THE UNIVERSITY OF HULL

Impact of flow regulation and habitat improvement on brown trout in Yorkshire rivers.

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by

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ABSTRACT

In the UK, reservoirs are a relatively common component of the landscape and provide a variety of functions, including the provision of water to meet industrial, agricultural and societal needs, as well as recreational activities such as sailing and fishing. However, reservoir function and operation impacts on downstream aquatic ecosystems by severely altering and degrading the flow regime. The importance of a natural flow regime in rivers is crucial not only to geo-morphological processes, but also to the life cycles and ecology of ichthyofauna present in these rivers. The introduction of the Water Framework Directive in 2000 (WFD: 2000/60/EEC) provided a framework by which the surface water in all 27 EU member states must achieve Good Ecological Status (GES) or Potential (GEP) by 2027. This thesis examines the drivers of population dynamics of brown trout (*Salmo trutta* L.) in rivers where the natural flow regime is severely degraded due to reservoir operation, as well as determining the effectiveness of introducing and amending reservoir release regimes and physical habitat restoration on the habitat quality and brown trout populations.

Using long term brown trout population data and flow, rainfall and temperature data the roles of density-dependent and density –independent regulation were investigated in three heavily impounded Yorkshire Rivers (Rivelin, Loxley and Holme). In each of the three rivers no relationship was found between monthly discharge rates and rainfall levels indicating that the natural flow regime was un-synchronised from that expected under un-impounded conditions. Investigations into density-dependent regulation of 0+ and ≥1+ brown trout, as well as length at age one, found that no meaningful density-dependent regulation was occurring in brown trout populations throughout the three study rivers; this was likely due to the low densities in which brown trout were found throughout the study sites. Using mixed effect linear models, it was found that the variability of flow regime in the summer period (June – September) was significantly correlated with 0+ brown trout densities, and that the flow variability during the emergence period (April –May) was also significantly correlated with the length of brown trout at age one. Temperature and flow during any other period of the year were not found to have any significant interaction with 0+ brown trout density or length at age one.

The introduction and amendment of compensation releases from impounding reservoirs is an established methodology for improving the ecological potential in downstream reaches of rivers. Long term effects on brown trout populations and estimates of habitat quality in rivers where new compensation release regimes were introduced were examined in three study rivers (Dibb, Dale Dike and Holme). In all three rivers the changes to densities of three age/size classes of brown trout (0+, ≥1+ <20cm and ≥1+ >20cm), length at capture of 0+ brown trout and habitat quality for the three age/size classes were examined to determine if the new compensation regime had any meaningful impact using a Before After Control Impact (BACI) study design. In the River Dibb, where a four stage seasonally varied release was introduced, there was no significant changes to the biological metrics but, there was a significant improvement in the habitat quality for ≥1+ (<20 cm) brown trout which could be attributed to the introduction of the compensation release. Operational constraints from Dale Dike reservoir were such that only an annual minimum flow could be established from the reservoir; in the study no significant changes to the biological metrics were detected, but there was a significant decrease to the 0+ brown trout habitat quality that was attributed to the introduction of the annual minimum compensation release. The findings suggest further amendments to the compensation release regime or alternative restoration methods (such as habitat restoration) should be considered to achieve GEP in Dale Dike. The compensation release from Brownhill and Digley reservoirs were revised to provide better flow conditions in the River Holme. There was no significant response from the biological metrics tested, however there was a significant improvement to the habitat quality of ≥1+ (> 20 cm) brown trout, suggesting that the revised compensation release provided better habitat conditions for larger, more fecund brown trout.

As it is not always possible to attain GEP in impounded rivers by amending reservoir release regimes due to operational circumstances the use of physical habitat restoration measures can be implemented to improve habitat conditions downstream of impounding reservoirs. In the River Washburn a BACI study was undertaken to detect if any meaningful change to brown trout populations and habitat quality had occurred following habitat improvement works in 2015. Immediately following the habitat improvement works the habitat quality for all brown trout age/size classes improved, but in late 2015 a 1-in-100-year flood event destroyed and damaged a majority of the instream works. There were significant improvements immediately following the habitat restoration work to the habitat quality for both ≥1+ size classes, but due low densities of brown trout following the habitat improvement there was a significant decline in ≥1+ (<20 cm) brown trout populations.

As it was found that there were significant changes to the habitat quality for brown trout across the four study rivers, further sampling of brown trout would be required to determine the biological response to the habitat changes. As the level of temporal and spatial variance was high in brown trout populations throughout this study, further monitoring would be required to draw robust conclusions as to the impact of Heavily Modified Water Body (HMWB) mitigation on brown trout populations. As habitat requirements for brown trout are different depending on the life stage, the complementary use of both habitat improvement and flow restoration techniques should be explored in future projects to mitigate HMWB status.

1 GENERAL INTRODUCTION

Rivers are a fundamental feature of the earths' geography draining over 70% of the land mass (Craig 2015). As a natural resource, rivers provide a range of ecosystem services e.g. climate regulation, provision of potable water, provision of water for agricultural and industrial processes and the provision of habitat for aquatic organisms and (Daily *et al.* 2012). At the most basic level, fish require aquatic habitat to survive and thrive, but riverine ecosystems can also provide a plethora of habitat types to support not only diversity amongst species but diversity amongst life stages (Cowx and Welcomme 1998).

For early human settlements, rivers were an important facet of civilisation, with evidence suggesting that modifications of rivers for human benefit were occurring at least 6000 years ago (Niehuis and Leuven 2001). With advances in technology, the nature in which humans alter and use river networks have also changed. In early European civilisations river modification consisted of the use of embankments and weirs to provide water supply and land drainage schemes for agriculture (Niehuis and Leuven 2001). Further advances in human society led to the canalisation of many rivers for navigable purposes, as well as further flow regulation through weirs and sluices to control flooding; eventually urban expansion resulted in the adoption of rivers as open sewers to transport effluent away from urbanised centres (Walsh 2000, Niehuis and Leuven 2001). The resultant high level of river degradation was particularly prevalent in developed and industrialised countries, where the development of urban centres and industry occurred with little or no consideration of the impacts that developments had on the rivers ecological status.

Natural river flow is one of the most important factors that governs the ecological integrity of the instream habitat (Poff *et al.* 1997a), with many critical physicochemical processes of river systems dependent on the intricacies of a natural flow regime (Power *et al.* 1995, Poff *et al.* 1997a, Resh *et al.* 2016). The pervasiveness of reservoirs and dams globally to manage, store and harness water resources is a major issue with investigations of river flow revealing that 85% of rivers in the United States and nearly 90% of rivers in the UK subjected to non-natural flow regimes (Petts 1988, Poff *et al.* 1997a, Acreman *et al.* 2009). The expansion of industrial processes in the UK, in particular water-driven cotton and woollen mills, led to vast over abstraction of the water resource in many rivers. Coupled with the expansion of urban areas surrounding these mills, a large number of reservoirs were commissioned over the last two centuries in the UK with the purpose of providing both adequate water supply and the provision of sufficient flow rates to meet the demands of water-driven industry. The releases from reservoirs to elevate flow levels in the river during working hours are termed as compensation flows, and despite the decline in water-driven mills compensation releases from reservoirs are still commonplace in reservoir regulated rivers to ensure the river levels and subsequent downstream abstraction levels are maintained (Gustard *et al.* 1987). In over 70% of reservoirs the compensation flows released are in the form of a fixed discharge rates that remains unchanged for seven days a week, 52 weeks a year (Gustard *et al.* 1987). In these reservoir-regulated rivers, the flow regime can deviate from what would be expected under natural, un-regulated conditions, potentially leading to ecological degradation. There is a general consensus amongst the scientific community and the lay public that the conservation and protection of river ecosystems is important (Poff *et al.* 1997b). International meetings of global leaders, such as at the Hague 2000, Johannesburg 2002, and Kyoto 2003, all served to highlight the need for a more harmonious and sustainable approach to water resource management to improve ecological integrity of river ecosystems whilst still meeting the water needs of people, agriculture and industry (Acreman *et al.* 2009). The Water Framework Directive (WFD: 2000/60/EEC) provides the legislative template in the European Union (EU) to more responsible water management. The goal of the WFD was for all waterbodies in the EU to achieve good ecological status or potential by 2015, this deadline has since been further extended to 2027 after it was apparent that numerous waterbodies would not reach the required good status within the original timeline.

In Yorkshire, a region in Northern England that comprises the counties of North and South Yorkshire as well as the East and West Ridings of Yorkshire, water supply and treatment are serviced by Yorkshire Water Services (YWS). Throughout the Yorkshire region, YWS maintain and operate over 120 water supply reservoirs across a catchment of approximately 11,900 km^2. The majority of these reservoirs are centuries old, with the construction of some occurring over 200 years ago, and infrastructure and operation on certain reservoirs remaining unchanged for the majority of their lifespan. The introduction of the WFD into UK legislation put the onus on water utility companies, like YWS, to invest considerable resources into managing the water resource with greater emphasis not just on economic benefits (i.e. providing drinking water at reasonable cost to the consumer), but wider issues such as societal (i.e. flood management) and ethical (i.e. environmental protection) needs of the water resource (Acreman 2001). Water utility companies (such as YWS) in conjunction with government agencies (i.e. Environment Agency (EA), Scottish Environmental Protection Agency (SEPA), Northern Ireland Environment Agency (NIEA) and Natural Resource Wales (NRW)) are also investing considerable resources in research surrounding effective water resource management as part of their statutory obligations to compliance with the WFD (Bowles and Henderson 2012). This research aims to advance our understanding of the interconnected network of ecological, societal, and economic pressures to aid implementation of more effective water resource management.

Brown trout (*Salmo trutta* L.) are a common fish species throughout Yorkshire river systems, making them an ideal candidate for measuring the biological response to habitat improvement and restoration techniques due to their sensitivity to a variety of ecological pressures (i.e. water quality, flow regulation and habitat modification) (Roni and Quimby 2005, Pont *et al.* 2006). Due to the wide geographic range of brown trout across Europe (Frost and Brown 1967, Jonsson and Jonsson 2011), it is a species commonly used by regulatory authorities for WFD testing compliance in upper river reaches. The global distribution of brown trout (Frost and Brown 1967, Jonsson and Jonsson 2011), allows for the findings and knowledge gained from the studies presented in this thesis to be applied in a much wider global context.

This thesis aims to develop and expand on the existing knowledge pertaining to brown trout population dynamics in HMWBs where flow is regulated by impounding reservoirs. The habitat and brown trout response to mitigation of reservoir regulated HMWB status through flow restoration and habitat modification is also assessed. This to determine the success of individual flow restoration and habitat modification methodologies and with the incorporation of the knowledge base (from not only this thesis but from existing studies and peer review literature) to provide a critical evaluation of the success of current mitigation measures which can be used in the future application of mitigation of reservoir regulated HMWBs.

This thesis is divided into two main sections, with the first section (chapter 3) analysing long term brown trout population data from three heavily modified rivers in Yorkshire and investigating the drivers behind density dependent and independent population regulation. Numerous studies, such by Elliott and Lobón-Cerviá (Elliott 1984, 1993, 1994, 2015, Lobón-Cerviá 2004, 2007, Lobón-Cerviá and Rincon 2004, Lobón-cerviá 2005) have compounded our understanding of brown trout population regulation mechanics in natural river systems. However, elucidating the processes behind population dynamics of brown trout in heavily regulated rives is important, and less well studied. Knowledge gained from this investigation can be applied to future management and conservation of brown trout in Heavily Modified Water Bodies (HMWB). The second section of this thesis (chapters 4 – 7) investigates the habitat and biological responses to four different flow restorations and rehabilitation techniques undertaken on HMWBs in Yorkshire. These chapters aim to highlight the importance of restoration and rehabilitation work as well as critically evaluate the design, implementation and impact that these restoration measures may have on both brown trout populations and habitat quality.

Chapter 2 reviews the life history and underlying mechanisms of population dynamics of brown trout, especially in relation to the importance of the natural river flow regime. The chapter further reviews the impact that the operation of water storage and flow management has on the natural river flow regime and the legislative and operational framework introduced to stimulate ecological improvement.

Chapter 3 examines the biotic and abiotic mechanisms that drive brown trout population dynamics in three heavily regulated rivers in Yorkshire. These results will then be used to identify any significant drivers of population variability, which can help advance current scientific knowledge of brown trout population dynamics in HMWBs.

Chapters 4-7 presents cases studies of mitigation of HMWB status in reservoir regulated Yorkshire Rivers. Each of these case studies investigates and evaluates the habitat change and biological response in the river downstream of an impounding reservoir. The mitigation measure applied varies in each case study and are: the introduction of a four-stage seasonally variable compensation release from Grimwith reservoir (Chapter 4), the introduction of an annual minimum compensation release from Dale Dike reservoir (Chapter 5), the revision of an existing compensation release from two reservoirs – Brownhill and Digley Reservoirs (Chapter 6) and the use of instream physical modification methods downstream of Swinsty Reservoir (Chapter 7).

Chapter 8 integrates and provides critical analysis of knowledge gained from Chapters 2 to 7, and subsequently provides recommendations and conclusions of the case studies presented in these chapters. These recommendations and conclusions will provide a strong foundation from which future projects to mitigate flow regulation in HMWB designated rivers can be based.

2 LITERATURE REVIEW

2.1 Brown trout life cycle and habitat requirements

Brown trout (*Salmo trutta* L.) is a fish species native to the United Kingdom and much of mainland Europe (Frost and Brown 1967). Brown trout is listed as a UK Biodiversity Action Plan (UKBAP) priority species (Biodiversity Reporting and Information Group 2007). The Environment Agency (EA) has also highlighted the need for protection and enhancement of wild brown trout as part of the national trout and grayling strategy (Environment Agency 2003). Due to brown trout abundance throughout many river systems globally (Frost and Brown 1967, Jonsson and Jonsson 2011), as well as its sensitivity to a variety of pressures (chemical pollution, flow regulation and habitat modification) it is a model study species (Pont *et al.* 2006), therefore brown trout were selected as the primary study species for this thesis. Brown trout has a diverse life history (Jonsson 1989), which can be broken down into the following phases; spawning, incubation and emergence, juveniles, parr and maturation.

2.2 Life history

2.2.1 Upstream migration and spawning of brown trout

Upstream spawning migration of brown trout typically occurs during late autumn and winter in UK rivers, the precise timing of which is usually dictated by environmental triggers such as decline in temperature and flow elevations (Frost and Brown 1967, Banks 1969, Crisp 2000, Jonsson and Jonsson 2011). There are several reasons for upstream migrations occurring during high flows. Despite having to swim against the higher flows, these aid upstream migrations by increasing longitudinal connectivity through further access upstream and ensuring potential barriers (such as weirs or natural barriers) are more easily ascended. The turbulence in the waters caused by greater flow also provide visual shelter to migrating individuals from aerial predators (Jonsson and Jonsson 2011). Colder, faster flowing water is preferable for brown trout spawning due to the increased concentration of dissolved oxygen at lower temperatures (Wetzel 2001). Elevated flows during this period also serve to maintain and provide suitable spawning substrate through the shifting and cleaning of fine sediment from within the gravels, which can aid in survival of eggs (Robison 2007, Jonsson and Jonsson 2011). Brown trout, like other salmonids, has strong spawning preference for gravels, which range from 32 -64 mm, but it is believed that female body size may be positively related to the mean particle size of the spawning substrate (Kondolf *et al.* 1993, Crisp 2000, Jonsson and Jonsson 2011), suggesting that larger females are capable of spawning in coarser gravels. Spawning of brown trout occurs typically in shallow waters ranging from 6 -91 cm (Cowx *et al.* 2004), and there is no clear evidence to suggest that larger brown trout spawn in deeper water, but it is believed that females display a tendency to select spawning grounds where the water depth fully submerses the female (Crisp 2000). Females use their caudal fins to cut and dig depressions into the gravel (redds) where eggs are deposited. Male trout do not exhibit nest building behaviour, instead they remain close to females during the spawning process, occasionally having aggressive interactions with other males (Jonsson and Jonsson 2011). Once the female has commenced shedding of eggs in the nest a dominant male will swim alongside and deposit sperm fertilizing the eggs. Access to females is controlled by the larger more dominant males (typically anadromous individuals, if the population holds them), but smaller males can sneak in and release their gametes after the larger male has swum off (Jonsson and Jonsson 2011). As soon as eggs are fertilized the female will cover with gravel and possibly excavate a new redd. This cycle repeats until males and females are fully spent, which can take from hours to days, depending on the population (Jonsson and Jonsson 2011). Brown trout are iteroparous - meaning that they do not die after spawning, as many fish will return to the same natal stream to spawn again unlike Pacific salmonids. The burying of eggs ensures they are not washed out of the redds by elevated flows (unless extreme), and also affords protection from any mechanical stress that would occur to the eggs or the newly hatched alevins if left exposed to high flows as well as floating debris (Cowx *et al.* 2004, Jonsson and Jonsson 2011, Unfer *et al.* 2011).

2.2.2 Incubation, hatching, and emergence of brown trout

Incubation time of salmonid eggs is proportional to temperature (Frost and Brown 1967, Crisp 1981, Elliott 1984, Jonsson and Jonsson 2011). The time from spawning to hatching decreases with increasing temperature (Frost and Brown 1967), while mortality of eggs increases with the temperature, with high mortality experienced at ≥14°C and few if any individuals completing embryonic development at temperatures ≥16°C (Ojanguren and Brana 2003). Hatching occurs not long after the eyes of the embryos are visible in the eggs. The physiology of the newly hatched trout (alevin) greatly differs from the adult form, as they are transparent, and carry a yolk sac beneath their abdomen. Even after they have hatched the alevins will remain in the gravels and this phase is known as the inter-gravel stage. During the inter-gravel stage alevins will move infrequently unless it is to light stimuli as they exhibit strong negative photo-taxis (Frost and Brown 1967, Jonsson and Jonsson 2011). Alevins also do not feed externally as they rely on their yolk as a food source. As more of the yolk is consumed the physiology changes, the fins and body shape becomes more developed into the adult form, and the skin becomes pigmented. When the yolk supply is almost fully exhausted, alevin behaviour changes to positive photo-taxis and they emerge from the gravels and orientate themselves to face the current (rheotaxis), whereupon they commence exogenous feeding, typically on drifting and epibenthic arthropods (Frost and Brown 1967, Jonsson and Jonsson 2011). During incubation to emergence phases, extreme high flow events, such as spates or floods, can cause increased mortality to young-of-year (Fry, alevins and parr) brown trout through washout, displacement and mechanical shock (Crisp 2000, Lobón-Cerviá 2004, Daufresne *et al.* 2005, Jonsson and Jonsson 2011). Investigations into the relationship between discharge and recruitment revealed that it is not just extreme high flow events that negatively impact on the incubation to emergence phases of brown trout, with analysis of long-term datasets in northern Spain revealing a parabolic relationship between discharge and recruitment success ((Figure 2.1) Lobón-Cerviá 2014). Extreme low flows can also have devastating effects on developing brown trout as they can reduce the interstitial flow of water in the spawning gravels (Jonsson and Jonsson, 2011). The level of interstitial flow is an important factor to developing brown trout, as these flows allow for the maintenance of dissolved oxygen in the egg pockets as well as removal of metabolites, such as ammonia, which is toxic to salmonids (Crisp 2000).

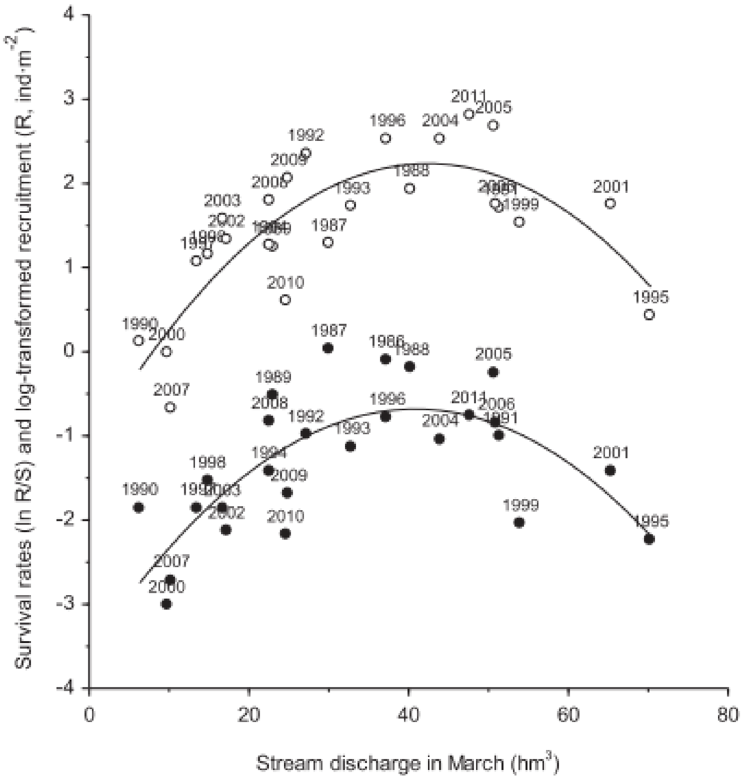

Figure 2.1 Parabolic relationships depicted by the mean annual survival rates (ln(R/S), open circles) and log-transformed mean annual recruitment averaged across sites (R, individuals (ind)·m−2, solid circles) against stream discharge in March (hm3). Taken from (Lobón-Cerviá 2014)

2.2.3 Brown trout juvenile and parr stages

Young brown trout parr establish feeding territories almost from the moment they begin exogenous feeding (Jonsson and Jonsson 2011). These territories are defended, won and lost through aggressive interactions between fish, with larger individuals holding the “best” territories for feeding. Feeding stations are maintained by individual parr, and maintained by the larger and more dominant individuals, which are in close proximity to shelter as well as fast flowing water (Grant and Kramer 1990, Jonsson and Jonsson 2011). From these feeding stations juveniles and parr feed on drifting or benthic invertebrates (Elliott 1994). Preferred habitat for juvenile brown trout is 20 to 30 cm in depth and with a velocity of <25 $cm.s^{-1}$ (Kennedy and Strange 1982, Cowx et al. 2004). A considerable amount of energy is expended on growth during the first year of brown trout life, with the average length of brown trout parr in English and Welsh rivers at the end of the year being approximately 80 mm (EA Growth Standards), with ranges from approximately 50 mm to 140 mm depending on a number of biotic and abiotic factors such as temperature and productivity of the habitat. Brown trout parr can be recognised by having between 9-10 vertical dark bars along the flank, commonly referred to as “parr marks”. Depending on brown trout genetics and environmental conditions the next stage in the life cycle can vary considerably. Many brown trout remain in their natal stream and reach maturity after their third summer (Jonsson and Jonsson 2011), while others can migrate into the main river channel or into lakes after their second or third summer, and remain resident there, reaching maturity at a similar age as stream-resident individuals and return to their natal streams only to spawn (Elliott 1994). Brown trout can also exhibit anadromous life stages, whereby they undergo a physiological and behavioural change called smolting and migrate downstream through the river network to the estuaries or the ocean to feed and are colloquially referred to as sea trout. It is believed that the process that drives the smolting and sea-ward migratory behaviour of sea trout is genetically inherited (Skaala and Nævdal 1989, Jonsson and Jonsson 2011), and that anadromous and stream-resident brown trout are genetically distinct from each other (Skaala and Nævdal 1989). As sea trout populations are genetically inherited, populations are only established in rivers and streams with relatively unimpeded access to the sea. Due to the significant number of migratory barriers throughout the river systems in the UK, particularly in Yorkshire, it is believed that sea trout are locally absent from all study sites covered in this research. Therefore, specifics of the divergent life history of sea trout as will not be covered in this literature review.

2.2.4 Brown trout maturation

There is no specific size or weight that brown trout must attain before reaching maturity (Thorpe and Metcalfe 1998). In some populations the size difference between the smallest and the largest adults can be around one order of magnitude, with mature adults being as small as 70 mm (Jonsson and Jonsson 2011). It is believed that both environmental and genetic factors influence the size variation in spawning adults (Thorpe *et al.* 1998). Thorpe *et al*. (1998) proposed a model for brown trout suggesting that certain physiological thresholds in both size and fat content must be met during a specific time-frame to begin the endocrinological processes that govern maturation. The largest variation in mature body size will be from populations that support both resident and anadromous forms of brown trout, whereby anadromous individuals several times larger than mature stream resident brown trout (Jonsson and Jonsson 2011). Inherited genetic traits and abiotic conditions also influence the variation in size at maturity, with populations from small shallow streams reaching maturity at a smaller size than other populations. Large individuals will be at greater threat from predation and reduced mobility in smaller shallower reaches (Jonsson *et al.* 2001) with length at maturity for stream resident males ranging from <150 mm to >230 mm depending on flow rate (Jonsson *et al.* 2001) Other external stimuli, such as temperature can also impact the size and age at maturation. Berrigan & Charnov (1994) stated that at reduced temperatures ectotherms reach maturity at a greater age than conspecifics, and at a greater size but the opposite is not fully true for fish at increased temperatures. While increased temperatures increase the speed of physiological processes in organisms, temperature increases above certain limits can have severe deleterious impacts on reproductive organs and processes (Pankhurst and King 2010). Lipid storage is also an essential factor in the maturation of trout, the development of both the primary and secondary sexual characteristics as well as the anorexia that is commonplace with migrating and spawning individuals (Kadri *et al.* 1995).

The primary sexual characteristics are present in brown trout at all life stages, but only develop during the maturation process. These consist of testes in male fish and ovaries in females. In the year of spawning, it is estimated that less than half of a resident brown trout's energy is allocated to reproduction (Jonsson and Jonsson 2011). This is due to the low energetic costs of small (if any) migrations to spawning grounds. By contrast those fish that have long return migrations (anadromous brown trout and Atlantic salmon) can expend around 70% of their total energy into the reproductive process. The change in size and energetic storage throughout the reproductive year is measured as a value of the Gonad Somatic Index (GSI = (gonadal mass/somatic mass)*100). This value is expected to be around 20-40%, depending on somatic size for female brown trout, but in male brown trout the GSI is expected to be around 2-4% (Jonsson and Jonsson 2003, 2011), indicating that the energetic expenditure for spawning for females is considerably greater than for males. Secondary sexual characteristics of brown trout show considerable variance and are sexually dimorphic. These characteristics tend to be changes in colouration and morphology to aid reproductive success for the individual. More dominant males start to display more striking and vivid colourations during spawning, with prominent red spots being commonplace amongst male resident trout. Females and subordinate males display a more “subdued” colouration with the flanks being varying shades of dark grey (Jonsson and Jonsson 2011). One of the most obvious morphological changes during maturation is the elongation of the jaw and the development of a hook to the lower jaw of the males. This structure is known as a kype, it develops late in the maturation process, only starting to appear once feeding and somatic growth has ceased (Witten and Hall 2002, Jonsson and Jonsson 2011). The histology of the kype is not analogous to that of the surrounding dentary bone and tissue, but post spawning the proximal parts will gradually converge into regular dentary bone, and begins to visually disappear, but in subsequent spawning years the pre-existing structure will contribute to greater jaw elongation and a larger kype (Witten and Hall 2003).

2.3 Behaviour and age-structured habitat use by brown trout

The overall density and carrying capacity for brown trout in rivers are in part regulated by territorial behaviour, which commences as soon as exogenous feeding begins (Frost and Brown 1967). Brown trout territories are held by aggressive interactions between conspecifics with subordinate fish having to migrate further afield (Elliott 1994, Landergren 2004). Brown trout are visual hunters and intruders to territories are only detected by sight (Jonsson and Jonsson 2011). Several experimental studies (Valdimarsson and Metcalfe 2001, Venter *et al.* 2008, Bonfil *et al.* 2010, Grabowski and Gurnell 2016) showed that visual isolation of individuals can reduce territory size and thus increase the overall population density, such isolation can include low light intensity, increased turbidity of water and increased substrate rugosity. However, this type of territorial behaviour does not extend through the entire life span of brown trout. Typically, beyond parr stage fish enter size-structured dominance hierarchies (Jonsson and Jonsson 2011), and in these hierarchies brown trout do not have individual territories but reside in a region described throughout literature as a “home range” (Bachman 1984, Jonsson and Gravem 1985, Jonsson and Jonsson 2011). Studies on the home ranges of brown trout have shown that the best feeding opportunities are exploited by the dominant resident who will forage throughout the range restricting feeding opportunities to subordinate fish (Elliott 1984, Landergren 2004). As brown trout become larger, they tend to occupy the deeper, slower flowing water of rivers as opposed to the shallow habitats occupied by juveniles (Keeley and Grant 1995). It is likely the utilisation of the deeper, faster flowing water for larger brown trout coincides with the reduced competition with smaller conspecifics as well as exploiting the greater feeding opportunities due to the increased abundance of drift (Crisp 2000). As previously mentioned, some juvenile brown trout move out of their natal streams due to lack of territory, and these fish are known as "movers" (Jonsson and Jonsson 2011). Landergren (2004) found that those brown trout that were displaced and had to move out of the territory were smaller than their resident conspecifics, but this was not always the case as some fish that dispersed downstream grew faster and were larger than resident individuals, lending credence to the hypothesis that in some cases dispersal can be voluntary and advantageous (Steingrímsson and Grant 2003).

2.4 Brown trout and the natural flow regime

Each unregulated river has a natural flow regime, that is dictated by climatic factors (especially rainfall) and catchment geography (i.e. geology, topography and land use), and is to be considered unique to each river (Poff and Zimmerman 2010). It is possible, however, due to synchrony and seasonality of weather patterns to generalize flow regimes both spatially and temporally (Poff *et al.* 2006). For instance, in northern hemisphere temperate rivers, the magnitude and duration of high flow events would be greater during winter than during summer. The natural flow regime can be broken into five key components that interact with each other to regulate the ecological and geo-morphological processes in riverine habitats: magnitude, frequency, duration, timing and rate of change (Poff *et al.* 1997).

Magnitude (also known as amplitude) represents the volumetric rate of water in any point in a river at any given time. It is not uncommon in natural river systems for there to be several orders of magnitude difference between the lowest and the highest flows. The magnitude of the flow in rivers plays an important role in the regulation not only of the brown trout populations but the habitat in which they reside. Flow conditions during low discharge releases can provide suitable conditions for juvenile brown trout and parr to thrive (Acreman *et al.* 2014). Newly emerged juvenile trout and parr have a reduced tolerance for higher water velocities and will seek out and establish feeding territories in areas of relatively low water velocity (Huntingford *et al.* 1988, Crisp 2000). As well as better territories for juvenile brown trout, water temperature is also negatively influenced by water velocity, as faster more turbulent water is colder (Sinokrot and Gulliver 2000). Water temperature plays a crucial role in juvenile brown trout development, influencing growth (Elliott *et al.* 1995, Heggenes 1999), with slower growth and maturation occurring at colder temperatures (Jonsson *et al.* 1991, Jonsson and Jonsson 2011). Large magnitude flow events are also crucial to the population dynamics of brown trout. There is strong evidence to suggest that the influence of discharge during the emergence phase of juvenile brown trout is a strong determinant of recruitment success (Lobón-Cerviá 2004, 2014, Daufresne *et al.* 2005, Richard *et al.* 2015), with extreme high discharge resulting in mortality and washout of newly emerged trout. High discharge rates, however, can also be beneficial to brown trout. During spawning the increased dissolved oxygen associated with lower temperatures and broken turbulent water, as well as the longitudinal connectivity afforded by elevated flows, aid in the upstream spawning migrations of brown trout (see section 1.1.1). As well as improved aeration of the redds, high discharge rates play a key role in the geo-morphological process of riverine ecology, allowing the removal of fine sediments and accumulated metabolites from within gravels (Crisp 2000, Cowx *et al.* 2004, Jonsson and Jonsson 2011, Newson *et al.* 2012).

Frequency (or return period) is described as how often flows above a specific magnitude can occur (Poff *et al.* 1997). For major flood events these can be categorised as the probability of occurrence within a specified unit of time, for instance the winter floods of 2015/2016 in Yorkshire were categorised as a 1 in 100-year flood event (Marsh *et al.* 2016). This means that a flood of this magnitude would not be expected to be seen more than once in a 100-year period. Periods of extreme flood events can be devastating both to brown trout populations and to the habitat quality they reside in, as extreme flows can scour the stream substrate and degrading nursery habitat; fish are also likely to be washed out if water velocities exceed levels that can be tolerated (Roghair *et al.* 2002).

Duration is how long a specific flow event lasts (Poff *et al.* 1997, Welcomme and Halls 2001). At extreme low discharge duration is a key component that can impact on brown trout populations. For instance, during droughts, the duration can be a critical factor for juvenile survival, as extended periods of drought can increase mortality through increased water temperature and decrease in suitable habitat (Elliott *et al.* 1997, Nicola *et al.* 2009). The duration of flood events can also influence brown trout recruitment. In Norwegian rivers it was found that shorter periods of high discharge negatively influence brown trout recruitment more than in longer periods of high discharge (Jensen and Johnsen 1999).

Timing of flow events is the regularity of which a flow of a pre-determined magnitude occurs, for instance, seasonal high flow events during winter (Poff *et al.* 1997). Brown trout, similar to many other riverine fish species, react to environmental cues, whether for individual survival or reproduction. It is believed that for brown trout and other salmonids, reduced photoperiod is a pivotal factor in the timings of maturation and spawning condition of brown trout (Crisp 2000). The timings of upstream migration of brown trout in the River Isma in Norway were found to be strongly correlated with high flow events (Jonsson and Jonsson 2002). Rainfall in temperate northern-hemisphere regions is relatively seasonal, with greater levels of rainfall expected in winter versus the summer. Therefore, there is a functional link between the environmental cues of photoperiod and rainfall to ensure the timings of maturation will be synchronized to seasons where the probability of elevated flows is much higher, increasing quality of spawning habitat, as well as upstream migrations (Crisp 2000, Jonsson and Jonsson 2002, 2011). As previously established (section 2.3.2) newly emerged brown trout are relatively poor swimmers, therefore the timing of extreme flow events during this period can have dramatic consequences through discharge-related mortality (Lobón-Cerviá 2004, Daufresne *et al.* 2005), as well as the reduction of sufficient micro-habitats for newly emergent brown trout (Lobón-Cerviá and Rincon 2004).

Rate of change (also known as flashiness) is described as the speed at which the flow regime can change from one magnitude to another (Poff *et al.* 1997). The flashiness of the riverine environment can have serious implications on the ecosystem, especially for brown trout. Sudden reductions in stream flow can reduce the longitudinal and lateral connectivity, which can cause stranding of fishes, especially of the juvenile trout that occupy marginal habitats. In the natural environment, sudden reductions in flow are rare, however, rapid increases in flow are more common. These rapid increases can be just as harmful to brown trout populations, as the rapid increase in flows can reduce the areas of slow moving or slack water, and lead to displacement of smaller brown trout that have a reduced capacity for holding station in faster moving water (Unfer *et al.* 2011, Richard *et al.* 2015).

2.5 Reservoir operation and natural flow regimes

The storage of water from rivers is designed for a variety of purposes including human consumption, agriculture, irrigation, industrial use, flood control, or for the generation of electricity, resulting in large quantities of water that should form part of the natural flow regime being stored in large reservoirs for later use (Acreman 2007). The impacts that reservoirs have on the natural flow regime can be profound and it is not uncommon for reservoirs to have the capability to store and hold back a high portion, if not all, of the flow from upstream for prolonged periods of time (Acreman 2007). The resulting flow regime downstream can, therefore, be substantially altered from that expected under natural conditions (Figure 2.2). It is still possible that extreme flow events, i.e. floods with high duration and magnitude, can influence the downstream flow regime, as reservoirs can only store finite supplies of water and overspill (overtopping, where reservoir exceeds capacity and excess flow bypasses the water storage scheme via spillway) does occur. The regulated hydrological regime contrasts starkly with un-regulated regimes (Figure 2.2) and lacks important flow elements that are crucial not only to the downstream geomorphology but also in maintaining the ecology of the downstream channel

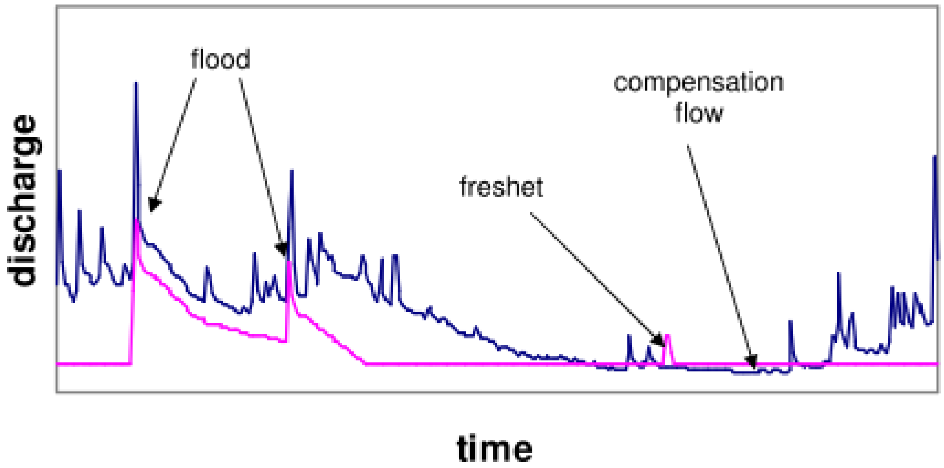

Figure 2.2. Natural (blue) and regulated (pink) flow regimes downstream of a hypothetical water supply reservoir (taken from Dunbar *et al.* 2008)
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2.5.1 Compensatory water release and reservoir operation in the UK

Since the onset of the Industrial Revolution in the United Kingdom, there has been cause for regulation of the flows in many river systems, through the storage and release of large volumes of water held in reservoirs. The earliest of these reservoirs were designed and built for the purpose of maintaining navigable channels and the expanding canal systems for the direct benefit of industry (Gustard *et al.* 1987). However, due to the diversion of water from rivers, flow rates decreased substantially, and downstream mills and factories (which relied on a steady water velocity to power machinery) were impacted and productivity was affected. Through Acts of Parliament, it was decided that compensatory releases of water from reservoirs would be required to maintain adequate flow for industry. Each reservoir release was governed by an individual Act of Parliament. The impacts that these releases had on the environment were not properly understood, but the priority at the time was to ensure sufficient flows to maximise mill productivity. For example, *The Bolton Waterworks Act 1824* stipulated the supply of water to Great Bolton, Little Bolton and Sharples, were designed to ensure a sufficient supply for the mills in this region. Originally compensation releases were scheduled around the working day, so there were no releases outside business hours or on bank-holidays and weekends. However, due to the accumulation of pollution and low flows at a local beauty spot, the *Halifax Corporation Waterworks act 1888* (Levin *et al*, 1888) gave provision for the release of water outside of work hours to improve the natural beauty of the popular tourist spot. Many more similar acts were introduced on other reservoirs and catchments soon after. The introduction of the *Water Act 1945* did not require the alteration of flows on new reservoirs to be implemented by Parliament, or end the practice of not releasing any compensation flows on weekends, a practice favoured by mill owners as all the stored water would only be released during the operational hours of the mills, which would maximise their outputs. The legislation that followed such as the *Water Resources Act 1963, Water Act 1973* and *The Drought Act 1976* invested the power that was held with Parliament into 10 Water Authorities (1973) and 26 River Authorities (1963) and these licenced the abstraction as well as determining suitable levels of release in these new authorities. This avoided the tedious introduction and amendment of legislation per reservoir that was required prior to 1945. The *Drought Act* 1976 expanded on the existing legislation to allow authorities to have powers to restrict abstraction and water use in emergency situations such as severe drought. A review into the reservoir operation in the UK revealed that compensation releases were present from 70% of UK reservoirs (Gustard *et al.* 1987, Acreman *et al.* 2009).

2.5.2 Habitat responses in reservoir regulated rivers and consequences to brown trout communities

As previously described (see Section 2.5), the natural flow regime in rivers is crucial to the maintenance and suitability of habitat for brown trout. Reservoir impoundment has a direct influence on the five elements of the natural flow regime (magnitude, timing, frequency, duration, and rate of change). In the USA, for example, the regulation of rivers by reservoir operation has on average reduced the annual peak discharge by 67% and the ratio between maximum and mean flow by 60% (Graf 2006), with the majority of freshwater river basins in the USA having at least 10% altered flow regimes due to water storage (Barlaup *et al.* 2008). Altered flow regimes can have deleterious effects on the spawning process of brown trout due to the loss of the high flow events that aid and can trigger upstream migration in mature brown trout (Aldvén *et al.* 2015). Due to severely altered run-off/flow regime relationships it is possible to lose environmental synchrony of the extreme flow events as the timing of peak flows may be altered, in some cases by up to six months (Graf 2006). This could lead to a de-synchronisation between the maturation of brown trout. The loss of magnitude can also have serious repercussions on the physical components of the river channel. The lower velocities and magnitudes of water in reservoir regulated reaches can have serious implications for the habitat of brown trout, at lower flows there is increased deposition of finer sediments such as sand and silts (Sear 1995, Poff *et al.* 1997). Silts can occupy the interstitial spaces within gravel beds, and this accumulation of fine sediment can have strong negative influences on the available spawning habitat (Sear 1995). Fine sediment can also clog the pores and gills of eggs and fry (Crisp 2000, Jonsson and Jonsson 2011), as well as juveniles pars and adults, where increasing the suspended solids can have lethal and sub-lethal consequences (Lloyd 1987, Servizi and Martens 1992).

The physical dimensions of the habitat are also altered in rivers downstream of impounding reservoirs. Johnson (1994) identified that in six large Nebraskan rivers the reduction in flow led to upwards of 10% reduction in channel width following flow regulation from impounding reservoirs. There is little literature to suggest that width is a crucial element to the life cycle of the brown trout, but the marginal areas of the river, which are key nursery grounds for brown trout (Crisp 2000, Jonsson and Jonsson 2011), may be lost. It is important to consider that the reduction in magnitude from reservoir operations will lead to an *ipso facto* reduction in the depth of the river channel. Cowx *et al.* (2004) identified that depth was a key element that could influence brown trout habitat selection, but whilst newly emergent trout and parr require shallower depths, reservoir regulation may lead to a reduction in the deeper habitat that is preferred by the larger, older individuals of brown trout populations.

The natural temperature regimes in rivers are also subject to influence due to reservoir regulation. For example, temperature regimes below Wimbleball Lake in south west England exhibited substantial deviations from the thermal regime of unregulated rivers in a similar climate (Webb and Walling (1997). The influence reservoirs exert on downstream temperature regimes is complex, for instance water temperature below Wimbleball Lake in summer was not uniformly higher or lower than would occur naturally. The draw-off level (the depth at which water is abstracted from the reservoir for release downstream) can play a significant role in the governance of thermal regimes downstream (Webb and Walling 1997) due to the potential thermal stratification of water within reservoirs. Water at the top of the water column in reservoirs (especially deep ones) during summer would be much warmer than the water taken from below the thermocline. It is possible therefore that surface of the reservoir would be warmer than present in un-regulated rivers and the converse true for beneath the thermocline. Webb and Walling (1997) determined that in the case of Wimbleball Lake, that in warm, dry summers, when more water was released from the reservoir into the downstream river there would be a warming effect. The ability though to release water from differing depths in the water column is in itself not a trait commonly shared amongst reservoirs globally, with most reservoirs having the draw-off point towards the bottom of the dam wall to ensure that abstraction is possible regardless of reservoir conditions (i.e. full or nearing empty). In the River Namoi in Australia the effects of draw-off from below the thermocline in thermally stratified reservoir (Keepit Dam) were that the thermal regime was not only 5°C cooler then in unregulated conditions, but the timings of annual daily maximum peak occurred 3 weeks after it occurred under un-regulated conditions (Preece and Jones 2002).

2.5.3 UK and European legislation

The purpose of reservoir operation has changed over the course of time, with a decreasing reliance on the provision of water in reservoirs to meet the needs of hydro-powered mills as was the case during the late 19th and early 20th centuries. Despite this there has been little change to the way that the reservoirs constructed for these purposes operate. Gustard *et al.* (1987) identified that a majority of reservoirs were (at the time of his investigation) still releasing compensation releases that were set when the reservoir was first commissioned, in some cases over a century previously. With the advancements in understanding of the way that human alterations of flows have impacted on ecological and geo-morphological process of the river systems there has been a shift towards restoring and improving modified ecosystems (Poff *et al.* 1997). Despite the shift in use of water in modern society, water is still a precious and in-demand resource; for example, in the Thames basin in south east England the level of precipitation versus the water demand from the human population leaves a per capita yield of water which is comparable to Ethiopia (Falkenmark and Widstrand 1992, Acreman 2001).

The introduction of the Water Framework Directive (WFD:2000/60/EEC) in 2000 by the European Commission, was aimed at providing a legislative framework from which the issues of sustainable water resource management and the associated ecological consequences could be addressed across the European Union (Acreman 2001). This European framework has been transposed into UK law in The Water Environment (Water Framework Directive) (England and Wales) Regulations 2003 and The Water Environment and Water Services (Scotland) Act 2003. The UK Technical Advisory Group (UKTAG) is an advisory group that makes recommendations to Governmental agencies on how best to implement WFD policies. In the United Kingdom, governmental agencies include DEFRA (Department for the Environment, Food and Rural Affairs) and the EA (Environment Agency), SEPA (Scottish Environment Protection Agency), NRW (Natural Resources Wales) and NIEA (Northern Ireland Environment Agency). Government agencies can work alongside local stakeholders and trusts to implement changes on a local scale. The fundamental goal of the WFD is for the surface water of 27 EU member states (including the UK) achieving a Good Ecological Status by 2027. The deadline for achieving GES was originally set for 2015, however, due to the slow rate at which GES was achieved across the 27-member states, this target was revised to 2027. Waterbodies in the EU can fall into one of five categories: High, Good, Moderate, Poor and Bad. The classification of High Ecological status is the best classification a waterbody can achieve and represents little or no form of human pressure influencing the biological, chemical, or morphological characteristics of the river. Good ecological status therefore represents that human pressures are present in the water body, but the level of deviation of the biological, chemical and hydrological characteristics from what would be expected under pristine natural conditions is slight (Bengtsson *et al.* 2012 seeFigure 2.3). The classifications further range from Moderate to Bad with increasing deviation of the surface water characteristics from natural conditions.

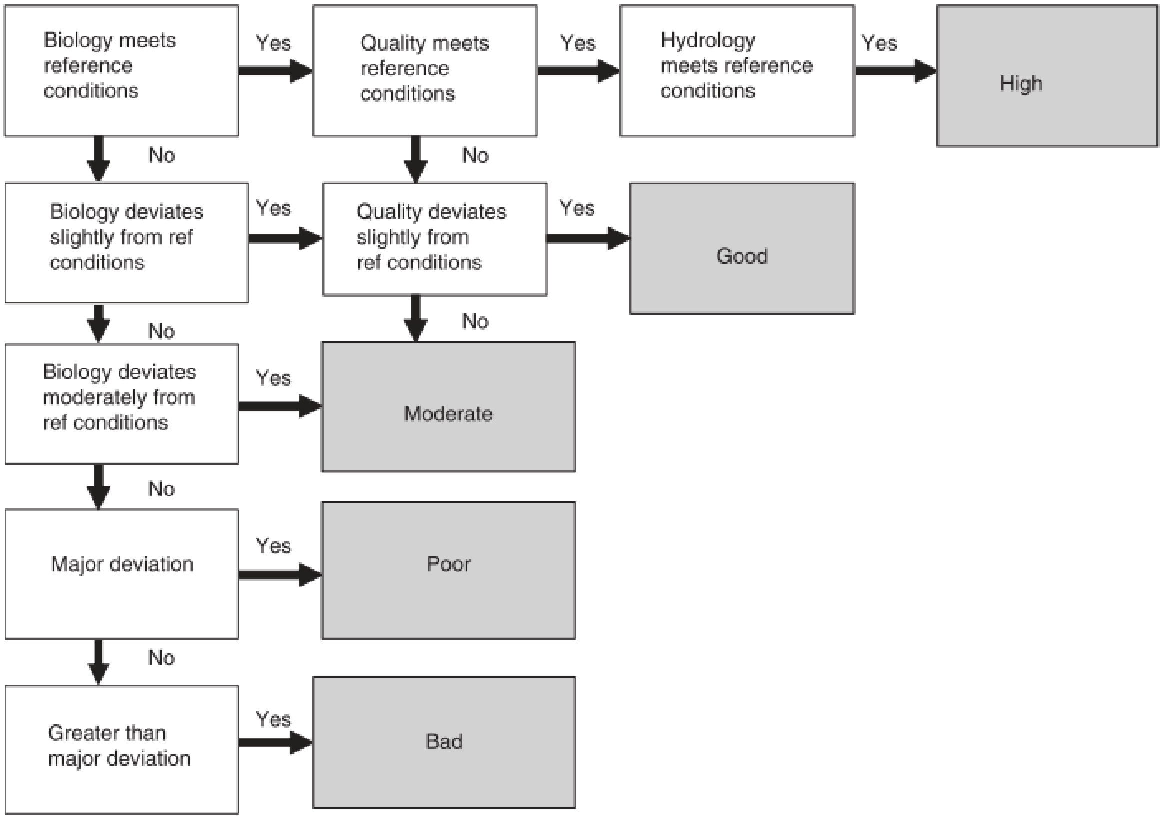

Figure 2.3. Process chart for the classification of the Ecological Status of water bodies (taken from Acreman and Ferguson (2010))

It may not always be possible to achieve GES in many waterbodies without severe consequences for water resource operations and those who rely and benefit from them, for instance flood defence schemes and people who reside in flood risk areas. Water bodies that fulfil these criteria (see Figure 2.4) are designated as Heavily Modified Water Bodies (HMWBs). In England and Wales over 3000 water bodies are now considered to be HMWB due to anthropogenic uses such as abstraction and storage for sanitation and agricultural purposes, as well major modification projects for flood defence purposes (UKTAG 2006).

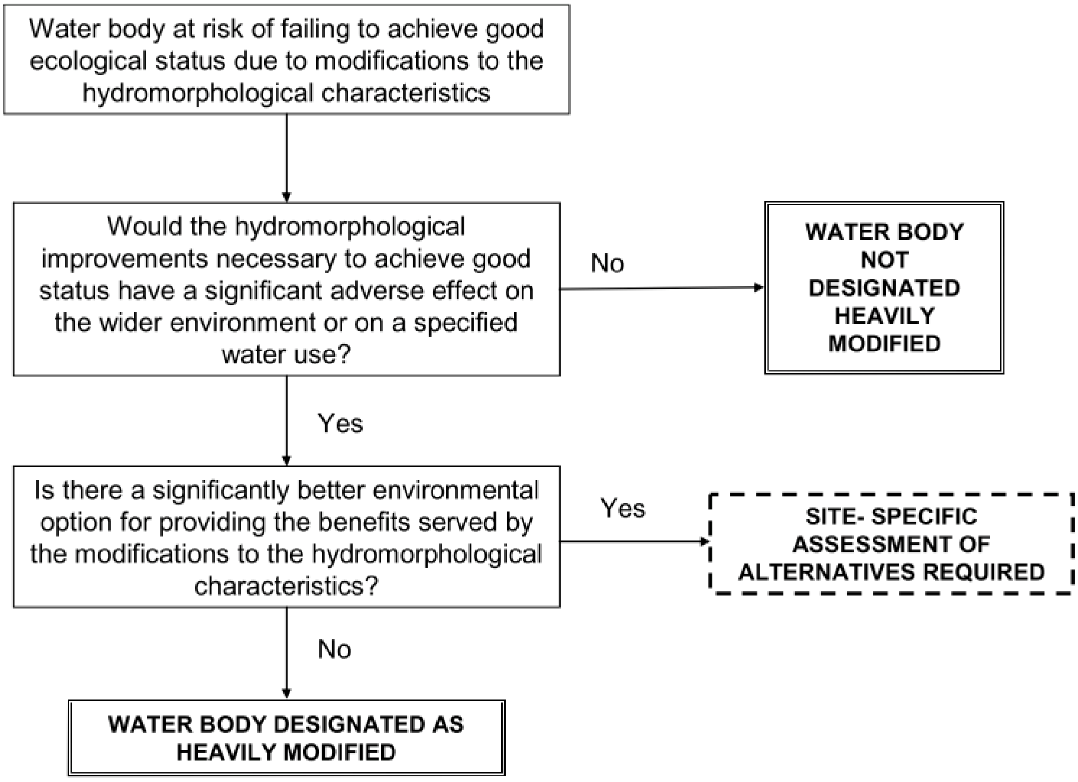

Figure 2.4. Example criteria and process chart for HMWB designation on UK waterbodies (taken from UKTAG (2006))

In these waterbodies where GES would not be achievable, WFD requires for the attainment of Good Ecological Potential (GEP). Like Ecological Status there are several categories of Ecological Potential that HMWBs can achieve: Maximum, Good, Moderate, Poor or Bad. To achieve maximum ecological potential Environmental Quality Standards (EQS) for biological, hydrogeomorphological and physicochemical elements when assessed must represent the best possible conditions that are achievable with appropriate mitigation measures in place. For instance, in the case of water storage reservoirs, MEP (Maximum Ecological Potential) for the hydrogeomorphological elements can be achieved with most appropriate mitigation measures in place (i.e. variable compensation release) that ensures the best approximation to natural ecology (Borja and Elliott 2007). For HMWBs the conditions that would achieve MEP are used as a reference condition against which the other ecological potential categories are judged, with GEP representing only slight deviation from MEP conditions, as with ecological status categories are then ranked from Moderate to Poor with increasing deviation from MEP (Borja and Elliott 2007).

2.6 Mitigation of HMWBs to achieve GEP with specific reference to water storage operations

Ultimately, the only way to return riverine habitats back to their original and natural state is to remove all forms of human pressure on the river system. Indeed with the ageing of dams and the increasing costs of infrastructure investment, as well as advancements in understanding and conservation of water resources, the removal of dams and reservoirs from waterbodies is becoming more prevalent (Poff and Hart 2002). In the USA over 160 dams and reservoirs were removed from water bodies from 1990 – 2000 (Poff and Hart 2002). This methodology is not easily implemented or relevant for HMWBs in the UK, especially as the HMWB designation criteria recognises the importance that the water storage provides both socially (i.e. fishing and boating lakes, as well as flood defence) and economically (revenue generated from water resource, electricity generated from hydroelectricity), as well as cultural heritage. Many of the structures associated with older reservoirs such as spillways, bridges and dam walls have achieved Grade II listed status, a legal framework that requires removal or alteration of improvement works only being allowed in exceptional circumstances (Historic England 2017).

Section 2.6 details the influences that reservoir operation has on the downstream natural flow regime, the homogenisation of flow caused by the absorption of the natural flow regime and the release of compensation flows seriously alter the natural flow regime. As part of the assessment of the hydrology for WFD classification, the flow regime downstream of an impounding reservoir is likely to be severely deviated from natural conditions; this deviation is a major barrier to be overcome to reach GES/GEP. In the Tang River basin in Southern China Yin *et al* (2016) demonstrated advancements in reservoir construction, and telemetry could potentially drive the development of a system by which the regulation of the downstream compensation flow was regulated by flow gauge data collected from upstream rivers. This methodology, however, still requires refinement, but does represent utilising the tools that advancements in engineering practices and telemetry can offer. In many cases in the UK this methodology would not be possible without serious investment in infrastructure surrounding reservoir operation. For instance, many reservoirs have been in operation for well over a century, and in many cases the compensation release (if there is one at all) has not changed since the reservoir was first commissioned (Gustard *et al.* 1987). Therefore, it is likely that the apparatus surrounding water release, such as release valves and control systems, are l as old as the reservoir itself. Indeed, personal communication with water resource managers in the UK revealed that compensation flows at some reservoirs are still adjusted by hand operation of release valves and set by eye with no form of flow gauge on the outflow.

The building block methodology (King *et al.* 2008) for setting compensation flow regimes provides a good balance between providing flow variability whilst working within the limitations of reservoir operation. The building block methodology (BBM) can be simplistically reduced down to the premise that the flow regime can be broken into flow components that biotic and geomorphic processes are reliant on. The most basic assumption in the BBM is that the rate at which water is stored in reservoirs will exceed the rate at which water is abstracted from reservoirs for human use, resulting in spare water (King *et al.* 2008). The BBM is not a one size fits all approach to achieving GEP/GES, but it does provide a framework that requires expert opinion from multiple scientific disciplines such as hydrology, geomorphology, and aquatic ecology, that can be used to attain GEP in HMWBs (Acreman and Ferguson 2010). In the BBM, there are five different flow blocks that can be implemented, with each building block providing ecological and hydrological benefits to the regulated waterbody (Table 2.1 and Figure 2.5). The number, timings and magnitude of the building block releases are all determined on a site-by-site basis using expert guidance

Table 2.1. Building blocks as well as overview of the biotic and hydraulic functions they serve (adapted from Acreman and Ferguson (2010)

Building Block	Purpose
Low Flows	Habitat for juveniles and prevention of invasive species
Maintenance Flows	Stimulates species migration, spawning and dispersal
Freshets	Stimulates species migration, spawning and dispersal
Small Floods	Sort river sediments, connect river and floodplain habitats
Large Floods	Remove undesired species, maintain channel structure and evolution

As part of the design and development phases of compensation flow regimes, it is crucial to understand the limitations of the impounding reservoir in terms of its ability to alter or introduce a compensation release and the volume of water that can be released without impacting on reservoir operations (King *et al.* 2008, Acreman *et al.* 2009, Acreman and Ferguson 2010). For instance, there would be little benefit in introducing a fully naturalised flow regime utilising all building blocks from a reservoir where release valves are operated by hand and outflow gauges do not exist, as there would be reduced capability to fine tune and measure the outflow as well as the expenditure in man hours to constantly re-adjust valves. The same is true when reservoir yield is only marginally higher than the abstraction demands and utilising the full suite of building blocks would severely compromise water resource operations (Acreman 2007). It is therefore important that during the planning phases, limitations of water storage operations are fully understood, and relevant building blocks introduced to maximise ecological benefits.

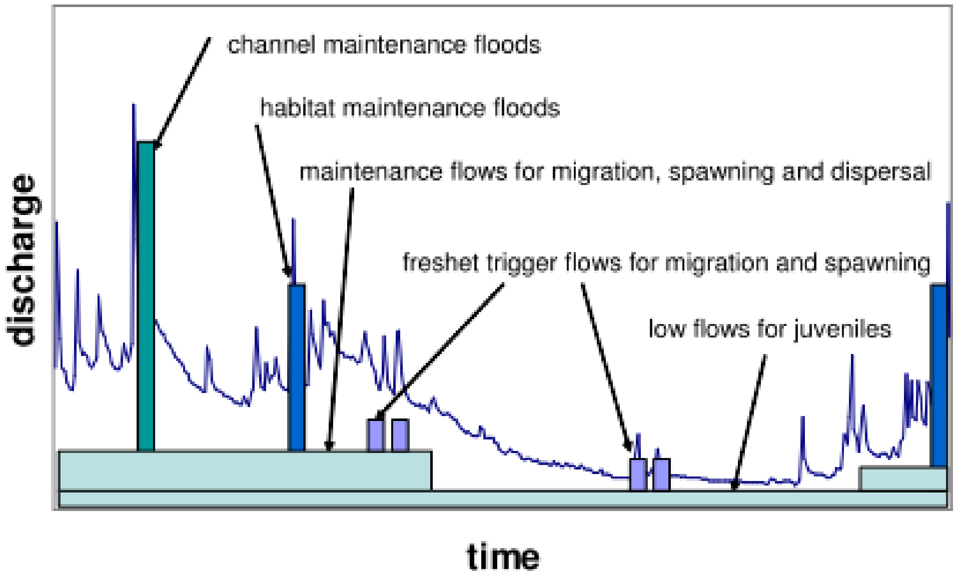

Figure 2.5. Natural flow regime (blue line) alongside the five building blocks and their function (taken from Acreman (2007))

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Whilst the driving purpose behind the implementation of the WFD is how to best improve ecology in water bodies across Europe, the framework of the legislation as well as the implementation at national levels does understand the need for working within the economic confines of water resource operations across Europe. Where disproportionate costs (either operationally or economically) would be incurred by introducing the best possible mitigation measures from impounding reservoirs, it is possible for derogation under article 4(5) of the WFD. In each river management cycle the process of classifying the ecological potential of a HMWB allows for the classification of GES/GEP if introducing any mitigation measure would have significant adverse effects on the wider environment (Figure 2.6).

2.6.1 Physical habitat restoration.

The introduction/amendment of compensation releases from reservoirs in HMWBs is not always achievable, for operational or economic reasons. Despite the processes in place to allow for derogation of legislation in these circumstances, advances in physical restorative measures allow for a viable alternative to achieving GEP in HMWBs. Physically altering the instream habitat as a method of improving the habitat is one method that, with advancements in knowledge base and technology, is becoming a popular tool to improving the riverine ecology (Whiteway *et al.* 2010), with investment in over 6000 instream restoration work projects in the USA alone (Bernhardt 2005). However, this methodology for improving the riverine environment can range from being relatively cheap to being very costly, with individual projects across European rivers requiring investment ranging from tens of thousands to millions of euros (Jähnig *et al.* 2010).

Physically altering the dimensions of the river channel is a practical method for achieving *in situ* changes in the flow characteristics without the need to adjust the outflow of any impounding structure. When the flow rate is constant due to the inflexibility of a release regime from an impounding reservoir, the water velocity can be influenced by changes to the cross-sectional area, in that decreasing the area of the river channel will provide a relative increase to the water velocity under the same flow conditions (Hammond *et al.* 2011)

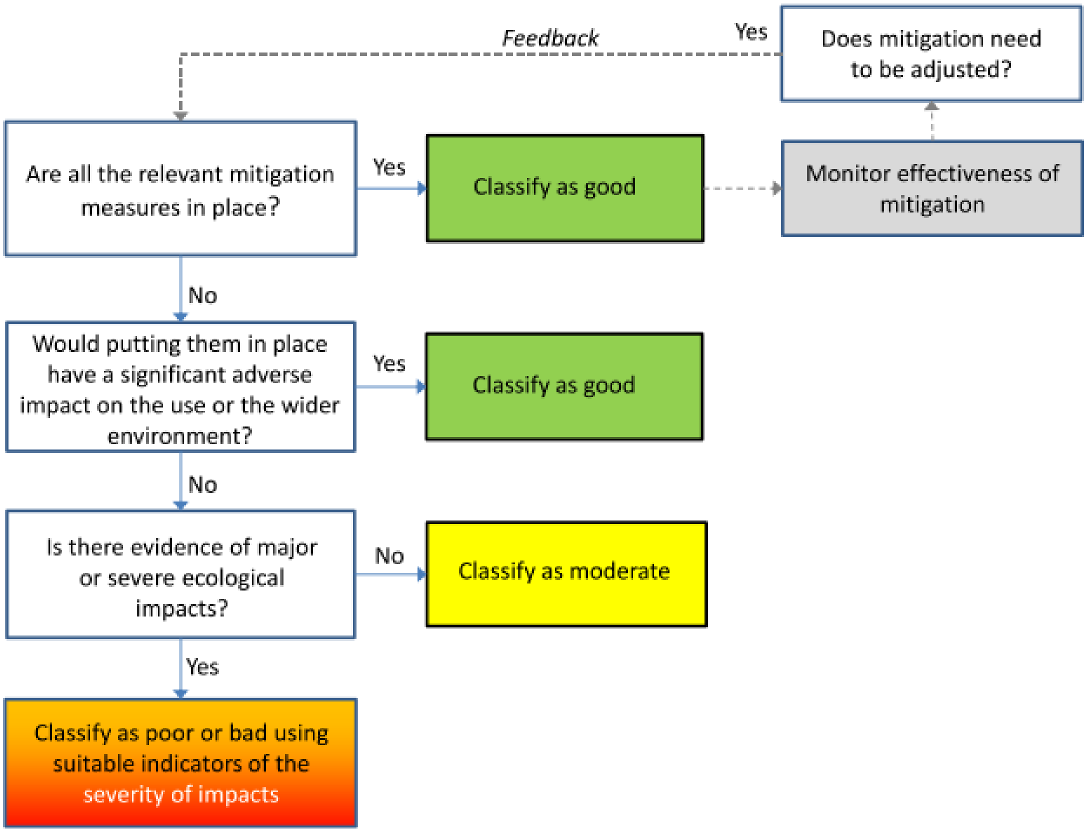

Figure 2.6. Classification of ecological potential of HMWBs (taken from UKTAG (2013))

The alteration of the flow velocity via changes to the physical channel structure can provide a useful mitigation tool by passing the deleterious habitat effects associated with low and stable flow regimes, such as increased sedimentation (Sear 1995) and increased temperature regimes associated with shallower water (Carron and Rajaram 2001). As well as physically altering the channel of the river, other methods such as utilisation of flow deflectors and berms and reintroducing suitable substrate are other methodologies to improve the geomorphology and ecology of HMWBs (Whiteway *et al.* 2010, Hammond *et al.* 2011).

2.6.2 Assessing responses to reservoir mitigation measures

Following the introduction of any HMWB mitigation measure, it is crucial to ensure that there is adequate monitoring of the success of the mitigation measure. Investigating biological responses is a common practice; in a meta-analysis of 111 studies of introduction/amendment of environmental flows by Olden *et al.* (2014), over 90% of studies reported biological responses to the change in flow from reservoir regulated rivers. Of these studies over 35% of the biological metrics tested were fishes. The natural stochasticity of fish – in particular salmonids – can, however, make it difficult to identify the community response to the environmental flow from the background noise (Daufresne *et al.* 2015). For instance, an investigation into the increase of the minimum flow in Douglas Creek, USA revealed that at only a third of monitoring sites did brown trout standing stocks deviate outside of the natural variability, suggesting that the introduction of a minimum flow significantly increased brown trout populations at one site and they significantly decreased at the others (Harris *et al.* 1991). This non-uniform change reveals that individual local pressures, such as cover (Harris *et al.* 1991) or changes to sediment (Daufresne *et al.* 2015), operating at a site to site level may interfere with monitoring of biological response to restoration measures. Conversely unexpected or undesired hydrological or geomorphological changes, such as alteration of the temperature regime, may have knock-on deleterious effects on the fish communities. This was given as a possible influence as to why the introduction of an environmental flow from Lostock and Chichester dams had no appreciable influence on fish communities in New South Wales, Australia. Alongside unintended hydro-geomorphological processes interfering with biological monitoring, large scale-regional climate conditions, such as increased rainfall, or warming can also influence fish assemblages, more so than the introduction or amendment of the environmental flow would (Rolls *et al.* 2010, Daufresne *et al.* 2015).

Physical restoration methods can provide a workable solution to attaining GEP/GES where reservoir operation is inflexible. The effectiveness of river restoration works is mixed, Whiteway *et al.* (2010) revealed that 73% of restoration projects using a variety of restoration measures, such as flow deflectors, large woody debris and boulder placement, have led to significant improvements in salmonid densities. The use of these restoration methods (e.g. woody debris, boulders) is aimed at improving nursery grounds which should promote better juvenile fish densities (Forseth and Harby 2014), but in Whiteway's *et al.* (2010) meta-analysis the biological response from salmonids was greater in the larger (>15 cm) salmonids than in the smaller juveniles. Champoux *et al.* (2003) revealed that the use of flow deflectors increased the relative water velocity, which in turn increased the scour of the river bed resulting in deeper faster flowing water that can provide better habitat for larger more fecund brown trout (Cowx *et al.* 2004), as well as reduce the accumulation of finer sediments within the substrate (Colby 1964).

A fundamental goal of any restoration project (via either physical habitat restoration works or environmental flows) is to provide better riverine ecological conditions for the betterment of surrounding aquatic flora and fauna. It is therefore crucial to achieve this goal that adequate monitoring programmes are established to determine that the desired goals are met. Guidance on the monitoring of mitigation measures by Acreman (2007) provides a hierarchical approach where time and financial assessment are balanced with certainty that the desired outcome of the mitigation measure is achieved. A desktop study requires no fieldwork, and utilises current advances in this field, scientific literature and expert opinion to provide the best possible mitigation measure with very low financial investment. Whilst being fiscally prudent there are downsides to the desk studies in that there is incredibly high uncertainty in the outcomes of the study. For instance, the environmental flows from Chichester and Lostock dams in Australia were found to have no appreciable effect on downstream fisheries (Rolls *et al.* 2010). Whilst this study was not a desk-based approach it highlights the importance of fisheries monitoring.

For physical restoration methods just under a quarter of studies were found to have deleterious effects on fish communities (Whiteway *et al.* 2010), amongst the reasons provided for negative fisheries responses, was poor study design and implementation. It can be argued that without any post-impact monitoring (such as in a desk-based study design) the poor methodologies that were implemented in these circumstances would be reused in further restoration programmes to the detriment of the overall goal of improving the ecological status of rivers. The lack of desired response from fish communities, the outcomes from these studies can still aid in our understanding and help shape the advancements in river restoration to help provide a more informed approach to reservoir mitigation in the future.

Olden *et al*. (2014) analysed flow trials and revealed that 72% of studies did include at least one form of post flow trial biotic or abiotic monitoring. 12% of studies that did include post impact monitoring were limited to less than a year following the change/introduction of the new flow regime. There is some question as to whether short post impact monitoring can provide reliable and robust conclusions. Sabaton *et al.* (2008) studied spatial and temporal variability both within and amongst streams and revealed that any responses to the brown trout populations would likely not be apparent until seven years after the change to the flow regime. Recommendations from a review of salmonid responses to habitat change in the Pacific north west of America suggested that viable and robust conclusions from salmonid monitoring should not be expected within a decade (Bayley 2002), which for these salmonids represents just over two generations. It is therefore crucial to consider that reduced sampling period will negatively interfere with the accuracy of the monitoring. Brown trout recruitment can naturally vary by orders of magnitude across years (Jenkins Jr *et al.* 1999), therefore without reliable temporal data the task of disentangling this "background noise" from any restoration or mitigation induced change becomes difficult.

Any responses to the change of habitat or flow regime following reservoir mitigation measures will not be solely manifested within biological metrics (i.e. population dynamics of fish). The very process of reservoir mitigation, whether flow trial or physical restoration represents a de facto change to the habitat. It is therefore prudent to ensure that physical habitat variables are monitored alongside biotic variables to ensure that the ecological responses as well as how the physical habitat properties that influence ecological variability respond to reservoir mitigation measures (Bayley 2002, Olden *et al.* 2014). Unlike biological responses, habitat responses to restoration measures can be detected fairly quickly; for instance, positive changes to the habitat were detected within two years of habitat restoration work on Lawrence Creek (US) (Champoux *et al.* 2003). In their meta-analysis of 211 stream restoration projects, Whiteway *et al.* (2010) revealed that alongside positive biological responses from salmonids there were also significant changes to the physical habitat with significant increases to pool size, mean depth and large woody debris detected following instream works. Physical habitat elements such as pools and depth are key to brown trout population dynamics (Cowx *et al.* 2004), as well as large woody debris providing key shelter and habitat for juveniles (Roni and Quinn 2001). A primary concern of physical restoration methods, however, is the longevity of the in-stream structures. A study into the rate of failure of physical restoration methods revealed that the size and type of instream structure can greatly influence the risk of failure in Pacific north-western American rivers, noting it is generally considered that the lifespan of instream structures is around 20 years (Frissell and Nawa 1992). This is important to consider when evaluating the longevity and resource allocation of a restoration project, as positive habitat responses can be seen, but the quality of the improvement is likely to deteriorate over time which is likely to have deleterious effects on the ecology of the river. In a study on the long-term effectiveness of restoration practices in Lawrence Creek, substantial deterioration to the habitat quality was detected in survey work undertaken 33 years after the original restoration work. This deterioration of the habitat was largely due to reduced longevity of the restoration structures, either through poor placement and utilisation of different structures. For instance, it is believed that whilst bank flow deflectors are appropriate for narrow sinuous channels, they are not suitable for over wider and irregular stretches of river as seen in the morainic section (Champoux *et al.* 2003). This demonstrates that assessments of previous studies, both the success and failures of which, can provide tremendous insight and help advance the knowledge of habitat restoration measures, to provide more targeted and efficient measures to achieve the goal of GES by 2027.

2.7 Summary

The life cycle of brown trout is intrinsically linked to the natural flow regime of rivers, with the timings of certain behaviours such as migrations and spawning coinciding with periods of elevated flow, to ensure longitudinal connectivity and optimal condition of spawning substrate. Alongside influencing behaviour, the natural flow regime plays an important role in density-independent regulation with research on unregulated river systems identifying a fundamental link between discharge during brown trout emergence and year-class strength (Unfer *et al.* 2011, Lobón-Cerviá 2014). The use of reservoirs to store and control the water resource for industrial, agricultural and societal needs has led to substantial degradation of the natural flow regime in many river systems in the UK. The flow regimes in many rivers in the UK are heavily regulated, with reservoir operation and condition playing an important role in determining the flow regime downstream. In these heavily regulated rivers elucidating the role that impounding reservoirs has on the population dynamics of brown trout is an important step that can enhance and aid in the conservation of brown trout in heavily regulated rivers. Chapter 3 of this research utilises long term population data from heavily regulated rivers in the UK to identify the role of density dependent and independent regulation on brown trout communities. The introduction of the WFD in 2002 across all European Union member states provides a legislative framework that puts the onus on governments and water resource managers to mitigate the deleterious effects that water management has on riverine ecology. The advancements in our understanding of hydrogeomorphological processes as well as brown trout population dynamics as led to the development of a suite of potential mitigation techniques that include flow restoration and physical habitat modification. This research uses four case studies where flow restoration (chapters 4 – 6) and physical habitat modification (chapter 7) to mitigate the deleterious effects that reservoir regulation has on the downstream habitat quality and brown trout populations.

3 DRIVERS OF BROWN TROUT POPULATION DYNAMICS IN HEAVILY MODIFIED WATER BODIES

3.1 Introduction

The construction of dams and weirs over the last 200 years in the UK has been detrimental to riverine ecology (Ligon *et al.* 1995). Reservoirs, constructed over a number of centuries, are typical anthropogenic modifications to the freshwater environment and can have profound influence on the downstream flow regime (Acreman 2007). Historically, reservoirs were mill ponds where water was abstracted from rivers and stored in large ponds for release to provide a rudimentary form of hydropower for mills (Gustard, *et al*, 1987). During the 19th Century and the Industrial Revolution in the UK, the size and number of reservoirs constructed in the UK increased to support the expanding industry and urban populous (Chapter 2.6). Despite industry moving towards electricity as a power source in the early 20th Century, reservoirs and compensation releases are still prevalent in the UK river network, with reservoirs now providing an important role in ensuring adequate supply of water for irrigation and human consumption, as well as managing water supply throughout. Compensation flows and reservoirs are not synonymous, with up to 30% of all UK reservoirs not releasing any routine flow downstream (Gustard *et al.* 1987).

There are a number of impacts that reservoirs have on rivers including changes to hydrology, geomorphology, chemistry and connectivity (Jager and Smith 2008), all of which can impact on the flora and fauna present in river systems further both upstream and downstream. Most notably reservoir operation can severely alter the natural flow regime with periods of high and low flow downstream more dependent on the operational state of the reservoir as opposed to climatic conditions. Periods of high rainfall upstream of the reservoir will only influence the downstream hydrology if the impounding reservoir is full and over-spilling. The same is true of natural low flow events, as an impounding reservoir with a compensation release may increase flows downstream to a higher magnitude than would be expected in an unregulated river. Conversely, in a reservoir without a compensation release, natural low flows downstream may be further exacerbated as flows would comprise solely of natural seepage/gather only from sources downstream of the reservoir, not the entire reach as found in unregulated rivers (Poff and Hart, 2002).

Reservoir operations not only influence downstream flows but also exert influence on the temperature regimes of rivers. Due to their size and depth, water temperatures are generally lower than in surrounding rivers (Carron and Rajaram 2001). Water released from reservoirs can therefore exert influence on the downstream thermal regime (Caissie

2006). Thermal regimes in downstream reaches during the summer period are typically suppressed (Webb and Walling 1993). As well as the suppression of temperature, cold-water releases can also eliminate synchronicity in thermal regimes between local rivers (Preece and Jones 2002).

Any deviation from a natural regime, both in terms of flow and temperature can be detrimental to the ecology of the river (see Chapter 2.6) and reflected in fish species, which are good indicator of environmental change. Brown trout is the most common fish in the upper reaches of Yorkshire Rivers and used in this study because it has low tolerance to sub-optimal environmental conditions, and is a good candidate indicator species by which the success of environmental restoration projects can be determined (UKTAG, 2008)

Brown trout, like most fish taxa, exhibit strong spatial and temporal variability (Elliott 1985, Lobón-Cerviá and Mortensen 2005, Ayllón *et al.* 2012). Understanding the complex interactions that take place in the population dynamics is a fundamental goal of fisheries management. In HMWBs impounded by reservoirs understanding and elucidating the role of density-dependent and density-independent mechanisms can aide in providing targeted management for conservation species.

Early studies into fish population dynamics by Ricker (1954) and Haldane (1956) noted that density dependent factors must exist in populations otherwise density would be able to increase indefinitely, but in many cases where the habitat is considered to be sub-optimal it is the abiotic elements of the habitat that is influencing population regulation. Studies on Black Brows Beck with brown trout in the Lake District, UK (Elliott 1984, 1993) suggested that the primary drivers of recruitment variability are density-dependent factors, as increasing egg density beyond a specific value was shown to negatively impact density of recruits surviving to their first winter. The self-thinning phenomenon (Elliott 1993) is due to only a finite abundance of food resources in any ecosystem meaning that the greater the population the lower the *per capita* share of food and therefore a reduced chance of individual survival and increased mortality. This method of regulation is at its most prevalent in populations with favourable habitats that approach the carrying capacity. Elliott (1984) noted that the abiotic conditions for brown trout fry in Black Brows Beck were favourable with temperature and water velocity never exceeding the upper tolerance limits. This relationship between recruitment and density can be simplistically described as following a curved relationship with reduced recruitment either side of an optimal spawning stock abundance, as described by Ricker (1954). The Ricker stock recruitment curve suggests that in all populations there is an optimal spawning stock size. As previously established in any fish population, an increase in abundance can be typified as a decrease in *per capita* share in both territory and food resources, but there are trade-offs. At lower abundance, there is an increased individual risk to predation as there is reduced protection from shoaling effect (Pitcher 1992), and in terms of lower spawning stock there is a risk of genetic drift and the alee effect when populations are maintained by a smaller genetic pool (Jonsson and Jonsson 2011). This explains the two-phase density-dependent relationship between adult and recruit density in the Ricker model with poor recruitment at both low and high adult densities.

Interactions between conspecifics can also influence the individual growth rate of brown trout (Bohlin *et al.* 2004). Growth is an important biotic factor to brown trout management as individual size and growth is intrinsically linked to survival and spawning size (Jonsson and Jonsson 2011). As for population density, regulation of growth rates of brown trout are driven by density-dependent mechanisms (Vøllestad 2002, Lobón-Cerviá 2007). Individual growth rates of 0+ brown trout have been found to be negatively influenced by increasing adult density (Jenkins Jr *et al.* 1999, Grant and Imre 2005, Richard *et al.* 2015). Limitations to growth arising from density-dependent factors is not just reported in juveniles but can operate at all age classes (Jenkins *et al.* 1999, Bohlin *et al.* 2002). Reduced growth at higher densities occurs due to the reduced per capita share of available food resources and increased energy expenditure on increased territorial interactions (Bohlin *et al.* 2002, Sundström *et al.* 2004).

Density-dependent processes are not the sole governance of trout population dynamics. For brown trout, the flow levels required at different life stages (Cowx *et al.* 2004), as well as the influence flows have on early development (Jensen and Johnsen 1999, Lobón-Cerviá 2004, Lobón-Cerviá and Rincon 2004, Unfer *et al.* 2011, Bret *et al.* 2016) are crucial to explaining temporal variability of the species. The timing of natural hydrological events during the incubation and emergence phases are critical to the recruitment success of the resident population (Lobón-Cerviá 2004, Lobón-Cerviá and Rincon 2004). High flows during the early phases of brown trout development can have a major influence on recruitment success. Studies into brown trout in the regulated Upper Ybbs River in Austria found that excessive velocities during the intra-gravel phase of brown trout were correlated with years of low recruitment (Unfer *et al.* 2011). A singular high flow event during these periods was able increase bed-sediment transport in the spawning gravels significantly enough to increase mortality associated with washout, and mechanical shock. In the Rio Esva drainage in northern Spain, there was a two-phase linear regression with flow during the emergence period of resident brown trout (Lobón-Cerviá 2004); both high and low periods of discharge were associated with lower levels of recruitment. Extreme high flow events are not commonplace in reservoir-regulated rivers due to the nature of water storage, but operational releases from reservoirs i.e. scours and spates as well as over-spilling events can lead to high flow events which may not correlate with the timings of high flows in unregulated river systems. Low flows are more typical of reservoir regulated rivers. From an ecological perspective, low flows during the emergence period can result in reduced wetted river habitat which can lead to higher levels of intraspecific and intercohort competition (Richard *et al.* 2015). During severe dewatering, marginal nursery habitat can become isolated from the main river channel, where conditions can quickly become hostile to juvenile trout as oxygen levels decrease, metabolites accumulate and temperatures can quickly rise beyond brown trout thermal tolerances (Jonsson and Jonsson 2011).

Water temperature is considered to be a crucial environmental factor in the regulation of growth of brown trout with a positive relationship between the two parameters, as long as the temperature remains within the thermal limits of the species (Elliott 1985, Elliott *et al.* 1995, Jonsson and Jonsson 2011). Studies in the Boiron River in Switzerland demonstrated that both temperature and flow rates during the summer were strong determinants of juvenile growth, with increased summer flows having a beneficial impact on brown trout growth (Richard *et al.* 2015). Droughts and extreme low flows negatively influence growth rate as individuals are likely to reside in highly fragmented marginal habitat during these periods, with a reduced level of prey items available and increased intraspecific competition (Vøllestad and Olsen 2008, Teichert *et al.* 2010). Both juvenile brown trout growth and density have been found to influence population dynamics of a population beyond the juvenile stage (Lobón-Cerviá 2009). The future growth rates of brown trout have been found to be heavily influenced by the rate of which growth occurred in the first year (Vincenzi *et al.* 2008), which in turn can influence a variety of important biological factors, i.e. fecundity, maturation, egg size and individual survival can all be attributed to growth of adult trout (Vincenzi et al 2012). Population densities of brown trout are crucial as well; studies on brown trout populations in the Rio Esva system in northern Spain highlighted the importance that recruitment levels played in forecasting the future population density, with there being a strong positive relationship between recruitment and future spawning stock (Lobón-Cerviá 2009). Temperature plays an important role in population dynamics in unregulated rivers. Temperature regimes in reservoir regulated rivers are typically lower and temporally distinct from unregulated rivers (Preece and Jones 2002), thus it is imperative to continued efforts in conservation of fisheries to understand how these processes interact with population dynamics in regulated HMWBs.

This chapter will investigate the drivers of temporal variability in resident brown trout populations in three Yorkshire rivers directly downstream of impounding reservoirs. In all three rivers, the flow regime is highly regulated. Under WFD classification all three rivers are classed as HMWBs where significant human impacts have resulted in the loss of diversity in flow and temperature regimes. Using long-term fisheries, flow and temperature data this chapter aims to elucidate the roles that density-dependent and density –independent processes have on brown trout populations. Understanding what is influencing the variability of the fish populations in these rivers will help aide environmental management in other HMWBs in the region.

3.2 Methods

3.2.1 Study Area

Brown trout populations in the rivers Rivelin, Loxley and Holme, in south Yorkshire were monitored annually in the periods 2002-2009 and 2012-2014. All rivers share the same characteristic of being classified as heavily modified and having at least one impounding reservoir in the upper reaches (Figure 3.1; Figure 3.2). All reservoirs have been in operation for a significant time, with the most recent reservoir, Digley (River Holme), being in operation for over 60 years and the oldest reservoir, Damflask (River Loxley), being operational for 172 years. The purpose of these reservoirs is storage of drinking water for the local communities as well as compensation flows to provide adequate river levels to ensure that water powered textile mills situated downstream could operate. Compensation releases from the reservoirs are still present on all three study rivers despite cessation of mill operation. The compensation releases from the reservoirs situated on the Rivelin and Loxley are a fixed daily release that only changes if levels in the reservoirs drop below their respective control lines. In 2004, the compensation release from the impounding dams on the Rivelin and Holme were amended under the Sustainable Compensation Releases project (Hull International Fisheries Institute 2011). The statutory release from the Rivelin Dams (river Rivelin) was increased from 2.24 ML/d to 10.28 ML/d, the release from the Brownhill and Digley reservoirs (River Holme) decreased from fixed releases of 6.82 ML/d and 6.56 ML/d, respectively, to 5.27 ML d^{-1} and 4.92 ML/d with a spate flow at the original release value in October to December. During this period, the compensation release from Damflask reservoir (River Loxley) remained unchanged at 36.1 ML/d (Table 3.1).

