IMPACTS OF FLOW REGULATION AND ARTIFICIAL FLOODS IN AN UPLAND STREAM ECOSYSTEM

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The candidate confirms that the work submitted is his own, except where work which has formed part of jointly authored publications has been included. The contribution of the candidate and the other authors to this work has been explicitly indicated below. The candidate confirms that appropriate credit has been given within the thesis where reference has been made to the work of others.

Chapter 3

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ABSTRACT

Mitigation of ecological impacts associated with stream regulation is now a legislative priority and Artificial Floods have been suggested as a potential tool to achieve this aim. However, understanding of the impacts of stream regulation and Artificial Floods on downstream ecology is currently limited. This thesis provides detailed reviews of both of these topics and identifies key contemporary research priorities. These priorities were subsequently addressed through assessment of the impact of stream regulation and Artificial Floods on downstream hydrology, physical chemistry, coarse sediment transport and benthic macroinvertebrates in an upland sub-catchment of the River Humber, UK. Evidence that regulation was associated with significant impacts on hydrology (e.g. flood frequency, rate of change), physical chemistry (particularly flood pH and diurnal stream temperature range) and macroinvertebrates was identified, but impacts were found to vary spatially and temporally, indicating the importance of site specific and temporal factors. Control of hydrological characteristics was demonstrated during Artificial Floods which generally resulted in reductions of electrical conductivity, dissolved oxygen and pH and no change in stream temperature. Evidence for coarse sediment transport in line with overspill events prior to Artificial Floods was identified, but little evidence for change in macroinvertebrate assemblage was found. Evidence for the use of Artificial Floods as a management tool was greatest for coarse sediment transport and pH but overall, limited potential was demonstrated, bringing into question their validity as management techniques in some regulated streams and provoking requirement for further research. The findings of this thesis, methodological developments, conceptual advances and recommendations are therefore considered to have advanced the science and understanding of regulated stream management. Such progress is vital in this rapidly developing research field.

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Abbreviations

AF	*Artificial Flood*
AIC	*Akaike information criterion*
aod	*Above ordnance datum*
BGS	*British Geological Survey*
CORINE	*Coordination of information on the environment*
CPOM	*Coarse particulate organic matter*
DO	*Dissolved oxygen*
EA	*Environment Agency*
EC	*Electrical conductivity*
Est.	*Estimate*
EU	*European Union*
GAMM	*Generalized Additive Mixed Model*
GLMM	*Generalized Linear Mixed Model*
GRPS	*General packet radio service*
ind.	*Individuals*
mins	*Minutes*
n.d.	*No date*
NE	*Natural Earth*
St.dev.	*Standard deviation*
UK	*United Kingdom*
USGS	*United States Geological Survey*
YW	*Yorkshire Water Ltd.*

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1 INTRODUCTION

1.1 Research context

Humans hold intrinsic value for the environment that surrounds them for recreational, spiritual, scientific, economic, historical or philosophical reasons (Rolston, 1994). The environment provides important services which enable our survival, for example, provision of food and oxygen (Salzman, 2009). Yet, paradoxically, through our activities, we threaten the environment. For example, overfishing of marine ecosystems threatens species with extinction (Jackson et al., 2001). However, we are aware of the threats we place on our environment and aim to counteract them through measures such as conservation and restoration (Meffe & Carroll, 1994).

Freshwater streams are important to humans as they provide services such as provision of water for consumption and agriculture (Araya et al., 2003). They are also of economic importance as they attract tourists to undertake activities such as fishing (e.g. Dalrymple, 2006) and can also be seen as socially, culturally and aesthetically important (Wallace et al., 2003). Stream environments are sensitive to anthropogenic pressures (Schmidt et al., 2009) and are considered to be among the most impacted ecosystems on the planet due to factors such as pollution, over-abstraction and dam construction (Malmqvist & Rundle, 2002). This has resulted in a growing drive to conserve and restore streams, reflected by the development of water focussed legislation such as the US Clean Water Act (USC, 2002) or the EU Water Framework Directive (EU WFD) (EC, 2000).

Whilst the drivers for implementation of mitigation of anthropogenic pressures on stream ecosystems are growing, there is recognition that the science to inform how, practically, these measures should be implemented is limited (Petts et al., 2006; Ormerod et al., 2007). Stream managers have highlighted the requirement for further research to underpin their decisions. For example, in the UK, representatives from water supply and treatment companies have recently recommended research is undertaken to better understand hydroecological relationships to enable them to meet their legislative targets with regards to practices such as reservoir management (Bowles & Henderson, 2012).

In regulated streams, environmental flows have been suggested as a potential tool to mitigate impacts to ecosystems downstream of dams (Dyson et al., 2003). Environmental flows can be defined as provision of a flow regime that meets desired ecosystem objectives (Acreman & Dunbar, 2004) and can include practices such as Artificial Flood (AF) implementation (i.e.

where discharge is increased to emulate a natural flood). The science of environmental flows is currently at an early stage, but their use has the potential to meet legislative targets (Dyson et al., 2003; Acreman & Ferguson, 2010) and there is therefore an urgent requirement to further understand the impact of implementation of such techniques.

1.2 Research gaps, aims, objectives and hypotheses

1.2.1 Aims

This thesis is driven by a requirement to better understand regulated stream ecosystems. The primary aims are to:

1. Assess the impact of regulation on downstream ecosystems, including hydrological, physical-chemical, morphological and biotic elements;

2. Assess the impact of environmental flows (AFs specifically) on downstream ecosystems.

1.2.2 Research gaps, objectives and hypotheses

This thesis addresses the key themes identified in the reviews undertaken in Chapters 2 and 3 by examining both the impact of regulation and Artificial Floods (AFs) on downstream ecosystems (Figure 1.1). Chapters 5 to 9 of this thesis detail the particular research gaps they address and these are summarised below. The final chapter of this thesis provides a general synopsis of the research detailed in Chapters 5 to 9.

Chapter 5

Understanding of the impacts of stream regulation at various temporal (e.g. diurnal, seasonal) and event based (i.e. flood) scales on downstream hydrology and physical-chemical parameters such as electrical conductivity (EC), dissolved oxygen (DO) and pH is currently limited. The impact of AFs on these facets is also poorly understood. This chapter reports on a detailed study of hydrology, EC, DO and pH in a catchment in upland UK over a two year period.

Objectives

1. Undertake an assessment of the impact of regulation on hydrology, EC, DO and pH through comparison with unregulated conditions;

2. Conduct a series of AFs to assess the potential for use of AFs as mitigation for any impacts identified.

Hypotheses

1. Regulation would reduce flood frequency, magnitude and duration and the impact would vary temporally;

2. Regulation would impact downstream physical chemistry and these impacts would vary temporally;

3. AFs would impact downstream physical chemistry thereby demonstrating control of downstream physical chemistry and potential for use as mitigation

Chapter 6

Identification of the complex drivers of stream temperature in regulated streams, variability of the impact of regulation on mean stream temperature and the limited number of studies examining the impact of multiple-reservoirs on downstream temperature has led to a requirement for a detailed assessment to address these topics. Further, the impacts of AFs on downstream temperature have varied globally and further studies are required to assess this impact. This chapter reports on a detailed study of stream temperature in a catchment in upland UK over one year.

Objectives

1. Undertake a spatio-temporal assessment of the impact of regulation on stream temperature using contemporary analytical techniques through comparison of several regulated streams with unregulated conditions in a multi-reservoir catchment;

2. Conduct an assessment of the impact of a series of AFs on downstream temperature.

Hypotheses

1. Regulation would reduce downstream temperature range and impact mean stream temperature according to season;

2. Impact would vary spatially;

3. Change in downstream temperature would be observed during AFs, thereby demonstrating the potential for use of AFs as mitigation.

Chapter 7

Process based understanding of sediment transport in streams has been cited as a key research priority (Petts & Lewin, 1979; Carling, 1988), yet few studies have addressed this need. Those that have (Gilvear, 1987; Lyons, 1992; Sear, 1993) have primarily focussed on transport of *suspended* sediment resulting in a dearth of process based understanding of coarse sediment transport in regulated streams. Additionally, the limited number of published observations concerning sediment transport *during* AFs is now driving the requirement for understanding at regional scales (Poff & Zimmerman, 2010; Gillespie et al., in review).

Objectives

1. Undertake a detailed study of coarse sediment transport in a regulated upland UK stream to better understand the sediment transport-discharge relationship and threshold discharges;

2. Assess the impact of a series of AFs of varying characteristics on coarse sediment transport and potential for use of AFs as morphological management tools.

