individual brown trout was recorded using location marker was placed on the river bank. The distance brown trout moved (to the nearest metre) between successive tracking trips was measured using a measuring tape and the location marker was relocated to the new position.

In 2012, brown trout were located daily (between 09.00 and 16.00) over a 40-day period (6 October – 16 November 2012), and every 30 minutes during the Brownhill Reservoir freshet release (15 November 2012) and a control day with no freshet release (14 November 2012) (Figure 4.7).

In 2013/2014, brown trout were located for five days after tagging (2-3 October 2013), three days prior to and after each freshet release and weekly at all other times until 19 February 2014 (Figure 4.7). Brown trout at Marsden Clough, Ramsden Clough and Mill Pond were also located hourly during Brownhill (16 October, 11 November and 11 December 2013) and Digley (17 October, 19 November and 12 December 2013) reservoir freshet releases and a control day with no freshet release (10 December 2013).

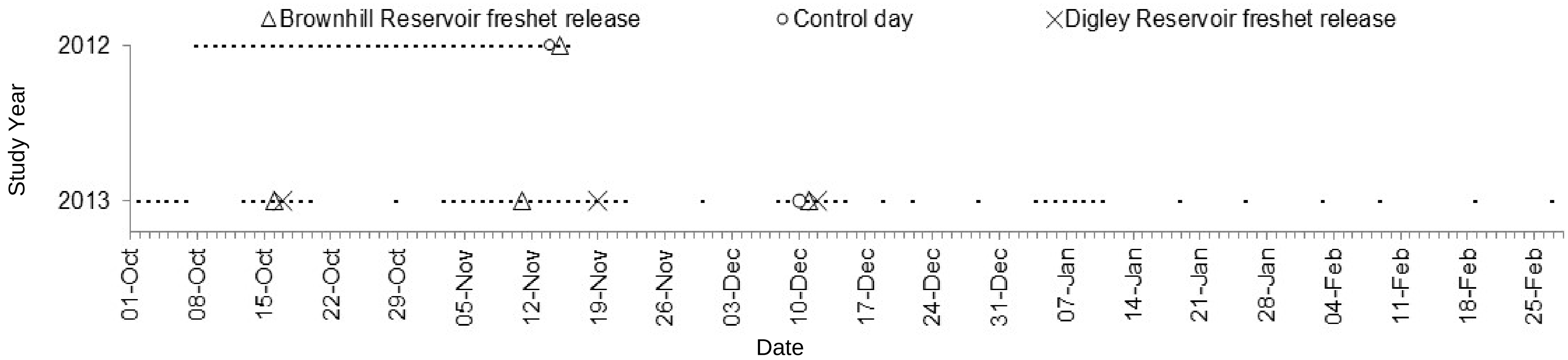
4.2.4 Missing brown trout / tag failure

It was not possible to locate all brown trout throughout the tracking period in each year of the investigation. In 2012, two brown trout were found dead, one tag failed (erratic beep frequency or weak signal) and 16 brown trout could not be found due to tag failure, predation or moving outside of the study reach (Table 4.1). In 2013/2014, three tags failed and a further 20 brown trout could not be found (Table 4.1).

Table 4.1. Number (n), date tagged and released, length (mean ± SD (range), mm), mass (mean ± SD (range), g), ratio (%) of tag weight of brown trout, release location National Grid Reference (NGR) and number of tags failed/lost brown trout.

Study year / Site name	n	Date	Brown trout length (mean ± SD (range), mm)	Brown trout mass (mean ± SD (range), g)	Tag / body wt ratio (mean (range), %)	Release location (NGR)	Tag fail/lost brown trout
2012							
Marsden Clough	15	7/10/12	185.8 ± 19.8 (164 – 238)	76.9 ± 29.0 (51 – 162)	1.4 (0.6 – 1.9)	SE 11564 06792	5
Ramsden Clough	15	7/10/12	198.5 ± 27.9 (163 – 256)	92.0 ± 41.4 (51 – 190)	1.2 (0.5 – 1.9)	SE 11949 06535	7
Co-op Lane	15	8/10/12	199.5 ± 21.5 (163 – 240)	96.3 ± 32.9 (53 – 167)	1.1 (0.6 – 1.8)	SE 12434 06934	7
2013/14							
Marsden Clough	10	2/10/13	179.2 ± 7.7 (169 – 196)	64.3 ± 8.2 (55 – 81.2)	1.5 (1.2 – 1.8)	SE 11641 06799	1
Ramsden Clough	10	2/10/13	189.4 ± 14.5 (171 – 222)	76.7 ± 21.7 (54 – 129)	1.3 (0.7 – 1.8)	SE 11949 06535	5
Co-op Lane	10	3/10/13	208.7 ± 16.7 (191 – 241)	102.3 ± 22.3 (76 – 150)	1.0 (0.6 – 1.3)	SE 12393 06916	8
Mill Pond	10	3/10/13	202.4 ± 29.9 (164 – 241)	99.6 ± 40.3 (49 – 148)	1.1 (0.6 – 2.0)	SE 12886 07228	7
Old Mill	10	3/10/13	203.3 ± 23.0 (168 – 239)	93.4 ± 27.6 (52 – 141)	1.1 (0.7 – 1.9)	SE 13192 07410	2

Figure 4.7. Days (dots) brown trout were located and intensive tracking during Brownhill and Digley reservoir freshet releases and control days without a reservoir freshet release in 2012 and 2013.



4.2.5 Data analysis

4.2.5.1 Brown trout movements

Analysis of spatial behaviour of radio-tracked brown trout during the full study and during freshet releases was based on a range of descriptors of the pattern and extent of movements (Table 4.2). All statistical analyses were carried out using SPSS version 22.

Table 4.2. Definitions of terms used when analysing data.

Term	Definition
General descriptors	
Home range	Calculated using the distance between the furthest upstream and downstream position recorded for each individual during the whole study period.
Range per day tracked	Calculated by dividing the home range by the number of days tracked, which describes the extent of river used, standardised for the period of tracking.
Total distance moved during the study period	Calculated by adding the total distance moved by an individual between each day the brown trout is located.
Daily distance moved	Calculated by dividing the total distance moved during the study period (calculated from the position recorded every day) by the period over which the brown trout was tracked, and reflects the overall level of movement.
Distribution during the study period	The distance from the release location. Positive values were assigned to brown trout located upstream of the release location and negative values to brown trout located downstream of the release location.
Descriptors during freshet releases	
Range during freshet release	Calculated using the distance between the furthest upstream and downstream position recorded for each individual during the freshet release.
Total distance moved during the freshet release	Calculated by adding the total distance covered by an individual brown trout between each location during the freshet release (non-directional).
Distribution after the freshet	Calculated from the location immediately prior to freshet release to the location immediately after the release. Positive values were assigned to brown trout that moved upstream during the freshet release and negative values to brown trout that moved downstream during the freshet release.
Descriptors related to flow	
Qn_{95}	The average flow for any one day expected to be greater for 95 days in any 100 days in a natural river system (ML day^{-1})
Q_{95}	The average flow for any one day expected to be greater for 95 days in any 100 days in a regulated river, based on compensation flow data or nearest flow gage (ML day^{-1})

When possible, parametric tests (independent samples t-tests) were performed but when variances were not equal (Levene statistic $P < 0.05$) non- parametric tests (Mann-Whitney U-tests and Kruskal–Wallis) were used to compare movements of brown trout (home range, range per day tracked, daily distance moved and total distance moved). Spearman rank correlation was used to compare brown trout length with home range, range per day tracked and daily distance moved. Chi-square tests were carried out to test the proportion of brown trout located upstream and downstream immediately after freshet releases in relation to the location occupied prior to the freshet release.

4.2.6 Temperature

Temperature profiles were interrogated using Mann-Whitney *U*-tests to ascertain if there were significant changes in temperature profiles in the downstream reaches (Marsden Clough, Ramsden Clough and River Holme) during freshet releases. Freshet temperature values were compared with control day values, and control days were compared to control days within the time frame of a freshet event (07:00 to 18:00).

4.3 RESULTS

Seven freshets were released from two different reservoirs over two study years. The range occupied and the total distance moved by brown trout during releases as well as the distribution of brown trout after freshets were analysed to assess the influence of freshet timing, magnitude and duration on spawning migrations. The general movements of brown trout at other times and temperature profiles were also considered.

4.3.1 Brown trout length

Brown trout length was not strongly correlated with home range (Spearman rank: 2012: $r = 0.227$, $n = 44$, $P = 0.139$ and 2013/2014: $r = 0.286$, $n = 49$, $P = 0.047$), range per day tracked (Spearman rank: 2012: $r = 0.103$, $n = 44$, $P = 0.506$ and 2013/2014: $r = -0.096$, $n = 49$, $P = 0.512$) or daily distance moved (Spearman rank: 2012: $r = 0.163$, $n = 44$, P

= 0.290 and 2013/2014: r = -0.097, n = 49, P = 0.509) in either study periods (Figure 4.20).

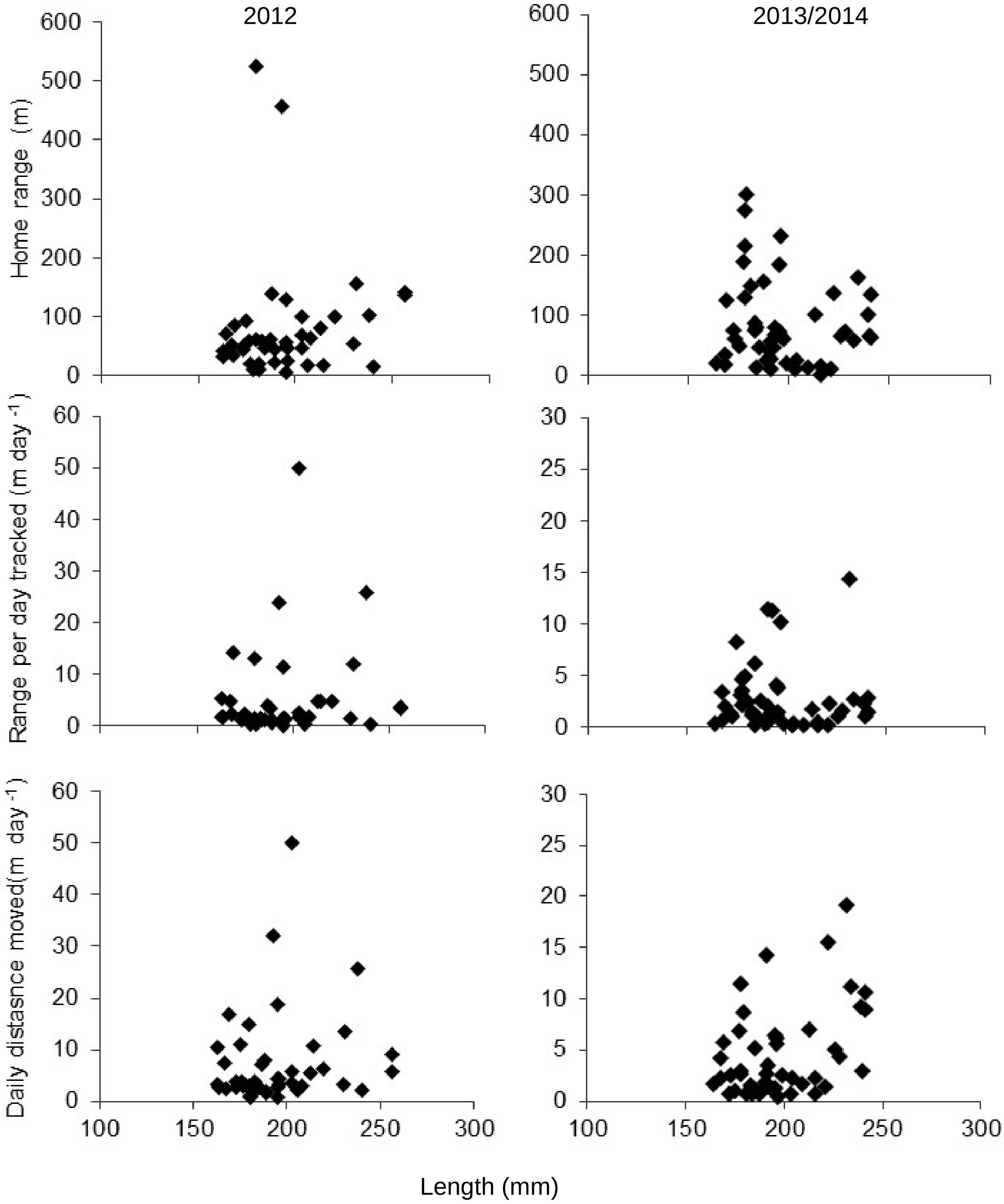


Figure 4.8. Relationship between brown trout length (mm) and home range (m; top), range per day tracked (m day^{-1}; middle) and daily distance moved (m day^{-1}; bottom) in 2012 and 2013/2014.

4.3.2 General movements

4.3.2.1 Daily movements

Radio tracked brown trout generally occupied small home ranges in 2012 (mean = 80 ± 99 m (range = 6 – 525 m)) and 2013/2014 (84 ± 71 m (2 – 300 m)), but there were some exceptions with 525 m and 300 m the largest home ranges in 2012 and 2013/2014, respectively (Figure 4.8a).

The range per day tracked, which describes the extent of river used standardised for the period of tracking, was comparable between 2012 (5 ± 9 m (0 – 50 m)) and 2013/2014 (3 ± 3 m (0 - 14 m)) (Mann-Whitney *U*-tests: $Z = -1.239$, $n = 93$, $P = 0.215$) (Figure 4.8b). The range per day tracked was comparable between study sections in 2012 (Kruskal-Wallis: $n = 44$, $X^2 = 0.294$, *d.f.* = 2, $P = 0.863$), but there was a significant difference in range per day tracked between Marsden Clough and Old Mill (*t*-test, $t = -3.516$, *d.f.* = 18, $P = 0.002$) and Marsden Clough and Co-op Lane (*t*-test, $t = -2.620$, *d.f.* = 18, $P = 0.017$) in 2013/2014 (Figure 4.8b).

Daily distance moved was not comparable between 2012 (8 ± 9 m (1 – 50 m)) and 2013/2014 (5 ± 4 m (0 – 19 m)) (Mann-Whitney *U*-tests: $Z = -2.462$, $n = 93$, $P = 0.014$). Daily distance moved between study sections were comparable in 2012 (Kruskal-Wallis: $n = 44$, $X^2 = 0.13$, *d.f.* = 2, $P = 0.994$) and 2013/2014 (Kruskal-Wallis: $n = 49$, $X^2 = 9.342$, *d.f.* = 4, $P = 0.053$) (Figure 4.8 c).

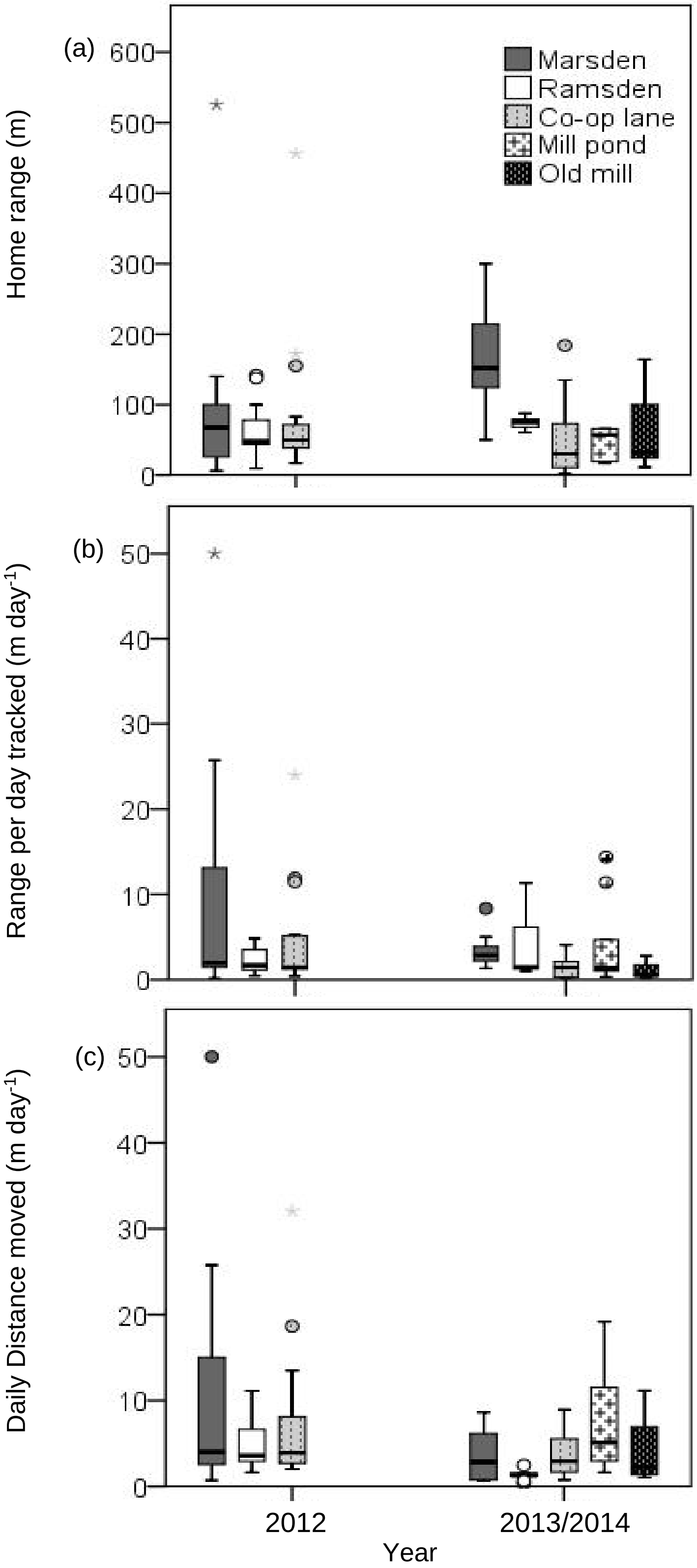


Figure 4.9. (a) Home range (m), (b) range per day tracked (m day^{-1}) and (c) daily distance moved (m day^{-1}) for brown trout in each study reach during 2012, 2013/2014.

4.3.2.2 Weekly movements

The movements of brown trout in the first three days after release were in both upstream and downstream directions (Figure 4.9 – 4.16), with the largest upstream and downstream movements performed by an individual in 2012 and 2013/2014 being 140 m upstream and 103 m downstream, 117 m upstream and 111 m downstream, respectively.

In 2012, the mean proportion of brown trout that did not move location between each day was 49% and brown trout movements were not disproportionately directional on any day, i.e. the number of brown trout moving in an upstream direction was similar to the number moving downstream (Figure 4.17a). The only exception was on 8 November 2012 when 4% of the brown trout moved downstream, 36% did not move and 60% moved upstream, although the largest upstream movement on that day was 5 m. On this particular day the water levels appeared to be low. There were 5 individual brown trout that occasionally performed uni-directional movements greater than 100 m in one day, with the furthest distance moved being 430 m downstream on 22 October 2012 (BT34; Figure 4.11). Of these five brown trout, one brown trout permanently relocated to a different location than previously found for the remainder of the study (BT29) (Figure 4.10), two brown trout returned to the area which they were usually found after one and two days (BT11 (Figure 4.9) and BT20 (Figure 4.10), respectively) and two brown trout relocated to another location 383 m and 430 m further downstream after nine and 15 days, respectively (BT4 (Figure 4. 9) and BT34 (Figure 4.11)). These relatively long-distance movements usually coincided with a period of natural elevated flow (rather than a reservoir freshet release) on 12, 13, 16 and 22 October and 3 November in 2012 (Figure 4.18a). One individual (BT10) relocated to a small side channel after a natural high flow event on 11 October 2012 (Figure 4.9). Temperature declined throughout the study period from 10.3 to 7.0ºC (Figure 4.18a), average daily flow and temperature throughout the study period had a strong positive relationship (r = 0.68, n= 39, P < 0.005) (Figure 4.18a). Despite such individual movements the distribution of all radio tagged brown trout in 2012 was consistently around the release location throughout the tracking period in 2012 (Figure 4.19a).

During the tracking in 2013/2014, the proportion of brown trout that did not move location between tracking weeks averaged 20%, but in weeks 1 and 3 100% of the brown trout moved from the previous weeks location (Figure 17b). The proportion of brown trout moving in a particular direction did not exceed 66 % and the proportion of brown trout moving upstream and downstream each week was similar (Figure 17b). For example, on 11 January 2014 only 3% of the tagged brown trout were found in the location they occupied on 4 January 2014, with 53% of the brown trout moving upstream and 43% moving downstream. The furthest individual upstream and downstream movements were made by the same brown trout (BT44), which moved upstream 184 m on 19 January and returned to the previous location 184 m downstream a week later (Figure 4.16). Greater movements by brown trout, both upstream and downstream, occurred from mid-December onwards and coincided with periods of elevated flow (Figure 4.17b). Two brown trout moved into different side channels; BT11 moved into a side channel after tagging (4 October 2013) and returned to the main river channel during two natural elevated flow events (22 December and 19 February) and once during a Digley Reservoir freshet release (17 October). BT22 moved into a side channel during a periods of naturally elevated flow (BT22; 16 October 2013), BT22 remained in the side channel throughout the rest of the study period. Temperature through the study period declined, the relationship between the average daily flow and temperature had a medium negative correlation (r= -0.49, n= 121, P = <0.005) (Figure 4.18b). The distribution of all radio tagged brown trout during the 2013/2014 tracking was consistently around the release location throughout the tracking period (Figure 19b).

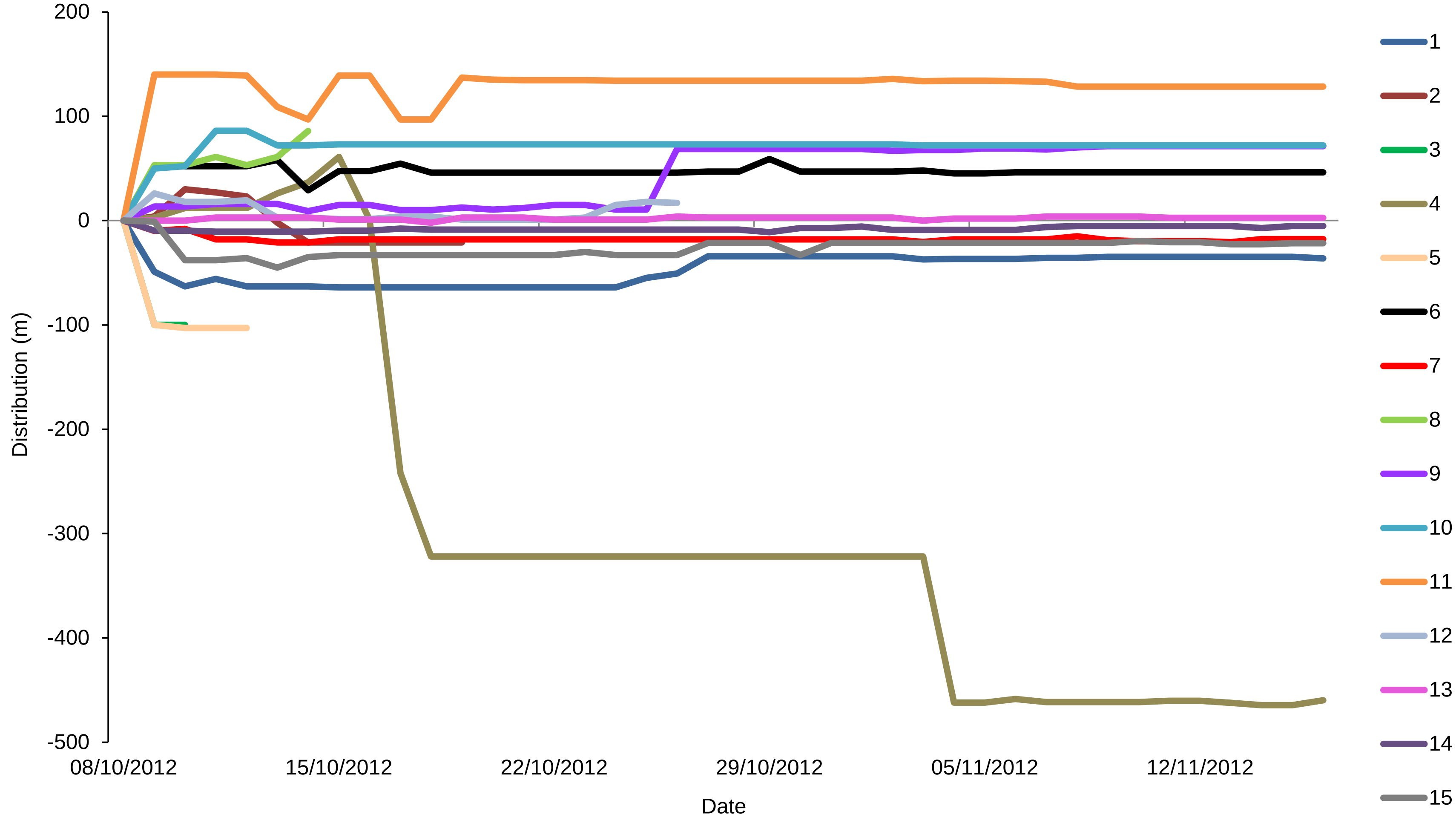


Figure 4.10. Movements (m) of brown trout from release site at Marsden Clough, from 7 October to 16 November 2012.

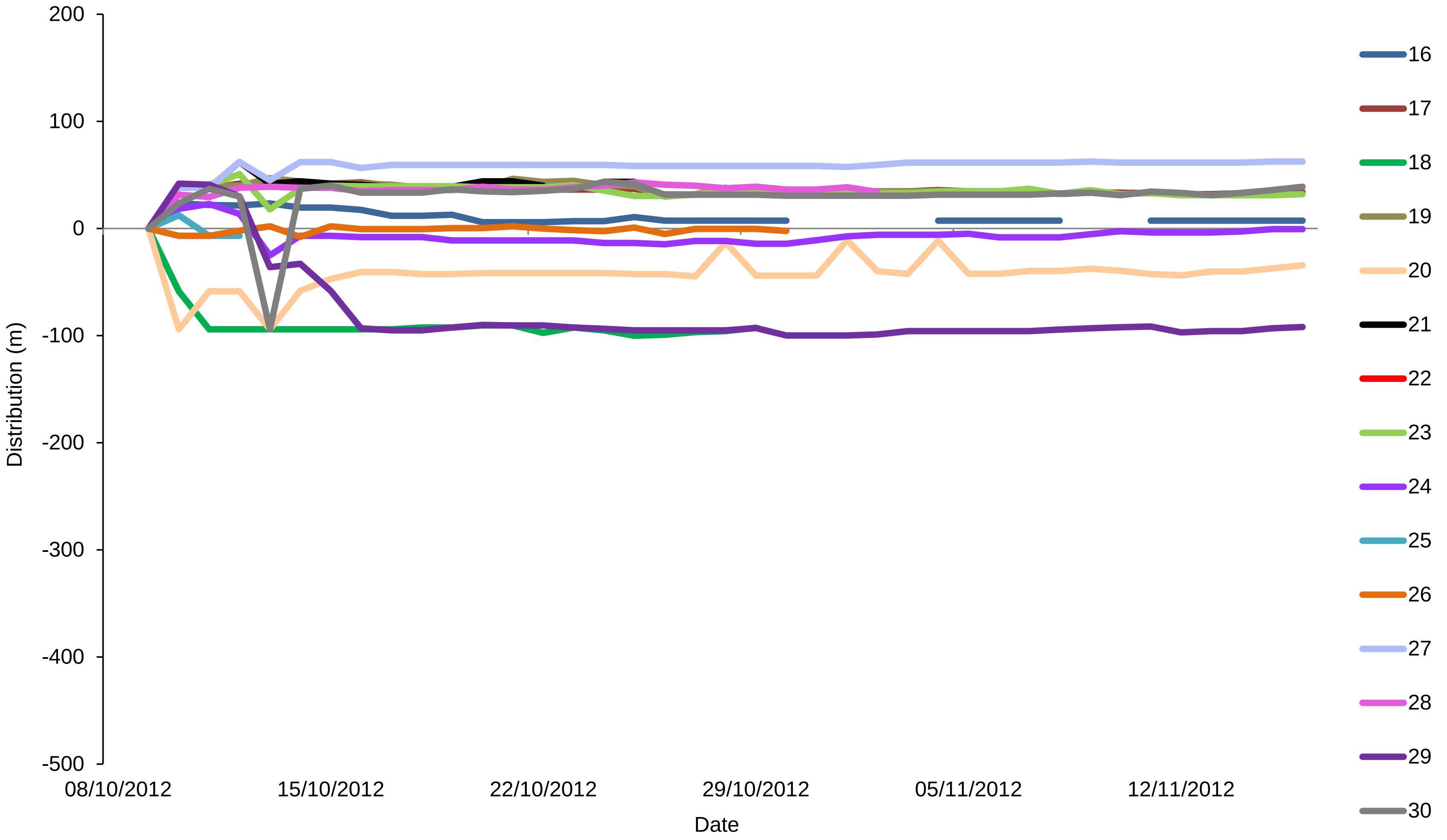


Figure 4.11. Movements (m) of brown trout from release site at Ramsden Clough from 7 October to 16 November 2012.

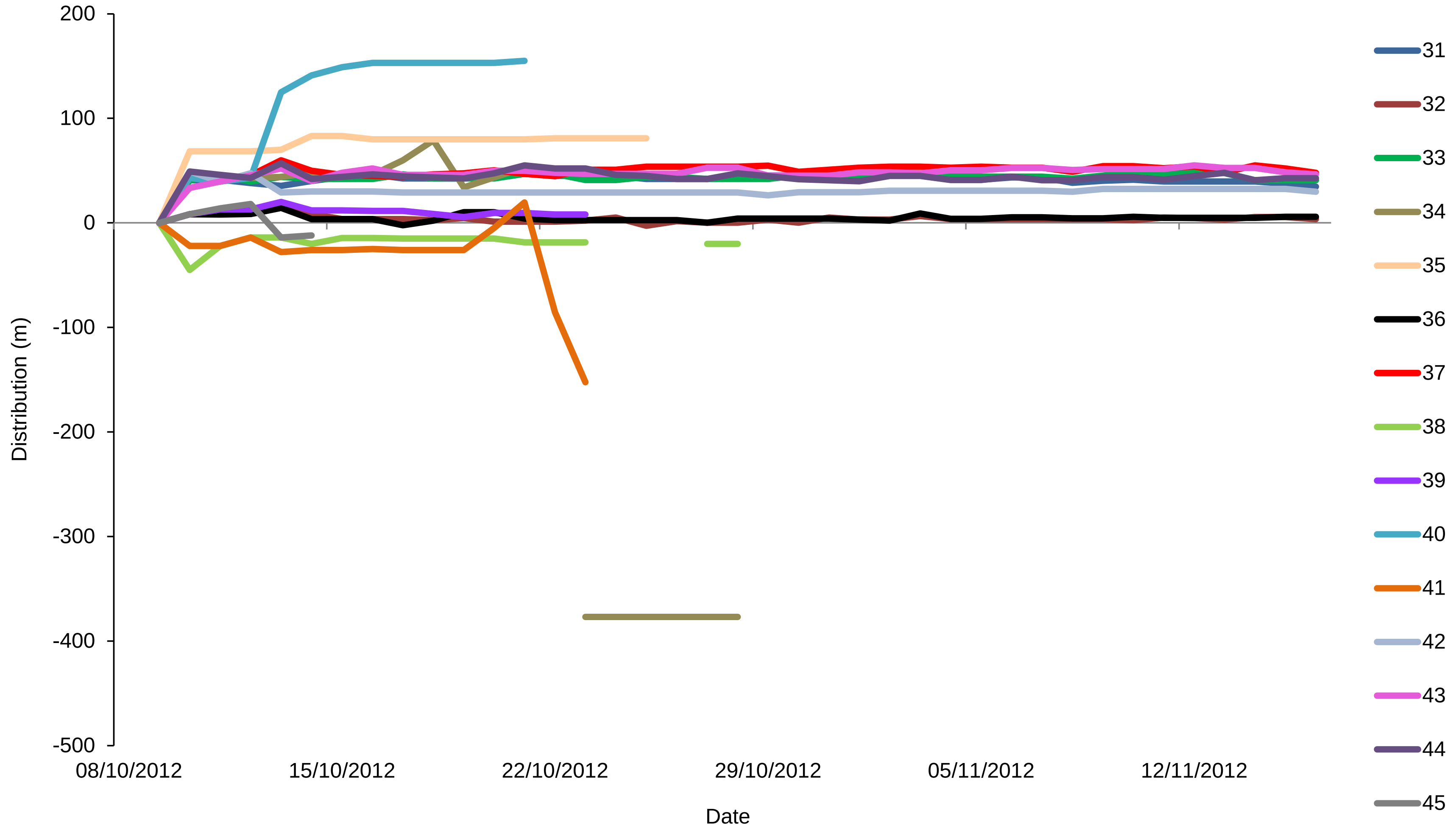


Figure 4.12. Movements (m) of brown trout from release site at Co-op Lane, from 7 October to 16 November 2012.

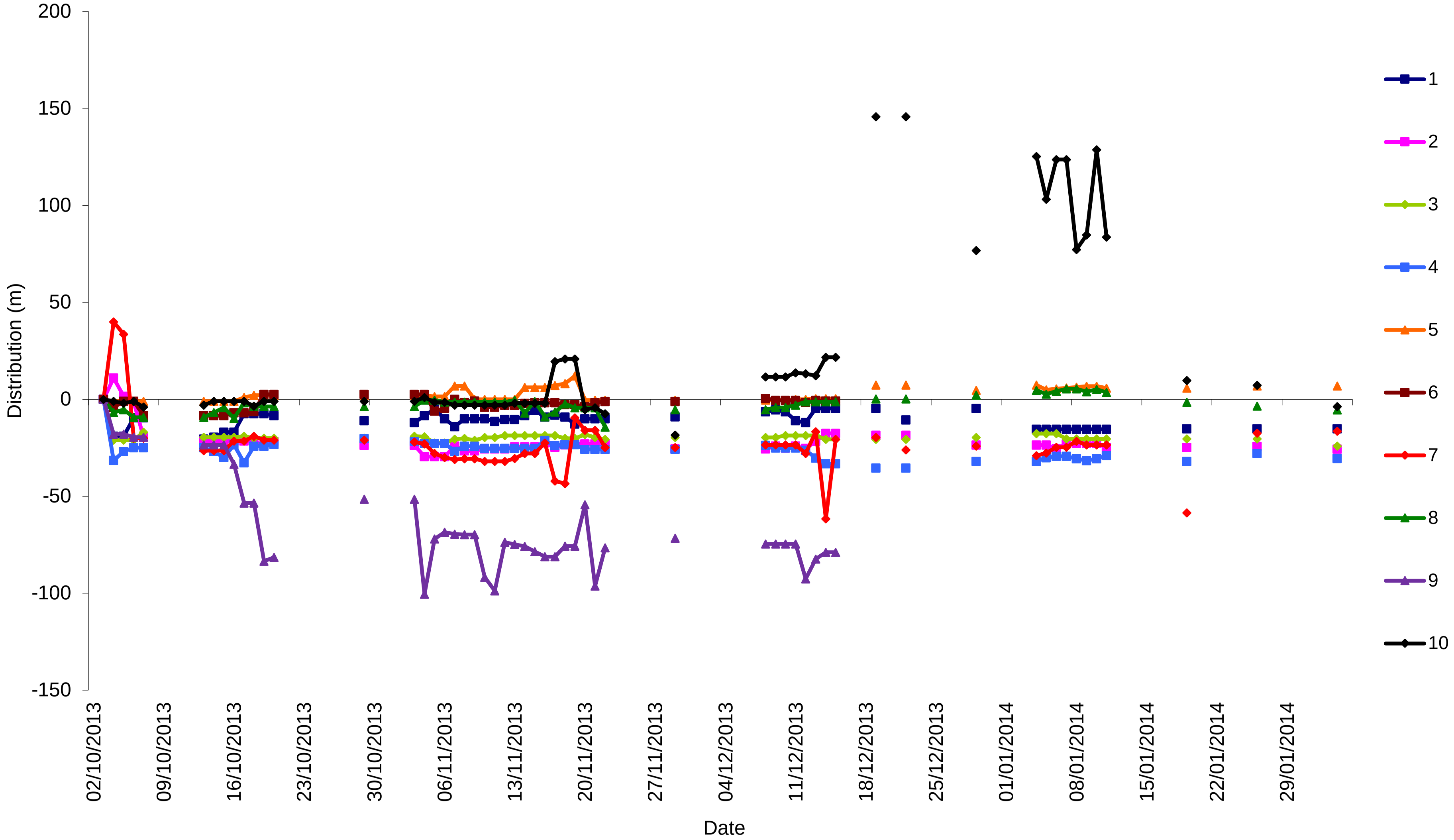


Figure 4.13. Movements (m) of brown trout from release site at Old Mill from 2 October 2013 to 19 February 2014.

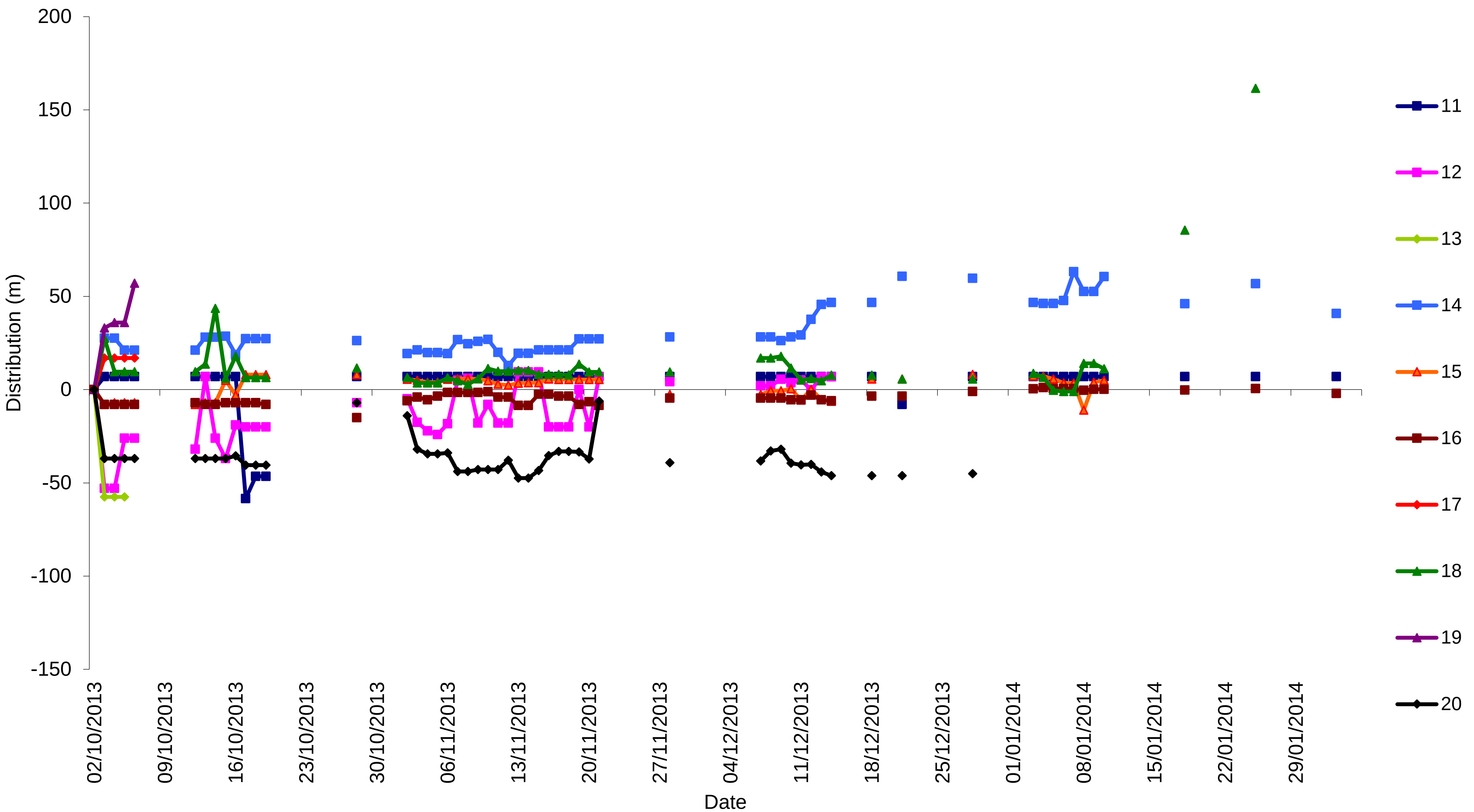


Figure 4.14. Movements (m) of brown trout from release site at Mill Pond from 2 October 2013 to 19 February 2014.

