UNDERSTANDING AND ADDRESSING THE ECOLOGICAL IMPACTS OF FLOW IMPOUNDMENT FOR RIVER SYSTEMS IN NORTHERN ENGLAND

By:

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A thesis submitted in partial fulfilment of the requirements for the degree of
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"For in the true nature of things, if we rightly consider, every green tree is far more glorious than if it were made of gold and silver."

- Martin Luther

ABSTRACT

Many rivers have undergone flow modification by impoundments to provide services such as water supply and hydropower. There is an established consensus that typical modified flow regimes do not sufficiently cater to the needs of downstream ecosystems, due to species having adapted to natural flow conditions. This may lead to changes in the biodiversity and functional composition of ecosystems, potentially compromising water quality and other river system services. More must be done to understand the relationship between flow and in-stream ecology, in order to mitigate the impacts of flow modification. The development of efficient methods of ecology-flow assessment is vital in order to meet current and future legislation, whilst considering other stakeholders and maintaining the resilience of the local water supply. This thesis combines statistical approaches applied to public datasets, and combined ecological-hydraulic modelling at a case study site, to propose environmental flow regimes. Univariate and multivariate analyses were performed on flow and macroinvertebrate sampling data from sites across northern England. The mean annual frequency of high flow events was identified as a particularly influential driver of functional composition and biodiversity metrics. Field data was gathered and a hydraulic-ecological model was also developed for a selected case study site in order to predict the responses of selected indicator species to flow. Spatial and temporal distributions of habitat quality with respect to flow were generated, allowing the impacts of various flow inputs to be assessed. These findings were integrated in order to generate recommended flow regimes for the case study site. It was demonstrated that the proposed regimes met or improved upon ecological metrics relative to impoundment outflow data, whilst also conserving significant quantities of water. Outcomes from this research demonstrate the potential of habitat suitability models, supplemented by knowledge of ecological-flow relationships, to inform environmental flow design decisions.

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Abbreviations

ADV - Acoustic Doppler Velocimeter

AMP – Asset Management Plan

BACI – Before/After, Control/Impact

BBM – Building Block Method

EA – Environment Agency

FST – Fliess Wasser Stammtisch (Approximate meaning: Flowing water table)

GDP – Global Domestic Product

GEP – Good Ecological Potential

GES – Good Ecological Status

HHS – Hydraulic Habitat Suitability

HMWB – Heavily Modified Water Body

HSI – Habitat Suitability Index

IHA – Indicators of Hydrological Alteration

OS – Ordinance Survey

PC – Principal Component

PCA – Principal Components Analysis

PHABSIM – Physical HABitat SIMulation

RIVPACS – River Invertebrate Prediction and Classification System

SRH - Sedimentation and River Hydraulics

STAR – Standardisation of River Classification

UK – United Kingdom

UKTAG – United Kingdom Technical Advisory Group

UU – United Utilities

WFD – Water Framework Directive

WUA – Weighted Usable Area

1. INTRODUCTION

1.1 Research context

It is generally recognised that we must manage our water bodies in a sustainable manner so that the ecosystem services of the system are not compromised. Ecosystem services is a term that has been used for some time but was popularised by the Millennium Ecosystem Assessment in the early 2000s, which introduces it in simple terms: "Ecosystem services are the benefits people obtain from ecosystems" (Reid et al., 2005). Ecosystem services are then broken down into the categories of *provisioning services* that provide us with resources such as timber or water; *regulating services* that affect local conditions such as water quality; *cultural services* that promote human wellbeing by, for example, facilitating leisure activities and creating beautiful surroundings; and *supporting services* such as nutrient cycling and soil formation (Reid et al., 2005). Throughout this thesis, when referring to the 'services' of a water body, this is in the context of ecosystem services. These services have increasingly been acknowledged, as scientists have documented ecosystem degradation and the compromising of key functions such as those described above. Many of these services are vital to human wellbeing, and thus their sustainability has increasingly been considered to be a priority (Costanza et al., 1998).

While ecosystem services are the benefits people get from ecosystems, the production of those benefits is supported by a multitude of ecological functions and processes; to maintain such services, a range of ecological elements and functions must be maintained. Maintenance of ecological functions is compromised through many forms of anthropogenic modification of the environment; sometimes even modifications to enhance one service can directly or indirectly impact others. One of the key modifications people make to water courses is flow modification – often in the interests of gaining some particular services. Such modification has potential consequences for ecological function, and therefore potentially for ecosystem services. Thus, forming a better understanding of the effects of flow modification, and how to manage it to reduce impact on ecological function, is a key step in maintaining the functions that support the services that have been described.

Flow modification and impoundment of river systems to meet water resource needs has become widespread throughout the world in response to increasing water demand. A majority of water bodies have been impounded by reservoirs for services such as water provision and hydropower, the United Kingdom having the highest density of impoundments in Western Europe (Lehner et al., 2011). Only in the past few decades have the environmental impacts of river impoundment and flow modification been seriously considered, and only more recently has the understanding of such impacts developed to a point where frameworks and methodologies may be put in place in order to begin quantifying and taking steps towards mitigating these impacts.

Appreciation for the value of ecosystems has resulted in an increasing focus upon improving the ecological condition of water bodies. This has revealed, however, that our current water management strategies have been found wanting. Through legislation such as the Water Framework Directive (WFD) (European Commission, 2000), alongside an increasingly stressed water supply infrastructure (BBC, 2019), it is beginning to be recognised that current flow regimes imposed by impoundments may both be detrimental to ecological health (Poff et al., 1997), and may be releasing unnecessary volumes of water in some contexts (thus spending water resources sub-optimally). Failure to meet

ecological targets, mandated in this context by the WFD, is likely under current flow regimes and may lead to penalties for water managers if measures are not taken to address this issue. Additionally, these traditional flow regimes could contribute towards water shortages, particularly during periods of drought, whilst being of no value to the downstream ecosystem. It is thus important to take steps towards better understanding the relationship between flow regime and ecological response. Such understanding would allow water managers to prescribe reservoir flow releases that promote downstream ecological condition, complying with WFD objectives, whilst doing so in an efficient manner that does not compromise the societal service provided by the reservoir (water supply, primarily). Due to conflicting stakeholder interests present in most riverine systems (Summers et al., 2015), optimising the way in which ecological needs are met is vital; environmental benefit must be maximised relative to volume of water spent by allocating water at magnitudes, timings and variabilities most suited to promote high ecological benefit relative to water expenditure.

The innate complexity of rivers as open systems brings high levels of uncertainty to the study of ecology-flow interaction (Konrad et al., 2011). Such uncertainty makes general ecology-flow relationships difficult to identify and thus manage for; rivers present highly diverse systems and there is no certainty that two different rivers will respond in the same way to management efforts, particularly when considering systems across a broad range of magnitudes (Monk et al., 2006). There is a need to formulate an approach towards the identification of ecologically beneficial flow regimes that moves beyond highly laborious site-specific prescription, whilst also avoiding over-generalised approaches that have a poor scientific basis, such as regimes that simply prescribe a constant minimum flow (such as those criticised by Arthington et al., 2006). The term used to describe flow provision for environmental purposes is "environmental flows". The term is defined by the Brisbane Declaration, 2007: *"Environmental flows describe the quantity, timing, and quality of water flows required to sustain freshwater and estuarine ecosystems and the human livelihoods and well-being that depend on these ecosystems."* (Overton et al., 2014). It is thus the task of water managers, ecologists and other disciplines involved in the field of ecohydraulics to meet the challenges previously described, in order to advance general knowledge and work towards common frameworks by which riverine systems may be assessed and have suitable environmental flows designated in a feasible manner, considering time and resource constraints.

The overarching aim of this investigation is to further understanding in the interactions between flow regime and ecological condition through investigation of the relationship between flows and ecological response; particularly focusing upon how flow regime impacts physical habitat quality through local flow conditions. In this investigation, benthic macroinvertebrates are utilised as indicators of ecosystem response. This is due to the fact that macroinvertebrates are an ideal indicator due to their abundance in most systems, the fact that they are well relatively understood in terms of their behavioural and morphological adaptations, and are widely used as indicators of ecological health (Barbosa et al., 2001). Additionally, macroinvertebrates are often neglected area in environmental flow research (Gillespie et al. 2015b).

1.2 Problem statement

There is a need to understand the impact of modified flows upon the native ecosystem and how flow regime might be altered, in order to mitigate these impacts so as to meet environmental objectives, maintain ecosystem services, and potentially conserve more water. Competing water resource

demands require innovative water management frameworks to meet the requirements of both society and the environment; traditional allocation has been based on volumetric requirements, but progress in the field of environmental flows has revealed that the ecosystem is dependent on other factors such as timing and variation of flow (Overton et al., 2014). There must therefore be a greater understanding in how ecosystem demands can be met at sites subject to flow modification, whilst also acknowledging societal demands for water supply provided by impounded water bodies. Thus, environmental flow needs must be identified and met efficiently, without compromising services provided at these sites. More specifically in a European context, there is also a need to define how "Ecological potential" defined under WFD targets may be quantified with regard to changes in parameters such as localised flow conditions and local physical habitat quality; this also applies to demands of environmental legislation internationally.

1.3 Research questions

1. What specific drivers are eliciting an ecological response?

Rivers are complex open systems in which components cannot easily be controlled or isolated; relationships between individual variables are difficult to quantify. This investigation aims to adapt and bring together recent developments in the field of environmental flows in order to identify key flow characteristics driving ecological response. The aim of this work is to inform future development of compensation flows to optimise ecological provision, particularly in developing a less data-intensive framework to assess a site's flow requirements. This would in turn aid water managers in meeting environmental objectives, and assist in maintaining the ecosystem services of the target water body.

2. How can localised hydraulic requirements of taxa be translated into an ecologically beneficial flow regime?

Taxa experience flow as localised forces as opposed to overall flow magnitudes, timings, etc. How can the requirements of these entities on a micro scale translate into an overall compensation flow and its inter-annual variation? This again feeds into the aim of developing a framework capable of integrating a diverse range of environmental and societal requirements, informing how water managers may move from general principles to more optimised holistic, adaptive management strategies.

3. What indicators can be used to interpret habitat quality, given its variation over time and different values between species?

The requirements of multiple species will be considered when designing an overall flow regime, generating a range of habitat quality metrics; how can these individual predictions be aggregated into an overall assessment of habitat quality? Solutions to this question will aid in flow regime designation and thus transferable mitigation measures, potentially countering ecosystem service degradation, and allowing water resources to be used efficiently within better-informed flow regimes.

4. How may outcomes such as habitat quality predictions be interpreted in light of legislative targets?

Approaches taken in this investigation will provide insights into how the ecology will respond to different flows. However, this does not immediately translate into how a site's flow regime may be

improved for the benefit of the native ecosystem. It is important to consider how the findings can be related to the demands of the Water Framework Directive, in order to provide guidance for water managers.

1.4 Anticipated outputs

The primary output of this thesis is the overall approach towards environmental flow designation that integrates more general ecology-flow trends with the predicted responses of indicator species to flow at a case study site, in order to design more holistic environmental flow regimes. Knowledge of environmental requirements will be generated through predictive hydraulic and ecological modelling, calibrated using field-based investigation, and complimented by a multi-site statistical analysis of flow drivers on macroinvertebrate populations across rivers of similar classifications. These approaches are detailed in later Chapters, and are described more briefly shortly in this Chapter. Findings of this investigation may see practical application with United Utilities (UU), the industrial sponsor for this project, as recommended flow regimes will be proposed for the case study site. The approach used in this thesis should promote further investigation into the methods used, and should it be fully validated, the approach may see application in future environmental flow regime designation.

Chapter 2: LITERATURE REVIEW

Chapter 2 investigates the history and current context of environmental flows, with a particular focus upon the context of flow impoundment impacts upon macroinvertebrates in UK river systems. This Chapter forms the foundations for the scientific basis and methodology of the rest of the investigation. The literature review provides insights into key areas of recent development, neglected areas of study, and current trends and best practices for environmental flow development.

Chapter 3: UNDERSTANDING AND QUANTIFYING THE IMPACTS OF FLOW MODIFICATION IN NORTHERN ENGLAND THROUGH MULTISITE ANALYSIS

Chapter 3 analyses multiple impounded systems in a desk-based multi-site study, drawing relationships between specific flow characteristics and ecological metrics. Conclusions from this study affirm or reject the question of whether or not flow modification causes significant ecological deviation in comparison with natural systems, thus highlighting the importance of greater flow regime naturalisation. Conclusions may also identify key flow drivers that may be focused upon in the environmental flow design framework described in Chapters 4 and 5.

Chapter 4: DEVELOPING A HYDRO-ECOLOGICALLY LINKED MODEL TO EVALUATE AND ADDRESS MACROINVERTEBRATE RESPONSE TO FLOW MODIFICATION AT A CASE STUDY SITE

Chapter 4 describes the justification and process of developing a 2D model by which habitat quality might be predicted at a case study site in response to channel hydraulic conditions. This Chapter affirms the scientific basis behind this approach, both by making reference to the current state of the field, suggestions from past studies, and the use of 2D modelling in other restorative contexts. This Chapter primarily demonstrates the accuracy of the model by describing the calibration process of both hydraulic and ecological models, their testing, and their subsequent application, feeding into Chapter 5 both by providing a basis from which various flow regimes can be assessed in terms of

ecological response (through changes in predicted habitat quality) and by providing evidence for the reliability of said responses.

Chapter 5: ADDRESSING IMPOUNDMENT-RELATED FLOW MODIFICATION AT A CASE STUDY SITE BASED ON HYDRO-ECOLOGICAL MODEL OUTPUTS AND ECOLOGICAL PRINCIPLES

Chapter 5 utilises the 2D model developed in Chapter 4 to assess the impact of a range of flow magnitudes and flow regimes upon habitat quality for a variety of macroinvertebrate indicator species. Temporal ecologically-relevant drivers such as flow event duration and frequency discussed in Chapter 3 are also considered. All analyses performed throughout the investigation, along with considerations of impoundment and water resource limitations, are integrated into proposed flow regimes with varying prioritisations between ecological provision and stakeholder interests. Proposed regimes are compared with historical reservoir outflows, and the implications of the new 'designer flows' are discussed.

Chapter 6: GENERAL DISCUSSION

Chapter 6 discusses the findings of previous Chapters, the implications of the overall work of this thesis both in terms of environmental provision and societal services, and possible future work going forward from this proposed methodology. Of particular interest is transferability of approaches used in this thesis to other river systems, and how the methodology might be adapted and scaled up when applied to more complex, larger systems. Thesis implications in light of legislation such as the Water Framework Directive are also forms a significant part of this Chapter.

Chapter 7: RESEARCH SYNOPSIS

Chapter 7 summarises the entire thesis, demonstrating research highlights and novelty, briefly discussing key outcomes and implications, and discussing extent to which the initial goals or research questions posed have been met.

2. LITERATURE REVIEW

Chapter overview

This Chapter investigates the history and current context of environmental flows (defined in Section 1.1), with a particular focus upon the context of flow impoundment impacts upon macroinvertebrates in UK river systems. The work described here forms the foundations for the scientific basis and methodology of the rest of the investigation. The core concepts relating to flow modification and restoration are discussed, an overview of the relevant legislation in a European context is also described along with how this drives academic and industrial action in the field. Lastly, the challenges faced when moving to the conceptual to the implementation stages of site restoration are discussed, and current gaps in research are identified.

2.1 Contemporary research context

The implications of flow modification are manifold, yet understanding of how flow may drive the ecological health of a river system has only relatively recently became a topic of wider interest. Flow, particularly in terms of magnitude, has been described as the "Master Variable" within the river system due to its direct influence upon river morphology, river habitat, sediment and nutrient transport, and physical forcing upon biota (Power et al., 1995). The brief review of literature that follows provides the background to concepts used in this project, and summarises current scientific understanding of this area.

Traditionally, reservoir impoundments have released a constant flow in order to sustain downstream river systems. Water managers have generally used a continuous prescribed compensation flow, for example Q95; a discharge value that is equalled or exceeded through 95% of the flow record. These approaches generally originate from historical commitments to downstream interests such as mills (many of which no longer exist), or from unfeasible demands placed upon ecologists and water managers to rapidly establish precise environmental requirements (Arthington et al., 2006). Such flows have no scientific basis with regard to ecological health, and do not consider a site's specific requirements and character. Current scientific knowledge suggests that such flows are very likely to result in ecological degradation due to the behavioural and morphological adaptations of species to natural flows, and the physiochemical regulatory role of variable flows within the riverine system (Poff et al., 1997, Bunn and Arthington, 2002, Alcazar and Palau, 2010), discussed in further detail in Section 2.2.

One of the most pertinent questions currently is how to bridge the gap between these unreliable "rule of thumb" principles and adaptive site management strategies which, whilst conceptually sound, are intensive, laborious and highly site-specific. It has previously been proposed that giving sites of similar characteristics a particular classification, and developing general trends within these classes, may serve as a middle-ground between over-generalised "rules of thumb" and very laborious site-specific investigation (Arthington et al., 2006). Such an approach could allow water managers to begin to work towards meeting legislatory targets in an economically feasible manner without the need to thoroughly assess every impacted site.

2.2 Fundamental concepts

2.2.1 Biological morphological and behavioural adaptation

It has been acknowledged that biota and the natural morphology of the river channel rely upon a river system's natural variation in order to be maintained; the flow variation introduced by predictable seasonal precipitation levels, or snow melt, are examples of this (Junk et al., 1989, Junk and Wantzen, 2004b). It has been stated that there is an intrinsic link between the natural flow regime and in-stream ecology; biota have developed life-history, behavioural and morphological adaptations in order to succeed within natural conditions that species have existed in for thousands, if not millions, of years (Poff et al., 1997, Lytle and Poff, 2004). When natural variation is removed, the behavioural and morphological adaptations of biota that previously aided in survival may in fact become a hindrance (Lytle and Poff, 2004). An example of behavioural adaptation is the timing at which eggs are laid or when eggs hatch, such events as hatching and recruitment and called life history events by ecologists and are generally adapted to coincide with seasonal patterns of flow in order to provide the most preferential conditions for early life stages of a species (Lytle and Poff, 2004).

Population numbers can also vary greatly season by season due to life histories leading to different life stages or behavioural patterns, with prolific numbers at certain times of year, and little to no presence in the river during other times (Beltran Epele et al., 2011, Raddum and Fjellheim, 1993). When the flow regime is altered, these seasonal patterns may be lost (Poff et al., 1997), leading to greater juvenile mortality due to hatching during non-ideal flow conditions (Lytle and Poff, 2004). In the context of environmental flows, it is thus important to assess the seasonality of target species within a river system prior to making flow allocations, so that flows will be relevant for biota present at the time of release. Temperature also has a major influence over species life history and seasonality, and is discussed in a Section 2.2.4. Another simple example of morphological adaptation is vegetation that develops brittle branches that will be broken in flood conditions, aiding downstream seed proliferation; in the absence of floods this morphology is of no advantage and simply results in a loss of biomass for the plant (Lytle and Poff, 2004). When their adaptations become non-advantageous, specialised native biota may be pushed out by more competitive generalist species, be they native or invasive (Lytle and Poff, 2004).

2.2.2 Influence upon sediment transport and associated impacts

Modified flows can prove disruptive to the natural supply and transport of sediments within a catchment network (Petts and Gurnell, 2005). Generally, it is expected that erosion beneath an impoundment will be enhanced due to a lack of sediment supply, and sediment deposition may be enhanced further downstream due to a lack of high flows that can flush out the system. Flow is the primary driver of the sediment regime and thus moving towards a more heterogeneous flow regime with incorporated disruption events would be expected to in turn bring greater naturalisation to sediment transport and improve ecological conditions through flushing and deposition events. Wood and Armitage (1997) report on the deleterious effects of fine sediment upon biota. Natural flows have significant variability; from the extremes of flooding to low flows and drought. In turn this brings about high variability in suspended solids concentration and sediment deposition, with a general pattern of erosion upstream and deposition downstream through longitudinal connectivity. This process may be disrupted when connectivity is broken and the flow regime is altered, for example by the presence of

a reservoir. In the event of a loss of connectivity, sediments from upstream are likely to be deposited behind the reservoir within the calm waters, whilst the reservoir releases clear, "hungry" water that has a high capacity for erosion due to its low sediment load and high excess energy (Kondolf, 1997). Thus where a reservoir has been installed, one may expect to see enhanced deposition behind a reservoir (and a gradual lowering of the reservoir's capacity), whilst erosion below a reservoir is enhanced leading to channel incision. Downstream of this enhanced erosion, deposition may be enhanced once more as the lack of high flows leads to deposited sediment from upstream and from tributaries not being re-mobilised by natural high flows. This may lead to problems associated with fine sediment (Kondolf, 1997).

Wood and Armitage (1997) identify the principal impacts of excess sediment deposition as loss in primary productivity, and faunal diversity and abundance. Most benthic macroinvertebrates have developed a resistance to short-term increases in suspended and benthic sediments. However, ongoing sediment regime modification may lead to severe consequences for benthic faunal communities. Suitability of the substrate for particular taxa may be altered by the penetration of fine sediments (Richards and Bacon, 1994); drift may also increase due to changes to the river substrate's characteristics and reduced habitat availability (Culp et al., 1986) as well as scouring due to suspended solids (Bilotta and Brazier, 2008); respiration may be affected by the presence due to silt deposition, affecting respirator structures and lowering oxygen concentrations due to oxygen being lost from the organic component (Bilotta and Brazier, 2008); finally, feeding activities may be adversely affected, either through impeded filter feeding or in reductions to the value or density or prey items (Wood and Armitage, 1997, Graham, 1990). In Wood and Armitage's study (1997), reviewing the findings of other research, it is suggested that the sediment issues most associated with compensation flows (and thus of most interest to this project) are silt deposition, siltation, and the infiltration of fines into the river substrate, as well as the general problem of sediment starvation that brings about issues such as enhanced erosion below the reservoir, coarsening of bed material, and the loss of spawning areas (Kondolf, 1997). Sediment starvation is a separate issue to flow as this relates to the reservoir structure obstructing river connectivity, and would need to be dealt with through sediment management strategies. The issue of fine sediment, however, could be somewhat mitigated through the incorporation of high flow disruption events into an environmental flow regime.

2.2.3 Hydraulic parameters

Flow cannot be characterised by a single general parameter such as magnitude when considering its effects on individuals; water is a dynamic fluid with properties that vary spatially and temporally when flowing down a river channel. Physical entities within the river system such as the bed, the channel and biota do not experience a "magnitude" of flow upon themselves but rather experience the changing velocities, depths and turbulence that come with varying volumes of flow. Therefore, it is important to understand which components of flow have most influence upon biotic response. In previous studies, ecologically relevant hydrological flow variables have been identified using literature grounded in concepts such as Indicators of Hydrologic Alteration (IHAs) (Monk et al., 2006, Richter et al., 1996). IHAs break flow data down into various components such as frequencies and durations of flow events, and indicators of flow magnitude over time. In order to avoid redundancy amongst variables, Principal Component Analysis (PCA) or correlation analysis may be ran in order to discount variables with overlapping explanatory power (Monk et al., 2006, Gillespie et al., 2015a, Chinnayakanahalli et al., 2011).

It has been suggested that macroinvertebrates are not primarily affected by magnitudes of flow forces, but rather by the turbulence of flow and local variation of turbulence about their habitat (Blanckaert et al., 2013). Unlike sediment particles that are much denser than water, which are only dislodged when forces exceed a given threshold, macroinvertebrates have a negligible submerged weight; they do not resist flow through mass alone. Macroinvertebrates instead resist dislodgement through a variety of morphological and behavioural traits that allow them to withstand or avoid flow forces; for example seeking out heterogeneous areas of riverbed that provide areas of shelter from the force of the flow (Blanckaert et al., 2013). Blanckaert et al. (2013) suggest that habitat models intending to reflect physical-biological coupling should focus upon two areas; (i) local substrate heterogeneity and its ability to "hide" invertebrates; (ii) peaks in flow forcing and the temporal variability of flow relating to the dominant turbulent structures. Other studies relating benthic ecology with flow suggest that longer-term bed shear stress is the primary driver of ecological response, particularly with regard to how these forces act as a control upon benthic habitat (Statzner and Muller, 1989, Schwendel et al., 2010). A case study has examined the impact of shear stress and other hydraulic parameters upon invertebrate distribution; shear stress was found to be one of the dominant parameters relating to taxon richness without accounting for the temporal variation described by Blanckaert et al. (2013) (Merigoux and Doledec, 2004). This investigation is more concerned with the general long-term relationships between ecology and flow, and thus the near-bed forces described by Statzner and Muller (1989) and Merigoux and Doledec (2004), amongst others, will be of primary concern here. The insights from Blanckaert et al. (2013) highlight, however, that invertebrates are sensitive to particular aspects of flow, and such features should be taken into account by water managers.

Monk et al. (2006) observes that flow magnitude is a dominant driver of ecological response when considering river systems across a broad range of scales. This is likely due to the fact that the magnitude of flow relates closely to flow velocity. Flow (m^3/sec) is calculated by velocity (m/sec) multiplied by area (m^2); changes to magnitude therefore correspond with changes to the flow velocity, which in turn corresponds to changes in bed shear stresses that benthic taxa experience. In addition to the flow magnitude's influence on channel hydraulics, it also exerts indirect influences upon biota; a reduction in flow can lead to increased species vulnerability to pollution due to lessened dilution of contaminants (Withers et al., 2011), and less flow may also lead to a smaller wetted area and depth within the channel, reducing connectivity and impacting species mobility (Lin et al., 2018, Shaw et al., 2016). Using the Richter et al.'s (1996) Indicators of Hydrologic Alteration (IHA) concept, there are a number of other regime characteristics that may be measured through specific statistical analysis of flow time series. Characteristics include magnitude of monthly conditions, magnitude and duration of annual extremes, timing of extremes, frequency and duration of high and low flow pulses, rate and frequency of water condition changes (Richter et al., 1996). Natural characteristics such as high and low flows are an integral part of ecologically beneficial regimes (Acreman et al., 2014). As can be seen from these variables, there is far more to be discerned from a flow time series than outright magnitude.

2.2.4 Temperature

The thermal regime of a river is defined by the distribution of temperature magnitudes within the system over time; the frequency with which particular temperatures occur, the time of day/year when certain temperatures occur, and the duration for which a stream is above or below a certain

temperature. Similar to a river's flow regime (Poff et al., 1997), thermal regime can be "split into the components of magnitude, frequency, duration, timing and rate of change" (Olden and Naiman, 2010). Much like the flow regime, biota have adapted to the natural diurnal and seasonal variation of the thermal regime in natural systems; for example the timing of development cycles to coincide with favourable seasonal temperatures (Olsson, 1982). Seasonal fluctuations in temperature are a major ecological driver, leading to flourishing ecological systems in warmer seasons, and a more dormant state in some systems during the cold of winter (Olsson, 1982). Flow modification may compromise this natural variation, impacting the ecological integrity of the lotic system (Olden and Naiman, 2010). Water temperatures have a direct impact upon the growth rates of river biota and have an influence in shaping species distributions. It has been mentioned that local populations have behavioural adaptations intended to synchronise life history events with particular trends in the thermal regime (Vannote and Sweeney, 1980); specific examples of this include thermal cues stimulating fish migration, spawning and hatching, whilst also directly influencing the survival and development time of eggs. Of primary interest to this study, changes to the thermal regime also have significant impact upon invertebrate communities. Key developmental cues can be eliminated by shifts in this regime, and the rate of egg development and juvenile growth can be hindered (Olden and Naiman, 2010).

Olden and Naiman (2010) discuss specifically the effect of river impoundment upon the riverine thermal regime. Of course, the extent of impact upon the thermal regime is largely dependent upon the operation and mechanism of water release by an impoundment. The volume of water being released into the system, the rate and frequency of release, the depth at which water is being drawn from and the size and depth of the reservoir all influence the downstream thermal regime. Stratification and depth of draw-off are of particular importance in determining the magnitude of influence these flow releases have. When reservoirs are of sufficient depth for stratification to occur, draw-off from below the thermocline can lead to releases of water significantly cooler than that within the natural stream system (Poole and Berman, 2001). Indirect influence is also exerted upon stream temperatures by changes to the flow regime; for example, alterations to discharge and stream volume change the rate at which water heats and cools through diurnal heat exchange (Petts, 1986). Though thermal impacts of impoundment-related compensation flows are well recognised, such impacts are examined much less often than outright hydrological impacts (Olden and Naiman, 2010). This may be partly due to the significant variance in thermal effects between sites, which can vary based on the position of the impoundment, mode of reservoir operation, release depth, and additionally environmental and geomorphological characteristics of the area (Olden and Naiman, 2010).

2.3 Recent approaches to flow regime design

In recent decades a number of important concepts concerning river structure and function have been advanced, in particular the Natural Flow Regime Paradigm (Poff et al., 1997), the River Continuum Concept (Vannote et al., 1980) and the Flood Pulse Concept (Junk et al., 1989). The Natural Flow Paradigm discusses the consequences of the modification of various flow components such as loss of flow heterogeneity, and the resulting ecological response within the system. The paradigm expands upon the concept that natural flows promote stable ecosystems, whilst over-regulated systems result in ecological harm due to direct and indirect response to flow (Poff et al., 1997). Specific reasons behind this will be discussed shortly. The River Continuum Concept discusses rivers as an open system, particularly regarding their longitudinal connectivity. Within natural systems, a river is generally well-connected to upstream areas, allowing for the transport of sediments and nutrients, in addition to

free migration of biota up and downstream. In regulated systems with structures such as dams and weirs this connectivity is disrupted, resulting in ecological harm as species migration is hindered and sediment/nutrient transport is disrupted (Vannote et al., 1980). The Flood Pulse Concept discusses the lateral connectivity displayed by natural river systems. This connectivity is usually brought about by flow variation and flooding events (i.e. a flood "pulse"), as parts of the flood plain undergo periods of inundation and drought. Not only does this periodic wetting maintain flood plain habitats, it also allows nutrients and sediments from the flood plain to be carried back into the river channel. It is suggested that flow modification due to impoundment may prevent such inundation from occurring (Junk et al., 1989, Junk and Wantzen, 2004a). This may bring about ecological harm both on the flood plain due to lack of habitat maintenance and wetting, and also in the channel due to the removal of a source of sediment and nutrients, as the natural pattern of inundation and recession "drags" resources from the floodplain into the channel (Junk et al., 1989).

Moving beyond the natural flow paradigm, the 'designer paradigm' is increasingly seen as an appropriate mitigation solution for modified systems; this paradigm acknowledges that full naturalisation of a heavily modified system is unfeasible due to the need for it to be utilised for societal services. Flows are instead designed around promoting positive and desired aspects of the ecosystem (such as native biodiversity) whilst inhibiting undesirable aspects (such as invasive species or algal blooms). This approach aims to develop efficient water solutions which intelligently allocate flows between conflicting interests for optimum benefit (Chen and Olden, 2017, Acreman et al., 2014). The 'designer paradigm' is most relevant to heavily modified water bodies, and therefore such an approach is being taken in this investigation. The pressures on riverine ecosystems described here also compromise a waterbody's services, in turn being detrimental to human well-being due to a decline in services (Overton et al., 2014). Therefore, the issues associated with impoundment do not merely represent a threat to native ecosystems, but over time may compromise water quality and other services provided by the water body, reducing the value of the river system and potentially posing a risk to human health. As such, the drive to improve ecological well-being is as much motivated by societal interest as it is by environmental conservation.

2.4 Legislative implications: The Water Framework Directive

Environmental legislation in recent decades has provided impetus for the restoration of environments impacted by anthropogenic activity. Most relevant to waterbodies within Europe is the Water Framework Directive (WFD), introduced in October 2000 (European Commission, 2000). The WFD implemented a classification system for water bodies, taking into account ecological as well as chemical quality. The goal for all member states under the WFD is to meet Good Ecological Status (GES) for all natural waters. GES is defined as:

"The values of the biological quality elements for the surface water body type show low levels of distortion resulting from human activity, but deviate only slightly from those normally associated with the surface waterbody type under undisturbed conditions." (European Commission, 2000).

In the case of waters being heavily modified in order to provide services, where complying with the standards of GES would be unfeasible, a different standard is set. Such waters are designated as Heavily Modified Water Bodies (HMWBs), and must instead comply with Good Ecological Potential (GEP). GEP is met when all proportional measures have been taken to bring the HMWB in line with

natural conditions. If the HMWB still deviates from close-to-natural conditions, it must be justified in that any further efforts towards restoration would be disproportionate, either economically, environmentally, or due to detrimental effects on the HMWB's primary service function (UKTAG, 2008). According to the Environment Agency, all relevant mitigation measures must be in place provided that these measures are practical in accordance with a given HMWB's characteristics, and do not have significant detrimental impact either to the water body's use or the wider environment (Environment Agency, 2011).

With the significant and growing amount of literature supporting the natural flow paradigm, and the pressure to meet ecological objectives, it may be seen as surprising that water managers and legislators have yet to adopt this principle as a key aspect of water resource management. Moving from conceptual to adoptable practice has proven to be a significant challenge, however. There are significant issues that must first be addressed before the natural flow regime concept can be well-implemented into management practices. Firstly, it is unfeasible to restore all of our water bodies to natural flow conditions as society is reliant upon the services that these waters provide. Thus, the question has been, is there a compromise in which natural flow can be partially restored in order to provide ecological benefit, without disproportional economic and service cost? It has been suggested that some modifications to flow release, by considering key ecological requirements, could provide an acceptable compromise; an example of this would be the Building Block Method (BBM); BBM proposes to introduce a level of intra-annual variation corresponding to the flow needs of native biota. King and Louw (1998) outlined BBM, which originated from a series of workshops in South Africa in which experts worked together to form recommended environmental flow regimes in a rapid manner, relative to previous approaches. The core feature of the method is the use of a series of workshops by which experts from multiple disciplines are brought together in order to "build" a flow regime based upon expert opinion and knowledge of individual flow requirements ("blocks") within the system (King and Louw, 1998). A recent report by the UK Technical Advisory Group (UKTAG) put BBM forward as suggested best practice in UK flow regime management, citing it as necessary to achieving ecological objectives within the United Kingdom (UKTAG, 2013).

2.5 Challenges in environmental flow implementation

Whilst BBM has been cited as best practice in UKTAG guidance, it has not yet been taken up as standard management practice by water managers, or made a legislative requirement by authorities such as the Environment Agency. Though BBM has been demonstrated to be a useful tool (King and Louw, 1998), it is also highly resource-intensive and site-specific, using a large number of experts who are limited in availability and number. This method therefore, while appropriate for attempting to rapidly restore important systems, cannot feasibly be applied to hundreds of sites on an individual basis. Therefore, how to best design flow regimes, and how this varies site-by-site, remains a pressing issue in the fields of ecohydraulics and environmental flows. Rivers present an extremely complex subject of study, being, as they are, open systems. A river is affected by several seemingly innocuous factors, due to connectivity to upstream parts of the system and the surrounding floodplain. Thus, changes in land use around the headwaters of a river may have significant detrimental impacts upon the downstream river system (Saunders et al., 2002). In addition to anthropogenic factors, the tributary contribution, geology, climate, and many other natural factors also lead to no two rivers being completely identical. This represents a major challenge to flow allocation, as designed flows

must either be tailored specifically to a site, or a transferable framework must be developed which allows for the application of general principles to rivers of certain classes.

As yet, no such framework has become standard practice within the field of ecohydraulics. Poff et al. (2017)'s update on the evolution of environmental flow science discusses growth of the field in almost every area, yet there is not yet a unified approach to environmental flow assessment. They also emphasise the need to extend from a local scale to basin-scale perspective (Poff et al., 2017). Previous regional frameworks have been attempted, but generally have used singular metrics of habitat suitability (Ceola and Pugliese, 2014). More recently, a framework for the strategic allocation of water in order to balance environmental flows and societal needs has been proposed (Sabzi et al., 2019), but within the scope of generalised "environmental considerations" which must be determined on a case by case basis. Arthington et al. (2018) discuss recent advances in environmental flows science, again emphasising the emerging focus upon regional consideration over local-scale solutions. A number of other key areas for advancement are detailed, including the use of a broader suite of ecological metrics to better assess the success of environmental flows (Arthington et al., 2018).

In addition to the open nature of the system, a river also tends to be intrinsically tied with societal services, such as water provision or recreation, and thus experimentation within the river system is limited by the fact that flow manipulations are constrained within a social context in which many stakeholders hold an interest in the state of the flow regime. Konrad et al. (2011) discusses the many challenges posed to large-scale flow experiments. He and other studies stipulate that new approaches must be adopted in order to adapt to the unique nature of rivers as a system to be studied (Summers et al., 2015, Konrad et al., 2011, Poff et al., 2003) due to the fact that many principles upheld by classical experimentation such as replication, randomisation and control of variables are not always feasible in the context of a large-scale flow experiment. Another issue that has been commented on is that such experimentation rarely has post-implementation monitoring in place where restorative measures proposed by a study can be fully validated (Gillespie et al., 2015b); this is due to the fact that ecosystem change can be a slow process, taking months to years, and few studies have been allocated the time or resources to investigate the long-term implications of flow change. Such monitoring would be of significant value in order to quantify the impact of proposed water management solutions. Additionally, societal interests add further complication to the environmental flow design process. As has been mentioned, there is a need to optimise flows, not simply meet ecological requirements. Increasing demand for water supply, the need for water security, and the fact that water is a profitable resource for utility companies mean that environmental needs are a contentious topic; water sent downstream for environmental purposes must be well-justified, and the "cost-benefit" in terms of water committed to environmental flows must be highly optimised in order to maximise the volume of water retained for societal interests (Harwood et al., 2018).

The need to adapt methodologies to optimise environmental flow solutions and account for system complexity has been demonstrated by the fact that most previous studies in this field have not provided data that can contribute to a general knowledge of ecology-flow interaction due to a lack of standardised method and high site specificity; this has been commented on in other reviews and studies (Arthington et al., 2006, Gillespie et al., 2015b). However, such studies have proven to be lessons in how methodology may be improved. Literature reviews provided by the likes of Poff et al. (2010) and Gillespie et al. (2015b) have used such studies to highlight why we cannot yet form general principles for implementing natural flow regimes, and where further research should be directed

towards. As knowledge in how approach and method may be improved has developed, it is becoming more likely that we will see a synthesis of future literature that might provide general knowledge and principles in ecology-flow interaction. From this, the practical application of the designer flow regime concept, and the implementation of transferable designer flow frameworks, may begin to take place. It has been suggested that the diverse influences of riverine ecology must be studied both through short-term mechanistic experiments and long-term explanatory studies in order to disentangle this complex web of interactions (Laini et al., 2018). Climate change and land use changes are also resulting in a shifting environment in which sustainable management of freshwater systems and security of water supply are all the more pressing (Li et al., 2018).

Few studies have actually monitored ecological quality within modified water bodies in a concerted effort to increase general understanding of ecological response to modified flow. Many of the current studies suffer from inconsistencies and potentially flawed methodologies (Gillespie et al., 2015b, Konrad et al., 2011, Summers et al., 2015). Because of this, attempts to synthesise current literature in order to draw out general trends have so far been unsuccessful (Gillespie et al., 2015b, Poff and Zimmerman, 2010). There is not yet reliable evidence beyond conceptual expert opinion that methods such as BBM will provide sufficient ecological improvement to heavily modified waterbodies to meet GEP, or that solutions in less modified systems will meet GES. There is a great need to increase general understanding of ecology and flow interaction, and for a quantified demonstration of the ecological impact of flow regimes post-implementation. This could inform management decisions in a broad practical context and allow water managers to move away from "rule of thumb" compensation flows such as Q95, into approaches which have a sounder scientific and evidential basis.

There are a number of interacting variables that might influence ecology and river conditions; principal drivers being flow alteration, deterioration of water quality, habitat degradation or modification, invasive species, and over-exploitation. Summers et al. (2015) present these drivers as an interacting "web" of pressures. This has presented an immense challenge that investigators are struggling to overcome; hydrologists and ecologists must identify new and suitable methods capable of better isolating key drivers, or mitigating the influence of external drivers, in order to draw quantitative relationships between flow regimes and biological metrics. In addition to the issues involved with the complexities of the open river system, there are also the challenges associated with the highly interdisciplinary nature of ecohydraulics, and the lack of centralised governance for river systems in most countries. This leads to disparate groups who are not necessarily in communication having responsibility for various aspects of water management and study, in addition to other stakeholders with potentially competing interests (Harwood et al., 2018). This adds a sociological aspect to flow allocation which cannot be neglected. "Integrated Water Resources Management" (IWRM) is the method by which water needs are balanced across competing interest groups; an early IWRM approach was that of the "Minimum Flow", an approach dating back to the 1990s where environmental flows were defined as the minimum downstream flow requirement to sustain ecosystems (Overton et al., 2014), whilst more recent work has moved towards the designer paradigm to efficiently balance societal and environmental interests.

The primary challenges currently faced within the field of environmental flows are structural limitations and political decision-making (e.g. how river quality is assessed and values, allocation of resources, regulatory pressures). How environmental services are valued is still an area of contention. Provision for the environment and ecosystems can be neglected when assessment focuses upon

maximising productivity or metrics such as impacts to global domestic product (GDP); such metrics inevitably undervalue the services provided by the riverine ecosystem both in terms of maintaining water quality and through the services they provide to human well-being (Overton et al., 2014). Major issues in maintaining water resources sustainability are predicted in the coming years if innovative solutions and a shift in perception of the importance of ecological sustainability are not brought about (Warner, 2014). This is an ever-evolving field, however, and there is a growing framework on how studies should approach these investigations, with a call for studies to standardise their methodology in order to allow synthesis into a more general knowledge base (Gillespie et al., 2015b, Summers et al., 2015, Konrad et al., 2011, Poff et al., 2003).

2.6 Current advances

As can be seen in Section 2.5, researchers are aware of the challenges inherent to the implementation of environmental flows. A number of studies have attempted to mitigate or resolve these challenges, and key examples are discussed below.

2.6.1 Trait-based analysis

Morphological and behavioural adaptations, or 'traits', of species within a system provide insight into the environment they are populating; species may be sorted into functional groups according to their traits. Functional diversity within a system may aid in determining whether the local ecosystem is healthy and balanced, and whether there are any significant issues driving out particular biota. Trait-based analysis can therefore act as an indicator of ecological health, particularly in modified systems in which their natural counterparts would be expected to host a variety of functional groups (Petchey and Gaston, 2006). Assuming no unusual natural biological filters are present, more functionally homogeneous systems may therefore suggest that the local environment has been modified; an example of this could be a polluted river in which only pollution-resilient species may survive. The functional composition of a system may provide insight into what pressures may be driving ecological response. The limits of the statistical analysis of species populations alone has been criticised for some time; particularly its difficulties in accounting for key ecological phenomena, restricting its ability to reveal the role of significant community-shaping drivers (James and McCulloch, 1990). As such, the trait-based approach, a method that has been emerging over the past decade or so (Petchey and Gaston, 2006), may be used to better provide insights into the ecological system present at a site (Ings et al., 2009, Alexandridis et al., 2017). This approach is based upon the Emerging Group Hypothesis (EGH), in which functional equivalence within groups of species is assumed, should they share particular traits (Lavorel et al., 1997). This method was used successfully to predict benthos responses to environmental change in the recent study by Alexandridis et al., (2017). Trait-based analysis allows for meaningful implications to be drawn on how a characteristic of flow may be affecting the ecosystem; for instance if size of biota is clearly being influenced by a certain aspect of the flow regime, one may draw out the possibly mechanisms that are causing the ecological response. An example of this could be that flow event frequency affects size, and a possible mechanism could be nutrient availability (or access to said nutrients) being altered by this particular flow event. In order to perform a trait-based analysis, one must first obtain data on the traits inherent to particular species. Such data may be obtained from species databases such as those found within the STAR (Standardisation of River Classification) Project, "Deliverable N2" (Bis and Usseglio-Polatera, 2004), described in greater detail in Section 3.2.

2.6.2 Habitat heterogeneity

Ecologists overwhelmingly support the concept of heterogeneity being the foundation of diversity, the idea being that a varied habitat mosaic will be occupied by varied biota. Theory on this is discussed by Ward et al. (2002) and Wiens (2002). Ward et al. (2002) discusses biodiversity in riverine landscapes. The study discusses the confidence ecologists have that habitat heterogeneity has a strong influence upon ecological composition, but explains that this is difficult to quantify *"due to complex interactions between disturbance regimes, spatial heterogeneity and biodiversity in riverine landscales"* (Ward et al., 2002). As such, more study is required in order to quantify these complex interactions. Wiens affirms the importance of habitat diversity, stating; *"Overall patterns of biodiversity that occur within riverine systems reflect organismal responses to landscape structure."* (Wiens, 2002)

More contemporary studies have also utilised the theory of habitat diversity as a key influence upon ecological composition and distribution:

A study by Dunbar et al. (2010a) highlights biodiversity as an important aspect of the river system, demonstrating that modification of river morphology (bed and bank in the case of this study) lead to a more significant response of macroinvertebrates to low flows (Q95). The study suggests that taxa with particular flow requirements must have their habitats preserved in order to maintain a stable ecosystem (Dunbar et al., 2010a).

Miller discusses how habitat heterogeneity has long been assumed as beneficial, but is not well studied in the case of macroinvertebrates, stating that macroinvertebrates are "...only recently receiving attention despite having a critical role in maintaining the stream ecosystem." The study found that a lack of good quality pre- and post-monitoring data for habitat restoration projects limited the study's ability to draw robust conclusions on macroinvertebrate response to restoration projects. That is, the paper assumes that such heterogeneity is a benefit, but we do not yet have the ability to predict and quantify a specific ecological response through meta-analyses due to a lack of rigorous, standardised study design and a lack of quality pre- and post- monitoring data for restorative studies (Miller et al., 2010).

Feld et al. (2014) examined benthic diversity along a hydromorphological gradient of alteration through a variety of different metrics. The study concluded that taxa were lost with increasing alteration but were often replaced by other taxa of similar function, leading to no great change in overall "diversity". Flow and habitat modification are thus clearly having an impact on the ecology, but this does not show up in certain metrics. Feld et al. state that an effort is needed to develop novel indicators of biodiversity, as current indicators do not account for redundancy of traits when they are present within the system (Feld et al., 2014).

2.6.3 Classification and regional-based analysis; improving transferability

The issue of site-specificity within many studies, and the need to increase transferability and synthesis of future research within the field of environmental flows and ecohydraulics (Konrad et al., 2011, Gillespie et al., 2015b) has led to the proposal of classification-based studies as a near-term solution to this challenge in order to move forward in conceptual development. The Water Framework Directive itself promotes river classification as a first step to defining reference communities for

specific river types. A number of ways by which river systems may be classified have been proposed, and numerous approaches were proposed throughout the early 2000's, as described by (Olden and Poff, 2003), attempting to group river systems into characteristics by which they may be compared. In 2006, Monk et al. observed that the "Magnitude" category of flow had the strongest influence over macroinvertebrate response, noting that assessing rivers by magnitude class potentially offers a means of better analysing the influence other variables when magnitude is not allowed to be the primary driving force. This in turn would help to inform management of fluvial systems, and principles could be transferred between sites of similar size and geological characteristics (Monk et al., 2006).

In addition to classification by flow, river systems can vary significantly due to geography through the influences of local geology, climatic conditions, and native taxa. Gonzalez and Garcia (2006) emphasise the importance of regional classification, listing possible classification levels and criteria such as size, geology, channel morphology and native vegetation type, demonstrating how Spain might be divided into nine distinct ecoregions (Gonzalez and Garcia, 2006). Regional-based analysis was also proposed by Arthington et al. (2006) as a method by which conceptual understanding may progress through transferable frameworks, and this was further developed by Alcazar et al. (2010) who applied this approach in order to propose regionally applicable environmental flows for regulated rivers within Spanish catchments. Regional flows were successfully proposed based upon several regional-based flow characteristics, and the study concludes that the methodology presented was a key outcome by which other studies might continue to build upon and develop understanding of regional and classification-based methods and application (Alcazar and Palau, 2010). Classification-based analysis remains a compelling approach by which to advance conceptual understanding of environmental flow designation, and currently represents the most efficient and feasible manner by which flow impoundment impacts might be mitigated, through environmental flows that may be applied across numerous systems of similar characteristics. There is still significant research required before such approaches become standard practice, and any studies capable of identifying ecology-flow relationships for a particular region would present a significant contribution to the broader field.

2.6.4 Hydraulic and ecological modelling

Computer modelling of riverine systems and their associated hydraulic characteristics is common practice in fields such as flood risk management due to a model's ability to modify flow inputs in a controlled environment and predict the outcome. Modelling has been increasingly utilised in the field of ecohydraulics for this reason (Schneider et al., 2016). Hydraulic and ecological modelling packages are key tools in the assessment of downstream impacts of flow modification, due to their ability to predict hydraulic conditions and expected ecological response outside of available observed data, without the need for intensive in-field experimentation. This is achieved through the input of a given flow or time series of flows, which is then computed into channel hydraulics such as velocity that subsequently inform ecological predictions such as habitability for a given species. 1D, 2D or 3D models may be utilised depending upon context and site complexity; 1D models are generally used in applications such as sediment transport or flooding assessments in which cross-channel spatial dynamics are not vital (Mashriqui et al., 2014, Sabatine et al., 2015), and are favoured for their simplicity, transferability, ease of calibration, and low computational demand. Examples of 1D hydraulic models include HEC-RAS 1D and SRH-1D. 2D models are widely applied within hydraulic and ecological assessments in which knowledge of detailed spatial dynamics are necessary; they are particularly appropriate when assessment requirements do not require, or are able to disregard, the

vertical component of flow (for example hydraulics in shallow river systems), and therefore 2D model assumptions are valid (Franz and Melching, 1997). 2D models are a middle-ground between 1D and 3D in terms of complexity and requirements, needing significantly more calibration and computational power than a 1D model, yet being significantly simpler than a 3D model. 2D models require boundary inflow and outflow data and relatively high-resolution bed geometry data (Lai, 2008), examples including SRH-2D and MIKE 21. 3D models offer the greatest amount of information within a system, offering vertical as well as lateral and longitudinal channel hydraulics, which have been observed to significantly affect habitat model predictions in larger river systems (Pisaturo et al., 2017). 3D models are the most data-intensive, computationally demanding, and difficult to calibrate of the three models however, and therefore tend to be used when the additional information provided is known to be necessary; for example in a deep river where vertical velocities have an important influence upon bed-level hydraulic forces.

Significant progress has been made both in terms of predicting the in-stream hydraulic response to flow regime inputs, and in predicting the ecological impacts of these hydraulics. Hydraulic models have grown in sophistication, with 3D models such as Delft3D providing highly advanced methods by which flow may be simulated. For application in environmental flow designation however, 2D modelling continues to be widely used; although recent studies have pointed out that in larger systems 3D models capture flow dynamics with greater accuracy (Pisaturo et al., 2017), 2D models such as SRH-2D and River2D continue to provide good results (Jowett and Duncan, 2012).

Ecological models have traditionally focused upon habitat quality, with the Physical Habitat Simulation system (PHABSIM) being one of the most widely used packages for this purpose (Reiser and Hilgert, 2018). Issues have been identified with this popular software (Beecher, 2017), but others such as Reiser and Hilgert (2018) argue that PHABSIM remains a valid approach for habitat suitability modelling. More recent models such as CASiMiR similarly model the habitat suitability of species based upon predicted hydraulics, and have advantages such as the use of fuzzy logic, which allows for a flexible ruleset by which the interplay of multiple variables may be accounted for with the aid of expert opinion (Schneider et al., 2010, Schneider et al., 2016), and the use of FST (Fliess Wasser Stammtisch) values in place of traditional velocity values, a surrogate for bed forces based upon river bed experimentation using weighted hemispheres to assess flow forces (Kopecki, 2008). Habitat quality models are the most common approach to ecological modelling currently. There have been calls for greater sophistication of model considerations (Anderson et al., 2006), pointing out that a more holistic consideration of ecological interactions (such as biotic components within the environment), arguing that flow forces alone do not fully explain the response of species to flow conditions, and may not accurately quantify resulting taxon distributions. However, this form of modelling has not yet seen widespread implementation, likely due to the complexity and high number of variables such an approach requires, in addition to the inability of current models to integrate the full spectrum of ecological dynamics such as physical habitat processes and biological interactions. Even were such a model available, much like 3D hydraulic models it would likely not be desirable for use unless necessary due to prohibitive data requirements, computational power demands and calibration difficulties. At this stage, physical habitat-quality based modelling remains the primary method by which hydraulic impacts are assessed (Pisaturo et al., 2017, Premstaller et al., 2017), and remains a "vital and well-utilized tool" in the field (Poff et al., 2017).

2.7 Current knowledge gaps

Despite current advancements, a number of knowledge gaps exist that cause environmental flow implementation to remain a challenge. As discussed in Section 2.6.3, transferability of results remains an issue; although regional and class-based transferability has been identified as a path forwards in the near-term, few studies have implemented approaches in this manner; those that have tend to address environmental flows without consideration of the temporal influence of flow. In the Piedmont region, Italy, a study designated minimum environmental flows at a regional level (Vezza et al., 2012), and whilst successfully implementing class-based regional assessment for flow designation, environmental flows were assigned in terms of steady flow minima. Solans and de Jalon (2016) also perform a regional analysis utilising the Ecological Limits of Hydrological Alteration (ELOHA) method (Poff et al., 2010); though the study presents ecology-flow relationships and trends on a regional scale, the ELOHA method focuses upon identifying thresholds for the extent to which particular flow characteristics can be modified before significant changes to ecological composition are observed, and deals with how environmental standards might be set; no specific flow regime is proposed. The approach represents one tool that might be used within flow regime designation, but may entail intensive field work to test the identified ecology-flow relationships. These studies do not directly consider the societal services and water security perspectives of environmental flow designation; this perspective appears to be an oft-neglected aspect of research. In this thesis I aim to integrate regional, class-based transferability with holistic environmental flow regime designation that considers the influence of flow magnitude, temporal flow characteristics such as flow event frequency, as well as the implications of working in a context with societal service and water security requirements. This is accomplished by using lessons learned from previous studies, identifying key areas of study outlined in studies such as Gillespie et al. (2015a) and Poff and Zimmerman, 2010 (Section 2.5), and adopting a number of recent approaches taken that have attempted to overcome the challenges posed in this field, as discussed in Section 2.6. Additionally, original research is used to identify influential flow characteristics and typical regional flow patterns in natural systems (discussed further in Chapters 3, 4 and 5) in order to develop a flow regime at a case study site. Flow regime designation integrates ecological flow magnitude requirements (identified through hydraulic and ecological modelling), the temporal influence of flows upon the ecosystem (identified through trait-based analysis), and impoundment storage capacity (tracked by a reservoir storage model) in order to achieve holistic proposed flows. The approach emphasises regional and river class-based assessment and is designed to be transferable to similar river systems, and possibly scaled up to larger systems, as discussed in Chapter 6.

2.8 Summary

Flow regime modification due to impoundment acts as a pressure upon riverine ecology through a number of direct and indirect drivers. It is important to gain a greater understanding in the relationship between these drivers and ecological response in order that cost-effective mitigation measures may be put in place to safeguard ecological health and work towards meeting freshwater legislative requirements. There is a great deal of challenge in the translation from existing conceptual understanding into practical investigatory frameworks, due to the highly complex nature of rivers and the numerous interdependent variables that contribute to the overall state of the system. This difficulty is further exacerbated by the lack of a cohesive approach to flow investigation; it has been

concluded that much literature cannot be compared with one another or synthesised into general understanding due to the high site-specificity of many investigations and the lack of uniform methodologies. This project has reviewed existing literature and will attempt to go about this investigation with the goal of furthering understanding on a more general transferable level, as opposed to being merely site-specific.

The focus of this study has been the relationship between flow modification and macroinvertebrate response within the region of Northern England. Macroinvertebrates were chosen as an indicator for a number of practical reasons. Firstly, ecological expertise present for this project lies in the study of macroinvertebrates. Secondly, the area of fish is a thoroughly investigated field, thus any findings of scientific novelty would necessitate extensive technical knowledge of the area; this would require expertise that, as previously mentioned, this investigation does not have access to. Thirdly, macroinvertebrates have been reported as a somewhat neglected field (Gillespie et al., 2015b); it was believed that this investigation with the resources available would be capable of much more scientifically valuable findings in the field of macroinvertebrates. A two-fold approach of statistical analysis, and ecohydraulic modelling throughout the investigation resulted in the following outputs; a framework for further investigations in this field; an enhanced general understanding of the relationship between flow and ecology; and recommended ecologically-beneficial flow regime recommendations for the case study site subject to investigation.

2.9 Research questions overview

Through a review of current literature, key research questions for this investigation were identified. These questions were outlined in detail in Chapter 1. Below, an overview of these questions and the Chapter(s) associated to them in this investigation is provided prior to moving into the research section of the thesis.

Table 2.1: Thesis research questions and associated Chapters

Research Question	Chapter 3	Chapter 4	Chapter 5	Chapter 6
1. What specific drivers are eliciting an ecological response?	✔			
2. How can localised hydraulic requirements of taxa be translated into an ecologically beneficial flow regime?		✔	✔	
3. What indicators can be used to interpret habitat quality, given its variation over time and different values between species?			✔	
4. How may outcomes such as habitat quality predictions be interpreted in light of legislative targets?				✔

3. UNDERSTANDING AND QUANTIFYING THE IMPACTS OF FLOW MODIFICATION IN NORTHERN ENGLAND THROUGH MULTISITE ANALYSIS

3.1 Introduction

Chapters 1 and 2 have discussed the issues and challenges inherent to the field of environmental flows. More specifically relevant to this Chapter, a significant knowledge gap was identified; this being a lack of understanding in how taxa respond to specific characteristics of flow, and which flow characteristics are the most impactful upon the ecosystem when a system is subjected to flow modification. This Chapter explores two of the key research questions that were previously identified in Section 1.3:

Research Question 1: How can ecology-flow interactions be better understood in a more transferable manner?

Research Question 2: Does impoundment flow modification impact the local ecosystem, and if so what specific flow characteristics are most influential upon benthic macroinvertebrate communities?

It was established in Chapter 2 that the wider implementation of environmental flows is hindered by a lack of general principles and methods that may be applied across sites (Gillespie et al., 2015b). Many smaller-scale impoundment systems exist, and many are currently not meeting legislated ecological targets (Voulvoulis et al., 2017). Intensive and site-specific investigation within such systems, given their number, is impractical. Better, and more general, information about ecology-flow relationships may reduce the amount of information that is required from a site before scientifically-grounded environmental flows may be developed, and regional principles for such relationships may allow for directly transferable environmental flows between sites of similar character (Arthington et al., 2006).

In this Chapter an assessment is undertaken, utilising knowledge from existing studies, by which quantitative general regional ecology-flow principles may be identified within a regionally relevant context (Northern England in this case). Such knowledge will aid in better understanding such relationships, and facilitate more efficient environmental flow designation by highlighting influential flow variables and their general relationship with a given ecological indicator (macroinvertebrates in this study). This may be of particular use at smaller sites in which intensive and site-specific investigation may not be feasible due to time and resource constraints. The key aims of this Chapter are to provide quantitative results that may be synthesised into the wider body of literature to aid in the identification of general principles in the relationship between flow modification and macroinvertebrate ecology within rivers of similar character, and to utilise these principles for environmental flow design in Chapter 5 of this investigation, while also establishing the degree to which quantified general relationships may be integrated into the designation of flow regimes. These outcomes will help to inform possible future water management solutions by contributing to the wider knowledge base, and more specifically by identifying ecology-flow trends within the region of Northern England which may potentially be applied across sites within the magnitude range studied,

as well as supplementing the modelling approach used in Chapter 5 in order to achieve a more holistic environmental flow designation methodology (research question 3).

In developing such a framework, a number of principles have been used to guide the approach; the first of these is to base analyses on similar river systems. Studies based on sites covering a wide range of conditions may encounter difficulties in detecting patterns as a result of the variation resulting from differences between classes or types of watercourse; magnitude of flow in particular may overwhelm other hydrological drivers when assessed across too broad a scale (Monk et al., 2006). River systems of a similar geology and geography, which experience the same climatic conditions, are expected to generally respond in a similar manner in terms of flow, thermal regime and physiochemical properties (Alcazar and Palau, 2010). Class-based investigation became more widely utilised in the early 2000s (Arthington et al., 2006, Alcazar and Palau, 2010, Gonzalez and Garcia, 2006), in which study sites are narrowed down to those sharing similar characteristics (as defined by the researcher) so as to constrain the number of variables likely to be influencing the ecosystem. This investigation focuses on rivers of a similar magnitude class, located across the region of Northern England.