Hypotheses

1. The sediment transport-discharge relationship and threshold discharge to invoke sediment transport would vary temporally reflecting antecedent flow conditions;

2. AFs would invoke sediment transport and therefore demonstrate potential for use as a morphological management tool.

Chapter 8

A lack of consensus exists with regards to the response of macroinvertebrate communities to upstream impoundment leading to a call for focussed regional-scale assessments (Poff & Zimmerman, 2010; Gillespie et al., in review). Additionally, previous research may have failed to differentiate between subtle changes in the extent to which a site is affected by regulation, potentially limiting the inferences made. Furthermore, recently developed indices (e.g. LIFE (Extence et al., 1999); PSI (Extence et al., 2013)) may have potential for use in identification of impacts of regulation.

Objectives

1. Identify relationships between the extent of stream regulation and macroinvertebrate communities using a multi-site, regional-scale approach;

2. Examine the utility of new, continuous, index representing the extent of stream regulation;

3. Evaluate two recently developed indices (LIFE and PSI) alongside established biomonitoring indices to consider their relative performance for assessment of the impacts of regulation.

Hypotheses

1. Macroinvertebrate indices and community composition would both be affected by upstream impoundment with some taxa increasing and others decreasing in abundance relative to their sensitivity to changes in flow;

2. A continuous index representing extent of regulation would be more sensitive to differences in community composition than categorical classifications;

3. LIFE and PSI would decrease as the extent of stream regulation increases, and demonstrate superior sensitivity to alternative indices such as diversity, dominance, BMWP and ASPT in detecting any impacts.

Chapter 9

In addition to the lack of consensus in response of macroinvertebrate communities to upstream regulation noted above, some of the assessments focussing on this topic have been of limited intensity (i.e. ≤ two samples per year (e.g. Englund & Malmqvist 1996; Maynard & Lane, 2012; Gillespie et al., in press (a))) potentially resulting in failure to identify intra-annual variation of impacts. Furthermore, an assessment of the impact of AFs on benthic macroinvertebrates in the UK is yet to be published.

Objectives

1. Conduct an assessment of intra-annual temporal dynamics of the impact of regulation on stream macroinvertebrates;

2. Examine the impact of AFs on downstream benthic macroinvertebrates in the UK.

Hypotheses

1. A difference in macroinvertebrate assemblages between regulated and unregulated sites could be observed in line with previous studies;

2. The difference would vary intra-annually reflecting taxon life-cycle attributes and environmental preferences;
3. Macroinvertebrate abundance, richness and diversity would decrease as a result of AFs;
4. Taxa would respond to AFs based on their specific environmental preferences resulting in a more disturbance resilient assemblage.

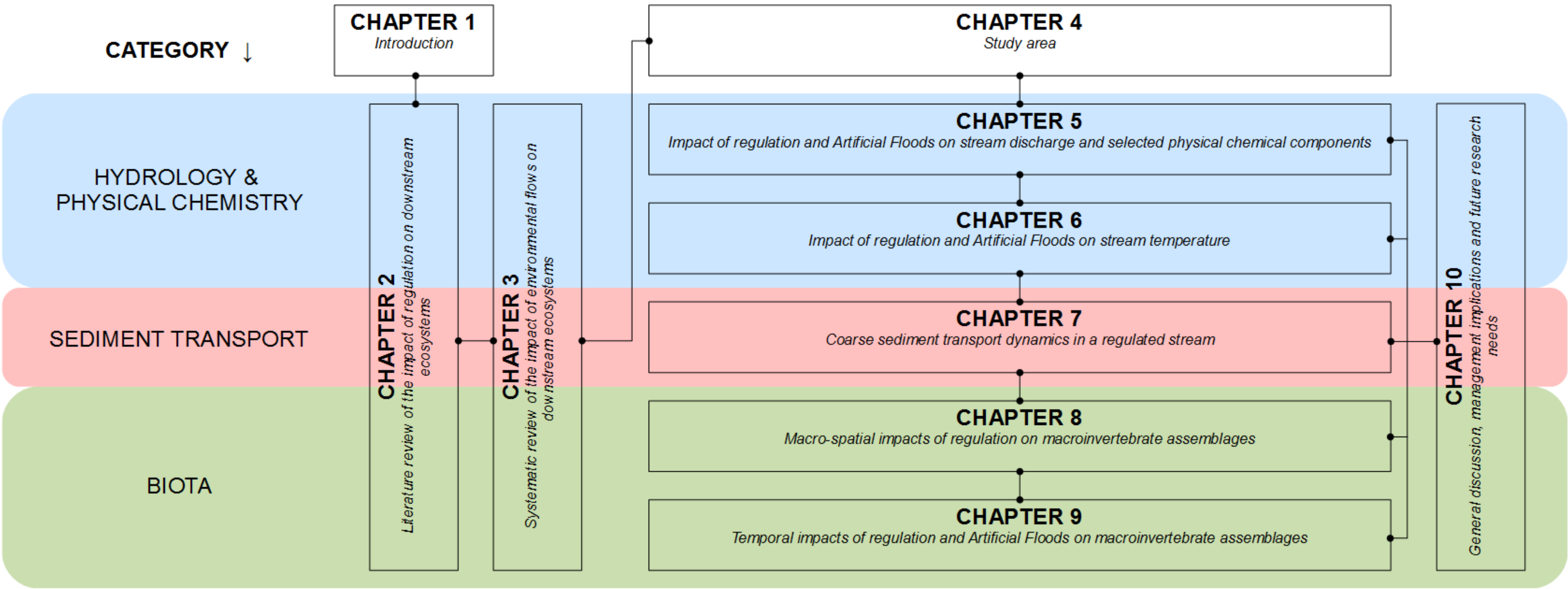

Figure 1.1: Thesis schematic structure.

2 THE IMPACT OF REGULATION ON DOWNSTREAM ECOSYSTEMS

2.1 Chapter overview

This chapter presents a global scale literature review of the impact of stream regulation on downstream ecosystems. First, to provide context, the history and spatial extent of regulation are presented. A section identifying and reviewing key research gaps in the literature regarding downstream impacts of stream regulation follows. For clarity, this section is split by topic: hydrology, physical chemistry, morphology and biota. This chapter is complementary to the following chapter which presents a strategic review of the global impacts of environmental flows on downstream ecosystems.

2.2 Introduction

Stream regulation has been defined as the alteration of natural fluvial dynamics (Ward et al., 1999) or a reduction in naturally variable flow patterns (Giller, 2005) and is achieved through installation of flow control structures such as dams, weirs, abstractions and sluices (Olive & Olley, 1997). Stream regulation through dam construction is the focus of this chapter and the aims are to (i) provide an overview of the history and global extent of stream regulation, (ii) review the literature regarding the impact of regulation on downstream ecosystems and, (iii) identify key research priorities where understanding is currently lacking.

2.2.1 The history of stream regulation

Humans have always altered the environment, whether directly or indirectly (McMichael, 1995; Leroux, 2005). Early examples of stream alteration date to the Neolithic period (c. 5800 – 2800 BCE) where irrigation was practised in Mesopotamia and Egypt (Mays et al., 2007). The earliest record of a dam is the Prosperpina in Spain, built c. 130 by the Romans to supply water to the city of Emerita Augusta (Mays et al., 2007; ICOLD, 2014). Dam construction during this period was typically small scale and associated with water diversion (Mays et al., 2007). However, the invention of the arch dam in c. 1280 and subsequent development of stronger and larger buttress and rock-filled dam types in the 18th and 19th centuries, enabled a rapid increase in the number of dams constructed. Construction was driven by activities such as the American Gold Rush, the Industrial Revolution and the growing requirement for water for expanding urban populations (Sheail, 1988; McConnell, 1999; Biswas & Tortajada, 2001). For example, in Britain, significant numbers of dams were completed in the late 18th and throughout the 19th and 20th centuries (Figure 2.1) where regular and predictable water supply to industry and expanding cities was required (Sheail, 1988).