Table 3.1 Compensation release rates in mega-litres per day (ML/d) for all impounding reservoirs for the three study rivers (2002 – 2014)

River		Holme		Rivelin	Loxley
Reservoir		Digley	Brownhill	Rivelin Dams	Damflask
Release 2002-2004	ML/d	6.82	6.56	2.24	36.1
Release 2004 – 2014	ML/d	5.27 + 6.82 spate	4.92 + 6.56 spate	10.28	36.1

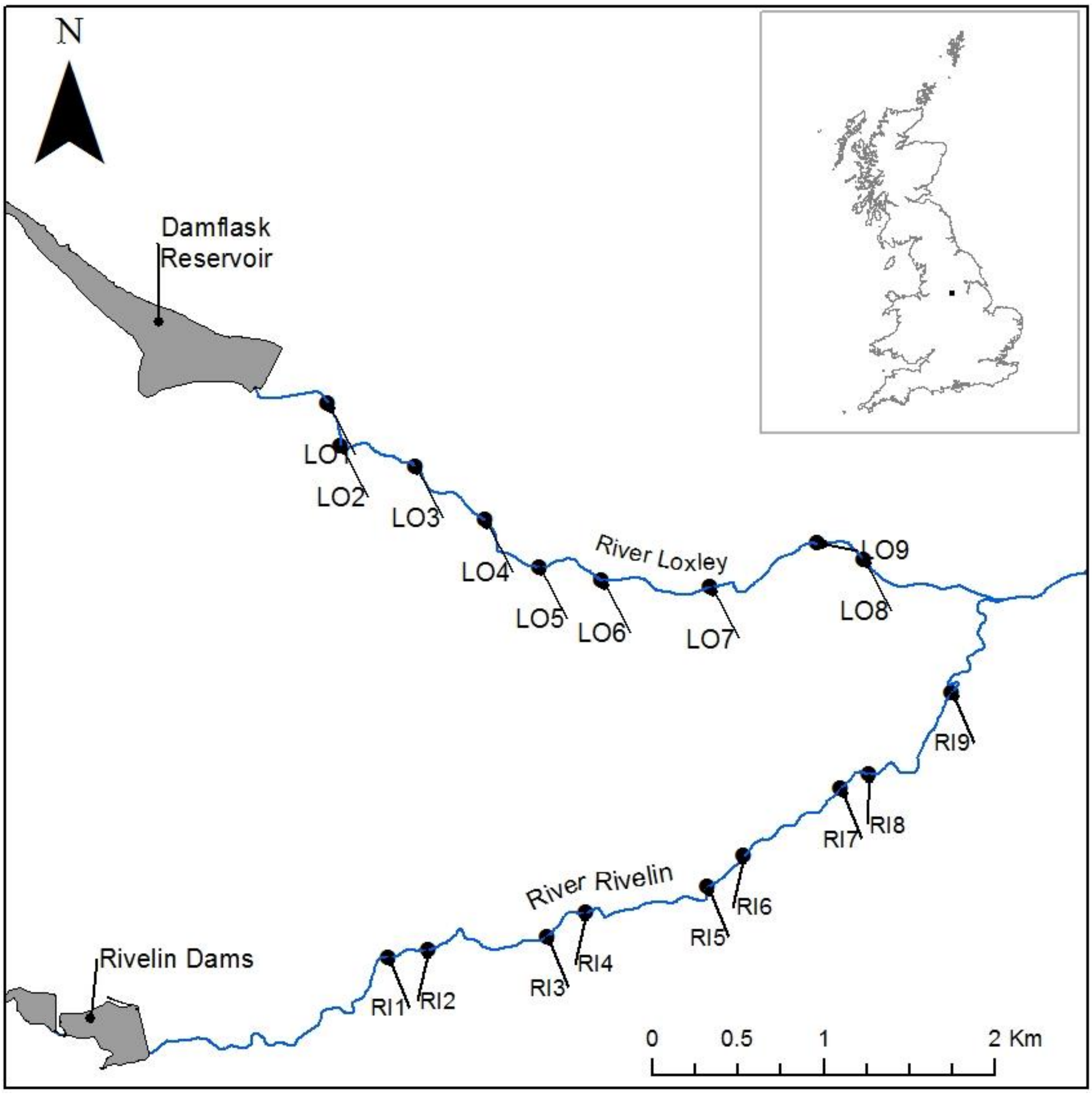

Figure 3.1 Location of survey sites on the River Loxley (LO1-LO9) and River Rivelin (RI1-RI9) in relation to each other and the impounding reservoirs upstream of each river.

The physical characteristics of each of the study rivers are similar in terms of wetted widths and depths. Land-use varies from agriculture and pasture to urban areas; the River Rivelin and River Loxley merge in and flow through the City of Sheffield before finally merging with the River Don. The River Holme flows through several small towns and surrounding countryside before joining the River Colne in the City of Huddersfield. The flow regimes of all of these rivers are highly regulated, as the impounding reservoirs maintain a constant flow in the river systems such that they are typically drawn low by the end of summer, and thus subsequently periods of high precipitation are not reflected in flow elevations in the rivers until the reservoirs refill from the natural gather in the headwaters. The only flow elevations linked to precipitation events are from local run off or over spilling events from the reservoirs.

Under the Environment Agency low flow monitoring programme, nine sites were selected on the rivers Rivelin and Loxley, and six on the River Holme, to represent the environmental heterogeneity of the reach. These sites were originally selected as part of the low flow monitoring programme and used in the Sustainable Compensation Releases (SCR) Project. The SCR Project was developed to look into the biological responses of brown trout to changes in compensation releases from Rivelin Dams, Brownhill and Digley reservoirs. The dataset associated with the SCR project represents a long-term data set to investigate brown trout population dynamics in HMWBs in the Yorkshire region.

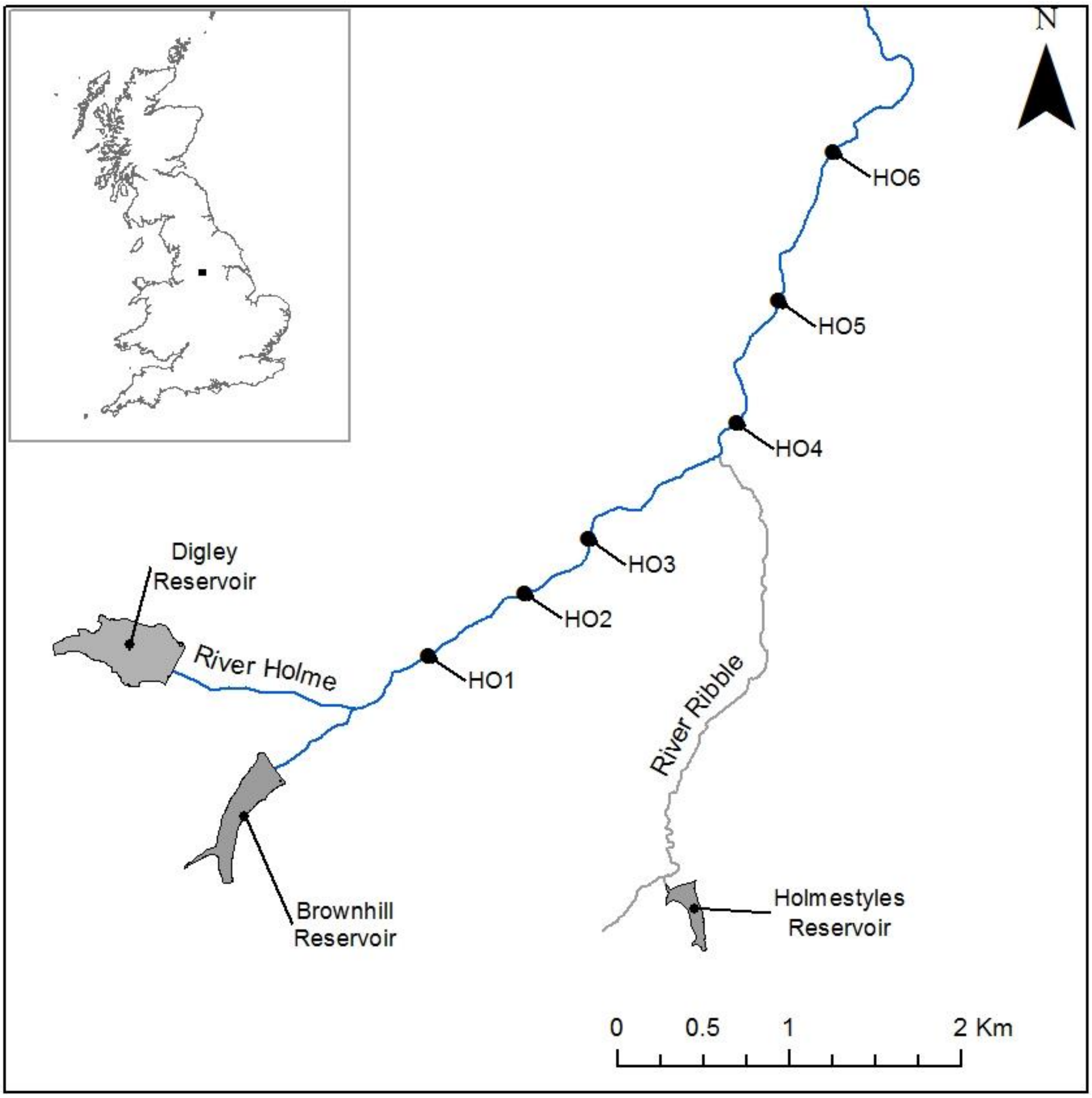

Figure 3.2 Location of the survey sites on the River Holme (HO1 – HO6) in relation to each other and the two impounding reservoirs in the upper reaches of the river Holme.

3.2.2 Fisheries monitoring

Fisheries surveys were undertaken at all sites in all study years (2002-2009, 2012-2014) using a quantitative electric fishing sampling strategy (estimates of absolute abundance based on a three-catch removal method; (Carle and Strub 1978). Sites were selected based on expert judgement by the Environment Agency prior to this study as part of the low flow monitoring programme, with particular attention to the suitability of the sites for monitoring fish populations and being representative of the river in the particular study reach. The quantitative electric fishing strategy 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 Easyfisher (EFU-1) or Electracatch control box producing a 240 V PDC output was employed. The study section was isolated prior to the start of the survey using block nets to prevent the emigration from or immigration into the site during the quantitative assessment. In some cases, barriers such as weirs or waterfalls were used as surrogates for block nets. 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. Following each survey, individual fish were identified to species level, fork length (mm) measured and scale samples removed for ageing purposes (using the appropriate Environment Agency Management System (EA-AMS, (Britton 2003)); the fish were then returned live to the river.

Density estimates of brown trout at each site were derived from estimates of absolute abundance based on the three-catch removal method. Estimates of populations of 0+ and ≥1+ brown trout were calculated by the Maximum Likelihood Method (Carle & Strub 1978). In all cases the population densities were expressed as numbers /100 m^2.

Derivation of density estimates was not possible for minor species as catches in the second and third runs were often greater than the first run, which contradicts one of the main assumptions of depletion sampling that the population is reduced on each sampling run. For example, depletion sampling for bullhead needs to be species specific and intensive because their cryptic behaviour causes them to become immobilised in situ underneath stones and makes them difficult to detect by survey operators, and therefore species-targeted surveys are recommended (Cowx & Harvey 2003).

The determination of the age and growth of fish is an important tool in the assessment of fish population dynamics (Bagenal 1978). The age and growth of brown trout in the study rivers was determined by the interpretation and counting of annual growth checks (annuli) which appear on the scales of the 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. If large numbers of scale samples were collected in surveys, sub-sampling of a representative size range was carried out accordingly to the EA-AMS (Britton 2003).

Scales from each individual 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. The analysis of the data involved assessment of the relationship between the length of the fish, scale radius to annuli and total scale radius (Dahl-Lea method, Francis (1990)):

Equation 1.

$$Li = \frac{SRi}{SRt} \times Lc$$

Where, *Li* is the length (mm) at year 1, *SRi* is scale radius at length *Li*, *Lc* the length at capture and *SRt* the scale radius at capture. For each brown trout, the length at age was calculated from the scale radius to each annuli at each age using equation 1. 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 pooled and displayed graphically for each survey river. To determine the level of spatial and temporal variance, one-way nested ANOVA tests were performed on all pooled length data in all years (2002-2009, 2012-2104) for all three rivers, where individual survey sites were nested within the term river, to ensure that the assumption of homogeny of variances is met the levene's test was ran prior to analysis using in the r package "car" (Fox and Weisberg, 2011). .

3.2.3 Environmental variables

No data exist for the timings of life history events in brown trout in the three study rivers. Using personal observations and UKTAG guidelines (Acreman 2007), three ecologically important timeframes for brown trout were determined for the purpose of this study – the incubation, hatching/emergence and summer periods, the Incubation period was defined as 1st January – 31st March, the Emergence period as 1st April – 31st May, and the summer period was defined as 1st June to 31st September. Each period was selected to encompass a critical life stage of the juvenile trout in the study rivers. Unfortunately, there are no data relating to the timings of spawning or emergence in the specific study rivers, therefore the timeframes chosen to encompass the incubation and emergence life stages were approximated from degree day sums (product of daily water temperature > 0°C for the entire period). In one study, 50% of emergence occurred at an average of 852°C days from spawning with water temperatures ranging from 3.9-6.7°C (Elliott 1984). Water temperature profiles for the study rivers were in a similar range and based on an assumed spawning period of late December, this degree day sum would put emergence typically in April and May each year. This assumption of emergence timing was reinforced from personal observations as post emergent brown trout were observed in late April to early May in another heavily modified water body in; the River Washburn in Yorkshire. The summer period was chosen to encompass the period of the year when abiotic factors are at their most optimum for growth (Elliott 1985, Neophitou 1986, Elliott *et al.* 1995).

Flow rates (m^3/s) for each river were recorded every 15 minutes from Environment Agency gauging stations at Hollins Bridge (River Rivelin), Rowell Lane (River Loxley) and Queens Mill (River Holme). From these data, the mean, maximum minimum, Q95 (5 percentile flow) and Q5 (95 percentile flow) values were determined for each period (incubation, emergence and summer) for all study years (2002-2009, 2012-2014) using the base statistics package in R (R Development Core Team 2016). The flow regimes for each of these rivers differed from each other significantly (ANOVA, $F_{(2)} = 1316$, P = <0.0001), therefore all flow data were standardised (mean centred) to allow for spatial and temporal comparison of flow between rivers. Flow duration curves were constructed using the "hydroTSM" package in R (Zambrano-Bigiarini 2014) and allowed for a visual comparison of the flow regime of the study rivers.

Water temperature data (°C) were recorded at 15-minute intervals from a temperature logger located in each of the study reaches. These loggers were installed and maintained by Yorkshire Water Services Ltd. From these data, it was possible to determine the mean, maximum and minimum daily water temperature ranges for each study reach. Due to operational constraints, the loggers were only in the river for a period of 2 years (2012-2014) and therefore only provided accurate data from a small duration of the study, but, due to the strong relationship between water and air temperature (Johnson *et al.* 2014) back calculation of water temperature using air temperature was possible. To ascertain the water temperature for the missing years (2002-2009), water temperature data were back calculated using an ordinary least squares regression with daily mean air temperature data provided by the UKCP09 climate database (Defra, 2017). All correlation coefficients between air temperature and river temperature for each river were significant (P = <0.01) but coefficients were lower than reported in other studies (Richard *et al* 2015) with r^2 values being 0.83, 0.63 and 0.66, respectively, in the rivers Rivelin, Loxley and Holme. These lower values were to be expected due the constant compensation release from the impounding reservoirs; during the summer water stored in reservoirs is cooler than surrounding water bodies due to the attenuation of solar heating at greater depths (Carron & Rajaram 2001).

Rainfall data were obtained from the Centre for Ecology and Hydrology (CEH) gridded Estimates of Area Rainfall dataset (CEH-GEAR) (Tanguy *et al.* 2016). An ordinary least squares regression model was constructed to test for the relationship between total monthly discharge days (cumulative discharge per month (m^3/s)), and total monthly rainfall (cumulative rainfall per month (mm) for the period of (2002-2014). The purpose of this was to investigate any relationship between rainfall and discharge in the three HMWBs

3.2.4 Density dependent factors influencing growth and abundance.

To determine the relationship between adult brown trout abundance and the following year's juvenile abundance, a simple linear regression model was constructed for juvenile abundance at year *t* and ≥1+ brown trout abundance in year *t*-1. This analysis was run separately on each of the three study reaches as it is not expected that density dependence operates uniformly across rivers. In the study period, this allowed the analysis of 63 cohorts (groups from the same age class per site) for each of the Rivelin and Loxley (nine sites per reach x seven years) and 42 cohorts for the Holme (six sites x seven years). 0+ and ≥1+ brown trout densities were also tested for their relationship with the first-year growth brown trout. In both cases a simple linear regression model was constructed of length (mm) at the end of year 1 against 0+ and ≥1+ brown trout densities.

3.2.5 Density independent factors influencing growth and abundance.

Density dependence is believed to always be operating in some level on the regulation of juvenile density, adult density and growth of juvenile brown trout, but it is not likely to be the exclusive driver of recruitment variance on populations in the study reaches (Lobón-Cerviá 2014, Richard *et al.* 2015). Thus, in addition to, density-dependant regulation, there are numerous explanatory variables that could influence abundance and first year growth, such as flow rates and temperature (Richard *et al.* 2015, Bergerot and Cattaneo 2016), therefore multivariate regression was chosen to identify any relationship between multiple elements of the environment and habitat to brown trout population dynamics.

Linear Mixed Effect models (LME) were chosen for this statistical analysis to test for the respective effects of environmental variables on 0+ and ≥1+ brown trout densities, as well as back calculated first year growth. The LME approach allows for the inclusion of random effects, including River and Years as random effects and allows for the LME model to account for the stochastic variability between rivers and years without which would lead to underestimations in the final model. Normalised Principal Component Analyses (PCA) were performed separately on the flow variables to allow for the synthesis of a large number of input variables on a small number of independent components. Mean, minimum, maximum Q5 and Q95 flow values (see section 3.2.4) were incorporated in the PCA analysis. Degree days were determined as the sum of average water temperatures for each period (incubation, emergence or summer) each year and separately for each river and were standardised (mean centred) to allow for spatial comparison. Analyses were run independently for the incubation, hatching and summer periods to allow for comparison between these periods. All PCA analyses were performed using the Base statistics package in R (R Core Team (2016)). For all periods, Principal Component axes 1 and 2 were selected as explanatory variables for flow, and degree days were selected as temperature variables for the incubation emergence and summer periods. Cumulative portion of variance explained by principal component axes 1 and 2 were >90% in all instances and was therefore suitable for use in analysis.

Linear Mixed effects models were constructed to test for the effects of the new principal component axis on the 0+ densities across the three study rivers, the models were built in R (R Core Team, 2016) using the package lme4 (Bates *et al*, 2015). Following the top down model selection (Zuur *et al* 2009), a beyond optimal model was constructed by including all the fixed and random effects listed in Table 3.2, using the equation 2.

Equation 2

$$M \sim Fe1 + Fe2 \ldots + (1|Re1) + (1|Re2)$$

Where M is the metric to be tested, i.e. $\log_e$(0+ density +1) or length (mm). The prefix Fe denotes Fixed Effects and Re denotes Random effects (Table 3.2).

Due to their being only one flow gauging station on each river and the 5-km spatial resolution of the UKCP09 dataset, only one overall relationship between flow and temperature existed for each river. 0+ densities from each of the study sites were then first averaged to give a mean juvenile density score for each river to avoid multicollinearity arising from the same flow and temperature values applied for individual sites. To account for the variability as well as zero values in density estimates for 0+ brown trout, these data were $\log_e$ +1 transformed prior to analysis. Furthermore, to allow for comparison of variability between them, the fixed effects were all standardised (mean centred) prior to inclusion in the model. The Beyond optimal was constructed using all nine fixed effects (Table 3.2) and then reduced iteratively by removing the fixed effect with the highest P value from a Chi squared analysis of deviance test performed on the model. This removal process was continued until only significant fixed effects remained in the final model or all insignificant fixed effects removed. The final model was finally tested using the restricted maximum likelihood estimation. The significance of each explanatory variable was tested using Wald Z tests. The assumptions of Linearity, Homogeny of variance and Normality of residuals were confirmed using visual examination of: residuals vs predictor plots (linearity), residuals vs fitted (homogeny of variance) and q-q plots (normality of variance)

Table 3.2 List and description of dependent variables, fixed effects and random effects used to create growth and density models

Dependent Variable (M)	**Fixed effects, notation and description**		**Random Effects description**
Log_e(Juvenile Density +1)	**IF1**	*Incubation flow PCA axis 1*	River
Log_e(Length at end of Year 1 +1)	**IF2**	*Incubation flow PCA axis 1*	Year
	EF1	*Emergence flow PCA Axis 1*	
	EF2	*Emergence flow PCA Axis 1*	
	SF1	*Summer flow PCA Axis 1*	
	SF2	*Summer flow PCA Axis 1*	
	IT	*Incubation degree days*	
	ET	*Emergence degree days*	
	ST	*Summer degree days*	

3.3 Results

3.3.1 River flow rates and temperature

The difference in the flow regimes for the study rivers is apparent on inspection of the flow duration curves (Figure 3.3). The River Loxley does not follow the sigmoidal curve displayed in the Rivelin and Holme. The low gradient seen between Q25 and Q90 shows that there is little variation in the flow for 65% of the time and is indicative of rivers where the compensation release makes up a large portion of the overall flow rate. The steepness of the gradient for the River Rivelin between Q90 and Q100 suggests that there is a large variation in the magnitude at low flow events. There is a disparity between the recorded flow rates in each of the three rivers. The rate of flow in the River Holme is greater than that of the River Loxley despite the Loxley having a compensation flow three times the magnitude. This disparity reflects the varying levels of natural input (i.e. seepage, natural gather and tributaries) in each of the three rivers.

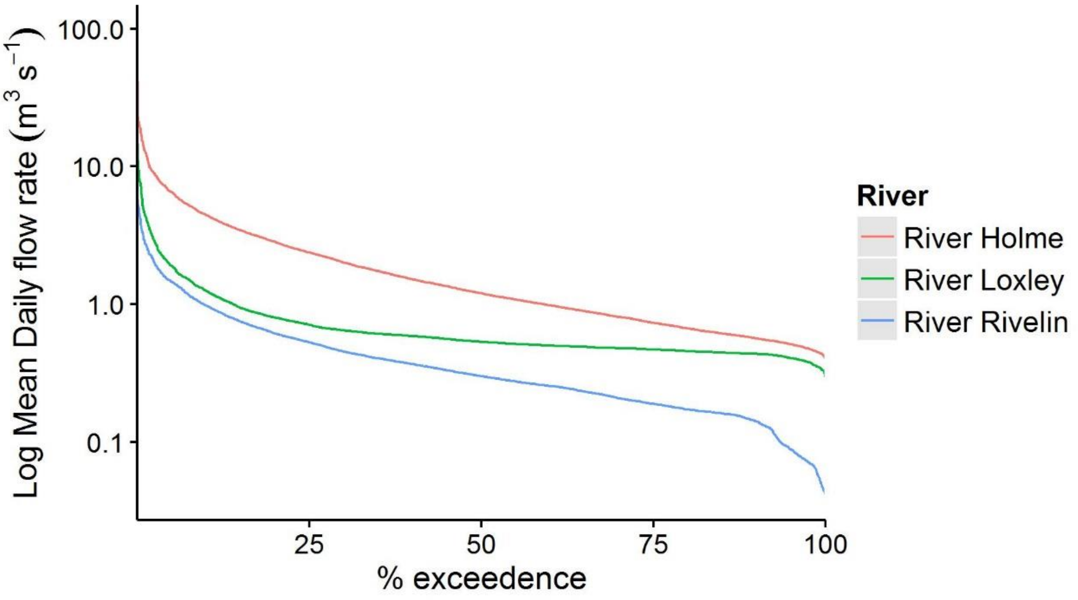

Figure 3.3 Flow duration curves for the Rivers: Rivelin, Loxley and Holme for the study period (2002-2009, 2012-2014)

Coefficients for the relationship between air and water temperature (Figure 3.4) displayed a high level of variance associated with high water temperatures in both the rivers Holme and Loxley typifying the relationship found in HMWBs; this variance reduced the predictive power of the model. All relationships are significant ($P < 0.001$). From this model water temperature for the missing years (2002-2009) were back calculated.

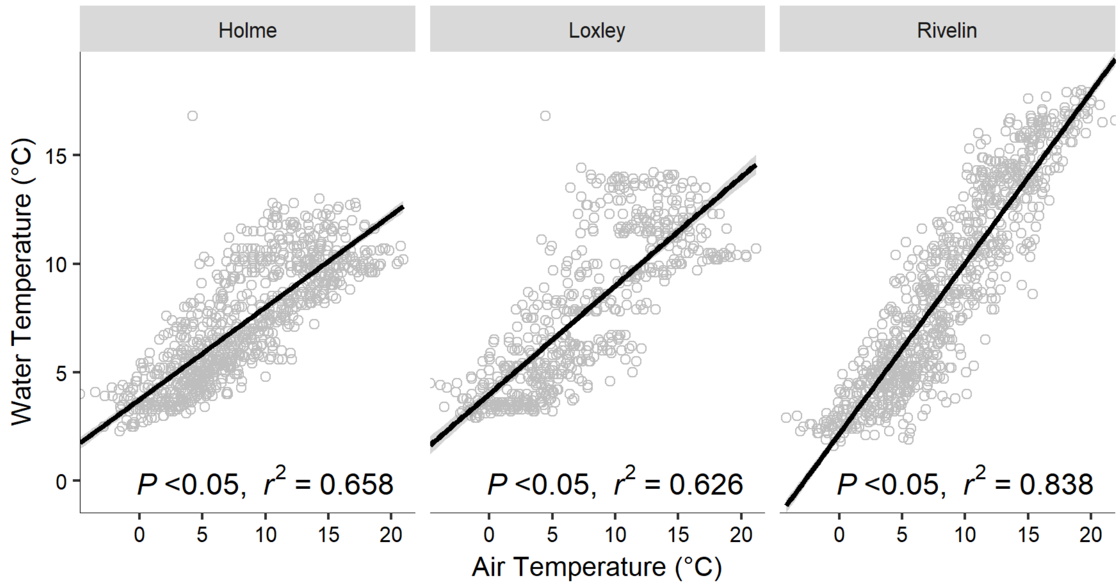

Figure 3.4 linear relationship (black line) between air temperature and water temperature for the rivers Holme, Loxley and Rivelin for the period of 2012-2014. Greyed bands represent 95% confidence limits. Please not that confidence limits for River Holme and River Rivelin are narrower than the trend line.

No relationship was found between precipitation (monthly rainfall (mm)) and flow levels (total monthly discharge (Discharge days (m^3/s). All three relationships were non-significant ($P > 0.05$) with the associated levels of explained variance extremely low ($r^2 < 0.01$) (Table 3.3).

Table 3.3 Correlation between total monthly discharge and total monthly rainfall in the River Holme, River Loxley and River Rivelin for the period of 2002-2015

River	r^2	P
River Holme	<0.001	0.862
River Loxley	0.005	0.385
River Rivelin	0.003	0.430

3.3.2 Density-dependent regulation of density and growth in brown trout

Mean densities of 0+ and ≥1+ brown trout varied both spatially (between rivers) and temporally (between years) throughout the study (Figure 3.5; Figure 3.6) A nested ANOVA (Years within Rivers) was constructed for both 0+ and ≥1+ brown trout. There was significant effect of both River ($F_{2,231} = 19.00$, $P < 0.001$) and Years ($F_{20,231} = 6.67$, $P < 0.001$) on 0+ brown trout throughout the study. The was a significant difference between ≥1+ brown trout population densities between the rivers ($F_{2,231} = 11.03$, $P < 0.01$), but not between years ($F_{20,231} = 1.08$, $P > 0.05$). Some variation in 0+ brown trout densities were found between sites in the study rivers ($F_{23,240} = 2.842$, $P < 0.001$) as well as ≥1+ densities ($F_{23,240} = 14.18$, $P < 0.01$).

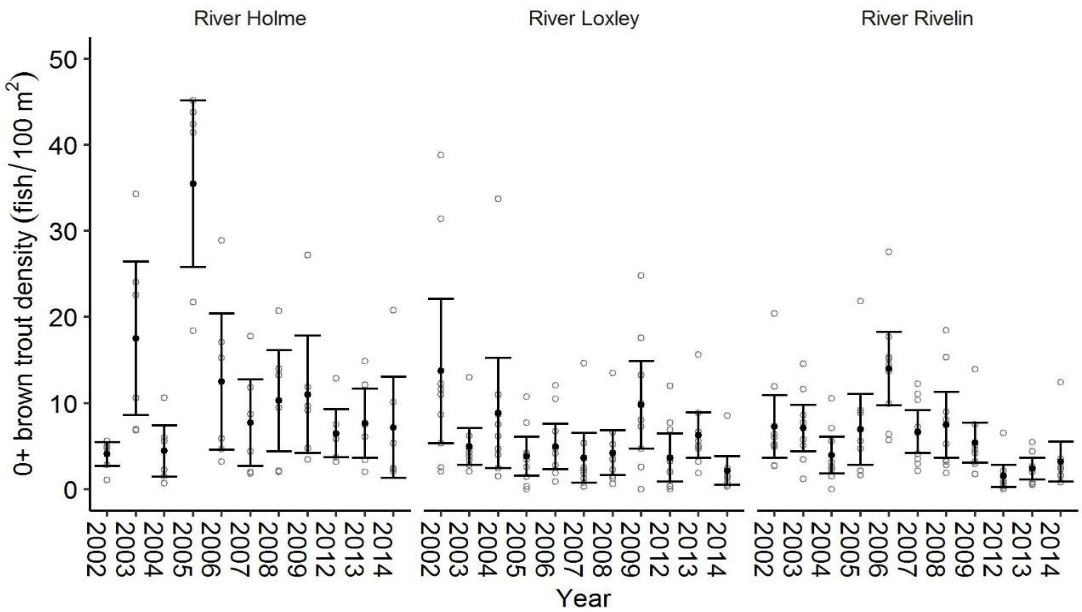

Figure 3.5 Mean 0+ brown trout density ± 95% C.L, for the rivers Holme, Loxley and

Rivelin for the study period (2002-2009, 2012-2014).

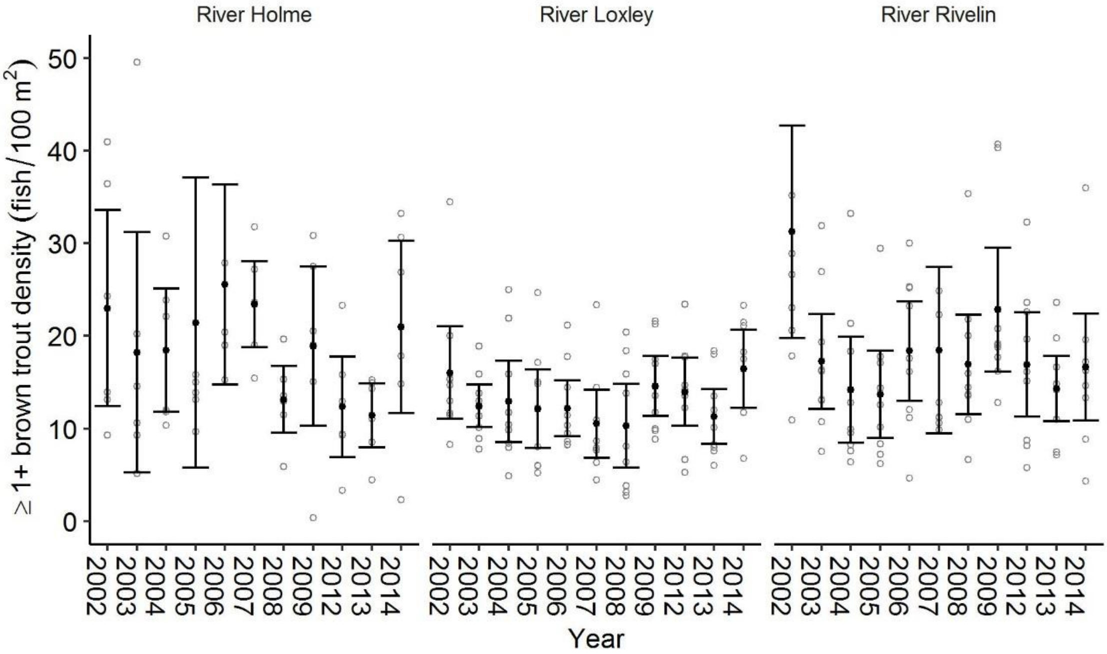

Figure 3.6 Mean ≥1+ brown trout density ± 95% C.L, for the Rivers Holme, Loxley and Rivelin for the study period (2002-2009, 2012-2014).

Length of brown trout at the end of first year (first year growth) varied both amongst year classes ($F_{10,20}$ = 3.24, P <0.05) and between rivers ($F_{2,28}$ = 4.64, P < 0.05) Figure 3.7), with the overall mean brown trout length in the rivers Holme, Loxley and Rivelin at age one being 71 ± 1.5 mm, 69 ± 1.2 mm and 65 ± 1.2 mm, respectively. In 2008, first year growth of brown trout was the lowest recorded in the rivers Rivelin and Loxley, while the smallest average length at age 1 was in the River Holme in 2009.

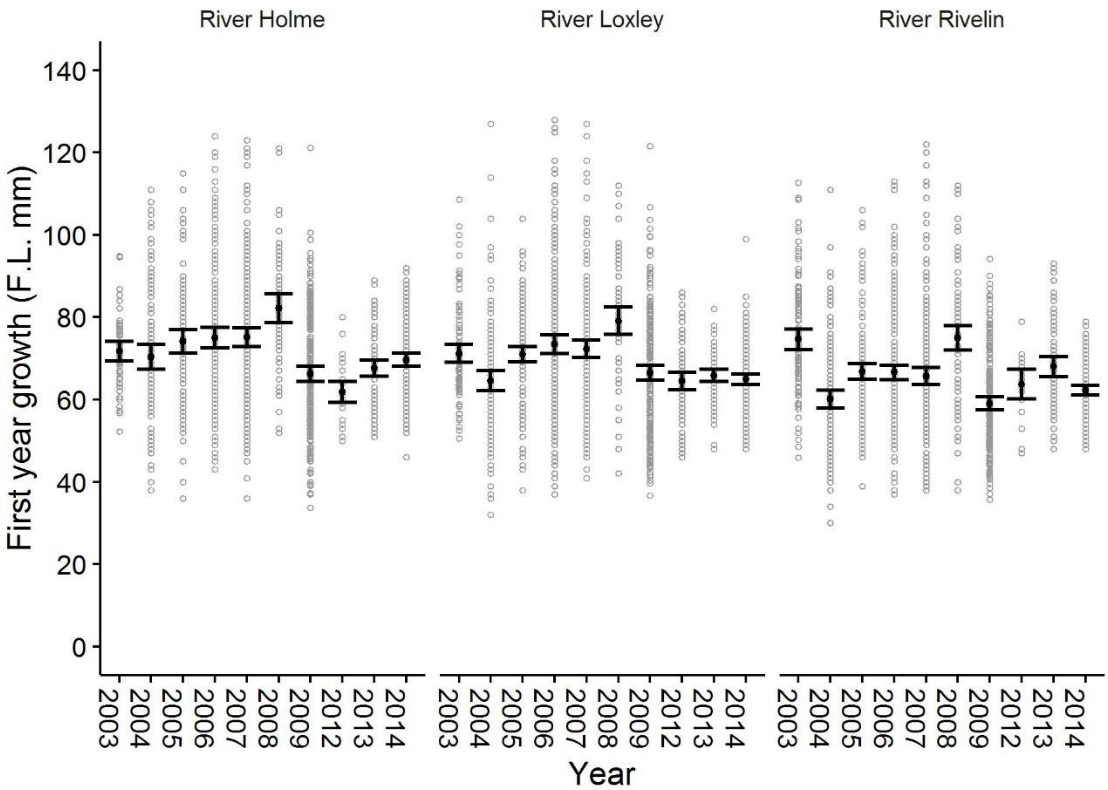

Figure 3.7 Mean length at end of first year ± 95% C.L for the Rivers Holme, Loxley and Rivelin for the study period (2002-2009, 2012-2014).

The role of density dependent regulation of population size on the study rivers varied. There were very weak positive relationships between ≥1+ densities the preceding year and 0+ densities in the Rivers Holme and Rivelin ((Table 3.4); P < 0.05) but the total variance explained was low at 9.2 % and 7.1% for the Holme and Rivelin, respectively. A significant relationship between 0+ and ≥1+ brown trout the preceding year was not found in the River Loxley (Table 3.4)

Table 3.4 Relationships between densities of ≥1+ brown trout densities in the preceding year and 0+ brown trout densities for the study period (20012 – 2009, 2012 – 2014) **Bold** denotes significant interactions (P<0.05)

Predictor variable	River Holme		River Loxley		River Rivelin	
	P	r^2	P	r^2	P	r^2
≥1+ brown trout densities at time t^{-1}	**0.025**	**0.092**	0.111	0.031	**0.016**	**0.071**

There was a significant positive relationship between 0+ densities and ≥1+ populations in the following year in all three rivers (P <0.05, Table 3.5). The total variance explained by these relationships though was very low 8.5 - 16.2% (Table 3.5). Suggesting that 0+

brown trout densities can influence the following year's ≥1+ densities, however the very low predictive power of this model suggests that other potential variables such as abiotic conditions, as well as dispersal and migration could be driving this relationship.

Table 3.5 Significance and predictive power of the influence of 0+ brown trout densities on ≥1+ brown trout densities at time t^{+1} for each of the study rivers. **Bold** denotes significant interaction (P<0.05)

Predictor variable	River Holme		River Loxley		River Rivelin	
	P	r^2	P	r^2	P	r^2
0+ brown trout densities	**0.001**	**0.162**	**0.008**	**0.085**	**0.002**	**0.110**

Relationships between density dependent processes were not found to be consistently influencing first year growth of brown trout across the three study rivers. All three predictor variables (0+ brown trout density, ≥1+ brown trout density and ≥1+ brown trout density at t^{-1}) were significantly correlated with first year growth in the River Holme, but only ≥1+ brown trout density at t was significantly correlated with first year growth in the rivers Loxley and Rivelin (Table 3.6). Despite the significance of the each of the models, the total variance explained was low <26% therefore, no meaningful density dependent regulation is in operation on first year brown trout growth in the three study rivers.

Table 3.6 Significance and R^2 values for linear regression models to determine role for density dependence in variability of first year growth. **Bold** denotes significant interaction (P < 0.05)

Predictor variable	River Holme		River Loxley		River Rivelin	
	P	r^2	P	r^2	P	r^2
0+ density	**0.013**	**0.10**	0.960	0.00	0.177	0.02
≥1+ density at t	**<0.001**	**0.26**	**0.006**	**0.08**	**0.001**	**0.11**
≥1+ density at t^{-1}	**<0.001**	**0.22**	0.657	0.00	0.01	0.04

3.3.3 Density independent regulation of density and growth in brown trout.

Principal Component Analysis (PCA) of flow variables

Principal component analysis was performed on the flow rate data for each of the three critical life stages of brown trout (incubation, emergence and summer) for each year of the study (2002-2009, 2012-2014) (Figure 3.8). For all three PCA models the variance explained by components 1 and 2 were >90%. For the incubation period and summer, all flow variables were negatively correlated with the first axis (IF1, SF1) in each of the rivers. For Incubation and summer periods the all variables were negatively correlated with the first PCA axis (Table 3.7), therefore higher values on the synthetic IF1 and SF1 represented years with lower flow rates, the converse is true for the emergence period and EF1. For the second axis, Minimum and Q95 values were negatively correlated, whilst Mean, Max and Q5 were positively correlated (Table 3.7). Higher values on the second PCA axis for all periods (IF2, EF2 and SF2) therefore represented periods with high flow rate variability (i.e. High maximum flow rates, but low minimum flow rates) (Figure 3.8).

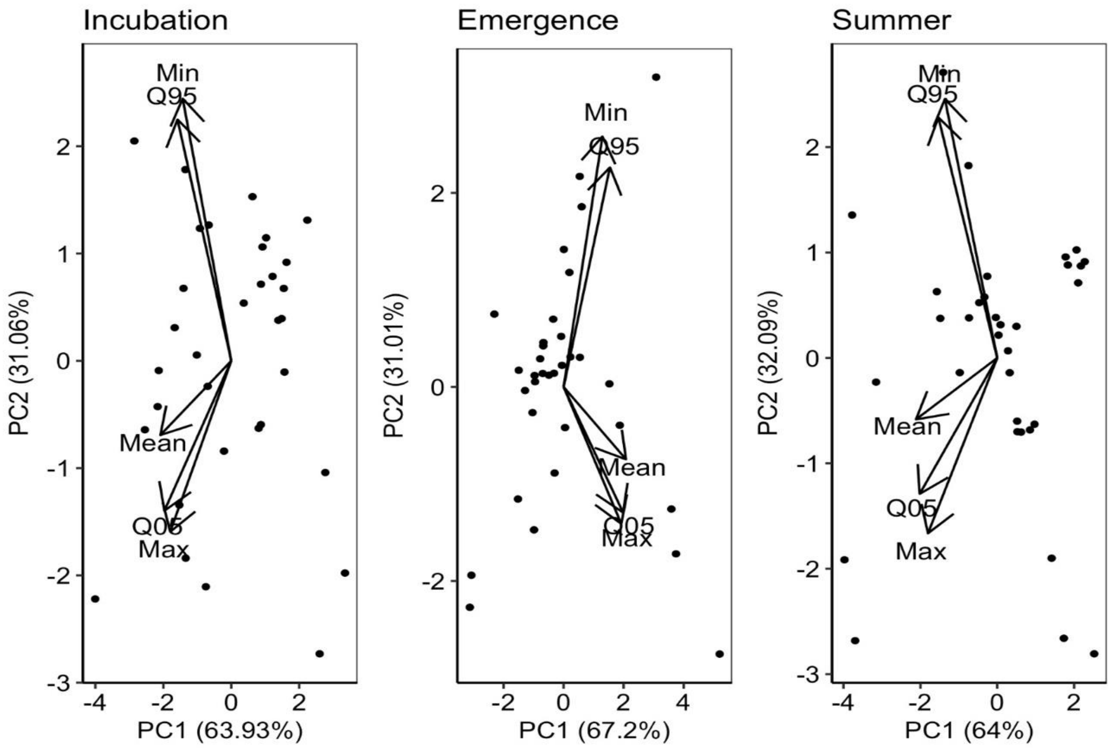

Figure 3.8 Principal Component Analysis of five environmental variables for each of the three ecologically important periods.

Table 3.7 Loadings and variance explained by principal component axis 1 and 2 for each of the three ecologically important periods.

	Incubation		Emergence		Summer	
Flow Variable (%Variance explained)	IF1 (63.9%)	IF2 (31.1%)	EF1 (67.2%)	EF2 (31.0%)	SF1 (64.0%)	SF2 (32.1%)
Q95	-0.40	0.56	0.39	0.57	-0.38	0.57
Q5	-0.49	-0.35	0.50	-0.32	-0.50	-0.32
Mean	-0.52	-0.17	0.52	-0.19	-0.53	-0.14
Maximum	-0.45	-0.39	0.48	-0.39	-0.45	-0.42
Minimum	-0.36	0.61	0.32	0.61	-0.34	0.61

Linear Mixed Effects Models into density independent regulation of 0+ density and first year growth

For both investigations into first year growth and 0+ density in brown trout, beyond optimal linear mixed effects models were constructed. In the case of the 0+ brown trout population density model all six synthetic axes from the PCA analysis, as well as degree days above zero for each period, were included in the model as fixed effects and Rivers and Years were included as random effects. For the first-year growth model the fixed effects for the period of incubation were removed prior to the analysis to reduce model complexity, as abiotic factors during incubation are not thought to influence first year growth.

Following the top-down model selection strategy (Zuur et al. 2014), non-significant variables were iteratively removed using a chi squared analysis of deviance test with a null-model. The order of removal and associated P values for non-significant explanatory variables is found in Table 3.8. For both models (density and length at age one), the top down model selection protocol yielded one significant explanatory variable. 0+ brown trout densities were positively correlated with the synthetic axis SF1 ($P < 0.001$) and thus related to flow variables during the summer period (Table 3.7) suggesting increased 0+ brown trout density in years with lower overall flow rates during the summer period (Figure 3.9). The total variance explained by the model was 38.3% ($R^{2fe} = 0.260$, $R^{2fm} = 0.383$) with 20% of this variance explained by the interaction of SF1 and juvenile density; the remaining 18% was explained by the random effects (Years and Rivers).

One variable remained for the growth model following the top down selection protocol (Table 3.8), the synthetic axis (EF1) derived from the first principal component of flow rate variables during the emergence period. The relationship between EF1 and mean length at age 1 was negatively correlated (Figure 3.9), indicating reduced length at age one in years with elevated levels of flow rate during the emergence period. The total amount of variance explained by the model being 87.6% ($R^{2fe} = 0.095$, $R^{2fm} = 0.876$) but only 8% of this variance could be explained by the fixed effect of EF1, the remaining 79% variance was explained by the random effects, (Years and Rivers).

Table 3.8. Fixed effects listed in the order of removal and with their associated significance values at the time of removal.

	0+ Densities (fish/100 m^2)		Length at age one (mm)	
Order of removal	Fixed Effect	P value	Fixed Effect	P value
1	IF1	0.599	SF1	0.525
2	EF1	0.677	SF2	0.251
3	ST	0.367	EF2	0.170
4	EF2	0.261	ST	0.110
5	ET	0.461	ET	0.284
6	IT	0.299	IT	0.270
7	IF2	0.058		
8	SF2	0.190		
Significant variables	**SF1**	**0.002**	**EF1**	**0.013**

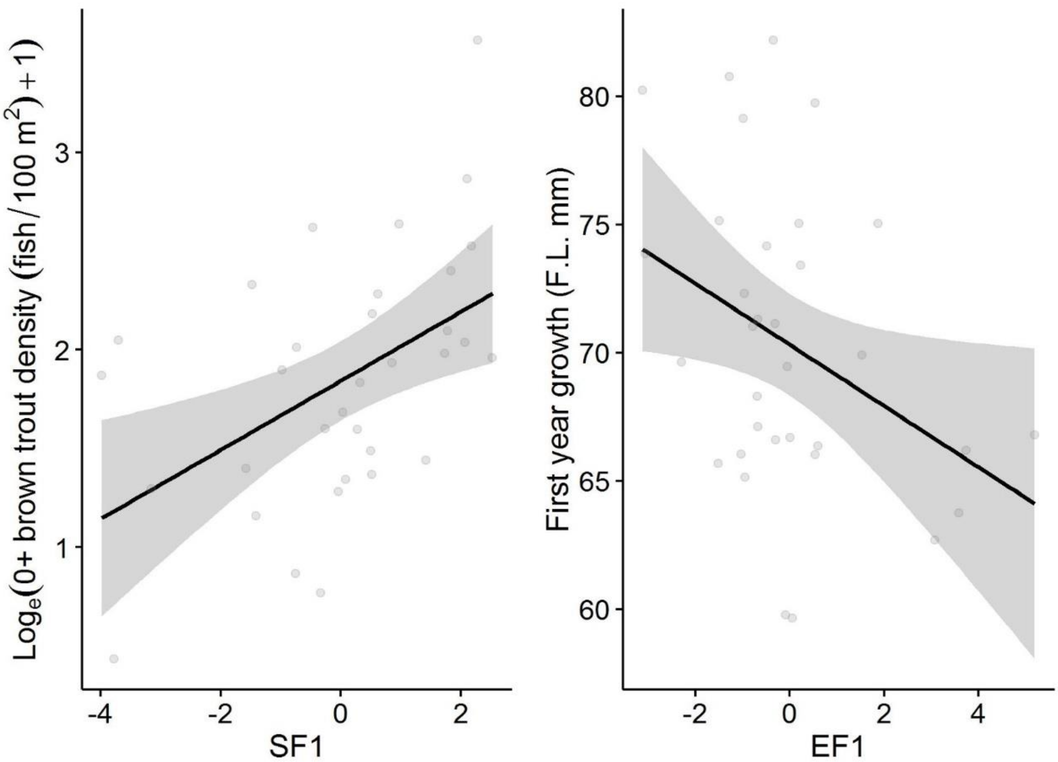

Figure 3.9 Relationship between $\log_e$ transformed 0+ brown trout density and summer flow rate (SF1) (left) and relationship between length at age one and emergence flow rate (EF1) (Right) for the Rivers Holme, Loxley and Rivelin during the study period (2002-2009, 2012-2014). Grey bands represent 95% confidence limits

3.4 Discussion

Both density dependent (i.e. inter and intra cohort competition) and density independent (i.e. flow rate) processes operate to influence the year-to-year variation on both size and abundance of juvenile brown trout in the three heavily modified rivers studied in Yorkshire. This was not unexpected given both density dependent and independent processes operate in tandem to regulate brown trout population (Richard *et al*. 2015). It would be rare if not impossible to have a population where only one of these processes is truly the sole driver behind population variability, as not only would the size of parental stock, inter-species and intra-cohort competition shape the size and structure of a year class, but the hydrology, chemistry and geomorphology of a habitat would also influence said year class. Elliott (1994) reported that density dependent processes are typically at their strongest shortly after emergence of brown trout or in populations close to or reaching carrying the capacity of the river. The scope of this study did not allow any investigation into the density dependent mortality of young of year via stock recruitment analysis as populations were only sampled yearly, precluding such analysis.

In unregulated rivers there is a strong relationship between both flow magnitude and precipitation level (Burn and Hag Elnur 2002). However, there was no evidence of any relationship between flow and rainfall for the three rivers in this study. This is because the nature of water storage operations in these systems is to ensure adequate water storage in large reservoirs (i.e. Brownhill, Digley, Damflask and Rivelin Dams) for months if not years thus impacting on natural streamflow (Collier et al. 1996). Other studies on the flow dynamics of river systems before and after the construction of large dams found that the occurrence of extreme flow events (droughts and floods) was substantially lower after the river was regulated by a dam (e.g. Gustard et al. 1987).

Despite the influence that the reservoirs exert on the hydrology of the three study rivers, the effects on their flow regimes are not similar. The disparity between the flow duration curves of the rivers Holme, Loxley and Rivelin reflects the differences between the operation and surrounding land-use of the three rivers. For instance, the River Loxley is a short stretch of river (between reservoir and gauging station) with no tributaries, so the level of natural gather from rainfall would be low; there is also very little overspill from Damflask reservoir (the high compensation release as well as abstraction ensures that the reservoir is drawn down most of the time). This is evident in the low level of variation in the magnitude between the 25^{th} and 75^{th} flow percentile in the River Loxley. On the River Rivelin it is slightly longer between reservoir and flow gauge and then the River Loxley, so again flow variability from natural sources would be minimal, but the compensation release from Rivelin Dams reservoir is much lower than in the Loxley, and as a consequence the reservoir overspills more often during periods of prolonged heavy rainfall providing a more varied flow. The River Holme is the longest stretch of river between reservoirs and flow gauge, but crucially the River Ribble (another regulated river) drains into the River Holme between sites HO3 and HO4, which will dissipate the influence that Brownhill and Digley reservoirs exert upon the study reach, explaining the increased magnitude of the flow rate on the River Holme compared to the River Loxley and Rivelin.

Environmental factors such as flow rate and temperature operating at a regional scale are believed to play a crucial role in “synchronising” population dynamics across a relatively large spatial extent (Moran 1953). This effect – known as the “Moran effect” has been reported in brown trout populations in French rivers (Cattaneo et al. 2003), but whilst 0+ and >1+ brown trout population densities in this study varied significantly spatially and temporally there was no evidence of synchronicity between the rivers. This is of particular interest because of the proximity of the rivers Loxley and Rivelin, highlighting that regional-scale climatic events do not exert strong influence on brown trout population dynamics in regulated river systems.

Understanding the underlying drivers behind population variability is crucial from a fisheries management perspective, especially providing a targeted management framework towards the attainment of good ecological potential for HMWBS. Lobón-Cerviá & Mortensen (2005) demonstrated that over 90% of the variation in density of cohorts across a lifetime could be explained by variation in the recruitment strength, suggesting that the greater the juvenile density the greater the density of the spawning stock will be in successive years and vice versa. This trend was observed in the rivers Holme, Loxley and Rivelin, but in all cases the level of variance explained was low (R^2 <0.20). Density dependent regulation of 0+ brown trout was also investigated by examining the relationship of ≥1+ densities the preceding year, to determine how potential spawning stock size influences recruitment (0+ densities). A positive but weak (<10%) significant relationship was detected in the rivers Holme and Rivelin, but in the River Loxley this relationship was non-significant. Investigations into the movement of brown trout on the River Holme (Taylor, 2017) found that resident brown trout occupied small home ranges and did not undertake seasonal spawning migrations. The fish in this study and that of Taylor (2017) were relatively slow growing when compared to the national average (Environment Agency, Unpublished Data) and therefore attain a greater age than faster growing brown trout found in lower latitude river systems (Jonsson, L’Abée-Lund, et al. 1991) (maximum observed age; River Holme = 6, River Loxley = 5, River Rivelin = 5). It is therefore possible that the younger year-classes may migrate and disperse downstream rather than compete with the larger older conspecifics, which might be able to occupy a home range within a study site for several years. While there are no data to support this, as the survey methodology (fixed sample sites on an annual basis) would not be able to effectively detect 0+ brown trout dispersal. It is, however, plausible and worth considering that the habitat dominance of larger, long living brown trout in the rivers Holme, Loxley and Rivelin are factors behind the low levels of variance explained by the model.

Density dependence does not fully explain the drivers behind year-to year variations in brown trout populations. Of the environmental factors studied, flow rates during ecologically sensitive periods were found to influence recruitment levels of brown trout. Summer flows were found to be significantly positively correlated with brown trout recruitment, largely driven by the synthetic variable SF1 (a variable that summarises the rate of flow during summer). This variable was negatively correlated with flow during the summer period, suggesting that lower summer flow rate would typically lead to higher 0+ fish densities at the end of the year. The importance of summer flows to juvenile brown trout has been previously reported (Jonsson and Jonsson 2011) with annual baseline flow levels and late summer elevations identified as important to the ecological success of brown trout (UKTAG). Previous studies have also reported the importance of flows during the incubation period (Unfer et al. 2011), and emergence (Jensen *et al.* 1997, Jensen and Johnsen 1999, Lobón-Cerviá and Rincon 2004, Lobón-Cerviá 2014, Bret *et al.* 2016). However, the variation in flows during the incubation and emergence periods were not found to have a significant effect on 0+ brown trout densities in the three study rivers. This does not mean that these phases are not critical to brown trout development but more likely reflects on the homogeneity of the flows during the incubation and emergence periods caused by the flow regulation.

Low summer flows and droughts can have catastrophic effects on brown trout (Jonsson and Jonsson 2011). The reduced wetted area of a river during a drought can increase competition between fish as the available habitat and food resources are decreased (Jonsson & Jonsson, 2011). However, other abiotic factors, such as reduced oxygen and increased temperature, are also associated with lower summer flow (Titus and Mosegaard 1992, Elliott et al. 1997). In the study rivers, low flow rates are determined by the baseline compensation level, and only in extremely rare occurrences (such as severe drought that depletes reservoir to emergency levels or fault in the reservoir itself) would the flow drop below the baseline compensation release. Consequently, it is unlikely that any significant impacts on flow and water temperature will arise in the study rivers during a drought under normal reservoir operation. The variability in the flow during summer most likely reflects variation in the flow above the baseline level, with higher SF1 values representing drier summers. Higher flow rate in an unregulated river during summer would potentially provide greater marginal habitat, which is the preferred nursery habitat for the juvenile brown trout (Jonsson & Jonsson, 2011). The relationship between 0+ density and summer flows suggest that increase flows during summer would have detrimental impacts on recruitment. As the compensation release in the three study rivers was constant around the year it is likely that the flow rate during summer was elevated from what would be expected under unregulated conditions. It is therefore unlikely that the parabolic relationship between recruitment and flow rate (Lobón-Cerviá and Rincon 2004, Lobón-Cerviá 2014) would be present in the regulated rivers in this study, as flow rate would rarely drop to low levels due to the compensation release. Also, the river reaches studies were short between barriers and the densities of fish were low, precluding the likelihood of density dependent factors intervening in population regulation (see Section 3.4.2). Increasing flow rate beyond the baseline flow during summer does have negative impacts on 0+ brown trout. These high flows are likely in a combination of heavy rainfall and reservoir operation, for instance, some reservoirs elevate their compensation release during drought to ensure adequate river levels in the river network. These flow elevations are probably creating sub-optimal habitat by reducing juvenile nursery habitat within the stream as well as increasing the rate of washout of the fry unable to maintain station in elevated water velocities, as seen in Norwegian regulated rivers (Saltveit et al. 1995)

First year growth of 0+ brown trout varied both spatially and temporally throughout the study, with brown trout in the River Rivelin being significantly smaller ($P<0.05$) then in the River Holme and River Loxley, but when compared to the national average (reference) brown trout in all three rivers were classed as slow growing. Brown trout growth in the three rivers ranged in size at the end of the first year from 65 ± 1.54 for the River Holme, 71 ±1.2 mm for the River Loxley and 60 ± 1.2 mm for the River Rivelin, but with individuals at either end of the size range being classified as extreme slow or fast first year growth when compared to the UK national averages. This level of phenotypic plasticity has been observed in Atlantic salmon populations in Norwegian rivers (Jonsson, Hansen, & Jonsson, 1991), where energy consumption and expenditure were considered more important factors in the variability of growth than inherited genetic traits. Several studies into density dependent growth (Grant and Imre 2005, Lobón-Cerviá and Mortensen 2005) demonstrated that there was a negative relationship between 0+ density and first year growth rates. No such relationships were found in the study rivers. This is largely because of the low densities of brown trout found and likely poor quality of the habitat and potential food resources because of the isolation of the stretches between reservoirs and lack of allochthonous and autochthonous production in the systems. Diet analysis of 0+ and ≥1+ brown trout in the study rivers would be required to confirm this hypothesis, but diet analysis was outside the scope of this study

To determine the influence that abiotic factors (flow rate and temperature) had on the first-year growth of brown trout, a linear mixed effects model was constructed using the synthetic flow rate variables for emergence and summer periods and degree-days for the incubation, emergence and summer periods. This model construction was very similar to the one used to investigate abiotic regulation of 0+ brown trout population density. The major difference between the structures of the two models is that the synthetic variables for incubation flow rate were removed for this (first year growth) model. The purpose of this was to ensure only biologically relevant variables were included to reduce the model complexity and reduce type I and type II errors. Temperature during incubation was retained in the first-year growth model as a biologically relevant variable as experiments into the muscle development and growth of hatchery reared Atlantic salmon have been shown to be influenced by temperature regimes during incubation (Albokhadaim et al. 2007). The final model concluded that growth of 0+ brown trout was negatively correlated with variation of flows during emergence, but the variance explained by the fixed effects was relatively low ($R^{2fe} = 0.1$). However, the random effects of both site and year explained a large portion of the residual error, so that the full model explained 88% ($R^{2fm} = 0.88$) of the variance in the data. Visual observation of the fixed effects showed that there was a high level of variability in growth in years with lower flow during emergence, but lower first year growth observed in years with higher flows during emergence. The interaction between flow rate and length can possibly be explained through increase bioenergetic expenditure to hold station at higher water velocities (Flore and Keckeis 1998, Nislow et al. 2004, Xu et al. 2010). It is important to consider that whilst low flows during incubation promoted better first year brown trout growth in the study rivers, these flows are still influenced by the compensation releases from the impounding reservoirs, therefore the devastating effects of low flows in unregulated rivers (Teichert et al. 2010, Kanno et al. 2016) are unlikely to be seen in the three regulated study rivers.

The impact that reservoirs exert on flow dynamics in the three heavily modified rivers has reduced the influence that local climate has on synchronising recruitment success on brown trout. As the three study rivers are indicative of other HMWBs throughout Yorkshire, it is likely that the reservoir status and operation now plays an important role in governing the recruitment success of brown trout in the region. From a management perspective, flows during the emergence and summer periods are crucial to brown trout

development and population dynamics. Unlike in unregulated river systems the flow regimes during incubation and emergence periods were not seen to significantly influence brown trout recruitment. This could be because either rainfall during these periods is able to elevate the flows sufficiently through run-off, seepage and reservoir overspill to ensure that the ecological requirements of developing 0+ brown trout are met, or the influence of the impounding reservoir is too strong to allow for any increases or decreases to flow magnitude that would significantly impact on 0+ brown trout populations. During the summer period flow rate significantly influenced recruitment, with flow elevations during the period negatively impacting brown trout recruitment.
This relationship should not be interpreted as lower flow rate in during the summer leads to better recruitment, as in all three study rivers the constant compensation release from the impounding reservoirs likely represented a sizeable portion of the summer flow rate. The effect that extreme low flow and drought events can have on brown trout population dynamics (Cowx and Gould 1989, Titus and Mosegaard 1992, Elliott *et al.* 1997) are potentially dissipated regulated rivers in the Yorkshire region, with the summer compensation flows possibly benefitting brown trout populations due to the provision of low and stable summer flows, which are crucial to post-emergence brown trout (Acreman 2007).

Biotic and abiotic processes regulating the variability of individual growth rates of brown trout were less obvious, with flows during the emergence period being negatively correlated with 0+ brown trout growth. The interpretation of this result is again similar to the interpretation of the influence of summer flows on 0+ brown trout density, in that these results should not be viewed as a preference for low flows as the compensation release ensures that low flows are rarely present in the three study rivers. It is more likely that the high flow events during these periods through either heavy rainfall, reservoir overspill, or operational release, negatively influence 0+ brown trout growth. An interesting result from the 0+ growth model was that Rivers and Years were strong predictors of 0+ brown trout growth, therefore there were significant differences between the growths between the three study rivers, possibly due to different levels of bio-energetic expenditure, ration size, and habitat quality, but these metrics were outside the scope of this study.

From a fisheries management perspective, compensation flows, and the operation of the impounding reservoirs alter the flow regime from what would be expected under natural conditions. The compensation release itself is providing stable flow rates throughout key life stages where drastic variability in the flow rates could negatively influence trout recruitment. As reservoir removal is costly and (in many cases) detrimental to society

(Poff and Hart 2002), mitigation measures for impounding reservoirs include at their most basic form introducing and revising compensation flow (see chapter 2.7). This study suggests that the compensation releases on the three impounded rivers maybe beneficial to brown trout population dynamics when compared to the alternative of no compensation release, but the isolation and lack of sediment replenishment, amongst other factors, are likely to have a profound impact. Further studies into the magnitude of the compensation release would be required to refine the release level for the best ecological gain.

4 IMPACT OF INTRODUCING A SEASONALY VARIED COMPENSATION RELEASE ON POPULATION DYNAMICS AND HABITAT QUALITY OF BROWN TROUT IN THE RIVER DIBB

4.1 Introduction

In natural river systems, hydrology is a key driver of fish population dynamics, with extreme variations in the flow regime a strong determinant of recruitment success (Lobón-Cerviá 2014). The analysis presented in chapter 2 revealed that brown trout populations in rivers or streams designated as HMWBs with large impounding reservoir(s) deviate from this natural functioning, as both brown trout populations and hydrology are influenced by reservoir operation. To attain "good ecological potential "in these river systems in compliance with the WFD environmental objectives, the least resource intensive mechanism is likely modification and/or introduction of compensation releases from the impounding reservoirs. To achieve GEP, water resource managers need to balance their legislative requirements with economic and social factors to provide mitigation measures. The building block approach (King *et al.* 2008) is an appropriate methodology adopted by UKTAG for designing compensatory releases for the benefit of riverine ecology (See Chapter 2.7).

Riverine fish are reliant on certain key flow events, e.g. for brown trout elevated flows during the winter are crucial in removing fine sediments and metabolites from spawning gravels as well as improving longitudinal connectivity (Jonsson and Jonsson, 2011) . These flow events can be broken down into several key building blocks, such as minimum flows, seasonal elevations, freshets and channel forming floods (Chapter 2.7). These building blocks can then be incorporated where necessary into new compensation flow regimes to replicate the biological, physical and chemical benefits of a natural flow regime (Acreman 2007). The socio-economic costs of achieving GEP must be accounted for alongside the ecological impacts when designing environmental flows. For instance, adopting an environmental flow that adjusts in real time to natural flow variation within a catchment, would require huge infrastructural investment in flow monitoring and the likely retrofitting of reservoir valves to allow for remote and frequent changes. The primary purpose for each impounding reservoir must be fully understood, for example, if the reservoir is required to provide water storage for human consumption there would be little benefit to providing a fully varied compensation flow to the detriment of water supply, as that demand would have to be met from other sources. Seasonally varied flow regimes can provide a good balance between the constraints of water resource and ecological demands (Yin *et al.* 2016), as a seasonally variable flow (in most cases) can be introduced with little impact on reservoir operation yet provide a reflection of natural flow variations that would be expected in unregulated conditions.

Where salmonids are the primary focus of flow restoration measures, a key objective would be to establish a flow regime that ensures that the key life cycle stages (such as spawning migrations, incubation, emergence and juvenile growth (UKTAG 2013)) are protected. For salmonids, behaviours such as upstream migration and spawning are triggered and aided by elevated flows during the winter period (Acreman 2007, Jonsson and Jonsson 2011), but these elevated flows during the emergence period can have drastic consequences on brown trout recruitment (Daufresne *et al.* 2005, Lobón-Cerviá 2014). Therefore, a seasonally varied flow regime (if set at appropriate rates) designed using the building block methodology can provide better hydraulic and geomorphic conditions for salmonid species compared with no or a uniform reservoir release.

The aim of this chapter is to determine if the introduction of a seasonally variable compensation flow from Grimwith reservoir has changed habitat quality and brown trout population dynamics using a Before After Control Impact (BACI) approach.

The specific objectives are to:

- Investigate 0+ and ≥1+ brown trout populations and habitat quality at sites along the River Dibb (downstream of Grimwith reservoir), before and after the introduction of the new compensation release;
- During the same timeframe, investigate 0+ and ≥1+ brown trout populations and habitat quality at sites whose flow regime is not influenced by Grimwith Reservoir (control sites) to estimate the temporal variability of brown trout populations in the region;
- Establish the influence (if any) that the introduction of the compensation release has had on brown trout populations and habitat quality in the River Dibb using a Before After Control Impact (BACI) methodology.

These findings aim to provide insights into the effectiveness of the introduction of a seasonally variable compensation flow on brown trout populations in a Heavily Modified Water Body (HMWB), which will aid in the development of the knowledge base of appropriate mitigation measures to achieve GEP in HMWBs regulated by reservoirs.

4.2 Methodology

4.2.1 Study reaches

Yorkshire Water Services, as part of their Adaptive Management Programme (AMP5), identified the River Dibb as a river downstream of an impounding reservoir where no compensation release was in place. Thus, Grimwith Reservoir (River Dibb) was selected as a suitable candidate for a flow trial where a seasonally variable compensation release would be introduced to mitigate any hydrological impacts downstream of the reservoir. To determine any biological responses to the flow trials, brown trout was selected as an indicator species due to its abundance across all study and control rivers, as well as its sensitivity to flow variation (see Chapter 2.5)

River Dibb (impact sites)

The River Dibb is 5.2-km long in North Yorkshire and a tributary of the River Wharfe. Grimwith Reservoir is located at the head of the river where three streams (Grimwith Beck, Blea Gill Beck, and Gate-up Gill) converge (Figure 4.1). Two streams (Birsta Gill Dike, and Stone Gill Dike) join the River Dibb downstream of Grimwith reservoir. Grimwith reservoir was originally constructed in 1864, and between 1970 and 1983 expansion work was undertaken to elevate the dam wall by a further 20 m allowing for a seven-fold increase in water storage. A hydroelectric generator was installed on outfall pipes during this expansion for electricity generation during any planned water release. Water is regularly released from Grimwith reservoir to ensure flow levels are maintained for water abstraction operations downstream in the River Wharfe. Water is also released when the reservoir approaches maximum capacity t to ensure any excess water that the reservoir could not hold would be used to generate electricity. Following assessments of the River Dibb catchment in 2011 by the EA, the ecological status of the river was classed as moderate under WFD criteria, with low fish populations believed to be the result of altered flow regimes from Grimwith Reservoir.

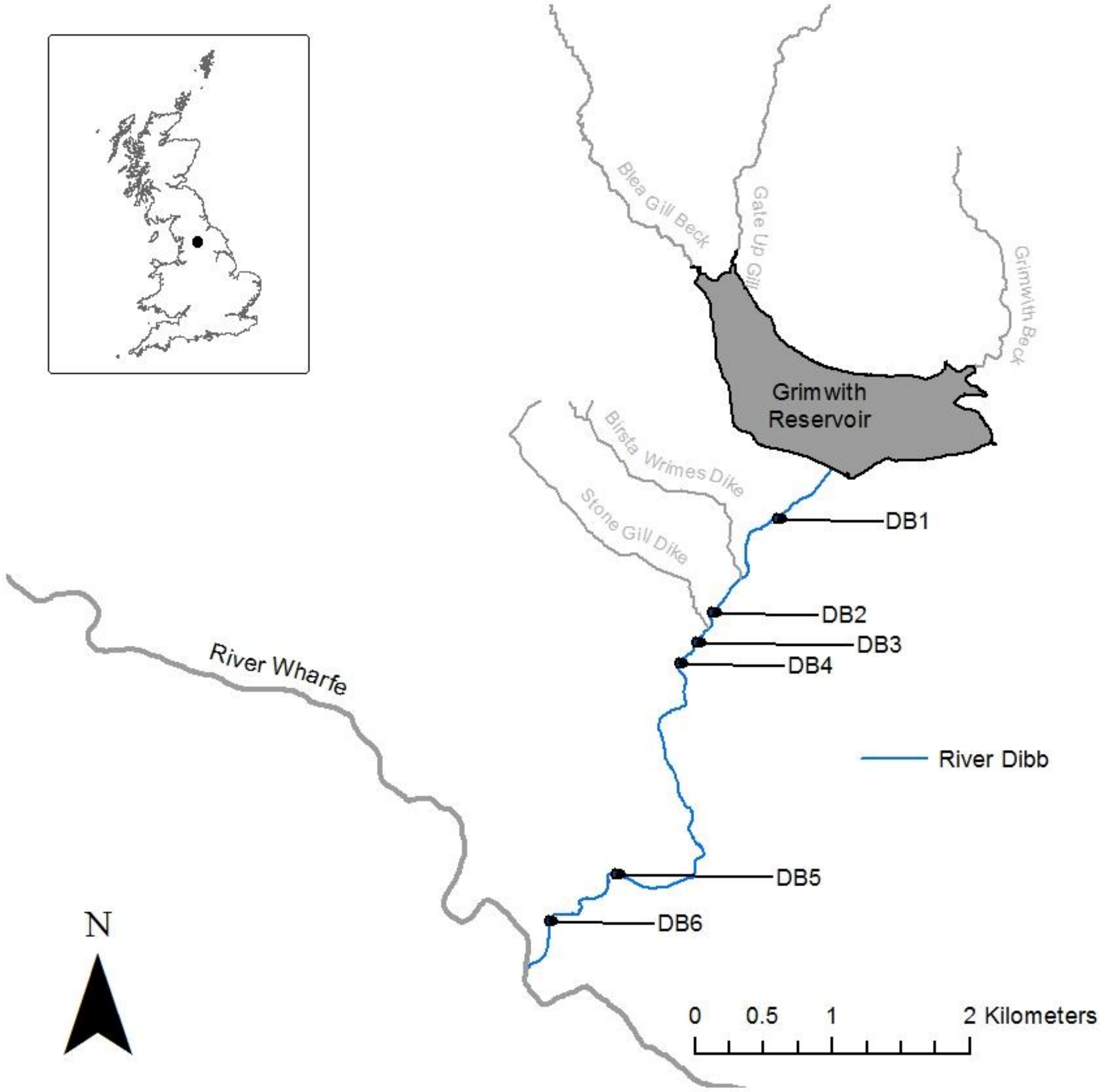

Figure 4.1. Location of all impact sites (DB1-DB6) in relation to Grimwith reservoir and surrounding tributaries.

For environmental (fisheries and habitat) investigations, six sites of 50 m in length (DB1-6) were selected along a 4.3-km section of the River Dibb with the most upstream site (DB1) 350 m downstream of the reservoir outfall, and the most downstream site (DB6) 500 m upstream of the confluence with the River Wharfe. The six sites were chosen to represent habitat heterogeneity of the River Dibb. The adjacent land use to the River Dibb was predominantly a mix of Heathland and rough pasture throughout, with deciduous woodland cover at sites DB5 and DB6, the average slope of the Dibb was 3.2%. Fisheries monitoring was undertaken at each site annually in October over a five-year period from 2012-2016. Monitoring was split into three years (2012-2014) to provide a baseline (*before*) and two years (2015 and 2016) after the start of the flow trial (*after*). In January 2015, a compensation flow was introduced from Grimwith Reservoir, with a seasonably variable minimum flow (Table 4.1). This building block flow regime was designed to follow flow trends seen in natural systems and provide, geomorphological physico-chemical and ecological benefits to the river in an attempt to increase the ecological potential to “good”.

Table 4.1. Magnitude (Ml/d) and timing of the seasonably variable compensation flow introduced from Grimwith reservoir. Week numbers relate to the ISO week data standard (ISO-8601) i.e. week 1 is the first week of January and week 52 last week of December

Period	Jan-Apr	Apr-May	May-Oct	Oct-Dec
weeks	1-15	16-18	19-40	41-52
Minimum flow (Ml/d)	15.12	7.74	3.80	7.80

Barden, Ings and Ashfold Side Becks (control sites)

To ensure that any change to fish assemblages following the introduction of a compensation flow is correctly understood, the study design required a number of control sites. The purpose of a control site is to provide a suitable estimation of temporal variance of the metrics that would be present in the River Dibb in the absence of the flow trial. This allows for a greater level of certainty surrounding any changes to fish populations as the study is able to discern between variance caused by the flow trial and variance that would persist if the river remained unchanged. When selecting control sites Physical (land use, cover and gradient) and biological (biological diversity and abundance of the fish fauna) characteristics should be considered to ensure control sites are broadly similar to the impact sites. Ideally control sites would be randomly selected from the stretch of river immediately upstream from the impounding reservoir to help reduce the level of spatial variability in the model. However, this was not appropriate for the Grimwith Reservoir flow trial as the streams upstream of the impounding reservoirs were deemed inappropriate as control sites due to dissimilarity between the physical characteristics of these sites and the River Dibb. To ensure that the control sites were subject to similar climatic events (e.g. rainfall, droughts) the selection of sites was limited to rivers and streams that were in close proximity to the River Dibb. All control sites chosen in this study were located within a 9 km radius to the River Dibb (Figure 4.2).

Nine, sites of 50 m (WR1-WR9) were selected from three rivers to be control sites for the Grimwith reservoir flow trial. The rivers selected were Barden Beck (WR1 and WR2, 2 sites) and Ings Beck (WR3 and WR4, 2 sites), both tributaries of the River Wharfe, and Ashfold Side Beck (WR5-WR9, 5 sites) which is a tributary of the River Nidd (Figure 4.2). The adjacent and upstream land use of the three control rivers were similar to the River Dibb i.e. Heathland and Rough pasture throughout the three reaches with areas of deciduous woodland in the lower sites (WR2, WR4 and WR9) with comparable stream gradient for the three control rivers 3.8%, 2.5% and 4.3% for Barden, Ings and Ashfold

Side beck respectively. The selection of sites and rivers was to ensure the maximum number of control sites were available from comparable and neighbouring rivers with similar hydrology riparian connectivity and land uses to create a more statistically robust BACI model. Both Ings Beck and Ashfold Side Beck are un-regulated streams with no impounding reservoir, but flow in Barden Beck is regulated by Lower Barden reservoir (Figure 4.2). Barden Beck was still selected as a suitable control site as there was no anthropogenic change to the flow regime during the study therefore temporal variation would reflect the natural conditions.

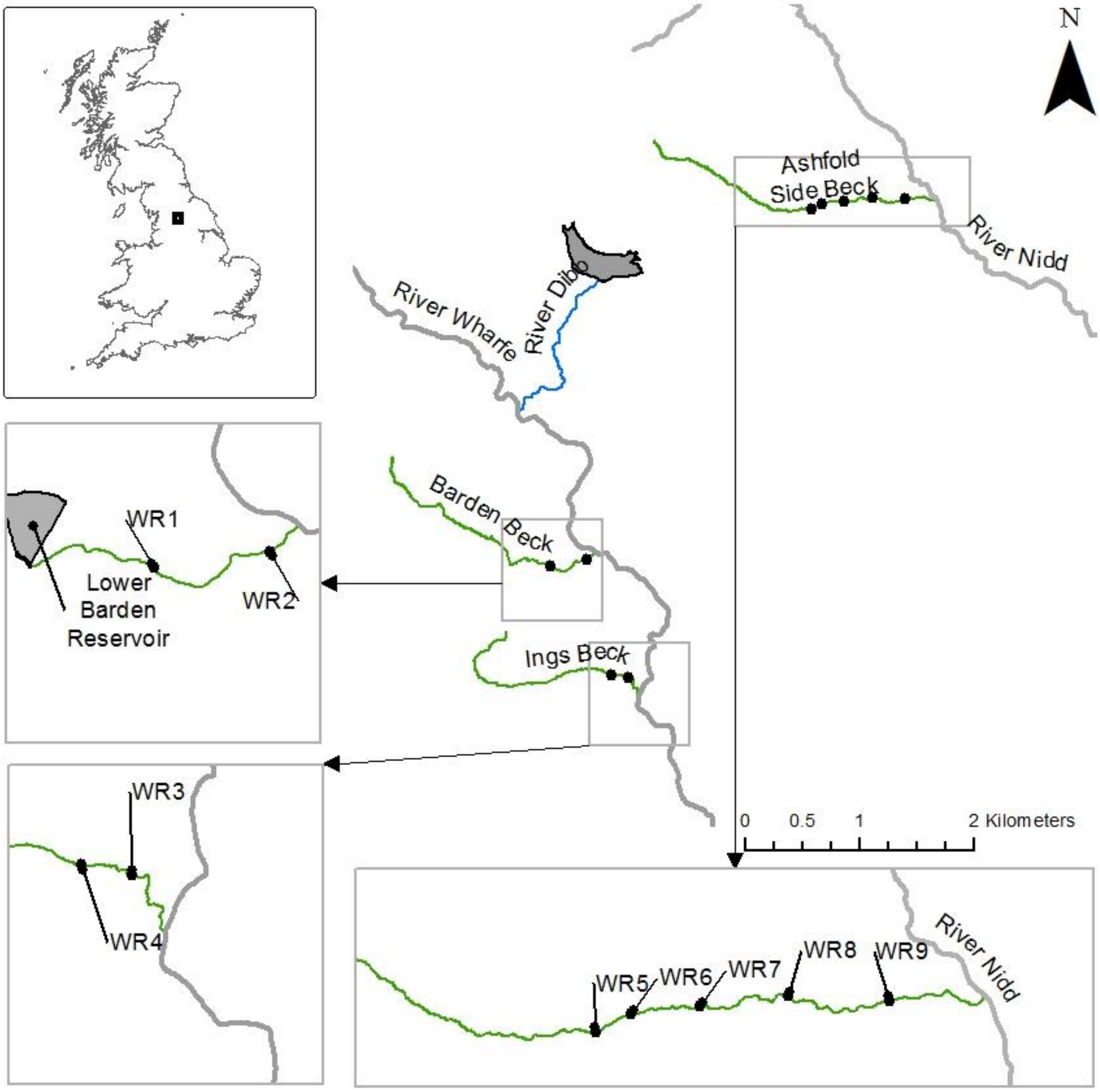

Figure 4.2. Location of the nine control sites (WR1 – WR9) in relation to each other, the River Dibb and Grimwith reservoir

4.2.2 Data collection

Environmental investigation

Brown trout populations in the study river (River Dibb) and the control rivers (Barden, Ings and Ashfold Side Beck) were monitored annually during before (2012-2014) and after (2015-2016) the flow modification using the protocols defined in Chapter 3 (See section 3.2.3). Density estimates were generated for 0+ and ≥1+ brown trout populations for all impact and control sites, additionally density estimates were generated for ≥1+ (< 20cm) and ≥1+ (> 20 cm) brown trout for use in HABSCORE analysis (Section 4.2.3.3)

Classification of population estimates

Density estimates derived from fisheries surveys were compared with the Environment Agency's Fisheries Classification Scheme (EA-FCS). This 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.* 1994a).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 4.2). These population estimates were used for spatial and temporal comparison not only as part of the BACI analysis (section 4.2.4) but also to identify any spatial trends within and between rivers (i.e. sites or rivers with consistently greater/poorer brown trout densities) or identify and regional synchronicity (i.e. uniform directional trends to brown trout populations across sties and rivers). It is important to note that this Classification scheme is derived from data collected on a national scale, and therefore there is a question as to its suitability when used on Yorkshire Spate rivers. The rationales behind using this scheme are that: regional classification schemes do not currently exist, applying the data collected to this national scale still provides useful information on any change to fisheries that can be compared at a national scale, and that this system of classification is currently used by the regulating authority (EA).

Table 4.2. 0+ and ≥1+ brown trout abundance (N/100 m²) 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
≥1+ brown trout	≥21.00	12.00-20.99	5.00-11.99	2.00-4.99	0.10-1.99	0.00

Length at capture

Determination of the age and growth of fish is an important tool in the assessment of fish population dynamics (Bagenal 1978). The methodology for the determination of age and growth of brown trout outlined in Chapter 3 (section 3.2.3),was not appropriate for this study because the growth rates are calculated by the proportional relationship between the length of the fish and the radius on the scale at which annuli are present (section 3.2.3). All 0+ fish are identified by the lack of annuli present on their scales, therefore all 0+ brown trout caught in this survey cannot be used to derive 0+ growth rates. As only fish that are ≥1+ can be used to determine 0+ growth rates of a year class it is not possible to present any growth data for the 2016 year class for this investigation. Length at capture was selected as an appropriate methodology to generate fish size metrics given the limited timeframe of the investigation, but these data should be interpreted with caution. Spatial and temporal variation of length at capture is much more likely to be influenced by the date at which the survey took place, for instance, length at capture is likely to be lower for fish caught in July than fish caught in September, as September fish would have 2 extra months to grow. To minimise this source of error, all surveys took place in September or October.

Habitat survey

HABSCORE is a system for measuring and evaluating stream salmonid habitat features based on empirical statistical models relating the population size of five salmonid species/age combinations (Table 4.3). Using the information from three HABSCORE questionnaires, the software produces a series of outputs, which includes estimates of the expected populations (the Habitat Quality Score, HQS) for each of five salmonid species/age combinations (Wyatt, Barnard & Lacey 1995). The HQS for each age class can be related to population estimates to provide a Habitat Utilisation Index (HUI), which provides a quantifiable relationship between the density of trout present and the density of trout expected under non-impacted conditions. To collect information for HABSCORE analysis, a questionnaire on the habitat found at each site was completed following electric fishing surveys at all sites in all years. The HABform questionnaire is based on physical measurements of widths and depths as well as observational data of the substrate flow and cover. The HABform questionnaire requires the habitat data to be recorded for 10-m segments. The wetted width and channel depth (at ¼, ½, and ¾ channel width) are recorded once per 10-m section. Substrate, flow and cover (Table 4.4) are recorded for each 10-m segment and classified to one of five ASCFD abundance categories (Absent – 0%, Scarce - <5%, Common - <20%, Frequent - <50% and Dominant - >50%).To complete the datasets for HABSCORE, a further questionnaire (MAPform) for each site required completion (Barnard & Wyatt 1995). MAPform is completed by collection of relevant information from OS Maps (1:50000) and River Water Quality Maps (1:25000). Unless it is believed that significant changes to the surrounding habitat have occurred (i.e. change in the number of upstream tributaries, or river gradient), the MAPform only needs to be completed once per site, with the HABSCORE programme retaining the information from this questionnaire for repeated site visits, however, the HABform questionnaire needs to be completed for every site in every year that fisheries data were collected (2012-2016). HQS densities can therefore be derived for all sites in all years allowing for spatial and temporal comparison of habitat quality for each of the age classes (Table 4.3). Salmon (*Salmo salar* (L.) were absent from the River Dibb as well as the control sites and therefore not assessed in this investigation. The kinumber of persons that completed HABforms during the study period were kept to a minimum, and where possible the same person would undertake the same surveys in multiple years. Where this was not possible, appropriate quality control measures i.e. exercises where multiple persons complete HABform and outputs were compared to detect any sources of variability, were put in place to minimise this source of error.

Table 4.3. Age and size classes for Salmonids used in the HABSCORE analysis (Wyatt *et al.* 1995)

Species	Brown trout (age/size class)	Salmon (age class)
	0+	0+
	≥1+ (< 20 cm)	≥1+
	≥1+ (> 20 cm)	

Table 4.4. Substrate, flow and cover parameters assigned an ASCDF abundance category during the HABform survey.