The second principle is to focus on functional, as well as taxonomic, measures of ecological community structure. This investigation uses trait-based ecological indices. Just focusing upon taxonomic composition may not detect the influences that flow exerts upon ecosystems in cases where composition is altered but overall richness is not (Chinnayakanahalli et al., 2011). There is increasingly a call for a broader suite of ecological metrics to fully assess ecological impact (Arthington et al., 2018). In this study, we employ a combination of taxonomic diversity measures, a flow velocity preference metric based on species traits, and LIFE scores (an existing flow preference metric; Extence et al., 1999) at the study sites.

A third principle is to try and take account of the transferability of results. Poff and Zimmerman (2010), following a wide meta-analysis of papers, comment:

"Our analyses do not support the use of the existing global literature to develop general, transferable quantitative relationships between flow alteration and ecological response; however, they do support the inference that flow alteration is associated with ecological change and that the risk of ecological change increases with increasing magnitude of flow alteration." (Poff and Zimmerman, 2010).

The demand for more transferable and relevant data is clear from past literature. The goal of increasing general understanding of ecological response to modified flows is therefore a key aim throughout this investigation. There is still much work to be done in working towards a general knowledge of ecology-flow relationships. A number of recent studies in the field of environmental flows are focused on specific applications, for example hydropeaking, that are not necessarily transferable to reservoir impoundment. Examples include ecosystem responses in rivers connected to wetland (Hickey et al., 2015), or the assessment of response following the implementation of a specific flow experiment (King et al., 2015)**.** Transferable data is therefore relatively scarce due to differing potential applications. The general consensus among recent studies appears to be a continuing need for more quantitative studies (Penaluna et al., 2017), transferable within their particular field of application, to increase the availability of data in all sub-fields of environmental flows. Additionally, further development of ecological assessment tools is desirable (Salmaso et al., 2018) to better identify different kinds of ecological impact (e.g. functional composition or richness as discussed in

Section 2.6.1), in order to increase the likelihood of significant ecological-flow relationships being identified.

In order to ensure a level of transferability, the approach of a multi-site desk-based analysis was taken utilising lessons from past literature such as the potential for class-based analysis to provide more useful ecology-flow information (Arthington et al., 2006), and the need for general principles that may be generated through an analysis of several systems (Gillespie et al., 2015b). A multi-site analysis approach allows trends to be observed across sites, rather than attempting to analyse specific sites which may lead to limited transferability of findings. It involves the statistical analysis of flow and ecological data across several sites of a particular magnitude and geographical class. The multi-site analysis breaks down flow at each site into several flow components, drawing from the Indicators of Hydrological Alteration (IHA) method (Richter et al., 1996), described further in section 3.2. As detailed in the methods, sites were selected at a similar scale and similar geographic region; this should in principle entail that statistically significant relationships identified across selected sites will have some level of transferability to other sites of a similar scale and in the local region, and perhaps might translate to systems beyond the region studied.

3.2 Methods

This study utilised an IHA-style breakdown of historical flow data (Richter et al., 1996) in order to identify hydrological characteristics at each site. Sites were characterised ecologically using species trait analysis focusing upon flow velocity affinity, Shannon's biodiversity index for species populations, and LIFE scores. The differences in site character and ecological properties between each site were analysed through multivariate linear modelling; variables or combinations of variables displaying a statistically significant influence upon ecology were noted. A key aim was to identify significant ecology-flow relationships in order for potential trends general to the region of Northern England to be proposed and applied practically in Chapter 5.

3.2.1 Site selection & data

Site briefings provided by United Utilities (UU), written by both a hired consultant, Grontmij, and Environment Agency assessors, and EA's online Catchment Data Explorer (Environment Agency, 2018), were utilised to select study sites for this investigation. Sites were chosen based upon their geographical similarity, similarity of flow magnitude, and abundance of available synchronous flow and macroinvertebrate data. A range within approximately one order of magnitude for annually-averaged mean daily flows was used for site selection; this range was used as magnitude classifications are typically defined within order of magnitude groupings, though no single classification method to date has been completely defined and accepted (Meybeck et al., 1996). Selected sites ranged from 0.31-4.3m^3/sec annual mean daily flow. Five years or more of data in order to obtain more robust seasonal mean species populations; a longer coverage would have been desirable, but long-term synchronised flow and ecological data is not abundant in the UK, particularly with the additional criteria discussed. Sites were also excluded if there were any apparent and significant external factors could influence populations, such as poor water quality, defined using the Environment Agency's online Catchment Data Explorer (Environment Agency, 2018) in which the EA's chemical classification for a site can be found. The sites selected for study were of "Good" chemical quality in the most recent

analyses. Site selection concluded with 20 selected sites located across the North of England as shown in Figure 3.1.

Figure 3.1: Locations of all study sites across the North of England

Sites OS grid reference locations are detailed below in Table 3.1:

Table 3.1: List of study sites and their flow gauging and ecological sampling OS locations, with distances between the two sampling sites.

Site name	Flow Gauging OS Location	Ecology Sample OS Location	Distance between Flow and Ecology sites
Blackfoss Beck	SE7249147392	SE7251947416	36m
Colne	SE1364416110	SE0910914447	830m
Crimple Blackstones	SE4013252956	SE3787951685	4km
Dearne	SE3497007279	SE3477007932	690m
Eastburn Beck	SE0203545263	SE0148144826	702m
Foulness	SE7797637277	SE7800738044	763m
Ryburn	SE0354718938	SE0404819773	970m
Skell	SE3157070949	SE3185270904	286m
Spen Beck	SE2247621023	SE2261920934	242m
Went	SE5506416309	SE5650116142	1.44km
Calder	SD4978643349	SD4988943319	108m
Church Beck	SD3063997190	SD3020097600	605m

Crake	SD2945588220	SD3067284861	3.5km
Douglas Wigan	SD5861706027	SD5860906011	19m
Eden	NY6045228312	NY6039128147	175m
Eea	SD3643176385	SD3610076600	390m
Heltondale	NY4943720421	NY4923520205	290m
Pendle Water	SD8366535152	SD8365535455	296m
Swindale Beck	NY5146113169	NY5360016300	3.8km

Following site selection, requests were sent to the UK's Environment Agency (EA), both for historical flow data (flow data was also obtained from the Centre of Ecology and Hydrology (CEH) National River Flow Archive) and benthos ecological sampling data. Flow data was in the form of mean daily flows, typically spanning over 10 years. Ecological data was in the form of a large database showing ecological indices and taxon abundance at a species or family level, typically with samples taken in spring and autumn each year and sampling spanning 5-10 years. The coordinates of the data were checked to ensure that the flow and ecology data were synchronised, had no significant intervening flow inputs such as tributaries between flow and ecological measurement sites, and were within a 4 kilometre proximity (see Table 3.1), within which overland flow would not be expected to significantly alter in-channel flow conditions between ecological and flow gauging sites. LIFE scores were immediately available from EA data, whilst other flow and ecological variables were derived from processing of the raw data.

3.2.3 Processing hydraulic data

Using mean daily flow data obtained from the Environment Agency or CEH, flows were characterised at each site following categories based on the IHA method of assessment using the IHA software (The Nature Conservancy, 2017). The indicators from Richter et al. (1996), discussed in the previous Chapter, Section 2.2.3, break down the characteristics of a flow time series into a number of ecologically-relevant flow characteristics. These flow components may then be related to taxa distributions (obtained from the Environment Agency) at a given site through various ecological metrics such as trait-based analysis and biodiversity. Using PCA, it will be possible to reduce the redundancy of flow characteristics and better isolate drivers of ecological response or categorise them into particular groups. Additional components such as the extent of river modification have been used in recent literature (Gillespie et al., 2015a) but such factors are beyond the control of water managers and thus would not contribute towards the overall goal of informing flow regime designation. Additionally, though such information would be scientifically interesting, it is beyond the scope of this investigation, given the desire to utilise a simple and transferable framework, and due to resources being committed to other avenues of investigation (see Chapters 4 and 5). Following redundancy analysis, the most influential drivers may be found through multivariate linear modelling. By identifying the most influential components of flow, this analysis will guide approaches used in 2D modelling in Chapters 4 and 5, and will contribute to the overall body of knowledge being accumulated within the field of ecohydraulics relating to reservoir impoundment and environmental flows.

The challenge of site-specificity and lack of general principles for ecologically beneficial flow regimes, due to the complexity of river systems, is discussed in Section 1.3 and further detailed in Chapter 2. In response to these difficulties, mitigation measures for flow modification based on river classification

have been suggested as a near-term solution (Arthington et al., 2006). River classifications involve sorting rivers into classes of similar character, for example a particular river class may share similar flow magnitudes and sediment characteristics. The meta-analysis hopes to identify such trends, and is thus one way in which this investigation hopes to overcome the issue of site specificity and provide site managers with general principles for ecological improvement that may be applied across a particular classification of river.

The multiple site analysis is not overly demanding in terms of data requirements, though significant processing of the data to form new metrics is required; daily flow averages obtainable from the Environment Agency and the Centre of Ecology and Hydrology, coupled with taxa sampling data in the form of species distributions are sufficient to achieve the fundamental goals outlined for this task. This method of matching daily or monthly flows coupled with biotic data has been seen in other recent publications (Solans and de Jalon, 2016) though, as described in Secton 2.7, their approach was more focused upon describing modification thresholds through the ELOHA method, as opposed to directly informing flow regime design as in this thesis. Though data requirements are not complex, the volume of data requires significant processing and formatting, and data availability is often an issue in these studies due to the need to find synchronised flow and ecological data that is acceptably accurate and has minimal additional ecological pressures present such as water quality issues. Index values by site are listed in Appendix ii. Table 3.2 describes the flow metrics utilised and corresponding characteristics they describe. These metrics were chosen due to their ecological importance described by Richter et al. (1996), evidence of this in other studies (Monk et al., 2006, Blanckaert et al., 2013, Rolls et al., 2012) and due to their relative simplicity to implement as part of a flow regime. For example, it is a simple matter for an impoundment to alter the frequency at which higher flows are outputted, whereas it is less feasible to control other categories identified by Richter et al. (1996) such as "Smoothness" or "Rapidity of Change" due to current impoundment infrastructure limitations and due to issues in communicating less intuitive regime changes to water managers.

Table 3.2: Flow metrics and their corresponding flow characteristics

Flow Index Category	Characteristic described
Mean annual flow	Describes general magnitude of flow
Q10	Describes very high-flows
Q25	Describes moderate high-flows
Q95	Describes very low-flows
Standard Deviation	Describes heterogeneity of annual flow
Mean annual minima	Describes extreme lows
Mean annual maxima	Describes extreme highs
Mean annual range	Describes general yearly range
Mean low flow frequency	How frequently low flow events occur (median flow -25%)

Mean low flow duration	How long low flow events tend to last (median flow -25%)
Mean high flow frequency	How frequently high flow events occur (median flow +25%)
Mean high flow duration	How long high flow events tend to last (median flow +25%)

Using the above flow index values, flow at each site could be well characterised by a number of flow metrics reported in literature to be ecologically influential (Worrall et al., 2014). Worrall et al. (2014) aimed to identify hydrological indices for characterising macroinvertebrate community response. They identified six flow variables that were able to explain ~95% of the ecological variance attributed to flows, though they emphasise that explanatory power and contribution of variables may change based upon regional characteristics such as groundwater contribution to flow. The six variables point to high flows, low flows, and temporal variation as key drivers within the systems investigated by the study; the variables selected in Table 3.1 are selected to characterise these flow components.

3.2.4 Processing ecological data

For the EA invertebrate data, there was variation in the taxonomic resolution used. In some years data was at species level whilst in others at a family level. Because of these differences, all data was converted to family level for consistency, and then the mean annual seasonal family abundances were calculated for each site in spring and autumn. Velocity affinity has been utilised in a number of ecological analyses (Schneider et al., 2016, Conallin et al., 2010), a similar approach was adopted here. Species preferences for the analysis were taken from the European Commission supported STAR project, within the output "Deliverable N2: Species Trait Analysis" (Bis and Usseglio-Polatera, 2004). This output deals specifically with species trait analysis of macroinvertebrates and contains one of the largest collections of species biological affinities currently available. The entirety of the STAR project is freely accessible online (www.eu-star.at). Preferences were assigned to families by taking the mean trait affinity value of all species present within that family. This approach has been used in other studies and, because of general levels of trait similarity within families, is considered a justified way of deriving family preferences or affinities (Resh et al., 1988).

Each family was also sorted into particular categories of flow preference, described in Table 3.2. These categories were based upon defined flow ranges similar to those defined by the STAR project (Bis and Usseglio-Polatera, 2004), with additional categories for more generalist species displaying a range of preferences (typically favouring a particular flow but having moderate affinity for a range of flow). The STAR project lists the following 4 categories of flow preference: null, slow (<25 cm/s), medium (25-50 cm/s), fast (>50 cm/s). However, these four categories each have affinity scores (0-3) assigned to them, so as to define how the species responds within each condition, rather than providing a discrete single categorical value per species. For example, a species might have an affinity score of '1' for 'null', '2' for 'slow' and 'medium', and '1' for 'high'. For this investigation, it was desirable that each family be assigned a discrete single categorisation in order for the statistical analysis (dividing the overall macroinvertebrate population into various flow preference categories in order to provide an overall site trait score) to proceed. As such, more than 4 categories of flow preference were necessary to represent the various affinity distributions.

Null flow was represented by the category of 'Null Flow', whilst high affinity scores of '3' for 'slow' or 'fast' were represented by 'Low Flow' and 'Fast Flow' respectively. Families with more general preferences (i.e. an affinity score of 2 or more for most STAR flow categories) were provided 'generalist' categories; most families were found to have at least a slight preference for lower or higher flows, however, and thus the categories to represent these families were designated 'Low-Medium Flow (generalists)' and 'Medium-Fast Flow (generalists), respectively. Lastly, families that were not generalists, but also did not display strong affinities strictly for 'slow' or 'fast' (e.g. perhaps had an affinity score of '2' for 'slow' and '2' for medium, and '1' or less for 'null' or 'fast') were put into the intermediate categories of Low-Medium, Medium, and Medium-Fast Flow, depending upon affinity score distributions. Categories are listed in Table 3.3.

Table 3.3: Trait score categories and associated weightings

Flow Velocity Preference	Trait Score	Weighting
Null Flow	1	10
Low Flow	2	7
Low-Medium Flow only	3	7
Low-Medium Flow (generalists)	4	4
Medium Flow only	5	1
Medium-Fast Flow (generalists)	6	4
Medium-Fast Flow only	7	7
Fast Flow	8	10

Populations within each category were summed up at each site based on the mean annual abundances of each family within a given category within spring and autumn. The distribution of abundances between categories provides an insight into functional composition of a site. Once population distributions across trait categories were calculated at each site, more extreme categories (e.g. very fast flow) were given a weighting due to the fact that taxa possessing extreme traits tend to be less common in typical conditions, yet the presence of even small numbers of such taxa is suggestive of a system's character (Petchey and Gaston, 2006). Generally across sites, species preferring medium flows were prolific, and thus weightings were used to better demonstrate fluctuations in functional distributions. Flow velocity categories were each given a score between 1 and 8. Completely lentic (still) flow was scored at '1', medium velocity '5' and very high velocity '8'. The abundances of families present in each category, relative to the total population, was multiplied by the weighted score. The sum of these values constituted the overall Trait Score. A trait score of '1' would indicate a site dominated by the lentic flow affinity species, whilst '8' would indicate that fast flow affinity species dominate. Values in between indicate some ratio between the two.

At low-medium flows, most species in the sampled regions appear to be generalists, with species of specific low-medium affinity being very rare. As such, the weighting for the Low-Medium affinity was weighted the same as the Low Flow affinity, which was also rare at most sites.

In Table 3.4 a trait score calculation is demonstrated:

Table 3.4: Example of trait score calculation on a fictitious site, prepared to demonstrate the calculation prior to data analysis of study sites.

Score	Flow Affinity	Average Abundance	Weighting	Weighted Population	Proportion	Proportion * Score
1	Null flow (0cm/s)	1	10	10	0.028	0.028
2	Low Flow (0-10cm/s)	1	7	7	0.020	0.040
3	Low-Medium Flow (10-20cm/s)	0	7	0	0.000	0.000
4	Low-Medium Flow (less selective)	5	4	20	0.057	0.226
5	Medium Flow (20-30cm/s)	97	1	97	0.274	1.370
6	Medium-Fast Flow (less selective)	49	4	196	0.554	3.322
7	Medium-Fast Flow (30-40cm/s)	6	7	24	0.068	0.475
8	Fast Flow (>40cm/s)	0	10	0	0.000	0.000
					TRAIT SCORE	5.461

It can be seen in Table 3.4 that a very large population of species with a medium-flow affinity dominates the site. However, also present are generalist species with a tendency to favour slightly higher flows, as well as a small population of species preferring medium-fast flows. Also present are generalists with tendencies for low flows, and singular individuals present for very low flows; these are much less significant. The final trait score of 5.461 for this site provides a good description of functional distribution; there is a slight favour for higher flows at this site, but it is still closest to the

integer score of 5. Were there a larger population of "medium-fast" affinity species, a score approaching 6 would be expected. This form of trait-based analysis allows for ecological characteristics to be compared across sites directly alongside flow characteristics, allowing hypotheses to be formed, for example, on how a particular flow characteristic may be driving a particular ecological characteristic. This method has been used in a number of previous studies, as demonstrated in the literature review (Alexandridis et al., 2017, Ings et al., 2009, Petchey and Gaston, 2006).

3.2.4.1 LIFE Score

The Lotic-invertebrate Index for Flow Evaluation (LIFE) score is a widely used metric for the ecological monitoring of benthic macroinvertebrates (Dunbar et al., 2010a). The method is based upon the flow affinities of macroinvertebrate species and families, recognized from established literature. Commonly identified British freshwater species are assigned to one of six flow groups shown below in Table 3.5 adapted from Extence et al. (1999):

Table 3.5: Benthic macroinvertebrates flow groups and associated characteristics, adapted from Extence et al. (1999)

Group	Ecological flow association	Mean current velocity
I	Taxa primarily associated with rapid flows	Typically >100 cm s^{-1}
II	Taxa primarily associated with moderate to fast flows	Typically 20-100 cm s^{-1}
III	Taxa primarily associated with slow to sluggish flows	Typically <20 cm s^{-1}
IV	Taxa primarily associated with flowing (usually slow) and standing waters	-
V	Taxa primarily associated with standing waters	-
VI	Taxa primarily associated with dry or drought impacted sites	-

Overall LIFE score for a site is calculated through the abundance of species by category as follows:

$$\text{LIFE} = \frac{\sum f_s}{n}$$

Where $\sum f_s$ is the sum of individual taxon flow scores for the whole sample, and n is the number of taxa used to calculate $\sum f_s$ (Extence et al., 1999).

3.2.4.2 Shannon's Diversity

Biodiversity is seen as a key indicator of ecological wellbeing (Levin, 2000, Benayas et al., 2009), therefore a metric for biodiversity was utilised to measure ecological response in this regard between

sites. Shannon's Diversity (H) is a widely used metric within the field of ecology to determine the diversity of species within a given population (Ludwig and Reynolds, 1988). This study determined diversity for each site from macroinvertebrate family data, again in spring and autumn respectively. An index of Biodiversity was calculated for each site, using the Shannon's Diversity Index (H) as a metric, the calculation being:

$$\mathrm{H}' = \sum_{i=1}^{s} p_i \ln p_i$$

Where S is the number of species present in the sample and p_i is the relative abundance of species i, i.e. it is the proportion of individual species relative to the total community (Magurran, 2004). At each site, biodiversity was calculated using the family population as 'species'.

3.2.6 Reducing redundancy of flow variables

A Principal Components Analysis (PCA) was undertaken with the aim of disentangling ecological and hydrological variables and reducing redundancy. PCA has been used in a number of statistical analyses relating to environmental flows as a method to overcome the complexity of the river system and its many interacting variables, by identifying the way in which independent variables relate to one another (Worrall et al., 2014, Mwedzi et al., 2017, Woznicki et al., 2015). PCA was performed in the software program R on data matrices containing IHA-processed flow data at each study site, in order to inform subsequent multivariate models; i.e. to avoid using redundant variables within the same model. Variables were sorted into distinct groups based upon their clustering in the PCA. Clustering occurs based upon how variables relate to one another; variables that are closely related will have similar vectors and cluster around the same area, inversely related variables will have opposing vectors, and variables sharing no relationship will be perpendicular to one another. Redundancy within these each of these groups was significant, and it was decided that in subsequent multivariate modelling, such variables should not be used within the same model; this is detailed further in Sections 3.2.7 and 3.3. The original flow variables were utilised as opposed to PCA-derived principal components (PCs) so as to be more interpretable and actionable; for example relating an original variable such as "Low Flow Frequency" to ecological response can provide clear implications and potential mitigation solutions within a designed flow regime, whereas an amalgamation of variables within a PC is more difficult to derive solutions from. Studies such as Mwedzi et al. (2017) similarly opt to retain individual variables in order to comment on potentially key drivers of ecological metrics.

3.2.7 Model fitting

Six data matrices were constructed, containing one of the three ecological indices (diversity, LIFE, or velocity preference) for either spring or autumn seasons. Each matrix contained all independent flow variables. Within the matrices, ecological indices vary seasonally due to shifts in the ecological community between seasons, whereas the flow variables do not vary as these flow characteristics are based upon year-round flow statistics. For each data matrix multiple linear regression was used to fit models for each ecological trait within each season. Linear and multiple linear regression models were created in R using the linear model function for all possible combinations of non-redundant variables along with each variable individually as univariate models. Redundant variables were defined as variables sharing the same category ("Magnitude" or "Temporal").

Model fitting was performed for each ecological dependent variable with combinations of flow variables as the independent variables. The best fitting models for each dependent variable, in spring and autumn respectively, were determined. These were based upon p-values, R-squared values, and the Akaike information criterion (AIC). The AIC is a measure of the relative quality of a statistical model, compared with other models applied to the same data. AIC takes into account the closeness of fit and also the complexity of the model, balancing model accuracy to the provided data with model transferability (Aho et al., 2014). AIC was used as the principal statistic by which best fitting model was identified relative to other models generated. Best-fitting models identified the most influential flow variables upon the ecological variables across the sites used in the study; ecological responses were predicted using the linear model coefficients of identified variables, and the results were recorded and plotted against the original data to compare predictions with what was observed in the field. All analyses were performed in R version 3.2.4 (R Core Team, 2016).

3.3 Results

Ecological metrics for each of the 20 study sites are shown below in Table 3.6.

Table 3.6: Site values for trait score, Shannon's index (H), and family LIFE scores at each site in spring and autumn

Site	Velocity Trait (Spring)	Velocity Trait (Autumn)	Diversity (Spring)	Diversity (Autumn)	Family LIFE (Spring)	Family LIFE (Autumn)
Blackfoss Beck	5.16	3.14	1.90	2.71	6.45	6.61
Colne	4.80	4.52	1.22	2.03	6.89	6.66
Crimple Blackstones	4.65	5.00	2.23	2.53	7.4	7.41
Dearne	5.02	2.83	1.39	2.03	6.29	6.56
Eastburn Beck	4.70	5.27	2.13	2.08	7.91	7.48
Foulness	4.57	2.27	2.03	1.16	6.3	5.81
Ryburn	4.20	4.62	2.19	2.47	7.92	7.49
Skell	4.14	3.97	2.57	2.53	8.14	7.55
Spen Beck	4.09	2.57	1.46	1.81	6.26	6.41
Went	5.59	3.03	2.01	2.35	6.18	6.03
Calder	No data	5.68	No data	2.50	No data	7.63
Church Beck	6.74	6.30	1.70	1.29	7.75	7.6
Crake	6.70	6.27	Omitted	Omitted	7.59	7.18
Douglas Wigan	6.27	5.99	1.87	1.83	6.69	6.89

Eden	6.11	6.75	2.31	2.64	7.9	7.73
Eea	6.43	7.10	2.40	1.85	7.43	7.07
Heltondale	6.89	7.01	1.84	2.18	8.3	8.29
Pendle Water	5.55	4.06	1.57	2.42	7.22	7.34
Swindale Beck	6.26	6.26	2.09	2.59	7.79	7.53

All hydrological metrics for each site are listed in Table 3.8 below. Metrics are as detailed in Table 3.7;

Table 3.7: Flow variables and descriptions of what they represent

Flow Variable		Description
Mean Daily	-	Annually-averaged mean daily flow
Q10	-	Annually-averaged 90th percentile of flow
Q25	-	Annually-averaged 75th percentile of flow
Q95	-	Annually-averaged 5th percentile of flow
STDEV	-	Annually-averaged standard deviation of flow
MINYR	-	Mean annual flow minima
MAXYR	-	Mean annual flow maxima
RNGYR	-	Mean annual flow range
LowFreq	-	Mean annual low flow event frequency
LowDura	-	Mean annual low flow event duration
HighFreq	-	Mean annual high flow event frequency
HighDura	-	Mean annual high flow event duration

Table 3.8: All study sites with their associated hydrological variables.

Site Name	Mean Daily (m³/sec)	Q10 (m³/sec)	Q25 (m³/sec)	Q95 (m³/sec)	STDEV	MINYR (m³/sec)	MAXYR (m³/sec)	RNGYR (m³/sec)	LowFreq	LowDura	HighFreq	HighDura
Blackfoss Beck	0.45	0.858	0.403	0.040	1.065	0.042	8.431	8.389	6.628	13.260	7.116	2.398
Colne	1.44	3.177	1.537	0.329	1.995	0.291	17.312	17.022	17.030	5.703	13.340	2.389
Crimple Blackstones	1.06	1.950	0.905	0.150	3.446	0.132	36.588	36.456	13.470	6.806	6.765	1.435
Dearne	1.36	2.860	1.410	0.273	2.165	0.258	18.937	18.679	11.510	8.001	10.540	2.520
Eastburn Beck	0.88	2.180	0.962	0.076	1.464	0.072	12.714	12.642	9.379	11.380	16.100	2.015
Foulness	1.28	3.117	0.772	0.049	3.081	0.045	18.183	18.138	5.000	19.810	6.824	3.675
Ryburn	0.61	1.249	0.556	0.201	0.877	0.190	7.833	7.643	9.306	9.208	9.222	3.167
Skell	1.51	3.703	1.795	0.149	2.295	0.132	18.010	17.878	6.677	13.060	12.770	2.679
Spen Beck	0.74	1.216	0.652	0.103	0.752	0.181	6.734	6.553	12.260	7.646	7.343	1.845
Went	0.57	1.005	0.567	0.162	0.842	0.165	7.224	7.059	10.430	8.297	7.757	2.868
Calder	1.007	2.570	0.979	0.045	1.380	0.057	12.103	12.046	10.600	4.320	22.650	2.080
Church Beck	0.848	3.680	1.300	0.092	1.073	0.082	8.705	8.623	11.070	4.250	25.710	2.210
Crake	4.214	9.010	5.860	0.630	3.458	0.530	19.839	19.309	5.350	12.140	8.070	7.810
Douglas Wigan	1.217	2.370	1.325	0.373	1.091	0.409	9.263	8.854	13.560	3.400	17.590	2.580
Eden	2.66	6.740	2.700	0.173	4.608	0.146	43.837	43.692	9.420	6.160	24.620	2.190
Eea	0.873	2.280	1.190	0.054	1.018	0.041	7.044	7.003	6.670	8.790	12.500	4.040
Heltondale	0.309	0.710	0.348	0.036	0.408	0.037	3.512	3.475	8.600	6.580	13.250	3.930
Pendle Water	2.825	6.830	2.910	0.464	3.958	0.435	38.119	37.684	12.640	4.550	22.140	2.460
Swindale Beck	1.201	3.150	1.320	0.076	1.938	0.066	15.189	15.123	13.610	4.080	30.720	1.890

PCA demonstrated redundancy between the flow variables; variables within close proximity, or with opposing vectors, are related or inversely related respectively. Vectors that are perpendicular have no discernible relationship. Thus, variables could generally be categorised into the two groups of "Magnitude" and "Temporal" as demonstrated below in Figure 3.2, with Mean Annual Magnitude generally having strong positive correlation between other non-temporal variables within the group, and weak relationships with other temporal variables. The Temporal category generally displayed positive correlation between variables of the same category (frequency or duration of event), and negative correlation between these two categories. The Magnitude group comprised of variables strongly driven by the overall flow magnitude; flow range, annual minima, annual maxima, Q10, Q25, Q95 and standard deviation. The Temporal group comprised of variables associated with flow event occurrence in time; low flow event frequency and duration, and high flow event frequency and duration.
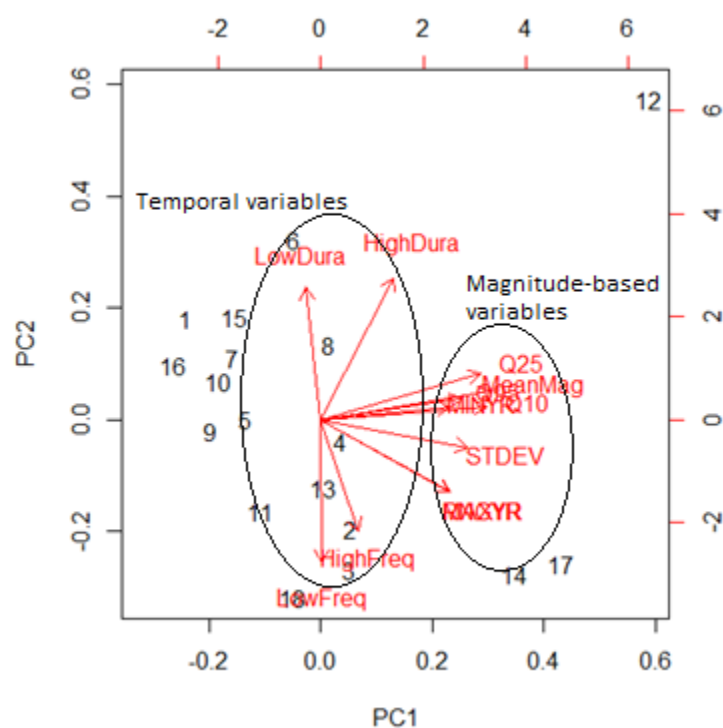
Figure 3.2: PCA bi-plot demonstrating redundancy between flow variables

Following the redundancy analysis and identification of variable clustering, fitting of linear models was performed for all possible non-redundant combinations of variables as described in Section 3.2.7. The best-fitting model was chosen for each dependent ecological variable, both in spring and autumn, based upon best (lowest) AIC value. Below in Tables 3.9 to 3.14, relationships between hydrological variables and ecological indicators are shown. R squared and AIC were not calculated for combinations of variables that were irrelevant due to high P values. Following this, Table 3.8 displays the best fitting models for each ecological metric along with model coefficients.

Table 3.9: Spring LIFE Score multivariate model fitting results

LIFE Scores - SPRING				
Variable 1	**Variable 2**	**P**	**R²**	**AIC**
HighFreq	None	0.056	0.161	40.256
HighFreq	MINYR	0.1366	0.131	41.72
HighFreq	Q25	0.138	0.13	41.743
HighFreq	MeanMag	0.1628	0.11	42.141
HighFreq	RNGYR	0.1697	0.105	42.241
HighFreq	MAXYR	0.1699	0.1052	42.243
LowDura	None	0.45	n/a	n/a
LowDura	Q25	0.5231	n/a	n/a
LowDura	MINYR	0.5235	n/a	n/a
LowFreq	None	0.55	n/a	n/a
HighDura	MINYR	0.5605	n/a	n/a
HighDura	None	0.6	n/a	n/a
LowFreq	Q25	0.6244	n/a	n/a
LowDura	MeanMag	0.636	n/a	n/a
LowFreq	RNGYR	0.6727	n/a	n/a
LowFreq	MAXYR	0.6766	n/a	n/a
LowDura	RNGYR	0.6808	n/a	n/a
LowDura	MAXYR	0.6848	n/a	n/a
HighDura	Q25	0.69	n/a	n/a
HighDura	RNGYR	0.7153	n/a	n/a
HighDura	MAXYR	0.7218	n/a	n/a
LowFreq	MeanMag	0.735	n/a	n/a
LowFreq	MINYR	0.7382	n/a	n/a
HighDura	MeanMag	0.822	n/a	n/a

Table 3.10: Autumn LIFE Score multivariate model fitting results

Variable 1	Variable 2	P	R^2	AIC
LIFE Score - AUTUMN				
HighFreq	None	0.0157	0.2561	34.872
LowDura	None	0.0543	0.1539	37.317
HighFreq	MINYR	0.0545	0.2181	36.666
HighFreq	Q25	0.0552	0.2167	36.699
HighFreq	RNGYR	0.0581	0.2117	36.82
HighFreq	MAXYR	0.0582	0.2116	36.823
HighFreq	MeanMag	0.0593	0.2097	36.869
LowDura	MINYR	0.1011	0.1552	38.135
LowDura	Q25	0.1298	0.1284	38.729
LowDura	RNGYR	0.1422	0.1184	38.946
LowDura	MAXYR	0.1434	0.1175	38.965
LowDura	MeanMag	0.1478	0.1141	39.038
HighDura	Q25	0.6611	n/a	n/a
LowFreq	RNGYR	0.8038	n/a	n/a
HighDura	RNGYR	0.8039	n/a	n/a
HighDura	MAXYR	0.8087	n/a	n/a
LowFreq	MAXYR	0.8088	n/a	n/a
LowFreq	Q25	0.8132	n/a	n/a
LowFreq	MINYR	0.8468	n/a	n/a
HighDura	MINYR	0.8698	n/a	n/a
HighDura	MeanMag	0.8727	n/a	n/a
LowFreq	None	0.8966	n/a	n/a
HighDura	None	0.9093	n/a	n/a
LowFreq	MeanMag	0.9224	n/a	n/a

Table 3.11: Spring Velocity Trait Score multivariate model fitting results

Velocity Trait Score – SPRING

Variable 1	Variable 2	P	R^2	AIC
HighFreq	None	0.0376	0.1959	49.79
HighFreq	Q25	0.055	0.2302	49.847
LowDura	Q25	0.0666	0.2103	50.307
LowDura	None	0.0793	0.1287	51.237
HighFreq	RNGYR	0.0802	0.1905	50.752
HighFreq	MAXYR	0.081	0.1893	50.778
HighFreq	MeanMag	0.0937	0.1735	51.126
HighDura	None	0.094	0.11	51.573
HighFreq	MINYR	0.1041	0.1618	51.378
LowDura	MeanMag	0.1187	0.147	51.693
LowDura	RNGYR	0.197	n/a	n/a
LowDura	MAXYR	0.197	n/a	n/a
LowDura	MINYR	0.225	n/a	n/a
HighDura	Q25	0.2314	n/a	n/a
HighDura	MINYR	0.252	n/a	n/a
HighDura	MeanMag	0.2573	n/a	n/a
HighDura	RNGYR	0.259	n/a	n/a
HighDura	MAXYR	0.259	n/a	n/a
LowFreq	Q25	0.3697	n/a	n/a
LowFreq	MeanMag	0.6084	n/a	n/a
LowFreq	None	0.77	n/a	n/a
LowFreq	MINYR	0.827	n/a	n/a
LowFreq	RNGYR	0.934	n/a	n/a
LowFreq	MAXYR	0.936	n/a	n/a

Table 3.12: Autumn Velocity Trait Score multivariate model fitting results

Velocity Trait Score - AUTUMN				
Variable 1	**Variable 2**	**P**	**R²**	**AIC**
HighFreq	None	0.01335	0.269	69.421
HighFreq	Q25	0.03	0.2742	70.135
LowDura	None	0.0352	0.1906	71.357
HighFreq	RNGYR	0.0467	0.2329	71.186
HighFreq	MAXYR	0.0468	0.2328	71.188
LowDura	Q25	0.046	0.2345	71.146
HighFreq	MeanMag	0.0458	0.2349	71.136
HighFreq	MINYR	0.0515	0.2235	71.417
LowDura	MeanMag	0.0752	0.1859	72.314
LowDura	MINYR	0.0981	0.1584	72.947
LowDura	MAXYR	0.1165	0.1401	73.354
LowDura	RNGYR	0.1165	0.1401	73.355
HighDura	None	0.3449	n/a	n/a
HighDura	Q25	0.4644	n/a	n/a
HighDura	MINYR	0.496	n/a	n/a
LowFreq	Q25	0.509	n/a	n/a
HighDura	RNGYR	0.527	n/a	n/a
HighDura	MAXYR	0.5277	n/a	n/a
HighDura	MeanMag	0.5445	n/a	n/a
LowFreq	MeanMag	0.75	n/a	n/a
LowFreq	None	0.8729	n/a	n/a
LowFreq	MAXYR	0.9712	n/a	n/a
LowFreq	RNGYR	0.9713	n/a	n/a
LowFreq	MINYR	0.984	n/a	n/a

Table 3.13: Spring Biodiversity multivariate model fitting results

Biodiversity - SPRING				
Variable 1	**Variable 2**	**P**	**R²**	**AIC**
LowFreq	None	0.0132	0.301	12.518
LowFreq	RNGYR	0.0278	0.3148	13.005
LowFreq	MAXYR	0.0281	0.3139	13.028
LowFreq	MINYR	0.0394	0.28	13.849
LowFreq	Q25	0.0426	0.2719	14.039
LowFreq	MeanMag	0.0472	0.2611	14.289
LowDura	MINYR	0.14	n/a	n/a
HighFreq	MINYR	0.14	n/a	n/a
HighDura	MINYR	0.14	n/a	n/a
LowDura	None	0.2016	n/a	n/a
LowDura	RNGYR	0.3757	n/a	n/a
LowDura	MAXYR	0.38	n/a	n/a
LowDura	Q25	0.42	n/a	n/a
LowDura	MeanMag	0.4515	n/a	n/a
HighDura	None	0.4613	n/a	n/a
HighDura	RNGYR	0.55	n/a	n/a
HighDura	MAXYR	0.5636	n/a	n/a
HighDura	Q25	0.76	n/a	n/a
HighDura	MeanMag	0.77	n/a	n/a
HighFreq	MAXYR	0.92	n/a	n/a
HighFreq	RNGYR	0.92	n/a	n/a
HighFreq	None	0.9322	n/a	n/a
HighFreq	MeanMag	0.97	n/a	n/a
HighFreq	Q25	0.99	n/a	n/a

Table 3.14: Autumn Biodiversity multivariate model fitting results

Biodiversity - AUTUMN				
Variable 1	**Variable 2**	**P**	**R²**	**AIC**
HighDura	None	0.18	n/a	n/a
HighDura	RNGYR	0.29	n/a	n/a
HighDura	MAXYR	0.29	n/a	n/a
LowDura	RNGYR	0.31	n/a	n/a
LowDura	MAXYR	0.32	n/a	n/a
LowDura	None	0.36	n/a	n/a
HighDura	Q25	0.39	n/a	n/a
HighDura	MeanMag	0.4	n/a	n/a
HighDura	MINYR	0.42	n/a	n/a
HighFreq	RNGYR	0.42	n/a	n/a
LowFreq	RNGYR	0.43	n/a	n/a
LowFreq	MAXYR	0.43	n/a	n/a
LowDura	MeanMag	0.6	n/a	n/a
LowDura	Q25	0.62	n/a	n/a
LowDura	MINYR	0.64	n/a	n/a
HighFreq	None	0.65	n/a	n/a
LowFreq	None	0.7	n/a	n/a
LowFreq	Q25	0.78	n/a	n/a
LowFreq	MeanMag	0.79	n/a	n/a
HighFreq	MAXYR	0.8	n/a	n/a
HighFreq	MeanMag	0.8	n/a	n/a
HighFreq	Q25	0.8	n/a	n/a
HighFreq	MINYR	0.9	n/a	n/a
LowFreq	MINYR	0.92	n/a	n/a

Table 3.15: Primary drivers and associated coefficients, identified for each ecological metric through model fitting, with model values of P, R^2 and AIC.

Ecological Variable	Identified primary driver(s)	Model values	Coefficients
Velocity Preference, Spring	High Flow Frequency	P: 0.03758 R^2: 0.1959 AIC: 49.79 On 16 degrees of freedom	Intercept: 4.53049 HighFreq: 0.06467
Velocity Preference, Autumn	High Flow Frequency	P: 0.01335 R^2: 0.269 AIC: 69.421 On 17 degrees of freedom	Intercept: 3.15375 HighFreq: 0.11896
Biodiversity (H), Spring	Low Flow Frequency	P: 0.01322 R^2: 0.301 AIC: 12.518 On 15 degrees of freedom	Intercept: 2.65649 LowFreq: -0.06898
Biodiversity (H), Autumn	No significant relationships identified (all P values >0.15)	n/a	n/a
Family LIFE, Spring	High Flow Frequency	P: 0.056 R^2: 0.161 AIC: 40.256 On 16 degrees of freedom	Intercept: 6.61227 HighFreq: 0.04513
Family LIFE, Autumn	High Flow Frequency	P: 0.01574 R^2: 0.2561 AIC: 34.872 On 17 degrees of freedom	Intercept: 6.44555 HighFreq: 0.06898

It can be seen in Table 3.15 that a number of statistically significant relationships were identified, with velocity trait score for both spring and autumn having P values below 0.04. Biodiversity in spring had a P value below 0.02, though no significant relationship was identified in autumn with all fitted models having P values over 0.15. For LIFE, a statistically significant relationship was observed in autumn with a P value below 0.02, and a possibly statistically significant relationship in autumn with a P value of 0.056.

Identified best fitting models were plotted against observed data in order to demonstrate how values generated by the fitted model compare with field data, and to better illustrate potential ecology-flow relationships (Figure 3.3).
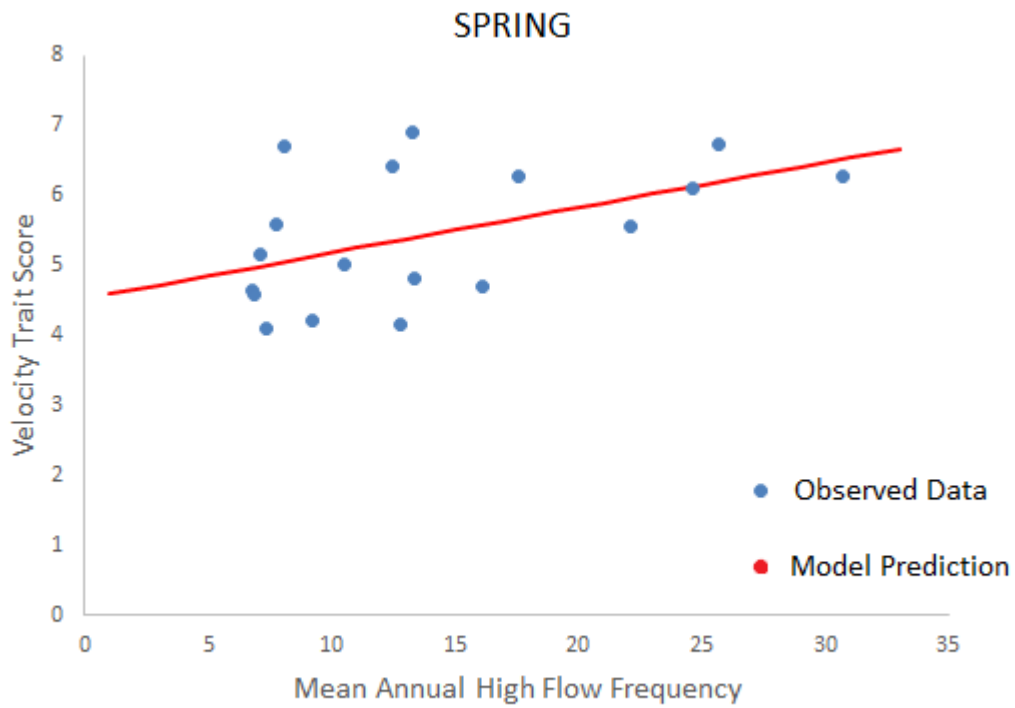
Figure 3.3: Velocity trait score plotted against mean annual high flow frequency, spring
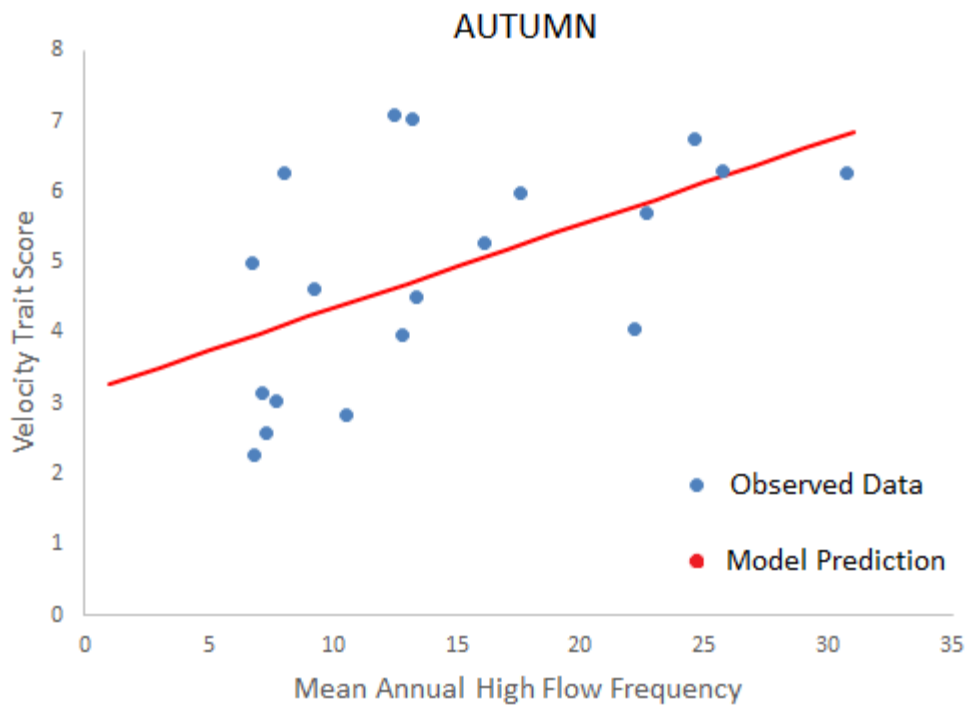
Figure 3.4: Velocity trait score plotted against mean annual high flow frequency, autumn

Figures 3.3 and 3.4 display spring and autumn ecological response in terms of velocity trait scores against mean annual high flow frequency, using the best fitted model from the multivariate analysis. These show that velocity trait score tends to increase as high flow event frequency increases; this is described further is Section 3.4.2.
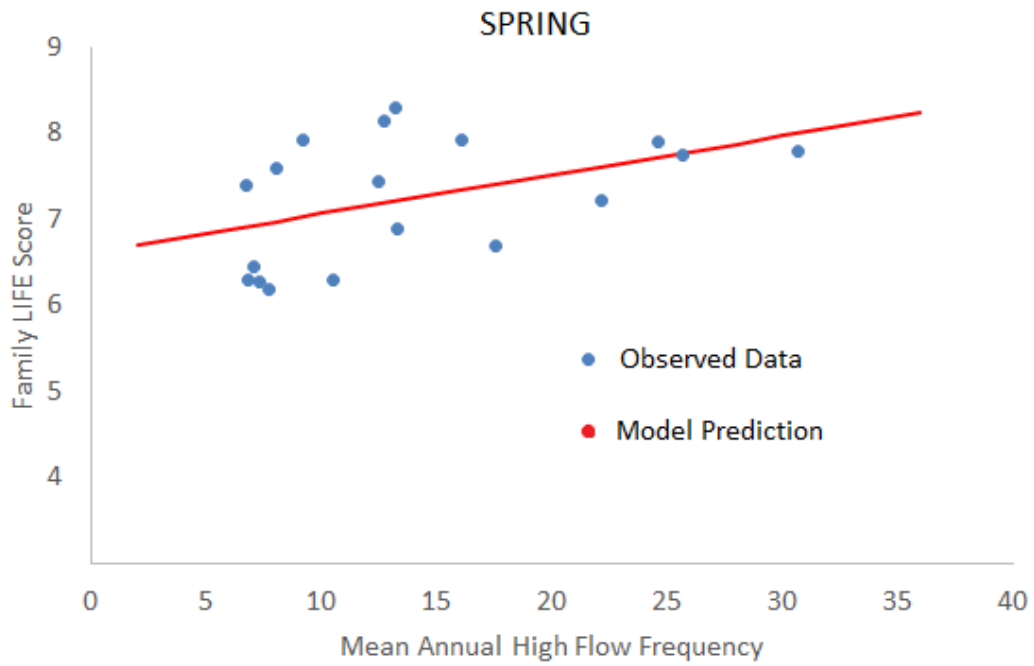
Figure 3.5: Family LIFE score plotted against mean annual high flow frequency, spring
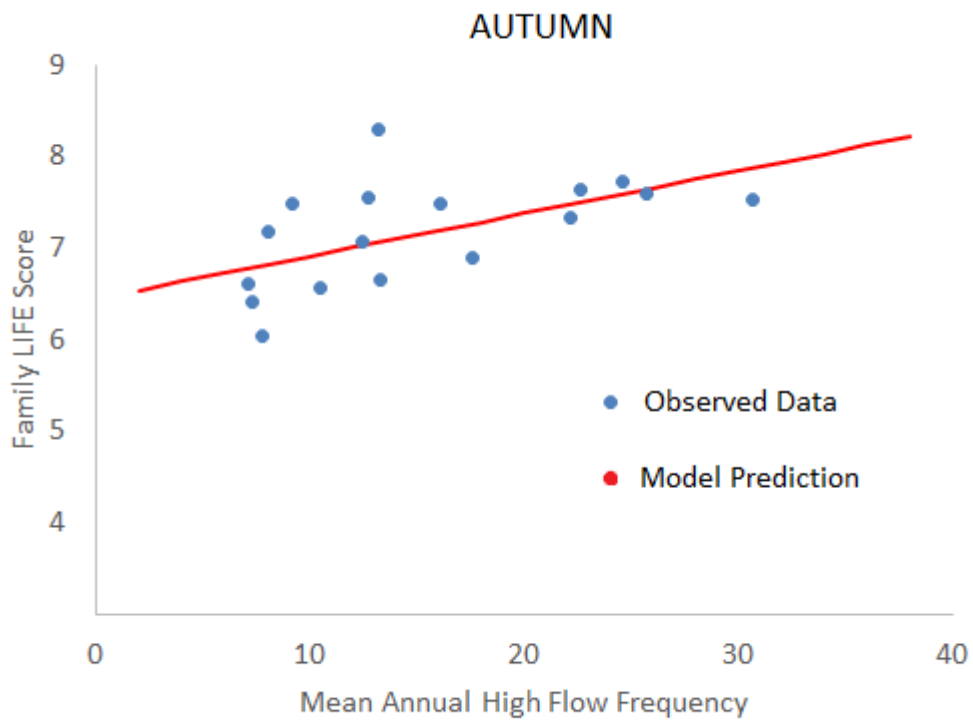
Figure 3.6: Family LIFE score plotted against mean annual high flow frequency, autumn

Figures 3.5 and 3.6 display spring and autumn ecological response in terms of family LIFE scores against mean annual high flow frequency, again using the best fitted model identified in the multivariate analysis. This relationship is very similar to the velocity trait score, increasing as high flow frequency increases. This is described further in Section 3.4.2.
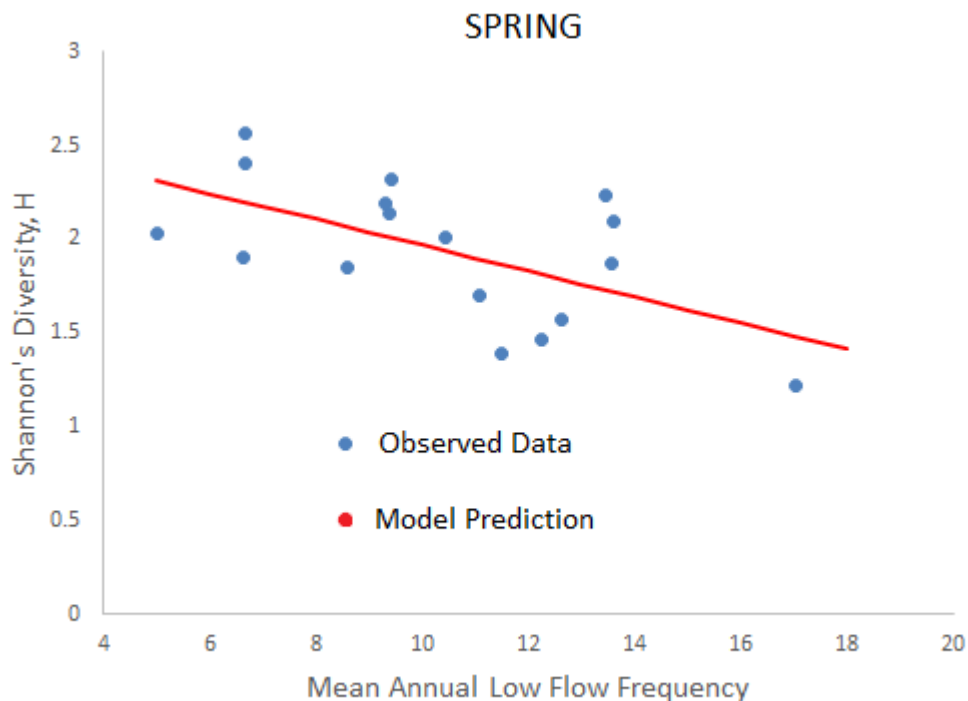
Figure 3.7: Shannon's Diversity plotted against mean annual low flow frequency, spring

Finally, Figure 3.7 displays the spring ecological response in terms of biodiversity using the best fitted hydrological variable of mean annual low flow frequency. Biodiversity tends to decrease as the frequency of low flow events increases. Section 3.4.2 describes this further and discusses possible mechanics driving the relationships observed. No relationship was found between biodiversity and hydrological variables in the autumn season as can be seen in Table 3.14, again Section 3.4.2 discusses possible reasons for this.

3.4 Discussion

This Chapter aimed to identify key flow characteristics influencing ecological response in the context of river impoundment flow modification. The results of this study support the consensus in literature, that deviation of flows from the natural regime are likely to lead to ecological changes in terms of biodiversity and/or functional composition (Poff and Zimmerman, 2010), as has been described in Section 2.2, due to the direct and indirect effects of flow such as bed forces acting upon benthic taxa and changes to connectivity within the river system (Shaw et al., 2016, Blanckaert et al., 2013). Statistically significant relationships were observed for both functional composition and biodiversity as seen in Section 3.3, with results described in Table 3.14. Relationships with P values below 0.05 were observed for all ecological variables except LIFE in spring, which had a P value of 0.056, and

biodiversity in autumn, where no discernible relationships were found. In this section, the results, implications, and limitations of this study are discussed.

3.4.1 Redundancy analysis

PCA demonstrated patterns of redundancy that were largely expected, and have also been identified in previous studies (Olden and Poff, 2003). "Magnitude" variables such as mean magnitude, Q95, Q10 and range tend to positively correlate; this is an intuitive result, given that a larger magnitude of river will have a larger mean magnitude and along with this higher percentile magnitudes (such as Q95), larger annual maxima, and generally higher annual minima. Magnitude variables are all primarily controlled by the catchment drainage area, ground water releases, and typical regional precipitation (Northwest River Forecast Center, 2006), assuming reservoir outflows are not dominating the entire flow regime at the site; such variables within any given river therefore share the same climactic drivers, hence high levels of redundancy between these variables.

The mean high and low flow event frequency variables strongly positively correlate and, conversely, duration variables for high and low flow events have strong negative correlation with frequency variables, whilst displaying strong positive correlation between one another. A likely cause of such behaviour is site morphology; a river may primarily have impermeable local geology leading to rapid surface runoff into the river system following rainfall events, with relatively little groundwater storage. This would generate short-lasting but frequent high and low flow events. A river with permeable local geology is likely to have few, but longer, high or low flow events due to permeable local geology; the long residence time of ground water fuelling flow event durations but being slow to respond to precipitation events (Dunne et al., 1991, Cardenas and Jiang, 2010). Rivers often display one of these two types of characteristics, and are described as either flashy or non-flashy respectively, based upon the rapidity and magnitude of river response to precipitation events.

3.4.2 Multivariate linear regression analysis

3.4.2.1 Velocity preference trait and LIFE Scores

Results suggest that, within the region investigated, high flow event frequency has a significant influence upon the functional composition of a system in terms of velocity preference of species. The levels of unexplained variation in the models suggest that external variables may also be playing a significant role in the functional composition of species at these sites. Previous studies have discussed the challenges presented by rivers as open systems and the degree of uncertainty often associated with studies investigating specific variables (Konrad et al., 2011, Arthington et al., 2006), and conclusive ecology-flow relationships have rarely been observed, partly due to such challenges and partly due to the need for further methodological development (Gillespie et al., 2015b). Results suggesting that singular variables among so many interdependent drivers might explain between 20-26% of total ecological variation are potentially an important new insight into the relationship between flow and macroinvertebrate ecology. IHA variables including the duration and frequency of high and low flow events were found to strongly influence stream macroinvertebrates in a similar study based on the ELOHA method in the U.S.A. using biological metrics primarily based on functional group composition such as measuring the percent of individuals adapted for filter feeding (Buchanan et al., 2013). Strong relationships were identified in Buchanan et al.'s study (2013) using Pearson's correlation between ecological metrics and IHA-derived flow variables. These similar findings suggest

that macroinvertebrates in general may be responsive to flow variation such as the frequency and duration of flow events, and this would highlight a significant problem with typical impoundment modification given the tendency of impounded system flow regimes to reduce flow variation as impoundment outflows are generally more homogeneous than natural flows (Poff et al., 2007). The ELOHA method is briefly detailed in Section 2.7.

The indication that high flow event frequency remains the primary driver between spring and autumn seasons, within both trait score and LIFE score variables, is suggestive of this flow characteristic being a primary ecological driver in terms of functional composition in Northern England. The likely mechanics causing this relationship can be deduced; preference for higher flow velocities increases with high flow event frequency according to model coefficients. This means that the more frequently high flows occur, the more resilient the general species population at a site becomes in terms of functional composition. One could ask whether this is due to species adapted to high flows flourishing in these environments, or whether this is due to less adapted species being unable to withstand the higher flow forces and thus being flushed downstream (thus high flow frequency acting as a biological filter). The latter is more likely to be the primary driving mechanic due to high frequency of event generally being associated with shorter event durations. A high frequency of flows means that high flow adapted species are exposed to a more variable flow regime, and are also experiencing lower flow conditions between flow peaks on a regular basis, and therefore are not constantly in a favourable environment. Therefore such species would not be in an environment in which they can take advantage of high flow conditions consistently. Washing out, however, is a rapid process caused by turbulent flow (Gabbud et al., 2019), and may act as a biological filter for species lacking resilience to high flow forces (Blanckaert et al., 2013). Frequent high flow events, therefore, are likely to enhance this biological filter and force taxon composition to become more populated by fast-flow affinity taxa. It has been proposed that it is not magnitude so much as the turbulence of flow that leads to macroinvertebrate dislodgement, due to turbulent flow being a stress upon behavioural and morphological adaptations (Blanckaert et al., 2013). High flow frequency, as opposed to flow magnitude-based variable, being a key driver of functional composition, supports the claim that frequent disturbance can have a significant impact upon benthic composition.

Higher R-squared and larger HighFreq coefficients in autumn suggest that high flow event frequency is a more influential driver during the autumn season, whilst influence is lessened in the spring period. This could be due to taxa in spring generally being more resilient to high flows, which is evidenced in the data as velocity trait score is higher in the spring period at 11 of the 18 comparable sites, with one site having the same score in both seasons and 6 having higher trait scores in autumn, though not significantly higher (>1.0). If the population is already adapted to high flow conditions to some extent, less ecological response might be expected as fewer taxa are being filtered by the mechanics discussed previously in this Section. Seasonal changes within the riverine environment may influence these differences in functional composition between seasons, or some variation may simply be due to life histories of the native taxa.