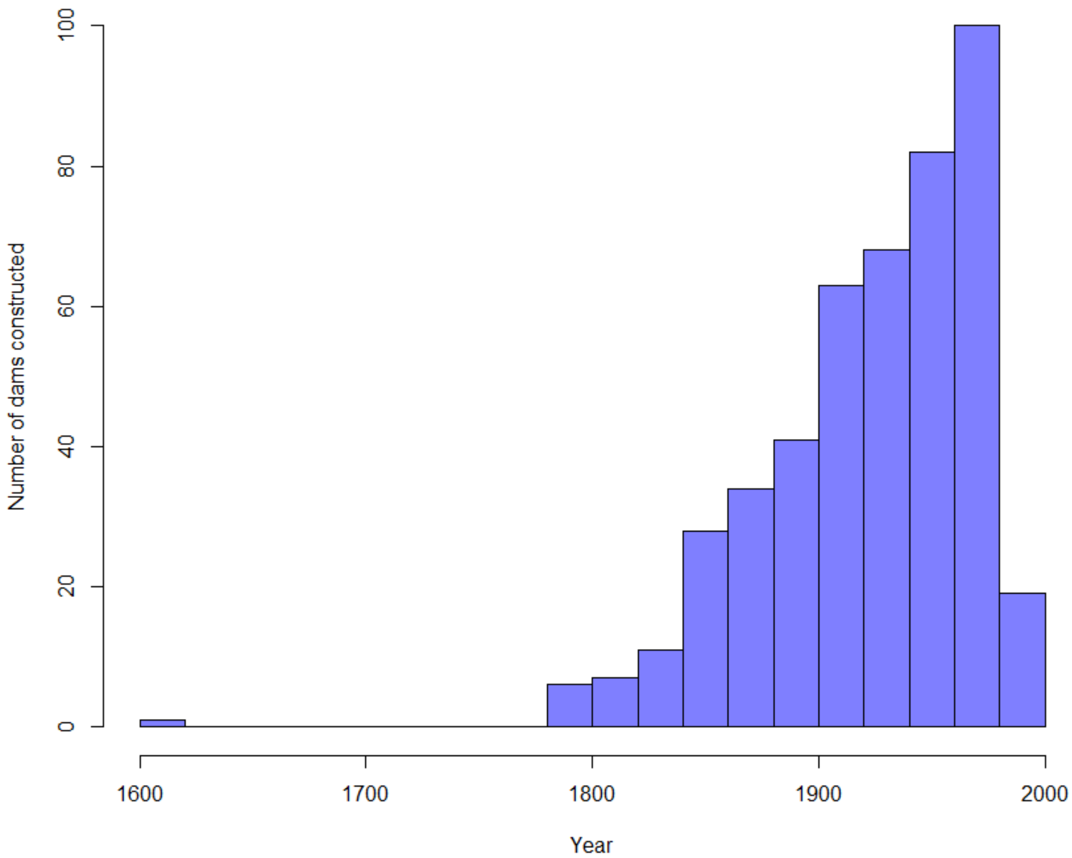

Figure 2.1: Number of dams (with a reservoir capacity > 1,000,000m^3) constructed in Britain each 20 years (source: Tedd & Hoton, 1994).

Post-1950 saw a dichotomy in the rate of dam building emerge: a decline in 'developed' nations such as the United States, Canada, Britain (Figure 2.1) and others in Western Europe was observed, and conversely, a rapid increase in dam construction rate in the majority of 'developing' nations driven by factors such as population growth, irrigation requirements and national pride (Bednarek, 2001; Biswas & Tortajada, 2001; Poff & Hart, 2002). This dichotomy, although evolved, still exists today. The growing appreciation for protection of the environment dominates dam construction/ management across the globe and, in 'developed' nations, dam mitigation and removal projects are increasing in number (Olden et al., 2014). Conversely, basic human needs remain unmet in 'developing' nations where large dams are currently being built (Biswas & Tortajada, 2001). Dam construction and management therefore remain contemporary issues that need to adapt to challenges driven by complex societal, environmental, economic and geographic issues, complicated by the 'developed-developing' nation dichotomy.

2.2.2 Contemporary extent of stream regulation

Stream regulation is now ubiquitous globally (Lehner et al., 2011). By country, the USA has the most dams (n = 9265), followed by China (n = 5191), India (n = 5101) and Japan (n = 3076) (ICOLD, 2014). There are few countries that have no recorded dams according to ICOLD (i.e. Alaska, Antarctica, Greenland and some within northern Africa) (Figure 2.2), but small dams are not recorded by this database and the true extent of global regulation is likely to be much larger (Lehner et al., 2011).

Within western Europe, Spain has the majority of dams (n = 987), followed by France (n = 622), Italy (n = 542) and the UK (n = 519) (ICOLD, 2014; Figure 2.3). Of these countries, the UK has the highest density of dams (2.13/ 1000 km^2 cf. Spain: 1.95, France: 0.97 and Italy: 1.80). The spatial extent of dams within Britain is however, not equal (Figure 2.4), with the majority sited in areas of relatively high altitude; taking advantage of gravity to transport water (McCulloch, 2004). Clusters of relatively large dams are present within both the Pennines (e.g. Scammonden Dam (73m)) and Welsh mountains (e.g. Llyn Brianne dam (91m)) (Tedd & Hoton, 1994; Figure 2.4).

Given the spatial extent of dams worldwide, there are few areas that have not felt their benefits: dams have provided cheap electricity, recreational opportunities, navigable channels and reduced severity of floods (Collier et al., 1996; Lehner et al., 2008). However, the environment downstream of dams has been cited as being profoundly impacted (Petts, 1984a). Indeed, Malmqvist & Rundle (2002) stated that dam construction was one of the key threats to stream ecosystems.

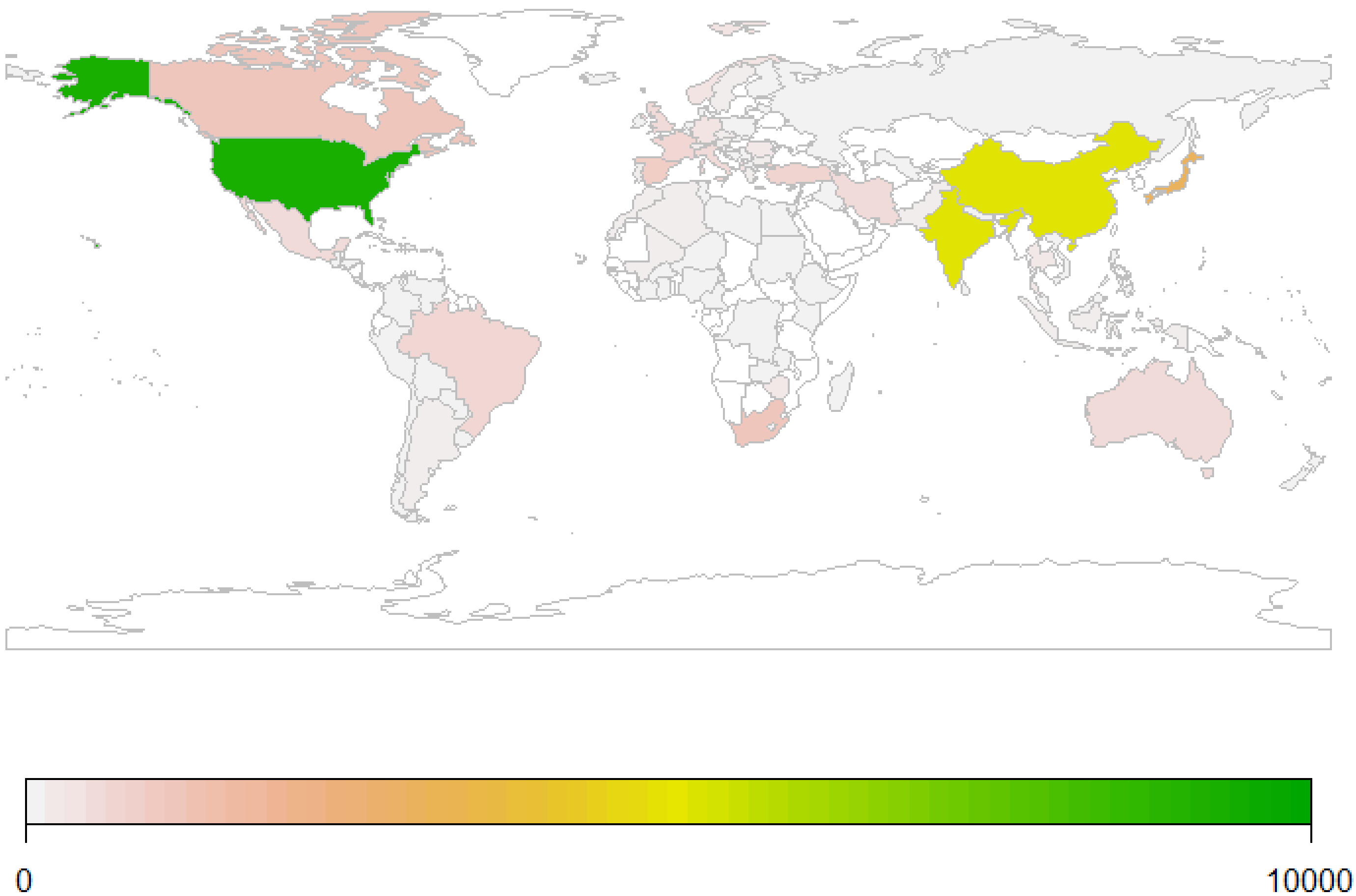

Figure 2.2: Number of dams by country (source: ICOLD, 2014).