Substrate	Flow	Cover
Compacted Clay	Slack (depth = <30cm)	Submerged Vegetation
Fine Sand / Silt	Slack (depth = >30cm)	Boulders, Cobbles,
Gravel / Coarse sand	Glide / Run (depth = <30cm)	Tree root systems
Cobbles	Glide / Run (depth = >30cm)	Branches and Logs
Boulders	Turbulent / Broken (depth = <30cm)	Undercut Banks
Bedrock	Turbulent / Broken (depth = >30cm)	Other Submerged Cover
	Cascade / Torrential	Overhang
		Area of Deep Water

HABSCORE analysis and outputs

HABSCORE data were collected in every year, corresponding to fisheries survey years, at all impact and control sites. Data from the two completed forms (HABform, and MAPform) at each site were entered into the HABSCORE for Windows program and the following outputs were produced for brown trout populations on an annual basis (definitions from Wyatt, Barnard & Lacey (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/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 was determined for three brown trout age and size categories (Table 4.3):

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 95% 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 non-impacted conditions. Conversely, a lower HUI confidence interval >1 indicates that the observed population was significantly higher than would normally be expected under non-impacted conditions.

Impact assessment and resource calculation

The purpose of an impact assessment is to provide statistically robust evidence that a meaningful change has occurred. To assess the effects of flow modification requires sufficient information from specific survey sites and over a sufficient duration to detect changes in fish population and community characteristics resulting from, in this case, flow modification. The effectiveness of detecting change depends on the variance in population characteristics between sample sites or over years. However, proving the change was caused by the pressure under observation and not attributed to coincidental temporal and spatial influences is difficult. Consequently, the study design must control/account, as far as possible, for extraneous influences by including data from the same sites before the impact to eliminate spatial differences and through comparison of an adjacent un-impacted control area to eliminate natural temporal variations. This study design is commonly known as the BACI design based on the before/after, control/impact configuration. Sedgewick (2006) documented the procedure to apply these principles to analyse fish population changes in space and time to determine resource requirements and perform impact assessments.

Understanding which facets of a fish population to monitor for a change is fundamental to the outcomes of this study. The analysis of long-term datasets from the Rivelin and Loxley studies presented in Chapter 2 demonstrated that both 0+ abundance and 0+ growth are two elements of fish communities that are susceptible to change in flows. The change in flow regime in a regulated river will not only exert influence on 0+ brown trout, a change to the flow regime would represent a change to the physical habitat of the river, therefore it would be prudent to investigate if the change to the flow regime has had significant impacts on ≥1+ brown trout as well as habitat quality. To allow for easier comparison between the population density and HQS, the BACI analysis was performed on the two HABSCORE age classes of ≥1+ (< 20 cm) and ≥ 1+ (> 20 cm).

A General Linear Mixed Models (GLMM) analysis was utilised to perform the BACI analysis. This allowed a greater understanding of how the sources of variation contribute to the eventual outcome of the flow trial. GLMM models were constructed in R using the "lme4" package (bates) with the following notation. To account for the large variability between density estimates and HQS densities for both 0+ and ≥1+ brown trout between impact and control sites, all data were transformed (natural log +1).

Equation 1

$$M \sim (1|Si{:}Ar) + (1|Yr{:}Pe) + Ar + Pe + (Ar \times Pe)$$

Where:

M is the biological metric to be tested

Si is the survey site

Yr is the survey year

Ar is the area i.e. impact or control site

Pe is the period i.e. before or after flow trial

The assumptions of the GLMM (linearity, homogony of variance and normality) were confirmed using visual examinations of plots such as: residuals vs predictor plots, residuals vs fitted. The natural log transformation of the data prior to analysis was to account for any skewness and ensured data were normally distributed. The premise of the GLMM BACI model is to determine a significant interaction between the two fixed variables of Area and Period. To reduce the levels of the residual error, random effects of Site and Year, both were nested within the fixed factors of Area and Period. An important tool in the designing and planning stage of habitat rehabilitation is the resource calculation. This is an iterative model that generates the variance of the impact and control sites before a flow trial and based on the assumption that the temporal variance detected persists, can determine if the number of years and number of sites proposed for post impact monitoring is adequate to detect a biologically relevant change to fish populations using the following model:

Residual Variance (Rvx) for impact before (Rvx_{tb}) and control before (Rvx_{rb}) is the EMS (Expected Mean Squares) of a 2 x 2 factorial ANOVA ($M \sim Yr + Si$i), these values are averaged to give a pooled variance estimate. Actual Variance is computed from pooled variance using equation 2

Equation 2

$$AVx = PVx \times \left(\frac{1}{(nYr_{Be} \times nSi_{Tr})} + \frac{1}{(nYr_{Af} \times nSi_{Tr})} + \frac{1}{(nYr_{Be} \times nSi_{Re})} + \frac{1}{(nYr_{Af} \times nSi_{Re})} \right)$$

Where:

nYr_{Be} = number of survey years before flow trial

nYr_{Af} = number of survey years after flow trial

nSi_{tr} = number of impact sites

nSi_{Re} = number of control sites

To determine the effectiveness of the study design, a target variance is generated from imposing a biologically meaningful impact on the before metrics. Due to the inherent variability in brown trout populations, a 50% change is accepted to be a biologically meaningful change that can be detected following habitat alteration (Cowx, 1996). 50% however, is a large change to be detected in any growth based metrics, i.e. length at capture. Therefore, a lower biologically meaningful change of 25% will be imposed on the length at capture data. A change of 25% was chosen in conjunction with the EA national growth standards for brown trout and represents a change that will represent a change in growth category, i.e. a 25% increase in a slow growing population would result in the population classified as having average growth.

Target variance (TVx) is determined by equation 3

Equation 3

$$TVx = \left(\frac{M \pm BMI}{\Phi \times \sqrt{2}} \right)^2$$

Where BMI is the imposed biologically meaningful impact on the metric (50% for Density and HABSCORE derived metrics and 25% for length at capture) and Φ is a value associated with the degrees of freedom which is estimated by (number control sites + number of impact sites) – 2.

Therefore, when target variance is less than actual variance the number of years proposed for the study and the number of impact and control sites is sufficient to detect a 50% change to the biological metrics at a 95% significance level. The process can be altered and used diagnostically during a full impact analysis to ensure that the levels of variance in the combination of impact and control sites during the *before* and *after* periods are sufficient to detect a change. This process can be achieved by estimating the RVx for both impact and control sites for the *after* period and pooling them together with the *before* period RVx to adjust the PVx value. The purpose of using a diagnostic resource calculation is to ensure that the intensity of sampling is adequate to detect a biologically relevant change to brown trout communities prior to running a full GLMM BACI analysis, this methodology will reduce the likelihood of a type II error from the GLMM model.

Nested ANOVA tests were utilised to determine if any there were any significant differences between the metrics spatially or temporally, prior to analysis tests to ensure the homogeny of variances were confirmed using levene's test, and normality with the shapiro-wilks test.

4.3 Results

4.3.1 Impacts of the introduction of a compensation release on 0+ brown trout densities

Baseline monitoring (Before)

Pre-flow trial, 0+ brown trout were caught at almost all impact sites in all years except sites DB1 and DB4 in 2012 (Table 4.5). At the remaining sites, 0+ brown trout densities were low between 2012 and 2014, ranging from 0.2 to 6.6 fish/100 m ^{2}. During this period, the populations were classified as either poor or fair/poor (Table 4.5), but there was overall a year-on-year increase in densities from 2012 to 2014 (Figure 4.3, Table 4.5) with a significant difference between 0+ brown trout densities between the years during this period ($F_{1,16} = 36.41$, $P < 0.05$), but there was no significant difference between the 0+ brown trout densities between the impact sites ($F_{5,12} = 0.53$, $P > 0.05$).

0+ brown trout densities at the control sites were much more varied amongst sites and years than impact sites; however, pooled densities for the control sites showed a temporal trend similar to the general increase seen in the impact sites. During the pre-flow trial (before) period 0+ brown trout densities in control sites varied from 0.9 to 47.0 fish/100 m ^{2} and ranged from poor to excellent (Table 4.4) however, there were no significant differences between the three control rivers ($F_{2,25} = 0.275$, $P > 0.05$) or between the sites ($F_{8,18} = 0.35$, $P > 0.05$). There was a significant difference between years at control sites ($F_{1,25} = 13.1$, $P < 0.05$). In 2012, 0+ brown trout populations were predominantly classified as poor and populations at two sites (WR2 to WR3) classified as fair/poor (Table 4.5). There was a general trend of pooled densities across sites improving from 2012-2014, however, populations at some sites did not follow this pattern. For example, 0+ brown trout densities at site WR6 increased from 2.2 to 47.0 fish/100 m ^{2} from 2012 to 2014, whereas in the same time period 0+ brown trout densities in a site further downstream (WR8) declined from 19.1 to 2.7 fish/100 m ^{2}; this high variability is reflected in the large confidence limits of the 2013 and 2014 pooled density estimates (Figure 4.3).

Before the commencement of the flow trial in 2015 a resource calculation was performed on the 0+ brown trout “before” data (Table 4.5). Under the assumption that the temporal variability in the impact and control sites persists, the resource calculation revealed that a ≥50% increase in 0+ brown trout populations would be detectable following 2 years of “after” monitoring (Table 4.6). Due to the variability and low densities in the impact sites it would not be possible to detect a ≥50% decrease in 0+ brown trout densities within 2 years, and a further year of monitoring (2017) would be required to ensure a statistically robust conclusion.

During compensation release (After)

0+ brown trout were absent from site DB4 in both 2015 and 2016 following the introduction of the compensation release, while at the remaining sites densities ranged from 0.4 to 5.0 fish/100 m ^{2} and were classified as poor to fair/poor (Table 4.5). Pooled density estimates decreased during this period with sites in 2015 and 2016 holding lower populations than in 2014 (Figure 4.3). Comparison of the populations between the two periods revealed a general decrease in 0+ pooled brown trout densities during the after period (2015 – 2016) compared with 2013-2014, however, the 0+ brown trout populations at the impact sites remained higher than the very low densities recorded in 2012 (Table 4.5).

Temporal trends in the control sites reflected those seen in the impact sites, despite the control sites remaining independent of the influence of the compensation release from Grimwith reservoir, suggesting that exogenous factors such as climate operating at a regional scale are the main factors influencing brown trout recruitment. During the “after” period, there was a general decrease in 0+ brown trout population densities across all control sites, with 0+ brown trout absent from four sites in 2016 (WR1, WR3, WR4 and WR8). At the remaining sites (WR2, WR5-WR7 and WR9 in 2016 and all sites in 2015) densities during the after period ranged from 0.6 – 20.7 fish/100 m ^{2}, with only one site classed as good, and the majority classed as either poor or fair/poor (Table 4.5). It should be noted that there was a significant decline ($P < 0.05$) in the pooled densities from 2015 to 2016 as 0+ brown trout were absent from four sites in 2016; this phenomenon was not observed in the impact sites. It is worth noting that the variability of 0+ brown trout densities is largely manifested temporally within sites. This is to say that there was no clear trend of some sites yielding continually higher 0+ brown trout populations than others (Table 4.5) as many sites showed considerable increases and decreases in 0+ densities over a short time period, which is likely a result of the inherent population variability that persists in many salmonid communities.

Table 4.5. Density estimates ± 95% C.L. of 0+ brown trout at impact and control sites during both the before (2012 – 2014) and after (2015 – 2016) periods. Colours denote EA-FCS abundance classification

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

River Name	Site ID	2012	2013	2014	2015	2016
		Before			After	
impact Sites						
River Dibb	DB1	0.00 ± 0.0	0.72 ± 0.3	3.31 ± 0.2	1.41 ± 0.3	0.43 ± 0
River Dibb	DB2	0.51 ± 0l0	3.23 ± 1.3	5.4 ± 0.7	0.63 ± 0	0.82 ± 0
River Dibb	DB3	0.45 ± 0.2	2.92 ± 0.0	5.23 ± 0.1	2.51 ± 0.5	4.98 ± 0.3
River Dibb	DB4	0.00 ± 0.0	1.25 ± 0.2	3.39 ± 0.4	0.00 ± 0.0	0.00 ± 0.0
River Dibb	DB5	0.45 ± 0.0	1.56 ± 0.4	6.52 ± 1.7	3.72 ± 0.3	1.03 ± 0.4
River Dibb	DB6	0.24 ± 0.0	6.16 ± 0.5	6.58 ± 1	3.06 ± 0.5	1.5 ± 0.1
Control Sites						
Barden Beck	WR1	1.48 ± 0.2	2.68 ± 0.4	27.53 ± 2	5.49 ± 0.9	0.00 ± 0.0
Barden Beck	WR2	3.02 ± 0.2	8.96 ± 0.5	2.98 ± 0.5	11.21 ± 0.3	0.6 ± 0.0
Ings Beck	WR3	3.88 ± 0.8	16.07 ± 0.4	13.78 ± 1.6	20.74 ± 0.9	0.00 ± 0.0
Ings Beck	WR4	1.5 ± 0.4	10.75 ± 1.0	1.67 ± 0.5	4.01 ± 0.5	0.00 ± 0.0
Ashfold Side beck	WR5	0.93 ± 0.4	3.99 ± 0.7	22.75 ± 2.3	2.5 ± 0.4	2.67 ± 0.4
Ashfold Side beck	WR6	2.23 ± 0.8	2.59 ± 0.1	47.02 ± 2.6	5.35 ± 1	3.55 ± 2.1
Ashfold Side beck	WR7	1.92 ± 0.0	10.19 ± 0.8	35.5 ± 2.3	7.26 ± 1.6	5.39 ± 1.7
Ashfold Side beck	WR8	0.97 ± 0.4	19.06 ± 1.4	2.75 ± 0.2	4.3 ± 0.4	0.00 ± 0.0
Ashfold Side beck	WR9	1.81 ± 0.0	5.9 ± 0.8	11.97 ± 0.5	1.33 ± 0.1	1.57 ± 0.1

Table 4.6 Actual Vx for *before* period and Target Vx to detect a ≥50% change in 0+ brown trout densities. **Bold and underlined** signifies that significant change to 0+ brown trout densities could be detected within the study parameters.

Actual Vx	Target Vx for ≥50% Increase	Target Vx for ≥50% Decrease
0.085	**<u>0.272</u>**	0.030

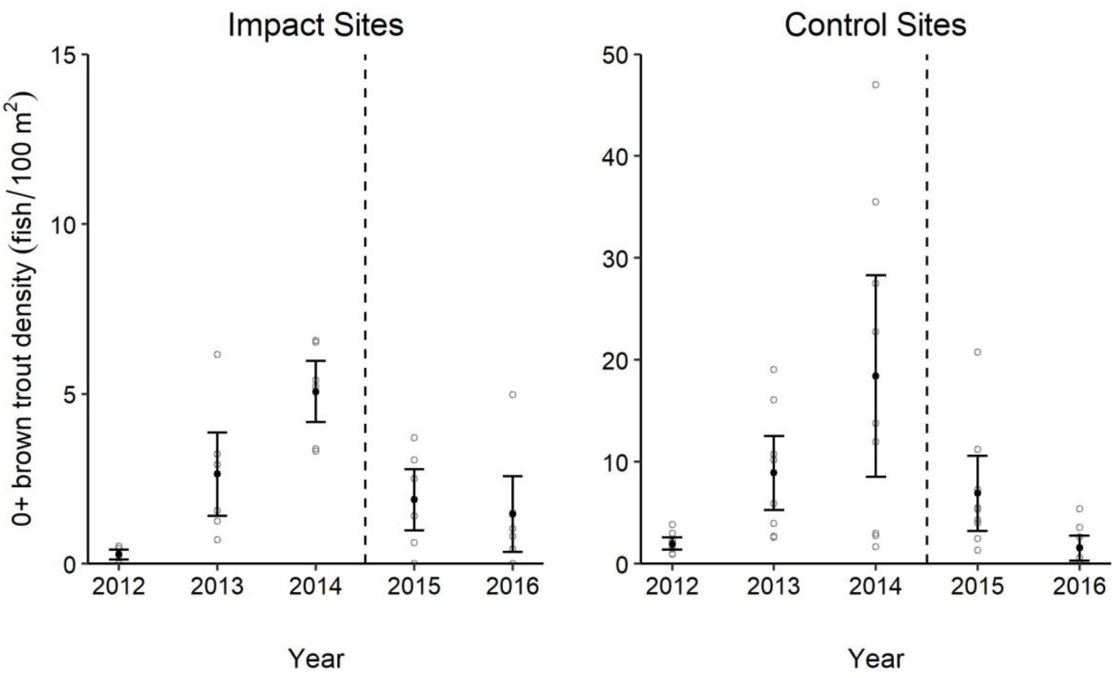

Figure 4.3. Average 0+ brown trout densities ± 95 % C.L. for all impact and control sites, throughout the study period. Dashed line represents the division between *before* and *after* periods. Greyed circles represent individual density estimates

Impact assessment (BACI)

Residual variance from both periods (before and after) were incorporated into a diagnostic resource calculation, which confirmed that a ≥50% increase in 0+ brown trout densities would be detectable given the variance in the current dataset, but identifying a ≥50% decrease to 0+ brown trout densities would not be achievable without further year's data collection (Table 4.7).

Table 4.7. Actual Vx for full study and Target Vx to detect a ≥50% change to 0+ brown trout densities. **Bold and underlined** signifies that significant change to 0+ densities could be detected within the study parameters.

Actual Vx	Target Vx for ≥50% Increase	Target Vx for ≥50% Decrease
0.086	**<u>0.272</u>**	0.030

0+ brown trout densities were lower during the flow trial in both impact and control sites (Figure 4.4), but not significantly (Table 4.8), and there was a significant difference between the 0+ brown trout densities in the impact and control sites (Figure 4.4) (Table 4.8). Whilst there was a decrease in 0+ densities in the River Dibb following the flow trial, the interaction between Area and Period was not significant (Table 4.8), indicating there was no significant change in 0+ brown trout densities in the River Dibb following the introduction of the compensation flow in 2015.

Table 4.8 BACI GLMM to detect a change in 0+ brown trout density following the introduction of the compensation flow from Grimwith Reservoir in 2015

	Sum Sq	Mean Sq	DoF	DenDF	F	P
Area	4.580	4.580	1	13.689	12.227	**<u>0.004</u>**
Period	0.1673	0.1673	1	3.012	0.447	0.551
Area:Period	0.696	0.696	1	55.000	1.8572	0.189

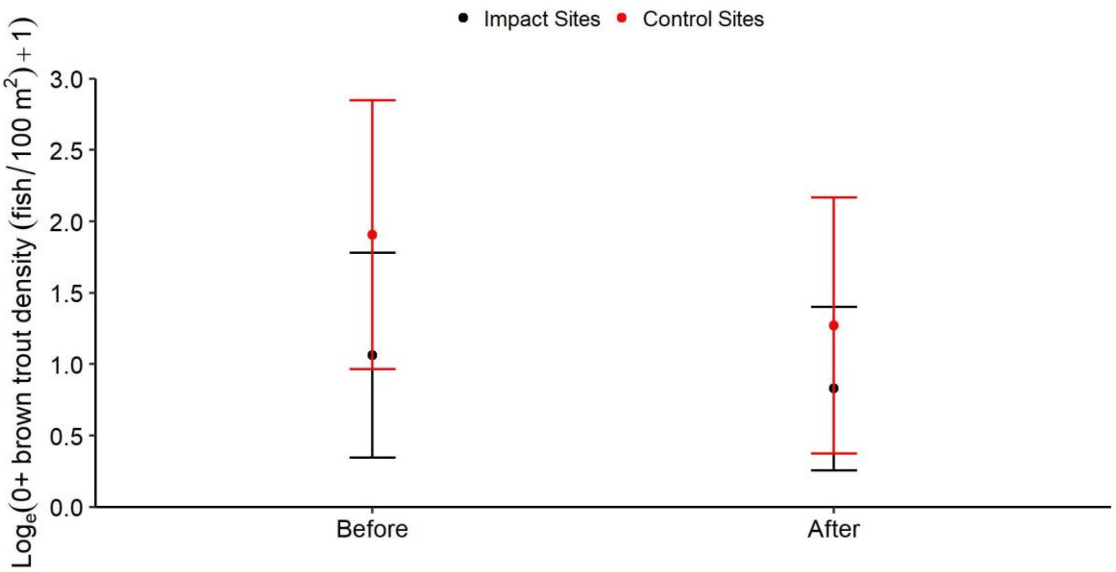

Figure 4.4. Mean (±95 %) 0+ brown trout densities before and after for impact (black) and control (red) sites. All densities are ln+1 transformed.

4.3.2 Impact of the introduction of a compensation flow on ≥1+ brown trout densities

To allow for comparison with HQS densities the ≥1+ brown trout densities are presented in two size based categories (≥1+ under 20cm, ≥1+ greater than 20 cm); ≥1+ brown trout (all sizes) densities and their respective EA-FCS classification are reported in the Appendix (Table A1) ≥1+ brown trout were caught at all sites in all years during this study, and were typically found in higher densities than juvenile conspecifics (appendix Table A.1). Brown trout were typically smaller than 20 cm in the ≥1+ age class, with this size class caught in significantly higher densities at all sites throughout the study (Figure 4.5, P <0.05). Prior to the flow trial ≥1+ (< 20 cm) brown trout densities varied between sites, but not significantly ($F_{5,12}$ = 0.641, P >0.05), but did vary significantly between years ($F_{1,16}$= 16.05, P< 0.05), ranging from 1.96 to 12.71 fish/100 m^2 across impact sites during the before period (Table 4.9). During the before period ≥1+ (< 20 cm) brown trout population densities improved year on year with pooled densities in 2013 significantly larger ($P < 0.05$) than the previous year during the before period (2012) (Figure 4.5). There was no significant difference spatially ($F_{5,12} = 1.575$, $P > 0.05$), but ≥1+ (> 20 cm) brown trout did significantly vary temporally ($F_{1,16} = 10.43$, $P < 0.05$). This size category was absent from site DB6 in 2012, densities at the remaining sites in 2012 and 2013 ranged from 0.67 to 4.04 fish/100 m^2 (Table 4.10).

≥1+ brown trout were present at all control sites during the before period and populations were more varied across the control sites compared to the impact sites, with total densities ranging from 0.39 to 29.76 fish /100 m^2 during the before period (appendix, Table A.1) and EA FCS classifications ranging from poor to excellent. Similar to the impact sites, ≥1+ (< 20 cm) brown trout dominated the ≥1+ age class (Figure 4.5) with densities ranging from 0.39 to 29.17 fish/100 m^2 (Table 4.9) which was significantly higher ($P < 0.05$) than the ≥1+ (> 20cm) brown trout that were absent from WR5 and WR6 in 2012, WR6 and WR7 in 2013, WR2 – WR4 in 2014 and WR8 throughout the before period. ≥1+ (> 20cm) brown trout densities at the remaining sites ranged from 0.36 to 2.99 fish/100 m^2 (Table 4.10). The three different rivers that comprise the control sites held significantly different ≥1+ brown trout populations ($F_{2,24} = 4.482$, $P < 0.05$), however, there was no significant difference temporally between of the three control rivers ($F_{1,25} = 0.211$, $P > 0.05$) suggesting that whilst each river supported significantly different ≥1+ (< 20 cm) brown trout densities there was no significant difference between years in the three rivers, i.e. the change in densities of ≥1+ brown trout in Barden Beck in 2012 to 2013 was not significantly different to the change seen in densities of ≥1+ brown trout in Ashfold Side Beck in 2012 to 2013. There was little variability within the ≥1+ (> 20 cm) brown trout at control sites with no significant difference between densities across years ($F_{1,25} = 0.646$, $P > 0.05$) or between the different rivers ($F_{2,24} = 0.059$, $P > 0.05$).

Before the commencement of the flow trial in 2015 a resource calculation was performed on the “before” ≥1+ brown trout population estimates: actual variance during this period (AVx) was lower than the target variance (TVx) (Table 4.11). Therefore, it was concluded, under the assumption that temporal variability seen in the impact and control sites persists, a ≥50% change to ≥1+ (< 20 cm) brown trout densities would be detectable following the introduction of a flow trial. Due to the low densities of ≥1+ brown trout (> 20 cm) in the control sites, further monitoring would be required before a ≥50 % decrease would be detectable, but a ≥50% increase to brown trout (> 20 cm) at the impact sites would be detectable within the timeframe of this study.

During compensation release (After)

≥1+brown trout were captured at all impact sites following the introduction of the compensation release in 2015. During this period densities ranged from 5.25 to 17.22 fish /100 m^2 and ≥1+ brown trout densities were classed as good at sites DB1 and DB2 and average at sites DB3-DB6 in 2015 and 2016. Despite a marginal decline in the ≥1+ (< 20 cm) brown trout densities in 2016 from 2015, there was no significant difference between ≥1+ (< 20 cm) brown trout densities between the two years ($F_{1,10}$ = 1.14, P> 0.05) or between the impact sites ($F_{5,6}$ = 2.848, P >0.05), suggesting ≥1+ (< 20 cm) brown trout populations were relatively stable following the introduction of the flow trial (Figure 4.5). Densities of this size category ranged from 4.12 to 13.7 fish/100 m^2 (Table 4.9). Similar to the before surveys, densities of ≥1+ (> 20 cm) brown trout were significantly lower than ≥1+ (< 20 cm) brown trout (P < 0.05). This size category was absent at site DB6 in 2014 and present in densities ranging from 0.26 to 9.04 fish /100 m^2 at the remaining sites in both years (Table 4.10). As seen with ≥1+ (< 20 cm) brown trout, there was no significant difference between densities both temporally ($F_{1,10}$ = 0.174, P > 0.05) or spatially ($F_{5,6}$ = 3.32, P >0.05), again suggesting a relatively stable population.

≥1+ brown trout densities were again more varied across the control sites compared to the impact sites during the after period. With ≥1+ brown trout densities ranging from 0.48 to 41.71 fish/100 m^2, population classifications during this period were either classed as poor, average, good or excellent (appendix, Table A.1). ≥1+ (< 20 cm) brown trout again dominated catches with densities of this age class significantly larger than ≥1+ (> 20 cm) brown trout (Table 4.9). Similar to observations at the impact sites, there was a non-significant marginal decrease in pooled ≥1+ (< 20 cm) brown trout at the control sites from 2015 to 2016 (Figure 4.5). There was a significant difference between the ≥1+ (< 20 cm) brown trout densities at the different control sites ($F_{8,9}$ = 6.51, P < 0.05), but there was no significant difference between pooled density estimates for each river ($F_{2,15}$ = 0.24, P>0.05). This suggests that spatial variance was operating at a site level as opposed to river level. ≥1+ (> 20 cm) brown trout were absent from populations at sites WR1-WR4 and WR8 in 2015 and sites WR2 – WR6 in 2016. At the remaining sites densities ranged from 0.40 to 1.88 fish/100 m^2 (Table 4.10). There was significant spatial (F = 3.917, P < 0.05) variability but no significant temporal ($F_{1,16}$= 0.176, P > 0.05) variation of ≥1+ brown trout of this size class (Figure 4.5).

Table 4.9. Density estimates ± 95% C.L. of ≥1+ brown trout (< 20 cm) at impact and control sites during both the before (2012 – 2014) and after (2015 – 2016) periods.

River Name	Site ID	2012	2013	2014	2015	2016
		Before			After	
impact Sites						
River Dibb	DB1	1.41 ± 0.4	5.02 ± 0.5	12.5 ± 0.6	13.7 ± 1.1	9.47 ± 1.5
River Dibb	DB2	2.31 ± 0.4	9.7 ± 0.8	12.08 ± 0.2	13.45 ± 0.6	11.84 ± 0.8
River Dibb	DB3	2.46 ± 0.3	1.33 ± 0.2	6.53 ± 0.4	5.03 ± 0.3	8.66 ± 0.9
River Dibb	DB4	1.51 ± 0	9.17 ± 2.8	12.71 ± 0.6	9.39 ± 0.6	4.63 ± 1.4
River Dibb	DB5	2.95 ± 0.1	2.5 ± 0.1	12.11 ± 1.5	9.46 ± 1.2	7.19 ± 0.4
River Dibb	DB6	1.96 ± 0.2	3.89 ± 0.6	5.64 ± 0.3	6.81 ± 1.3	4.12 ± 1.7
Control Sites						
Barden Beck	WR1	9.84 ± 1.7	9.65 ± 0.2	13.43 ± 0.7	12.8 ± 1.4	7.79 ± 0.6
Barden Beck	WR2	5.17 ± 0.1	5.8 ± 0.1	3.35 ± 0.3	5.14 ± 0.4	6.41 ± 0
Ings Beck	WR3	4.74 ± 0.6	9.06 ± 0.5	10.2 ± 0	7.45 ± 0.1	0.48 ± 0
Ings Beck	WR4	3.5 ± 0.4	3.96 ± 0.3	1.11 ± 0.3	1.72 ± 0.2	1.71 ± 0
Ashfold Side beck	WR5	14.02 ± 1	11.16 ± 2.1	22.16 ± 2.1	32 ± 0.7	22.99 ± 1.1
Ashfold Side beck	WR6	23.46 ± 2	7.28 ± 0.6	29.17 ± 0.2	41.71 ± 3.9	21.3 ± 1.8
Ashfold Side beck	WR7	17.69 ± 0.4	7.9 ± 0.4	21.12 ± 0.6	19.35 ± 0.9	18.42 ± 1.6
Ashfold Side beck	WR8	8.71 ± 1.7	9.44 ± 0.4	0.39 ± 0	9.32 ± 0.7	13.3 ± 0.9
Ashfold Side beck	WR9	11.59 ± 1.1	9.76 ± 0	11.61 ± 0.5	11.96 ± 0.3	9.09 ± 0.5

Table 4.10 Density estimates ± 95% C.L. of ≥1+ brown trout (> 20 cm) at impact and control sites during both the before (2012 – 2014) and after (2015 – 2016) periods.

River Name	Site ID	2012	2013	2014	2015	2016
		Before			**After**	
impact Sites						
River Dibb	DB1	1.69 ± 0.2	1.08 ± 0	0 ± 0	3.51 ± 0.3	9.04 ± 2.6
River Dibb	DB2	0.77 ± 0.1	1.85 ± 0.2	0 ± 0	2.82 ± 0.1	2.04 ± 0.4
River Dibb	DB3	0.67 ± 0.3	0.8 ± 0	0 ± 0	1.26 ± 0.1	0.26 ± 0.2
River Dibb	DB4	1.21 ± 0.2	3.75 ± 0.3	0 ± 0	2.04 ± 0.3	0.84 ± 0.2
River Dibb	DB5	0.68 ± 0.3	0.94 ± 0.3	0 ± 0	0.68 ± 0.2	0.68 ± 0
River Dibb	DB6	0 ± 0	1.3 ± 0	0 ± 0	0 ± 0	1.12 ± 0.1
Control Sites						
Barden Beck	WR1	0.98 ± 0	0.54 ± 0.3	1.34 ± 0	0 ± 0	0.65 ± 0.4
Barden Beck	WR2	0.43 ± 0	0.53 ± 0	0 ± 0	0 ± 0	0 ± 0
Ings Beck	WR3	0.86 ± 0.4	0.41 ± 0	0 ± 0	0 ± 0	0 ± 0
Ings Beck	WR4	1 ± 0	0.57 ± 0	0 ± 0	0 ± 0	0 ± 0
Ashfold Side beck	WR5	0 ± 0	0.4 ± 0	2.99 ± 0	0.5 ± 0	0 ± 0
Ashfold Side beck	WR6	0 ± 0	0 ± 0	0.6 ± 0	0.53 ± 0	0 ± 0
Ashfold Side beck	WR7	0.38 ± 0	0 ± 0	0.45 ± 0	0.4 ± 0	0.45 ± 0
Ashfold Side beck	WR8	0 ± 0	0 ± 0	0 ± 0	0 ± 0	0.4 ± 0
Ashfold Side beck	WR9	0.36 ± 0	1.56 ± 0	1.09 ± 0	1 ± 0.1	1.88 ± 0.8

Table 4.11. Actual Vx for *before* period and Target Vx to detect a ≥50% change to ≥1+ brown trout densities. **Bold and underlined** signifies that significant change to ≥1+ densities could be detected within the study parameters.

	Actual VX	Target Vx for ≥50% Increase	Target Vx for ≥50% Decrease
< 20 cm before	0.051	**0.722**	**0.080**
> 20 cm before	0.038	**0.058**	0.001

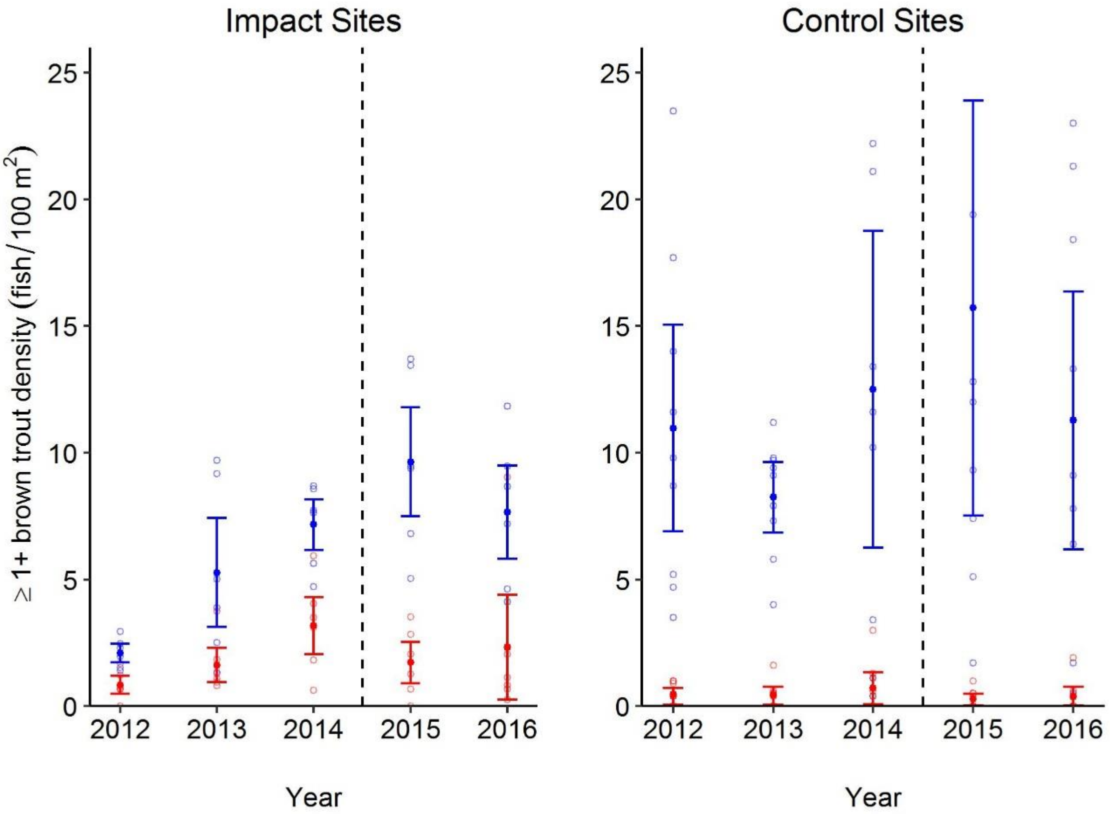

Figure 4.5 Average ≥1+ brown trout densities for all impact sites and control sites. ≥1+ (< 20 cm) data presented in blue, and ≥1+ (> 20 cm) in red. Dashed line represents the division between before and after periods. Hollow circles represent individual data points.

Impact assessment (BACI)

Residual variance from both periods (before and after) were incorporated into a diagnostic resource calculation, which confirmed that identifying a ≥50% change to ≥1+ (< 20 cm) brown trout would be achievable within the timeframe of this study. For the ≥1+ (> 20 cm) brown trout, inclusion of the after dataset increased the level of actual variance therefore any increase or decrease detected to this size class in the impact sites cannot be confidently determined as resulting from the introduction of a compensation release (Table 4.12).

Table 4.12. Actual Vx for the combined before and after periods and Target Vx to detect a ≥50% change to ≥1+ brown trout densities. **Bold and underlined** signifies that significant change to ≥1+ densities could be detected within the study parameters.

	Actual VX	Target Vx for ≥50% Increase	Target Vx for ≥50% Decrease
< 20 cm	0.042	**0.722**	**0.080**
> 20 cm	0.110	0.058	0.001

The BACI GLMM found no significant difference between the densities of ≥1+ (< 20 cm) brown trout between the impact and control sites or the before or after period, the interaction between area and period in the GLMM was also non-significant (Table 4.13), therefore the introduction of the compensation flow has not had any appreciable influence on ≥1+ (< 20 cm) brown trout densities. Despite the low densities it was possible to detect a marginal increase in the ≥1+ (> 20 cm) brown trout densities at the impact sites and a marginal decrease at the control sites (Figure 4.7). The BACI GLMM revealed that the ≥1+ (> 20 cm) brown trout population at impact sites was significantly different from those populations in the control sites (Table 4.13). The interaction between area and period in the GLMM was not significant (Table 4.13), suggesting that there was no change to the ≥1+ (> 20 cm) brown trout densities as a result of the flow trial.

Table 4.13 BACI GLMM to detect a change in ≥1+ brown trout densities following the introduction of a compensation flow from Grimwith Reservoir in 2015. **Bold and underlined** signifies a significant (P<0.05) interaction in the GLMM

	Sum Sq	Mean Sq	DoF	DenDF	F	P
≥1+ (< 20 cm)						
Area	0.461	0.461	1	13.191	1.878	0.193
Period	1.236	1.236	1	3.147	5.0342	0.106
Area:Period	0.932	0.932	1	55	3.797	0.056
≥1+ (> 20 cm)						
Area	2.048	2.048	1	13.321	0.373	**0.001**
Period	0.047	0.047	1	3.097	0.373	0.583
Area:Period	0.046	0.046	1	55.000	0.363	0.549

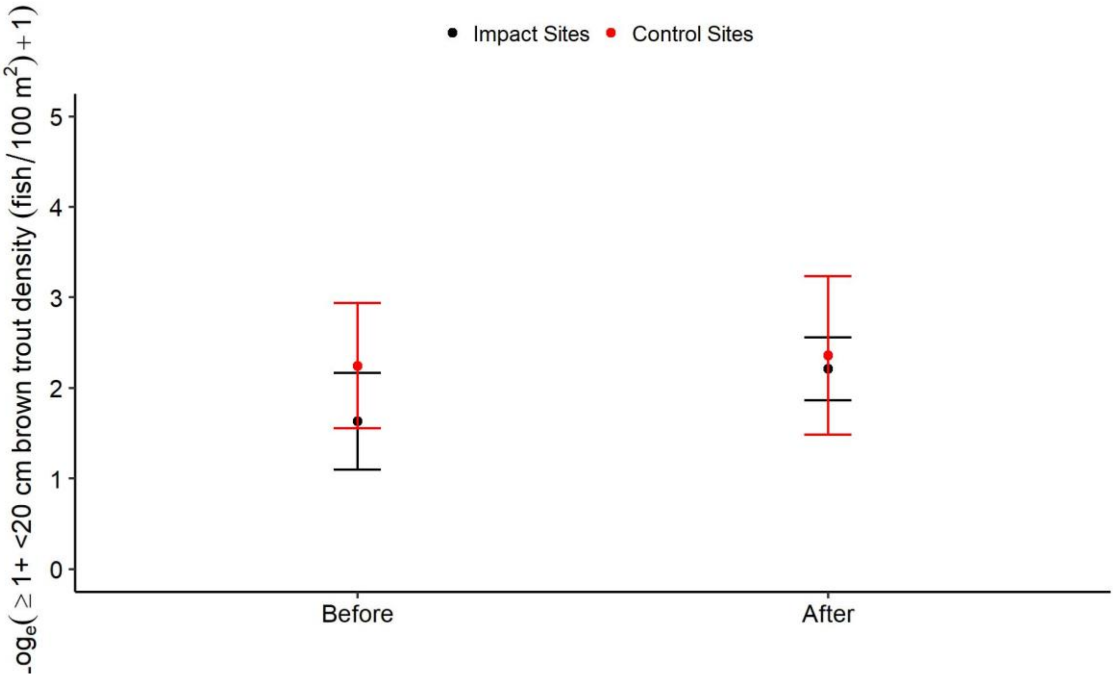

Figure 4.6. Mean (±95%) ≥1+ (< 20 cm) brown trout densities for the before and after periods for impact (black) and control (red) sites.

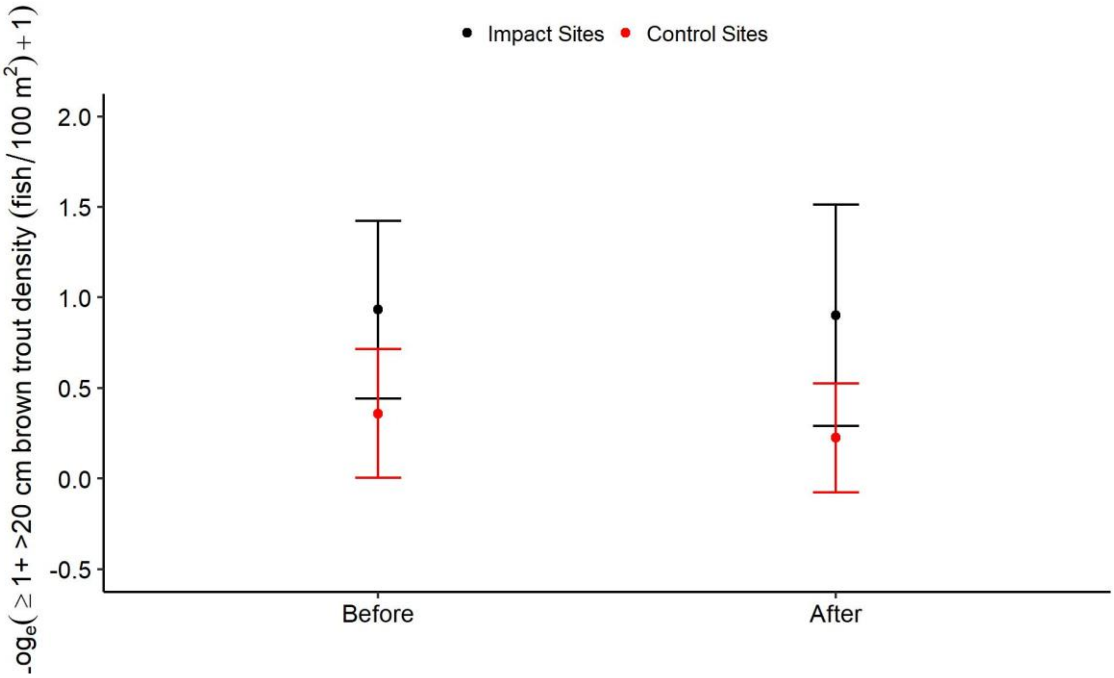

Figure 4.7 Mean (±95%) ≥1+ (> 20 cm) brown trout densities for the before and after periods for impact (black) and control (red) sites.

4.3.3 Impacts of the introduction of a compensation release on the length at capture of 0+ brown trout

Baseline monitoring (Before)

The length at which 0+ brown trout were captured varied considerably both spatially and temporally. Before the introduction of the flow trial 0+ brown trout were captured in the size range of 62 to 110 mm at impact sites. During this period, the average length of 0+ brown trout displayed sizeable temporal variability, with no significant spatial ($F_{5,10}$ = 1.545, P >0.05) or temporal ($F_{1,14}$ = 0.05, P > 0.05) variance exhibited at the impact sites. However, pooled length at capture for 0+ brown trout in 2013 was significantly smaller than 2012 and 2014 (P <0.05) (Figure 4.9).

0+ brown trout were caught in a much broader size range across the control sites compared to impact sites with length ranging from 44 to 103 mm, but length at capture between the impact and control sites was not significantly different (Figure 4.8, P > 0.05). There was a similar trend during the before period in the control sites with the average length at capture of brown trout in 2013 lower than other years, but there was no significant difference in length at capture at control sites in the years between 2012 and 2014 inclusive ($F_{1,25}$ = 0.055, P > 0.05) (Figure 4.8). Size ranges for all sites in all years are found in Appendix, Table A.2.

Prior to the flow trial, a resource calculation was performed on the baseline data to test if a ≥25% change in 0+ brown trout length at capture could be detected. The target variance was significantly lower than the actual variance, therefore given the persistence of the current level of temporal variability, a ≥25% change to brown trout length at capture would be detectable (Table 4.14).

Table 4.14. Actual Vx for the before period and Target Vx to detect a ≥25% change to 0+ brown trout lengths. **Bold and underlined** signifies that significant change to 0+ brown trout lengths could be detected within the study parameters

Actual Vx	Target Vx for ≥25% Increase	Target Vx for ≥25% Decrease
12.221	**1603.189**	**178.132**

During compensation release (After)

During the flow trial, 0+ brown trout were caught in the size range 60 to 102 mm, the average length of 0+ brown trout in 2015 was lower than in 2014 and 2016, with all fish in this year caught in a size range 60-83 mm (Figure 4.8). Lengths of 0+ brown trout were significantly smaller in 2015 than in 2016 at impact sites ($F_{1,8,}$= 5.942, P<0.05), but there were no significant differences in 0+ brown trout length at capture between impact sites ($F_{4,5}$=0.684, P>0.05)

0+ brown trout length at capture at control sites during the after period, were still highly varied amongst sites within the size range 49 – 103 mm (Figure 4.8), but there was no significant variation amongst control sites ($F_{5,8}$=3.052, P>0.05) or between years ($F_{1,12}$=0.017, P>0.05). Size ranges for individual sites and years are found in Appendix Table A.2

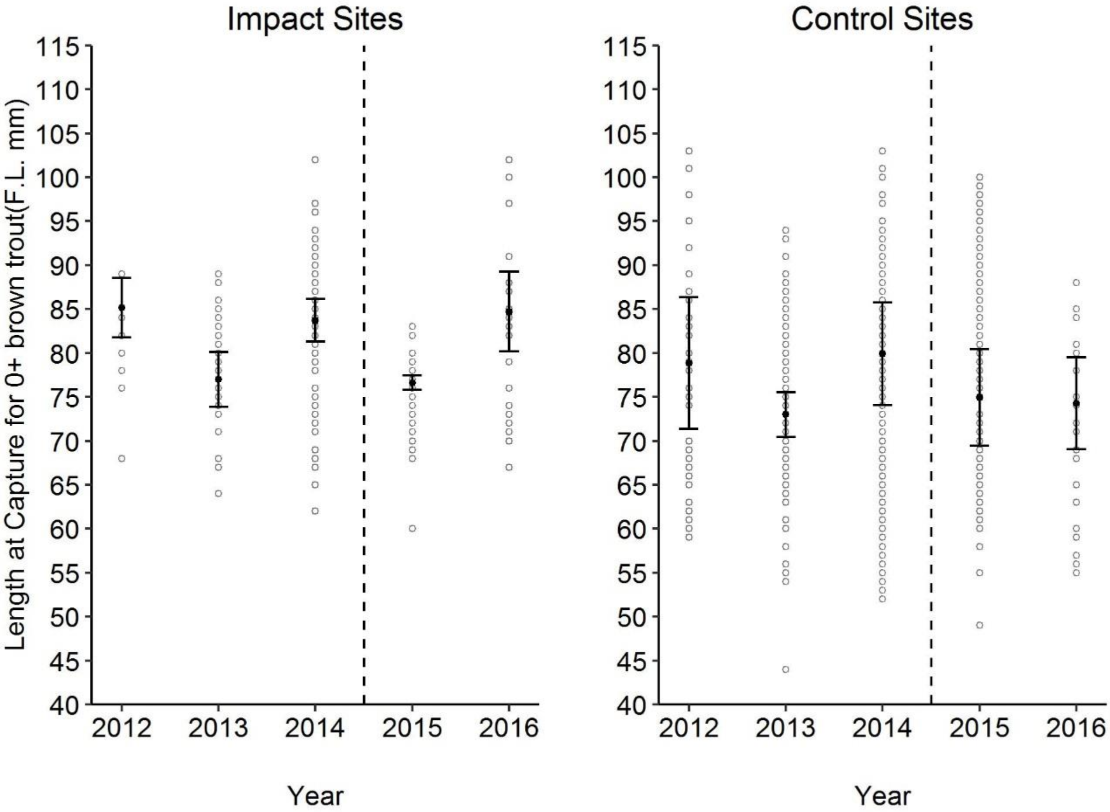

Figure 4.8. Lengths at capture for individual 0+ brown trout (greyed circles). As well as average length (± 95% C.L.) for impact and control sites. Dashed line represents division between before and after period

Impact assessment (BACI)

The resource calculation performed on the full data set confirmed that a 50% change to length at capture for 0+ brown trout would be detectable within the timeframe of this study (Table 4.15).

Table 4.15. Actual Vx for the full study and Target Vx to detect a ≥25% change to 0+ brown trout lengths. **<u>Bold and underlined</u>** signifies that significant change to 0+ lengths could be detected within the study parameters

Actual Vx	Target Vx for ≥25% Increase	Target Vx for ≥25% Decrease
2.503	**<u>1603.189</u>**	**<u>178.132</u>**

There was no significant difference between the impact and control sites, or the before and after period (Table 4.16). There was no significant interaction between Area and Period in the BACI GLMM (Table 4.16), therefore there was no significant change in length at capture of 0+ brown trout that can be attributed to the introduction of a flow trial in 2015.

Table 4.16 BACI GLMM to detect a change in 0+ brown trout length at capture following introduction of compensation release from Grimwith reservoir in 2015

	Sum Sq	Mean Sq	DoF	DenDF	F	P
Area	79.384	79.384	1	13.424	2.211	0.160
Period	2.225	2.225	1	3.158	0.062	0.819
Area:Period	0.100	0.100	1	48.243	0.003	0.958

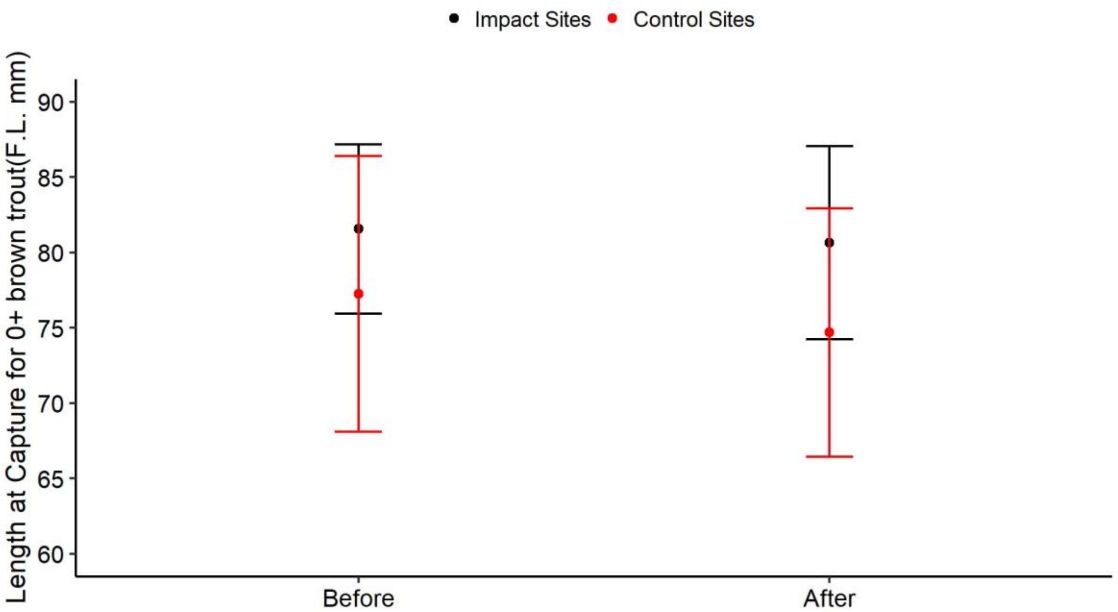

Figure 4.9. Mean (±95%) 0+ brown trout lengths *before* and *after* for impact (black) and control (red) sites.

4.3.4 Impacts of the introduction of a compensation release on the habitat quality (HQS density) for 0+ brown trout

Baseline Monitoring (Before)

Habitat Quality Score (HQS) varied between sites and years in both the impact and control sites during the before period. In the River Dibb, 0+ HQS densities were lower in 2012 than would be expected under non-impacted conditions and were classed as fair/poor (Table 4.17). Habitat quality for brown trout improved in 2013 and 2014 (Table 4.17), with densities expected under non-impacted conditions typically classified as average, with the exception of site DB3 where the habitat still remained fair/poor (Table 4.17). A significant difference in HQS densities was found between the three years of the before period at the impact sites ($F_{1,16} = 25.13$, $P < 0.05$), but during this period there was no significant variation between sites ($F_{5,12} = 0.647$, $P > 0.05$).

The Habitat Utilisation Index (HUI) revealed that 0+ brown trout densities were lower than predicted at all impact sites during the study (Figure 4.11). With densities at all sites in 2012, DB1, DB2, DB4 and DB5 in 2013, and DB1 and DB4 in 2014 significantly lower than the HQS prediction ($P < 0.05$).

At the control sites, 0+ brown trout habitats were typically better than those in the River Dibb as reflected by the higher HQS densities. However, there were significant differences between the three control rivers ($F_{2,25} = 6.77$, $P < 0.05$), but there were no significant differences between sites within rivers ($F_{8,18} = 1.937$, $P > 0.05$), suggesting that the higher HQS densities recorded at each site were indicative of the river. Unlike in the impact sites, there was no significant year to year variability of the 0+ brown trout HQS ($F_{1,25} = 1.36$, $P > 0.05$). The range of the HQS densities was greater in the control sites (Table 4.17) with population classifications under non-impacted conditions ranging from fair/poor to excellent (Table 4.17). Similar to the impact sites, the HUI revealed that observed densities of 0+ brown trout were typically lower than the HQS at the control sites during the before period. Observed densities were greater than the HQS at sites WR7 and WR8 in 2013 and WR3, WR5, WR6 and WR7 in 2014, but there were no instances when observed 0+ brown trout densities were significantly higher than predicated, although densities at sites WR1, WR4 WR6, WR7, WR8 and WR9 in 2012, WR1 and WR6 in 2013 and WR2, WR4 and WR8 in 2014 were all significantly lower than the HQS.

A resource calculation performed on the “before” data revealed that the intensity of sampling would be adequate to detect a 50% change to the HQS densities following an introduction of a flow trial. (Table 4.18). As target VX for both 50% increase and decrease was greater than Actual Vx.

During compensation release (After)

0+ brown trout HQS densities remained largely similar to previous years, with the 0+ brown trout population status expected under non-impacted conditions being predominantly average, with the exception of site DB3 in 2015, and DB2 in 2016, which were fair/poor, and site DB6 in 2016 where expected 0+ brown trout populations were good (Table 4.17). There was little variation either between sites or between years for 0+ brown trout HQS (Figure 4.10), with no significant differences found in HQS densities either between sites ($F_{5,6} = 2.02$, $P > 0.05$) or between years ($F_{1,10} = 0.44$, $P > 0.05$). The HUI suggested that the observed trout densities were lower than the HQS at all sites, with densities at all sites but DB3 significantly lower than predicted. (Figure 4.11). 0+ brown trout habitat quality at control sites varied, but there were no significant differences between HQS densities between sites ($F_{8,9}=1.296$, $P>0.05$), or between years ($F_{1,10}=0.453$, $P>0.05$). Observed densities of 0+ brown trout were lower than predicted at all control sites during the after period, with the exception of site WR3, where observed densities were marginally higher than predicted from the HQS densities (Figure 4.11).

Table 4.17. 0+ HQS densities related to the EA-FCS categories for impact and control sites during the before (2012 2014) and after (2015 – 2016) periods.

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

River Name	**Site ID**	**2012**	**2013**	**2014**	**2015**	**2016**
		Before			**After**	
impact Sites						
River Dibb	DB1	4.54	12.39	15.05	10.51	15.47
River Dibb	DB2	5.26	15.63	12.91	12.41	4.30
River Dibb	DB3	3.92	5.71	7.21	6.56	10.14
River Dibb	DB4	5.20	9.79	14.79	10.63	10.95
River Dibb	DB5	3.79	9.38	11.29	14.03	16.73
River Dibb	DB6	5.02	10.93	13.37	13.70	20.90
Control Sites						
Barden Beck	WR1	15.99	41.81	30.27	42.99	45.52
Barden Beck	WR2	11.12	32.06	14.49	24.72	10.26
Ings Beck	WR3	5.96	30.02	7.97	9.94	4.54
Ings Beck	WR4	11.20	14.91	16.12	15.58	13.78
Ashfold Side beck	WR5	8.67	6.16	12.45	11.30	11.51
Ashfold Side beck	WR6	10.75	9.23	12.87	11.19	17.39
Ashfold Side beck	WR7	11.27	10.62	14.37	8.75	15.98
Ashfold Side beck	WR8	7.82	15.07	13.17	10.79	11.96
Ashfold Side beck	WR9	12.18	14.74	15.35	9.94	13.41

Table 4.18. Actual Vx for the before period and Target Vx to detect a ≥50% change to 0+ HQS Densities. **Bold and underlined** signifies that significant change to 0+ HQS Densities could be detected within the study parameters

Actual Vx	Target Vx for ≥50% Increase	Target Vx for ≥50% Decrease
0.021	**<u>1.207</u>**	**<u>0.134</u>**

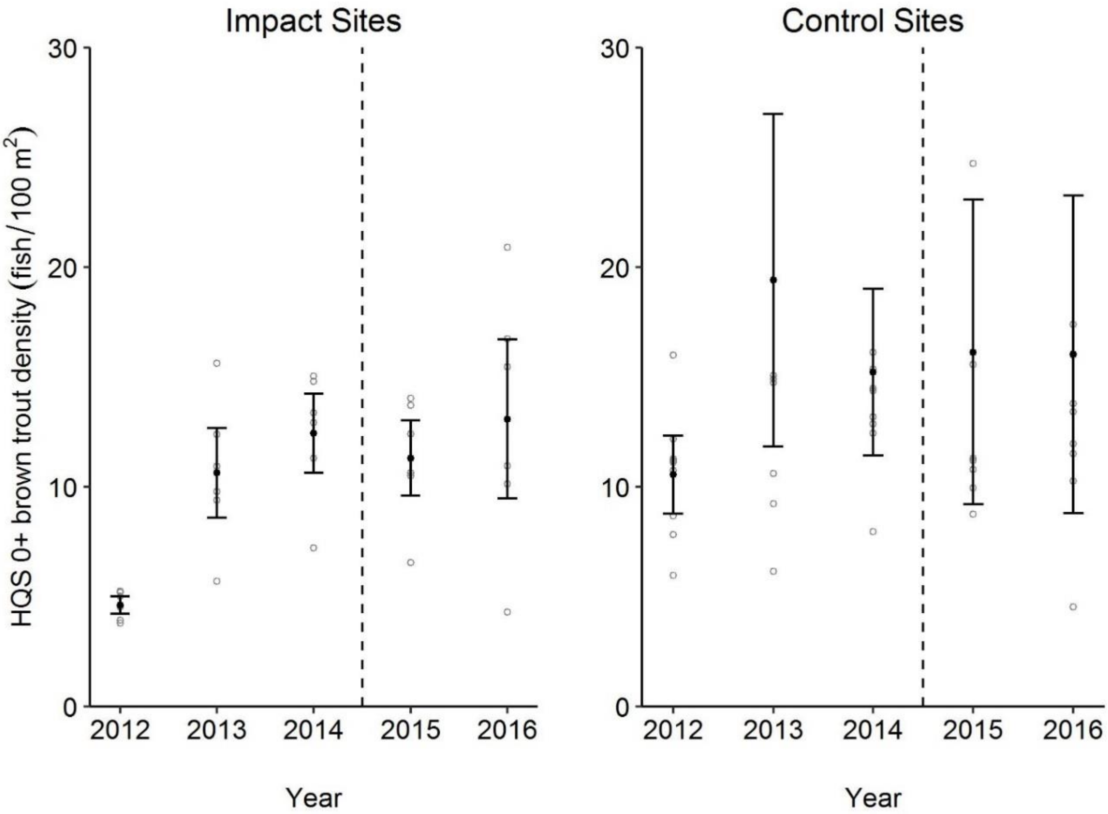

Figure 4.10. Average 0+ brown trout HQS densities ± 95 % C.L. for all impact and control sites, throughout the study period. Dashed line represents the division between *before* and *after* periods. Greyed circles represent individual density estimates

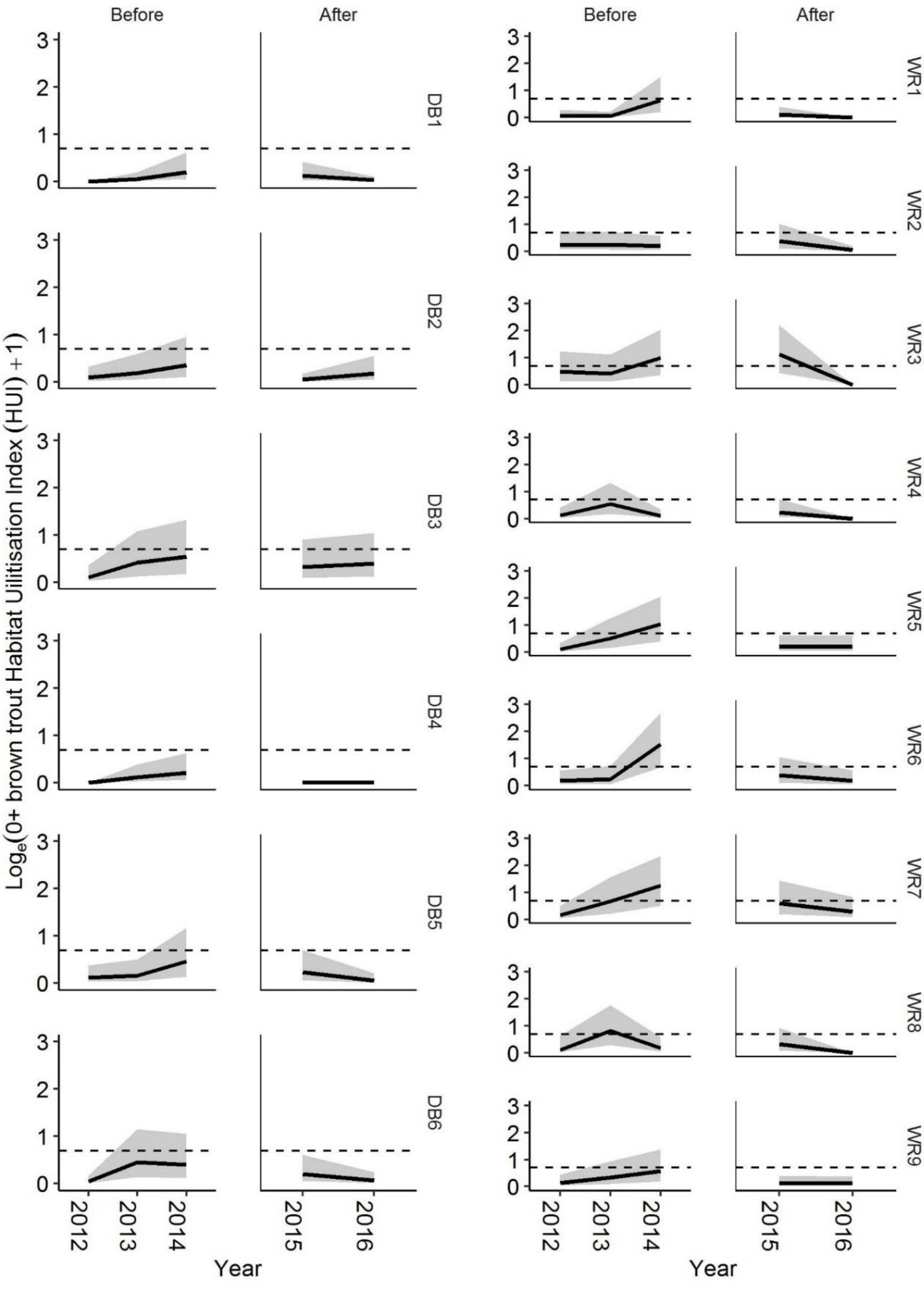

Figure 4.11. Loge + 1 transformed HUI for 0+ brown trout in the impact (left) and control sites (right) during the before (2012 – 2014) and after (2015 – 2016) period. Grey bands represent 95% confidence limits. Dashed line represents HUI value where observed and expected densities are equal.

Impact assessment (BACI)

A diagnostic resource calculation performed on the full data set revealed a marginal increase to the actual Vx present across the study sites, but crucially was lower than target Vx for both ≥50% increase and decrease in 0+ brown trout HQS densities.

Table 4.19. Actual Vx for the full study and Target Vx to detect a ≥50% change to 0+ HQS Densities. **Bold and underlined** signifies that significant change to 0+ HQS Densities could be detected within the study parameters

Actual Vx	Target Vx for ≥50% Increase	Target Vx for ≥50% Decrease
0.021	**<u>1.207</u>**	**<u>0.134</u>**

During the study there was a significant difference between 0+ brown trout HQS densities for the River Dibb and control sites (Table 4.20) despite there being a marginal decrease in 0+ brown trout HQS at the control sites and a marginal increase in the impact sites (Figure 4.12.). There was no significant effect of period (Table 4.20) in this study. The interaction between Area and Period in the GLMM was not significant (Table 4.20) therefore there was no change to the 0+ brown trout habitat in the River Dibb that could be attributed to the introduction of the compensation flow in 2015.

Table 4.20 BACI GLMM to detect a change in 0+ brown trout HQS density following the introduction of the compensation release from Grimwith reservoir in 2015

	Sum Sq	Mean Sq	DoF	DenDF	F	P
Area	0.601	0.601	1	13.396	4.706	**<u>0.049</u>**
Period	0.056	0.056	1	3.033	0.439	0.555
Area:Period	0.284	0.284	1	55.000	0.221	0.141

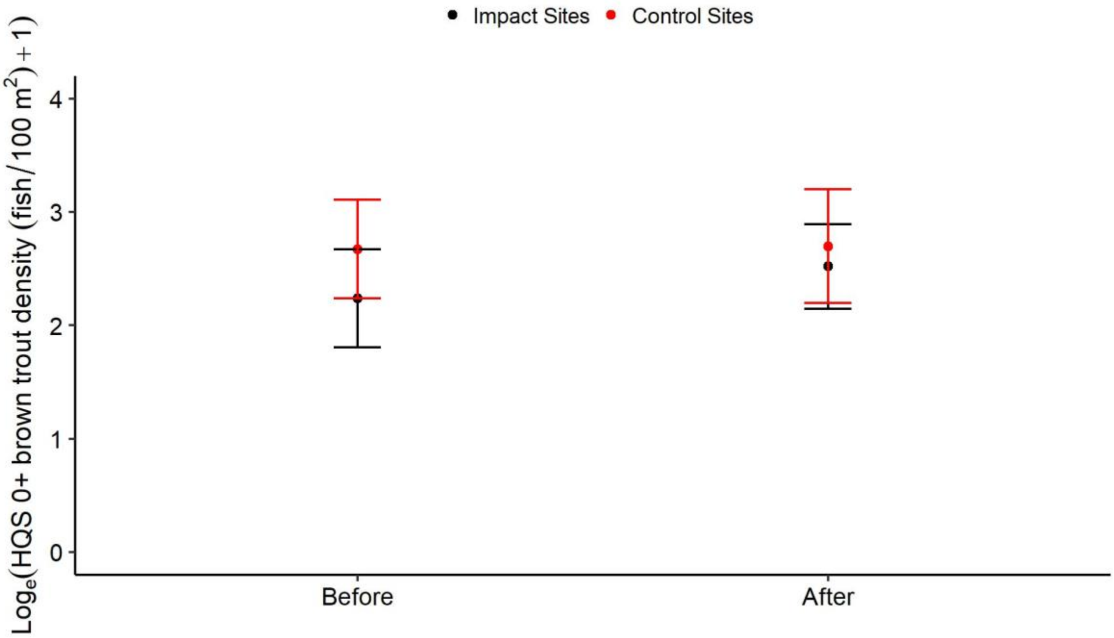

Figure 4.12. Mean (±95%) 0+ brown trout HQS densities during before and after periods for impact (black) and control (red) sites.

4.3.5 Impacts of the introduction of the compensation flow on the habitat quality (HABSCORE) for ≥ 1+ brown trout

Baseline monitoring (Before)

During the “before” period, HQS densities for ≥1+ brown trout varied between sites and years. For the impact sites expected densities of ≥1+ (< 20 cm) brown trout ranged from 0.68 to 8.04 fish/100 m^2 (Table 4.21). HQS densities typically displayed year to year increases during this period (Figure 4.13) with ≥1+ brown trout habitat quality significantly poorer in 2012 than other years during the before period ($P < 0.05$) (Figure 4.13). Despite the low HQS in 2012 the observed densities of ≥1+ (< 20 cm) brown trout were equal to or greater than those expected from the habitat quality, and in the case of site DB5 in 2012 and 2014 significantly greater (Figure 4.14). The habitat quality for ≥1+ (> 20 cm) brown trout was typically lower as indicated by lower HQS densities, which ranged from 0.27 to 1.9 fish/100 m^2 (Table 4.22). There was little variation in the ≥1+ (> 20 cm) brown trout HQS with no significant variation between impact sites ($F = P > 0.05$) or years ($F = 0.77$, $P > 0.05$), although pooled HQS for this size class did reveal a marginal decline from 2013 to 2014 (Figure 4.13). Despite the low number of expected ≥1+ (> 20 cm) brown trout densities, the HUI variables did fluctuate between sites and years. With observed densities at impact sites greater than predicted at all sites in 2012, with the exception of site DB4 and DB6, which were significantly lower than predicted. In 2013, densities at four sites (DB1, DB2, DB3 and DB5) were lower than predicted, of which sites DB3 and DB5 were significantly lower. The remaining two sites (DB4 and DB6) had higher than predicted ≥1+ (> 20 cm) brown trout densities with densities at site DB4 significantly higher. In 2014 the observed densities of ≥1+ (> 20 cm) brown trout were significantly lower than the HQS at all sites as this size class was absent from each brown trout population (Figure 4.15).

The expected densities for ≥1+ (< 20 cm) brown trout at control sites was typically better than predicted from the habitat with expected densities ranging 2.16 to 24.30 fish/100 m^2 during this period (Table 4.21); there was no significant difference in the ≥1+ (< 20 cm) brown trout HQS between the three study rivers ($F = 1.43$, $P > 0.05$). The observed densities of ≥1+ (< 20 cm) brown trout in the control sites were equal to or greater than the habitat quality at all control sites during the before period, with the exception of site WR2 in 2013 and 2014 and sites WR4 and WR8 in 2014 (Figure 4.14). The habitat quality for ≥1+ (> 20 cm) brown trout revealed that expected densities of ≥1+ (> 20 cm) brown trout would be typically lower than the < 20 cm conspecifics. There was no significant difference between ≥1+ (> 20 cm) brown trout HQS at control sites during the after period ($F = 2.68$, $P>0.05$) with densities ranging from 0.74 to 10.16 fish/100 m^2 (Table 4.22).

There were significant differences between the sites (F = 4.017, P<0.05) but no significant difference between the three rivers (F = 2.14, P> 0.05) suggesting habitat at individual sites were more suitable for ≥1+ (> 20 cm) brown trout opposed to habitat across entire rivers. The HUI revealed that ≥1+ (> 20 cm) brown trout densities were lower than predicted at all sites with the exception of site WR4 in 2012 and WR5 in 2015 (Figure 4.15). Of these sites where ≥1+ (> 20 cm) brown trout were lower than predicted, all were significantly lower (P>0.05) with the exception of WR1 in all years, WR2 in 2012 and 2013, WR3 in 2012 (Figure 4.15). In no instance during the before period was the observed densities of ≥1+ (< 20 cm) brown trout at the control sites significantly different from the HQS.

The resource calculation performed in 2014 revealed that the target variances for a ≥50% increase or decrease were greater than the level of actual variance in the dataset, therefore a ≥50% change would be detected within the timeframe of this study for ≥1+ (> 20 cm) brown trout (Table 4.23). However, due to the low densities of ≥1+ (> 20 cm) brown trout the level of actual variance in the data set was greater than the target variance to detect a ≥50% decrease. Therefore, if the current level of spatial and temporal variance persists a biologically relevant change to ≥1+ (> 20 cm) brown trout habitat quality would not be detected with statistical robustness within the timeframe of the study (Table 4.23).

During flow trial (After)

Habitat quality of ≥1+ (< 20 cm) brown trout at impact sites remained within the variability observed during the before period, although ≥1+ (< 20 cm) brown trout habitat quality was marginally lower in 2015 than 2013 and 2014, but still significantly higher than the poor habitat quality recorded in 2012 (P < 0.05) (Figure 4.13). Expected densities of ≥1+ (< 20 cm) brown trout at impact sites during the after period ranged from 0.99 to 7.85 fish/100 m^2 (Table 4.21). There were no significant differences in the HQS for ≥1+ (< 20 cm) brown trout resulting from spatial variability ($F_{5.12}$ = 2.181, P>0.05), but they did vary significantly temporally ($F_{1,16}$ = 5.611, P>0.05). The observed densities of ≥1+ (< 20 cm) brown trout were greater than predicted from the habitat quality at all sites in both years (2015 – 2016) with the exception of site DB6 where observed densities of ≥1+ (< 20 cm) brown trout were marginally, but not significantly, lower than predicted from the habitat. The HUI at site DB3 in 2015 indicated that observed ≥1+ (< 20 cm) brown trout densities were significantly greater than predicted (Figure 4.14). As seen during the before period at impact sites, habitat quality for ≥1+ (> 20 cm) brown trout was significantly lower (P<0.05) than for the ≥1+ conspecifics < 20 cm (Figure 4.13). Whilst there was site to site variation of the habitat quality for ≥1+ (> 20 cm) brown trout, this variability was not considered to be significant ($F_{5,12} = 0.19$, $P > 0.05$). Habitat quality for ≥1+ (> 20 cm) brown trout did improve significantly from 2015 to 2016 at the impact sites ($P<0.05$), with expected densities during this period ranging from 0.27 –1.53 fish/100 m^2 (Table 4.22). The degree of habitat utilization of ≥1+ (> 20 cm) brown trout varied from site to site (Figure 4.15), with ≥1+ (> 20 cm) brown trout densities significantly lower than predicted at sites DB3, DB5 and DB6 during the after period and at site DB2 in 2015. Of these, densities of ≥1+ (> 20 cm) brown trout were significantly lower than predicted at site DB2 in 2015, DB3 in 2016 and DB6 in 2015. At the remaining sites and years (DB1 in both years, DB2 in 2016 and DB4 in both years) ≥1+ (> 20 cm) brown trout densities were all significantly higher than predicted from the habitat quality ($P<0.05$) (Figure 4.15).

There was a marginal decrease in ≥1+ (< 20 cm) brown trout habitat quality at control sites in 2014 compared with previous years (2012-2014). During this period, there were no significant variations in habitat quality for ≥1+ (< 20 cm) brown trout manifested either spatially ($F_{8,9} = 2.878$, $P> 0.05$) or temporally ($F_{1,16} = 2.799$, $P>0.05$). HUI revealed that observed densities of ≥1+ (< 20 cm) brown trout were greater than expected at sites WR2, WR5-WR9 during both after years and site WR3 in 2015 (Figure 4.14). Of these sites the observed density of ≥1+ (< 20 cm) brown trout was significantly higher than predicted at sites WR5 – WR7 and WR9 (Figure 4.14) and at the remaining sites (WR1 and WR4 as well as WR3 in 2016) observed densities of ≥1+ (< 20 cm) brown trout were lower than predicted and in the case of site WR3 in 2016 significantly lower ($P<0.05$) (Figure 4.14). ≥1+ (> 20 cm) brown trout densities were typically lower at control sites during the after period, but not significantly so (Figure 4.13). The habitat quality for ≥1+ (> 20 cm) brown trout did not vary significantly between years ($F_{1,16} = 1.46$, $P >0.05$). HQS densities for ≥1+ (> 20 cm) brown trout varied significantly both between the three control rivers ($F_{2,18}=3.798$, $P<0.05$) and the sites within them ($F_{6,18}=4.090$, $P<0.05$). It can be assumed that the difference between sites is manifested in individual site variability as well as variability between the control rivers. The HUI revealed that observed ≥1+ (> 20 cm) brown trout densities at all sites were significantly lower ($P<0.05$) than predicted from the habitat quality with the exception of site WR9, where observed densities were marginally greater than predicted in both years (Figure 4.15).

Table 4.21 HQS densities for ≥1+ (<20cm) brown trout at all impact and control sites for the before (2012-2014) and after (2015-2016)

River Name	Site ID	2012	2013	2014	2015	2016
		Before			**After**	
impact Sites						
River Dibb	DB1	1.23	3.9	3.06	3.21	6.14
River Dibb	DB2	1.46	8.04	5.95	4.75	3.12
River Dibb	DB3	0.68	1.01	1.83	0.99	3.87
River Dibb	DB4	1.35	4.22	6.68	4.82	4.42
River Dibb	DB5	0.46	0.66	1.15	1.44	3.42
River Dibb	DB6	1.50	3.82	4.75	6.76	7.85
Control Sites						
Barden Beck	WR1	4.95	24.3	12.75	13.85	22.76
Barden Beck	WR2	2.22	6.7	4.81	3.62	3.34
Ings Beck	WR3	3.36	11.76	3.61	4.00	1.61
Ings Beck	WR4	2.16	4.8	3.50	2.89	3.14
Ashfold Side beck	WR5	8.34	5.35	9.00	4.75	13.23
Ashfold Side beck	WR6	14.12	5.91	8.71	4.65	13.35
Ashfold Side beck	WR7	8.74	5.09	5.02	2.65	9.08
Ashfold Side beck	WR8	3.18	5.83	4.4	2.20	6.81
Ashfold Side beck	WR9	3.42	6.78	3.75	2.08	5.75

Table 4.22. HQS densities for ≥1+ (>20cm) brown trout at all impact and control sites for the before (2012-2014) and after (2015-2016)

River Name	Site ID	2012	2013	2014	2015	2016
		Before			After	
impact Sites						
River Dibb	DB1	1.01	1.9	0.37	0.9	0.79
River Dibb	DB2	0.96	1.41	0.27	0.63	1.3
River Dibb	DB3	0.54	1.3	0.63	0.43	2.72
River Dibb	DB4	1.09	1.01	0.48	0.34	1.11
River Dibb	DB5	1.09	0.47	1.52	0.28	1.69
River Dibb	DB6	0.9	0.37	0.94	0.31	2.04
Control Sites						
Barden Beck	WR1	1.47	1.75	1.54	1.13	1.62
Barden Beck	WR2	0.95	0.74	0.84	0.47	0.5
Ings Beck	WR3	1.37	1.46	1.81	1.39	1.3
Ings Beck	WR4	0.82	1.83	1.6	0.99	2.03
Ashfold Side beck	WR5	6.32	3.89	1.55	1.96	2.86
Ashfold Side beck	WR6	10.16	6.1	2.74	3.13	3.18
Ashfold Side beck	WR7	3.19	1.86	1.21	1.27	1.79
Ashfold Side beck	WR8	1.26	0.92	0.84	0.73	1.19
Ashfold Side beck	WR9	1.96	1.58	1.04	0.72	1.53

Table 4.23 Actual Vx for *before* period and Target Vx to detect a ≥50% change to ≥1+ brown trout Habitat Quality (HQS). **<u>Bold and underlined</u>** signifies that significant change to ≥1+ densities could be detected within the study parameters.

	Actual VX	Target Vx for ≥50% Increase	Target Vx for ≥50% Decrease
< 20 cm Before	0.031	**<u>0.345</u>**	**<u>0.038</u>**
> 20 cm Before	0.016	**<u>0.092</u>**	0.010

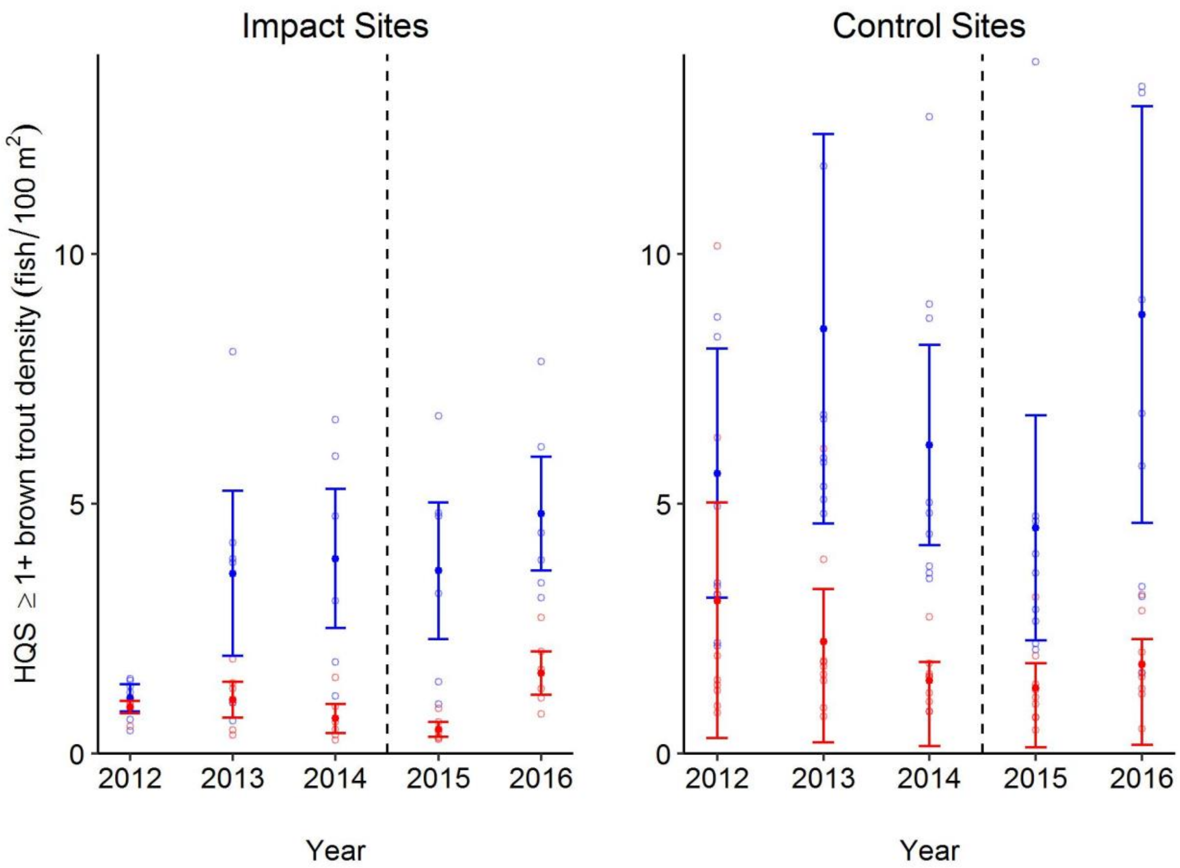

Figure 4.13. Relationship between average HQS densities scores ≥1+ < 20 cm (blue) and ≥1+ > 20 cm (red) for impact and control sites, error bars represent 95% confidence

intervals, dashed line represents introduction of the compensation flow.