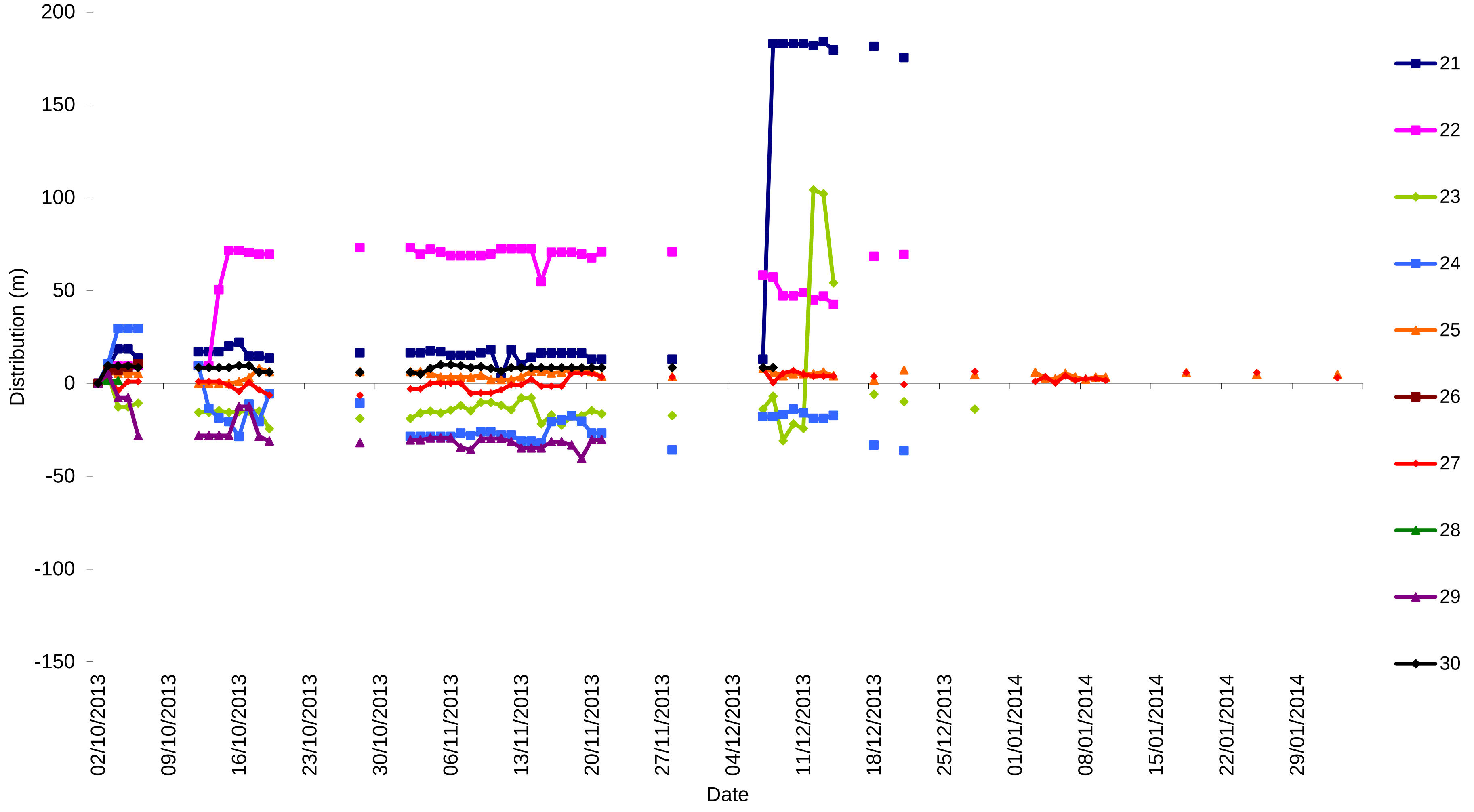


Figure 4.15. Movements (m) of brown trout from release site at Co-op Lane from 2 October 2013 to 19 February 2014.

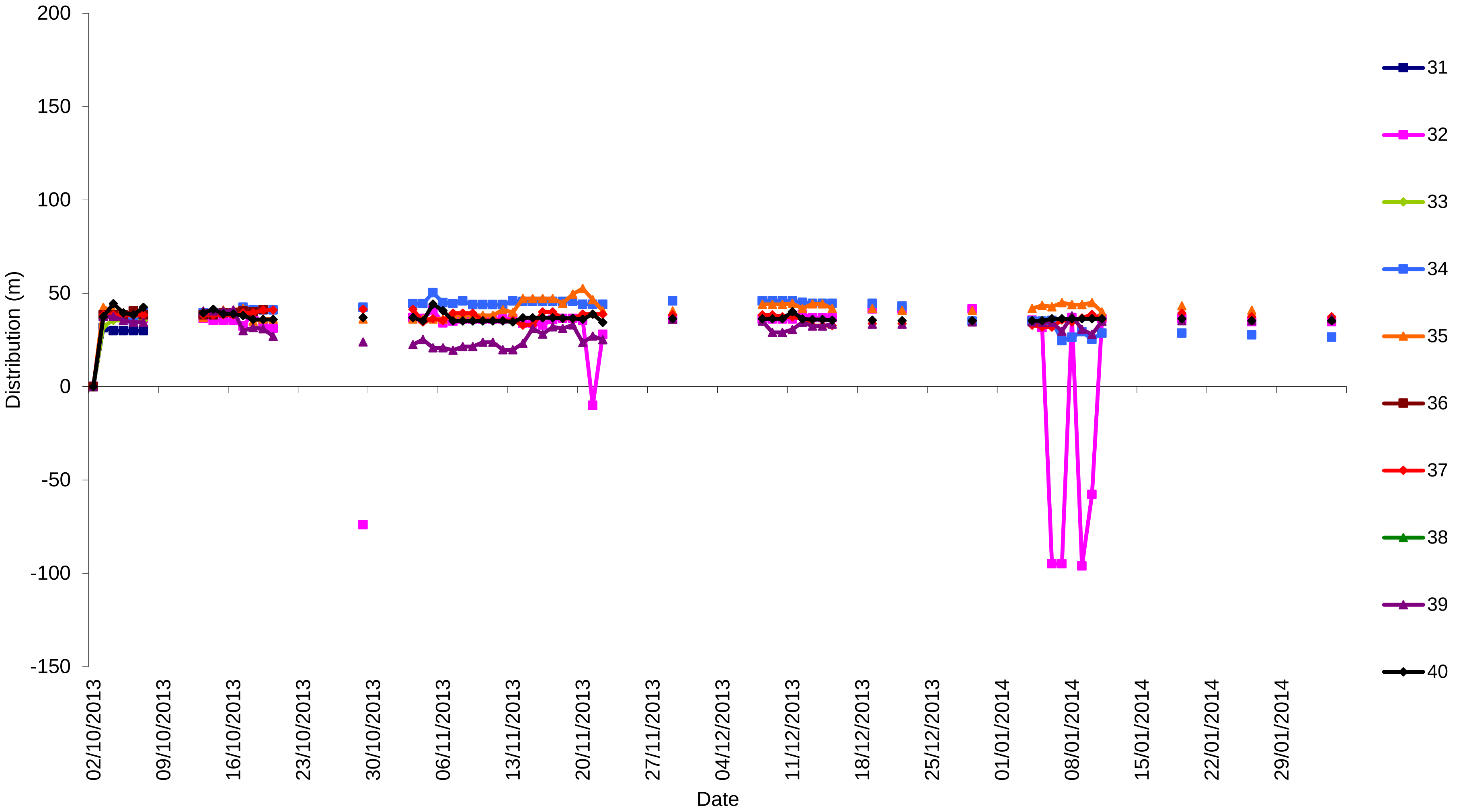


Figure 4.16. Movements (m) of brown trout from release site at Ramsden Clough from 2 October 2013 to 19 February 2014.

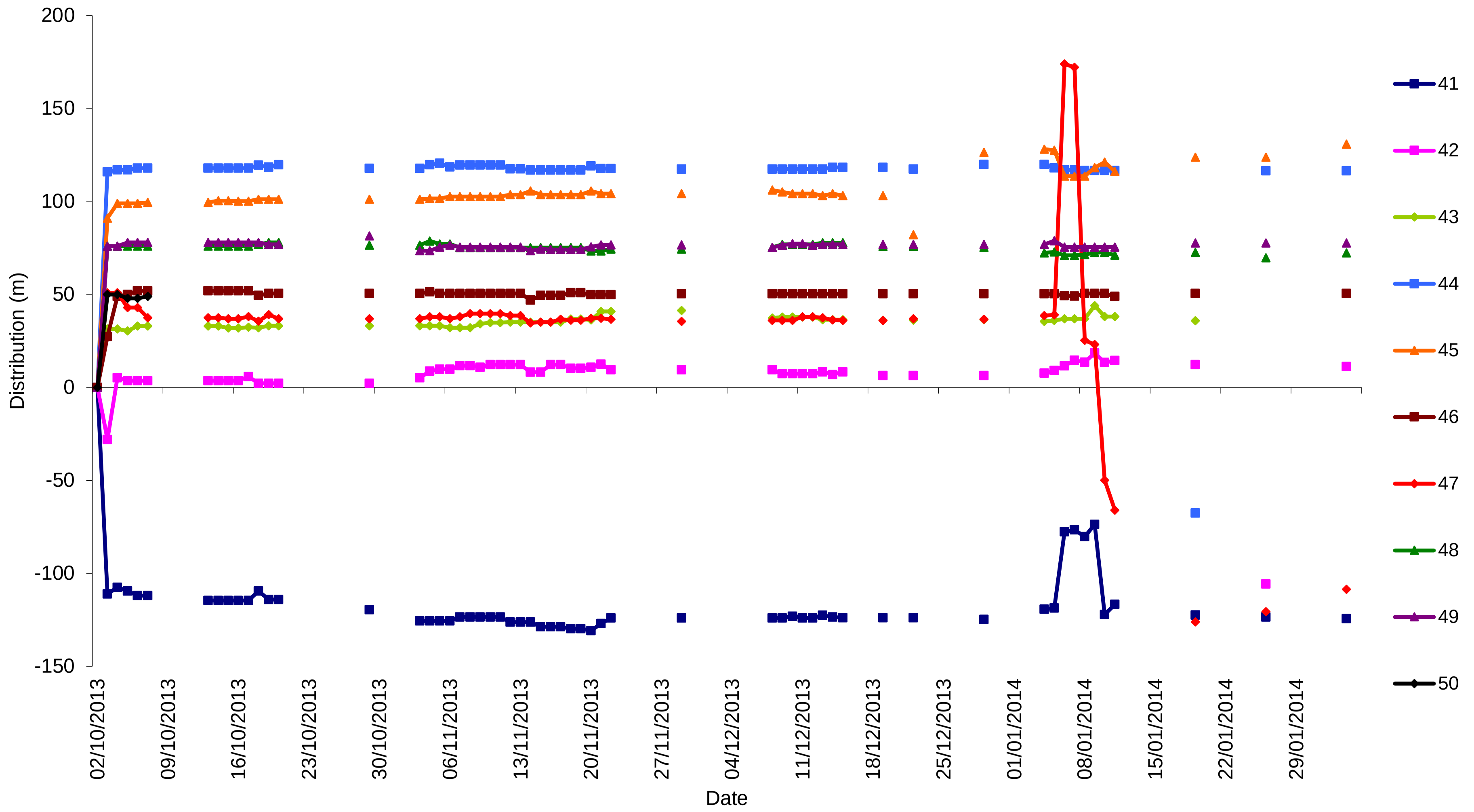


Figure 4.17. Movements (m) of brown trout from release site at Marsden Clough from 2 October 2013 to 19 February 2014.

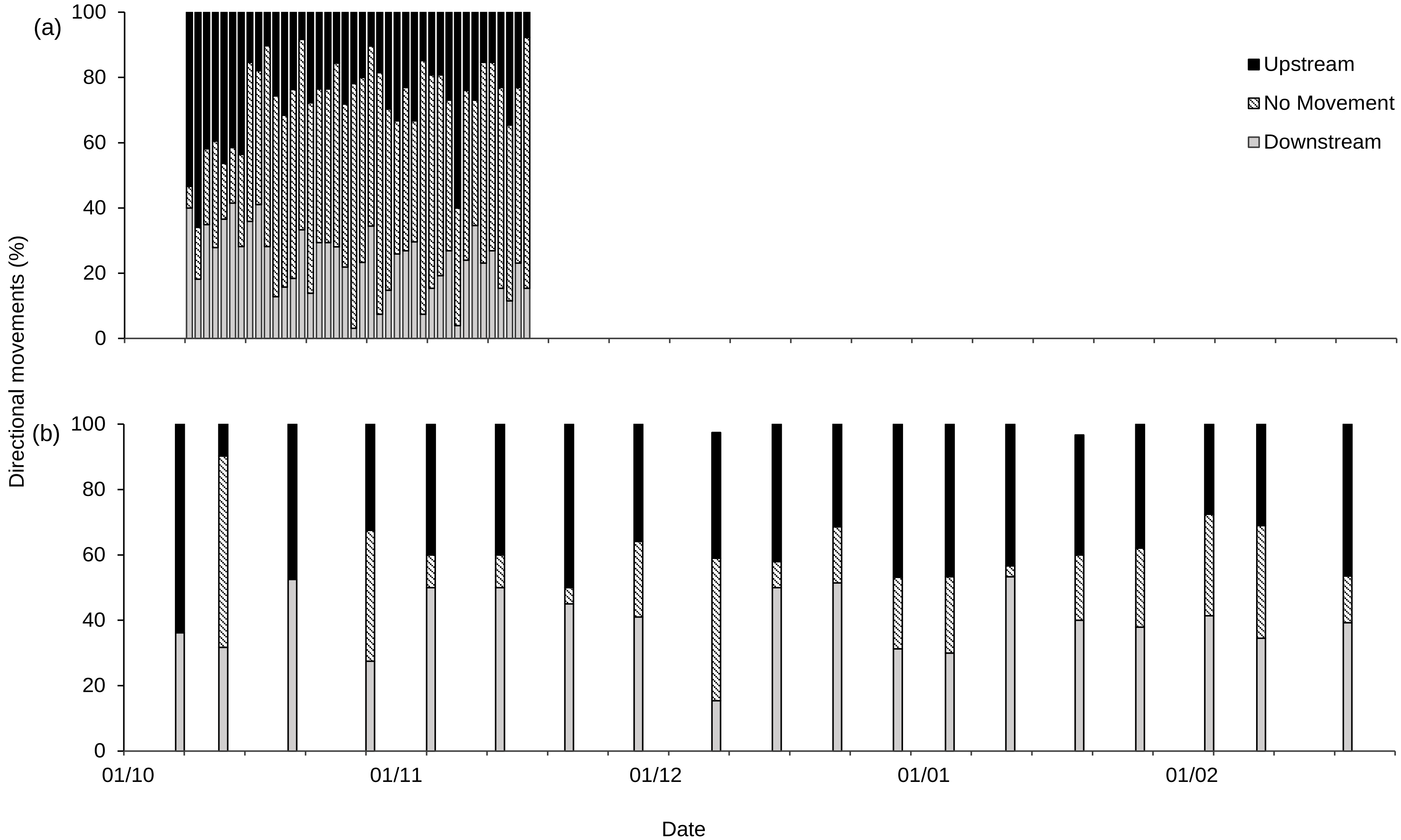


Figure 4.18. Proportion of (a) daily (b) weekly directional movements for all tagged brown trout in (a) 2012 and (b) 2013/2014.

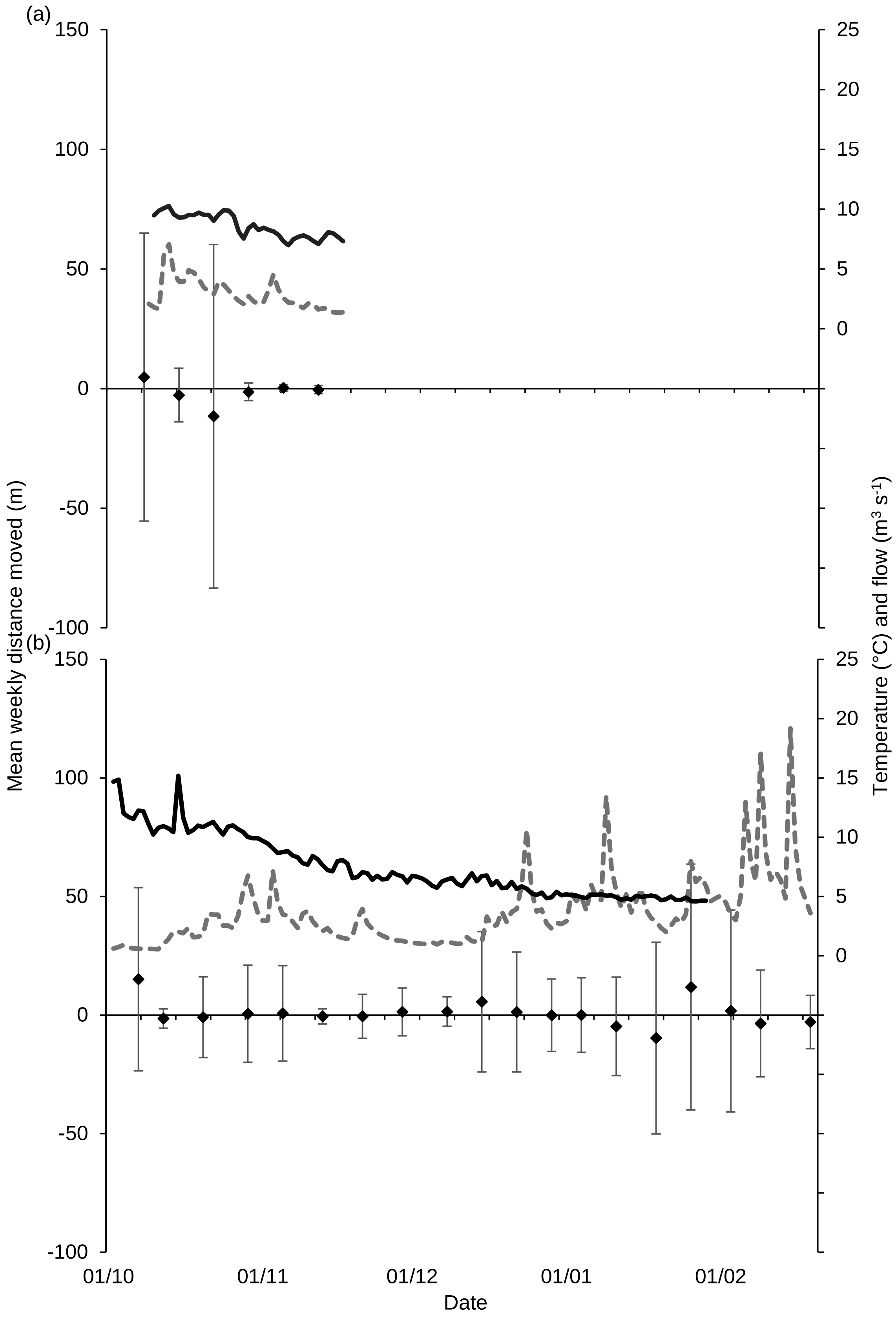


Figure 4.19. Mean daily temperature (°C; solid line), flow ($m^3\ s^{-1}$; dotted line) and mean weekly distance moved (diamond markers ± SD) by all tagged brown trout in (a) 2012 and (b) 2013/2014.

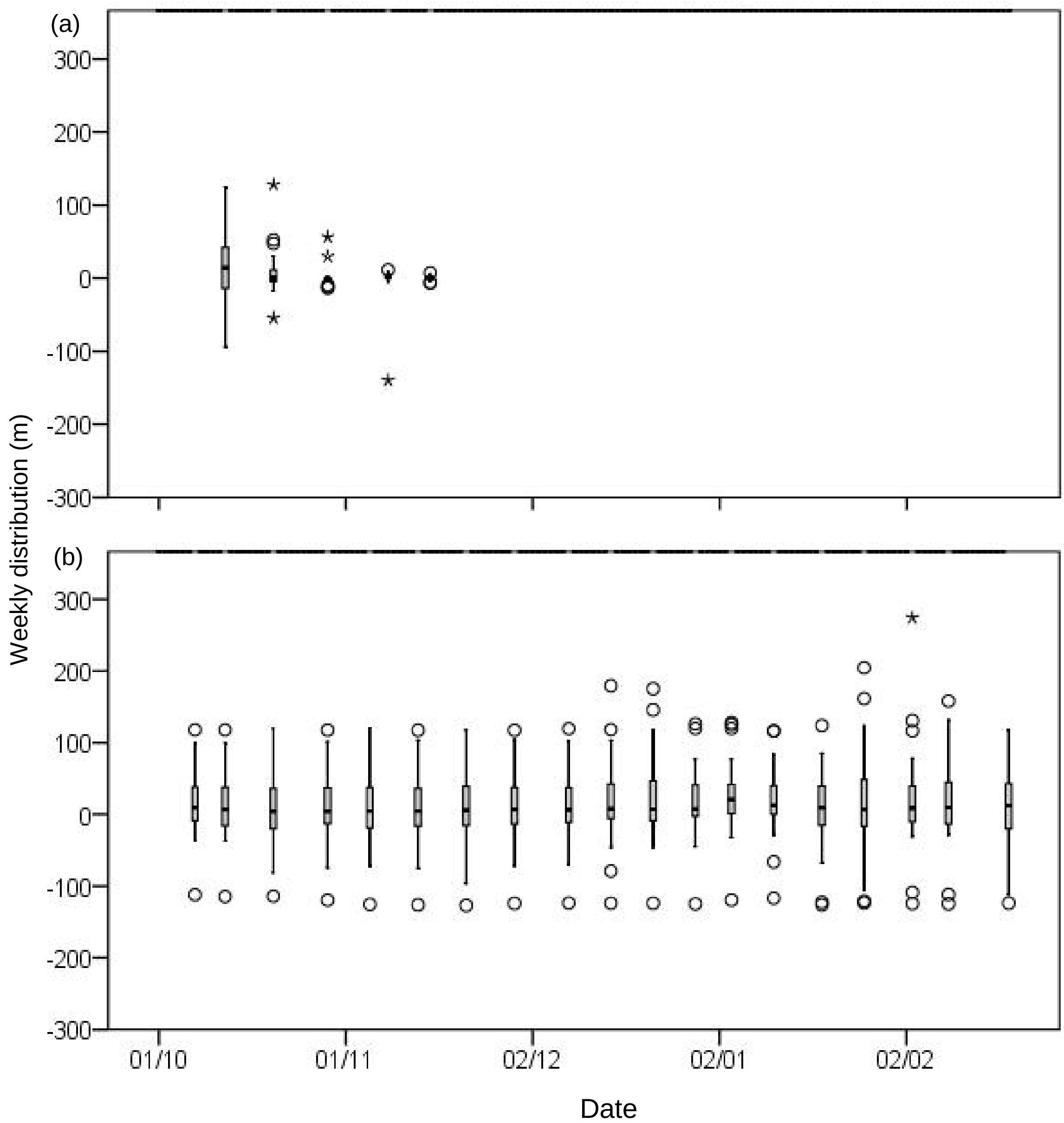


Figure 4.20. Weekly distribution of all brown trout in (a) 2012 and (b) 2013/2014.

4.3.3 Influence of reservoir freshet releases on brown trout movements

Directional movements of brown trout exclusively in an upstream or downstream direction were not observed during the Brownhill Reservoir freshet release in November 2012, as brown trout tended to occupy a relatively small extent of river, with 9 m and 15 m being the largest range recorded by any individual during the reservoir freshet release and control days, respectively (Table 4.3). The total distance moved by an individual during the Brownhill Reservoir freshet release and on a control day was also 20 m and 18 m, respectively, in the Ramsden Clough study reach (Table 4.4). Indeed, the total distance moved by brown trout in Ramsden Clough (Mann-Whitney *U*-test: Z = 1.629, n

= 16, P = 0.105), Co-op Lane (Mann-Whitney U-test: Z = 0.578, n = 16, P = 0.574) and Marsden Clough (control reach) (Mann-Whitney U-test: Z = 1.706, n = 20, P = 0.143) during the Brownhill Reservoir freshet release in November 2012 was not significantly different to movements in the same reach on the control day without a freshet release. Furthermore, brown trout location immediately after the freshet release was not disproportionally upstream (55%) or downstream (20%) of the location occupied prior to the freshet release (X^2 = 13.8, $d.f$ = 12, P = 0.314) with no brown trout >6 m from the location they occupied immediately prior to the freshet release (Figure 4.21a).

The largest extent of river used by brown trout during Brownhill and Digley Reservoir freshet releases in 2013 was 21 m in October in Mill Pond and 41 m in December in Marsden Clough, respectively (Table 4.3). Despite the relatively small range of river occupied by brown trout during freshet releases, brown trout in the reaches exposed to freshets had larger ranges than brown trout in the control reaches on the same day and in the same reach on the control day (Table 4.3). The largest total distance moved by an individual brown trout during Brownhill and Digley Reservoir freshet releases in 2013 occurred in October and was 59 m in Mill Pond and 32 m in both Ramsden Clough and Mill Pond, respectively (Table 4.4). In 2013, brown trout moved more in reaches impacted by freshet releases than those in the control reaches on the same day (Table 4.4). For example, the total distance moved by brown trout in Ramsden Clough was significantly larger than those in Marsden Clough during freshet releases from Brownhill Reservoir in October 2013 (t-test, t = 7.7, $d.f.$ = 14, P = 0.000) and December 2013 (t-test, t = 7.4, $d.f.$ = 13, P = 0.000), but not in November 2013 (t-test, t = -0.099 $d.f.$ = 13, P = 0.922). Likewise, the total distance moved by brown trout in Marsden Clough was significantly larger than those in Ramsden Clough during freshet releases from Digley Reservoir in October 2013 (t-test, t = 2.8, $d.f.$ = 14, P = 0.015, November 2013 (t-test, t = 4.3, $d.f.$ = 13, P = 0.001) but not in December 2013 (t-test, t = 1.3, $d.f.$ = 13, P = 0.215) . The total distance moved by brown trout in the Mill Pond reach did not differ significantly between freshet releases from Brownhill and Digley reservoirs in 2013 (t-test, October: t = -1.460 $d.f.$ = 12 P = 0.170, November: t = -0.972 $d.f.$ = 12 P = 0.350, December: t = -0.623 $d.f.$ = 12 P = 0.545). More than 60% of tagged brown trout were within 5 m of the location they occupied prior to any of the Brownhill (Figure 4.21b) and Digley (Figure 4.21c) reservoir freshet releases in 2013 and the furthest distance a brown trout relocated was 38 m upstream after the Digley Reservoir freshet release in December 2013.

Table 4.3. Range (mean ± SD (range), m) of individual brown trout in each study reach during Brownhill Reservoir and Digley Reservoir freshet releases in October, November and December and a control day with no reservoir freshet release in 2012 and 2013. Shaded represents control data.

Study year	Brownhill Reservoir freshet release			Digley Reservoir freshet release			Control day
/ Site name	October	November	December	October	November	December	
2012	-	-	-	-	-	-	-
Marsden Clough	-	0 ± 1 (0 – 4)	-	-	-	-	0 ± 1 (0 – 5)
Ramsden Clough	-	5 ± 3 (0 – 9)	-	-	-	-	2 ± 2 (0 – 7)
Co-op Lane	-	3 ± 2 (0 – 6)	-	-	-	-	3 ± 5 (0 – 15)
2013							
Marsden Clough	0 ± 1 (0 – 3)	0 ± 1 (0 – 2)	1 ± 1 (0 – 2)	7 ± 5 (1 – 15)	6 ± 5 (0 – 13)	7 ± 13 (0 – 41)	0 ± 1 (0 – 2)
Ramsden Clough	8 ± 3 (6 – 14)	4 ± 3 (0 – 9)	3 ± 2 (1 – 6)	2 ± 2 (0 – 5)	1 ± 1 (0 – 3)	2 ± 1 (1 – 4)	1 ± 1 (0 – 3)
Co-op Lane	-	-	-	-	-	-	-
Mill Pond	7 ± 7 (0 – 21)	4 ± 3 (0 – 9)	6 ± 4 (0 – 13)	12 ± 9 (0 – 25)	10 ± 10 (0 – 25)	5 ± 5 (0 – 14)	2 ± 2 (0 – 7)
Old Mill	-	-	-	-	-	-	-

Table 4.4. Total distance moved (mean ± SD (range), m) by brown trout in each study reach during Brownhill Reservoir and Digley Reservoir freshet releases in October, November and December and a control day with no reservoir freshet release in 2012 and 2013. Shaded are represents control data.

Study year	Brownhill Reservoir freshet release			Digley Reservoir freshet release			Control day
/ Site name	October	November	December	October	November	December	
2012							
Marsden Clough	-	2 ± 5 (0 – 15)	-	-	-	-	1 ± 2 (0 – 5)
Ramsden Clough	-	9 ± 7 (0 – 20)	-	-	-	-	5 ± 6 (0 – 18)
Co-op Lane	-	6 ± 4 (0 – 12)	-	-	-	-	5 ± 3 (0 – 8)
2013							
Marsden Clough	1 ± 2 (0 – 6)	1 ± 1 (0 – 2)	1 ± 1 (0 – 2)	14 ± 11 (1 – 27)	12 ± 8 (0 – 25)	12 ± 14 (0 – 46)	0 ± 0 (0 – 2)
Ramsden Clough	18 ± 9 (10 – 32)	1 ± 1 (0 – 3)	8 ± 3 (3 – 12)	3 ± 2 (0 – 5)	1 ± 1 (0 – 3)	4 ± 2 (2 – 8)	0 ± 1 (0 – 3)
Co-op Lane	-	-	-	-	-	-	-
Mill Pond	12 ± 11 (0 – 32)	9 ± 9 (0 – 24)	8 ± 7 (0 – 20)	22 ± 19 (0 – 59)	17 ± 18 (0 – 45)	10 ± 9 (0 – 27)	0 ± 1 (0 – 7)
Old Mill	-	-	-	-	-	-	-

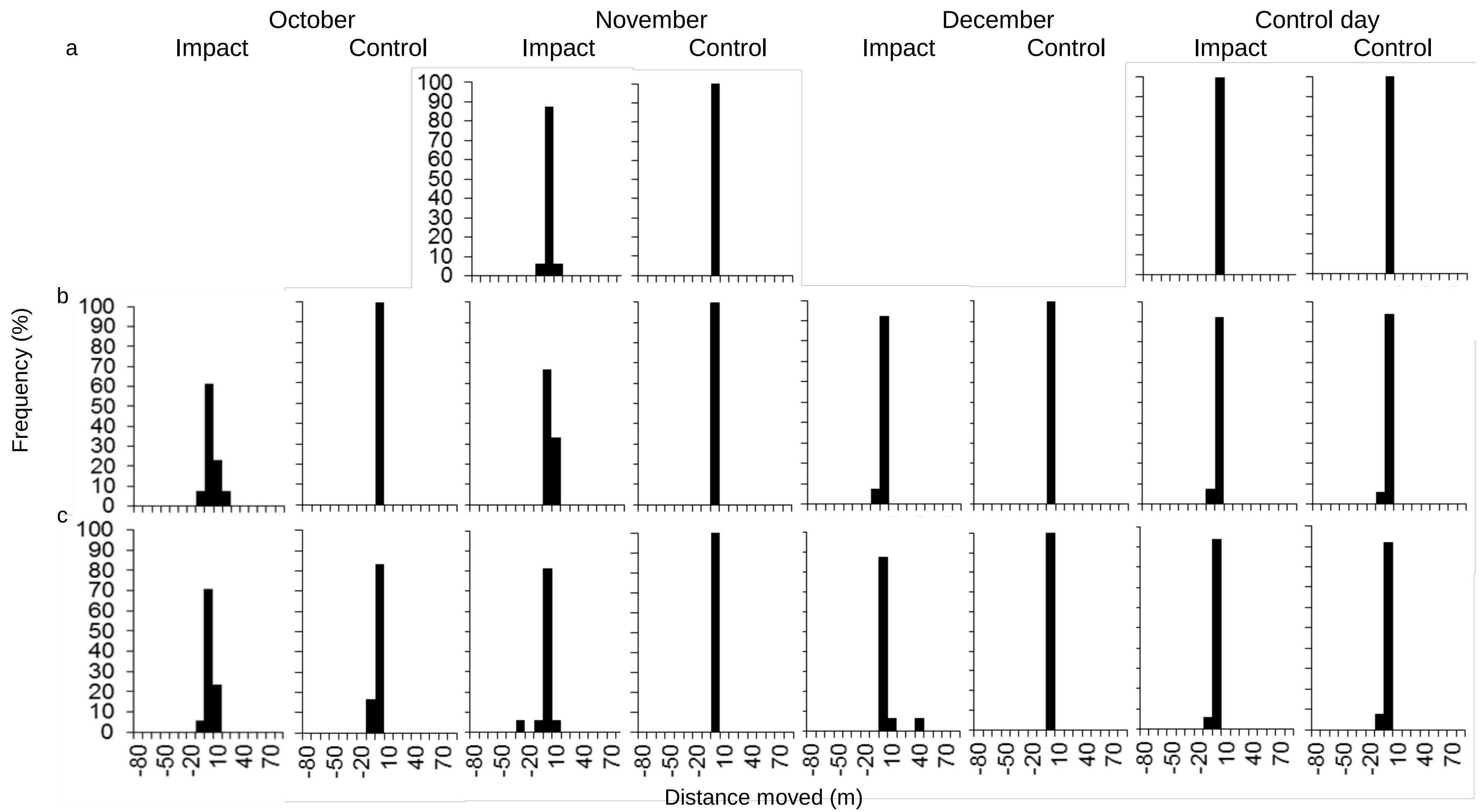


Figure 4.21. Frequency distribution of brown trout after Brownhill Reservoir freshet releases 2012 (a) (n= 26) and 2013 (b) (n= 23 October and n= 22 November and December) in impact (Ramsden Clough, Co-op Lane (2012) and Mill Pond (2013)) and control (Marsden Clough) sites and Digley freshet release (c) in impact (Marsden Clough and Mill Pond) and control (Ramsden Clough) sites in October, November and December. Gaps represent no freshet release performed.

4.3.4 Temperature profiles during freshet releases

4.3.4.1 Daily temperature profiles

In 2012 and 2013/2014 the water temperature decreased throughout the study period at all three sites. Ramsden Clough water temperature appeared to be slightly warmer than the other two sites at the start of the study period and at the end appeared to be cooler. (Figure 4.22a, 4.22b, 4.23a and 4.23b). Temperature profiles between Ramsden and Marsden Clough appeared to be very similar.

4.3.4.2 Freshet temperature profiles

It was speculated that the freshet releases could have suddenly changed the temperature profiles, they did not reflect a natural high flow event and this sudden change could have been the influencing factor to why brown trout did not move in response to freshet releases.

The water temperature profile on the control day in 2012 appeared to steadily warm throughout the day (Figure 4.22c), whereas on the day of the Brownhill freshet release, the freshet appeared to have a cooling effect at Ramsden and Co-op Lane (Figure 4.22d). Mann-Whitney U Tests revealed that temperature profiles at every site in 2012 were significantly different between freshet and control day even at Marsden Clough (control site) but not between two control days (Appendix Table 9.2).

In 2013 the water temperature profiles at all three sites during the freshet releases appeared to be similar to control temperature profiles on the same day. The Brownhill Reservoir freshet releases had minimal effect on the temperature profiles at Ramsden Clough, there was a slight increase in temperature in the River Holme in response to the Brownhill Reservoir release in October (Figure 4.24a) but no noticeable change in November (Figure 4.24b) and December (Figure 4.24c) 2013. The Digley Reservoir freshet release in October appeared to have a cooling effect on the water temperatures at Marsden Clough and the River Holme (Figure 4.26a) and a warming effect in November when water temperatures had dropped to around 6.5°C (Figure 4.25b). Whereas in December the Digley freshet release had no noticeable change on water temperatures at Marsden Clough and the River Holme (Figure 4.25c).

Despite any similarities seen in temperature profiles during freshet releases from Brownhill and Digley reservoirs and during control days in 2013, significant differences persisted in all incidences (Appendix Table 9.2), because of this result it makes it difficult to say whether the lack of fish movement was a result of sudden changes in temperature from the reservoir freshet releases.

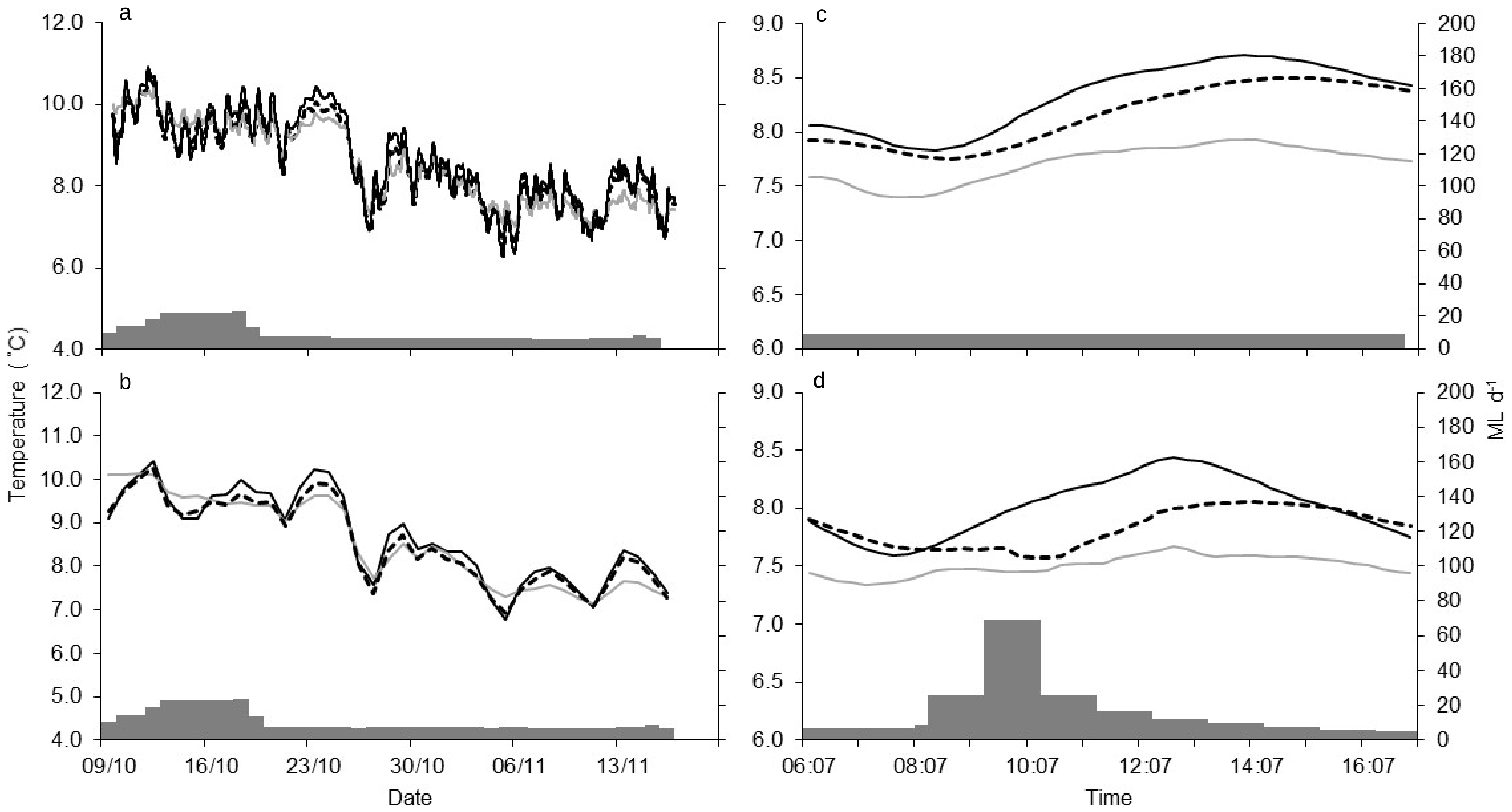


Figure 4.22. Every (a) 15 minute and (b) average daily temperature (˚C) profiles at Marsden Clough (black line), Ramsden Clough (grey line) and River Holme (dashed black line), including Brownhill Reservoir releases (grey bar) (ML day^{-1}) throughout the tracking period (October – November 2012). Brownhill Reservoir (c) control and (d) freshet release data (grey bar) (ML day^{-1}) and 15 minute temperature profiles at Marsden Clough, Ramsden Clough and River Holme, November 2012.