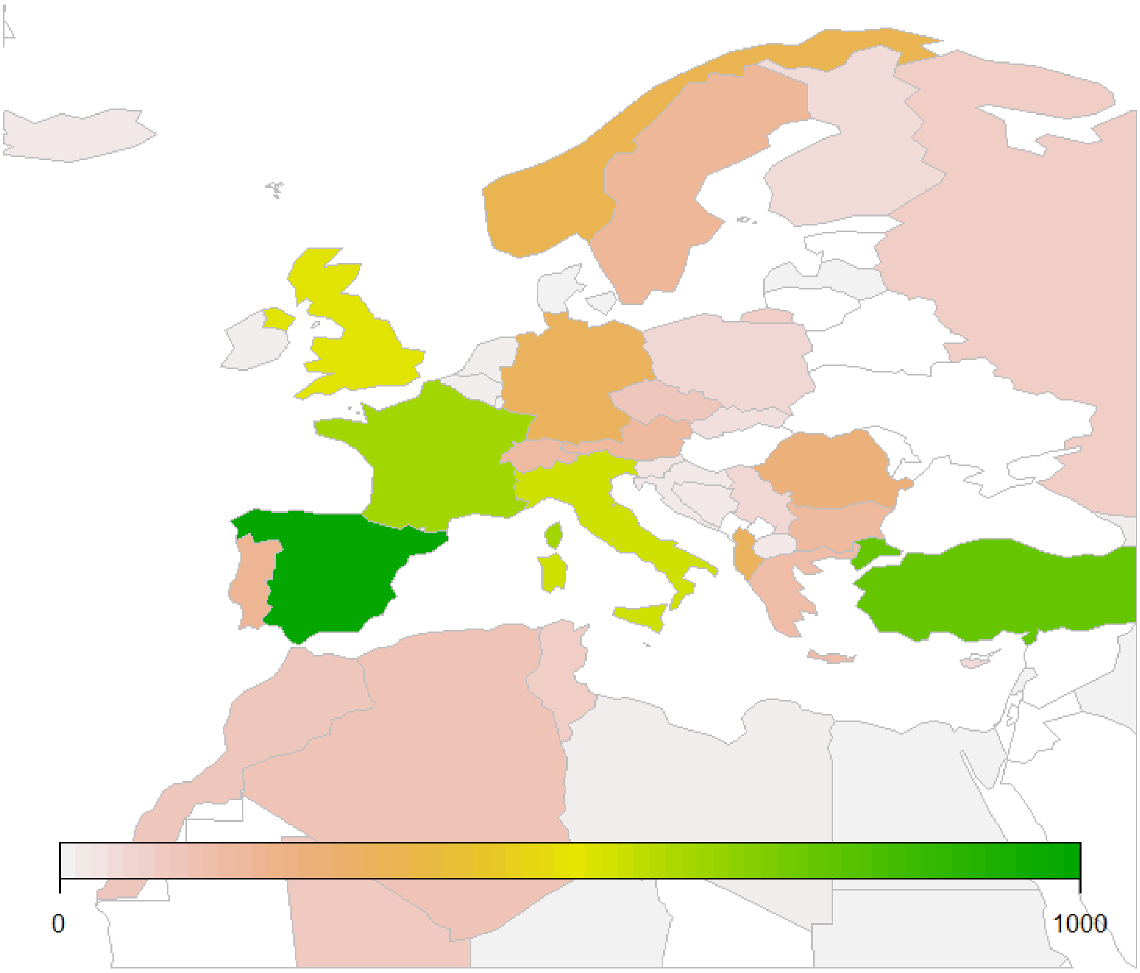

Figure 2.3: Number of dams by country in western Europe (source: ICOLD, 2014).

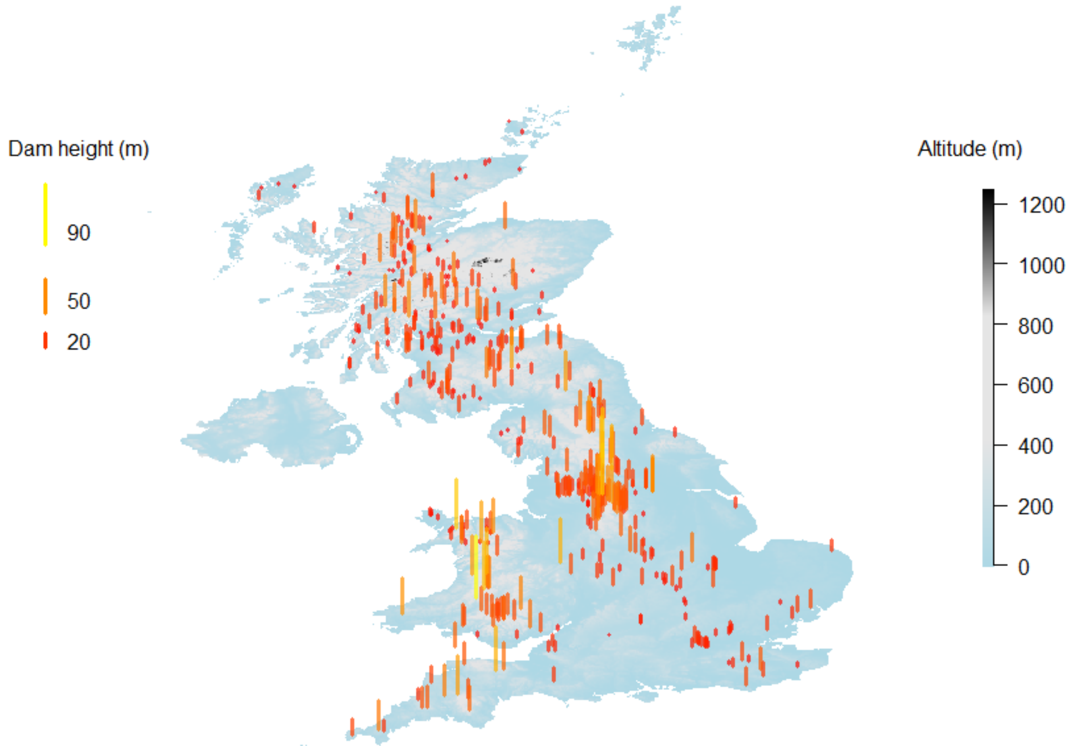

Figure 2.4: Distribution and height of dams in Britain (source: Tedd & Hoton, 1994).

2.3 Impacts of regulation on downstream ecosystems

2.3.1 Hydrology

The flow regime is perhaps the most important controlling factor of stream ecological integrity. It can be considered a “master variable” that influences physical chemistry, energy sources, physical habitat and biotic interactions which in turn affect ecological integrity (Figure 2.5) (Poff et al., 1997; Saunders et al., 2002). The over-riding principal of the natural flow regime is that of dynamism and five key characteristics can be used to describe it: magnitude, duration, frequency, timing and rate of change (Poff et al., 1997; Petts, 2009). Variability in discharge over a multitude of time scales and the frequency of extreme high and low flow events, exert control over stream ecological integrity (Petts, 1984a). It is this dynamism that can be affected by dam construction and a vast literature is dedicated to describing the impacts.

Lehner et al. (2011) estimated that 575,900 km or 7.6% of the world's streams with average flows >1 cumec are affected by upstream regulation. Nilsson et al. (2005) estimated that impoundments are capable of holding back approximately 15% of the world's total annual runoff. This regulation modifies natural flow regimes primarily through redistributing discharge through time (Petts, 1984a; Higgs & Petts, 1988). However, exact effects of an impoundment on the downstream flow regime are determined by factors such as inflow dynamics, reservoir characteristics (e.g. spillway shape) and reservoir function (e.g. hydropower, water supply) (Petts, 1984a). These factors combine to affect the occurrence and characteristics of flow-specific facets such as pulse-releases, compensation flows and flood absorption resulting in spatially and temporally complex impacts as conceptualised by Petts (1984a) (Figure 2.6).

Typically regulation reduces annual flow variability due to the attenuating ability of a reservoir and subsequent reduction in high flows (Baxter, 1977; Petts, 1984a); indeed, Graf (2006) noted reductions in the magnitude of annual peak discharges by 90%. However, over shorter time scales, flow variability can be increased due to management of the reservoir (e.g. hydropower ramping (e.g. Andrews & Pizzi, 2000)). Regulation has also been observed to reduce mean annual discharge by up to 80%, alter the timing of annual extremes and, typically after hydropower dam installation, unnatural pulse discharge events have been introduced (Petts, 1984a; Table 2.1).