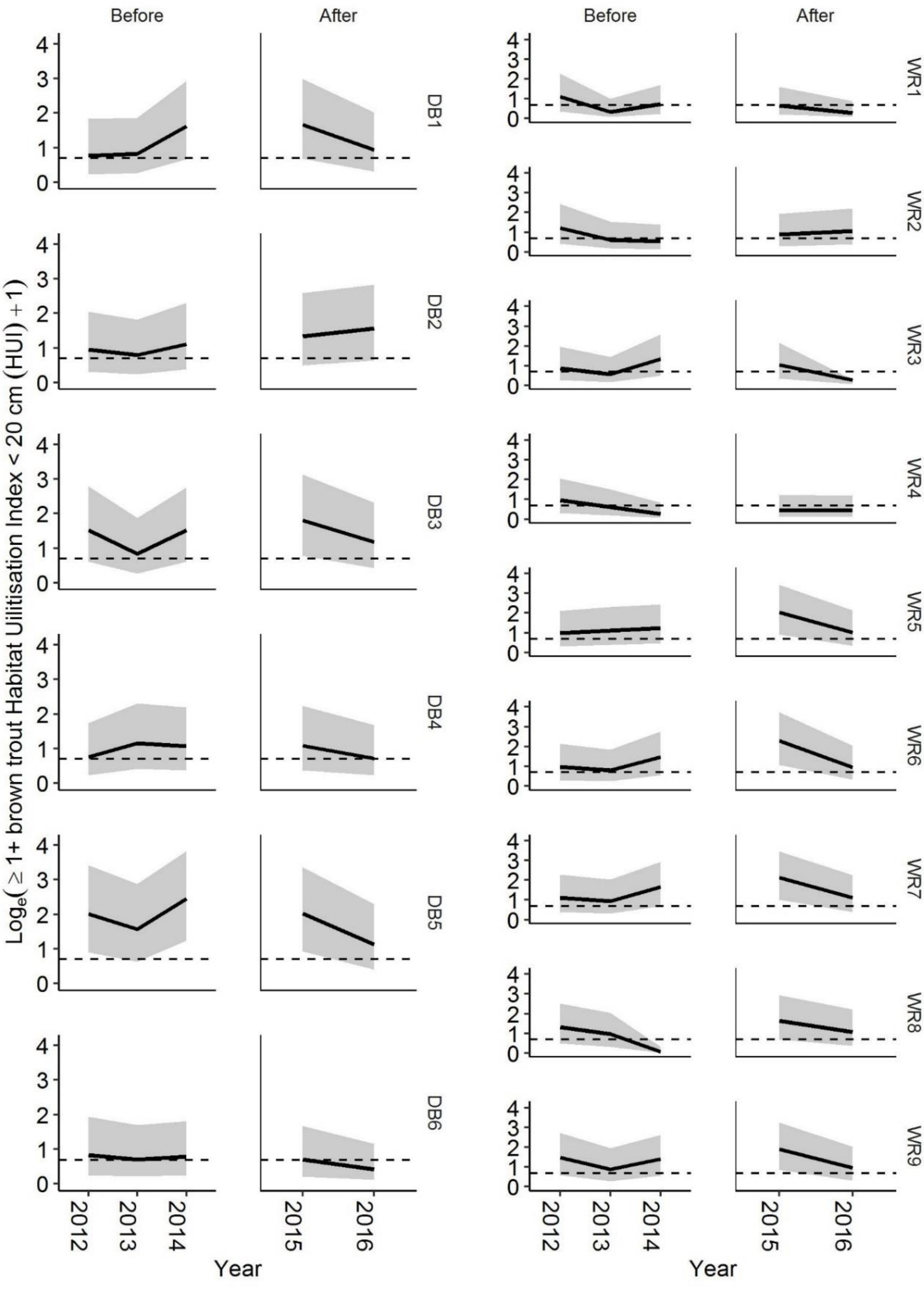

Figure 4.14 Log$_e$ + 1 transformed HUI for ≥1+ (< 20 cm) brown trout in the impact (left) and control sites (right) during the before (2012 – 2014) and after (2015 – 2016) period. Grey bands represent 95% confidence limits. Dashed line represents HUI value where observed and expected densities are equal.

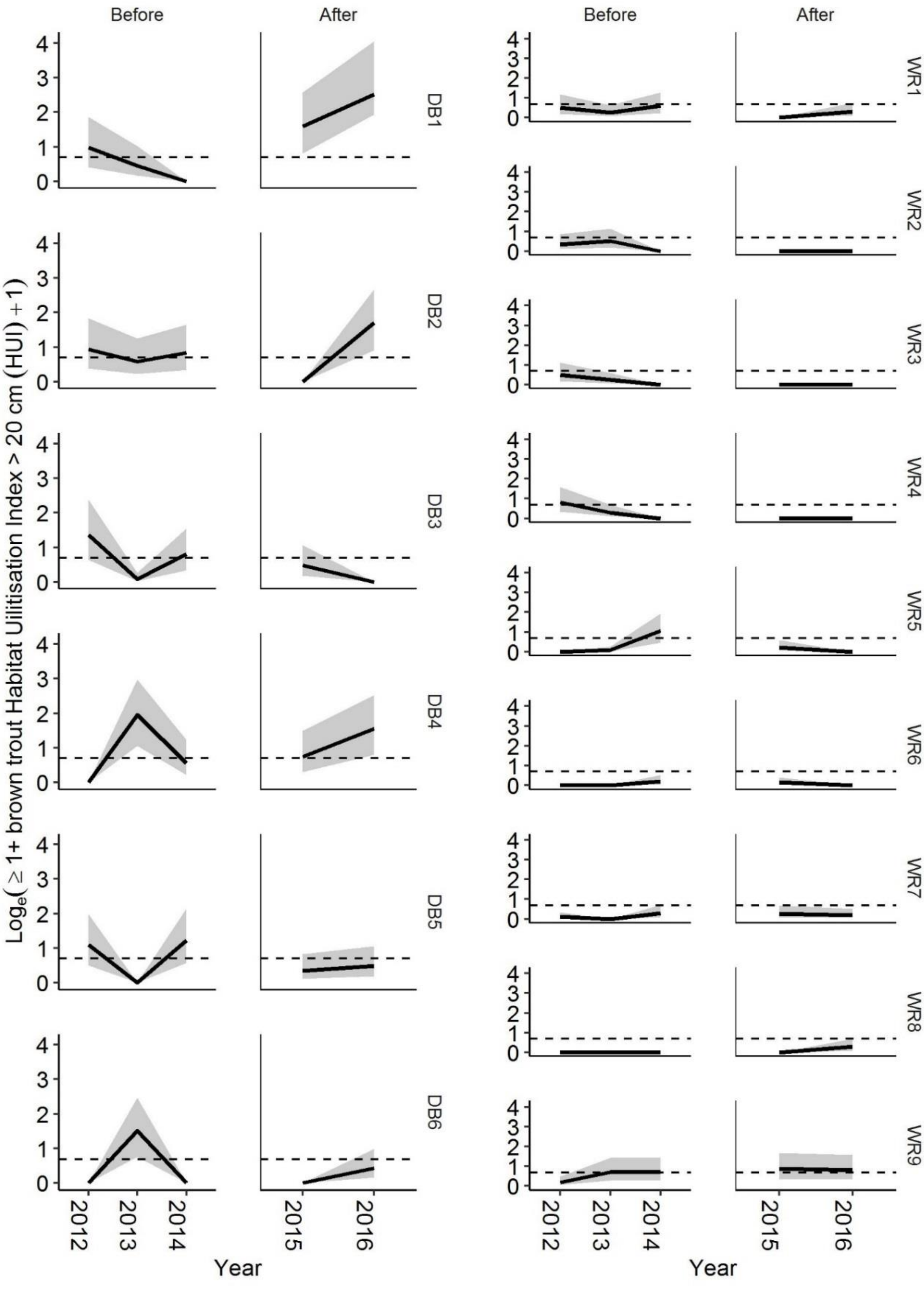

Figure 4.15. Loge + 1 transformed HUI for ≥1+(> 20 cm) brown trout in the impact (left) and control sites (right) during the before (2012 – 2014) and after (2015 – 2016) period. Grey bands represent 95% confidence limits. Dashed line represents HUI value where observed and expected densities are equal.

Impact Assessment (BACI)

Prior to the impact assessment, a resource calculation was performed to determine if the intensity of sampling was adequate to detect a biologically meaningful change to ≥1+ brown trout habitat quality. Following the collection of the after data (2015-2016) the level of actual variance increased for both size categories. Consequently, it was not possible to detect a ≥50% decrease to ≥1+ brown trout habitat quality from the natural variability inherent in the data (Table 4.24).

Table 4.24 Actual Vx for *before* and after period and Target Vx to detect a ≥50% change to ≥1+ brown trout Habitat Quality (HQS). **Bold and underlined** signifies that significant change to ≥1+ densities could be detected within the study parameters.

	Actual VX	Target Vx for ≥50% Increase	Target Vx for ≥50% Decrease
< 20 cm Full	0.040	**<u>0.345</u>**	0.038
> 20 cm Full	0.068	**<u>0.092</u>**	0.010

Habitat quality for ≥1+ (< 20 cm) brown trout was significantly lower (Table 4.25) in the impact sites than the control sites. Following the introduction of the compensation flow there was a marginal increase to the ≥1+ (< 20 cm) brown trout habitat quality (Figure 4.16); as the interaction between Area and Period in the BACI GLMM was significant (Table 4.25) it can be said that there had been a significant change to ≥1+ (< 20 cm) brown trout habitat quality in the River Dibb following the introduction of the flow trial in 2015. The same cannot be said for the habitat quality for ≥1+ (> 20 cm) brown trout. The BACI GLMM revealed that the habitat quality for ≥1+ (> 20 cm) brown trout was significantly lower (Table 4.25) (Figure 4.17). Despite the increase to ≥1+ (> 20 cm) brown trout habitat quality in the after period, the interaction between area and period in the GLMM was not significant (Table 4.25), therefore the increase to ≥1+ (> 20 cm) brown trout habitat quality in the River Dibb is more likely representative of natural variability as opposed to the introduction of the compensation release in 2015.

Table 4.25 BACI GLMM to detect a change in ≥1+ brown trout HQS density following the introduction of the compensation release from Grimwith reservoir in 2015

	Sum Sq	Mean Sq	DoF	DenDF	F	P
≥1+ (< 20 cm)						
Area	0.659	0.659	1	13.168	5.226	**0.039**
Period	0.042	0.042	1	3.029	0.335	0.603
Area:Period	0.973	0.973	1	55.000	7.7049	**0.007**
≥1+ (> 20 cm)						
Area	0.267	0.267	1	13.147	4.894	**0.045**
Period	0.009	0.009	1	3.030	0.159	0.717
Area:Period	0.180	0.180	1	55.000	3.289	0.0752

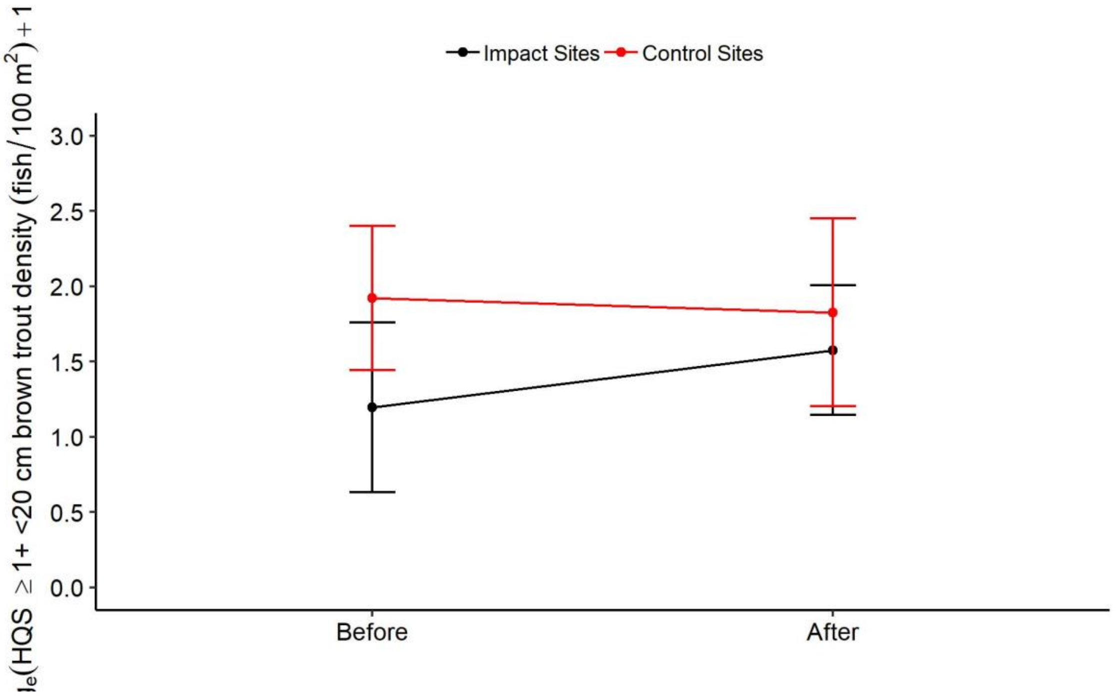

Figure 4.16. Mean (±95%) ≥1+ (< 20 cm) brown trout HQS densities during *before* and *after* periods for impact (black) and control (red) sites. Trendlines represent significant interaction between Area and Period in BACI GLMM.

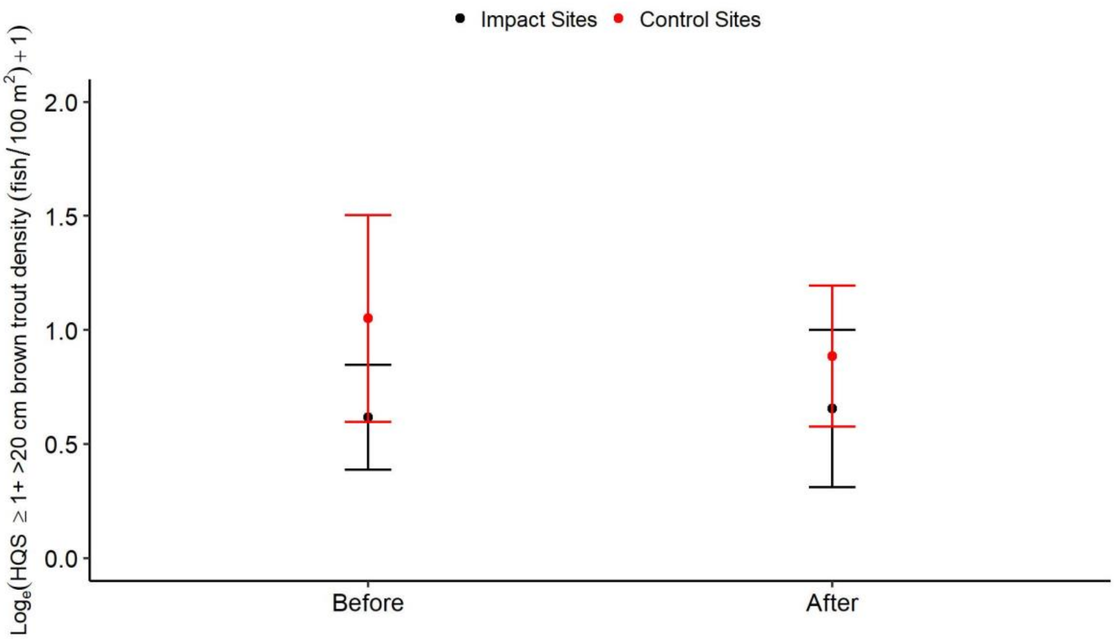

Figure 4.17 Mean (±95%)≥1+ (> 20 cm) brown trout HQS densities during *before* and *after* periods for impact (black) and control (red) sites.

4.4 Discussion

The introduction of the four-stage, seasonally variable compensation flow from Grimwith reservoir in 2015 was aimed at improving populations of brown trout of all age and size classes and the habitat quality throughout the River Dibb. Observed and expected densities (from HABSCORE) of brown trout varied both spatially and temporally across the impact and control sites during the study. 0+ brown trout were typically found in low densities at the impact sites with this age class absent from two sites in 2012 and one site in 2015 and 2016, suggesting poor recruitment within the River Dibb. Observed densities of 0+ brown trout were classed as either poor or fair/poor compared to the national averages (EA-FCS). At the control sites observed densities of 0+ brown trout were much more varied with densities classed from poor to excellent, but this age class was found to be absent from four control sites in 2016. When compared to the national averages it was apparent that the 0+ brown trout densities at control sites were far more varied not only between sites but between years, with one site varying by 44.5 fish/100 m^2 during the study. This suggests there was much more habitat heterogeneity in the control sites, but populations were much more prone to large fluctuations derived from the natural variability in brown trout recruitment. However, it is not uncommon for salmonid populations to fluctuate by orders of magnitude (Platts and Nelson 1988). 0+ brown trout densities were very low across all impact and control sites in 2012, it is suspected that abnormally high summer flows during this period (Parry *et al.* 2013) were responsible for the low 0+ brown trout densities. High levels of rainfall can lead to increased rates of discharge-related mortality in brown trout through washout and displacement (Daufresne *et al.* 2005, Jonsson and Jonsson 2011, Lobón-Cerviá 2014). Unfortunately, no fisheries data were recorded prior to this study (commencing in 2012) at any of the impact or control sites, therefore, density dependent factors such as poor spawning stock densities cannot be discounted. The low densities at the impact sites may also be explained by the impoundment from Grimwith reservoir. Prior to 2015, Grimwith reservoir operation did not account for ecological impacts resulting from flow. Flow levels in the River Dibb fluctuated between high flows stemming from planned reservoir release to low flows when the release was shut off completely, leading to reduction of wetted widths and depths of the river channel. Whilst width is thought to play little significance in the population dynamics of brown trout, depth is crucial to certain key life phases of brown trout (see Chapter 2.2; Cowx *et al.* 2004, UKTAG 2013). Similar to 0+ brown trout densities, ≥1+ brown trout densities were relatively low across the impact sites prior to 2015, particularly so in 2012 probably due to the dual pressures of inclement weather and reservoir operation resulting in dispersal and drift to more suitable, and deeper, reaches of the river (Crisp 2000). By contrast, densities of ≥1+ brown trout were higher at the control sites and displayed no significant loss of brown trout in 2012 compared with other years, which is not surprising given that older larger trout have greater ability to maintain station within the water column at higher water velocities (Jonsson and Jonsson 2011).

Habitat quality at impact sites in the River Dibb indicate that 0+ brown trout populations were expected to be higher, and in many cases significantly higher, than observed suggesting that there are other limiting factors beyond physical habitat (i.e. temperature water chemistry or flow rate) that may also be operating on 0+ brown trout densities in the River Dibb. It is important, however, to understand fully the limitations of the habitat surveying methodology when interpreting these results. It has been previously established that the River Dibb is prone to large fluctuations in flow rate due to reservoir operation, and that these elevated flows could lead to less suitable 0+ brown trout habitat, but these higher flows were not recorded during the habitat surveying as any release from Grimwith reservoir was disabled prior to instream work to ensure the safety of the field work team. This is imperative when interpreting the HABSCORE data as the HQS and subsequently HUI values are representative of habitat conditions with no release from Grimwith Reservoir. Whilst no data exist regarding the habitat conditions in the River Dibb during a release, personal observations of the habitat revealed high water marks at many sites in excess of 30 cm higher than the river level during surveying. Therefore, it is plausible that the low HUI values for all impact sites reflect the high flow conditions that occurred outside the scope of habitat monitoring. The converse is potentially responsible for the high HUI of ≥1+ (< 20cm) brown trout across the impact sites, in that the deeper water in the River Dibb resulting from the reservoir release provides habitat that can support larger densities of ≥1+ (< 20 cm) brown trout that are still present within survey sites following the release shut off prior to any surveying work. Unlike the 0+ and smaller (< 20 cm) brown trout the ≥1+ (> 20 cm) brown trout exceeded what was expected from the habitat quality at the majority of impact sites in 2012 and 2014. It is likely that in 2012 the elevated rainfall and reservoir operation offered better habitat (i.e. faster deeper flowing water) for larger trout (Cowx *et al.* 2004, UKTAG 2013) that again may not have been entirely reflected in the habitat surveys.

Observed densities of 0+ brown trout at control sites were lower than expected from the HQS at a number of sites in 2012 and 2013, but several control sites in 2014, supported densities of brown trout that exceeded those predicted from the HQS. These findings reflected the natural variability of both brown trout and habitat quality. Similar to the impact sites, observed densities of ≥1+ (< 20 cm) brown trout typically exceeded what was expected from the habitat quality. Observed densities of the largest size class of brown trout (> 20 cm) were lower than expected from the HQS in most years both during the before and after stage reflecting that the habitat may not be the limiting factor behind the low densities of the ≥1+ (> 20 cm) brown trout, although the lower than expected densities do reflect the lack of suitable habitat (i.e. deeper faster flowing water) that large brown trout favour (Cowx *et al.* 2004, UKTAG 2013).

There was some intra-cohort variation in the length at capture of 0+ brown trout, with the smallest individuals being around 50% of the total length of the largest individuals of the year class. The level of length variation of 0+ brown trout at both the impact and control sites is consistent with Newman's (1993) prediction that individual growth is based on a distribution of feeding site quality, as territoriality between the conspecifics would lead to smaller individuals feeding at less profitable territories exacerbating the size difference within cohort. It also cannot be ruled out that genetic variability within a population is an important mechanism behind variability of length at capture (Jonsson and Jonsson 2011). 0+ brown trout at capture were typically larger in impact sites than the control sites, but it cannot be ascertained with any certainty if this is a result of better growth conditions in the impact sites or that the impact sites were surveyed 2-3 weeks later in the year than control sites. 0+ brown densities were higher in control sites compared to the impact sites therefore this disparity in growth could possibly be due to density dependent processes as a negative correlation between growth and density has been identified in other salmonid populations (Newman 1993, Bohlin *et al.* 2004). There was a marginal decrease to the average length of 0+ brown trout in the River Dibb following the introduction of the compensation flow in 2015, but there was also a decrease in the average length of 0+ brown trout at capture in the control sites during the same time period. The variability of 0+ brown trout lengths across sites and reaches were within the boundaries to statistically detect a decrease to the population, but GLMM analysis revealed that there was no change to the lengths of brown trout in the River Dibb that could be attributed to the flow trial.

The GLMM BACI model revealed that there were no significant changes to brown trout population dynamics (density and length at capture) for the 0+ age class or changes to the densities of larger brown trout (< 20cm, >20cm) that can be attributed to the introduction of the compensation release. In the case of the 0+ and ≥1+ brown trout age classes the low densities and high level of variability at both the impact and control sites made it impossible to disentangle any compensation-induced increase from natural variability. Side by side comparison of the four population dynamics (densities for the three age classes and 0+ length at capture) from the before and after period revealed that only ≥1+ (< 20 cm) brown trout densities improved in the after period with the remaining three metrics marginally declining from the before to after period. In comparison densities for 0+ and ≥1+ (> 20 cm) brown trout as well as length at capture for 0+ brown trout declined at the control sites in the after period, whereas densities of ≥1+ (> 20 cm) brown trout marginally increased, suggesting that the introduction of the flow trial has had the opposite effect on brown trout population dynamics to that expected. This result, however, should be treated with caution. The introduction of the compensation release did not occur until March 2015, therefore the spawning between October 2014 and December 2014 and intra-gravel phases between October 2014 and March 2015 were not influenced by the compensation release. The timing of the introduction of the compensation release meant that only the 2016 0+ brown trout cohort had every facet of their life cycle influenced by the compensation release. It is important to note that the direction of the change to the population dynamics metrics at the impact sites were also seen in the control sites. In 2016, 0+ brown trout were absent from four of the nine control sites, suggesting that 2016 was a poor year for recruitment in the region, indeed in 2016 low 0+ brown trout densities were also observed at many regulated and un-regulated rivers in Yorkshire (HIFI, Unpublished data, 2016). In late 2015 and early 2016 the UK was subject to severe and protracted flooding due to unprecedented levels of rainfall, with flow rates in many northern rivers eclipsing previous record levels (Marsh *et al.* 2016). In the River Wharfe for instance the average flow rate during this period was >2.5 times greater than the long-term average. The timings of these floods will have almost certainly interacted with spawning and or incubation of brown trout eggs. It has been demonstrated in alpine rivers that high flow velocities can lift and scour spawning gravels causing washout or mechanical shock to the eggs contained within (Unfer *et al.* 2011). It is also possible that the prolonged high flows severely reduced the available spawning habitat as gravels get washed out (Roghair *et al.* 2002), but this hypothesis is not fully supported as habitat quality at the study sites was not significantly different in 2016 from previous years. It is important to note that there was a decline in 0+ brown trout in 2016 at control sites that was not seen at impact sites, with some sites holding better stocks of 0+ brown trout than in 2015. It has been established (see chapter 2.5) that extreme high flows can be somewhat reduced in reservoir regulated rivers, it is possible that the influence of Grimwith Reservoir in reducing extreme flows in the River Dibb reduced the degradation to brown trout populations as seen in other local rivers and streams. There was no significant change to either size class of ≥1+ brown trout in 2016 at either the control or impact sites, it is most likely that the larger body size of the older brown trout aided in their ability to withstand higher flow velocities to reduce any potential washout (Jonsson and Jonsson 2011).

The habitat quality for the three HABSCORE age/size classes at the impact sites also improved following the introduction of the compensation release with the habitat quality for the ≥1+ (< 20 cm) showing a significant change from prior to the flow trial. It is worth noting that habitat quality for the three HABSCORE age/size classes at the control sites all declined between the before and after periods. This gives more weight to the assumption that the improvement in habitat quality (especially for the ≥1+ (< 20 cm) brown trout) is linked to the introduction of the baseline compensation release. The compensation release was introduced to mitigate the altered flow regime caused by the operation of Grimwith Reservoir, but despite its introduction the underlying factors that alter the flow regime from its natural state (i.e. release to ensure abstraction on the Wharfe and hydro-electric generation during over topping events) still occur in the River Dibb and are operational requirements for the system. In unregulated natural river systems sediment can be replenished throughout the river network by large flow events transporting and depositing sediment components downstream, but with all but the finest of sediments accumulating in reservoirs, this replenishment cycle does not occur downstream of reservoirs leading to coarsening of the streambed (Poff *et al.* 1997). High flow events can also remove marginal vegetation and cover from the survey sites (Stromberg *et al.* 2007). Whilst it is encouraging to see that the establishment of the minimum compensation release from Grimwith has been shown to significantly improve the habitat quality for 1+ (< 20 cm), the issue of large operational releases that are unsynchronised from the natural conditions persist from Grimwith reservoir. It is likely that the deleterious impacts from these releases (such the armouring and coarsening of the substrate and lack of refuge areas for juvenile fish) need to be addressed before the desired biological responses are seen in the River Dibb.

5 IMPACT OF INTRODUCING COMPENSATION RELEASE ON BROWN TROUT POPULATION DYNAMICS AND HABITAT QUALITY IN DALE DIKE

5.1 Introduction

Achieving the goal of “good” ecological potential in all HWMBs by 2027 requires appropriate mitigation measures are undertaken in affected river systems. For rivers impounded by reservoirs with a significant alteration to the flow regime, the amendment or introduction of compensation releases from upstream reservoirs is typically the least resource intensive measure applicable to meet the “good” ecological potential goal.

The previous chapter (Chapter 4) covered the introduction of a compensation release from an impounding reservoir to help improve brown trout fisheries in the river downstream. As with all flow trials and rehabilitation measures it is key to ensure that the socio-economic costs are assessed, and the most appropriate measure is then chosen. Reservoirs in Yorkshire are not recent additions to the environment, with many large reservoirs in Yorkshire (i.e. Damflask and Rivelin see Chapter 3) commissioned in the mid-nineteenth century, and many of these reservoirs having little to no infrastructure improvement during their lifetimes. In the case of Damflask (Chapter 3) and Dale Dike (introduced in this chapter) reservoirs, the rate of release from the reservoir can only be set manually by a YWS engineer physically opening and closing the outlet valves by hand. In the case of Dale Dike (prior to 2014), there was also no system in place to determine the volume of water released as flow levels were set and visually maintained. The associated costs of retrofitting and maintaining new valves can be costly and must consider not only the costs of fitting and maintenance but also the knock-on effects throughout the reservoir supply system as the reservoir would be shut down or at least operated at reduced capacity for the duration of the engineering works. As part of the design and planning phase of achieving “good” ecological potential, an assessment is undertaken of the release capacity of an impounding reservoir, as well as the potential impact that altering/introducing the release pattern will have on reservoir yield and flood management (Acreman 2007). The purpose of this is to ensure that any proposed change to reservoir operation in compliance with WFD would not incur disproportionate costs or impacts on the role of the reservoir for water storage or flood defence. After this assessment, it may become apparent that the introduction of a fully naturalised flow regime would not be appropriate, therefore further assessment would be required to provide the best ecological improvement (i.e. seasonally variable release, annual minimum flow, physical river restoration) without incurring disproportionate socio-economic costs.

The specific objectives of this chapter are to:

- Assess the status of 0+ and ≥1+ brown trout populations and habitat quality at sites along Dale Dike (a stretch of river downstream of Dale Dike Reservoir) before and after the introduction of a new compensation release.
- During the same timeframe compare the status of 0+ and ≥1+ brown trout populations and habitat quality at control sites where the flow regimes are not influenced by Dale Dike Reservoir to determine the level of temporal variability of brown trout populations in the region.
- Use a Before After Control Impact (BACI) analysis to establish the influence (if any) that the introduction of the compensation release has had on the brown trout populations and habitat quality in Dale Dike.

Outputs will be used to determine the effectiveness of introducing an annual minimum compensation release from Dale Dike reservoir on improving the Habitat quality and brown trout populations in Dale Dike

These findings aim to provide insights into the effectiveness of the introduction of an annual minimum compensation flow on brown trout populations in a Heavily Modified Water Body (HMWB), namely Dale Dike. This will aid in the development of the knowledge base of appropriate mitigation measures to achieve GEP of HMWBs regulated by reservoirs.

5.2 Methodology

5.2.1 Study reaches

Yorkshire Water Services, as part of their Adaptive Management Programme (AMP5), identified Dale Dike as a stream located downstream of an impounding reservoir where no compensation release was in place. Consequently, Dale Dike Reservoir (Dale Dike) was selected as a suitable candidate for a flow trial where a compensation release could be introduced to mitigate hydrological impacts downstream of the reservoir. Brown trout was selected as an indicator species to determine any biological responses to the flow trials, due to its abundance across all study and control rivers, as well as its sensitivity to flow regimes (See Chapter 2.5).

Dale Dike (Dale Dike Reservoir)

Dale Dike is a 3-km long stream located between Strines and Damflask Reservoirs in South Yorkshire. Dale Dike reservoir is part of a chain of reservoirs that eventually flow into Damflask Reservoir, which in turn flows into the River Loxley and the River Don.

Dale Dike Reservoir is situated 250 m downstream of the outfall from Strines Reservoir (Figure 5.1) and was constructed in 1875. Prior to this flow trial, there was a constant release from the reservoir into a stilling basin directly beneath the outfall. This stilling basin is the abstraction point for Loxley Water Treatment Works (LWTW). Typically, the level of release from the reservoir matches the level abstracted by LWTW so it can be considered that this release from the reservoir has no appreciable effect on the flow downstream. In 2011, the ecological status of Dale Dike was classed as poor with fish populations failing, likely because of the low flow levels and flow homogenisation resulting from reservoir operation.

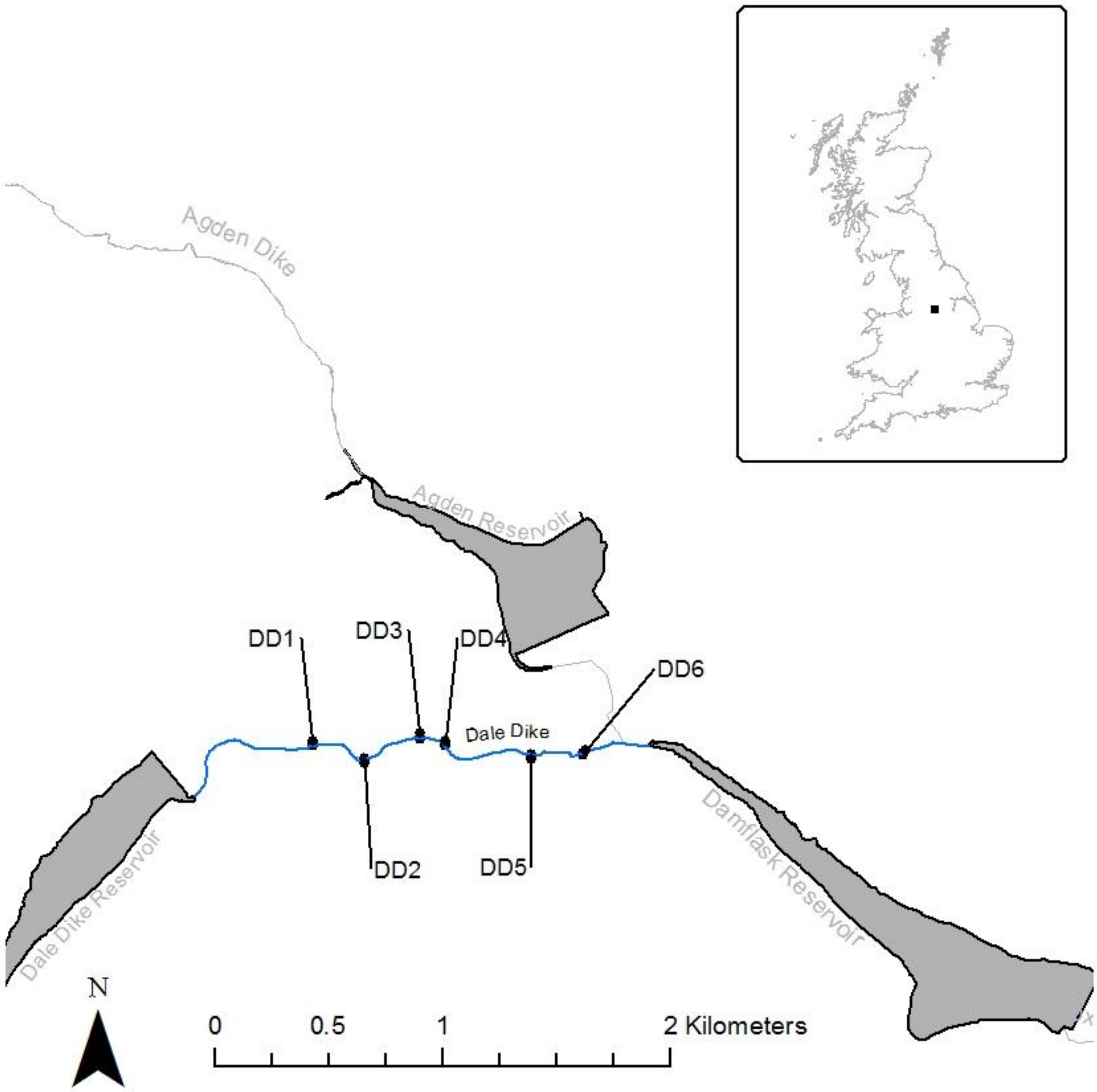

Figure 5.1. Location of all impact sites (DD1-DD6) in relation to Dale Dike reservoir and surrounding reservoirs and tributaries.

In March 2014, a baseline compensation release set at 3.3 Ml/d (0.039 m^3/s) was introduced from Dale Dike reservoir. Unlike the Grimwith reservoir (Chapter 4) release, this does not vary by season, but it was presumed that elevating the baseline flow would provide benefits to the geomorphological, physicochemical, and ecological elements of the environment, and improve the ecological status of Dale Dike to "Good".

Six sites of 50 m(DD1 – DD6) were selected along a 2.4-km section of Dale Dike between Dale Dike Reservoir and Damflask Reservoir for the monitoring. The most upstream site was 900 m downstream of Dale Dike reservoir and the most downstream site was 200 m upstream of the confluence with Agden Dike and 300 m upstream of Damflask Reservoir (Figure 5.1). The land surrounding Dale Dike is almost exclusively rough pasture and grazing, with moderate areas of deciduous woodland, with an average stream gradient of 3.6%, the six monitoring sites were selected to represent the heterogeneity of the aquatic habitat in Dale Dike. Annual monitoring was undertaken in September or October over a five-year period from 2012-2016, consisting of two years baseline fisheries data collection and three years flow trial data collection.

Ewden Beck, Little Don, River Don, River Sheaf and Wyming Brook (Control sites)

Seven sites from five rivers were selected as control sites for the Dale Dike reservoir flow trial. The streams selected were Ewden Beck (DR1 and DR2, 2 sites), Little Don (DR3, 1 site), River Don (DR4 and DR5, 2 sites), River Sheaf (DWR1, 1 site) and Wyming Brook (DWR2, 1 site) (Figure 5.2). Rivers in the immediate vicinity of Dale Dike were not suitable as controls sites due to the difference in habitat characteristics. It was difficult to find stretches of river in the vicinity of Dale Dike that were long enough to allow for multiple sampling sites whilst being representative of the impact sites. The land use surrounding and upstream of the control sites was, a mix of rough pasture and grazing (River Don, Little Don Ewden Beck and Wyming Brook), whilst the river sheaf had a more urbanized setting the surrounding and upstream land use was dominated by deciduous woodland. The gradient of the 5 control rivers were varied with Ewden beck, the little don and river don being 3.1%, 2.2% and 3.5% respectively. The gradient of the river sheaf was relatively shallow (1.2%). Compared to the impact sites the gradient of Wyming brook is relatively steep (8.8%). Despite the varied gradients of the 7 control sites, there were still deemed as appropriate impact sites due to the similarities in other physical characteristics (substrate, wetted widths and depths) as well as comparable fish populations. All control sites were on regulated river systems. The sites on Ewden Beck, Little Don and Wyming Brook were located downstream from impounding reservoirs but were still considered as suitable control sites as there was no anthropogenic change to the habitat during the study so temporal variation in the data set would be considered to be a reflection of natural conditions.

Environmental investigation

Brown trout populations in the study river (Dale Dike) and control rivers (Ewden beck, Little Don, River Don, River Sheaf and Wyming Brook) were monitored annually during both the before (2012-2013) and after (2014-2016) phases of this study using the fish survey methodology detailed in Chapter 3.2.3. The methodology for the classification of population estimates, juvenile growth rates, habitat surveying, HABSCORE analysis and impact assessment (BACI) methodology, detailed in Section 4.2.4 – 4.2.8, were identical in this study so will not be repeated.

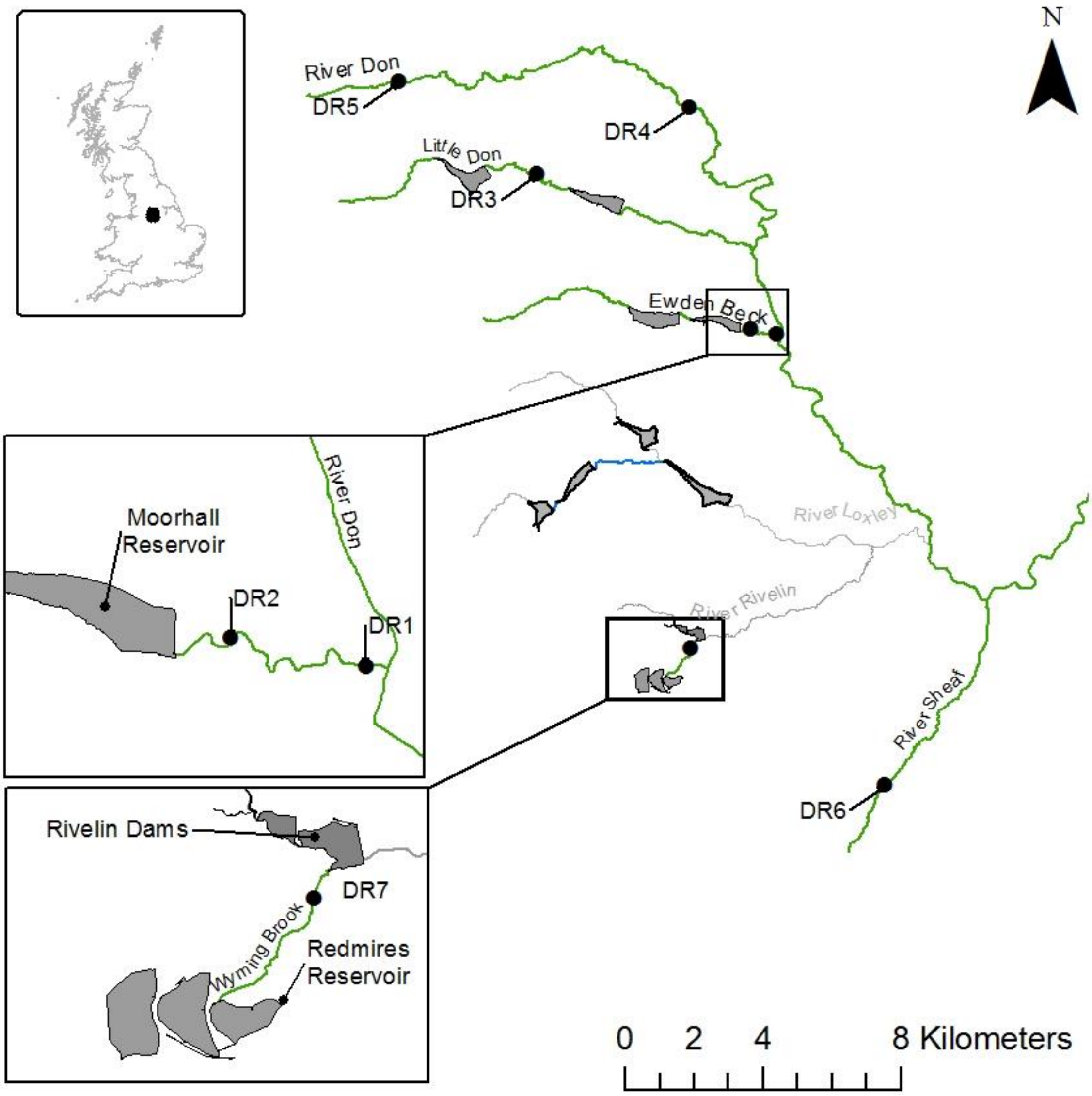

Figure 5.2. Location of the seven control sites (DR1 – DR7) in relation to each other, and Dale Dike (Blue)

5.3 Results

5.3.1 Impact of introduction of compensation release on 0+ brown trout densities

Baseline monitoring (Before)

During the baseline period, 0+ brown trout were caught at all impact sites except for DD6 in 2012. At the remaining sites 0+ brown trout densities ranged from 0.7 to 13.7 fish/100 m^2 (Table 5.1), however, there was no significant variation between the sites ($F_{5,6}$ = 1.201, P > 0.05). During this period, 0+ brown trout populations were predominantly poor, especially in 2012. 0+ brown trout populations improved significantly in 2013 compared to 2012 (P<0.05), increasing at all impact sites, with three sites (DD1, DD3 and DD5) classed as average, two sites as fair/poor (DD4 and DD6) and one site (DD2) remaining as poor. This site level improvement of 0+ brown trout populations led to significantly (P <0.05) increased pooled 0+ brown trout densities in 2013 (Figure 5.3).

0+ brown trout population densities were also low in the control sites with this age class absent in site DR3 in 2013. At the remaining sites, 0+ brown trout densities ranged from 0.3 to 14.3 fish/100 m^2 during this period (Table 5.1) and improved in 2013 compared to 2012 at five sites (DR1, DR4-DR7), but not significantly (P>0.05). There was also no significant difference between 0+ brown trout densities between the five control rivers ($F_{6,7}$ = 1.19, P>0.05). Pooled 0+ brown trout densities followed a similar trend in the control sites as observed in the impact sites, but due to the variability of population estimates the differences between the 2012 and 2013 were not significant (Figure 5.3).

Prior to the commencement of the flow trial in 2014, a resource calculation was performed on the 0+ brown trout 'before' density data. Under the assumption that the temporal variability in the impact and control sites persists, the resource calculation indicated that a ≥50% increase to 0+ brown trout populations would be detectable following three years of 'after' monitoring (Table 5.2). Due to the variability and low densities in both the impact and control sites, a ≥50% decrease to 0+ brown trout densities would not be detected within the timeframe of this study.

During compensation release (After)

0+ brown trout were present in low densities at all impact sites during the after period, except for site DD5 in 2016 when this age class was absent. 0+ brown trout densities at the remaining sites ranged from 0.6 to 12.6 fish/100 m^2 during this period and were predominately classed as poor (Table 5.1). The pooled 0+ brown trout density estimates varied little during this period (Figure 5.3) with no significant ($P>0.05$) differences between years. Unlike the before period, there were significant differences between the sites ($F_{5,12}$= 15.38, $P<0.05$), for example, one site (DD1) consistently held greater (average classification) 0+ brown trout densities than the other impact sites (Table 5.1).

0+ brown trout populations varied at control sites during the after period with significant differences between the years ($F_{1,19}$ = 4.68, $P<0.05$). This age class was absent from site DR2 in 2016. At the remaining control sites 0+ brown trout densities ranged from 0.8 to 13.5 fish/100 m^2 with sites predominantly classed as poor (Table 5.1). There was an apparent decline in 0+ brown trout densities during the after period (Figure 5.3). There was some between site variability, with some sites consistently supporting lower 0+ brown trout densities than others (i.e. sites DR1 and DR2 held populations that were poor – fair/poor for the duration of the study) (Table 5.1), however, 0+ brown trout populations were not significant different between the control rivers ($F_{4,14}$=2.51, $P>0.05$) or between sites nested within them ($F_{2,14}$=2.46, $P>0.05$).

Table 5.1. Density estimates ± 95% C.L. of 0+ brown trout at impact and control sites during both the before (2012 – 2013) and after (2014 – 2016) periods. Colours denote EA-FCS abundance classification

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

River Name	Site Identifier	2012	2013	2014	2015	2016
		Before			After	
impact Sites						
Dale Dike	DD1	8.6 ± 0.9	13.71 ± 1.7	8.72 ± 0.6	11.22 ± 0.7	12.59 ± 1.2
Dale Dike	DD2	1.64 ± 0.3	2.2 ± 0.1	1.73 ± 0	2.92 ± 0	1.36 ± 0.0
Dale Dike	DD3	1.14 ± 0.0	10.05 ± 2.3	3.07 ± 0.2	3.23 ± 0	1.13 ± 0.3
Dale Dike	DD4	1.31 ± 0.2	4.19 ± 0.4	2.68 ± 0.3	2.32 ± 0.1	1.16 ± 0.3
Dale Dike	DD5	0.66 ± 0.4	9.73 ± 1.3	5.3 ± 0.6	2.61 ± 0.0	0.00 ± 0.0
Dale Dike	DD6	0.00 ± 0.0	5.84 ± 0.4	0.57 ± 0.0	0.83 ± 0.0	2.75 ± 0.5
Control Sites						
Ewden Beck	DR1	4.97 ± 0.6	4.16 ± 0.4	2.42 ± 0.4	2.85 ± 0.6	1.81 ± 0.0
Ewden Beck	DR2	1.22 ± 0.1	0.34 ± 0.0	1.37 ± 0.4	1.13 ± 0.5	0.00 ± 0.0
Little Don	DR3	2.63 ± 0.5	0.00 ± 0.0	3.33 ± 0.4	1.98 ± 0.3	1.12 ± 0.2
River Don	DR4	0.67 ± 0.2	2.63 ± 0.9	3.39 ± 1.3	3.09 ± 0.4	0.75 ± 0.1
River Don	DR5	4.67 ± 0.3	9.53 ± 1.4	13.18 ± 0.4	13.16 ± 1.2	0.89 ± 0.3
River Sheaf	DR6	0.60 ± 0.0	7.85 ± 0.5	13.53 ± 0.6	11.37 ± 0.7	2.49 ± 0.0
Wyming Brook	DR7	3.00 ± 0.3	14.32 ± 1.8	7.85 ± 0.6	1.6 ± 0.6	4.29 ± 0.2

Table 5.2. Actual Vx for *before* period and Target Vx to detect a ≥50% change in 0+ brown trout densities. **Bold and underlined** signifies that significant change to 0+ brown trout densities could be detected within the study parameters.

Actual Vx	Target Vx for ≥50% Increase	Target Vx for ≥50% Decrease
0.109	**<u>0.513</u>**	0.057

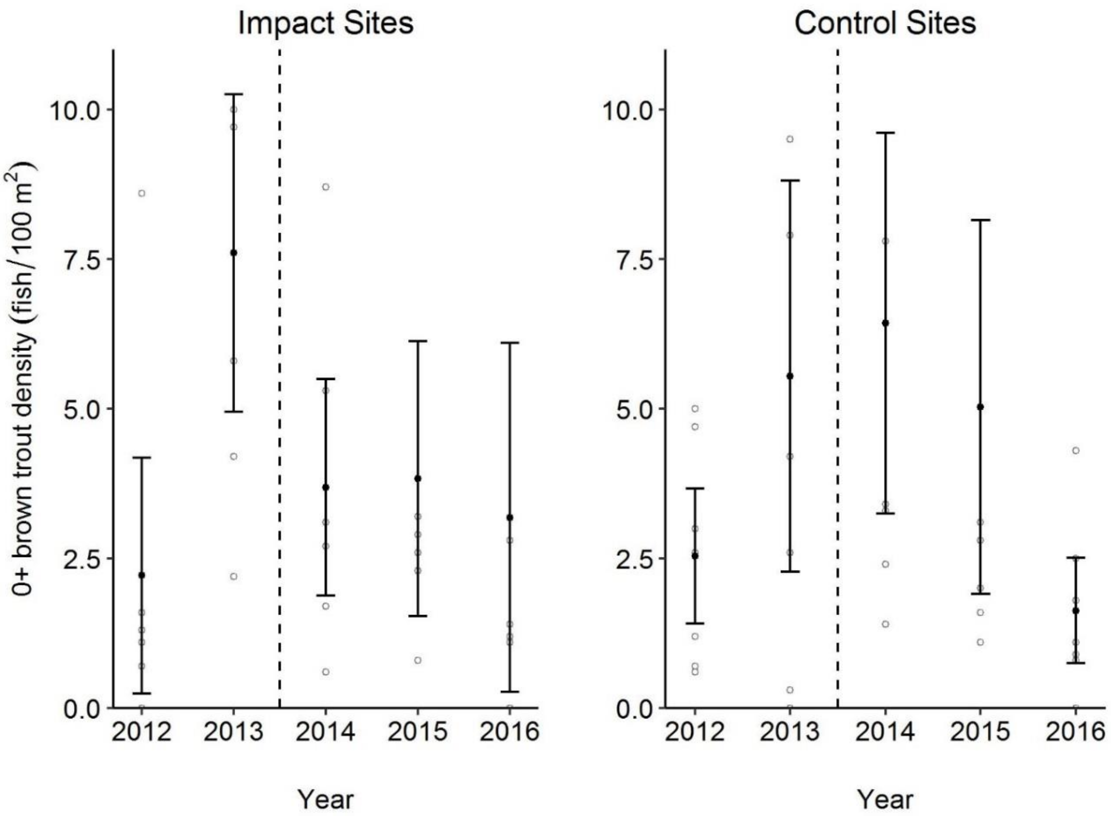

Figure 5.3 Average 0+ brown trout densities ± 95 % C.L. for all impact and control sites, throughout the study period. The dashed line represents the division between *before* and *after* periods. Grey circles represent individual density estimate

Impact assessment (BACI)

Residual variances from both periods (before and after) were incorporated into a diagnostic resource calculation, which confirmed that a ≥50% increase to 0+ brown trout densities would be detectable given the variance in the current dataset. However, due to the low densities and variability between the sites, a ≥50% decrease to 0+ brown trout densities would not be identified without further sampling during the after period (Table 5.3).

Table 5.3. Actual Vx for full study and Target Vx to detect a ≥50% change to 0+ brown trout densities. **<u>Bold and underlined</u>** signifies that significant change to 0+ densities could be detected within the study parameters.

Actual Vx	Target Vx for ≥50% Increase	Target Vx for ≥50% Decrease
0.083	**<u>0.513</u>**	0.057

There was no significant difference in 0+ brown trout densities between the impact and control sites (Table 5.4) and t there was no significant effect of period (before and after) on the densities (Table 5.4). The effect of the flow trial on the impact sites was also not significant (Table 5.4). Therefore, the flow trial did not have a significant influence on 0+ brown trout densities in Dale Dike, as the high level of variability within 0+ brown trout densities across impact and control sites prevented the isolation of any changes caused by the compensation release from natural variation.

Table 5.4 BACI GLMM to detect a change in 0+ brown trout density following the introduction of the compensation flow from Dale Dike Reservoir in 2014

	Sum Sq	Mean Sq	DoF	DenDF	F	P
Area	<0.001	<0.001	1	11.168	0.002	0.962
Period	0.006	0.006	1	3.005	0.0214	0.893
Area:Period	0.247	0.247	1	47.00	0.874	0.354

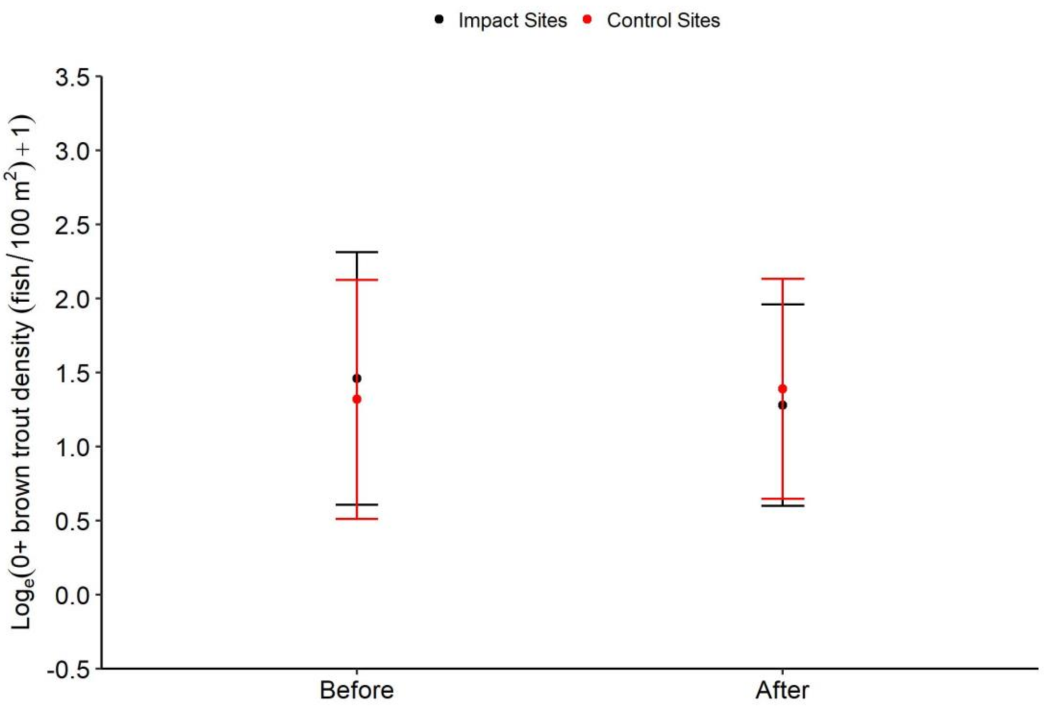

Figure 5.4. Mean (±95%) 0+ brown trout densities before and after for impact (black) and control (red) sites.

5.3.2 Impact of introduction of compensation flow on ≥1+ brown trout densities

≥1+ brown trout were caught at all sites in all years during this study and were typically found in higher densities than juvenile conspecifics (Appendix Table A.3). Comparison of actual ≥1+ brown trout densities with HQS densities was undertaken in two size-based categories (≥1+r <20cm, ≥1+ >20 cm).

Baseline monitoring (Before)

≥1+ brown trout dominated catches at impact sites and were caught at all sites in both years during the before period with densities ranging from 3.67 to 19.54 fish/100 m^2 (Appendix, Table A.3). Of these, the ≥1+ (< 20cm) densities ranged from 2.9 to 17.2 fish/100 m^2 (Table 5.5). There was a marginal, but not significant, decline in ≥1+ (< 20 cm) brown trout densities from 2012 to 2013 ($F_{1,10}$ = 1.808, P>0.05) and no significant difference between sites ($F_{5,6}$=2.798, P>0.05). ≥1+ (> 20cm) brown trout densities were significantly lower than ≥1+ (< 20cm) brown trout densities at impact sites in the before period (P<0.05), with densities ranging from 0.6 to 7.6 fish/100 m^2 (Table 5.6). There were no significant differences between ≥1+ (> 20cm) brown trout densities between the before years ($F_{1,10}$ = 0.124, P>0.05) (Figure 5.5), but there were significant differences in ≥1+ (> 20cm) brown trout densities between impact sites ($F_{5,6}$ = 6.00, P<0.05).

≥1+ brown trout densities were much more varied at control sites compared to the impact sites (Figure 5.5), ranging from 0.98 to 42.98 fish/100 m^2; populations of ≥1+ brown trout at control sites ranged from poor to excellent (Appendix, Table A.3). ≥1+ (<20 cm) brown trout dominated catches, but there were no significant differences between the two size classes in the control sites unlike observations in the impact sites (Figure 5.5). ≥1+ (< 20cm) brown trout were absent from site DB3 in 2013 (Table 5.5). At the remaining control sites, ≥1+ (< 20cm) brown trout densities ranged from 0.65 to 35.28 fish/100 m^2 (Table 5.5). Densities of ≥1+ (< 20cm) brown trout varied significantly between sites ($F_{2,7}$= 7.679, P<0.05), but there were no significant differences between densities of ≥1+ (< 20cm) brown trout across the five control rivers ($F_{4,7}$ = 3.841, P>0.05). Therefore, it cannot be said that any of the five control rivers held better densities of ≥1+ (< 20cm) brown trout than any other sites. Densities of ≥1+ (< 20cm) brown trout did not vary significantly between the two before years ($F_{1,12}$=0.498, P>0.05). ≥1+ (> 20cm) brown trout were typically found at lower densities in the control sites than ≥1+ (< 20cm) brown trout, with densities of this larger size class ranging from 0.33 to 6.89 fish/100 m^2 during the before period. Unlike the smaller ≥1+ (< 20cm) brown trout there were significant differences between the densities of ≥1+ (> 20cm) brown trout across the five control rivers, with densities of ≥1+ (> 20cm) brown trout lower in sites at the Little Don and River Sheaf compared to the other rivers (Table 5.6).

Prior to the introduction of the flow trial in 2014, a resource calculation was performed on both size classes of ≥1+ brown trout to determine if the intensity of sampling was adequate to detect a biologically meaningful change (≥50%) to ≥1+ brown trout densities in both size classes. In both cases actual variance was lower than the target variance, therefore if the level of temporal variance persists a statistically significant change to densities of either ≥1+ brown trout size classes would be detected within the timeframe of this study (Table 5.6)

During compensation release (After)

There was a marginal improvement in ≥1+ brown trout densities in the impact sites during the after period, with densities ranging from 5.8 to 19.5 fish/100 m^2, with all populations classed as either average or good (appendix, Table A.3). ≥1+ (<20 cm) brown trout densities showed no significant variability between 2014 to 2016 ($F_{1,16}$= 4.372, P>0.05). Densities at the impact sites ranged from 5.42 to 17.24 fish/100 m^2 (Table 5.5). However, density estimates varied significantly between sites ($F_{5,12}$=3.416, P<0.005), for example, densities of ≥1+ brown trout at site DD1 were typically higher than at other sites (Table 5.5). Densities of ≥1+ (>20 cm) brown trout were significantly lower than densities of ≥1+ (<20 cm) brown trout (P<0.05) (Figure 5.4). Densities of ≥1+ (>20 cm) brown trout varied significantly between sites ($F_{5,12}$ = 44.44, P<0.05), for example, site DD4 typically yielded the highest densities of ≥1+ (>20 cm) brown trout and the most upstream sites (DD1 and DD2) typically yielded the lowest (Table 5.6). Unlike the ≥1+ (< 20 cm) brown trout, there was little difference between the densities of ≥1+ (>20 cm) brown trout between years ($F_{1,16}$ = 0.011, P>0.05) (Figure 5.4).

≥1+ brown trout in the control sites more varied than the impact sites, with densities ranging from 2.7 to 32.4 fish/100 m^2 during the after period (appendix, Table A.3). ≥1+ (<20 cm) brown trout densities ranged from 1.13 to 31.04 fish/100 m^2 at control sites during the after period (Table 5.5). As seen in the before period, there were significant difference between ≥1+ (<20 cm) brown trout densities amongst the control sites during the after period, but there was no significant difference between the five control rivers, suggesting spatial variance was manifested in between site differences and not that certain rivers held better ≥1+ (<20 cm) brown trout densities. Unlike at the impact sites, there was no significant difference between the ≥1+ (<20 cm) brown trout densities between the years. ≥1+ (>20 cm) brown trout were caught at all control sites in the after periods, and in densities that were not significantly lower than the ≥1+ (<20 cm) brown trout size class, with densities ranging from 0.33 to 10.62 fish/100 m^2 (Table 5.6). There were significant differences between the ≥1+ (>20 cm) brown trout densities between the five control rivers ($F_{4,14}$ = 9.819, P<0.05) and sites nested within them ($F_{2,14}$=12.681, P<0.05), suggesting that different rivers supported greater ≥1+ (>20 cm) densities compared to others. Similarly, to the ≥1+ (<20 cm) densities, there were no significant fluctuations of the ≥1+ (>20 cm) brown trout densities between years (Figure 5.4).

Table 5.5. Density estimates ± 95% C.L. of ≥1+ brown trout (< 20 cm) at impact and control sites during both the before (2012 – 2013) and after (2014 – 2016) periods

River Name	**Site Identifier**	**2012**	**2013**	**2014**	**2015**	**2016**
		Before			**After**	
impact Sites						
Dale Dike	DD1	12.04 ± 0.8	11.81 ± 0.5	14.36 ± 3.6	12.24 ± 0.2	11.33 ± 1.3
Dale Dike	DD2	4.52 ± 0.6	2.93 ± 0	7.36 ± 0.8	5.42 ± 0.1	6.35 ± 1.7
Dale Dike	DD3	10.98 ± 0.7	8.89 ± 0.6	17.24 ± 0.5	11.11 ± 0	8.65 ± 0.1
Dale Dike	DD4	11.83 ± 0.5	4.79 ± 0.1	11.37 ± 0.6	10.81 ± 0.5	6.99 ± 0.5
Dale Dike	DD5	7.94 ± 0.8	7.96 ± 0.4	6.62 ± 0.3	9.57 ± 0.1	6.84 ± 0.2
Dale Dike	DD6	13.31 ± 0.6	9.03 ± 0.4	13.78 ± 0.3	11.57 ± 1.4	9.64 ± 0.2
Control Sites						
Ewden Beck	DR1	8.17 ± 0.6	8.95 ± 0.5	8.47 ± 0.7	4.47 ± 0.42	4.34 ± 0.73
Ewden Beck	DR2	35.28 ± 1.7	14.95 ± 0.6	11.99 ± 0.5	11.28 ± 0.7	18.36 ± 1.93
Little Don	DR3	1.46 ± 0.1	0 ± 0	1.67 ± 0.1	6.34 ± 0.29	1.86 ± 0.23
River Don	DR4	3.69 ± 1	2.43 ± 0.7	3.57 ± 0.9	3.09 ± 1.31	1.13 ± 0.13
River Don	DR5	16.53 ± 1.2	19.06 ± 0.6	21.87 ± 1	31.04 ± 1.45	18.47 ± 1.4
River Sheaf	DR6	3.3 ± 0.3	0.65 ± 0	2.85 ± 0.2	4.01 ± 0	3.56 ± 0.15
Wyming Brook	DR7	12.99 ± 2	9.55 ± 1.8	15.04 ± 2.2	10.69 ± 1.81	4.83 ± 1.06

Table 5.6. Density estimates ± 95% C.L. of ≥1+ brown trout (>20 cm) at impact and control sites during both the before (2012 – 2013) and after (2014 – 2016) periods.

River Name	Site Identifier	2012	2013	2014	2015	2016
		Before			After	
impact Sites						
Dale Dike	DD1	0.57 ± 0	0.95 ± 0.4	0.51 ± 0	0.51 ± 0.5	1.26 ± 0.3
Dale Dike	DD2	0.41 ± 0	0.73 ± 0.2	0.43 ± 0	0.42 ± 0.3	0.45 ± 0
Dale Dike	DD3	2.27 ± 0.3	1.55 ± 0	2.30 ± 0.1	1.43 ± 0	3.01 ± 0.2
Dale Dike	DD4	7.56 ± 1.4	6.29 ± 0.5	7.69 ± 0.5	5.79 ± 0	7.76 ± 0.7
Dale Dike	DD5	1.32 ± 0	3.54 ± 0.3	4.64 ± 1	4.35 ± 0.3	4.56 ± 0.2
Dale Dike	DD6	4.93 ± 0.1	1.06 ± 0.6	2.87 ± 0	3.31 ± 0.5	2.07 ± 0.6
Control Sites						
Ewden Beck	DR1	6.04 ± 1.2	3.84 ± 0.4	3.23 ± 0	4.07 ± 0.4	1.09 ± 0
Ewden Beck	DR2	6.89 ± 0.9	6.45 ± 0.3	10.62 ± 0.5	7.52 ± 0.63	7.98 ± 0.8
Little Don	DR3	1.46 ± 0.3	1.24 ± 0.1	1.00 ± 0	1.19 ± 0.33	1.12 ± 0.17
River Don	DR4	4.53 ± 1.8	3.64 ± 0.3	4.46 ± 0.2	4.54 ± 1.03	2.45 ± 0.29
River Don	DR5	1.8 ± 0.2	1.12 ± 0.1	2.4 ± 0	1.35 ± 0	2.38 ± 0.36
River Sheaf	DR6	0.6 ± 0.3	0.33 ± 0	0.36 ± 0	0.33 ± 0	0.71 ± 0
Wyming Brook	DR7	6.49 ± 0.6	3.82 ± 0.4	7.85 ± 0.6	6.42 ± 2.03	2.15 ± 0.7

Table 5.7. Actual Vx for *before* period and Target Vx to detect a ≥50% change to ≥1+ brown trout densities. **Bold and underlined** signifies that significant change to ≥1+ densities could be detected within the study parameters.

	Actual VX	Target Vx for ≥50% Increase	Target Vx for ≥50% Decrease
< 20 cm Before	0.019	**<u>1.143</u>**	**<u>0.127</u>**
> 20 cm Before	0.023	**<u>0.277</u>**	**<u>0.030</u>**

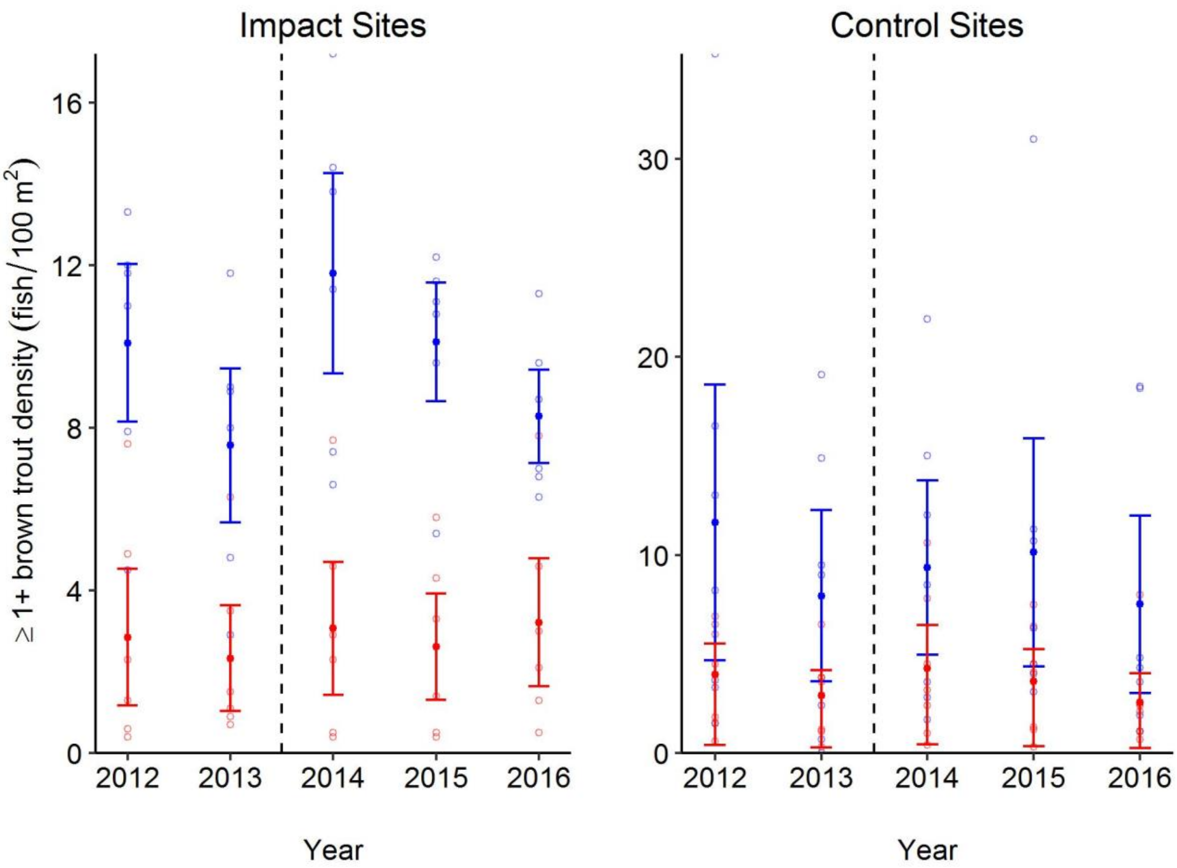

Figure 5.5. Average ≥1+ brown trout densities for all impact sites and control sites. ≥1+ (< 20 cm) data presented in blue, and ≥1+ (>20 cm) in red. The dashed line represents the division between before and after periods. Hollow circles represent individual data points.

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Residual variances from both periods (before and after) were incorporated into a diagnostic resource calculation. The introduction of the data from the after period introduced more variability into the model for both size categories. The actual variance for the ≥1+ (<20 cm) brown densities remained below the target variance, but the increased spatial and temporal variability in the ≥1+ (>20 cm) brown trout densities elevated the actual Vx greater than the target variance (Table 5.8). Therefore, whilst a biologically meaningful change would be detected on the ≥1+ (<20 cm) brown trout in the impact sites, it was not possible to disentangle any flow trial induced change from natural variability within the timeframe of this study. Crucially the actual variance was still lower than the target variances in both instances, confirming that identifying a ≥50% change in ≥1+ brown trout densities would be achievable within the timeframe of this study (Table 5.8).

Table 5.8. Actual Vx for the combined before and after periods and Target Vx to detect a ≥50% change to ≥1+ brown trout densities. **<u>Bold and underlined</u>** signifies that significant change to ≥1+ densities could be detected within the study parameters

	Actual VX	Target Vx for ≥50% Increase	Target Vx for ≥50% Decrease
< 20 cm	0.036	**<u>1.143</u>**	**<u>0.127</u>**
> 20 cm	1.102	0.277	0.030

There were marginal increases in densities of both size classes of ≥1+ brown trout at the impact sites following the introduction of the flow trial in 2014 (Figure 5.6 & Figure 5.7). In breaking down the BACI GLMM outputs, there were no significant differences between the impact and control sites for ≥1+ (<20 cm) brown trout (Table 5.9), or ≥1+ (>20 cm) brown trout (Table 5.9). The interaction between area and period was not significant for both <20 cm (Table 5.9) and >20 cm (Table 5.9) ≥1+ brown trout densities. Therefore, it is probable that the introduction of the compensation release had no influence on densities of ≥1+ (<20 cm) and ≥1+ (>20cm) brown trout. It is worth noting that the level of variance in the ≥1+ (>20 cm) brown trout densities was so high that any changes to the densities could not be isolated from natural variability (Table 5.8).

Table 5.9 BACI GLMM to detect a change in ≥1+ brown trout density following the introduction of the compensation flow from Dale Dike Reservoir in 2014. **<u>Bold and underlined</u>** signifies a significant ($P<0.05$) interaction in the GLMM

	Sum Sq	Mean Sq	DoF	DenDF	F	P
≥1+ (<20 cm)						
Area	0.074	0.074	1	11.045	0.721	0.414
Period	0.035	0.035	1	3.007	0.343	0.599
Area:Period	0.026	0.026	1	47.000	0.252	0.618
≥1+ (>20 cm)						
Area	0.017	0.017	1	11.033	0.246	0.630
Period	0.011	0.011	1	3.027	0.166	0.711
Area:Period	0.742	0.742	1	47.000	1.067	0.307

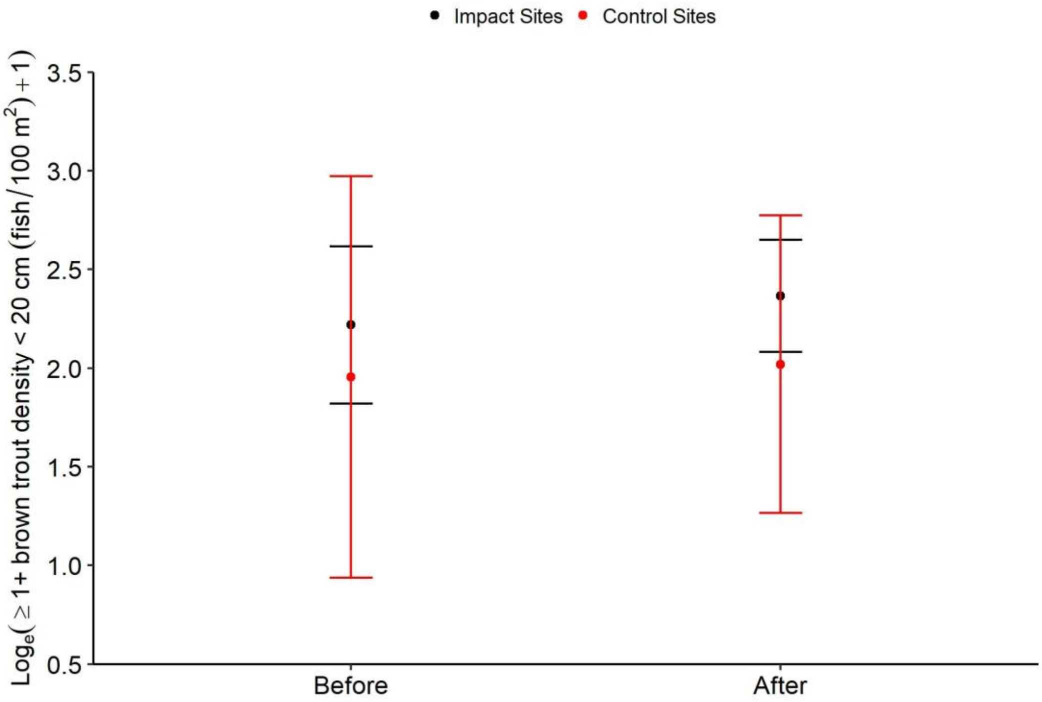

Figure 5.6. Mean (±95%) ≥1+ (< 20 cm) brown trout densities for the before and after periods for impact (black) and control (red) sites.

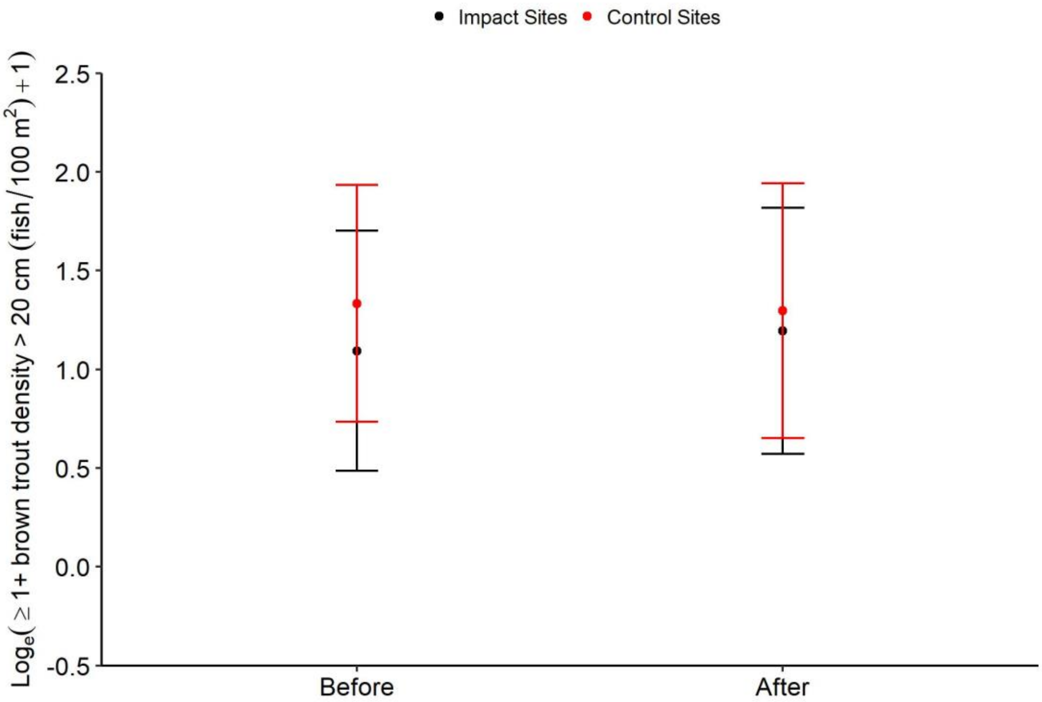

Figure 5.7. Mean (±95%) ≥1+ (> 20 cm) brown trout densities for the before and after periods for impact (black) and control (red) sites.

5.3.3 Impacts of the introduction of compensation flow on the length at capture of 0+ brown trout

Baseline monitoring (Before)

The length at which 0+ brown trout were captured varied spatially and temporally throughout the study. The length at capture of 0+ brown trout in Dale Dike during the study period ranged from 41 to 89 mm with an average length of 57 mm in 2012 and 2013 (Figure 5.8). Despite the large range at which 0+ brown trout were caught, there were no significant differences between impact sites ($F_{5,5} = 4.842$, $P>0.05$) during the before period. The size of 0+ brown trout in the control sites was much more varied, with lengths at capture ranging from 42 – 96 mm during the baseline period. Average lengths of 0+ brown trout were significantly larger in the control sites than the impact sites ($P< 0.05$) (Figure 5.8). For the control sites, there were significant differences between the length at capture of 0+ brown trout both between sites ($F_{6,6} = 5.85$ $P<0.05$) and between rivers ($F_{4,8} = 4.46$, $P<0.05$). The smallest fish in both the impact and control sites were caught in 2013, with these individuals 5 and 8 mm smaller than the next smallest fish caught in other years of the study (Figure 5.8). However, the differences between the length at capture of 0+ brown trout between years at impact sites ($F_{1,9} = 0.011$, $P>0.05$) or control sites ($F_{1,11} = 0.479$, $P>0.05$) prior to the introduction of the compensation release were not significant. Size ranges for all sites in all years are in Appendix (Table A.4)

Prior to the introduction of the compensation release a resource calculation was constructed to determine if the level of before monitoring was adequate to detect a ≥25% change to lengths at capture of 0+ brown trout in the impact sites. The target variance was lower than the actual variance in both instances, therefore, a significant change to lengths of 0+ brown trout would be detectable if the level of temporal and spatial variance present in 2012-2013 persists (Table 5.10).

Table 5.10 Actual Vx for the before period and Target Vx to detect a ≥25% change to 0+ brown trout lengths. **<u>Bold and underlined</u>** signifies that significant change to 0+ lengths could be detected within the study parameters

Actual Vx	Target Vx for ≥25% Increase	Target Vx for ≥25% Decrease
3.772	**<u>420.547</u>**	**<u>281.519</u>**

During compensation release (after) 0+ brown trout remained within a similar size range (46 to 87 mm) in the impact sites during the flow trial as during the baseline study (Figure 5.8), with no significant differences between the length at capture of 0+ brown trout at impact sites ($F_{5,11}$=1.509, P>0.05). Average length at capture of 0+ brown trout in the impact sites in 2014 and 2015 were within a similar size range as in the baseline period (58 mm in both years), but the average length in 2016 was marginally larger at 62 mm. Average length at capture of 0+ brown trout was marginally larger in 2016 than in other years, with the mean 2016 length significantly larger than in 2013 (P< 0.05) (Figure 5.8), but there were no significant differences between the three after period years ($F_{1,15}$=1.823, P>0.05).

There was more variation in the length at capture in the control sites than at impact sites during this period, similar to observations in the before period, with 0+ brown trout caught in the size range of 50 to 97 mm (Figure 5.8). Unlike the before period, there were no significant differences between length at capture for 0+ brown trout between control sites ($F_{6,13}$=1.636, P>0.05) and rivers ($F_{4,15}$=1.241, P>0.05). However, there was a similar trend in the control sites as observed in the impact sites, with the average length at capture being larger in 2016 than other years, but not significantly so (P >0.05) (Figure 5.8).

Size ranges for 0+ brown trout for all sites and all years are in Appendix (Table A.4)

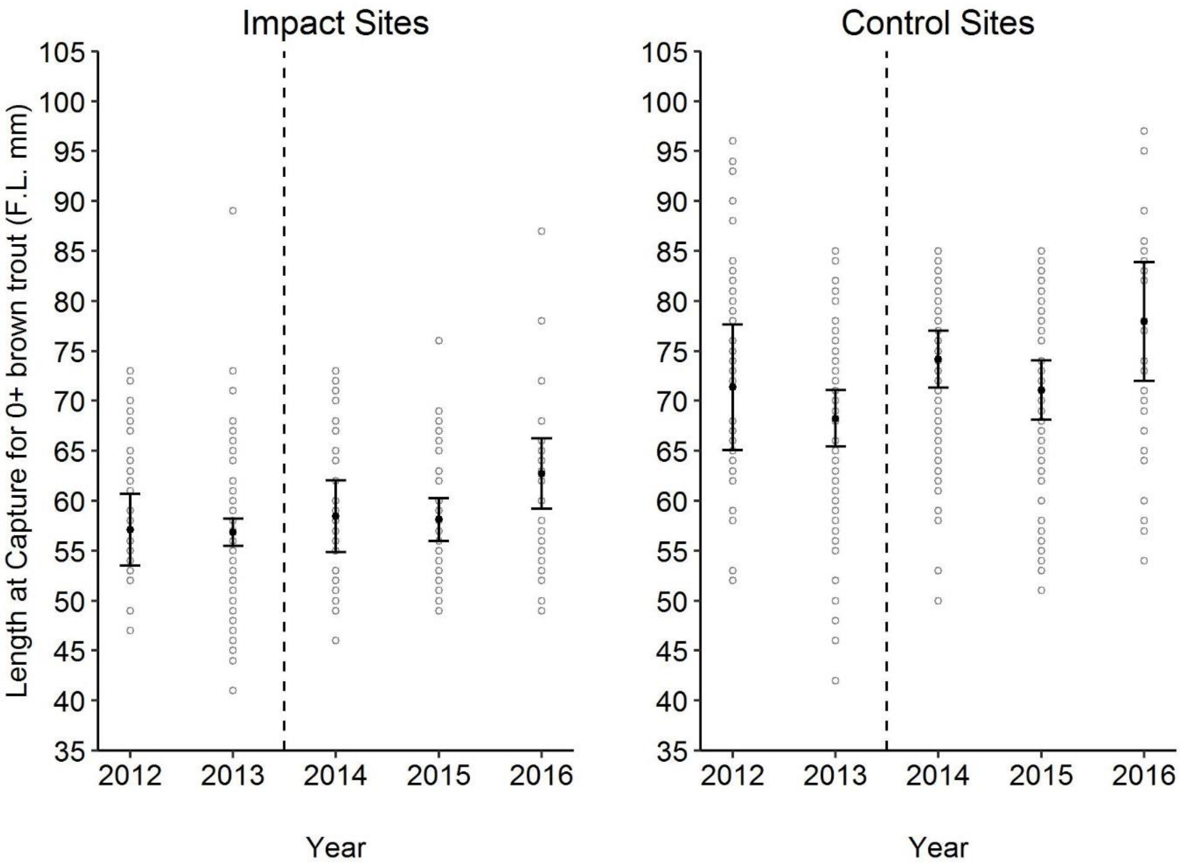

Figure 5.8. Lengths at capture for individual 0+ brown trout (greyed circles). As well as average length (± 95% C.L.) for impact and control sites. Dashed line represents division between before and after period

Impact assessment (BACI)

The addition of after (2014-2016) year's data introduced greater variance into the dataset, as the level of actual variance (AVx) was greater than in the previous resource calculation. However, this level of variance was still considerably lower than the target variance in both instances, therefore a ≥25% change to 0+ brown trout sizes could be detected within the timeframe of this study (Table 5.11).

Table 5.11. Actual Vx for the full study and Target Vx to detect a ≥25% change to 0+ brown trout lengths. **<u>Bold and underlined</u>** signifies that significant change to 0+ lengths could be detected within the study parameters

Actual Vx	Target Vx for ≥25% Increase	Target Vx for ≥25% Decrease
5.819	**<u>420.547</u>**	**<u>281.519</u>**

Throughout the study, 0+ brown trout were caught at significantly larger sizes in the control sites than the impact sites (Table 5.12) (Figure 5.9). There was a marginal increase in the lengths at capture between the impact and control sites, but this increase was no significant (Table 5.12). Crucially there was no significant interaction between area and period in the BACI GLMM (Table 5.12), therefore no significant change to length at capture for 0+ brown trout in Dale Dike can be attributed to the introduction of the flow trial from Dale Dike reservoir.