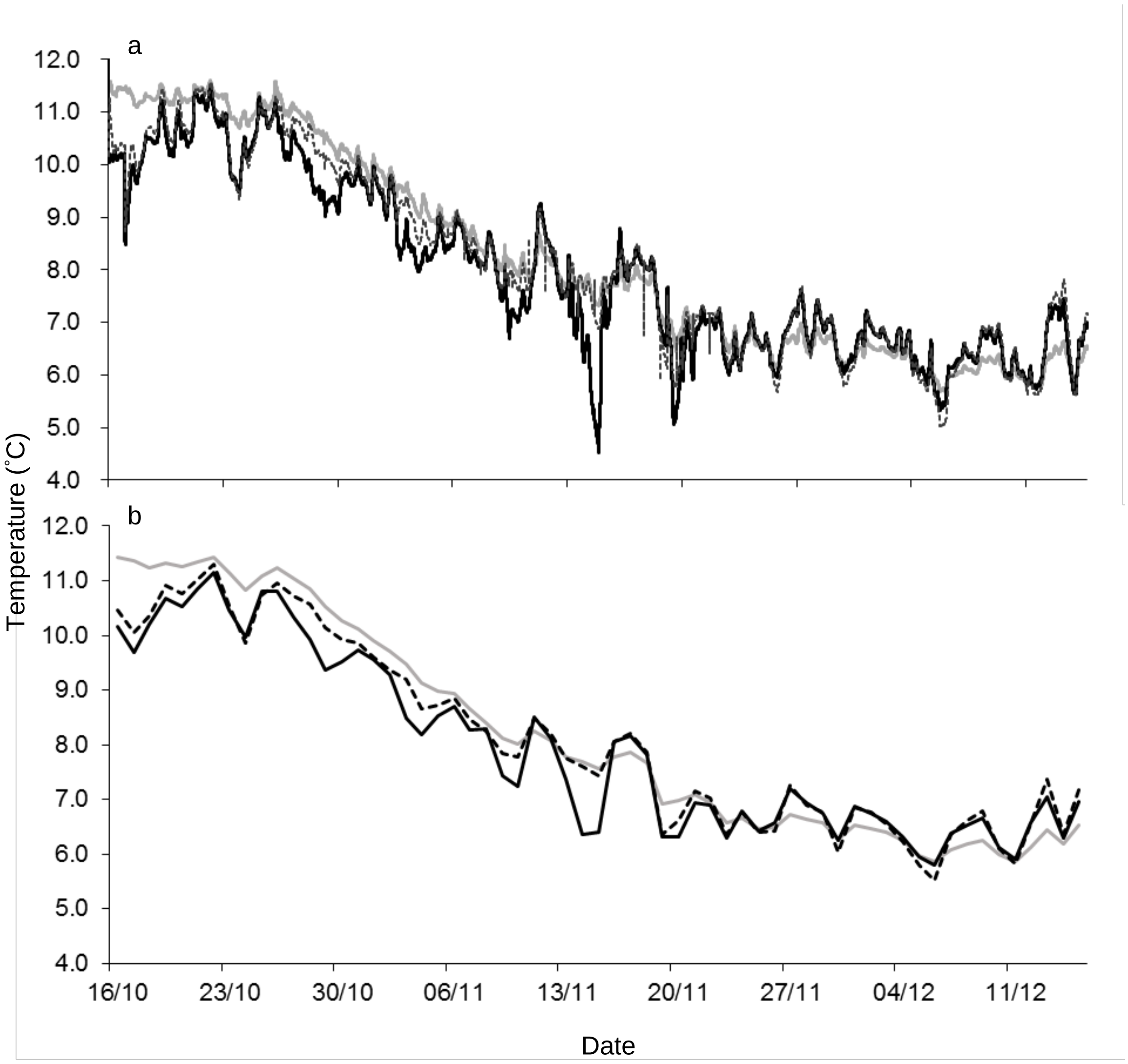


Figure 4.23. Every 15 minute (a) and average daily (b) temperature (˚C) profiles at Marsden Clough (black line), Ramsden Clough (grey line) and River Holme (dashed black line).

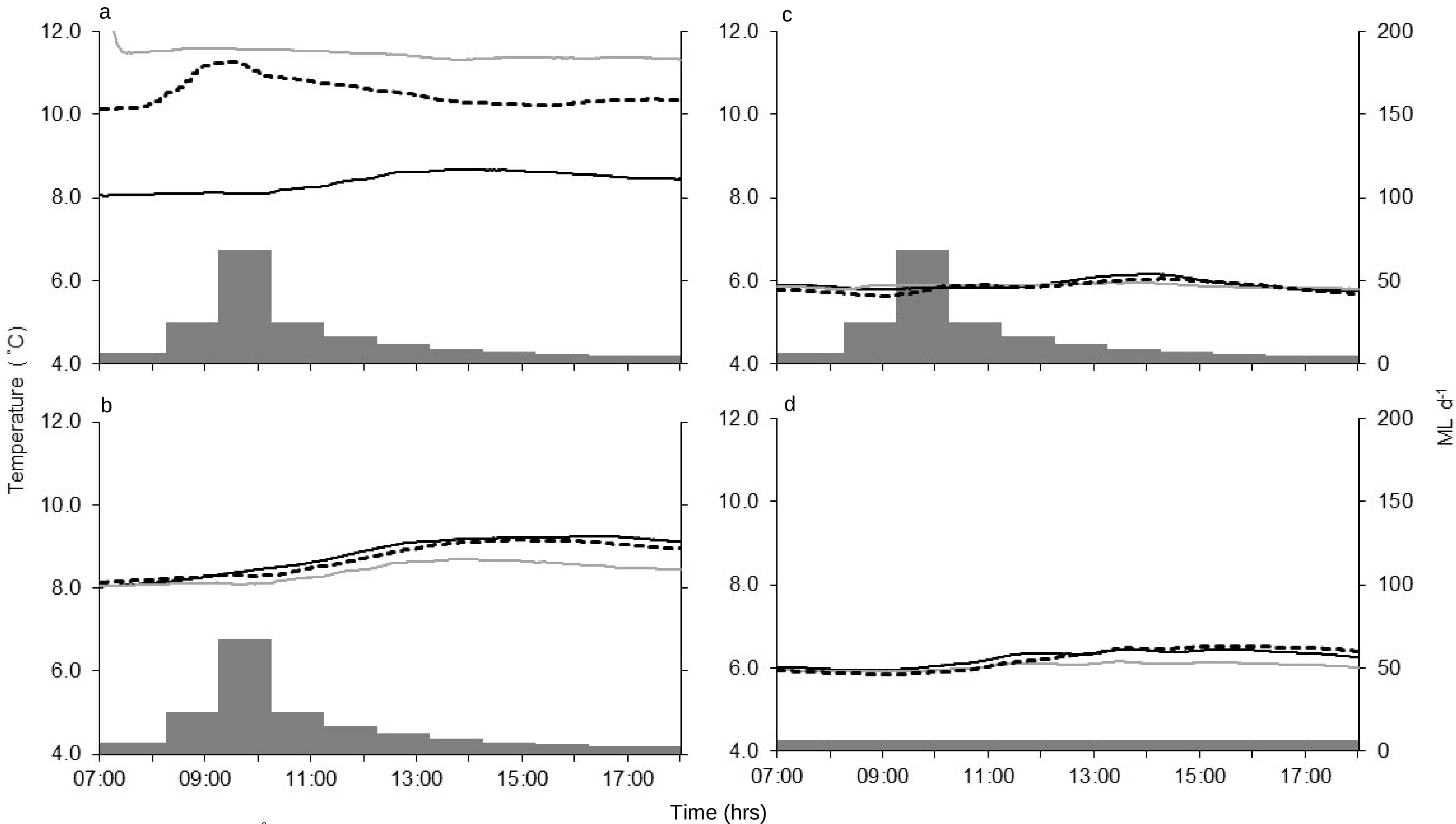


Figure 4.24. Temperature (˚C) profiles at Marsden Clough (black line), Ramsden Clough (grey line) and River Holme (dashed black line), including Brownhill Reservoir release data (grey bar) (ML day^{-1}) during freshet release in (a) October, (b) November, (c) December and (d) control day with no reservoir freshet release, 2013.

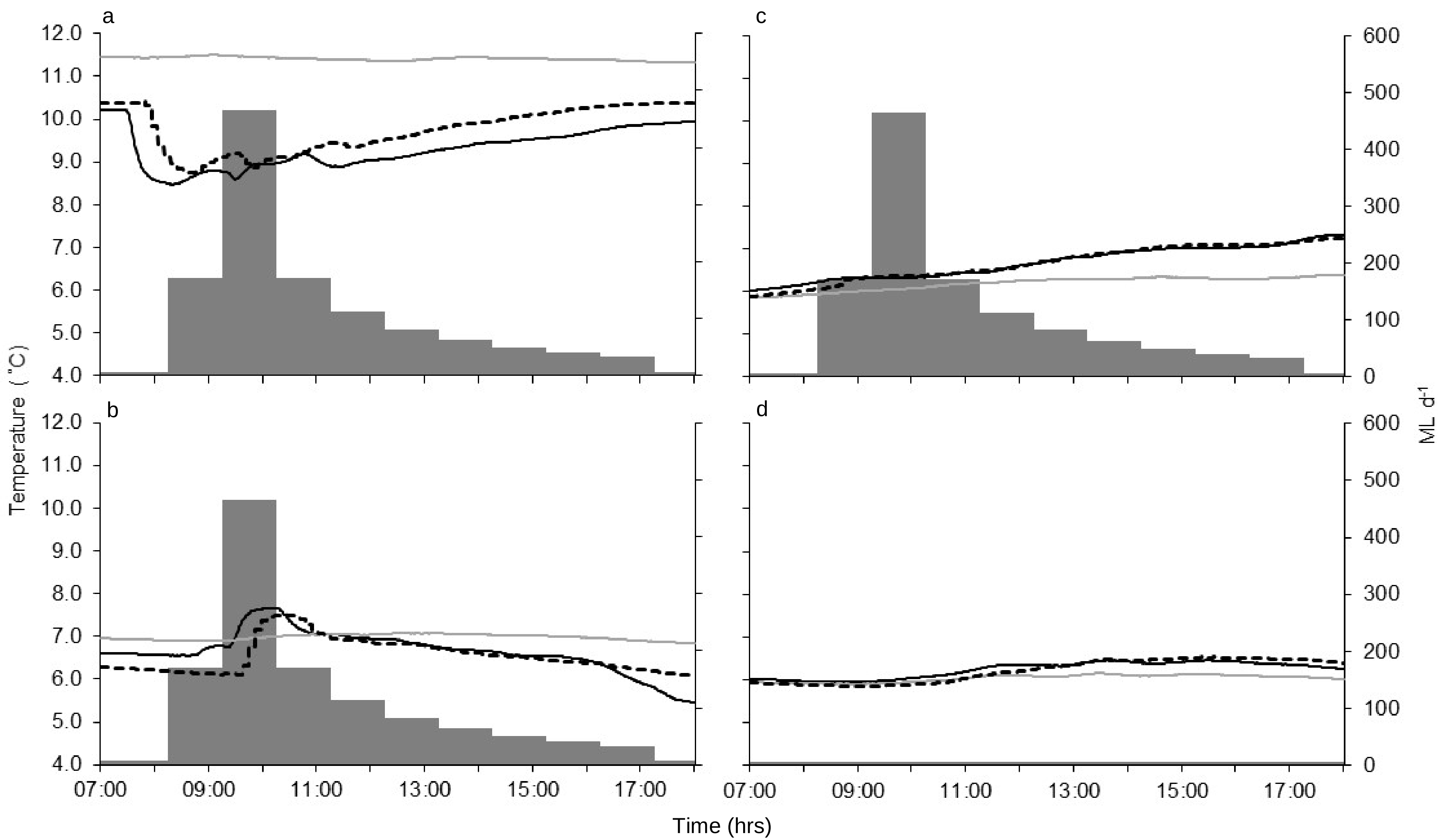


Figure 4.25. Temperature (˚C) profiles at Marsden Clough (black line), Ramsden Clough (grey line) and River Holme (dashed black line), including Digley Reservoir release data (grey bar) (ML day^{-1}) during freshet releases in (a) October, (b) November, (c) December and (d) control day with no reservoir freshet release, 2013.

4.4 DISCUSSION

4.4.1 Overview

Radio tracking brown trout in the River Holme, Ramsden and Marsden Clough monitored general movements and movements in response to Brownhill and Digley reservoir freshet releases. Monitoring over the two study periods 2012 and 2013/14 indicated that general movements of brown trout were relatively small during autumn and winter, and, despite changes to the reservoir release programme in 2013/2014, no long distance spawning migrations were identified during freshet releases. Findings are discussed in relation to other studies and UKTAG guidance (UKTAG, 2013).

4.4.2 General movements and individual behaviours of brown trout

Throughout the two study periods, brown trout in the River Holme catchment moved relatively short distances and occupied small home ranges. This finding was consistent with other studies that reported brown trout occupying small, well-defined home ranges during the majority of their life (Solomon & Templeton, 1976; Bachman 1984; Burrell *et al*., 2000; Knouft & Spotila 2002; Stakėnas *et al*., 2013). Solomon & Templeton (1976) found that 71 % of 111 tagged brown trout were recaptured within a 100-m section of the location they occupied three months earlier and a further 20 % within 300 m. Bachman (1984) found that brown trout were commonly found in the same sites for three consecutive years. Knouft & Spotila (2002) reported 95.5% of all brown trout recaptures and contacts were within 800 m of initial tagging and a study carried out in the UK found that the mean distance moved by brown trout in winter was only 100 m (Stakėnas *et al*., 2013).

Brown trout in the River Holme were, on the whole, located within 100 m of the release location in both study periods, with <10% of all the radio tracked brown trout (*n*= 95) moving >100 m from the release location. A year-long tracking study in the Chattooga River watershed in South Carolina USA found that brown trout established small home ranges (<270 m), apart from during spawning migrations (Burrell *et al*., 2000). Body size has been known to influence home range in freshwater fishes; larger individuals are reported to have larger home ranges in comparison to smaller individuals (Minns, 1995). The length of the brown trout studied in the River Holme were of a similar size, and there was no relationship between home range, range per day tracked, daily distance moved and length of brown trout. It is possible that if larger brown trout were studied they could have moved greater distances. Höjesjö *et al*. (2007) found that dominant (aggressive) brown trout moved longer distances and occupied larger home ranges than less aggressive subordinates, thus it is possible that brown trout in this study that were more active and had larger home ranges were the more dominant individuals. Also, male brown trout are less migratory than females (Jonsson & Jonsson, 2011), during the tagging procedure it was noted that the majority of tagged brown trout in the current study were male and they may be less likely to migrate. There were cases in this study where brown trout underwent single long distance movements, and this same behaviour was also found with brown trout in the River Orthe (Ovidio *et al*., 1998) and Chattooga River (Burrell *et al*., 2000). Ovidio *et al*. (1998) found that a large proportion of brown trout made large spawning migrations over several kilometres. Seven out of nine tagged brown trout in the River Ourthe, Belgium, travelled between 5.6 to 23.0 km upstream into side tributaries between the 7th of October to the 15th of November 1996, though daily journeys never exceeded 300 m (Ovidio *et al*., 1998). Such shifts in home range were found in the River Holme and there were cases of brown trout moving further than 100 m (2012 = three brown trout and 2013 = ten brown trout). In this study the brown trout were not restricted by any large dams and the weirs present were <1 m high, which was presumed to have little impact on brown trout movement. Other possible reasons behind the absence of large spawning migrations by brown trout in the River Holme, are covered in section 4.4.3 - 4.4.8 (Flow requirements), section 4.4.9 (water temperature) and section 4.4.10 (habitat suitability).

4.4.3 Influence of freshets on brown trout movement

In this study, the purposes of autumn/ winter freshets was to support the migration of brown trout to their spawning grounds and to flush away the build-up of fine sediments. According to UKTAG guidance (UKTAG, 2013), autumn and winter flow elevations should have a 6 x Qn95 magnitude, a 12-hour duration and occur once per week at night in October, November and December (Table 4.5). It was apparent that the freshet released in 2012 and freshets released in 2013 did not trigger any long distance spawning migrations of brown trout in the River Holme.

Table 4.5. UKTAG recommendations for autumn and winter flow elevations to reach good ecological potential (UKTAG, 2013).

Autumn & Winter flow elevations		
Purposes		To support the migration of adult salmon, sea trout, river lamprey and sea lamprey into rivers and the migration of these species and brown trout in rivers to their spawning grounds.
Default flow	**Magnitude**	6x Qn95
Ascending and descending limbs of flow rise to mimic those of comparable natural flow rises.		
Period		October, November, December
Duration		12 hours if no obstacles to migration are present. If a number of obstacles are present, two to three days.
Frequency		Once per week at night of 12-hour duration and, where possible, synchronised with catchment rainfall events.

4.4.4 Frequency of reservoir freshet releases

Since 2004, Yorkshire Water Services has released a single freshet in November each year from Brownhill and Digley reservoirs. Brown trout movements were monitored in Ramsden Clough and the River Holme in 2012 and it was apparent that this single freshet from Brownhill Reservoir was insufficient to trigger spawning migrations. Therefore the number of freshets was increased and performed in October November and December. In 2013 Digley Reservoir was also operational (maintenance works meant that Digley Reservoir could not release freshets in 2012) meaning the brown trout in Marsden Clough could be monitored during freshets in 2013. The frequency of freshets were 2 separate days in each month (October, November and December) in the River Holme (receiving Brownhill and Digley reservoir freshets) and only one day a month in Ramsden (receiving Brownhill freshets) and Marsden Clough (receiving Digley freshets). The number of freshets differed between the two study periods but there was no indication that the increased number of freshets in 2013/14 resulted in longer distance movements of brown trout. The recommended frequency according to the UKTAG guidance is once a week in October, November and December, meaning that the 2013 flow release programme did not meet the requirements of the UKTAG guidance. This guidance is difficult to achieve for licencing and logistical reasons. Yorkshire Water Services are restricted to the amount of water they can release, to ensure there's enough water for human consumption and secondly and reservoir freshets are manually operated, thus coordinating and resourcing freshet releases that frequently would be costly and demanding. In an unregulated system the frequency of higher flow releases would likely occur more regularly in the autumn period, and reservoirs curtail this frequency b impounding the water, thus suppressing rainfall events. Increasing the frequency of freshet releases could provide more of an opportunity for brown trout to perform spawning migrations (Jonsson & Jonsson, 2011), but of this basis of the present study this does not appear to occur, possibly because of the limited habitat in the affected reach.

Other benefits by increasing the frequency of freshet releases would to flush sediment deposited in riffle habitats (Gilvear et al., 2002; UKTAG, 2013). If these habitats are not flushed on a regular basis they become clogged with finer sediment between the gravels, macrophytes become established and further bind the finer sediment together making it less likely to be flushed away; eventually the riffle habitat is lost (Arthington & Zalucki, 1998). The invertebrate community will also benefit from increasing the frequency of freshet releases; density has been reported to increase with the frequency of floods three times greater than the median flow (Acerman and Ferguson, 2010).

4.4.5 Duration of reservoir freshet releases

The duration of the freshets was over an 8-hour period from Brownhill Reservoir (2012 and 2013/2014) and a 10-hour period at Digley Reservoir (2013/2014). The difference in duration (and magnitude) of freshet releases between the two reservoirs did not result in more movement. The recommended magnitude of the freshet releases from Brownhill Reservoir exceeded 6 x Qn95 for only a 2-hour period (08:15am – 10:15am), 10 hours shorter than the UKTAG recommended period. The 6 x Q_{95} (22.8 ML day^{-1}) from Digley Reservoir was exceeded for a period of 8 hours during the freshet release but the duration of the freshet release at the desired magnitude fell sort of the recommended 12-hour period (Table 4.5). Longer duration releases at Digley and Brownhill reservoirs are problematic because the valves are manually operated and it is costly paying a member of staff to be present for longer duration freshets. Public perception could also be a factor to consider as members of the public might be concerned about the risk of flooding (personal communication Yorkshire Water). Longer flow durations are, however, important to enable salmonids to shift between microhabitats and the current short term flow changes may not provide sufficient opportunity to make those shifts (Kemp *et al*., 2003; Ayllon, 2014). An increased period of a flow event can also provide more opportunities for feeding and spawning movements, increasing growth and survival (Welcomme, 1985).

4.4.6 Magnitude of reservoir freshet releases

Habitat availability and connectivity are increased at higher flow magnitudes, which is important for refuge, feeding opportunities and spawning activities (Kennard *et al*., 2007). During this study, fish did not move further upstream during small (17.3 x Qn95) or large (122.4 x Qn95) magnitude releases. This is similar to Heggenes *et al.* (2007) who found brown trout in the River Måna, Norway, had small home ranges with few individuals performing sporadic larger movements during extreme magnitude flow events. The peak magnitudes of freshet releases from Digley and Brownhill reservoir were 122 x Qn95 and 17 x Qn95 respectively, many times higher than the recommended 6 x Qn95. Flows this high could potentially discourage brown trout from moving because of high energetic costs. Also smaller brown trout could be potentially displaced or stranded by the high magnitude (Acreman & Ferguson, 2010; Jonsson & Jonsson, 2011). None of the brown trout radio tracked in this studied were displaced by the freshets but there could have been a risk to smaller individuals. Vehanen *et al*. (2000) found increased flow increased displacement of juvenile brown trout during a controlled flume experiment during winter. Other studies indicated that adult and juvenile brown trout are displaced downstream by floods but returned upstream as the flow had declined (Dare *et al*., 2002).

4.4.7 Rate of change of reservoir freshet releases

UKTAG guidance(UKTAG, 2013) recommends that the rising and falling limb (rate of change) of the freshet must mimic those of comparable natural elevated flow events because sudden changes in flow can cause smaller fish and invertebrates to become displaced (Acreman & Ferguson, 2010; Jonsson & Jonsson, 2011). Halleraker *et al*. (2003) found that a reduction in ramping rate was needed to reduce the number of individuals from stranding. None of the brown trout radio tracked in this studied became displaced by the freshets but there could have been a risk to smaller individuals, 0+ fish having have limited swimming abilities making them more susceptible to becoming displaced by freshets (Wolter & Arlinghaus, 2003). The freshets, particularly from Digley Reservoir in 2013, had a rapid change in flow increasing to 45 x Qn95 in the first hour and then to 122 x Qn95 in the second hour quickly descending back to 45 x Qn95 in the third hour and thereafter declining quite gradually. These releases did not mimic a natural spate release, and could be considered to be unsuccessful releases. Not having an unregulated control site makes it difficult to conclude. The reason why the rate of change of releases was so drastic from Digley and Brownhill reservoirs was because they are manually operated: the operator has to manually open and close a valve, making it difficult to control. Given the manual operational constraints it would be very difficult to exactly mimic a natural spate release. There are no specific guidelines provided by the UKTAG other than ascending and descending limbs of flow rise to mimic those of comparable natural flow rises (Table 4.5).

4.4.8 Timing of reservoir freshet releases

In 2012 the timing of the single freshet was during November, which could have been at the wrong time of the year, although it was designed to match spawning migration. In the second study period (2013/2014) the freshets were released in October through to December 2013. Comparing the timing of freshets between October November and December found no difference in the distances moved by brown trout. The UKTAG guidance suggests that the release should occur in October through to December as the timing can be crucial for triggering spawning migrations (Ziegler & Schofield, 2007). The second study period met the recommended time period but no long distance migrations were observed. Exact timings may vary between rivers and sub- catchments due to genetic differences (Acreman & Ferguson, 2010) and it is possible that the brown trout spawn later in January (Ovidio *et al*., 1998). Freshet releases were not performed in this month but there were noticeable periods of heavier rainfall and natural overtopping events from late December 2013 and through January 2014 and six individuals made larger movements in January. Time timing of the freshet releases was during the day and it is possible that these freshets were failing to trigger larger movements because of the time of day. Freshets releases are recommended by the UKTAG guidance to be performed at night, when the risk of predation in is lower (Alanärä *et al*., 2001) and the risk of stranding of juvenile brown trout is reduced (Saltveit *et al*., 2001; Halleraker *et al*., 2003) so this again may be a reason for the lack of detected movements..

4.4.9 Water temperature

River temperatures were monitored throughout the study and during the freshet releases. Temperature profiles at Marsden Clough, Ramsden Clough and the River Holme were comparable throughout the study period.

When comparing temperatures on an hourly basis during the period of a freshet release, temperatures were significantly different to control days and reaches. On the day of the Brownhill freshet release, the freshet appeared to have a cooling effect at Ramsden and Co-op Lane. During freshet releases the water was sourced from the bottom of the reservoirs (hypolimnial) and this could mean that the water released would be warmer in winter than water coming into the river via rainfall events or overtopping events. Warmer winter reservoir releases were documented by Dare *et al*. (2003), and this effected species distribution, behaviour and abundance.

Comparing temperature variations between control days, significant differences still persisted. This indicated that temperature differences were attributed to day to day temperature variations despite days when freshets were performed, thus possibly having little influence on brown trout behaviour. The daily variation could result from the diurnal atmospheric heating cycle, especially as Carron & Rajaram (2001) reported that even under steady flow conditions temperature varies significantly due to the diurnal atmospheric heating cycle.

Importantly in this study temperatures recorded were within spawning temperature ranges. Ovidio *et al*. (1998) found that brown trout spawning migrations were triggered by a combination of high variations of water temperature, water level and need to be within the thermal range of 10-12˚C. In 2012 in the River Holme water temperatures in the first week of the study were on average less than 10˚C (9.7˚C) and in 2013 water temperatures declined below 10˚C on average after 29 October. This would suggest that brown trout in the river reaches monitored should have performed spawning migrations before October in 2012 and during October 2013 when temperatures were in the suggested temperature range. However, Jonsson and Jonsson (2011) reported that brown trout can still spawn in temperatures as low as 5ºC, suggesting temperatures were sufficient for spawning throughout the entire study.

4.4.10 Habitat suitability

The River Holme is a small river around 14 km long and brown trout were tagged between 400 m and 3 km downstream of Brownhill and Digley reservoirs. Man-made barriers such as weirs and dam walls could have restricted any long-distance movements in the River Holme, Ramsden and Marsden Cloughs. The distance between impassable weirs was 500 m at Marsden Clough, 200 m at Ramsden Clough, and for the three reaches further downstream could potentially be 1800 m or 2010 m if ascending up Ramsden Clough or Marsden Clough, respectively. It was possible for brown trout to drop down below weirs, in 2012 BT4 dropped below the first downstream weir from the release site in Marsden Clough. This demonstrated that brown trout could access downstream reaches increasing their home range and potentially spawn with populations downstream. It would have been very unlikely that they were able to gain access back up into the reach in which it had been captured and tagged. These barriers could have restricted movements and home ranges of brown trout but there was no evidence that they were trying to ascend any of them or spending time in the pools downstream of them.

Spawning migrations are dependent on the location of spawning gravels and suitable adult habitat (Northcote, 1992). It is possible, in this study, that there could have been sufficient suitable spawning and adult habitat to support the brown trout population in the specific reaches of the River Holme, suggesting brown trout may not have to migrate large distance to spawn, a possible reason why no large movements of brown trout were undertaken. Furthermore, it is not an obligatory requirement for either individual brown trout or the entire population in the study river to perform a spawning migration to complete their life-cycle (Jonsson & Jonsson, 2011). Older studies concluded that stream resident salmonids complete their whole life cycle in one single pool (Miller, 1957) or short stream sections (Bachman, 1984), and this could be the case in this study, supporting the reason behind the lack of movement performed by brown trout. Brown trout could have been behaving normally but it is difficult to conclude without an unregulated reference site to provide a comparison. Improving the study design to account for this is latter recommended in Chapter 7.

The brown trout populations studied may be able to withstand high fluctuations in flow, on an annual basis, and have adapted to withstand such stresses (Gasith & Resh, 1999). The River Holme is a flashy river meaning that when there are periods of high rainfall there is a short lag time between the rains falling in the catchment and making the way downstream the river. There is a possibility that the freshets performed during this study were just a few events occurring around other natural rain induced elevated flow events and therefore brown trout did not respond to the freshets and may respond to the natural elevated flow events. In 2013/2014 study rainfall and overtopping events were more frequent in late December and January it appeared that brown trout were more active during this time. This appears that natural high flow events are more meaningful than the freshets releases.

4.5 SUMMARY

In 2012 manual radio tracking was used to obtain a detailed knowledge of the general movements and impact of the freshet release on the spatial distribution, and movement of adult brown trout downstream of Brownhill and Digley reservoirs. Radio tracked adult brown trout occupied small home ranges and the freshet releases did not result in substantial longitudinal movements (migration). It was hypothesised that the freshet releases in November 2012 were not performed at the appropriate time of year and at the right magnitude to promote brown trout migration, and thus releases were performed in October, November, and December 2013. Although occasional longer distance (>100m) were observed in 2013, reservoir freshet releases still did not result in long distance longitudinal movements. This suggests that the freshet releases in 2012 and 2013, which lasted eight hours, provided brown trout with very little opportunity to move a reasonable distance. Further, the hydrograph did not resemble a natural freshet, with the rising and falling limb reaching maximum and minimum too quickly. Since this study was carried out, the UKTAG guidance has been published, and the magnitude, duration, diel timing and frequency for the freshet releases from Brownhill and Digley reservoir do not currently meet the flow profile recommended by UKTAG for autumn and winter flow elevations to support spawning migrations (UKTAG, 2013).

5 HABITAT IMPROVEMENT FOR FISHERIES IN RIVERS DOWNSTREAM OF RESERVOIRS IN YORKSHIRE

5.1 INTRODUCTION

Modifying flow releases from reservoirs is one way in which flow conditions for fish can be improved in heavily modified water bodies (HMWB) to achieve good ecological potential (GEP) as part of the Water Framework Directive (WFD) environmental objectives (European Commission, 2015) (Chapter 2.4). In many instances modifying flows from reservoirs, even those with dam walls up to 60 m high are implemented relatively quickly and easily without retrofit or restoration of outlet works, for example, a dam operator opening a sluice gate to increase the downstream release (Dyson *et al*., 2003). In other instances modifying flows can take considerably more time and effort to allow structural and operational changes to be made to outlets (Dyson *et al*., 2003). Methods for implementing modified flows from reservoirs have been discussed extensively in the literature, but progress putting them into action has been limited (Reilly & Adamowski, 2014).

There are potential positive and negative outcomes when designing modified flows from reservoirs, making it difficult to decide whether to implement them into the flow regime (Morrison & Stone, 2015). Morrison & Stone (2015) concluded that modifying flows from reservoirs to the Rio Chama basin, New Mexico would have minimal impact on hydropower production and whitewater rafting on the system, but would decrease the annual reservoir storage and increase median flows. Finding a balance between ecological and social requirements for water can be challenging; WFD objectives for river improvement works are ecologically driven and do not focus on social objectives, which may be less acceptable to local stakeholders (Acreman & Ferguson, 2010). Another limiting factor when modifying flows from reservoirs from existing reservoirs is the cost and who should pay for any modification to structures. The modification of flow releases may involve a cost associated with retrofitting or rehabilitating the valves to allow them to operate and modify the releases from the reservoir (Dyson *et al*., 2003). However, costs are minimal when no operational constraints are present and a dam operator can open a valve to modify the flow. In addition, releasing additional water to provide benefits to the downstream environmental is costly for water services, there could be a reduction in the amount of water available for power generation, human consumption or even water available for irrigation (Dyson *et al*., 2003). The costs to the services lost must be outweighed by the benefits gained by restoring or maintaining ecosystem services.

In some instances it is not operationally or economically possible to modify flow releases from reservoirs to improve the ecology of the receiving watercourse. In this case alternative solutions need to be found to ensure compliance with WFD downstream of reservoirs where flows cannot be modified. This chapter focuses on two reservoir systems in Yorkshire, UK where the flow regime downstream of the reservoirs could not be modified. Alternative techniques assessed in this chapter involve habitat improvement works which aim to improve flow and provide habitat diversity, trialled on two rivers downstream of impounding reservoirs where fish populations were deemed to be failing under WFD.

Habitat improvement works occurs frequently across the UK but monitoring is rarely carried out or when carried out has not been undertaken in a statistically robust and meaningful way (Hammond *et al*., 2011). One of the main reasons for the lack of monitoring is cost, making it difficult to draw comparisons and learn from successes and failures of river improvement works. Traditionally habitat improvement works often lack pre- and post-monitoring of fish populations to determine success of the works (Verdonschot *et al*., 2013). Consequently, the principle aim of this chapter was to ascertain a baseline of fish populations prior to habitat improvement works and to develop a suitable monitoring programme for assessing fish population change following the works.

Specific objectives (O): -

- Using case studies determine the appropriate fish monitoring requirements to ensure post programme changes in population metrics can be detected(using resource calculations);
- Provide recommendations on fish monitoring programme design to ensure habitat improvement works are assessed in the appropriate manner and with scientific rigor

This chapter is a prerequisite for a full BACI design which will detect the effectiveness of habitat improvement works as an alternative to flow modification using brown trout as an indicator.

5.2 METHODOLOGY

5.2.1 Study reaches

Yorkshire Water Services identified two river reaches downstream of reservoirs, where modification of compensation flows into the downstream rivers was not possible due to operational constraints. The rivers downstream of the reservoirs are both subject to slow flows and low water levels, as well as high levels of siltation, poor water quality and lack of physical habitat for fish (Figure 5.1 & 5.2). Habitat improvement techniques were proposed to improve habitats for fish.

5.2.1.1 Ingbirchworth

Ingbirchworth Dike is located in the River Don catchment, with the proposed habitat improvement reach consisting of 0.5 km of channelised river downstream of Ingbirchworth Reservoir (Figure 5.1 & Figure 5.3). The WFD ecological status of the river is 'moderate' with fish populations failing. The river has been channelised, and it's slow flow has meant that the river has become clogged with vegetation (macrophytes), leading to high levels of siltation. There are a small number of isolated pools available for larger brown trout and very little potential spawning habitat or nursery habitats for 0+ brown trout. The aim of the habitat improvement work was to help improve the ecological status of Ingbirchworth Dike to "good potential". Planned habitat improvement techniques for Ingbirchworth Dike include remeandering, involving excavators, creating a new meandering channel to increase flow diversity. The installation of brushwood bundles to provide overhead cover for fish and the, installation of fences to prevent cattle poaching the river banks which will help reduce the amount of bank erosion and sediment entering Ingbirchworth Dike. Five (IBW1 – IBW5 (Figure 5.1 a – e)) of the six sites sampled at Ingbirchworth Dike were within the habitat improvement reach, one site on Ingbirchworth Dike (IBW6, (Figure 5.1f) was located downstream of the habitat improvement reach. Sites were selected based on habitat features and field boundaries, where fencing restricted access. Site six sampled downstream of the habitat improvement section was requested to be sampled by Yorkshire Water Services.

5.2.1.2 River Washburn

The River Washburn, located in the River Wharfe catchment, has a proposed habitat improvement reach consisting of 1 km of river downstream of Swinsty Reservoir (Figure 5.2 & 5.4). The current WFD ecological status of the river is 'moderate' with fish populations being the failing element due to the impacted flow regime. The river is over widened, and resulted in slow flowing water with high levels of siltation, and a lack of suitable habitats for different life stages. For example; isolated pools for larger trout and very little areas of riffle habitat for smaller brown trout. The aim of habitat improvement work is to help improve the ecological status of the River Washburn to ‘good potential’. In the River Washburn planned habitat works include channel narrowing to increase flow and reduce siltation problems, introducing gravels to provide suitable spawning habitat for brown trout, removal of some trees to reduce shading with trees cut down placed in the river to provide overhead cover for fish and narrow the channel to increase flow. Habitat improvement aims to provide a more diverse habitat for fish improving areas for spawning, juveniles and adult populations of brown trout. Three (SW1 – SW3 (Figure 5.2a - c)) of the six sites sampled on the River Washburn were within the habitat improvement reach, the other three sites (SW4 – SW6 (Figure 5.1d - e) were located downstream of the habitat improvement works. Sites 1 – 3 were evenly spaced out along the habitat improvement section, the three sites sample downstream of the proposed improvement section were monitored to see if there was any impact on fish populations despite no improvement works been carried out.

5.2.1.3 Reference sites

Reference sites were selected for Ingbirchworth Dike (Figure 5.3) and the River Washburn (Figure 5.4) to account for environmental variability and temporal trends found in both the reference and impact sites, increasing the ability to differentiate the effects of habitat improvement from natural variability (Smith *et al.,* 1993), i.e. change in impact reach – change in reference reach. The selection of control sites would have ideally been upstream of the impact sites but habitat improvement works were planned directly downstream both Ingbirchworth and Swinsty reservoirs. Consequently, it was decided that reference sites should be selected on the nearest most comparable rivers. Initially 14 reference sites were sampled and then later narrowed down based on comparable brown trout densities.

Figure 5.1. Ingbirchworth Dike sample sites (IB1 – IBW6 (a – f)), all photographs taken in September, upstream view at downstream limit of survey site (pre-habitat improvement).