In addition to reservoir specific factors (Figure 2.6), local climatic and weather conditions are also important in determining downstream impacts of flow regulation (Higgs & Petts, 1988). For example, rainfall intensity (which can be spatially variable (e.g. Abrahams & Parsons, 1991)) affects the rate at which reservoirs fill and therefore the likelihood of overspill (Higgs & Petts, 1988). Given the number of factors which can potentially affect downstream flow regimes in regulated catchments, a requirement for local scale, detailed assessments exists to inform future management plans (e.g. River Basin Management Plans) under contemporary freshwater legislation (e.g. the EU WFD).

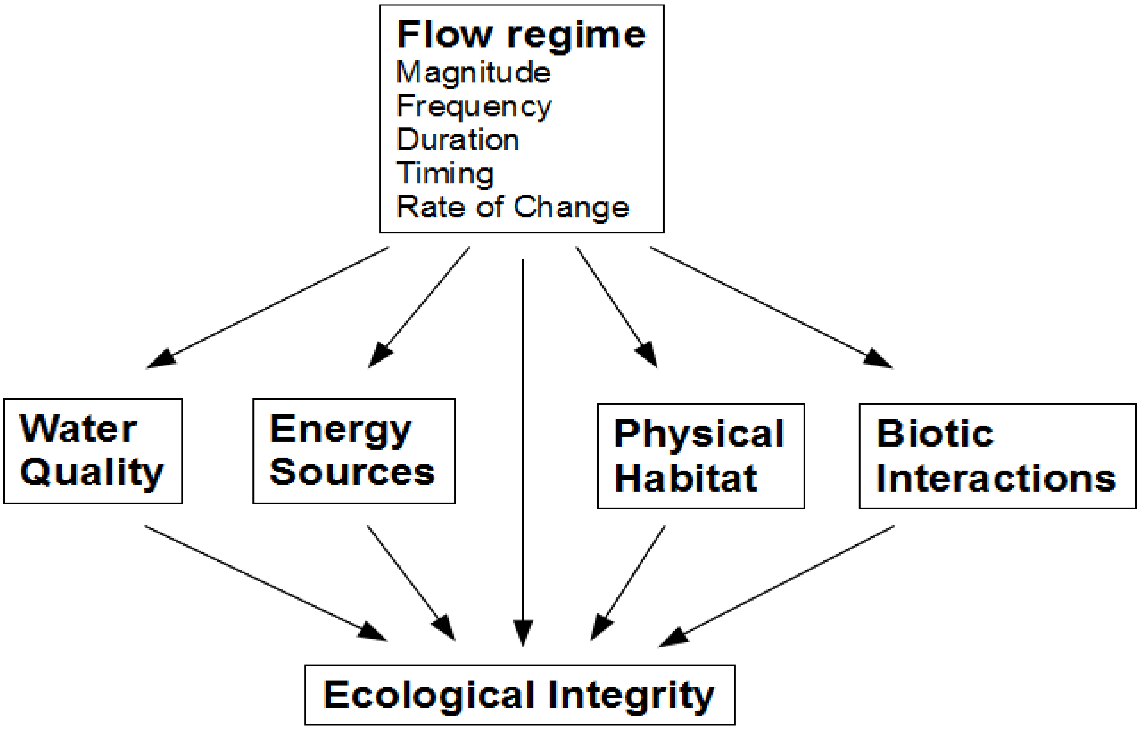

Figure 2.5: Links between five facets of the flow regime and ecological integrity (modified from Poff et al., 1997).

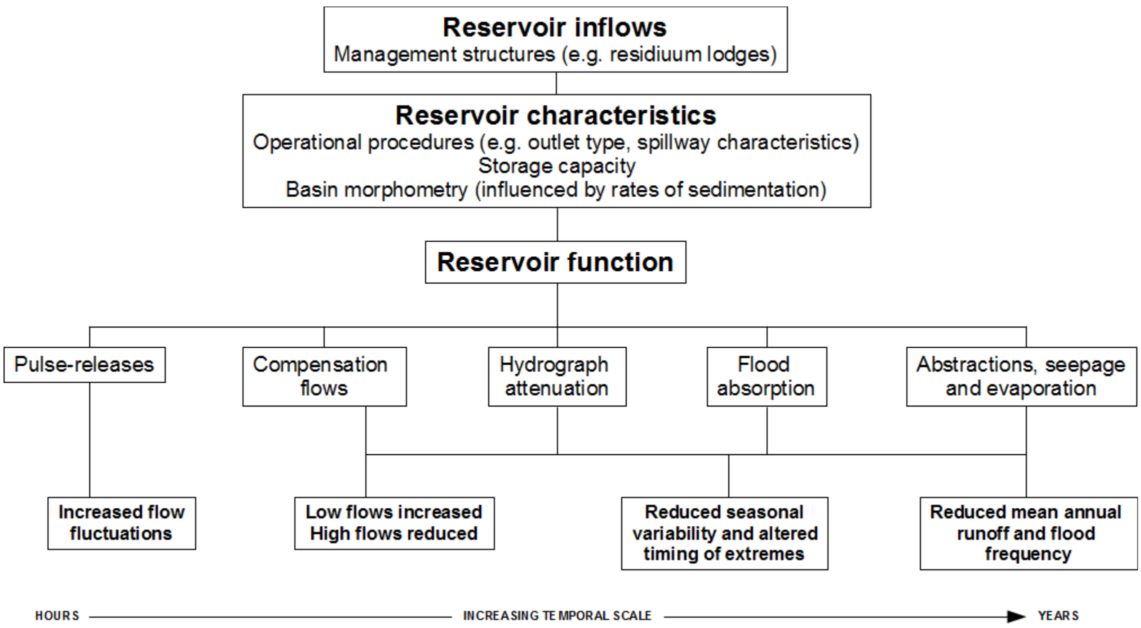

Figure 2.6: Reservoir specific factors that can affect downstream flow regimes (modified from Petts, 1984a).

Table 2.1: Examples of hydrological alteration attributed to upstream regulation.

Stream, Country	Observed hydrological changes	Source
Reduced average annual runoff		
River Zambezi, Mozambique	Saltwater incursion in the coastal floodplain and delta were induced by reduced freshwater discharges	Hall, 1977
Yangtze River, China	Slightly reduced annual discharge due to diversion of water from reservoirs for agricultural use	Chen et al., 2001
Indus River, Pakistan	Reduced by nearly 80% after headwater dam construction	Farnsworth & Milliman, 2003
Flow variability		
River Ebro, Spain	Reduced annual variability due to reduced autumn and winter peaks and increased flows during summer to aid downstream agriculture	Batalla et al., 2004
Colorado River, USA	Reduced annual, but increased daily flow variability due to hydropower installation	Andrews & Pizzi, 2000
Altered timing of annual extremes		
Salt River, USA	Shift in annual period of high flows from March - May to March - September	Fenner et al., 1985
River Jordan, Israel	High flow period has shifted from winter to summer due to irrigation and power demands on reservoirs	Ortal & Por, 1978
Reduced flood magnitudes		
Salt River, USA	Overall reduction in high flow events	Fenner, et al., 1985
Murray-Darling basin, Australia	The magnitude of average annual floods (annual exceedance probability 50%) has been reduced by over 50%	Maheshwari et al., 1995
McKenzie River, USA	Since the construction of the two dams in the 1960s, peak discharges have been reduced by more than 50%	Ligon et al., 1995
Imposition of unnatural pulses		
Rio Tera, Spain	In March, daily flow fluctuations of between 10 and 210 m^3/s are typical due to demand for hydroelectric power	Garcia De Jalon et al., 1994
Colorado River, USA	Daily fluctuations of water-depth by about 1.5 m	Turner & Karpiscak, 1980

2.3.2 Physical-chemical impacts

Stream physical chemistry (also known as 'water quality') is a function of a variety of parameters such as water temperature, inorganic chemistry, metals and organic compounds (Binkley & Brown, 1993). Its combined characteristics are one of the primary controllers of stream biological integrity (Karr & Dudley, 1981) (Figure 2.7), for example, affecting reproductive success and competitive ability (Karr & Dudley, 1981). The characteristics of physical chemistry are therefore commonly measured to assess the 'health' or 'quality' of stream ecosystems (Norris & Thoms, 1999).