Table 5.12 BACI GLMM to detect a change in 0+ brown trout length at capture following introduction of compensation release from Dale Dike reservoir in 2014

	Sum Sq	Mean Sq	DoF	DenDF	F	P
Area	818.43	818.43	1	11.092	33.447	**<u><0.001</u>**
Period	63.000	63.000	1	2.925	2.575	0.209
Area:Period	4.37	4.37	1	44.021	0.179	0.674

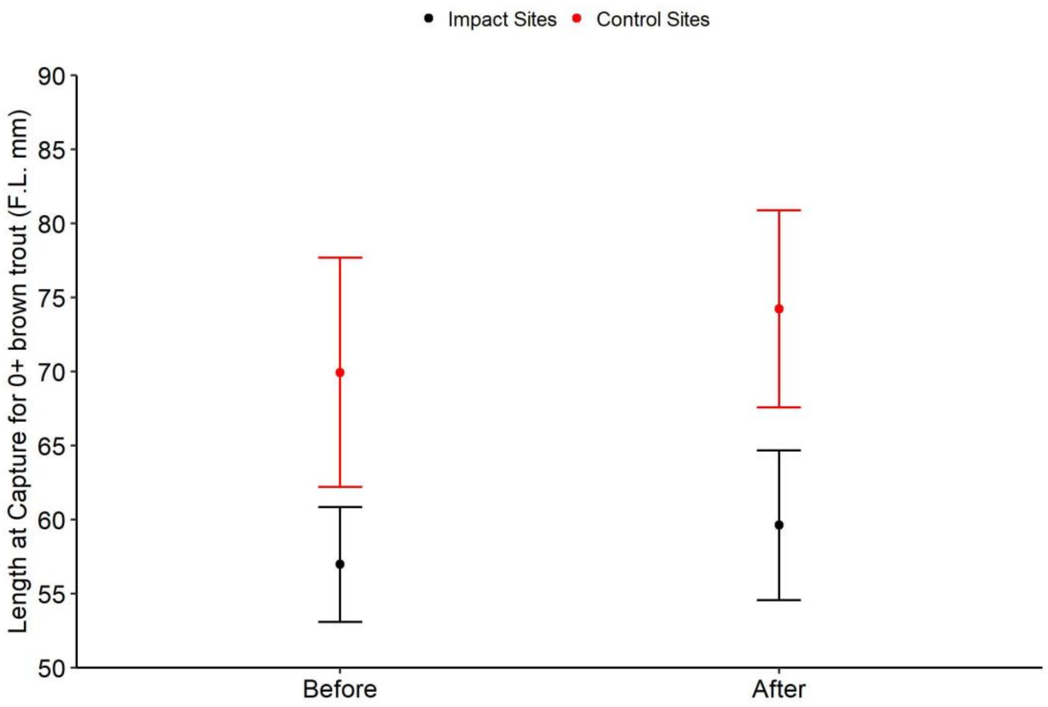

Figure 5.9. Mean (±95%) 0+ brown trout lengths *before* and *after* for impact (black) and control (red) sites.

5.3.4 Impacts of the compensation release on the habitat quality (HQS density) for 0+ brown trout

Baseline monitoring (Before)

During the before period the 0+ habitat quality score (HQS) varied between sites and years in the impact sites, with expected 0+ brown trout populations in pristine conditions ranging from 4.09 – 19.4 fish/100 m^2, with these populations predominantly classed as average (Table 5.13). Habitat quality did not vary significantly between 2012 and 2013 in the impact sites ($P>0.05$). Observed densities of 0+ brown trout were lower than expected at all impact sites during the before period, with the exception of sites DD3 in 2013. Observed 0+ brown trout densities at sites DD2 –DD6 was significantly lower than expected in 2012 ($P<0.05$) (Figure 5.11).

0+ brown trout habitat quality was lower in the control sites than the impact sites, with 0+ brown trout populations expected under pristine conditions ranging from 1.7 to 13.5 fish/100 m^2 in 2012 (Table 5.13). There were no significant differences between the habitat quality in the five control rivers during the before period ($F_{4,7}$= 1.753, $P>0.05$) or sites nested within them ($F_{2,7}$=3.137, $P>0.05$), suggesting that the level of heterogeneity between habitats for 0+ brown trout across the control sites was low. Unlike in the impact sites, habitat quality improved in 2013 from 2012 (Figure 5.10), but the majority of sites during this period were classed as fair/poor (Table 5.13); the observed densities of 0+ brown trout were lower than predicted from the habitat at all sites in 2012, but only significantly so at sites WR2 and WR6. Observed densities in 2013 were higher than predicted at sites WR4-WR7, and marginally lower at site WR1. Observed densities at sites WR2-WR3 were significantly lower than predicted (Figure 5.11).

Prior to the introduction of the flow trial in 2014, a resource calculation was constructed to determine if the level of before sampling was adequate to detect a ≥50% change to 0+ HQS at impact sites in Dale Dike. The actual variance in the dataset was lower than the target variance in both instances (Table 5.14), therefore, based on the assumption that the current level of spatial and temporal variance persists, a ≥50% to 0+ brown trout could be detected within the timeframe of this study.

During compensation release (After)

During the after period the spatial and temporal variance of the 0+ brown trout HQS remained similar between years, but the expected 0+ brown trout densities were significantly different between the impact sites during the after period ($F_{5,12}$ = 5.447, $P<0.05$). Expected 0+ brown trout populations ranged from 3.73 to 19.6 fish/100 m^2 with populations predominantly classed as average during this period. The pooled 0+ brown trout HQS densities remained similar throughout the after period (10.7, 10.9, 11.0 fish/100 m^2, for 2014, 2015 and 2016 (Table 5.13)) demonstrating that there was no significant difference between the three after years ($F_{1,16}$=0.012, $P>0.05$) at impact sites. Observed densities of 0+ brown trout were lower than expected at all sites during the after period, with densities at sites DD5 and DD6 in 2015 and DD3 – DD5 in 2016 significantly lower than predicted (Figure 5.11).

Control sites were more varied from 2014-2016, with expected 0+ brown trout densities under ranging from 2.4 to 17.9 fish/100 m^2, with these populations predominantly classed as average (Table 5.13). Unlike the before period, 0+ brown trout habitat quality varied significantly between rivers ($F_{4,14}$=7.830, P<0.05) but not sites nested within them ($F_{2,14}$=3.619, P>0.05), suggesting a stratification of habitat quality between the five control rivers. Pooled habitat quality scores for the control sites suggest that there was no significant change to HQS densities in the after period (Figure 5.10). There were no significant differences between HQS throughout the after period ($F_{1,19}$ = 0.01, P>0.05). The observed densities of 0+ brown trout in the control sites were higher than expected at sites WR4-WR6 in 2014 and 2015, and lower than expected at the remainder of sites and years in the after period (Figure 5.11), with expected populations significantly lower at sites WR1 and WR2 in 2012, WR1-WR3 and WR7 in 2015, and at sites WR1-WR3, WR5 and WR6 in 2016 (Figure 5.11).

Table 5.13. 0+ HQS densities related to the EA-FCS categories for impact and control sites during the before (2012 - 2013) and after (2014 – 2016) periods.

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

River Name	Site Identifier	2012	2013	2014	2015	2016
		Before		After		
impact Sites						
Dale Dike	DD1	19.39	14.83	9.83	10.63	14.90
Dale Dike	DD2	7.95	6.35	6.71	9.74	3.73
Dale Dike	DD3	13.32	7.82	9.48	7.08	11.49
Dale Dike	DD4	10.90	4.09	8.13	8.40	9.57
Dale Dike	DD5	12.17	19.17	15.06	17.75	19.64
Dale Dike	DD6	16.60	12.93	15.20	11.82	6.70
Control Sites						
Ewden Beck	DR1	11.62	8.16	10.36	14.14	13.43
Ewden Beck	DR2	7.52	4.33	8.45	9.77	9.69
Little Don	DR3	5.85	9.83	7.91	12.58	4.20
River Don	DR4	2.21	2.42	3.04	2.59	2.35
River Don	DR5	5.11	7.19	7.42	8.45	6.60
River Sheaf	DR6	4.57	6.15	5.61	7.80	12.62
Wyming Brook	DR7	8.89	4.09	16.21	17.90	10.75

Table 5.14. Actual Vx for the before period and Target Vx to detect a ≥50% change to 0+ HQS Densities. **Bold and underlined** signifies that significant change to 0+ HQS Densities could be detected within the study parameters

Actual Vx	Target Vx for ≥50% Increase	Target Vx for ≥50% Decrease
0.023	**<u>1.207</u>**	**<u>0.134</u>**

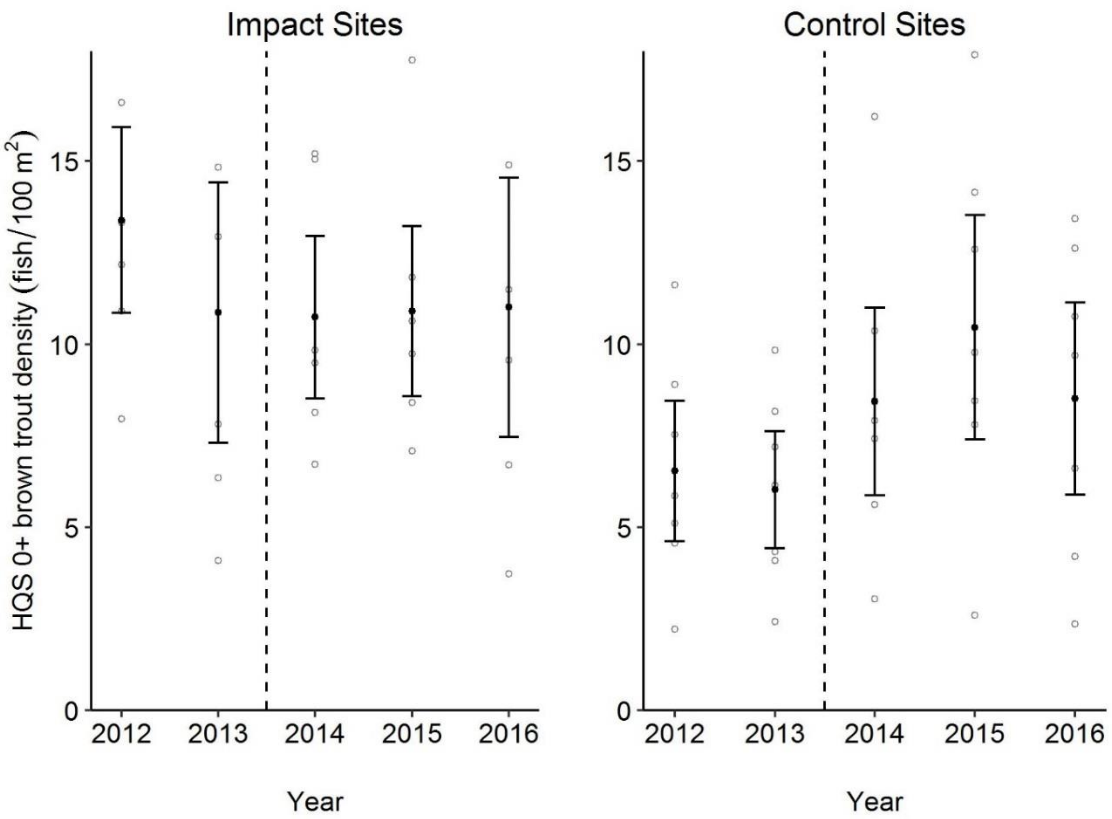

Figure 5.10. Average 0+ brown trout HQS densities ± 95 % C.L. for all impact and control sites, throughout the study period. The dashed line represents the division between *before* and *after* periods. Greyed circles represent individual density estimates

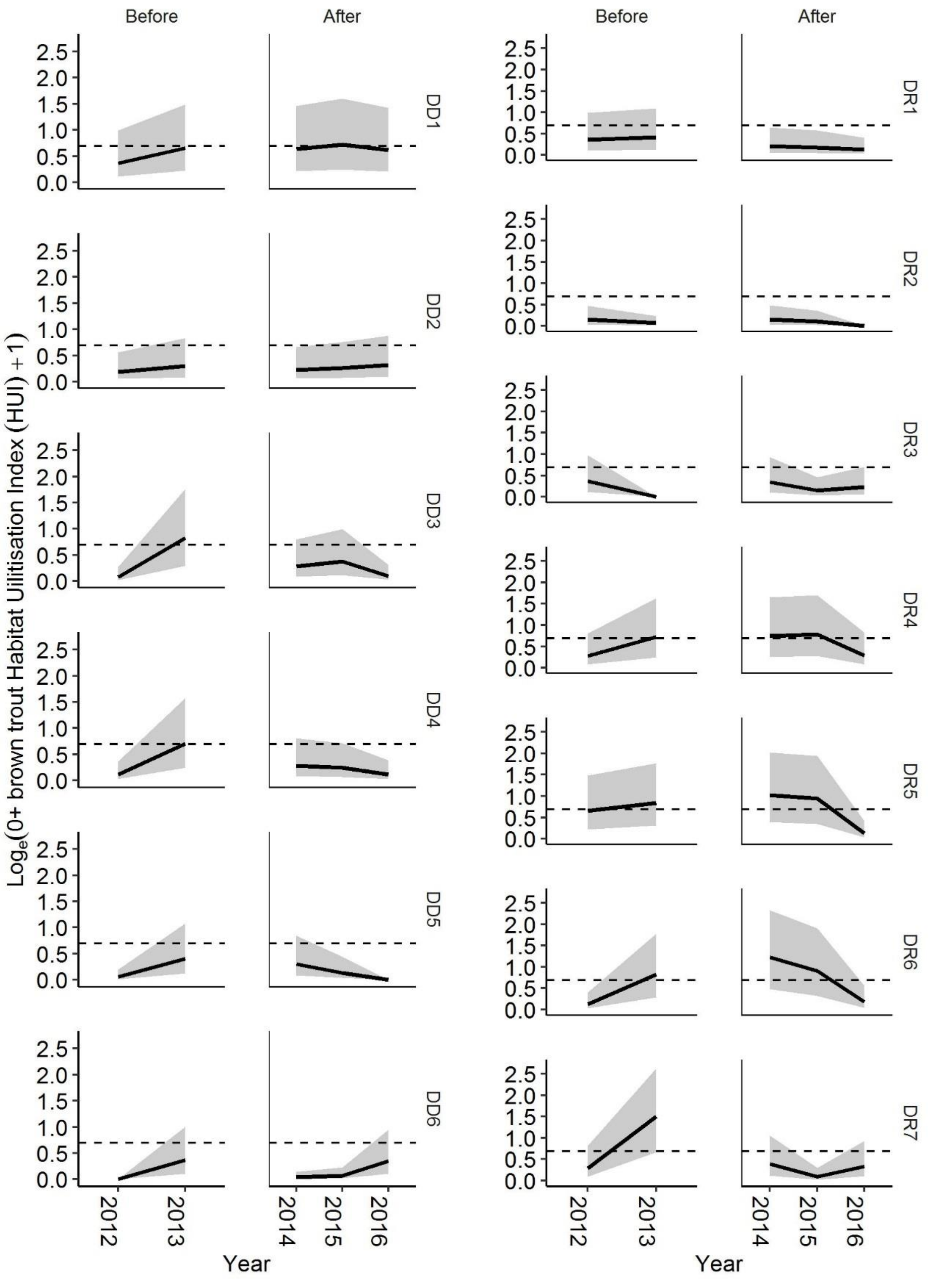

Figure 5.11. Log$_e$ + 1 transformed HUI for 0+ brown trout in the impact (left) and control sites (right) during the before (2012 – 2013) and after (2014 – 2016) period. Grey bands represent 95% confidence limits. The dashed line represents HUI value where observed and expected densities are equal.

Impact Assessment (BACI)

A resource calculation performed on the full data set revealed a marginal decrease in the actual variance after the addition of the after-period dataset. Therefore a ≥50% change to 0+ brown trout habitat quality could be detected within the timeframe of this study (Table 5.14).

Table 5.15. Actual Vx for the full study and Target Vx to detect a ≥50% change to 0+ HQS Densities. **<u>Bold and underlined</u>** signifies that significant change to 0+ HQS Densities could be detected within the study parameters

Actual Vx	Target Vx for ≥50% Increase	Target Vx for ≥50% Decrease
0.021	**<u>1.207</u>**	**<u>0.134</u>**

0+ brown trout habitat quality decreased between the two study periods, and during this same timeframe the quality of the 0+ habitat at the control sites increased (Figure 5.12). it was apparent that the habitat quality was marginally better in Dale Dike than the control sites during both periods of the study, but not significantly so (Table 5.16). The direction of the change to 0+ habitat quality was different between the two periods for both impact and control sites but not significantly (Table 5.16) (Figure 5.12). However, there was a significant interaction between area and period in the BACI GLMM (Table 5.16), therefore it can be said that there was a significant decrease in 0+ habitat quality in Dale Dike during the study period.

Table 5.16 BACI GLMM to detect a change in 0+ brown trout HQS density following the introduction of the compensation release from Dale Dike Reservoir in 2014

	Sum Sq	Mean Sq	DoF	DenDF	F	P
Area	0.277	0.277	1	11.102	3.671	0.081
Period	0.148	0.148	1	3.030	1.958	0.255
Area:Period	0.572	0.572	1	47.000	7.589	**<u>0.008</u>**

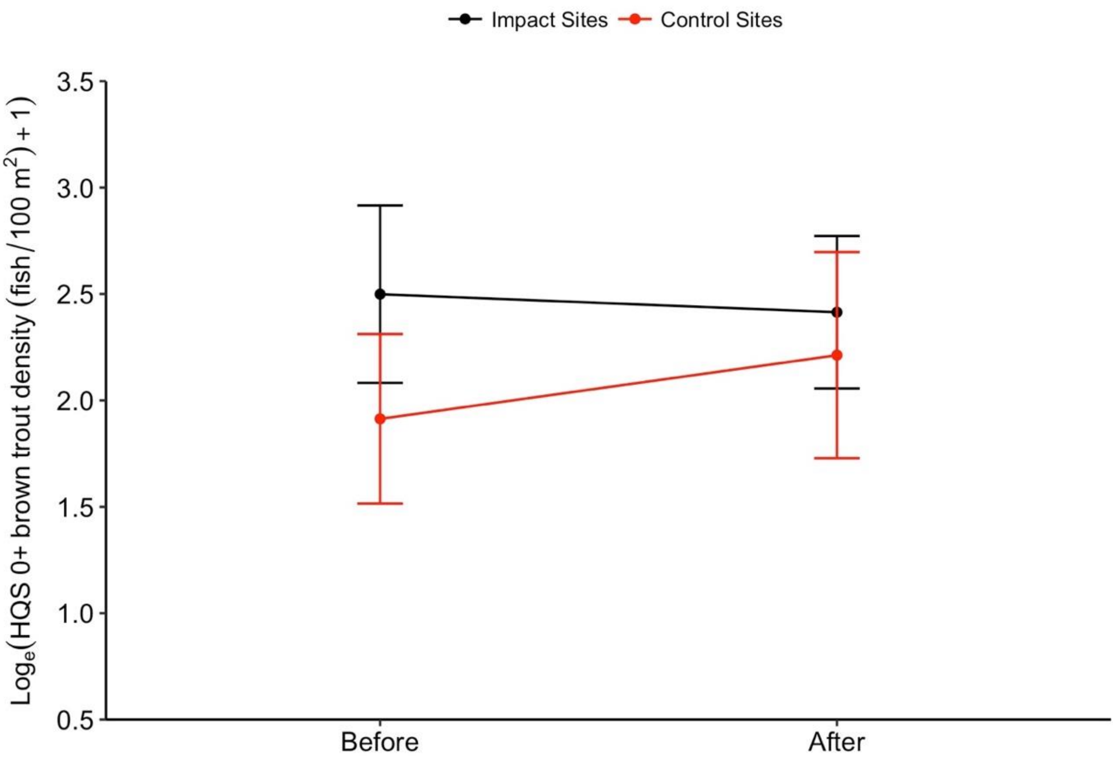

Figure 5.12. Mean (±95%) 0+ brown trout HQS densities during before and after periods for impact (black) and control (red) sites. Trendlines represent significant interaction between Area and Period in BACI GLMM.

5.3.5 Impacts of the introduction of the compensation flow on the habitat quality (HABSCORE) for ≥ 1+ brown trout

Baseline monitoring (Before)

During the before period the HQS for ≥1+ brown trout varied between sites and years. For the impact sites expected densities of ≥1+ (<20 cm) brown trout ranged from 0.75 to 20.8 fish/100 m^2 (Table 5.17). The quality of habitat for ≥1+ (<20 cm) brown trout varied between the two years, with expected densities of brown trout significantly lower in 2012 than 2013 ($P<0.05$) , but there were no significant differences in habitat quality for ≥1+ (<20 cm) brown trout at site level ($F=0.29$, $P>0.05$). Observed densities of ≥1+ (<20 cm) brown trout were greater than indicated from the habitat quality at all impact sites, and significantly greater at sites DD2, DD3, DD4 and DD5. However, in 2013 the observed densities of ≥1+ (<20 cm) brown trout were lower than expected at all impact sites (Figure 5.13). The expected densities of ≥1+ (>20 cm) brown trout ranged from 0.75 to 5.59 fish/100 m^2 at the impact sites in 2012-2013 (Table 5.18, Figure 5.13). Unlike the ≥1+ (<20 cm) brown trout HQS, there was no significant increase in ≥1+ (>20 cm) brown trout habitat quality from 2012 to 2013 (Figure 5.16). There were no significant variations between ≥1+ (>20 cm) brown trout habitat quality between the sites, however, the expected densities of ≥1+ (>20 cm) brown trout were significantly lower in 2013 than ≥1+ (<20 cm) brown trout ($P<0.05$) (Figure 5.17). At three sites (DD1 – DD3) in 2012 and 2013 as well as DD5 and DD6 in 2013 the observed densities of brown trout were lower than predicted, with the observed densities of ≥1+ (>20 cm) brown trout at site DD6 in 2013 significantly lower than predicted from the habitat (Figure 5.17).

HQS densities at control sites during the before period were less varied than the impact sites. HQS densities of ≥1+ (<20 cm) brown trout ranged from 1.13 to 10.85 fish/100 m^2 during this period (Table 5.17). HQS densities for ≥1+ (<20 cm) brown trout marginally improved from 2012 to 2013 however not significantly ($P>0.05$) (Figure 5.13). There were no significant differences between the habitat qualities at either a site or river level in the control sites suggesting that there was homogeneity between the control site habitats. Habitat Utilisation Indexes for ≥1+ (<20 cm) brown trout varied amongst control sites during the before period.

A resource calculation was performed on the HQS data for both size classes of ≥1+ brown trout to determine if a ≥50% change could be detected within the timeframe of the study. For both ≥1+ (<20 cm) brown trout HQS and ≥1+ (>20 cm) brown trout HQS, the level of actual variance was lower than the target variance for both a 50% increase or decrease, suggesting that if the current level of temporal and spatial variance persists a biologically meaningful change would be detected within the timeframe of this study (Table 5.19). In 2012 the observed densities of ≥1+ (<20 cm) brown trout were higher than predicted from the habitat at five sites (DR1, DR3, DR5, DR6 and DR7), but not significantly so; at the remaining two sites (DR2 and DR4) observed densities of ≥1+ (<20 cm) brown trout were lower than predicted. In 2013, observed densities of ≥1+ (<20 cm) brown trout densities were only higher than predicted from the habitat at three sites (DR1, DR2, DR5) with densities at site DR5 significantly higher than predicted. At the remaining four sites (DR3, DR4 DR6 and DR7) observed densities of ≥1+ (<20 cm) brown trout were lower than predicted, but not significantly (Figure 5.14). Habitat quality for ≥1+ (>20 cm) brown trout suggests that the expected densities of ≥1+ (>20 cm) brown trout should be lower than the densities of ≥1+ (<20 cm) brown trout with HQS densities ranging from 0.56 to 6.33 fish/100 m^2 (Table 5.18). As observed in the smaller (≥1+ <20 cm) brown trout, HQS densities for ≥1+ (>20 cm) brown trout marginally (but not significantly ($P>0.05$)) improved in 2013 compared to 2012 (Figure 5.13). There was also no significant difference between the HQS densities at either a river or site level, again reflecting the habitat homogeneity of the control sites. The observed densities of ≥1+ (>20 cm) brown trout at control sites varied in relation to the HQS densities. In 2012, the observed densities of ≥1+ (<20 cm) brown trout at three sites (DR1, DR4 and DR7) were significantly higher than the HQS densities; at the remaining sites (DR2, DR3, DR5, DR6) the observed densities were lower than the HQS densities and significantly lower at sites DR2 and DR6 (Figure 5.15). In 2013, observed densities of ≥1+ (<20 cm) brown trout were greater than the HQS densities at four sites (DR1, DR2, DR4 and DR5) and lower than predicted at the remaining three sites (DR3, DR6 and DR7). In 2013, there was no significant deviation between observed densities and HQS densities at the control sites (Figure 5.15).

During compensation release (After)

The level of temporal variability of the habitat quality for ≥1+ (<20 cm) brown trout was not as dramatic as seen in the before period at impact sites, with no significant differences between the three years. The expected densities of ≥1+ (<20 cm) brown trout ranged from 3.04 to 20.21 fish/100 m^2, but did not deviate significantly between sites ($F = 0.14$, $P>0.05$). There was an overall trend of decline of expected densities of ≥1+ (<20 cm) brown trout from 2013 to 2015 (Figure 5.13). The observed densities of ≥1+ (<20 cm) brown trout were higher than predicted from the habitat quality at all impact sites during the after period, with the exception of site DD3 in 2016, and DD5 in 2014 and 2016, there were, however, no instances of densities significantly higher or lower than predicted at the impact sites (Figure 5.14). The habitat quality of ≥1+ (>20 cm) brown trout suggested that the expected densities of ≥1+ (>20 cm) brown trout should be significantly lower than the densities of ≥1+ (<20 cm) brown trout ($P<0.05$) (Figure 5.15). The habitat quality of ≥1+ (>20 cm) brown trout suggests that densities of ≥1+ (>20 cm) brown trout should range from 0.57 to 3.51 fish/100 m^2. Habitat quality for ≥1+ (>20 cm) brown trout did not significantly deviate between sites ($F = 0.65$, $P>0.05$) or years ($F=0.69$, $P>0.05$) (Figure 5.13). Observed densities of ≥1+ (>20 cm) brown trout were higher than expected from the habitat quality at sites DD3 – DD6 in all years of the after period, with the observed densities at site DD4 and DD5 in 2014 significantly higher. Observed densities of ≥1+ (>20 cm) brown trout at the remaining two sites DD1 and DD2 were lower than expected in all years of the after period but significantly lower in 2014 compared to 2012 and 2013 (Figure 5.15).

During the after period, the ≥1+ brown trout habitat quality remained similar to those recorded in the before periods with ≥1+ (<20 cm) brown trout densities under pristine condition ranging from 0.70 to 11.48 fish/100 m^2 (Table 5.17. There was no significant variation between ≥1+ (<20 cm) brown trout HQS densities between 2014-2016 and similarly there was no significant deviation between ≥1+ (<20 cm) brown trout HQS densities between the five control rivers, however there were significant differences between the sites, suggesting that the spatial variances are manifested in individual site variability and do not operate at a river level. The fluctuations between the observed brown trout densities and the habitat quality for ≥1+ (<20 cm) brown trout at the control sites caused variations in the HUI values. Observed ≥1+ (<20 cm) brown trout densities at sites DR2, DR5 and DR7 were consistently higher than predicted from the habitat for the duration of the after period and observed densities of ≥1+ (<20 cm) brown trout were consistently lower than predicted from the habitat at site DR1 and DR2. At the remaining two sites (DR4 and DR6) observed densities of ≥1+ (<20 cm) brown trout were higher than predicted in 2014 and 2015 and lower than predicted in 2016. In only one instance (DR5 in 2015) was ≥1+ (<20 cm) brown density significantly different from expected from the habitat quality, where they were significantly higher (Figure 5.14). The habitat quality for ≥1+ (>20 cm) brown trout revealed that expected densities should be significantly lower in all three years of the after period than the ≥1+ (<20 cm) brown trout (Figure 5.13). There was little variation in the expected densities of ≥1+ (>20 cm) brown trout with estimates ranging from 0.5 to 3.51 fish/100 m^2 (Table 5.18). There was no significant difference between the years of the after period, but there were significant differences between both control rivers ($F = 6.43$, $P < 0.05$) and sites ($F = 5.94$, $P < 0.05$) suggesting that the differences in habitat quality for ≥1+ (>20 cm) brown trout were manifested both at a site and river levels. Observed densities of ≥1+ (>20 cm) brown trout in the control sites fluctuated between years and sites. At sites DR2, DR4 and DR5 observed densities of ≥1+ (>20 cm) brown trout were higher than expected for the duration of the after period. Similarly observed densities of ≥1+ (>20 cm) brown trout were lower than predicted at sites DR3 and DR6 for the duration of the after period. The observed densities of ≥1+ (>20 cm) brown trout at sites DR1 and DR7 were higher than predicted in 2014 and 2015, but lower than predicted in 2016 (Figure 5.15).

Table 5.17 HQS densities for ≥1+ (<20cm) brown trout at all impact and control sites for the before (2012-2013) and after (2014-2016)

River Name	Site Identifier	2012	2013	2014	2015	2016
		Before			**After**	
impact Sites						
Dale Dike	DD1	0.57 ± 0	0.95 ± 0.4	0.51 ± 0	0.51 ± 0.5	1.26 ± 0.3
Dale Dike	DD2	0.41 ± 0	0.73 ± 0.2	0.43 ± 0	0.42 ± 0.3	0.45 ± 0
Dale Dike	DD3	2.27 ± 0.3	1.55 ± 0	2.3 ± 0.1	1.43 ± 0	3.01 ± 0.2
Dale Dike	DD4	7.56 ± 1.4	6.29 ± 0.5	7.69 ± 0.5	5.79 ± 0	7.76 ± 0.7
Dale Dike	DD5	1.32 ± 0	3.54 ± 0.3	4.64 ± 1	4.35 ± 0.3	4.56 ± 0.2
Dale Dike	DD6	4.93 ± 0.1	1.06 ± 0.6	2.87 ± 0	3.31 ± 0.5	2.07 ± 0.6
Control Sites						
Ewden Beck	DR1	6.04 ± 1.2	3.84 ± 0.4	3.23 ± 0	4.07 ± 0.4	1.09 ± 0
Ewden Beck	DR2	6.89 ± 0.9	6.45 ± 0.3	10.62 ± 0.5	7.52 ± 0.63	7.98 ± 0.8
Little Don	DR3	1.46 ± 0.3	1.24 ± 0.1	1 ± 0	1.19 ± 0.33	1.12 ± 0.17
River Don	DR4	4.53 ± 1.8	3.64 ± 0.3	4.46 ± 0.2	4.54 ± 1.03	2.45 ± 0.29
River Don	DR5	1.8 ± 0.2	1.12 ± 0.1	2.4 ± 0	1.35 ± 0	2.38 ± 0.36
River Sheaf	DR6	0.6 ± 0.3	0.33 ± 0	0.36 ± 0	0.33 ± 0	0.71 ± 0
Wyming Brook	DR7	6.49 ± 0.6	3.82 ± 0.4	7.85 ± 0.6	6.42 ± 2.03	2.15 ± 0.7

Table 5.18 HQS densities for ≥1+ (>20cm) brown trout at all impact and control sites for the before (2012-2013) and after (2014-2016

River Name	Site Identifier	2012	2013	2014	2015	2016
		Before			After	
impact Sites						
Dale Dike	DD1	0.57 ± 0	0.95 ± 0.4	0.51 ± 0	0.51 ± 0.5	1.26 ± 0.3
Dale Dike	DD2	0.41 ± 0	0.73 ± 0.2	0.43 ± 0	0.42 ± 0.3	0.45 ± 0
Dale Dike	DD3	2.27 ± 0.3	1.55 ± 0	2.3 ± 0.1	1.43 ± 0	3.01 ± 0.2
Dale Dike	DD4	7.56 ± 1.4	6.29 ± 0.5	7.69 ± 0.5	5.79 ± 0	7.76 ± 0.7
Dale Dike	DD5	1.32 ± 0	3.54 ± 0.3	4.64 ± 1	4.35 ± 0.3	4.56 ± 0.2
Dale Dike	DD6	4.93 ± 0.1	1.06 ± 0.6	2.87 ± 0	3.31 ± 0.5	2.07 ± 0.6
Control Sites						
Ewden Beck	DR1	6.04 ± 1.2	3.84 ± 0.4	3.23 ± 0	4.07 ± 0.4	1.09 ± 0
Ewden Beck	DR2	6.89 ± 0.9	6.45 ± 0.3	10.62 ± 0.5	7.52 ± 0.63	7.98 ± 0.8
Little Don	DR3	1.46 ± 0.3	1.24 ± 0.1	1 ± 0	1.19 ± 0.33	1.12 ± 0.17
River Don	DR4	4.53 ± 1.8	3.64 ± 0.3	4.46 ± 0.2	4.54 ± 1.03	2.45 ± 0.29
River Don	DR5	1.8 ± 0.2	1.12 ± 0.1	2.4 ± 0	1.35 ± 0	2.38 ± 0.36
River Sheaf	DR6	0.6 ± 0.3	0.33 ± 0	0.36 ± 0	0.33 ± 0	0.71 ± 0
Wyming Brook	DR7	6.49 ± 0.6	3.82 ± 0.4	7.85 ± 0.6	6.42 ± 2.03	2.15 ± 0.7

Table 5.19 Actual Vx for *before* period and Target Vx to detect a ≥50% change to ≥1+ brown trout Habitat Quality (HQS). **<u>Bold and underlined</u>** signifies that significant change to ≥1+ densities could be detected within the study parameters.

	Actual VX	Target Vx for ≥50% Increase	Target Vx for ≥50% Decrease
< 20 cm Before	0.080	**<u>0.852</u>**	**<u>0.090</u>**
> 20 cm Before	0.302	**<u>0.092</u>**	**<u>0.034</u>**

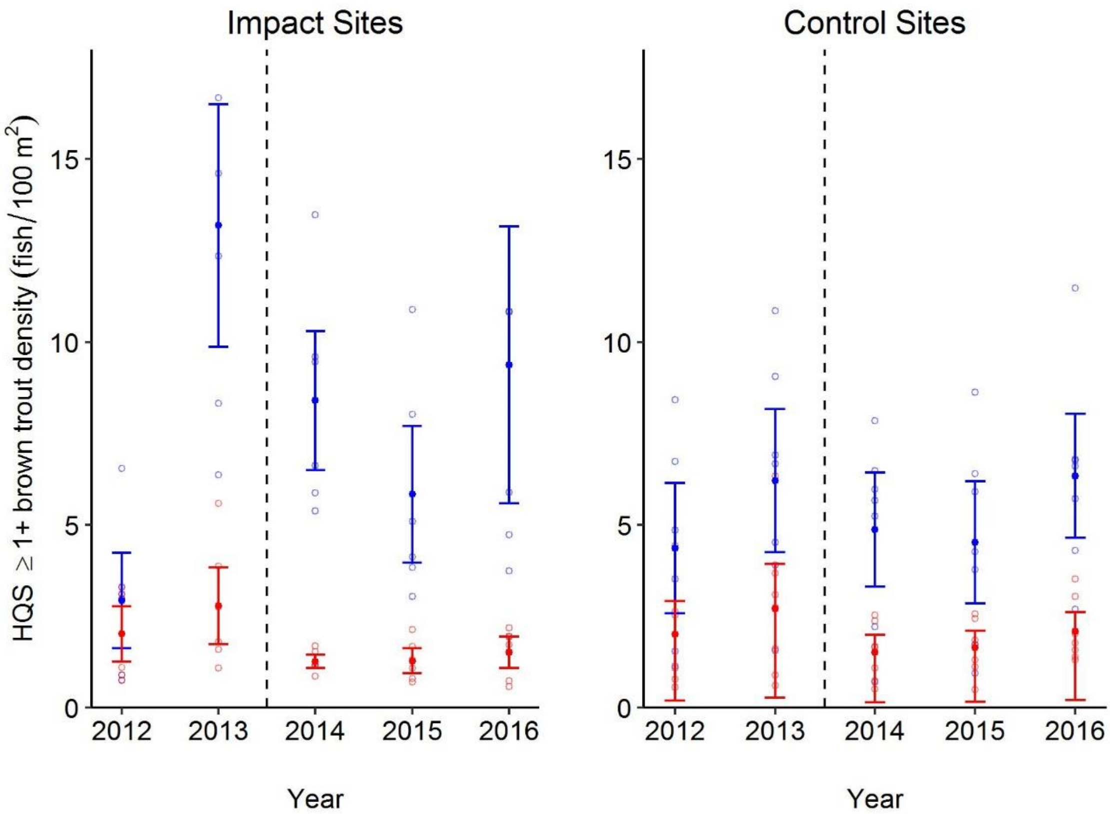

Figure 5.13 Relationship between average HQS densities scores ≥1+ <20 cm (blue) and ≥1+ >20 cm (red) for impact and control sites, error bars represent 95% confidence intervals, dashed line represents introduction of the compensation flow.

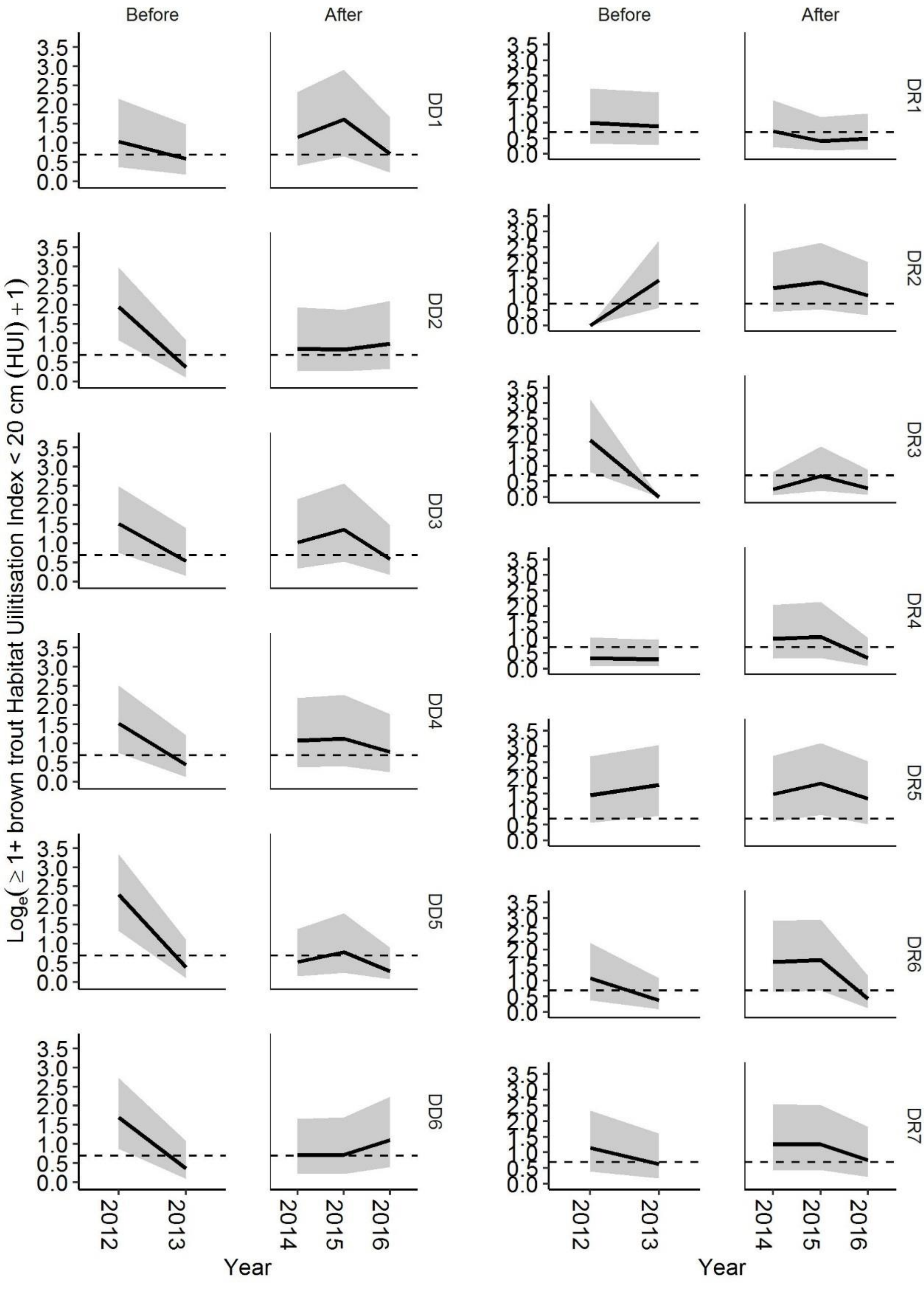

Figure 5.14 Log$_e$ + 1 transformed HUI for ≥1+ (< 20 cm) brown trout in the impact (left) and control sites (right) during the before (2012 – 2014) and after (2015 – 2016) period. Grey bands represent 95% confidence limits. The dashed line represents HUI value where observed and expected densities are equal.

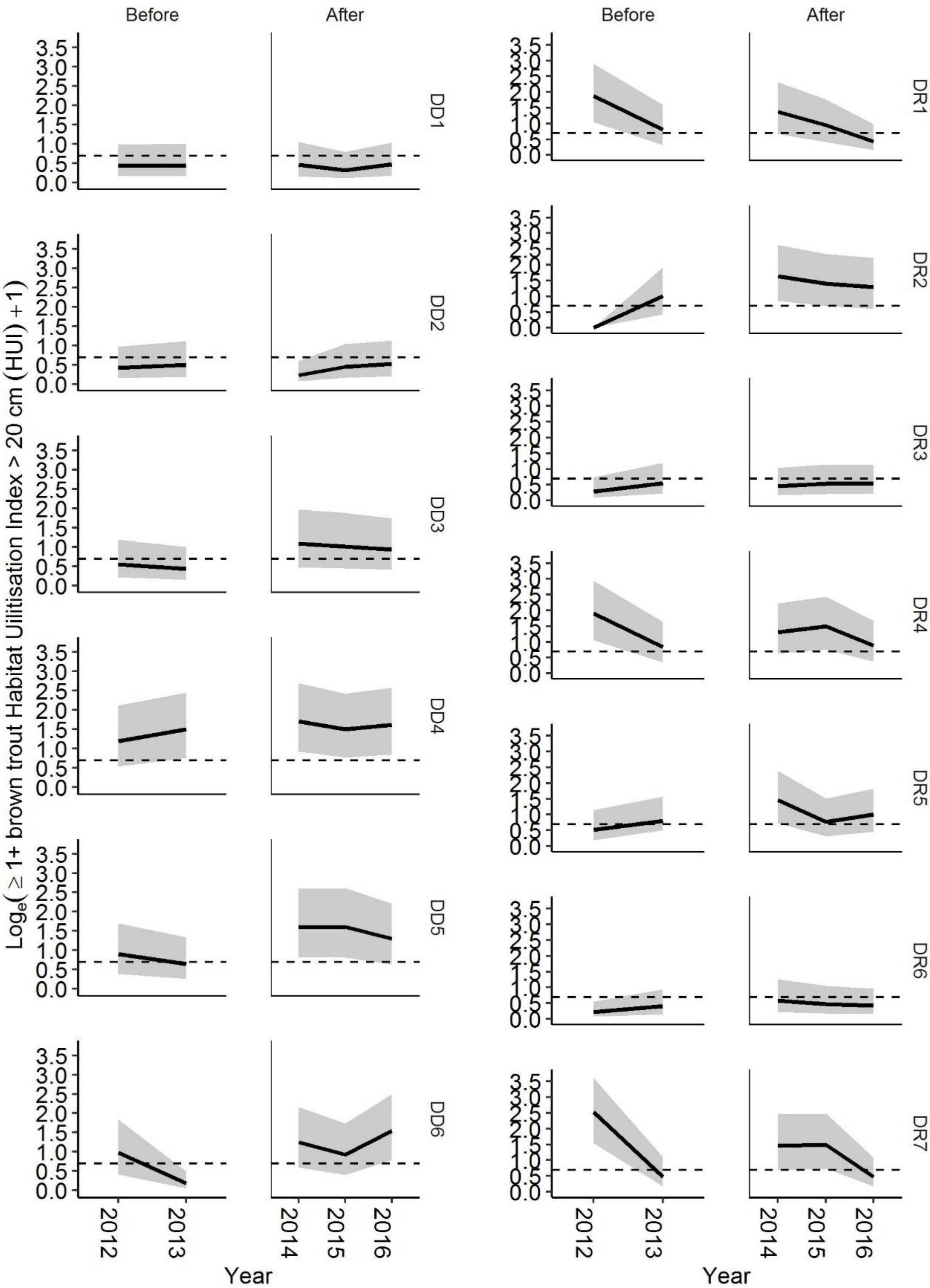

Figure 5.15 Log$_e$ + 1 transformed HUI for ≥1+(> 20 cm) brown trout in the impact (left) and control sites (right) during the before (2012 – 2014) and after (2015 – 2016) period. Grey bands represent 95% confidence limits. The dashed line represents HUI value where observed and expected densities are equal.

Impact assessment (BACI)

A resource calculation performed on both the before and after periods did revealed that the introduction of after years data reduced the level of variance in the dataset for the HQS densities for both ≥1+ (<20 cm) and ≥1+ (> 20cm) classes of brown trout. This, therefore, confirms that a biological meaningful change to ≥1+ brown trout HQS densities could be detected within the timeframe of this study (Table 5.20)

Table 5.20 Actual Vx for *before* and after period and Target Vx to detect a ≥50% change to ≥1+ brown trout Habitat Quality (HQS). **Bold and underlined** signifies that significant change to ≥1+ densities could be detected within the study parameters.

	Actual VX	Target Vx for ≥50% Increase	Target Vx for ≥50% Decrease
< 20 cm Before	0.040	**<u>0.852</u>**	**<u>0.090</u>**
> 20 cm Before	0.020	**<u>0.092</u>**	**<u>0.034</u>**

The habitat quality for ≥1+ (<20 cm) brown trout were similar in both the impact and control sites during the before period. Following the introduction of the compensation release in 2014, the expected densities of ≥1+ (<20 cm) brown trout declined at the impact sites, but this decline was also detected in the control sites (Figure 5.16). The interaction between area and period in the BACI GLMM was not significant (Table 5.21) and therefore revealed that the decline in ≥1+ (<20 cm) brown trout HQS densities in Dale Dike could not be attributed to the introduction of the compensation release. There was little change between the ≥1+ (>20 cm) brown trout HQS densities at the impact sites between the before and after period, but during the same period there was an overall marginal increase to ≥1+ (>20 cm) brown trout HQS densities. The interaction between area and period in the BACI GLMM was not significant (Table 5.21), therefore no change to ≥1+ (>20 cm) brown trout HQS densities in Dale Dike could be attributed to the introduction of the compensation release in 2014 (Figure 5.17).

Table 5.21 BACI GLMM to detect a change in ≥1+ brown trout HQS density following the introduction of the compensation release from Dale Dike reservoir in 2014

	Sum Sq	Mean Sq	DoF	DenDF	F	P
≥1+ (<20 cm)						
Area	0.722	0.722	1	11.742	3.110	0.104
Period	0.019	0.019	1	2.992	0.096	0.788
Area:Period	0.077	0.077	1	46.492	0.332	0.567
≥1+ (>20 cm)						
Area	0.078	0.078	1	11.111	0.209	0.656
Period	0.616	0.616	1	3.003	5.883	0.094
Area:Period	0.103	0.103	1	46.396	0.983	0.327

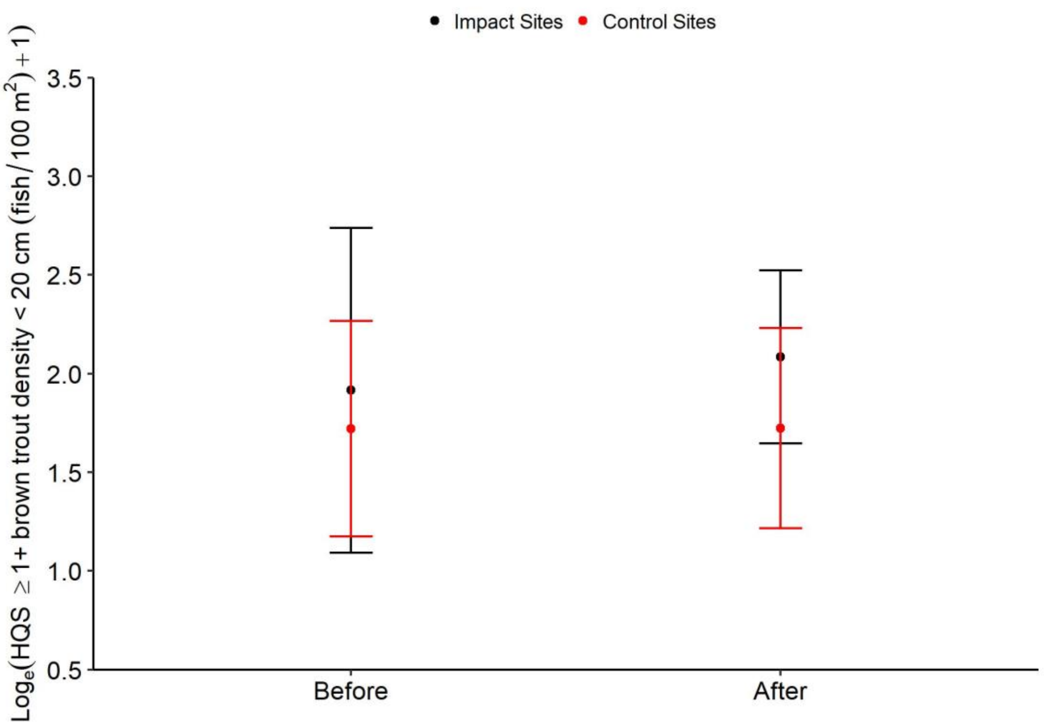

Figure 5.16 Mean (±95%) ≥1+ (<20 cm) HQS densities during *before* and *after* periods for impact (black) and control (red) sites.

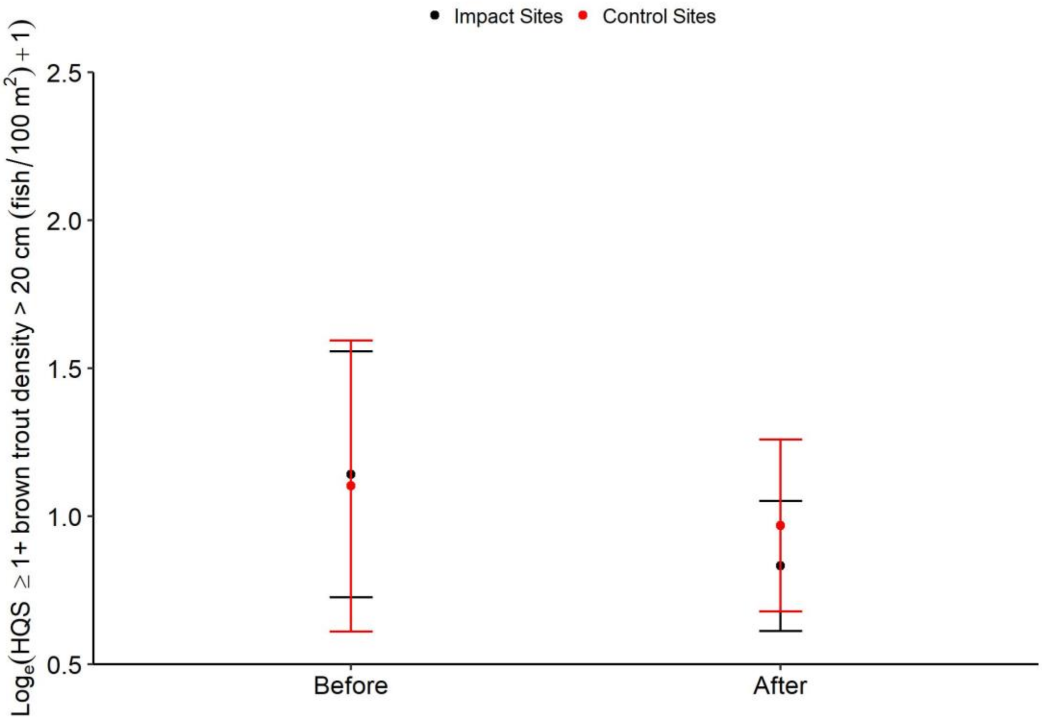

Figure 5.17 Mean (±95%) ≥1+ (>20 cm) HQS densities during *before* and *after* periods for impact (black) and control (red) sites.

5.4 Discussion

During the study period the observed and expected densities (derived from HABSCORE) varied spatially and temporally across the impact and control sites. Densities of 0+ brown trout were particularly low in 2012 across both impact and control sites (a trend that was identified in the brown trout populations in other study rivers (see section 4.4.1)). As elaborated previously it was considered that the exceptionally high rainfall across the Yorkshire during April to July in 2012 (Parry *et al.* 2013) played a strong role in the poor recruitment of brown trout resulting from discharge related mortality and washout (Daufresne *et al.* 2005, Jonsson and Jonsson 2011, Lobón-Cerviá 2014). Investigations into the habitat quality at the impact sites suggested that the habitat quality for 0+ brown trout was at the highest of the study period in 2012, therefore the actual degree of habitat utilization by 0+ brown trout was significantly low at all but one of the impact sites and lower in all control sites, and significantly so in two instances. Whilst it is entirely likely that the poor weather conditions during April to July 2012 were responsible for the low 0+ densities, the time at which the habitat survey took place (September) occurred outside of this period of high rainfall, therefore the habitat quality in September may not be reflect the extreme conditions that occurred earlier in the year. There was a significant improvement in 0+ brown trout densities in 2013 compared to 2012 across the impact sites, and similar improvement was seen at the control sites, but this increase was not significant. The habitat quality for 0+ brown trout at impact and control sites did not reflect this increase, indeed there were marginal decreases in the HQS densities of 0+ brown trout across both impact and control sites in 2013. The mechanisms behind this increase in 0+ brown trout densities at both impact and control sites cannot be fully explained by the ecological data for this study, as there were no significantly higher populations of older larger trout in 2012 that would suggest a greater spawning stock. it is plausible that the extreme high flows resulting from the high rainfall in 2012 may have resulted in better spawning habitat within the impact and control sites, but this was not reflected in the HQS score for both size categories of ≥1+ brown trout. No hydrological data exist for this period, so it was not possible to correlate any flows during the emergence and summer period to assess if optimal conditions for 0+ brown trout occurred in 2013 (See chapter 3.2.6) (Lobón-Cerviá and Rincon 2004). Despite this large increase in 0+ brown trout densities in 2013, HQS data suggested observed densities of 0+ brown trout were still lower than predicted in all but one (DD3) impact sites in 2013. The same was seen in control sites with observed densities at four sites (DR4-DR7) higher than predicted. This suggests that in most impact and some control sites, factors outside the scope of detection within the HABSCORE methodology may be limiting 0+ brown trout demographics, such as temperature regimes (Coleman and Fausch 2007) and water chemistry (Enge *et al.* 2017).

≥1+ brown trout were typically found in higher densities than 0+ brown trout across impact and control sites throughout the study. Of this group, the ≥1+ (< 20 cm) brown trout were the dominant size/age class with ≥1+ (> 20 cm) typically found in much lower densities than ≥1+ (< 20 cm) brown trout. In impact sites ≥1+ (< 20 cm) brown trout were found in similar densities in 2012 and 2013. Habitat quality for ≥1+ (< 20 cm) brown trout varied significantly between the two years prior to the introduction of the compensation release, with expected densities of ≥1+ (< 20 cm) brown trout significantly lower in 2012 than 2013. The mechanisms behind this significant difference are not fully understood, as sites were sampled at a similar time each year, and in both instances, took place before any change from Dale Dike reservoir was made. Potentially the extreme weather during the summer period of 2012 could have scoured the impact sites, as high flow events have been known to scour and remove substrate elements as well as riparian vegetation (Stromberg *et al.* 2007), which are both sources of cover for brown trout. The poor habitat quality found across the impact sites in 2012 is consistent with what would be expected in a relatively short impounded river, with no compensation release (i.e. shallow, slow flowing, with accumulation of fine substrate particles), which contribute to poorer ≥1+ brown trout habitat (Cowx *et al.* 2004, Jonsson and Jonsson 2011, UKTAG

2013). However, the significantly higher habitat quality for ≥1+ (< 20 cm) brown trout in 2013 suggests that the low HQS densities in 2012 were not primarily driven by the regulated nature of the river; these fluctuations in the HQS and density of ≥1+ (< 20 cm) brown trout that were reflected in the HUI values. ≥1+ (< 20 cm) densities in the control sites were much more varied in 2012 and 2013, but this large variability was not reflected in the HQS densities, with HUI values suggesting the observed densities ranged from significantly higher to significantly lower than predicted from the habitat.

0+ brown trout length at capture was used as a metric to gauge the growth of 0+ brown trout throughout the study. Surveys at both impact and control sites were undertaken annually within a 3-week time window in September, therefore the effect of time i.e. trout at some sites being larger because they were caught later in the year should be negligible; the lengths at capture of 0+ brown trout were significantly lower at impact sites thane control sites, but there was considerable site to site variability amongst impact and control sites. The range of length at capture of 0+ brown trout was much smaller in the impact sites than the control sites, suggesting that feeding opportunities and or food availability, as well as habitat, may be poorer across sites in Dale Dike than the control sites. There was no significant difference in 0+ brown trout length at capture in 2012 compared to other years, suggesting that the very poor 0+ brown trout densities at impact sites in 2012 did not afford reduced density-dependent growth regulation as seen in other populations (Newman 1993, Utz and Hartman 2009), i.e. and increased *per capita* share in available feeding opportunities and availability leading to increased growth rates. There were significant differences in 0+ brown trout length at capture from control sites, suggesting that mechanisms such as prey abundance and habitat quality may be profoundly different amongst the five control rivers and their respective sites.

During the after period the BACI GLMM revealed that the habitat quality for 0+ brown trout displayed a significant change at the impact sites that could be attributed to the introduction of the compensation release. However, the level of temporal and spatial variance in both the impact and control sites was high enough that it could not be ruled out that any significant change that was identified was, as a result of, natural stochasticity. Following the formalization of the compensation release in 2014, 0+ brown trout densities displayed fairly uniform temporal variance, with average 0+ brown trout densities in Dale Dike varying little amongst years. Within the BACI GLMM the average 0+ brown trout densities of the before and after periods for the impact sites suggests that there was a marginal (and non-significant) decline following the formalisation of the compensation flow in 2014. This result, however, should be treated with caution, the two years of fisheries data during the before period were highly varied, with significant differences between the two years (2012 and 2013). It is not fully understood why the temporal variability of 0+ brown trout populations in Dale Dike was low, especially in comparison to the 0+ brown trout populations at the control sites, where there was a significant difference between the 0+ brown trout populations in 2014 and 2016. As demonstrated in previously (Chapter 3; Lobón-Cerviá 2004, Unfer *et al.* 2011, Richard *et al.* 2015), variability in hydraulic regimes are key mechanisms underpinning recruitment dynamics. Investigations into the physical habitat at sites in Dale Dike suggested there was little temporal variability during the after period, as indicated by the 0+ brown trout HQS densities. However, the average HQS density for sites in Dale Dike was lower than during the before period, suggesting that the formalisation of the compensation release has had deleterious effects on the habitat quality for 0+ brown trout in Dale Dike. It is therefore plausible that the formalization of an annual minimum compensation release in 2014 homogenised the flow regime removing the key drivers of recruitment dynamics i.e. high flows for spawning migrations and lower flows to provide suitable nursery habitat.

In the first season following the implementation of the compensation release in 2014, ≥1+ (<20 cm) brown trout densities were better than during the before period, but these high densities were not maintained during the after period, and ≥1+ (<20 cm) densities declined in the successive years albeit not significantly. As recruitment can strongly influence successive years' adult populations (Lobón-Cerviá 2005), it is plausible that the increased ≥1+ (<20 cm) densities in 2014 were the result of the relatively high densities of 0+ brown trout in 2013 propagating into this age class. Estimated ≥1+ (<20 cm) brown trout as indicated by the habitat quality did not track trends exhibited in the observed brown trout population, with no significant differences between the years due to high levels of between site variability. Both observed and expected densities of ≥1+ brown trout (>20 cm) were low across impact and control sites following the implementation of the compensation flow in 2014. This suggests that the nature of the revised compensation flow had little influence on the larger (>20 cm) brown trout. Expected densities of ≥1+ (>20 cm) brown trout were significantly lower in 2014 and 2015 than 2013 across Dale Dike. The same trend was found across control sites, but the differences were not significant, which suggests that exogenous climate conditions across the Yorkshire region in combination with the heavily regulated nature of both impact and control rivers perhaps result in poor habitat quality for larger (>20 cm) brown trout. That being said, the decline of habitat quality for ≥1+ (>20 cm) brown trout was greater in the impact sites than the control sites suggesting that the implementation of the compensation release in 2014 led to a decline in faster deeper areas of habitat that larger brown trout favour (Cowx *et al.* 2004, Jonsson and Jonsson 2011), but as the BACI GLMM revealed that this decline could not be significantly attributed to the change in flow regime in 2014. Therefore, no robust conclusions concerning the influence of the flow change on habitat quality for larger brown trout can be drawn at this time.

Following the conclusion of this study in 2016, Yorkshire Water identified that the operation of the abstraction of water for Loxley Water treatment Works (LWTW) was causing water to spill into Dale Dike from the stilling basin. No hydrological data were recorded at Dale Dike prior to the introduction of the flow trial in 2014, therefore the timings and magnitude of this overspill from the stilling basin is currently only speculative, however YWS estimate the overspill is in the region of 2-3Ml/d and has probably been occurring daily for at least the duration of this study (pers. comm., YWS, 2016). The outfall from Dale Dike reservoir was set manually by YWS engineers and flow adjustments were made by opening or closing the valves based on visual observation of the stilling basin, in real terms, there may not have been a dramatic change to the flow regime in Dale Dike due to this prior overspill, which may not have stimulated the desired biological response. It is still important to consider that the compensation flow from Dale Dike reservoir has gone from an unmeasured and potentially irregular release to a formal, consistent measured compensation release, which is important to the development and implementation of future mitigation measures for Dale Dike in future river basin planning cycles.

With the exception of observed 0+ brown trout densities, it was not possible to identify any significant changes between the before and after periods for the fisheries metrics tested in this study. This is not surprising given that at the end of the study it was possible that no substantial change occurred to the flow regime in Dale Dike, due to the releases for Loxley treatment works. Spatial differences existed amongst the population, length and habitat metrics tested at impact and control sites during the after period, but, as explained in Chapter 3.4), it was difficult to detect spatial synchrony in Yorkshire region HMWBs over a relatively small spatial extent (4 km) unlike in unregulated French rivers where the study was carried out over a much greater, 75-km, spatial extent (Bret *et al.* 2016).

The introduction of the annual minimum compensation release from dale dike represents a basic form of flow modification as it only introduces one element from the building block methodology (King *et al.* 2008, UKTAG 2013). This compensation release regime from dale dike was chosen with the limitations of the reservoir release system in mind, i.e. adding more elements of the building block methodology, such as freshets and winter elevations would require significant investment in man-hours to operate the manual release valves. Nevertheless, the inclusion of further building blocks into the Dale Dike release regime would likely promote better habitat quality which can lead to a better biological response from brown trout of all age classes. The implementation of physical habitat restoration methods, such as the use of flow deflectors and woody debris throughout the stream could be used to scour the main channel providing deeper faster flowing areas of habitat while providing more suitable marginal nursery habitat.

6 IMPACT OF MODIFYING COMPENSATION FLOWS ON HABITAT QUALITY AND POPULATION DYNAMICS OF BROWN TROUT IN THE RIVER HOLME

6.1 Introduction

Feedback and assessment of mitigation measures is an important process as part of the prioritisation of rivers for improvement works to achieve good ecological potential (UKTAG 2013). Consequently, improvement of waterbodies under WFD recommendations are under constant review. As demonstrated in Chapters 4 and 5, attempts to improve the ecological status of rivers by modifying flows alone does not necessarily guarantee successful outcomes. Mitigation measures used for ecological improvement are implemented using current knowledge from advances in the field of ecology and hydrology but constrained by operational and infrastructural limitations of the impounding reservoir(s), and this may lead to sub-optimal outcomes. For instance, a study into the potential for ecological improvement of brown trout in the River Holme, Yorkshire between 2002 and 2009 concluded that there were no significant changes to brown trout populations following the revision of a compensation flow in 2004 (Hull International Fisheries Institute 2011). The compensation flow introduced in 2004 was a two-stage seasonally varied flow regime with minimum release levels altering between a summer and winter period. It was concluded that the magnitude of the flow change from the impounding reservoirs was not adequate to detect changes in the fish populations downstream (Taylor 2017). However, given time, it may be possible that advances in understanding of population ecology of brown trout, as well as advances in technology, may provide new insights or methodologies to improve the ecology of impounded rivers (Chapter 2.7). Seasonally varied compensation flows can, for example, provide an adequate balance between ecological demands and water resource constraints (Yin *et al.* 2016), demonstrated that a varied compensation flow regime is likely to provide more beneficial habitat for riverine ecology, namely the study species – brown trout, but a new release regime not only needs to provide temporal variance in the flow but also variability in magnitude to ensure that crucial flow elements (Cowx *et al.* 2004, UKTAG 2013) are met.

The aim of this chapter is to examine the habitat (HABSCORE HQS) and ecological (brown trout) responses to the revision of an existing compensation flow to mimic better natural hydrological events using the building block approach (King *et al.* 2008, UKTAG 2013) in the River Holme.

The specific objectives of this chapter are to:

- Investigate 0+ and ≥1+ brown trout populations and habitat quality at sites along River Holme (a stretch of river downstream of Digley and Brownhill Reservoir) before and after the modification of the existing compensation flow.

- During the same timeframe, monitor 0+ and ≥1+ brown trout populations and habitat quality at sites where the flow regime is not influenced by Digley and Brownhill reservoirs (control sites) to estimate the level of temporal variability of brown trout populations in the region.
- Use a Before After Control Impact (BACI) analysis to establish the influence (if any) that the introduction of the compensation flow has had on the brown trout populations and habitat quality in River Holme.

Outputs will determine the effectiveness of revising the compensation release from Brownhill and Digley reservoirs on the habitat and brown trout population in the River Holme.

6.2 Methodology

6.2.1 Study reaches

Yorkshire Water as part of the sustainable compensation release programme identified the compensation flow from Brownhill and Digley reservoirs as suitable candidates for revision of flows to provide better habitat for fish communities downstream in the River Holme. Baseline monitoring began in 2004 as part of the Sustainable Compensation Releases (SCR) programme with Yorkshire Water and the Environment Agency (Hull International Fisheries Institute, 2011). The new flow regime introduced in 2004 with five year's post flow trial monitoring investigating the biological response from brown trout populations at six monitoring sites selected throughout the reach. Following the conclusion of this study in 2009 (Hull International Fisheries Institute 2011), it was decided that the compensation releases from both Brownhill and Digley reservoirs resulted in negligible changes to brown trout populations in the River Holme and could, therefore, be further amended to provide greater ecological benefits to the resident brown trout population.

River Holme (Digley and Brownhill Reservoir)

The River Holme is a 20-km stretch of river that is formed from the merging of two tributaries from Digley and Brownhill reservoirs (Figure 6.1) and is itself a tributary of the River Colne. The study section is a 5-km stretch of the upstream extent of the Holme with the first site (HO1) located 500 m downstream of the confluence of the Digley and Brownhill streams. Prior to 2004 there was an annual fixed compensation release from both Digley and Brownhill reservoirs that totalled 13.38 Ml/d in the River Holme (Table

6.1). During the sustainable compensation releases programme, the compensation level was lowered from January to September to provide better environmental conditions for brown trout in the River Holme. However, in 2012 a revision of the flow was planned from both Digley and Brownhill reservoirs to have a seasonally variable release pattern with seven different flow releases per year (Table 6.1).

For the environmental monitoring, six 50-m sites (HO1 – HO6) were selected along a 5-km section of the River Holme (Figure 6.1) using the same sites that were originally used as part of the SCR project (see Chapter 3). As these sites were already subject to repeated annual monitoring studies, they were identified as ideal candidates for this investigation as they represent the full heterogeneity of the river habitat as well as holding fish stocks suitable for this study.). Despite the urbanized setting of the river, the surrounding land use is a mix of urban and rough pasture and grazing, with areas deciduous woodland throughout the study area, with an average stream gradient of 3%
The flow regime at the three most upstream sites (HO1 – HO3) is influenced entirely by natural gather and the compensation releases from Digley and Brownhill reservoirs (Figure 6.1). However, there is a confluence upstream of site HO4 where the River Ribble (reservoir regulated river) merges with the River Holme, therefore, the flow at the three most downstream sites represents natural gather, the compensation releases as well as the flow from the River Ribble.

Table 6.1. Monthly values in Ml/d for the compensation release from Digley and Brownhill Reservoir, for the three time periods of; pre-2004 (No flow trial), 2004-2013 (SCR Flow trial) and 2014-present (Revised flow trial)

Time	Reservoir	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Pre - 2004	Digley	6.82											
	Brownhill	6.56											
	Total	13.38											
2004 - 2014	Digley	5.4									6.82		
	Brownhill	5.1									6.86		
	Total	9.6									13.38		
2014 - present	Digley	10.6	5.8				4.5	3.2	5.8			10.6	
	Brownhill	6.9			5.4		4.8		5.4			6.9	
	Total	17.5	12.7		11.2		9.3	8.0	11.2			17.5	

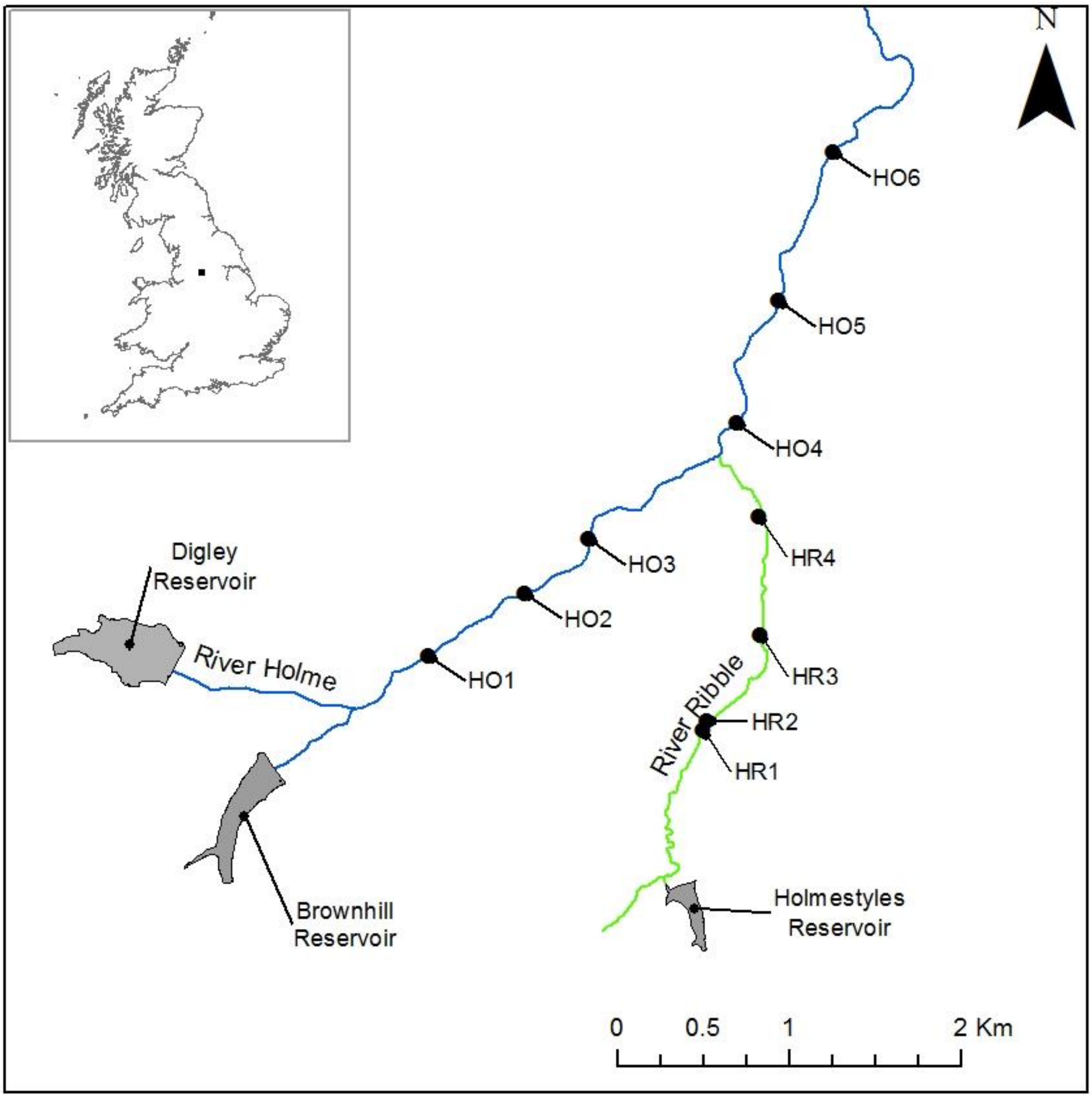

Figure 6.1 Location of all impact sites (HO1-HO6) in relation to Digley and Brownhill reservoir, as well as all control sites (HR1-HR4) in relation to the River Holme

The River Ribble (Control sites)

Four sites were selected from the River Ribble as representative control sites for the River Holme flow trial (Figure 6.1). The streams above both impounding reservoirs were not physically or topographically representative of the habitat in the River Holme, so were therefore discounted for consideration as control sites. The River Ribble, as a tributary of the River Holme is geographically proximal to the impact site and maintains similar habitat features with land-use a mix of urban/rough pasture and grazing along with areas of deciduous woodland along the river, the average stream gradient of the Ribble is slightly steeper than the River Holme at 4.8%, but the geographic proximity, land use and fish composition identified the River Ribble to be a suitable control for the River Holme. Due to land access and the length of the control river, it was only possible to monitor four, 50-m sites along the River Ribble. The River Ribble is a regulated river with the flow regime influenced by a compensation release from Holmestyles reservoir, but within the duration of this study, no changes to the release from Holmestyles reservoir was made. Therefore, temporal variation in the fisheries populations in the River Ribble is assumed to be from natural variability and not induced by anthropogenic changes to land use and reservoir operation.

Environmental investigation

Brown trout populations in the impact river (River Holme) and the control river (River Ribble) were investigated annually during both the before (2012-2014) and after (2014-2016) phases of this study using the fisheries methodology detailed in Chapter 3.2.3. The methodologies for the classification of population estimates; juvenile growth rates; habitat surveying HABSCORE analysis and impact assessment methodology are detailed in Sections 4.2.4 – 4.2.8 and were identical in this study so will not be repeated.

6.3 Results

6.3.1 Impact of the revision of a compensation release on 0+ brown trout densities

Baseline Monitoring (Before)

During the baseline period, 0+ brown trout were caught at all impact sites at densities ranging from 1.99 to 14.87 fish/100 m^2 (Table 6.2); 0+ brown trout populations were predominantly classed as fair/poor. There were no differences spatially ($F_{5,6}$ = 1.515, P>0.05) or temporally ($F_{1,10}$ = 0.219, P>0.05) (Figure 6.2).

Densities of 0+ brown trout were significantly higher at the control sites during this period (P<0.05, Figure 6.2). 0+ brown trout were caught at all control sites in 2012 and 2013 with densities ranging from 13.40 to 74.8 fish/100 m^2 (Table 6.2). These densities were categorised under EA FCS as ranging from average to excellent. 0+ brown trout densities at site HR1 in 2012 and 2013 were exceptionally high compared to other sites and national averages (50.0 to 74.8 fish/100 m^2 (Table 6.2)). Despite this large range in 0+ brown trout densities at the control sites during 2012 and 2013 there were no significant differences between either site ($F_{3,4}$ = 3.554, P>0.05) or years ($F_{1,6}$ = 2.112, P>0.05), suggesting that recruitment in the River Ribble is highly variable (Figure 6.2).

Before the change in compensation release regime, a resource calculation was undertaken to determine if the intensity of sampling during the before period was adequate to detect a biologically relevant change to 0+ brown trout densities within 3 years of the compensation release change. Actual variance was lower than the target variance for both ≥50% decrease and ≥50% increase to 0+ brown trout densities (Table

6.3), therefore if temporal and spatial variance persists in both the impact and control sites a biologically relevant change to 0+ brown trout densities in the River Holme resulting from the compensation flow change would be identified within three years monitoring.

During Amended Compensation release (After)

Following the introduction of the new compensation release regime from Digley and Brownhill reservoirs, there were large changes to the densities of 0+ brown trout both spatially and temporally during the after period. In 2014, this age class was absent at site HO4, but found in much higher densities at the site immediately downstream (HO5). During the 3 years of the after period, brown trout densities ranged from 0.37 to 20.77 fish/100 m^2 (Table 6.2).The pooled density estimates revealed a general decline from 2014 to 2016, with 0+ brown trout found in significantly lower densities in 2016 than in 2015 ($P>0.05$), as well as the 2 years preceding the flow trial (2012-2013) (Figure 6.2). 0+ brown trout densities declined at control sites in the after period compared to the before period with densities ranging from 2.5 to 47.41 fish/100 m^2 (Table 6.2). Pooled 0+ brown trout density estimates suggested that 2015 and 2016 densities were significantly lower than in 2013 ($P<0.05$) they not significantly lower than in 2012 (Figure 6.2). Individual site density estimates for the control sites revealed a significant change occurred at sites HR1 and HR2, which previously held good to excellent populations of 0+ brown trout in 2012 and 2013, but only held fair/poor to average 0+ brown trout populations in 2014-2015. A nested ANOVA of sites nested within period found significant differences at control sites between the before and after periods ($F_{4,12}= 7.243$, $P<0.05$), with0+ brown trout densities at the control sites were significantly higher than the impact sites during the after period ($P<0.05$).

Table 6.2. Density estimates ± 95% C.L. of 0+ brown trout at impact and control sites during both the before (2012 – 2013) and after (2014 – 2016) periods. Colours denote EA-FCS abundance classification

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

River Name	Site Identifier	2012	2013	2014	2015	2016
		Before		After		
impact Sites						
River Holme	HO1	3.69 ± 0.2	1.99 ± 0.4	5.3 ± 0.1	0.4 ± 0.2	0.37 ± 0.2
River Holme	HO2	3.19 ± 0.3	7.48 ± 0.6	2.06 ± 0.3	4.57 ± 0.9	1 ± 0.2
River Holme	HO3	5.89 ± 0.7	3.42 ± 1.2	0 ± 0	8.38 ± 0.6	1.32 ± 0.3
River Holme	HO4	5.82 ± 0.4	12.12 ± 2.7	2.44 ± 0.6	5.77 ± 0.5	3.62 ± 1.2
River Holme	HO5	7.5 ± 0.8	14.87 ± 2.4	20.77 ± 2.9	6.43 ± 0.4	3.32 ± 1.1
River Holme	HO6	12.86 ± 1.7	6.06 ± 0.5	10.14 ± 2.8	11.3 ± 1.1	1.95 ± 0.5
Reference Sites						
River Ribble	HR1	50 ± 1.5	74.8 ± 3.1	7.64 ± 0.2	13.6 ± 0.9	22.48 ± 0.9
River Ribble	HR2	27.99 ± 0.7	41.5 ± 6.6	10.68 ± 2.6	9.92 ± 1.2	2.5 ± 0
River Ribble	HR3	13.72 ± 0.8	34.76 ± 2.1	47.41 ± 15.5	36.84 ± 1.5	25.51 ± 7.7
River Ribble	HR4	13.4 ± 0.8	30.82 ± 1.8	26.05 ± 4.9	32.51 ± 3.2	5.06 ± 0.2

Table 6.3. Actual Vx for *before* period and Target Vx to detect a ≥50% change in 0+ brown trout densities. **Bold and underlined** signifies that significant change to 0+ brown trout densities could be detected within the study parameters.

Actual Vx	Target Vx for ≥50% Increase	Target Vx for ≥50% Decrease
0.043	**0.819**	**0.091**

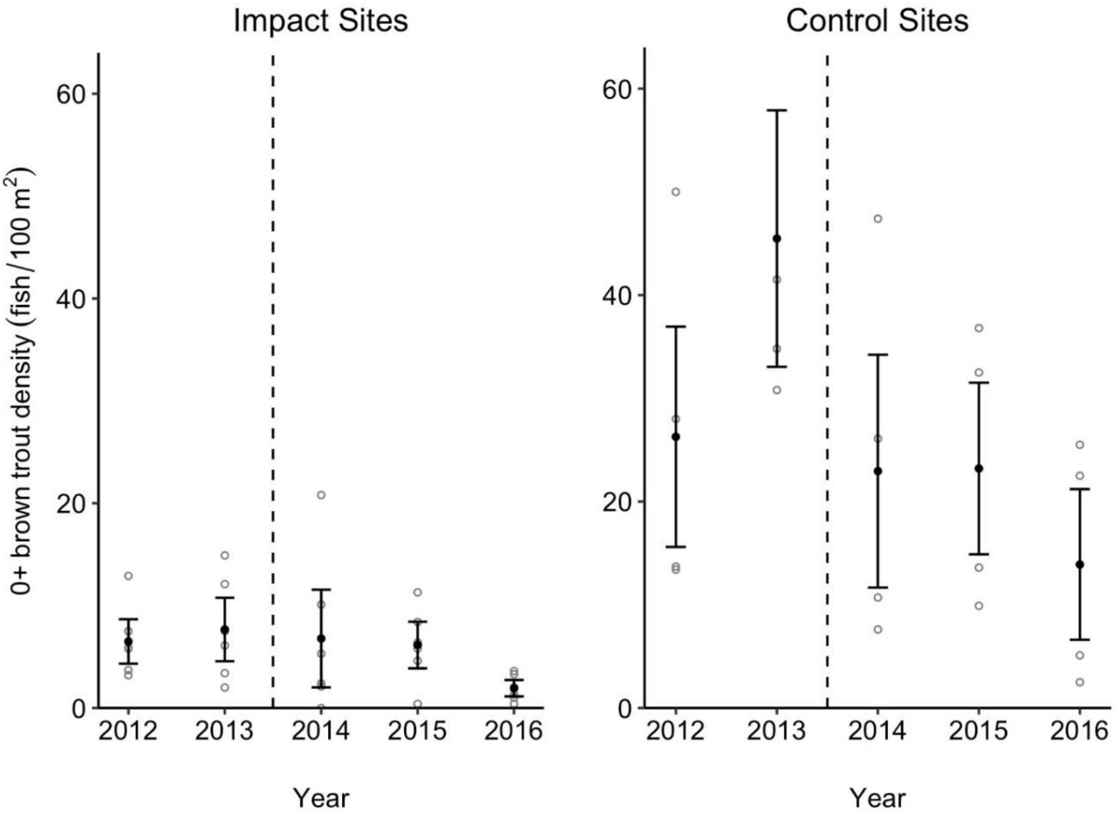

Figure 6.2 Average 0+ brown trout densities ± 95 % C.L. for all impact and control sites, throughout the study period. Dashed line represents the division between *before* and *after* periods. Grey circles represent individual density estimates

Impact assessment (BACI)

Following the conclusion of the after period it was apparent that the temporal and spatial variance of 0+ brown trout seen during the before period did not persist, especially at the control sites. A resource calculation on the full study revealed that the level of actual variance was greater than the target variance to detect a decrease in 0+ brown trout densities. The actual variance was still sufficiently low to ensure that an increase in 0+ brown trout densities could be detected (Table 6.4).

Table 6.4. Actual Vx for full study and Target Vx to detect a ≥50% change to 0+ brown trout densities. **Bold and underlined** signifies that significant change to 0+ densities could be detected within the study parameters.

Actual Vx	Target Vx for ≥50% Increase	Target Vx for ≥50% Decrease
0.61	**<u>0.819</u>**	0.091

The BACI GLMM identified that during the study 0+ brown trout densities were significantly lower in the River Holme than in the River Ribble (Table 6.5). Following the introduction of the amended compensation release from Digley and Brownhill reservoirs in 2014. Unfortunately, due to the high temporal and spatial variance of 0+ brown trout densities at the impact and control sites, it was not possible to conclude to any degree of statistical certainty as to the impact that the amended flow trial had on 0+ brown trout densities.

Table 6.5 BACI GLMM to detect a change in 0+ brown trout density following the revision of the compensation flow from Digley and Brownhill Reservoirs in 2014

	Sum Sq	Mean Sq	DoF	DenDF	F	P
Area	8.861	8.861	1	8.251	24.187	**<u>0.001</u>**
Period	1.276	1.276	1	3.077	3.483	0.157
Area:Period	0.087	0.087	1	35.00	0.242	0.626

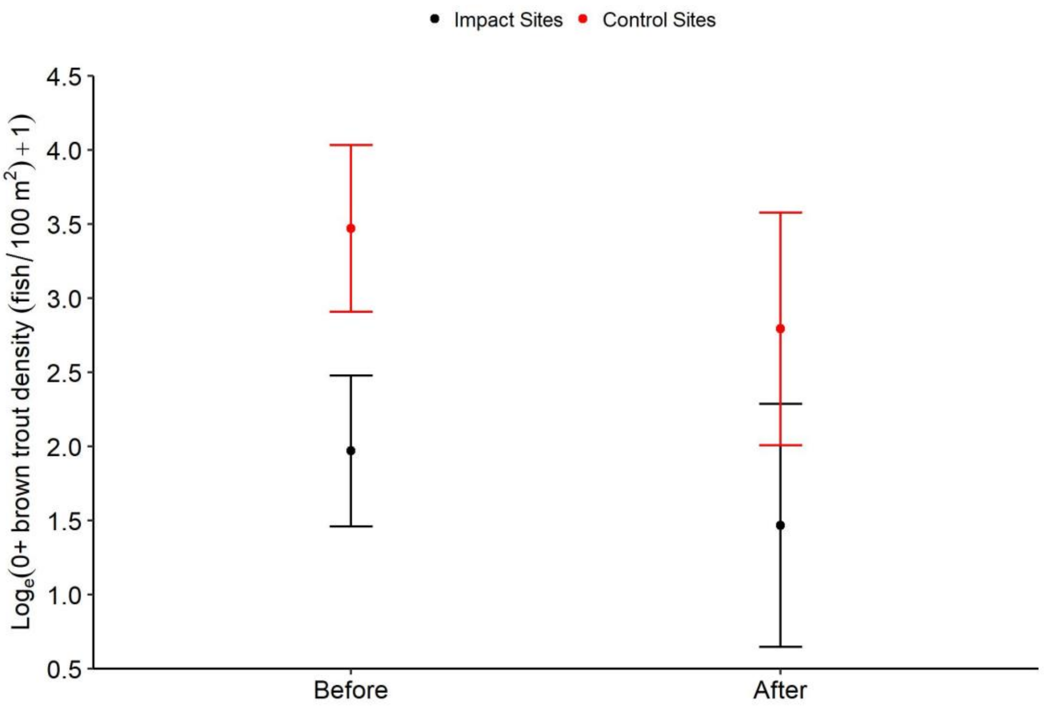

Figure 6.3. Mean (±95%) 0+ brown trout densities before and after for impact (black) and control (red) sites.