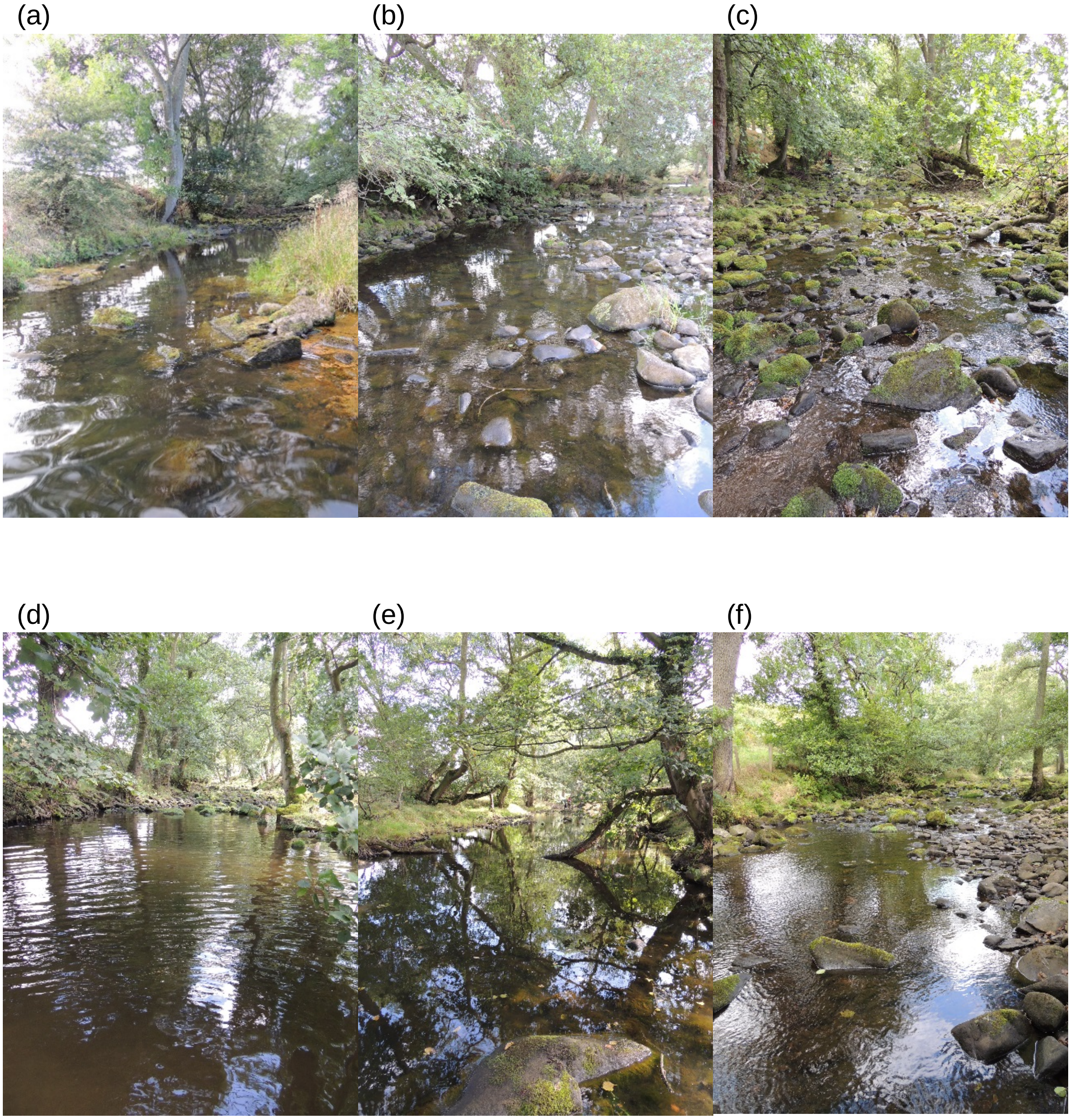


Figure 5.2. River Washburn sample sites (SW1 – SW6) (a – f)), all photographs taken in September, upstream view at downstream limit of survey site (pre-habitat improvement (a – c)) (site not being improved (d – f)).

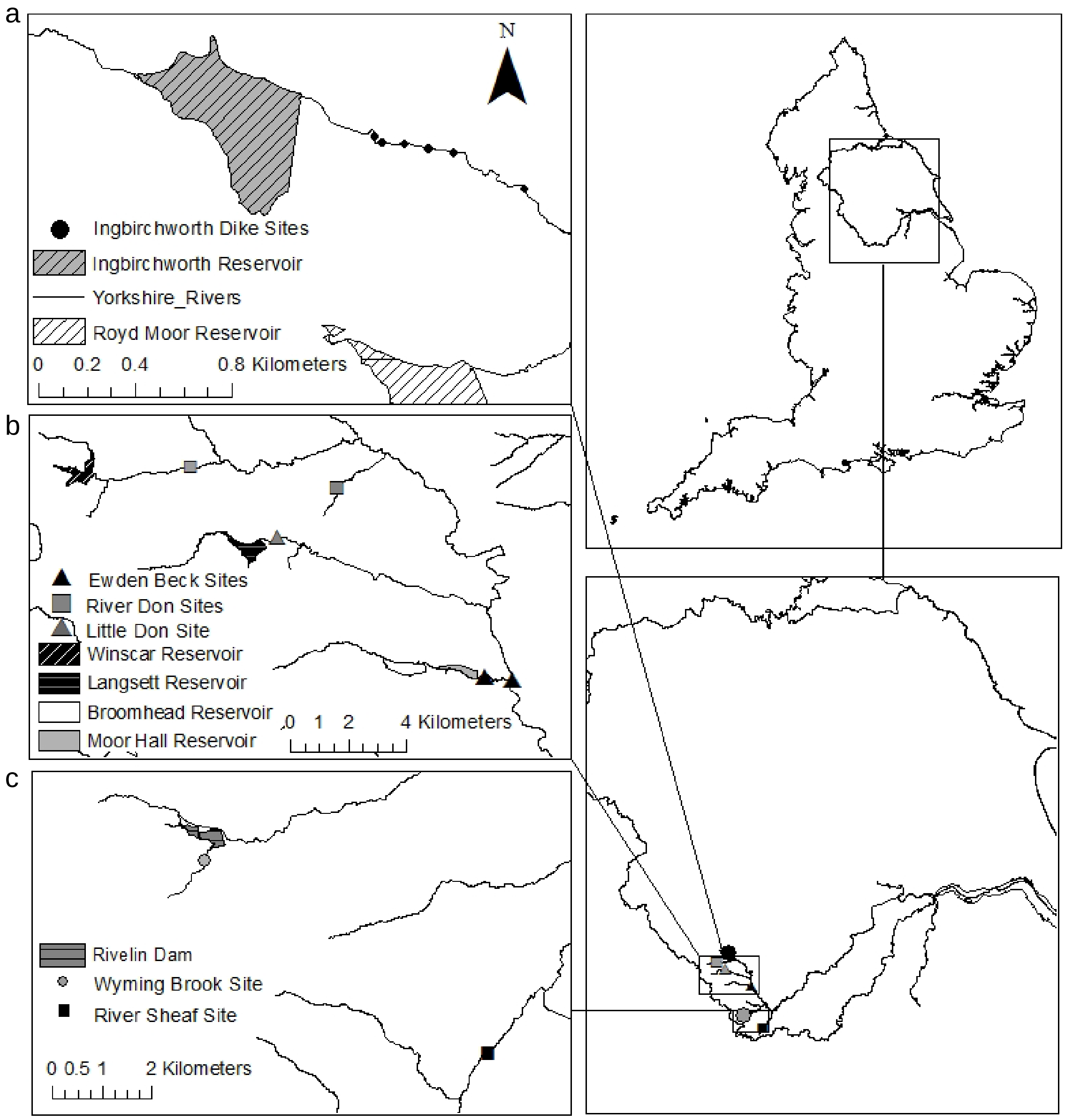


Figure 5.3. Location of the Ingbirchworth Dike study are in England and detailed maps of (a) Ingbirchworth Dike habitat improvement sites and (b & c) reference sites.

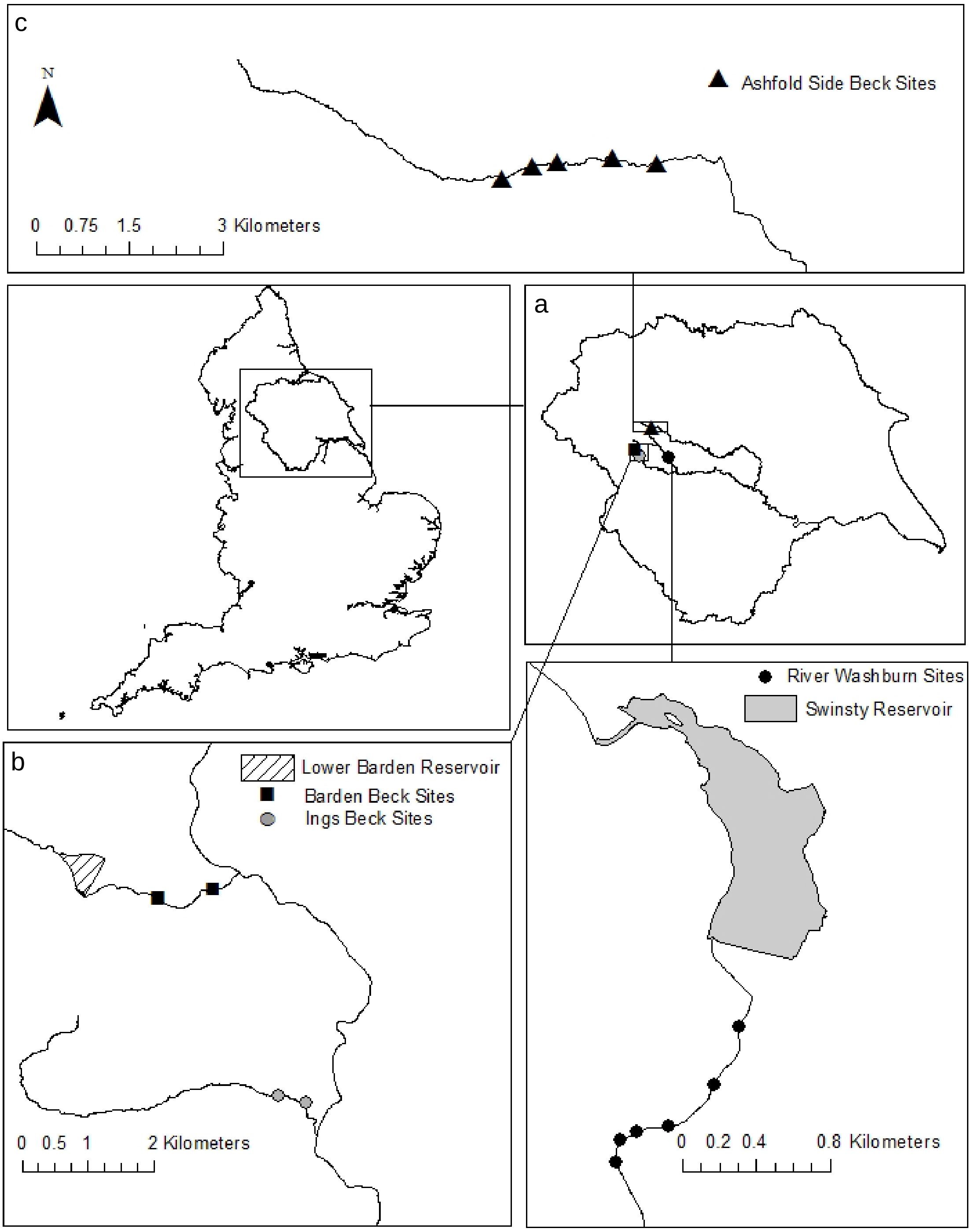


Figure 5.4. A map of England showing the location of River Washburn study and more detailed maps of (a) River Washburn habitat improvement sites and (b & c) reference sites.

5.2.2 Fish survey methodology

Fisheries surveys at study sites (Tables 5.1 & 5.2) were carried out in 2012, 2013 and 2014, prior to habitat improvement, using quantitative electric fishing (estimates of absolute abundance based on a three-catch removal method (Carle & Strub, 1978)

(Chapter 3.2.2). The focus of the study was primarily on brown trout, but other species were captured and recorded. at Ingbirchworth Dike (three-spined stickleback (*Gasterosteus aculeatus* L.), stone loach, roach (*Rutilus rutilus* (L.)) and perch (*Perca fluviatilis* L.)) and the River Washburn (three-spined stickleback, stone loach, perch, roach, minnow (*Phoxinus phoxinus* (L.)), bullhead and chub (*Leuciscus cephalus* (L.)). . As mentioned previously, brown trout are the model study species, they make long distance migrations making them an ideal species to monitor when investigating habitat suitability and accessibility. They require various habitats at different stages of their life cycle, for example, clean gravels for spawning and deep pools for large adult trout. Thus YWS will be creating various habitats through habitat improvement works and by monitoring brown trout populations, it can be measured how successful the habitat works have been based on the brown trout populations. Extensive photographs were taken at the survey sites to allow identification of the study reaches and for comparison following habitat improvement works.

Table 5.1. Fisheries survey site details for Ingbirchworth Dike habitat improvement works (n/s = not sampled).

Impact/reference sites River Name	Site identifier	NGR	Year surveyed 2012	 2013	 2014
Impact					
Ingbirchworth Dike	IBW1	SE 219 059	✓	✓	✓
Ingbirchworth Dike	IBW2	SE 220 058	✓	✓	✓
Ingbirchworth Dike	IBW3	SE 221 058	✓	✓	✓
Ingbirchworth Dike	IBW4	SE 222 058	✓	✓	✓
Ingbirchworth Dike	IBW5	SE 223 058	✓	✓	✓
Site downstream of habitat improvement reach					
Ingbirchworth Dike	IBW6	SE 225 057	n/s	✓	✓
Reference					
Ewden Beck	R23	SK 299 955	✓	✓	✓
Ewden Beck	R24	SK 290 956	✓	✓	✓
Little Don	R25	SE 219 004	✓	✓	✓
River Don	R26	SE 237 021	✓	✓	✓
River Don	R27	SE 189 028	✓	✓	✓
River Sheaf	R28	SK 330 824	✓	✓	✓
Wyming Brook	R29	SK 272 863	✓	✓	✓

5.2.3 HABSCORE data collection

Habitat parameters at all sites surveyed at Ingbirchworth Dike, the River Washburn and reference sites were collected and used to determine habitat quality and usage using the HABSCORE programme (Chapter 3.2.3).

Table 5.2. Fisheries survey site details for the River Washburn habitat improvement works.

Impact/reference sites River Name	Site identifier	NGR	Year surveyed 2012	2013	2014
Impact					
River Washburn	SW1	SE 195 523	✓	✓	✓
River Washburn	SW2	SE 195 520	✓	✓	✓
River Washburn	SW3	SE 192 518	✓	✓	✓
Sites downstream of habitat improvement reach					
River Washburn	SW4	SE 190 518	✓	✓	✓
River Washburn	SW5	SE 188 518	✓	✓	✓
River Washburn	SW6	SE 188 516	✓	✓	✓
Reference					
Barden Beck	R6	SE 047 562	✓	✓	✓
Barden Beck	R7	SE 056 564	✓	✓	✓
Ings Beck	R8	SE 064 533	✓	✓	✓
Ings Beck	R9	SE 068 531	✓	✓	✓
Ashfold Side Beck	R13	SE 118 661	✓	✓	✓
Ashfold Side Beck	R14	SE 123 661	✓	✓	✓
Ashfold Side Beck	R15	SE 127 663	✓	✓	✓
Ashfold Side Beck	R16	SE 135 664	✓	✓	✓
Ashfold Side Beck	R17	SE 143 663	✓	✓	✓

5.2.4 Data analysis

5.2.4.1 Density estimates of brown trout

Estimates of 0+ and >0+ brown trout densities were derived annually between 2012 and 2014 for all sites surveyed on Ingbirchworth Dike and the River Washburn as well as all reference sites. Data were compared to determine population densities prior to habitat improvement works. The methodology for calculating densities is described in Chapter 3.2.4.2.

5.2.4.2 Classification of population estimates of brown trout

Methodology for classification of population estimates can be found in Chapter 3.2.4.3.

5.2.4.3 Length distributions of brown trout

Length distributions of brown trout at each site were produced. The methodology involved assigning each fish length of a particular species into a size class and determining the total number of fish in each size class. Brown trout were assigned to 5-

mm size classes, to allow for clear identification of age classes. Length distributions derived from surveys in each year were compared

5.2.4.4 Length at age and determination of growth rate of brown trout

Calculation of growth rates of brown trout was facilitated by the collection of scale samples from a representative number of fish from Ingbirchworth Dike and the River Washburn (three brown trout form every 10mm size class) (Britton, 2003). Methodology for determination of age and growth rate of fish is described in Chapter 3.2.4.4.

5.2.4.5 HABSCORE analysis and outputs

See Chapter 3.2.4.5.

HABSCORE outputs at each site in this report were derived using habitat measurements in 2014 and incorporating annual fish survey data between 2012 and 2014 surveys; this improves the HABSCORE outputs by taking into account some of the observed temporal variability of the trout populations at each site.

5.2.4.6 Resource calculations

Ideally a full before/after, control/impact (BACI) study design would have been performed to provide statistical evidence that the habitat improvement works carried out at Ingbirchworth Dike and the River Washburn resulted in a biologically meaningful change. Unfortunately due to delays the habitat improvement works were not carried out within the timeframe of this study, this was outside of control of the study, which meant a full BACI analysis could not be performed. Despite of this resource calculations were carried out to determine the level of future sampling, and to demonstrate how it would be carried out when running a fully BACI analysis.

The analysis was performed on 0+ brown trout densities obtained from Ingbirchworth Dike and the River Washburn, 0+ densities areconsidered to be an important measure of recruitment and are also sensitive to change. To separate the components of true temporal, spatial and interaction variance it was necessary to obtain data from a number of sites on more than one occasion (Sedgwick, 2006). Consequently three years of before sampling at five impact sites (IBW1 – IBW5) on Ingbirchworth Dike (Table 5.1) and three years of before sampling at six impact sites (SW1 – SW6) on the River Washburn were used (Table 5.2). Three years of before sampling at seven reference sites (Table 5.1) for Ingbirchworth Dike and three years of before sampling at nine reference sites (Table 5.2) for the River Washburn were selected. It is imperative that sufficient sites are sampled enough times to identify a population change (i.e. the precision level) within a stated level of probability (e.g. 0.8 or 80%) and statistical power (e.g. 0.05 or 5%). This can be determined using resource calculations, a pre requisite to a full BACI analysis. It is important to consider the precision level that must be achieved. In this context, precision is associated with the "noise" (usually expressed as the variance) generated by the spatial and temporal variations in fish populations, and is usually reduced by larger sample sizes or repetitive surveys. A reliable estimate will have a low variance. The precision level deemed biologically meaningful and within the realms of feasible resource allocation was 50% of the mean pre-impact density (Cowx, 1996). The resource calculations using three years pre data from impact and reference sites detected how many years of post-habitat improvement sampling was required to detect a 50% population change.

The following steps were followed:

1. The mean density was calculated for 0+ brown trout at impact sites at Ingbirchworth Dike and the River Washburn based on three years of sampling before river habitat improvement works.

2. The target variance (Sedgwick, 2006) for a 50% change in the mean pre-impact density for 0+ brown trout was calculated:

   (50% of mean density before/ (Ø*SQRT(2)))2 Equation 5.1

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

3. The actual variance (V(x)) (Sedgwick, 2006); Equation 5.2) of the full BACI quadrant for 0+ brown trout was calculated.

   V(x) = (Vytr)*(1/(mB*nT)+1/(mA*nT)+1/(mB*nC)+1/(mAnC)) Equation 5.2

   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 with the target variance to identify how many years of post-habitat improvement sampling was required to detect a 50% population change.

5. If the actual variance was lower than the target variance, a statistically significant 50% change in the mean pre-impact density could be identified.

5.3 RESULTS

5.3.1 Ingbirchworth Dike and reference sites

5.3.1.1 Species composition, density estimates and classification

Species composition

Five fish species (three-spined stickleback, stone loch, brown trout, roach and perch) were captured in Ingbirchworth Dike; numbers varied between sites and years (Figure 5.5). Species composition at Ingbirchworth Dike reference sites included several other additional species, viz. grayling, minnow, bullhead, rainbow trout and ruffe (*Gymnocephalus cernua* (L.)). These additional species were not present at all reference sites (Figure 5.6).

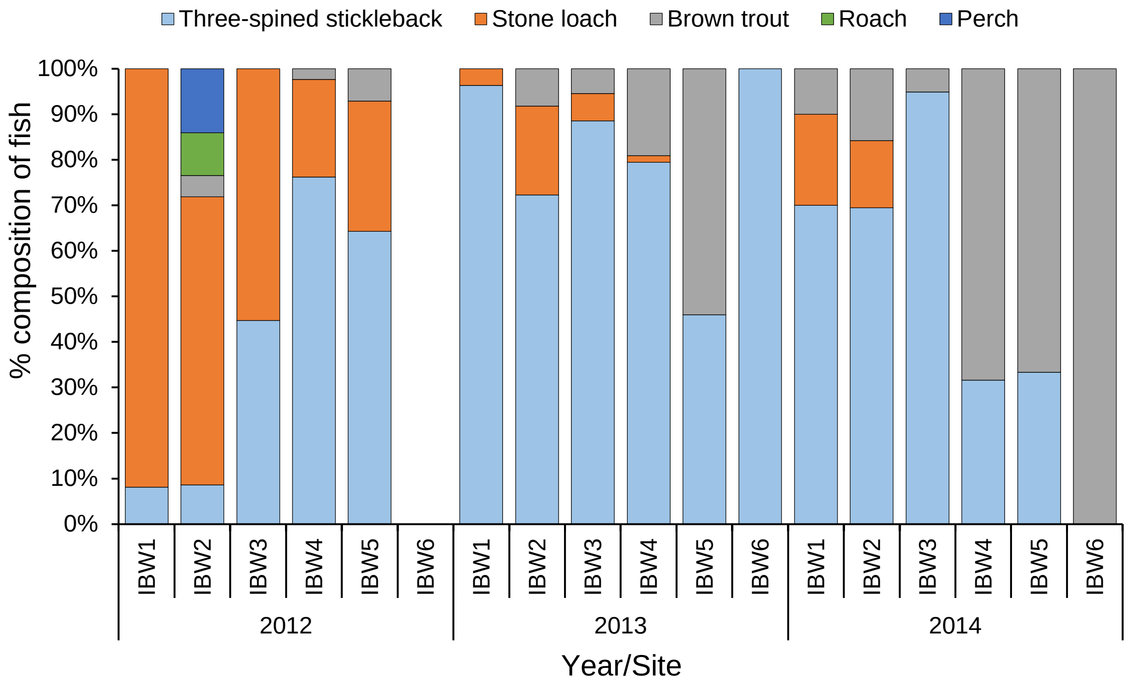


Figure 5.5. Percentage composition of fish captured in Ingbirchworth Dike annually between 2012 and 2014 (Site identifiers in Table 5.1).

At Ingbirchworth Dike three-spined stickleback (*Gasterosteus aculeatus* L.) was the most abundant species at the majority of sites and years, except at IBW1, IBW2 and IBW3 in 2012, where stone loach was most abundant, and IBW4 (2014) and IBW5 (2013 and 2014) where brown trout was most abundant (Figure 5.5).

Unlike at Ingbirchworth Dike, brown trout were the most abundant species at the majority of the reference sites, instances where brown trout were not the most abundant species included; 2012 the most abundant species found at R23, R26 and R28 was bullhead, in 2013 minnow was most abundant at sites R26 and R28 and in 2014 bullhead was the most abundant species at R23 and R28 and perch was the abundant species at R24 (Figure 5.4). Three-spined stickleback were found in far lesser numbers at reference sites than at Ingbirchworth Dike (Figure 5.6 and 6.5).

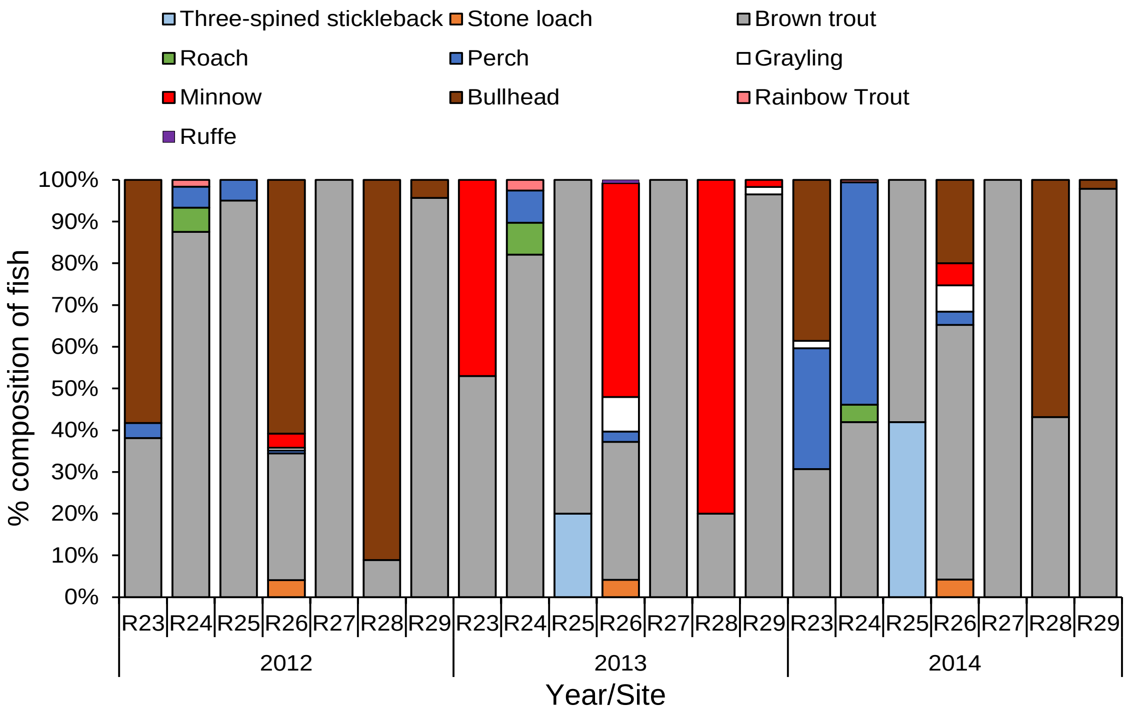


Figure 5.6. Percentage composition of fish captured at Ingbirchworth Dike reference annually between 2012 and 2014 (Site identifiers in Table 5.1).

Brown trout density estimates and classifications

0+ brown trout were absent from all sites sampled in Ingbirchworth Dike in 2012; 0+ brown trout populations were poor at IBW2-IBW4 (1.0 – 1.9 fish per 100 m^2), average at IBW5 (13.9 fish per 100 m^2) and absent at IWB6 in 2013. In 2014, 0+ brown trout were absent from all sites except IBW2 where populations were poor as in 2013 (Table 5.3).

0+ brown trout were present at all Ingbirchworth Dike reference sites and all years except R25 in 2013. 0+ brown trout densities were stable at reference site R24 over the study period (0.3 – 1.4 fish per 100 m^2) whereas at other reference sites there was some variability in densities. For example at R28 brown trout populations in 2012 were 0.6 fish per 100 m^2 (class E) and then increased to 13.5 fish per 100 m^2 (class C) in 2014 (Table 5.3).

In summary, when 0+ brown trout were caught at Ingbirchworth Dike they were of comparable densities to reference sites but on occasions reference sites were higher.

Table 5.3. 0+ and >0+ brown trout density (numbers of fish per 100m2) and abundance classification at survey site locations for the Ingbirchworth Dike habitat improvement works (n/s = not sampled).

A (excellent)	B (good)	C (average)	D (fair/poor)	E (poor)	F (fishless)

River Name	Site identifier	0+ brown trout			>0+ brown trout		
		2012	2013	2014	2012	2013	2014
Impact sites							
Ingbirchworth Dike	IBW1	0.0	0.0	0.0	0.0	0.0	0.9
Ingbirchworth Dike	IBW2	0.0	1.9	0.5	10.3	10.0	7.4
Ingbirchworth Dike	IBW3	0.0	1.0	0.0	0.0	10.0	1.9
Ingbirchworth Dike	IBW4	0.0	1.0	0.0	0.7	12.1	13.6
Ingbirchworth Dike	IBW5	0.0	13.9	0.0	1.9	17.0	16.9
Ingbirchworth Dike	IBW6	n/s	0.0	0.0	n/s	0.0	1.4
Reference							
Ewden Beck	R23	5.0	4.2	2.4	14.2	12.8	11.7
Ewden Beck	R24	1.2	0.3	1.4	43.0	21.4	22.6
Little Don	R25	2.6	0.0	3.3	2.9	1.2	2.7
River Don	R26	0.7	2.6	4.3	8.7	6.3	8.2
River Don	R27	4.7	9.5	13.2	18.3	20.5	24.3
River Sheaf	R28	0.6	7.9	13.5	3.9	1.0	3.2
Wyming Brook	R29	3.0	14.3	7.8	20.0	13.4	22.9

Ingbirchworth Dike >0+ brown trout densities varied between sites and years from absent (IBW1 & IBW3 in 2012 and IBW1 &IBW6 in 2013) to good (IBW4 & IBW5 in 2013 & 2014). IBW2 was the most consistent site for >0+ brown trout densities over the three sampling years, but not the most abundant (Table 5.3).

>0+ brown trout populations at Ingbirchworth reference sites were generally stable between 2012 and 2013, except at sites R25 and R28 where populations declined from fair/poor to poor. >0+ brown trout populations at site R24 were excellent throughout the study period while at sites R27 and R29 populations were the highest in 2014 (Table 5.3).

Overall, Ingbirchworth reference sites had higher densities of >0+ brown trout throughout the whole study in comparison to the Ingbirchworth Dike sites.

5.3.1.2 Length distributions of brown trout

0+ brown trout were only captured in 2013 and one individual in 2014 (IBW2, 79mm), in 2013 individuals caught were 83 mm (IBW3) and 76 mm (IBW4), where more than one brown trout were caught 0+ brown trout were captured in similar size ranges 73-76 mm at IBW2 and 78-87 mm at IBW5 (Figure 5.7).

>0+ brown trout in 2012 were caught in small numbers at IBW2 (size range 205 – 485 mm), IBW 3 (size range 220 – 225 mm) and IBW5 (size range 195 – 225 mm). Only one >0+ brown trout was caught at IBW4 (222 mm) and no >0+ brown trout were captured at sites IBW1 and IBW3 in 2012. In 2014 >0+ brown trout caught at IBW2 were caught in a larger size range (93-463 mm) than the other sites, 95-167 mm at IBW4 and 91-130 mm at site IBW5; two >0+ brown trout were captured at IBW3 of 128 mm and 177 mm. One >0+ brown trout was caught at IBW1 (134 mm) and another individual at IBW6 (104 mm) (Figure 5.7).

Length distributions of brown trout in the Ingbirchworth Dike reference sites varied over the study period (Figures 5.8 and 5.9). Sites R23, R24, R27 & R28 had similar size classes of 0+ brown trout over the three study years typically ranging from 50 – 91mm (Figures 5.8). Site R25 had the largest 0+ brown trout captured in 201 (72-100 mm) (Figures 5.8) and R29 in the same year had the smallest 0+ brown trout (42-78mm) (Figures 5.9).

Overall site R26 (132-375 mm in 2012, 148-311 mm in 2013 and 138-324 mm in 2014) and had larger >0+ brown trout, in comparison to the other Ingbirchworth reference sites which typically had smaller >0+ brown trout with very few >300mm, for example at R27 size ranges were 108-235 mm in 2012, 104-210 mm in 2013 and 98-234 mm in 2014 (Figure 5.8). R29 had brown trout >300mm but the ranges were also very large with small >0+ individuals (128-340 mm in 2012, 103-241 mm in 2013 and 98-340 mm in 2014) (Figure 5.9)

Comparing length distributions at Ingbirchworth Dike impact sites to the reference sites was clear that there were more brown trout captured with greater size ranges (Figure 5.7 - 5.9). There was some overlap in length distributions for >0+brown trout but due to the low numbers at Ingbirchworth Dike it makes it difficult to compare. Where numbers were slightly higher in 2012, ranges were smaller, typically between 195 – 225mm with a few larger individuals up to 485mm at IBW2. In 2014, smaller >0+ trout were captured at Ingbirchworth sites similar to those ranges found at R29.

5.3.1.3 Growth rates of brown trout

Overall, growth rates appear to become more widely distributed the older the brown trout become (Figure 5.10 a-c & 5.11 a-c). In the first year, growth rates were similar (Slow) between reference sites and Ingbirchworth Dike sites (Figure 5.10 a-c & 5.11 a-c). In the second year of life brown trout at Ingbirchworth sites (excluding IBW2 in 2013) were slow to average over the study period as were reference sites(Figure 5.10 a-c & 5.11 a-c). In the third year brown trout growth were slow for the Ingbirchworth sites (excluding IBW2 in 2012), and slow to fast for the reference sites (Figure 5.10 a-c & 5.11 a-c). IBW2 had fast growing brown trout in the second year in 2013, there were no comparable reference sites in that year. In the later years (year 3 plus), brown trout at IBW2 growths were average to very fast, no other reference sites had growths very fast but sites R24 and R26 in 2012 and R29 were fast growing at age 4.

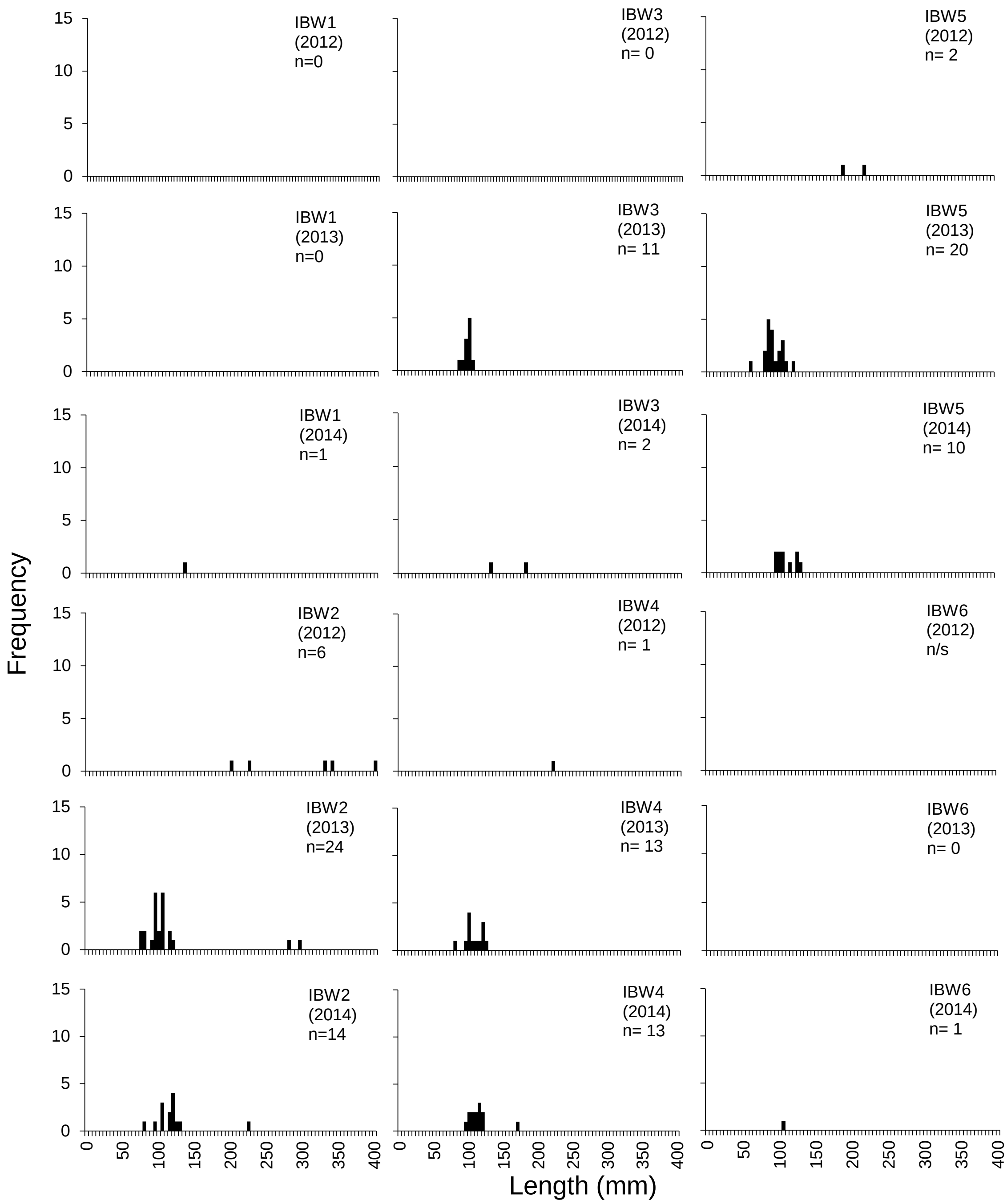


Figure 5.7. Length distribution of brown trout at sites IBW1 – IBW6 on Ingbirchworth Dike, 2012- 2014 (n/s = not sampled) (one brown trout of 482 was caught at IBW2 but not displayed on the graph).

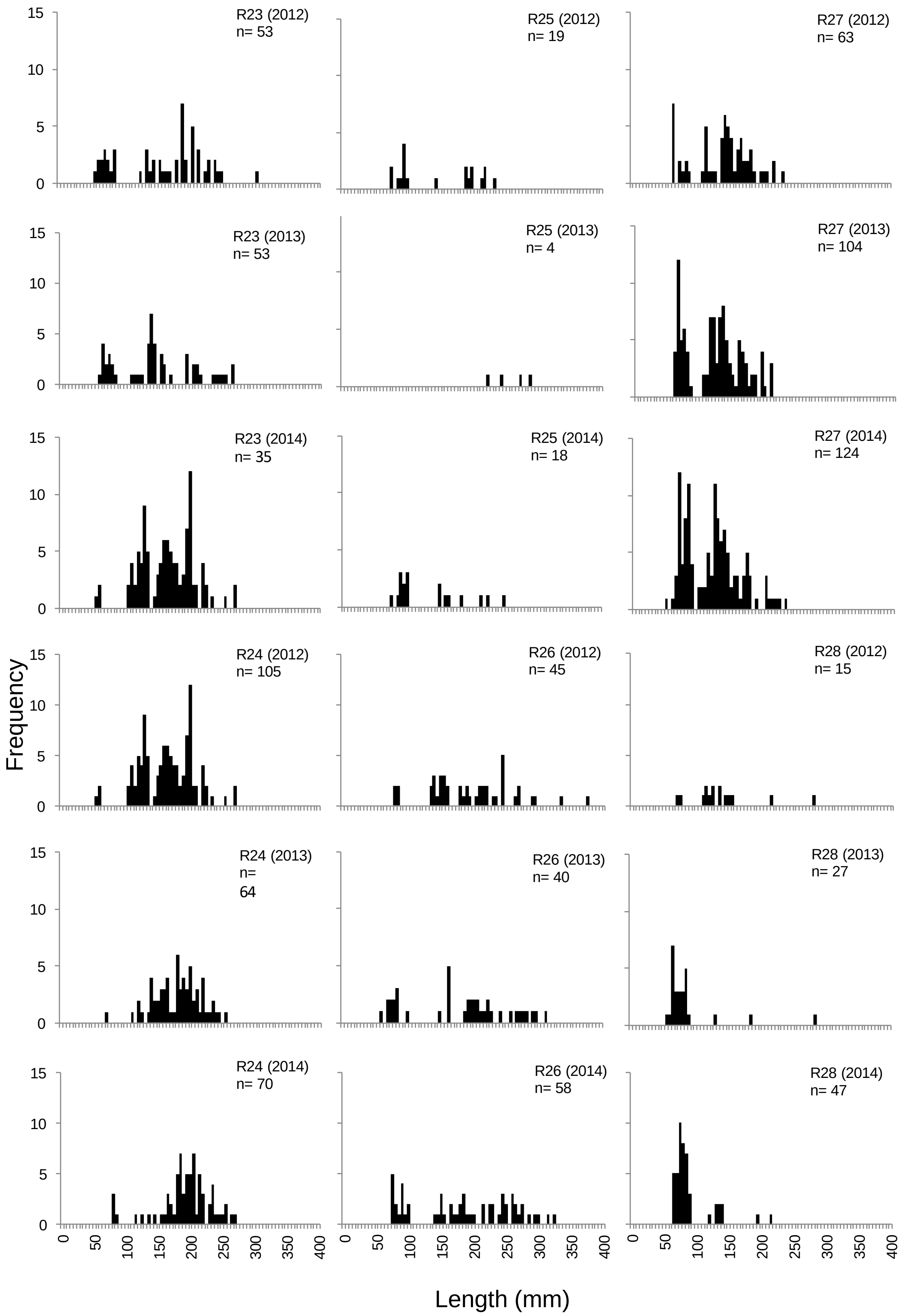


Figure 5.8. Length distributions of brown trout in Ewden Beck (R23-24), Little Don (R25), River Don (R26-R27) and River Sheaf (R28), 2012-2014.