The influence of high flow event frequency as an ecological driver may have significant implications for water managers, particularly when applied to the region of Northern England. A lack of high flow events within a modified system may lead to a lack of an important biological filter. This could lead to systems being dominated by highly competitive generalist species, be they native or invasive, as discussed by a number of studies examining river deviation from natural flows (Summers et al., 2015,

Lytle and Poff, 2004). Incorporating a moderate frequency of high flows events into environmental flow regimes to mitigate the impacts of modification through impoundments may serve to balance a system's functional composition, and be one facet in ensuring a stable and diverse ecosystem. Conversely, systems with an over-abundance of high flow events will lead to a system dominated by rheophilic (fast flow) species resilient to disturbance events. Modified systems such as hydropower may cause such effects; downstream ecosystems in these environments may benefit from a reduction in high flow frequency, as is seen in a recent restorative study for a hydropower dam in Italy (Premstaller et al., 2017).

The robustness of quantified ecology-flow relationships may be increased as studies integrate larger quantities of data into analyses and provide results that may be compared with one another. However, an ongoing limitation for studies such as this is data availability; sites to be studied must have high quality available flow and ecological data (mean daily flows, annual spring and autumn macroinvertebrate sampling), that is temporally synchronised and in close proximity to one another. The site must also be relatively free of external ecological pressures such as water quality issues, and be within a particular range of flow magnitudes relative to the scale of rivers being investigated. Due to this somewhat demanding criteria, difficulties may be encountered in obtaining a sufficient quantity of sites across a region capable of facilitating robust statistical analysis. This is unfortunate as rivers, being complex open systems, tend to display significant variation between sites (Konrad et al., 2011) and therefore trends may be difficult to identify if observations are few. It may be the case that this applies even more so to larger scale river systems where greater complexity might be expected due to the wider range of potential influencing variables.

3.4.2.2 Biodiversity, H

Results suggest that low flow event frequency is significant in spring time in influencing site biodiversity. In autumn, no significant relationships were detected. This may be due to varying conditions between the two seasons; it has already been demonstrated that functional composition differed significantly between the two seasons either due to life history or external drivers, and again response to the flow modification may vary due to these differences in composition between seasons. Coefficients from the model suggest that more frequent low flows lead to lower biodiversity. It is not surprising that low flow frequency is identified here as a significant ecological driver; four mechanics by which low flows in general influence ecology have been investigated and their impacts are discussed by Rolls et al. (2012); low flows control the extent of physical habitat, thus impacting biotic composition, trophic structure and river carrying capacity. Low flows may also mediate change in physical and chemical conditions, in turn driving distribution and recruitment trends in biota. Further, low flows influence system productivity and biotic composition by altering the sources and exchange of materials and energy within the ecosystem, and can also restrict connectivity and limit habitat diversity (Rolls et al., 2012). Another study also suggested that low flows may influence oxygen levels within the riverine system and therefore impacts biodiversity (Pardo and Garcia, 2016). Pardo and Garcia (2016) observed a seasonal element to this impact, summer in their case, due to flow modification varying between seasons at their study site.

In addition to these general impacts from low flows, the frequency of low flow in itself is described by Rolls et al. (2012) as a "key biological filter". With increasing low flow frequency, species of low tolerance for such flows are wiped out, leaving more resilient species to dominate. Conversely, in the

absence of low flows, low flow affinity species may be outcompeted by higher flow preferring species (Rolls et al., 2012). Low flow events also have significant influence upon riverine physicochemistry, though this varies with site based upon other climatic variables. Low flows have been associated with changes in river temperature due to the smaller amount of water being heated (van Vliet et al., 2011). They may also influence water quality in various ways; increased pollutant concentration due to lack of dilution, (particularly problematic in agricultural catchments) or decreased concentrations due to pollutants not being flushed into the river due to a lack of runoff (Caruso, 2002). Quantifying such impacts are beyond the scope of this investigation, but do provide some insights as to why low flow frequency appears to be a significant driver of ecological response. Where high flow event frequency drives functional composition yet appears to be of little influence on species diversity, frequency of low flows appears to have the opposite function; a significant impact on biodiversity, but seemingly a lesser impact upon functional composition. This could be due to low flow frequency not being a cause of mortality specifically for fast-flow affinity species, provided that the events are not of prolonged duration, in contrast to high flows rapidly washing out slow-flow affinity species. Instead, low flows, as stated in literature, mediate chemical conditions and system productivity, and alter the physical habitat (Rolls et al., 2012). This may lead to long-term impacts upon site conditions, in contrast to the "immediate" impact of high flow disturbance, and thus the entire ecosystem is impacted as opposed to a particular functional group.

The influence of low flow event frequency, at least applied to the region of Northern England, could have significant implications for water managers wishing to increase biodiversity within managed systems. Whilst the model makes clear that a high number of low flow events is detrimental to system biodiversity, the impact of very few events is less clear due to the small number of times within this lower frequency range, and no sites falling below 4 low flow events per year. Due to the role played by low flows within natural river systems, it would be expected – as stated in literature – that such components would play a regulating role upon the ecosystem, preventing dominance by specialised and highly competitive species (Poff et al., 1997, Richter et al., 1996). A fuller understanding of the impact of an inordinately low frequency of low flow events upon biodiversity might be reached by specifically targeting a significant number of sites with few low flows (<6/year). One might hypothesise that an "optimal" regional frequency of low flows exists, below which biodiversity tends to fall as particularly competitive species flourish, free of low flow regulation. The investigation of such a hypothesis is beyond the scope and timeframe of this investigation, but could be a potential area of further study. As noted regarding functional composition, this study and others attempting to expand understanding on the influence of flow characteristics upon biological metrics may be limited by the availability of data in terms of sites that have sufficient data to be investigated.

3.4.3 General discussion

Results in this Chapter have provided evidence that there are key flow categories that are strongly associated with ecological response, and that significant predictive relationships can be found on a regional scale. The questions initially proposed by Research Questions 1 and 2 asked how we can assess sites in a more generalised manner, and can we develop a better understanding of specific ecology-flow relationships. These questions have been answered in part, and have provided results of scientific value both to the following Chapters in this thesis and to the field as a whole. However, more research will be required to fully resolve these questions, and this Chapter represents a foundation from which further investigation might be prompted. This Chapter has suggested one means by which

ecology-flow interactions may be better understood, and has identified and quantified specific drivers for a particular region and for particular indicator species, though there are inherent limitations with the approach. A limiting factor of this investigation may be the somewhat narrow scope of variables utilised; though this is advantageous in identifying significant flow characteristics, the use of exclusively flow-related drivers likely fails to explain the majority of variation within the system, and may fail to provide any significant relationships in some circumstances, such as in the case of biodiversity in the autumn period for this study. Interactions between flow, ecology, physical and chemical conditions exert influence upon taxon response (Summers et al., 2015), and this influence may not be well explained or accounted for when these categories of drivers are not accounted for. Bringing together such a complex web of interactions would bring its own challenges in terms of uncertainty and model complexity, and at this stage in the field of ecohydraulics it is likely best to begin with a narrow focus before attempting to derive ecological principles defined by physiochemical-flow interactions. Yet, caution should be taken when deriving ecological responses exclusively from hydrological characteristics; external variables such as river morphology and extent of anthropogenic modification also play significant roles and should be considered in studies for management and restoration purposes. Many factors contribute towards statistical variance in ecological data, and therefore it should be expected that statistical analyses generally explain only a portion of the overall variance between sites (Worrall et al., 2014).

Due to the limitations in any single approach, further research is required in order to gain an exhaustive understanding of these topics. Results do however affirm the general assumption that the magnitude, timing, duration and variability of flows influence biodiversity and functional composition – though transferable evidence is still not as common as desired. It can be inferred from this observation that highly modified flows, such as those observed within impounded systems, are liable to foster an ecosystem that diverges significantly from the waterbody's natural state. This conclusion is similar to findings from other studies that have investigated the effects of increasing flow modification and regulation upon downstream ecosystems (Gillespie et al., 2015a, Haxton and Findlay, 2008, Nichols et al., 2006). This Chapter also affirms the suggestion of Chinnayakanahalli (2011), that taxon richness and functional composition respond differently to flow alternation, and using only one of these metrics may fail to recognise significant changes within the ecosystem. A good example of this observed within this investigation is the contrast between the functional composition variables (trait and LIFE scores) in comparison to Shannon's Diversity in the autumn period. Shifts in functional composition relating to flow are identified in both seasons, but most strongly in the autumn period; in contrast, a significant relationship between Shannon's Diversity and flow is identified in spring but not in autumn. If diversity alone had been the metric used in this investigation, the strong influence of high flow event frequency upon ecological response would not have been identified. This investigation echoes the call of other recent studies, that a broader suite of ecological metrics are required in order to fully evaluate changes within the ecosystem (Arthington et al., 2018, Poff et al., 2017).

As far as I am aware, no other studies have demonstrated statistically significant, quantitative hydrological response relationships for macroinvertebrates for this particular region and magnitude class of river. Results from this study may be used for water management decisions across the North of England, to supplement future studies in the region, or to draw comparisons between other river classes. It could also be tested to what extent such relationships hold true beyond Northern England; throughout England, or even throughout temperate climates in Europe. The exhortation in recent

literature to standardise and present a greater body of transferable methods and results through focused studies is not yet showing itself to be evident; the most recent studies on environmental flows rarely focus on this drive towards general regional ecology-flow principles. Those that do are spread across different fields such as hydropower or abstraction and are of limited transferability between one another due to the significant differences in the nature of flow modification taking place. A number of studies are site-specific, or have focuses very different to this investigation, such as flow regulation impacts upon floodplain rivers (Hayes et al., 2018), riverine impacts during dam construction (Santos et al., 2017) and fish trait analysis in the context of impoundment (Lima et al., 2017). Other studies review recent progress in the field (Poff et al., 2017, Arthington et al., 2018). Within such a broad field of research, there is a need for more studies to orientate themselves towards developing generalised flow-ecological relationships within a reservoir-impoundment context. Within this context much potential for study exists, both for a variety of ecological indicators (e.g. fish, macroinvertebrates and macrophytes), and for a variety of regions, environments, and river classes.

Results from this study offer potential for further research. From the results, one can derive principles for similar sites within this region, such as higher flow frequency generally leading to a functional composition shift towards fast-flow preferring species. The breadth of possible application would be an interesting potential research question; could derived principles apply across similar rivers throughout England, or perhaps Western Europe and other temperate regions? An important aspect of investigations such as this is their ability to be compared, contrasted and tested with others through meta-analyses, potentially enabling broader relationship principles to be identified. Thus far such analyses have not met success in this regard (Poff and Zimmerman, 2010, Gillespie et al., 2015b), and it is therefore necessary for further regional and class-based studies to be proliferated for the benefit of general academic knowledge. This investigation is necessarily limited in its explanatory scope to the region of Northern England, for rivers of similar magnitude, due to its focus upon this particular class of river. As has been discussed, magnitude is a particularly strong ecological driver, impacting river morphology, physicochemistry and native taxon composition (Monk et al., 2006), therefore attempting to apply principles across a broad range of magnitudes will likely prove difficult unless the influence of magnitude can be identified and accounted for; an area in itself requiring further intensive study.

3.5 Conclusions

This Chapter finds significant relationships between flow characteristics typically affected by flow regime modification, particularly high and low flow event frequencies, and ecological responses. These results are from sites within Northern England, comprising rivers with an average discharge of 1.34m^3/sec, and a mean discharge range between 0.31-4.21m^3/sec. The results offer potential principles by which ecology-flow interactions may be better understood, specifically the tendency of functional composition to shift towards faster-flow affinity species as high flow event frequency increases, and the tendency for biodiversity to decrease with as low flow event frequency increases. Broader application of the principles derived from my results would first require verification of these results based upon a statistical analysis of ecology-flow response within a new region or class of river.

Statistical analysis is limited in its predictive ability; channel morphology plays an important role, interacting with magnitude and leading to a diverse range of hydraulics dependent upon the structure of the river channel. Further investigation and developments in the field of ecohydraulic statistical

analyses will help to improve the effectiveness of environmental flow planning to promote sustainable in-stream habitat conditions. One must always keep in mind that solutions provided through such methods are probabilistic in nature and thus a risk-based approach should be taken when drawing conclusions from the data (Turner and Stewardson, 2014). Results and implications from this Chapter contribute towards Chapter 5, having highlighted that temporal flow characteristics are very likely to have a significant impact upon riverine ecological response. An outcome of this is that the magnitude-based predictions of ecological response used in Chapter 5 must be further supplemented by temporal variation across the flow regime; the methodology for achieving this is discussed in Chapter 5. This Chapter has thus worked towards meeting two key aims within this investigation; the first being to better understand and attempt to quantify general ecology-flow relationships, the second being to design an environmental flow regime at a particular case study site.

4. DEVELOPING A HYDRO-ECOLOGICALLY LINKED MODEL TO EVALUATE AND ADDRESS MACROINVERTEBRATE RESPONSE TO FLOW MODIFICATION AT A CASE STUDY SITE[*]

[*]Flow designation through the approach described throughout Chapters 4 and 5, and the results generated (particularly in Section 5.3), are of significant worth and represent a promising area for further research; a paper comprising a condensed version of Chapters 4 and 5 has been accepted and published by the journal Ecological Indicators, see Appendix 9.1.

4.1 Introduction

A primary goal of this thesis is to present a potential transferable methodology by which impoundment-modified river systems may be assessed, and environmental flows designated. Chapter 3 identified statistically significant relationships between flow characteristics and macroinvertebrate functional composition and biodiversity, however, these findings alone are not sufficient to inform an environmental flow regime. A more detailed understanding of how flows affect river channel hydraulics (and in turn impact local benthic taxa) both spatially and at a reach-wide scale, is required in order to prescribe flow changes with a firm scientific basis. This Chapter contributes towards the overall goal of flow designation through the development of a 2D hydraulic-ecological linked model at a chosen study site, capable of predicting habitat quality for chosen indicator species based on flow inputs. The model's capabilities are demonstrated by comparing model predictions with observed species populations from field sampling. This feeds into model outputs, findings, and the environmental flow regime design process described in Chapter 5. The methodology aims to be transferable in the context of similar systems, addressing the challenge of site-wide flow regime designation through a novel combination of habitat quality prediction (based on 2D ecological model outputs), flow event timings, habitat diversity, and flow event heterogeneity, whilst also making efforts to actively conserve water relative to current outflows. The methodology proposed takes steps towards an answer for environmental flow designation and implementation; designed flows should in principle provide significant benefit to the ecosystem (Richter et al., 1996), whilst also conserving water resources; both highly desirable outcomes within our current economic and social climate. The impoundment addressed, Holden Wood, and the study reach it sustains, Ogden Brook, were selected to test this methodology, and are described in greater detail shortly.

This chapter aims to address the following research questions initially posed in Chapter 1:

Research Question 1. How can ecology-flow interactions be better understood in a more transferable manner?

Research Question 3. How can localised hydraulic requirements of taxa be translated into an ecologically beneficial flow regime?

Chapter 2 discussed the implications of flow modification causing systems to deviate from their natural conditions, and the challenges associated with the mitigation of flow modification impacts. While it may not be feasible to return flows to their natural regimes in most cases, an increasingly popular approach has been that of 'Designer Flows', by which flow patterns are created to provide desired benefits, within practical constraints (Chen and Olden, 2017), also discussed in Chapter 2. Such an approach is followed in this Chapter and the subsequent Chapter, 5, for the development and

implementation of an hydraulic-ecologically linked model by which an environmental flow regime can be designated. Macroinvertebrate species are utilised as ecological indicators, due to their relative neglect in the field when compared to taxa such as fish (Gillespie et al. 2015b), and the fact that they are a more prolific indicator at the scale used. These taxa experience flow as localised forces as opposed to overall magnitudes, timings, etc. This raises the question posed in Chapter 1; how can the requirements of invertebrates on a micro scale translate into an overall compensation flow and its inter-annual variation? Habitat quality models are an increasingly utilised approach in restorative studies (Reiser and Hilgert, 2018, Schneider et al., 2016, Conallin et al., 2010), yet may not account for life history requirements and temporal flow characteristics experienced by taxa such as the frequency and duration of flow events. A broader suite ecological indices are required to achieve robust environmental flow designs (Chinnayakanahalli et al., 2011, Arthington et al., 2018), and methodological progress is required in order to determine their implementation; how are conflicting flow needs to be resolved, and how does one judge whether or not a flow regime is "good"?

Competing interest groups and increasing demands for water supply mean that environmental needs are a contentious topic; water sent downstream for environmental purposes must be well-justified, and the "cost-benefit" in terms of water committed to environmental flows must be acceptable in order to maximise the volume of water retained for societal use (Harwood et al., 2018). The lack of transferable ecology-flow principles can necessitate time- and cost-intensive site-specific investigation (Anderson et al., 2017), and the impracticality of scaling up such an approach to larger or multiple sites is readily apparent. One potential solution gaining favour is to use regional-based methods as discussed in Section 2.6.3. However, even these relationships have proven difficult to extract from the current body of literature, largely due to challenges associated with a lack of standardised approaches and difficulties in the synthesising current available data, as discussed in Section 2.5.

Conventional methods of developing environmental flows, such as the before/after control impact approach (BACI), working under a flow naturalisation paradigm, are concerned with maintaining a status quo within the river system by comparing an impacted site with a non-impacted control site (Underwood, 1991). An example of this would be bringing the flow of a regulated river more in line with the flow regime of a similar control site. Such an approach requires a valid control site that corresponds well with the impact site, both rivers displaying similar flow magnitudes, hydraulics and geomorphology. However, such a site is not often readily available, making it difficult to determine how the river would differ from its current state were it not regulated. Another drawback of BACI methods is that it is impossible to predict the long-term response of taxa within the river system to changes in flow, it can only bring an impact site "in line" with a theoretical ideal in which ecological response is uncertain due to river complexity and potential external drivers. The naturalised "ideal" may not be the most beneficial regime for the ecosystem. This means that BACI approaches would likely require laborious and site-specific in-stream flow experimentation before an ongoing flow regime is determined. Additionally, such ideals sought under the paradigm of flow naturalisation are often neither feasible nor the optimal solution in a world where river modification for societal use is the case for a majority of river systems; the designer paradigm (discussed in Chapter 2) focused upon maximising benefits (such as desirable species), and mitigating anthropogenic impact, is increasingly preferred (Acreman et al., 2014).

This thesis proposes predictive modelling as a viable alternative to the site-limited BACI approach, resource-intensive BBM approach, and typical current "rule of thumb" (Arthington et al., 2006) flow

solutions such as percentile flow thresholds. The approach proposed involves using models to design optimised flow provision both for both environmental and societal needs. The CASiMiR ecological model will be used as a tool for the development of environmental flows at a case study site. The individual merits of CASiMiR will be discussed later, and the general merits of modelling discussed first. Computer modelling of a system allows for environmental flows to be developed and system response predicted without the need for a suitable reference site (which often are not available) or the use of flow experiments (which are costly, labour intensive and require long-term assessment). One of the primary aims of this project was to provide a water utility company with a potential methodology to establish environmental flow regimes, with accompanying recommendations. As the time and resources required to perform flow experiments within physical catchment systems is intensive and therefore generally unfeasible for numerous smaller scale sites, an alternative method is proposed which may be transferable across such sites. Flow modelling of a case study site enabled simulated flow manipulations and predicted responses to be evaluated, providing insights into site responses to changing flow regime and enabling rapid outputs and greater control when compared to in-stream flow experimentation. A simulation model allows for a high degree of control over the variables within a system, allowing singular variables to be selectively manipulated and the outcomes observed. Thus a number of flow requirements can be identified and individually altered with the aim of eliciting the best ecological response within the model. Subsequent ecological response to flow regimes over, for instance, a year, can then be outputted in short order in the form of habitat suitability maps, calculated by changes in local hydraulics across the modelled reach – a significant advantage in contrast to the long-term approach required in field monitoring of ecological response.

A significant output from this task was the end-product model, results, and associated framework. With guidance as to how it may be adapted to other sites, it might see application at other United Utilities sites in order to inform ecology/flow interactions; particularly the many small-scale sites similar to Holden Wood present across the UK which could not all feasibly undergo real-time flow experimentation and assessment. As mentioned, the end goal of the model was to produce recommended flow regimes which optimally balance the potentially conflicting interests of societal use and environmental flow provision.

4.2 Choice of study site

The Holden Wood impoundment and its associated water body, Ogden Brook, were selected from a number of possible sites managed by United Utilities. The choice of sites was narrowed down due to the presence of a consulting company, Cascade Consultants (here on referred to as "Cascade"), working on a handful of sites during the course of this investigation (Cascade Consulting, 2016), providing the opportunity for collaboration and data sharing. Three sites were visited with Cascade; Holden Wood, Fernilee, and Etherow. Holden Wood was favoured both due to a lack of practical constraints such as site access or complications posed due to private land around the site. Additionally, Ogden Brook is a small scale site (fully detailed in Section 4.3) with flow regulated almost entirely by its upstream impoundment, being within 100m of the impoundment outlet. In contrast, most previous studies in the literature have focused upon the investigation of larger river systems; it is difficult to isolate or control ecologically-influential factors at this scale (for example, tributary flow inputs). It was believed that a small scale study site (approx. 40m) may present value by allowing the development of a foundational approach towards environmental flow development that can later be scaled and adapted to account for complexity in larger systems. The site also presented no significant

external ecological pressures such as water quality issues, and little to no vegetation in the channel which may act to influence ecological distribution and flow hydraulics, again aiding to isolate ecologically-influential factors as much as possible.

4.3 Site description

The study site Ogden Brook (Figures 4.1a and 4.1b) is a stream system in the North West of England, directly downstream of the impounding reservoir Holden Wood, approximately 27.2km North of Manchester, Northern England, and located near the village of Haslingden, OS grid reference SD776220. The site was chosen due to data availability, lack of significant external pressures such as wastewater inflows, the dominance of upstream reservoir compensation flows, and its appropriate scale for the scope of the investigation. Historical background has been adapted from a consultancy report provided by the regional water company, United Utilities (Phillips, 2016). Typical flow conditions at the site remain in the range of Q=0.02-0.04 m^3/s, with mostly shallow depths between 0.1-0.25m, though recorded depths in pools reached as high as 0.8m. The reach under investigation is approximately 40m in length, primarily chosen due to the presence of a downstream tributary avoided in this study so as to retain flow contribution solely from reservoir outflows. The study is thus performed on a small scale; this fits the aims of this investigation, which focuses upon the response of taxa at a micro-habitat level similar to studies such as LeCraw and Mackereth (2010) who utilised study reaches of 10m to observe localised ecological responses, or other fish and macroinvertebrate restoration studies utilising 100m reach scales (Pretty et al., 2003, Harrison et al., 2004).
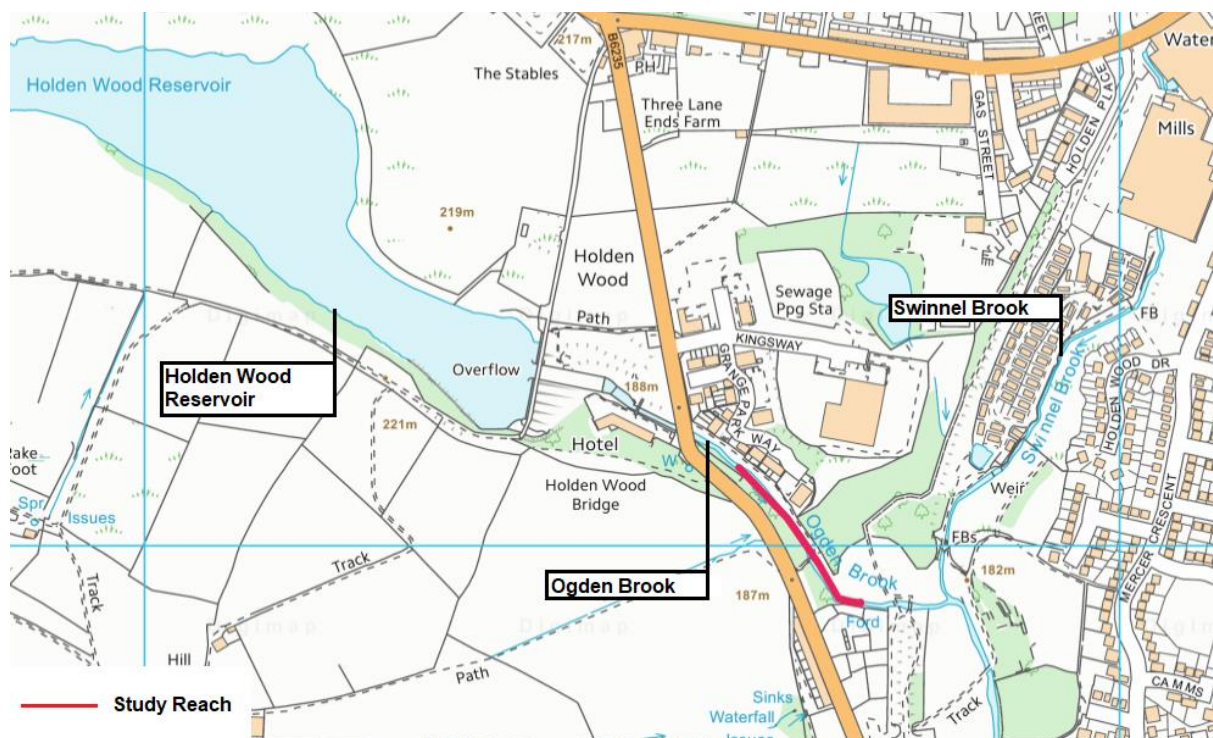
Figure 4.1a: The Holden Wood reservoir and Ogden Brook, OS Map produced using Digimap, developed by EDINA, original map data from Ordinance Survey (GB) (Digimap Ordinance Survey Service, 2018)

Figure 4.1b: A cross-section at the Ogden Brook site, pictured in November 2017

The water company United Utilities manages Holden Wood reservoir and releases into Ogden Brook, the river under investigation. Ogden Brook runs along a narrow band of woodland surrounded by light urban development (Figure 4.2). The riverbed itself is mostly gravel, with top-layer sediment ranging from small pebbles (~1cm) to larger stones (up to ~10cm), with a few larger rocks (up to ~20cm) scattered throughout the reach. A lower layer of finer sediment lies beneath the stones and cobbles. The river channel has little to no vegetation.

Figure 4.2: Satellite view of Holden Wood Reservoir and Ogden Brook

Presently, Holden Wood is required to release 3.46 megalitres per day ($0.041m^3$/s) of flow during times of the impoundment being within 2 metres of its maximum water level, and 1.84 megalitres per day ($0.0215m^3$/s) when water depth is below this point. Prior to 2016, release requirements were lower; within the range of 0.01-0.02 m^3/s. Impoundment releases are the sole major contribution to the studied reach of Ogden Brook, aside from small amounts of direct runoff insignificant relative to overall flow. Mean daily flow data for outflows from Holden Wood between 2014 and 2017, and inflows between 2010 and 2014 were provided by United Utilities, derived from cumulative inflow and outflow metres read and recorded daily; an outflow meter on the spillway measures the volume of spill events when these occur, and both outflow metres are added together for overall reservoir

outflow. Macroinvertebrate single-point, three-minute kick sampling data from spring and autumn of 2016 conducted by a local environmental consultancy within the analysis reach were also made available; this was used to assess typical seasonality of native taxa.

United Utilities holds an abstraction licence 2569001165 which governs abstraction for public water supply from Calf Hey, Ogden and Holden Wood reservoirs and also Musbury Brook. The licence allows abstraction from Holden Wood Reservoir via surface mounted pumps, when needed, up to a maximum of 7.160 Ml/d1 and 40,915 Ml/week. The site (figure 4.2) was listed as a potential drought source in 2008 but was later discounted, as the new compensation flow set in the impoundment licence resulted in no spare water being available for abstraction from Holden Wood Reservoir. Section 4 of the Bury and District Joint Water Board 1928 required a compensation flow release from Holden Wood Reservoir to the downstream Ogden Brook as provision for downstream mill owners. The Act required a release of 6.66 cubic feet/sec for 10.5hrs every working day, except for 7.5hrs on Saturdays (this equates to 5.767 Ml/d), and a further 2.478 cubic feet/sec of water could be requested by the mill owners, up to maximum release of 6.66 cubic feet/second. North West Water Authority purchased the downstream mill and applied for a licence variation on 22 March 1991 to eliminate the requirement to release water from Holden Wood Reservoir. However, a voluntary flow release was maintained for Ogden Brook.

Due to the need to undertake major structural improvements to the spillway at Holden Wood Reservoir, United Utilities (UU) had to apply to the Environment Agency for an impoundment licence. This was issued on 3 February 2012. The impoundment licence set a compensation flow release requirement of 0.58 Ml/d until 31 December 2016, increasing to 5.766 Ml/d from 1 January 2017 (equivalent to the original release rate required by the Bury and District Joint Water Board 1928). However, Holden Wood Reservoir is unable to sustain a release of 5.776 Ml/d, as its reliable output during a drought event has been assessed to only be 1.94 Ml/d (Grontmij, 2013). Given that there is no abstraction from Holden Wood Reservoir at this time, this assessment represents the volume of water that could be reliably released during the worst drought on record without the reservoir running out of water. An assessment of the natural low flow at Holden Wood Reservoir was undertaken using the Environment Agency's Low Flows Enterprise software package which gave a Qn95 value of 3.54 Ml/d (Grontmij, 2013). Therefore United Utilities wished to complete a study to determine the impact of different flow releases on the downstream Ogden Brook, in particular the brown trout population. The aim of the study was to gather evidence to support an application to the Environment Agency, before 1 January 2017, to reduce the compensation flow that is required. The risk of releasing a flow in excess of 1.94 Ml/d is that at times of drought, the reservoir could empty and run out of water, resulting in very low or even no flow to the downstream river. As there is no abstraction from the reservoir, its water level will only vary as water is released to the downstream river – the higher the compensation flow, the greater the extent and duration of any reservoir drawdowns. As a result of this, the consultancy company Cascade suggested that a lower compensation flow will allow the reservoir to spill more frequently, providing a more variable flow regime in the downstream river. This is largely true, though this will be much more the case in seasons of higher precipitation and less so in times of lower precipitation, and it is the opinion of this thesis that such a regime may not be suitable in terms of environmental provision due to the reliance upon spills; such events are not predictable and may not behave in the same manner as natural flow within an unmodified system, and therefore it may be difficult to justify this approach as a complete solution to the issues posed by flow modification.

As of the time of writing, Holden Wood is required to release 41 L/s ($0.041m^3$/s) of flow during times of the reservoir being within 2 meters of the maximum depth limit, and 21.5 L/s ($0.0215m^3$/s) when reservoir depth is below this point. These values were taken from one of the site engineers in February 2017. Field measurements are close to these values, with flow being measured at $0.028m^3$/s in the observations of this investigation. The additional flow is likely due to runoff from a recent rainfall event; though reservoir flows are dominant, runoff could make a small contribution to flow when the area has been significantly wetted. Background data was available and utilised in this study. Mean daily flow data for outflows from Holden Wood between 2014 and 2017, and inflows between 2010 and 2014 were provided by United Utilities, derived from cumulative inflow and outflow meters read and recorded daily. Macroinvertebrate single-point, three-minute kick sampling data from spring and autumn of 2016 were also made available by the consultancy firm Cascade, used to verify primary field data collected and to assess typical seasonality of native taxa.

4.4 Model description and rationale

As mentioned in section 4.1, a deeper understanding of the influence of channel hydraulics on river benthos was required in order to translate the flow requirements of taxa into a flow regime with defined flow magnitudes. Macroinvertebrates utilised in this thesis live on the channel bed, and therefore experience flows as near-bed forces. Flow velocity is an appropriate surrogate for these forces within typical systems (Lancaster and Hildrew, 1993). Therefore, the primary concept of model development was to identify the flow preferences of selected indicator species, define these preferences as flow velocities, and identify the flow magnitude inputs that result in good ecological conditions to form a significant part of overall environmental flow regime designation.

4.4.1 Choice of model: 1, 2, or 3 Dimensions?

Predictive modelling can be performed using 1D, 2D or 3D systems. Each of these choices have significant advantages and limitations. 1D modelling offers simplest approach of the three choices, being much easier to develop, having significantly lower data requirements, and being easier to calibrate. Additionally, 1D models are the least computationally demanding of the three options and are thus able to feasibly model on larger scales than 2D or 3D, such as at catchment-level or regional scales. This can be particularly useful when making assessments at a macro-level that do not require in-stream information, such as flood modelling (Mashriqui et al., 2014) or the differing impacts of morphology on flow over a region (Saleh et al., 2013). Examples of 1D hydraulic models include HEC-RAS 1D and SRH-1D; generally such models see use in flood modelling and sediment transport, where lateral spatial dynamics are less important, flood modelling mainly being concerned with channel overflow, sediment transport mainly being concerned with volume and type of sediment taken up or deposited (Mashriqui et al., 2014, Sabatine et al., 2015). However, at the scale this investigation is working on a 1D model is less appropriate due to it providing far less information, such as how hydraulics might vary laterally across the river channel. Section 4.2 described the choice of scale for the study site; a smaller site allows for the desired ecological driver (i.e. flow velocity) to be isolated without further complications being introduced such as changing sediment, or flow inputs external to the impoundment. In turn, level of detail required for the model was informed by site scale and the requirements of the investigation. Information such as lateral flow velocity distribution is vital to this investigation, as we wish to consider both spatial and whole-site ecological impacts of flow. The advantage of large scale coverage in a 1D model was not relevant to the site used in this investigation,

whilst the level of detail provided by such a model was insufficient for the requirements of the approach, and therefore 1D modelling was not utilised.

3D modelling stands in contrast to the 1D approach, in that it is the most complex of the three, providing data for x, y and z dimensions. As such, more data is required, such as highly detailed channel geometry and observations of in-stream turbulence; calibration is intensive as there are more factors that must conform to field observations. There is also significant computational demand for running such models. These data, labour and computational requirements made 3D modelling a less ideal choice. It has been claimed that 3D models provide most robust predictions, and that the z dimension can be an important aspect of ecological pressure and response (Pisaturo et al., 2017). However, in the case of Pisaturo et al, the study was performed within a much larger river system of significant depth, magnitude, and velocity. Discussion with experts, and the success of studies utilising 2D models even in larger river systems (Jowett and Duncan, 2012) leads this thesis to propose that in smaller-scale systems such as Holden Wood, the 2D modelling approach is more desirable and that in these environments no vital information is overlooked by discounting the z dimension. An example of a 3D model is Delft-3D; such models are commonly used relating to the transport and dispersion of pollutants, nutrients, or saline water in environments where there is significant variation in the vertical flow field, resulting in the material in question not being well-mixed throughout the entire water column, or relating to dispersion at sea where depth cannot be discounted (Zhao et al., 2013), and more commonly within industrial environments such as turbines in which 3D velocities are an important factor (Gartner et al., 2016).

2D modelling lies between 1D and 3D approaches in terms of complexity, labour requirements, and information output. 2D models require boundary inflow and outflow data and relatively high-resolution bed geometry data. Calibration and verification involves assessing the spatial distribution of model accuracy; flow velocity in the case of this investigation. 2D models provide the information this investigation is primarily interested in; the spatial distribution of flow velocity both laterally and longitudinally in the modelled reach, and this in turn can be used to predict spatial habitat quality distributions, using the assumption that depth-averaged velocity has a close relationship with near-bed velocity in rivers of this type. This assumption generally holds true in typical river systems, though the relationship can become less defined with increasing bed complexity. Possible variation in velocity with depth is demonstrated in Figure 4.3.
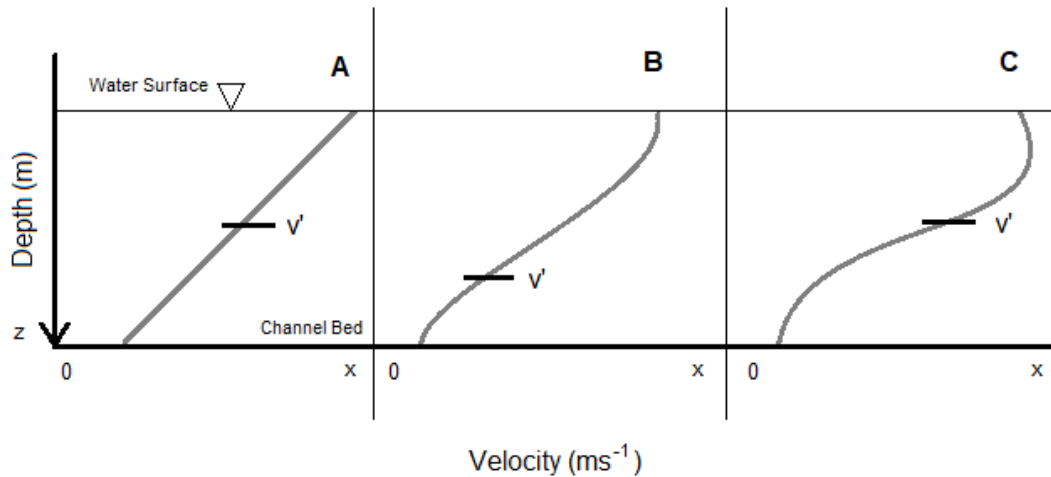
Figure 4.3: Possible scenarios demonstrating variation of velocity throughout the depth of the water column within different systems of varying morphologies; hypothetical systems A (flume-like), B (typical natural system), and C (complex system).

Due to this a 3D model would be ideal were resources, data and computational power not limiting factors, due to a 3D model's ability to display flow velocities throughout the river depth, predict possibly turbulence in flows, and thus reduce uncertainty. In contrast, a 1D model provides the least detail of the three models, simply providing possible ecological response to flow with no spatial component across the channel. Given resource limitations, current computational capacity, and the fact that hydraulic assumptions are expected to hold true within the chosen study site, a 2D model was seen as an appropriate choice and a good compromise between level of detail and overall practicality. Choice of specific 2D model, and why one particular model was chosen, is discussed below.

4.4.2 The SRH 2D/SMS hydraulic model

The SRH-2D model with the SMS interface was chosen primarily for practical reasons. The CASiMiR ecological model was selected first, and has a small selection hydraulic models already integrated with the software. This investigation therefore had the choice of SRH-2D, River 2D, or a GIS-based module as viable options. The GIS module required advanced knowledge of GIS and was not recommended for unfamiliar users, whilst having no advantage over the other choices, so this option was avoided. River 2D was initially selected as it is a free piece of software, but was discounted due to the fact that it was not regularly updated, did not have significant support, and produced questionable results when the model was initially tested, possibly due to compatibility issues with Windows 10. SRH-2D is a licenced piece of software with a user interface facilitated by SMS; it is well supported and fully compatible with the operating system used, and is a well acknowledged model in the field of hydraulics (Lai, 2008, Zarrati et al., 2005), thus the selection of this model was seen to be well justified.

The following describes the SRH 2D model as discussed by its distributor, AQUAVEO;

"SRH-2D is a hydraulic model developed by the U.S. Bureau of Reclamation that incorporates very robust and stable numerical schemes with a seamless wetting-drying algorithm. The model uses a flexible mesh that may contain arbitrarily shaped cells, both quadrilateral and triangular elements, which promotes solution accuracy while minimizing computing demand. SRH-2D modelling

applications include flows with in-stream structures, through bends, with perched rivers, with side channel and agricultural returns, and with braided channel systems. SRH-2D is well suited for modelling local flow velocities, eddy patterns, flow recirculation, lateral velocity variation, and flow over banks and levees."

A list of the major features of SRH-2D, below, has been taken from the SRH-2D Theory and User's Manual from the US Bureau of Reclamation (Lai, 2008):

"• 2D depth-averaged dynamic wave equations (the standard St. Venant equations) are solved with the finite-volume numerical method;

• Steady state (with constant discharge) or unsteady flows (with a flow hydrograph) may be simulated;

• An implicit scheme is used for time integration to achieve solution robustness and efficiency;

• The model may make use of a quadrilateral mesh, a purely triangular mesh, or a combination of the two. Cartesian or raster meshes may also be used. In most applications, a combination of quadrilateral and triangular meshes is the best in terms of efficiency and accuracy;

• All flow regimes, i.e., subcritical, trans-critical, and supercritical flows, may be simulated simultaneously without the need for special treatments;

• Robust and seamless wetting-drying algorithm; and

• Solved variables include water surface elevation, water depth, and depth averaged velocity. Output variables include the above, plus Froude number, bed shear stress, critical sediment diameter, and sediment transport capacity."

SRH-2D is a 2D model, and is therefore of particular use when addressing problems that have important 2-dimensional aspects (Lai, 2008). Within this thesis, I wished to explore the lateral and longitudinal impact of flow input, particularly relating to its effects upon local flow velocities, thus SRH-2D's outputs are highly relevant to the problem presented.

SRH-2D has the following limitations:

• Only flow is modelled in the version utilised by this project. Mobile-bed sediment transport and temperature and vegetation modules are available but are beyond the scope of this investigation (Lai, 2008);

• SRH-2D does not have its own user interface. Users need to have access to other software for mesh generation and result post-processing (Lai, 2008). This investigation uses SMS for this role, the Surface-Water Modelling System, available from Aquaveo. SRH-2D is also compatible with other graphical post-processing software such as ArcGIS and TECPLOT.

4.4.3 SRH-2D equations

Much of the following explanation is adapted from the SRH-2D handbook (Lai, 2008):

Most open channel flows are relatively shallow and the effect of vertical motions is negligible for the purposes of this investigation. The general three-dimensional Navier-Stokes equations may therefore

be vertically averaged to obtain a set of depth-averaged two-dimensional equations (Lai, 2008). This leads to the following 2D St. Venant equations:

$$\frac{\partial h}{\partial t} + \frac{\partial hU}{\partial x} + \frac{\partial hV}{\partial y} = e \qquad (1)$$

$$\frac{\partial hU}{\partial t} + \frac{\partial hUU}{\partial x} + \frac{\partial hVU}{\partial y} = \frac{\partial hT_{xx}}{\partial x} + \frac{\partial hT_{xy}}{\partial y} - gh\frac{\partial z}{\partial x} - \frac{\tau_{bx}}{\rho} = D_{xx} + D_{xy} \qquad (2)$$

$$\frac{\partial hV}{\partial t} + \frac{\partial hUV}{\partial x} + \frac{\partial hVV}{\partial y} = \frac{\partial hT_{xy}}{\partial x} + \frac{\partial hT_{yy}}{\partial y} - gh\frac{\partial z}{\partial y} - \frac{\tau_{by}}{\rho} + D_{xx} + D_{xy} \qquad (3)$$

In the above, t is time, x and y are horizontal Cartesian coordinates, h is water depth, U and V are depth-averaged velocity components in x and y directions, respectively, e is excess rainfall rate, g is gravitational acceleration, T_{xx}, T_{xy}, and T_{yy} are depth-averaged turbulent stresses, D_{xx}, D_{xy}, D_{yx}, D_{yy} are dispersion terms due to depth averaging $z = z_b + h$ is water surface elevation, z_b is bed elevation, ρ is water density, and τ_{bx}, τ_{by} are the bed shear stresses (Lai, 2008). Bed friction is calculated using the Manning's roughness equation as follows:

$$\begin{pmatrix} \tau_{bx} \\ \tau_{by} \end{pmatrix} = \rho C_f \begin{pmatrix} U \\ V \end{pmatrix} \sqrt{U^2 + V^2}; \qquad C_f = \frac{gn^2}{h^{1/3}} \qquad (4)$$

Where n is the Manning's roughness coefficient.

Turbulence stresses are based on the Boussinesq equations as:

$$T_{xx} = 2(\vartheta + \vartheta_t)\frac{\partial U}{\partial x} - \frac{2}{3}k$$

$$T_{xy} = (\vartheta + \vartheta_t)\left(\frac{\partial U}{\partial y} + \frac{\partial V}{\partial x}\right) \qquad (5)$$

$$T_{yy} = 2(\vartheta + \vartheta_t)\frac{\partial V}{\partial y} - \frac{2}{3}k$$

Where ϑ is kinematic viscosity of water; ϑ_t is turbulent eddy viscosity; and k is turbulent kinetic energy (Lai, 2008).

A turbulence model is used to compute the turbulent eddy viscosity. Two turbulence models may be used (Rodi, 1993): the depth-averaged parabolic model and the two-equation k-ε model. With the parabolic model, $\upsilon_t = C_t U_* h$ in which U_* is the bed frictional velocity. The model constant C_t ranges from 0.3 to 1.0, and a default value of $C_t = 0.7$ is used by SRH-2D; but its value may be changed by altering the _DIP.dat file within the software folder. Note that terms with k are dropped in equation (5) (Lai, 2008).

If k-ε model is used, turbulent viscosity is calculated with $\vartheta_t = C_\mu k^2 / \varepsilon$. Two additional equations are solved as follows:

$$\frac{\partial hk}{\partial t} + \frac{\partial hUk}{\partial x} + \frac{\partial hVk}{\partial y} = \frac{\partial}{\partial x}\left(\frac{h\vartheta_1 \partial k}{\sigma_k \partial x}\right) + \frac{\partial}{\partial y}\left(\frac{h\vartheta_1 \partial k}{\sigma_k \partial y}\right) + P_h + P_{kb} - h\varepsilon \tag{6}$$

$$\frac{\partial h\varepsilon}{\partial t} + \frac{\partial hU\varepsilon}{\partial x} + \frac{\partial hV\varepsilon}{\partial y} = \frac{\partial}{\partial x}\left(\frac{h\vartheta_1 \partial \varepsilon}{\sigma_\varepsilon \partial x}\right) + \frac{\partial}{\partial y}\left(\frac{h\vartheta_1 \partial \varepsilon}{\sigma_\varepsilon \partial y}\right) + C_{\varepsilon l}\frac{\varepsilon}{k}P_h + P_{\varepsilon b} - C_{\varepsilon 2}h\frac{\varepsilon^2}{k} \tag{7}$$

The following definitions and coefficients are used (Rodi, 1993):

$$P_h = h\vartheta \left[2\left(\frac{\partial U}{\partial x}\right)^2 + 2\left(\frac{\partial V}{\partial y}\right)^2 + \left(\frac{\partial U}{\partial y} + \frac{\partial V}{\partial x}\right)^2 \right] \tag{8}$$

$$P_{kb} = C_f^{-1/2} U^{.3}; \qquad P_{\varepsilon b} = \frac{C_{\varepsilon F} C_{\varepsilon 2} C_\mu^{\frac{1}{2}} C_f^{-\frac{3}{4}} U^{.4}}{h} \tag{9}$$

$$C_\mu = 0.09, \qquad C_{\varepsilon 1} = 1.44, \qquad C_{\varepsilon 2} = 1.92, \qquad \sigma_k = 1, \qquad \sigma_\varepsilon = 1.3, \qquad C_{\varepsilon F} = 1.8{\sim}3.6 \tag{10}$$

The terms P_{kb} and $P_{\varepsilon b}$ and are added in order to account for the generation of turbulent energy and dissipation caused by to bed friction for the case of uniform flows (Lai, 2008).

The dispersion terms result from the process of depth-averaging, and may be important when considering secondary flows (Flokstra, 1976). The effect of the secondary flow may be accounted for indirectly; this is achieved by increasing the coefficient of momentum exchange in the horizontal plane (Mihn Duc et al., 1996). The Manning's roughness coefficient must also be discussed when explaining the model. In SRH-2D, the Manning's coefficient is a local constant that may vary spatially depending on bed type; the value is unaffected by flow (Lai, 2008).

4.4.4 The CASiMiR model

A number of ecological models types exist; the conventional hydraulic habitat model was chosen as the expected change in hydraulics with respect to habitat is the extent of the scope of this investigation and the primary context of the case study site. Of the hydraulic habitat models, PHABSIM is perhaps the most prolific, but has received criticism in recent years (Beecher, 2017) due to limitations such as the model's inability to utilise inputs other than depth, velocity, and substrate, though others have refuted some these criticisms as misunderstandings or limitations inherent to older versions of the software (Reiser and Hilgert, 2018). Regardless, PHABSIM's limited input options do offer a significant drawback in comparison to newer models such as CASiMiR in terms of potential application in more complex systems.

CASiMiR presents a number of features that made it an attractive model to utilise. In contrast to PHABSIM, developed by the US Fish and Wildlife Service in the 1970s, CASiMiR is a relatively recent habitat suitability model that provides a wide array of results outputs, from plan views of the river system to graphical breakdowns of habitat suitability with flow. It has seen application in numerous studies (Conallin et al., 2010, Schneider et al., 2016, Pisaturo et al., 2017), and integrates fuzzy logic and "Fliesswasserstammrisch" (FST), both of which boast advantages over traditional habitat

modelling approaches and are discussed shortly. Although these approaches were eventually discarded in favour of the more traditional method of directly using flow velocity and associated preference curves due to the context of the site and its size making this an equally viable and simpler option, the availability of these approaches within CASiMiR offers flexibility and transferability. Fuzzy logic and the FST metric are effective tools to address increasing complexity should the methods used in this study be applied at a larger scale. Fuzzy logic and FST are further detailed in the CASiMiR Handbook (Schneider et al., 2010). Previous studies have utilised CASiMiR without these advanced features and have provided robust results using standard velocity data and velocity preferences (Pisaturo et al., 2017). In addition to these capabilities, CASiMiR, like PHABSIM, is also capable of integrating additional variables such as substrate type, cover (e.g. branches in the river), and shade (e.g. overhanging trees), in addition to morphodynamic modelling which can play an important role in habitat suitability but is not integrated into older models such as PHABSIM (Almeida and Rodriguez, 2009). Again, although the case study site does not require the application of these features, the use of CASiMiR enables this approach to operate in more complex contexts without disregarding important ecological influences.

Much of the following explanation of the CASiMiR model is adapted from the thesis of Dr.-Ing Ianina Kopecki; "Calculational Approach to FST-Hemispheres for Multiparametrical Benthos Habitat Modelling."(Kopecki, 2008), and other materials obtained through a CASiMiR training course given by Dr.-Ing Matthias Schneider in November 2016 (Schneider et al., 2010, Schneider et al., 2016).

The CASiMiR model is a computational approach to habitat quality prediction. It utilises simulation of a river system's hydraulics and geomorphology, along with ecological preferences of selected indicator species in order to provide the proportion of suitable habitat for investigated species, both in spatial and temporal distributions (Schneider et al., 2010). An accurate hydraulic model is required before CASiMiR itself is utilised. This investigation uses SRH-2D as a hydraulic model; this software is supported and can be directly imported into CASiMiR. Following the development of the hydraulic model, species preference data can be inputted into CASiMiR. Other data such as substratum type can be entered into the model, but this is beyond the scope of this flow-ecological response investigation, and it was recommended by the developers that we focus upon species velocity preferences.

Species preferences can be presented in a few formats. The most conventional form of input is velocity, depth and substratum preference curves. These take the form of habitat suitability ratings from 0 to 1 (0 being completely unsuitable, 1 being most ideal) plotted against a given parameter such as velocity. Other parameters such as temperature may be added if these are found to be relevant to the investigation (Kopecki, 2008). Figure 4.4, below, is an example of flow force preferences for two species of benthos, one adapted to low velocities, the other to high.
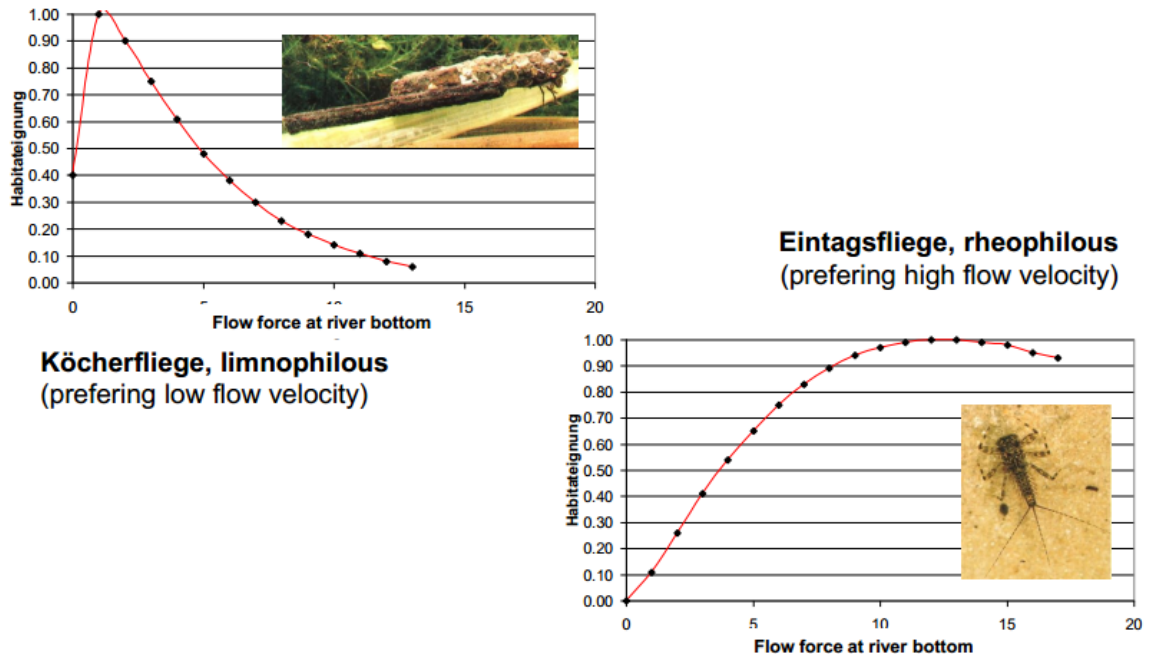
Figure 4.4 Examples of Flow Force Preference Curves, from Schneider et al., (2016). Axes represent flow force (x) vs habitat suitability (y). "Flow force" represents bed shear stress, generally correlating closely with flow velocity.

Once the preferences are inputted, the model will calculate the habitat quality values across the modelled area for the species provided. The simplest and most conservative method for doing this is the product of the habitat parameters being simulated; habitat suitabilities will simply be multiplied together and the final value is the habitat suitability (still from 0 to 1). For example, a point found to be 0.7 suitability for depth and 0.5 suitability for velocity will give be calculated as 0.7 * 0.5 = 0.35 as an overall habitat suitability. The other calculation method is the geometric mean of the parameters. The suitability of the previous example would be $\sqrt{0.7 * 0.5}$ = 0.59. As can be seen, this is a significantly higher value than the product method. However, at the Holden Wood site, a low-magnitude stream, parameters such as depth have a negligible influence on habitat suitability. This is why the developers have recommended utilising the single variable of velocity preference for this investigation, greatly reducing model complexity whilst introducing little additional uncertainty.

Another form of preference curve that can be used, and is primarily used by CASiMiR developers, is preference relating to FST values. FST values act as a surrogate for bed-level flow forces in the river system. Flow forces at bed-level relate well to habitat suitability due to the varying morphological and behavioural adaptations of different species; some macroinvertebrates are capable of avoiding or resisting strong or turbulent bed forces resulting from flows and flourishing in such an environment, whereas species less adapted to such conditions are likely to be dislodged and washed downstream (Blanckaert et al., 2013). Initially proposed by Statzner and Muller (1989) as a method to quickly characterise flows in stream reaches, the value was originally determined through the use of "FST hemispheres" in the field. Each hemisphere is given a number, higher numbers representing higher density. A hemisphere is placed upon a plate on the river bed and increasingly dense hemispheres are placed upon the plate until one the flow cannot move the given hemisphere. This is then the assigned FST value for this area of the river bed. FSTs can replace velocity preferences as they have been found to better represent flow forces at bed-level, and is the standard form of preference curve used by CASiMiR developers, SJE (Statzner and Muller, 1989, Kopecki, 2008). However, this option is limited

by the fact that FST preferences are laborious to obtain and current available preference curves are is limited in terms of species coverage; SJE possesses preferences for species generally found in Germany, and FST preferences are not available for many of the species used in this investigation. Velocity preferences remain a widely used metric within models such as PHABSIM, and as discussed in Section 4.4.1 the depth-averaged flow velocity generally relates closely to near-bed forces in typical river systems, and remains a well-studied metric that has produced good results (Brooks and Haeusler, 2016, Jowett and Duncan, 2012).

The final method used to define habitat preferences involves the use of fuzzy logic. Fuzzy logic allows a flexible and multivariate approach to habitat suitability prediction and is particularly useful in environments where data is sparse (Kopecki, 2008). However, this method is likely more suitable when considering multiple variables, as the main strength of fuzzy logic is its additional nuance due to the fact that fuzzy logic is able to consider the interactions between a number of variables. For instance, a fuzzy rule might state that beyond a certain depth, habitat is always unsuitable for a given species regardless of other variables. Another may state that a particular combination of depth and velocity is highly ideal as they may exclude a species' primary competition. Such rules necessitate significant ecological expertise, however. Whilst such an approach would be highly effective in larger rivers where variables such as depth, vegetation and shade have much more influence, these variables play less of a role at the investigated site and similar rivers of a low flow magnitude (average discharge of >1m^3/sec), and thus this research will focus on the univariate velocity preference curve approach, with justification for this provided in the following section.

4.4.5 Chosen model inputs

In this investigation, river geometry and discharge will be inputted into the model along with the ecological input of velocity preference curves, outlined in further detail in the next section. Other potentially ecologically influential variables that could have been included include depth, sediment type, and vegetation. It was decided that such variables would not be of significant benefit to the accuracy of model predictions due to the relatively small size and simplicity of the Holden Wood site. Depth across the study site was not found to exceed 80cm, and rarely exceeds 20cm under general flow conditions. This ranges of depth observed in most river systems is rarely relevant to benthic macroinvertebrates, with depth not even being considered in the STAR project trait analysis (Bis and Usseglio-Polatera, 2004). The impact of depth for macroinvertebrates in rivers is rarely considered in literature, with flow and thermal regime typically cited as key drivers (Poff et al., 1997, Poff and Zimmerman, 2010, White et al., 2017), though depth is considered in lakes where the range is much greater, with observed differences at depths above 2m (Rieradevall et al., 1999). Fish are influenced by depth in rivers much more significantly, due to the fact that they move within the body of water. Shallow waters may inhibit mobility, or even isolate larger fish from the main river system, whilst deeper waters may affect visibility and are more likely to be used by predators.

Sediment, though more significant to macroinvertebrate habitat, did not vary sufficiently at the site to be considered; the bed is rocky across the entire modelled reach, and whilst various sizes of rock are mixed, size distributions are approximately uniform across the bed. This is expected due to the small scale of the site, the relatively large sediment size, and the lack of flow features that might encourage significant micro-scale sediment disturbance or deposition. Due to the fact that sediment does not change across the site, and the fact that this investigation is principally concerned with flow

regime influence on river hydraulics, it was decided that the model would most benefit from only considering the one variable, flow velocity, rather than implementing secondary variables that provide little towards model outputs or the goals of this investigation.

In terms of vegetation, again the relative simplicity of the Holden Wood site is favourable. There is no vegetation present on the banks or within the water of the river. Whilst depth and sediment may be discounted at many sites for similar investigations of this scale, it is expected that vegetation plays a key role on the ecosystem at any site where it is present. Vegetation acts as a nutrient source for other organisms, provides refugia, and also has a significant influence upon channel hydraulics and sediment processes (Curran and Hession, 2013). Because of this, we would recommend including vegetation as an ecological variable at any modelled site in which significant bodies of vegetation occur.

The scale and simplicity of Holden Wood serves to allow us to isolate the variable of flow velocity, for reasons described. This provides an opportunity to directly assess the influences of flow regime change upon benthic macroinvertebrate habitat, without further complications brought about by variables external to the scope of the investigation.

4.4.6 Model inputs and outputs in concept

The concept behind this modelling approach is to begin with flow conditions at the Holden Wood case study site, that is, volume of flow passing through a section of the channel, along with channel geometry, and ending with a metric of ecological condition, that being habitat quality for selected indicator species.