6.3.2 Impact of the revision of a compensation flow on ≥1+ brown trout densities

Baseline monitoring (Before)

≥1+ brown trout were caught at all impact sites during the before period and typically dominated catches, with densities ranging from 3.31 to 23.31 fish/100 m^2, and population classifications ranging from fair/poor to good, with most sites classed as average (Appendix, Table A.5). ≥1+ (<20 cm) brown trout were the more abundant size class for ≥1+ brown trout with densities ranging from 2.94 to 13.13 fish/100 m^2 (Table 6.6). There was a marginal, but not significant ($P>0.05$), increase in ≥1+ (<20 cm) brown trout densities from 2012 to 2013 (Figure 6.4). There was, however, a significant difference between the ≥1+ (<20 cm) brown trout densities amongst the impact sites during the before period ($F_{5,6} = 22.940$, $P<0.05$) with densities at sites HO5 and HO6 nearly double the densities seen at the upstream sites, i.e. HO1 (Table 6.6). ≥1+ (>20 cm) brown trout were found in lower densities than the ≥1+ (<20 cm) in 2013 (Figure 6.4), with densities of ≥1+ (>20 cm) brown trout ranging from 0.37 to 10.18 fish/100 m^2 (Table 6.7). Densities of ≥1+ (>20 cm) brown trout declined from 2012 to 2013, but the decline was not significant. However, unlike the ≥1+ (<20 cm) brown trout there were no significant differences between densities of ≥1+ (>20 cm) brown trout between the impact sites ($F_{5,6}=1.339$, $P>0.05$).
≥1+ brown trout densities at control sites were in a similar range as the impact sites, ranging from 3.91 to 25.54 fish/100m^2. Population classifications of ≥1+ brown trout were typically average to good at control sites, but at site HR1 in 2012 ≥1+ brown trout were classed as fair/poor (Appendix, Table A.5). During the before period ≥1+ (<20 cm) brown trout were found in significantly higher densities than ≥1+ (>20 cm) brown trout at control sites, with ≥1+ (<20 cm) brown trout densities ranging from 3.91 to 25.54 fish/100 m^2 (Table 6.6). Densities of ≥1+ (<20 cm) brown trout increased in 2013 compared to 2012, but this increase was not significant (Figure 6.4). Despite the variability of this ≥1+ (<20 cm) brown trout age class at control sites, there was no significant difference between sites ($F_{3,4}=0.788$, $P<0.05$) ≥1+ (>20 cm) brown trout were absent from site HR1 in 2012 and sites HR2 and HR3 in 2013, while at the remaining sites in 2012 and 2013, densities of ≥1+ (>20 cm) brown trout densities ranged from 0.73 to 4.68 fish/100 m^2 (Table 6.7). As found in the impact sites, there was no significant differences in ≥1+ (>20 cm) brown trout densities at the control sites between years ($F_{1,6}=0.078$, $P>0.05$) or sites ($F_{3,4}=6.061$, $P>0.05$) (Figure 6.4).
Before the revision of the compensation flow in 2014, a resource calculation was constructed to determine if the intensity of sampling at impact and control sites was adequate to detect a biological relevant change to both size classes (<20cm, >20 cm) of ≥1+ brown trout. The resource calculation revealed that the actual variance was

adequate to detect a ≥50% increase or decrease in the ≥1+ (<20 cm) brown trout densities, but given the low densities of ≥1+ (>20 cm) brown trout if the current level of temporal variance persists only a 50% increase in this size class could be detected (Table 6.8).

During amended compensation release (After)

Following the revision to the compensation release from Digley and Brownhill reservoirs, ≥1+ brown trout densities were more varied, ranging from 2.34 to 30.79 fish/100 m^2 with population classifications ranging from poor to excellent (Appendix, Table A.5). ≥1+ (<20 cm) brown trout were found in significantly higher densities ranging from 1.56 to 30.27 fish/100 m^2 at all impact sites during the flow trial compared to ≥1+ (>20 cm) brown trout (P<0.05) (Figure 6.4; fish/Table 6.6). Unlike the before period there were no significant differences in ≥1+ (<20 cm) brown trout densities between sites ($F_{5,12}$=2.475, P>0.05) or between years ($F_{1,16}$=2.524, P>0.05) but densities in 2014 were significantly higher than in 2012, and densities in 2015 and 2016 exhibited a marginal decline each year (Figure 6.4). ≥1+ (>20 cm) brown trout densities ranged from 0.78 to 11.46 fish/100 m^2 (Table 6.7) and displayed marginal increases year by year from 2014 to 2015. However, this was not significant (P>0.05) (Figure 6.4). There were, however, significant differences between densities of ≥1+ (>20 cm) within the six impact sites during the after period ($F_{5,12}$=8.487, P<0.05).
≥1+ brown trout were found in a similar density range at the control sites to those at the impact sites ranging from 6.00 to 30.79 fish/100 m^2 (Appendix, Table A.5). ≥1+ brown trout population classifications during this period ranged from fair/poor to good (appendix, Table A.5). Of the ≥1+ brown trout caught at the control sites during this period ≥1+ (<20 cm) brown trout were found in significantly higher densities than ≥1+ (>20 cm) brown trout (P<0.05) (Figure 6.4). Densities of ≥1+ (<20 cm) brown trout ranged from 5.25 to 25.79 fish/100 m^2. As observed in the before period, ≥1+ (<20 cm), brown trout densities at control sites were significantly different ($F_{3,8}$=11.260 P<0.05) amongst the sites, with the two downstream sites (HR3 and HR4) typically holding higher densities than the two upstream control sites (HR1 and HR2). As found at the impact sites, densities of ≥1+ (<20 cm) brown trout were highest during 2014 with populations showing a marginal decline in 2015 and 2016. However, this decline was not significant (P>0.05) (Figure 6.4). ≥1+ (>20 cm) brown trout were absent from sites HR2 for the duration of the after period, and from site HR3 in 2016, but densities of ≥1+ (<20 cm) brown trout at the remaining sites and years ranged from 0.64 to 5.21 fish/100 m^2 (Table 6.7). There was little variation in ≥1+ (>20 cm) brown trout densities between the years ($F_{1,10}$=0.592, P>0.05), but significant differences were found in ≥1+ (>20 cm) densities between the sites ($F_{3,8}$=16.69, P<0.05).

Table 6.6. Density estimates ± 95% C.L. of ≥1+ brown trout (< 20 cm) at impact and control sites during both the before (2012 – 2013) and after (2014 – 2016) periods

River Name	Site Identifier	2012	2013	2014	2015	2016
		Before			After	
Impact Sites						
River Holme	HO1	5.03 ± 0.4	6.82 ± 0.3	17.42 ± 0.2	14.62 ± 0.5	11.16 ± 3.7
River Holme	HO2	7.71 ± 0.3	7.22 ± 0.4	13.47 ± 0.2	10.29 ± 1.1	6 ± 1.1
River Holme	HO3	2.94 ± 0.3	3.55 ± 0.2	1.56 ± 0.0	5.1 ± 0.1	11.42 ± 1.6
River Holme	HO4	9.38 ± 2.1	12.43 ± 1.5	26.83 ± 1.1	9.01 ± 0.4	10.2 ± 3.5
River Holme	HO5	13.13 ± 1.6	12.52 ± 2.1	30.27 ± 1.2	23.48 ± 0.9	13.3 ± 0.9
River Holme	HO6	11.54 ± 1.1	12.98 ± 2.7	14.79 ± 0.9	14.62 ± 3.1	13.99 ± 2.2
Reference Sites						
River Ribble	HR1	3.91 ± 0	10.17 ± 0.9	13.38 ± 2.6	6.4 ± 0.2	5.25 ± 1.1
River Ribble	HR2	6.83 ± 0.9	25.54 ± 1	15.26 ± 0.3	7.44 ± 0.2	6.67 ± 0.2
River Ribble	HR3	14.4 ± 0.9	11.83 ± 0.2	22.41 ± 1.3	20.3 ± 1	18.22 ± 0.8
River Ribble	HR4	19.45 ± 1.3	14.83 ± 0.8	25.58 ± 0.6	19.21 ± 0.4	25.79 ± 3.6

Table 6.7. Density estimates ± 95% C.L. of ≥1+ brown trout (>20 cm) at impact and control sites during both the before (2012 – 2013) and after (2014 – 2016) periods.

River Name	Site Identifier	2012	2013	2014	2015	2016
		Before			After	
Impact Sites						
River Holme	HO1	4.36 ± 1.3	4.26 ± 0.4	9.47 ± 0.4	11.46 ± 0.1	5.95 ± 0.6
River Holme	HO2	1.59 ± 0.4	1.34 ± 0	1.37 ± 0.1	1.37 ± 0	1.5 ± 0
River Holme	HO3	0.37 ± 0	0.92 ± 0	0.78 ± 0.2	2.55 ± 0.1	2.2 ± 0.1
River Holme	HO4	3.23 ± 0.6	2.49 ± 0.2	3.83 ± 0.1	2.52 ± 0.3	3.29 ± 0.7
River Holme	HO5	10.18 ± 0.3	2.09 ± 0	2.97 ± 0.2	7.4 ± 0.4	7.25 ± 0.4
River Holme	HO6	3.96 ± 0.5	2.02 ± 0.3	3.01 ± 0.3	2.66 ± 0.2	5.21 ± 0.4
Reference Sites						
River Ribble	HR1	0 ± 0	0.73 ± 0.7	0.64 ± 0	0.75 ± 0	0.7 ± 0
River Ribble	HR2	2.05 ± 0.3	0 ± 0	0 ± 0	0 ± 0	0 ± 0
River Ribble	HR3	1.37 ± 0	0 ± 0	1.72 ± 0	0.8 ± 0	0 ± 0
River Ribble	HR4	3.46 ± 0	4.68 ± 0	5.21 ± 0	3.4 ± 0.1	3 ± 0

Table 6.8. Actual Vx for *before* period and Target Vx to detect a ≥50% change to ≥1+ brown trout densities. **Bold and underlined** signifies that significant change to ≥1+ densities could be detected within the study parameters.

	Actual VX	Target Vx for ≥50% Increase	Target Vx for ≥50% Decrease
< 20 cm Before	0.047	**<u>1.021</u>**	**<u>0.113</u>**
> 20 cm Before	0.084	**<u>0.333</u>**	0.036

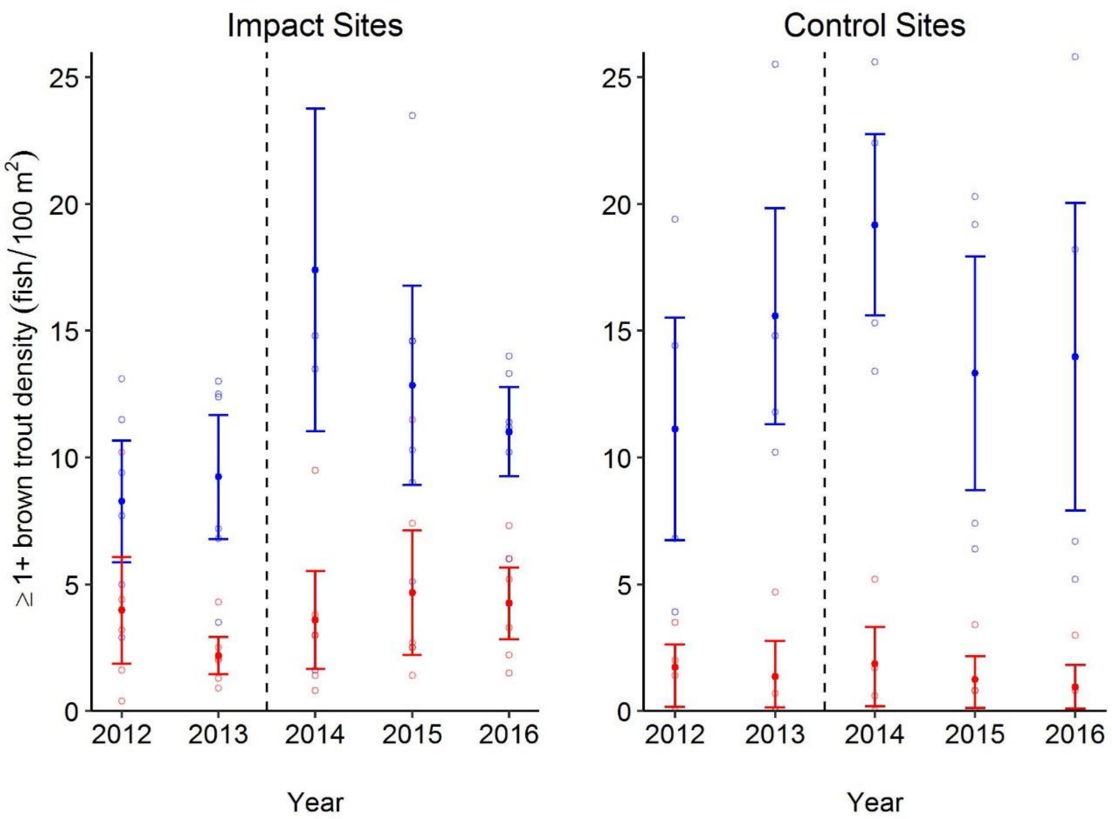

Figure 6.4. Average ≥1+ brown trout densities for all impact sites and control sites. ≥1+ (< 20 cm) data presented in blue, and ≥1+ (>20 cm) in red. Dashed line represents the division between before and after periods. Hollow circles represent individual data points.

Impact assessment (BACI)

Following the conclusion of the study, levels of actual variance amongst the ≥1+ (<20 cm) and ≥1+ (> 20 cm) brown trout had changed with the addition of the after periods fisheries data. In both cases, actual variance declined, but not sufficiently enough to ensure that a biologically relevant decline to ≥1+ (>20 cm) brown trout could be detected within the study. However, all other changes to both ≥1+ brown trout size classes could be detected within the timeframe of this study (Table 6.9).

Table 6.9. BACI GLMM to detect a change in ≥1+ brown trout density following the revision of the compensation release from Digley and Brownhill Reservoirs in 2014. **Bold and underlined** signifies a significant ($P<0.05$) interaction in the GLMM

	Actual VX	Target Vx for ≥50% Increase	Target Vx for ≥50% Decrease
< 20 cm	0.027	**1.021**	**0.113**
> 20 cm	0.052	**0.333**	0.036

There were marginal increases in the density of ≥1+ (<20 cm) brown trout at impact sites following the revision of the compensation release from Digley and Brownhill reservoirs in 2014. This increase was also seen in the control sites; the BACI GLMM revealed that there were no significant differences to the ≥1+ (<20 cm) brown trout densities either between the impact and control sites or between the before and after period (Table 6.10). The BACI GLMM, therefore, confirmed that the increase to ≥1+ (<20 cm) brown trout densities in the River Dibb following the compensation release revision was as consequence of natural variability (Table 6.10). There was again no significant difference in ≥1+ (>20 cm) brown trout densities between impact and control sites or between the before and after periods (Table 6.10), despite densities of ≥1+ (>20 cm) brown trout increasing at the control sites, but marginally declining at the impact sites during the after period (Figure 6.6). The BACI GLMM revealed that the change to 1≥+ (>20 cm) brown trout densities in relation to the control sites were not significantly attributed to the revision of the compensation flow in the River Holme (Table 6.10). However, the resource calculation identified that it would not be possible to isolate any changes to the ≥1+ (>20 cm) brown trout densities from natural variability during the timeframe of the study (Table 6.9).

Table 6.10 BACI GLMM to detect a change in 0+ brown trout density following the restoration work on the River Washburn in 2015

	Sum Sq	Mean Sq	DoF	DenDF	F	P
≥1+ (<20 cm)						
Area	0.091	0.019	1	8.089	0.705	0.425
Period	0.348	0.348	1	3.123	2.689	0.196
Area:Period	0.150	0.150	1	35.000	1.150	0.291
≥1+ (>20 cm)						
Area	0.463	0.463	1	8.055	3.816	0.086
Period	0.075	0.075	1	3.202	0.615	0.487
Area: Period	0.256	0.256	1	35.000	2.124	0.154

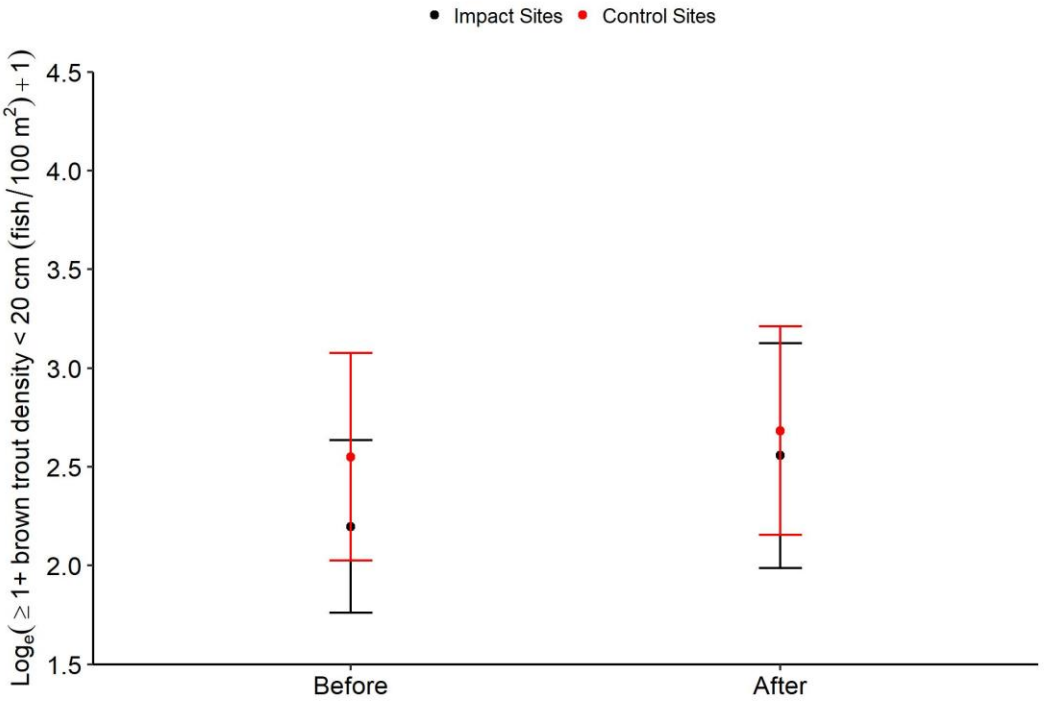

Figure 6.5 Mean (±95%) ≥1+ (< 20 cm) brown trout densities for the before and after periods for impact (black) and control (red) sites.

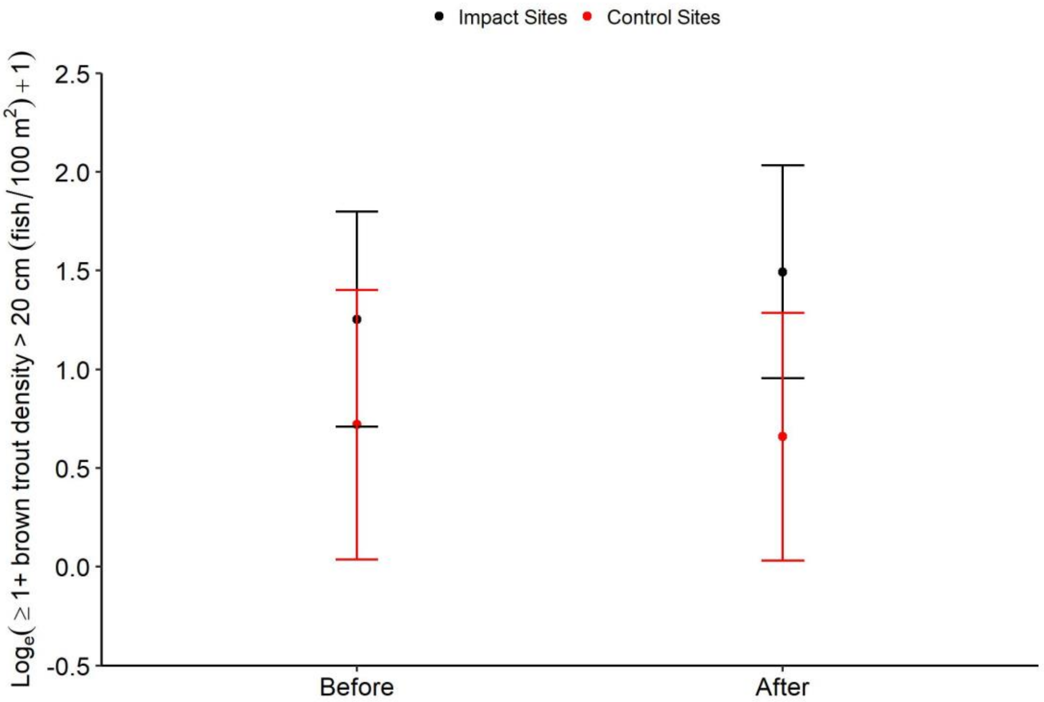

Figure 6.6 Mean (±95%) ≥1+ (> 20 cm) brown trout densities for the before and after periods for impact (black) and control (red) sites.

6.3.3 Impacts of the revision of a compensation flow on the length at capture of 0+ brown trout

Fisheries surveys at both control and impact sites were completed within a 14-day time window in September of each year. Therefore, any variance in length at capture stemming from the time at which the 0+ brown trout were captured (i.e. more likely that fish caught earlier in the season would be smaller due to reduced growth window) should be minimised. Size ranges of 0+ brown trout at capture for both impact and control sites during the before and after period are in Appendix Table A.6.

Baseline Monitoring (Before)

The length at which 0+ brown trout were captured varied spatially and temporally throughout the study at both impact and control sites. For impact sites it ranged from 45 to 95 mm during the before period, and there were no significant differences between the length of capture amongst sites, although the length at capture of 0+ brown trout ($P<0.05$) decreased significantly between 2012 and 2013 (Figure 6.7).

0+ brown trout were caught in a similar size range at control sites as in impact sites, with the length at capture ranging from 39 to 104 mm. There were no significant differences between the lengths at capture between control sites. There were no significant differences between the length at capture for 0+ brown trout between the impact ($F_{5,6}=0.217$, $P>0.05$) and control ($F_{3,4}=0.256$, $P>0.05$) sites during the before period (Figure 6.7). Before the revision of the compensation flow in 2014, a resource calculation was used to determine if the intensity of sampling was sufficient to detect a biologically meaningful change to 0+ brown trout length at capture. The actual variance was well below the target variance, therefore if the temporal and spatial variance present in the impact and control sites persists, a change to 0+ brown trout length at capture would be detectable within the timeframe of this study.

Table 6.11 Actual Vx for the before period and Target Vx to detect a ≥25% change to 0+ brown trout lengths. **Bold and underlined** signifies that significant change to 0+ lengths could be detected within the study parameters

Actual Vx	Target Vx for ≥25% Increase	Target Vx for ≥25% Decrease
3.772	**<u>420.547</u>**	**<u>281.519</u>**

During amended compensation release (After)

Following the revision of the flow trial in 2014, there was still considerable variation in the length at capture of 0+ brown trout. 0+ brown trout were caught in the size range 47 to 91 mm at reference sites, and there were no significant variations between lengths of capture of 0+ brown trout either between years in the after period or between impact sites(Figure 6.7).

At control sites, 0+ brown trout were caught in the size range 46 to 89 mm. There were no significant differences between the length at capture of 0+ brown trout between the three years of the after period or between the control sites. Mean length at capture for 0+ brown trout revealed marginal increases from 2013 (in the before period) to 2014 and again from 2014 to 2015. None of these increases were significant (Figure 6.8). The length at capture of 0+ brown trout was marginally lower in 2016 compared to the other years of the after period (Figure 6.7).

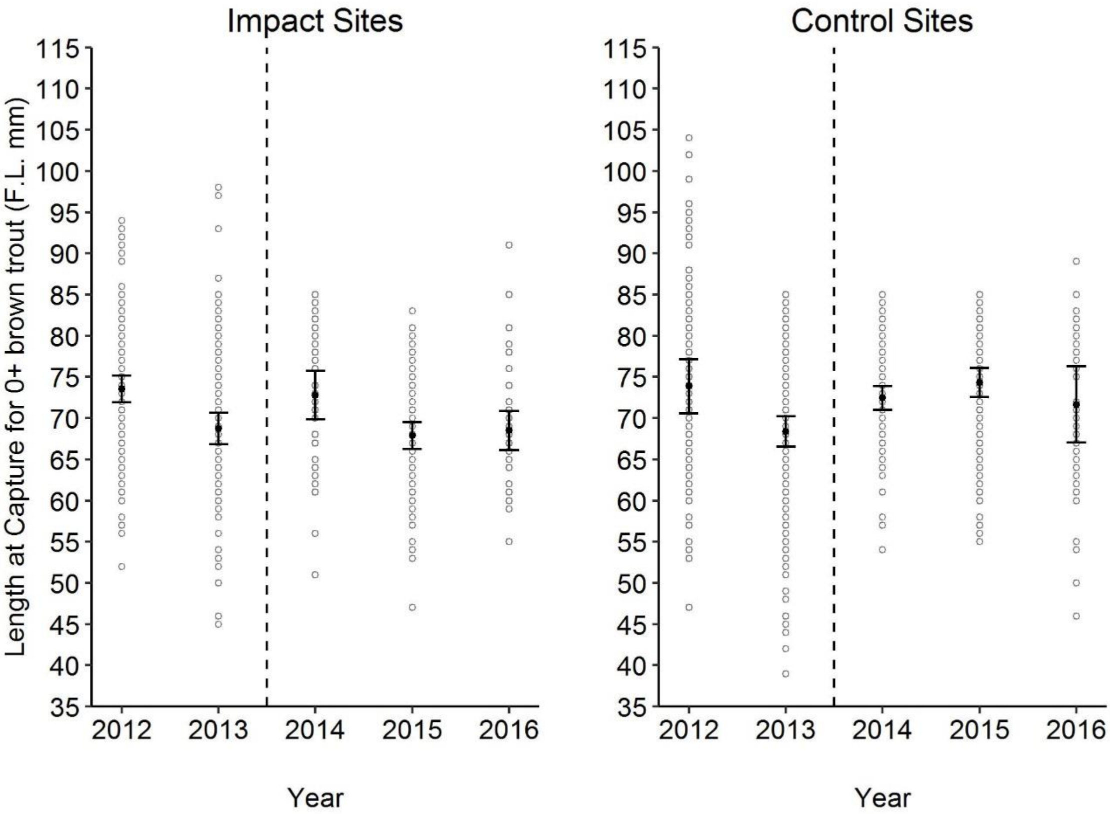

Figure 6.7. Lengths at capture for individual 0+ brown trout (greyed circles). As well as average length (± 95% C.L.) for impact and control sites. Dashed line represents division between before and after period

Impact assessment (BACI)

The addition of after-period data introduced more variability as the actual variance for the full data set increased. However, this level of variance was still considerably lower than the target variance, therefore, a biologically meaningful change to the length at capture of 0+ brown trout could be detected within the timeframe of the study (Table 6.12).

Table 6.12. Actual Vx for the full study and Target Vx to detect a ≥25% change to 0+ brown trout lengths. **Bold and underlined** signifies that significant change to 0+ lengths could be detected within the study parameters

Actual Vx	Target Vx for ≥25% Increase	Target Vx for ≥25% Decrease
5.819	**420.547**	**281.519**

During the before period 0+ brown trout were caught at very similar sizes in both the impact and control sites (Figure 6.8). Following revision of the compensation release from Digley and Brownhill reservoirs in 2014, the average length at which 0+ brown trout were caught declined, whereas the mean length at capture for the control sites increased during this same timeframe. In both cases the change in 0+ brown trout length at capture between the before and after periods was not significant (Figure 6.8). The interaction between area and period in the BACI GLMM (Table 6.16) revealed that the decrease to length at capture of 0+ brown trout in the River Holme could not be attributed to the revision of the compensation release from Digley and Brownhill reservoir.

Table 6.13 BACI GLMM to detect a change in 0+ brown trout length at capture following revision of the compensation release from Digley and Brownhill reservoirs in 2014

	Sum Sq	Mean Sq	DoF	DenDF	F	P
Area	2.55	2.55	1	13.605	15.228	**0.002**
Period	0.04	0.04	1	3.023	0.242	0.657
Area:Period	0.114	0.114	1	54.177	0.683	0.412

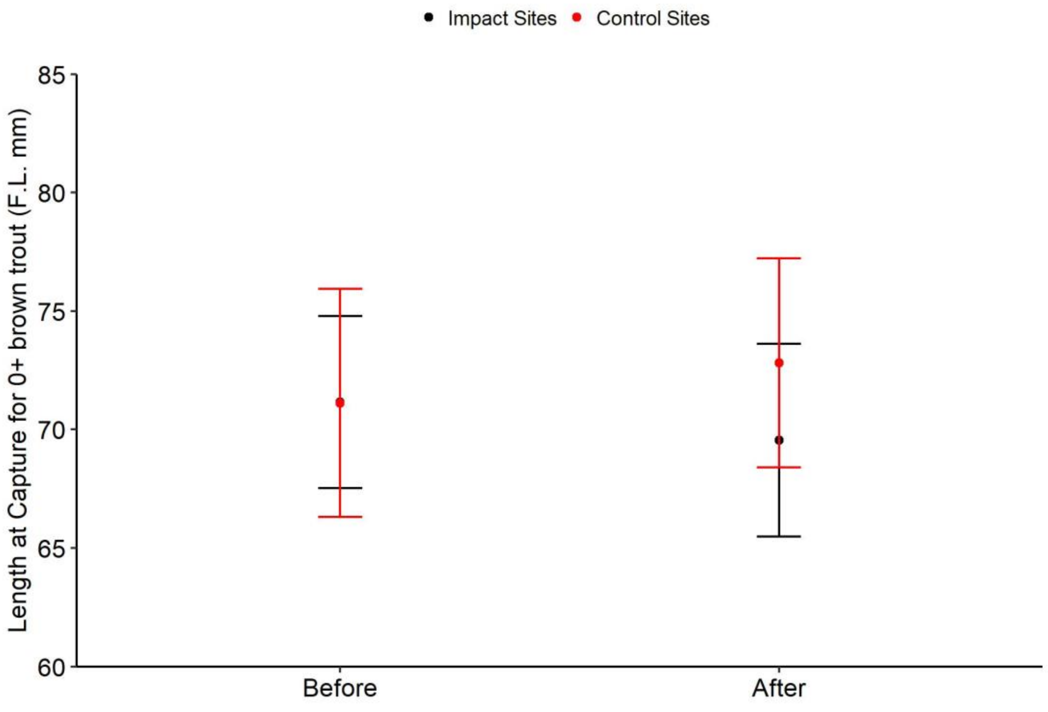

Figure 6.8 Mean (±95%) 0+ brown trout lengths *before* and *after* for impact (black) and

control (red) sites.

6.3.4 Impact of the revision of the compensation release on the habitat quality (HQS density) for 0+ brown trout

Baseline Monitoring (Before)

There was no variation in 0+ brown trout HQS densities between sites and years at impact sites during the before period, with expected densities ranging from 4.01 to 9.07 fish/100 m^2. Populations were predominately classed as fair/poor, except three sites (HO1 in 2012, and HO3 and HO4 in 2013), which were classed as average (Table 6.14). Pooled 0+ brown trout HQS densities were marginally, but not significantly, higher in 2012 than 2013 ($P>0.05$) (Figure 6.9). There was also no significant variation in 0+ brown trout HQS densities between sites. Observed densities of 0+ brown trout varied between this period and fluctuated at impact sites during the before period, with observed densities lower than expected at site HO1 and HO2 in 2012 and at sites HO1, HO3 and HO6 in 2013. At the remaining impact sites in the before years (2012 – 2013) observed densities were greater than expected 0+ brown trout HQS densities (Figure 6.10). However, at impact sites during the before period there were no instances of observed 0+ brown trout densities significantly deviating from the HQS densities (Figure 6.10).
0+ brown trout HQS densities at the control sites were typically higher than at the impact sites, but not significantly so (Figure 6.9) with expected 0+ brown trout densities ranging from 4.27 to 21.77 fish/100 m^2 (Table 6.14). Expected populations at the control sites ranged from fair/poor to good under EA FCS classifications. Most 0+ brown trout were classed as average, with the exception of site HR1 in 2012 and HR3 in 2013, which was classed as fair/poor, and expected densities at site HR2 in 2013, which were classed as good (Table 6.14). Observed densities of 0+ brown trout were very high at some control sites during the before period (6.3.1.1), but these high densities were not reflected in the HQS densities estimated from habitat quality. Habitat utilisation revealed that observed densities were higher than predicted from the HQS densities at all control sites during the before period, with observed 0+ brown trout densities significantly higher at site HR1 in 2012 and 2013, and also HR3 in 2013 (Figure 6.10).
Before the revision of the compensation release from Digley and Brownhill reservoirs, a resource calculation was constructed to determine if the intensity of sampling during the before period was adequate to detect a biologically meaningful change at the impact sites. The level of actual variation was considerably lower than the target variation for a ≥50% increase, however, was marginally larger than the target variance to detect a ≥50% decrease to expected 0+ brown trout densities within the timeframe of the study (Table 6.15).

During amended compensation release (After)

0+ brown trout HQS densities were marginally higher with expected densities of 0+ brown trout ranging from 3.97 to 14.48 fish/100 m^2 (Table 6.14). There were significant differences between the HQS densities amongst the impact sites with HQS densities of 0+ brown trout better at site HO3 and HO4 and consistently better than at other impact sites ($F_{5,12}$= 7.602, P<0.05). Expected populations ranged from fair/poor to average but expected 0+ brown trout densities were better in 2015 (Figure 6.9), with all sites classed as average (Table 6.14). There were no significant differences between 0+ brown trout HQS densities at the impact sites during the after period. Habitat utilisation by 0+ brown trout at the impact sites was again low as seen in the before period, with observed densities of 0+ brown trout lower than the HQS densities at sites HO1 – HO4 for the duration of the after period. At site, HO5 observed densities were marginally higher than the HQS densities in 2014 but lower in 2015 and 2016, and at site HO6 the observed densities were only marginally higher than the HQS densities in 2014 and 2016, but again lower in 2016. Observed densities were significantly lower than the HQS densities at site HO3 and HO4 in 2014, site HO1 in 2015 and site HO1 and HO3 in 2016. There were no instances of observed densities significantly higher than the HQS densities at impact sites during the after period (Figure 6.10)

Habitat quality at control sites during the after period suggested that the expected densities of 0+ brown trout should be higher than the before period, with expected 0+ brown trout densities significantly higher in the three after years compared to 2012 (P<0.05) (Figure 6.9). Expected densities of 0+ brown trout during this period ranged from 9.11 to 23.04 fish/100 m^2 during at the after period. Expected densities were classed as either average or good (Table 6.14). Expected densities of 0+ brown trout varied amongst years, average 0+ brown trout HQS densities in 2014 and 2015 were similar, but in 2016 HQS densities improved significantly from the previous year (P<0.05) (Figure 6.9). Observed densities were higher than predicted from the habitat at site HR3 in all after years, and at site HR1 in 2016, and HR4 in 2014 and 2015 (Figure 6.10). Observed densities only significantly deviated from the HQS densities at site FR4 in 2014, where observed densities were significantly higher than expected (Figure 6.10).

Table 6.14 0+0+ HQS densities related to the EA-FCS categories for impact and control sites during the before (2012 - 2013) and after (2014 – 2016) periods.

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

River Name	Site Identifier	2012	2013	2014	2015	2016
		Before			After	
impact Sites						
River Holme	HO1	8.83	7.61	7.22	11.17	7.18
River Holme	HO2	4.93	7.25	6.57	8.07	3.97
River Holme	HO3	5.52	8.15	8.77	9.23	9.67
River Holme	HO4	5.80	9.07	13.53	14.48	12.02
River Holme	HO5	6.25	6.46	7.53	9.28	5.92
River Holme	HO6	4.01	6.88	7.23	8.16	6.44
Reference Sites						
River Ribble	HR1	4.27	13.85	14.21	17.12	18.57
River Ribble	HR2	10.83	21.77	13.49	12.54	18.4
River Ribble	HR3	9.03	4.83	18.98	15.24	23.04
River Ribble	HR4	9.77	9.45	11.52	9.11	13.13

Table 6.15. Actual Vx for the before period and Target Vx to detect a ≥50% change to 0+ HQS Densities. **<u>Bold and underlined</u>** signifies that significant change to 0+ HQS Densities could be detected within the study parameters

Actual Vx	Target Vx for ≥50% Increase	Target Vx for ≥50% Decrease
0.102	**<u>0.891</u>**	0.099

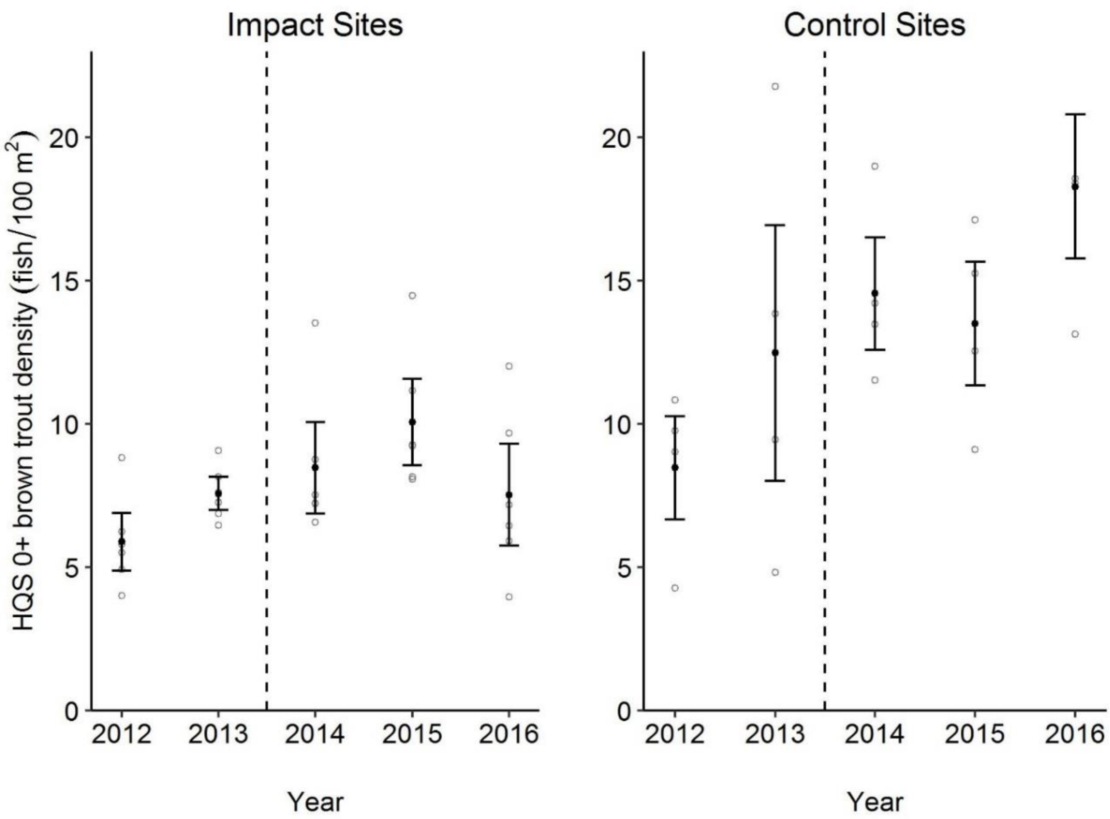

Figure 6.9. Average 0+ brown trout HQS densities ± 95 % C.L. for all impact and control sites, throughout the study period. Dashed line represents the division between *before* and *after* periods. Greyed circles represent individual density estimates

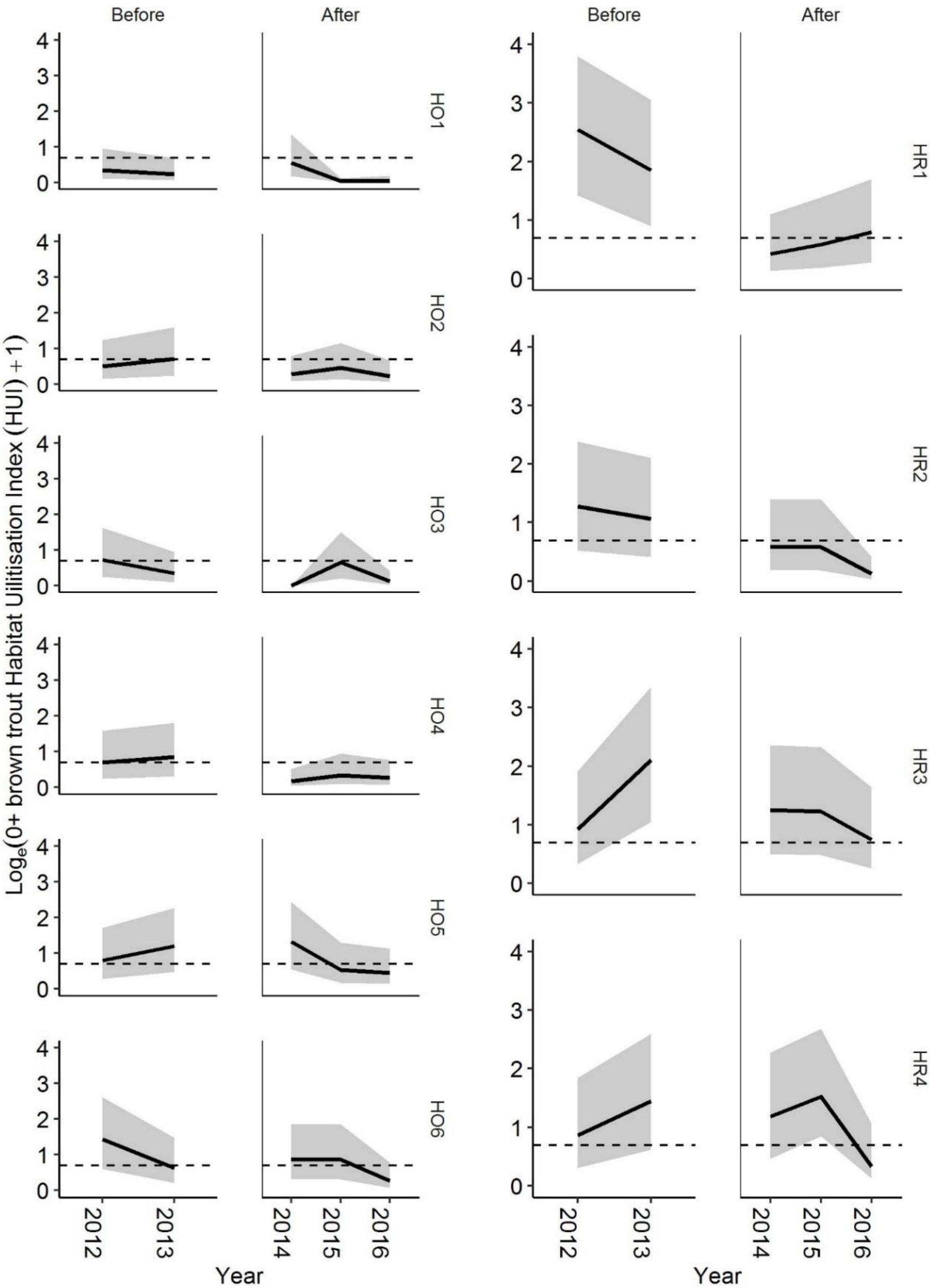

Figure 6.10 Log$_e$ + 1 transformed HUI for 0+ brown trout in the impact (left) and control sites (right) during the before (2012 – 2013) and after (2014 – 2016) period. Grey bands represent 95% confidence limits. Dashed line represents HUI value where observed and expected densities are equal.

Impact Assessment (BACI)

Following the addition of the after-period data to the resource calculation, the level of actual variance in the 0+ brown trout HQS densities declined. The resource calculation concluded that a biologically relevant decrease and increase to 0+ brown trout HQS densities would be detected within the parameters of this study (Table 6.16).

Table 6.16. Actual Vx for the full study and Target Vx to detect a ≥50% change to 0+ HQS Densities. **Bold and underlined** signifies that significant change to 0+ HQS Densities could be detected within the study parameters

Actual Vx	Target Vx for ≥50% Increase	Target Vx for ≥50% Decrease
0.074	**<u>0.891</u>**	**0.099**

Following the revision of the compensation release from Digley and Brownhill reservoirs in 2014, habitat quality for 0+ brown trout marginally improved as seen by the increase in expected 0+ brown trout densities during the study (Figure 6.11). However, habitat quality also increased by a greater magnitude across the control sites during the same period (Figure 6.11), suggesting that the increase seen in the River Holme reflected natural variability. The BACI GLMM revealed that there were significant differences in the 0+ brown trout HQS density at the impact and control sites (Table 6.17) as well as between the two study periods (before and after) (Table 6.17), however, the interaction between area and period in the BACI GLMM was not significant (Table 6.17). Therefore, the improvement of habitat quality for 0+ brown trout in the River Holme could not be attributed to the revision of the compensation flow from Brownhill and Digley reservoirs.

Table 6.17 BACI GLMM to detect a change in 0+ brown trout HQS density following the revision of the compensation release from Digley and Brownhill reservoir in 2014

	Sum Sq	Mean Sq	DoF	DenDF	F	P
Area	1.151	1.151	1	8.359	15.823	**<u>0.003</u>**
Period	0.756	0.756	1	3.16	10.391	**<u>0.045</u>**
Area:Period	0.141	0.141	1	35.00	1.938	0.173

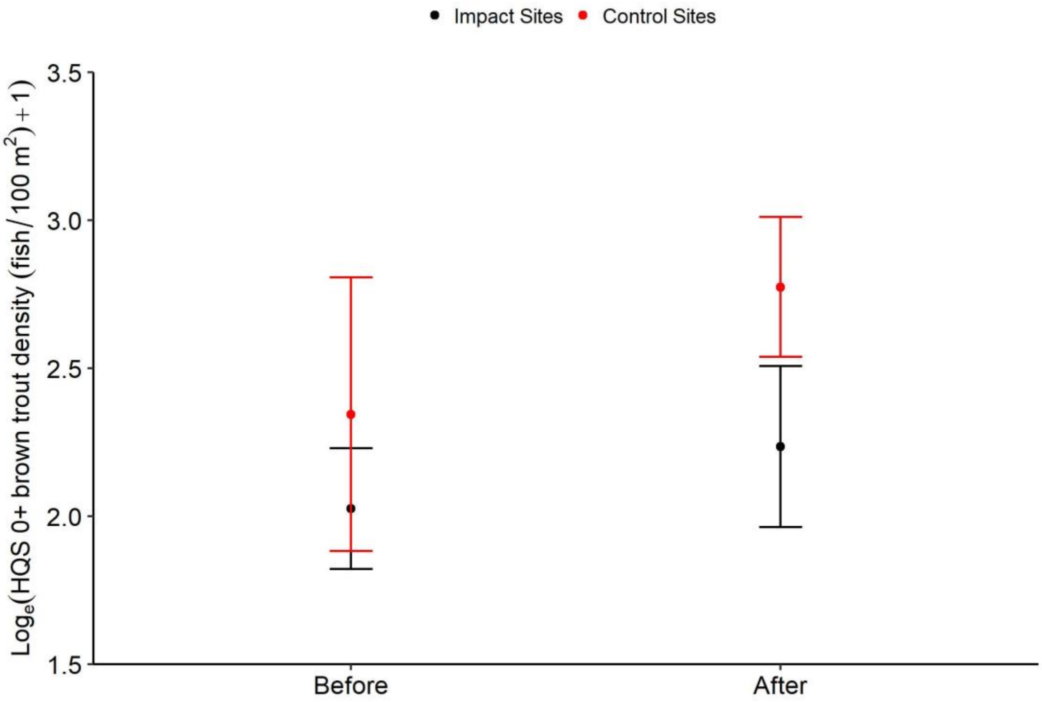

Figure 6.11. Mean (±95%) 0+ brown trout HQS densities during before and after periods for impact (black) and control (red) sites.

6.3.5 Impacts of the revision of the compensation flow on the habitat quality (HABSCORE) for ≥ 1+ brown trout

Baseline monitoring (Before)

During the before period, HQS densities for ≥1+ brown trout varied between sites and years. At the impact sites, the expected densities of ≥1+ (<20 cm) brown trout ranged from 0.36 to 4.00 fish/100 m^2. There were variations between the two before years for the HQS densities of ≥1+ (<20 cm) brown trout with expected densities of this size class significantly higher in 2013 than in 2012 at impact sites (Figure 6.14) ($P<0.05$). Despite these significant differences between years, there were no significant differences in ≥1+ (<20 cm) brown trout HQS densities amongst the sites ($F_{5,6}=0.191$, $P>0.05$) (Table 6.18). Observed densities of ≥1+ (<20 cm) brown trout were greater than expected from the habitat quality at all impact sites for the duration of the before period. Observed densities of ≥1+ (<20 cm) brown trout were significantly higher at sites HO4 and HO5 in 2012 and 2013 and at site HO6 in 2012 (Figure 6.13). The quality of habitat for ≥1+ (>20 cm) brown trout was poorer than the ≥1+ (<20 cm) brown trout at impact sites during the before period, as denoted by the significantly lower ($P<0.05$) ≥1+ (>20 cm) brown trout HQS densities (Figure 6.12). Expected densities of ≥1+ (>20 cm) brown trout ranged from 0.36 to 1.5 fish/100 m^2 across the impact sites during the before period (Table 6.19). Only at impact site HO3 during the before period was observed density of ≥1+ (>20 cm) brown trout lower than predicted. At the remaining sites, observed densities of ≥1+ (>20 cm) brown trout were higher than expected, and at sites HO1, HO4-HO6 in 2012 and HO1 and HO4 in 2013 were observed densities of ≥1+ (>20 cm) brown trout significantly ($P>0.05$) higher (Figure 6.14).

At control sites, the habitat quality for ≥1+ brown trout was much more varied across the sites during the before period (Figure 6.12). Expected densities of ≥1+ (<20 cm) brown trout ranged from 1.64 to 18.02 fish/100 m^2 during the before period (Table 6.18). Despite this large variability there were no significant differences amongst the control sites during this period ($F_{3,4}=1.696$, $P>0.05$). The HQS densities of ≥1+ (<20 cm) brown trout increased from 2012 to 2013, but this increase was not significant (Figure 6.12), as HQS densities of ≥1+ (<20 cm) brown trout remained largely unchanged at sites HR3 and HR4 during the before period (Table 6.18). Despite the relatively poor habitat quality for ≥1+ (<20 cm) brown trout, expected densities of ≥1+ (<20 cm) brown trout were higher than predicted at all sites and were significantly higher at site HR4 for the duration of the before period (Figure 6.13). HQS densities for ≥1+ (>20 cm) brown trout were lower than for ≥1+ (<20 cm) brown trout at control sites during the before period, however not significantly so (Figure 6.12). HQS densities for ≥1+ (>20 cm) brown trout ranged from 0.37 to 5.00 fish/100 m^2 at control sites (Table 6.19). A marginal increase was seen in the HQS densities of ≥1+ (>20 cm) brown trout at impact sites. However, this increase was not significant ($P<0.05$) (Figure 6.12). There was also no significant ($F_{3,8}=0.979$ $P>0.05$) variations amongst sites. Observed densities of ≥1+ (>20 cm) brown trout at control sites were typically lower than expected from the HQS ≥1+ (>20 cm) brown trout densities, with densities at sites HR1 and HR3 in 2012 and 2013 and site HR2 in 2014 all lower than the HQS ≥1+ (>20 cm) brown trout densities (Figure 6.14). Of these, the observed densities of ≥1+ (>20 cm) brown trout were significantly lower ($P<0.05$) at sites HR1 in 2012 and HR2 and HR3 in 2013. Observed densities of ≥1+ (>20 cm) brown trout were higher than predicted from the habitat at control site HR4 for the duration of the before period (Figure 6.14)

During amended compensation release (After)

Following the revision of the compensation release from Digley and Brownhill reservoirs, HQS densities for ≥1+ (<20 cm) brown trout increased across impact sites and ranged from 1.28 to 14.77 fish/100 m^2 (Table 6.18). There were no significant variations amongst the years ($F_{1,16}= 0.913$, $P>0.05$), but HQS densities of ≥1+ (<20 cm) brown trout were significantly higher ($P<0.05$) in 2016 than during the before period (Figure 6.12). There were significant variations amongst the impact sites ($F_{5,12}=3.183$ $P<0.05$), with ≥1+ (<20 cm) brown trout HQS densities consistently lower at sites HO1 and HO6 than other sites (Table 6.18). Observed densities of ≥1+ (<20 cm) brown trout at the impact sites were again typically greater than the HQS densities during the after period, with observed densities at HO1 and HO4 – HO6 greater than the HQS densities for the duration of the study period. Observed densities of ≥1+ (<20 cm) brown trout were also greater than the HQS densities at site HO2 in 2014 and 2015, and HO3 in 2015. Observed densities of ≥1+ (<20 cm) brown trout at sites HO2 in 2016 and HO3 in 2014 and 2016 were lower than the HQS densities. Observed densities of ≥1+ (<20 cm) brown trout deviated significantly from the HQS densities at site HO1, HO5 and HO6 in 2014, and at HO4 in 2015. Observed densities of ≥1+ (<20 cm) brown trout at site HO3 in 2014 were significantly lower ($P<0.05$) than the HQS densities (Figure 6.13). HQS densities for ≥1+ (>20 cm) brown trout during the after period at impact sites were again significantly lower ($P<0.05$) than the HQS densities. Temporally the HQS densities of ≥1+ (>20 cm) brown trout followed a similar trend to the ≥1+ (<20 cm) brown trout HQS densities (Figure 6.12). Expected densities of ≥1+ (>20 cm) brown trout ranged from 0.43 to 3.18 fish/100 m^2 (Table 6.19), with habitat quality (HQS) for ≥1+ (>20 cm) brown trout significantly greater in 2016 than seen in the before period (Figure 6.12). There were no significant variations in the HQS densities amongst impact sites in the after period ($F_{5,12}=2.358$, $P>0.05$). Observed densities of ≥1+ (>20 cm) brown trout were greater than indicated by the habitat quality at sites HO1, HO4 – HO6 for the duration of the after period and at site HO2 in 2014 and 2015, and HO3 in 2015. Observed densities of ≥1+ (>20 cm) brown trout were significantly greater than the HQS densities at site HO1 and HO6 for the duration of the after period, and at sites HO4 in 2015, and HO5 in 2015 and 2016. Observed densities of ≥1+ (>20 cm) brown trout were significantly lower than the HQS densities at site HO3 in 2014 (Figure 6.14).

Habitat quality of ≥1+ (<20 cm) brown trout at the control sites showed no change from the before levels in 2014 and 2015 and expected densities of ≥1+ (< 20 cm) brown trout were significantly higher ($P<0.05$) in 2016 than 2014 and 2015 (Figure 6.12). During the after period HQS densities of ≥1+ (< 20 cm) brown trout ranged from 1.72 to 14.77 fish/100 m^2 (Table 6.18), but there were no significant variations amongst sites ($F_{3,8}=0.321$, $P>0.05$). Observed densities of ≥1+ (< 20 cm) brown trout were greater than the HQS densities at sites HO3 and HO4 for the duration of the after period, and at site HR1 in 2014 and HR2 in 2014 and 2015. Observed densities being significantly greater ($P<0.05$) at sites HR2 and HR3 in 2014 and at site HR4 in 2014 – 2016 (Figure 6.13). Expected populations of ≥1+ (>20 cm) brown trout derived from habitat quality were significantly smaller than the ≥1+ (<20 cm) brown trout densities (Figure 6.12), with expected densities as indicated by HQS for ≥1+ (>20 cm) brown trout ranging from 0.74 to 5.51 fish/100 m^2 (Table 6.19). Unlike the HQS densities of ≥1+ (<20 cm) brown trout at the control sites or the HQS densities of ≥1+ (>20 cm) at the impact sites there was no significant variation ($F_{3,8}$=1.38, P>0.05) amongst the sites or years (Figure 6.12). Analysis of HUI values revealed that the observed densities of ≥1+ (>20 cm) brown trout were significantly lower at sites HR1 –HR3 during the after period with observed densities significantly lower at site HR2 in 2014 -2016 and at site HR3 in 2015 and 2016. Observed densities of ≥1+ (>20 cm) brown trout at site HR4 were higher than the HQS densities for the duration of the after period, with observed densities in 2014 significantly higher than the HQS densities (P<0.05) (Figure 6.14).

Table 6.18. HQS densities for ≥1+ (<20cm) brown trout at all impact and control sites for the before (2012-2013) and after (2014-2016)

River Name	Site Identifier	2012	2013	2014	2015	2016
		Before			After	
impact Sites						
River Holme	HO1	2.46	2.63	2.23	3.15	4.67
River Holme	HO2	2.3	3.6	5.12	5.1	6.78
River Holme	HO3	1.67	2.23	9.24	4.22	11.65
River Holme	HO4	1.46	2.67	4.72	1.28	3.1
River Holme	HO5	2.11	2.46	4.31	4.87	4.88
River Holme	HO6	0.36	4	2.65	2.2	5.55
Reference Sites						
River Ribble	HR1	1.64	7.15	2.92	5.81	12.66
River Ribble	HR2	5.59	18.02	2.73	2.89	14.77
River Ribble	HR3	3.41	3.27	3.91	4.76	13.22
River Ribble	HR4	1.68	2.03	2.21	1.72	7.16

Table 6.19. HQS densities for ≥1+ (>20cm) brown trout at all impact and control sites for the before (2012-2013) and after (2014-2016)

River Name	Site Identifier	2012	2013	2014	2015	2016
		Before			After	
impact Sites						
River Holme	HO1	0.56	0.88	0.6	0.65	1.17
River Holme	HO2	0.73	1.24	1.39	1.19	2.27
River Holme	HO3	0.87	1.5	3.18	1.45	2.57
River Holme	HO4	0.56	0.74	1.78	0.43	2.19
River Holme	HO5	0.37	0.88	1.01	1.33	1.75
River Holme	HO6	0.36	0.73	0.77	0.67	1.51
Reference Sites						
River Ribble	HR1	0.37	1.78	0.74	1.14	1.91
River Ribble	HR2	1.84	5.5	1.19	1.5	2.98
River Ribble	HR3	3.29	1.15	2.13	2.41	5.51
River Ribble	HR4	1.65	1.76	1.09	0.94	3.51

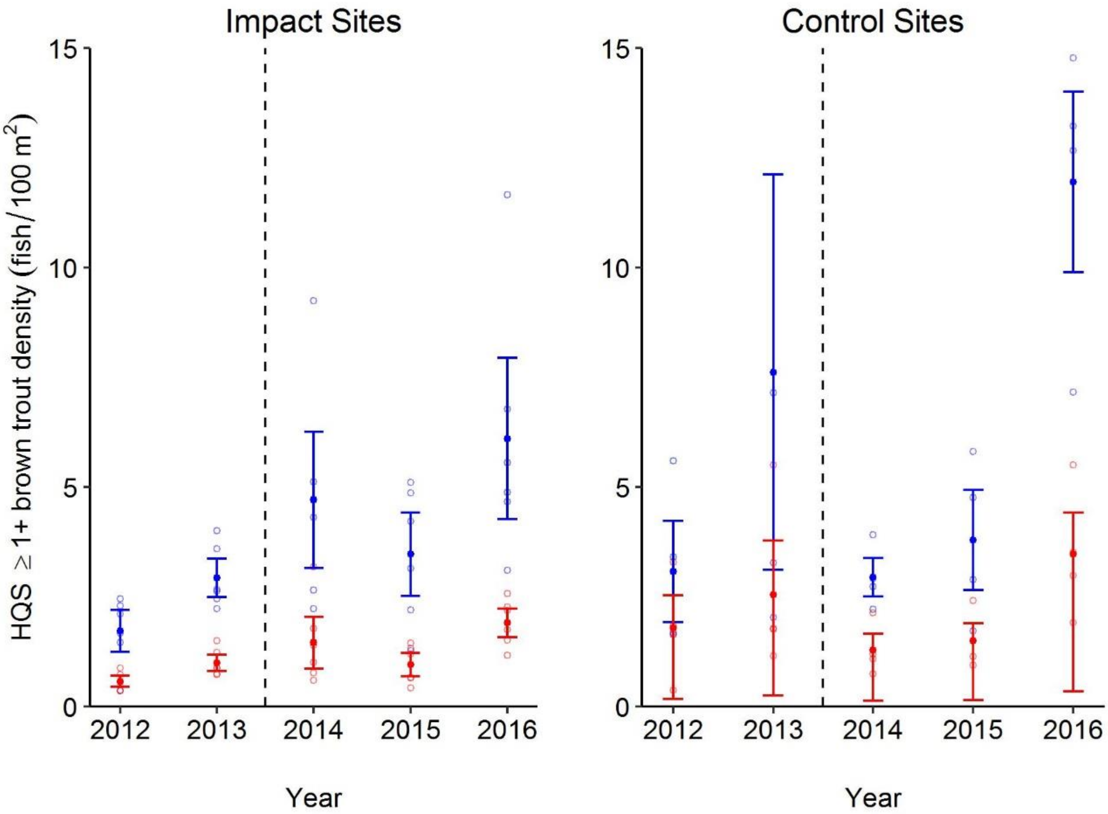

Figure 6.12 Relationship between average HQS densities scores ≥1+ <20 cm (blue) and ≥1+ >20 cm (red) for impact and control sites, error bars represent 95% confidence intervals, dashed line represents introduction of the compensation flow.

Table 6.20 Actual Vx for *before* period and Target Vx to detect a ≥50% change to ≥1+ brown trout Habitat Quality (HQS). **<u>Bold and underlined</u>** signifies that significant change to ≥1+ densities could be detected within the study parameters.

	Actual VX	Target Vx for ≥50% Increase	Target Vx for ≥50% Decrease
< 20 cm Before	0.080	**<u>0.852</u>**	**<u>0.090</u>**
> 20 cm Before	0.302	**<u>0.092</u>**	**<u>0.034</u>**

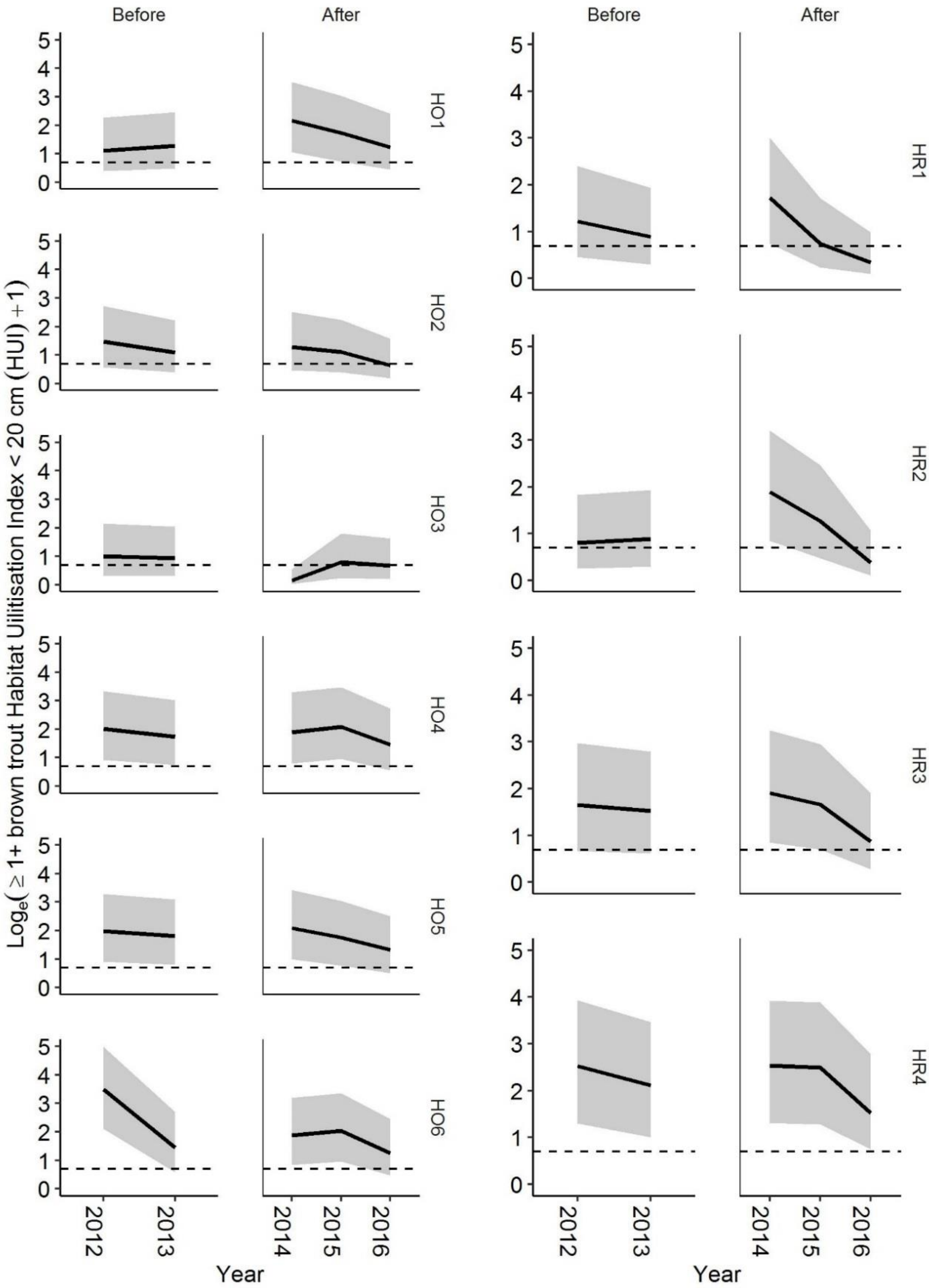

Figure 6.13. Log$_e$ + 1 transformed HUI for ≥1+ (< 20 cm) brown trout in the impact (left) and control sites (right) during the before (2012 – 2014) and after (2015 – 2016) period. Grey bands represent 95% confidence limits. Dashed line represents HUI value where observed and expected densities are equal.

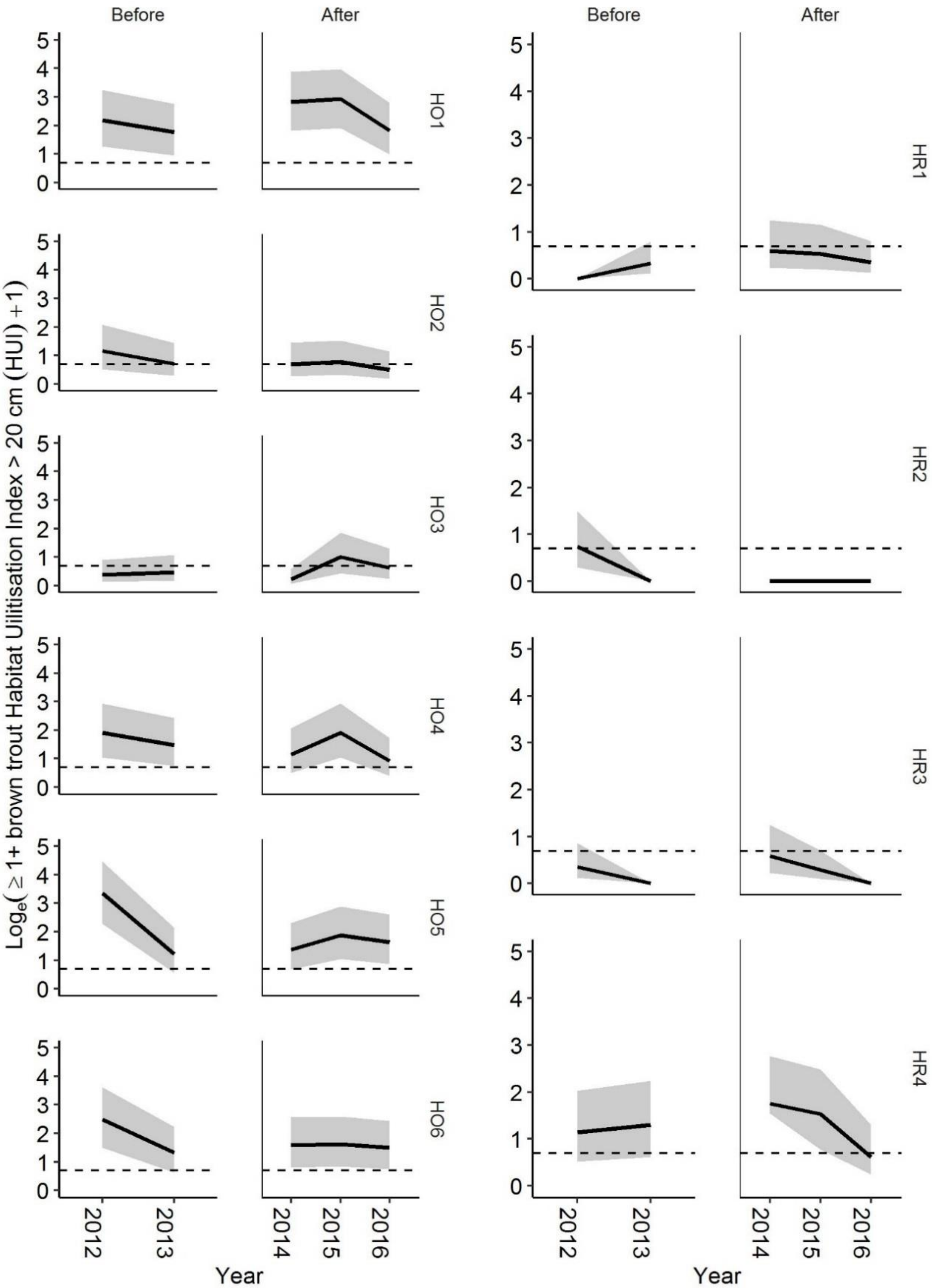

Figure 6.14. Log$_e$ + 1 transformed HUI for ≥1+(> 20 cm) brown trout in the impact (left) and control sites (right) during the before (2012 – 2014) and after (2015 – 2016) period. Grey bands represent 95% confidence limits. Dashed line represents HUI value where observed and expected densities are equal.

Impact assessment (BACI)

A resource calculation performed on both the before and after periods confirmed that the intensity of sampling was adequate to detect a biologically meaningful change to ≥1+ (<20 cm and >20cm) brown trout within the timeframe of this study. The addition of the after period to the dataset reduced the level of actual variance for HQS densities for both size classes of ≥1+ brown trout (Table 6.21).

Table 6.21. Actual Vx for *before* and after period and Target Vx to detect a ≥50% change to ≥1+ brown trout Habitat Quality (HQS). **Bold and underlined** signifies that significant change to ≥1+ densities could be detected within the study parameters

	Actual VX	Target Vx for ≥50% Increase	Target Vx for ≥50% Decrease
< 20 cm Before	0.040	**<u>0.852</u>**	**<u>0.090</u>**
> 20 cm Before	0.020	**<u>0.092</u>**	0.034

Habitat quality at both impact and control sites before the revision of the compensation release from Digley and Brownhill reservoirs in 2014 suggested that the expected populations of ≥1+ (<20 cm) brown trout were very similar (Figure 6.15). Following the change to the flow regime in 2014 there was only a marginal increase to the expected densities ≥1+ (<20 cm) brown trout, however, during the same period the expected densities of ≥1+ (<20 cm) brown trout increased by a greater magnitude at the control sites. The BACI GLMM revealed that this marginal increase to ≥1+ (<20 cm) brown trout HQS densities in the River Holme were significantly attributed to the change in flow regime from the Digley and Brownhill reservoirs (Table 6.22). This result, however, should be interpreted with caution; the purpose of the control sites are to gauge the trends to fisheries and habitat metrics that would be seen in the impact sites if no compensation release change occurred. It must, therefore, be considered that the marginal increase to the ≥1+ (<20 cm) brown trout would be seen in greater magnitude if no change to the compensation regime occurred. Therefore, the revision to the compensation can be viewed in real terms as detrimental to the habitat quality for ≥1+ (<20 cm) brown trout. The change to habitat quality for ≥1+ (>20 cm) brown trout as reflected in the HQS densities were less complex to address. There HQS densities for ≥1+ (>20 cm) brown trout were much lower at impact sites than at reference sites before the compensation release change, however following the change HQS densities of ≥1+ (>20 cm) brown trout increased at impact sites and decreased at control sites (Figure 6.16). The BACI GLMM identified that the interaction between area and period was significant (Table 6.22), and therefore the increase to the ≥1+ (<20 cm) brown trout habitat quality in the River Holme can be significantly attributed to the revision of the flow trial from Digley and Brownhill reservoir in 2014.

Table 6.22 BACI GLMM to detect a change in ≥1+ brown trout HQS density following the revision of the compensation release from Digley and Brownhill Reservoir in 2014

	Sum Sq	Mean Sq	DoF	DenDF	F	P
≥1+ (<20 cm)						
Area	4.350	4.350	1	43.000	22.303	**<u><0.001</u>**
Period	<0.001	<0.001	1	3.019	0.003	0.957
Area:Period	2.699	2.699	1	43.00	13.844	**<u><0.001</u>**
≥1+ (>20 cm)						
Area	0.656	0.686	1	43.000	7.197	0.010
Period	0.284	0.284	1	3.107	3.017	0.173
Area:Period	0.467	0.467	1	43.000	5.107	**<u>0.029</u>**

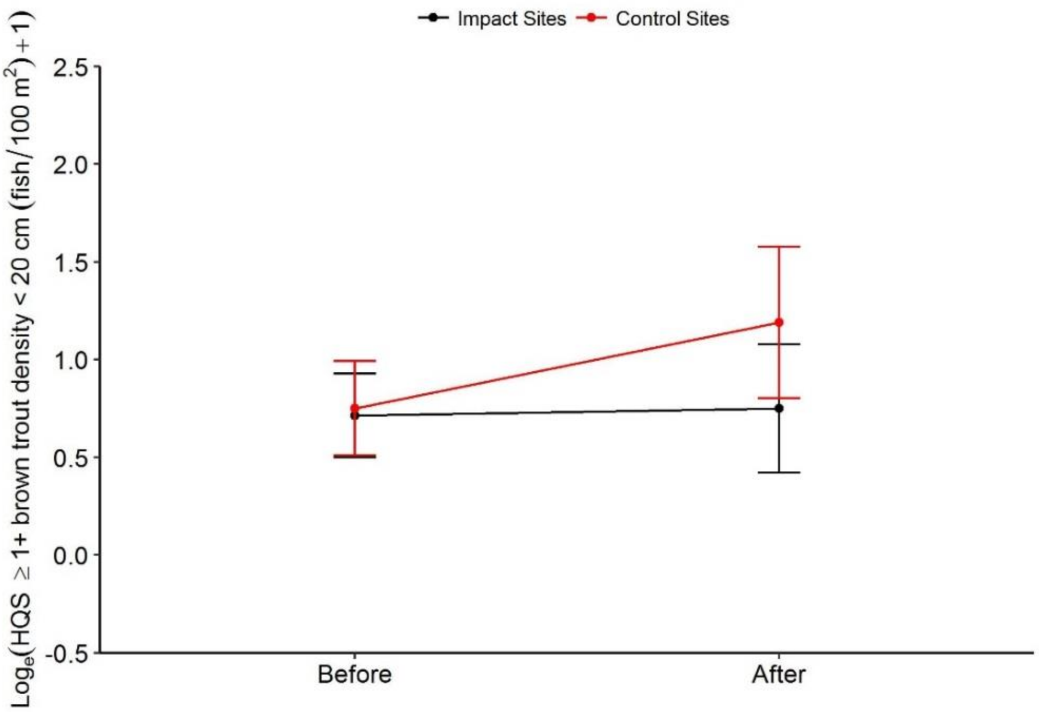

Figure 6.15 Mean (±95%) ≥1+ (<20 cm) HQS densities during *before* and *after* periods for impact (black) and control (red) sites. Trend lines represent significant interaction between Area and Period in BACI GLMM.

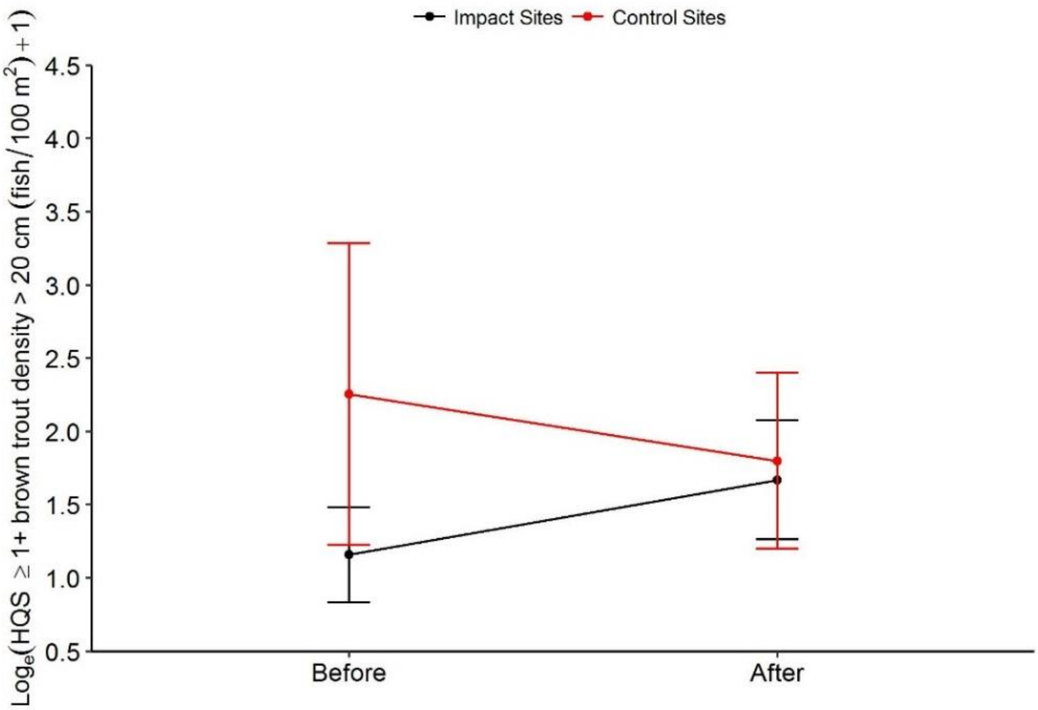

Figure 6.16. Mean (±95%) ≥1+ (<20 cm) HQS densities during *before* and *after* periods for impact (black) and control (red) sites. Trend lines represent significant interaction between Area and Period in BACI GLMM

6.4 Discussion

Densities of 0+ brown trout were significantly lower in the River Holme than the River Ribble throughout the before period; the poor densities of 0+ brown trout, however, were not exclusively representative of the habitat quality as expected densities were higher than typically observed at the three most upstream survey sites (HO1 – HO3) in the River Holme. Densities of ≥1+ brown trout were also found in low densities at these upstream sites. Low recruitment therefore, is likely not driven by density-dependent factors, such as competition for food and territory (Lobón-Cerviá 2005, Daufresne and Renault 2006, Jonsson and Jonsson 2011). It is possible that abiotic factors outside the scope of detection within the HABSCORE model are limiting factors to recruitment at the impact sites HO1 –HO3. For example, sites HO1 and HO3 are adjacent to roads and urban developments and run-off from urbanised areas are usually contaminated (Gobel *et al.* 2007), which can lead to reduced levels of recruitment (Woodward *et al.* 1995). Poor water quality is of particular concern for the study sites on the River Holme mainly due to the heavily urbanized surrounding land use, as well as the industrial development, such as mills and mines – many of which have been in operation for decades and likely attributing to poor water quality (Jarvis and Younger 2000). This is not to say that the remaining sites were isolated from the adverse effects of urbanization, with site HO4 located adjacent to a large car park, as well as potential urban runoff present at sites HO5 and HO6, but there are likely some influences outside the scope of this study that are limiting recruitment of 0+ brown trout.

In 2012 and 2013 the observed densities of 0+ brown trout in the River Ribble were exceptionally high, exceeding 50 fish/100 m^2. Investigations into the habitat quality revealed that the expected densities of 0+ brown trout at these control sites should have been significantly lower. This out-performance of the expected densities of 0+ brown trout in the River Ribble was seen across all sites, but only significantly so at site HR1. There was no apparent explanation for the exceptionally high 0+ densities in the upstream sites in the River Ribble; it is not uncommon for 0+ brown trout populations to vary by orders of magnitudes between years (Jenkins *et al.* 1999), therefore these high-density estimates are likely a reflection of natural variability. ≥1+ brown trout were present in lower densities across the control sites, but they were still found in higher densities than predicted from the habitat, except for brown trout >20 cm, which were found in very low densities and underutilizing the available habitat in the River Ribble, likely due to the shallow nature of the river at sites HR1 – HR3. >20 cm brown trout were found to be in greater numbers at site HR4, due to the presence of a large deep weir pool at the top of this site. The land surrounding the two upstream sites on the River Ribble (HR1 and HR2) was dominated by grazing pasture, and studies into land-use and their effects on macroinvertebrates reveal that the macroinvertebrate densities are positively influenced by this type of land use, with higher expected macroinvertebrate densities surrounding grazing land due to higher nutrient loadings from the runoff (Hepp *et al.* 2010). Investigations into other salmonid populations in laboratory stream channels revealed that increasing macroinvertebrate density reduced individual territory size and aggressive interactions between conspecifics (Slaney and Northcote 1974, Nicola *et al.* 2015). It is possible that these land-use and macroinvertebrate interactions are influencing the high 0+ densities on the River Ribble. However, macroinvertebrate data would be required to support this hypothesis. The length at capture of 0+ brown exhibited considerable variation amongst individuals, but no significant differences were found either between sites or between impact and control sites, suggesting that the growth of brown trout was not influenced by site differences, i.e. better feeding opportunities and food availability at one site compared to another (Newman 1993).

Following the revision of the compensation release in 2014, observed densities of 0+ brown trout declined at sites on the River Holme. In the first year of the new flow regime, 0+ brown trout densities exhibited substantial variation in between sites, with population classifications ranging from fishless to good. The absence of 0+ brown trout from site HO3 in 2014 is not fully understood, before the change in the flow regime 0+ brown trout were present, albeit in low densities. The same is true for ≥1+ brown trout densities, which were very low at site HO3 in 2014. HABSCORE analysis of the site revealed that the habitat quality of this site was better than in previous years, as indicated by the higher than expected densities of 0+, ≥1+ (<20 cm) and ≥1+ (>20 cm) brown trout. It is possible that the low numbers of trout present at this site was due to an incident of environmental degradation possibly a pollution event. Immediately upstream of site HO3 is a Livestock Auction Market, underneath which runs a storm drain that drains into the River Holme 20 m upstream of site HO4. Point source pollution, such as detergents and other cleaning agents, are known to have deleterious effects on brown trout populations (Madsen *et al.* 2001). Unfortunately, no water quality data were available for the River Holme. Therefore, this hypothesis is purely speculative. However, it must also be considered that this loss of 0+ brown trout was because of natural stochasticity, as both sites HO2 and HO4 held low densities of 0+ brown trout.