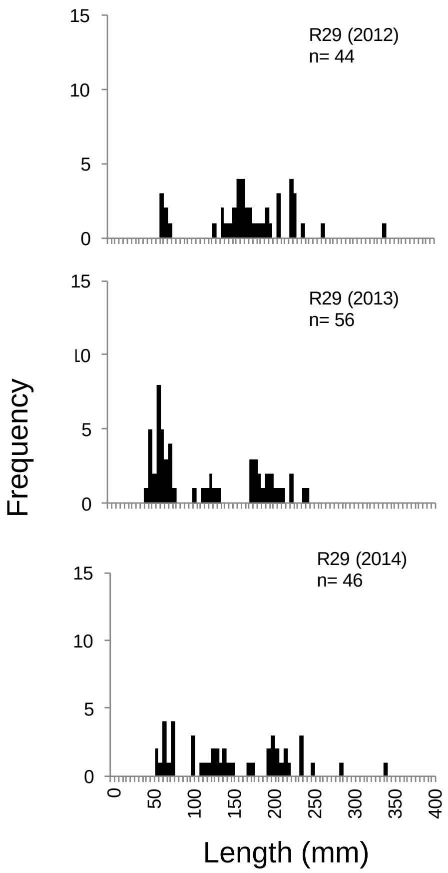


Figure 5.9. Length distributions of brown trout in Wyming Brook (R29), 2012-2014.

5.3.1.4 HABSCORE analysis and outputs

Ingbirchworth Dike

HABSCORE outputs for the sites on the Ingbirchworth Dike revealed variations in the observed densities, predicted densities and habitat utilisation by brown trout (Tables 5.4-6.6). HABSCORE data indicate that 0+ brown trout densities at all sites were lower than predicted from the Habitat Quality Score (HQS); 0+ brown trout densities were significantly lower (HUI lower C.L. <1) at sites IBW1, IBW3, IBW4 and IBW6 (Table 5.4). Overall actual 0+ brown trout populations were zero to 3.56 fish per 100 m^2 (absent to fair/poor) while HQS predicts 4.31 to 23.16 fish per 100 m^2 (fair/poor to good)

Observed densities of >0+ trout (<20 cm) were lower than predicted from the HQS at sites IBW1, IBW3, IBW4 and IBW5, however the density was only significantly lower (HUI lower C.L. <1) at site IBW1 (Table 5.5). IBW1 observed densities were 0.89 fish per

100 m^2 and the HQS predicted 13.86 fish per 100 m^2 (class B (Good)). The observed density of >0+ trout (<20 cm) was higher than predicted from the HQS at sites IBW2 and IBW4 although these were not significant (Table 5.5). The observed densities of >0+ trout (>20 cm) were lower than predicted from the HQS at all sites; >0+ trout (>20 cm) densities were significantly lower (HUI lower C.L. <1) at sites IBW3, IBW4 and IBW6 (Table 5.6). Overall actual 0+ brown trout populations were zero to 1.74 fish per 100 m^2 (absent to poor) while HQS predicts 2.69 to 9.65 fish per 100 m^2 (fair/poor to average).

Ingbirchworth Dike reference sites

HABSCORE outputs for the sites on Ingbirchworth Dike reference sites revealed variations in the observed densities, predicted densities and habitat utilisation by brown trout (Tables 5.7-5.9). 0+ brown trout densities lower than predicted was found at Ingbirchworth Dike reference sites, except at site R27 where observed 0+ brown trout densities were good (26.9 fish per 100 m^2), higher than predicted HQS density (fair/poor (7.42 fish per 100 m^2)); 0+ brown trout densities were significantly lower (poor(2.3 per 100 m^2)) (HUI upper C.L. <1) at R24 than predicted by HQS (average (8.45 per 100 m^2)) (Table 5.7). Observed densities of >0+ trout (<20 cm) were higher than predicted at all sites, except site R25, however the densities were not significantly different than predicted (Table 5.8). Observed densities of >0+ trout (>20 cm) were higher than predicted at sites R23, R24, R26, R27 and R29, significantly so at site R23 (HUI lower C.L. >1), observed density was 11.8 per 100 m^2 (average) and the HQS predicted 1.09 per 100 m^2 (poor) (Table 5.9).

Comparing HQS at Ingbirchworth Dike to reference sites, results reveal that on the whole Ingbirchworth Dike HQS was on average higher for all three brown trout categories and the highest HQS were always scored at the Ingbirchworth Dike sites. On occasions HQS were similar between Ingbirchworth Dike and reference sites; 0+ and >0+ (<20cm) brown trout populations at IBW2 had similar HQS to those at R26, both classified as (fair/poor). IBW2 and IBW4 had similar HQS for >0+ (>20cm) brown trout to R24 and R29, classified as poor.

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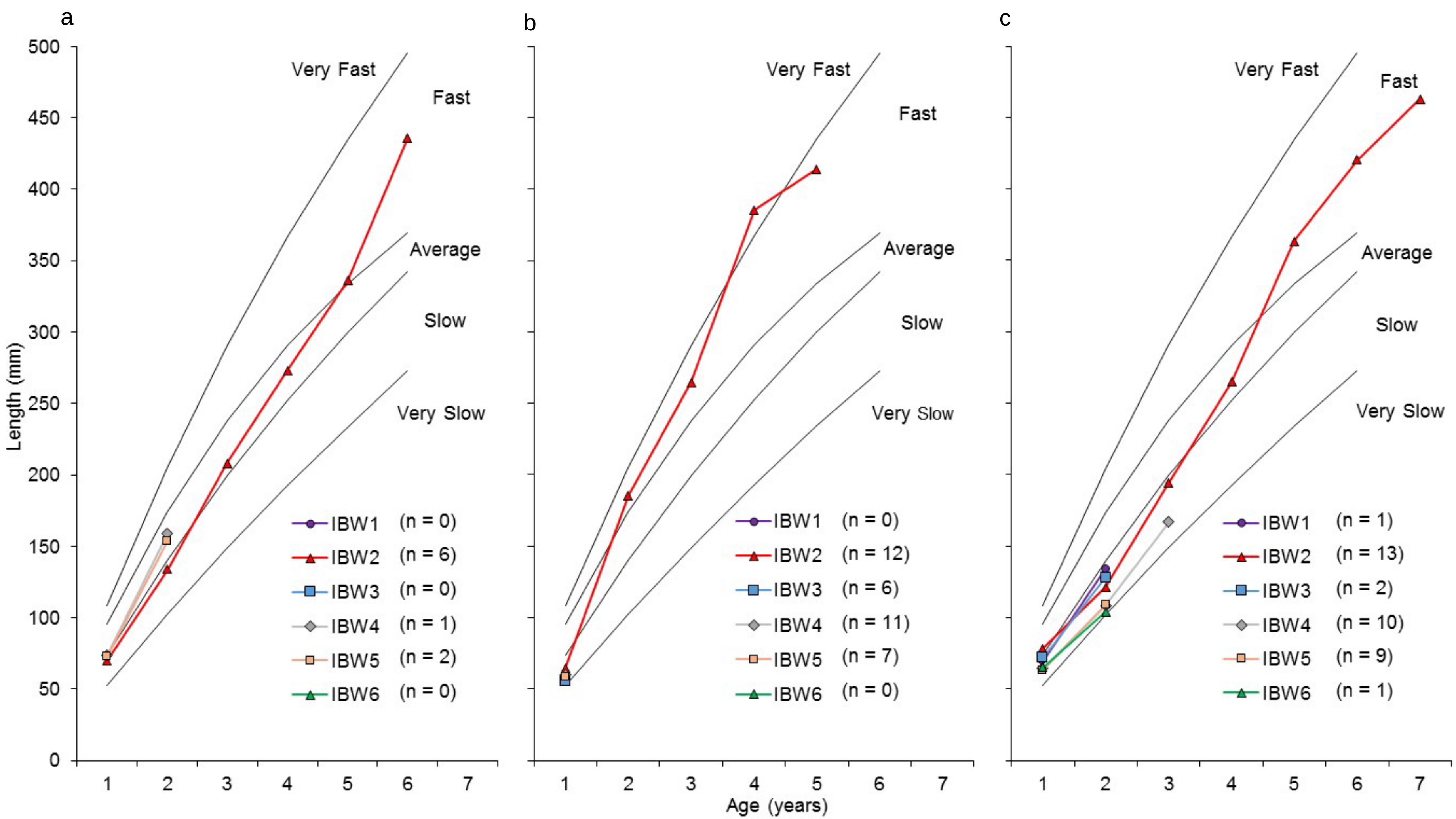

Figure 5.10. Growth rates of brown trout at Ingbirchworth Dike in (a) 2012, (b) 2013 and (c) 2014 compared to EA National Fisheries Laboratory standards (unpublished data).

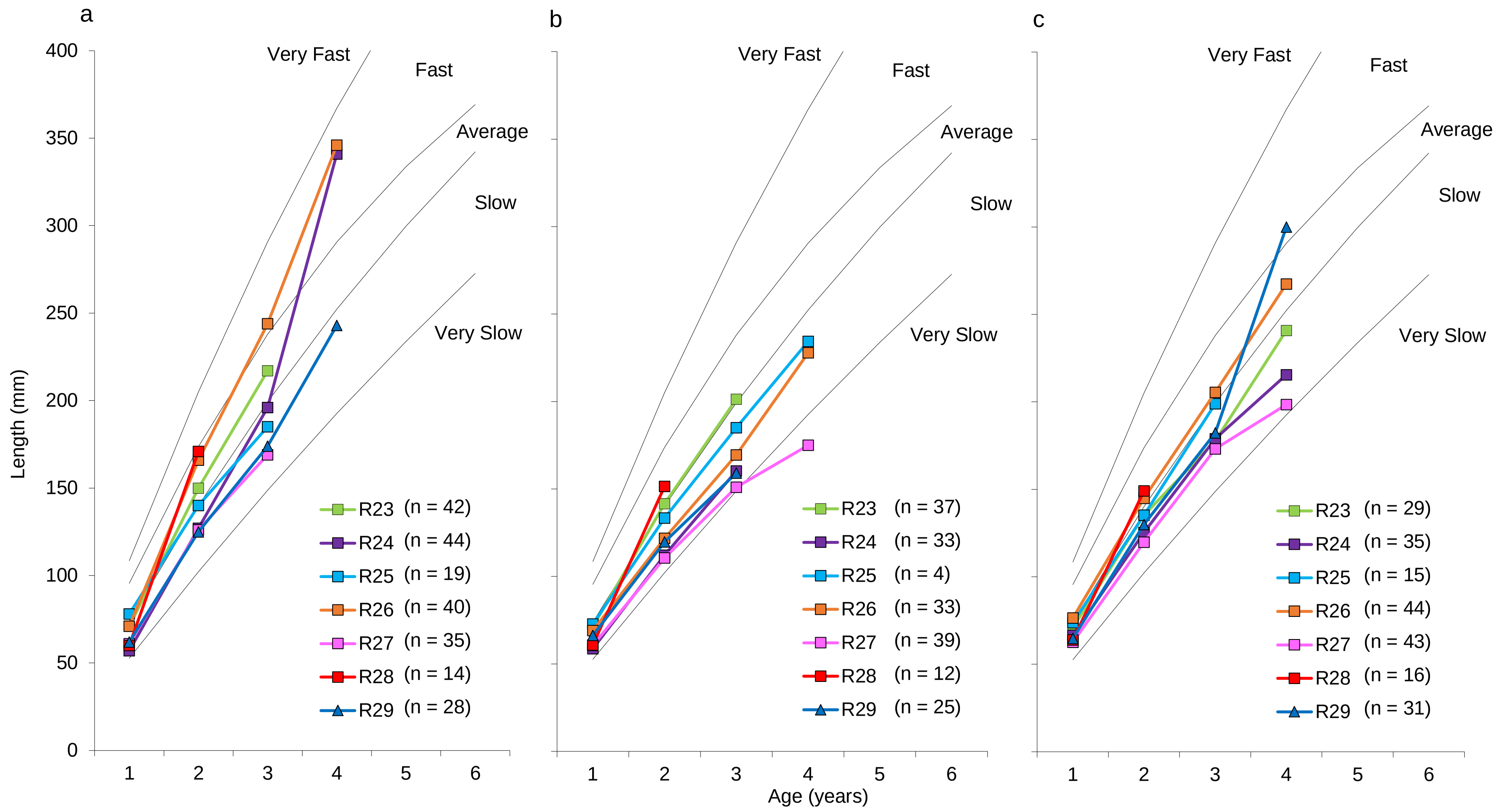

Figure 5.11. Growth rates of brown trout at Ewden Beck (R23-24), Little Don (R25), River Don (R26-R27), River Sheaf (R28) and Wyming Brook (R29), in (a) 2012, (b) 2013 and (c) 2014 compared to EA National Fisheries Laboratory standards (unpublished data).

Table 5.4. HABSCORE outputs for 0+ brown trout in Ingbirchworth Dike, 2014 habitat data incorporating fish data from annual surveys between 2012 and 2014. Shaded area represents sites where the observed population was significantly higher (HUI lower C.L. column; blue) or lower (HUI upper C.L. column; red) than would be expected under pristine conditions.

Site identifier	Observed Number	Observed density	HQS (density)	HQS lower C.L.	HQS upper C.L	HUI	HUI lower C.L.	HUI upper C.L.	Ln (HUI)
IBW1	0.0	0.00	13.86	3.62	53.04	0.06	0.01	0.43	-2.84
IBW2	1.6	0.84	4.31	1.15	16.11	0.19	0.03	1.28	-1.66
IBW3	1.0	0.91	13.01	3.53	47.99	0.07	0.01	0.46	-2.65
IBW4	1.0	1.00	14.46	3.82	54.66	0.07	0.01	0.46	-2.65
IBW5	2.1	3.56	15.24	4.06	57.23	0.23	0.04	1.55	-1.46
IBW6	0.0	0.00	23.16	6.04	88.75	0.07	0.01	0.48	-2.65

Table 5.5. HABSCORE outputs for >0+ (<20 cm) brown trout in Ingbirchworth Dike, 2014 habitat data incorporating fish data from annual surveys between 2012 and 2014. Shaded area represents sites where the observed population was significantly higher (HUI lower C.L. column; blue) or lower (HUI upper C.L. column; red) than would be expected under pristine conditions.

Site identifier	Observed number	Observed density	HQS (density)	HQS lower C.L.	HQS upper C.L	HUI	HUI lower C.L.	HUI upper C.L.	Ln (HUI)
IBW1	1.0	0.89	13.86	3.62	53.04	0.06	0.01	0.43	-2.81
IBW2	6.0	3.17	2.59	0.62	10.87	1.22	0.21	7.20	0.19
IBW3	2.7	2.47	10.54	2.56	43.39	0.23	0.01	1.36	-1.46
IBW4	5.5	5.52	11.90	2.83	50.01	0.46	0.08	2.75	-0.77
IBW5	4.8	8.20	7.45	1.80	30.79	1.10	0.19	6.43	0.09
IBW6	1.0	1.64	7.60	1.79	32.24	0.22	0.4	1.29	-1.51

Table 5.6. HABSCORE outputs for >0+ (>20 cm) brown trout in Ingbirchworth Dike 2014, habitat data incorporating fish data from annual surveys between 2012 and 2014. Shaded area represents sites where the observed population was significantly higher (HUI lower C.L. column; blue) or lower (HUI upper C.L. column; red) than would be expected under pristine conditions.

Site identifier	Observed number	Observed density	HQS (density)	HQS lower C.L.	HQS upper C.L	HUI	HUI lower C.L.	HUI upper C.L.	Ln (HUI)
IBW1	0.0	0.00	2.69	0.88	8.25	0.33	0.11	1.01	-1.10
IBW2	3.3	1.74	5.10	1.65	15.73	0.34	0.11	1.05	-1.07
IBW3	0.0	0.00	2.78	0.92	8.37	0.33	0.11	0.98	-1.10
IBW4	1.0	1.00	5.45	1.78	16.70	0.18	0.06	0.56	-1.71
IBW5	1.0	1.71	4.65	1.48	14.57	0.37	0.12	1.16	-0.99
IBW6	0.0	0.00	9.65	2.99	31.21	0.17	0.05	0.55	-1.77

Table 5.7. HABSCORE outputs for 0+ brown trout in Ingbirchworth Dike reference sites, 2014 habitat data incorporating fish data from annual surveys between 2012 and 2014. Shaded area represents sites where the observed population was significantly higher (HUI lower C.L. column; blue) or lower (HUI upper C.L. column; red) than would be expected under pristine conditions.

Site identifier	Observed number	Observed density	HQS (density)	HQS lower C.L.	HQS upper C.L	HUI	HUI lower C.L.	HUI upper C.L.	Ln (HUI)
R23	10.3	3.86	10.36	2.65	40.48	0.38	0.05	2.54	-0.97
R24	2.3	0.78	8.45	2.23	32.01	0.09	0.01	0.62	-2.41
R25	4.5	1.51	7.91	2.14	29.26	0.19	0.03	1.25	-1.66
R26	10.0	1.76	3.04	0.80	11.59	0.58	0.09	3.90	-0.54
R27	26.9	8.30	7.42	2.01	27.40	1.12	0.17	7.33	0.11
R28	12.2	4.41	5.61	1.45	21.69	0.79	0.12	5.32	-0.24
R29	12.9	7.31	16.21	4.18	62.86	0.45	0.07	3.06	-0.80

Table 5.8. HABSCORE outputs for >0+ (<20 cm) brown trout in Ingbirchworth Dike reference sites, 2014 habitat data incorporating fish data from annual surveys between 2012 and 2014. Shaded area represents sites where the observed population was significantly higher (HUI lower C.L. column; blue) or lower (HUI upper C.L. column; red) than would be expected under pristine conditions.

Site identifier	Observed number	Observed density	HQS (density)	HQS lower C.L.	HQS upper C.L	HUI	HUI lower C.L.	HUI upper C.L.	Ln (HUI)
R23	23.8	8.94	7.85	1.86	32.36	1.14	0.19	6.77	0.13
R24	51.2	17.40	5.23	1.27	21.53	3.33	0.57	19.39	1.20
R25	2.9	0.99	5.66	1.38	23.27	0.17	0.03	1.01	-1.77
R26	17.4	3.08	2.22	0.53	9.27	1.39	0.23	8.18	0.33
R27	61.1	18.86	6.47	1.56	26.77	2.91	0.50	17.03	1.07
R28	5.6	2.02	0.70	0.16	3.07	2.89	0.47	17.75	1.06
R29	22.9	12.93	5.97	1.27	28.21	2.16	0.33	14.11	0.77

Table 5.9. HABSCORE outputs for >0+ (>20 cm) brown trout in Ingbirchworth Dike reference sites, 2014 habitat data incorporating fish data from annual surveys between 2012 and 2014. Shaded area represents sites where the observed population was significantly higher (HUI lower C.L. column; blue) or lower (HUI upper C.L. column; red) than would be expected under pristine conditions.

Site identifier	Observed number	Observed density	HQS (density)	HQS lower C.L.	HQS upper C.L	HUI	HUI lower C.L.	HUI upper C.L.	Ln (HUI)
R23	11.8	4.42	1.09	0.35	3.39	4.03	1.30	12.51	1.39
R24	21.6	7.33	2.53	0.82	7.82	2.90	0.94	8.96	1.06
R25	3.9	1.32	1.68	0.55	5.11	0.79	0.26	2.39	-0.24
R26	23.0	4.07	1.66	0.55	5.07	2.45	0.80	7.50	0.90
R27	5.4	1.68	0.73	0.24	2.25	2.28	0.75	6.98	0.82
R28	1.3	0.45	0.51	0.16	1.62	0.90	0.28	2.91	-0.11
R29	10.8	6.09	2.37	0.72	7.80	2.57	0.78	8.46	0.94

5.3.1.5 Resource calculation

The resource calculation indicated that the level of "before" sampling in the study reach was adequate to statistically detect a 50% change in 0+ fish density two years after the habitat improvement works (Table 5.10), provided that the same impact (IBW1-IBW5; five sites) and reference sites (R23-R29; seven sites) are sampled and the spatial and temporal variance persist after the stream has been rehabilitated.

Table 5.10. The target variance and the number of years sampling to achieve the actual variance required to statically detect a 50% change in 0+ fish density downstream of Ingbirchworth Reservoir. Red text denotes when the actual variance is below the target variance.

Target Variance	Actual Variance for a specified number of years after habitat improvement works		
	2	3	4
0.109	0.107	0.085	0.075

5.3.2 River Washburn and reference sites

5.3.2.1 Species composition, density estimates and classification

Species composition

Eight fish species were captured in the River Washburn prior to river habitat improvement works; the number of each species caught varied between sites and years (Figure 5.12). At the River Washburn reference sites there were seven fish species captured, unlike the River Washburn sites there was less variation in numbers and brown trout were most abundant species at all reference sites excluding R9 where bullhead dominated the catches in all three years (2012-2014) and R8 in 2012 and R17 in 2013 (Figure 5.13). Stone loach were more abundant than brown trout in 2013 at the reference site R9. One European brook lamprey (*Lampetra planeri* Bloch) was caught at R9 in 2013. Three-spined stickleback was the most abundant species at SW1 and SW2 in all study years. Brown trout were only abundant at SW3 in 2012, SW5 in 2013 and SW6 in 2012. Bullhead was most abundant species at SW3 in 2013 and 2014, SW4 in every study year, SW5 in 2104 and SW6 in 2013 and 2014 (Figure 5.12). Other species captured on the River Washburn in smaller numbers were chub (*Leuciscus cephalus* (L.)), roach, perch, stone loach and minnow.

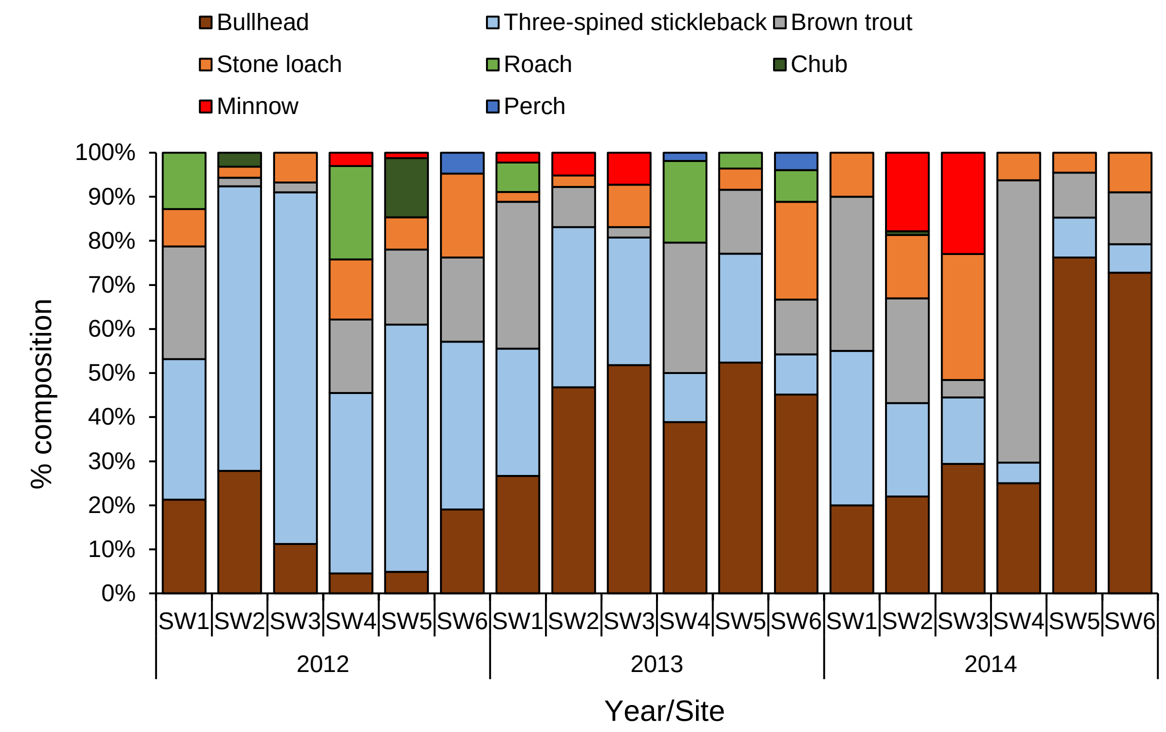

Figure 5.12. Percentage composition of fish captured in the River Washburn annually between 2012 and 2014 (site identifiers in Table 5.2).

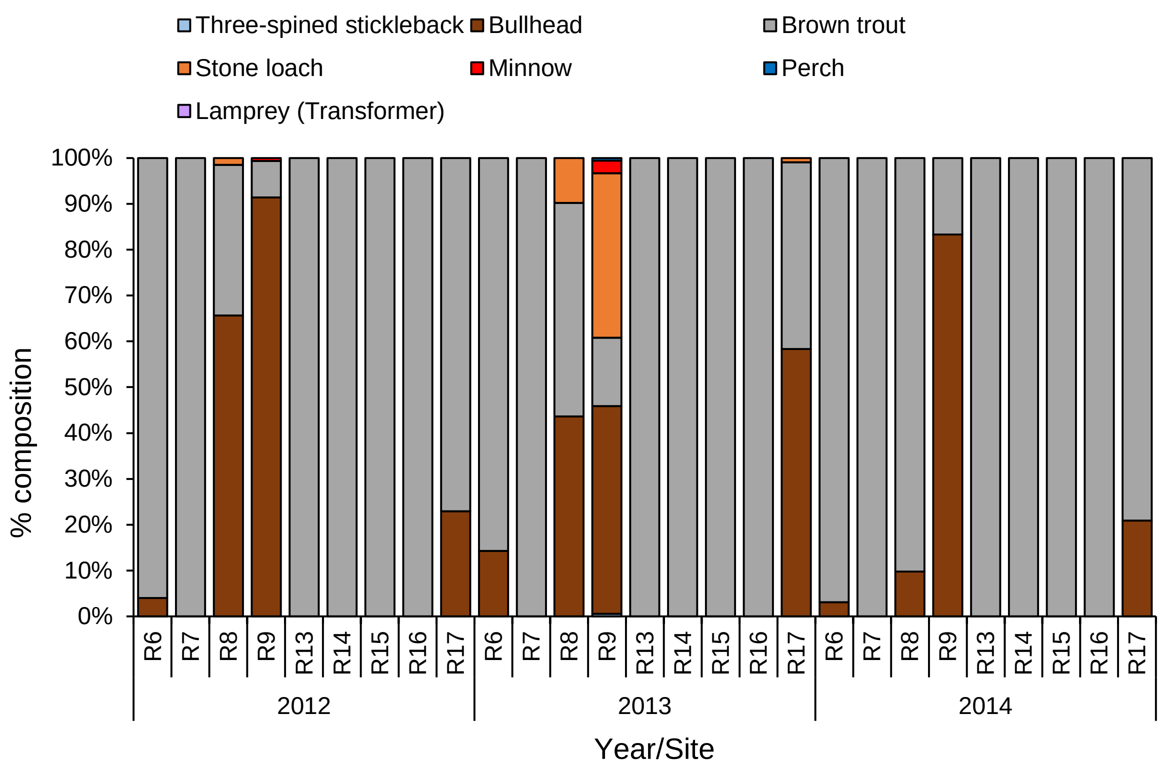

Figure 5.13. Numbers of fish captured in the River Washburn reference sites annually between 2012 and 2014 (site identifiers in Table 5.2).

Brown trout density estimates and classifications

0+ brown trout were absent from several River Washburn sites over the tree study years (SW1 (2012 & 2014) and SW2, SW3 & SW5 (2014)). In 2012 and 2013, 0+ brown trout densities in the River Washburn were poor (0.2 - 2.4 fish per 100 m^2), except at SW6 where populations were average (9.8 fish per 100 m^2) in 2012 and fair/poor (6.6 fish per 100 m^2) in 2013 & 2014 (7.1 fish per 100 m^2) (Table 5.11). 0+ brown trout densities at reference sites ranged from poor to fair/poor in 2012 (0.9 – 3.9 fish per 100 m^2), similar to those in the River Washburn (Table 5.11). In subsequent years (2013 & 2014), 0+ brown trout densities increased over the study period at reference sites, densities ranging from poor (2.6 fish per 100 m^2) excellent (47.0 fish per 100 m^2) at R14 in 2014 (Table 5.11).

Table 5.11. 0+ and >0+ brown trout density (numbers of fish per 100m^2) and abundance classification at survey site locations for the River Washburn habitat improvement works programme.

A (excellent)	B (good)	C (average)	D (fair/poor)	E (poor)	F (fishless)

River Name	Site identifier	0+ brown trout 2012	0+ brown trout 2013	0+ brown trout 2014	>0+ brown trout 2012	>0+ brown trout 2013	>0+ brown trout 2014
Impact sites							
River Washburn	SW1	0.0	0.0	0.0	3.1	0.8	0.1
River Washburn	SW2	0.2	0.9	0.0	2.4	2.1	2.4
River Washburn	SW3	0.5	0.6	0.0	4.2	1.6	1.0
River Washburn	SW4	2.4	1.8	0.4	1.8	5.6	6.4
River Washburn	SW5	0.8	0.8	0.0	1.0	7.3	1.6
River Washburn	SW6	9.8	6.6	7.1	1.3	3.8	5.9
Reference							
Barden Beck	R6	1.5	2.7	27.5	10.3	10.2	14.8
Barden Beck	R7	3.0	9.0	3.0	5.6	6.3	3.4
Ings Beck	R8	3.9	16.1	13.3	5.6	9.5	10.7
Ings Beck	R9	1.5	10.8	1.7	4.5	4.5	1.1
Ashfold Side Beck	R13	0.9	4.0	22.8	14.0	11.6	25.1
Ashfold Side Beck	R14	2.2	2.6	47.0	23.5	7.3	29.8
Ashfold Side Beck	R15	1.9	10.2	35.5	18.1	7.9	21.6
Ashfold Side Beck	R16	1.0	19.1	2.7	8.7	9.5	0.4
Ashfold Side Beck	R17	1.8	5.9	12.0	11.6	11.4	12.7

>0+ brown trout populations varied from poor to average throughout the three study years at River Washburn sites (0.1 – 7.2 fish per 100 m^2) (Table 5.11). Whereas at the >0+ brown trout populations at the River Washburn reference sites were mostly classified as average over the three study years. There were instances where the River Washburn reference sites were far better for example, densities were good (14.0 (R13)

& 18.1 (R15) fish per 100 m^2) and excellent (23.5 fish per 100 m^2 (R14)) in 2012 (Table 5.11). The range of densities were greater at reference sites (0.4 – 29.8 fish per 100 m^2) in comparison to the River Washburn sites.

5.3.2.2 Length distributions of brown trout

Brown trout were captured at all sites (SW1-SW6) in 2012 - 2014 allowing determination of length distributions (Figure 5.14). Discrimination between 0+ & >0+ was difficult therefore ageing was used to facilitate discrimination of age groups (Figure 5.6). 0+ brown trout were absent from SW1 in all three study years.

In 2012, 0+ brown trout were captured in similar size ranges at SW4-SW6 (82-99mm), smaller 0+ brown trout were captured at SW2 (73 mm) and SW3 (50-56 mm) (Figure 5.14). In 2013, size ranges of 0+ brown trout were all similar and comparable to 2012 at SW2, SW3, SW4 and SW6 in the size range 86-104 mm, 91-108 mm, 81-98 mm and 76-104 mm respectively (Figure 5.14). 0+ brown trout were absent from the three most upstream sites in 2014. 0+ brown trout at sites SW5 and SW6 were in the size range 76-88 mm and 80-97 mm respectively, only one 0+ brown trout was caught at SW4 (87 mm) (Figure 5.14).

In 2012, River Washburn reference sites had smaller and more varied size ranges (60 – 103 mm) of 0+ brown trout in comparison to the River Washburn Sites. In 2013, size ranges of 0+ brown trout at River Washburn reference sites were again more varied, particularly at site R8 where the range was between 37-91 mm. In 2014, size ranges of 0+ brown trout were less varied and larger at sites R6-R9 and R16 than in previous years and were more comparable to River Washburn sites, whereas sites R13-R15 and R17 had smaller 0+brown trout captured (52-54 mm).

>0+ brown trout were captured at all sites (SW1- SW6) in 2012, with size range 94-247 mm (SW1), 95-268 mm (SW2), 84-184 mm (SW3), 100-249 mm (SW4), 154-224 mm (SW5) and 101-181 mm (SW6). In 2013, >0+ brown trout were caught at all sites in the size range 98-203 mm (SW1), 144-198 mm (SW2), 143-226 mm (SW3), 135-240 mm (SW4), 141-239 mm (SW5) and 134-170 mm (SW6). In 2014, >0+ brown trout were captured at all study sites, size ranges were similar at SW1 (147-238 mm), SW2 (142-257 mm), SW4 (126-263 mm) & SW5 (148-230 mm), smaller >0+ brown trout were captured at SW3 (102-161 mm) and SW6 (104-198 mm).

Length distributions of brown trout in the reference sites varied over the study period (Figure 5.15). Size ranges caught at reference sites were similar to those at the River Washburn sites. In 2012, >0+ brown trout were caught as small as 91 mm at R14 and as large as 306 mm at R6. Larger than any brown trout captured at the River Washburn sites. In 2013, size ranges of >0+ brown trout were similar between reference sites and were only slightly larger at R17 (97-245 mm) (Figure 5.6). Frequently more >0+ trout were caught <100 mm at reference sites in comparison to River Washburn sites. In 2014, similar sized >0+ were captured at River Washburn sites to those caught in 2013 (Figure 5.5 & 5.6).

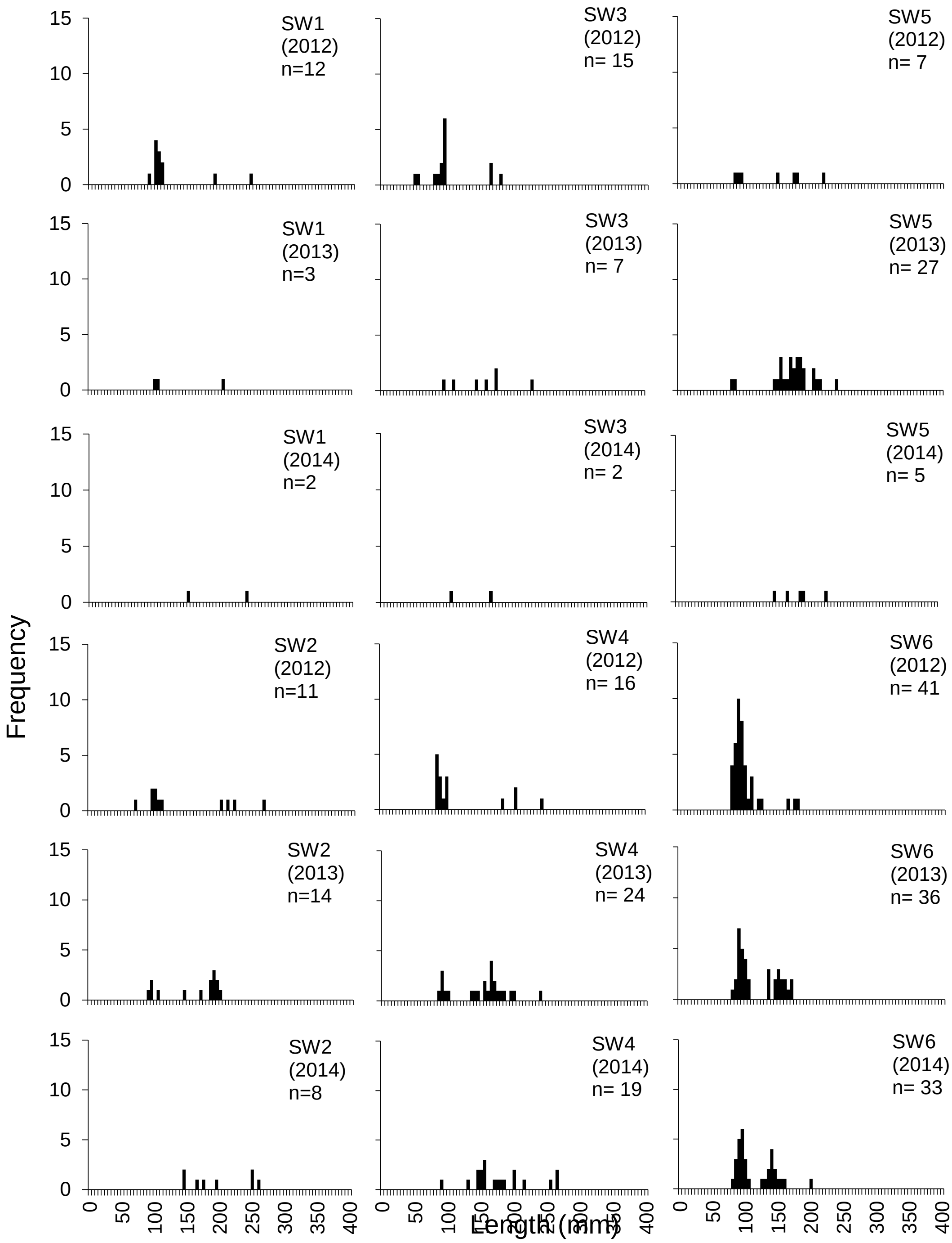

Figure 5.14 Length distribution of brown trout at sites SW1 - SW6 on the River Washburn, 2012- 2014.

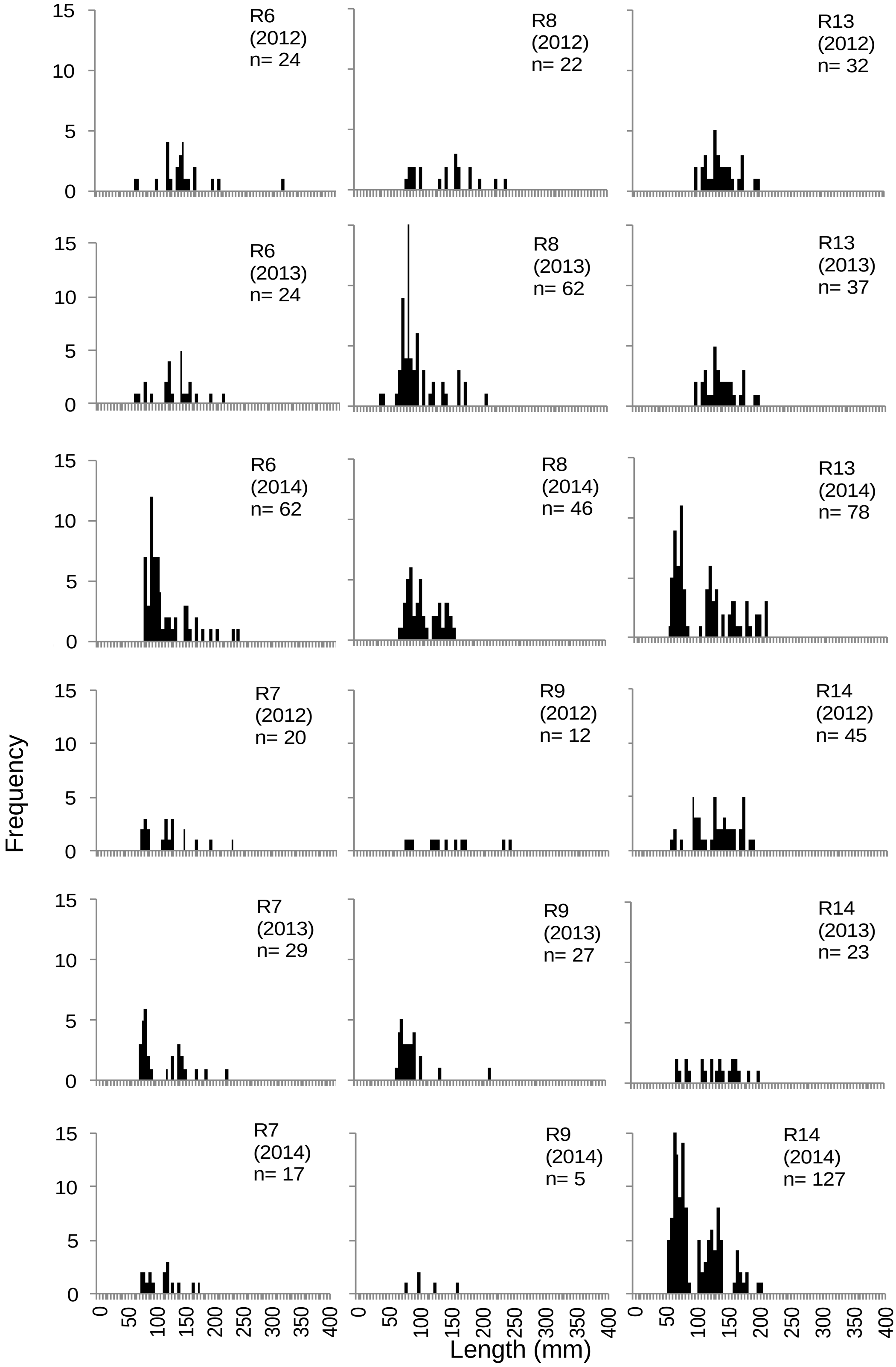

Figure 5.15. Length distributions of brown trout in Barden Beck (R6-R7), Ings Beck (R8-R9) and Ashfold Side Beck (R13-R14), 2012-2014.