In order to achieve this, channel geometry is taken along with initial flow conditions as flow inputs and outputs (i.e. flow "in", water level "out"). SRH-2D predicts how the inflows interact with river geometry, generating spatial distributions of flow from which flow velocities may be taken across the entire modelled reach. This distribution of velocities can be translated into habitat quality by using macroinvertebrate flow velocity preference curves, derived from an ecological database as described in Section 4.5.2, in the ecological model. Finally, these distributions of habitat quality can be computed into a single-value metric of ecological condition for a given species at a given flow magnitude input. Thus, we start with a single value of flow (such as $0.1m^3/sec$), this is translated into velocity distributions, which are then translated into spatial habitat quality distributions using the flow preference curve inputs. These spatial preferences can then be translated into a single value of habitat suitability for the entire reach; this stage of output it performed in Chapter 5 and is described in Section 5.2.3.

4.5 Methods

An ecological model was constructed using the CASiMiR model (Schneider et al., 2010) to develop an understanding of the macroinvertebrate response (habitat quality) to flow at the site. This required the development of a hydraulic model of the site in order to understand the velocity regime. River geometry, velocity and ecological data was gathered for model development, calibration and testing. Once calibrated, habitat predictions were utilised and supplemented by an integrated consideration of taxon requirements (habitat quality metrics and anticipated responses to temporal flow characteristics) in order to design potential environmental flows for the Holden Wood site. In this Chapter, the design, calibration, and application stages of model development are described and discussed, with model outcomes described in Chapter 5.

4.5.1 Model development: SRH-2D hydraulic model

The SRH-2D (Sedimentation and River Hydraulics) modelling package was used to develop an understanding of the hydraulic complexity of the study reach. SRH-2D is based on the numerical solution of the two dimensional depth averaged St. Venant equations, providing calculations of depth and velocity at each computational cell based on model boundary conditions, reach topography and bed roughness (Lai, 2008). SRH-2D has recently seen widespread use in the field of river restoration and eco-hydraulics (Erwin et al., 2017, Stone et al., 2017, Lane et al., 2018).

Bed elevations at the study site were obtained using a Total Station surveyor (Leica Geosystems, 2009). Bed elevations were taken using a scatter-based method, taking elevation readings that adapted in resolution according to bed complexity. A total of 2069 geometry data points were collected over the reach. Bed elevations were uploaded into the SRH-2D model using the SMS interface (Aquaveo LLC, 2013) and a fine mesh was generated with cell sizes approximately 30x30cm. In particularly complex rivers, meshes as fine as 10x10cm have been utilised (Lange et al., 2015). Most ecological studies using SRH-2D have used 30x30cm mesh sizes for detailed sections, with typical mesh sizes of around 250x250cm or higher in larger rivers (Bandrowski et al., 2014, Stone et al., 2017, Lane et al., 2018).

Model calibration was performed using direct velocity measurements, utilising a Nortek Vectrino Acoustic Doppler Velocimeter (ADV), which is typically expected to provide velocity values accurate to within 5% in field conditions (Dombroski and Crimaldi, 2007). The ADV probe was secured to an adjustable surveying tripod, allowing for stable positioning at any point of measurement. The probe was capable of taking simultaneous measurements of three orthogonal velocity components at a frequency of 20 Hz, providing temporally averaged velocity data as well as standard turbulent statistics. An initial convergence test was conducted, ensuring that representative, reliable data was possible at each point. A sampling period of 60s was used, due to little hydraulic complexity at the site and due to readings typically stabilising within 30s of deployment. For each measurement, the probe was orientated as such to obtain primary (x) velocity in the main channel direction (with the y dimension normal to the river bank). Raw ADV data were processed in WinADV 32 (Wahl, 2000), and the phase space threshold de-spiking filter was applied prior to data analysis (Goring and Nikora, 2002).

Eight cross-sections were measured along the reach, with flows being taken at 3 to 5 points along each cross-section depending on channel width. Measurements were taken at 0.6 of the depth to obtain a

depth-averaged reading (Hewlett, 1982). A total of 31 readings obtained in total allowed for moderate coverage along the entire reach at a high resolution relative to many studies; SRH-2D has been successfully calibrated in larger rivers with significantly fewer observation points (Deslauriers and Mahdi, 2018). At the time of measurement, flow into the river was measured as $0.024 m^3$/sec, based on impoundment outflow data provided by the site operator. This discharge is generally consistent throughout the autumn season, unless the impoundment is close to capacity, at which point flow releases are elevated and spill events are possible.

Upstream and downstream boundary conditions were established based upon straight, stable areas of flow within the study reach. The upstream boundary condition was set as the inflow ($0.024 m^3$/sec), and the velocity was defined using SRH-2D's Conveyancing approach in which flow direction is assumed to be normal to the inlet boundary (Lai, 2018), and the velocity is uniformly distributed. The downstream boundary condition was set as the measured water level (185.02m above sea level), again assuming flow normal to the boundary. Manning's roughness values were initially assigned with appropriate ranges based upon literature values (Chow, 1959) based on substrate type at the site.

River geometry and flow at the time of measurements were loaded into SRH-2D and the predicted steady velocities were compared with the observed temporally-averaged velocities. Manning's roughness values for the river channel were calibrated based on established best practices (Van Waveren et al., 1999) initially testing homogeneous roughness across the entire reach, and later adjusting small areas where observed changes in substrate led to discrepancies in velocity. Final calibration saw the majority of the river assigned a Manning's value of 0.05, whilst patches of the riverbed had roughnesses ranging from 0.04 to 0.07. These values are appropriate for streams with generally little vegetation, steep banks, trees and scrub at the banks, and cobbles and large stones within the channel (Chow 1959).

4.5.2 Model development: CASiMiR

The CASiMiR model framework is modular and integrates hydraulic and structural parameters from a hydraulic model for the calculation of habitat suitability for indicator organisms. Aquatic habitat suitability in this study is derived by the use of univariate flow velocity preference curves, and this is later calibrated through species population distributions observed in the field (Schneider et al., 2016). Preference curves were based on flow velocity affinities found in the STAR Project, a large-scale investigation supported by the European Commission in order to resolve challenges posed by the Water Framework Directive, using the study "Deliverable N2" (Bis and Usseglio-Polatera, 2004).. This study involved the aggregation of macroinvertebrate traits into one of the largest species trait databases available (Bis and Usseglio-Polatera, 2004). In the STAR project, velocity preferences are described in the range of Null (0cm/s); Low (>0-25cm/s); Medium (>25-50cm/s) and High (>50cm/s) based upon flow affinity, i.e. how well a species is adapted to particular flow conditions. Affinities range from 0 (lowest) to 3 (highest). These affinities were interpreted into Habitat Suitability Index (HSI) values ranging from 0.00 (lowest possible affinity) to 1.00 (highest possible affinity). In this study, flow velocity was selected as the sole parameter for driving ecological response. Depth and substratum are also used as key parameters in larger river systems, but at the scale investigated at this study site substratum can be assumed to be homogeneous, and changes in depth are not significant in terms of macroinvertebrate sensitivity.

CASiMiR can be calibrated through small adjustments to preference curve inputs (Schneider et al., 2010), due to possible variations in biological behaviour from site to site caused by external drivers. This was not necessary for this study due to species behaving in accordance to established preference values; as demonstrated in Section 4.6, and discussed in Section 4.7, predicted species behaviour fell within reasonable expectations. The model was tested by comparing observed species sample populations, taken using the standard 3-minute kick sample method (Murray-Bligh, 1999) in November 2017 at a flow rate of 0.024m^3/sec. 15 measurements were taken using single-point kick sampling from a range of microhabitats distributed across the reach, as displayed in Figure 4.5.

Total length: 39.83 m
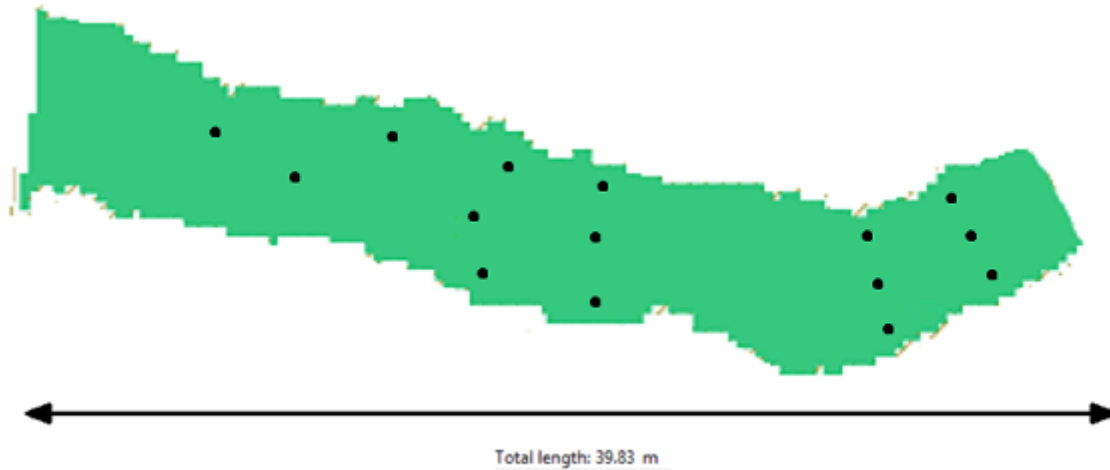
Figure 4.5: 15 Kick sample locations across the study reach, Ogden Brook, represented by points on the reach map.

Habitat predictions based upon the same flow rate. Such testing under a single flow condition was deemed reasonably justified due to the minimal variation of flow at the site, and the fact that samples demonstrated similar species composition proportions to those observed in 2016 sampling data provided by the consultants (described in section 3.2). Three species, *Gammarus pulex, Polycentropus flavomaculatus, and Hydropsyche siltalai,* were chosen for model testing based upon their occurrence at most sample sites, and their range of flow preferences. Agreement between model predictions in the form of HSI, and observations in terms of species sample populations at the same point, was assessed.

4.6 Results

4.6.1 Hydraulic model calibration

Below in Figure 4.6 model predictions are shown in terms of primary velocity vectors.
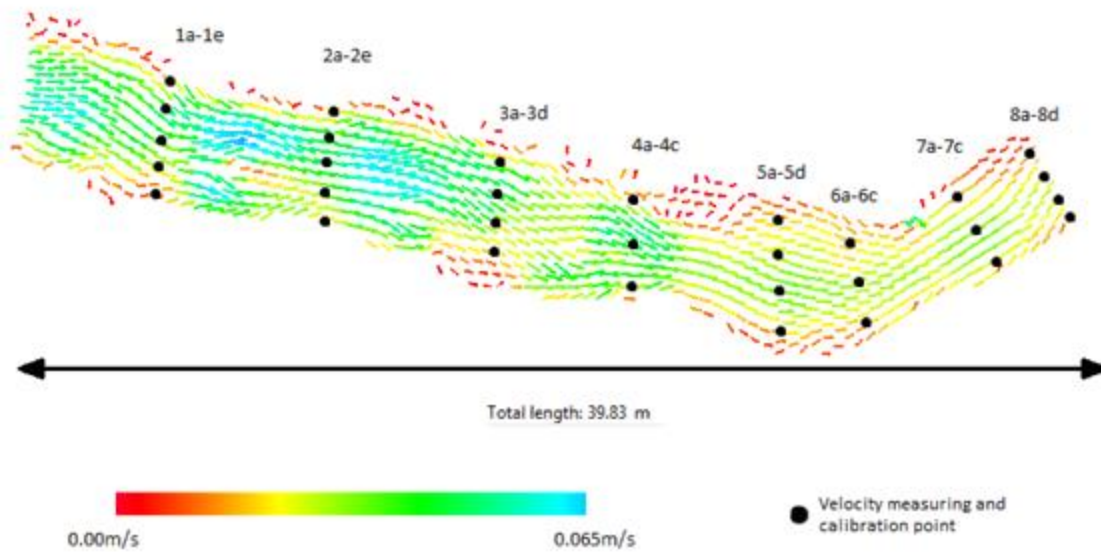
Figure 4.6: SRH-2D hydraulic predictions, post-calibration, showing primary velocity in m/sec for an inflow of 0.024m³/sec

Model predictions of steady primary depth-averaged velocity were tested by comparison with field point-observations as shown in Figure 4.7. It can be seen that there is broadly good agreement between predictions and observed values. Anomalous readings tend to be at the highest ranges of velocity. Given that these results are not clustered around a particular area of the river, these high-velocity anomalies may be caused by sudden, localised changes in bed geometry, either not accounted for at the mesh scale used, or not detected during bed geometry measurements, such areas of faster flow (>5cm/s) may be highly localised and difficult to account for; for instance above a large rock causing a small shallow area of increased velocity, or a cleft between stones through which flow is funnelled. The most erroneous point, 3c, had been noted during field velocity measurement to be an area of particularly fast and complex local flow due to the presence of nearby rocks. There is also a possibility that errors arise from measurement inaccuracies inherent to characterisation of the depth-averaged velocity at a single depth.
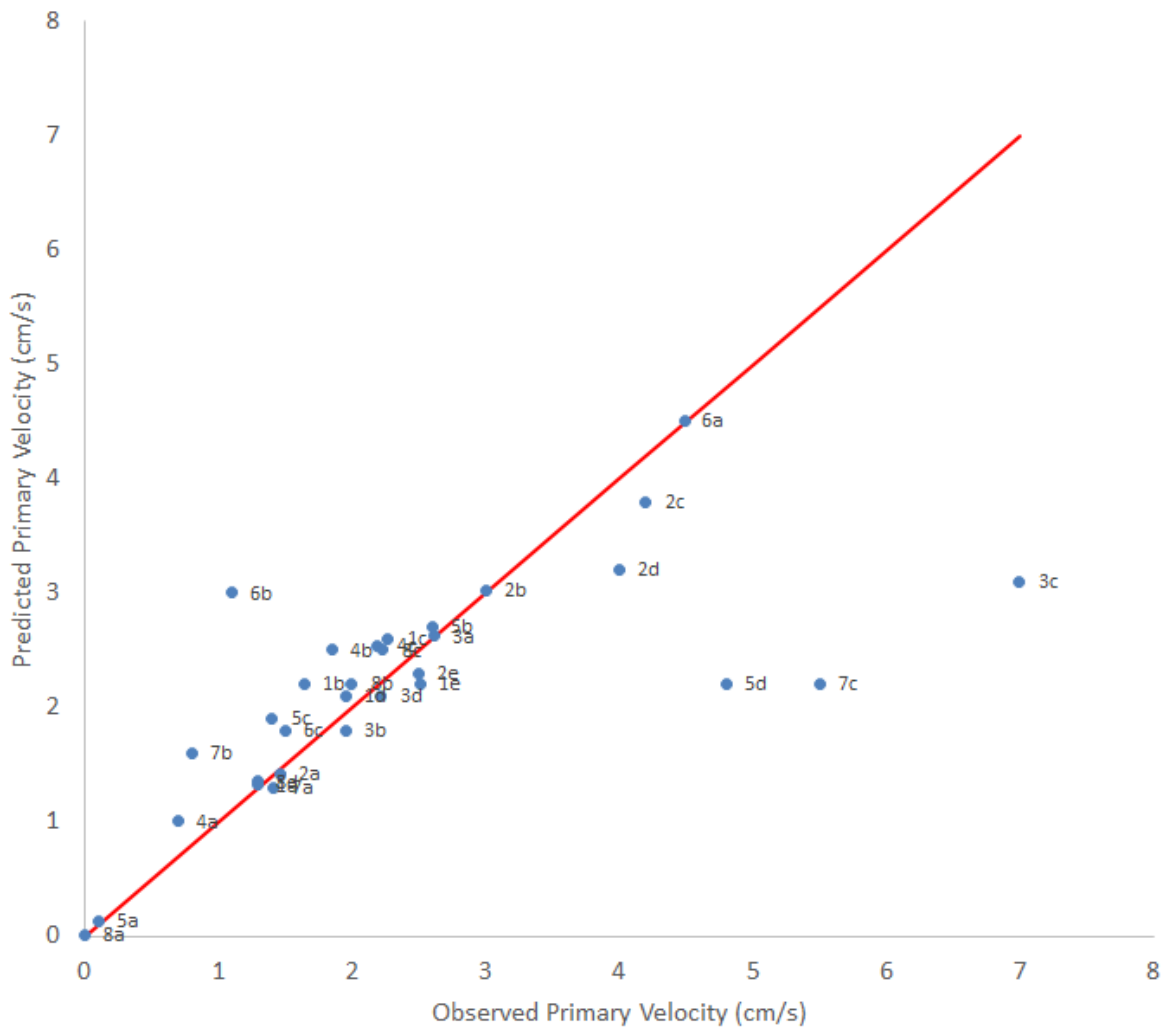
Figure 4.7: Post-calibration SRH-2D model predictions vs observed field primary depth-averaged velocities

4.6.2 CASiMiR ecological model verification

Table 4.1 and Figure 4.8 below shows observed populations of three indicator species, compared with CASiMiR outputs of predicted HSI. Figure 6 demonstrates the features that were expected of the HSI predictions; particularly poor areas never accommodate significant numbers of a particular species, whilst population numbers between moderate and good are often somewhat stochastic. Peak populations appear to be associated with a HSI of at least moderate quality, and also of significant area. Small patches of good HSI, whilst having significant populations, do not tend to host peak populations. This may be due to a lack of habitat permanence or simply the somewhat stochastic nature of ecological distributions. Reasons for this manner of distribution are discussed in detail in the Discussion section.

Table 4.1: Observed macroinvertebrate populations compared with predicted habitat quality.

Sample	Gammarus pulex		Hydropsyche siltalai		Polycentropus flavomaculatus	
	Observed	HSI	Observed	HSI	Observed	HSI
1	9	0.22	3	0.13	10	0.53
2	16	0.33	11	0.34	22	0.64
3	7	0.25	2	0.23	2	0.59
4	2	0.22	0	0.10	2	0.50
5	29	0.34	8	0.28	9	0.62
6	27	0.40	12	0.44	17	0.70
7	67	0.37	31	0.45	25	0.65
8	10	0.33	13	0.35	5	0.62
9	11	0.28	0	0.12	10	0.63
10	28	0.37	7	0.21	7	0.60
11	0	0.35	136	0.45	5	0.64
12	18	0.35	7	0.19	15	0.61
13	18	0.32	4	0.16	8	0.57
14	57	0.41	57	0.50	6	0.63
15	22	0.40	14	0.47	9	0.67
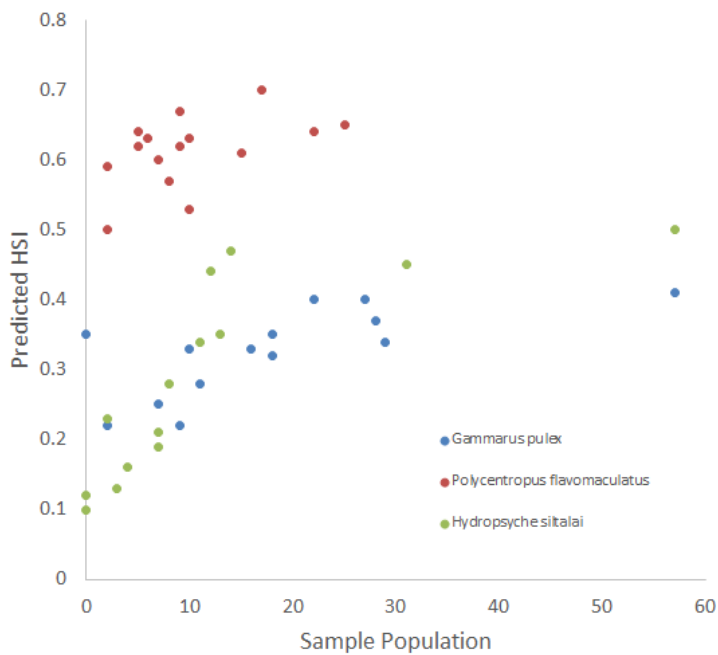
Figure 4.8: CASiMiR verification, comparing the HSI and observed sample populations of 3 macroinvertebrate species at the Holden Wood site

It can be seen that predicted HSI and observed species populations are in good general agreement. Correlation coefficients for the relationships in Figure 4.8 are 0.623, 0.577, and 0.488 for *Gammarus pulex, Hydropsyche siltalai, and Polycentropus flavomaculatus* respectively. This is good evidence for the utility of the model predictions. The clear correlation is strong evidence for the reliability of model predictions, particularly due to the often stochastic nature of macroinvertebrate distributions due to external drivers and species tendency to cluster together in certain areas whilst sparsely populating others. Overall, predicted HSI and observed species populations are in good general agreement. The clear correlation is strong evidence for the robustness of model predictions, particularly due to strong relationships being observed in spite of potential confounding factors that have been mentioned, and are detailed in the Discussion.

4.7 Discussion

4.7.1 Hydraulic Model

The hydraulic model demonstrates the flow patterns present at Holden Wood. Figure 4.9 below demonstrates that flow is relatively simple at the site, with primary velocity vectors generally going downstream, with some exceptions at the river banks. This shows that there are few, if any, complex flow features such as vortices present in the modelled section of river. This gives justification to the assumption that depth-averaged velocities at the site can be treated as surrogates to near-bed flow due to an anticipated lack of vertical flow complexity, and affirms the use of a 2D model. These flows were anticipated, due to the shallow gradient and relative flatness of much of the river bed.

0.00m/sec   0.065m/sec
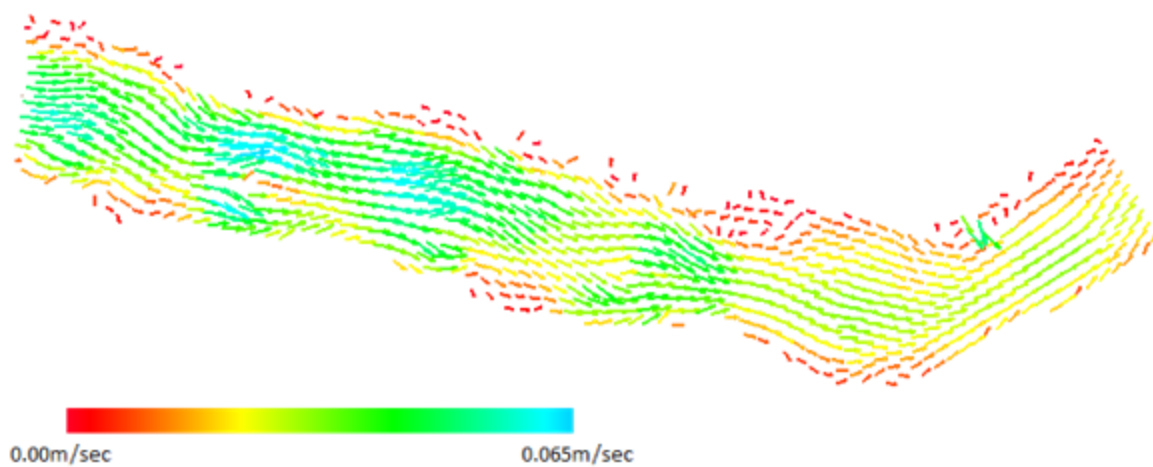
Figure 4.9: Primary velocity vector predictions in SRH-2D at the Holden Wood site, at a flow of 0.024m³/sec (typical site conditions)

As demonstrated in Figure 4.10, there is little change in the pattern of flow within the general ranges expected at Holden Wood (0.02 − 0.1 m³/s), other than significantly higher flow velocities and changes to behaviour at the river banks.
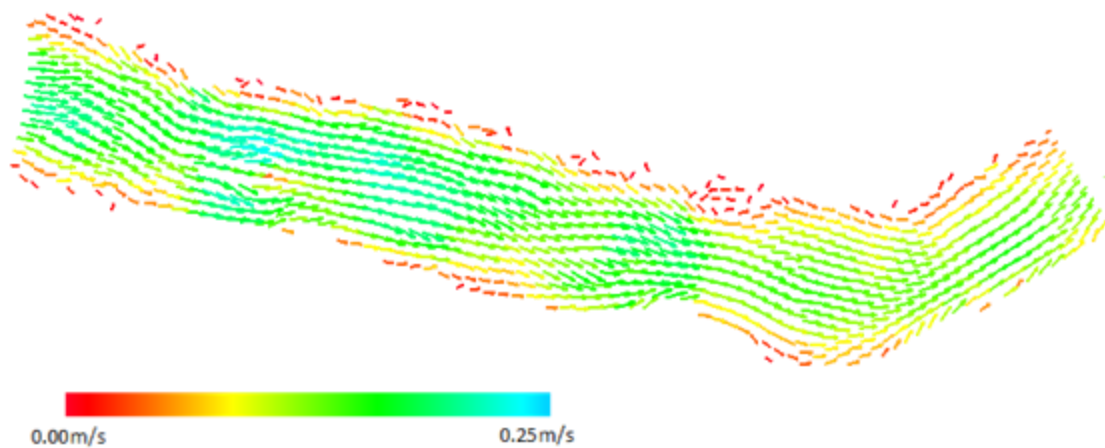
Figure 4.10: Primary velocity vector predictions in SRH-2D at the Holden Wood site, at a flow of 0.1m3/sec (significantly above typical site conditions)

Results from the hydraulic model predictions have been shown in relation to field observations. In the %-based reading, there are a significant number of seemingly deviant predictions. However, most of these high % errors are found at points of very low flow, thus even small discrepancies in velocity lead to high % error. Most of the severe discrepancies are resolved when looking at model performance through the perspective of deviation based upon cm/s. 71% of predictions fall within a 1cm/s margin of error. Some discrepancies still exist, and this is likely due to the point-based nature of measurements that were taken, and their limited number. Due to the significant time required to set up the ADV probe at each point, only 31 points of measurement were taken, in comparison to the 2069 points for geometry (which can be taken almost instantly with the Total Station). It is likely that points where significant error is present fell within areas that differed significantly from the surrounding conditions; this could be due to small-scale geometry fluctuations caused by the river bed, such as submerged rocks causing the probe to be at a different elevation in relation to the surrounding area. Such error could have been somewhat mitigated through a higher resolution of flow readings, and this study would suggest quicker (albeit less precise) methods of flow velocity measurements in future studies, to allow for a much higher number of point flow readings. Higher resolution geometry data would also mitigate this issue, though acquiring such data would be both labour- and resource-intensive, and would require a higher resolution model mesh which would in turn increase computational demand. This said, uncertainty is always a significant issue in flow measurement, as has been noted in the literature, especially in areas of particularly high or low flow (Acreman et al., 2014), and thus some degree of uncertainty is expected.

The predicted data, when compared with observed data, follows velocities within the channel closely for a majority of readings. However, the model has difficulty in modelling localised high flows and very low flows. This may be due to localised complexities at the river bed level, such as water being forced through a narrow point between two elevated surfaces which were not detected when assessing river bed geometry. Despite using a Total Station surveyor, the resolution of the points (around 10cm) is still insufficient to detect significant changes in elevation at a fine scale, e.g. a 5cm "gorge" between two higher points. The resolution provided by the Total Station was limited by time and the battery

power available for the equipment, and as mentioned mesh resolution is also limited by computational demand. This source of uncertainty could be mitigated through a greater commitment of time and resources; using a surveyor system with longer battery life and spending more days in the field could allow for very fine bed elevation data to be generated, though more time would also need to be given to the computation of a finer scale model.

Additionally, a greater number of flow velocity data points would have resulted in a more robust verification of the Holden Wood model. Data point collection was limited by the time-consuming setup and reading time of the ADV probe at each point. Future studies may wish to consider flow reading methods that generate a greater quantity of data points, for example through the use of a propeller-type current meter. Whilst this may sacrifice some accuracy for the individual data points, a significant increase in overall data would almost certainly be a worthwhile payoff due to the fact that a primary concern for a hydraulic model utilised for this purpose is avoiding serious miscalculations which would generate misleading data. Thus, thorough coverage of flow velocities across the modelled reach should be a priority for calibration purposes in modelling of this nature so that anomalous readings (e.g. due to fine scale changes in bed geometry) do not obscure overall model reliability.

Overall, the hydraulic model performed well with a few exceptions. Adjustments to the methodology in potential future work would be expected to generate an even more accurate model. With a calibrated hydraulic model, it was then possible to begin work on the ecological model, CASiMiR.

4.7.2 CASiMiR

As demonstrated in the Results section, quantified macroinvertebrate populations from sampling data were compared to HSI predictions made through CASiMiR. There are likely to be a number of sources of error in the relationship between these. Due to the nature of HSI predictions (HSI expresses probability of species being present, it does not predict abundance directly. We must assume that areas of low habitat suitability for a given species would correspond to there being zero, or few individuals of that species present, whilst the species would be expected to be most abundant in areas of high habitat suitability. In addition, given that CASiMiR only models the influence of flow, other drivers such as food availability, ecological interactions and temperature may also alter the distribution of species (Ferreiro et al., 2011, Alba-Tercedor et al., 2017). For instance, while habitat suitability may be ideal for a species at a given point, the species may be pushed out by a more competitive species or a predator that also finds the habitat suitable. This leads to a somewhat stochastic population distribution that cannot be accounted for within the scope of this investigation.

Gammarus pulex displayed the strongest correlation between population and HSI, though this trend plateaus at the upper limit of population number, possibly due to the stochastic nature of population distributions and external drivers such as biological interactions. *Polycentropus flavomaculatus* had the weakest trend between population number and HSI, but retains a clear correlation. The weakness in the trend is likely due to a lack of low HSI values for this species, which are prohibitive to populations, as well as the limited number of the specie present. Both of these factors pronounce the stochastic nature of ecological distribution, whereas the presence of lower HSI's would be expected to lead to low populations in such areas, and the presence of higher populations might lead to colonies that can only be sustained within areas of higher habitat quality.

Some level of stochastic distribution of species was expected, both due to the probabilistic nature of HSI predictions, and due to external complexities not accounted for within the CASiMiR model. It is thus especially noteworthy that CASiMiR HSI predictions generally correspond to the trends in species distribution; poor HSI always corresponds to low species population, whilst moderate to good HSI typically hosts significant species populations, with some degree of stochastic distribution due to complexities introduced by external variables, for example biological interaction, which may restrict access of species to certain areas, or otherwise influence population distributions. It is expected that poor HSI should be restrictive to species populations due to it being an inhospitable environment resulting in high mortality, thus low HSI should invariably be associated with low population numbers. As HSI increases and the environment becomes more tolerable, species distribution becomes less predictable due to species being capable of surviving, whilst being influenced by complex, interacting variables.

When discussing HSI predictions, the assumptions and limitations inherent to this metric must be considered. The use of HSI to establish the response of biota to flow change assumes that changes in the habitat suitability metric are proportional to changes in the abundance of the target biota. Results in Section 4.6.2 give support to this assumption, though it is not at a 1:1 ratio, and nor does it remain consistent (potentially for reasons described above). Other studies such as Kelly et al., 2014 found that the weighted usable area (WUA) metric (described in Section 5.2.3) which is derived from HSI values, followed this trend, but also not in a clear 1:1 ratio.

There is also the assumption that the general preference curves used as inputs for the HSI calculations (as opposed to site-specific preference curves) are transferable and valid within the system in which they are being utilised. In many cases, general preference curves have proved to be successful (Tomsic et al., 2007, Pisaturo et al., 2017), though in some cases such as Kelly et al. (2014), such general inputs were found to lead to insensitive ecological metrics, and site-specific inputs were necessary. The relationship observed between HSI and species abundances in Section 4.6.2 is some evidence that the river under investigation, Holden Wood, is suitable for generalised habitat preference inputs. However, given that ecological sampling could not take place across a range of flow conditions, it must be assumed that this would be the case across typical river flow conditions. Should time and resources be sufficient, ideally site-specific habitat suitability inputs would be preferable, but generalised inputs are found to provide valid results.

In terms of limitations, inputs such as preference curves are utilised for the calculation of HSI, as discussed in Section 4.5.2. Due to this, HSI in a given investigation is limited by whatever inputs are provided for the metric's calculation; this thesis utilised flow velocity alone (as discussed in Section 4.4.5), while larger or more complex systems may also utilise depth, substratum type, and perhaps thermal preferences for indicator species. As mentioned above, if non-site-specific inputs are utilised, some degree of uncertainty may be introduced (Kelly et al., 2014); it would then follow that the more variables are utilised to calculate HSI, the more uncertainty there may be. In such cases, a thorough validation of HSI prior to its use in predictive modelling would be essential. Obtaining a number of preference variables, and subsequently validating them, however, may prove to be laborious. Conversely, should HSI be calculated through a small number of inputs (as in this thesis), the habitat model is effectively "blind" to ecological influences that have not been included. Again, it is therefore important to have good knowledge of the site in question and whether assumptions about ecological

influences present at the site are valid, and a validation of calculated HSIs and their relationship to target biota abundances would help to provide confidence in the metric.

Macroinvertebrate sampling took place during the autumn season; practical constraints restricted invertebrate sampling to a single season (autumn), but the use of autumn samples is given some validity by Cascade consultancy data, and it can be stated with confidence that macroinvertebrate samples taken at this time are representative of the reach as a whole. Due to the lack of disturbance events at the site, the highly regulated nature of flow, and the associated thermal regime being relatively consistent due to the dominance of the reservoir releases, it is believed that macroinvertebrate distributions are unlikely to be significantly altered seasonally, except with regard to life cycle behaviour which can be accounted for using Cascade's seasonal data, alongside further information from the literature. Spring sampling data provided by Cascade provides insight into species numbers present during this period. Most indicator species utilised in this investigation are present in both seasons. From this, it is believed that the assumption that the samples used to test the CASiMiR model are representative of the studied area is a valid one.

4.7.3 Model outputs

We have developed a framework by which a small-scale impounded river system may be investigated, and the influence of reservoir releases assessed in terms of predicted ecological response based upon changes to habitat quality across the modelled area, whilst also being mindful of the practical constraints of the reservoir. Results have given support to the assumption that velocity-based habitat quality is a significant driver of species population distributions, and thus our manipulations of HSI with flow demonstrates the likely response of selected taxa. The use of the HSI metric has made it possible to predict the likely outcome of flow modification for the habitat quality of selected indicator species. Figure 4.11 demonstrates spatial HSI predictions for *Gammarus pulex* at two flow magnitudes, $0.024m^3$/sec and $0.10m^3$/sec, as an example of how CASiMiR outputs might be utilised. As described by the results in Section 4.6.2, the species *Gammarus pulex* is sensitive to low flows, with low HSI values across most of the reach during typical site flow conditions at $0.024m^3$/sec. It can be seen that the species has a significant response in terms of HSI with significantly elevated flows at $0.10m^3$/sec, due to the species' affinity for higher flows. Such outputs allow this investigation to work towards optimising flows for an environmental flow regime. Chapter 5 deals with the question as to how the various outputs, and the requirements of different indicator species, come together in order to achieve ecological benefit whilst addressing issues of water security and societal needs.

Habitat Suitability Index for *Gammarus pulex*, at 0.024m³/sec

Habitat Suitability Index for *Gammarus pulex*, at 0.10m³/sec

Legend:
- 0.00 <= SI < 0.10
- 0.10 <= SI < 0.20
- 0.20 <= SI < 0.30
- 0.30 <= SI < 0.40
- 0.40 <= SI < 0.50
- 0.50 <= SI < 0.60
- 0.60 <= SI < 0.70
- 0.70 <= SI < 0.80
- 0.80 <= SI < 0.90
- 0.90 and better
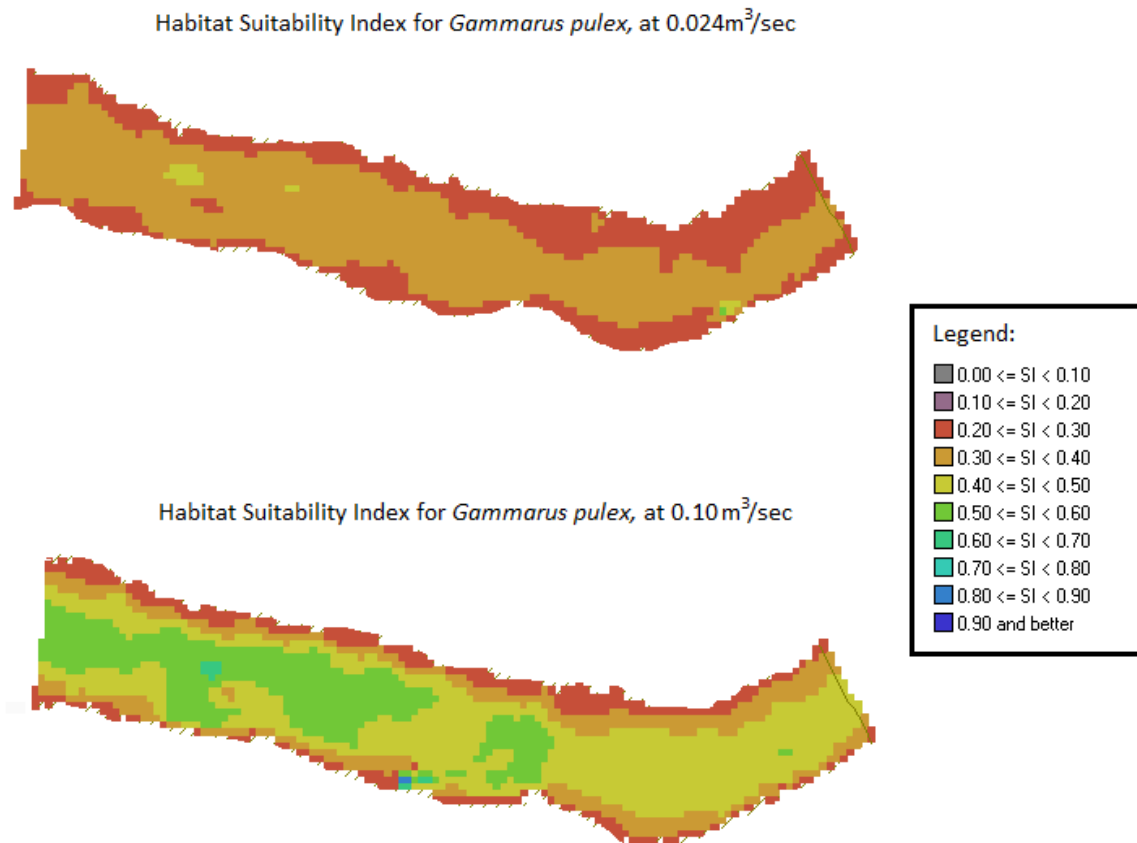
Figure 4.11: Predicted Habitat Suitability Index for Gammarus pulex across the study site in two flow conditions, generated by CASiMiR

These initial results from model calibration and its associated implications lead into the next stage of this investigation, detailed in Chapter 5, which is the development of efficient environmental flows designed to optimise HSI both in terms of flow volume required and in terms of flow timings (time of year, frequency, and duration). CASiMiR is limited in its capacity to model temporal dynamics, and thus other approaches must be used to implement these factors into flow regime design.

4.8 Conclusions

This stage of the investigation has been concerned with developing and calibrating both a hydraulic and an ecological model of a case study site, for the purpose of predictively assessing the impacts of flow regime upon the native macroinvertebrate ecosystem, with the purpose of optimising the ratio of water spent to ecological provision provided, balancing the conflicting interests of environmental needs and societal use. It has been demonstrated that the hydraulic model developed is capable of predicting flow velocities across the river reach to an acceptable accuracy, though some discrepancies exist at more extreme high flows, with possible reasons discussed in Section 4.7. The ecological model outputs were found to be in generally good agreement with field observations, giving confidence in the capabilities of the model. However, this study has been met with a number of limitations, and model accuracy and robustness could certainly be improved upon in future work, allocating more time and resources to take more measurements and create a finer-scale model, as has been discussed in Section 4.7.

Model testing could have been even more robust through the processing of more ecological sampling data; though it must be considered that ecological sampling and sorting is a significant time and resource investment and this may limit the number of samples that can be taken. As stated in Section 4.7, the calibrated 2D model provides predictions of habitat suitability, not of defined species abundance. CASiMiR model outputs were evaluated and acted as the primary source of information for individual species provision (based upon sensitivity thresholds) and spatial habitat response to flow (habitat diversity, connectivity and persistence) from which principles for environmental flows at Holden Wood were developed. This represents a significant step forwards in the task of bringing together knowledge of ecology-flow response relationships in order to optimise ecological provision in terms of timings and appropriate magnitudes. Other temporal drivers such as flow event frequency and duration are not considered by CASiMiR and are incorporated through insights gained from statistical analysis covered in Chapter 3, and additional knowledge from literature discussed in Chapter 2.

As we have discussed, the outputs of the CASiMiR model have provided evidence that the assumption that depth-averaged flow velocity is an acceptable surrogate for near-bed forces, and relates well to habitat quality experienced by benthic biota, is a valid one in this context. HSI predictions have been shown to correspond well with field measurements of species population distributions, and because of this we can have confidence in the assumption that changes to predicted HSI are likely to denote significant changes in macroinvertebrate populations. Therefore, one of the goals of subsequent flow design – to optimise HSI benefit from reservoir releases – is a well-justified one. We are now able to move into the next stage of this investigation, flow regime design, with good foundations and confidence that the outputs of our developed model are sensible, and changes to these outputs should be expected to elicit ecological change accordingly.

5. ADDRESSING IMPOUNDMENT RELATED FLOW MODIFICATION AT A CASE STUDY SITE BASED ON HYDRO-ECOLOGICAL MODEL OUTPUTS AND ECOLOGICAL PRINCIPLES[*]

[*]Flow designation through the approach described throughout Chapters 4 and 5, and the results generated (particularly in Section 5.3), are of significant worth and represent a promising area for further research; a paper comprising a condensed version of Chapters 4 and 5 has been accepted and published by the journal Ecological Indicators, see Appendix 9.1.

5.1 Introduction

The aims of the 2D model developed and calibrated in Chapter 4 were to aid in better understanding how benthic macroinvertebrates within the system respond to flow alteration and to develop recommendations for more efficient flow regimes that make provision for the environment, whilst retaining as much water as is possible. Moving from established science to implementation has been identified as a particular challenge in the field of environmental flows (Overton et al., 2014). This investigation proposes a novel methodology by which findings may be implemented. Conflicting stakeholder interests present in most riverine systems are a particular challenge to environmental flow implementation. The provision of societal services is a significant priority that cannot be neglected; therefore water managers must identify ways in which to maximise environmental benefit relative to volume of water released as impoundment outflows (Konrad et al., 2011, Summers et al., 2015). In the previous Chapter river geometry, velocity and ecological data were gathered for model development and calibration. In this Chapter, habitat predictions are utilised and supplemented by a further consideration of ecosystem requirements, based on findings in Chapter 3, in order to design potential environmental flows for the Holden Wood site. These designer flows were compared with past and current impoundment outflows to demonstrate their advantages.

5.2 Methods

This section outlines the process by which flow regimes were designated, including an evaluation of the impacts of the Holden Wood impoundment, an analysis of indicator species requirements both in terms of magnitude and timing, and a more general analysis and application of ecological principles such as the need for flow variation, habitat heterogeneity, and a consideration for the physicochemical regulation and ecological stability provided by more natural and varied flow regimes. As the Results section deals specifically with the resulting observations post-flow regime design, the insights provided by the above analyses are given here and their integration into flow regime design is detailed. This section ends with the proposal of designer flow regimes and their rationale.

5.2.1 Preliminary analysis: natural vs modified flow

Prior to designing new flow regimes, the 2014 flow regime at Holden Wood reservoir was compared to 2014 inflows into the reservoir using mean daily flow data obtained from United Utilities, the outflows being measured by a cumulative flow gauge at the site that is read daily by a site operator, who calculates the flow for the last 24 hours and converts this into flow rate per second, and the inflows being calculated by a gauging station placed before Holden Wood reservoir. Reservoir inflows

can be approximated to natural flow into Ogden Brook, were no impoundment were in place. 2014 data was chosen due to these being the latest synchronised datasets made available. A semi-quantitative analysis was performed between the two datasets, focusing upon the primary alterations typically caused by flow modification; changes to overall flow magnitude, changes to inter-annual seasonality and flow variability, and changes to the frequency and duration of individual high flow and low flow events (typically defined as the upper and lower 25th percentiles of flow respectively) (Poff et al., 1997, Richter et al., 1996). This analysis provided insights into the extent of flow regime modification, and into what characteristics within the regime are most altered. There are significant differences between the two time series, as demonstrated in Figure 5.1 and Table 5.1 below:
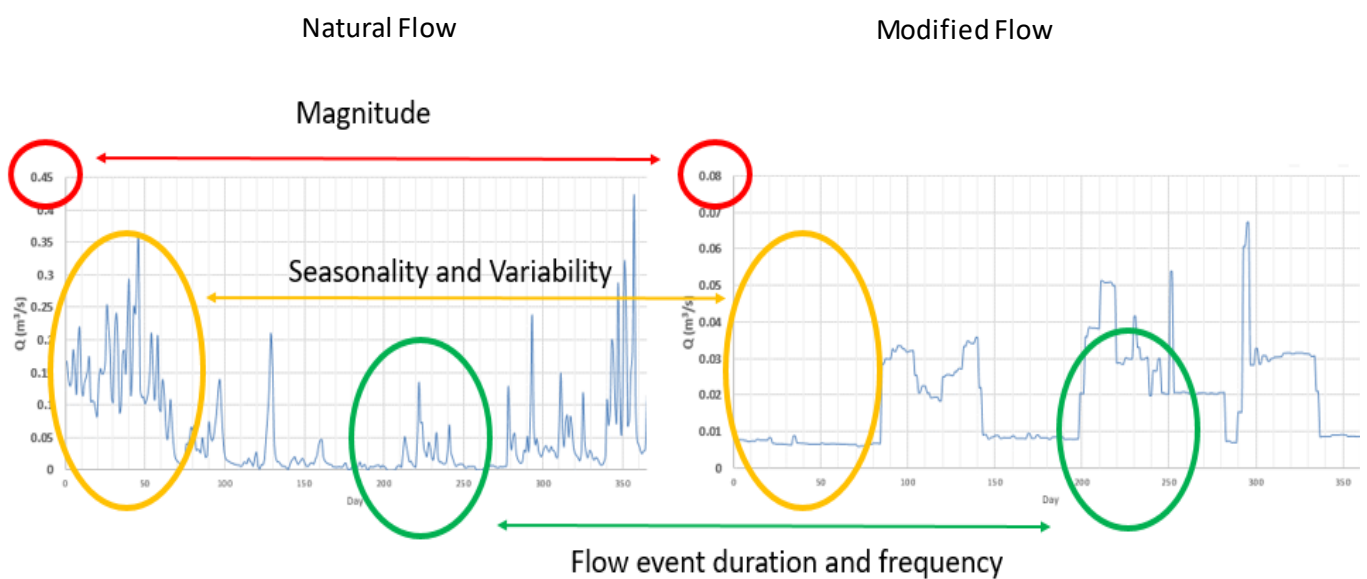
Figure 5.1: Reservoir inflows (left) compared with Reservoir outflows (right), with associated flow features

Table 5.1, Comparative analysis of high and low flow events between 2014 reservoir inflows and outflows

INFLOWS Flow Parameters		OUTFLOWS Flow Parameters		DEVIATION FROM NATURAL
Low pulse count	16	Low pulse count	5	-11
Low pulse duration (days)	3	Low pulse duration (days)	7	4
High pulse count	15	High pulse count	11	-4
High pulse duration	2	High pulse duration	4	2

Figure 5.1 and Table 5.1 show that modified flow at Holden Wood deviates from the natural flow regime in a number of significant ways, in agreement with previous literature (Richter et al., 1996). A

primary difference is that "natural" flows amounted to 1,786,596m^3 over the course of 2014, whereas impoundment outflows released 600,283m^3 during the same period. Due to licence changes under the EA, outflows have increased (detailed in Section 5.2.8 and ecological impacts predicted in Section 5.3) but inflow records for this time were not made available, leading 2014 to be the most feasible year for comparison. It can also be seen in Table 5.1 that low pulse events are significantly lower in reservoir compensation flows, likely due to the baseline flow already being at a low level due to the artificial nature of the flows. High pulse events are also reduced, though to a lesser extent (this may also be due to the low baseline causing the threshold of what defines a high flow to be very low compared to natural conditions). The duration of flow events are also significantly altered, with low and high pulse durations lasting 4 and 2 days longer on average, respectively, than would be expected in natural conditions. The impacts of regulation and their implications will be discussed in greater length within Section 5.4.1. The main aim of this initial analysis was to illustrate the impact of flow impoundment upon river flow, and therefore the extent of regulation was not fully quantified as in some studies (Gillespie et al., 2015a); quantified extent of regulation is generally a more useful metric when comparing between sites.

5.2.2 CASiMiR outputs and supplementary data

Utilising the hydraulic-ecologically linked model developed in Chapter 4, it is possible to predict the ecological response of indicator species to flow based on their flow velocity preferences inputted into the CASiMiR model as described in Section 4.5.2. A steady flow can be inputted into the model in order to generate a response curve of flow vs habitat suitability using the hydraulic and ecological inputs described in Section 4.5, or a flow regime can be inputted and a temporal output of habitat suitability over the year (or length of time inputted) can be generated. This provided insights into species flow sensitivities and their requirements for optimal habitat suitability. The final proposed flow regimes required the incorporation of many factors, however. In addition to the role of flow magnitude on species habitat quality, the influences of flow on habitat heterogeneity and connectivity were also considered, as were temporal variation, reservoir functions, and the potentially conflicting requirements of environmental and societal needs.

5.2.3 Species requirements: flow

Chapter 4 discussed the selection of indicator species based upon their range of flow requirements and their presence throughout the study reach CASiMiR's outputs were utilised to identify improved flows for the provision of indicator species requirements. The Hydraulic Habitat Suitability (HHS) index was utilised to provide an intuitive dimensionless value of overall habitat quality across the site, between 0 and 1. HHS is based upon weighted usable area (WUA) metric (Kelly et al., 2015), divided by the total wetted area. WUA in turn is based on the Habitat Suitability Index (HSI) (Oldham et al., 2000) by multiplying habitat type by area, with greater weighting for higher HSI values. The equations for WUA and HHS are described below as provided by Schneider et al. (2010):

$$WUA = \sum_{i=1}^{n} A \cdot HSI$$

Where A is the area of the i_{th} cell (m^2) and HSI is the habitat suitability value of the i_{th} cell.

$$HHS = \frac{1}{A_{ges}} \sum_{i=1}^{n} A \cdot HSI$$

Where A_{ges} is the area of the entire modelled area (m^2) and the remainder of the equation is equivalent to the calculation for WUA.

In their proposal of HSI, Oldham et al. (2000) state that a direct correlation between HSI value and the species abundance; this also applies to HHS. Whilst this assumption generally holds true, it may be expected that weakening correlation occurs due to external drivers such as biological interactions; high habitat quality facilitates but does not guarantee habitation. Poor quality habitats by definition are unsuitable for significant species populations, and therefore could not sustain a high species abundance, though external drivers may also have an influence upon species distributions within these habitats. The HHS for indicator species was plotted with flow magnitude (Figure 5.2).
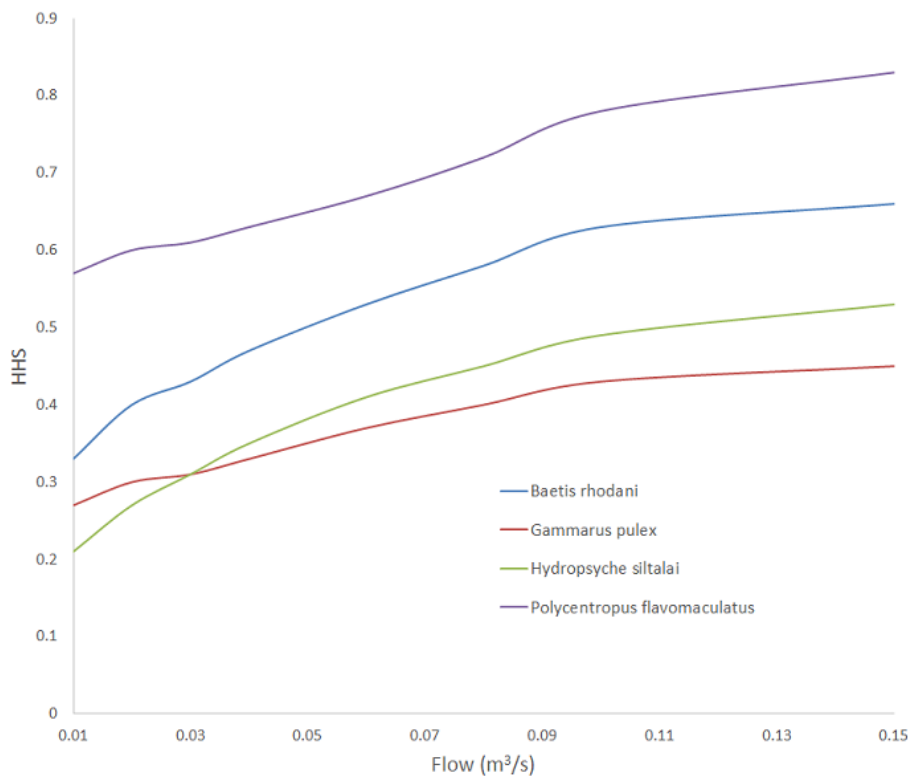
Figure 5.2: HHS response against flow for each indicator species at the case study site

Some species were sensitive to changes in flow; at the low end of the flow range, increasing flow from 0.01m³/sec to 0.05 m³/sec resulted in a HHS increase from 0.21 to 0.45 for *Hydropsyche siltalai*, whilst the same increase in flow resulted in a HHS increase from 0.28 to 0.31 for *Gammarus pulex*. This difference in response is quite significant, particularly at low HHS ranges where increases of in habitat quality may mean the difference between an untenable species population that is constantly danger of being wiped out (for example by predation or sudden flow events), and a small but sustainable population that is large enough to recover from disturbance events (McMullen et al., 2017). Such differences in HHS response suggest that certain species at the site are more vulnerable to changes in flow while some are more resilient. Levels of responsiveness at the flow ranges present within Ogden

Brook (approximately 0.01-0.10 m³/sec) suggest that some species will respond favourably to small increases in flow, whereas others will show little response, particularly at the lowest ranges of flow magnitude. Such findings may optimise flow designations depending upon seasonal species distributions.

2014 Outflows vs 2014 Inflows

Comparisons were made between the Holden Wood data for 2014 outflows and inflows, the inflows acting as a surrogate for a naturalised system free of impoundment. The response of species' HHS to the temporal variation in flows were derived (Figures 5.3 and 5.4).
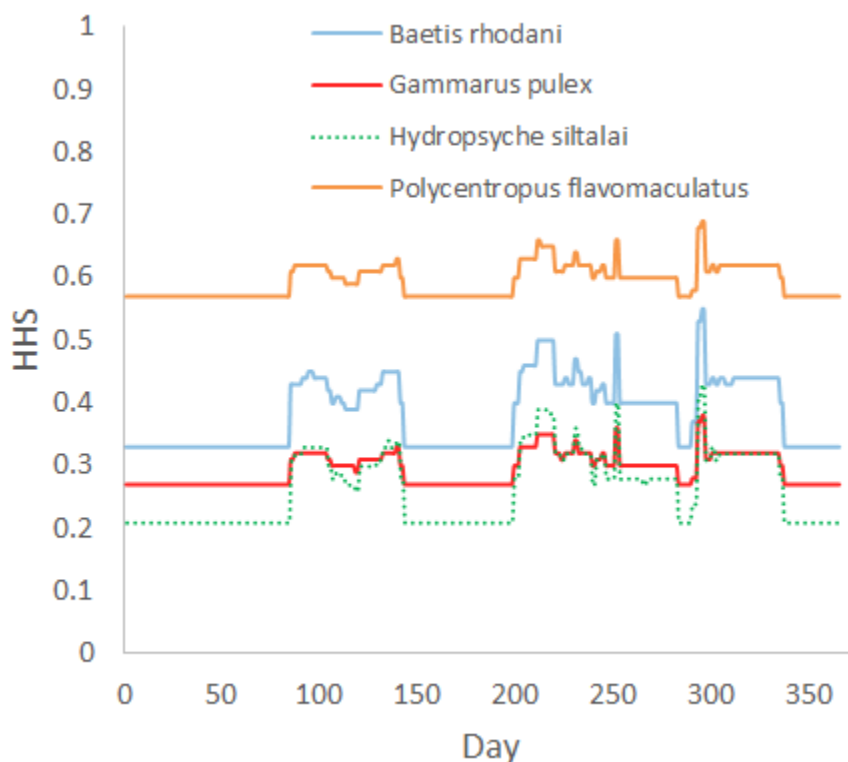
Figure 5.3: Predicted HHS values of indicator species, in response to 2014 Outflows at Holden Wood
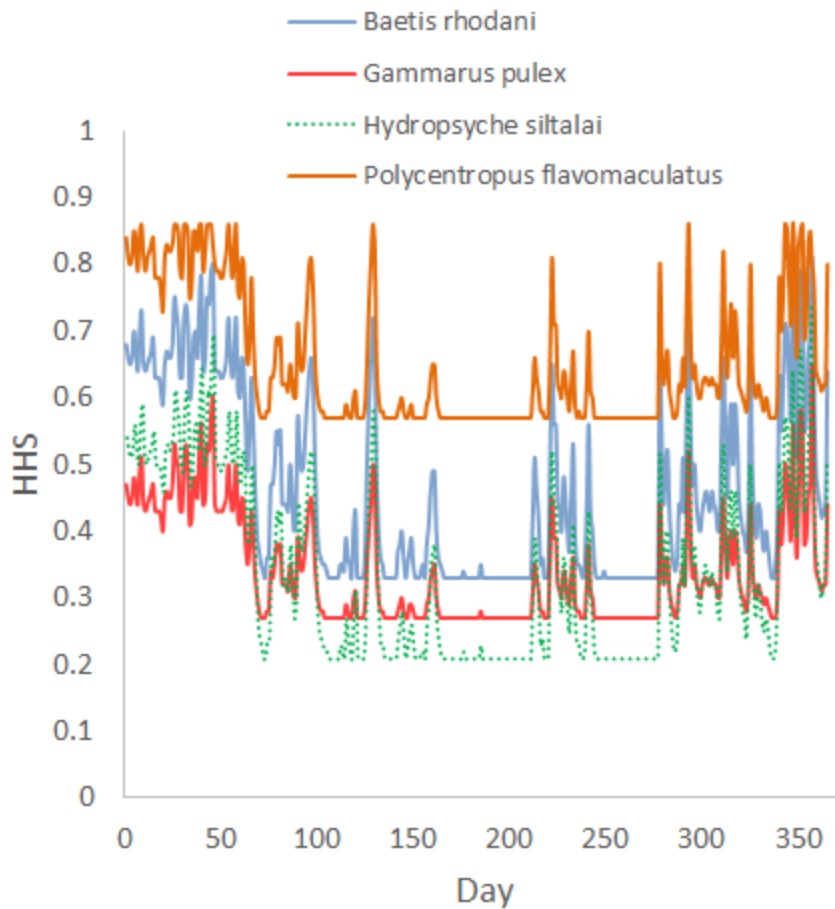
Figure 5.4: Predicted HHS values of indicator species, in response to 2014 Inflows at Holden Wood

Magnitude vs Habitat Heterogeneity

Differences in flow preferences, and responses to flow change, among species also highlights the potential importance of flow heterogeneity in promoting biological diversity (Ward et al., 2002). Homogeneity of flow velocity was identified as an issue associated with the modified flow regime at the study site. To address this, CASiMiR was also used to calculate the flow diversity of available habitat at across range of flows.

An index for habitat heterogeneity was developed using Shannon's Diversity Index (H). The index was applied to the range of velocity distributions present within the river channel at a given discharge, as demonstrated in Figure 5.5. Ranges of velocity reflect the range of flow environments and thus habitats present within the system. H is calculated using:

$$H' = \sum_{i=1}^{s} p_i \ln p_i \qquad (1)$$

Where S is the number of flow categories present in the sample and p_i is the relative proportion of habitat in the ith category (Magurran, 2004).