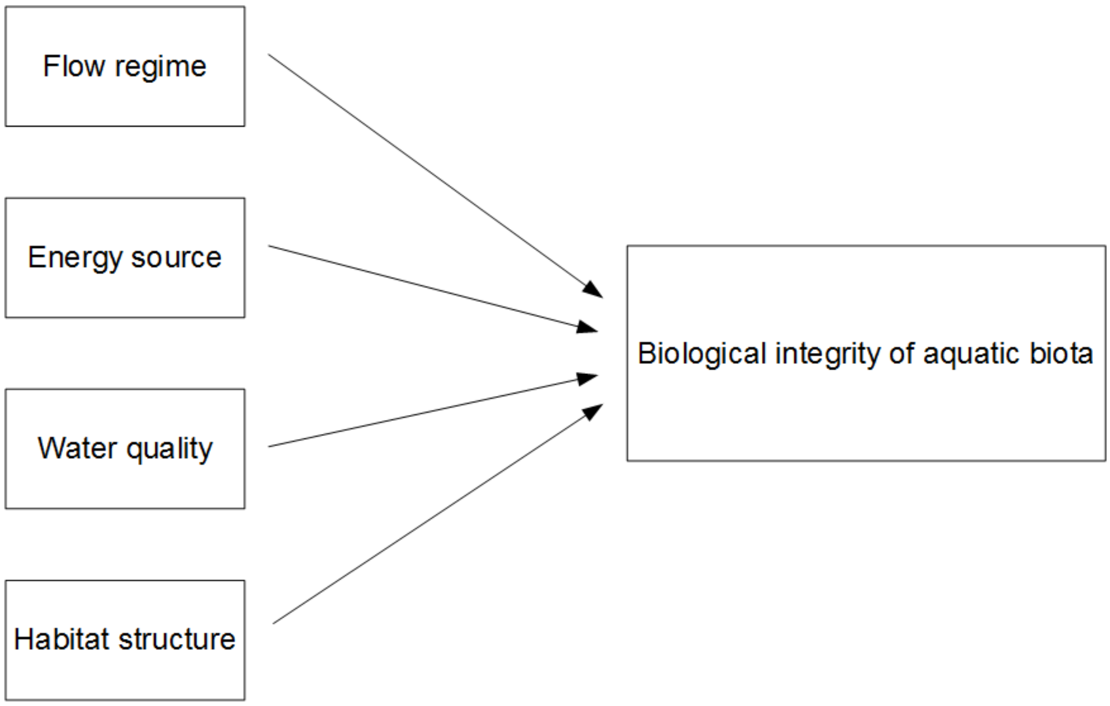

Figure 2.7: Primary variables affecting biological integrity of aquatic biota. Modified from Karr & Dudley (1981).

Stream regulation can significantly impact downstream physical chemistry and the type and magnitude of these impacts are primarily driven by a combination of (i) the physical chemistry of upstream reservoirs, and (ii) environmental factors independent of any reservoirs (e.g. stream temperature may be affected by air temperature in addition to reservoir water temperature) (Friedl & Wüest, 2002). When a stream is dammed, it undergoes a transformation from lotic to lentic environment immediately upstream of the impoundment; this change can drastically impact both the physical environment and the prevailing processes (Baxter, 1977; Friedl & Wüest, 2002). The determinants of reservoir physical chemistry are complex, but Hannan (1979) identified six key factors : (i) the physical-chemical characteristics of reservoir inputs, (ii) thermal, biological and chemical processes, (iii) water residence time, (iv) waterbody size and shape, (v) catchment characteristics (e.g. geology, vegetation and micro-climate) and (vi) reservoir operation and age.

One of the most important factors that can influence facets of reservoir physical chemistry is thermal stratification (Petts, 1984a; Cassidy, 1989). Perhaps the most important is de-oxygenation of the hypolimnion through processes such as aerobic respiration. This has a direct impact of lowering the dissolved oxygen content of the water, but it can also indirectly cause the production of hydrogen sulphide, the release of carbon dioxide, a reduction in pH and increased electrical conductivity, alkalinity and orthophosphate through anaerobic breakdown of organic matter (Petts, 1984a). These conditions can also lead to the dissolution of metals such as iron and manganese that were previously adsorbed to sediment (Petts, 1984a; Inverarity et al., 2003). This factor, combined with others can directly influence downstream physical chemistry and some of the commonly observed impacts are discussed below.

Water temperature is considered a key characteristic of streams as it affects the metabolism of organisms both directly (Beschta et al., 1987) and indirectly (Macan, 1963), is an important influencer of other physical-chemical properties of water (e.g. dissolved oxygen) and has economic significance (Webb et al., 2008). Stream regulation has been documented to impact water temperature regimes at a variety of temporal and spatial scales but, prior to examining these impacts, it is important to appreciate the physics of stream temperature.

Caissie (2006) identified an abundance of factors which can affect stream water temperature (Figure 2.8), but the typical primary determinants are climate (e.g. solar radiation, air temperature), stream morphology, groundwater influences and riparian canopy cover (Sullivan & Adams, 1991). In natural streams, these determinants typically result in the following spatial and temporal patterns (Caissie, 2006): (i) mean daily temperature generally increases with distance from source; (ii) temperature varies on both a diurnal and annual scale. However, when a stream is dammed, these characteristics can be interrupted and, in some cases, extirpated.

Stream regulation has been observed to have mixed impacts on mean stream temperature: Lavis & Smith (1972) found negligible downstream impact, Cowx et al. (1987) observed reduced mean temperatures during summer and Dickson et al. (2012) found increased mean temperatures during both winter and summer, suggesting local factors are important in determining impact. However, a general consensus that reservoirs reduce the annual and diel range of stream temperature (e.g. Lehmkhul, 1972; O' Keeffe et al., 1990 and Webb and Walling, 1993 respectively) is evident. This is primarily due to the relatively large heat capacity of a reservoir cf. a stream and tendency for water to be drawn from the relatively thermo-stable hypolimnion resulting in relatively little variation in temperature of water from the reservoir outlet (Petts, 1984a; Cassidy, 1989). However, although general impacts on temperature range are understood, more recent studies have identified complex spatio-temporal impacts on downstream temperature regimes attributed to factors such as reservoir operation and groundwater influence (e.g. Webb & Walling, 1997).

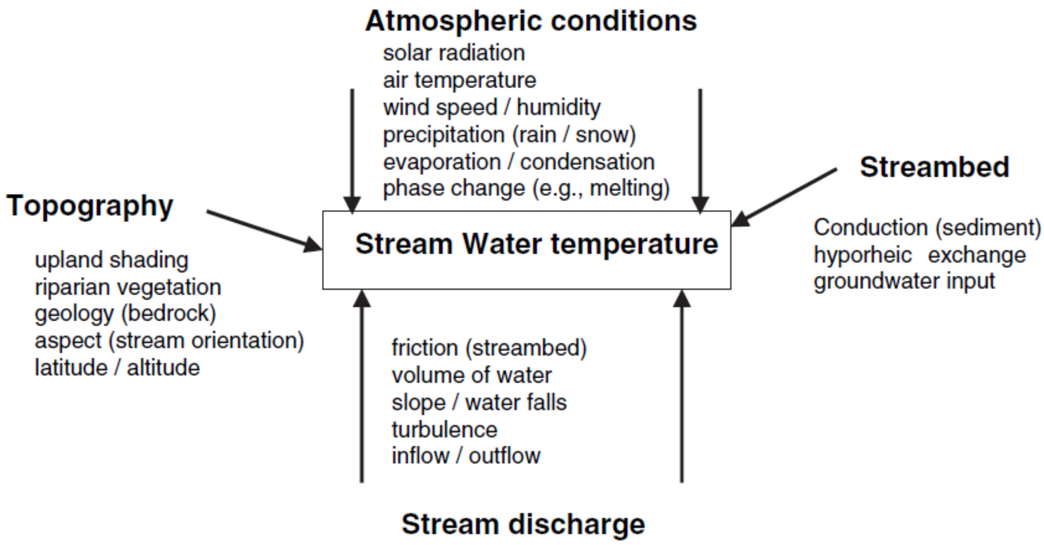

Figure 2.8: Determinants of stream temperature (Caissie, 2006).

Changes to the dissolved oxygen (DO) regime of streams post regulation have also been reported, but consistency in observations is lacking. A reduction in DO has been most commonly noted (Walker, 1985; Palmer & O'Keeffe, 1990; Bunn & Arthington, 2002), but no impact (Crisp, 1977) and increased DO due to gas supersaturation caused by aeration under pressure at the reservoir outlet (Petts, 1984a; Cassidy, 1989; Lutz, 1995) have also been observed indicating the potential importance of site specific factors.