Substantial declines in 0+ brown trout densities were also seen in the control sites HR1 and HR2, in 2014 compared to 2013 with densities at site HR1 an order of magnitude lower than in previous years. It is likely that this poor recruitment reflected the high natural variability of brown trout (Jenkins Jr *et al.* 1999). Again, no water quality data were recorded in the River Ribble, so it is not possible to rule out that these declines in 0+ densities were correlated to any changes to water chemistry. HABSCORE data revealed that expected densities should be higher than those observed, therefore habitat elements recorded by the HABSCORE analysis were not thought to be limiting to the poor recruitment in 2014. In general terms there was a decline in 0+ brown trout densities across both impact and control sites, the BACI GLMM revealed that the change to the compensation release was not significantly linked to this decline in recruitment in the River Holme, as it was seen in the control populations and more likely as a result of natural population variability and spatial synchrony (Bret *et al.* 2016). This decline in 0+ brown trout densities was not thought to be induced by habitat degradation either, as expected 0+ brown trout densities derived from the habitat quality scores, suggest that there was an improvement in 0+ brown trout habitat quality across the impact and control during the period 2014 – 2016. The BACI GLMM could not isolate the change to 0+ brown trout habitat from natural variability as seen in the control sites, so the decrease of 0+ brown trout densities could not be attributed to the revision in the compensation release in 2014. It is, however, possible that the revision in the flow regime in the River Holme provides better 0+ brown trout habitat, but HABSCORE data, due to the timings of the surveys, were not able to account for the lower summer flows in June and July compared to the old regime. Investigations into population dynamics of 0+ brown trout in HMWBs, including historical River Holme data, revealed that 0+ brown trout populations were negatively correlated with summer flow rates, suggesting that better habitat for 0+

brown trout was present under lower summer flows in these heavily regulated rivers (see chapter 3).

During the after-period, densities of ≥1+ (<20 cm) brown trout marginally improved across the River Holme, but a greater magnitude increase of this age/size class was seen in the River Ribble. The BACI GLMM revealed that this increase was not significantly related to the revision of the compensation flow. Habitat quality was not believed to play an important factor in the increase of ≥1+ (<20 cm) brown trout densities during the after period as the quality of habitat for this age/size class showed no change in the River Holme following the change to the compensation release. It is probable that the change to ≥1+ (<20 cm) brown trout densities were related to spatial synchrony between the populations (Bret *et al.* 2016). The negligible change to habitat quality for ≥1+ (<20 cm) brown trout in the River Holme suggests that the change in flow was not of sufficient magnitude to induce any meaningful changes to ≥1+ (<20 cm) brown trout habitat quality. As habitat quality improved at control sites, the BACI GLMM interprets this result as a significant decline in habitat quality across sites in the River Holme, as the BACI model assumes that changes to the control sites represent changes that would likely be seen in the impact sites if there were no flow trial. By contrast, improvements to ≥1+ (>20 cm) brown trout densities were observed in the River Holme in contrast declines seen in the River Ribble. However, this change to ≥1+ (>20 cm) brown trout densities could not be significantly attributed to the new compensation flow. The increase in the River Holme was significantly related to the revision of the compensation release from Digley and Brownhill reservoirs. As habitat surveys were undertaken at the same time of year as the fisheries surveys, in September, it is possible that the improvement of the habitat quality for ≥1+ (>20 cm) brown trout was reflected in the elevated flows in this month following the revision of the compensation release. Higher flow rates will lead to deeper, fast-flowing water habitat that larger brown trout favour (Cowx *et al.* 2004). This is important for brown trout communities as larger brown trout are more fecund and can play important roles in the future recruitment of brown trout communities (Jonsson and Jonsson 2011).

It was previously speculated that no reliable conclusions can be drawn on whether the surrounding land use is influencing the aquatic ecosystem in any meaningful way. Therefore, further in-depth investigations into the water quality of both rivers would be required to determine the validity of this hypothesis. It is as likely that as declines were seen in across the impact and control sites between the two periods of this study that synchronised regional events are also influencing population dynamics of brown trout between the impact and control sites (Bret *et al.* 2016). Both habitat quality for 0+ and >1+ (> 20 cm) also improved following the revision of the compensation release. However, the habitat quality for 0+ brown trout also increased at control sites. The improvement of habitat quality for ≥1+ (>20 cm) brown trout and associated density of brown trout was significantly attributed to the revision of the compensation release in 2014. The conclusions form this study are encouraging, as improvements to the habitat quality were seen in the River Holme following the revision of the compensation release. It is recommended that this revised release regime from Brownhill and Digley reservoirs remain in place following the conclusion of this study.

7 PHYSICAL HABITAT RESTORATION ON BROWN TROUT POPULATION DYNAMICS AND HABITAT QUALITY IN THE RIVER WASHBURN (DOWNSTREAM OF SWINSTY RESERVOIR)

7.1 Introduction

Improving instream habitat is a fundamental strategy to help achieve "Good Ecological Potential" (GEP) in waterbodies designated as heavily modified (HMWBs). In rivers that are heavily regulated by impounding reservoirs, the introduction/amendment of a compensation release is commonplace (Acreman *et al.* 2009) as studied in Chapters 4, 5 and 6. In scenarios where flow releases from reservoirs cannot be modified, alternative approaches, such as physical habitat restoration, are required to attempt to improve GEP in associated waterbodies.

Physical habitat restoration methods are popular for improving habitat in rivers (Roni *et al.* 2006, Whiteway *et al.* 2010) and may be considered as an appropriate HMWB mitigation measure where flow releases cannot be modified. Physical habitat restoration is a broad term that describes a number of different concepts and techniques that can be used individually or conjunctively to alter the physical environment with the goal of supporting better ecology (Forseth and Harby, 2014). Typical examples of physical habitat restoration consists of the addition/removal of cover, boulders and large woody debris, as well as alteration of substrates and earth works to alter the flow dynamics and physical dimensions of the river (Cowx and Welcomme 1998, Roni and Quimby 2005, Whiteway *et al.* 2010, Thompson *et al.* 2017). For HMWBs where high levels of habitat degradation have occurred and where alteration of the flow regime is not possible from impounding reservoirs, habitat restoration can be considered as a suitable alternative to achieving GEP (Wharton and Gilvear 2007). It is believed that the biological response from fish communities in the heavily modified waterbodies following physical restoration projects would be positive, with meta-analysis of physical restoration projects revealing that 73% of projects resulted in increased densities of salmonids (Whiteway *et al.* 2010). A major concern of habitat restoration works across the UK is that assessment often lacks adequate monitoring at appropriate spatial and temporal scales (Hammond *et al.* 2011, Verdonschot *et al.* 2013). This study focuses on the restoration of 1-km reach of the River Washburn, a HMWB in North Yorkshire. In this study, fisheries and habitat data were collected across sites along the River Washburn and suitable control sites, to undertake a BACI model to provide robust assessment of the habitat and biological responses to restoration works in the River Washburn.

Specific objective

- Investigate 0+ and ≥1+ brown trout populations and habitat quality at sites along the River Washburn (downstream of Swinsty reservoir) before and after habitat restoration works completed in summer 2015
- During the same timeframe, investigate 0+ and ≥1+ brown trout populations and habitat quality at sites where the habitat quality would not be influenced by the restoration works (control sites) to determine the level of temporal variability of brown trout populations and habitat quality in the region.
- establish the impact (if any) that restoration works had on the brown trout populations and habitat quality using a Before After Control Impact (BACI) methodology.

Outputs will determine the effectiveness of the physical habitat restoration in the River Washburn on the habitat quality and brown trout populations.

7.2 Methodology

7.2.1 Study reaches

Yorkshire Water Services, as part of the MM6 river restoration programme, identified a stretch of the River Washburn, downstream of an impounding reservoir (Swinsty reservoir) where the habitat was degraded, and the flow regime regulated. Due to operational constraints, the issue of low homogenised flows in the River Washburn could not be addressed by modification of the compensation release from Swinsty reservoir. Therefore, river restoration methods were proposed to attempt to improve the habitat and resident fish populations. To determine any biological responses to the restoration works, brown trout were selected as indicator species due to their abundance in both the impact and control sites as well as their sensitivity to habitat requirements. (Chapter 2.5)

River Washburn (Downstream of Swinsty Reservoir)

The River Washburn is a tributary of the River Wharfe in North Yorkshire, currently impounded by four large reservoirs (dam wall >10 m) – Thruscross, Fewston, Swinsty and Lindley Wood reservoirs. Water storage operations in the sequence of reservoirs on the River Washburn are complex, with these reservoirs not only providing water storage and supply for YWS but also a vast array of leisure activities, such as sailing (Thruscross Reservoir) and fishing (Fewston and Swinsty reservoir). Regular high magnitude flow releases also occur from Thruscross Reservoir to provide fast flowing water for canoe slalom competitions in the River Washburn immediately downstream. However, these do not have an impact on the study reach. No formal compensation release exists from Swinsty Reservoir and there is a minimal release of water into the downstream study reach. The river habitat downstream has been altered resulting in an over-widened, slow-moving stretch of river with high levels of siltation and lack of habitat diversity. Operational demands of the reservoir network determined that it was not feasible to introduce a compensation flow from Swinsty Reservoir into the River Washburn to counter low flows. Physical habitat improvement measures were, therefore, determined to provide better ecological improvement opportunities with the aim of increasing the ecological potential of the River Washburn to "good". Proposed habitat alterations included:

- Narrowing and reprofiling the river channel in areas to provide areas of faster flowing water in between areas of wider deeper habitat (pool riffle sequence)
- Creation of Gravel bars within the channel to help support salmonid ecology
- Seeding of finer substrates to promote sediment transport
- Riparian management, limbing and felling of deciduous trees along the bank to reduce the level of cover. Felled trees were recycled into groynes and flow deflectors to provide areas of marginal habitat during high flow events. Root ball structures (root systems from felled trees) were also introduced to channel to provide complex marginal habitat.

These proposed restoration measures were planned for a 1-km section of the River Washburn, downstream of Swinsty reservoir. The average gradient of the study reach was fairly shallow (1%) The surrounding land is exclusively used as rough pasture and grazing. The restoration work commenced in April 2015 and the finished in August 2015. To monitor the effectiveness of the measures, six 50 m sites (SW1 – SW6) were selected along a 1.3-km segment of the River Washburn, with three sites SW2 – SW4 within the restoration zone. The three sites outside the restoration zone were sampled to determine if any changes to fish assemblages occurred outside the habitat restoration zone. The downstream sites (SW5 and SW6) outside of the restoration zone may be considered to have passive improvement through dispersal of gravels and increase in water velocities.

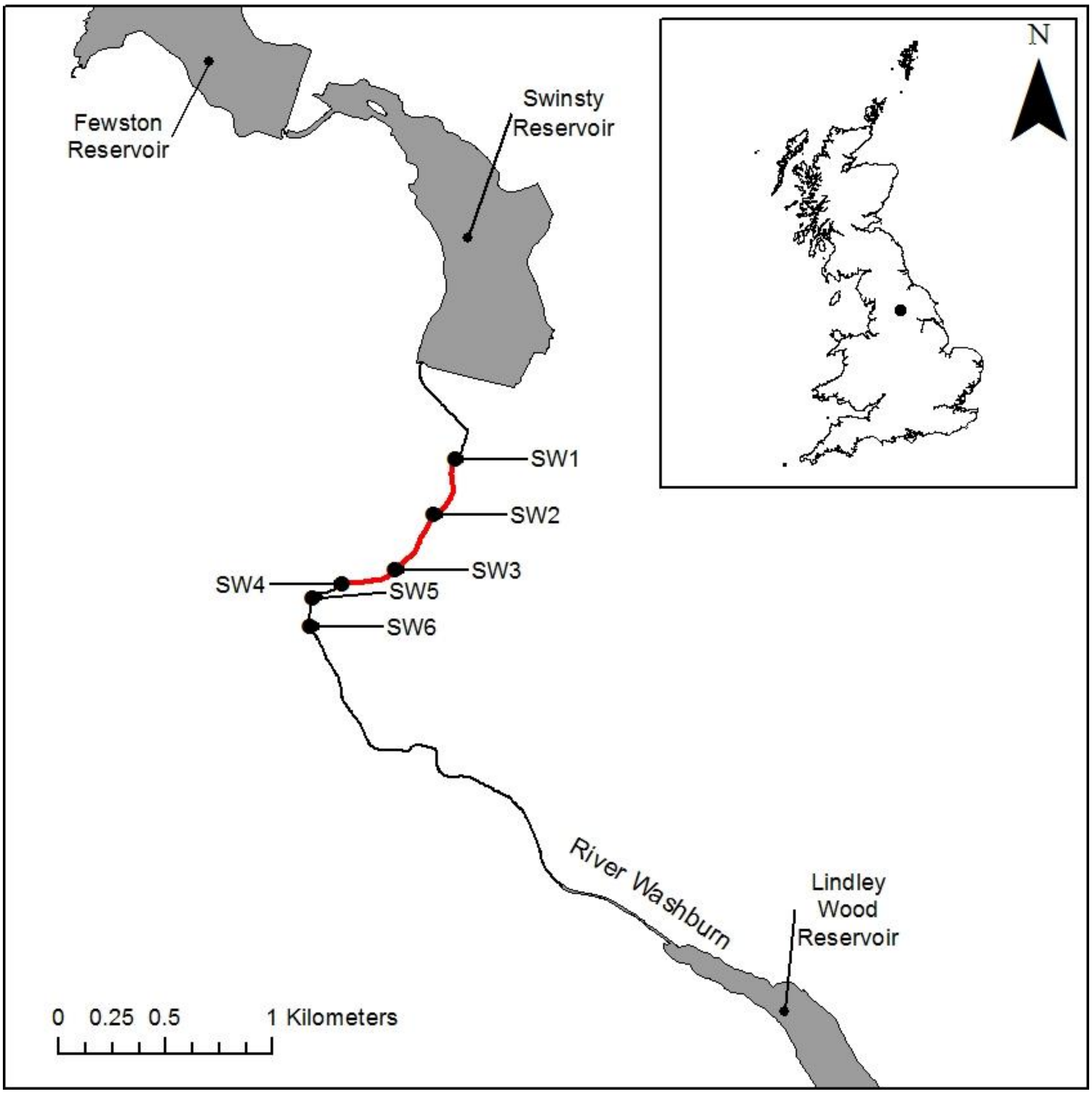

Figure 7.1. The Location of the six impact sites (SW1 – SW6) on the River Washburn in relation to Fewston, Swinsty and Lindley Wood Reservoirs. Red section of the river denotes the 1km habitat restoration section.

Barden, Ings and Ashfold Side Becks (Control sites)

No reaches of the River Washburn were identified as being suitable control sites for the restoration programme. Downstream sites on the River Washburn were also discounted due to the potential influence of the restoration works violating the independence that is required of the control sites. Due to the proximity of the River Washburn to the River Dibb and the studies running concurrently, the nine control sites selected for the Grimwith flow trial study (WR1 – WR9 (see section 4.1.1.1) at Barden Ings and Ashfold Side Beck were deemed appropriate control sites. The spatial proximity between the impact and control sites was reasonably large (16 km), but as synchronising environmental factors have found to operate at much larger scales (75-km) (Bret *et al.* 2016). The average gradient of the control sites was much steeper (4.8%) than the impact sites, but as the predominant land use (rough pasture and grazing) was similar to the River Washburn these nine sites were deemed acceptable control sites.

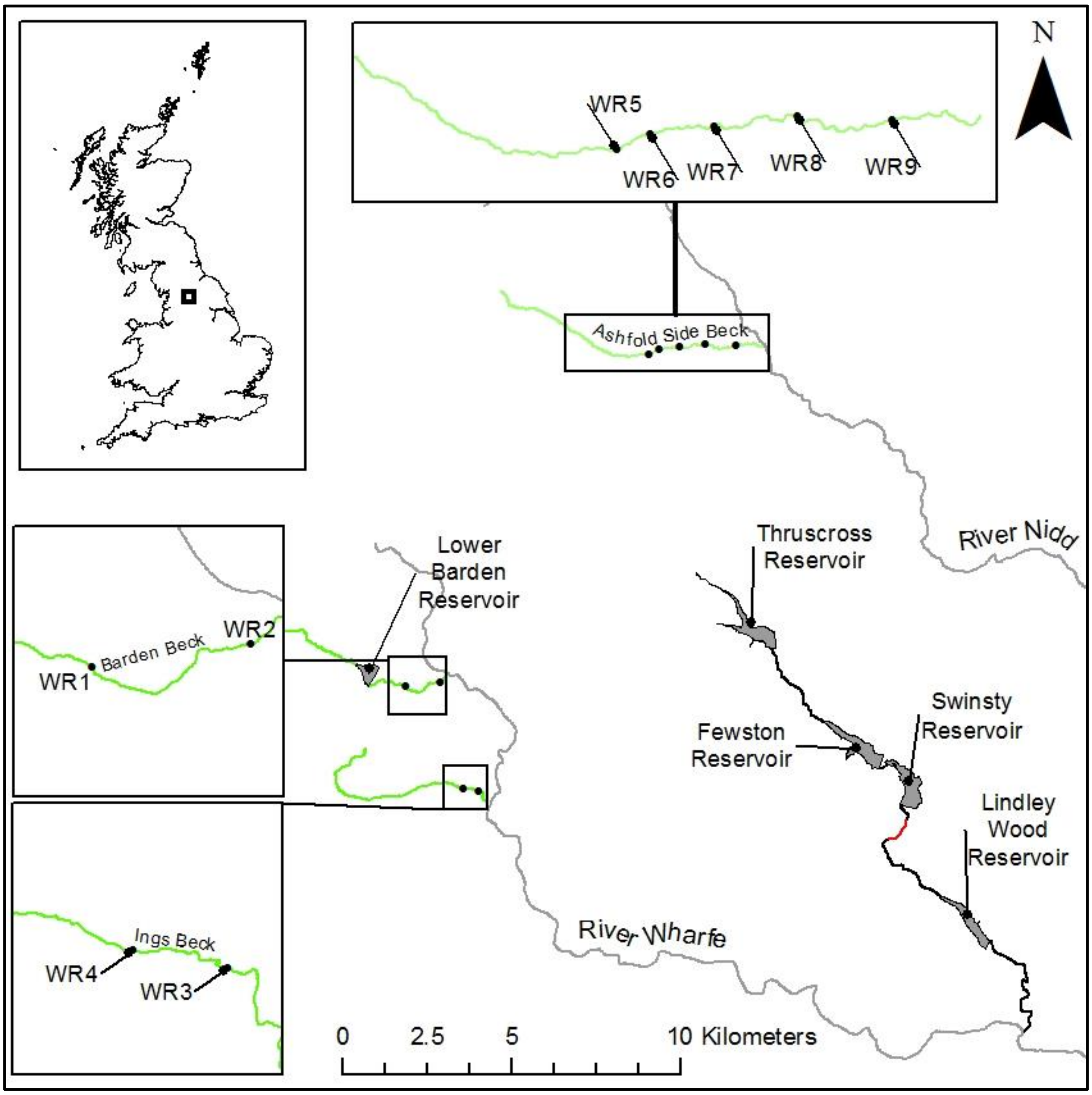

Figure 7.2. Location of the nine control sites (WR1 – WR9) in relation to each other as well as the River Washburn and proposed restoration reach (red).

Environmental Investigation.

Fisheries and habitat investigations in the impact (River Washburn) and control sites (Barden, Ings and Ashfold Side Becks) were undertaken annually during the before (2012 – 2014) and after (2015 – 2016) periods. The fish survey methodology used was identical to the methodology outlined in Chapter 3.2.3. All fisheries and HABSCORE methodologies are described in Chapter 4.2 and will not be repeated. However, due to exceptionally low numbers of 0+ brown trout in the River Washburn in some years, as well as variations of the timing of surveys on the River Washburn (August – October), the reliability of 0+ brown trout length at capture information was poor; i.e. the high error associated with estimates derived from few individuals. Therefore, a BACI analysis on the change of length at capture of 0+ brown trout was not considered viable.

7.3 Results

7.3.1 Impact of restoration works on 0+ brown trout densities

Baseline Monitoring (before)

During the before period, 0+ brown trout were found in very low densities in the River Washburn, with this age class absent from sites SW1 in all years and sites SW1-SW3 and SW5 in 2014. Where caught, 0+ brown trout densities ranged from 0.24 to 9.81 fish/100 m^{2}. (Table 7.1); at sites SW2 – SW5 0+ brown trout densities were classed as poor, but at site SW6 average in 2012 and fair/poor in 2013 and 2014. While there was significant ($F_{5, 11}$ = 48.854, P<0.05) variation amongst the sites, there was no significant difference amongst the three years of the before years ($F_{1, 16}$ = 0.374, P>0.05) suggesting that densities of 0+ brown trout were consistently low (Figure 7.3).

0+ brown trout densities at the control sites were much more varied between sites and years than in the impact sites. During the pre-restoration (before) period, 0+ brown trout densities in control sites varied from 0.9 to 47.0 fish/100 m^{2} and ranged from poor to excellent (Table 7.1); a nested ANOVA found no significant differences between the three control rivers ($F_{2,18}$ = 0.233, P>0.05) or sites within rivers ($F_{6,18}$=0.395, P>0.05). There was a significant difference between years at the control sites ($F_{1, 25}$ = 13.100, P < 0.05). In 2012, 0+ brown trout populations were predominantly poor and populations at two sites (WR2 to WR3) were classified as fair/poor (Table 7.1). Densities across sites were generally average, improving from 2012-2014, but individual populations at some sites did not follow this pattern. For example, 0+ brown trout densities at site WR6 increased from 2.2 to 47.0 fish/100 m^{2} from 2012 to 2014, whereas in the same period 0+ brown trout densities in a site further downstream (WR8) declined from 19.1 to 2.7 fish/100 m^{2}. This high variability is reflected in the large confidence limits of the 2013 and 2014 average density estimates (Figure 7.3).

Before the commencement of the flow trial in 2015, a resource calculation was performed on the 0+ brown trout "before" data (Table 7.2). Under the assumption that the temporal variability in the impact and control sites persists, the resource calculation revealed that a ≥50% increase in 0+ brown trout populations would be detectable following two years of "after" monitoring (Table 7.2). Due to the variability and low densities in the impact sites, it would not be possible to detect a ≥50% decrease in 0+ brown trout densities within two years, and further years of monitoring would be required to ensure a statistically robust conclusion.

Post-Restoration works (After)

In the two years following the restoration works, 0+ brown trout were only captured on three occasions, at sites SW1 and SW6 in 2015 and SW6 in 2016, with densities at these three sites ranging from 0.25 to 2.88 fish/100 m^2 (Table 7.1). 0+ brown trout were absent from all the remaining sites. Crucially, no 0+ brown trout were present in the restoration zone (SW2 – SW4). Despite this, 0+ brown trout densities were not found to be significantly lower than in previous years (Figure 7.3), with a nested ANOVA (years within Period) revealing that there were no significant changes to 0+ brown trout densities either between the period ($F_{1,26}$ = 2.69, P>0.05) or years nested within period ($F_{2,26}$ = 0.368, P>0.05).

During the “after” period in the control sites, there was a general decrease in 0+ brown trout population densities across all control sites, with 0+ brown trout absent from four sites in 2016 (WR1, WR3, WR4 and WR8). At the remaining sites (WR2, WR5-WR7 and WR9 in 2016 and all sites in 2015), densities during the after period ranged from 0.6 to 20.7 fish/100 m^2 with only one site classed as good, and the majority of sites classed as either poor or fair/poor (Table 7.1). It should be noted that there was a significant decline ($P < 0.05$) in the average densities from 2015 to 2016 as 0+ brown trout were absent from four sites in 2016; this phenomenon was not observed in the impact sites. It is worth noting that the variability of 0+ brown trout densities was largely between years ($F_{1,16}$ = 6.693, P<0.05), i.e. there was no clear trend of some sites yielding continually higher 0+ brown trout populations than others (Figure 7.1). Many sites showed considerable increases and decreases in 0+ densities over a shorter time period, which is likely a result of the inherent population variability that persists in many salmonid communities.

Table 7.1 Density estimates ± 95% C.L. of 0+ brown trout at impact and control sites during both the before (2012 – 2014) and after (2015 – 2016) periods. Colours denote EA-FCS abundance classification

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

River Name	Site Identifier	2012	2013	2014		2015	2016
		Before				After	
impact Sites							
River Washburn	SW1	0.00 ± 0.0	0.00 ± 0.0	0.00 ± 0.0		2.88 ± 0.3	0.00 ± 0.0
River Washburn	SW2	0.24 ± 0.1	0.86 ± 0.2	0.00 ± 0.0		0.00 ± 0.0	0.00 ± 0.0
River Washburn	SW3	0.53 ± 0.3	0.64 ± 0	0.00 ± 0.0		0.00 ± 0.0	0.00 ± 0.0
River Washburn	SW4	2.48 ± 0.2	1.77 ± 0.3	0.36 ± 0.0	0.00 ±	0.00 ± 0.0	0.00 ± 0.0
River Washburn	SW5	0.75 ± 0.3	0.85 ± 0.3	0.00 ± 0.0		0.00 ± 0.0	0.00 ± 0.0
River Washburn	SW6	9.81 ± 0.0	6.55 ± 0	7.09 ± 0.0		0.63 ± 0.0	0.25 ± 0.10
Control Sites							
Barden Beck	WR1	1.48 ± 0.2	2.68 ± 0.4	27.53 ± 2.0		5.49 ± 0.9	0.00 ± 0.0
Barden Beck	WR2	3.02 ± 0.2	8.96 ± 0.5	2.98 ± 0.5		11.21 ± 0.3	0.6 ± 0.0
Ings Beck	WR3	3.88 ± 0.8	16.07 ± 0.4	13.78 ± 1.6		20.74 ± 0.9	0.00 ± 0.0
Ings Beck	WR4	1.5 ± 0.4	10.75 ± 1.0	1.67 ± 0.5		4.01 ± 0.5	0.00 ± 0.0
Ashfold Side beck	WR5	0.93 ± 0.4	3.99 ± 0.7	22.75 ± 2.3		2.50 ± 0.4	2.67 ± 0.4
Ashfold Side beck	WR6	2.23 ± 0.8	2.59 ± 0.1	47.02 ± 2.6		5.35 ± 1.0	3.55 ± 2.1
Ashfold Side beck	WR7	1.92 ± 0	10.19 ± 0.8	35.5 ± 2.3		7.26 ± 1.6	5.39 ± 1.7
Ashfold Side beck	WR8	0.97 ± 0.4	19.06 ± 1.4	2.75 ± 0.2		4.3 ± 0.4	0.00 ± 0.0
Ashfold Side beck	WR9	1.81 ± 0	5.9.0 ± 0.8	11.97 ± 0.5		1.33 ± 0.1	1.57 ± 0.1

Table 7.2. Actual Vx for *before* period and Target Vx to detect a ≥50% change in 0+ brown trout densities. **<u>Bold and underlined</u>** signifies that significant change to 0+ brown trout densities could be identified within the study parameters.

Actual Vx	Target Vx for ≥50% Increase	Target Vx for ≥50% Decrease
0.084	0.065	0.007

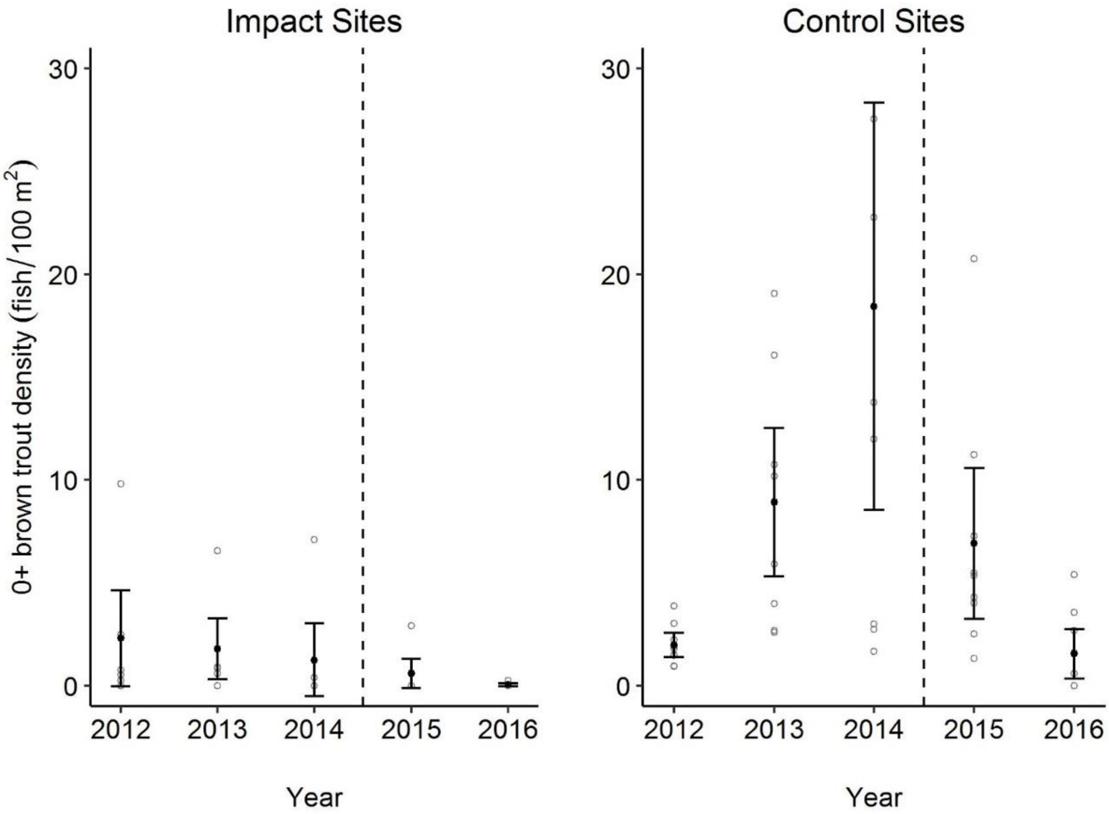

Figure 7.3. Average 0+ brown trout densities ± 95 % C.L. for all impact and control sites, throughout the study period. The dashed line represents the division between *before* and *after* periods. Greyed circles represent individual density estimates

Impact assessment (BACI)

Actual variance decreased in the 0+ brown trout dataset following the addition of the after years data. Therefore, it would be possible to detect a biologically meaningful increase in 0+ brown trout densities. Due to the low densities, it would still not be possible to prove with statistical certainty if a ≥50% decrease to 0+ brown trout in the River Washburn was attributed to the restoration works (Table 7.3).

Table 7.3. Actual Vx for full study and Target Vx to detect a ≥50% change to 0+ brown trout densities. **<u>Bold and underlined</u>** signifies that significant change to 0+ densities could be detected within the study parameters.

Actual Vx	Target Vx for ≥50% Increase	Target Vx for ≥50% Decrease
0.049	**<u>0.065</u>**	0.007

0+ brown trout densities were significantly lower in the impact sites than the control sites (Table 7.4). There were no significant changes to 0+ brown trout densities at either impact or control sites in the after period (Table 7.4), despite densities declining dramatically in 2016 from previous years (Figure 7.4). The interaction between area and period in the BACI GLMM was not significant (Table 7.4). It is, therefore, not possible to link the habitat improvement works to the decline in 0+ brown trout densities in the River Washburn with any degree of statistical confidence.

Table 7.4 BACI GLMM to detect a change in 0+ brown trout density following the restoration work on the River Washburn in 2015

	Sum Sq	Mean Sq	DoF	DenDF	F	P
Area	15.82	15.82	1	13.722	29.217	**<u><0.001</u>**
Period	0.934	0.934	1	3.041	1.725	0.279
Area:Period	0.086	0.086	1	55	0.16	0.691

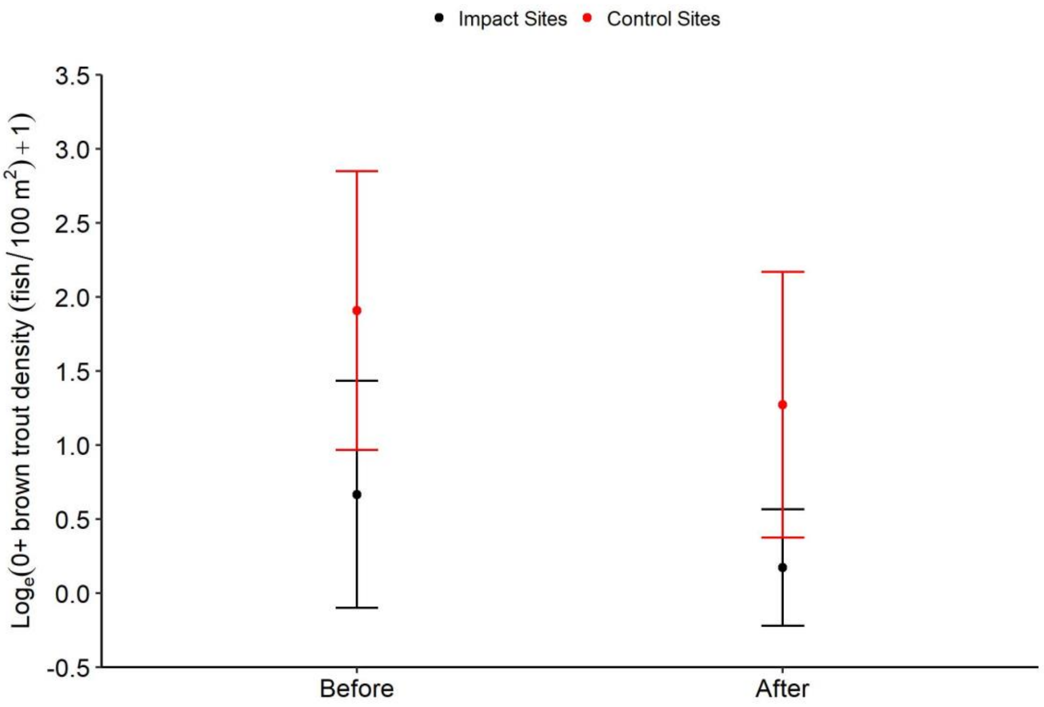

Figure 7.4. Mean (±95%) 0+ brown trout densities before and after for impact (black) and control (red) sites. All densities are ln+1 transformed.

7.3.2 Impact of restoration works on ≥1+ brown trout densities

To compare HQS values, ≥1+ brown trout densities are presented in two size-based categories (≥1+ under 20cm, ≥1+ greater than 20 cm). However, ≥1+ brown trout (all sizes) densities and their respective EA-FCS classifications are presented in Appendix Table A.7. ≥1+ brown trout were caught at all control sites for the duration of the study typically in higher densities than 0+ brown trout, but ≥1+ brown trout were caught in low densities in impact sites at similar densities to 0+ brown trout. ≥1+ brown trout were absent from sites SW1 in 2015 and 2016 and SW3 in 2014 (Appendix, Table A.7).

Baseline monitoring (before)

≥1+ brown trout were caught in low numbers at all impact sites prior to the restoration works, with densities ranging from 0.65 to 7.33 fish/100 m^2 (Appendix, Table A.7); these populations ranged from poor to average. ≥1+ (<20 cm) brown trout were found in slightly higher densities compared to the larger (> 20 cm) brown trout, but not significantly higher ($P>0.05$) (Figure 7.5); densities of this size class (> 20 cm) ranged from 0.46 to 5.92 fish/100 m^2 (Table 7.5). During the before period there were no significant deviations between the years ($F_{1,16} = 0.137$, $P>0.05$), but ≥1+ (<20 cm) brown trout densities were marginally higher in 2013 compared to other years. There were also no significant differences in ≥1+ (<20 cm) brown trout densities amongst sites ($F_{5,12} = 0.682$, $P>0.05$). ≥1+ (>20 cm) brown trout were absent from sites SW3 and SW6 in 2012, SW2 and SW6 in 2013 and SW3, SW5 and SW6 in 2014, and where captured, were caught in the range 0.26 to 1.78 fish/100 m^2 (Table 7.6) prior to the restoration works. Despite ≥1+ (>20 cm) brown trout being absent from site SW6 for the duration of the before period, there were no significant differences between the sites ($F_{5,12} =2.119$, $P>0.05$). There was also insignificant variation amongst years ($F_{1,16} = 0.232$, $P>0.05$) (Figure 7.5).

≥1+ brown trout were present at all control sites during the before period and populations were more varied across the control sites compared to the impact sites. ≥1+ brown trout densities ranged from 0.39 to 29.76 fish/100 m^2 during the before period (Appendix, Table A.7) and EA FCS classifications ranging from poor to excellent. Similar to the impact sites, ≥1+ (< 20 cm) brown trout dominated the ≥1+ age class (Figure 7.5), with densities ranging from 0.39 to 29.17 fish/100 m^2 (Table 7.5). This was significantly higher ($P < 0.05$) than the ≥1+ (> 20cm) brown trout densities, with e this age/size group absent from sites WR5 and WR6 in 2012, WR6 and WR7 in 2013, WR2 – WR4 in 2014 and WR8 throughout the before period. ≥1+ (> 20cm) brown trout densities at the remaining sites ranged from 0.36 to 2.99 fish/100 m^2 (Table 7.6). The three different rivers that comprised the control sites supported significantly different ≥1+ brown trout densities ($F_{2,18} = 6.242$, $P < 0.05$) to each other, but there were no significant differences between sites nested within rivers ($F_{6,18} = 2.57$, $P>0.05$). There was, however, no significant difference between control sites temporally ($F_{1,25} = 0.211$, $P > 0.05$). This suggests that, whilst each river held significantly different ≥1+ (< 20 cm) brown trout densities, there was no significant difference between years in the three rivers, i.e. the change in densities of ≥1+ brown trout in Barden Beck in 2012 to 2013 was not significantly different to the change seen in densities of ≥1+ brown trout in Ashfold Side Beck in 2012 to 2013. There was little variability within the ≥1+ (> 20 cm) brown trout densities at control sites with no significant difference between densities across years ($F_{1,25} = 0.646$, $P >0.05$) or between the different rivers ($F_{2,24} = 0.059$, $P > 0.05$). Prior to the commencement of the restoration work in 2015, a resource calculation was constructed to determine if the intensity of sampling would be adequate to detect a biologically meaningful change. At impact sites for both size classes of ≥1+ brown trout, densities were very low therefore a ≥50% decrease in densities of these size classes would not be detected within the timeframe of this study. Only a ≥50% increase in ≥1+ (<20 cm) brown trout densities would be detected within the timeframe of this study if the currently observed levels of temporal and spatial variance persisted following the restoration work (Table 7.7).

Post-restoration work (After)

Following the restoration work in summer 2015, observed densities of ≥1+ brown trout were low or zero at impact sites, with ≥1+ brown trout absent at site SW1 in 2015 and 2016, and site SW3 in 2015. At the remaining sites ≥1+ brown trout were found in densities ranging from 0.25 to 3.11 fish/100 m^2, with these populations classed as poor or fair/poor (Appendix, Table A.7). ≥1+ (<20 cm) brown trout were found in marginally higher densities than larger ≥1+ (>20 cm) brown trout but were absent from sites SW1 – SW3 in 2015 and site SW1 in 2016. Where caught, ≥1+ (<20 cm) brown trout densities ranged from 0.25 to 2.82 fish/100 m^2 (Table 7.5). ≥1+ (<20 cm) brown trout were caught at more sites in 2016 than 2015, but in lower densities (Table 7.5). Due to the low densities persisting across the impact sites there were no significant differences between either sites ($F_{5,6}$ = 1.964, P>0.05), or years ($F_{1,10}$ = 1.268, P>0.05) for ≥1+ (<20 cm) brown trout (Figure 7.5). ≥1+ (>20 cm) brown trout were absent from a number of impact sites following the restoration work (sites SW1, SW3, and SW6 in 2015, and SW1 – SW3 and SW6 in 2016), and where captured, densities ranged from 0.25 to 1.05 fish/100 m^2 (Table 7.6). There was a marginal decrease in average density estimates from 2015 to 2016, but this decline was not significant (P>0.05) (Figure 7.5). There were significant differences amongst the impact sites ($F_{5,6}$ = 4.439, P<0.05) as ≥1+ (>20 cm) brown trout were consistently absent from sites SW1, SW3 and SW6 during the after period (Table 7.6).

≥1+ brown trout densities were again more varied across the control sites compared to impact sites during the after period, with ≥1+ brown trout densities ranging from 0.48 to 41.71 fish/100 m^2. Population classifications during this period were either poor, average, good or excellent (appendix, Table A.7). ≥1+ (< 20 cm) brown trout again dominated catches with densities of this age class at control sites significantly larger than the ≥1+ (> 20 cm) brown trout (Table 7.5). Like observations at the impact sites, there were non-significant marginal decreases in average ≥1+ (< 20 cm) brown trout densities at the control sites from 2015 to 2016 (Figure 7.5). There was a significant difference between ≥1+ (< 20 cm) brown trout densities at the different control sites ($F_{8,9}$ = 6.510, P < 0.05), but there was no significant difference between average density estimates for each river ($F_{2,15}$ = 0.240, P>0.05). This suggests that spatial variance was operating at a site level as opposed to river level. ≥1+ (> 20 cm) brown trout were absent from sites WR1-WR4 and WR8 in 2015 and sites WR2 – WR6 in 2016. At the remaining sites, densities ranged from 0.40 to 1.88 fish/100 m^2 (Table 7.6). There were no significant differences between rivers ($F_{2,1} = 0.933$, $P > 0.05$), or sites nested within rivers ($F_{14,1} = 0.526$, $P>0.05$). There were no significant differences ($F_{1,16} = 0.176$, $P > 0.05$) of ≥1+ brown trout densities of this size class between years (Figure 7.5).

Table 7.5. Density estimates ± 95% C.L. of ≥1+ (< 20 cm) brown trout at impact and control sites during both the before (2012 – 2014) and after (2015 – 2016) periods.

River Name	Site Identifier	2012	2013	2014	2015	2016
		Before			After	
impact Sites						
River Washburn	SW1	2.85 ± 0.4	0.53 ± 0.0	0.46 ± 0.0	0.00 ± 0.0	0.00 ± 0.0
River Washburn	SW2	1.42 ± 0.4	2.14 ± 0.3	1.50 ± 0.2	0.00 ± 0.0	0.25 ± 0.2
River Washburn	SW3	4.22 ± 0.3	1.28 ± 0.2	0.98 ± 0.0	0.00 ± 0.0	0.24 ± 0.0
River Washburn	SW4	1.10 ± 0.0	4.72 ± 0.5	4.62 ± 0.7	1.05 ± 0.5	0.76 ± 0.2
River Washburn	SW5	0.75 ± 0.1	5.92 ± 1.1	0.65 ± 0.2	2.85 ± 1.2	0.26 ± 0.0
River Washburn	SW6	1.26 ± 0.2	3.82 ± 0.4	5.91 ± 0.3	2.82 ± 1.3	1.26 ± 0.6
Control Sites						
Barden Beck	WR1	9.84 ± 1.7	9.65 ± 0.2	13.43 ± 0.7	12.8 ± 1.4	7.79 ± 0.6
Barden Beck	WR2	5.17 ± 0.1	5.80 ± 0.1	3.35 ± 0.3	5.14 ± 0.4	6.41 ± 0.0
Ings Beck	WR3	4.74 ± 0.6	9.06 ± 0.5	10.20 ± 0.0	7.45 ± 0.1	0.48 ± 0.0
Ings Beck	WR4	3.50 ± 0.4	3.96 ± 0.3	1.11 ± 0.3	1.72 ± 0.2	1.71 ± 0.0
Ashfold Side beck	WR5	14.02 ± 1.0	11.16 ± 2.1	22.16 ± 2.1	32.00 ± 0.7	22.99 ± 1.1
Ashfold Side beck	WR6	23.46 ± 2.0	7.28 ± 0.6	29.17 ± 0.2	41.71 ± 3.9	21.30 ± 1.8
Ashfold Side beck	WR7	17.69 ± 0.4	7.90 ± 0.4	21.12 ± 0.6	19.35 ± 0.9	18.42 ± 1.6
Ashfold Side beck	WR8	8.71 ± 1.7	9.44 ± 0.4	0.39 ± 0.0	9.32 ± 0.7	13.30 ± 0.9
Ashfold Side beck	WR9	11.59 ± 1.1	9.76 ± 0.0	11.61 ± 0.5	11.96 ± 0.3	9.09 ± 0.5

Table 7.6 Density estimates ± 95% C.L. of ≥1+ (>20 cm) brown trout at impact and control sites during both the before (2012 – 2014) and after (2015 – 2016) periods.

River Name	Site Identifier	2012	2013	2014	2015	2016
		Before			After	
impact Sites						
River Washburn	SW1	0.26 ± 0.0	0.26 ± 00	0.46 ± 0.3	0.00 ± 0.0	0.00 ± 0.0
River Washburn	SW2	0.95 ± 0.0	0.00 ± 0.0	0.90 ± 0.0	0.81 ± 0.2	0.00 ± 0.0
River Washburn	SW3	0 .00± 0.0	0.32 ± 0.0	0.00 ± 0.0	0.00 ± 0.0	0.00 ± 0.0
River Washburn	SW4	0.83 ± 0.4	0.59 ± 0.4	1.78 ± 0.3	1.05 ± 0.0	0.76 ± 0.4
River Washburn	SW5	0.25 ± 0.0	1.41 ± 0.2	0.00 ± 0.0	0.26 ± 0.0	0.53 ± 0.1
River Washburn	SW6	0.00 ± 0.0	0.00 ± 0.0	0.00 ± 0.0	0.00 ± 0.0	0.00 ± 0.0
Control Sites						
Barden Beck	WR1	0.98 ± 0.0	0.54 ± 0.3	1.34 ± 0.0	0.00 ± 0.0	0.65 ± 0.4
Barden Beck	WR2	0.43 ± 0.0	0.53 ± 0.0	0.00 ± 0.0	0.00 ± 0.0	0.00 ± 0.0
Ings Beck	WR3	0.86 ± 0.4	0.41 ± 0.0	0.00 ± 0.0	0.00 ± 0.0	0.00 ± 0.0
Ings Beck	WR4	1.00 ± 0.0	0.57 ± 0.0	0.00 ± 0.0	0.00 ± 0.0	0.00 ± 0.0
Ashfold Side beck	WR5	0.00 ± 0.0	0.40 ± 0.0	2.99 ± 0.0	0.50 ± 0.0	0.00 ± 0.0
Ashfold Side beck	WR6	0.00 ± 0.0	0.00 ± 0.0	0.60 ± 0.0	0.53 ± 0.0	0.00 ± 0.0
Ashfold Side beck	WR7	0.38 ± 0.0	0.00 ± 0.0	0.45 ± 0.0	0.40 ± 0.0	0.45 ± 0.0
Ashfold Side beck	WR8	0.00 ± 0.0	0.00 ± 0.0	0.00 ± 0.0	0.00 ± 0.0	0.40 ± 0.0
Ashfold Side beck	WR9	0.36 ± 0.0	1.56 ± 0.0	1.09 ± 0.0	1.00 ± 0.1	1.88 ± 0.8

Table 7.7. Actual Vx for *before* period and Target Vx to detect a ≥50% change to ≥1+ brown trout densities. **Bold and underlined** signifies that significant change to ≥1+ densities could be detected within the study parameters.

	Actual VX	Target Vx for ≥50% Increase	Target Vx for ≥50% Decrease
< 20 cm Before	0.074	**<u>1.76</u>**	0.019
> 20 cm Before	0.024	0.014	0.001

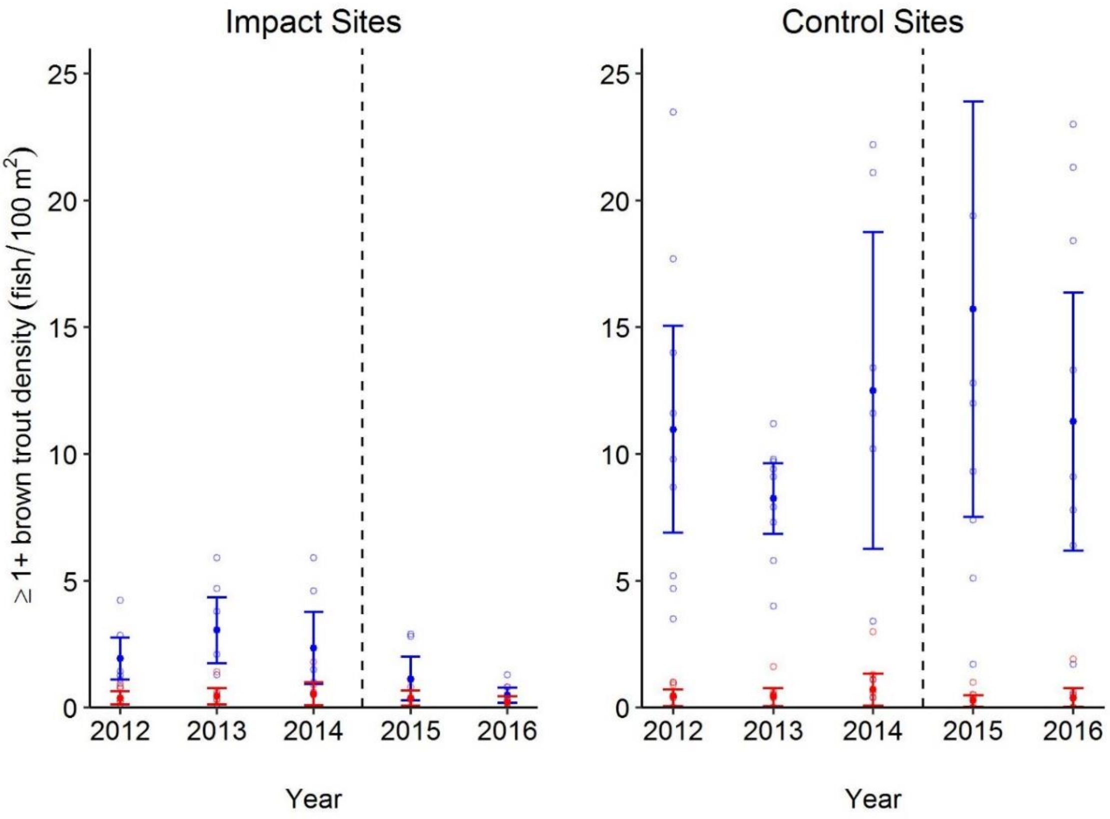

Figure 7.5. Average ≥1+ brown trout densities for all impact sites and control sites. ≥1+ (< 20 cm) data presented in blue, and ≥1+ (>20 cm) in red. The dashed line represents the division between before and after periods. Hollow circles represent individual data points.

Impact assessment (BACI)

Following the collection of the after-period data, a resource calculation was constructed to determine if any changes to the level of actual variance had occurred and whether a biologically relevant change could be detected within the timeframe of this study. For both size classes of ≥1+ brown trout actual variance increased, so therefore only a biologically meaningful increase in ≥1+ (<20 cm) brown trout could be detected within the timeframe of this study due to the high levels of variability within the impact and control populations

Table 7.8. Actual Vx for the combined before and after periods and Target Vx to detect a ≥50% change to ≥1+ brown trout densities. **Bold and underlined** signifies that significant change to ≥1+ densities could be detected within the study parameters

	Actual VX	Target Vx for ≥50% Increase	Target Vx for ≥50% Decrease
< 20 cm	0.086	**<u>1.76</u>**	0.019
> 20 cm	0.044	0.014	0.001

Densities of ≥1+ (<20 cm) brown trout were significantly lower than those in the control sites for the duration of this study (F = 27.86, P<0.05). Following the restoration work in 2015, ≥1+ (<20 cm) brown trout densities decreased in the impact sites, while marginally increasing in the control sites (Figure 7.6). Outputs from the BACI GLMM found that the interaction between area and period was significant. Therefore, the decline seen in ≥1+ (<20 cm) brown trout densities in the River Washburn could be attributed to the habitat restoration work in 2015. However, this result should be treated with caution as resource calculations identified that the actual variance of the ≥1+ (<20 cm) brown trout densities were too high to confidently isolate the change caused by the restoration works from the variability of the populations (Table 7.8). ≥1+ (>20 cm) brown trout densities increased marginally in the impact sites compared to a marginal decline at the control sites (Figure 7.7), unlike the smaller ≥1+ (<20cm) brown trout where there were no significant differences between the densities at impact and control sites. The BACI GLMM found that the interaction between area and period was not significant therefore the marginal increase in ≥1+ (>20 cm) brown trout densities in the River Washburn could not be attributed to the habitat restoration work in summer 2015.

Table 7.9 BACI GLMM to detect a change in ≥1+ brown trout density following the restoration work on the River Washburn in 2015. **Bold and underlined** signifies a significant (P<0.05) interaction in the GLMM

	Sum Sq	Mean Sq	DoF	DenDF	F	P
≥1+ (<20 cm)						
Area	7.112	7.112	1	13.185	27.862	**<u><0.001</u>**
Period	1.206	1.206	1	58	4.727	**<u>0.034</u>**
Area:Period	2.493	2.493	1	58	9.768	**<u>0.003</u>**
≥1+ (>20 cm)						
Area	0.005	0.005	1	13.314	0.065	0.803
Period	0.241	0.241	1	58	3.203	0.079
Area:Period	0.005	0.005	1	58	0.064	0.80

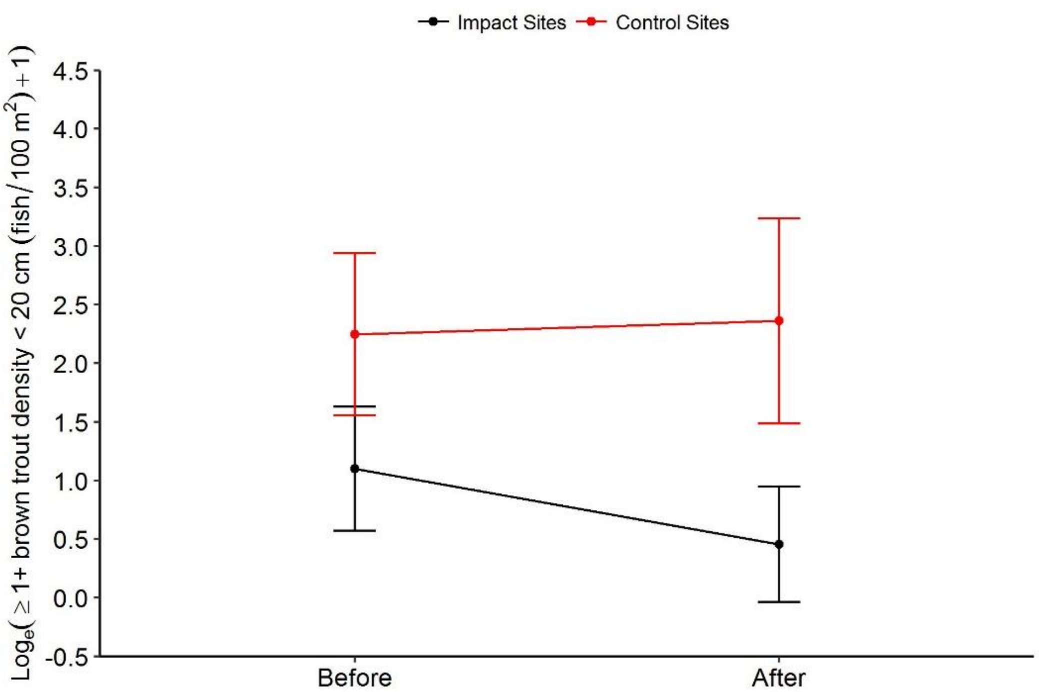

Figure 7.6. v ≥1+ (< 20 cm) brown trout densities for the before and after periods for impact (black) and control (red) sites. All densities are Log$_e$ +1 transformed. Trend lines represent significant interaction between Area and Period in the BACI GLMM

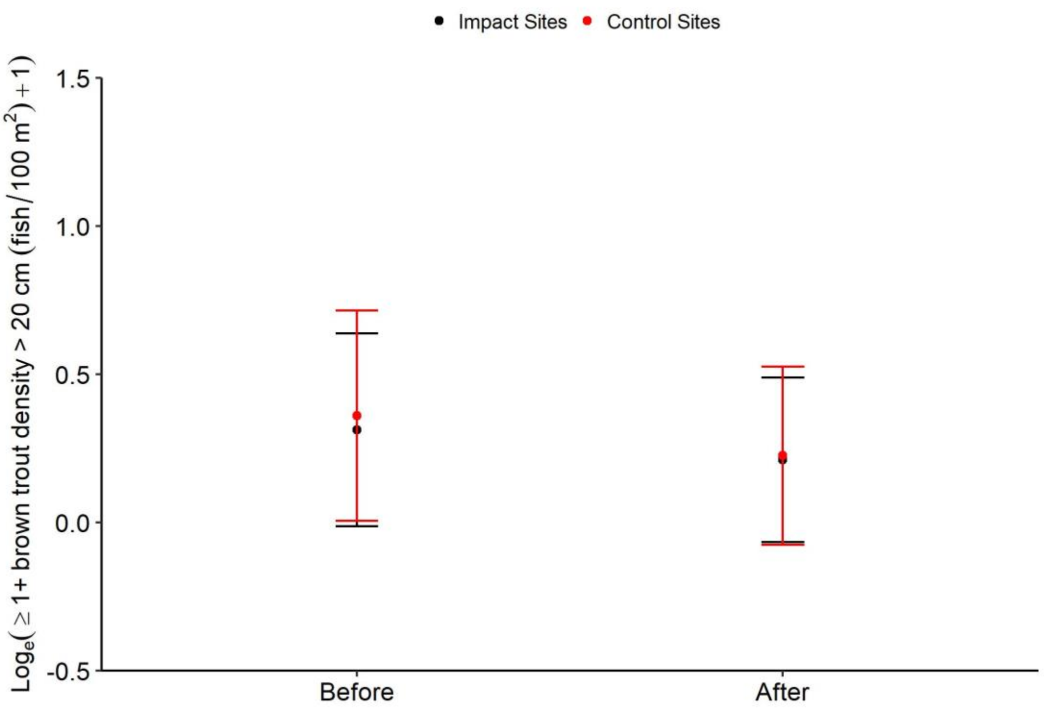

Figure 7.7. Mean (±95%) ≥1+ (> 20 cm) brown trout densities for the before and after periods for impact (black) and control (red) sites. All densities are Log$_e$ +1 transformed.

7.3.3 Impacts of the restoration work on the habitat quality (HQS density) for 0+ brown trout

Baseline monitoring (Before)

Habitat Quality Score (HQS) varied between sites and years in both impact and control sites during the before period. At impact sites, there was considerable variation between the years, with expected densities of 0+ brown trout, as indicated from the HQS, significantly higher ($P<0.05$) in 2014 than in 2012 (Figure 7.8). No HABSCORE data were available for site SW2 in 2012 but expected densities of 0+ brown trout at the other impact sites during the before period ranged from 3.48 to 14.3 fish/100 m^2 which suggest 0+ brown trout densities in 2012 and 2013 are fair/poor, but average in 2014 (Table 7.10). Observed densities of 0+ brown trout were significantly lower ($P<0.05$) than the HQS densities at sites SW1-SW3 and SW5 for the duration of the before period, and at site SW4 in 2014. At site SW4 in 2012 and 2013 and site SW6 in 2013 and 2014 observed 0+ brown trout densities were lower than the HQS densities but not significantly so ($P>0.05$). Due to the low 0+ brown trout HQS density estimate at site SW6 in 2012 observed densities were greater than expected but not significantly (Figure 7.9) ($P>0.05$).

At the control sites, 0+ brown trout habitats were typically better than those in the River Washburn as reflected by the higher HQS values. However, there were significant differences between the three control rivers ($F_{2,18} = 6.043$, $P < 0.05$). 0+ brown trout HQS densities ranging from 5.96 to 41.81 fish/100 m^2 (Table 7.10), but there were no significant differences between sites within rivers ($F_{6,18} = 0.596$, $P > 0.05$) suggesting that the higher HQS values recorded at each site are indicative of the river. Unlike in the impact sites, there was no significant year to year variability of the 0+ brown trout HQS ($F_{1,25} = 1.36$, $P > 0.05$). The range of the HQS values was greater in the control sites (Figure 7.8) with population classifications under non-impacted conditions ranging from fair/poor to excellent (Table 7.10). Similar to the impact sites, observed densities of 0+ brown trout were typically lower than the HQS at the control sites during the before period. Observed densities of 0+ brown trout were greater than the HQS at sites WR7 and WR8 in 2013 and WR3, WR5, WR6 and WR7 in 2014, but there were no instances where observed 0+ brown trout densities were significantly higher than predicted, although densities at sites WR1, WR4 WR6, WR7, WR8 and WR9 in 2012, WR1 and WR6 in 2013 and WR2, WR4 and WR8 in 2014 were all significantly lower than the HQS. Prior to the restoration work, a resource calculation was constructed to determine if the current level of sampling would be adequate to detect a biologically relevant change to 0+ brown trout HQS densities following the habitat restoration work. The actual variance was lower than the target variances; therefore, if the current level of spatial and temporal variance persists, a biologically meaningful change could be detected in 0+ brown trout HQS densities in the River Washburn (Table 7.11).

After habitat restoration work (After)

0+ brown trout HQS densities varied in the two years following the restoration works. In 2015, 0+ brown trout HQS densities were at their highest in the impact sites for the entire study, with the expected densities at the three sites within the restoration zone (SW2 – SW4) higher than previous years (Table 7.10). In 2015, the expected 0+ brown trout densities as indicated by HABSCORE ranged from 3.56 to 28.91 fish/100 m^2 and were classed as fair/poor to good, with the majority of the HQS densities classed as average (Table 7.10). However, 0+ brown trout HQS densities significantly declined in 2016 compared to 2015 with expected densities at sites ranging from 1.85 to 8.61 fish/100 m^2 which are classified as poor to average, with a majority of sites classed as fair/poor (Table 7.10). Due to the very low numbers of 0+ brown trout present at impact sites following the restoration works the observed densities of 0+ brown trout were significantly lower than predicted from the habitat at all sites for the duration of the after period (Figure 7.9).

0+ brown trout HQS values mostly remained similar to previous years with no significant changes between the two after years (2015 and 2016) ($F_{1,16}$ = 0.00, P >0.05). Expected 0+ brown trout populations were typically classed as average (Table 7.10). Habitat quality for 0+ brown trout was better in Barden beck with expected populations at sites WR1 and WR2 ranging from average to excellent (Table 7.10). As seen in the before period there were significant differences in the 0+ brown trout HQS values between the three control rivers ($F_{2.1}$ = 458.02, P < 0.05), but there were no significant differences between sites within rivers ($F_{14,1}$ = 55.55, P > 0.05) (Figure 7.8). During the after period the observed densities of 0+ brown trout were lower than expected from the HQS at all sites, except site WR3 in 2015. At the remaining sites only 0+ brown trout densities at sites WR2, WR4, WR6 and WR8 in 2015 were not significantly lower than predicted (Figure 7.9).

Table 7.10. 0+ brown trout HQS densities related to the EA-FCS categories for impact and control sites during the before (2012 - 2014) and after (2015 – 2016) periods. N/S denotes no HABSCORE data available.

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

River Name	Site Identifier	2012	2013	2014		2015	2016
		Before				After	
impact Sites							
River Washburn	SW1	4.75	5.91	14.3		13.03	1.85
River Washburn	SW2	N/S	3.59	7.65		10.96	5.88
River Washburn	SW3	4.12	7.79	14.50		28.91	6.18
River Washburn	SW4	5.16	6.01	10.28	0.00 ±	15.61	8.61
River Washburn	SW5	4.67	4.84	5.77		3.56	6.53
River Washburn	SW6	3.48	8.30	13.06		10.95	5.95
Control Sites							
Barden Beck	WR1	15.99	41.81	30.27		42.99	45.52
Barden Beck	WR2	11.12	32.06	14.49		24.72	10.26
Ings Beck	WR3	5.96	30.02	7.97		9.94	4.54
Ings Beck	WR4	11.20	14.91	16.12		15.58	13.78
Ashfold Side beck	WR5	8.67	6.16	12.45		11.30	11.51
Ashfold Side beck	WR6	10.75	9.23	12.87		11.19	17.39
Ashfold Side beck	WR7	11.27	10.62	14.37		8.75	15.98
Ashfold Side beck	WR8	7.82	15.07	13.17		10.79	11.96
Ashfold Side beck	WR9	12.18	14.74	15.35		9.94	13.41

Table 7.11. Actual Vx for the before period and Target Vx to detect a ≥50% change to 0+ HQS Densities. **Bold and underlined** signifies that significant change to 0+ HQS Densities could be detected within the study parameters

Actual Vx	Target Vx for ≥50% Increase	Target Vx for ≥50% Decrease
0.021	**<u>0.604</u>**	**<u>0.067</u>**

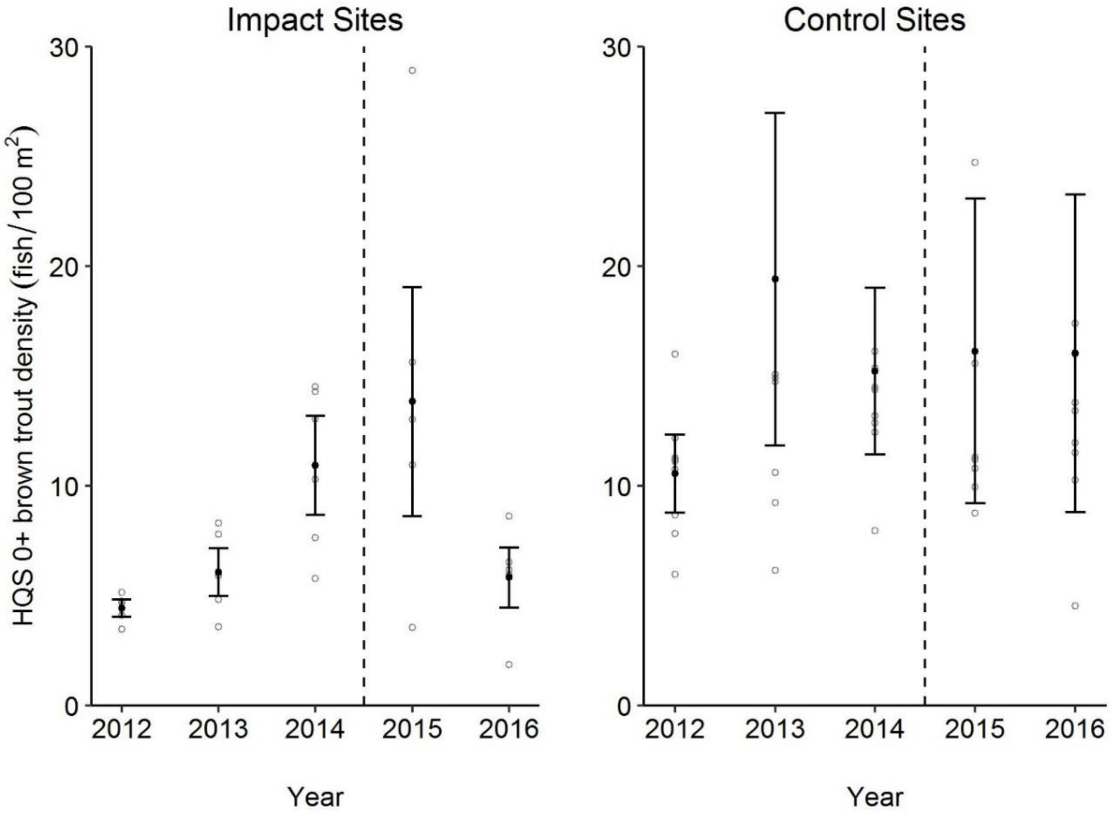

Figure 7.8. Average 0+ brown trout HQS densities ± 95 % C.L. for all impact and control sites, throughout the study period. The dashed line represents the division between *before* and *after* periods. Greyed circles represent individual density estimates

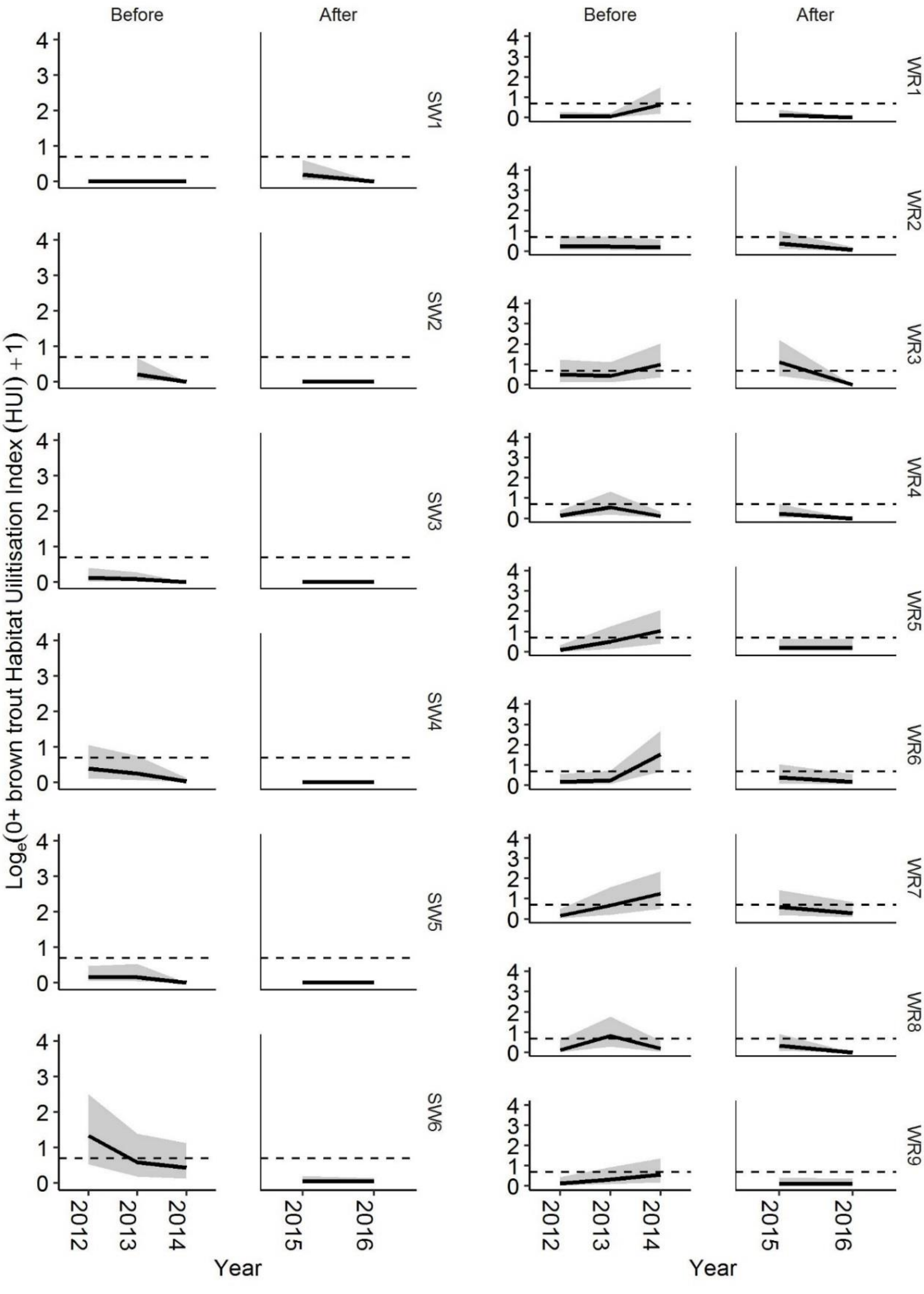

Figure 7.9. Loge + 1 transformed HUI for 0+ brown trout in the impact (left) and control sites (right) during the before (2012 – 2014) and after (2015 – 2016) period. Grey bands represent 95% confidence limits. The dashed line represents HUI value where observed and expected densities are equal.

Impact assessment

The addition of the 0+ brown trout HQS densities from the after period increased the level of actual variance in the dataset. However, the actual variance was still lower than the target variances. This confirms that a biologically meaningful change to the habitat quality for 0+ brown trout in the River Washburn could be isolated from the natural variance of the dataset (Table 7.12)

Table 7.12. Actual Vx for the full study and Target Vx to detect a ≥50% change to 0+ HQS Densities. **Bold and underlined** signifies that significant change to 0+ HQS Densities could be detected within the study parameters

Actual Vx	Target Vx for ≥50% Increase	Target Vx for ≥50% Decrease
0.046	**0.604**	**0.067**

The habitat quality for 0+ brown trout was significantly lower in the River Washburn compared to the control sites throughout the study (Figure 7.10). There was, however, a marginal increase in the 0+ brown trout habitat quality at sites in the River Washburn following the restoration work, but this change was not significant (P>0.05). The interaction of area and period in the BACI GLMM was not significant therefore the increase in 0+ habitat quality in the River Washburn could not be attributed to the restoration works that occurred in 2015 with any degree of confidence.

Table 7.13 BACI GLMM to detect a change in 0+ brown trout HQS density following the restoration work on the River Washburn in 2015

	Sum Sq	Mean Sq	DoF	DenDF	F	P
Area	2.55	2.55	1	13.605	15.228	**0.002**
Period	0.04	0.04	1	3.023	0.242	0.657
Area:Period	0.114	0.114	1	54.177	0.683	0.412

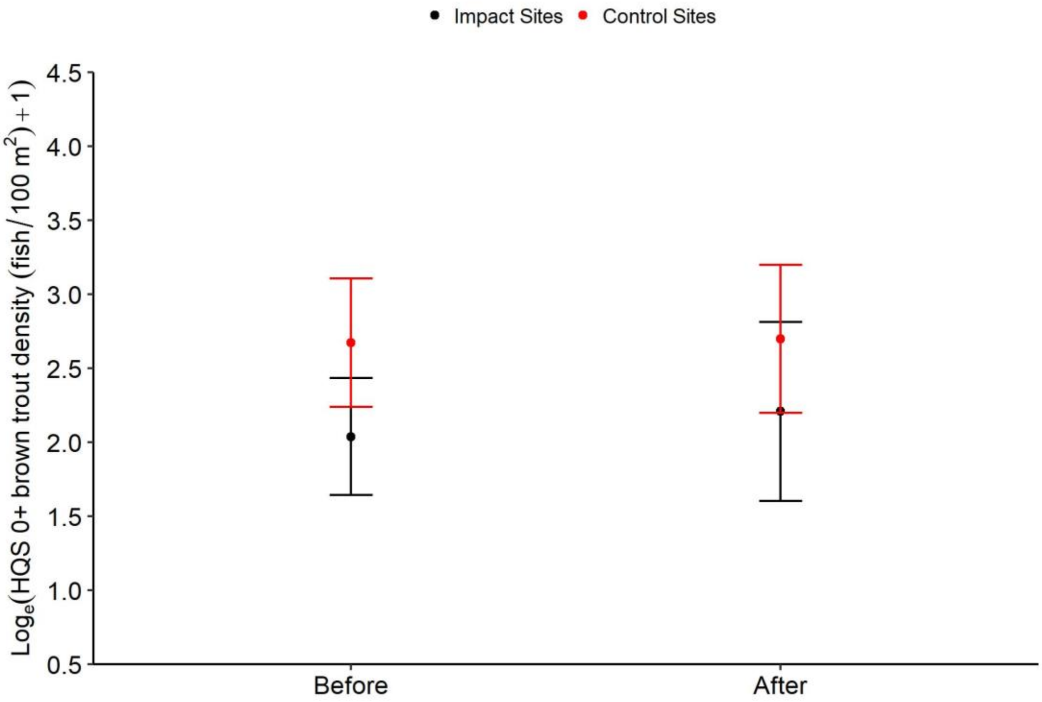

Figure 7.10. Mean (±95%) 0+ brown trout HQS densities during before and after periods for impact (black) and control (red) sites.

7.3.4 Impacts of the restoration work on the habitat quality (HABSCORE) for ≥ 1+ brown trout

Baseline monitoring (Before)

During the before period, HQS densities for ≥1+ brown trout varied between sites and years. At impact sites, the expected densities of ≥1+ (<20 cm) brown trout ranged from 0.2 to 3.72 fish/100 m^2 (Table 7.14). There was considerable variation between years with expected densities of ≥1+ (<20 cm) brown trout increasing significantly ($P<0.05$) for each of the three before years (2012-2014) (Figure 7.11), but there were no significant differences ($F_{5,11} = 0.064$, $P>0.05$) amongst the impact sites. The observed densities of ≥1+ (<20 cm) brown trout varied in relation with habitat quality, with densities at sites SW4 and SW6 higher than the expected for the duration of the before period. At sites, SW1 and SW3 in 2012 and at site SW5 in 2013 observed densities were higher than expected from the HQS, and in the case of site SW3 in 2012 significantly higher ($P>0.05$) (Figure 7.12). At the remaining sites and years observed densities were lower than predicted from the HABSCORE, with ≥1+ (<20 cm) brown trout densities at site SW1 in 2013 and 2014 significantly lower ($P<0.05$) than expected from the habitat quality (Figure 7.12). Habitat quality for ≥1+ (>20 cm) brown trout was typically lower at impact sites than the larger >20cm brown trout, but not significantly so ($P>0.05$) (Figure 7.11). ≥1+

(>20 cm) brown trout HQS densities improved annually from 2012 to 2014, but this increase was not significant ($P>0.05$) (Figure 7.11). During this period, the expected densities of ≥1+ (>20 cm) brown trout brown trout ranged from 0.34 to 3.59 fish/100 m^2 (Table 7.15) and did not differ significantly between the sites ($F_{5,11} = 1.353$, $P>0.05$). Observed densities of ≥1+ (>20 cm) brown trout were lower than expected from the HQS at all sites during the before period, except site SW4 in 2014 when densities were marginally higher than expected (Table 7.15). At site SW3 in 2012, SW1 and SW3 in 2013 and at SW3 and sW5 in 2014 as well as SW6 for the duration of the before period observed densities of ≥1+ (>20 cm) brown trout were significantly lower than the HQS densities (Table 7.15).

The available habitat for ≥1+ (< 20 cm) brown trout at control sites was typically better with expected densities ranging from 2.16 to 24.30 fish/100 m^2 during this period (Table 7.14). There was no significant difference in the ≥1+ (< 20 cm) brown trout HQS between the three study rivers ($F_{2,18} = 1.763$, $P> 0.05$) or between sites within the control rivers ($F_{6,18} = 1.924$, $P>0.05$). The observed densities of ≥1+ (< 20 cm) brown trout in the control sites were equal to or greater than the habitat quality at all control sites during the before period, except site WR2 in 2013 and 2014 and sites WR4 and WR8 in 2014 (Table 7.14). The habitat quality for ≥1+ (> 20 cm) brown trout revealed that expected densities of ≥1+ (> 20 cm) brown trout would be typically lower than the < 20 cm conspecifics. There was no significant difference between ≥1+ (> 20 cm) brown trout HQS at control sites during the before period ($F = 2.68$, $P>0.05$) with densities ranging from 0.74 to 10.16 (Table 7.15). There were significant differences between the rivers ($F_{2,18}=3.798$, $P<0.05$) and between sites within rivers ($F_{6,18}=4.090$, $P<0.05$) suggesting that habitat at individual sites were more suitable for ≥1+ (> 20 cm) brown trout as opposed to habitat across entire rivers. The HUI revealed that ≥1+ (> 20 cm) brown trout densities were lower than predicted at all sites except site WR4 in 2012 and WR5 in 2015 (Figure 7.14). Of these sites where ≥1+ (> 20 cm) brown trout were lower than predicted, all were significantly lower ($P>0.05$) except WR1 in all years, WR2 in 2012 and 2013, WR3 in 2012 (Figure 7.15). In no instances during the before period was the observed densities of ≥1+ (<20 cm) brown trout at the control sites significantly different from the HQS.

A resource calculation was constructed prior to the restoration work in 2015 to ensure that the intensity of sampling would be adequate to detect a biologically meaningful change to habitat quality of ≥1+ brown trout within the timeframe of this study. Due to the low HQS densities at impact sites for both size classes of ≥1+ brown trout, it would not be possible to isolate any decrease at impact sites caused by the restoration work from the natural variability of the populations. However, it will be possible to detect a biologically meaningful increase to ≥1+ brown trout habitat quality within the timeframe of this study (Table 7.16)

After habitat restoration work (After)

Following the restoration works in 2015, there was a marginal increase in habitat quality (as reflected by the HQS densities) for ≥1+ (<20 cm) brown trout across impact sites. However, the increase was significantly higher ($P<0.05$) than HQS densities in 2012, but not significantly higher ($P>0.05$) than HQS densities in 2013 and 2014 (Figure 7.11). Following the restoration work, expected ≥1+ (<20 cm) brown trout densities ranged from 0.75 to 7.88 fish/100 m^2 (Table 7.14). Unlike the before period, there were no significant differences between sites ($F_{5,6}$=1.685, $P>0.05$) or years ($F_{1,10}$=0.219, $P>0.05$) at impact sites (Figure 7.11). Observed densities of ≥1+ (<20 cm) brown trout were lower than expected from the habitat quality at all impact sites following the restoration work, except site SW5 and SW6 in 2015, where observed densities of ≥1+ (<20 cm) brown trout were marginally higher. Observed densities were significantly lower than HQS densities at sites SW1 – SW4 for the duration of the after period, and at site SW5 in 2016 (Figure 7.12). Expected densities of ≥1+ (>20 cm) brown trout indicated that there was improvement to the habitat quality for ≥1+ (>20 cm) brown trout at impact sites following the restoration works, but the HQS densities during the after period were only significantly higher ($P>0.05$) than those in 2012 (Figure 7.11). Expected densities of ≥1+ (>20 cm) brown trout during this period ranged from 0.56 to 4.69 fish/100 m^2 (Table 7.15). Observed densities of ≥1+ (>20 cm) brown trout, in comparison, were significantly lower ($P>0.05$) than the HQS densities at all sites for the duration of the after period, with the exception of SW2 and SW4 in 2015, and densities at site SW5 in 2016, where observed densities of ≥1+ (>20 cm) brown trout were still lower than expected, but not significantly (Figure 7.13).

There was a marginal decrease in ≥1+ (< 20 cm) brown trout habitat quality at control sites in 2014 compared to the previous years (2012-2014). During this period, there was significant variation in habitat quality for ≥1+ (< 20 cm) brown trout manifested either spatially ($F_{8,9}$ = 2.88, $P > 0.05$) or temporally ($F_{1,16}$ = 2.80, $P>0.05$). HUI revealed that observed densities of ≥1+ (< 20 cm) brown trout were greater than expected at sites WR2, WR5-WR9 during both after years and site WR3 in 2015 (Figure 7.12). Of these sites, the observed density of ≥1+ (< 20 cm) brown trout was significantly higher than predicted at sites WR5 – WR7 and WR9 (Figure 7.12). At the remaining sites (WR1 and WR4 as well as WR3 in 2016) observed densities of ≥1+ (< 20 cm) brown trout were lower than predicted and in the case of site WR3 in 2016 significantly lower ($P<0.05$) (Figure 7.12). ≥1+ (> 20 cm) brown trout densities were typically lower at control sites during the after period, but not significantly so (Figure 7.13). The habitat quality for ≥1+ (> 20 cm) brown trout did not vary significantly between years (F = 1.46, P >0.05). There was a significant difference between sites ($F_{8,9}$ = 6.930, P >0.05), but as there was no significant difference between the three control rivers ($F_{2,15}$ = 0.508, $P>0.05$). It can be assumed that the difference between sites was manifested in individual site variability as opposed to all sites in one of the three rivers having better ≥1+ (> 20 cm) brown trout habitat quality. The HUI revealed that observed ≥1+ (> 20 cm) brown trout densities at all sites were significantly lower ($P<0.05$) than predicted from the habitat quality except site WR9, where observed densities were marginally greater than predicted in both years (Figure 7.13.

Table 7.14 HQS densities for ≥1+ (<20cm) brown trout at all impact and control sites for the before (2012-2014) and after (2015-2016)

River Name	Site Identifier	2012	2013	2014	2015	2016
		Before			After	
impact Sites						
River Washburn	SW1	0.69	1.88	3.80	3.09	0.75
River Washburn	SW2	N/S	1.32	2.84	3.58	2.40
River Washburn	SW3	0.37	2.21	3.72	6.50	2.03
River Washburn	SW4	1.04	2.51	3.11	4.21	7.88
River Washburn	SW5	1.39	1.58	2.09	1.07	2.13
River Washburn	SW6	0.20	2.70	2.85	2.30	1.98
Control Sites						
Barden Beck	WR1	4.95	24.3	12.75	13.85	22.76
Barden Beck	WR2	2.22	6.70	4.81	3.62	3.34
Ings Beck	WR3	3.36	11.76	3.61	4.00	1.61
Ings Beck	WR4	2.16	4.80	3.50	2.89	3.14
Ashfold Side beck	WR5	8.34	5.35	9.00	4.75	13.23
Ashfold Side beck	WR6	14.12	5.91	8.71	4.65	13.35
Ashfold Side beck	WR7	8.74	5.09	5.02	2.65	9.08
Ashfold Side beck	WR8	3.18	5.83	4.40	2.20	6.81
Ashfold Side beck	WR9	3.42	6.78	3.75	2.08	5.75

Table 7.15. HQS densities for ≥1+ (>20cm) brown trout at all impact and control sites for the before (2012-2014) and after (2015-2016)

River Name	Site Identifier	2012	2013	2014	2015	2016
		Before			After	
impact Sites						
River Washburn	SW1	0.34	1.03	0.98	1.20	1.02
River Washburn	SW2	N/S	3.59	0.84	1.23	1.38
River Washburn	SW3	0.30	0.87	1.42	3.41	1.53
River Washburn	SW4	0.87	1.65	1.53	2.93	4.69
River Washburn	SW5	0.49	1.34	0.74	0.75	1.25
River Washburn	SW6	0.36	0.69	1.29	0.93	0.56
Control Sites						
Barden Beck	WR1	1.47	1.75	1.54	1.13	1.62
Barden Beck	WR2	0.95	0.74	0.84	0.47	0.5
Ings Beck	WR3	1.37	1.46	1.81	1.39	1.3
Ings Beck	WR4	0.82	1.83	1.60	0.99	2.03
Ashfold Side beck	WR5	6.32	3.89	1.55	1.96	2.86
Ashfold Side beck	WR6	10.16	6.10	2.74	3.13	3.18
Ashfold Side beck	WR7	3.19	1.86	1.21	1.27	1.79
Ashfold Side beck	WR8	1.26	0.92	0.84	0.73	1.19
Ashfold Side beck	WR9	1.96	1.58	1.04	0.72	1.53

Table 7.16 Actual Vx for *before* period and Target Vx to detect a ≥50% change to ≥1+ brown trout Habitat Quality (HQS). **<u>Bold and underlined</u>** signifies that significant change to ≥1+ densities could be detected within the study parameters.