This was applied by calculating the total wetted area and the wetted area covered by each flow velocity category over a range of discrete steady inflow discharges. CASiMiR defines 8 velocity categories, from "Very Low" to "Extreme". These categories are defined by flow ranges set by CASiMiR for each category, from 0.00-5.00cm/s for Very Low, up to >80.00cm/s for Extreme. The proportion of each velocity category was determined and used in Equation 1 to derive a measure of "flow diversity" for the study reach (Figure 5.5).
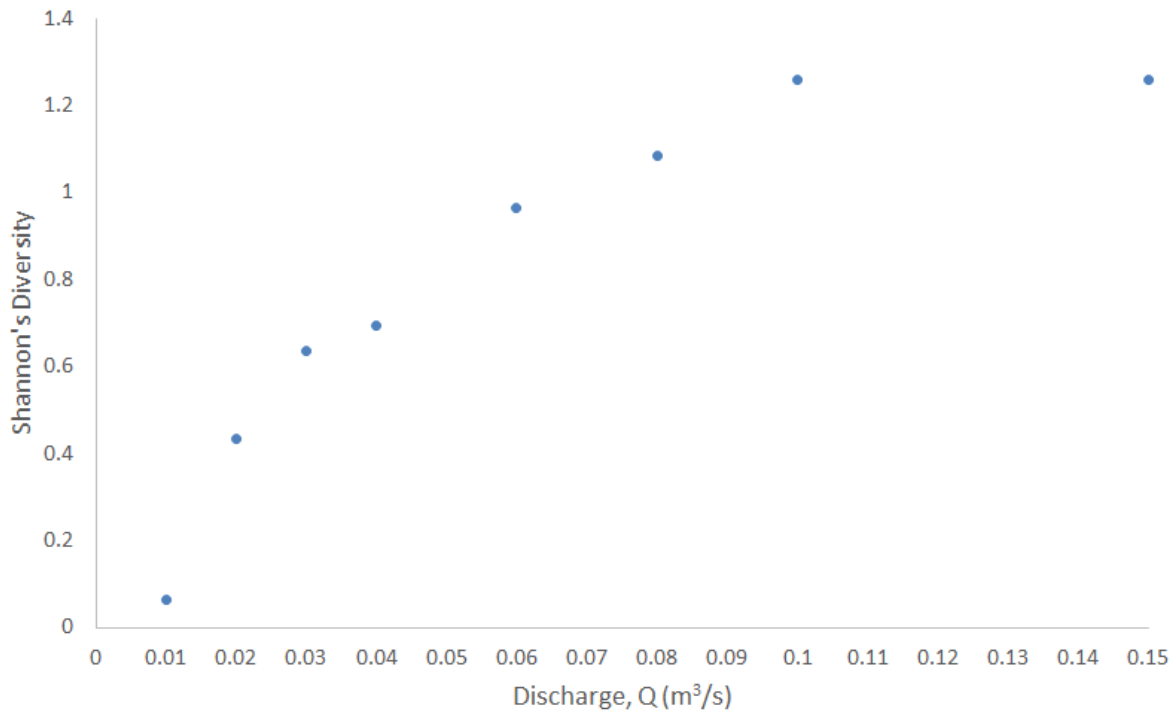
Figure 5.5: Diversity of habitat; Shannon's Index of depth-averaged flow velocity across the river channel vs channel inflow

It was found that habitat diversity increases with flow rapidly in the lower flow ranges, but this trend diminishes and eventually plateaus. Beyond Q=0.1m^3/sec, flow expenditure gives little benefit in terms of habitat diversity, and at higher flow ranges flow-habitat diversity decreases as the river becomes more uniformly fast-flowing.

Ecological Requirements Informing Environmental Flow Design

The response to flow in terms of individual species HHS values, and the response of the ecological conditions more generally in terms of habitat diversity, informed environmental flow regime design. Due to the diminishing returns observed in terms of increasing habitat diversity with flow, alongside the reduced responsiveness of indicator species at higher flows, and due to local infrastructure design being based upon historical flows, designed flows were limited to a maxima of 0.1m^3/sec. Mean diversity across the range of flows (up to 0.1m^3/sec) is approximately 0.75. In order to define a lower bound for designed flows, a critical diversity value was defined as an approximately 80% loss of habitat diversity below the mean (i.e. a diversity value of 0.15), which corresponds to a flow threshold of approximately 0.015m^3/sec. It is recognised that the relative nature of Shannon's index, and the difficulty in quantifying the impact of heterogeneity of habitat upon the ecosystem (Yin et al., 2017), means that meaningful habitat diversity (and thus flow) thresholds are difficult to define objectively. In this study the threshold is designed to act as a buffer to prevent complete habitat homogeneity,

and regime-specific flow regime minima are designated through a combination of habitat diversity and more quantitative species sensitivities identified through HSI values (see earlier in this section). Depending upon the information available for a given system, the approach towards such thresholds and the emphasis placed upon particular metrics may be varied.

It should also be noted that the hydraulic model for the site is calibrated at a significantly lower magnitude than the upper natural flow range ($0.024m^3$/sec vs $0.41m^3$/sec); model results at magnitudes similar to natural conditions may therefore not provide accurate hydraulic predictions. Additionally, local infrastructure has developed alongside the current state of the flow regime; "natural" flow ranges in reservoir inflow data would be unsuitable for the current state of the river channel and could increase the risk of flooding in the surrounding urban area. This threshold was applied to spring, summer and autumn. Due to lower biological activity in winter (White et al., 2017, Olsson, 1982) as discussed in Section 2.2.4, a lower flow of $0.01m^3$/sec was deemed acceptable. During high flow periods, a maxima of $0.1m^3$/sec gives optimal habitat diversity prior to a plateau in the flow-diversity relationship. A full account of flow regime designation is detailed in Sections 5.2.8 and 5.2.9.

5.2.4 Temporal flow requirements

Habitat modelling provides a prediction of ecological response to changes in flow magnitudes. However, this alone is not sufficient to derive holistic environmental flow regimes. Chapter 3 highlighted the fact that the timings, frequencies, and durations of flow events must be considered in terms of ecological requirements. Sections 3.3 and 3.4 in particular demonstrated the importance of flow event frequency as a driver of ecological response. Additionally, as discussed throughout Chapters 1 and 2, societal needs and practical constraints must be considered in terms of impoundment operation and storage. Such factors cannot be considered within the CASiMiR model alone, and are often unique to a particular river or region (Konrad et al., 2011). In these cases, species requirements from literature, and natural flows from other river systems in the North West of England, were used to supplement model outputs and were integrated into flow regime development.

Ecological stability can be compromised by the loss of natural flow characteristics (Poff et al., 1997), and therefore supplementary data was required to inform flow regime design in terms high flow event frequencies and durations. As river systems of a similar geology and geography experience the same climatic conditions and tend respond to a given flow in a similar manner in terms of thermal regime and physicochemical properties (Alcazar and Palau, 2010, Arthington et al., 2006), it is expected that the biota at Holden Wood should respond favorably to high flow event frequencies and durations that are approximate to typical naturalised flow regimes within the region (low flow events were not considered due to baseline impoundment outflows already being comparable to low flow events). This approach is comparable to the Before/After Control Impact approach (Underwood, 1991), but is applied on a more general regional level and does not require extensive conformity with specific reference conditions. Long-term Holden Wood inflow data was not available, and a transferable "regional" set of conditions was desired; therefore flow data was obtained from 7 non-heavily regulated rivers across the North West of England, around the Greater Manchester and Lancashire areas, through the CEH NRFA website (Centre for Ecology and Hydrology, 2018), and the typical frequency and duration of high flow events in the region were identified. Rivers with an average daily flow above $1m^3$/s were excluded, ensuring rivers of similar magnitude to Holden Wood's natural state (derived from impoundment inflow data). This flow data, spanning on average 37 years, was processed

using IHA software (The Nature Conservancy, 2017). The particular variables of "High pulse frequency" and "High pulse duration" were extracted from software outputs, and the median of these values was taken for each of the 7 sites. "High flows" or "high pulses" are defined in this study as flows that exceed 75% of the mean daily flow record. Analysis outputs are shown in Figure 7. Mean standard deviation of sites was 5.648 from the mean high pulse count across sites, and 0.488 for high pulse duration (measured in days).
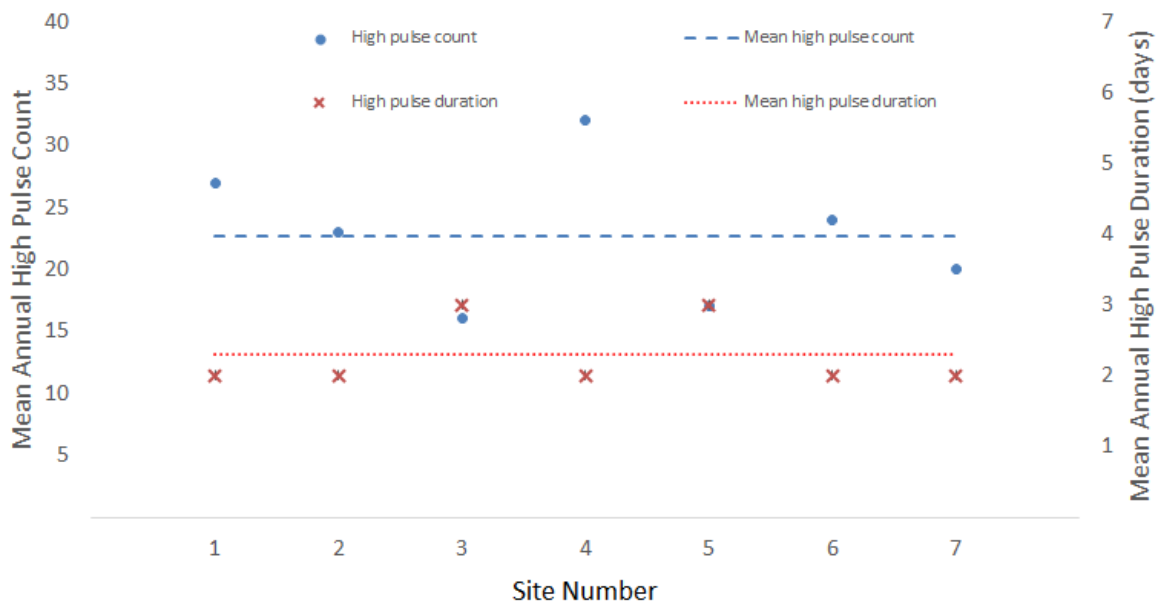
Figure 5.6: High pulse event frequency and duration at 7 non-regulated sites, demonstrating extent of similarity of conditions in the North West Greater Manchester and Lancashire region.

It can be seen from Figure 5.6 that mean annual high pulse frequencies across the seven sites range from approximately 15 to 33, with an average of approximately 23 events per year. Mean annual high pulse duration is less varied, with most sites having high pulse durations of two days, and two having a duration of three days. For comparison, Holden Wood 2011-2014 inflows were found to have an annual mean high pulse frequency of 15 per year and a mean high pulse duration of 2.5 days through the IHA software. The reservoir inflows utilised as reference conditions for a natural system are therefore within the flow condition range typical to the North West England region.

In addition to the variability of flow over a period of time such as a season, another key question was species occurrence over the course of the year. A preliminary analysis of temporal species persistence was performed using Cascade consultancy ecological sampling data from 2016, based on single-point, three minute kick samples taken in April, May and October (Table 5.2). Species life history information from literature was also assessed (Beltran Epele et al., 2011, Raddum and Fjellheim, 1993, Welton, 1979, Andersen and Klubnes, 1983, Berg and Hellenthal, 1992). Variation in species populations were considered between seasons. The peak population for each species within the river throughout the year was calculated. A period in which a species is at peak abundance within the river system is assumed to be a key time of year for that species, where it will be susceptible to changes in flow. Periods of very low species abundance (due to the species being at a life stage where most individuals have emerged as terrestrial adults) are unlikely to be times of key importance or sensitivity for that

species. Other key periods may exist, such as egg laying, which are not clearly reflected by mere population numbers. Such features and timings were searched for within previously cited literature.

Table 5.2: 2016 Data on species distributions in spring and autumn for single-point 3-minute kick samples, provided by Cascade Consultants

Species	Spring abundance	Autumn abundance
Orthocladiinae	196	20
Tanypodinae	11	5
Tanytarsini	30	7
Baetis rhodani	75	30
Gammarus pulex	32	360
Hydropsyche siltalai	12	145
Polycentropus flavomaculatus	0	67

At Holden Wood, literature and observed data were not in agreement in terms of species seasonal abundances; differences in river temperature can be a strong influence on life history timings (Poole and Berman, 2001). Water temperature may be modified by the reservoir itself due to thermal stratification, and this may lead to desynchronization between observed species activity and general trends reported in literature. While seasonal variation is apparent in the data seen in Table 5.2, it is not clear how this accords with expectation from life-history information in the literature, and the data set is insufficient to base seasonally specific flow targeting upon. Therefore no differentiation was made between the targeted seasons of spring and autumn in flow regime development.

5.2.5 Habitat distribution, connectivity, and persistence

In addition to the considerations of heterogeneity, the more temporal considerations of habitat connectivity and persistence also play a role in ecological diversity and stability. Areas of high habitat quality are of little use to an individual if they are not persistent and not connected to alternative habitats, as the individual would then be left in poor conditions without access to refugia, assuming that a given species lacks the mobility to relocate to a more preferable location in a timely manner – this is a valid assumption for many macroinvertebrate species, as macroinvertebrate mobility varies greatly across species (Mackay, 1992). Other influences unrelated to habitat quality also necessitate connectivity to other suitable habitat; high levels of competition or predation, for instance, may make it desirable for individuals to move if possible (McCabe, 2010). At the flow ranges expected at the Holden Wood site, habitat quality and connectivity almost unanimously increases with flow. Thus, the most intuitive and productive question appeared to be, how does habitat persistence and connectivity change with decreasing flow?

Flows were initially set as 0.06 m^3s^{-1}, close to the maximum flow currently observed at Holden Wood. Flow was then reduced in steps to 0.04, 0.03, 0.02 and finally 0.01 m^3s^{-1} and the change in spatial distribution of habitat quality (HSI) was assessed for each individual species. Flow diagrams of habitat distributions at each flow were created and the connectivity and persistence of good to moderate habitat for each individual species was assessed and the implications considered. The HSI for each species was predicted at a range of flows, starting at a maxima and decreasing. Predictions were visualised using CASiMiR's Plan View to show changing connectivity across the river reach. Connectivity in terms of the spatial distribution of high-quality habitat decreased with decreasing flow for all species, though at different rates. Examples of habitat connectivity and persistence with decreasing flow for individual species are shown below in Figures 5.7-5.9.

The species *Baetis rhodani*, *Gammarus pulex*, and *Hydropsyche siltalai* are presented below. *Polycentropus flavomaculatus* was omitted due to the species' resilience to low flow conditions, which simply led to increasingly homogeneous good habitat conditions as flow decreased:
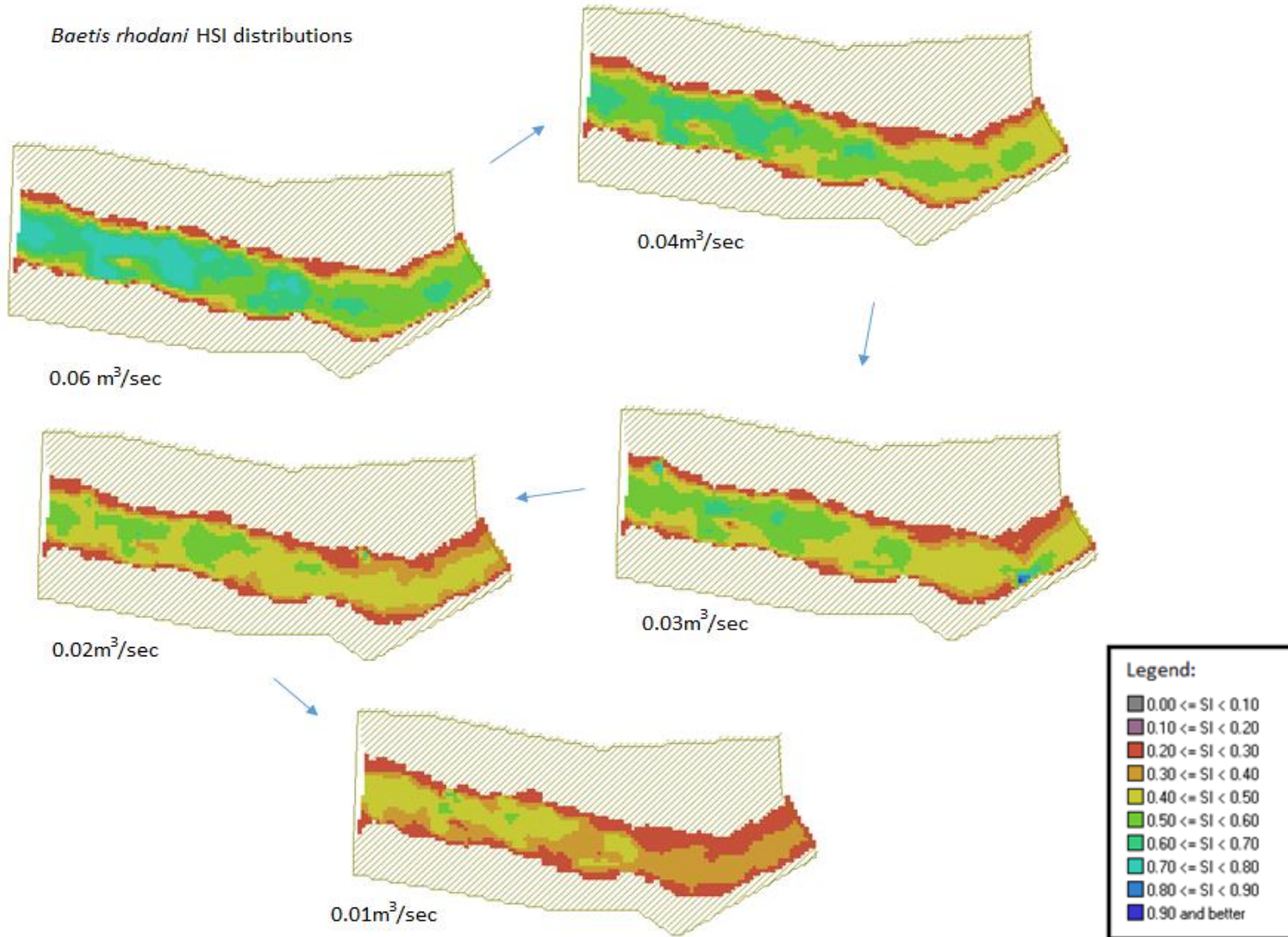
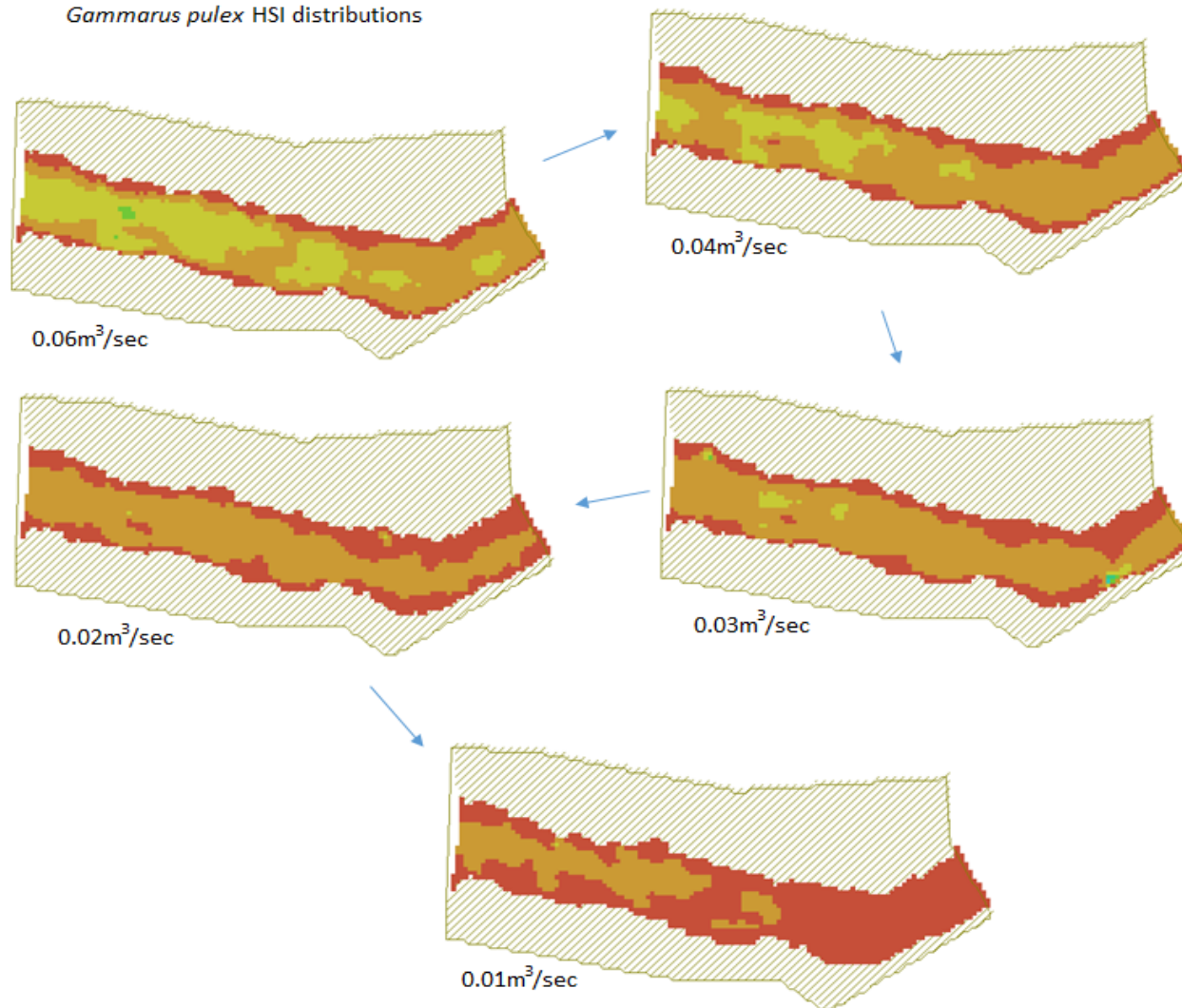Figure 5.7: Habitat connectivity and persistence for Baetis rhodani

Figure 5.8: Habitat connectivity and persistence for Gammarus pulex
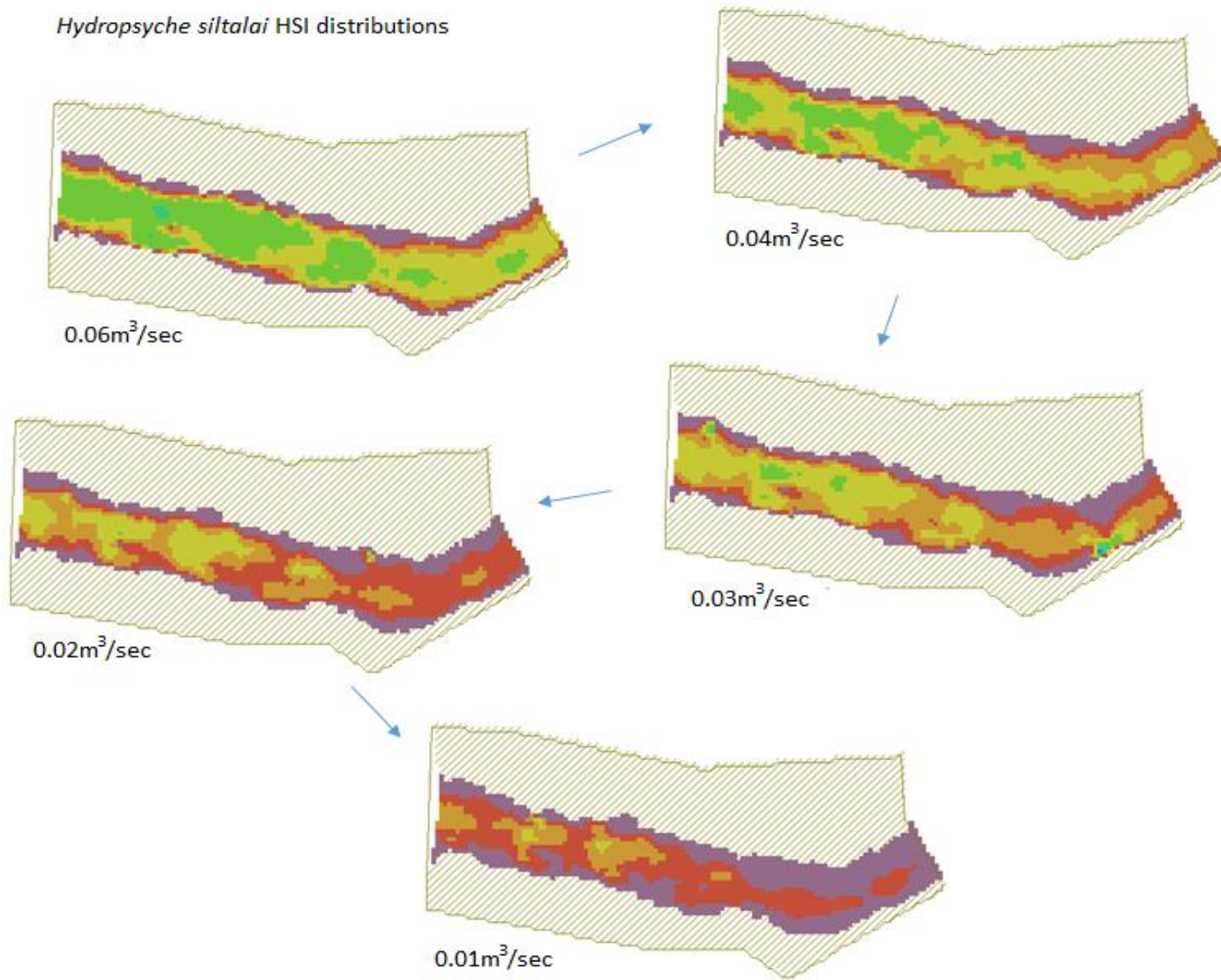
Figure 5.9: Habitat connectivity and persistence for Hydropsyche siltalai

Figure 5.7 demonstrates that *Baetis rhodani* has abundant and well-connected habitat (HSI >0.5), at 0.06 and 0.04m^3/sec. As flow is reduced further to 0.03m^3/sec, connectivity begins to diminish, particularly in the downstream region of the assessed river channel. At 0.02m^3/sec, favourable habitat begins to break up into pockets, with lower SI (<0.4) being the dominant condition in the river. Finally, at 0.01m^3/sec, it can be seen that the downstream habitat is entirely poor, whilst upstream remains habitable but lacks areas of habitat where the species might be expected to flourish.

Baetis rhodani is one of the more resilient indicator species; *Hydropsyche siltalai* and *Gammarus pulex* fare much more poorly at low flows, and the spatial changes in HSI likely have a much greater impact. Figures 5.8 and 5.9 demonstrate that as flow calls to 0.02m^3/sec, the majority of the channel is of poor habitat quality, and at 0.01m^3/sec most of the channel is at a HSI of 0.2 or lower. Should these species be unable to move elsewhere, it is likely that these macroinvertebrates in these patches will see a significant decline, or even total loss, of population. This could particularly be the case in the downstream area of the river that sees the lowest levels of HSI in both cases. Such spatial modelling may help to inform water managers as to the impact flow might have upon populations throughout an investigated river system that might not be clear from site-wide metrics such as HHS. This may in turn inform flow regime decisions. From the example provided here, one might conclude that a flow of 0.02m^3/sec would be detrimental to all three indicator species, particularly the more sensitive *Hydropsyche siltalai* and *Gammarus pulex*, and flow should be kept above this where possible. Further, one might conclude that flow falling to 0.01m^3/sec leads to critical conditions in a large part of the system (the downstream section) for these two species, and such flows should be avoided if at all possible.

Table 5.3 summarises some insights that a water manager might take from the spatial HSI information. Both *Hydropsyche siltalai* and *Gammarus pulex* were sensitive to low levels of flow. In the case of *Hydropsyche siltalai*, connectivity decreased to a critical point where it was lost almost entirely. From these outcomes, flow thresholds might be developed for recommended flows in order to avoid a critical loss of connectivity. These thresholds could be adapted season by season, making provision for a given species composition. In the table, "Soft Threshold" represents a recommended low flow threshold for a species, which should usually not be crossed due to likely detrimental impacts. "Hard Threshold" similarly is a low flow threshold, but one that should be avoided if at all possible, as below this threshold flow high levels of mortality would be expected. This thesis acknowledges that this is a somewhat qualitative approach to spatial HSI representation, and further thoughts on this matter are discussed in Section 5.4.6.

Table 5.3: Flow Thresholds for indicator species, based on habitat persistence, connectivity and heterogeneity

Species	Soft Threshold (m^3/sec)	Hard Threshold (m^3/sec)
Baetis rhodani	0.01	0.005
Gammarus pulex	0.04	0.02
Hydropsyche siltalai	0.03	0.02
Polycentropus flavomaculatus	N/A – Resilient	N/A – Resilient

5.2.6 External variables

In addition to these approaches, the complexity of the river system means that other influences on ecological response are beyond the scope of this investigation. External variables include the influences of land use, water chemistry, nutrient availability, light availability, water temperature, sediment transport and biological interactions. Such external variables must be acknowledged, and changes to flow regime must consider whether the above variables may also be changed as a result, and what the implications of those changes might be. Quantifying such variables, however, is beyond the scope of this investigation, and selection of study site was significantly influenced by its smaller scale, and its flow being primarily controlled by impoundment releases (as discussed in Sections 4.2 and 4.3); this allowed the influence of external variables to be minimised as much as possible. It is impossible to control most external variables without a disproportionate amount of effort being required, and therefore this thesis has focused upon flow, a variable that can be feasibly controlled and manipulated. Having been performed within a (relatively) controlled environment within Ogden Brook and laying out the foundations of this approach, further study may adapt the method for larger and/or more complex systems, as discussed in Section 6.2.2. This said, potential external variables to be considered by water managers are listed with their possible impacts in Table 5.4:

Table 5.4: External variables and their associated impacts within the river system

External Variable	Influence	Citation
Temperature	Productivity, growth, life history timings	(Olden and Naiman, 2010)
Nutrient availability	Productivity, growth, population distributions, water quality in cases of over saturation (eutrophication)	(Vannote et al., 1980)
Biological interactions	Species distributions and numbers	(McCabe, 2010)
Shade	Species distributions, Productivity	(McCabe, 2010)
Oxygen concentration	Productivity, species numbers, nutrient availability	(McCabe, 2010)
pH	Plant growth / productivity, microbial activity, climate for pH-sensitive species	(Fondriest Environmental Inc, 2013)
Sediment type and transport	Species distribution, visibility (fine sediment in water), nutrient retention, spawning, river morphology	(Wampler, 2012)

As well as considering ecologically-important variables, the practical constraints of the reservoir must also be regarded. The following section discusses how reservoir storage was integrated into flow regime design decision making.

5.2.7 Impoundment storage model

When designing managed outflow from impoundments based on ecological modelling, the practical consideration of the impoundment structure and operational rules must be considered. In this case both minimum permitted water levels as well as the operational capacity of Holden Wood must be accounted for. Failure to utilise the impoundment capacity sustainably could result in drainage down to the extent at which the impoundment is no longer able to drain under gravity, a point below which additional storage is termed "dead water". This would breach the impoundment contract set by the Environment Agency, and would lead to prosecution if not mitigated. Flow regimes were designed this constraint in mind. "Dead water" Holden Wood is below 37,000m^3 (Maddison, 2012), therefore a significant buffer from this water level was set. A threshold of 100,000m^3 was designated, that Holden Wood should not drop below, based on discussions with United Utilities. A simple model was therefore developed to understand storage levels as a function of both measured inflows and simulated 'designed' outflows over each period of analysis. This also allows the calculation of the 'efficiency' of each designed flow regime in terms of maintaining impoundment water levels. The model operated using impoundment inflow data paired with outflow data (historical or proposed flow regimes), impoundment storage capacity, and volume of spill from the volume of inflow per day that exceeds reservoir storage capacity. At each daily time step the change in storage within the impoundment is calculated as:

$$\frac{dV}{dt} = I - (O + Sp)dt$$

Where V is current impoundment storage volume (m^3), t is time (s), I is daily inflow (m^3/s), O is daily prescribed outflow (m^3/s), and Sp is overflow spill rate (m^3/s). Based on the capacity of the impoundment, Sp = 0 when level is below reservoir capacity (367,000m^3), and when above this capacity, is defined as total volume of capacity exceedance. At the start of each period of analysis the storage volume is set based on known values taken from historical records kept by United Utilities. Water levels are monitored for each simulation across a proposed outflow time series, such that the total released volume of water over each period is known to ensure that levels do not fall below the prescribed minimum threshold. The storage model assumes that daily excess water (Sp) is released within one day, as would be expected in all but the most extreme climatic conditions.

5.2.8 Flow regime design

Various aspects of this investigation have informed particular facets of flow regime design. Proposed regime designs encompass better provision for species flow requirements, encouragement of greater diversity of habitat, increased temporal flow heterogeneity (particularly with regard to high flow frequency), and also acknowledge infrastructural constraints and the need to reduce annual outflows relative to 2017 impoundment releases due to economic and water security constraints.

2017 outflows at Holden Wood released 1,180,460m^3 of water over the course of a year under the new impoundment licence. Previously, 2014 outflows released 600,284 m^3. HHS was assessed under both of these conditions and a goal was set to maintain or exceed ecological metrics, whilst using significantly less water than 2017 releases. Three regimes are proposed with different design focuses. Each designed regime was compared to historical inflow data, and the ecological benefit of each regime was considered alongside volume of water released and optimal use of flow; for instance high

flows in winter may not have a significant impact upon overall ecology due to the biologically-limiting effects of the thermal regime during this period (Olsson, 1982, Vannote and Sweeney, 1980).

5.2.9 Flow designation

Individual species requirements, habitat diversity, typical regional flow event duration and frequency, and practical reservoir and site constraints were considered in order to design annual flow regime magnitude and timings with the aim of optimising ecological provision relative to volume of water released. Due to the diverse interests that may be present at a site, and due to the potential conflict between societal needs and environmental requirements, three flow regimes were proposed. These designed regimes (A, B and C) vary based on their balance between ecological provision and water conservation focus, thus demonstrating the utility of the approach and allowing it to be adapted to the context of application; for example providing a vital societal service that must ensure water security, or ensuring provision for a site of special scientific interest and therefore maximising habitat suitability of a selected species as a priority.

Regime A aims to maximise habitat diversity and HSI during flow maxima whilst releasing a similar overall volume of water to 2017 outflows; Regime B aims to balance the two priorities, retaining a modest amount of water and maintaining moderate habitat diversity and HSI; Regime C retains more than 50% of the water released in 2017 outflows, but ecological metrics are at threshold values. A full account of regime design characteristics and rationale is provided in Table 5.5. All regimes follow the same general design shown in Figure 5.10, with five high flow pulses occurring in spring and in autumn respectively, with magnitude varying with regime. This pulse frequency and duration criteria is based on values identified in Section 3.4. Summer and winter retain constant flow rates (not including impoundment spills); in the case of summer, the season lacks biological information and there is a need to retain as much water as possible due to lower rainfall, increased water demand, and the risk of drought, and in the case of winter the cold seasonal climate leading to dormancy among many taxa suggests lower flow requirements in this season, additionally, supplementary flow from spill events is common in this season due to elevated rainfall.
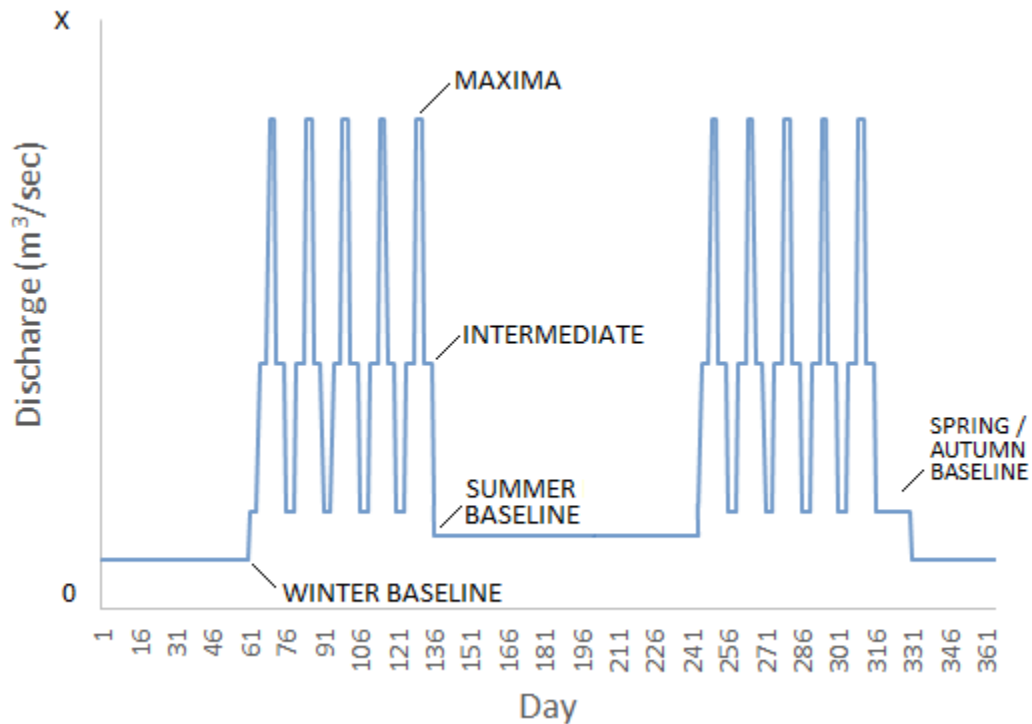
Figure 5.10: General design of proposed flow regimes; A, B, and C.

Table 5.5: Breakdown of individual flow regime design characteristics with their rationale

Characteristic	Rationale
Maxima A – 0.1m³/s B – 0.075m³/s C – 0.04m³/s	Periods of high flow cultivate elevated habitat diversity and high mean HHS values across indicator species for short, repeated periods in spring and autumn. Such flows also aid in regulating the system's physicochemical properties (Alcazar and Palau, 2010).
Intermediate A – 0.05m³/s B – 0.03m³/s C – 0.025m³/s	Based on good habitat diversity and moderate-high HHS values across indicator species whilst remaining within annual flow target, prolong the period of higher flow, prevent the flow increases being too sudden and disruptive to the native ecosystem (Blanckaert et al., 2013).
Spring / Autumn Baseline A – 0.02m³/s B – 0.02m³/s C – 0.015m³/s	Based on threshold for most sensitive species present in these seasons, *Gammarus pulex* and Hydropsyche siltalai, identified in the seasonal analysis of consultant data. HSI becomes poor (>0.03) below flows of 0.02m³/sec (see HSI vs Flow in Figure 5). HSI above 0.02 is maintained in Regime C, a habitat of low carrying capacity but still tolerable (Oldham et al., 2000)

Reduced Summer Baseline (Regimes A and B only) 0.015m³/s	Threshold based on habitat diversity and critical habitat quality responses to flow. Season lacks biological information and there is a need to retain as much water as possible due to lower rainfall, increased water demand, and the risk of drought.
Reduced Winter Baseline 0.01m³/s (Regimes A, B and C)	Lower productivity, and dormancy among many taxa during winter, suggests lower flow requirements in this season (Olsson, 1982). Elevated rainfall regularly supplements winter flow with spill events.

5.3 Results

Figure 5.11 demonstrates the influences of Regimes A, B and C upon Holden Wood storage. The reservoir begins the year full due to high winter precipitation levels leading this to typically being the case; this is based upon reservoir records from the previous year (2013), supplied by United Utilities, in which the reservoir was at full capacity towards the end of December.
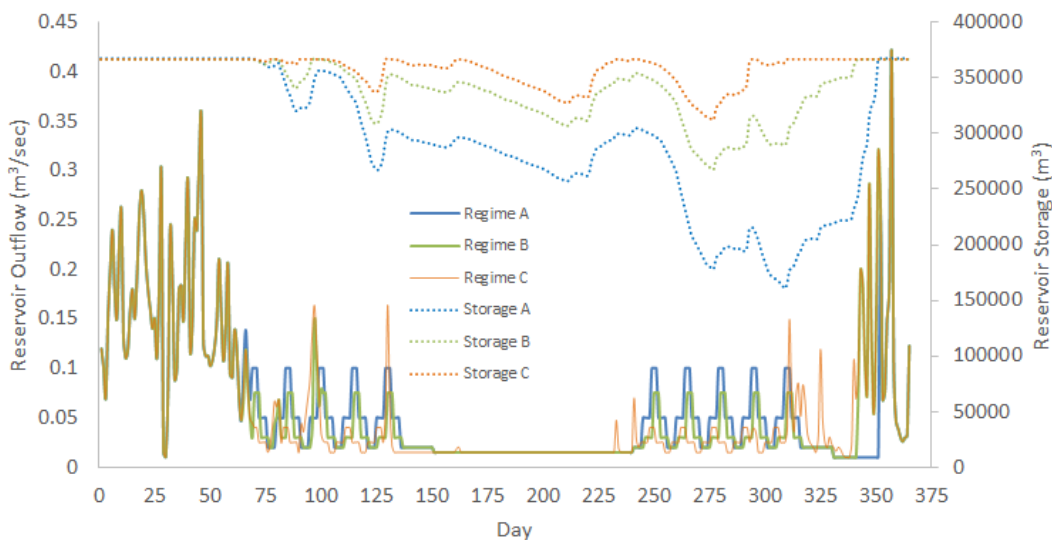
Figure 5.11: Flow regimes A, B and C with associated changes in reservoir storage

It can be seen in the above figure that spills play a significant role throughout the winter period, and also have some influence upon the flow regime during the spring and autumn periods under smaller flow regimes B and C. Spills are a result of the reservoir being at capacity and water therefore being released from the reservoir's overflow channel into Ogden Brook. These spills elevate the flow within the river system for a short time and therefore have a sporadic influence upon the ecosystem. The predicted levels of spill demonstrated in Figure 5.11 are predicted to be of short-term ecological benefit (see Figure 5.12, discussed shortly), but due to spills being caused primarily by heavy precipitation events, this thesis does not consider spills to be a reliable mechanism by which ecological requirements might be met, particularly in terms of timings, durations, and frequencies.

Historical flows are also considered and compared with proposed regimes; based on historical measured data, 2017 outflows at Holden Wood released 1,180,460m³ of water over the course of a

year under the current impoundment licence. Under a previous licence agreement, 2014 outflows released 600,284 m^3. The increase in flow under the current licence is largely motivated by environmental concerns; 2017 outflows thus provide a good example of the continued use of the traditional steady outflow approach for ecological provision. It is therefore possible to demonstrate potential ecological benefits provided by increased flow magnitudes under the new licence, and to demonstrate how ecological needs may be met more efficiently under proposed designer flows. As a reference case the yearly variation in HHS based on the CASiMiR model was assessed under the conditions defined by 2014 and 2017 outflows. Figure 5.12 demonstrates the outcomes in terms of mean HHS between the 4 indicator species for each of the designed flow regimes and historical outflow data.
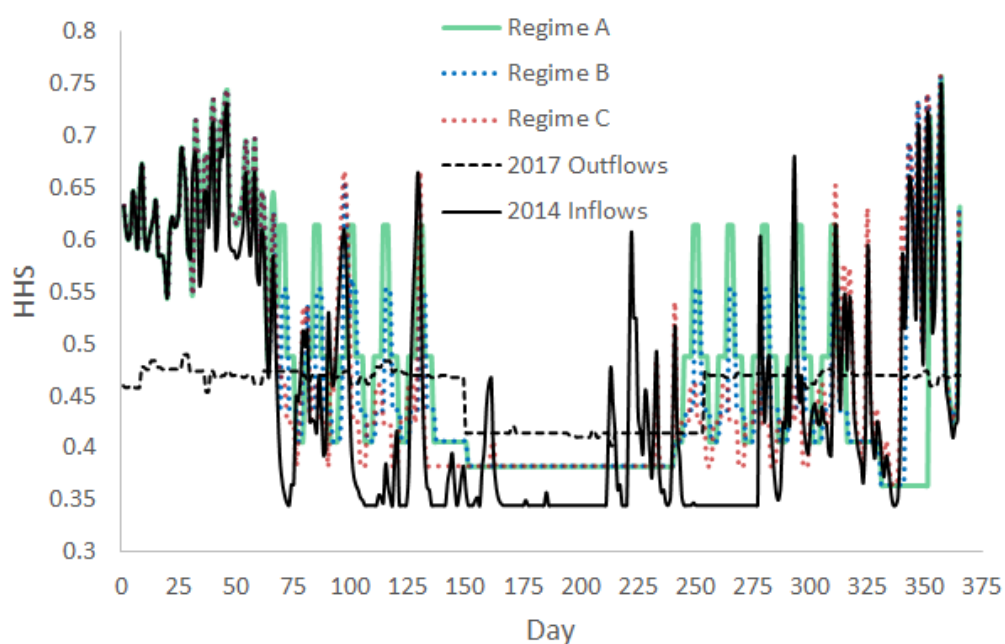
Figure 5.12: Mean HHS predictions resulting from implementation of flow regimes A, B and C, alongside mean HHS based on 2017 outflows and 2014 inflows (values include effects of predicted impoundment spills)

It can be seen from the above results that Regime A maintains good to moderate mean HHS (~0.5-0.6) for much of the spring and autumn period, whilst Regime B maintains lower-moderate values (~0.45) with periods of higher HHS approaching 0.55 during pulse maxima. Regime C maintains lower-moderate values for much of the two seasons (~0.40-0.45), with minima values dropping to 0.35; approaching the lower end of the tolerable HHS range. The more water that is conserved within a given regime, the more likely it is that spill events will occur due to impoundment capacity. These events are determined by annual precipitation however, and may not be a reliable supplementary provision due to this inherent unpredictability. The HHS values of indicator species were assessed between flow regimes, evaluating the average HHS under uniform flow and the three proposed designs, and also evaluating the peak HHS achieved under the same flows. These results are displayed in Tables 5.6 and 5.7.

Table 5.6: Average HHS for 4 indicator species at Holden Wood under historical and designated flow regimes, displaying each regime's annual flow output in cubic metres.

Average HHS

	2014 Outflow (600,284m^3/yr)	2017 Outflow (1,180,460m^3)	A (924,480m^3)	B (721,440m^3)	C (565,488m^3)
Baetis	0.38	0.46	0.4	0.39	0.37
Gammarus	0.29	0.33	0.3	0.3	0.29
Hydropsyche	0.26	0.34	0.28	0.27	0.25
Polycentropus	0.59	0.63	0.61	0.6	0.59

Table 5.7: Peak HHS for 4 indicator species at Holden Wood under historical and designated flow regimes, displaying each regime's annual flow output in cubic metres.

Peak HHS					
	2014 Outflow (600,284m^3)	2017 Outflow (1,180,460m^3)	A (924,480m^3)	B (721,440m^3)	C (565,488m^3)
Baetis r.	0.55	0.5	0.61	0.57	0.47
Gammarus p.	0.38	0.35	0.43	0.39	0.31
Hydropsyche p.	0.43	0.39	0.49	0.44	0.35
Polycentropus f.	0.69	0.65	0.78	0.71	0.63

HHS values between 2014 and 2017 flows show limited response to flow increase; *Baetis rhodani* shows the greatest change, and even here an increase of only 0.08 HHS is observed; a definite improvement, but requiring over 400,000m^3 more water to be sent downstream per year. Designated regimes are shown to be capable of maintaining average annual ecological metrics at acceptable levels, while conserving significant quantities of water and providing frequent habitat quality maxima within the most ecologically-relevant seasons (based on Environment Agency sampling procedure). Habitat quality maxima demonstrate a dramatic improvement in terms of applied ecological principles; flow variation is far greater, with ten high pulse events in contrast to the two or three throughout the year in 2014 and 2017 outflow data, and pulse magnitude is significantly higher in regimes A and B, with above a 100% increase (approximately 0.045m^3/sec up to 0.10m^3/sec) for Regime A, and an approximate 66% increase for Regime B (up to 0.075 m^3/sec). Regime C maintains pulses in spring and autumn seasons similar to those of 2017 outflows (though with lower duration and more flow fluctuation), despite releasing less than half the amount of water annually.

Spring and Autumn Seasonal HHS

Ecological target seasons of spring and autumn were analysed and the flow regimes from 2014 and 2017 were compared with designer regimes in terms of daily HHS distributions, as shown in Figure 5.13.
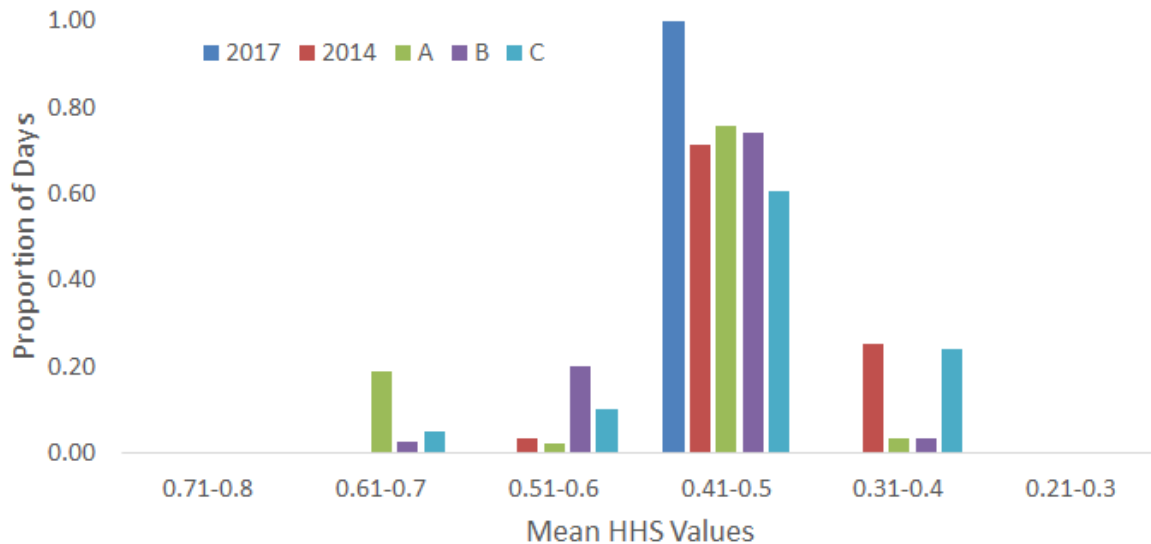
Figure 5.13: Ranges of HHS across spring and autumn seasons for historical and proposed flow regimes

Throughout spring and autumn, 100% of 2017 daily outflows generated a mean HHS in the range of 0.41-0.50; both release rates remained in one HHS range. For 2014 flows, 3% of daily flows were in the range of 0.51-0.60 HHS, likely due to spills. 72% were in the range of 0.41-0.50, and 25% were in the range of 0.31-0.40 due to alternating flow between two release rates. For Regime A, 19% of daily flows were in the range of 0.61-0.70 HHS, mainly associated with the maxima, 2% were in the range of 0.51-0.60, associated with spills, 76% were in the range of 0.41-0.50, associated with intermediate flow and baseline, and 3% were in the range of 0.31-0.40, associated with reductions in flow just prior to winter. For Regime B, 3% of daily flows were in the range of 0.61-0.70, associated with spills, 20% were in the range of 0.51-0.60, mainly associated with the maxima, 74% were in the range of 0.41-0.50, associated with the intermediate and baseline, and 3% were in the range of 0.31-0.40, associated with reductions in flow just prior to winter. For Regime C, 5% of daily flows were in the range of 0.61-0.70, associated with spills, 10% were in the range of 0.51-0.60, again associated with spills, 61% of flows were in the range of 0.41-0.50, associated with maxima and intermediate, and 24% were in the range of 0.31-0.40, mainly associated with baseline.

5.4 Discussion

The results of this study support the premise behind criterion-driven flow design encompassing both temporal and magnitude-based requirements. Despite greatly increased outflows in 2017 historical data compared to other regimes, HHS did not increase in favourable proportion. Whilst 2017 outflows have increased significantly relative to 2014, they remain largely homogeneous and fail to integrate natural variation such as high flow pulses. Thus, whilst more than 400,000m^3 of additional water is released, ecological improvement relative to this is minimal. A holistic approach to environmental flow design is necessary to efficiently provide for ecological requirements in a world with increasingly pressing and conflicting water resources demands. This is consistent with findings from other recent studies (Gillespie et al., 2015a, Gillespie et al., 2015b, Worrall et al., 2014, Brooks and Haeusler, 2016).

5.4.1 Comments on "Natural inflow" and "Impoundment outflow" 2014 data sets

Primary differences between expected natural flows and impoundment releases were identified using 2014 impoundment inflow and outflow data, and demonstrate the loss of natural conditions in the modified system. Expected natural flows demonstrate significantly higher magnitudes of flow; natural flow reaches a maximum of 0.42m^3/sec throughout the year, whilst impoundment flow releases reach a much lower maximum of 0.068m^3/sec. Mean natural magnitude throughout the year is 0.057m^3/sec, whilst mean flow magnitude from impoundment releases is 0.018m^3/sec. The total flow release throughout the year further demonstrates differences in magnitudes; while the natural system would release 1,786,596m^3 over the course of 2014, impoundment outflows released 600,283m^3 throughout the year. Natural flow magnitudes are also highly variable, and therefore rather than being a single ongoing moderate flow, we observe frequent shifts between high and low flows in the ranges discussed. This is in contrast to outflow data, and is discussed further below.

The flow records show that periods of low or high flow in the modified system remain relatively steady for weeks to months throughout the year. Flow event frequency is very low, with only four significant flow events taking place over the course of the year. This is in contrast to the natural system, where flow events are generally short but frequent. There is in nature much variation throughout all seasons, and flow rarely remains constant for more than a few days. Differences in the natural inflow data when compared with modified outflows clearly show significant deviations from expected natural conditions at the Holden Wood site, and little likelihood that modified flows are sufficient to sustain a healthy native ecosystem, both in terms of magnitude and flow regime timings and variation; the physiochemical and ecological regulation provided by these flow features is discussed in further detail below. The clear implication of the HHS results for environmental flow development at our study site is that higher magnitude (within the range of the natural time series) is almost always ecologically beneficial for the species being assessed; the question is how great the ecological benefit is relative to how much water is "spent" in flow modification mitigation. Magnitude plays a number of roles within the riverine system with direct and indirect impacts upon ecology; it is seen by many as the primary influential flow characteristic, particularly when assessing rivers across very large ranges of magnitude (Monk et al., 2006). Magnitude is responsible for many ecologically-relevant mechanics within the river system such as sediment transport and shear stress (Richter et al., 1996, Merigoux and Doledec, 2004), and thus increasing it to more natural levels not only serves as provision for species flow velocity requirements, but also maintains a more natural environment through other means.

Seasonality and flow variability are inherent to natural systems due to predictable precipitation patterns and events such as snow melt (Junk et al., 1989, Junk and Wantzen, 2004a); due to this predictable seasonal aspect and regular variation within natural water bodies, it is of little surprise that riverine taxa have adopted biological adaptations to best take advantage of predictable and varying flow patterns (Poff et al., 1997, Lytle and Poff, 2004). There is thus a need to adopt such variability into any proposed flow regime, though in a somewhat limited state due to economic and functional limitations imposed by the reservoir itself. While it is not possible to change the flow as gradually or as frequently as seen in the natural time series, the recommended flow attempted to implement more flow transitions (compared to the current modified flow regime), and such seasonality and variation may be further supplemented by reservoir spills should periods of high rainfall occur. These proposals are also consistent with findings from the statistical analysis of flow patterns in a range of rivers (Chapter 3), in which flow event frequency and duration were identified

as key drivers of ecological composition and diversity, which in essence tie in closely with flow variability; longer durations imply less variability whilst higher frequency implies more.

5.4.2 Model development and implementation: assumptions and limitations

A number of assumptions are made in order to generate 2D model predictions. For hydraulic predictions, channel hydraulics were assumed to be simplistic enough for depth-averaged velocity to be valid. In more complex river systems, more extensive velocity measurements at multiple depths may be required in order to properly represent river hydraulics. Normal velocity distributions were assumed at the inflow and outflow boundaries; this assumption was valid in this study due to the identification of ideal boundary locations upstream and downstream at the reach. In complex, winding channels other velocity distribution methods may be necessary. For CASiMiR model predictions, it was assumed that species preferences at the site correspond to those found in the STAR project database. Subsequent model predictions, compared with observed sampling data, suggest that this assumption is valid at the study site, although further sampling under a greater range of flow conditions (such as those described under proposed regimes) would be necessary to fully validate this assumption. Overall, predictions of ecological response to flow were as expected and no divergent or anomalous response was encountered.

It has been claimed that 3D models provide more robust predictions, and that the z dimension can be an important aspect of ecological pressure and response (Pisaturo et al., 2017). However, in the case of Pisaturo et al, the study was performed within a much larger river system of significant depth, magnitude, and velocity. The continued success of studies utilising 2D models even in larger river systems (Jowett and Duncan, 2012) leads this investigation to propose that in a smaller-scale system such as Holden Wood, the 2D modelling approach is more desirable, requiring significantly less calibration whilst also holding true to 2D assumptions such as simplistic z-dimension hydraulics. The lesser requirements of the 2D modelling approach also entails easier transferability; a desirable advantage given the aim of this framework to be appropriate in a regional context. Designed flow regimes derived from model results and ecological considerations are based on the assumption that precipitation patterns reflect typical annual precipitation. During particularly wet or dry years, adaptive management should address cases in which proposed flows are not appropriate for current conditions; perhaps flows must be reduced to baseline levels during droughts, or elevated flows must be prolonged during wet periods when the reservoir is near capacity. During such extreme conditions, the expertise of the water managers may adapt the regime accordingly, or flows may be set to pre-defined values based on flow, similarly to 2017 outflows being defined by water level.

5.4.3 Individual species requirements – flow

Species flow requirements tie closely into habitat connectivity and persistence; the latter of which most informed the overall environmental flow regime recommendations. However, the flow requirements of individual species served to highlight the responsiveness of individual species to flow magnitude changes in the modified system, and also showed species response within natural flow ranges. This gave further insights into flow provision for individual species; for example *Gammarus pulex* does not respond significantly to elevated flows, but is sensitive to low flows. One could conclude from this that *Gammarus pulex* does not require high flows; it simply needs to be kept above a certain flow threshold to remain at moderate HHS levels. The primary limitation to identifying

individual flow requirements has been CASiMiR's inability to consider the temporal aspect of flow, such as the frequency or duration of flow events. This has meant that we cannot directly predict species response to certain characteristics of the flow time series outside of magnitude. Although the statistical analysis aspect of this investigation in Chapter 3 has had some success in identifying and quantifying key ecological drivers, we are unable to directly predict species response to temporal events with certainty, and thus cannot refine these aspects of flow to the same extent as we can for spatial and overall river reach habitat quality predictions corresponding to a given value of flow magnitude.

5.4.4 Individual species requirements – timing

Due to time and resource constraints, there are significant limitations in identifying the flow timing requirements of the species present at Holden Wood; Cascade consultants seasonal sampling data showing populations across Spring and Autumn have been the best tool for assessing requirements at the Holden Wood site. Primary field work for this investigation was performed in autumn only, but reflects species distributions found by Cascade in autumn, giving further confidence to the data provided. In the seasons of winter and summer, this investigation has taken a risk-averse approach due to lack of data, maintaining constant levels of flow that stay above thresholds particular to each season. Winter allows for the lowest base-flow due to the lower levels of biological activity (e.g. macroinvertebrate dormancy) that is typical to the season (Olsson, 1982), whilst in summer a moderate base-flow has been set as biological activity is likely, though a number of species may be in flight. Such flows were decided using a combination of literature (see Section 5.2.4) and expert knowledge.

With increased time and resources in a more extensive site-based investigation, ecological sampling could be performed across all four seasons alongside ecological expertise to determine life history timings (such as the time of year a species takes flight), and flow regime could be tailored to individual species across all four seasons. It could be argued however that the cost of this would outweigh the benefit, given that ecology is not sampled by the Environment Agency outside of spring and autumn (and therefore the legislator focus is upon these seasons), is unlikely to be active in winter, and many species may be on the wing or undergoing other undetectable lifecycle stages for a significant period over summer. Even if most species were present in the river during summer season, economic and water security constraints likely would limit environmental flows over this period, due to issues such as increased water demand and risk of drought.