Evidence of increased concentrations of metals (e.g. iron and manganese) (Petts, 1984a; Scullion et al., 1982), increased pH (Palmer & O'Keeffe, 1990) and reduced annual and seasonal range of electrical conductivity (EC) (Soja & Wiejaczka (2013) and Palmer & O'Keeffe (1990) respectively) has also been reported. Furthermore, regulation has been noted to both increase and decrease nutrient concentrations downstream of separate reservoirs within the same region (O'Keeffe, 1990). It is important, however, to note that typically, these observations are based on sampling strategies of short duration (i.e. sub-seasonal) (e.g. Scullion et al., 1982) or low frequency (i.e. daily/ monthly measurements) (e.g. Soja & Wiejaczka, 2013 and Palmer & O'Keeffe, 1990) resulting in a lack of understanding of these particular physical-chemical impacts over a variety of time scales (e.g. diurnal, weekly and seasonal) and during particular events (e.g. floods).

This review of publications reporting impacts of stream regulation on downstream physical chemistry has highlighted variation in impact on mean stream temperature, the importance of site specific factors in determining stream temperature and DO, the limited number of publications focussing on other physical-chemical facets such as pH and EC and potential methodological limitations. There is therefore a requirement for detailed, site specific studies to be undertaken (Poff & Zimmerman, 2010) to address these factors. The completion of such studies will enable informed management at a local level to be undertaken which is essential to the achievement of the aims of contemporary freshwater legislation (e.g. the EU WFD).

2.3.3 Morphological impacts

Fluvial geomorphology has been defined as:

> "the study of sediment sources, fluxes and storage within the river catchment and channel over short, medium and longer time scales and of the resultant channel and floodplain morphology" (Newson & Sear, 1993).

It can be conceptually modelled through the interaction between channel form, flow, sediment transport and grain size (Figure 2.9) (Newson & Sear, 1993). Stream morphology is important as it influences biological integrity (Figure 2.7) through control of nutrient and energy flux and habitat for vegetation, periphyton, invertebrates and fish (Ligon et al., 1995). Anthropogenic factors such as stream restoration and regulation influence natural facets of fluvial geomorphology (Figure 2.9). Understanding these influences is therefore crucial to enable informed management of these systems to mitigate any adverse impacts and meet targets of contemporary legislation such as the EU WFD.

Understanding the morphological changes associated with stream regulation has been described as the key to comprehending long term ecological consequences (Ligon et al., 1995). Dams can influence downstream morphology in a number of ways, for example, they can act as a barrier to movement of fine and coarse sediment (Baxter, 1977; Simons, 1979; Petts, 1984a) and modify the flow regime which affects bed shear stress, bedload transport, erosion and deposition and ultimately channel form (Figure 2.9) (Carling, 1988; Newson & Sear, 1993). Interaction of these factors can result in a number of key morphological changes which are discussed in the following text.

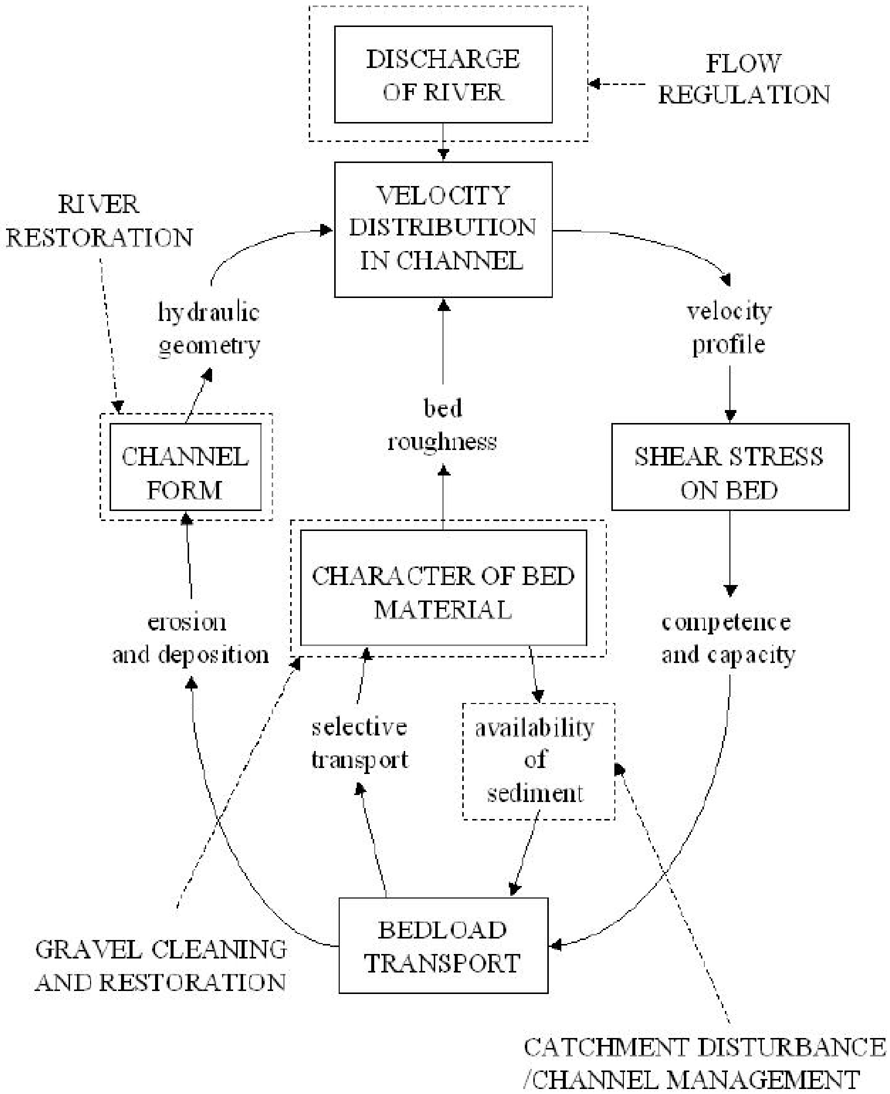

Figure 2.9: Factors that can influence fluvial geomorphology. Source: Newson & Sear (1993).

Stream regulation has been associated with three key downstream morphological impacts. First, water released from a reservoir is often devoid of suspended sediment (Petts, 1984a). This can result in erosion in the downstream channel (e.g. downstream of the Three Gorges Dam, China (Xu et al., 2006)) as it has a high capacity to entrain and transport sediment (Petts, 1984a). This results in a coarsening of the streambed which is commonly termed bed 'armouring' (Carling, 1988; Brandt, 2000). Second, transport of almost all coarse sediment to the downstream reach can be eliminated as the reservoir acts as a sink (Petts, 1984a). When this is combined with high flow events, degradation (removal of all sediment clast sizes) of the downstream bed can occur as sediment is not replenished (Brandt, 2000). Last, dams typically reduce/ eliminate the number and magnitude of high flow events resulting in bed stabilisation as threshold discharges required to rework sediments are not met (Carling, 1988; Brandt, 2000; Wellmeyer et al., 2005).

The interaction between discharge and the capacity to transport sediment is therefore an important concept in determining the impacts of regulation. Brandt (2000) presented nine cross-sectional morphological scenarios based on this relationship (Figure 2.10) and Table 2.2 demonstrates the range of impacts that have been observed globally.

Downstream morphological impacts of regulation can change at both spatial and temporal scales. Longitudinally, morphological impacts reduce as regulated streamflow influence reduces. For example, Petts (1980) suggested that for the UK, where the impounded catchment area was less than 35–40% of the total drainage area, no significant impact could be identified. Temporally, significant downstream morphological impacts have been observed relatively soon after dam-closure (e.g. < two months (Williams & Wolmann, 1984)) but such changes are more typically observed within two years (Brandt, 2000). This period of change is termed the 'relaxation period' (Petts, 1987). During the relaxation period, changes are dynamic and are driven by episodic events reflecting the frequency of major discharge events and periods of stability (Petts, 1987). These temporal changes have been conceptualised by Petts (1987) as the Transient System Model (Figure 2.11). After the relaxation period, a quasi-equilibrium period is reached where relatively small scale changes occur, but major change in system status is not observed (Petts, 1987).