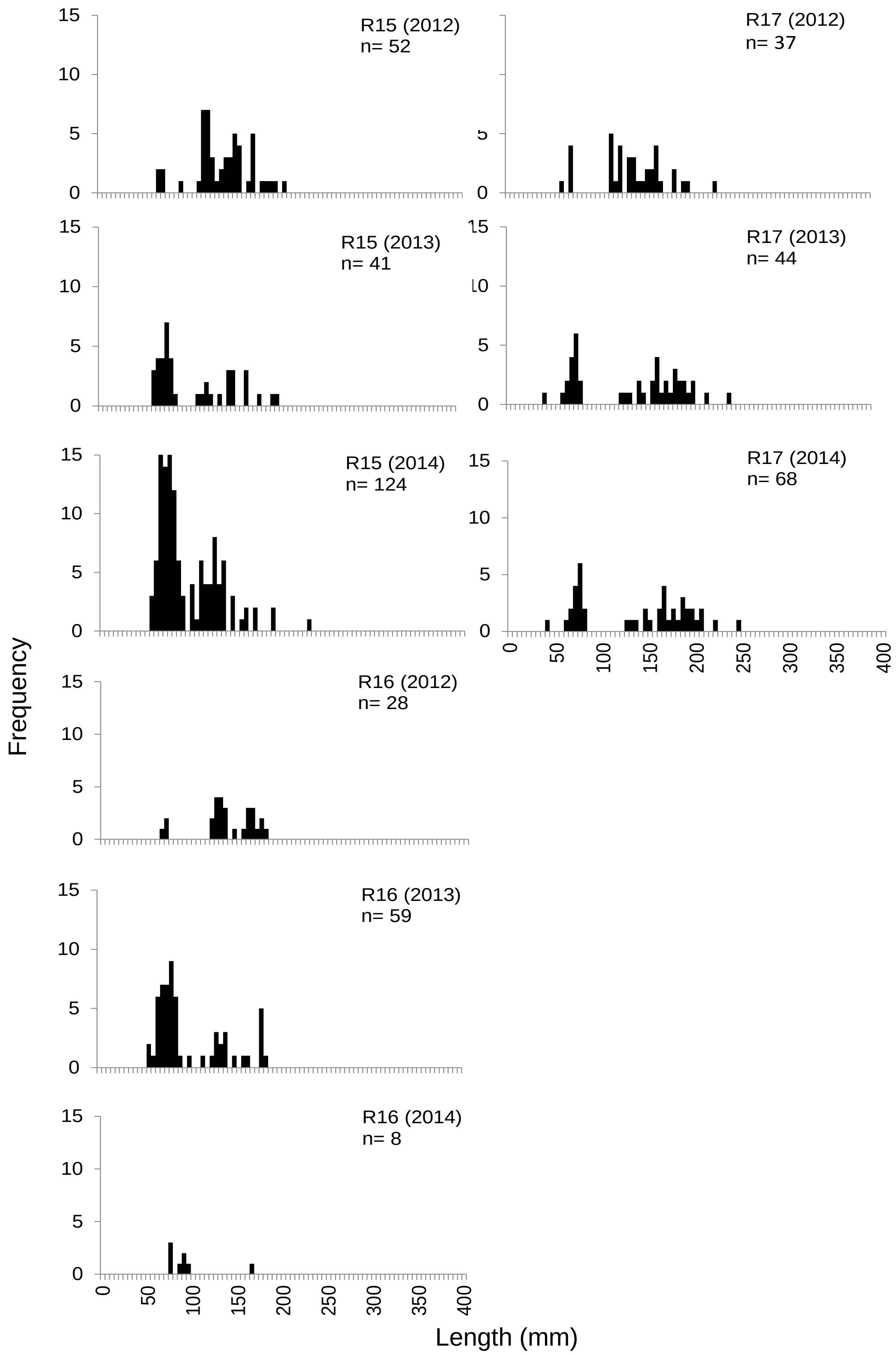

Figure 5.16. Length distributions of brown trout in Ashfold Side Beck (R15-R17), 2012-2014.

5.3.2.3 Growth rates of brown trout

In the River Washburn, growth rates of brown trout in 2012 were slow, except at SW2 at age 2 where growth rates were average (Figure 5.17). In 2013, growth was generally average in the first year of life, except at site SW1 where growth was slow, and was average for older age groups at sites SW1, SW2, SW3, and SW4 (Figure 5.17). Growth of brown trout at site SW5 was above average in older age groups. In 2014 growth of brown trout was slow at all sites in the first two years of life, and at sites where older aged trout were present (SW1, SW2 and SW4) growth remained slow at ages 3 and 4 (Figure 5.17). It should be emphasised the data be treated with caution at sites SW1, SW2, SW3 and SW6 in 2013 and SW1-SW3 and SW5 in 2014 due to the low sample size available for calculation of growth rates (Figure 5.17).

Growth rates of brown trout in the first year of life in the River Washburn reference sites in 2012 were average at sites R8 and R9, very slow at site R13 and slow at all other sites (Figure 18a). In the second year of life growth rates of brown trout were above average at site R9 and slow at all other sites, while in the third year of life growth rates were slow at site R13, very slow at site R14 but fast at R6 (Figure 18a). In 2013 growth rates of brown trout at all River Washburn reference sites were slow in all age groups except at sites R15 and R16, which were very slow in the third year of life and R9 where growth was average in the second year (Figure 18b). Growth rates of brown trout in 2014 were average in the first year of life at site R9 and slow at all other sites and in all other age groups (Figure 18c).

Grown rates of brown trout between River Washburn and the reference sits, were comparable, generally been classified as slow to average, with only a few brown trout being very slow and fast (Figure 5.17 & 5.18).

5.3.2.4 HABSCORE analysis and outputs

HABSCORE outputs for the sites on the River Washburn revealed variations in the observed densities, predicted densities and habitat utilisation by brown trout (Tables 6.12 – 6.14). HABSCORE data indicated that 0+ brown trout densities at all sites were lower than predicted from the Habitat Quality Score (HQS); 0+ brown trout densities were significantly lower (HUI lower C.L. <1) at sites SW1-SW5 (Table 5.12). The observed densities of >0+ trout (<20 cm) were lower than predicted from the HQS at sites SW1, SW2, SW3, and SW5 and marginally higher than predicted at sites SW4 and SW6, however the densities were not significantly lower or higher at any sites (Table 5.13).

The observed densities of >0+ trout (>20 cm) were lower than predicted from the HQS at all sites; densities at SW6 were significantly lower (0.92 fish per 100 m^2) than the predicted HQS (1.29 fish per 100 m^2) (Table 5.14).

HABSCORE outputs reference sites revealed variations in the observed densities, predicted densities and habitat utilisation by brown trout (Tables 5.15-17). HABSCORE data indicated that 0+ brown trout densities at all sites were lower than predicted from the Habitat Quality Score (HQS), except at site R8 where observed 0+ brown trout densities (10.27 per 100 m^2) were higher than predicted (HQS (9.97 per 100 m^2)); in all cases densities were not significantly different (Table 5.15). At reference sites observed densities of >0+ trout (<20 cm) were higher than predicted at sites R8, R13, R14, R15 and R17 and lower than predicted at sites R6, R7, R9 and R16, however the densities were not significantly lower or higher than predicted (Table 5.16). At reference sites observed densities of >0+ trout (>20 cm) were lower than predicted at all sites, significantly so at site R14 (0.60 per 100 m^2) (HUI upper C.L. <1) (Table 5.17).

Comparing HQS between the River Washburn and the reference sites, predicted 0+ brown trout densities were similar between most sites but predicted to be much higher at R6 and lower densities at SW5. >0+ (< 20 mm) brown trout predicted HQS on the River Washburn sites were comparable to most reference sites (fair/poor), other than R6 which predicted much higher densities, classified as good (12.75 per 100 m^2) and average at R13 (9.00 per 100 m^2) and R14 (8.71 per 100 m^2). HQS on the River Washburn and all of the reference sites provided very little predicted >0+ (>20 mm) brown trout habitat classified as poor. Whereas Reference site R14 was predicted to be slightly better providing fair/poor habitat for >0+ (>20 mm) brown trout.

5.3.2.5 Resource calculation

The resource calculation indicated that the level of “before” sampling downstream of Swinsty Reservoir was adequate to detect statistically a 50% change in 0+ fish density two years after the habitat improvement works (Table 5.18), provided that the same impact (SW1-SW6; six sites)) and reference sites (R6-R9 and R13-R17; nine sites) are sampled and the spatial and temporal variance persist after the River Washburn has been rehabilitated modified.

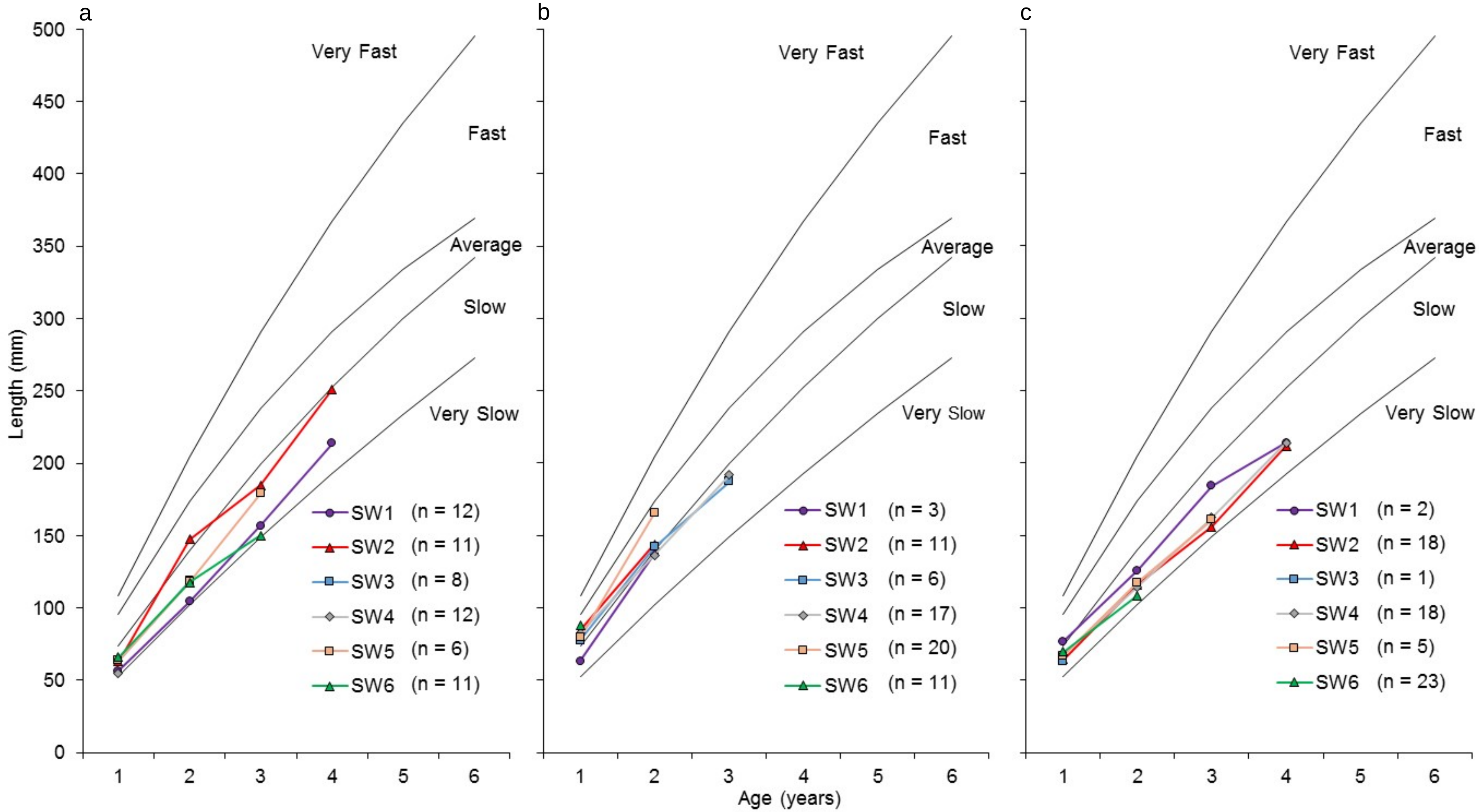

Figure 5.17. Growth rates of brown trout in the River Washburn, (a) 2012, (b) 2013 and (c) 2014 compared to EA National Fisheries Laboratory, unpublished data.

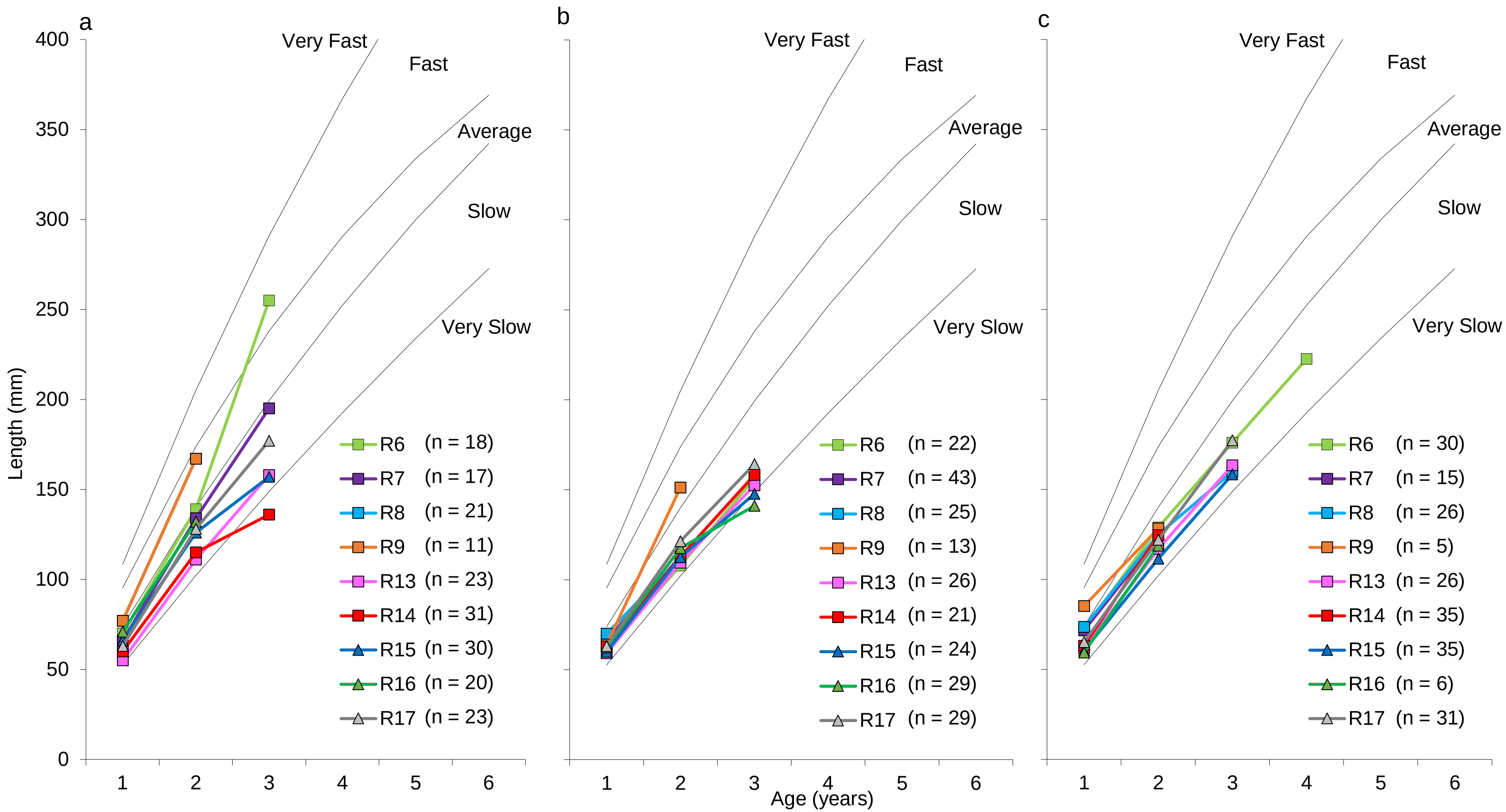

Figure 5.18. Growth of brown trout in Barden Beck (R6-R7), Ings Beck (R8-R9) and Ashfold Side Beck (R13-R17), (a) 2012, (b) 2013 and (c) 2014 compared to EA National Fisheries Laboratory, unpublished data.

Table 5.12. HABSCORE outputs for 0+ brown trout in the River Washburn, 2014 habitat data incorporating fish data from annual surveys between 2012 and 2014. Shaded area represents sites where the observed population was significantly higher (HUI lower C.L. column; blue) or lower (HUI upper C.L. column; red) than would be expected under pristine conditions.

Site identifier	Observed number	Observed density	HQS (density)	HQS lower C.L.	HQS upper C.L	HUI	HUI lower C.L.	HUI upper C.L.	Ln (HUI)
SW1	0.0	0.00	14.30	3.72	54.91	0.03	0.00	0.20	-3.50
SW2	1.6	0.48	7.65	2.03	28.90	0.06	0.01	0.41	-2.81
SW3	1.6	0.82	14.50	3.80	55.25	0.06	0.01	0.38	-2.81
SW4	3.8	1.31	10.28	2.71	38.93	0.13	0.02	0.85	-2.04
SW5	2.1	0.65	5.77	1.51	22.12	0.11	0.02	0.76	-2.20
SW6	25.0	9.62	13.06	3.40	50.13	0.74	0.11	4.96	-0.30

Table 5.13. HABSCORE outputs for >0+ (<20 cm) brown trout in the River Washburn, 2014 habitat data incorporating fish data from annual surveys between 2012 and 2014. Shaded area represents sites where the observed population was significantly higher (HUI lower C.L. column; blue) or lower (HUI upper C.L. column; red) than would be expected under pristine conditions.

Site identifier	Observed number	Observed density	HQS (density)	HQS lower C.L.	HQS upper C.L	HUI	HUI lower C.L.	HUI upper C.L.	Ln (HUI)
SW1	43.5	1.53	3.80	0.91	15.94	0.40	0.07	2.39	-0.91
SW2	7.1	2.13	2.84	0.68	11.75	0.75	0.13	4.40	-0.28
SW3	5.0	2.59	3.72	0.89	15.47	0.70	0.12	4.11	-0.35
SW4	9.4	3.25	3.11	0.75	12.97	1.05	0.18	6.15	0.04
SW5	6.3	1.98	2.09	0.50	8.78	0.95	0.16	5.60	-0.05
SW6	10.2	3.92	2.85	0.68	11.90	1.37	0.23	8.09	0.31

Table 5.14. HABSCORE outputs for >0+ (>20 cm) brown trout in the River Washburn, 2014 habitat data incorporating fish data from annual surveys between 2012 and 2014. Shaded area represents sites where the observed population was significantly higher (HUI lower C.L. column; blue) or lower (HUI upper C.L. column; red) than would be expected under pristine conditions.

Site identifier	Observed number	Observed density	HQS (density)	HQS lower C.L.	HQS upper C.L	HUI	HUI lower C.L.	HUI upper C.L.	Ln (HUI)
SW1	1.0	0.43	0.98	0.32	2.95	0.45	0.15	1.34	-0.79
SW2	2.0	0.60	0.84	0.28	2.54	0.71	0.23	2.16	-0.34
SW3	1.0	0.51	1.42	0.46	4.37	0.36	0.12	1.11	-0.35
SW4	3.1	1.07	1.53	0.50	4.67	0.70	0.23	2.17	-0.35
SW5	1.7	0.54	0.74	0.24	2.23	0.73	0.24	2.21	-0.31
SW6	0.0	0.00	1.29	0.42	3.97	0.30	0.10	0.92	-1.20

Table 5.15. HABSCORE outputs for 0+ brown trout in the River Washburn reference sites, 2014 habitat data incorporating fish data from annual surveys between 2012 and 2014. Shaded area represents sites where the observed population was significantly higher (HUI lower C.L. column; blue) or lower (HUI upper C.L. column; red) than would be expected under pristine conditions.

Site identifier	Observed number	Observed density	HQS (density)	HQS lower C.L.	HQS upper C.L	HUI	HUI lower C.L.	HUI upper C.L.	Ln (HUI)
R6	8.5	5.60	30.27	7.77	117.92	0.19	0.03	1.26	-1.66
R7	9.8	3.79	14.49	3.77	55.62	0.26	0.04	1.76	-1.35
R8	21.2	10.27	7.97	2.05	31.00	1.29	0.19	8.75	0.25
R9	5.6	3.10	16.12	4.22	61.58	0.19	0.03	1.29	-1.66
R13	9.1	5.13	12.45	3.34	46.38	0.41	0.06	2.72	-0.89
R14	12.4	7.46	12.87	3.4	48.69	0.58	0.09	3.87	-0.54
R15	20.9	9.38	14.37	3.77	54.82	0.65	0.10	4.37	-0.43
R16	9.4	3.76	13.17	3.47	49.94	0.29	0.04	1.91	-1.24
R17	13.5	5.07	15.35	4.02	58.62	0.33	0.05	2.21	-1.11

Table 5.16. HABSCORE outputs for >0+ (<20 cm) brown trout in the River Washburn reference sites, 2014 habitat data incorporating fish data from annual surveys between 2012 and 2014. Shaded area represents sites where the observed population was significantly higher (HUI lower C.L. column; blue) or lower (HUI upper C.L. column; red) than would be expected under pristine conditions.

Site identifier	Observed number	Observed density	HQS (density)	HQS lower C.L.	HQS upper C.L	HUI	HUI lower C.L.	HUI upper C.L.	Ln (HUI)
R6	19.3	12.72	12.75	3.01	53.97	1	0.17	5.94	0.00
R7	10.6	4.08	4.81	1.13	20.40	0.85	0.14	5.05	-0.16
R8	16.9	8.21	3.61	0.83	15.59	2.28	0.38	13.78	0.82
R9	4.6	2.58	3.50	0.84	14.58	0.74	0.13	4.32	-0.30
R13	31.4	17.66	9.00	2.13	37.97	1.96	0.33	11.65	0.67
R14	32.5	19.51	8.71	1.98	38.29	2.24	0.37	13.75	0.81
R15	33.9	15.23	5.02	1.20	21.02	3.03	0.51	17.87	1.11
R16	8.1	3.24	4.40	1.05	18.43	0.74	0.12	4.36	-0.30
R17	29.5	11.04	3.75	0.90	15.70	2.94	0.50	17.36	1.08

Table 5.17. HABSCORE outputs for >0+ (>20 cm) brown trout in the River Washburn reference sites, 2014 habitat data incorporating fish data from annual surveys between 2012 and 2014. Shaded area represents sites where the observed population was significantly higher (HUI lower C.L. column; blue) or lower (HUI upper C.L. column; red) than would be expected under pristine conditions.

Site identifier	Observed number	Observed density	HQS (density)	HQS lower C.L.	HQS upper C.L	HUI	HUI lower C.L.	HUI upper C.L.	Ln (HUI)
R6	1.6	1.05	1.54	0.50	4.73	0.68	0.22	2.11	-0.39
R7	1.0	0.38	0.84	0.27	2.64	0.46	0.15	1.45	-0.78
R8	1.3	0.61	1.81	0.58	5.67	0.34	0.11	1.06	-1.08
R9	1.3	0.70	1.60	0.53	4.86	0.44	0.14	1.33	-0.82
R13	1.7	0.96	1.55	0.50	4.87	0.62	0.20	1.94	-0.48
R14	1.0	0.60	2.74	0.85	8.85	0.22	0.07	0.71	-1.51
R15	1.0	0.45	1.21	0.39	3.73	0.37	0.12	1.14	-0.99
R16	0.0	0.00	0.84	0.27	2.61	0.47	0.15	1.46	-0.76
R17	2.3	0.86	1.04	0.34	3.21	0.83	0.24	2.55	-0.19

Table 5.18. The target variance and the number of years sampling to achieve the actual variance required to statically detect a 50% change in 0+ fish density downstream of Swinsty Reservoir. Red text denotes when the actual variance is below the target variance.

Target Variance	Actual Variance for a specified number of years after habitat improvement works		
	2	3	4
0.180	0.065	0.052	0.045

5.4 DISCUSSION

The results from surveys at Ingbirchworth Dike, River Washburn and associated reference sites provided an indication of the baseline status of the fish populations and habitat present in the rivers prior to habitat improvement works. Species composition and brown trout densities varied at all planned habitat improvement sites and between the three study years. Data collected were sufficient to carry out resource calculations to determine the number of years of post- habitat improvement data collection required to detect a 50% change in 0+ fish density.

5.4.1 Ingbirchworth Dike

The EA and Yorkshire Water Services identified Ingbirchworth Dike fish populations to be failing for WFD. This was confirmed by the surveys carried out where many of the brown trout densities were poor and even absent, and recruitment of 0+ brown trout was weak of absent.

Annual variations in the brown trout population densities were identified at Ingbirchworth Dike. For example in 2012, 0+ brown trout were absent at all sites but only low in 2013 and 2014. In comparison, densities were poor in 2012 at several of the reference sites suggesting a poor year for recruitment. Rainfall in in the UK from April to July 2012 was reported to be exceptionally wet (Parry *et al*., 2013) and periods of high rainfall leading to flooding can be detrimental to brown trout during emergence periods when juvenile brown trout are very vulnerable and easily displaced (Daufresne *et al*., 2005; Acreman & Ferguson, 2010; Jonsson & Jonsson, 2011). >0+ brown trout were comparable to 0+ brown trout densities, in that densities in 2012 were lower than the subsequent years, particularly at sites IBW4 and IBW5 where >0+ brown trout densities were poor in 2012 and good in 2013 and 2014. Populations could not have increased due to recruitment from the previous year because there were no 0+ brown trout captured in 2012. The most likely explanation is >0+ brown trout moving upstream from downstream reaches into the sampling area. >0+ brown trout densities at reference sites were fairly consistent over the three sampling years, highlighting the importance of reference sites to help eliminate the temporal and spatial variability in brown populations when carrying out impact assessment of the habitat improvement works.

Prior to fish surveys, lack of habitat diversity was considered to be one of the contributing factors for the poor densities of brown trout. This was confirmed by the HABSCORE outputs; according to the HQS overall predicted brown trout densities were often only predicted to be poor/fair and average, providing evidence that there is potential for habitats to be improved for brown trout and that they are limited by the habitat available. For example, predicted densities of >0+ (>20cm) were average to fair/poor, suggesting that the river habitat available supports only a small number of larger trout. Lower population densities of larger trout could be a result of the lack of deep pooled habitats, although the largest of the brown trout caught was located at IBW2. At this site there were deeper habitats (approx. 10%) in comparison to the other sites with no deep areas apart from IBW4 (with approx. 6%). Limited availability of pools can restrict the presence of large brown trout, which could have led to them moving out of the study reach to more suitable habitats (Heggenes, 1996; Jonsson & Jonsson, 2011).The available cover was limited at most of the sites and absent at IBW1 and IBW6. Shelter availability can be a factor regulating fish density, and also influence growth of salmonids (Millidine *et al*., 2006; Finstad *et al*., 2007, 2009). The growth of brown trout in Ingbirchworth Dike was generally slow in the first year of life. Flow at all sites was dominated by shallow glides and slack areas, with very little (<20%) to no (IBW3) shallow turbulent areas. It is possible that brown trout at Ingbirchworth Dike have been competing with each other and older brown trout for the limited availability of habitat resulting in low population densities and slow growth. Competition could have led to high levels of mortality, especially in the first year, thus being the reason for low levels of recruitment (Lobón-Cerviá, 2007). The sample size suitable for ageing of trout was, however, small and this might not be a true reflection so this conclusion should be treated with caution.

Although brown trout densities were low in Ingbirchworth Dike it was evident that there is some suitable brown trout habitat in the study reach. HABSCORE analysis reinforced this; for example, HQS outputs suggested that 0+ brown trout populations at four of the six sites should be better under pristine conditions. The HQS predicted densities of 0+ brown trout to be between fair/poor and good, whereas in reality the densities were absent to poor. This was also found for >0+(>20cm) brown trout suggesting that habitat availability is not the only factor limiting brown trout population abundance, but pressures including flow regulation, river engineering, high nutrient loading from surrounding land, siltation and lack of food resources are also likely contributory (Pulg *et al*., 2013).

Ingbirchworth Reservoir regulates the flow of the Ingbirchworth Dike, the consequences of flow regulation on the dike has meant that the flow regime is less varied and, as discussed previously, less varied flow regimes can have negative impacts (Chapter 2.3.1). Silt issues are associated with slow flowing and less variable flow regimes (Schindler Wildhaber *et al*., 2014), which was apparent at Ingbirchworth Dike. Substrate types throughout the habitat improvement reach were dominated by cobles and gravel but the levels of silt were common. Siltation can reduce recruitment success as a result of fine sediment reducing the supply of oxygen, blocking the micropores in the egg surface and accumulating in the gravels reduces the permeability of the gravels reducing intra-gravel flow and reoxygenation (Greig *et al*., 2005; Schindler Wildhaber *et al*., 2014). Sediment issues on Ingbirchworth Dike were exacerbated by cattle poaching eroding the banks of the Dike. Ingbirchworth Dike is heavily modified not only as a result of it being regulated by Ingbirchworth Reservoir but at site IBW2 the river bed was artificial (concrete bed) and the channel was over widened. Despite these issues IBW2 was the site where the largest brown trout was caught in the three consecutive sampling years, suggesting this site was suitable for larger trout. Nonetheless, HABSCORE results indicated the potential for larger trout populations to be present at the other rehabilitated sites. Although it was not monitored, poor water quality could have been a contributing factor to why the brown trout populations were lower than expected at Ingbirchworth Dike; slow flowing rivers can reduce water quality and oxygen levels (Jonsson & Jonsson, 2011). Any sources of pollution, both point source and diffuse pollution, may over-ride any benefits gained from instream river habitat improvement works (Cowx & Portocarrero Aya, 2011). Macroinvertebrate densities at Ingbirchworth Dike were classified as failing. Drifting invertebrates are the primary food source for brown trout, although size and species vary through growth and development of individual trout, thus lack of available food resources could have limited brown trout populations in Ingbirchworth Dike and also influence the growth rates (Baerum *et al*., 2013; Elliott, 2015).

5.4.2 River Washburn

The EA and Yorkshire Water Services identified the River Washburn for fish populations to be failing for WFD. This was apparent in the surveys carried out where many of the brown trout densities found were poor and even absent, particularly for 0+ brown trout, which suggests failure to recruitment.

Brown trout population densities were fairly stable between years in the River Washburn. 2014 was the worst year for 0+ brown trout populations; they were absent in the top three proposed habitat improvement sites (SW1-SW3), and at SW5, SW2, SW3 and SW5 densities were poor and absent at SW1 in surveys in previous years. As speculated for 2012 for Ingbirchworth Dike, weather and rainfall could have effected 0+ brown trout densities in 2014. Severe winter storms occurred frequently in the UK throughout December, January and February 2013/2014, resulting in widespread flooding from January onwards. This could have caused damage to brown trout redds, by flattening the redds and influx of fine sediment, which has been documented to affect the water exchange, reducing hyporheic flows and the permeability of the gravels in the redds. (Greig *et al.*, 2005; Schindler Wildhaber *et al.*, 2014). By contrast, brown trout densities at reference sites were more variable; 0+ brown trout densities were lowest in 2012 and were higher in 2013 & 2014. The improvements in densities seen at the reference sites in 2013 and 2014 suggest that there is a bottleneck to recruitment in the River Washburn, again highlighting the importance of reference sites accounting for temporal and spatial variability in brown trout populations.

In the River Washburn lack of habitat was considered to be one of the contributing factors to the poor densities of brown trout, which was reflected in the HABSCORE outputs for >0+ brown trout. Predicted HQS for >0+ brown trout < 20 cm and > 20 cm were similar to actual densities of brown trout caught. This suggests that the current habitat on the River Washburn can only support low densities of this size/age group of brown trout. Brown trout population densities of larger than 20 cm were generally low, because the dominant flow types were shallow glides and slack areas, highlighting the lack of sufficient deep pool habitats that larger trout require (mentioned previously in section 5.4.1).

HASCORE results at all River Washburn sites indicated that 0+ brown trout populations should be higher under pristine conditions, significantly so at sites SW1-SW5. Throughout the 3-year study period, site SW6 was the best for 0+ brown trout populations, suggesting that the habitat available was much more suitable for 0+ brown trout, especially in comparison to SW1. The most frequently occurring substrate at SW6 was boulder and coble with flows of turbulent broken shallow sections and shallow glides. The other sites had similar substrate types but flows were frequently or dominated by shallow glides and slack areas, with little (approx. 25%) or no (SW2 and SW5) shallow turbulent broken flows. Contributing factors that could have resulted in low numbers of 0+ brown trout are similar to those suggested at Ingbirchworth, including low flows, siltation issues, river engineering and poor water quality (section 5.4.1). Growth rates of brown trout in the River Washburn were average to slow in the first year, but declined to slow in older brown trout age classes. This decline could be attributed to low levels of available food sources or lack of suitable habitat increasing competition for habitat and food. Again like Ingbirchworth Dike, there were relatively low numbers of brown trout suitable for ageing so these results should be treated with caution.

Good populations of bullhead were found in the River Washburn, with evidence of recruitment in 2013 suggesting there are important areas of suitable habitat for this conservation species. Lower number of 0+ bullheads were caught in 2012 and 2014 in comparison to 2013. The reasons for this could have been due to higher flows in 2012 as it was a wet year and the reservoir was over topping, which could have displaced small 0+ bullhead downstream. Bullhead swimming capabilities are one of the lowest of European freshwater species, which makes them more likely to be displaced (Cowx & Gould, 1985). Bullhead rely on their pectoral fins to anchor themselves to the bottom of riverbed (Tudorache *et al*., 2007). In 2014 the fish surveys took place four weeks earlier (26 August) than in 2013 (30 September), thus in 2014 fewer bullhead might have grown to a suitable size to be catchable. The reason for this earlier sampling date was the habitat improvement works were due to commence from September 2014 onwards.

5.4.3 Habitat improvement

It is widely acknowledged that habitat improvement techniques in rivers to provide fish habitat, increase fish populations (Fausch & Northcote, 1992; Cowx & Welcomme, 1998; Roni, 2005; Whiteway *et al*., 2010), therefore it can would be expected that the river habitat improvement works at Ingbirchworth Dike and the River Washburn will increase fish population density. It is important that the habitat improvement works carried out provide a wide range of fish habitats for key species in the river, e.g. brown trout in Ingbirchworth Dike and brown trout and bullheads in the River Washburn. A number of studies found that increasing available overhead cover, spawning habitat, the number and size of pools increased salmonid populations (Solazzi *et al*., 2000; Roni *et al*., 2008; Whiteway *et al*., 2010). Whiteway *et al*. (2010) found that 73% of 211 stream improvement studies showed an increase in salmonid population densities following habitat improvement (weirs, deflectors, cover structures, boulder placement, and large woody debris). Increasing the number of pooled habitats at Ingbirchworth Dike and the River Washburn will provide suitable areas for larger trout to inhabit, potentially supporting larger densities of brown trout populations. In the River Washburn there are plans to cut down trees to reduce shading, these trees will be utilised and secured into the river to narrow the channel, deflect the flow and provide instream cover for the fish. Meandering Ingbirchworth Dike and installing brushwood bundles will create river channel diversity and increase available instream cover for fish. The extent of biological benefit is related to the amount of river rehabilitated (Fausch & Northcote, 1992), in theory the more natural features created within the River Washburn and Ingbirchworth Dike the increase in biological benefit. Additionally, fencing off the river to reduce cattle poaching will help to protect riparian vegetation, reduce erosion, sediments and nutrient input from the land (Naiman & Decamps 1997; Pusey & Arthington 2003). Additionally, fencing can preventing cattle from straying, providing benefits to farmers and improving social acceptability (and cost effectiveness) around fencing (Angelopoulos *et al*., 2015). In the River Washburn there are plans to introduce gravels to provide suitable spawning habitat for brown trout, throughout the reach sampled the most common substrate type were boulders and cobbles, suggesting the introduction of gravels to be a valuable improvement method to perform. Barlaup *et al*. (2008) found that spawning success was >80% after introduction of spawning gravel, it was also noted that gravel additions are even successful in areas of sub-optimal flow and water depth.

Positive results from habitat improvement works are not always seen. Whiteway *et al*. (2010) found that 27% of projects showed a decrease in salmonid density and stated that this was a result of poor study design, for example poorly chosen reference reach, short monitoring programme. This is unlikely to be the root cause but appropriate planning is needed, in some studies there were unexpected responses after being subject to hydromorphological and biological processes and poor water quality, as a result efforts to improve habitat was of little worth (e.g. decrease in depth and decrease in spawning gravels and contaminated land reducing water quality during periods of high rainfall) (Cowx & van Zyll de Jong, 2004; Whiteway *et al*., 2010; Hering *et al*., 2015). Although Ingbirchworth Dike and the River Washburn suffer from low flows, during periods of high rainfall the reservoirs fill up to the point where they overspill down a designated route to the river below, meaning the downstream rivers are exposed to higher flows. There is a risk that the rehabilitated techniques carried out at Ingbirchworth Dike and the River Washburn could be potentially wash out or re shaped due to higher flows. Barlaup *et al*. (2008) found that three of seven locations were unsuitable for the introduction of gravels after gravel was totally or partially displaced after a flood event. The gravels planned to be introduced in the River Washburn could be displaced further downstream, because the River Washburn is regulated by Swinsty Reservoir, sediment replacement is much less than in a more natural system. Vörösmarty *et al*. (2003) estimated that more than 53% of the global sediment flux in regulated systems is potentially trapped by reservoirs. Some studies have found that species diversity and density did not differ in response to a number of habitat improvement techniques (river widening, creation of instream structures, flow enhancement, remeandering and side-channel reconnection) (Schmutz *et al*., 2014). Schmutz *et al*. (2014) found that multiple habitat improvement techniques resulted in a shift in species composition, the proportion of rheophilic species increased and eurytopic species decreased. The planned habitat improvement works in the River Washburn are designed to narrow the channel and increase the flow, this may change the species composition, from species that prefer slow flowing water (e.g. roach) to species such as brown trout that prefer faster flowing water.

To detect this change it is important that fish population monitoring continues after habitat improvement, it is important that both impact and control sites are monitored, due to the large temporal variability of brown trout populations between 2012 and 2014. The resource calculations indicated that three years pre monitoring of control and impact sites was sufficient to detect a 50% change in 0+ fish density two years after habitat improvement. Other studies have found that longer periods of monitoring are recommended; for example, Jones & Schmitz (2009) reviewed 240 studies and found that the mean recovery was between 10-20 years, this length of monitoring is rarely feasible due to the costs involved (Hammond *et al*., 2011). Schmutz *et al*. (2014) discussed this and stated that fish populations were most responsive in the first 3 and 12 years of post-habitat improvement and less so in the mid-term (3-12 years), it was further said that in their habitat improvement sites even in the longer recovery periods (10 years) the mean increase of species was only one. This provides supporting evidence that 3 years post monitoring at Ingbirchworth Dike and the River Washburn is sufficient.