	Actual VX	Target Vx for ≥50% Increase	Target Vx for ≥50% Decrease
< 20 cm Before	0.028	**<u>0.155</u>**	0.017
> 20 cm Before	0.017	**<u>0.068</u>**	0.008

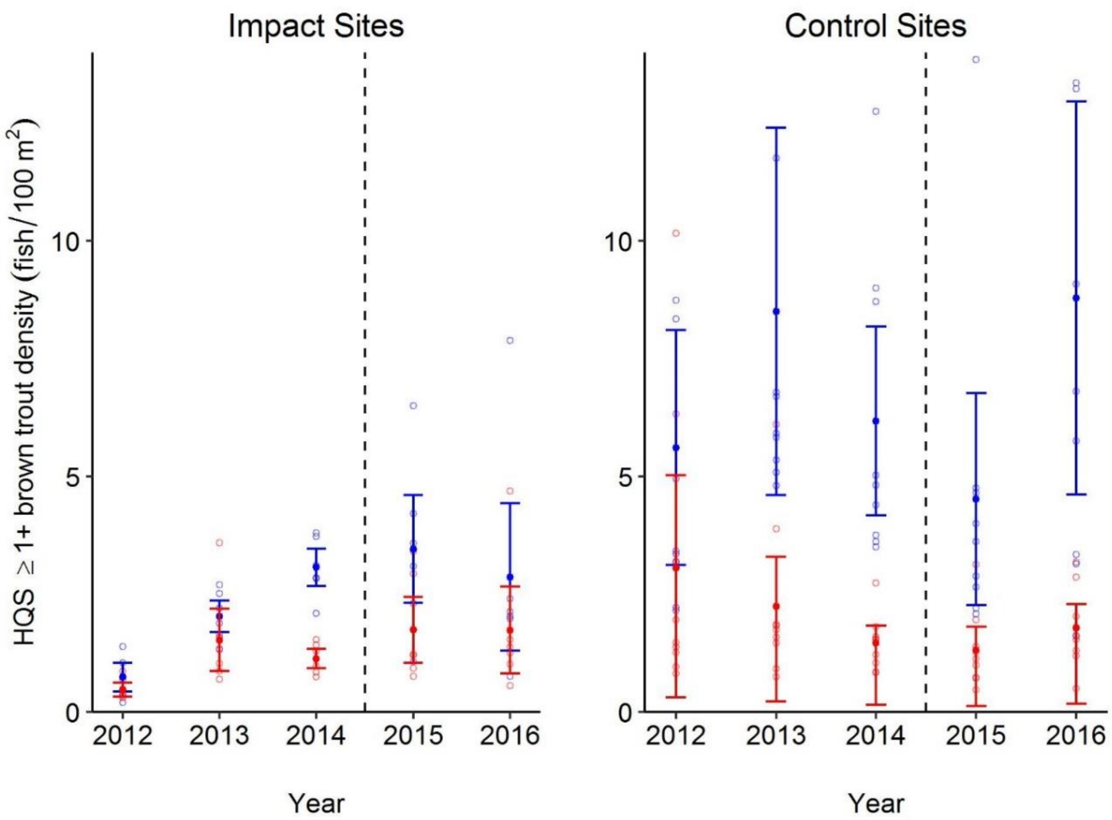

Figure 7.11. The relationship between average HQS densities scores ≥1+ <20 cm (blue) and ≥1+ >20 cm (red) for impact and control sites, error bars represent 95% confidence intervals, dashed line represents introduction of the compensation flow.

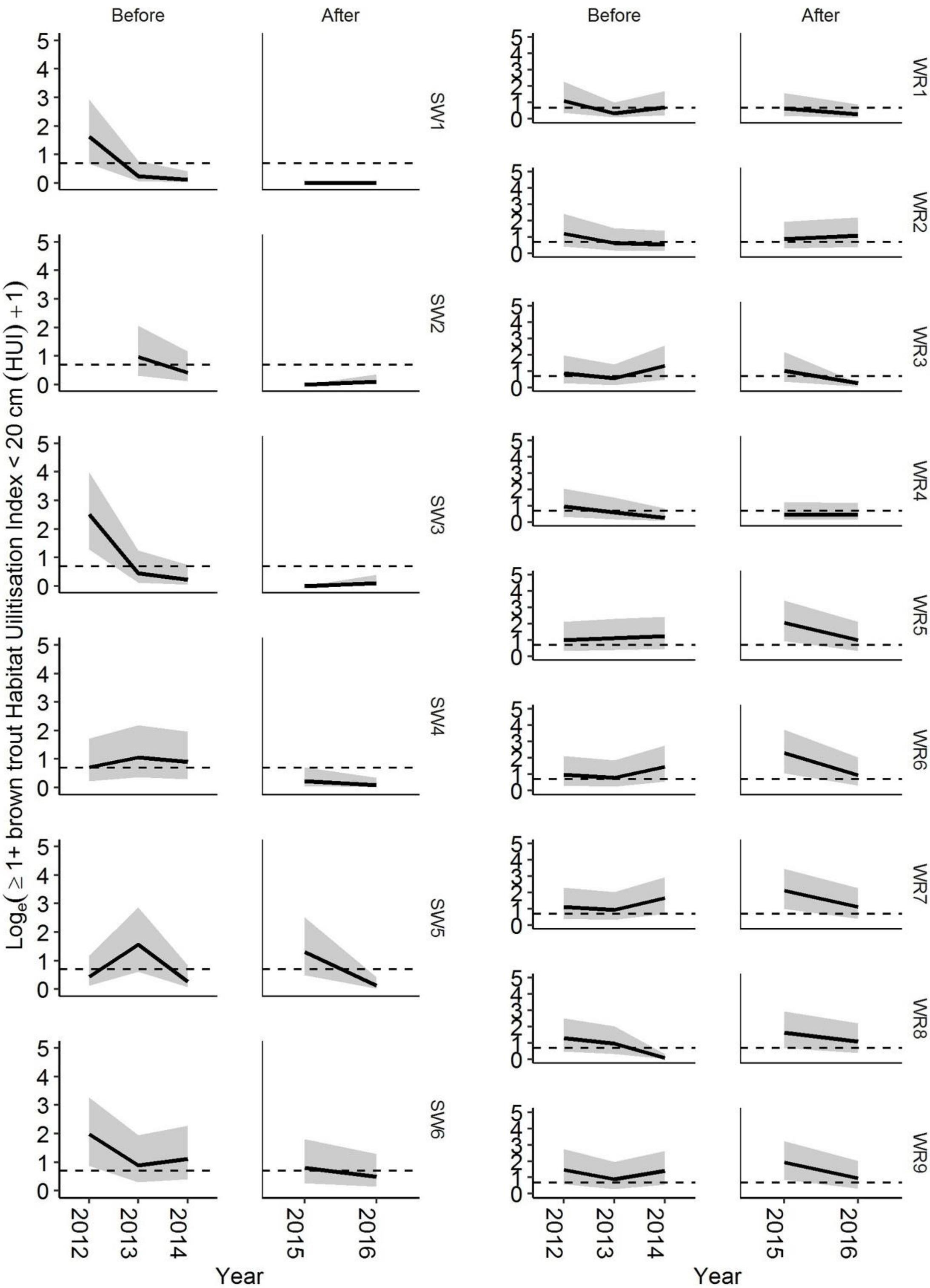

Figure 7.12 Loge + 1 transformed HUI for ≥1+ (< 20 cm) brown trout in the impact (left) and control sites (right) during the before (2012 – 2014) and after (2015 – 2016) period. Grey bands represent 95% confidence limits. The dashed line represents HUI value where observed and expected densities are equal.

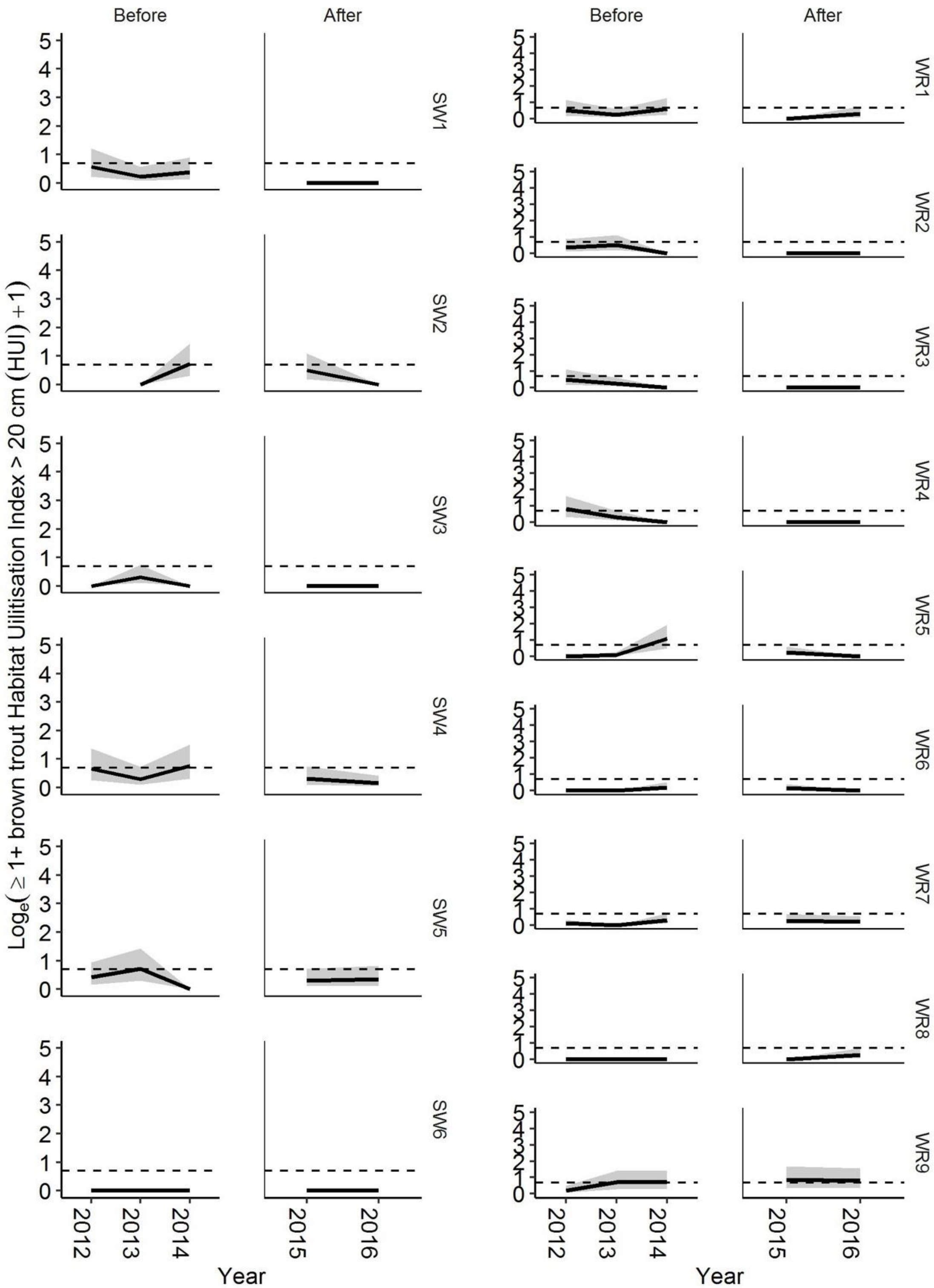

Figure 7.13. Loge + 1 transformed HUI for ≥1+(> 20 cm) brown trout in the impact (left) and control sites (right) during the before (2012 – 2014) and after (2015 – 2016) period. Grey bands represent 95% confidence limits. The dashed line represents HUI value where observed and expected densities are equal.

Impact assessment (BACI)

Following the addition of the data collected in 2014 and 2015 the actual variance in the habitat quality for ≥1+ (<20 cm) and ≥1+ (>20 cm) brown trout increased from the levels seen in just the before period. Therefore, only a biologically meaningful increase of the habitat quality for ≥1+ (<20 cm) brown trout would be able to sufficiently disentangle from the level of natural variance (Table 7.17).

Table 7.17 Actual Vx for *before* and after period and Target Vx to detect a ≥50% change to ≥1+ brown trout Habitat Quality (HQS). **Bold and underlined** signifies that significant change to ≥1+ densities could be detected within the study parameters.

	Actual VX	Target Vx for ≥50% Increase	Target Vx for ≥50% Decrease
< 20 cm Full	0.039	**<u>0.155</u>**	0.017
> 20 cm Full	0.073	0.068	0.008

Habitat quality (as indicated by HQS densities) increased for both <20 cm (Figure 7.14) and >20 cm (Figure 7.15) ≥1+ brown trout following the restoration works on the River Washburn. During the same time periods, the habitat quality for both size classes of brown trout decreased at control sites. The BACI GLMM revealed that for both size classes the interaction between area and period was significant ($P<0.05$) for <20cm ($F = 4.21$, $P<0.05$) and >20 cm ($F=9.77$, $P<0.05$). For ≥1+ (<20 cm) brown trout, it can be confidently asserted that the increase in habitat quality in the River Washburn for this size class was because of the habitat restoration works. However, due to the level of high variance within the HQS densities for ≥1+ (> 20 cm) brown trout the significant interaction between area and period in the BACI GLMM should be regarded with caution, as it was identified in the resource calculation that any biologically meaningful change could not be isolated from the levels of background natural variation.

Table 7.18 BACI GLMM to detect a change in ≥1+ brown trout HQS density following the restoration work on the River Washburn in 2015

	Sum Sq	Mean Sq	DoF	DenDF	F	P
≥1+ (<20 cm)						
Area	2.373	2.373	1	13.407	15.042	**<u>0.002</u>**
Period	0.04	0.04	1	3.014	0.256	0.648
Area:Period	0.664	0.664	1	54.199	4.21	**<u>0.045</u>**
≥1+ (>20 cm)						
Area	0.067	0.067	1	13.209	0.927	0.353
Period	0.018	0.018	1	3.256	0.253	0.647
Area:Period	0.702	0.702	1	54.258	9.771	**<u>0.003</u>**

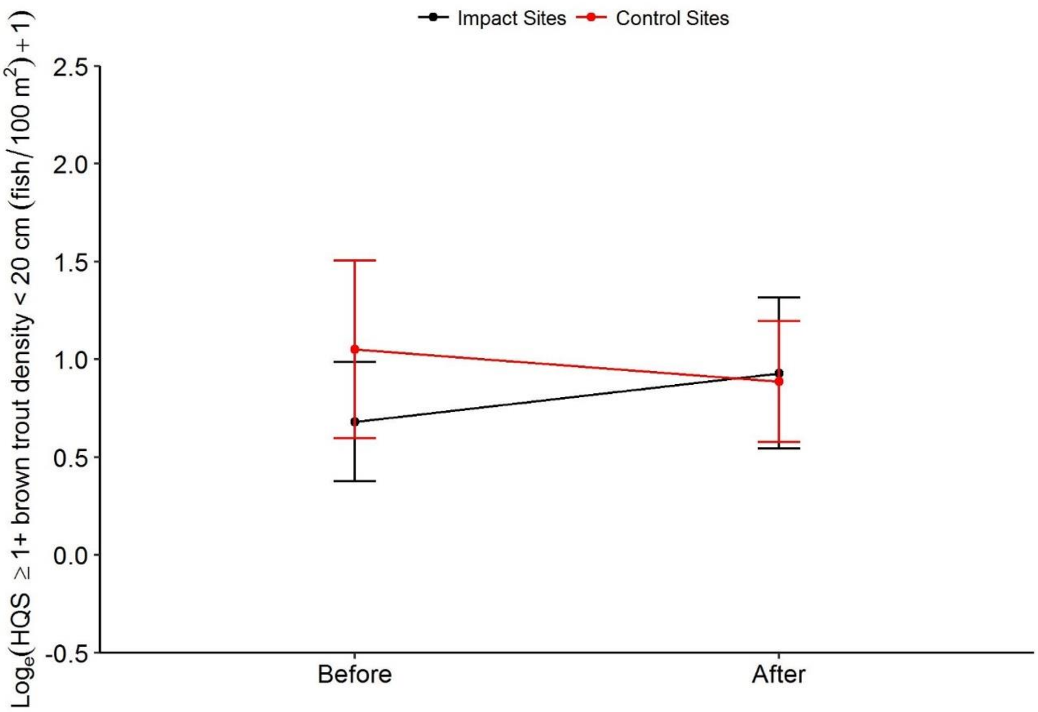

Figure 7.14. Mean (±95%) ≥1+ (<20 cm) HQS values during *before* and *after* periods for impact (black) and control (red) sites. Trend Lines represent significant interaction between Area and Period on the BACI GLMM.

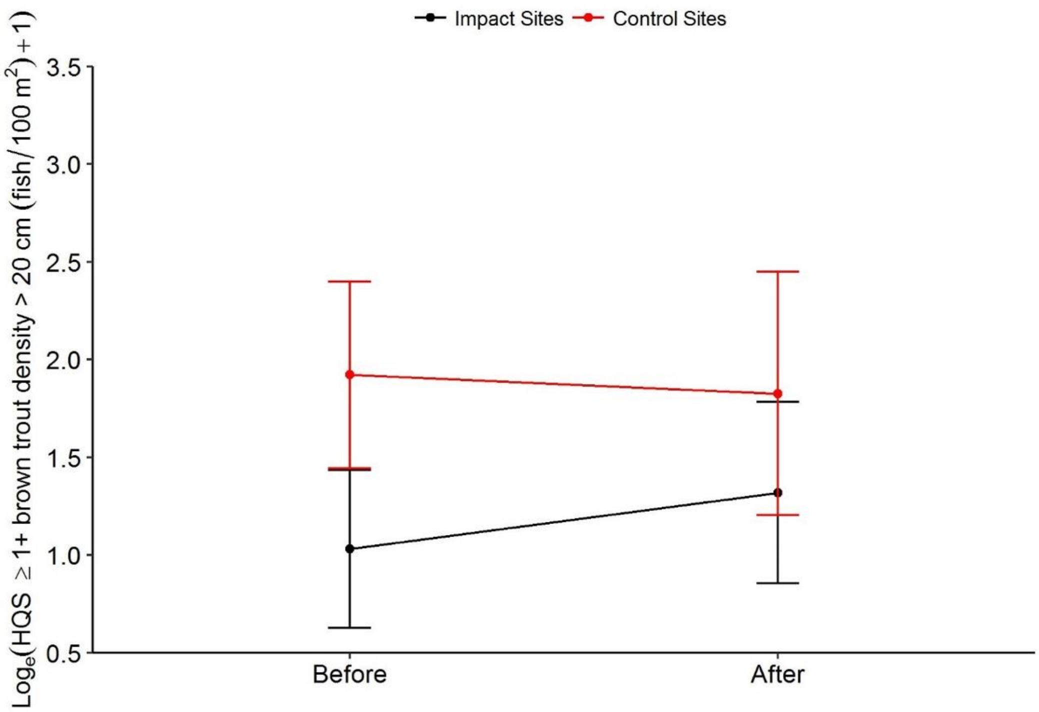

Figure 7.15. Mean (±95%) ≥1+ (<20 cm) HQS values during *before* and *after* periods for impact (black) and control (red) sites. Trendlines represent significant interaction between Area and Period in BACI GLMM.

7.4 Discussion

Analysis of the baseline data (2012 – 2014) revealed that densities of 0+ and ≥1+ brown trout were poor in the River Washburn due to the poor habitat and lack of compensation flow, reaffirming the need for restoration works. ≥1+ brown trout were found in higher densities than 0+ conspecifics across the study sites, with 0+ either in lower densities or absent, suggesting poor recruitment within the river. This poor recruitment is likely a combination of poor densities of spawning stock as well as poor habitat conditions. Little variation was found amongst brown trout densities for the 0+ and ≥1+ age classes during the baseline period. In contrast to the control sites, and across other studies in Yorkshire, 2012 did not stand out as a poor year for recruitment. This is despite the exceptionally high rainfall throughout the summer period (Parry *et al.* 2013), which is believed to have been the cause for low 0+ brown trout densities across numerous study sites throughout Yorkshire (Chapter 4 and Chapter 5). It is probable that the operations of Thruscross, Fewston and Swinsty reservoir led to reduced flow elevations in the Washburn as the resulting flow elevations from the high rainfall would of be "captured" in these three upstream impounding reservoirs.

Site selection for spawning brown trout is strongly governed by the available substrate with spawning females showing a strong preference for substrate particle diameters around 10% of their total length (Kondolf and Wolman 1993, Jonsson and Jonsson 2011). Limited habitat, such as low prevalence of suitable spawning gravels and dominance of silt in the marginal nursery grounds, were believed to be a key contributor for the low brown trout densities in the River Washburn in 2012-2014. HABSCORE data showed that boulders and cobles were the predominant substrate in the River Washburn, with large levels of siltation in the marginal areas. This was reflected in the poor HQS densities seen at some impact sites, but despite this, observed densities of 0+ brown trout were lower than predicted at sites SW1 to SW5, and in many cases significantly so. It is likely that other abiotic elements outside the scope for detection within the HABSCORE analysis, such as water chemistry (Enge *et al.* 2017), macroinvertebrate densities and prey availability (Jonsson and Jonsson 2011), or water temperature (Coleman and Fausch 2007) or lack of suitable spawning stock are responsible for poor brown trout densities.

≥1+ (<20 cm) brown trout were present at all impact sites before the restoration works, again in low densities, and habitat degradation resulting from low flows from Swinsty Reservoir are likely to play a key role in the poor densities. However, in many cases across the impact sites observed densities were greater than predicted HQS densities, with two instances in SW3 and SW6 in 2012 where observed densities of this age class were significantly higher than predicted from the habitat. This suggests that the poor densities were in line with expectation from the habitat within the reach. Densities of ≥1+ (>20 cm) brown trout were absent from a number of impact sites before the restoration work, and where captured were only captured in low densities. These low densities reflect a lack of suitable habitat, i.e. lack of deep pools (Cowx *et al.* 2004) that large brown trout favour, but like the 0+ brown trout densities the degree of habitat utilization was low for ≥1+ (>20 cm) brown trout, with observed densities lower than predicted from the habitat and significantly lower in at sever sites. This suggests the poor habitat, as well as habitat variables outside the scope for detected within the HABSCORE analysis, were influencing ≥1+ (>20 cm) brown trout densities.

There was greater variability in brown trout densities and habitat quality amongst the nine control sites used in this study, with strong temporal and spatial variability between the biotic and abiotic metrics used in this study. Densities and habitat quality were poor in 2012, particular so for 0+ brown trout, and is likely that the heavy rainfall in 2012 (Parry *et al.* 2013) was responsible for washout and discharge related mortality of 0+ brown trout (Saltveit *et al.* 1995, Daufresne *et al.* 2005, Unfer *et al.* 2011). The increase in 0+ brown trout densities at control sites from 2012 – 2014 suggest that the regulation of the River Washburn by Swinsty reservoir is a major bottleneck to natural recruitment variability as seen in other regulated rivers (chapter 3).

In-stream restoration measures have been shown to be very effective in improving both habitats for, and densities of, salmonid populations (Whiteway *et al.* 2010). As previously established, depth and water velocity are considered to be major limitation to recruitment of brown trout in the River Washburn, as well as the dearth of clean and suitable spawning gravels. In many areas, the wetted width exceeded 10 m in the River Washburn, yet average depths at these sites was around 30 cm. While these depths are within the tolerances of brown trout at all life stages, they are well below preferred ranges for brown trout (Cowx *et al.* 2004). The possible reduction of the width of these over-widened segments of the river aims to provide deeper and faster flowing water more suited to brown trout populations. To ensure that suitable nursery grounds for 0+ brown trout are available, the additions of groynes throughout the restoration zone allow for shallower and slower moving water better suited for 0+ brown trout, as well as providing cover and shelter from higher flows occurring from heavy rainfall or reservoir release (Frissell and Nawa 1992, Forseth and Harby 2014).

Studies in Sweden demonstrated a strong link between recruitment success and abundance of suitable spawning grounds (Palm and Lepori 2007). Analysis of HABSCORE data revealed that not only were spawning gravels in low abundance, they were also in poor condition (i.e. accumulation of silts and sands amongst them). As there is little opportunity for replenishment of the gravels from upstream (due to reservoir operation), the addition of suitable spawning gravel throughout the restoration reach provided improved spawning habitat for brown trout. Also, an increase in water velocities resulting from the narrowing of the river channel should provide suitable flows to ensure sufficient interstitial flow to reduce the accumulation of sand and silts

Fisheries surveys were undertaken one week after the completion of the restoration work in 2015, to assess the remaining stocks of fish after the habitat modification. In 2015, 0+ brown trout were absent from four of the six study sites (SW2 – SW5), and low densities of this age class were found at site SW1 and SW6; both sites that were outside of the restoration zone. The physical alteration of 1-km of habitat in the River Washburn was a considerable undertaking and would likely have had consequences on the downstream habitat. For instance, in the restoration reach the placement of large boulders and wooden groynes used a Spider excavator (which weighs more than 12,000 kg), and this was driven through the river channel. The level of sediment disturbance that occurred through deployment of this heavy industrial lifting and moving equipment in the river channel was high. Increased levels of silt can have lethal effects on brown trout (Servizi and Martens 1992, Bash and Berman 2001b), with juveniles more susceptible than larger fish (Lloyd 1987, Bash and Berman 2001a). There was, however, no evidence to suggest that the disturbance during the restoration works had lethal effects on brown trout (i.e. no carcass observed in the study reach or downstream). It is possible that the brown trout resident both within and downstream of the restoration zone undertook downstream migrations in response to the disturbance and deteriorating water quality. Densities of 0+ and ≥1+ brown trout remained very low in 2016, one year after the restoration works, suggesting there has been poor utilisation of the reach following the loss of individuals from the previous years. In similarly restored rivers in America, recolonisation process of native fishes (including the brown trout) to levels reflective of natural conditions took less than one year (Moerke and Lamberti 2003), but this was not observed in the River Washburn. This may be because the study section of the River Washburn is bookended by two large reservoirs (Swinsty upstream and Lindley Wood Downstream), as significant barriers to migration, and these reservoirs pose major obstacles to the potential recolonisation of brown trout into the River Washburn. While the restoration work was undertaken on 1 km of river, the deleterious impacts that flow regulation has on the habitat is likely present throughout the entire 3-km reach of the River Washburn. While no fisheries surveys were undertaken in the 2-km stretch downstream of the study reach, it is plausible that poor habitat quality and densities persist throughout the River Washburn. Therefore, any recolonisation of brown trout within the study area would be from a relatively limited fisheries stock. These populations further declined in 2016, with this age class absent from five of the six survey sites (SW1 – SW5), and only very low numbers of 0+ brown trout present at site SW6. This indicates that some brown trout spawning occurred within the reach, albeit low. ≥1+ brown trout were only absent from one site in 2016 and were present at two sites (SW2 and SW3) where they were absent from in previous years. At the remaining sites (SW4 – SW6) they were found in lower densities than in 2015. Declines in the fish populations following physical habitat restoration is not a rare occurrence, indeed Whiteway *et al.* (2010) reported that 27% of restoration activities resulted in lower salmonid densities. Poor study design and reference site selection, as well as unexpected physical changes to the habitat, were amongst the most citied reasons for the poor biological response to the restoration work. HABSCORE data in 2015 revealed that habitat quality for both 0+ and ≥1+ brown trout was better than in previous years, with habitat quality at site SW3 significantly better for 0+ brown trout, suggesting the instream works had the desired effects of improving on the physical river habitat, as indicated by improved habitat quality for all age classes within the restoration reach. However, the process by which these goals were achieved must be understood. It is highly likely that the decreased densities of all age classes of brown trout in the River Washburn, particularly in 2015, resulted from the disruption to the ecosystem caused by the instream restoration works themselves.

0+ brown trout habitat quality significantly declined in 2016 to levels similar to those seen before the restoration work (2012-2014); an unexpected scenario. Unfortunately, exceptionally high levels of rainfall were seen in winter 2015/16 throughout Yorkshire, with rainfall during this period >175% of the average for this region. This resulted in large floods with a return period of over 100 years and these coincided with this restoration study. Observations of the restoration reach following these high winter floods highlighted serious degradation of the restoration work. Some wooden groynes as well as boulders had been displaced and washed downstream. Areas where the river had been re-channelized had been washed out increasing the wetted width throughout the study, and suitable spawning gravels had been washed-out due to high flow velocities (Unfer *et al.* 2011). It is possible this flood event was linked to the poor recruitment of brown trout in 2016, as washout of spawning gravels caused mortality of incubating eggs (Jensen and Johnsen 1999, Nicola *et al.* 2009). This hypothesis was supported by poor fisheries recruitment across rivers in Yorkshire in 2015/16 (Chapters 4, 5, 6). However, it is possible that brown trout recruitment may still have been poor in 2016, due to the low numbers of ≥1+ brown trout, particularly of larger more fecund >20 cm brown trout (Milner *et al.* 2003). Therefore, the issue of low numbers of mature brown trout for spawning was compounded by the high flows potentially washing out 0+brown trout and suitable habitats.

The extreme weather in 2015/16 was believed to play a key role in the loss of 0+ brown trout from a number of control sites and lower densities at the remaining sites through discharge related mortality (Jensen and Johnsen 1999, Nicola *et al.* 2009), suggesting the loss of 0+ brown trout from the River Washburn would have been experienced regardless of the restoration work. For ≥1+ brown trout, the extreme flows at the control sites had marginal impacts, highlighting increased ability of larger brown trout to tolerate greater flow velocities (Jonsson and Jonsson 2011). This was seen in the River Washburn albeit at lower densities, suggesting that available shelter from high flows was low in the River Washburn despite the restoration work.

The recovery of brown trout populations in the River Washburn is likely to take several years. It is unsure as to whether the loss of brown trout in 2015 stemmed from migration of fish away from the disruption or increased mortality from the lethal effects of high siltation and large-scale industrial machinery deployed within the river. As it is assumed that the size of the brown trout populations during the baseline period is representative of the River Washburn, it was expected that brown trout would be found at very low densities throughout the reach.

Other work focussing on habitat restoration of salmonid habitats have noticed that whilst the restoration work may yield a much higher carrying capacity in the river, the time taken for the habitat to become fully exploited will take a long time if initial densities are low (Johnson *et al* 2005, Luhta *et al* 2012). Therefore, a single translocation event, where mature brown trout are taken from downstream and released into the study area is a potential area for future study to aid in improving trout populations in the River Washburn over a much shorter time.

8 GENERAL DISCUSSION AND RECOMMENDATIONS

8.1 Introduction

Habitat diversity as well as the natural flow regime in rivers are fundamental mechanisms driving the population dynamics of all life stages of brown trout (Heggenes 1999, Lobón-Cerviá and Rincon 2004, Richard *et al.* 2015). Rivers and surrounding land use have been subjected to anthropogenic modifications for millennia, and this has disrupted and degraded the natural hydrological processes of river networks leading to the loss of habitat diversity globally (Poff *et al.* 1997, Cowx and Welcomme 1998). Degradation of the riverine habitat and resulting loss of biological diversity became a major cause for concern over the last 30 years, and with advancements in understanding of the mechanics of the riverine ecosystem led to increasing ecological awareness of the impacts that humankind has on the natural world (Poff *et al.* 1997, Cowx and Welcomme 1998). Applying and evaluating current knowledge and research of brown trout population dynamics has led to the development of a suite of restorative measures that can be undertaken to improve degraded habitat for the species. This thesis investigated long-term brown trout populations to elucidate the drivers of population dynamics in heavily degraded rivers in Yorkshire, as well as investigating and evaluating the biological and habitat responses from multiple restorative techniques deployed on heavily modified rivers in Yorkshire.

This chapter discusses the knowledge and insights that were gained from the previous chapters and provides conclusions and recommendations for further study and management pertaining to the mitigation of reservoir operation on the natural flow regime and physical habitat modification, with the goal of achieving GEP in the habitat downstream of impounding reservoirs.

8.1.1 The influence of flow regulation on the natural flow regime

Local climate is a key driver behind the variability in the natural flow regime of rivers, i.e. periods of prolonged rainfall in the catchment will lead to elevated flow magnitude in un-regulated river systems (Burn and Hag Elnur 2002). Investigations into “long-term” flow data of three heavily regulated Yorkshire rivers and their respective catchment rainfall data failed to identify any relationship between these variables (Chapter 3). Large dams and reservoirs are capable of storing the streamflow of upstream rivers and streams for periods of months to years (Collier *et al.* 1996). It is this mode of operation that contributes to the loss of variability caused by local climate effects on river flow downstream of impoundments.

8.1.2 Density-dependent regulation of brown trout population dynamics in heavily regulated rivers

Density-dependent regulation of brown trout populations has been observed in numerous populations across the UK and Europe (Vincenzi *et al.* 2008, Lobón-Cerviá 2009, Elliott 2015, Richard *et al.* 2015). In three heavily regulated rivers in Yorkshire (River Holme, River Loxley and River Rivelin), there were weak relationships between 0+ and successive years ≥1+ brown trout densities, however no relationship was seen between preceding years ≥1+ brown trout and 0+ brown trout. Due to the low densities of brown trout it is likely that the carrying capacity of the river has not been met, therefore, density-dependant processes are not playing a pivotal role in driving population dynamics of brown trout (Chapter 3.). Following investigations into brown trout movement in heavily regulated Yorkshire rivers, Taylor (2017) hypothesised that the presence of older, larger resident brown trout displaced the smaller competitively inferior 0+ brown trout outside of the study sites. Length at age is a biotic factor of brown trout population dynamics as variability in brown trout length can influence factors such as fecundity and mortality (Vincenzi *et al.* 2008, 2010). Across all three study rivers, the length at age 1 of brown trout was negatively related to ≥1+ brown trout population size, suggesting that 0+ brown trout had slower growth in the study rivers in the presence of higher ≥1+ brown trout density. This suggests possible reduced feeding opportunities and increased competitive interactions maybe limiting growth of 0+ brown trout.

8.1.3 The role of density-independent regulation of brown trout population dynamics in heavily regulated rivers.

The natural flow regime is a fundamental driver of brown trout population dynamics (Lobón-Cerviá 2004, Hayes *et al.* 2010). Long term population studies revealed that the influence of discharge variability during the crucial emergence phase of 0+ brown trout can significantly influence the survival rate of the year class, through increased washout, displacement and mechanical shock related mortality (Lobón-Cerviá and Rincon 2004, Unfer *et al.* 2011, Richard *et al.* 2015, Bergerot and Cattaneo 2016). The natural flow regime of a river is also crucial in regulating the habitat variability and suitability for brown trout (Chapter 2). The disruption of the natural flow regime of rivers through the impoundment and abstraction of water is of concern for the conservation of biological diversity (Poff *et al.* 1997, Williams *et al.* 2000).

The length at age 1 of brown trout was found to be significantly influenced by flow rate during the emergence period in three heavily regulated rivers (chapter 3), as high flows during this period were found to negatively influence 0+ brown trout growth; this is likely due to the increased bioenergetic expenditure by newly hatched brown trout holding station at higher water velocities (Flore and Keckeis 1998, Nislow *et al.* 2004, Xu *et al.* 2010).

Long term studies into population dynamics of brown trout under natural conditions have elucidated how important flow variability is for explaining temporal variability of brown trout recruitment success (Lobón-Cerviá *et al.* 1997, Lobón-Cerviá and Rincon 2004). The local climate is largely responsible for regulating flow variability in unregulated river systems. Investigation into long-term population trends of brown trout in three heavily regulated rivers River Holme, River Loxley and River Rivelin) did not identify flow variability during the emergence period as having any influence on recruitment strength across any of the three study rivers (Chapter 3). Importantly, variability in the flow during the summer period significantly influenced brown trout recruitment success (Chapter 3). This study found elevated flow rates during the summer period negatively influenced brown trout recruitment. It is believed that the baseline compensation release from the impounding reservoirs on all three study rivers were higher than what would be expected under un-regulated conditions. Therefore, flow increases above the baseline compensation release may lead to sub-optimal deeper and faster flowing habitat and increased washout of 0+ brown trout. This finding is not surprising given that it is estimated that compensation flows are typically 22% higher than the pre-impoundment natural low flows (Gustard *et al.* 1987). The findings in Chapter 3 give support that the future focus of HMWB mitigation should investigate if the lowering of compensation flows would provide benefits to downstream ecology

8.1.4 Introducing/Amending compensation releases

The use of introducing/amending compensation releases from reservoirs is a commonly used tool by water resource managers in to mitigate the effects of flow regulation from impounding reservoirs (Hull International Fisheries Institute, 2011). The development of the Building Block Methodology (King *et al*. 2008, UKTAG 2013) has allowed for the development of a framework to design and implement new compensation releases that allows for a targeted approach to improving the aquatic ecology of flow regulated HMWBs that can account for specific pressures and constraints on water resource plans.

Given the relatively short period of time allowed to assess the environmental impact of the flow trials it is not surprising that a majority of the biological responses to the introduction and amendment of compensation releases from impounding reservoirs was inconclusive. In Rolls *et al* (2010) the introduction of artificial flood flows from impounding reservoirs was not found to have a significant impact on downstream fish fauna in a similar timeframe to those in this study and in Robinson and Uehlinger (2008) ecological impact of artificial flood flows were only apparent after 3 years of post-impact monitoring. Analogues for changes to the physical habitat were collected alongside fisheries data in the form of HQS (Wyatt *et al* 1995). The HQS values were used as a method to track the spatial and temporal variance of habitat quality across the study sites and determine if mitigation measures had significantly influenced the quality of available habitat for brown trout. The habitat response to the three flow trials in Chapters 4-6 were mixed. Improvements to the habitat quality for the ≥1+ (<20 cm) brown trout were identified in the River Dibb (Chapter 4) and River Holme for the ≥1+ (>20 cm) brown trout age class, this result was most likely manifested in the provision of more water that was able to provide deeper areas of habitat that these age/size classes favour. Significant losses to habitat quality were also identified for 0+ brown trout in Dale Dike (Chapter 5) and for the ≥1+ (<20 cm) brown trout in the River Holme (Chapter 6). The significant interaction to habitat quality for ≥1+ (<20 cm) brown trout in the River Holme was largely manifested in significant improvements to the habitat quality for this age/size class in the control sites that wasn't seen in the impact sites, suggesting that the amended compensation release from Digley and Brownhill reservoir supressed improvements to habitat quality that would have occurred if no change to the flow had taken place.

The flow trials presented in this thesis have been developed incorporating various aspects of the Building Block Methodology (King *et al.* 2008, UKTAG 2013). The habitat responses from studies that incorporated more elements of the building block such as season variability and winter and spring elevations (Chapters 4 and 6) were largely positive. The significant decrease of 0+ brown trout habitat quality in Dale Dike is also a result of importance. It is likely that the increase in flow from the compensation release has led to a reduction in suitable marginal and nursery habitat in Dale Dike due to flow homogenisation, highlighting the need to ensure that the variability of the natural flow regime is incorporated into the future application of mitigation measures. Understanding why mitigation measures have failed to have the desired effect is an important part of the evaluation process. Therefore, even though the results from the flow trials presented in Chapters 4-6 are mixed, the application of the lessons learnt from these studies is important to the to guide future management of regulated flow regimes.

8.1.5 Physical habitat restoration

Physical habitat restoration can be achieved on a variety of scales, indeed, similar to the methodology to introducing and amending compensation releases there is no established *one size fits all approach* to restoring rivers they can range from relatively small scale works such as the addition of large woody debris (Johnson *et al.* 2005, Thompson *et al*. 2018) to larger scale civil engineering works such as the Alt Lorgy restoration work in Scotland (Friberg *et al.* 2015), and can include a variety of techniques to help modified river channels return to a more *natural-like* state (Forseth and Harby, 2014). It is important that prior to the commencement of any instream work it is vital that the factors contributing to habitat degradation are fully understood and identified to ensure a target application of resources to improving the riverine habitat. In cases where it is operationally and economically unfeasible to introduce/amend a compensation release from an impounding reservoir, the use of physical habitat restoration is a recognised alternative to achieving GEP in HMWBs. In Chapter 7 a physical habitat restoration project was undertaken on a 1-km stretch of the River Washburn downstream of Swinsty reservoir. The biological responses to the restoration works were poor, the BACI model revealed that there was a significant decline of ≥1+ (<20 cm) brown trout densities following two years post impact monitoring. Within months of the completion of the restoration work, the River Washburn was subjected to a substantial flooding event in the winter of 2015, with water levels in the River Washburn rising several metres higher than the levels measured during HABSCORE surveys. It is likely that the compounding factors of an intensive programme of instream works and extreme flood flow are responsible for the decline in of ≥1+ (<20 cm) brown trout. Whilst it is possible that there were declines in 0+ and ≥1+ (>20 cm) brown trout densities (there was an increase in the occasions that trout of these age classes were absent from impact sites) the low densities recorded during the before period was likely responsible for the lack of significant result.

BACI models revealed that there were significant improvements to the habitat quality for both size classes of ≥1+ brown trout following the restoration, considering that the restoration work encompassed creating more areas of deeper and faster flowing habitat, this result is not surprising. It does however demonstrate the resilience of the habitat restoration work, as personal observations following the 2015 floods revealed that elements of the restoration work, such as large woody debris and flow deflectors had been damaged and washed out, contributing to the lack of change in the 0+ brown trout habitat quality as the loss to the areas of shallow and slack nursery areas that are crucial for recruitment success of brown trout (Jonsson 1989, Cowx *et al.* 2004, Garcia de Leaniz *et al.* 2007).

The application of physical habitat restoration is not exclusive to addressing the issue of flow modification, in fact in many the goals of many habitat restoration projects have been to remedy issues surrounding, water quality, flood plain and riparian management and historic anthropogenic modifications to the water body (Bernhardt, 2005, Johnson *et al.* 2005, Friberg *et al.* 2015, Thompson *et al*. 2018) This approach for large scale restoration works was understood to be used as a case study for YWS to provide evidence into the efficiency of habitat restoration and highlighted its viability as an alternative to introduction and modification of flow releases for future HMWB mitigation projects (personal communication, 2018) where socio-economic pressures such as:, loss of yield to water companies, flood risk management concerns or outdated infrastructure constrain the ability to implement sufficient elements of the building block approach (King *et al.* 2008, UKTAG, 2013).

8.1.6 Study Design and industrial constraints

With the exception of length of the capture of 0+ brown trout, a majority of the biological data collected in Chapters 4-7 were too varied to determine the significance of any impact stemming from the introduction of a HMWB mitigation measure, therefore, a majority of the biological responses investigated in these case studies were inconclusive. It is likely that the heavy rainfall and subsequent flood events in 2015 play a role in the high temporal variance in the case studies presented in this thesis. Extending the study period to include more post impact years will reduce the unexplained variance in the BACI models to allow for robust conclusions to be drawn from these studies. Extra years of fisheries monitoring would be required to establish conclusively the biological responses to the HMWB mitigation measures, the requirement however, would need to be balanced against the needs of water resource managers and regulating authorities. It is important to note that there is an expectation from water resource managers and the regulating authorities (EA, DEFRA, OFWAT etc) that these projects can be delivered within the timeframe of a 5-year AMP (Adaptive Management Period). For each of the flow trials in Chapters 4-6 the licences granted to Yorkshire Water to vary/introduce a compensation flow from a reservoir were issued on a Time-Limited License (TLL). A TLL is issued with a firm expiry date at the end of an AMP, whereby the water resource manager must decide to either return to the pre-trial compensation release regime or formalise the trial flow into a statutory compensation release via an application to the EA. These 5-year AMP cycles are also important in the context of wider ecological management as River Basin Management Plans, water resource business plans, and WFD assessment cycles are all tied into each AMP, therefore, projects that are not delivered within their AMP deadline can impact on the data available for these environmental management and water resource plans.

8.1.7 Longitudinal connectivity

Despite not being the primary focus of this research, the fragmentation of habitat throughout the UK river network because of weirs and dams is an issue that is likely impacting on brown trout populations across, all study sites as large weirs and dams are present within all study catchments. There is a consensus that barriers such as weirs and dams are extremely detrimental to upstream migration of brown trout (Aarestrup and Jepsen 1998, Rustadbakken *et al.* 2004, Gosset *et al.* 2006) with this research highlighting that larger brown trout from the main river channel were unable to complete their upstream migration into smaller tributaries due to significant barriers. Large adult brown trout were found in very low densities across all study sites, which is probably due to poorer habitat quality available within the streams, but it is also likely that larger and more fecund trout are unable to migrate into the study rivers from the main river channel leading to reproductive isolation within the study streams.

8.1.8 Management implications of future HMWB mitigation measures

The process of mitigating flow regulation and HMWB status in UK waterbodies can be costly to water resource managers (Acreman *et al.* 2009). A key requirement in the attainment of GEP in these water bodies is, therefore, to ensure a targeted and sustainable approach to flow regulation mitigation. Guidance and recommendations from expert opinion and existing literature, as well evaluation of scientific studies such as those presented in this thesis, can help inform water resource managers, but the application of this knowledge into real life scenarios can prove to be challenging, especially given the expectation for the project to be delivered within relatively tight AMP deadlines.

Whilst there is no standard practice for fisheries assessment to monitor the impact of HWMB mitigation measures, the assessment of 0+ fish populations appear to be the favoured biological metric in BACI study designs of this nature (Johnson *et al*. 2005, Bradford *et al.* 2011, Hull International Fisheries Institute 2011, Conner *et al.* 2015). In this thesis the collection of habitat quality data (derived from the HABSCORE programme) revealed that it was possible to identify significant changes to the habitat quality for salmonids during the timeframe of the study. This is a novel application of the HABSCORE data and demonstrates that significant changes to the riverine habitat can be detected in a much shorter time frame. The incorporation of this methodology into future HMWB mitigation projects would mean informative outputs on the impacts of the mitigation measure are available in a much shorter timeframe to water resource managers and regulators in line with industry deadlines. Whilst the application of HABSCORE data to a BACI design is a novel approach, the collection of HABSCORE data is commonly undertaken alongside fisheries monitoring in England and Wales, as the HABSCORE programme is widely used as standard practice by the EA and NRW. This thesis therefore, demonstrates that robust conclusions can be drawn into the effectiveness of HMWB mitigation measures in a shorter timeframe using habitat assessment tools that are readily available and routinely collected. As HABSCORE was developed specifically for rivers in England and Wales, the application of this specific methodology is limited, however the existence of other habitat assessment tools such as the Habitat Quality Index (Binns, and Eiserman, 1979) or the Rapid Method for Trout Stream Habitat Assessment (Lanka *et al* 1985) in other countries demonstrates that the principles of the methodologies in this thesis may be applied on a much wider scale.

The application of physical habitat restoration (Chapter 7) demonstrated that the methodologies used present a viable alternative to addressing the impacts of flow modification then just the introduction and amendment of compensation releases from impounding reservoirs. It is important to consider though that this methodology should not be viewed purely as an alternative. Smaller scale habitat restoration projects using individual elements of the suite of tools used on the river Washburn have shown to yield significant improvements to both habitat quality and salmonid populations in other non-regulated rivers. The creation of complex marginal habitat using large woody debris can be implemented on a relatively small scale and was shown to lead to significant improvements in salmonid density (Roni *et al.* 2015, Thompson *et al.* 2017) by creating areas of refuge and nursery habitat as well as increasing biomass of basal resources and invertebrates increasing autochthonous production within the habitat. The success of these small-scale habitat restoration projects demonstrates that future programmes of HMWB mitigation should explore a more complimentary approach, such as identifying where small-scale physical restoration efforts can be used to complement changes to the compensation regime to provide a better habitat and biological response.

8.2 Conclusions and recommendations

8.1.1 Density dependent and independent regulation of brown trout population dynamics in HMWBs

Investigations into the population dynamics of brown trout in three HMWBs found that there was a significant correlation between 0+ brown trout populations and successive years >1+ brown trout populations but no correlation was found between 0+ brown trout populations and the preceding years >1+ brown trout populations. It is speculated that 0+ brown trout were being displaced downstream through either washout from elevated flows or competition with larger competitively superior resident brown trout. **It is recommended that further investigation using drift traps during the emergence period and flow data would be able to elucidate if any relationship exists between flow conditions and 0+ brown trout displacement in regulated rivers.** To reliably gauge the emergence period, **temperature data must be collected as well as habitat surveying to identify the location of redds, to estimate time to emergence based on the degree day model by Elliott (1984).** The flow regime was also found to influence 0+ brown trout densities and growth in the three HMWB studied. Unlike other studies, the role of flow variability during the emergence period was not significantly correlated with 0+ brown trout densities but flows during the summer period were in these highly regulated study rivers. It is proposed that, due to reservoir operation, there was insufficient flow variability during brown trout emergence to elicit any changes to the 0+ brown trout populations. This study found there was no significant influence of rainfall on flow rates, and it is suggested that the water storage and reservoir operation were likely responsible for the supressed flow variability in the downstream river reaches. **It is therefore recommended that flow data are recorded in the catchment upstream of the regulating reservoirs to elucidate the extent that reservoir operation suppresses the natural flow regime.** Water temperature did not have any influence on brown trout population dynamics in the study rivers. Unfortunately, water temperature data were not available for the full period of the study and had to be modelled from air temperature. Whilst water temperature was related to air temperature the level of variance explained by these models for the two rivers were low (~65%), suggesting that other influences were exerting affecting this relationship. It is likely that the temperature that water is released from the reservoir is an important factor in these relationships. **It is therefore recommended that temperature loggers are located at the reservoir outfall to gauge the temperature at which the compensation flow is released, as well as placing loggers downstream to provide long-term water temperature data where the rivers have mixed.**

8.2.1 Mitigating flow regulation with compensatory releases

The introduction of a four-stage, seasonally varied compensation release from Grimwith Reservoir into the River Dibb yielded significant improvements in the habitat quality for ≥1+ (<20 cm) brown trout, but there were no significant responses from other biological or habitat metrics. **It is recommended that the current compensation release from Grimwith Reservoir is maintained and brown trout populations are continued to be assessed to elucidate any subsequent changes.** Reservoir operation and water resource demands still require flow releases from Grimwith Reservoir outside of the compensation release for hydroelectric generation, as well as maintaining abstraction levels throughout the Wharfe catchment. **It is recommended that where possible these hydropower flows are minimised to ensure a flow profile in line with UKTAG guidance.** The water level for abstraction on the Wharfe must be maintained to ensure adequate abstraction, however it is **recommended that balancing releases from other reservoirs throughout the Wharfe catchment, i.e. Lower Barden Reservoir, is investigated to reduce the impact to the flow regime on the River Dibb.** The influence of high flows, particularly in the upper reaches of the River Dibb, and the lack of substrate replenishment from upstream of Grimwith Reservoir, has led to dominance of larger substrate, such as boulders and cobbles. It is **recommended that reintroducing suitable spawning gravels into the upper reaches of the River Dibb would help promote suitable habitat and brown trout recruitment.** This action of replenishing spawning substrate would have to be an ongoing management intervention due to the continual washout from high flows and the lack of substrate transport from the upper reaches stemming from reservoir impoundment. As the quantity of water released from Grimwith reservoir to support downstream abstraction is high it is also **recommended that the option of habitat intervention is explored to ensure that the habitat is able to effectively retain the reintroduced gravels.**

A significant decline in 0+ brown trout habitat quality in Dale Dike was found after the introduction of an annual minimum flow compensation release form Dale Dike Reservoir It was unclear however, as to the cause. It was not possible to determine if there was any meaningful change to the hydrodynamics of Dale Dike following the introduction of the compensation release in 2014 due to a pre-existing, un-gauged, flow releases from the reservoir to ensure adequate operation of Loxley Water Treatment Works. The annual minimum flow represents the most basic form of compensation flow as it incorporates only a single element of the building block methodology (King *et al.* 2008, UKTAG 2013). **It is recommended further inclusion of elements from the Building Block Methodology i.e. winter elevations and freshets, are incorporated into the Dale Dike release regime to create greater habitat diversity for brown trout.** The reduction 0+ habitat quality in Dale Dike is probably a result of the formalisation of the annual minimum flow providing consistently faster, deeper-flowing, areas of habitat that are unsuitable for 0+ brown tout (Cowx *et al.* 2004), however, due to the absence of flow data prior to 2014 it was not been possible to substantiate this claim. **It is recommended**

that when undertaking investigations, quantitative data pertaining to the flow regime prior to any introduction or amendment of a compensation release is collected to determine if any meaningful change to the flow regime has occurred. The lack of suitable 0+ brown trout habitat will likely impact on the brown trout population, **therefore it is recommended that physical restoration of habitat, such as the introduction of flow deflectors and woody debris instream in Dale Dike, is undertaken to provide greater 0+ brown trout habitat.**

The amendment of an existing compensation flow from Brownhill and Digley reservoirs improved habitat for both size classes of ≥1+) brown trout. However, habitat quality for ≥1+ (<20 cm) brown trout was significantly lower than predicted from the temporal trends across the control sites, but in real terms represented a reliable change to habitat quality between the two periods (before and after) of the study. **It is recommended that the current compensation release is maintained from Brownhill and Digley reservoirs,** but as part of the river basin management cycle this compensation should be re-evaluated periodically. With advances in the scientific understanding of measure to mitigate the influence of impounding reservoirs, it may become apparent over the course of time that more appropriate methodologies need to be developed to ensure the goal of GEP is achieved in the River Holme**.** Improvements in 0+ brown trout habitat quality in the River Holme were found following the amendment of the flow trial, but this increase was deemed representative of regional trends in habitat quality, i.e. trends in the impact sites were similar to those seen across reference sites, and therefore not as a result of the amended flow trial.

The loss of longitudinal connectivity with the main river channels in the study rivers is likely contributing to the reproductive isolation of brown trout populations within the study reaches. This isolation, coupled with low adult densities, are likely contributing to poor recruitment across many study sites. For the River Dibb and River Holme (Chapters 4 and 6, respectively), these migratory barriers are mostly in the form of large weirs throughout the downstream catchments. **It is recommended that migratory barriers throughout the Wharfe and Calder catchments are assessed and where possible mitigated** to restore longitudinal connectivity in the rivers Dibb and Holme and allow larger more fecund brown trout from the main river channel to spawn in these smaller tributaries. Damflask Reservoir and numerous weirs on the River Loxley and throughout the Don catchment present significant barriers preventing and larger brown trout from the lower catchment migrating into Dale Dike for spawning. There is potential for upstream colonisation and migration from trout in Damflask Reservoir, but the condition of brown trout populations in Damflask as well as any migratory barriers in the lower reaches of Dale Dike remain unknown. **It is recommended that further research is undertaken to establish the presence and condition of any brown trout populations within Damflask Reservoir.**

8.2.2 Mitigation of flow regulation with instream habitat modification

The physical restoration of 1 km of river was deemed the most prudent form of mitigation of the habitat homogeny in the River Washburn due to the operational inability to alter the flow regime from Swinsty reservoir. The restoration work aimed at reducing the river width to provide areas of faster deeper flowing habitat for larger brown trout (Cowx *et al.* 2004, UKTAG 2013). Habitat surveys undertaken in 2015, immediately after the restoration work revealed improvements to the habitat quality for all age/size classes of brown trout. The introduction of groynes, berms and flow deflectors, as well as re-introduction of smaller gravel, provided greater habitat heterogeneity across the restoration sites (Cowx and Welcomme 1998). Unfortunately, a 1-in-100-year flood occurred between the 2015 and 2016 fish survey seasons and subsequently most of the instream restoration structures were damaged or destroyed, degrading the habitat quality. **It is recommended that the instream structures are repaired or replaced to ensure habitat heterogeneity in the River Washburn.** The initial brown trout densities in the River Washburn prior to the restoration works were very low for all age/size classes. These populations further deteriorated following the instream work, with brown trout absent from a number of sites in 2015 and 2016. It is believed that the invasive nature of the instream works, i.e. utilising heavy machinery, led to disturbance of the sediment and drastically increased the suspended solids within the water column. It is possible that this poor habitat quality during the restoration works increased the risk of mortality or more likely displacement of brown trout further downstream as the habitat within the restoration zone and immediately downstream became uninhabitable. **It is recommended that during future instream restoration works, care is exercised when undertaking major earthworks to minimise impacts on resident brown trout populations or translocating resident fish population to undisturbed habitat for the duration of the instream work.** Due to the heavily impounded nature of the River Washburn, barriers to migration may impede recolonization of the restoration reach. It is therefore **recommended that any barriers to migration on the River Washburn are properly assessed and where possible mitigated to allow fish passage.** The initial low densities of brown trout across the survey sites in the river Washburn may also pose an obstruction to speedy recolonization of brown trout into the restoration reach. If following further years of post-restoration work investigation there has been no noticeable improvement to the brown trout populations. **It is recommended that the**

option of relocation of brown trout into the restoration zone would help stimulate recolonization and habitat utilisation of the improved habitat is explored.

To ensure the success of relocation, **it is recommended that brown trout population investigations are undertaken in the lower reaches of the River Washburn and Lindley Wood Reservoir** to determine if any brown trout populations in the lower reaches are viable candidates for relocation. Populations upstream of Swinsty reservoir (between Thruscross and Fewston reservoir) are also viable candidates for relocation.

8.2.3 Study design

Investigations into the biological response to flow regulation mitigations in Chapters 3-7 failed to identify any significant change to brown trout population dynamics within the timeframe of the study (with the exception of a significant decrease in ≥1+ (<20 cm) brown trout densities in Chapter 7, (River Washburn)). It is likely that given the time constraints of this study only large and abrupt changes to brown trout populations would be detected within 2-3 years of post-change monitoring (Bayley 2002). Indeed, resource calculations revealed that for a majority of the biological metrics (with the exception of length at capture for 0+ brown trout) the level of temporal and spatial variance was too high to provide definitive conclusions to the change in brown trout densities. **It is recommended that investigations into brown trout population dynamics are continued across all control and impact sites until a minimum of 4-years post-change monitoring is completed.** This will allow further understanding of the temporal and spatial variance and more robust conclusions regarding the biological response to the mitigation measures to be drawn. However, this recommendation must be balanced with the practicalities that water resource managers and regulating authorities face. A major challenge that was faced for these projects is that there was an expectation that these projects will be completed within the cycle of a 5-year AMP (Adaptive Management Programme) cycle. Therefore, for many WFD projects (such as the ones detailed in Chapters 4-7), regulatory deadlines expected the projected to be delivered using a total of 5 years (pre and post impact) ecological monitoring. The rationale behind the minimum 4-year post-change monitoring is taken from prior studies, such as the sustainable compensation releases programme (Hull International Fisheries Institute, 2011) where 4 years of post-change data was deemed appropriate to draw robust conclusions from the biological data. It is understood that the extension of the post-change monitoring will likely not fit with regulatory deadlines but will provide valuable evidence into fisheries response to change in habitat quality, as well as highlight pressures on the ecology outside the scope of detection within the HABSCORE model. These outputs would be valuable in the planning of future AMP cycles. **It is recommended that alongside**

fisheries data, habitat quality data (HABSCORE) is included in the BACI study design. Whilst the primary focus of the fisheries monitoring is to determine the biological response to the HMWB migration measures, the case studies in this thesis have demonstrated that changes to habitat quality can be detected in a much smaller timeframe than fisheries responses. The implementation of this methodology would allow for informative outputs to the water resource managers and their regulators within the timeframes of their AMP cycles.

It is also important to determine the carry capacity of the rivers to highlight if any biotic or abiotic factors outside the scope of detection within this study are limiting brown trout populations in the River Washburn. **It is recommended that water quality macro invertebrate sampling assessment are undertaken** more than once a year to help identify seasonal trends that would not be apparent with sampling at yearly intervals. The collection of these data will serve to highlight if there are any underlying issues pertaining to water chemistry, i.e. pH, dissolved oxygen, and phosphorous and nitrogen content, that can be limiting to brown trout populations.

In Chapters 4-7 0+ and ≥1+ brown trout densities were reported alongside their respective EA-FCS classification. The EA-FCS is a national classification tool used in England and Wales to allow for a comparison between the observed densities and national averages. As brown trout occupy a wide range of river types in the UK i.e. Yorkshire Spate rivers, Chalk streams, low land rivers (Frost and brown, 1967, Mann *et al*. 1989, Cowx *et al*. 2004, Jonsson and Jonsson 2011), it is possible that densities of trout from high energy spate rivers will be classified as poor/fair poor even though brown trout would be naturally expected to be found in much lower densities. This may indicate to environmental managers that there is factor in the habitat limiting trout populations when in reality none exist **It is therefore recommended that further investigation is given into developing regional classification schemes to allow for better classification and comparison of brown trout densities that accounts for the varied habitat types across the UK.**

None of the control sites used in the investigations in Chapters 3-7 were located on the same river as the impact sites for many reasons, ranging from unrepresentative instream habitat absence of brown trout, poor to no access for survey teams, or not enough river to reasonable accommodate the number of sampling sites to ensure habitat heterogeneity. Some control sites were therefore located in different catchments, making the impact assessments performed sub-optimal due to the increased levels of spatial variance amongst the fish populations in the control reaches. **It is recommended that**

in future studies, control reaches are selected to ensure that the spatial and temporal variation in fish and habitat metrics is as representative as possible of the impact sites. This would allow for more robust conclusions to be drawn in a shorter timeframe. This recommendation again should be balanced against the requirements of the study, and whilst control sites located upstream of the impounding reservoir are deemed optimal control sites, the requirement for representative habitat and fisheries populations in control sites is paramount.

Whilst this research assessed either changing the reservoir release regime or physical habitat restoration (where releases could not be modified), future attempts to mitigate HMWB status on regulated rivers should investigate the application of both these methodologies in conjunction. In the River Dibb, Dale Dike and River Holme case studies an improvement to ≥1+ brown trout habitat quality but not to 0+ habitat quality, the use of physical habitat restoration methods to improve 0+ habitat i.e. flow deflectors and large woody debris to provide greater marginal nursery habitat could illicit greater habitat quality improvement for all age/size classes of brown trout .**Therefore, it is recommended that in the future implementation of mitigation measures both flow restoration and physical habitat restoration should not be viewed as mutually exclusive**

This study has advanced the knowledge of the drivers of population dynamics of brown trout in heavily modified water bodies as well as the effectiveness of a range of flow regulation and habitat mitigation measures and their impacts on habitat quality and brown trout population dynamics. It has also demonstrated that the application of routinely collected habitat quality data can be applied to provide indications of the success of HMWB mitigation measures within the timeframes set by water resource managers and regulating authorities.

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Table A.1. Density estimates ± 95% C.L. of >0+ brown trout at impact and control sites during both the before (2012 – 2014) and after (2015 – 2016) periods. Colours denote EA-FCS abundance classification

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

River Name	Site Identifier	2012	2013	2014	2015	2016
		Before			After	
impact Sites						
River Dibb	DB1	3.1 ± 0.5	6.09 ± 0.4	12.5 ± 0.6	17.22 ± 1.2	19.37 ± 4
River Dibb	DB2	3.08 ± 0.4	11.55 ± 0.8	12.08 ± 0.2	16.27 ± 0.5	14.29 ± 1.4
River Dibb	DB3	3.13 ± 0.5	2.12 ± 0.1	6.53 ± 0.4	6.28 ± 0.3	8.92 ± 0.9
River Dibb	DB4	2.72 ± 0.2	12.92 ± 2.9	12.71 ± 0.5	11.43 ± 0.7	5.47 ± 1.5
River Dibb	DB5	3.63 ± 0.2	3.44 ± 0.2	12.11 ± 1.4	10.14 ± 1.3	7.87 ± 0.4
River Dibb	DB6	1.96 ± 0.2	5.18 ± 0.5	5.64 ± 0.3	6.81 ± 1.3	5.25 ± 2.1
Control Sites						
Barden Beck	WR1	10.33 ± 0.9	10.19 ± 0.3	14.77 ± 0.7	12.8 ± 1.4	8.44 ± 0.8
Barden Beck	WR2	5.6 ± 0.1	6.33 ± 0.1	3.35 ± 0.3	5.14 ± 0.4	6.41 ± 0
Ings Beck	WR3	5.6 ± 0.9	9.48 ± 0.5	10.2 ± 0	7.45 ± 0.1	0.48 ± 0
Ings Beck	WR4	4.5 ± 0.4	4.53 ± 0.2	1.11 ± 0.3	1.72 ± 0.2	1.71 ± 0
Ashfold Side beck	WR5	14.02 ± 0.9	11.56 ± 2.1	25.15 ± 2.0	32.5 ± 0.7	22.99 ± 1.1
Ashfold Side beck	WR6	23.46 ± 2.0	7.31 ± 0.5	29.76 ± 0.2	42.25 ± 3.9	21.3 ± 1.9
Ashfold Side beck	WR7	18.08 ± 0.3	7.94 ± 0.4	21.57 ± 0.6	19.76 ± 0.8	18.87 ± 1.6
Ashfold Side beck	WR8	8.71 ± 1.8	9.49 ± 0.4	0.39 ± 0.0	9.32 ± 0.7	13.7 ± 0.8
Ashfold Side beck	WR9	11.96 ± 1.1	11.36 ± 0	12.7 ± 0.4	12.96 ± 0.3	10.97 ± 1.1

Table A.2. Length Range for 0+ brown trout captured at all impact and control sites for the Grimwith Reservoir study (2012 – 2016). N/S denotes no fish caught

Year	Site	River	Length Range
2012	DB1	River Dibb	N/S
2012	DB2	River Dibb	78 - 84 mm n = 2
2012	DB3	River Dibb	68 - 76 mm n = 2
2012	DB4	River Dibb	N/S
2012	DB5	River Dibb	80 - 89 mm n = 2
2012	DB6	River Dibb	82 mm n = 1
2012	WR1	Barden Beck	65 - 98 mm n = 3
2012	WR2	Barden Beck	74 - 84 mm n = 7
2012	WR3	Ings Beck	78 - 103 mm n = 9
2012	WR4	Ings Beck	80 - 89 mm n = 3
2012	WR5	Ashfold Side Beck	N/S
2012	WR6	Ashfold Side Beck	59 - 75 mm n = 4
2012	WR7	Ashfold Side Beck	63 - 86 mm n = 5
2012	WR8	Ashfold Side Beck	61 - 70 mm n = 3
2012	WR9	Ashfold Side Beck	60 - 70 mm n = 5
2013	DB1	River Dibb	64 - 75 mm n = 2
2013	DB2	River Dibb	71 - 80 mm n = 6
2013	DB3	River Dibb	67 - 89 mm n = 11
2013	DB4	River Dibb	73 - 82 mm n = 3
2013	DB5	River Dibb	68 - 79 mm n = 5
2013	DB6	River Dibb	78 - 88 mm n = 8
2013	WR1	Barden Beck	64 - 78 mm n = 4
2013	WR2	Barden Beck	66 - 88 mm n = 17
2013	WR3	Ings Beck	44 - 94 mm n = 40
2013	WR4	Ings Beck	64 - 89 mm n = 19
2013	WR5	Ashfold Side Beck	54 - 77 mm n = 10
2013	WR6	Ashfold Side Beck	66 - 88 mm n = 6
2013	WR7	Ashfold Side Beck	58 - 82 mm n = 23
2013	WR8	Ashfold Side Beck	55 - 89 mm n = 39
2013	WR9	Ashfold Side Beck	60 - 78 mm n = 15
2014	DB1	River Dibb	72 - 91 mm n = 8
2014	DB2	River Dibb	62 - 92 mm n = 15
2014	DB3	River Dibb	68 - 94 mm n = 14
2014	DB4	River Dibb	80 - 93 mm n = 5
2014	DB5	River Dibb	65 - 89 mm n = 17
2014	DB6	River Dibb	67 - 102 mm n = 18
2014	WR1	Barden Beck	71 - 100 mm n = 40
2014	WR2	Barden Beck	74 - 91 mm n = 8
2014	WR3	Ings Beck	70 - 103 mm n = 24
2014	WR4	Ings Beck	82 - 103 mm n = 3
2014	WR5	Ashfold Side Beck	54 - 82 mm n = 37
2014	WR6	Ashfold Side Beck	52 - 87 mm n = 77
2014	WR8	Ashfold Side Beck	N/S
2014	WR7	Ashfold Side Beck	53 - 90 mm n = 76
2014	WR9	Ashfold Side Beck	54 - 98 mm n = 33

Table A.2 Continued

Year	Site	River	Length Range
2015	DB1	River Dibb	71 - 82 mm n = 4
2015	DB2	River Dibb	78 - 78 mm n = 2
2015	DB3	River Dibb	60 - 82 mm n = 10
2015	DB4	River Dibb	N/S
2015	DB5	River Dibb	68 - 83 mm n = 11
2015	DB6	River Dibb	60 - 83 mm n = 7
2015	WR1	Barden Beck	71 - 88 mm n = 9
2015	WR2	Barden Beck	62 - 81 mm n = 24
2015	WR3	Ings Beck	71 - 99 mm n = 39
2015	WR4	Ings Beck	78 - 100 mm n = 7
2015	WR5	Ashfold Side Beck	55 - 90 mm n = 5
2015	WR6	Ashfold Side Beck	49 - 75 mm n = 10
2015	WR7	Ashfold Side Beck	55 - 86 mm n = 16
2015	WR8	Ashfold Side Beck	60 - 85 mm n = 12
2015	WR9	Ashfold Side Beck	61 - 89 mm n = 4
2016	DB1	River Dibb	88 mm n = 1
2016	DB2	River Dibb	83 - 87 mm n = 2
2016	DB3	River Dibb	67 - 97 mm n = 19
2016	DB4	River Dibb	N/S
2016	DB5	River Dibb	70 - 85 mm n = 3
2016	DB6	River Dibb	87 - 102 mm n = 4
2016	WR2	Barden Beck	88 mm n = 1
2016	WR5	Ashfold Side Beck	63 - 75 mm n = 5
2016	WR6	Ashfold Side Beck	55 - 81 mm n = 6
2016	WR7	Ashfold Side Beck	57 - 80 mm n = 11
2016	WR9	Ashfold Side Beck	59 - 85 mm n = 5

Table A.3 Density estimates ± 95% C.L. of >0+ brown trout at impact and control sites during both the before (2012 – 2013) and after (2014 – 2016) periods. Colours denote EA-FCS abundance classification

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

River Name	Site Identifier	2012	2013	2014	2015	2016
		Before			After	
impact Sites						
Dale Dike	DD1	3.1 ± 0.5	6.09 ± 0.4	12.5 ± 0.6	17.22 ± 1.2	12.59 ± 1.2
Dale Dike	DD2	3.08 ± 0.4	11.55 ± 0.8	12.08 ± 0.2	16.27 ± 0.5	1.36 ± 0.0
Dale Dike	DD3	3.13 ± 0.5	2.12 ± 0.1	6.53 ± 0.4	6.28 ± 0.3	1.13 ± 0.3
Dale Dike	DD4	2.72 ± 0.2	12.92 ± 2.9	12.71 ± 0.5	11.43 ± 0.7	1.16 ± 0.3
Dale Dike	DD5	3.63 ± 0.2	3.44 ± 0.2	12.11 ± 1.4	10.14 ± 1.3	0.00 ± 0.0
Dale Dike	DD6	1.96 ± 0.2	5.18 ± 0.5	5.64 ± 0.3	6.81 ± 1.3	2.75 ± 0.5
Control Sites						
Ewden Beck	DR1	10.33 ± 0.9	10.19 ± 0.3	14.77 ± 0.7	12.8 ± 1.4	1.81 ± 0.0
Ewden Beck	DR2	5.6 ± 0.1	6.33 ± 0.1	3.35 ± 0.3	5.14 ± 0.4	0.00 ± 0.0
Little Don	DR3	5.6 ± 0.9	9.48 ± 0.5	10.2 ± 0.0	7.45 ± 0.1	1.12 ± 0.2
River Don	DR4	4.5 ± 0.4	4.53 ± 0.2	1.11 ± 0.3	1.72 ± 0.2	0.75 ± 0.1
River Don	DR5	14.02 ± 0.9	11.56 ± 2.1	25.15 ± 2	32.5 ± 0.7	0.89 ± 0.3
River Sheaf	DR6	23.46 ± 2.0	7.31 ± 0.5	29.76 ± 0.2	42.25 ± 3.9	2.49 ± 0.0
Wyming Brook	DR7	18.08 ± 0.3	7.94 ± 0.4	21.57 ± 0.6	19.76 ± 0.8	4.29 ± 0.2

Table A.4. Length Range of 0+ Brown trout captured at impact and control sites during the Dale Dike reservoir Study (2012 – 2016). N/S denotes no fish caught

Year	Site	River	Length Range
2012	DD1	Dale Dike	47 - 73 mm n = 15
2012	DD2	Dale Dike	47 - 52 mm n = 4
2012	DD3	Dale Dike	55 - 62 mm n = 3
2012	DD4	Dale Dike	53 - 70 mm n = 4
2012	DD5	Dale Dike	54 mm n = 1
2012	DR1	Ewden Beck	53 - 83 mm n = 14
2012	DR2	Ewden Beck	52 - 58 mm n = 3
2012	DR3	Little Don	73 - 96 mm n = 9
2012	DR4	River Don	76 - 84 mm n = 4
2012	DR5	River Don	62 - 88 mm n = 13
2012	DR6	River Sheaf	66 - 72 mm n = 2
2012	DR7	Wyming Brook	63 - 72 mm n = 6
2013	DD1	Dale Dike	46 - 68 mm n = 28
2013	DD2	Dale Dike	41 - 61 mm n = 6
2013	DD3	Dale Dike	44 - 68 mm n = 24
2013	DD4	Dale Dike	47 - 73 mm n = 14
2013	DD5	Dale Dike	44 - 68 mm n = 11
2013	DD6	Dale Dike	48 - 89 mm n = 12
2013	DR1	Ewden Beck	58 - 81 mm n = 13
2013	DR2	Ewden Beck	68 mm n = 1
2013	DR4	River Don	59 - 84 mm n = 10
2013	DR5	River Don	56 - 85 mm n = 32
2013	DR6	River Sheaf	55 - 84 mm n = 23
2013	DR7	Wyming Brook	42 - 78 mm n = 29
2014	DD1	Dale Dike	50 - 73 mm n = 17
2014	DD2	Dale Dike	46 - 65 mm n = 4
2014	DD3	Dale Dike	57 - 71 mm n = 8
2014	DD4	Dale Dike	56 - 73 mm n = 8
2014	DD5	Dale Dike	50 - 70 mm n = 8
2014	DD6	Dale Dike	49 mm n = 1
2014	DR1	Ewden Beck	59 - 83 mm n = 5
2014	DR2	Ewden Beck	77 - 82 mm n = 4
2014	DR3	Little Don	72 - 82 mm n = 2
2014	DR4	River Don	71 - 83 mm n = 8
2014	DR5	River Don	50 - 84 mm n = 40
2014	DR6	River Sheaf	62 - 85 mm n = 35
2014	DR7	Wyming Brook	53 - 74 mm n = 12

Table A.4 Continued

Year	Site	River	Length Range
2015	DD1	Dale Dike	50 - 76 mm n = 22
2015	DD2	Dale Dike	49 - 69 mm n = 7
2015	DD3	Dale Dike	52 - 59 mm n = 9
2015	DD4	Dale Dike	53 - 66 mm n = 6
2015	DD5	Dale Dike	51 - 53 mm n = 3
2015	DD6	Dale Dike	60 - 62 mm n = 2
2015	DR1	Ewden Beck	56 - 83 mm n = 7
2015	DR2	Ewden Beck	64 - 67 mm n = 3
2015	DR3	Little Don	68 - 80 mm n = 5
2015	DR4	River Don	63 - 85 mm n = 13
2015	DR5	River Don	53 - 84 mm n = 37
2015	DR6	River Sheaf	51 - 73 mm n = 34
2015	DR7	Wyming Brook	69 - 76 mm n = 2
2016	DD1	Dale Dike	49 - 72 mm n = 20
2016	DD2	Dale Dike	55 - 87 mm n = 3
2016	DD3	Dale Dike	58 - 63 mm n = 3
2016	DD4	Dale Dike	63 - 78 mm n = 3
2016	DD5	Dale Dike	N/S
2016	DD6	Dale Dike	54 - 62 mm n = 4
2016	DR1	Ewden Beck	69 - 97 mm n = 5
2016	DR2	Ewden Beck	N/S
2016	DR3	Little Don	77 - 84 mm n = 3
2016	DR4	River Don	77 - 95 mm n = 4
2016	DR5	River Don	57 - 82 mm n = 3
2016	DR6	River Sheaf	73 - 86 mm n = 7
2016	DR7	Wyming Brook	54 - 71 mm n = 8

Table A.5 Density estimates ± 95% C.L. of >0+ brown trout at impact and control sites during both the before (2012 – 2013) and after (2014 – 2016) periods. Colours denote EA-FCS abundance classification

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

River Name	Site Identifier	2012	2013	2014	2015	2016
		Before			After	
impact Sites						
River Holme	HO1	9.39 ± 1.3	11.08 ± 0.5	26.89 ± 0.3	26.09 ± 0.4	17.11 ± 3.7
River Holme	HO2	9.3 ± 0.4	8.55 ± 0.3	14.85 ± 0.2	11.66 ± 1.1	7.5 ± 0.8
River Holme	HO3	3.31 ± 0.3	4.47 ± 0.2	2.34 ± 0.4	7.65 ± 0.2	13.61 ± 1.4
River Holme	HO4	12.94 ± 2.6	14.92 ± 1.4	30.66 ± 1.1	11.53 ± 0.5	14.47 ± 4.9
River Holme	HO5	23.31 ± 1.5	14.35 ± 1.6	33.24 ± 1.3	30.87 ± 1	20.55 ± 1
River Holme	HO6	15.83 ± 1.6	15.29 ± 2.9	17.81 ± 0.9	16.95 ± 2.5	19.19 ± 2
Reference Sites						
River Ribble	HR1	3.91 ± 0	10.89 ± 1.1	14.01 ± 2.5	7.2 ± 0.2	6 ± 1.2
River Ribble	HR2	8.87 ± 1	25.54 ± 1.1	15.26 ± 0.3	7.44 ± 0.3	6.67 ± 0.2
River Ribble	HR3	15.78 ± 0.8	11.83 ± 0.2	24.14 ± 1.3	21.05 ± 1	18.22 ± 0.7
River Ribble	HR4	22.9 ± 1.2	19.51 ± 0.6	30.79 ± 0.5	22.66 ± 0.4	28.32 ± 3

Table A.6 Length Range of 0+ Brown trout captured at impact and control sites during the River Holme Study (2012 – 2016). N/S denotes no fish caught

Year	Site	River	Length Range
2012	HO1	River Holme	61 - 94 mm n = 11
2012	HO2	River Holme	58 - 84 mm n = 12
2012	HO3	River Holme	61 - 93 mm n = 16
2012	HO4	River Holme	56 - 85 mm n = 18
2012	HO5	River Holme	52 - 91 mm n = 27
2012	HO6	River Holme	63 - 89 mm n = 37
2012	HR1	River Ribble	63 - 94 mm n = 64
2012	HR2	River Ribble	58 - 104 mm n = 41
2012	HR3	River Ribble	54 - 85 mm n = 20
2012	HR4	River Ribble	47 - 83 mm n = 31
2013	HO1	River Holme	54 - 84 mm n = 7
2013	HO2	River Holme	45 - 98 mm n = 30
2013	HO3	River Holme	53 - 78 mm n = 22
2013	HO4	River Holme	50 - 85 mm n = 34
2013	HO5	River Holme	46 - 93 mm n = 52
2013	HO6	River Holme	58 - 78 mm n = 21
2013	HR1	River Ribble	39 - 81 mm n = 100
2013	HR2	River Ribble	53 - 83 mm n = 37
2013	HR3	River Ribble	54 - 85 mm n = 43
2013	HR4	River Ribble	45 - 85 mm n = 68
2014	HO1	River Holme	61 - 84 mm n = 13
2014	HO2	River Holme	51 - 79 mm n = 9
2014	HO3	River Holme	N/S
2014	HO4	River Holme	67 - 83 mm n = 7
2014	HO5	River Holme	68 - 85 mm n = 42
2014	HO6	River Holme	63 - 84 mm n = 22
2014	HR1	River Ribble	65 - 82 mm n = 12
2014	HR2	River Ribble	57 - 78 mm n = 12
2014	HR3	River Ribble	54 - 85 mm n = 43
2014	HR4	River Ribble	58 - 85 mm n = 34
2015	HO1	River Holme	65 mm n = 1
2015	HO2	River Holme	53 - 74 mm n = 19
2015	HO3	River Holme	47 - 83 mm n = 11
2015	HO4	River Holme	54 - 81 mm n = 13
2015	HO5	River Holme	57 - 80 mm n = 19
2015	HO6	River Holme	53 - 83 mm n = 32
2015	HR1	River Ribble	63 - 85 mm n = 14
2015	HR2	River Ribble	74 - 84 mm n = 12
2015	HR3	River Ribble	60 - 85 mm n = 40
2015	HR4	River Ribble	55 - 85 mm n = 55

Table A.6 continued

Year	Site	River	Length Range
2016	HO1	River Holme	72 mm n = 1
2016	HO2	River Holme	59 - 68 mm n = 4
2016	HO3	River Holme	62 - 78 mm n = 3
2016	HO4	River Holme	55 - 91 mm n = 10
2016	HO5	River Holme	60 - 85 mm n = 10
2016	HO6	River Holme	67 - 76 mm n = 6
2016	HR1	River Ribble	46 - 81 mm n = 30
2016	HR2	River Ribble	62 - 69 mm n = 3
2016	HR3	River Ribble	70 - 85 mm n = 24
2016	HR4	River Ribble	70 - 89 mm n = 10

Table A.7 Density estimates ± 95% C.L. of >0+ brown trout at impact and control sites during both the before (2012 – 2014) and after (2015 – 2016) periods. Colours denote EA-FCS abundance classification

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

River Name	Site Identifier	2012	2013	2014	2015	2016
		Before			After	
impact Sites						
River Washburn	SW1	3.11 + 0.2	0.79 + 0	0.92 + 0.2	0.00 + 0.0	0.79 + 0.0
River Washburn	SW2	2.37 + 0.1	2.14 + 0.3	2.4 + 0.2	0.4 + 0	2.14 + 0.2
River Washburn	SW3	4.22 + 2.8	1.6 + 0.2	0.98 + 0.0	0.00 + 0.0	1.6 + 0.0
River Washburn	SW4	1.93 + 0.6	5.6 + 1.2	6.39 + 0.7	1.58 + 0.0	5.6 + 0.4
River Washburn	SW5	1 + 0.3	7.33 + 1.1	0.65 + 0.2	1.04 + 0.0	7.33 + 0.1
River Washburn	SW6	1.26 + 0.4	3.82 + 0.5	5.91 + 0.3	0.31 + 0.0	3.82 + 0.6
Control Sites						
Barden Beck	WR1	10.33 ± 0.9	10.19 ± 0.3	14.77 ± 0.7	12.8 ± 1.4	8.44 ± 0.8
Barden Beck	WR2	5.6 ± 0.1	6.33 ± 0.1	3.35 ± 0.3	5.14 ± 0.4	6.41 ± 0
Ings Beck	WR3	5.6 ± 0.9	9.48 ± 0.5	10.2 ± 0	7.45 ± 0.1	0.48 ± 0
Ings Beck	WR4	4.5 ± 0.4	4.53 ± 0.2	1.11 ± 0.3	1.72 ± 0.2	1.71 ± 0
Ashfold Side beck	WR5	14.02 ± 0.9	11.56 ± 2.1	25.15 ± 2.0	32.5 ± 0.7	22.99 ± 1.1
Ashfold Side beck	WR6	23.46 ± 2.0	7.31 ± 0.5	29.76 ± 0.2	42.25 ± 3.9	21.3 ± 1.9
Ashfold Side beck	WR7	18.08 ± 0.3	7.94 ± 0.4	21.57 ± 0.6	19.76 ± 0.8	18.87 ± 1.6
Ashfold Side beck	WR8	8.71 ± 1.8	9.49 ± 0.4	0.39 ± 0.0	9.32 ± 0.7	13.7 ± 0.8
Ashfold Side beck	WR9	11.96 ± 1.1	11.36 ± 0	12.7 ± 0.4	12.96 ± 0.3	10.97 ± 1.1