5.4.5 Habitat heterogeneity

The importance of habitat heterogeneity has been extensively explored by ecologists (Dunbar et al., 2010b, Miller et al., 2010, Feld et al., 2014, Ward et al., 2002, Wiens, 2002). The core premise is that in order to host a diverse range of species, all of whom may have varying flow preferences, the river channel must present a heterogeneous environment for a diverse set of species to populate. Once a profile of habitat diversity vs flow is developed for a river, a critical threshold for diversity may be identified and flow targets may be set. This could potentially be an effective first step in increasing biodiversity in river systems across the UK and beyond. Thus, diversity of habitat has been an integral consideration when assessing the interaction between flow and habitat within Holden Wood, and in the development of an environmental flow regime.

Shannon's Index has been used widely in ecology (Beisel and Moreteau, 1997), including as a metric for habitat diversity. Application to the field of environmental flows, and more specifically reservoir releases, is potentially a novel method of flow evaluation which has not been applied elsewhere to as far as I am aware. The habitat diversity vs flow relationship shown in section 5.2.3 does not directly apply to all rivers; indeed each river will have its own flow-diversity profile based upon channel geometry. In the case of Holden Wood, diversity has peaks at $0.1m^3$/sec, and is followed by a plateau in diversity as the flow velocity range narrows (lower velocities are eliminated at higher flows). River morphology may significantly alter this relationship; for example in wider rivers, flow magnitude would likely need to be much higher to achieve a peak in habitat diversity, and the range of velocities across the channel would likely be greater.

The implications of this are that the method of considering habitat diversity response to flow will need to be considered on a site-by-site basis. This would require flow and habitat modelling, and could allow habitat diversity thresholds to be set across many rivers. It is possible, perhaps even likely, that rivers of a similar magnitude class and geometry would display similar diversity-flow relationships. Confirming this is beyond the scope of this investigation, but if such a thing were the case, approximate diversity flow thresholds could be set for all rivers within a given magnitude and geometry class, derived from a single modelled site, following validation of this.

5.4.6 Habitat distribution, connectivity, and persistence

The importance of connectivity and habitat persistence have long been discussed as a key ecological issue in river systems (Junk et al., 1989, Poff et al., 1997). As such, these too have been a key object of investigation and an important facet in designing Holden Wood environmental flows. The spatial outputs CASiMiR provides for the assessment of habitat connectivity and persistence, demonstrated in Section 5.2.5, have been highly informative in identifying sensitive species within the system; complimenting HHS graphs and providing further detail. For instance, the species *Hydropsyche siltalai* is shown in the Section 5.3 to be sensitive to low flows in the modified flow regime, but has a significant HHS increase during periods of elevated flow. Some explanation of this is provided by the spatial habitat quality results for the species found in Section 5.2.5; we see that reduced flow leads to increasingly lower flow velocities beginning at the edges of the river channel. Beyond a certain threshold, connectivity downstream is completely lost and much of the observed river reach is inhospitable to *Hydropsyche siltalai*. Such detail allows us to set thresholds as we can see from what point connectivity and moderate habitat quality is lost; this is 0.02 m^3/s in the case of *Hydropsyche siltalai*.

As mentioned in Section 5.2.5, this approach is somewhat qualitative and formed only a small part of flow regime design in this thesis (and was not published as part of the paper found in Appendix 1 for this reason). It is possible that this method could be emphasised to a greater extent in flow regime design; this may be particularly useful in more complex river systems in which habitat might be influenced by flow in unexpected ways. Should spatial HSI information be applied to such sites and utilised to communicate information to water managers and other stakeholders, a more quantitative metric might be useful as a final output, in order to produce a result that is both more precise and more rapidly understood, without the need for extensive explanation.

Such metrics could be developed through the utilisation of image assessment software; this would allow images to be broken down in terms of the spatial distributions of habitat provided by CASiMiR. This would allow the characteristics of these distributions to be quantified using a range of metrics. Software for such an approach is already available and it utilised within Ecology; a widely used example of this is FRAGSTATS, a piece of landscape ecology software that has the specific goal of quantifying landscape patterns (Kupfer, 2012). It is believed that such capabilities could be applied to the visual outputs provided by CASiMiR.

The FRAGSTATS documentation (McGarigal, 2015) describes various metrics that may be applied to a landscape; two categories are of particular interest to this thesis. The first are *composition-based* metrics, referring to the variety and abundance of certain patch types (habitat values, in the case of this thesis). Though this category is less directly relevant to connectivity, the metrics of proportional abundance and evenness within this category could be powerful metrics in evaluating the heterogeneity of habitat, the importance of which was discussed in Section 2.6.2, and which this thesis quantified by another method in Section 5.2.3. Proportional abundance is somewhat self-explanatory in that it provides a metric of the abundance for a certain class (i.e. habitat quality) relative to the entire mapped site. Evenness on the other hand is a measure of the heterogeneity or homogeneity of site conditions; an evenness of "1" would represent a perfectly homogeneous habitat across the system, and would approach "0" the less homogeneous conditions became. The second category of metrics are somewhat more complex to measure, but relate much more closely to habitat connectivity. This category is *spatial configuration*, which is concerned with the spatial character and arrangement of classes within the site. Of particular interest to this thesis are the metrics of aggregation and isolation. Aggregation measures the degree to which the values of a particular class (again, habitat quality in this case) tend to clump together, and is concerned with the adjacency of patches of a given class type; this clearly has significant overlap with the concept of connectivity, and a higher level of aggregation would be expected to correlate with the connectivity of a particular habitat. Isolation metrics are in contrast to aggregation, in that they are concerned with the tendency of patches to be relatively isolated, or distant, from similar patch types. Again, such a metric would be expected to relate closely to connectivity, this time in an inverse manner, as isolated patches of habitat are the very definition of a lack of connectivity (McGarigal, 2015).

The metrics described, provided by software such as FRAGSTATS, would provide quantitative indices by which to assess the impacts of flow regime alteration, rather than relying upon the subjective interpretation of visual outputs. This would be highly advantageous, as thresholds for particular metrics might be set in order to inform flow regime recommendations. For instance, if flows at a site were being increased, a water manager may utilise aggregation metrics in order to identify the aggregation level that satisfies the ecological requirements at a given site. If it is desirable for flows to be reduced, the isolation metric may be used; perhaps flows could only be reduced to the point of a set threshold level of isolation for an important habitat type. Where these thresholds would be set is a matter of debate, and would likely require a degree of experimentation and investigation within the ecohydraulics community in order to reach a consensus on this matter, and it is expected that no universal thresholds would be ideal; optimum thresholds would be subject to the class of river and the priorities of water managers and stakeholders at a given site.

5.4.7 Environmental flow design

Flow requirements of indicator species presented in the Methods show that generally, at the ranges of flows studied, there are diminishing returns of predicted habitat quality response to increasing flow at the study site. Beyond 0.07m^3/s a reduction in responsiveness is observed, and beyond 0.09m^3/s HHS is generally beginning to plateau. This implies that magnitude increases, based solely upon species preference curves relating flow to HHS, are not an efficient solution for the ecological improvement of a system at the flow ranges studied at the Ogden Brook site, and becomes increasingly less efficient the longer the flow is maintained. Current impoundment outflows at Holden Wood do not demonstrate a consideration for seasonal variation in productivity and taxon composition; this study has proposed that a criterion-based flow design may target the key ecological timings for a system, and provide less flow at other times such as biologically less active periods (e.g. winter) or periods when stricter water resource conservation is necessary (e.g. summer). Allocating flows in this manner may allow for ecological provision that both improves ecological metrics, and also addresses the conflict between environmental flows and the societal need for water resource conservation. In contrast, uniform increases to flow lead to small improvements in ecological metrics yet disproportionately high flow expenditure, as has been the case between 2014 and 2017 Holden Wood compensation flows. Re-allocation and optimisation of flow releases in designated regimes are able to meet or exceed historical flow HHS values at key times, provide in a more varied flow regime (beneficial in principle but difficult to quantify within the scope of this project), and retains large quantities of water for other uses.

This retention of large volumes of water, whilst causing little change in terms of average HHS (as seen in Table 5.6 in Section 5.3), might raise questions as to the sensitivity of the HHS metric. Although HHS changes little in terms of annual averages, with changes of 0.06, 0.01, and 0.02 from 2017 flows, to Regime A, to B, to C, respectively, it can be observed in results such as Figure 5.12 that changes on short timescales are far more significant, for example with a HHS change of 0.2 during autumn and spring maxima and minima flows in Regime A. This demonstrates that HHS is indeed responding to flow changes. Little change in annual average HHS, despite a large reduction in flow, is due to the more adaptive flow designation implemented within Regimes A, B, and C; flow is utilised in such a way so as to maximise HHS during key periods, with the trade-off of reducing HHS during less vital periods. As mentioned previously in this discussion, maintaining moderate levels of HHS throughout the year (as seen in 2017 outflows) requires a large quantity of flow. In contrast, elevating HHS to high levels for key periods, with fluctuations in flow for the sake of flow heterogeneity, and allowing HHS to drop to lower levels at less important times of year, requires much less flow whilst not having a significantly detrimental impact on the annual average HHS.

The above said, there are some sources of uncertainty to be aware of when utilising HHS, and these primarily stem from the assumptions inherent to the metrics from which it is derived; HSI and WUA. The assumptions and limitations of HSI are discussed in Section 4.7.2. WUA shares the assumptions and limitations of HSI, but also assumes that the greater weighting of higher HSI values, multiplied the extent of wetted area each HSI covers (as seen in the equation in Section 5.2.3), provides a metric that is proportional to the abundance of the target biota. This assumption has been found to be valid in other studies such as Kelly et al., 2014, although Kelly et al. (2014) did find that WUA did not have a 1:1 proportion with biota abundance; it tended to underestimate biota response to flow. This is likely due to the limitations of the habitat suitability methods commonly used, as was discussed in Section

4.7.2 regarding HSI. The HSI metric, which directly influences WUA, was found to be proportional with target biota abundance, as seen in Section 4.6.2. However, lacking the opportunity to validate predicted changes in HSI in response to flow with actual changes in biota abundances, this thesis cannot directly attest to the sensitivity of HSI under changing flow conditions.

HHS itself is simply a normalisation of WUA. As such, the conversion from WUA to HHS removes the influence of the river's wetted area. This is generally seen as advantageous (Schneider et al., 2010), as the added factor of wetted area can lead to inconsistent index values; for example, a small but high quality area of habitat received the same value as a larger but poorer quality habitat. The one assumption of HHS this study makes in addition to those already present for WUA, therefore, is the assumption that changes in wetted area across a range of flows are not exerting a significant influence upon the ecosystem that is distinct from other changes in habitat quality. This is seen as valid in the system studied; extreme changes in wetted area were not observed (for instance, the river almost drying up), and therefore ecological influence is expected to be primarily derived from changes in flow velocity. Cases in which the assumption might be said to be less valid could include instances where a river overflows its banks, or where flow is greatly reduced, resulting in significant changes to the wetted area. It is also possible that the extent of wetted area may be useful information in large rivers, where wetted area may vary greatly depending upon flow. I believe that HHS is a valid metric within small to medium scale river systems, and further, due to normalisation, HHS can be directly compared between systems (Garbe and Beevers, 2017). Given one of the goals of this thesis being to promote transferable methodologies, HHS holds a significant advantage over WUA in this regard.

The homogeneity of steady regimes not only requires disproportionate volumes of water relative to the HHS achieved, but also reduces the range of flow (and thus habitat) conditions at a site. Section 5.3 demonstrates this; 2017 outflows result in peak HHS values most similar to Regime C, despite releasing more than double the quantity of water throughout the year. Again, this supports the premise that such flows may release a great deal of water, yet do not address important ecological requirements. Variation in flow and more naturalised high pulses influence the physicochemical properties of the riverine system such as the sediment and thermal regimes, nutrient content, and water pH, and such flows may influence species populations by preventing the dominance of single-flow specialists (Petts and Gurnell, 2005, Richter et al., 1996). Frequent periods of elevated flow also generate greater diversity of habitat in areas of previously homogeneous low flows. As greater habitat diversity facilitates greater biodiversity (Ward et al., 2002), flows throughout spring and autumn periods in designated regimes would in principle be expected to improve biodiversity metrics, assuming the periods of low flow between intermediate and maxima do not remove established biota. High flow pulses also aid in river connectivity, transferring nutrients between the main channel and periodically wetted areas (Junk et al., 1989, Junk and Wantzen, 2004), as well as varying connectivity between different river sections that may be separated by barriers such as weirs (Shaw et al., 2016). Lacking such mechanisms, it is unlikely that the functional composition or level of biodiversity within current modified systems will resemble that of their natural counterparts (Gillespie et al., 2015a, Poff et al., 1997).

Results suggest that preference curves alone give a limited view on the impact of flow upon ecological response. Considering the impact of flow magnitude based upon the flow requirements of individual species is only one aspect of ecological health within a complex interacting system; habitat heterogeneity and temporal flow variation are key factors which may be overlooked when considering

physical habitat suitability based upon flow magnitude alone. Increasing flow magnitude does not always result in ecological improvement; its benefits may plateau or increases may become detrimental to some species at higher ranges. Flow event durations and frequencies may play a key ecological role, creating more temporally heterogeneous environment where a single species cannot dominate (Levin, 2000), driving sediment transport mechanics and their associated impacts (Kondolf, 1997), driving connectivity of the river with the surrounding flood plain (Junk and Wantzen, 2004). In an attempt to better naturalise the flow in proposed regimes, this study adopted flow characteristics from regional rivers of similar magnitude. While approaches such as Before/After, Control/Impact utilise site-specific reference sites, this study is not aware of typical regional conditions being utilised as a more general indicator of natural conditions, and we suggest that this approach may be integrated into environmental flow assessment methods in cases where systems are relatively similar.

Whilst raw flow magnitude has a very substantial influence upon benthic ecology, the temporal aspects of flow such as frequency and duration of events, based upon local natural trends, should in principle provide ecological provision better suited to ecological requirements. Systems with homogeneous flows have demonstrated decreased biodiversity (Wiens, 2002), and it is unlikely that flow magnitude divorced from natural conditions can ensure a healthy ecosystem capable of meeting ecological targets (Acreman et al., 2014). A key challenge to the implementation of environmental flows has been the increased labour such flow designs would entail. A transferable framework based upon general regional principles, such as that proposed in this study, could help to alleviate some of these labour requirements by allowing environmental flows to be designated efficiently across numerous small-scale sites with minimal adaptation between them; sites which may otherwise be unfeasible for restoration on a specific case by case basis. The similarity of natural river system behaviour observed in the North West of England lend support towards this possible transferability, though further research and flow experimentation would be necessary to confirm this with confidence. Initial work may be required to validate the transferability of certain aspects of flow regime design, such as the relationship between flow and habitat diversity discussed in Section 5.4.5, but this study stipulates that if transferability between similar systems can be validated, the efficiency of restoration measures can be significantly increased by dealing with systems on a class-by-class basis, as opposed to site-by-site.

5.4.8 External variables

This investigation has briefly detailed some of the primary external, ecologically-influential variables expected to be encountered within the riverine system in Section 5.2.6. Whilst it is outside of the scope of this investigation to quantify or predict how these variables may alter ecological response, it is acknowledged that the influence of these variables is a potential source of uncertainty, and we have outlined the ways in which these variables may impact the ecosystem. This investigation acknowledges that environmental flows are one aspect of influence upon the riverine ecosystem; other issues such as sediment transport (an issue associated with reservoir impoundments) must also be properly assessed and mitigation measures decided within a holistic context. Such measures must be implemented (or disregarded due to impracticality) if a system is to meet the legislatory target of Good Ecological Potential.

There is a limit to what can be asserted about external variables, however, due to the complex interplay of all variables within the river. We have discussed already the challenges presented in

riverine study due to the open systems these waterbodies present to us. External variables (and indeed hydrological ones) all influence and feed into one another. Hydrological events alter sediment transport and storage, which in turn may alter river morphology (among other things), leading to further changes to flow, which again influences sediment, and so on. Such feedback may magnify or counteract the original causal event, leading to complexities and nuances far outside the scope of this project. Thus, we must rely on general principles accepted by ecologists and hydrologists regarding such variables (e.g. McCabe, 2010), and, as described in Section 5.2.6, attempt to minimise the influence of external variables in this stage of the research in order to construct the foundations of an approach to environmental flows that can later be built upon and adapted to address further complexity. This could be accomplished through the tools made available in CASiMiR, such as the ability to consider species preferences of sediment, or the use of fuzzy logic, as described in further detail in Section 6.2.2. The extensive quantification of such variables and their interactions with flow and ecology would require significant investigation in and of itself, and could be a subject of further research.

5.4.9 Outcomes and Implications

The framework by which an overall environmental flow recommendation using predictive modelling has been developed for the case study site, Holden Wood, has been discussed. The outcome from this framework is a flow regime that recommends flow events at a given duration and frequency over two seasons of the year, spring and autumn, whilst setting low flow thresholds across the other two seasons, winter and summer. I have discussed the rationale behind these flow recommendations and the limits inherent to our assessment methods. There are also limitations in the application and transferability of the overall framework which must be considered, along with potential options for the mitigation of such limitations. This work has demonstrated the feasibility of 2D predictive modelling of ecohydraulics in developing solutions to the anthropogenic impact of flow modification, and meeting new environmental legislation as the understanding of the relationship between flows and ecology continues to develop. This investigation presents the initial steps in such an endeavour, and does not presume to be a complete solution. As has been mentioned, meeting GEP is a holistic undertaking that will require the assessment of all potential impacts brought about by water body modification. However, studies such as this bring us ever closer to meeting such targets, demonstrating to legislators that water utilities are in the process of considering mitigation measures, and also building up firm basis for continued investigation and resources to be directed towards this manner of investigatory framework.

A primary limitation of this framework is that it is not a "stand-alone" mitigation solution; it must be integrated into an overall holistic water management framework. This limitation will be partially mitigated should future work expand upon the use of predictive modelling for the development of environmental flows; with more time and resources it will be possible to integrate sediment transport, shade, local geology and other variables into an ecological model. This would come at the cost of increased data requirements and computational power demand, however. "Over-engineering" a model may hinder a model's ability to predict outside of observed conditions, thus there is a balance to be struck between model complexity and feasibility, and other solutions alongside modelling may be required. Data demonstrates that average HHS changes little, likely because a majority of the time series is similar (summer / winter), but there is significant change in the range of HHS values. Peak HHS increases significantly over Target values in all three suggested regimes. This is significant as peak

HHS represents four weeks of the year, with eight further weeks of the year being at elevated but not peak HHS, and spring/autumn seasons are highly ecologically important.

5.4.10 Cost analysis

Implementation of the flow regimes recommended in this investigation would have their associated cost or savings primarily based in loss or gain of water yield for use by United Utilities, relative to current operations. Capital-associated costs should not be a factor, as no further engineering development should be required for these flow ranges. Costs of water yield for environmental solutions, assuming the HMWB services are not compromised, are significantly cheaper than engineering capital expense in the short term, and in many cases where historical compensation flows have been set, savings may be made through flow release reductions. Below in tables 5.8 and 5.9, change in yield is costed and compared with the capital-associated cost of an engineering solution, a fish pass installation, to put these price magnitudes into relation with other water management operations. Difference in flow from designer regimes is calculated by the 2017 overall annual outflow from Holden Wood, 1,180,460m^3.

Table 5.8: Approximate water yield, based on industrial standard cost per megalitre (ML)

Flow Regime	Reduction in Flow (m^3)	Water Cost per ML (1,000m^3) (£)	Annual Gain in Yield (£)	Gain over AMP cycle (£)
A (924,480m³)	255,980	1,000,000 (Industrial standard price used in most utility cost-benefit analysis, based on cost of infrastructure development to transport water to required area in the event of water shortage)	255,980,000	1,279,900,000
B (721,440m³)	459,020		459,020,000	2,295,100,000
C (565,488m³)	614,972		614,972,000	3,074,860,000

Table 5.9: Approximate costs for fish pass installation, taken from a UU Cost Benefit Analysis by Grontmij (2013)

Fish pass type		Embankment Height (m)		
		5	10	15
Alaska	Project Cost	£ 6,193,479	£ 7,510,634	£ 11,128,399
	Cost/m	*£ 1,238,696*	*£ 751,063*	*£ 741,893*
Larinier	Project Cost	£ 6,112,861	£ 7,457,347	£ 10,638,230
	Cost/m	*£ 1,222,572*	*£ 745,735*	*£ 709,215*
Pool and Traverse	Project Cost	£ 6,479,127	£ 9,417,739	£ 13,567,758
	Cost/m	*£ 1,295,825*	*£ 941,774*	*£ 904,517*
Fish Lock	Project Cost	£ 7,431,521	£ 11,409,155	£ 15,268,921
	Cost/m	*£ 1,486,304*	*£ 1,140,915*	*£ 1,017,928*

It must be noted that the figures in Table 5.8 correspond to industrial-standard pricing of £1 million per megalitre of water, which considers the cost of contingency infrastructure for water transport in the event of a water shortage (for example pumping the water from elsewhere); these yields therefore do not translate directly into profit; actual cost of water in the UK is £1.50-3.00/m^3, depending on household or industrial use. However, these figures can be interpreted as potential savings due to increased water security, avoiding the need for extremely costly contingencies in an increasingly water-stressed country that could be expected to see water shortages within 25 years under current water management practices (BBC, 2019). It is therefore reasonable to base water costs in these terms. As can be seen in this comparison, significant savings may be accumulated over a number of years, and within one or two 5-year Asset Management Plan (AMP) cycles sufficient savings could be made to invest in infrastructural needs within the region. Operating costs for engineering would also add to the cost over time of engineering works. These results suggest that addressing flow modification through environmental flow regimes is an efficient solution to flow impoundment ecological pressures which could result in financial gain in the long-term, particularly when considering the consequences of unsustainable water management practices. The difficulty, however, lies in developing such regimes with the current available knowledge within the field. As understanding and methods continue to develop in this field, there may be a proliferation of environmental flow investigations at modified sites, both due to the need to meet legislative targets, and due to financial incentive.

5.4.11 Further Implications

This Chapter demonstrates the use of 2D hydro-ecological modelling and its potential, particularly for small-scale sites where vertical complexity is minimal and an efficient approach is necessary due to resource constraints. A significant outcome from this study has been the demonstrated potential for significant quantities of water to be conserved through designer regimes, whilst anticipated ecological response should be improved, both due to criteria-based flow allocation and greater naturalisation of the regime. Regimes A, B and C promote ecological provision, with varying prioritisations. This demonstrates the utility of this approach; it is possible to define design criteria, which may be adapted to accommodate water demands and diverse interests of stakeholders present within a given system. Should this method be validated by physical flow experiments, it is believed that such flows could be applied regionally to similar river systems with minimal field investigation requirements. Such transferability may allow for smaller scale impoundments across the UK to implement environmental flows, where previously this was unfeasible due to the quantity of impoundment systems and the intensity of labour required to assign environmental flows to individual sites.

This framework currently focuses upon application for sites impacted by impounding reservoirs; it could be possible to adapt it for use in other site restoration assessments such as hydropower-impacted sites by incorporating the unique challenges and priorities of the given modification into the design stage of the environmental flow regime. An example of such a consideration would be the necessity of disruptive high flows from hydropower releases; perhaps a regime design for such an application may focus upon dampening and prolonging these high flows according to what is feasible without compromising the service of the dam. It is also acknowledged that flow is not the sole driver of ecological response, and other stressors such as water chemistry likely play a significant role at

many sites. It has been suggested that the diverse influences of riverine ecology must be studied both through short-term mechanistic experiments and long-term explanatory studies in order to more fully understand this complex web of interactions (Laini et al., 2018). Changes to climate and land use are also driving further changes in ecological metrics (Li et al., 2018). As understanding of these interactions grows, it would be possible to integrate further mitigation methods into the framework presented in this study. The ability to integrate ecological requirements according to context, and make adjustments according to new knowledge, offers significant utility within this framework.

5.5 Conclusions

This study has presented a methodology by which a study site is assessed and environmental flows are proposed based upon a combination of species response to flow (through preference curves), the influence of magnitude upon habitat diversity, and typical unregulated regional flow characteristics in order to form a holistic ecological solution. Results suggest that uniform increases in magnitude over long periods result in disproportionately little ecological benefit relative to volume of water released, and affirms the use of optimised and targeted high flow events. Though there is a rich literature detailing the concepts considered, I am not aware of any studies suggesting a similar approach by which such a combined range of flow requirements within a particular site, and potentially region with further development, may be assessed. Poff et al. (2017)'s update on the evolution of environmental flow science discusses progress in almost every area, but there is not yet a unified approach to environmental flow assessment. They emphasise the need to extend from a local scale to basin-scale perspective (Poff et al., 2017). Previous regional frameworks have been attempted, but generally have used singular metrics of habitat suitability (Ceola and Pugliese, 2014). More recently, a framework for the strategic allocation of water in order to balance environmental flows and societal needs has been proposed (Sabzi et al., 2019), but within the scope of generalised "environmental considerations" which must be determined on a case by case basis.

Amidst this rapidly developing field in which numerous frameworks and methods for environmental flow assessment are emerging, this study presents a novel approach towards efficient annual flow regime designation by bringing together flow and habitat suitability modelling tools, in combination with trait-based analysis, in order to assess the potential ecological effects of varying flow regimes within a river reach. The approach has scope for regional transferability for sites of similar scales due to their anticipated similarity of response, though this would first require validation through further research. Many previous studies have focused upon specific ecological response from a given study site, or have focused upon the conceptual development of particular aspects of ecological response; this study has attempted to draw together knowledge accumulated through recent academic progress, and combines this with the considerations of practical constraints and stakeholder interests faced by the water industry. This therefore offers the first steps towards a potential regional water management solution addressing the issue of impoundment-modified flow impacts considering both ecological wellbeing and water resources conservation. There is scope for this framework to be scaled up to larger river systems, though this would require the incorporation of other variables significant at such a scale, such as substrate type and variation and the sediment transport regime, in addition to

greatly increased topographic data requirements. Fish may also be considered in CASiMiR should they be present in the system. This investigation suggests that 2D habitat modelling remains a tool with great potential when incorporated into such holistic practices, and shows great promise as water managers move into transferable, regional-focused forms of investigation.

6. GENERAL DISCUSSION

6.1 Introduction

Chapters 3, 4 and 5 have described two complementary approaches by which the relationship between flow and ecology, in the context of flow modification through impoundment of river systems, was investigated. Firstly, as described in Chapter 3, desk-based analysis across multiple sites was utilised in order to identify potential regional, general principles based upon flow characteristics (Richter et al., 1996) and ecological metrics (Extence et al., 1999, Petchey and Gaston, 2006). Findings from this analysis informed the second approach, in which this investigation explored the use of a methodology by which environmental flow regimes for a site may be designated through a combination of 2D hydraulic-ecological linked modelling and existing ecological principles, with development described in Chapter 4 and application described in Chapter 5. Chapter 3 informed and supplemented the environmental flow regime development in Chapter 5 through the identification of temporal ecology-flow dynamics that are not described through the modelling approach alone. A key aim of this thesis was to provide further understanding and solutions for environmental flows with utility and transferability; the methodology was tested at a case study site but demonstrates significant potential for broader application in a regional context within rivers of a similar scale, with the possible potential for scaling up. This Chapter will discuss the key findings of this investigation, how findings compare with existing literature within the field, implications of the research, and how the research might see broader application in a future where both water resources and ecological systems are increasingly stressed (European Commission, 2000, BBC, 2019).

6.2 Chapter Discussion

6.2.1 Discussion of findings for Chapter 3

Chapter 3 aimed to answer research questions 1 and 2 initially proposed in Chapter 1:

1: How can ecology-flow interactions be better understood in an efficient and transferable manner?

2: What specific flow drivers are eliciting an ecological response?

A desk-based analytical approach by which ecology-flow relationships might be derived has previously been utilised in literature (Gillespie et al., 2015b), though results from such studies have so far have been inconclusive. Studies have called for more research to be performed, with a focus upon transferability, in order to overcome the challenges associated with riverine investigations (discussed in Chapter 2) and contribute towards the overall knowledgebase and thus aid future meta-analyses (Poff and Zimmerman, 2010, Konrad et al., 2011, Gillespie et al., 2015b). Chapter 3 assessed river systems within a particular range of conditions; Monk (2006) proposes that magnitude is a dominant variable that masks the influence of other flow characteristics if site comparisons vary over too great a range of magnitudes. Due to the role of magnitude as the primary driver of riverine morphology, and regulator of other systems such as the sediment regime (Kondolf, 1997), the investigation in Chapter 3 explored sites within a defined range of magnitudes (0.5-4.3m^3/sec). Within this range, the "masking" of the magnitude variable was minimised, and therefore the influence of other drivers such as the temporal characteristics of flow could be assessed. The investigation also focused upon a

particular geographic region, Northern England, in order to ensure that sites shared a degree of similarity in terms of climate and geological conditions, allowing for a higher degree of comparability between sites (Arthington et al., 2006, Monk et al., 2006) by "controlling" as many variables as possible and thus reducing the masking of significant ecology-flow relationships. That said, this study acknowledges that some differences between sites do exist between sites across northern England, and should one desire even greater similarity between sites a more selective process would need to be undertaken during site selection. This may, however, greatly limit the amount of data available for analysis. Rivers with significant water quality issues were also discounted from the investigation in order to limit external influences upon the ecosystems being assessed. Constraining the investigation through lessons previously learned in literature has resulted in an approach that can be applied to a particular region and magnitude range to generate results capable of being compared within a wider knowledge base. It is therefore proposed that Chapter 3 provides an answer for research question 1; desk-based and data-driven investigation within a framework constructed with a good scientific basis is capable of analysing ecology-flow interactions in an efficient (not overly labour-intensive, yet with reliable results) and transferable manner. Results from Chapter 3 are a demonstration of this, and in turn help to answer research question 2.

In the Chapter, statistically significant flow drivers were observed to be influential both upon functional composition and biological diversity. These relationships varied between seasons. The two influential flow characteristics identified were the annual frequencies of high and low flow events, these being the upper 25th percentile and lower 25th percentile of annual flow respectively. Few other studies have identified statistically significant relationships between flow and macroinvertebrates; partly due to macroinvertebrates being less commonly investigated in such studies relative to other indicators such as fish (Gillespie et al., 2015b), and partly due to the methodology for such investigation still being developed, leading to meta analyses that have attempted to draw such general relationships being unable to do so due to difficulties in transferability between methodologies (Poff and Zimmerman, 2010). Because of this, there are few studies to contrast the findings in Chapter 3 with; Gillespie et al.'s (2015b) review of 76 studies comparing ecological response and flow revealed that the majority of studies (55) focused upon magnitude-based modification, whilst the remainder focused upon the cumulative impacts of flow modification; the literature review reports no typical ecological responses to a specific flow variable, other than magnitude. This thesis identified a potential significant relationship between macroinvertebrate response and flow event frequency; section 3.4.2 of Chapter 3 discusses the significance of high and low flow events and the possible mechanics by which these events drive ecological response.

As mentioned, the possible reason for previous studies failing to observe significant ecological responses could be due to the masking of individual variables due to the dominance of flow magnitude, or due to methodologies not yet implementing the regional and class-based analytical approach that has been suggested by studies such as Arthington (2006) and Monk (2006). Some studies have utilised other approaches, such as investigating the impact of sudden flow increases in the context of hydropeaking (Cereghino et al., 2004) in terms of taxon drift; such investigations do not identify specific ecology-flow relationships, but provide useful water management information within their particular context. A majority of studies over the last decade are site-specific case studies involving flow experimentation (Gillespie et al., 2015b). Studies that have utilised desk-based approaches such as IHA often demonstrate the presence of significant flow modification, but do not encompass quantitative ecological response within the scope of the study (Zhang et al., 2014); this

demonstrates the relative infancy of this field that is still in the process of methodological development. Chapter 3 therefore presents a potentially novel answer to research question 2, "What specific flow drivers are eliciting an ecological response?" though it is by no means a comprehensive one. The observations were made within the context of sites in Northern England, within the particular magnitude range utilised, and without the presence of significant external pressures such as water quality issues. However, the findings in Chapter 3 are likely to apply to a broader range of systems, given the fact that species utilised within the analysis are prolific across many riverine systems. The findings therefore represent a significant step forwards in understanding ecology-flow relationships, and further study might test the relationships identified in this thesis within other river systems. It is hoped that future studies may utilise the framework proposed (or an adaptation of it) in order to continue to generate regional ecology-flow relationship data that may in turn inform broader meta-analyses and environmental flow implementation.

The methods utilised in Chapter 3 could be utilised by water managers, regulators, and utility companies to identify key flow drivers within a given region. This would enable an assessment of the appropriateness of current activities resulting in flow modification by suggesting how likely flow alteration is impacting native downstream riverine ecosystems; outputs from such analyses may also inform water managers how current activity might be altered in order to generate more ecologically beneficial environmental flows. As demonstrated by this investigation, such desk-based analysis may form a foundation for more focused study such as a modelling approach within a river system. Upscaling the approach of this investigation would entail a greater time and possibly resource investment in order to obtain a greater quantity of data; the approach itself would largely remain the same. There would be limitations to the extent the approach could be scaled, however, due to the necessity of constraining data within particular criteria (for example, magnitude ranges or geographic area), and due to the limited amount of data of this nature available (as discussed in Chapter 3, synchronised flow and ecological data is not common). Therefore investigations would be limited to increasing the number of sites within a given region (depending upon data availability), or expanding the criteria or geographical area assessed, though this may compromise the robustness of results. Alternatively, studies with significant time and resources could perform an analysis across multiple regions simultaneously. For example, while Chapter 3 only assessed Northern England, a larger assessment might analyse Scotland, Northern England, Wales, and Southern England as individual regions, and then perform a meta-analysis between these regions.

In addition to answering research questions 1 and 2, Chapter 3 also informed approaches used in Chapters 4 and 5. Within the flow range studied, Chapter 3 implies that temporal flow variables are as important, perhaps even more important, than flow magnitude. Due to the statistically significant influence of temporal flow drivers suggested that habitat quality modelling alone (based upon magnitude at a given point in time) would not sufficiently predict the ecological impact of altering the hydraulic conditions at the Ogden Brook study site, and therefore the flow regime design process was expanded to encompass temporally-related flow characteristics.

6.2.2 Discussion of findings for Chapter 4

Chapter 4 aimed to answer research questions 1 and 3 initially proposed in Chapter 1:

1. How can ecology-flow interactions be better understood in an efficient and transferable manner?

3. How can localised hydraulic requirements of taxa be translated into an ecologically beneficial flow regime?

In this Chapter both a hydraulic and ecological model were developed, calibrated, and the outputs were tested. They were linked together in order to predict ecological response to varying flow conditions in the form of the habitat suitability metrics HSI and HHS, as described in Chapter 4. Habitat suitability modelling has been using previously within modified systems (Doledec et al., 2007, Merigoux and Doledec, 2004, Hatten and Parsley, 2009), but these studies tend to focus upon detecting particular responses from biota, or identifying the most influential hydraulic parameters within the model, whilst acting as a foundation for future development in water management approaches. This Chapter focuses more upon the potential of modelling directly within a water management context, working towards the optimisation of flows to both meet ecological objectives and conserve as much water as possible within the impoundment and in turn directly informing proposed flow regimes for a case study site. The merits of this approach for such a use is discussed, and the calibration and process is demonstrated. Chapter 4 acts as a foundation for Chapter 5 to expand upon in terms of model outputs, as it presents the study site and demonstrates the reliability, as well as the limitations, of the model developed.

The Chapter offers another answer for research question 1, presenting the modelling approach as a potential analytical tool by which the relationship between ecology-flow interactions may be better understood. This approach has been utilised previously, for example in in a site-specific context for hydropeaking (Pisaturo et al., 2017), but has yet to be proposed for wider application as an alternative to typical approaches such as BBM (described in Chapter 2). The modelling approach potentially has significant advantages over the workshop-driven BBM approach, particularly in the context of smaller scale systems where the human resource requirements of BBM are likely to be unfeasible and overly site-specific; an approach that can be performed by a single individual or small team, with a degree of transferability to potentially be applied rapidly to similar systems in the local region, is possibly a more appropriate solution when addressing environmental concerns at this scale. Additionally, moving towards a more criteria-driven and quantified approach towards environmental flow designation through specific metrics such as HHS would allow for the development of implementable principles across sites that might be actioned by water managers without the need for expert opinion, which may not be readily available and is not as readily justifiable (e.g. through definite results such as habitat suitability responses). Following studies that have emphasised the need for transferability and regional-based analysis (Gonzalez and Garcia, 2006, Arthington et al., 2006, Alcazar and Palau, 2010), this investigation proposes that frameworks such as the modelling-based form of assessment utilised in Chapter 4 may become increasingly desirable as an option to assess multiple sites in a resource-efficient manner, both due to pressures imposed by legislation (European Commission, 2000), and due to methodological and computational improvements within the field (Schneider et al., 2016).

Chapter 4 also partially answers research question 3, translating the hydraulic requirements of taxa at a local scale into metrics that may be interpreted to inform an overall flow regime in terms of flow. Chapter 5 expands upon this by describing how flow might vary over time within an annual regime, discussed later. Chapter 4 discusses how the flow requirements of taxa, in the form of preference curves, are inputted into the ecological model CASiMiR, these preferences interact with the hydraulics for a given flow input, predicted by SRH-2D, which generate habitat suitabilities in the form of the HSI metric (Oldham et al., 2000). HSI can be spatially assessed through the interface provided by CASiMiR,

or can be reduced to a more general site-wide indicator of ecological response through the further HSI-derived metrics the software provides; WUA or HHS. This investigation utilised HHS as a dimensionless metric by which the entire modelled site can be assigned a quantified value of habitat quality. WUA and HHS have been used in other studies such as (Kelly et al., 2015) in order to assess ecological response to changing flow conditions. In this investigation, HHS is proposed as a potential answer to research question 3, accounting for the hydraulic requirements of taxa either individually or on average. The robustness of HSI is demonstrated in the outputs of the CASiMiR model, where an agreement between HSI and observed population numbers at the site is demonstrated. Therefore, the HSI-derived metric of HHS is expected to be an effective tool for site assessment, though spatial habitat distributions may also require analysis to ensure desirable habitat distributions (e.g. ensuring HSI is not skewed by a relatively small but very high quality area). CASiMiR provides tools to make this assessment rapidly, so this requirement is not a significant issue.

This investigation suggests that site-wide ecological metrics such as WUA or HHS could have the potential to be used effectively for the restorative assessment of sites, when considered alongside spatial outputs. Such metrics allow values of flow to be related to a single-value variable (e.g. HHS) and therefore theoretical optimum values of flow can be proposed within a range of flow values. Such an approach may forgo the need for time- and labour-intensive in-stream flow experimentation, or the necessity of expert opinion across various disciplines. However, as discussed, flow values alone are not the only drivers of ecological response; ecosystems are also influenced by the variability of flow with time. Consequently, predicted HHS-flow relationships may be used to propose theoretical flow thresholds such as maxima and minima, as discussed in Chapter 5, but cannot by themselves construct the entire regime. While the use of metrics such as HHS is proposed as an answer to research question 3, such metrics are to be used alongside other criteria such as temporal requirements in order to create a holistic framework for flow regime designation. That said, magnitude of flow is certainly a principal driver of ecological response and is the primary variable under the direct control of site operators; an appropriate magnitude of flow acts as the foundation for a suitable habitat, which they then be adjusted based upon temporal variation inherent to natural systems.

The findings in Chapter 4 primarily demonstrate the potential of hydraulic-ecological linked modelling at the chosen case study site. Although there is some uncertainty inherent to predictions of channel hydraulics (Acreman et al., 2014), and a degree of error associated with depth-averaged velocity, utilised in this investigation. HSI predictions were found to be in generally good agreement with observed species populations taken from field kick samples, with weakening relationships at higher HSI values due to external drivers such as biological interactions (discussed more fully in Chapter 4). This could be used as evidence to promote the use of such a method in other studies operating within river systems of a similar magnitude class, partly fulfilling an aim of this investigation which was to offer a transferable approach to environmental flow designation particularly for smaller-scale yet numerous flow-modified systems.

Scaling this methodology up to assess higher magnitude class river systems would very likely require adaptation of the approach, which may entail a greater investment of time and resources. In-field velocity measurements would require multiple readings across the depth at each point due to the more influential and variable component of vertical velocity in water bodies of greater depth, therefore providing more robust depth-averaged velocity calculations. Larger systems may host a greater variety of biota, and therefore the type of indicator species selected must be considered; fish

may be present and act as an important aspect of the ecosystem (Cheimonopoulou et al., 2011, Harris, 1995), or macrophytes might be used for analysis as in other studies (Onaindia et al., 2005). CASiMiR allows for the assessment of fish, and a specialised package for CASiMiR, CASiMiR-vegetation (Benjankar et al., 2012), allows for the assessment of macrophytes. Additionally, larger systems will likely present a more heterogeneous environment; ecologically important characteristics such as sediment and vegetation may vary along larger reaches and therefore be an active influence within the system (ASCE, 1992, Shucksmith et al., 2010). Such features must be properly surveyed in the field, and can then be incorporated into the CASiMiR model, which has features to predict the influence of both vegetation and substrate type. In order to account for these additional channel characteristics, the user must obtain preference curves or set fuzzy rules for each variable; e.g. vegetation type or sediment type. The capability of CASiMiR to account for further variables such as depth, substrate, in-stream cover, and shade, in addition to the software's ability to utilise FST values and fuzzy logic (Schneider et al., 2016), give CASiMiR a great deal of utility to meet the complexities presented in larger or less typical systems; this is a distinct advantage when compared to modelling software such as PHABSIM, which is limited to the use of velocity, depth, substrate, and cover. CASiMiR is also capable of integrating bed structure variables to predict morphodynamics such as erosion or sedimentation (SJE, 2019); a potentially important influence upon ecosystems, that is not accounted for in many other habitat suitability models (Almeida and Rodriguez, 2009).

Whilst it has been argued that 3D models may be more appropriate within larger river systems (Pisaturo et al., 2017), the computational demand of such models, particularly within the context of longer river reaches, makes the feasibility of 3D modelling questionable in applications where time and resources are limited. Pisaturo et al. (2017)'s study reach was 150m in length; detailed habitat suitability on a larger scale, such as a 1km reach, would likely have unfeasibly high computational demands at this point in time. As discussed in Chapter 4, the 2D modelling approach is less computationally demanding and thus is capable of processing larger stretches of river within a practical timeframe, is easier to calibrate and validate due to the lesser complexity of the model, and offers greater transferability between similar river systems due to less adaptations between systems being required.

In addition to contributing towards research questions 1 and 3, Chapter 4 formed the basis of subsequent investigation in Chapter 5 through the development of the hydraulic and ecological models for the case study site. Results suggest that there is a good basis for utilising such models within the context of smaller-scale systems, and there may be potential for the approach to be scaled up (though such a claim would require calibration and testing within a larger river system). Application of the model at the Ogden Brook site demonstrated that habitat quality predictions relative to flow were acceptably accurate, and therefore gave validity to subsequent findings in Chapter 5, discussed below.

6.2.3 Discussion of findings for Chapter 5

Chapter 5 aimed to answer research questions 1, 3, 4, and 5 initially proposed in Chapter 1:

1. How can ecology-flow interactions be better understood in an efficient and transferable manner?

3. How can localised hydraulic requirements of taxa be translated into an ecologically beneficial flow regime?

4. What indicators can be used to interpret habitat quality, given its variation over time and different values between species?

In this Chapter, the model developed in Chapter 4 was utilised to inform environmental flow designations for the case study site Ogden Brook. The individual requirements of species were considered, with particular regard to sensitive species *Hydropsyche siltalai* and *Gammarus Pulex*; these species helped to inform baseline flow thresholds to ensure that sensitive species within the system were not below critical habitat quality values. Additionally, the spatial effect of flow upon habitat was analysed through CASiMiR's output data, and Shannon's Diversity was used to assess the heterogeneity of flow (and thus habitat) across the entire reach. A high degree of habitat diversity (i.e. heterogeneity) was desired, as a diverse range of habitat conditions should in principle encourage biodiversity at the site (Dunbar et al., 2010b, Ward et al., 2002). This metric helped to inform flow maxima and intermediate flows between maxima and baseline, as the flow regime aimed to reach optimal habitat diversity within an appropriate range of flow magnitudes. Due to the influence of temporal flow characteristics, evidenced in Chapter 3 and also in literature (Poff et al., 1997, Summers et al., 2015), the temporal aspect of the flow regime was also considered in order to ensure robust ecological provision by proposed flows. Such consideration is not possible within current habitat suitability models, and therefore supplementary approaches were taken alongside Q vs HSI and HHS metrics to improve flow variability through the frequency and duration of flow maxima events. The primary output of Chapter 5 was the proposal of a flow designation framework that was informed and assisted by the other two research Chapters, 3 and 4. The framework was tested through the development of environmental flows at a case study site, and the potential benefits of the flows when compared with typical flow regimes at the site were discussed.

Bringing together findings from Chapters 2, 3 and 4, Chapter 5 provides perhaps the fullest solution to research question 1 within this investigation. As the culmination of the investigation, it demonstrates how principles in current literature (Chapter 2), general regional principles (Chapter 3), and hydraulic-ecological outputs (Chapter 4) can together inform the flow designation at a case study site. This process was briefly reviewed above, and is discussed more fully in Chapter 5.

For research question 3, Chapter 5 expands upon how the localised hydraulic requirements of taxa might be translated into an ecologically beneficial flow regime; in addition to the integration of localised hydraulic requirements by individual species, represented by preference curves (described above discussing Chapter 4), this Chapter also integrates the proposal from Chapter 3 and established literature, that temporal hydraulics also play a vital role within ecologically beneficial flow regimes (Poff et al., 1997, Richter et al., 1996). This may be done by assessing conditions typical to similar non-regulated river systems within the region (Lancashire and Greater Manchester in North West England, in this case). In addition to temporal characteristics, habitat heterogeneity was also considered as mentioned, due to the ecological benefits it could provide in principle (Wiens, 2002). Therefore Chapter 5 describes a more holistic approach to flow regime designation than ecological provision in terms of Q as a steady flow, and suggests an approach that integrates magnitude requirements, typical natural conditions, and habitat heterogeneity at the site. This may result in a flow regime similar to some designated through BBM (King and Louw, 1998), but in a less site-specific manner that is achievable with fewer resources, and has a quantified and criterion-driven basis that may be more readily implementable by water managers. As mentioned, such an approach may be most appropriate for smaller scale sites where the advice of expert panels is not feasible and where more assumptions

can be validly made due to smaller sites in general being less complex, but if appropriately scaled up may also prove to have advantages over established approaches such as BBM. Chapter 5 proposes the alternative approach taken by this thesis that may allow for the rapid designation of flow regimes for smaller scale modified systems, with the possibility of transferable general regime design and requiring only a small team for the investigation to be performed.

Research question 4 follows from research question 3; assuming an ecological metric is identified, how is it to be interpreted, given the variation of such metrics between species and over time? CASiMiR outputs offer individual ecological metrics for each indicator species; in order to consider the ecosystem as a whole, the user must decide how to interpret model outputs. For example, they must decide whether the mean HHS of all indicator species provides a meaningful metric; in some contexts the mean HHS may be valid, whereas in others a mixture of highly resilient and sensitive species may be present and render mean HHS less relevant. In this investigation, flow maxima were informed by habitat diversity and mean HHS values, while baseline low flows were based upon the individual HHS values of sensitive species. Frequencies and durations of flow events within the regime were based upon typical regional conditions within non-regulated rivers, and seasonal variation was determined by practical constraints (e.g. high demand for water security in summer) and biological requirements (e.g. low productivity in winter). In another example of recent, large-scale flow regime design, Chen and Olden (2017) took an approach in which native fish populations were maximised in terms of flow needs based upon functional regression models predicting fish populations in response to flow magnitudes, whilst non-native species populations were minimised through flow designs working against their flow preferences. Additionally, the study considered human water-use requirements and aimed to optimise the flow regime for both societal and environmental needs. Flow over time was based upon an equation balancing the varying flow needs of human, native fish and non-native fish throughout the year (Chen and Olden, 2017). The study had notable differences from the approach taken within this thesis. Firstly, the study targeted fish as indicator species, as opposed to macroinvertebrates, which could be argued to be less transferable due to the prolific nature of macroinvertebrates. Chen and Olden (2017)'s approach is also highly site-specific, targeting certain fish (with specific requirements that may not apply to hydrologically similar sites hosting different species) and certain societally-driven objectives and prescribing daily flow releases for the specific site. Their approach is highly detailed with specific criteria for flow regime designation, but represents a method that is most feasible within the largest and most vital systems (the study worked on a very large scale with mean daily flows of up to 250m^3/sec), in contrast with this thesis that has first worked to develop a method that may be appropriate for rapid environmental flow designation within smaller and more numerous systems.

Chapter 5 presents the approach of a criterion-driven flow regime design, encompassing both temporal and magnitude-based requirements. Results demonstrated that historical flows from 2017 released 300,000m^3 more flow than previously implemented flows at the site observed in 2014 data, yet only marginally improved ecological metrics. This investigation suggests that a holistic approach to environmental flow design is necessary to efficiently provide for ecological requirements in a world with increasingly pressing and conflicting water resources demands, consistent with claims from other recent studies (Gillespie et al., 2015b, Gillespie et al., 2015a, Worrall et al., 2014, Brooks and Haeusler, 2016).

Outputs from this Chapter have provided insights into how environmental flow regimes might be designated with relatively few resources or manpower requirements, and suggests that there may be significant research potential in this area, both for the generation of academic knowledge and for the improvement of water management practices. Such potential may stimulate further study into this particular approach in order to test its application in different contexts and validate the robustness of its outputs. Should studies more fully affirm and refine the approaches utilised in this investigation, this might facilitate the efficient designation of environmental flow regimes within ecologically-impacted smaller-scale impounded systems which could not previously be mitigated due to the disproportionate cost of doing so; many systems have ecological issues identified but fail to be actioned at the cost-benefit analysis stage (Grontmij, 2013).

The potential to improve ecological metrics while conserving water may stimulate interest in the field of environmental flows for other stakeholders outside of academia; rather than being another protocol to follow, Chapter 5 suggests that environmental flows may be an opportunity for the water industry to conserve water while also working to meet environmental legislation. Such findings may also be of interest to downstream interest groups such as fishing communities and conversation groups; instead of finding a compromise between environmental and societal use as is often suggested (McManamay et al., 2016), in some cases it may be possible to benefit both. This is particularly the case in contexts such as that at the case study site Ogden Brook, where reservoir flow licences are based upon historical downstream needs (that typically no longer exist), rather than on a firm scientific basis. Additionally, the utility of the approach, demonstrated through the development of three flow regimes in Chapter 5 with varying degrees of balance between saving water and making provision for ecological requirements, allows for varying prioritisation of stakeholder interests and therefore may be applied within a broad range of contexts.

As well as utility, the approach may also allow for adaptability. The three regimes developed in Chapter 5 (or developed at any other site) could in practice be swapped between to allow for a more adaptable and flexible form of water management. For example, in a particularly dry year the regime may change to the minimal environmental flow, whereas in a particularly wet year the highest flows might be chosen. This approach has already been proposed in the area of abstractions, where studies promoted reforms to current abstraction practices that operate more sustainably; protecting the environment when necessary but allowing water to be utilised when it is readily available. This has been referred to as "smart licencing" (Wilby et al., 2011). Following formal consultation, the UK government concluded that water abstraction practices were to be reformed, setting out a water abstraction plan in 2017 and aiming to have updated all abstraction licencing strategies under the Environment Agency by 2027 (Department for Environment Food & Rural Affairs, 2017). Many of the same principles apply for flow modification through flow impoundments as they do to water abstractions; an adaptive approach to environmental flows that makes use of water resources when they are freely available, and retains these resources when they are scarce, is likely going to be an expected practice for water managers in coming years similar to abstraction reforms. Devised approaches to flow regime design should therefore have an adaptive element to them; in the case of the approach devised by this thesis, adaptive management could be a simple matter of designing a series of flow regimes and switching between them as required in response to changing water resource availability. For impoundments, such decisions may be based upon current reservoir capacity, and hydrological forecasting. Changing our current practices is a particularly relevant topic within the context of water shortages being expected to occur in England within 25 years under the current trend of water consumption (BBC,

2019), in addition to growing concern for the state of riverine ecosystems and the introduction of firmer environmentally-driven legislation such as the Water Framework Directive and the drive to meet GEP within heavily modified water bodies (European Commission, 2000).

We have discussed the potential for approaches in Chapter 5 to be applied in similar contexts to that of the case study site utilised in the investigation, yet there is also the question of the potential for the methodology's wider implementation to a larger scale of system. As discussed above regarding Chapter 4, it would be possible to scale up the modelling aspect of this investigation for application in rivers of higher magnitude classes, with adaptations required such as more accurate depth-velocity measurements and the consideration of the impacts of sediment and vegetation, and possibly the inclusion of other biota such as fish within the ecological analysis. Assuming a model is developed for a given site, it would then be a simple matter to also incorporate the assessment of habitat heterogeneity through model outputs, and this would not require further adaptation at larger scales as the principle remains the same. A larger system would likely require a greater number of indicator species to represent the range of biota present, and this would both entail a more labour-intensive field campaign and likely greater consideration being required when translating individual species requirements into an overall flow regime. Within the Ogden Brook system, species did not display conflicting interests in terms of flow due to a general preference for higher flows than were currently in place. In a larger system, it is likely that ecosystem preferences will not be so straight-forward; therefore water managers must consider possible trade-offs between maximising overall biodiversity, and mitigating impacts upon sensitive species. In Chen and Olden's (2017) study discussing "designer flows", an approach is proposed by which desirable characteristics (such as the protection of an endangered species) of the riverine system are maximised through flow, while undesirable characteristics (such as the presence of invasive species) are eliminated or minimised. Such decisions would likely require expert opinion and would increase the human resources demand of the investigation. It is not easy to quantify the extent of resources required for a large-scale application of the methods proposed in Chapter 5, and therefore it is challenging to compare this approach with other established frameworks such as BBM (King and Louw, 1998), which, as previously mentioned, is an appropriate approach for individual systems but is constrained by its intensive human resource demand due to expert-driven workshops being a central component of the approach, and lacks a defined criteria or process justified by data that may be preferred by some water managers. Due to the quantity of impoundments that exist across the UK, having the highest density of impoundments in Western Europe (Lehner et al., 2011), it is likely that more quantitative, computational approaches, actionable without reliance upon the presence of an interdisciplinary team of experts, will be a desirable alternative for water managers, particularly as pressure grows to meet environmental objectives before defined deadlines (UKTAG, 2013).

6.3 General discussion

The majority of previous studies have generally focused upon site-specific flow experimentation (often disproportionately focusing upon the impacts of flow magnitude change), or have attempted to quantify the extent of flow regulation at a given site through methods such as IHA (Gillespie et al., 2015b). With the foundations provided by such studies and lessons taken from meta-analyses such as Gillespie et al. (2015), and Poff and Zimmerman (2010), this thesis goes beyond existing literature by identifying potential quantitative ecology-flow relationships in Chapter 3, and applying these concepts alongside habitat suitability modelling in Chapters 4 and 5 in order to achieve a methodology by which

environmental flows can be designated, demonstrated using a case study site. There is still a great deal of work required before the ecological response to flow is clearly and widely understood, and environmental flow implementation can become a routine procedure for water managers.

6.3.1 Thesis outputs in the context of ecosystem services

In Chapter 1, the concept of ecosystem services was introduced as one of the main drivers of research within the field of ecohydraulics. Ecosystem services were described as "benefits people obtain from ecosystems", and encompass the areas of provisioning, regulating, cultural, and supporting services (Reid et al., 2015). The findings of each of the chapters within this thesis have been described and discussed, and the implications of these findings should be considered in light of the main motivation behind much of the research occurring in this field; that being the sustainable management of water body (rivers, specifically in this thesis) ecosystems in order to maintain their ongoing services. It has been described how this thesis has presented the methodology for environmental flow designation, and as has been described above, the primary research aims for this thesis have been met, though a number of limitations within this work must be considered. The question remains, however: how do these findings relate to the general goal of maintaining ecosystem services?

Chapter 2 described the potential impacts of flow modification through river impoundment, and how the ecosystem – being adapted to natural conditions – may deteriorate in terms of metrics such as biodiversity or species abundance. This may in turn compromise multiple ecosystem services, for example an ecological imbalance driven by flow modification may lead to a decline in fish populations; this could in turn impact the cultural services of a water body (in terms of aesthetics and fishing, for instance), and may lead to a degradation in regulating services due imbalances in the food chain – resulting in issues such as a deterioration in physical, chemical, and biological conditions (Haines-Young and Potschin, 2013), which may in turn impact the provisioning services of the water body in terms of fresh water.

The research goals of this thesis, and indeed much of the field of ecohydraulics in general when dealing with freshwater systems, find context within the maintenance of ecosystem services. Of the outputs generated by this thesis, the following are likely to have the most significant impact in terms of maintaining ecosystem services, and each of the implications will be discussed in turn:

- General ecology-flow relationships
- Ecological modelling methodology
- Environmental flow designation
- Potential water conservation

The importance of generalisation of ecology-flow principles are discussed throughout this thesis, i.e. the value of being able to consider mitigation measures without the need for site-specific investigation; this is of particular importance in smaller-scale systems where resources may not be available for such study. These smaller-scale systems, whilst not providing significant provision of fresh water, often are of great cultural value to the surrounding area, and play a vital role within the broader ecosystem in the surrounding floodplain (Junk et al., 1989). As discussed in Chapter 3, the findings from this thesis highlight some statistically significant ecology-flow relationships that may be considered for river systems of a similar scale and climate, and, perhaps more importantly, feed into a broader meta-analysis that may generate more robust results. The relationships found, and those

that might be found through future analyses, would greatly assist water managers to assess potential impacts and mitigation solutions at a more generalised desk-based level, potentially offering cost-effective maintenance of ecosystem services within numerous small-scale systems that would otherwise be difficult to evaluate. The generalised observations may, for example, lead to compensation flow best practices that improve upon the "rule of thumb" flows that are typical in these systems currently (Arthington et al., 2006). It could therefore be expected that some pressure upon these ecosystems would be lifted, thus better maintaining the ecosystem services of such waterbodies. These relationships also feed into the other outputs of this investigation, particularly the final environmental flow regime proposals, as is discussed throughout this thesis.

Next the ecological modelling methodology that was proposed in Chapter 4 will be considered. The nature of this approach, being primarily desk-based (with the exception of field samples and measurements taken for model calibration), again should aid in the wider application of environmental flows, as the resource requirements of habitat modelling are less than other common ecology-flow investigation approaches such as flow experimentation, which requires long-term monitoring, potentially expensive in-situ sensors, and numerous excursions into the field to observe changing flow conditions and the impacts upon the ecology. Such approaches also require a prolonged period of time before data can be generated, due to the greater need to sample across seasons and flow conditions. These approaches also benefit from having clear metrics from which to base or communicate mitigation solutions; this being in contrast to approaches such as BBM which rely upon expert knowledge, with associated uncertainties and lack of transferability that come about due to the subjective nature of expert opinion. The expected implications of the modelling methodology are therefore similar to those described for the ecology-flow relationship observations; essentially acting as a tool that increases the cost-effectiveness of flow modification investigations. It is also noted that ecological modelling in and of itself is not original to this thesis, and the output model's implications tie closely to the next output of this work that will be discussed; the designation of environmental flows.

Environmental flow designation is the culmination of much of the work of this thesis, and carries some of the most direct implications for the maintaining of ecosystem services. The designer paradigm approach to the allocation of flows is expected to increase flow heterogeneity and meet the needs of target species at key times. This has the potential to better maintain ecosystem services in general due to greater flow 'naturalisation' in the form of lessened homogeneity, and in specific application through targeting certain species; for instance improving ecological conditions for desirable species. As was seen in Chen and Olden (2017), this can also be expanded to conversely bring about less favourable conditions for species that are detrimental to the system, such as invasives. Altering flow to reduce invasive populations, whilst supporting desirable species populations, would impact all levels of ecosystem services, and would be expected to alleviate pressures on the sustainability of the ecosystem; for instance removing or reducing the influence of invasive species that may otherwise outcompete and displace desirable native species. The final key output of the thesis comes about from the results of environmental flow designation; this being the demonstration that significant amounts of water may be conserved through proposed flows, depending upon the priority water managers ascribe towards conserving water within an impoundment.