5.5 SUMMARY

Population densities brown trout at Ingbirchworth Dike and the River Washburn were very low and growth rates were slow during the three sampling years. These low densities and slow growth rates were attributed to lack of suitable habitat, particularly spawning and juvenile riffle habitats, lack of deeper pooled areas for larger brown trout and lack of available cover. Other factors influencing densities and growth included low flows, siltation and water quality. The habitat improvement works planned should help improve flow, habitat and sediment issues at rehabilitated sites. It was concluded that three years of pre monitoring of control and impact sites was enough to detect a 50% change in 0+ fish density two years after the habitat improvement works are carried out, but more years of post-sampling would reduce the variance and outputs would be more statistically robust.

6 BROWN TROUT RESPONSE TO HABITAT MODIFICATIONS AT MALIN BRIDGE

6.1 INTRODUCTION

Extensive land use changes due to human activity have meant that the majority of river catchments have been modified. The most dramatic activities that have occurred include deforestation, intensification of agriculture and industrial activity and the modification of river channels to improve navigation and reduce flood risk (Cowx & Van Zyll de Jong, 2004). Habitat improvement measures are frequently used to mitigate anthropogenic disturbances and following the introduction of the WFD, habitat improvement measures are a way in which GES/GEP can be achieved by 2027 (Gilvear *et al*., 2013).

The floods in 2007 severely affected people, buildings and businesses in the Don catchment, Meadowhall shopping centre was closed and businesses were affected for many months (Mayes, 2008). Following these floods, efforts were made to reduce flood risk whilst trying to rehabilitate the River Don and its catchment. Malin Bridge in Sheffield was one site in which the EA identified as a priority area to carry out flood defences works as this site was heavily impacted upon in the 2007 floods. Proposed flood defences works to reduce the risk of flooding in the future included gravel bar and tree removal. Rehabilitation works were incorporated into the planning to reinstate habitat features and return brown trout populations to their base line before flood defence works were carried out.

In this chapter brown trout populations were monitored at Malin Bridge in Sheffield on the Rivers Rivelin and Loxlely following the first phase of flood defences works and second phase rehabilitation works. The aim of the rehabilitation works were to mitigate the impacts of the flood defence works and ensure the river did not deteriorate as a result of the flood defence works. Brown trout are the dominant species at Malin Bridge and monitoring populations pre flood defence and post flood defence and post rehabilitation will act as indicator to ensure flood defence and rehabilitation works have not cause any deterioration on the Rivers Loxley and Rivelin.

The aim of this study was to compare brown trout populations and habitat parameters pre and post flood defence and rehabilitation works in the rivers Loxely and Rivelin at Malin Bridge in Sheffield to assess the impact of the works on the fish population status. Specific objectives were to: -

- Identify whether habitat rehabilitation can mititgate the impacts of food defence works and provide comparable brown trout population densities and population structures to those prior to flood defence works.
- Identify whether habitat rehabilitation can mititgate the impacts of food defence works and provide comparable habitat for brown trout population prior to flood defence works using HABSCORE.

Finding will support the conclusions to whether habitat improvement works are required to reduce the impacts of flood defence works and return brown trout populations to their baseline before flood risk management works were carried out.

6.2 METHODOLOGY

6.2.1 Study area

Malin Bridge is at the confluence of the rivers Rivelin and Loxley, located in the River Don catchment (Figure 6.1). In 2010, the study reach underwent habitat modifications (gravel bar and tree removal) to reduce flood risk. These modifications drastically altered the river channel and habitat features (Figure 6.2 a - j). In the modification process, trees were removed (Figure 6.2 d – f), which previously provided shade and cover for fish (Figure 6.2 a – c), the river channel was widened, and gravel bars (areas of shallow water created by the deposition of sediment) were removed, resulting in the river becoming shallower, changing from a heterogeneous habitat to a homogenous habitat. During the modification process, a weir (Burgon/Ball) was uncovered on the River Loxley (Figure 6.2 f), previously buried by sediment. In 2010, the EA rehabilitated the rivers at Malin Bridge post flood defence works, from a very uniform depth and flow to a more varied depth and flow to try and return it to a more natural topography. This was completed by channel re-profiling, installation of instream boulders and a rock-riffle to increase flow and habitat diversity for aquatic biota (Figure 6.2 g). Large boulders were used to frame the rock-riffle, these were large enough so they did not move under flood flows and smaller material was used to fill in between the large boulders (Figure 6.2 g). Vegetation was expected to recolonise over time and this would be managed in the future; evidence of recolonisation can be seen in Figure 6.2 h – j.

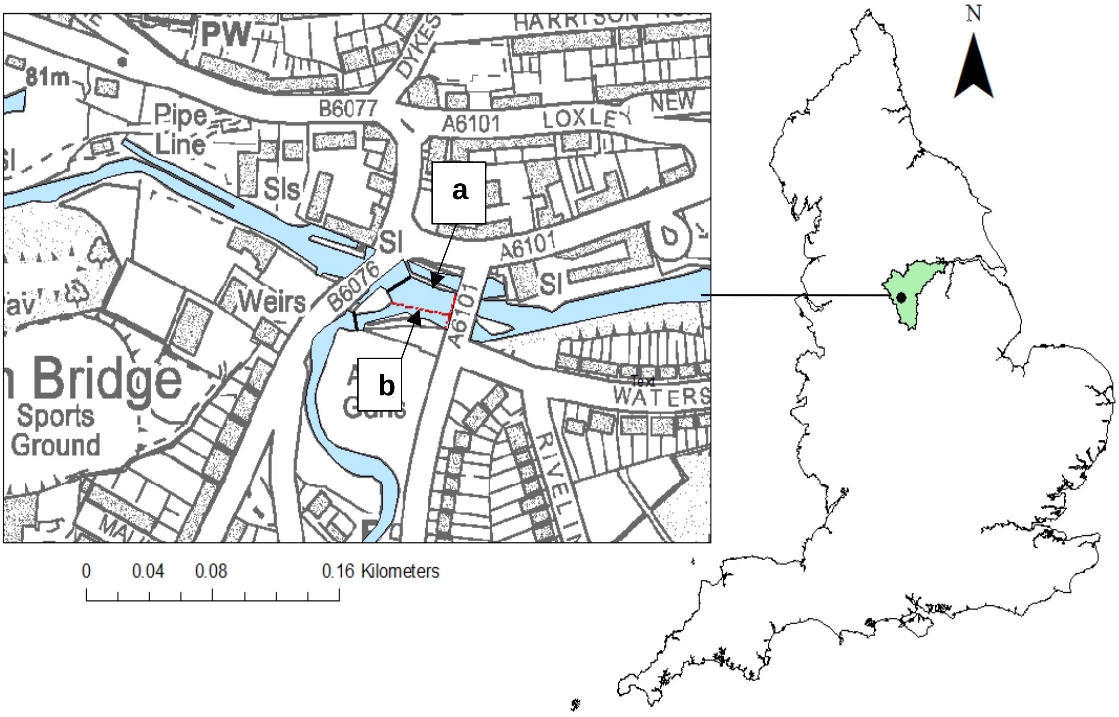

Figure 6.1. A map of England showing the outline of the Don catchment (green) and location of the study (●), and a more detailed map showing the location of the study Malin Bridge, including the survey sites Rivers (a.) Loxley and (b.) Rivelin isolated by stop nets (---) and weirs (━).

6.2.2 Fish survey methodology

Fisheries surveys were carried out by HIFI on 3rd July 2009 (prior to flood defence works) (Figure 6.2 a - c), 21st July 2010 (post flood defence works) (Figure 6.2 d - f) and 21st July 2011 (Figure 6.2 g), 19th July 2012 (Figure 6.2 h), 20th August 2013 (Figure 6.2 i) and 14th August 2014 (Figure 6.2 j) (post rehabilitation works) using quantitative electric fishing (estimates of absolute abundance based on a three-catch removal method (Carle & Strub, 1978) (personally involved in 2010 – 2013 survey years). The reach surveyed was at the confluence of the Rivers Loxley and Rivelin, and to ensure adequate coverage of the site, the two rivers were sampled separately. In fish surveys between 2009 and 2014 in the River Loxley, the survey reach was isolated with a downstream stop net (at the A6101 bridge) and the upstream limit was Burgon and Ball weir, which provided a barrier to fish movements. In fish surveys in the River Rivelin between 2009 and 2012, the survey reach was isolated with a downstream stop net (at the A6101 bridge) and the upstream limit was Rivelin weir, which provided a barrier to fish movements; the rivers Loxley and Rivelin were isolated from each other with a stop net running from the island downstream to the A6101 road bridge (Figure 6.1). However, in 2013, following high winter flows, a substantial build-up of coarse substrate changed the River Rivelin channel meaning the river joined the Loxley at the base of the island and the confluence did not extend down to the A6101 road bridge. In this instance the Rivelin channel was isolated by a stop net at the base of the island and Rivelin Weir. The isolation of sections during the survey ensured there was no escape of fish from, or migration into, the sample area. The quantitative electric fishing strategy on the River Loxley involved three operatives (one anode operator and two people netting fish) fishing in an upstream direction, with a fourth operator on the bank supervising safe operation of the electric fishing equipment. A 2kVA generator powering an Electracatch control box producing a 220 V DC output was employed. During the fishing exercise as many fish as possible were caught in dip nets by operatives positioned either side, and downstream, of the anode; the process was repeated for each run of the three-catch removal method with catches kept separate for data collection. The same methodology was used to survey the River Rivelin once the catch had been processed from the River Loxley. Following each survey, individual fish were identified to species level, fork length (mm) measured and scale samples removed for ageing purposes (three brown trout from every 10-mm size class (Britton 2003)); the fish were then returned to the river. After each electric fishing survey, habitat and environmental data were collected at each site and recorded on standard forms used by the EA. Extensive photographs were taken at the survey site to allow

a – River Rivelin, July 2009, upstream view at downstream limit of survey reach (pre-flood defence works).

b – River Rivelin (left) and River Loxley (right), July 2009, upstream view from A6101 road bridge (pre-flood defence works).

c – River Loxley, July 2009, upstream view from A6101 road bridge (pre-flood defence works).

d – River Rivelin, July 2010, upstream view from downstream limit of survey site (post-flood defence works).

e – River Rivelin (left) and River Loxley (right), July 2010, upstream view from A6101 road bridge (post-flood defence works).

f – River Loxley, July 2010, upstream view from A6101 road bridge (arrow indicates newly uncovered Burgon/Ball weir (post-flood defence works).

Figure 6.2. Pre and post flood defence and rehabilitation works on the rivers Rivelin and Loxley 2009-2014 (continued onto next page)

g – River Rivelin and River Loxley confluence in July 2011 post completion of river rehabilitation works

h – River Rivelin and River Loxley confluence in July 2012

i – River Rivelin and River Loxley confluence in August 2013

j – River Rivelin and River Loxley confluence in August 2014

Figure 6.2 (Continued). Pre and post flood defence and rehabilitation works on the rivers Rivelin and Loxley 2009-2014

6.2.3 HABSCORE data collection

Habitat parameters in the Rivers Loxley and Rivelin survey reaches were collected and used to determine habitat quality and usage using the HABSCORE programme (Chapter 3.2.3).

6.2.4 Data analysis

6.2.4.1 Density estimates

Density estimates of 0+ and >0+ brown trout were derived for the survey reaches in the rivers Loxley and Rivelin, and the rivers combined, annually between 2009 and 2014. Data were compared to determine population density pre and post flood defence works and post rehabilitation works. The calculation of density estimates were described in Chapter 3.2.4.2.

6.2.4.2 Classification of population estimates

The classification of population estimates were described in Chapter 3.2.4.3.

6.2.4.3 HABSCORE analysis and ouputs

Habitat data were collected and analysed for the Rivers Loxley and Rivelin with outputs as detailed in Chapter 3.2.4.5.

HABSCORE outputs were derived using a combination of habitat data and fisheries data collected during the survey period to account for the temporal changes to the habitat. In 2009, HABSCORE outputs were derived using habitat and fisheries data collected in 2009 (pre-flood defence works) while in 2010, HABSCORE outputs were derived using habitat data collected in 2010 and only fisheries data collected in 2010 due to the changes in the habitat. The same approach was used to derive HABSCORE outputs from 2011 surveys due to the changes in the habitat because of the rehabilitation works, i.e. habitat data collected in 2011 and only fisheries data collected in 2011. In 2012, to allow for temporal changes in fish populations, HABSCORE outputs in 2012 were derived using habitat data collected in 2012 and fisheries data from 2011 and 2012 as no instream habitat modifications were made. In 2013, the HABSCORE outputs were planned to be derived using habitat data from 2013 coupled with annual fisheries data from 2011-2013, but because of the dramatic change in habitat structure due to high flows, particularly in the River Rivelin, it was deemed appropriate to only use fisheries data from 2013. In 2014, HABSCORE outputs were derived using habitat and fisheries data collected in 2014 only.

Analysis of Variance (ANOVA) was used to compare variation in depths and widths pre and post flood defence and rehabilitation works, *post hoc* tests (Tukey HDS) were used to find where the significant differences were. When variances were not equal (Levene statistic $P < 0.05$) non-parametric tests were used (Kruskal–Wallis). Results allowed brown trout densities to be compared with response in relation to habitat changes.

6.3 RESULTS

6.3.1 Brown trout population density trends, Malin Bridge 2009 - 2014

Prior to flood defence works (2009) brown trout densities were similar in the rivers Loxley and Rivelin and spawning of brown trout occurred in both rivers in 2009, as indicated by the presence of 0+ fish (Table 6.1, Figure 6.2). In both rivers, >0+ brown trout dominated catches in 2009 and the presence of average to good densities of >0+ brown trout indicated good survival of brown trout from recruitment in previous years. Derivation of abundance categories revealed 0+ brown trout populations in both the rivers Loxley and Rivelin in 2009 were fair/poor (class D), while >0+ brown trout populations were good (class B) in the River Loxley and average (class C) in the River Rivelin (Table 6.1). Overall in 2009 the reach at Malin Bridge contained fair/poor (class D) 0+ brown trout populations and good (class B) >0+ brown trout populations (Table 6.1, Figure 6.2).

Post flood defence works (2010), overall densities of brown trout varied in the rivers Loxley and Rivelin, and contrasted markedly with 2009 (Table 6.1). In both rivers, 0+ brown trout dominated catches in 2010 with the River Rivelin having the greatest densities of 0+ brown trout; 0+ brown trout densities in both rivers were greater than recorded in 2009 (Table 6.1, Figure 6.2). >0+ brown trout densities were higher in the River Rivelin in 2010 than 2009, but >0+ brown trout densities were lower in the River Loxley in 2010 than 2009 (Table 6.1, Figure 6.2). Abundance categories of 0+ brown trout populations in the rivers Loxley and Rivelin in 2010 were good (class B) and excellent (class A), respectively. In both cases an improvement in 0+ abundance was found compared with 2009 (Table 6.1). >0+ brown trout populations in 2010 were fair (class C) in the River Loxley and good (class B) in the River Rivelin (Table 6.1). Overall in 2010 the reach at Malin Bridge contained good (class B) 0+ brown trout populations and average (class C) >0+ brown trout populations (Table 6.1).

Table 6.1. Total population estimate (N) and population density (D) (± 95% C.L. at quantitative sites) of trout derived from fisheries surveys in Rivers Loxley and Rivelin at Malin Bridge in July 2009, 2010, 2011, 2012, 2013 and 2014 (density of fish given as numbers per 100m^2). Flood defence and rehabilitation works indicated by red and green lines respectively. Colours represent EA-FCS grading scheme.

A (excellent)	B (good)	C (average)	D (fair/poor)	E (poor)	F (fishless)

Site Identifier	Total Population (N)		Population density (D)	
	0+	>0+	0+	>0+
Loxley 2009	10±3	39±6	3.22±1.02	12.57±2.12
Loxley 2010	121±18	25±1	26.29±3.98	5.43±0.23
Loxley 2011	21±2	49±1	6.03±0.61	14.06±0.38
Loxley 2012	3±1	53±1	0.80±0.12	14.17±0.34
Loxley 2013	20±7	52±5	4.32±1.60	11.24±1.02
Loxley 2014	13±2	36±1	2.71±3.63	7.50±1.12
Rivelin 2009	14±5	52±4	3.21±1.15	11.94±1.07
Rivelin 2010	168±10	67±3	46.03±2.73	18.36±0.72
Rivelin 2011	32±3	104±4	9.20±0.91	29.89±1.15
Rivelin 2012	0±0	74±1	0.00±0.00	14.95±0.30
Rivelin 2013	34±3	30±3	12.38±1.20	10.92±1.21
Rivelin 2014	2±0	25±2	0.82±0.00	10.30±3.96
Loxley/Rivelin combined 2009	27±9	92±8	3.62±1.32	12.34±1.12
Loxley/Rivelin combined 2010	290±15	92±3	35.14±1.91	11.15±0.36
Loxley/Rivelin combined 2011	54±5	153±4	7.75±0.66	21.97±0.59
Loxley/Rivelin combined 2012	3±1	127±2	0.41±0.06	17.45±0.25
Loxley/Rivelin combined 2013	54±8	83±6	7.32±1.05	11.25±0.85
Loxley/Rivelin combined 2014	14±1	61±2	1.94±1.88	8.44±3.31

After rehabilitation works were carried out at Malin Bridge prior to the 2011 surveys, the overall densities of brown trout in the rivers Loxley and Rivelin were lower in 2011 than 2010 but higher than in 2009 (Table 6.1). 0+ brown trout densities were higher in the River Rivelin compared with the River Loxley and were lower than in 2010 but higher than in 2009 in both rivers (Table 6.1, Figure 6.3). In both rivers, >0+ brown trout dominated catches in 2011 with the River Rivelin having the greatest densities of >0+ brown trout; >0+ brown trout densities were higher in 2011 than in 2009 or 2010 (Table 6.1, Figure 6.3). Abundance categories of 0+ brown trout populations in the Rivers Loxley and Rivelin in 2011 were fair/poor (class D) and average (class C) (Table 6.1). >0+ brown trout populations in 2011 were good (class B) in the River Loxley and excellent (class A) in the River Rivelin (Table 6.1). Overall in 2011 the reach at Malin Bridge contained fair/poor (class D) 0+ brown trout populations and excellent (class A) >0+ brown trout populations (Table 6.1).

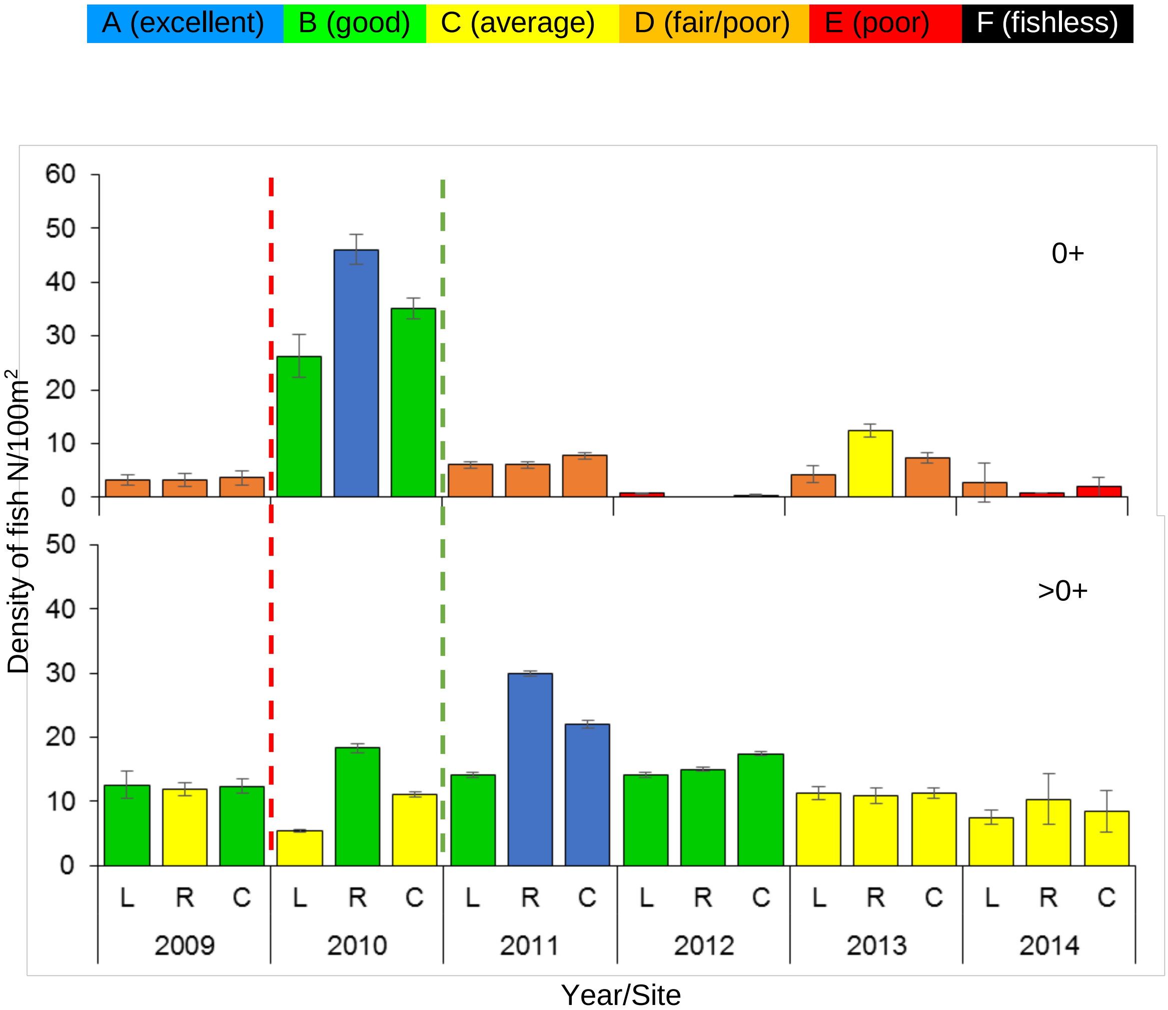

Figure 6.3. Density (± 95% C.L at quantitative sites) estimates of 0+ and >0+ brown trout in Rivers Loxley (L), Rivelin (R) and both sites combined (C) between 2009 and 2014. Flood defence and rehabilitation works indicated by red and green dashed lines respectively. Colours represent EA-FCS grading scheme.

Overall densities of brown trout in the rivers Loxley and Rivelin were lower in 2012 than 2011 and 2010 and were similar to those found in 2009 (Table 6.1). 0+ brown trout densities in 2012 were lower than other years in the River Loxley, and 0+ brown trout were absent from the River Rivelin in 2012 (Table 6.1, Figure 6.3). In both rivers in 2012 >0+ brown trout dominated. >0+ brown trout densities in the River Rivelin were lower in 2012 than 2011 and 2010 but were higher than in 2009 (Table 6.1, Figure 6.3). In the River Loxley >0+ brown trout densities were higher in 2012 than in previous years, albeit densities in 2012 were marginally higher than in 2011 (Table 6.1, Figure 6.3). Abundance categories of 0+ brown trout populations in the rivers Loxley and Rivelin in 2012 were absent (class F) and poor (class E) respectively (Table 6.1, Figure 6.3). >0+ brown trout populations in 2012 were good (class B) in the River Loxley and River Rivelin

(Table 6.1). Overall in 2012 the reach at Malin Bridge contained poor (class E) 0+ brown trout populations and good (class B) >0+ brown trout populations (Table 6.1).

The overall densities of brown trout in the rivers Loxley and Rivelin in 2013 were similar to 2012, and were lower than in 2011 and 2010, but higher than in 2009 (Table 6.1).In the River Rivelin 0+ brown trout marginally dominated catches; contrasting markedly with findings in 2012 when 0+ brown trout were absent (Table 6.1, Figure 6.3). In addition 0+ brown trout densities in 2013 were higher than in 2012, 2011 and 2009 but lower than in 2010. >0+ brown trout densities in 2013 on the River Rivelin were the lowest recorded albeit only marginally lower than densities in 2009.In the River Loxley 0+ brown trout densities were higher in 2013 than in 2012 and 2009. River Loxley in 2013 >0+ brown trout dominated catches, densities in 2013 were lower than in 2012 and 2011, higher than in 2010, and similar to those in 2009 (Table 6.1, Figure 6.3). Abundance categories of 0+ brown trout populations in the Rivers Loxley and Rivelin in 2013 were fair/poor (class D) and average (class C)) respectively (Table 6.1). >0+ brown trout populations in 2013 were average (class C) in the River Loxley and River Rivelin (Table 6.1). Overall in 2013 the reach at Malin Bridge contained fair/poor (class D) 0+ brown trout populations and average (class C) >0+ brown trout populations (Table 6.1).

In 2014 the overall brown trout densities in the Rivers Loxley and Rivelin were the lowest out of all of the years previously sampled (Table 6.1). In both rivers in 2014 >0+ brown trout dominated catches and were similar to those in 2013. In the Rivers Loxley and Rivelin 0+ brown trout densities in 2014 were lower than in 2009, 2010, 2011 and 2013 but not 2012. In the River Loxley >0+ brown trout densities were lower in 2014 than in all other years apart from 2010; while 2014 >0+ brown trout densities in the River Rivelin were lower than all other years (Figure 6.3, Table 6.1). Abundance categories of 0+ brown trout populations in the River Loxley and Rivelin in 2014 were fair/poor (class D) and poor (class E) (Table 6.1). >0+ brown trout populations in 2014 were average (class C) in the Rivers Loxley and Rivelin (Table 6.1). Overall in 2014 the reach at Malin Bridge contained poor (class E) 0+ brown trout populations and average (class C) >0+ brown trout populations (Table 6.1).

Overall, confidence limits indicated that densities would not have been notably different and classifications would have been similar given the variation.

6.3.2 Habitat overview

Prior to the flood defence works in 2009, the rivers Rivelin and Loxley were heavily shaded (80-90 %) by riparian vegetation. Post flood defence works the trees and shrubs that provided any shading were removed, but over time vegetation has recolonised and by 2013 vegetation had begun to shade the rivers (Table 6.2 and Figure 6.1 a - j). The width of the River Loxley following the flood defence and rehabilitation works was constantly wider than prior to the works; this was to increase the carrying capacity of the channel. There was a significant difference in widths between years prior to flood defence works (2009), post flood defence works (2010) and post rehabilitation works on the River Loxley (ANOVA: $F_{2,13}$ =13.182, P= 0.001). Post-hoc comparisons using Tukey HSD test indicated that mean score for 2009 (M = 6.6, SD = 1.13) was significantly different from 2010 (M = 11.8, SD = 1.43). Post rehabilitation works river widths (2011) (M = 8.5, SD = 2.40) were no different to 2009 and 2010. The River Rivelin on average didn't change in width before and after the flood defence and rehabilitation works, although it was at its widest in 2013, (Kruskal-Wallis: n = 22, X^2 = 0.956, $d.f.$ = 2, P = 0.620).

The mean depth of the River Loxley was deepest in 2009 (31cm±8.58) and 2011 (31cm±9.99) prior to the flood defence works and post rehabilitation works and shallowest in 2010 (18cm±7.26) (Table 6.2), demonstrating that the rehabilitation works re-introduced some deeper habitats for larger trout although depth did decrease in 2012 and remained the same through to 2014. There was no significant difference in depths between 2009 and 2011 in the River Loxley (ANOVA: $F_{5,\,87}$ =1.56, P= 0.18). The depth in the River Rivelin did not vary so much, but was slightly deeper in 2009 prior to any works and shallowest in 2013 (17cm±4.80) (Table 6.2), although there was no significant difference in depth between 2009-2011 (ANOVA: $F_{2,63}$ =1.69, P= 0.19). The variation of the width and depth of the channel can be a result of sediment deposition; in the River Loxley several gravel bars were formed in 2012 (Figure 6.1 h) and in the River Rivelin the gravel bar in 2010 after the flood defence works shifted from the right hand side of the channel to the left in 2011 (Figure 6.1 d & g) post rehabilitation works and an additional gravel bar formed in 2013 (Figure 6.1 i). Vegetation had established on all of the gravel bars formed after the instream channel works in 2013 and 2014 (Figure 6.1 i & j). Sources of cover for brown trout >10cm in the Rivelin and Loxely prior to the instream channel works in 2009 were 5% and 8%, respectively, dominated by boulders and cobbles (Figure 6.4). In 2010, after the flood defence works, sources of cover were at their lowest in the River Loxley but were lowest in the River Rivelin in 2011 post rehabilitation works. In subsequent years sources of cover gradually increased in both rivers with values of 32% in the Rivelin and 23% in the Loxley, dominated by boulder/cobbles and deep water (Figure 6.4).

Table 6.2. Loxley (L) and Rivelin (R) total % riparian shading, average width (m) and average depth (cm) for each section from 2009-2014.

	2009		2010		2011		2012		2013		2014	
	L	R	L	R	L	R	L	R	L	R	L	R
Shading (%)	90	80	0	0	0	0	0	0	5	5	0	15
Width (m)	6.6	6.5	11.8	6.3	8.6	6	8.2	6.88	11.8	8.2	11.7	5.9
Depth (cm)	31.8	27.9	18.5	23	31.1	22.3	24.6	25.6	23.1	17.9	24.7	25.9

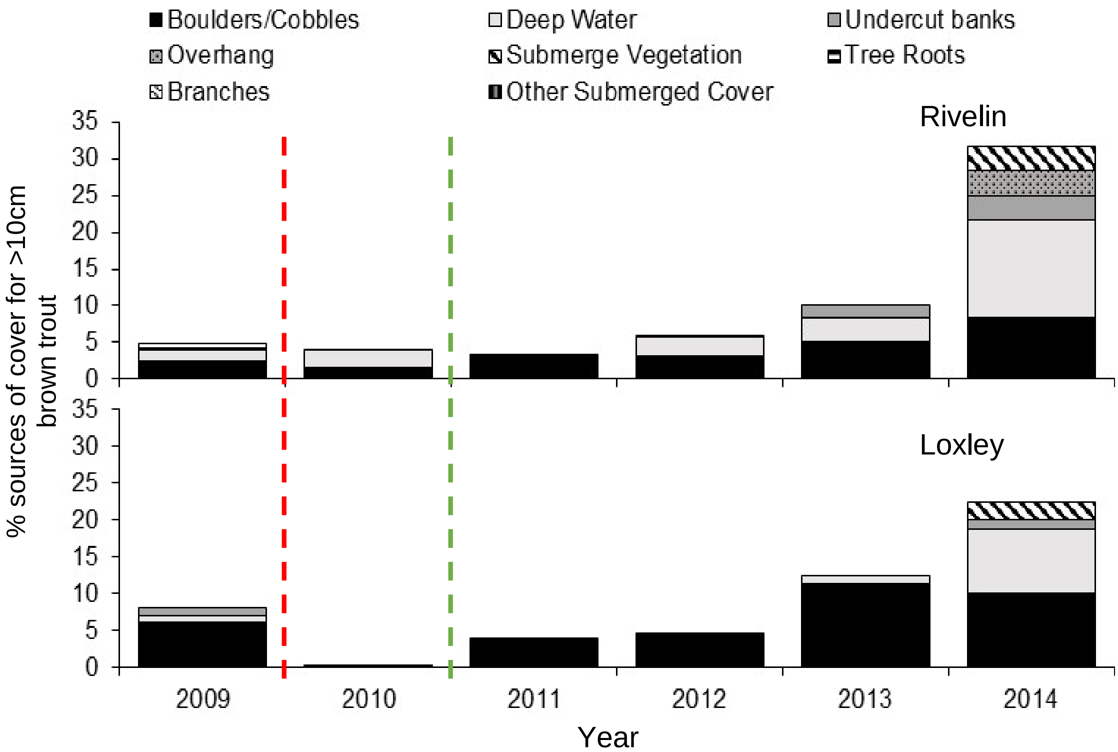

Figure 6.4. Average % sources of cover for >10 cm brown trout in the rivers Rivelin and Loxley 2009-2014. Information obtained from HABSCORE raw data. Red dashed line indicates when the flood defence works were carried out and the green dashed line when the rehabilitation works were carried out.

6.3.3 HABSCORE outputs

HABSCORE outputs for the sites on the Rivers Loxley and Rivelin revealed variations in the observed densities, predicted densities and habitat utilisation by brown trout (Figures 6.5 & 6.6).

Prior to flood defence works (2009) on the River Loxley, brown trout observed densities were similar to expected for 0+ and >0+ (>20 cm), whereas >0+ (<20 cm) brown trout observed densities were significantly higher than expected under pristine conditions (Figure 6.5). In the River Rivelin in 2009, 0+ brown trout observed densities were slightly lower than expected, slightly higher for >0+(<20 cm) and similar to expected for >0+(>20 cm), none of which were significantly different (Figure 6.6).

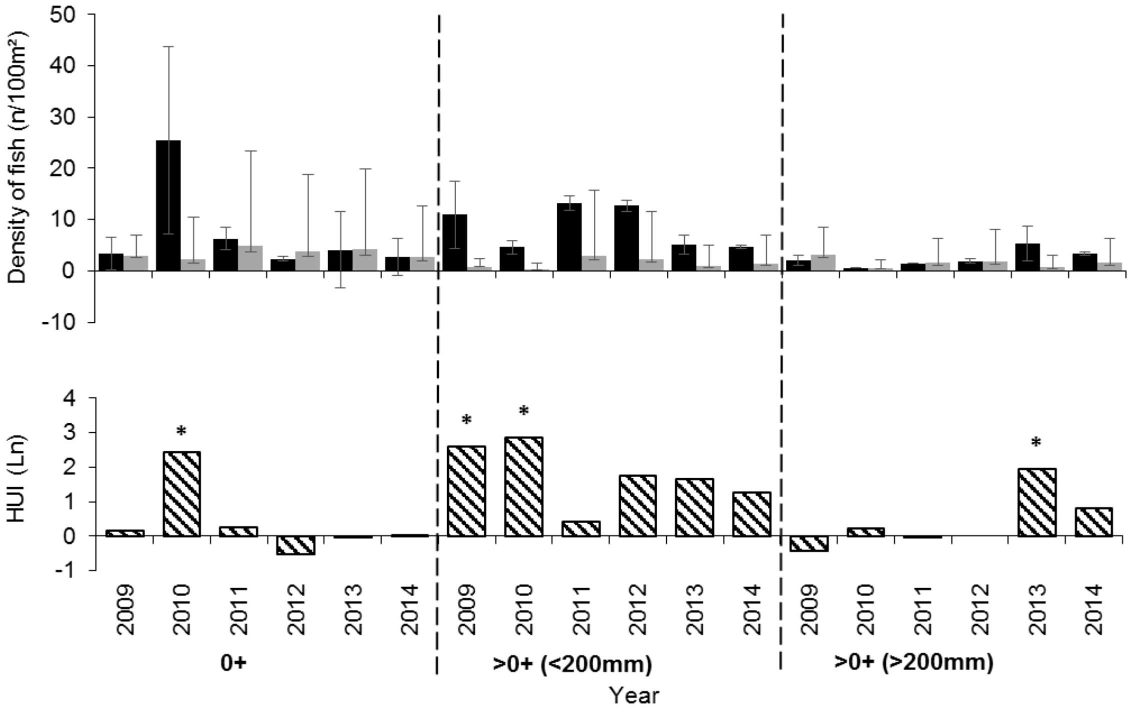

Figure 6.5. HABSCORE outputs of observed brown trout density (± 95% C.L) (black), HQS (± 90% C.L) (grey) and HUI (stripe) for the River Loxley at Malinbridge. *represents sites where the observed population was significantly higher than would be expected under pristine conditions.

Post flood defence works (2010), 0+ and >0+ (<20 cm) brown trout in the River Loxley and River Rivelin were higher than predicted from the Habitat Quality Score (HQS), whereas observed densities of larger >0+ brown trout (>20 cm) were similar to expected (Figures 5.5 & 5.6).

Post rehabilitation works (2011) in the River Loxley, 0+ and >0+ (>20 cm) brown trout were similar to expected, whereas observed densities for >0+ (<20 cm) brown trout were higher than expected but not significantly so (Figure 6.5). 0+ and >0+ (>20 cm) brown trout densities were similar to and slightly higher than predicted from the HQS in the

River Rivelin in 2011, whereas >0+ (<20 cm) remained significantly higher post the rehabilitation works (Figure 6.6).

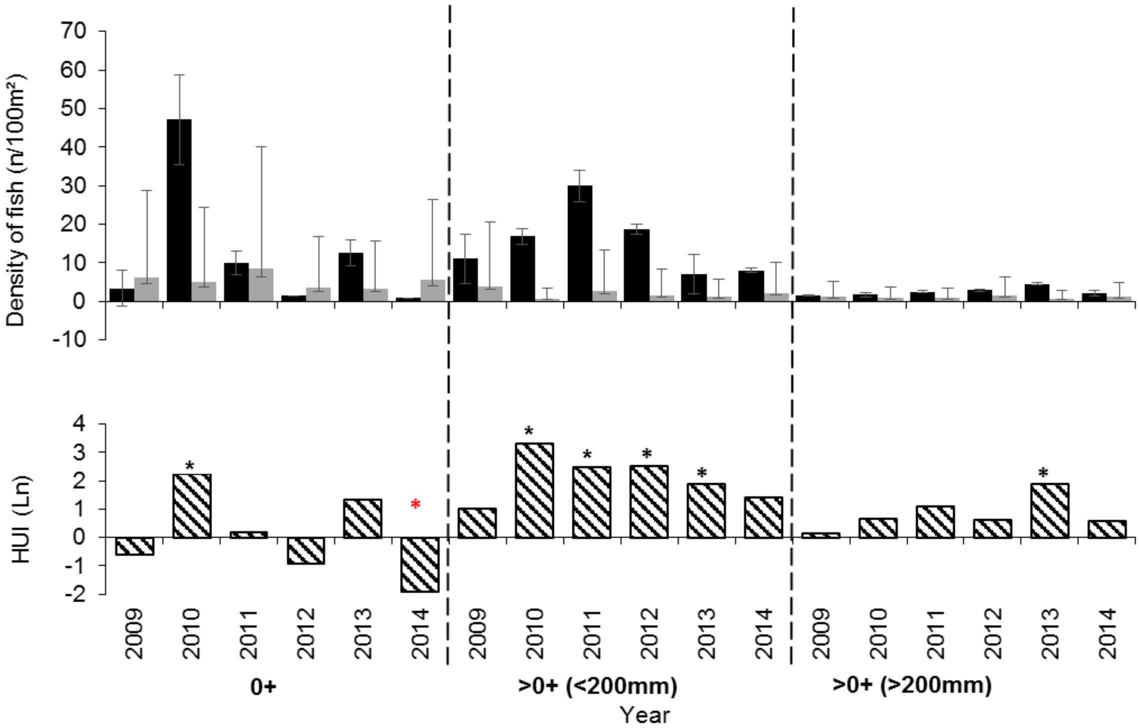

Figure 6.6. HABSCORE outputs of observed brown trout density (± 95% C.L) (black), HQS (± 90% C.L) (grey) and HUI (stripe) for the River Rivelin at Malin Bridge. *represents sites where the observed population was significantly higher and * represents sites where the observed population was significantly lower than would be expected under pristine conditions.

In 2012, two years after post rehabilitation works, 0+ brown trout densities in the River Loxley were slightly lower than predicted, densities of >0+ (<20 cm) brown trout were higher than predicted neither of which were significant (Figure 6.5). >0+ (>20 cm) brown trout densities were exactly as predicted from the HQS in the River Loxley (Figure 6.5). 0+ brown trout densities in the River Rivelin were slightly lower than predicted, densities of >0+ (>20 cm) brown trout were higher than predicted neither of which were significant; >0+ (<20 cm) brown trout observed densities remained significantly higher than predicted from the HQS (Figure 6.6).

Three years after post rehabilitation works (2013), 0+ brown trout observed densities in the River Loxley were similar to predicted by the HQS, densities were higher for >0+ (<20 cm) brown trout and significantly so for >0+ (>20 cm) (Figure 6.5). Observed densities were higher than predicted for 0+ and >0+ brown trout in the River Rivelin in 2013 and significantly so for >0+ (>20 cm) (Figure 6.6).