Potential water conservation within impoundments is likely of major interest to certain stakeholders such as the water utility companies managing said impoundments. These utilities often face pressure

relating to water security, particularly during times of drought, and there are concerns that water resources may be greatly strained in the coming years (BBC, 2019). Therefore provisioning ecosystem services are vital to the ongoing wellbeing of the populations reliant upon these water resources, and the outcome from this thesis, that potentially large quantities of water may be conserved through changing the way in which we designate compensation flows, could significantly contribute towards future water security.

These four outputs, as discussed, have the potential to be a significant contribution towards the maintenance and future sustainability of ecosystem services, which is the fundamental driver for attempting to understand ecology-flow relationships within this thesis. However, the work presented in this thesis has limitations, and represents initial first steps as opposed to a fully proposed solution to flow modification. The following section of this chapter will discuss the limits of this thesis, and will provide a perspective on potential work that could be seen in the future, which would build upon the foundations set out in this work. ***6.3.2 Thesis limitations***

The thesis has devised and tested an approach towards environmental flow regime designation within a small-scale, relatively controlled environment (as described in Section 4.2 and 4.3) in which external variables are minimised as much as possible. Methodological limitations have been discussed in their respective Chapters, but additionally the approach proposed has general limitations, and those methodological limitations previously discussed must be considered in terms of how they limit the potential applications of this thesis as a whole, and how the outcomes of the thesis might be interpreted.

Firstly, due to environmental flows being a wide subject with a number of branches such as impoundments, hydropeaking, wetlands, abstraction, etc. most studies, including this thesis, may be of limited application to areas outside of their primary focus (impoundments in this case), due to differing forms of flow modification, differing ecological impacts, and differing infrastructural management requirements. Secondly, this thesis has focused upon two areas of environmental flows and ecohydraulics; the quantification of ecological-flow relationships, and an approach by which environmental flows might be designed. A number of other concerns exist within the field that have been beyond the current scope of the thesis. The influences of the sediment and thermal regime, biological interactions, or the role of extreme events have not been considered here, nor has stochasticity and uncertainty inherent to environmental water processes or ecological outcomes been quantified; these issues were also highlighted as pressing challenges in the field alongside the need for improved understanding of ecology-flow interaction and methodological development (Arthington et al., 2018). In the process of adapting the approach posed by this thesis for application in more complex and possible larger system, such concerns would likely require addressing due to their greater influence upon the ecosystem relative to the case study site utilised here, as will be discussed in Section 6.3.3.

Variables such as thermal regime or biological interactions discussed by Arthington et al. (2018) as areas requiring further research may possibly be integrated into this approach as understanding of these phenomena develop, but currently such factors must be associated with uncertainty within the methodology.

The thermal regime, as described in Section 2.2.4, is considered to be an important ecological driver within riverine systems (Olden et al., 2010), yet it was decided that investigation into the impacts of, and potential mitigation measures for, the modification of the thermal regime would not be incorporated into this thesis. There are some assumptions and limitations associated with this, but also a firm rationale for this omission. The key limitation of not considering the thermal regime is, of course, that this thesis is unable to comment upon the area of thermal regime modification, and the solutions proposed by this thesis may not be transferable to systems that have issues associated the thermal regime. This imposes a limit upon the transferability and the scope of the investigation methods and limits the applicability of mitigation solutions presented in this thesis. There is also a key assumption associated to not investigating thermal impacts upon ecosystems; with the omission of it as a driver in ecological modelling, it is assumed that thermal regime modification is not obscuring or overriding the impacts of flow modification, and that the flow regime changes proposed in this thesis will not introduce detrimental thermal regime modification impacts.

Results in Section 4.6.2 suggest that biota are responding as expected to flows within the Holden Wood system, and given that proposed flow regimes have constraints in place to prevent too much draw down in water level within the reservoir for the purposes of water security, and to ensure sufficient water is available for compensation flows in the long-term, as shown in Section 5.3. The new regimes are unlikely to lead to a change in the current thermal regime active at Holden Wood; be it currently modified or unmodified. Therefore, the thermal regime is unlikely to be obscuring ecology-flow relationships, and is unlikely to be changed by the environmental flow regimes suggested in this thesis.

Having discussed the limitations, let us move on to the rationale behind not investigating the influence and potential mitigation measures for thermal regime modification. The thermal regime was not considered both due to logistical and practical factors, and due to a potentially detrimental impact upon the accuracy of ecology-flow relationship predictions. Firstly the logistical and practical issues will be discussed. This thesis aimed to directly influence current industrial mitigation measures; as such, the findings of the thesis should be actionable within the system that was under investigation, that being Holden Wood. Currently, the thermal regime cannot easily be manipulated through mitigation measures; Olden et al. (2010) acknowledges that modification of the thermal regime is very much an area in its infancy, with significant uncertainty (more so than flow regime modification, an area already fraught with uncertainty). The study identifies a number of challenges to be overcame before thermal regime modification impacts can be quantified and effectively mitigated, the major challenges discussed by Olden et al., 2010 are outlined below:

- We currently have a poor understanding of how impoundments alter system thermal regimes
- We currently have a poor understanding of ecosystem response to modified thermal regimes
- Should we wish to mitigate thermal regime modification in a highly regulated system, it is likely that engineering modification to the impounding structure would be necessary
- Still unclear how thermal criteria may be incorporated into environmental flow designation
- Still unclear how the thermal regime's influence might be investigated within an environmental flow assessment

Due to the above challenges, and the fact that research is still inconclusive on these issues (Gillespie et al., 2015b), it is believed that investigation into thermal regime modification is beyond the scope of this thesis, given this thesis' focus upon the flow regime modification. Incorporation of variables that

consider the thermal regime's impact upon ecological metrics are unlikely to produce actionable results, and in fact would likely introduce greater levels of uncertainty into the investigation, both due to the current uncertainty as to how ecosystems respond to thermal regime modification, and due to the fact that such inputs would require a further generalised input of thermal preferences into the CASiMiR model. In Section 4.7.2 it was discussed how such generalised preferences may introduce some uncertainty, and these additional preferences would possibly obscure ecology-flow relationships that this thesis set out to investigate.

While the thermal regime is perhaps the key ecological driver alongside flow, a number of other variables exist that may exert varying degrees of influence upon the ecosystem; as discussed by Poff et al., 1997, rivers are complex open systems that can contain a large number of interacting variables. This thesis took steps to limit and 'control' these external variables as much as possible, through river classification in Chapter 3, and site selection in Chapter 4, for instance. However, the influences of external variables are likely not entirely negated, and in the process of mitigating their influence to an extent, the thesis may be limited in its scope of application. Here, we will consider these two forms of limitations: Limitations imposed by uncertainty associated to external variables, and limitations imposed necessarily by the scope of the work performed for this thesis.

Water quality – water quality encompasses a variety of factors contributing to the chemical state of the river system. This includes pH, the presence of pollutants, and other influences that may impact the water's chemical composition; for example dense woodland surrounding the river is likely to be a significant source of carbon, and can trigger other chemical changes within the river system, due to the products of decaying leaf litter that will be washed into the river system (O'Brien et al., 2017). Such chemical changes may be beneficial (for instance, increased nutrient availability) or detrimental (for instance, the presence of toxic chemicals) to the various species within the ecosystem. As stated both in the methodology for the multi-site analysis of Chapter 3 and the case study for Holden Wood in Chapter 4, this thesis took measures to avoid systems in which water quality was acting as a potential detriment to the ecosystem. This is due to the fact that most water quality issues are not directly related to reservoir impoundment (though flow modification may play a role in water quality degradation if the ecosystem regulating services are compromised), and therefore would require alternative mitigation solutions beyond the scope of this thesis. The fact that systems with water quality issues were avoided in this study, however, does entail that likewise the application of the results from this study are necessarily limited to systems that do not suffer from poor water quality. In such systems, it is unlikely that solutions proposed here will bring about desirable outcomes if water quality remains as a significant pressure upon the ecosystem and its associated services. Care should also be taken in applying the principles in this thesis directly to systems of good chemical status yet significantly different chemical composition; such as an aforementioned dense woodland system. Iit is advised that one should validate whether a system of very different chemical character responds similarly to changes in flow regime, before undertaking a full restoration project based on the principles discussed in this thesis.

Biological interactions - as mentioned previously, particularly when calibrating the ecological model in CASiMiR, investigated species are not isolated entities; their populations and distributions within a system are often driven by biological interactions, such as predation, competition, or a species' tendency to group together in colonies (McCabe, 2010). Such interactions present a complex system of mutual correlation between the various members of the ecosystem, and as such are beyond the

scope of consideration in this thesis. Because of this, biological interactions present a source of uncertainty within this investigation, and species distributions were treated as having a degree of stochastic nature to them. In general, this would entail that caution should be used when interpreting or applying principles of this investigation, as one should have with any experiment involving some degree of uncertainty. Further, there may be cases in which biological interactions are more influential, for example a scenario in which a predative invasive species is introduced to a system. In such a scenario, species populations and distributions may be driven more by the disruption caused by the new species than by adjustments to the flow regime. As such, care should be taken in simply applying the methodology of this thesis to a given system; prior knowledge of the state of the ecosystem and its component species would first be beneficial.

Vegetation – as mentioned in Section 6.2.2, this thesis did not consider the role of vegetation upon the ecosystem and upon flow dynamics (Shucksmith et al., 2010) due to the fact that this thesis aimed to first address the fundamentals of a novel approach within a simple, small-scale system. I discussed the need to consider the influence of vegetation in larger, more complex systems. The approach of this thesis in its current state is limited to smaller-scale sites with sparse vegetation within the river channel; application to more vegetated channels would require adaptation of the methods proposed to incorporate vegetation's role, perhaps most importantly upon flow dynamics, and perhaps also its role as refugia for species, as this could influence species distributions. Adaptations such as this are discussed in Section 6.3.3.

Shade – Whether or not an area within a channel is exposed to direct sunlight can significant impact the ecosystem within that area. In systems where there are areas of shade, due to tree canopies for instance, the growth of vegetation will be altered, the distribution of species will be altered, and nutrient availability may be affected (McCabe, 2010). The Holden Wood site had no tree coverage, and therefore shade did not play a significant role within the system and was not considered. As such, systems that have extensive tree coverage might expect a different ecosystem response to flow changes as a result of interactions with external drivers such as shading

Sediment – The sediment regime can have a significant influence upon the ecosystem due to its impact upon species habitat, visibility within the water, nutrient retention, spawning, and river morphology (Wampler, 2012). It was discussed previously that flow modification can impact the sediment regime, though the impact upon sediment is often caused by the physical barrier of the impoundment itself (Kondolf, 1997), and will not be completely mitigated by changes in flow releases. Systems with severe sediment regime issues would therefore require sediment restoration schemes in addition to environmental flows in order to maintain ecosystem services, and the findings from this thesis would be of limited impact in such scenarios – although should the sediment issues be resolved, the findings in this work may then become more applicable.

Next, the uncertainty presented due to limitations inherent to the scope of this project will be discussed, which will again entail some caution as to the interpretation or application of the results in this thesis in contexts outside of the scope of the work undertaken.

Geography – It has been discussed in previous chapters that geographical area can significantly influence the character of a river system, and this is the reason why this thesis has focused upon the area of Northern England. The influence of geography is due to regional factors such as predominant underlying geology and local climate (Gonzalez and Garcia, 2006). As such, findings in this thesis should

be interpreted or applied with caution in areas beyond the region studied; species may respond to flows in a different manner and therefore regional calibration may be necessary. Caution should increase the further a region deviates from the conditions in the initial study; for instance, it could be expected that minimal adaptation of the method may be required for systems in temperate Southern England, whilst a river in a tropical climate may require an entirely different approach towards mitigating flow modification impacts.

Scale – The scale of a system is one of the major factors of differentiation between river systems, and was proposed as one of the main factors for the class-based approach towards environmental flow investigation (Arthington et al., 2006). This is due to the fact that the scale of a river has a dominant influence upon its character; larger rivers entail larger flows, and higher flows result in river morphology, local flow velocity conditions, water depth, and other variables, being very different from conditions observed within smaller-scale systems. As discussed previously, Monk et al. (2006) described flow magnitude as a dominant ecological driver, masking other variables that may be influential at a finer scale (i.e. within certain flow ranges). Thus, due to the fact that this thesis has focused upon smaller-scale systems such as Holden Wood, it is not thought that principles identified in this work will be directly transferable to larger systems. It is very likely that significant adaptation of the approaches used in this investigation will be required when investigating rivers of a higher flow magnitude, and some potential adaptation solutions have been discussed both in Section 6.2, and in the Discussion sections of previous chapters.

Metrics used (HSI / HHS) – This thesis has approached ecology-flow relationships and how they may be interpreted to inform environmental flows by using a particular perspective, that being habitat preference in the form of the HSI and HHS metrics. Sections 4.7.2 and 5.4.7 discussed the assumptions and limitations of these metrics. Whilst it is expected that improvement in these metrics will result in overall benefit for the ecosystem, and the maintenance of its associated services, caution must be used in how the metrics are obtained and interpreted within systems; appropriate indicator species within the context of a given river system must be utilised, and water managers should keep in mind other potential factors influencing the ecosystem, such as the external factors described above.

Temporal dynamics – it was not possible to directly model the impacts of temporal dynamics within CASiMiR, as discussed in Chapter 5, and therefore the ecological principles behind the benefits of flow heterogeneity were adopted and flow variation was set based upon flow regime characteristics within natural systems. Though there is a good scientific basis for integration of flow heterogeneity into environmental flows, predictions of ecosystem response on this basis cannot be provided. It is likely that the degree of ecosystem response to increased flow heterogeneity will vary between systems, and therefore caution should be used and expectations managed in the incorporation of temporal flow dynamics until validation is performed upon the system in question to monitor the post-implementation impacts of a proposed flow regime. Validation is a key need in order to advance beyond the findings of this thesis, as I will discuss further in Section 6.3.3.

Having now discussed a number of limitations within this thesis that should provide caution for potential applications, let us consider potential next steps from the work that has been laid out, and consider what other developments might be expected within the area of environmental flows in the coming years.

6.3.3 Potential future developments

This thesis has presented a novel approach to environmental flow designation within impounded river systems; as an initial proposal, I have focused upon relatively simple conditions and a limited number of metrics and variables. This investigation has also been somewhat resource-limited, which has influenced methods that have been utilised. Should the approach proposed by this thesis gain wider acceptance, there is significant scope to further develop the methods within this thesis, in terms of greater investments of resources to achieve higher accuracy, or to scale up the approach and apply it to larger and/or more complex systems.

Before moving into additional developments and increased complexity, a key next step in furthering the findings in this thesis and making them more robust would be to fully validate the approach and findings presented. The ecological response to the range of flows described in Chapters 4 and 5 were not observed within the field; calibration of the model took place at the steady flow of $0.024m^3$/sec. Therefore, whilst calibration suggests that model predictions are sensible and justified, the approach requires full validation through long-term flow experimentation before the response of ecology to the proposed regime can be quantified and compared with model predictions, in order to confirm that the designated flow regime achieves its objectives of improving ecological metrics whilst conserving water. This would entail ecological and hydrological monitoring over a number of months, perhaps up to a one or two years, due to the time taken for ecosystem response to reach equilibrium and adjust to new conditions. Upon validation of the environmental flow regime(s), further research might more confidently be considered in applying the approach to other systems to assess its transferability and potential for scaling up for application in larger systems.

Having discussed the importance of validation, a number of potential future developments will be discussed that could potentially aid in improving robustness or expanding the potential applications of the approach (e.g. to larger and more complex systems). Firstly, let us consider the potential of CASiMiR in accounting for more variables, and thus accommodating investigations within more complex systems.

A number of external variables can be accounted for through adaptation of variables utilised by the ecological model, easily achievable due to features provided by CASiMiR, such as its ability to account for variation of bed sediment, the influence of channel vegetation, or the impacts of shade. These impacts can be accounted for within the model if the relevant preference curves or fuzzy rules are inputted. The limitation in accomplishing this would be the attaining of robust preference curves, which may currently prove to be difficult; preference curves calibrated to a particular class of conditions have been found to be more reliable than universally general preference curves (Kelly et al., 2014), but these are not readily available in many systems. That said, availability of such preference curves should improve with time as further contributions are added to the existing knowledge base.

In addition to further ecologically-influential variables, the complexities of natural flow inputs such as tributaries that contribute to the river flow alongside the compensation flow may be accounted for through hydrological forecasting of precipitation and the resulting flow from the tributary in order to predict likely flow inputs into the main channel. As technology continues to develop, this forecasting would inevitably become more robust. In time, options for feasible long-term in-situ measurements of flow may become available, allowing flow to be measured directly in real time. Flow inputs, forecasted or measured, could be accounted for through an adaptive management approach of

impoundment outflows; potentially reducing flow outputs at times when tributary inputs are elevated, or conversely increasing flows during dryer periods.

As well as integrating further variables into the modelling component of the approach, there is scope to improve other aspects of environmental flow investigation through more advanced technology and additional resources that may be available for future studies. One example of this would be the ability to better integrate river connectivity in terms of its relation to flow and the impact upon the ecosystem. In section 5.4.6 it was mentioned that the potential use of colour extraction software to achieve this end. Such integration of river connectivity could be particularly important within larger river systems with extensive floodplains, and thus the capability to assess the impact of connectivity and wetted area with software solutions would increase the applicability of the approach proposed by this thesis at a larger scale.

Digital elevation mapping took significant effort within this investigation, and this could be a limiting factor upon the scale at which modelling is performed. Technological advances may resolve this issue; drones - specifically, unmanned aerial vehicles (UAVs) - have become widespread and publically available in recent years. UAVs have recently been proposed to assist in generating digital elevation models, as well as other in-field data such as the presence of riparian vegetation, sediment characteristics, and flow velocity; a study by Rhee et al. (2018) provides a thorough overview of recent advances in UAV applications within the fluvial environment. Such technology could not only facilitate efficient data collection at larger scales, and produce higher-resolution data than that typically collected by manual approaches such as the Total Station (through the use of specialised bathymetric LiDAR technology), but could also perform various other remote sensing tasks that would otherwise require extensive field surveys or measuring campaigns. Examples of this include the detection of algal blooms, or utilising aerial velocimetry methods such as the Large-scale Particle Image Velocimetry method to determine flow velocity (Rhee et al, 2018).

Rhee et al.'s 2018 study provides some promising perspectives, but acknowledges that such technology currently suffers from limitations. For instance, the use of LiDAR is limited due to the weight of the hardware, and LiDAR specialised for bathymetric study has not yet been perfected. As these technologies develop, UAVs may revolutionise researcher's approaches to in-field monitoring and measurement; they may facilitate far more detailed data collection, or they may greatly increase the efficiency of the current levels of data collection. In the case of the former, this may aid in the analysis of larger and more complex systems, whereas in the case of the latter, it may become feasible to monitor many smaller-scale systems that would otherwise have been overlooked due to resource or time constraints. It is possible that as such technologies see application, the field of environmental flows will see a vast increase in available data. This leads us to another form of technology that is showing great future potential in the area of data analysis: machine learning.

Studies have lamented the lack of available data for synthesis in the field of environmental flows (Gillespie et al., 2015b), but as mentioned above, technological advances may lead to an increase in monitoring data such that innovation in data analysis will be required in order to process complex, high-dimensional data. The laborious data analysis approaches such as that seen in Chapter 3 may be revolutionised through advances in field of artificial intelligence, specifically machine learning. Machine learning (ML), which has seen significant advances over the last decade, could present a highly efficient method of data analysis within the field of ecohydraulics. Fields such as Chemistry have

recently felt the impact of ML; an editorial letter in the Journal of Physical Chemistry touts the success of ML in its 2018 issue, citing studies in which molecular bonds have been mapped to energy levels, particle size and composition were related to catalytic activity, and a neural network representation of an alloy was used to predict a material's composition and temperature-dependence surface segregation (Schneider and Guo, 2018).

The potential of machine learning has not gone unnoticed in the field of Biology either, as the letter written in Conservation Biology by Cheng et al. (2018), demonstrates. In this letter Cheng et al. (2018) discuss the rapid growth of environmental research and the growing interest in machine learning that has emerged alongside this. They mention that ML may greatly improve the process of data synthesis and effectively "sort the wheat from the chaff" for researchers seeking relevant literature to draw upon. As machine learning becomes more accessible to non-programmers, it has the potential to become a prolific tool in data-driven studies; efficiently sorting through large quantities of data and identifying relationships among many variables that may have otherwise been difficult or highly laborious to discern. Interfaces are being developed by which experts in other fields may more intuitively utilise ML; an example of this is Tableau, a commercial widely-used piece of software that facilitates machine learning for non-specialists (Murray et al., 2013) and cloud computing through virtual machines is facilitating the computational demand of ML algorithms, thus removing the prohibitive skill and resource requirements that previously limited the use of ML. Virtual machines are widely commercially available, one of the top providers being Microsoft, through the Microsoft Azure services (azure.microsoft.com/en-gb/services/virtual-machines). This thesis envisions a future in which class-based data may be accumulated and processed in this manner, and following this ML may also assist in inter-class meta-analyses in order to discern broader principles that exist on a more general scale; such principles would greatly aid water managers in maintaining ecosystem services and mitigating the impacts of flow modification.

With the rapid advancement of technologies, particularly in the fields of drones and AI, we may see significant changes in both field- and desk-based methodologies, as drones and easier-to-utilise in situ analysers become more widely available and intuitive to use for field measurement, and advances in the area of machine learning transform the way we analyse data. The embracing of technology to assist environmental research may occur all the more rapidly as a changing climate and increasingly resource-hungry populations further place pressure upon ecological systems. Legislation such as the WFD will continue to aid in communicating this pressing matter to government organisations and water managers; in the next section, the possible interpretations of the outcomes of this thesis within the context of legislation such as the WFD will be discussed.

6.3.4 Thesis within the context of the Water Framework Directive

Thus far this thesis has largely dealt with the academic understanding of flow-ecological relationships and its application in a water management context. A significant driver of environmental flows, environmental legislation (specifically WFD within Europe), also has major implications for environmental flow implementation; environmental objectives dictate the priority and the extent of restorative measures. A key question for water managers is how they might meet legislative targets through the application of knowledge and methodological development. The next section discusses this is greater detail.

Previous Chapters have yet to answer the final research question posed:

5. How may outcomes such as habitat quality predictions be interpreted in light of legislative targets?

In this Chapter, the implications of legislative targets and subsequent interpretation of the outputs of this investigation are discussed.

Chapter 2 provides an overview of the Water Framework Directive and its associated challenges. The concept of Ecological Potential, as opposed to Ecological Status, was described; this being an ecological designation applied to heavily modified water bodies, in which the economic feasibility and service provided by a particular water body is considered alongside ecological metrics before a rating is assigned. The legislator target of good ecological potential (GEP) is therefore decided based upon a water body having all reasonable mitigation measures in place against ecological impact, and an ecosystem with minimal deviation from the expected natural system, within cost-benefit and practical constraints. This leads to a somewhat subjective method by which HMWBs are assessed in terms of ecological potential, as it is the decision of the assessor as to whether or not the water managers have done everything reasonably possible to prevent ecological impact, and it can be difficult to fully quantify the value of the ecosystem and associated societal services provided in terms of a monetary value. It can also be difficult to determine the natural state of the ecosystem, particularly within systems that have been modified for long periods of time with reference conditions no longer available.

Other authors have attempted to define GEP and present steps by which it may be assessed, but the very apparent challenges are widely acknowledged (Borja and Elliott, 2007), and recently it has been proposed that the WFD cannot be properly implemented without a complete revision of the way in which ecological restoration is performed (Voulvoulis et al., 2017); the highly ambitious WFD legislation was put forward with "great expectations" as Voulvoulis et al. (2017) says, yet almost two decades following its implementation there has not yet emerged a standardised method by which targets such as ecological potential might be achieved. New ways of thinking must be put forward, such as "systems thinking", described as a "system to think about systems" (Arnold and Wade, 2015); or, elaborating further:

"Systems thinking is a set of synergistic analytic skills used to improve the capability of identifying and understanding systems, predicting their behaviors, and devising modifications to them in order to produce desired effects. These skills work together as a system." Arnold and Wade (2015)

Such a shift in thinking calls for a move away from approaches focused upon singular criteria, which might be illustrated in an impoundment release context through steady, "rule of thumb" flows (Arthington et al., 2006), into approaches that more broadly assess an ecosystem and consider how interdependent variables may affect outcomes. This investigation has also proposed a move away from singular flow release (Q) values, yet is by no means comprehensive in evaluating the riverine system; a more pragmatic view might consider such the task of deeply understanding the river system as a whole disproportionately labour-intensive relative to potential outcomes in the context of many river systems, and therefore the current implementation of restorative measures currently must rely upon the development of robust, transferable frameworks that act as a compromise between complete reductionism and comprehensive understanding. As knowledge, methods, and available tools continue to improve however, it is possible that we may come closer to a fuller understanding

of the complexities of the riverine system even at smaller scales. Increasing computational power, ever-improving hydraulic and ecological modelling software, and new field surveying tools such as geological survey drones (Baglione, 2016) allow for more efficient desk-based and field-based analyses, and will assist with progress within this field of science.

However, currently, due to the fact that a standardised framework does not yet exist for the implementation of WFD within HMWBs, we can only offer suggestions as to how the proposed flow regimes at Holden Wood might be interpreted in terms of meeting ecological potential. Given that ecological potential assessment considers the "reasonable efforts" of water managers attempting to mitigate ecological impacts, as opposed to being based entirely upon ecological metrics within the impacted system, it is believed that the method of flow regime designation proposed by this study would qualify as a significant measure taken towards improving the local ecosystem within Ogden Brook, given the scientific rationale provided for proposed flow regime characteristics, and the demonstrated validity of the ecological model utilised. However, as Chapter 5 demonstrated, flow regimes may be adapted relative to their priorities; regimes may release more annual flow to maximise ecological provision (as in regime A), or less in order to maximise water conservation (as in regime C); the appropriate regime for GEP would have to be assigned based upon a thorough cost-benefit analysis at the site. Additionally, ecological impacts in addition to the flow regime may need to be considered and actioned in order to meet GEP; flow modification is only one aspect of the system modification imposed by an impoundment.

The sediment regime may be disrupted due to the loss of connectivity upstream behind the impoundment (Junk et al., 1989), the resulting water with less dissolved particles may enhance erosion and affect river morphology close to the impoundment outlet (Kondolf, 1997), the thermal regime may be impacted due to thermal stratification within the reservoir (Olden and Naiman, 2010), and the transport of nutrients from upstream to downstream may also be disrupted (Vannote et al., 1980). Therefore, the designation of GEP may not be resolved by any single mitigation measure, but must consider the impacts of system modification in their entirety. This investigation has proposed a solution for flow regime modification; an area that is still in its infancy in terms of mitigation methodology (Gillespie et al., 2015b, Poff et al., 2017), but such a solution acts as a contribution towards GEP, not a guarantee of it. As previously mentioned, implementation methodologies for WFD must continue to be improved upon and encompass the system as a whole, if environmental targets are to be met at a national scale.

6.4 Conclusions

This Chapter has discussed the key findings, implications, and possible applications of this thesis. It has also discussed the interpretation of these in light of WFD legislation, and possible direction for future research. We believe that this investigation has provided outcomes that will be significant for the scientific community. The first major outcome from this work has been the demonstration of quantitative general, regional relationships for the impact of particular flow characteristics upon both functional composition and biodiversity in the region of northern England, within a particular magnitude range, following approaches proposed in previous literature (Arthington et al., 2006, Monk et al., 2006). This research both provides insights into the implications of flow modification in this

region, and, it is hoped, stimulate similar research in order to develop a broader knowledge base that may be synthesised in future meta-analysis studies.

The second major outcome has been the demonstration of the ongoing potential for hydraulic-ecological modelling for the assessment of environmental flow regimes at a case study site, particularly in the context of smaller river systems, having demonstrated the agreement between model-predicted individual species habitat suitabilities and in-field observed species populations; this may stimulate similar investigations that may continue to explore the ability of such models to provide insights into ecological response to flow. Such models could be of particular use in systems where resource-intensive field or expert-driven investigation is unfeasible.

Lastly, the third major outcome of this investigation has been the development and application of a flow regime designation framework by which model predictions and supplementary data are considered in order to propose environmental flows for the case study site that was investigated. These flows were found to both maintain ecological metrics during important periods at the case study site, and also conserve significant quantities of water within the upstream impoundment. Such a result is expected to be of interest to both industrial and environmental stakeholders, and it is hoped that such an outcome will motivate further research into the area of "designer flows" (Chen and Olden, 2017), which may contribute towards a solution to the oft-cited conflict between societal and environmental needs (Summers et al., 2015). It is hoped that studies in this field will continue to advance to meet the challenges posed by impoundment-related flow modification, both in terms of generalised principles to apply to flow-ecological relationships, and in terms of environmental flow designation methodology. Additionally, academics, water managers, and legislators must continue and expand upon collaboration efforts in order to interpret research outcomes in light of legislation such as the WFD and expected best practice (UKTAG, 2013).

7. RESEARCH SYNOPSIS

7.1 Introduction

This thesis has investigated the impact of flow modification due to impoundment both in terms of general relationships across a region (Chapter 3) and more specifically through a framework by which a case study site is assessed and environmental flows are proposed (Chapters 4 and 5). A general discussion in Chapter 6 on the overall findings, implications, comparisons with literature, and possible applications and future research coming about due to this work has been detailed. This Chapter will

provide a synopsis of the entire thesis, briefly discussing key outcomes and implications, and the extent to which the initial goals of the project have been met.

7.2 Novel Outcomes

Chapters 3, 4 and 5 have described the process of developing an understanding of ecology-flow relationships and developing an approach for environmental flow regime designation, based upon desk-based analysis of multiple systems and through findings at a case-study site through hydraulic-ecology modelling. The key, novel outcomes of this thesis are as follows:

1. Quantified ecology-flow relationships linking flow event frequency to the ecological metrics of functional composition and biodiversity. High flow frequency was identified as a particularly influential driver, impacting functional diversity in both spring and autumn seasons using both LIFE and trait score metrics.

2. An integrated approach for environmental flow development, utilising model outputs as a basis for flow magnitude values, combined with ecological principles derived from influential flow characteristics identified in Chapter 3, and the analysis of natural systems in Chapter 5, as a basis for flow event frequencies and durations.

3. The integrated approach described in 2 was utilised to design three flow regimes for the case study site; these were the tangible output of the research performed in this thesis and provide a demonstration of the capabilities of the proposed approach to environmental flows.

7.3 Research questions

This investigation began by posing the following research questions in Chapter 1:

1) What specific drivers are eliciting an ecological response?

Chapter 3 is the major contributor towards this question; significant statistical relationships were observed between biological metrics and high and low flow event frequencies. Potential mechanics behind this were discussed with evidence from scientific literature, providing a compelling argument for the influence of the frequency of high or low flow events upon ecological response, both directly through experience of flow forces, and indirectly through flow influence upon physiochemical riverine properties (Rolls et al., 2012, Lytle and Poff, 2004). It is difficult to isolate these variables from other temporal flow characteristics, however. For instance, the duration of flow events tend to be inversely correlated to flow event frequency. Therefore, it is possible that influence exerted by the duration of flow events also contributes to the variance observed. The influence of flow event duration was also analysed however, and did not demonstrate significant statistical relationships with biological metrics, suggesting that flow frequency is a much stronger influence.

2) How can localised hydraulic requirements of taxa be translated into an ecologically beneficial flow regime?

Chapters 4 and 5 work together to answer this question; the hydraulic requirements of taxa in terms of magnitude can be well represented through the habitat quality metric, derived from species velocity affinities recorded in literature. Through subsequent CASiMiR model predictions, it was

demonstrated that habitat quality predictions correlate well with species population numbers sampled at the same point at the study site. Good correlation was observed in spite of the stochastic nature of species distributions due to external variables such as nutrient availability and biological interactions, providing good evidence that habitat quality is capable of reliably representing magnitude-based hydraulic requirements. Chapter 5 discusses how magnitude alone is neither a suitable nor efficient means to address in-stream taxon requirements. A holistic integration of magnitude-based hydraulic requirements and requirements for the temporal fluctuation of said hydraulics must be considered; the influence of temporal fluctuation cannot be simulated within current habitat quality models and therefore must be translated into ecologically beneficial flow regime design by other means. Chapter 5 demonstrates a process by which this may be achieved, through analysis of regional unmodified water bodies and their temporal flow characteristics to identify typical conditions in the region. It is assumed with conceptual evidence from literature (Poff et al., 1997, Acreman et al., 2014, Wiens, 2002) that this approximation of natural conditions is a significant improvement upon traditional steady compensation flows. This integration of temporal variation into the flow regime alongside magnitude-based provision still requires validation through in-stream flow experimentation, however ecological benefit relative to water released suggests that such regimes are highly efficient based upon HSI metrics and total water released; this is not accounting for further benefit assumed by the more naturalised flow variation.

3) What indicators can be used to interpret habitat quality, given its variation over time and different values between species?

Chapter 5 works to answer this question. It is difficult to interpret holistic habitat quality with great precision through a single metric; this investigation has developed an understanding of expected habitat quality through a combination of individual HSI scores, diversity of habitat, and the extent of site correspondence with natural conditions. Current methods of ecological assessment do not provide a holistic analysis of the state of the ecosystem, and recent studies have called for the development of a broader suite of ecological metrics due to this problem (Poff et al., 2017, Arthington et al., 2018). This investigation affirms and repeats this call, for whilst it has been possible to provide a qualitative assessment of anticipated ecological quality by combining the metrics discussed, it is beyond the scope of expertise of this project to develop a quantitative measure of ecological health integrating all of these systems.

4) How may outcomes such as habitat quality predictions be interpreted in light of legislative targets?

This question remains a challenging one that will likely require further advancement in methodological and conceptual knowledge within the field of environmental flows to properly address. Whilst it is possible to demonstrate expected ecological response and the anticipated implications within the system, it is not possible to predict a quantified ecological response such as absolute shifts in biodiversity or functional composition. Given that current legislative assessment is site-specific and is performed through biological sampling and subsequent comparison with reference conditions or predicted conditions (such as those produced by RIVPACs), it is difficult to conceptualise how suggested restorative measures might translate into legislative targets without first deriving quantitative predictions for how the ecosystem is expected to respond. This would likely require in-stream flow experimentation beyond the scope of this investigation. It should be noted however that

within the systems being investigated - that is HMWBs - water managers have little interest in quantifying the extent to which legislative targets might be met, and often lack the internal expertise to do so; it is the interest of water managers to demonstrate to regulatory bodies that they are doing what can feasibly be expected to mitigate ecological impact within the context of the HMWB, within financial constraints and whilst maintaining the services provided by the system.

7.4 Summary of findings

Chapter 3: Multi-site statistical analysis: Quantitative relationships from a particular region; insight into flow characteristic interactions

Highlights:

1. Flow modification associated with river impoundment is likely to cause a significant deviation from natural, native macroinvertebrate population compositions.

2. In addition to flow magnitude, the temporal characteristics of flow also play a key role in maintaining ecological stability.

3. It is possible to identify specific influential flow characteristics driving macroinvertebrate population composition, and quantify these ecology-flow relationships.

Chapter 4: Model Development: Framework for assessment with potential for transferability

Highlights:

1. The localised hydraulic forces brought about by a given compensation flow, experienced by taxa, can be simulated through the consideration of taxa flow requirements using a metric for local habitat quality.

2. Predictions of the habitat quality metric will generally correlate with observed species populations in the field under the same flow conditions; e.g. high habitat quality is associated with high species population numbers.

Chapter 5: Model Implementation: Framework for integration, with outcomes from case study site

Highlights:

1. A magnitude-based ecological model is not sufficient to provide mitigation solutions for the impacts of flow impoundment modification.

2. A degree of flow naturalisation is a necessary part of environmental flow design when such implementation is feasible.

3. A holistic framework considering flow magnitude and temporal characteristics enables the design of optimal environmental flow regimes.

4. Designer environmental flows may increase ecological metrics whilst also conserving impounded water resources.

7.5 Research Implications

Flow regime modification due to impoundment acts as a pressure upon riverine ecology through a number of direct and indirect drivers. Chapter 2 highlighted the importance of gaining a greater understanding in the relationship between these drivers and ecological response, and also emphasised the challenge in the translation from current conceptual understanding into practical investigatory frameworks, due to the highly complex nature of rivers and the numerous interdependent variables that contribute to the overall state of the system. This difficulty is further exacerbated by the lack of a cohesive approach to flow investigation; it has been concluded that much literature cannot be compared with one another or synthesised into general understanding due to the high site-specificity of many investigations and the lack of uniform methodologies. This project set out to further understanding in this area on a more general transferable level, as opposed to being merely site-specific. Chapter 2 may serve to guide future research in a similar manner to how it has directed this thesis. Studies should aim to move away from highly site-specific investigation, and work towards frameworks that are easily transferable between systems. Currently, studies are tending towards regional-based investigation. This opens up many research opportunities, as every new region studied has the potential to expand general knowledge of ecology-flow relationships, either through similarities or differences between other regions. With sufficient regional studies, the field may in future form universal principles through a meta-analysis of regional findings across the globe.

This thesis has established that the riverine system is a complex web of interactions (Summers et al., 2015) and the ecosystem is likely to be driven by many interacting variables including flow regime, sediment regime, thermal regime, shade, nutrient availability, and species interactions. The models that were developed in Chapter 3 are an attempt to identify key influential characteristics within the flow regime; they do not claim to perfectly predict ecological response in its entirety, but rather examine the extent to which particular flow characteristics influence this response. In instances where a model of high explanatory power was not found, this is likely due to the limited scope of this investigation, and the influence of external variables not examined in this investigation, such as those mentioned above. The Chapter has affirmed the hypothesis presented by the likes of Poff et al. (1997) and Richter et al. (1996), that there are key flow categories that strongly influence ecological response, and that significant predictive relationships can be found in many circumstances on a regional scale (Arthington et al., 2006). This supports the general assumption within the field of environmental flows, though transferable empirical evidence is still not as common as desired; the assumption that the magnitude, timing, duration and variability of flows directly influence biodiversity and functional composition. It can be inferred from this observation that highly modified flows, such as those observed within impounded systems, are liable to foster an ecosystem that diverges significantly from the waterbody's natural state. This conclusion is similar to findings from other studies that have investigated the effects of increasing flow modification and regulation upon downstream ecosystems (Gillespie et al., 2015a, Haxton and Findlay, 2008, Nichols et al., 2006). As far as I am aware, no other studies have demonstrated definitive hydrological response relationships for macroinvertebrates for this particular region and magnitude class of river. Data from this study may be used for water management decisions across the North of England, to supplement future studies in the region, or to draw comparisons between other river classes. It could also be tested to what extent such relationships hold true beyond the region studied; throughout England, or even throughout temperate climates across Europe.

The exhortation of recent literature to standardise and present a greater body of transferable data through focused studies is not yet showing itself to be evident; the most recent studies on environmental flows rarely focus on this drive towards general regional ecology-flow principles. Those that do are spread across different fields such as hydropower or abstraction and are of limited transferability between one another due to the significant differences in the nature of flow modification taking place. A literature search for recent studies between 2017-2018 shows a number of site-specific studies, or studies with focuses very different to this investigation, such as flow regulation impacts upon floodplain rivers (Hayes et al., 2018), riverine impacts during dam construction (Santos et al., 2017) and fish trait analysis in the context of impoundment (Lima et al., 2017), in addition to reviews of recent progress in the field (Poff et al., 2017, Arthington et al., 2018). With such a diverse breadth of potential research, many studies with varying focuses must orientate themselves towards developing generalised flow-ecological relationships within reservoir-impounded systems, both for fish and macroinvertebrate species and for different regions, environments, and river classes.

Chapter 4 has been concerned with developing and calibrating both a hydraulic and an ecological model of a case study site, for the purpose of predictively assessing the impacts of flow regime upon the native macroinvertebrate ecosystem, with the purpose of optimising the ratio of water spent to ecological provision provided, balancing the conflicting interests of environmental needs and societal use. It has been demonstrated that the hydraulic model developed is capable of predicting flow velocities across the river reach to an acceptable accuracy, though some discrepancies exist at more extreme high flows, likely due by sudden changes in geometry at a spatial scale smaller than the resolution of the model, and the fact that significant uncertainty is generally involved in river velocity measurements (Acreman et al., 2014). The ecological model was also calibrated, and outputs were found to be in generally good agreement with model observations. This study acknowledges the significant degree of stochastic distribution present in ecological populations, and reasons for this have been discussed in Section 4.7.2; biological interactions and the tendency of some species to cluster together are examples of how population distribution may not be entirely driven by habitat suitability. The primary indicator of CASiMiR predictions being sensible is the fact that areas of poor HSI do not have any significant species populations within observed data, whilst moderate to good HSI areas have varying distributions of a given species. Such a result was expected; the stochastic nature of distributions means that "Good" HSI is not guaranteed to correspond to high population numbers due to habitat being only one influencing factor upon species populations, however, "Poor" HSI should almost always correspond to low species numbers due to the inhospitable environment. Species occurring in significant numbers within poor HSI areas would have caused concern about the validity of the model, but it has been demonstrated that poor HSI consistently corresponds to low observed population numbers for all indicator species. Despite external complexities, graphs in the Results section demonstrate clear trends between increasing HSI and population number for all 3 species.

It was confirmed that model outputs are valid for the Holden Wood site, and have sufficient reliability to generate insights into ecological response to flows. However, this study has been met with a number of limitations, and model accuracy and robustness could certainly be improved upon in future work, allocating more time and resources to take more measurements and create a finer-scale model. The CASiMiR model provides predictions of habitat suitability, not of defined species numbers. Efforts to improve habitat suitability, in the opinion of this investigation, represents the best effort for ecological provision, short of channel-invasive methods such as direct channel modification or direct

interference with the ecosystem (e.g. introducing or removing particular species) which are both cost- and labour-intensive, whilst also risking unexpected and possibly irreversible harm to the native species.

With the models successfully calibrated and tested, the outputs of the CASiMiR model were evaluated and acted as the primary source of information for individual species provision (based upon sensitivity thresholds) and spatial habitat response to flow (habitat diversity, connectivity and persistence) from which principles for environmental flows at Holden Wood were developed. This represents a significant step forwards in the task of bringing together knowledge of ecology-flow response relationships in order to optimise ecological provision in terms of timings and appropriate magnitudes. Other temporal drivers such as flow event frequency and duration are not considered by CASiMiR and are incorporated through insights gained from statistical analysis covered in Chapter 3, and additional knowledge from literature discussed in Chapter 2. The outputs of the CASiMiR model has justified the use of depth-averaged flow velocity is an acceptable surrogate for near-bed forces in the case of small-scale, shallow river systems. HSI predictions have been shown to correspond well with field measurements of species population distributions, and because of this we can have confidence in the assumption that changes to predicted HSI are likely to denote significant changes in macroinvertebrate populations. Therefore, one of the goals of subsequent flow design – to optimise HSI benefit from reservoir releases – is a well-justified one. This Chapter formed the basis of Chapter 5 in which flow regimes were designed, with good foundations and confidence that the outputs of our developed model are sensible, and changes to said outputs should be expected to elicit ecological change accordingly.

This study affirms use of 2D modelling, particularly for small-scale sites where vertical complexity is minimal and an optimal approach is necessary due to resource constraints. Designed regimes A, B and C promote ecological provision with varying prioritisation, demonstrating the utility of this framework; it is possible to define design goals, and these may be adapted to accommodate water demands and diverse interests of stakeholders present within a given system. A key next step in this research would be to fully validate the model through physical flow experiments, obtaining field observations at the flow magnitude range used in the model and comparing them with model outputs. Once validated at the study site, this approach could be applied to similar river systems within the region of Northern England, utilising the proposed regimes with some adaptations for site-specific contexts. Should the outcomes in these systems again be validated, the transferability of this approach may allow for smaller scale impoundments across the UK to implement environmental flows, where previously this was unfeasible due to the quantity of impoundment systems and the intensity of labour required to assign environmental flows to individual sites.

This thesis currently focuses upon application for sites impacted by impounding reservoirs; it could be possible to adapt it for use in other site restoration assessments such as hydropower-impacted sites by incorporating the unique challenges and priorities of the given modification into the design stage of the environmental flow regime. An example of such a consideration would be the necessity of disruptive high flows from hydropower releases; perhaps a regime design for such an application may focus upon dampening and prolonging these high flows according to what is feasible without compromising the service of the dam. It is also acknowledged that flow is not the sole driver of ecological response, and other stressors such as water chemistry likely play a significant role at many sites. It has been suggested that the diverse influences of riverine ecology must be studied both

through short-term mechanistic experiments and long-term explanatory studies in order to disentangle this complex web of interactions (Laini et al., 2018). Climate change and land use change are also resulting in a shifting environment, further driving changes in ecological metrics (Li et al., 2018). As understanding of these interactions grows, it would be possible to integrate further mitigation methods into the framework presented in this study. The ability to integrate ecological requirements according to context, and make adjustments according to new knowledge, offers significant utility within this framework.

An approach has been presented by which a study site is assessed and environmental flows are proposed based upon a combination of species response to flow (through preference curves), the influence of magnitude upon habitat diversity, and typical unregulated regional flow characteristics in order to form a holistic ecological solution. Results suggest that uniform increases in magnitude over long periods result in disproportionately little ecological benefit relative to volume of water released, and affirms the use of optimised and targeted high flow events. Though there is a rich literature detailing the concepts considered, we are not aware of any studies suggesting a similar framework by which such a combined range of flow requirements within a particular site or region may be assessed. Poff et al. (2017)'s update on the evolution of environmental flow science discusses progress in almost every area, yet there is not yet a unified approach to environmental flow assessment. They emphasise the need to extend from a local scale to basin-scale perspective (Poff et al., 2017). Previous regional frameworks have been attempted, but generally have used singular metrics of habitat suitability (Ceola and Pugliese, 2014). More recently, a framework for the strategic allocation of water in order to balance environmental flows and societal needs has been proposed (Sabzi et al., 2019), but within the scope of generalised "environmental considerations" which must be determined on a case by case basis.

Amidst this rapidly developing field in which numerous frameworks and methods for environmental flow assessment are emerging, this study offers a novel approach in optimal annual flow regime designation, aiming towards regional transferability. Whilst many previous studies have focused upon specific ecological response from a given study site, or have focused upon the conceptual development of particular aspects of ecological response, this study provides an uncommon perspective in drawing together knowledge accumulated through recent academic progress, and combines this with the considerations of practical constraints and stakeholder interests faced by the water industry. We offer the first steps towards an actionable regional water management solution to the issue of impoundment-modified flow impacts that is desirable both for the purpose of ecological and water resources conservation. There is scope for this framework to be scaled up to larger river systems, though this would require the incorporation of other variables significant at such a scale, such as substrate type and variation and the sediment transport regime. Fish may also be considered in CASiMiR should they be present in the system. This investigation suggests that 2D habitat modelling remains a tool with great potential when incorporated into such holistic practices, and shows great promise as water managers move into transferable, regional-focused forms of investigation.

7.6 Concluding Remarks

This thesis began with a single question; how can we better understand the relationship between ecology and flow within impoundment-modified systems? Literature in this field guided the focus of

this investigation into two avenues; a regional statistical analysis of similar river systems aiming to identify key flow characteristics driving ecological response, and the design and implementation of a 2D predictive model by which improved flow regimes might be suggested at a case study site. This thesis has affirmed the study of regional ecology-flow relationships as a promising ongoing area of investigation, and has demonstrated a potential framework by which environmental flows may be designed through the requirements of indicator species, widely held ecological principles, and typical local or regional conditions through the proposal of environmental flow regimes for a case study site.

A breadth of principles and lessons from recent literature within the field of ecohydraulics have informed this investigation, and this thesis has drawn these together in order to develop a novel approach towards the study of ecology-flow relationships within impounded systems. Transferability has been a key focus throughout this investigation, as this is a pressing need within the field (Arthington et al., 2006, Arthington et al., 2018, Gillespie et al., 2015b), therefore both the statistical analysis and case study utilised more generalised concepts applicable to a region or transferable to similar sites. Chapter 3 utilises IHA-style flow characteristics that can be processed from flow data in any river system, along with species affinities that should also be accessible given the availability of ecological sampling data, and from this regional principles are drawn through quantified linear modelling relationships. Chapters 4 and 5 utilise river hydraulics, indicator species velocity affinities, general ecological principles (such as the ecological benefits provided by natural flow variation), and typical unmodified regional flow characteristics within a given magnitude range. All of this information is widely accessible, with the most labour-intensive processes being the mapping of local geometry for hydraulic modelling, and field sampling for indicator species at the study site.

A consideration that one might raise is that of indicator species, and this leads into a philosophical question posed within the field of environmental flows as a whole; given that the system in question is already modified, and lacking natural reference conditions, can we have confidence that the indicator species that have been used here reflect the biological composition of the system if it was not modified? Are they suitable guides towards improving ecological health? Given that the indicators utilised are species commonly found throughout UK waters (NatureSpot, 2015), and given that flow modification by impoundments has not been found to alter the biological composition of a system to such a drastic extent as to be unrecognisable relative to natural conditions (Gillespie et al., 2015a, Poff and Zimmerman, 2010), I am confident that the indicator species reflect taxa that would be found within a similar, natural, system. Additionally, indicator species primarily inform the magnitude requirements of the ecosystem; these call for an increase in magnitude, which is in agreement with the reservoir inflow data that demonstrates the higher magnitude flows of the system prior to meeting the impoundment. Other aspects of the flow regime such as the duration and frequency of flow events were designed through the analysis of other river systems within the region, and are therefore not related to the question of ecological indicator choice.

Important next steps from the findings in this thesis would be first to fully validate one of the proposed flow regimes depending on desired prioritisation between ecological provision and water conservation, through in-field flow experiments. Should the proposed regime demonstrate desirable outcomes, impounded systems of similar scale and geography might be identified and local ecology investigated. The general flow design from Holden Wood may be adapted new sites according to their channel geometries and the requirements of the native species, though one would expect species composition to be largely similar if the two sites are in close proximity. Ecological response at such

sites may then be examined, and should outcomes again prove desirable then adaptations of this regime might be implemented across Northern England with minimal site-specific investigation.

Another area of interest could be to investigate how flow regime design might differ under this framework when applied to another region, such as mainland Europe, or for a larger scale of river, which could possibly contribute to the development of universal principles for flow impoundment mitigation and restorative environmental flows. A number of adaptations to the proposed methodology are likely required should it be scaled up to assess larger systems. Due to the smaller scale at the current study site, a number of simplifications and assumptions were made such as depth and sediment type not being impactful variables, as detailed in Chapter 4, along with a lack of significant amounts of vegetation or the presence of fish meaning that in-stream macrophytes and fish were not considered within ecological assessment. Within a larger system, these variables are likely to be of importance and thus should be considered. CASiMiR is capable of integrating features such as sediment type and fish if such data is provided, but hydraulic modelling in this case would need to shift to 3D software, and vegetation hydraulics may need to be considered. Additionally, larger sites require increasingly more labour with size due to the need to survey river geometry and obtain proper site coverage of in-field flow and ecological measurements. It may be the case that the approach taken in this investigation could have other methods integrated into it from past approaches more focused upon large-scale systems, such as expert workshops that form the core of the Building Block Method (King and Louw, 1998), in order to assess the flow requirements of larger sites in a more rapid and efficient manner, whilst maintaining this approaches focus upon possible transferability to similar systems, and an emphasis upon flow optimisation for water resources conservation.

It is hoped that regional ecology-flow relationship studies continue to expand upon the current body of knowledge within the field of ecohydraulics, and that developments such as the integration of previously disparate concepts such as species trait analysis, IHA metrics and regional-based investigation become common practice in future studies. Such ideas aid in increasing the standardisation of methodologies within this relatively young field of science, and in turn maximise the transferability of research and its potential for synthesis into the broader knowledge-base. It is also hoped that water managers will take note of the potential of the proposed environmental flow design framework that has been presented, given the significant implications for the water industry and riverine ecology due to the possibility of providing for ecological needs in a manner that conserves water resources when compared to the traditional approach of constant, unvarying compensation flows. The findings of this thesis and the conceptual advances made are likely to be of use in a world where (i) water resources are an increasingly stressed resource due to growing populations, changing land use, and a changing climate, and (ii) contemporary legislation continues to evolve and increasingly will pressure water managers to understand and mitigate the impacts of impoundments and other flow-modifying systems.

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9. APPENDIX

9.1: The following is a published Paper accepted by Ecological Indicators (Elsevier), covering research from Chapters 4 and 5, co-authored by myself and my two supervisors, Dr. James Shucksmith and Prof. Phil Warren.

Designing an environmental flow framework for impounded river systems through modelling of invertebrate habitat quality*

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ABSTRACT

Many rivers have undergone flow modification by impoundments to provide services such as water supply and hydropower. There is an established consensus that typical modified flow regimes do not sufficiently cater to the needs of downstream ecosystems, and more must be done to understand and mitigate their associated impacts. This study presents a novel, transferable framework by which a small-scale impoundment in North West England is assessed through the use of linked hydro-ecological modelling in SRH-2D and CASiMiR, utilising flow velocity measurements and macroinvertebrate sampling data. Model predictions of habitat quality were supplemented by established ecological principles such as the importance of flow heterogeneity. Results are used to design environmental flow regimes, with the aim of improving ecological metrics whilst considering conflicting water demands. Based on an analysis of historical flow records, the implementation of designer flows over a 12 month period demonstrated increased peak species habitat qualities of 23–26%, characteristics such as flow heterogeneity were more naturalised, and 22% less water was released from the impoundment. Should outcomes be validated by in-stream flow experiment, there is great potential for further development and application of this method,

including regional transferability for the rapid designation of environmental flows across a number of sites of similar magnitude and geography.

Keywords

Environmental flows; Flow impoundment; Macroinvertebrate; Eco-hydraulics

1. Introduction

Flow modification and impoundment of river systems has become widespread throughout the world in response to increasing water demand and energy requirements. Over recent decades it has increasingly been recognised that typical flow regimes imposed even by small impoundments and hydropower schemes may have impacts on riverine ecology (Anderson et al., 2015, Poff et al., 1997, Summers et al., 2015). It is thus important to understand the relationship between flow regime and ecological response, and develop efficient frameworks to mitigate the impact of flow modification. The needs for impoundment are unlikely to decrease, hence a key question is how we might maximise environmental benefit for a volume of water released as impoundment outflow (Konrad et al., 2011). Since its proposal in the late 1990s, the Natural Flow Paradigm (Poff et al., 1997, Acreman et al., 2009) has formed the basis of the environmental flow concept. Poff et al. (1997) discuss the likely consequences of the alteration of natural flow components such as flow heterogeneity and the resulting ecological response within the system, and propose that natural flows promote stable ecosystems, whilst over-regulated systems result in ecological impacts due to direct and indirect responses to altered flow. Examples of natural flow variation include flows driven by predictable seasonal precipitation levels, or snow melt (Junk et al., 1989, Junk and Wantzen, 2004). Poff et al. (1997) argue that there is an intrinsic link between the natural flow regime and in-stream ecology as a result of the biota having developed life-history, behavioural and morphological characteristics adapted to be successful in their native environment. In contrast, impounded systems have traditionally based their regulated outlet flow regimes upon "rule of thumb" percentile-based values such as the Q95 (the 5th percentile flow) of the non-modified river system (Arthington et al., 2006), or upon historical flow licences that had been in place to sustain downstream interests such as mills, many of which no longer exist (Gustard, 1989). Such flows neglect the natural fluctuation of flow, and some behavioural and

morphological adaptations of biota may no longer be appropriate for their environment (Lytle and Poff, 2004).

While it may not be feasible to return flows to their natural regimes in most cases, an increasingly popular approach has been that of 'Designer Flows', by which flow patterns are created to provide desired benefits, within practical constraints (Chen and Olden, 2017). Designed environmental flows are unlikely to match the variation of their natural counterparts, either in magnitude or heterogeneity, but can significantly improve ecological quality. This can be achieved by accounting for particular ecological requirements such as periods of elevated flow, and integrating them into the flow regime. The "Building Block Method" (BBM) approach proposes that such requirements can come together as individual "blocks" to create an overall regime, originating from South African restorative studies and later seeing international application (King and Louw, 1998). The UK advisory group UKTAG have discussed BBM in recent years and propose the approach as best practice for the mitigation of impacts arising from impoundments (UKTAG, 2013). Despite such conceptual frameworks, the implementation of environmental flows remains a major challenge; this is largely due to the lack of any defined, standardised protocol by which these flows are to be implemented. Part of this difficulty may be that most studies have focused upon the investigation of larger river systems; it is difficult to isolate ecologically-influential factors at this scale (for example due to tributary flow inputs). This study utilises a small scale study site to allow the development of a foundational approach towards environmental flow development that can later be scaled and adapted to account for further complexity in larger systems.

This study utilises macroinvertebrate species as ecological indicators, due to their relative neglect in the field when compared to taxa such as fish (Gillespie et al., 2015b), and the fact that they are a more prolific indicator at small scale sites. These taxa experience flow as localised forces as opposed to overall magnitudes, timings, etc. This raises the question, how can the requirements of invertebrates on a micro scale translate into an overall compensation flow and its inter-annual variation? Habitat quality models are an increasingly utilised approach in restorative studies (Reiser and Hilgert, 2018, Schneider et al., 2016, Conallin et al., 2010), yet may not account for life history requirements and temporal flow characteristics experienced by taxa, such as the frequency and duration of flow events. A broader suite ecological indices are required to achieve robust environmental flow designs (Chinnayakanahalli et al., 2011, Arthington et al., 2018), and methodological progress is

required in order to determine their implementation; how are conflicting flow needs to be resolved, and how does one judge whether or not a flow regime is "good"?

Competing interest groups and increasing demands for water supply mean that environmental needs are a contentious topic; water sent downstream for environmental purposes must be well-justified, and the "cost-benefit" in terms of water committed to environmental flows must be acceptable in order to maximise the volume of water retained for societal use (Harwood et al., 2018). The lack of transferable ecology-flow principles can necessitate time- and cost-intensive site-specific investigation (e.g. Anderson et al., 2017), and the impracticality of scaling up such an approach to larger or multiple sites is readily apparent. One potential solution gaining favour is to use regional-based methods (Summers et al., 2015). These recognise that whilst general principles may remain elusive, it should be possible to identify commonalities between approaches for rivers of similar magnitude and geography (Arthington et al., 2006). However, even these relationships have proven difficult to extract from the current body of literature, largely due to a lack of standardised approaches and challenges in the synthesis of current data (Poff and Zimmerman, 2010, Gillespie et al., 2015b).

This paper presents a potential transferable methodology by which impoundment-modified river systems may be assessed, and environmental flows designated. Here, we test this method of environmental flow designation at a case study site, addressing the challenge of site-wide flow regime designation through a novel combination of habitat quality prediction (based on 2D ecological model outputs), flow event timings, habitat diversity, and flow event heterogeneity, whilst also making efforts to actively conserve water relative to current outflows, with a methodological design emphasising future transferability to other sites. The proposed methodology takes steps towards an answer for generic environmental flow designation and implementation based on the principle that designed flows should provide significant benefit to the ecosystem (Richter et al., 1996), whilst also conserving water resources.

2. Site description

The study site Ogden Brook (Fig. 1a, Fig. 1b) is a stream system in the North West of England, directly downstream of the impounding reservoir Holden Wood, approximately 27.2 km North of Manchester, Northern England, and located near the village of Haslingden, OS grid reference SD776220. The site was chosen due to data availability, lack of significant external

pressures such as wastewater inflows, the dominance of upstream reservoir compensation flows, and its appropriate scale for the scope of the investigation. Historical background information has been adapted from a consultancy report provided by the regional water company, United Utilities (APEM, 2016).
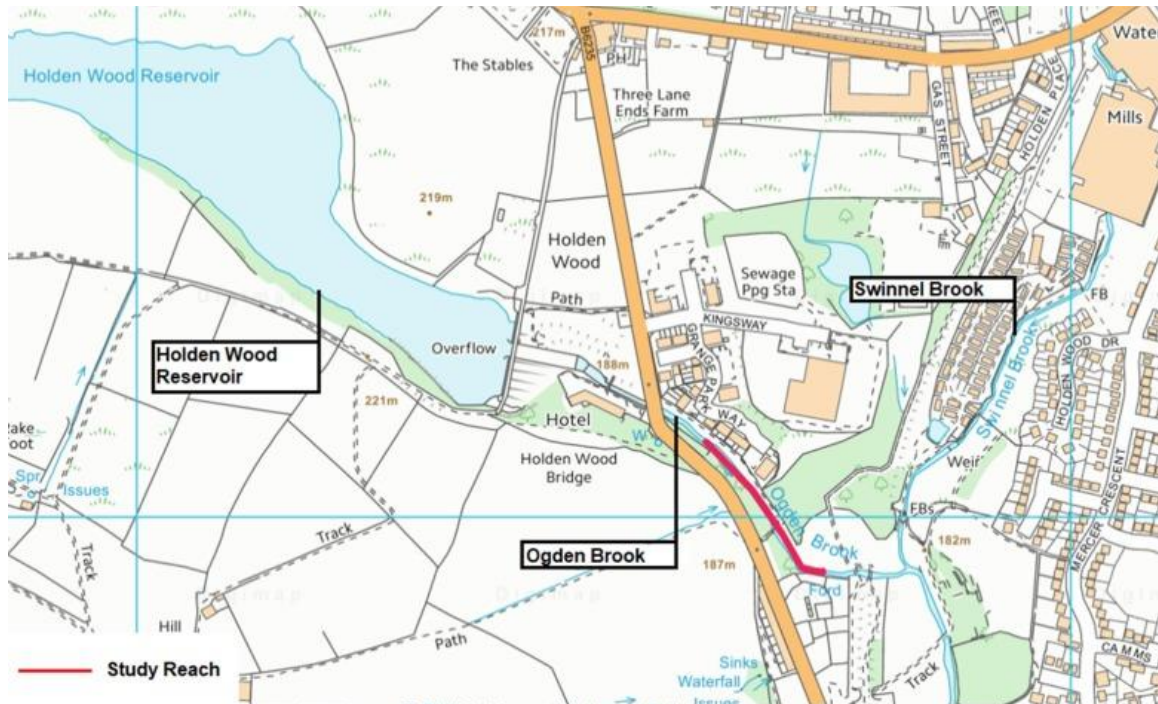

Fig. 1a. The Holden Wood reservoir and Ogden Brook, (Digimap Ordinance Survey Service, 2018).

Fig. 1b. A cross-section at the Ogden Brook site, pictured in November 2017.

Typical flow conditions at the site remain in the range of Q = 0.02–0.04 m³/s, with mostly shallow depths between 0.1 and 0.25 m, though recorded depths in pools reached as high as

0.8 m. The reach under investigation is approximately 40 m in length, primarily chosen to avoid the presence of a downstream tributary, so as to retain flow contribution solely from reservoir outflows. The study is thus performed on a small scale; this fits the aims of this investigation, which focuses upon the response of taxa at a micro-habitat level similar to studies such as LeCraw and Mackereth (2010) who utilised study reaches of 10 m to observe localised ecological responses, or other fish and macroinvertebrate restoration studies utilising 100 m reach scales (Pretty et al., 2003, Harrison et al., 2004).

The water company United Utilities manages Holden Wood Reservoir. Compensation flow releases are made to the Ogden Brook, in line with the conditions of an impoundment licence granted by the Environment Agency. Ogden Brook runs along a narrow band of woodland surrounded by light urban development. The riverbed itself is mostly gravel, with top-layer sediment ranging from small pebbles ($\sim$1 cm) to larger stones (up to $\sim$10 cm), with a few larger rocks (up to $\sim$20 cm) scattered throughout the reach. A lower layer of finer sediment lies beneath the stones and cobbles. The river channel has little to no vegetation. Presently, Holden Wood is required to release 3.46 ML per day (0.041 m^3/s) of flow during times of the impoundment being within 2 m of its maximum water level, and 1.84 ML per day (0.0215 m^3/s) when water depth is below this point. Prior to 2016, release requirements were lower; within the range of 0.01–0.02 m^3/s. Impoundment releases are the sole major contribution to the studied reach of Ogden Brook, aside from small amounts of direct runoff insignificant relative to overall flow. Mean daily flow data for outflows from Holden Wood between 2014 and 2017, and inflows between 2010 and 2014 were provided by United Utilities, derived from cumulative inflow and outflow metres read and recorded daily; an outflow meter on the spillway measures the volume of reservoir spill events when these occur, and both outflow metres are added together for overall reservoir outflow. Macroinvertebrate single-point, three-minute kick sampling data from spring and autumn of 2016, taken within the analysis reach, were provided by United Utilities; this data was used to assess typical seasonality of native taxa.

3. Methods

An ecological model was constructed using the CASiMiR model (Schneider et al., 2010) to develop an understanding of the macroinvertebrate response (habitat quality) to flow at the site. This required the development of a hydraulic model of the site in order to determine the velocity regime. River geometry, velocity and ecological data was gathered for model development and calibration. Once model accuracy was assessed, habitat predictions were

utilised and supplemented by an integrated consideration of taxon requirements (habitat quality metrics and anticipated responses to temporal flow characteristics) in order to design potential environmental flows for the Holden Wood site. These designer flows were compared with past and current impoundment outflows in terms of flow event characteristics (e.g. flow variability) and impact upon predicted habitat quality in order to demonstrate the differences in ecological response between designer flows and typical compensation flows, relative to annual volume of water released.

3.1. 2D hydraulic modelling

The SRH-2D (Sedimentation and River Hydraulics) modelling package was used to develop an understanding of the hydraulic complexity of the study reach. SRH-2D is based on the numerical solution of the two dimensional depth averaged St. Venant equations, providing calculations of depth and velocity at each computational cell based on model boundary conditions, reach topography and bed roughness (Lai, 2008). SRH-2D has recently seen widespread use in the field of river restoration and eco-hydraulics (Erwin et al., 2017, Stone et al., 2017, Lane et al., 2018).

Bed elevations at the study site were obtained using a Total Station surveyor (Leica Geosystems, 2009). Bed elevations were taken using a scatter-based method, taking elevation readings that adapted in resolution according to bed complexity. A total of 2069 geometry data points were collected over the reach. Bed elevations were uploaded into the SRH-2D model using the SMS interface (Aquaveo LLC, 2013) and a fine mesh was generated with cell sizes approximately 30×30 cm. In particularly complex rivers, meshes as fine as 10×10 cm have been utilised (Lange et al., 2015) however most ecological studies using SRH-2D have used 30×30 cm mesh sizes for detailed sections, with typical mesh sizes of around 250×250 cm or higher in larger rivers (Bandrowski et al., 2014, Stone et al., 2017, Lane et al., 2018).

Model calibration was performed using direct velocity measurements, utilising a Nortek Vectrino Acoustic Doppler Velocimeter (ADV), which is typically expected to provide velocity values accurate to within 5% in field conditions (Dombroski and Crimaldi, 2007). The ADV probe was secured to an adjustable surveying tripod, allowing for stable positioning at any point of measurement. The probe was capable of taking simultaneous measurements of three orthogonal velocity components at a frequency of 20 Hz, hence providing temporally averaged velocity data as well as standard turbulent statistics. A convergence test was

conducted to determine an appropriate sampling period for the acquisition of reliable data at each point. A resulting sampling period of 60 s was used, due to low hydraulic complexity with readings typically stabilising within 30 s of deployment. For each measurement, the probe was orientated as such to obtain primary (x) velocity in the main channel direction (with the y dimension normal to the river bank). Raw ADV data was processed in WinADV 32 (Wahl, 2000), and the phase space threshold de-spiking filter was applied prior to data analysis (Goring and Nikora, 2002).

Eight cross-sections were measured along the reach, with flows being taken at 3–5 points along each cross-section depending on channel width. Measurements were taken at 0.6 of the depth to obtain a depth-averaged reading (Hewlett, 1982). A total of 31 readings obtained allowed for moderate coverage along the entire reach at a high resolution relative to many studies; SRH-2D has been successfully calibrated in larger rivers with significantly fewer observation points (Deslauriers and Mahdi, 2018). At the time of measurement, flow into the river was measured as 0.024 m^3/s, based on impoundment outflow data provided by the site operator. This discharge is generally consistent throughout the autumn season, unless the impoundment is close to capacity, at which point flow releases are elevated and spill events are possible.

Upstream and downstream model boundary conditions were established based upon straight, stable areas of flow within the study reach. The upstream boundary condition was set as the measured inflow (0.024 m^3/s), and the velocity was defined using SRH-2D's Conveyancing approach in which flow direction is assumed to be normal to the inlet boundary (Lai, 2008), and the velocity is uniformly distributed. The downstream boundary condition was set as the measured water level (185.02 m above sea level), again assuming flow normal to the boundary. Manning's roughness values were initially assigned with appropriate ranges based upon literature values (Chow, 1959) based on substrate type at the site.

Manning's roughness values for the river channel were calibrated based on established best practices (Van Waveren et al., 1999) initially testing homogeneous roughness across the entire reach, and later adjusting small areas where observed changes in substrate led to discrepancies in velocity. Final calibration saw the majority of the river assigned a Manning's value of 0.05, whilst patches of the riverbed had roughnesses ranging from 0.04 to 0.07. These values are appropriate for streams with generally little vegetation, steep banks, trees and

scrub at the banks, and cobbles and large stones within the channel (Chow, 1959). Fig. 2 presents post calibration model outputs in terms of predicted flow velocity vectors.
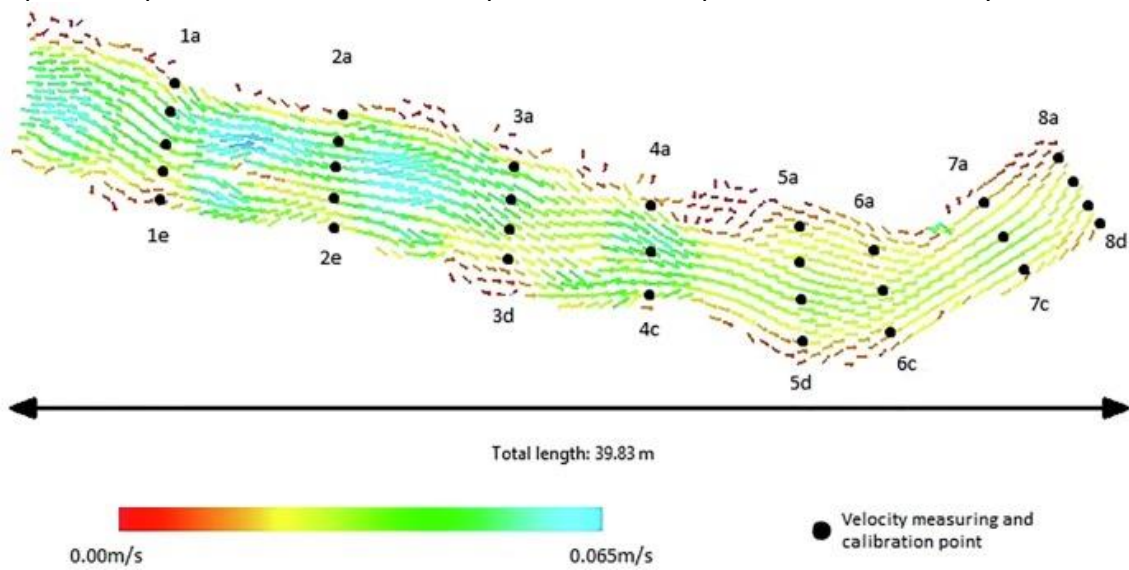

Fig. 2. SRH-2D hydraulic predictions, post-calibration, showing predicted velocity in m/s for an inflow of 0.024 m³/s, with in-field velocity measuring point positions.

Model predictions of calibrated depth-averaged velocity were tested by comparison with field point-observations of primary, temporally-averaged flow velocity taken by the ADV. Observed and modelled primary (x dimension) velocity values are plotted in Fig. 3. It can be seen that there is broadly good agreement between predictions and observed values. Anomalous readings tend to be at the highest ranges of velocity, which may indicate deviations in model predictions at higher flows. However, these high-velocity anomalies may also be caused by localised changes in bed geometry, either not accounted for at the mesh scale used, or not detected during bed geometry measurements, such areas of faster flow (>5 cm/s) may be highly localised and difficult to account for; for instance above a large rock causing a small shallow area of increased velocity, or a cleft between stones through which flow is funnelled. The most erroneous point, 3c, had been noted during field velocity measurement to be an area of particularly fast and complex local flow due to the presence of nearby rocks. It is possible that errors also arise from inaccuracies inherent to characterisation of the depth-averaged velocity at a single measurement point, which may be more significant in areas of irregular topography or cross currents which lead to complex velocity distributions.
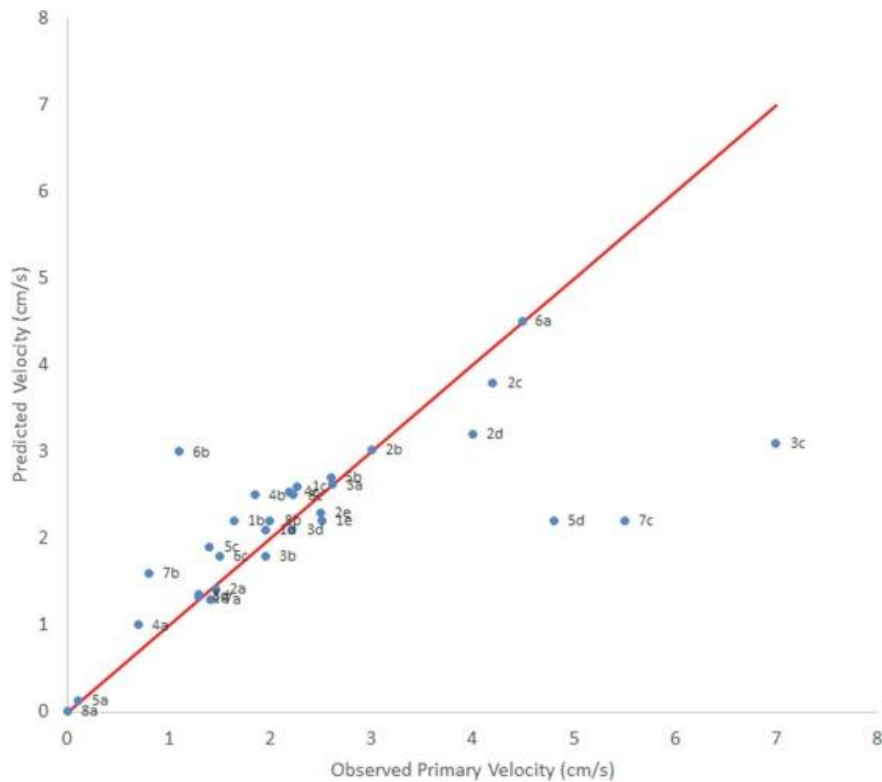
Fig. 3. Post-calibration SRH-2D primary velocity vs observed field primary velocities with field measuring points and associated model predictions (see Fig. 2 for spatial positions).

3.2. 2D ecological modelling

The CASiMiR model framework is modular and integrates hydraulic and structural parameters from a hydraulic model for the calculation of habitat suitability for indicator organisms. Aquatic habitat suitability in this study is derived by the use of univariate flow velocity preference curves, and this is later compared with species population distributions observed in the field (Schneider et al., 2016). Preference curves were based on flow velocity affinities found in the STAR Project, a large-scale investigation supported by the European Commission in order to resolve challenges posed by the Water Framework Directive, using the study "Deliverable N2" (Bis and Usseglio-Polatera, 2004). This study involved the aggregation of macroinvertebrate traits into one of the largest species trait databases available (Bis and Usseglio-Polatera, 2004). In the STAR project, velocity preferences are described in the range of Null (0 cm/s); Low (>0–25 cm/s); Medium (>25–50 cm/s) and High (>50 cm/s) based upon flow affinity, i.e. how well a species is adapted to particular flow conditions. Affinities range from 0 (lowest) to 3 (highest). These affinities were interpreted into Habitat Suitability Index (HSI) values ranging from 0.00 (lowest possible affinity) to 1.00 (highest possible affinity). In this study, flow velocity was selected as the sole parameter for driving ecological response. Depth and substratum are also used as key parameters in larger river systems, but at the scale

investigated at this study site substratum can be assumed to be homogeneous, and changes in depth are not significant in terms of macroinvertebrate sensitivity.

CASiMiR can be calibrated through small adjustments to preference curve inputs (Schneider et al., 2010), due to possible variations in biological behaviour from site to site caused by external drivers. This was not necessary for this study due to species behaving in accordance to established preference values. The model was tested using observed species sample populations, taken using the standard 3-min kick sample method (Murray-Bligh, 1999) in November 2017 at a flow rate of 0.024 m^3/s. 15 measurements were taken using single-point kick sampling from a range of microhabitats distributed across the reach. Habitat predictions were then generated based upon the same flow rate. Testing under a single flow condition was deemed reasonably justified due to the minimal variation of flow at the site, and the fact that samples demonstrated similar species composition proportions to those observed in 2016 sampling data provided by the consultants (described in Section 2). Three species, *Gammarus pulex*, *Polycentropus flavomaculatus*, and *Hydropsyche siltalai*, were chosen for model testing based upon their occurrence at most sample sites, and their range of flow preferences. A comparison between model predictions in the form of HSI, and observations in terms of species sample populations at the same point, is presented in Fig. 4.
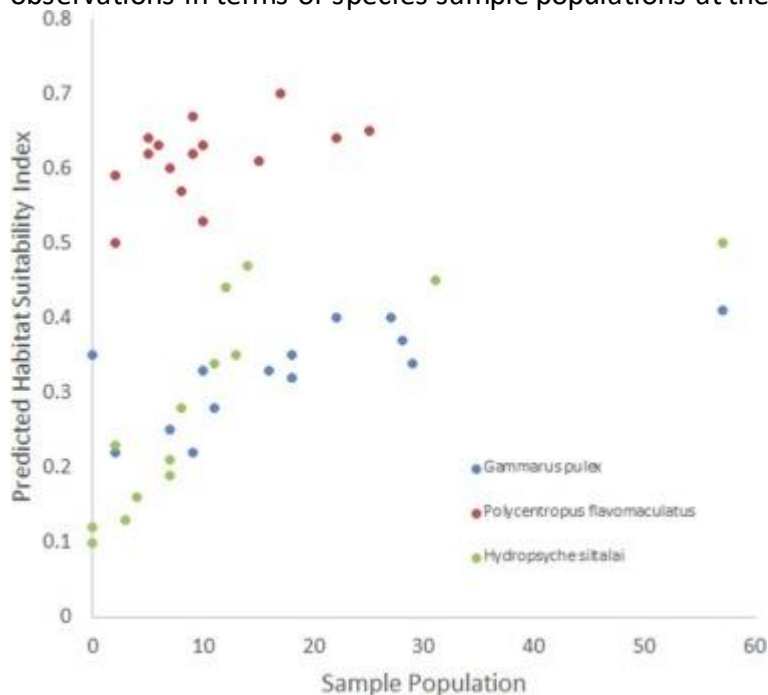

Fig. 4. CASiMiR testing, comparing the HSI at 0.024 m^3/s with observed number of individuals sampled at each point for 3 macroinvertebrate species at the Holden Wood site.

A positive correlation can be observed between predicted HSI and measured species populations. Pearson correlation coefficients for the above figure are 0.62302, 0.57719, and

0.48843 for *Gammarus pulex*, *Hydropsyche siltali*, and *Polycentropus flavomaculatus* respectively. It should be noted that whilst HSI expresses the suitability of a flow regime for a given species, it does not assert that species *should* be present in any particular number. Therefore, it is not expected that HSI predictions should correspond perfectly to field data of measured populations. The relationship between HSI and species population is expected to be strongest in areas of low predicted HSI, as the conditions in these areas actively prohibit species occuptation through their unfavorable habitat. Areas of high predicted HSI may be ideal for a given species, but it does not follow that a species will occupy that area; the stochastic nature of species colonisation, or external drivers such as predation, may lead to areas of high HSI being sparsely populated. It can be said that whilst not all good habitat is populated, all large species populations should be found within good habitat capable of accomodating them. Given that the current approach only models the influence of flow, other drivers such as nutrient availability, ecological interactions and temperature may also alter the distribution of species (Ferreiro et al., 2011, Alba-Tercedor et al., 2017). Therefore, given the nature of the relationship between HSI and species populations, the current results are seen as good evidence for the utility of the model predictions.

For an analysis of flow effects, four indicator species were chosen based upon their presence in primary sampling data at most sampling sites across the reach, a range of velocity affinities, and numbers present in consultant sampling data. The four consisted of the three used in the model testing plus *Baetis rhodani*; this latter species does not occur in significant numbers in autumn, when sampling took place, so could not be utilised for testing, but did so in spring as demonstrated by consultancy data, described in Section 2, in which both autumn and spring samples were taken. *Gammarus pulex* and *Hydropsyche siltalai* display rheophilic preferences, *Polycentropus flavomaculatus* displays more limnophilic preferences, and *Baetis rhodani* displays intermediate preferences. A modelling analysis was subsequently conducted to investigate how ecological metrics for these species vary with flow.

3.3. Flow regime development – flow/ecology response

CASiMiR's outputs were then utilised to identify flows for the provision of indicator species requirements. The Hydraulic Habitat Suitability (HHS) index was utilised to provide an intuitive dimensionless value of overall habitat quality across the site, between 0 and 1. HHS is based upon weighted usable area (WUA) metric (Kelly et al., 2015), divided by the total wetted area. WUA in turn is based on the Habitat Suitability Index (HSI); by multiplying habitat type by area, with greater weighting for higher HSI values. In their proposal of HSI, Oldham et al.

(2000) state that a direct correlation between HSI value and the species carrying capacity of a habitat is assumed; this also applies to HHS. Whilst this assumption generally holds true, at higher values this correlation may level out due to external drivers such as biological interactions; high habitat quality facilitates but does not guarantee habitation, whilst poor quality habitats by definition are unsuitable for significant species populations as discussed in Section 3.2. CASiMiR-predicted HHS for indicator species was calculated as a function of flow magnitude. The resulting individual responsiveness of species to flow is presented in Fig. 5.
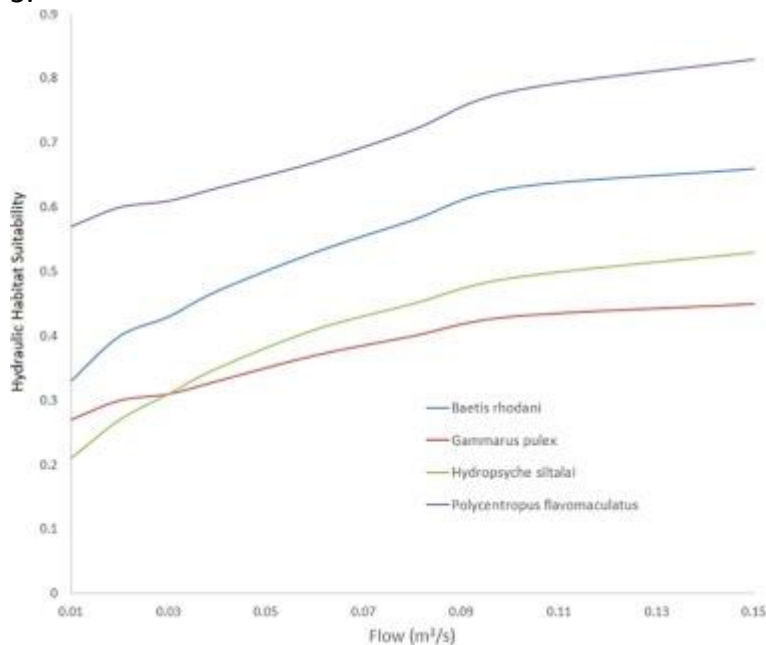

Fig. 5. CASiMiR-predicted indicator species Hydraulic Habitat Suitability values plotted against steady reach inflow.

Some species were sensitive to changes in flow; at the low end of the flow range, increasing flow from 0.01 m^3/s to 0.05 m^3/s resulted in a HHS increase from 0.21 to 0.45 for *Hydropsyche siltalai*, whilst the same increase in flow resulted in a HHS increase from 0.28 to 0.31 for *Gammarus pulex*. This difference in response is quite significant, particularly at low HHS ranges where increases of in habitat quality may mean a transition from intolerable to tolerable habitat (Oldham et al., 2000). Such differences in response suggest that certain species at the site are more vulnerable to changes in flow while some are more resilient. Levels of responsiveness at the flow ranges present within Ogden Brook (approximately 0.01– 0.10 m^3/s) suggest that some species will respond favourably to small increases in flow, whereas others will show little response, particularly at the lowest ranges of flow magnitude. Such findings may optimise flow designations depending upon seasonal species distributions.

Differences in flow preferences, and responses to flow change, among species also highlights the potential importance of flow heterogeneity in promoting biological diversity (Ward et al., 2002). Homogeneity of flow velocity was identified as an issue associated with the modified flow regime at the study site. To address this, CASiMiR was also used to calculate the flow diversity of available habitat across range of flows. An index for habitat heterogeneity was developed using Shannon's Diversity Index (H) (Magurran, 2004). The index was applied to the range of velocity distributions present within the river channel at a given discharge, as demonstrated in Fig. 6. Ranges of velocity reflect the range of flow environments and thus habitats present within the system. H is calculated using:

$$\mathrm{H}' = \sum_{i=1}^{S} p_i \ln p_i$$

Where S is the number of flow categories present in the sample and p_i is the relative proportion of habitat in the ith category (Magurran, 2004).
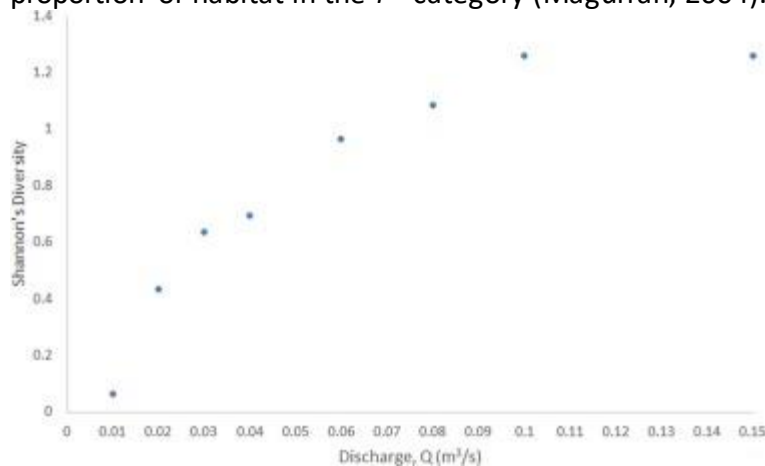

Fig. 6. Diversity of habitat; Shannon's Index of depth-averaged flow velocity across the river channel vs channel inflow.

This was applied by calculating the total wetted area and the wetted area covered by each flow velocity category over a range of discrete steady inflow discharges. CASiMiR defines 8 velocity categories, from "Very Low" to "Extreme". These categories are defined by flow ranges set by CASiMiR for each category, from 0.00 to 5.00 cm/s for Very Low, up to >80.00 cm/s for Extreme. The proportion of each velocity category was determined and used in Eq. (1) to derive a measure of "flow diversity" for the study reach (Fig. 6).

It was found that habitat diversity increases with flow rapidly in the lower flow ranges, but this trend diminishes and eventually plateaus. Beyond Q = 0.1 m³/s, flow expenditure gives little benefit in terms of habitat diversity, and at higher flow ranges flow-habitat diversity decreases as the river becomes more uniformly fast-flowing. Due to these diminishing

returns, alongside the reduced responsiveness of indicator species at higher flows, and due to local infrastructure design being based upon historical flows, designed flows were limited to a maxima of 0.1 m^3/s. Mean diversity across the range of flows (up to 0.1 m^3/s) is approximately 0.75. In order to define a lower bound for designed flows, a critical diversity value was defined as an approximately 80% loss of habitat diversity below the mean (i.e. a diversity value of 0.15), which corresponds to a flow threshold of approximately 0.015 m^3/s. It is recognised that the relative nature of Shannon's index, and the difficulty in quantifying the impact of habitat availability and heterogeneity upon the ecosystem (Yin et al., 2017), means that habitat diversity (and thus flow) thresholds are difficult to define objectively. In this study the threshold is designed to act as a buffer to prevent complete habitat homogeneity, and regime-specific flow regime minima are designated through a combination of habitat diversity and more quantitative species sensitivities identified through HSI values (see Section 3.6). Depending upon the information available for a given system, the approach towards such thresholds and the emphasis placed upon particular metrics may be varied.

It should also be noted that the hydraulic model for the site is calibrated at a significantly lower magnitude than the upper natural flow range (0.024 m^3/s vs 0.41 m^3/s); model results at magnitudes similar to natural conditions may therefore not provide accurate hydraulic predictions. Additionally, local infrastructure has developed alongside the current state of the flow regime; "natural" flow ranges in reservoir inflow data would be unsuitable for the current state of the river channel and could increase the risk of flooding in the surrounding urban area.

3.4. Flow regime development – flow naturalisation

Habitat modelling provides a prediction of ecological response to changes in flow magnitudes. However, this alone is not sufficient to derive holistic environmental flow regimes. The desired timings, frequencies, and durations of flow events must be considered in terms of ecological requirements, and practical constraints must be considered in terms of impoundment operation and storage. Such factors cannot be considered within the CASiMiR model alone, and are often unique to a particular river or region (Konrad et al., 2011). In these cases, species requirements from literature, and natural flows from other river systems in the North West of England, were used to supplement model outputs and were integrated into flow regime development.

Ecological stability can be compromised by the loss of natural flow characteristics (Poff et al., 1997), and therefore supplementary data was required to inform flow regime design in terms high flow event frequencies and durations. As river systems of a similar geology and geography experience the same climatic conditions and tend respond to a given flow in a similar manner in terms of thermal regime and physicochemical properties (Alcazar and Palau, 2010, Arthington et al., 2006), it is expected that the biota at Holden Wood should respond favorably to high flow event frequencies and durations that are approximate to typical naturalised flow regimes within the region (low flow events were not considered due to baseline impoundment outflows already being comparable to natural low flow events). This approach is comparable to the Before/After Control Impact approach (Underwood, 1991), but is applied on a more general regional level and does not require extensive conformity with specific reference conditions. Long-term Holden Wood inflow data was not available, and a transferable "regional" set of conditions was desired; therefore flow data was obtained from 7 non-heavily regulated rivers across the North West of England, around the Greater Manchester and Lancashire areas, through the CEH NRFA website (Centre for Ecology and Hydrology, 2018), and the typical frequency and duration of high flow events in the region were identified. Rivers with an average daily flow above 1 m^3/s were excluded, ensuring rivers of similar magnitude to Holden Wood's natural state (derived from impoundment inflow data). This flow data, spanning on average 37 years, was processed using IHA software (The Nature Conservancy, 2017). The particular variables of "High pulse frequency" and "High pulse duration" were extracted from software outputs, and the median of these values was taken for each of the 7 sites. "High flows" or "high pulses" are defined in this study as flows that exceed 75% of the mean daily flow record. Analysis outputs are shown in Fig. 7. Mean standard deviation of sites was 5.648 from the mean high pulse count across sites, and 0.488 for high pulse duration (measured in days).
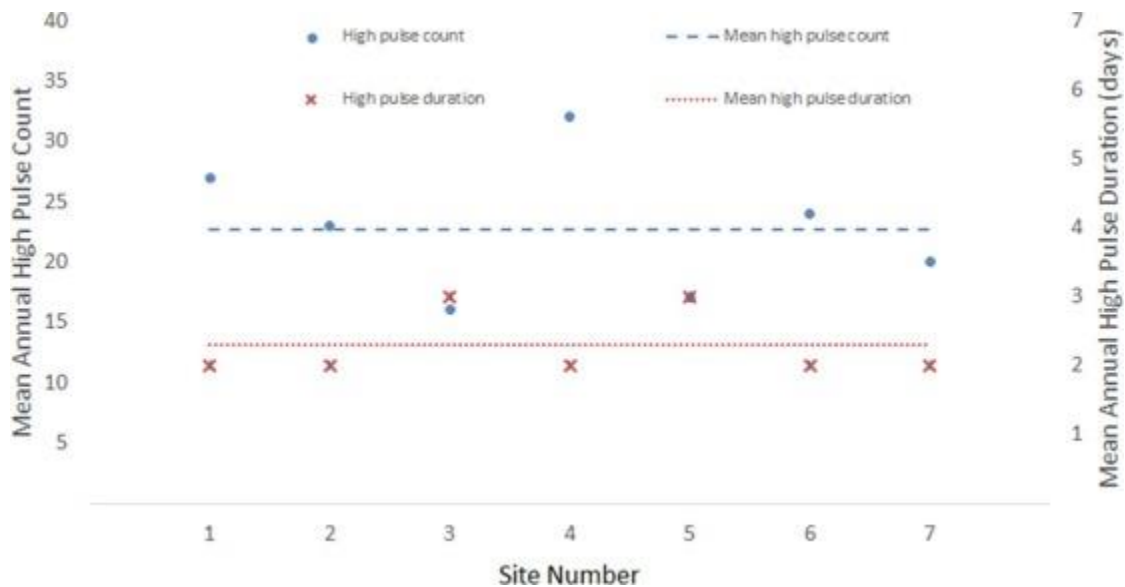
Fig. 7. High pulse event frequency and duration at 7 non-regulated sites, demonstrating extent of similarity of conditions in the North West Greater Manchester and Lancashire region.

3.5. Flow regime development – impoundment storage and water efficiency

When designing managed outflow from impoundments based on ecological modelling, the practical consideration of the impoundment structure and operational rules must be considered. In this case both minimum permitted water levels as well as the operational capacity of Holden Wood must be accounted for. Failure to utilise the impoundment sustainably could result in drainage down to the extent at which the impoundment is no longer able to continue to release the required levels of compensation flow. This would breach the impoundment licence set by the Environment Agency, and would lead to prosecution if not mitigated via water transfers from other impoundments. Flow regimes were designed this constraint in mind. "Dead water", at which point Holden Wood can no longer drain under gravity, is below 37,000 m^3 (Maddison, 2012), therefore a significant buffer above this water level was set. Based on discussions with the operator (United Utilities) a minimum threshold of 100,000 m^3 was designated. A simple model was therefore developed to understand storage levels as a function of both measured inflows and simulated 'designed' outflows over a simulation period of 1 year. This also allows the calculation of the 'efficiency' of each designed flow regime in terms of maintaining impoundment water levels. The model operated using historical impoundment inflow data from 2014 paired with outflow data (historical or proposed flow regimes), impoundment storage capacity, and volume of spills (calculated based on exceedance volume above reservoir storage capacity). At each daily time step the change in storage within the impoundment is calculated as:

$$\frac{dV}{dt} = I - (O + Sp)dt$$

where V is current impoundment storage volume (m^3), t is time (days), I is daily inflow (m^3/day), O is daily prescribed outflow (m^3/day), and Sp is overflow spill rate (m^3/day). Sp = 0 when the storage volume is below reservoir capacity (367,000 m^3); when above this level Sp is based on the total volume of capacity exceedance. I.e. the storage model assumes that excess water above reservoir capacity is released within one day, as would be expected in all but the most extreme climatic conditions. At the start of each simulation the storage volume is set based on a known value on 1st Jan 2014, taken from historical records. Water levels are calculated for each simulation across the proposed time series (until 31st Dec 2014), such that the total released volume of water over the period is known, and to ensure that levels do not fall below the prescribed minimum threshold.

3.6. Flow regime design

Individual species requirements, habitat diversity, typical regional flow event duration and frequency, and practical reservoir and site constraints were considered to design annual flow regime magnitude and timing in order to optimise ecological provision relative to volume of water released. Designed regimes (A, B and C) follow the same general design shown in Fig. 8, with five high flow pulses occurring in spring and in autumn respectively, with magnitude varying with regime. The pulse frequency and duration criteria are based on values identified in Section 3.4. Summer and winter retain constant flow rates (not including impoundment spills); in the case of summer, the season lacks biological information and there is a need to retain as much water as possible due to lower rainfall, increased water demand, and the risk of drought. In winter the cold thermal regime leads to dormancy among many taxa suggesting lower flow requirements in this season, additionally supplementary flow from spill events are common in this season due to elevated rainfall. The three regimes are informed by modelling outputs described in Section 3.3, and vary based on their balance between ecological provision and water conservation focus. Regime A aims to maximise habitat diversity and HSI during flow maxima whilst releasing a similar volume of water to 2017 outflows; Regime B aims to balance the two priorities, retaining a modest amount of water over 2017 levels and maintaining moderate habitat diversity and HSI; Regime C retains more than 50% of the water released in 2017 outflows, but ecological metrics are at threshold values. A full account of regime design characteristics and rationale is provided in Table 1.
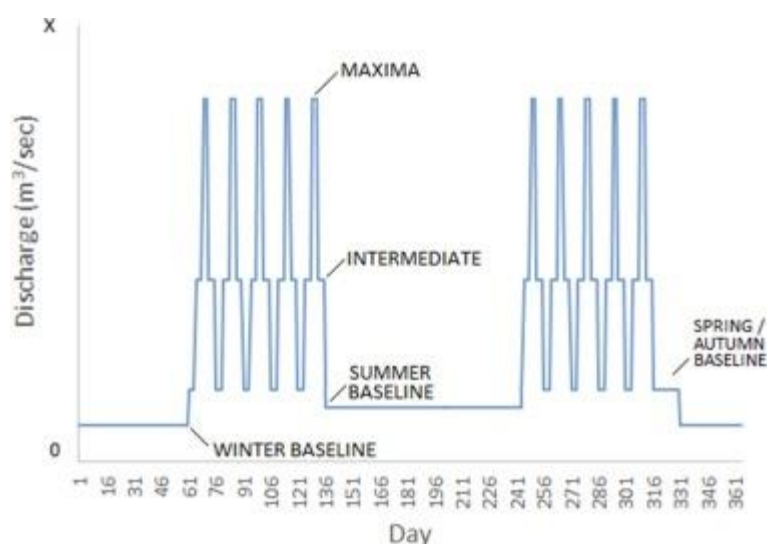
Fig. 8. General design of proposed flow regimes; A, B, and C. Day = 1 represents the 1st January.

Table 1. Breakdown of individual flow regime design characteristics with their rationale.

Characteristic	Rationale
Maxima A – 0.1 m³/s B – 0.075 m³/s C – 0.04 m³/s	Periods of high flow cultivate elevated habitat diversity and high mean HHS values across indicator species for short, repeated periods in spring and autumn. Such flows also aid in regulating the system's physicochemical properties (Alcazar and Palau, 2010)
Intermediate A – 0.05 m³/s B – 0.03 m³/s C – 0.025 m³/s	Based on good habitat diversity and moderate-high HHS values across indicator species whilst remaining within annual flow target, prolong the period of higher flow, prevent the flow increases being too sudden and disruptive to the native ecosystem (Blanckaert et al., 2013)
Spring/Autumn Baseline A – 0.02 m³/s B – 0.02 m³/s C – 0.015 m³/s	Based on threshold for most sensitive species present in these seasons, *Gammarus pulex* and *Hydropsyche siltalai*, identified in the seasonal analysis of consultant data. HSI becomes poor (>0.03) below flows of 0.02 m³/sec (see HSI vs Flow in Fig. 5). HSI above 0.02 is maintained in Regime C, a habitat of low carrying capacity but still tolerable (Oldham et al., 2000)
Reduced Summer Baseline (Regimes A and B only) 0.015 m³/s	Threshold based on habitat diversity and critical habitat quality responses to flow. Season lacks biological information and there is a need to retain as much water as possible due to lower rainfall, increased water demand, and the risk of drought
Reduced Winter Baseline 0.01 m³/s (Regimes A, B and C)	Lower productivity, and dormancy among many taxa during winter, suggests lower flow requirements in this season (Olsson, 1982). Elevated rainfall regularly supplements winter flow with spill events

4. Results

Fig. 9 presents mean HHS over the 4 indicator species for each of the designed flow regimes, historical reservoir inflow data from 2014 (which approximate to natural flow conditions), and 2017 outflows (i.e. measured flow into Ogden Brook). Outputs were generated first by defining inflows for a given model simulation, obtaining hydraulic regimes through the calibrated SRH-2D model based on the inputted flow time series, then importing this data to CASiMiR in to generate temporal habitat quality predictions.
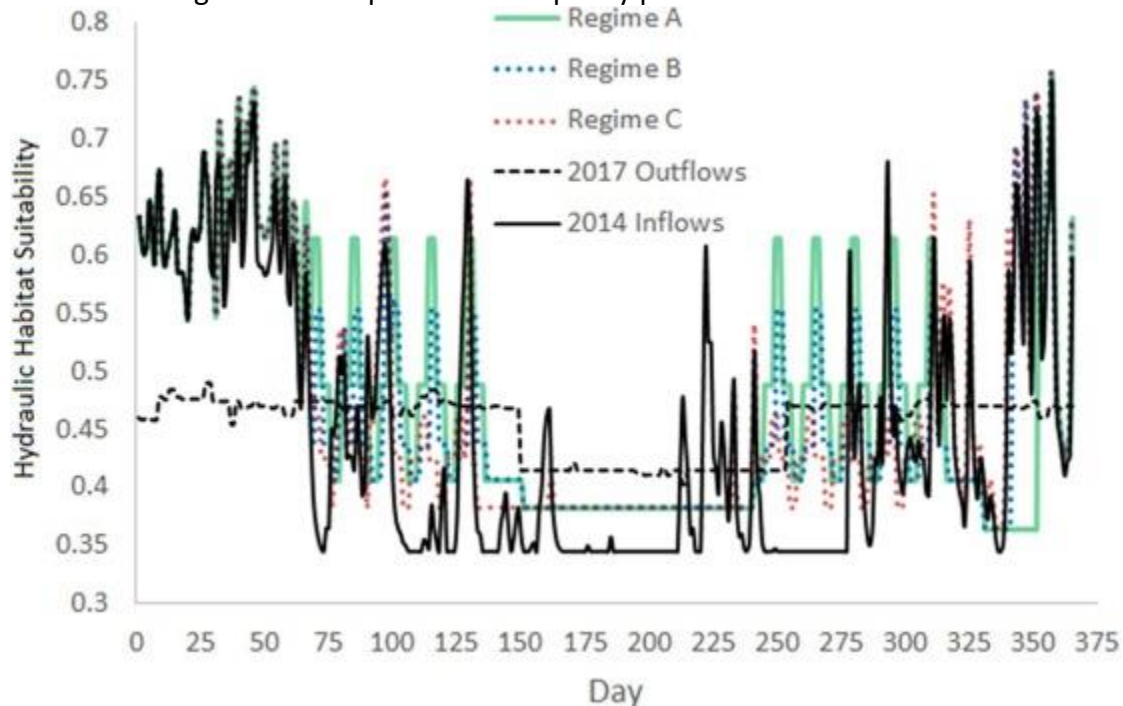

Fig. 9. Mean HHS predictions resulting from implementation of flow regimes A, B and C, alongside mean HHS based on 2017 outflows and 2014 inflows (values include effects of predicted impoundment spills).

Results show that Regime A maintains good to moderate mean HHS (~0.5–0.6) for much of the spring and autumn period, whilst Regime B maintains lower-moderate values (~0.45) with periods of higher HHS approaching 0.55 during pulse maxima. Regime C maintains lower-moderate values for much of the two seasons (~0.40–0.45), with minima values dropping to 0.35; approaching the lower end of the tolerable HHS range. The more water that is conserved within a given regime, the more likely it is that spill events will occur due to limited impoundment capacity. However as these events are determined by annual precipitation they may not be a reliable supplementary provision due to their inherent unpredictability. Fig. 10 demonstrates the influences of Regimes A, B and C upon Holden Wood storage levels based on 2014 inflow data.
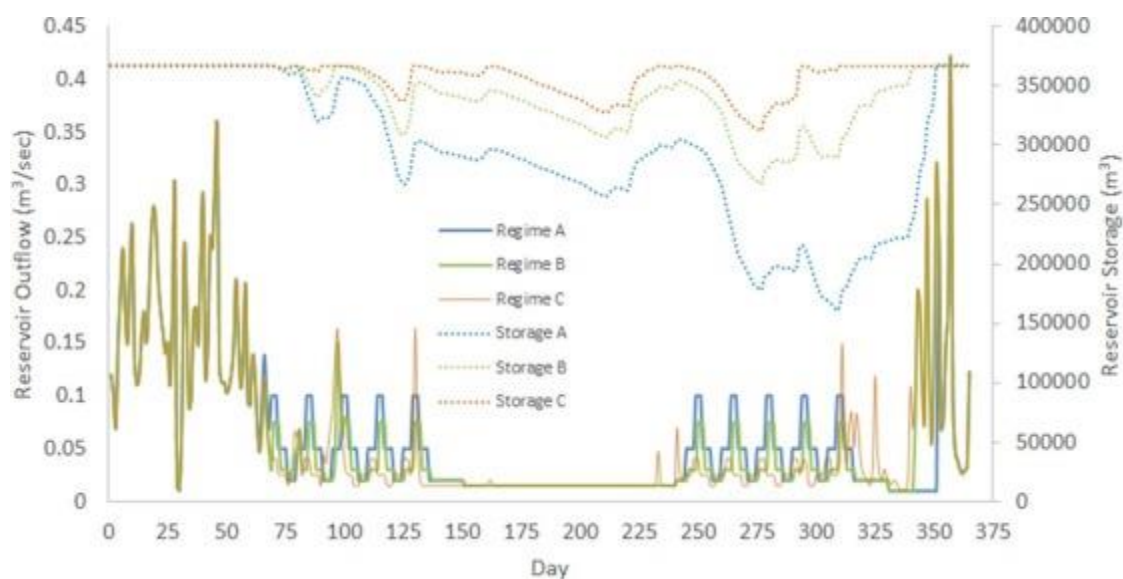
Fig. 10. Flow regimes A, B and C with associated changes in reservoir storage (based on 2014 inflow data).

Based on historical measured data, 2017 outflows at Holden Wood released 1,180,460 m³ of water over the course of a year under the current impoundment licence. Under a previous licence agreement, 2014 outflows released 600,284 m³. The increase in flow under the current licence is largely motivated by environmental concerns; 2017 outflows thus provide a good example of the continued use of the traditional steady outflow approach for ecological provision. It is therefore possible to demonstrate potential ecological benefits provided by increased flow magnitudes under the new licence, and to demonstrate how ecological needs may be met more efficiently under proposed designer flows. As a reference case the yearly variation in HHS based on the CASiMiR model was assessed under the conditions defined by 2014 and 2017 outflows.

The HHS values of indicator species were assessed between flow regimes, evaluating the mean and peak HHS over the period of analysis. These results are displayed in Table 2, Table 3.

Table 2. Average HHS for 4 indicator species at Holden Wood under historical and designated flow regimes, displaying each regime's annual flow output in cubic metres.

Mean HHS

	2014 Outflow (600,284 m³/yr)	2017 Outflow (1,180,460 m³)	A (924,480 m³)	B (721,440 m³)	C (565,488 m³)
Baetis	0.38	0.46	0.4	0.39	0.37
Gammarus	0.29	0.33	0.3	0.3	0.29
Hydropsyche	0.26	0.34	0.28	0.27	0.25
Polycentropus	0.59	0.63	0.61	0.6	0.59

Table 3. Peak HHS for 4 indicator species at Holden Wood under historical and designated flow regimes, displaying each regime's annual flow output in cubic metres.

Peak HHS

	2014 Outflow (600,284 m³)	2017 Outflow (1,180,460 m³)	A (924,480 m³)	B (721,440 m³)	C (565,488 m³)
Baetis	0.55	0.5	0.61	0.57	0.47
Gammarus	0.38	0.35	0.43	0.39	0.31
Hydropsyche	0.43	0.39	0.49	0.44	0.35
Polycentropus	0.69	0.65	0.78	0.71	0.63

Mean HHS values between 2014 and 2017 flows show limited response to flow increase; *Baetis rhodani* shows the greatest change, and even here an increase of only 0.08 HHS is observed; a definite improvement, but requiring over 500,000 m³ more water to be sent downstream per year. Designated regimes are shown to be capable of maintaining average annual ecological metrics at acceptable levels, while conserving significant quantities of water and providing frequent habitat quality maxima (demonstrated in Fig. 8, Fig. 9, Fig. 10) within the most ecologically-relevant seasons (based on Environment Agency sampling procedure).

Habitat quality maxima demonstrate a dramatic improvement in terms of applied ecological principles; flow variation is far greater, with ten high pulse events in contrast to the two or three throughout the year in 2014 and 2017 outflow data, and pulse magnitude is significantly

higher in regimes A and B, with above a 100% increase (approximately 0.045 m^3/s up to 0.10 m^3/s) for Regime A, and an approximate 66% increase for Regime B (up to 0.075 m^3/s). Regime C maintains pulses in spring and autumn seasons similar to those of 2017 outflows (though with lower duration and more flow fluctuation), despite releasing less than half the amount of water annually.

5. Discussion

The results of this study support the premise behind criterion driven flow design encompassing both temporal and magnitude-based requirements; despite greatly increased outflows in 2017 historical data compared to other regimes, HHS did not increase in favourable proportion. Whilst 2017 outflows have increased significantly relative to 2014, they remain largely homogeneous and fail to integrate natural variation such as high flow pulses. Thus, whilst more than 500,000 m^3 more water is released, ecological improvement relative to this is minimal. A holistic approach to environmental flow design is necessary to efficiently provide for ecological requirements in a world with increasingly pressing and conflicting water resources demands. This is consistent with findings from other recent studies (Gillespie et al., 2015a, Gillespie et al., 2015b, Worrall et al., 2014, Brooks and Haeusler, 2016).

5.1. Assumptions and limitations

A number of assumptions are made to generate 2D model predictions. For hydraulic predictions, channel hydraulics were assumed to be simplistic enough for depth-averaged velocity characterisation to be appropriate. In more complex river systems, more extensive velocity measurements at multiple depths may be required to represent river hydraulics. Normal velocity distributions were assumed at the inflow and outflow boundaries; this assumption was valid in this study due to the identification of ideal boundary locations upstream and downstream at the reach. In complex, winding channels other velocity distribution methods may be necessary.

It has been claimed that 3D models provide more robust predictions, and that the z dimension can be an important aspect of ecological pressure and response (Pisaturo et al., 2017). However, in the case of Pisaturo et al. (2017), the study was performed within a much larger river system of significant depth, magnitude, and velocity. The continued success of studies utilising 2D models even in larger river systems (Jowett and Duncan, 2012) leads this investigation to propose that in a smaller-scale system such as Holden Wood, the 2D

modelling approach is more appropriate. The lesser requirements of the 2D modelling approach entails easier transferability; a desirable advantage given the aim of this framework to be appropriate in a regional context. This is particularly the case should this approach see more typical application within larger systems in which the computational demands of 3D modelling would become unfeasible for most users.

Designed flow regimes derived from model results and ecological considerations are based on the assumption that precipitation patterns reflect typical annual precipitation. During particularly wet or dry years, adaptive management should address cases in which proposed flows are not appropriate for current conditions; perhaps flows must be reduced to baseline levels during droughts, or elevated flows must be prolonged during wet periods when the reservoir is near capacity. During such extreme conditions, the expertise of the water managers may adapt the regime accordingly, or flows may be set to pre-defined values based on demand, similarly to 2017 outflows being defined by water level.

5.2. Environmental flow design

Flow requirements of indicator species presented in the Methods show that generally, at the ranges of flow studied, there are diminishing returns of predicted habitat quality response to increasing flow at the study site. Beyond 0.07 m^3/s a reduction in responsiveness is observed, and beyond 0.09 m^3/s species response is generally beginning to plateau. This implies that magnitude increases, based solely upon species preference curves, are not an efficient solution for the ecological improvement of a system at the flow ranges studied at the Ogden Brook site; and becomes increasingly less efficient the longer the flow is maintained. Current impoundment outflows at Holden Wood do not demonstrate a consideration for seasonal variation in productivity and taxon composition; this study has proposed that a criterion-based flow design may target the key ecological timings for a system, and provide less flow at other times such as biologically less active periods (e.g. winter) or periods when stricter water resource conservation is necessary (e.g. summer). Allocating flows in this manner may allow for ecological provision that both improves ecological metrics, and also addresses the conflict between environmental flows and the societal need for water resource conservation. In contrast, uniform increases to compensation flows can lead to small improvements in ecological metrics yet disproportionately high flow expenditure, as was observed to be the case between 2014 and 2017 Holden Wood outflows.

The homogeneity of steady regimes reduces the range of flow (and thus habitat) conditions at a site. Section 4 demonstrates this; 2017 outflows result in peak HHS values most similar to Regime C, despite releasing more than double the quantity of water throughout the year. Again, this supports the premise that such flows may release a great deal of water, yet do not address important ecological requirements. Variation in flow and more naturalised high pulses serve to regulate the physicochemical properties of the riverine system such as the sediment and thermal regimes, nutrient content, and water pH, and such flows may control species populations by preventing the dominance of single-flow specialists (Petts and Gurnell, 2005, Richter et al., 1996). Peak HHS during Regime A flow maxima are significantly higher (increases of 0.08–0.13) and more frequent than peak HHS during 2017 outflows; these periods of elevated HHS may allow taxa to better establish themselves within the reach, whilst remaining resilient to short-term low flows between flow maxima periods due to biological adaptations to natural flow variation (Poff et al., 1997). Regime B shows similar but less pronounced improvements, whilst Regime C sees a slight reduction in peak HHS relative to 2017 outflows, yet utilises less than 50% the total annual outflow by comparison. Frequent periods of elevated flow also generate greater diversity of habitat in areas of previously homogeneous baseline flows. As greater habitat diversity facilitates greater biodiversity (Ward et al., 2002), flows throughout spring and autumn periods in designated regimes would in principle be expected to improve biodiversity metrics, assuming the periods of low flow between intermediate and maxima do not remove established biota. High flow pulses also aid in river connectivity, transferring nutrients between the main channel and periodically wetted areas (Junk et al., 1989, Junk and Wantzen, 2004), as well as varying connectivity between different river sections that may be separated by barriers such as weirs (Shaw et al., 2016). Lacking such mechanisms, it is unlikely that the functional composition or level of biodiversity within current modified systems will resemble that of their natural counterparts (Gillespie et al., 2015a, Poff et al., 1997). Whilst raw flow magnitude has a very substantial influence upon benthic ecology, the temporal aspects of flow such as frequency and duration of events, based upon local natural trends, should in principle provide more holistic ecological provision. Flow event durations and frequencies may play a key ecological role, creating more temporally heterogeneous environment where a single species cannot dominate (Levin, 2000), driving sediment transport mechanics and their associated impacts (Kondolf, 1997), and driving connectivity of the river with the surrounding flood plain (Junk and Wantzen, 2004). Systems with homogeneous flows have demonstrated decreased biodiversity (Wiens, 2002), and it is unlikely that flow magnitude divorced from natural conditions can ensure a healthy ecosystem capable of meeting ecological targets (Acreman et al., 2014). A key

challenge to the implementation of environmental flows has been the increased labour such flow designs would entail. A transferable framework based upon general regional principles, such as that proposed in this study, could help to alleviate some of these labour requirements by allowing environmental flows to be designated efficiently across numerous small-scale sites with minimal adaptation between them; sites which may otherwise be unfeasible for restoration on a specific case by case basis. The similarity of natural river system behaviour observed in the North West of England lend support towards this possible transferability, though further research and flow experimentation would be necessary to confirm this with confidence.

5.3. Further implications

This study demonstrates the potential of ecology-flow principles as a promising ongoing area of investigation. Such investigation could be performed through a number of means; desk-based analyses utilising IHA-style flow characteristics and ecological metrics could investigate trends between study sites, or in-field flow experimentation could attempt to apply a designer flow across similar sites and monitor ecosystem response. We have demonstrated the potential of linked hydro-ecological modelling, particularly for small-scale sites where vertical complexity is minimal and an efficient approach is necessary due to resource constraints. A significant outcome from this investigation has been the demonstrated potential for significant quantities of water to be conserved through designer regimes, whilst anticipated ecological response should be improved, both due to criteria-based flow allocation and greater naturalisation of the regime. Regimes A, B and C promote ecological provision, with varying prioritisations. This demonstrates the utility of this approach; it is possible to define design criteria, which may be adapted to accommodate changing water demands and diverse interests of stakeholders present within a given system. Validation of this approach through post-implementation in-stream flow experiments, in order to assess ecological response to proposed flow regimes, is a key next step. Should this method be validated, it is believed that such flows could be applied regionally to similar river systems with minimal field investigation requirements. Such transferability may allow for smaller scale impoundments across the UK to implement environmental flows, where previously this was unfeasible due to the quantity of impoundment systems and the intensity of labour required to assign environmental flows to individual sites.

Scaling this methodology up to assess higher magnitude class river systems would likely require adaptation of the approach. A larger number of field velocity measurements are

recommended for more robust calibration, and the influence of vertical velocity may also have to be considered in some cases (Pisaturo et al., 2017). Larger systems may host a greater variety of biota, and therefore the type of indicator species selected must be considered; fish may be present and act as an important aspect of the ecosystem (Cheimonopoulou et al., 2011, Harris, 1995), or macrophytes might be used for analysis as in other studies (Onaindia et al., 2005). Further system ecological model complexity might be added, including processes that may be more relevant at larger scales, e.g. heterogeneity of bed sediment; CASiMiR is able to consider such influences if species affinities are inputted. The flow contributions of any downstream tributaries or depleted reaches to the site of interest would also require consideration, and may entail a more adaptive approach to the reservoir flow regime due to the variability of natural flow that is introduced (which could for example be informed using hydrological modelling).

This framework currently focuses upon application for sites impacted by impounding reservoirs; it could be possible to adapt it for use in other site restoration assessments such as hydropower-impacted sites by incorporating the unique challenges and priorities of the given modification into the design stage of the environmental flow regime. An example of such a consideration would be the necessity of disruptive high flows from hydropower releases; perhaps a regime design for such an application may focus upon dampening and prolonging these high flows according to what is feasible without compromising the service of the dam. It is also acknowledged that flow is not the sole driver of ecological response, and other stressors such as water chemistry likely play a significant role at many sites. It has been suggested that the diverse influences of riverine ecology must be studied both through short-term mechanistic experiments and long-term explanatory studies in order to disentangle this complex web of interactions (Laini et al., 2018). Climate change and land use change are also resulting in a shifting environment, further driving changes in ecological metrics (Li et al., 2018). As understanding of these interactions grows, it would be possible to integrate further mitigation methods into the framework presented in this study. The ability to integrate ecological requirements according to context, and make adjustments according to new knowledge, offers significant utility within this framework.

6. Conclusions

This study has presented a methodology by which a study site is assessed and environmental flows are proposed based upon a combination of species response to flow (through preference curves), the influence of magnitude upon habitat diversity, and typical

unregulated regional flow characteristics, in order to form a holistic ecological solution. Results suggest that uniform increases in magnitude over long periods result in disproportionately little ecological benefit relative to volume of water released, and affirms the use of optimised and targeted high flow events. Though there is a rich literature detailing the concepts considered, we are not aware of any studies suggesting a similar approach by which such a combined range of flow requirements within a particular site or region may be assessed. Poff et al. (2017)'s update on the evolution of environmental flow science discusses progress in almost every area, but there is not yet a unified approach to environmental flow assessment. They emphasise the need to extend from a local scale to basin-scale perspective (Poff et al., 2017).

Amidst this rapidly developing field in which numerous frameworks and methods for environmental flow assessment are emerging, this study offers a novel approach in efficient annual flow regime designation, aiming towards regional transferability. We offer the first steps towards an actionable regional water management solution to the issue of impoundment-modified flow impacts that is desirable both for the purpose of ecological and water resources conservation. Future priorities include the detailed validation of such an approach by implementation of a derived flow regime at a case study site, and the monitoring of ecological response in comparison to model predictions. There is also scope for this framework to be scaled up to larger river systems, though this may require the incorporation of other variables significant at such a scale, such as substrate type and variation and the sediment transport regime. Fish may also be considered in CASiMiR should they be present in the system. This investigation suggests that 2D habitat modelling remains a tool with great potential when incorporated into such holistic practices, and shows great promise as water managers move into transferable, regional-focused forms of investigation.

Acknowledgements (Paper)

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