Four years after rehabilitation works (2014), in the River Loxley 0+ brown trout observed densities in the River Loxley were similar to predicted by the HQS, densities were higher for >0+ (<20 cm) and >0+ (>20 cm) brown trout but not significantly(Figure 6.5). In the River Rivelin 0+ brown trout densities were significantly lower than predicted than the HQS, densities for >0+ brown trout were higher than predicted but not significantly (Figure 6.6).

6.4 DISCUSSION

Improved scientific understanding of the impacts of river engineering has changed the methods of flood risk management and helped to develop and enforce 'softer' approaches (Wharton & Gilvear, 2007). The rehabilitation works post flood defence works at Malinbridge is an example of these newer approaches. The aim of this study was determine the impact of these rehabilitation works on brown trout populations and habitat parameters in the rivers Lovely and Rivelin at Malin Bridge in Sheffield. Fisheries and habitat surveys in 2009 provided a baseline for the brown trout populations and habitat availability, suitability and usage before post flood defence (2010) and rehabilitation works (2011) in the rivers Loxley and Rivelin.

Brown trout populations in 2009 at Malin Bridge were on average classified as fair and the presence of 0+ individuals and good numbers of >0+ brown trout, indicating that the reach provided a variety of habitats for a range of brown trout age and size classes.

Post flood defence works at Malin Bridge in 2010 there was a drastic visual change in the flow and habitat in comparison to 2009, from a diverse pool/riffle sequence with overhanging vegetation and variable substrate in 2009 to a reach of uniform shallow depth, substrate and flow in 2010.

Brown trout populations at Malin Bridge in 2010 (post flood defence works) was nearly three times higher than that in 2009 which could suggest that the flood defence works improved brown trout populations. The overall increase was a result of 0+ brown trout dominating the brown trout populations and being nearly 10 times higher than in 2009. This suggests that the flood defence works potentially provided more suitable habitat for 0+ brown trout or possibly the high >0+ brown trout population in 2009 could have meant there were a high number of adult spawners that resulted in a high number of 0+ brown trout in 2010. It is also possible that 2010 could have been the result of good recruitment of brown trout in the river in general.

The overall width in the River Loxley was significantly higher in 2010 than in 2009. There is little evidence to suggest that brown trout favour wider rivers, although Rahel & Nibbelink (1999) found that brown trout were more successful in larger streams (>4m in wetted width) as it increased their opportunity to escape harsh conditions. This should be treated with caution and it cannot be assumed for this case study as wetted widths were >4m both pre and post flood defence works. Although not significant, depths at both sites between 2009 and 2010 were noticeably different, particularly on the River Loxley. In 2010 the overall depth was shallower, it has been reported that 0+ brown trout select shallower habitats to maximise growth (Ayllón *et al*., 2009; 2010). Although it cannot be proved from this study, available spawning habitat could have increased, resulting in higher densities of 0+ brown trout, or it is possible that that the increase in 0+ brown trout could have drifted downstream from upstream spawning habitats, Bujold *et al*. (2004) found that soon after emergence some juvenile Atlantic salmon drift downstream whereas others establish territories and become resident. The latter point could explain why the density of 0+ brown trout was significantly higher than predicted from the HQS.

Despite this positive rise in the number of 0+ fish, there was a decline in >0+ brown trout abundance in the River Loxley with classifications declining from good (class B) in 2009 to fair (class C) in 2010. These data suggest the flood defence works caused deterioration in quality and/or quantity of habitat for larger trout, probably as a result of the shallowing of the river bed meaning limited deeper pool habitat for larger trout. Limited availability of pools can restrict the presence of large brown trout, which could have resulted in them moving out of the study reach to more suitable habitats (Heggenes, 1996; Jonsson & Jonsson, 2011). Brown trout are very territorial and a decrease in habitat complexity could potentially forcing less dominant >0+ (>20 cm) brown trout to move out of the study reach due to increased competition for the limited pool habitats available (Höjesjö *et al*., 2007). This is supported by the level of cover (e.g. overhanging trees, large boulders and deep water) available for >10cm brown trout being dramatically reduced in 2010. A number of studies found that increasing available overhead cover, spawning habitat, the number and size of pools increased salmonid population abundance (Solazzi *et al*., 2000; Roni *et al*., 2008; Whiteway *et al*., 2010). This would reduce competition for food resources and increase available habitat for brown trout to establish territories. In addition, brown trout could have been more susceptible to predation. Ayllón *et al*. (2009, 2010) reported that adult brown trout selected pools and visually covered habitats to reduce the risk of predation. Pools play in important role during periods of drought serving as refuge habitats when water levels drop (Williams *et al*., 2015), thus the presence of pools will ensure the survival of brown trout helping to maintain population densities through fluctuating water levels.

Removal of gravel bars, instream and bank side vegetation to reduce the risk of flooding, consequently reducing the overall habitat diversity did not appear to have a negative effect on >0+ brown trout (<20 cm) densities in 2010. HABSCORE revealed significantly higher densities of >0+ brown trout (<20 cm) than predicted by the HQS, indicating that the populations were generally greater than would be expected. When 2010 observed densities were compared to 2009 predicted densities, 0+ and >0+ (< 20 cm) densities were greater in 2010 than the predicted habitat densities in 2009. Again suggesting that the flood defence works did not have a negative effect on brown trout populations. It is possible that the >0+ brown trout (<20 cm) were concentrated in the pools below the weirs and not evenly distributed through the entire stretches sampled.

Rehabilitation works in 2011 aimed to improve brown trout habitat by channel re-profiling and installation of instream boulders. In both the rivers Loxley and Rivelin in 2011, >0+ brown trout dominated catches with densities greater than in 2009 and 2010. These data suggest the rehabilitation works may have improved the quality of habitat for larger brown trout, probably as a result of the deepening of the river bed in places meaning an increase in deeper pools, which adult brown trout prefer (Cowx *et al.,* 2004), plus the addition of boulders, which increased the available cover for larger brown trout. Although the depths between 2011 and 2010 were not significantly different. Another possible explanation for higher densities of >0+ brown trout may 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 than 2010, but 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 revealed higher densities of 0+ trout and >0+ trout (<20 cm) than predicted in both the rivers Rivelin and Loxley, indicating that the populations were generally greater than would be expected. There was a positive result in the River Rivelin with >0+ trout (>20 cm) densities higher than predicted but still lower than predicted in the River Loxley in 2011.

Fish surveys undertaken in 2012, 2013 and 2014 allowed further assessment of the status of the fish populations post rehabilitation works in 2010/11. Between 2012 and

2014, >0+ brown trout dominated catches in the River Loxley, while in the River Rivelin >0+ brown trout dominated catches in 2012 and 2014 but were marginally lower than 0+ brown trout catches in 2013.

0+ brown trout densities were lowest in 2012 and absent from the River Rivelin. The year 2012 was a particularly bad year for 0+ brown trout in comparison to all other years. This was further supported by the HABSCORE results which revealed 0+ trout populations were lower than predicted in the rivers Loxley and Rivelin. The very low 0+ brown trout densities in 2012 initially suggest a decline in the populations at Malin Bridge, but surveys carried out by HIFI at 18 other sites on the Rivers Rivelin and Loxley also revealed fair/poor or poor 0+ trout populations throughout the rivers (Harvey *et al.*, 2014). Indeed 0+ brown trout populations at nine sites surveyed on the River Rivelin upstream of Malin Bridge revealed the lowest densities recorded since monitoring began in 2002 (Harvey *et al.*, 2014). This suggest that the 0+ brown trout populations were low as a result of some additional factor. Rainfall from April to July in the UK was reported to be exceptionally wet, breaking previous rainfall records and causing several flood events (Parry *et al*., 2013). It is possible that 0+ brown trout densities were particularly low in 2012 due to rainfall and flood events during the period of emergence and at an age where brown trout would be easily displaced (Daufresne *et al*., 2005; Acreman & Ferguson, 2010; Jonsson & Jonsson, 2011). >0+ trout (<20 cm) and 0+ trout (>20 cm) populations were higher than predicted in both rivers, significantly so in the case of >0+ trout (<20 cm) in the River Rivelin. Older brown trout densities might have been higher because they were more able to withstand the higher flow conditions in 2012, due to their swimming capabilities and behavioural responses being more developed than 0+ brown trout.

In 2013, 0+ and >0+ brown trout densities were at a similar level to those found in 2009 prior to the instream river works. >0+ trout (<20 cm) and >0+ trout (>20 cm) populations were higher than predicted in both rivers, significantly so in the case of >0+ trout (>20 cm) in the River Loxley and >0+ trout (<20 cm) and 0+ trout (>20 cm) in the River Rivelin. It is possible that in this particular year, larger brown trout were concentrated in the deeper pooled areas below the weirs; artificial pools created by weirs are reported to have higher number density and biomass of fishes below weirs (Katano *et al*., 2005). It should be noted that the habitat in the reach in 2013 was considerably different to that found in 2012 due to accumulation of substrate, especially in the River Rivelin. This was most probably deposited following high flow events (Williams *et al*., 2015).

There was a decline in brown trout populations at Malin Bridge in 2014, the second lowest density of brown trout other than 2012 in the study period. As speculated for 2012, weather and rainfall could have effected 0+ brown trout densities in 2014. Throughout December, January and February 2013/14, the UK was affected by frequently severe winter storms, resulting in widespread flooding from January onwards. This could have caused damage to brown trout nests, by flattening the redds and influx of fine sediment, which has been documented to affect the water exchange in redds, reducing hyporheic flows and the permeability of the redd gravel. (Greig *et al*., 2005; Schindler Wildhaber *et al*., 2014).

The available cover for >10cm brown trout in 2014 was at its highest, but, despite the increase in available cover, the overall brown trout densities were one of the lowest. A possible cause could be the low numbers of 0+ fish in 2012 having an effect on recruitment resulting in lower numbers of >0+ brown trout in 2013 and 2014, or habitat became more suitable for larger trout and less so for 0+. According to the HABSCORE results the densities of 0+ brown trout were significantly lower than expected meaning that there is suitable habitat available for 0+ brown trout.

Repeating these surveys over a number of year's highlights the variation in brown trout populations and how the habitat features (e.g. regrowth of vegetation and reappearance of gravel bars) and biological processes over time restore the river towards its previous state (Hughes *et al*., 2005). The removal of gravel bars and vegetation in 2010 naturally reappeared following the high winter flows of 2012, not to the same extent as 2009 but the channel of the River Rivelin narrowed considerably with a large accumulation of substrate restricting the channel width. Natural regrowth of vegetation occurred, replanting was deliberated by the EA but the method was considered more costly, and the species that became established at Malin Bridge included the non-native species, Himalayan balsam, an opportunistic species that outcompetes native plants. The result of this would mean that the EA would have to maintain the site because leaving the site to re-establish itself could increase the flood risk and has already led to non-native plant species becoming established. The constant maintenance works that will be involved would be costly and potentially disturb the fish populations at Malin Bridge (Ayres *et al*., 2014).

6.5 SUMMARY

Brown trout densities varied throughout the study but overall there was no deterioration or improvement in the brown trout population densities following flood defence and rehabilitation works at Malin Bridge. It is concluded that rehabilitation works returned brown trout populations back to their baseline classification following flood defence and rehabilitation works at Malin Bridge. This demonstrates that the potential impacts associated with flood defence works can be reduced when incorporating habitat improvement works in to flood risk management, leading to positive outcomes.

7 GENERAL DISCUSSION AND RECOMENDATIONS

7.1 INTRODUCTION

Variation in natural flow regime and habitat diversity are important factors regulating various life stages of brown trout (Heggenes *et al*., 1999). Anthropogenic activities have altered the natural flow regime and reduced habitat diversity in many rivers in the UK, therefore undertaking management measures to reduce the impact of altered flow regimes (e.g. introduction of more variable flows) and habitat deterioration (e.g. by using habitat improvement measures), whilst ensuring needs for water between human and wildlife are balanced is a fundamental requirement (Chapter 2). The WFD has helped to prevent further deterioration and wherever possible promotes opportunities to enhance the quality of Europe's waters (Hering *et al*., 2015). Modifying reservoir flow releases is one method of flow management that aims to maintain certain aspects of the natural flow regime, for example periods of higher flows through autumn and winter, to facilitate fish migration and spawning (Chapters 3 & 4) and seasonal compensation releases to maintain sufficient flow and sufficient water depth to support fish population recruitment and growth (Chapter 3). Habitat improvement works are another river management technique used to improve river habitats for fish that have previously been degraded from human activities, and brown trout population monitoring provides a good ecological indicator of the success of such habitat improvement works (Chapters 5 & 6).

This chapter discusses the knowledge gained from the previous chapters providing key conclusions and recommendations for further study and future management of flows from reservoirs as well as suitability of habitat improvement measures for attaining good ecological potential in sections of river below reservoirs.

7.1.1 The influence of flow on brown trout movements and key population parameters

Chapter 2 provided an overview of the importance of a natural flow regime in rivers and how the various aspects (magnitude, frequency, duration, timing and predictability) are vital for fish and maintaining river habitats. The construction and operation of reservoirs reduces the natural flow variability in downstream reaches of rivers, but modifying flow releases from reservoirs can be an important management measure to improve ecological status because key elements of a natural flow regime can be reintroduced to rivers downstream of reservoirs.

7.1.1.1 Brown trout movements in response to reservoir freshet releases

Few studies have investigated the impact of freshets on resident adult brown trout movements in response to modified flow releases from reservoirs. Radio telemetry employed in the River Holme identified spatial distribution and temporal movements of brown trout, particularly in response to reservoir freshet releases (Chapter 4). Over the two study periods general movements of brown trout were small, with the range per day tracked ≤ 50 m in 2012 and ≤14 m in 2013/2014. In the River Holme reservoir freshet releases from Brownhill and Digley reservoirs, performed during autumn and winter, did not promote long distance spawning migrations for brown trout (Chapter 4). The largest total distance moved by a single individual was 59 m in October at Mill Pond. It appears that single freshet releases (Brownhill Reservoir) in November 2012 and six freshet releases over October to December (once a month from Brownhill and Digley reservoirs) in 2013, which lasted eight hours, provided brown trout with very little opportunity to make an active migration. This contrasts with days without freshet releases when movements of brown trout in the River Holme were relatively small (<100m), but during natural high flow events following periods of rainfall and during periods when reservoirs were overtopping, brown trout were considered to be making larger distance movements than when the reservoirs were not over topping (personal observation).

7.1.1.2 Brown trout population parameters

Introducing environmental flows is costly to water supply companies (Acreman *et al*., 2009) and it is important that evidence is available to support whether flow modifications provide favourable hydrological conditions for fish populations in rivers affected by flow regulation (e.g. Alonso-González *et al*., 2008). The introduction of seasonal compensation flows and a single freshet release in November since 2004 on the rivers Holme and Ribble was considered insufficient to cause a significant difference in brown trout population density and growth. (Chapter 3). IHA outputs comparing flow before and after flow change in 2004 did not identify any significant changes in the various flow parameters assessed. The flow parameter that had changed the most following the flow change was the reduction in extreme low flows, potentially protecting brown trout populations from these. Extreme low flows can reduce connectivity, restricting movement of brown trout and potentially expose, brown trout redds (Crisp, 1996; Mathews & Richter, 2007). Any variations in brown trout populations were considered to be a result of annual variation, sufficient non-impacted/control sites would have been ideal to compare variations in brown trout populations under natural conditions verses those in the Rivers Holme and Ribble.

7.1.2 The influence of habitat modifications on brown trout population parameters

Habitat availability is vital for all life stages of brown trout, as habitat requirements change with brown trout development. HABSCORE enabled assessment of brown trout populations and whether densities were above or below what was expected based on the available habitat. In Chapter 5, flows were unable to be modified from Ingbirchworth and Swinsty reservoirs, thus habitat improvement was considered to be an alternative method to introduce flow and habitat diversity. The lack of available pool habitats were identified in both the River Washburn and Ingbirchworth Dike. Pools are important for reducing competition, increasing survival during periods of extreme low flow and providing protection from predators, thus the presence of pools will ensure the survival of brown trout helping to maintain population densities during fluctuating water levels. The River Washburn was over-wide, increasing the risk of redds becoming exposed when water levels drop following higher flow events. By narrowing the river channel, water levels will be maintained protecting redds, flows will be increased ensuring intragravel flows are maintained for eggs and alevins, and fine sediment deposition is reduced preventing clogging spawning gravels. On the River Washburn there was a lack of suitable spawning gravels, as spawning gravels have probably been washed away over time in higher flow events and never replaced because the reservoir has trapping sediment and reduced the rate of sediment replacement in the River Washburn. Plans to reintroduce gravels into the River Washburn need to ensure that the hydromorphological process will not result in the same gravel displacement. Habitat availability was not always the limiting factor for brown trout populations, suggesting other parameters such as water quality and siltation are key drivers. Chapter 5 demonstrated the influence of habitat modification works to reduce flood risk on brown trout populations. Flood defence works were considered not to be sympathetic to habitat requirements of brown trout, which resulted in habitat improvement works measures being implemented in 2011 following the flood defence works in 2010. Integrating habitat improvement works into flood risk management is still in its infancy, highlighting the need to identify best practices (Cowx *et al*., 2013). Following flood defence works the river habitat was more suitable for 0+ brown trout populations, the shallower river bed provided better spawning and nursery habitat for 0+brown trout. However, the flood defence works caused deterioration in quality and/or quantity of habitat for larger trout (>20 cm), probably a result of the shallowing of the river bed meaning limited deeper pools for larger trout. The flood defence works uncovered a weir, creating a potential barrier for migrating fish, to what extent the weir impacted upon fish migration was not investigated . Following habitat improvement works at Malin Bridge in 2011, brown trout populations were similar to those found prior to flood defence works in 2009. Habitat improvements resulted in range of habitats including; shallower riffle areas suitable for 0+ brown trout, deeper areas and increased available cover for >0+ brown trout, providing suitable habitat for a range of brown trout sizes and age classes.

7.1.3 The influence of freshets on temperature profiles

Temperature is considered to be a controlling factor triggering spawning migrations, thus it was t monitored throughout the study and during the freshet releases in the River Holme (Chapter 4). Temperatures recorded were within natural spawning temperature ranges, suggesting optimal temperatures for spawning were experienced. Temperature differences were found between freshet and control days but differences were found between control days suggesting differences in temperature were attributed to day to day temperature variations and not as a result of freshet releases potentially influencing trout behaviour.

7.1.4 Management considerations when carrying out flow modifications and habitat improvement in rivers

It is important to consider the feasibility and restrictions around carrying out environmental flow releases from reservoirs and habitat improvement measures. Applying guidance and recommendations from literature, coupled with experimental studies as carried out in this thesis, into real life scenarios can be difficult for managers to implement.

Yorkshire Water Services were able to introduce seasonal compensation flows and one single freshet to the rivers Holme and Ribble in November from 2004 onwards (Chapter 3). Chapter 4 explored the effectiveness of the single freshet in 2012 from Brownhill Reservoir, the flow regime was not promoting long distance migrations of brown trout. In 2013, Yorkshire Water Services increased the frequency of events from Brownhill Reservoir into the River Holme to once a month in October, November and December. The same number of freshets were released from Digley Reservoir but at a higher magnitude than Brownhill Reservoir. After changes were made in 2013 the freshet releases were again not triggering brown trout long distance migrations. Possible reasons why spawning migrations were not observed are the time of day of the release, duration of event, rapidity of change, frequency and magnitude. When rethinking the possible modifications to the flow regime, restrictions around flow releases need to be consider. Yorkshire Water Services are restricted to the amount of water they can release from Brownhill and Digley reservoir on a daily basis, and it is possible that performing high flow events over a longer period of time or increasing the frequency of events might exceed the limits set by Yorkshire Water. Certain components of the winter freshet programme might never be achievable in the set limits, for example; magnitude would have to be compromised to extend the duration of the freshet event to ensure limits set are not exceeded (Chapter 4). Currently Yorkshire Water Services only release seasonal compensation flows and autumn/winter freshet releases from Brownhill and Digley reservoir, other environmental flow building blocks need to be explored, such as introducing late summer flows to flush away fine sediments. It is important to consider that in periods when rainfall has been low and water resources are limited, particularly in summer, that there could be tighter restrictions around environmental flows. For example, the frequency, magnitude and/or duration of freshets might need to be reduced or even completely withdrawn to ensure there is enough water for potable supply.

Freshet releases are manually operated which requires a Yorkshire Water Services member of staff to operate the reservoir freshet releases, and as a result the freshet release hydrographs were not controlled, the ascending and descending limbs of freshet did not mimic those of comparable natural freshets, the changes were stepped rather than curved due to the manual operation of release valves (Chapter 4). This meant that the rapidity of change was sudden, not providing any opportunity or cue for brown trout to seek protection from the high flows (especially smaller 0+ brown trout) and the falling limb is considered to be the time in which spawning migration occurs, highlighting the importance of freshets to mimic natural comparable events (UKTAG, 2013).

Carrying out reservoir flow releases and habitat improvement techniques can be expensive. Releasing water from reservoirs is costly to Yorkshire Water, the water that is designated for compensation and freshet releases are valuable resources that would be otherwise used for a number of other purposes for example human consumption and irrigation. There is also a cost involved when employing a member of staff to operate the releases, the more complex and frequent the releases are from the reservoirs the more money that would need to be spent manning the station. Considering if this was on a larger scale with, Yorkshire Water Services operating over 120 reservoirs, if freshet releases across Yorkshire would be very costly to implement. A solution would be to install automatic flow valves to reduce the man hours required to meet flow requirements, but this also can be expensive, and may be considered cost prohibitive (Dyson, 2003). To reduce costs, rivers could be prioritised in order of importance or rivers that have less variable flow prioritised over rivers that have more complex flow regimes, in that way not all rivers would require environmental flow releases reducing costs. Management plans need to be established to ensure cost benefits are achieved. The budget available for improving river habitat quality dictates the extent of river to be improved along with the quality and quantity of materials required to implement the measure. It is ecologically more beneficial if a greater section of river is rehabilitated, for example in Chapter 6 where only a small section of the Rivers Loxley and Ribble were rehabilitated the benefit to brown trout populations was limited, whereas in Chapter 5 a larger sections of rivers were planned to be rehabilitated, and thus in theory a greater benefit should be seen as a result (Hering *et al*., 2015). However, only the top three sections (1 km) in the River Washburn have planned habitat improvement works, instead of focusing habitat improvement efforts in one continuing stretch of river, there could be a greater benefit distributing habitat improvement measures in intervals over the six sites monitored (3.5 km) and ensure all sites are connected hydraulically.

Flow releases and habitat improvement measures in this study were focussed primarily on brown trout but it is important to consider the possible conflicts with, and impacts on, other species. This is where a balance needs to be established and that any mitigation or habitat improvement measure implemented does not put other fish species or taxonomic groups at risk, for example where the release of freshets in late spring/early summer to flush fine sediments may cause negative impacts on invertebrates on exposed riverine sediments (Bradley *et al*., 2012) or have negative implications on conservations species, for example bullhead. Not only do other species need to be considered but various life stages, brown trout occupy a range of meso-habitats (e.g. riffle, pool, run and glide), it is important to consider the potential effects of environmental flows on smaller and younger brown trout to ensure they are not being displaced downstream by freshets or habitat availability does not favour one life stage of brown trout (e.g. pools and large trout). For example, understanding the impact of rapid change in flow when freshets are release to ensure 0+ or smaller (<20 cm) brown trout become displaced or stranded. The flow releases and habitat improvement measures put in place need to provide multi benefits to the species present in the study reaches.

7.2 CONCLUSIONS AND RECOMMENDATIONS

7.2.1 Reservoir flow release trials

To-date, freshet releases from Brownhill and Digley reservoirs have not resulted in long-distance uni-directional migrations of brown trout. At this stage it is uncertain if artificial freshets will result in long-distance uni-directional migrations of brown trout, but the magnitude, duration, diel timing and frequency do not currently meet the flow profile recommended by UKTAG for autumn and winter flow elevations to support spawning migrations (UKTAG, 2013). For the next series of trial reservoir releases from Brownhill and Digley reservoirs **it is recommended that the reservoirs should release 6x Qn_{95} autumn/winter freshets once per week at night of 12-hour duration and, where possible, synchronised with catchment rainfall events during the months of October to December**. If licencing restrictions are non-negotiable and limit such an approach, a similar investigation should be performed downstream of another water storage reservoir with a freshet release, preferably with a short reach prior to the confluence with a main stem river, to elucidate if this current case study is representative of other reservoirs in Yorkshire. More generally across the UK, **it is recommended that flow trials should be undertaking experimenting with all aspects of the flow profile in accordance with UKTAG guidance.** Such an approach would require the natural flow regime (Qn) to be modelled downstream of each reservoir, or possibly at the downstream limit of the WFD waterbody, to help determine the recommended magnitude of the freshet releases. It is possible that not all aspects (annual minimal flow, flood flow, late summer flows, autumn /winter flows and spring flow) of the UKTAG guidance can be implemented in all UK rivers due to operational constraints. It is therefore **recommended that where possible environmental flow building blocks should be implemented or alternative habitat improvement measures should be carried out**, as this would at least provide some benefit to fish communities located downstream of reservoirs.

In the current study, there was a lack of flow data specific to the reaches under investigation so **it is recommended that future flow trial studies have individual flow loggers in each study reach.** Real time flow data from each study reach would have supported some of the assumptions and observations made. For example, it was observed from the Queens Mill flow gauge that the River Home is a flashy river and increased rainfall and overtopping events late December 2013 through to January 2014 increased brown trout activity. Queens Mill flow gauge was used as a surrogate to investigate flow in the River Holme, but the gauge was located approximately 14 km downstream of Brownhill and Digley reservoirs, and thus the natural gather and variation seen at Queens Mill Flow gauge would not be a true reflection of the flow within the reaches studied; using flow loggers in the reach would improve the accuracy.

In Chapter 4 it was speculated that brown trout should have migrated to suitable spawning habitat but in this study there was no measure of how much suitable spawning habitat was available for them to migrate to and how far away it was, **It is recommended that visual walkover surveys should be carried out to locate and map brown trout habitat.** Using habitat categories defined in Hendry and Cragg-Hine (1997) would help to understand the amount of habitat types available for different brown trout life stages within the study reach. This would help identify whether barriers to migration (weirs) were preventing brown trout form accessing different habitats and/or clarify that all the habitats that brown trout require are accessible within the study reach, supporting the theory that long distance spawning migrations were not necessary as suitable spawning habitats were in close proximity to radio tracked brown trout.

7.2.2 Improving habitat for brown trout through river habitat improvement works

A range of habitats are required to support the various life stages of a brown trout (pools and glides for adult brown trout and riffles for eggs and 0+ brown trout). In Chapters 5 and 6 it was identified that there was a lack of pooled habitats and overhead cover available for larger brown trout (>20 cm), a lack of suitable spawning and nursery habitats for eggs and 0+ brown trout. I**t is recommended when river habitat improvement works are carried out that a range of habitat types are available to support the different life stages of a brown trout.** A range of techniques can be used to achieve this, for example, placement of log or boulders, installation of brushwood, introduce gravels and remaindering a straight channel (Roni, 2013).

This thesis investigated the effectiveness of reservoir releases and habitat improvement works separately, but Hannaford & Acreman (2007) suggested that habitat modifications maybe required to work with reservoir releases to meet biological requirements, where feasible and appropriate. I**t is recommended that reservoir flow releases and river habitat improvement should be carried out concurrently where possible** to maximise the ecological benefits.

Weirs were present within study reaches (Chapter 4 and 6), it was uncertain to what extent the weirs influence brown trout movements (Chapter 4) and brown trout population densities (Chapter 6). **It is recommended that any barriers to migration and population isolation should be removed or modified to allow fish passage.** Carrying out this recommendation would require monitoring around the impact the weir has on fish movements and if weirs are passable during certain flows, this could be potentially investigated using radio tracking or even cheaper techniques such as visible implant elastomer tags (Enders *et al*., 2007; Pépino *et al*., 2012). These techniques can be further used to detect efficiency of fish passes once installed or removal of weir has taken place.

7.2.3 Water resource management (Future planning, prioritisation and climate change)

Pressure on water resources and environmental flow releases will more than likely be a challenge in the future, especially considering the thermal impact of climate change and the lack of water resources to mitigate the impacts of climate change (Olden & Naiman, 2010). Water resources may be limited in periods of low rainfall and drought, and in this instance freshets releases nay be constrained and not available at the recommended duration, frequency or magnitude. To best maximise the resources available **it is recommended that reservoir releases should be timed to occur with rainfall events** (Alfredsen *et al*., 2012). This approach would utilize the residual flow and give a real time reflection of environmental flow. Less water from the reservoirs would be required or releasing water from the reservoir might be unnecessary if the natural high flow event achieves desired freshet profile. This would require monitoring the forecast and the uncertainty of predicting the weather may cause some difficulty when considering this method. A way to overcome the predictability issue is to base environmental flows on the last five to seven days of discharge, this would be better for authorities and water manages to control and document but there would be a lag time between the natural inflow and release (Alfredsen *et al*., 2012). Another recommended approach is to monitor discharge in an unregulated system, to determine whether a wet, dry or normal period is occurring and select flow releases accordingly (Alfredsen *et al*., 2012). This would utilize the residual flow and provide a close to real time environmental flow. Many of the rivers in the UK are heavily modified and finding an unregulated river within some systems could be difficult. Arthington *et al*. (2006) advised site specific monitoring, this is especially important when prioritising rivers and whether they are eligible for flow trials.

7.2.4 Study Design

In Chapter 3 there were no reference sites to account for temporal and spatial variation in brown trout populations on the rivers Holme and Ribble. **It is recommended when implementing flow trials reference sites should be sampled to** eliminate the natural variation and help identify any changes in brown trout populations in response to the changes in reservoir flow releases. In Chapter 4, brown trout were only monitored in regulated river reaches. I**t is recommended that in future radio tracking studies fish should be monitored in regulated and unregulated reaches** to provide a comparison of behavioural responses to flow variation. In Chapter 5 there was a lack of baseline data and reference sites, thus **for future flood defence and habitat improvement sites at least two years of baseline data should be collected and reference sites should**

be sampled to account for environmental variability and temporal trends in brown trout populations. Ideally a full BACI design (monitoring multiple controls and impacts before and after) should be applied when carrying out habitat improvement works. The approach includes spatial and temporal replication, increasing statistical power to further detect treatment effects from natural variability; which is critically important when studying salmonid populations. A number of parameters could be explored such as impact on 0+ and >0+ densities, numbers, growth, survival rates and proportion of stock densities as well as, flow and temperature further understanding the effectiveness of mitigation and habitat improvement measures. Numbers of impact sites and reference sites will vary from study to study and depending on which parameters are tested. More reference sites should be sampled in the first year of study and discarded based on how comparable they are to the impact sites.

7.3 OTHER FACTORS INFLUENCING BROWN TROUT POPULATIONS

In this thesis flow and habitat were the main aspects focused on which could influence brown trout populations but there are many other parameters which could have influenced brown trout populations.

Temperature was investigated in Chapter 4 looking at changes in temperature during freshet events, other parameters such as water quality could have influenced movements in brown trout but also influence populations in other chapters, **it is recommended when reservoir releases and/or river habitat improvement works are carried out, water quality should be measured**, dissolved oxygen and pH are the most commonly measured when assessing water quality from reservoir releases and additionally when habitat improvements are carried out, phosphorus and nitrate sampling should be carried out to detect agricultural runoff, this monitoring will detect any underlying water quality issues that could be preventing brown trout populations from improving.

Food availability for brown trout could have been a contributing factor to variations in population densities and growth between years, invertebrates are the primary food source of brown trout. Fluctuations in invertebrate populations between years could have influenced brown trout population densities, in addition limited food resources could have restricted growths, **it is recommended that invertebrate sampling should take place once a month,** to allow for comparison between invertebrate samples and brown trout population densities and growth but to also act as another biological indicator to changes in flow and habitat.

It must be acknowledged that predation, parasites and disease could have influenced brown trout populations. Predation could be in the forms of avian (e.g. heron & cormorant) and/or mammalian (e.g. otter & mink) predation which could be monitored by making a note of any predators present or associated indicators such as spraint or scats. When carrying out fisheries surveys and identify any scaring or injuries on brown trout individuals that could have occurred due to predation, predation could also occur through other fish eating eggs and larval fish, monitoring this would be challenging but it's important to acknowledge. Parasites can be lethal and kill fish, making brown trout more susceptible to predation and less able to perform, day to day activities such as feeding, when carrying out surveys brown trout should be examined for any potential parasites. Diseases could have increased mortality in particular years influencing densities, monitoring this would require taking blood and pathology samples, this method might not be necessary for this study but it is worth acknowledging to be a potential influencer of brown trout population change.

The collection of this information may be cost prohibitive and these aspects were not within the scope of this thesis, but further investigations by Yorkshire Water Services and the EA are being carried out to monitor these parameters.

7.4 CLIMATE CHANGE

Water resource management and flood risk management will become increasingly important in the future with concerns around climate change. Predicted changes due to climate change include; changes in rainfall, surface runoff and increased air and water temperatures. The extent to which climate change will impact freshwater ecosystems will be difficult to estimate, some impacts include increased low flow events, increased evaporation, higher and more frequent flood events, increased pollution and siltation form more intense run off events and increased water temperatures (Palmer *et al*., 2009). Ecological impacts include pressures upon life cycles and growth rates. It has been recommended that monitoring changes and rivers response is a priority and development of local scenario-building exercises that take land use and water use into account. To ensure the security of environmental flows by purchasing or leasing water rights and/or altering reservoir release patterns and incorporating rehabilitation projects to protect existing resources to minimize effect off climate change (Palmer *et al*., 2009). The long term future of brown trout in England is uncertain and with predicted temperate set to rise and Q_{95} to reduce during the summer. It is important to focus efforts on investigating summer compensation flows or have drought plans in place to prioritise compensation programmes in rivers with good fish populations and protected species. Climate change was not the main focus within this thesis but it is worth highlighting the importance of climate change to water resource management and how it will become increasingly more relevant in the future.

This thesis has helped to gain knowledge on the effectiveness of current reservoir releases and habitat improvement works by monitoring brown trout populations, this has provided guidance and support to Yorkshire Water Services and the EA on future water management and habitat improvement works decisions.

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9 APPENDIX

Table 9.1. Analysis of Variance (ANOVA) results comparing length at age one and two of brown trout between years (1998-2013) on the River Holme and Ribble (* the mean difference is significant at the 0.05 level).

(I) yearclass	(J) yearclass	Std. Error	Sig.	Std. Error	Sig.	Std. Error	Sig.	Std. Error	Sig.
		Holme		Ribble		Holme		Ribble	
		Length at age one				Length at age two			
1998									
1999	2001								
	2002								
	2006							11.57	0.310
	2007							12.97	0.016
	2008	2.77	0.005						
	2009	3.24	0.002					11.57	0.004
	2010	2.81	0.001					10.87	0.001
	2011					4.92	0.010	12.37	0.016
	2012	2.65	0.026						
	2013	2.62	0.012			4.28	0.001	9.78	0.000
2000	2002					4.42	0.009	9.67	0.000
	2003							9.67	0.000
	2008	2.20	0.000						
	2009	2.77	0.000			4.38	0.000	10.57	0.000
	2010	2.26	0.000					12.21	0.000
	2011	2.09	0.001			3.56	0.004		
	2013	2.05	0.000			4.21	0.005		
2001	2002			4.63	0.031	3.44	0.000		
	2003					3.61	0.003		
	2004					3.56	0.000		
	2005					3.87	0.000	11.06	0.000
	2010					3.41	0.000		
	2011					3.37	0.028		
	2013					3.12	0.027		
2002	2003			3.96	0.037	3.28	0.017		
	2004							6.90	0.045
	2005					3.41	0.007		
	2006			4.04	0.001			10.66	0.000
	2007			4.44	0.015			10.90	0.000
	2008			4.13	0.000			10.61	0.000
	2009							12.21	0.000
	2010	2.06	0.023	4.63	0.000			11.55	0.000
	2011			4.11	0.000			17.92	0.000
	2012			4.07	0.000	4.52	0.000	12.97	0.000
	2013			4.10	0.000	3.81	0.000	10.53	0.000
2003	2008	1.80	0.001	2.44	0.001				

	2009	2.46	0.002			3.96	0.000	10.43	0.000
	2010	1.87	0.000	3.21	0.000	3.90	0.000	10.43	0.000
	2011					3.92	0.000	11.27	0.000
	2012	1.62	0.006			4.13	0.000		
	2013	1.57	0.001	2.39	0.002	3.34	0.000	6.40	0.001
2004	2006			2.51	0.003				
	2008			2.65	0.000				
	2009	2.49	0.044			3.51	0.000	6.23	0.000
	2010	1.91	0.004	3.37	0.000	3.45	0.000	6.23	0.001
	2011			2.61	0.000	3.47	0.000	7.56	0.019
	2012			2.55	0.000	4.10	0.000		
	2013			2.60	0.000	3.30	0.000	6.79	0.026
2005	2008	1.70	0.000	2.50	0.000				
	2009	2.39	0.001			3.47	0.000	6.63	0.001
	2010	1.78	0.000	3.26	0.000	3.41	0.000	6.63	0.016
	2011	1.56	0.041	2.46	0.001	3.43	0.000		
	2012	1.51	0.003	2.39	0.001	3.90	0.000		
	2013	1.46	.000			3.04	0.000	6.32	0.020
2006	2008	1.87	0.017						
	2009	2.52	0.016			3.23	0.000	6.15	0.001
	2010	1.94	0.001			3.16	0.000	6.15	0.012
	2011					3.18	0.000		
	2012					4.58	0.001		
	2013	2013	1.65			3.88	0.000		
	2008	2.21	0.001						
2007	2009	2.78	0.001			4.03	0.000		
	2010	2.27	0.000			3.98	0.011		
	2011					3.99	0.000		
	2012	2.06	0.010			4.61	0.001		
	2013	2.03	0.003			3.91	0.000		

Table 9.2. statistical output of Mann-Whitney U Test comparing temperature of control days to freshet releases in each study reach during Brownhill Reservoir and Digley Reservoir freshet releases (7am - 6pm) in October, November and December and a control day with no reservoir freshet release in 2012 and 2013. Shaded represents control data compared with control data.

Study year	Brownhill Reservoir freshet release			Digley Reservoir freshet release			Control day
/ Site name	October	November	December	October	November	December	
2012							
Marsden Clough		(Z = 7.215, n = 90, P = 0.000)					(Z = -0.565, n = 90, P = 0.572)
Ramsden Clough		(Z = 5.794, n = 90, P = 0.000)					(Z = -0.5121, n = 90, P = 0.904)
Co-op Lane		(Z = 5.803, n = 90, P = 0.000)					(Z = -0.791, n = 90, P = 0.429)
2013							
Marsden Clough	Z =-29.312, n = 1322, P = 0.000)	Z =31.476, n = 1322, P = 0.000	Z =-31.477, n = 1322, P = 0.000	(Z = 30.438, n = 1320, P = 0.000)	(Z = 31.306, n = 1322, P = 0.000)	(Z = -18.808, n = 1322, P = 0.000)	(Z = -31.477, n = 1322, P = 0.000)
Ramsden Clough	(Z = -4.517, n =1263, P = 0.000)	(Z = -21.169, n = 1322, P = 0.000)	(Z = 29.443, n = 1382, P = 0.000)	(Z = -3.876, n = 1321, P = 0.000)	(Z =-5.642, n = 1322, P = 0.000)	(Z =-17.888, n = 1382, P = 0.000)	(Z = -17.888, n = 1382, P = 0.000)
Co-op Lane	(Z =-2.142, n = 190, P = 0.032)	(Z =-11.938, n = 191, P = 0.000)	(Z = 9.171, n = 208, P = 0.000)	(Z = 8.450, n = 191, P = 0.000)	(Z =12.146, n = 198, P = 0.000)	(Z = -6.085, n = 208, P = 0.000)	(Z = -12.460, n = 208, P = 0.000)