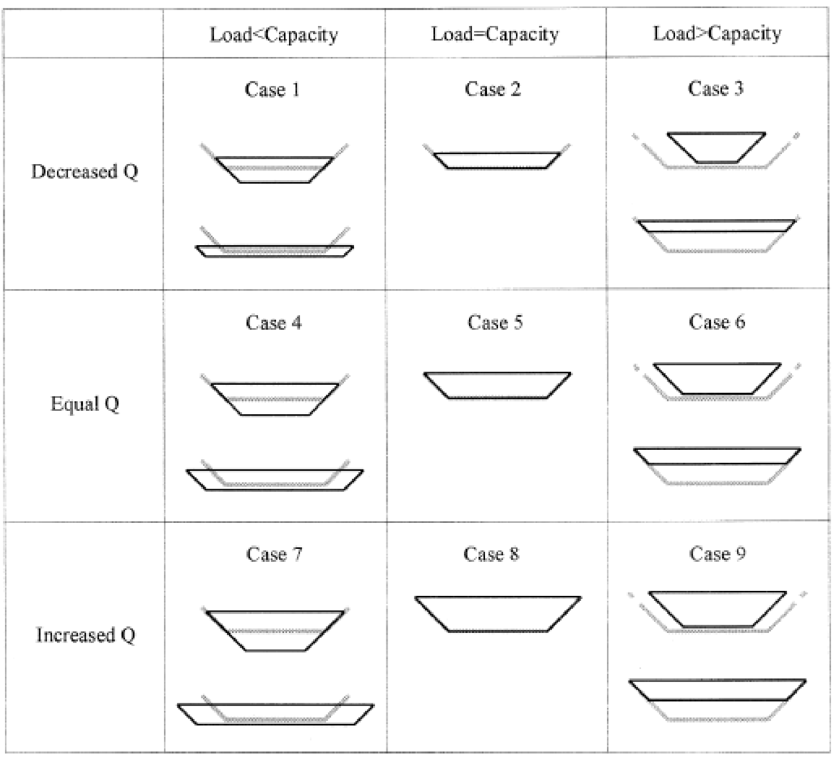

Figure 2.10: Cross sectional morphological possibilities based on the interaction between discharge (Q) and sediment load and capacity of the stream water. Grey = pre-impoundment, black = post-impoundment. Source: Brandt, 2000.

Table 2.2: Examples of stream morphological impacts of upstream impoundment and respective Brandt (2000) classification.

Reservoir, stream, country	Observation(s)	Source	Brandt classification
Clathworthy, Tone, UK	Reduced cross-sectional capacity, little evidence of scour	Gregory & Park, 1974	Case 1
Catcleugh, Rede, UK	Reduced channel width and cross sectional area	Petts et al., 1993	Case 1
Kielder, North Tyne, UK	Incision of riffles and bed armouring immediately downstream of dam	Sear, 1995	Case 1
River Dee, between Farndon and Bangor, UK	Reduced channel width and increased sedimentation since the 1960s (the period of highest flow regulation)	Gurnell, 1997	Case 1
Nant-y-Moch, Rheidol, UK	Transport only of fine gravels and a resultant coarsening of the bed material	Greenwood et al., 1999	Case 1
Nant-y-Moch, Rheidol, UK	Deposition at a tributary confluence	Petts, 1984b	Case 2
Various, China	Redistribution of sediment transport through time resulting in deposition and bed aggradation	Chien, 1985	Case 3
Various, China	Erosion downstream of reservoir due to increased transport capacity of sediment stripped water	Chien, 1985	Case 4
Various, USA	Deepening of channel due to hydropower operation	Williams and Wolman, 1984	Case 5
Gebidem, Massa, Switzerland	Deposition due to flushing of reservoir to remove sediment	Boillat et al., 1996	Case 6
Cachi, Reventazon, Costa Rica	Deposition due to flushing of reservoir to remove sediment	Brandt & Sweening, 1999	Case 6
Leighs, Ter, UK	Increase in channel capacity after a 10-fold increase in low-flow discharge	Petts & Pratts, 1983	Case 7
Various, Canada	Increased cross-sectional area due to increased discharge and inter-basin diversion	Kellerhals et al., 1979	Case 8

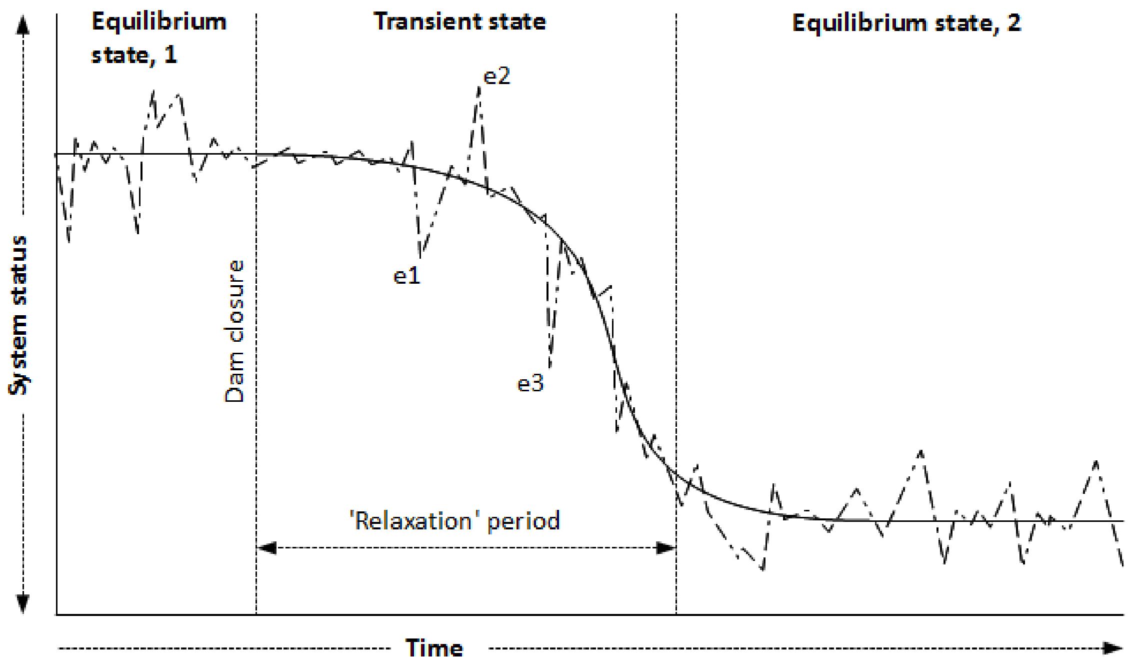

Figure 2.11: Petts' Transient System Model of morphological changes after dam closure driven by episodic events (e1-3) (modified from Petts, 1987).

Variation in observations of geomorphological change combined with the spatial-temporal complexities associated with morphological impacts of regulation on downstream systems have resulted in a clear requirement for detailed assessments of local scale or catchment based approaches to inform management and enable achievement of legislative targets (e.g. EU WFD (EC, 2000); Australian Water Act (2007)). Such assessments should not just focus on morphological impacts, but also aim to build a process (e.g. sediment transport) based understanding of how impacts manifest (Petts & Lewin, 1979; Carling, 1988). To date, limited research has focussed on understanding such processes and those that have (e.g. Gilvear, 1987; Lyons, 1992; Sear, 1993) have primarily focussed on transport of suspended sediment, most likely reflecting the limitations associated with monitoring the coarse component (Reid et al., 2007; Turowski & Rickenmann, 2011). This has resulted in a dearth of process based understanding of coarse sediment transport in regulated streams. A detailed understanding of elements of the relationship between discharge and sediment transport, including an appreciation of temporal variability and controlling factors (e.g. antecedent conditions), is therefore required to inform and enable successful management of these systems. For example, implementation of sediment transport specific mitigation measures under the EU WFD (UKTAG, 2008).

2.3.4 Biota

Stream biota are primarily affected by the flow regime, energy sources, physical chemistry and habitat structure of a stream (Figure 2.7; Karr & Dudley, 1981). They are therefore seen as an indicator of the 'health' of a system and are increasingly used over, or in combination with, alternative indicators such as physical chemistry (Norris & Thoms, 1999) and their monitoring has accelerated in recent years due to integration into worldwide legislation (e.g. EU WFD (EC, 2000), Clean Water Act (USC, 2002)) (Norris & Thoms, 1999). Biota are also important from a conservation perspective as conserving biodiversity is seen as essential in maintaining ecosystem function (Srivastava & Vellend, 2005). Biota also provide ecosystem services which are valued by humans, for example, stream leaf litter breakdown by organisms provides resources for food webs which sustain fish that are consumed by humans (Meyer et al., 2005). Identification and assessment of anthropogenic impacts on stream biota (e.g. stream regulation) is therefore a key priority in terms of maintaining ecosystem health, meeting legislative targets, conserving biodiversity and preserving ecosystem function and services.

It is useful to consider a number of general biological concepts to understand how stream regulation may affect biota. In general terms, natural streams can be considered as continuums where organisms are spatially arranged in a relatively predictable order from headwater to stream mouth (Vannote et al., 1980). However, Ward & Stanford (1983; 1995) suggested streams rarely follow this paradigm and theorised that streams should be viewed as an alternating series of lotic and lentic reaches. This theory was then used to suggest how structures such as dams could affect stream biota and was subsequently found to provide generally accurate predictions (Standford & Ward, 2001). Central to this concept was a recognition of the importance of disturbance (Resh et al., 1988). Disturbance can be defined as:

> “any relatively discrete event in time that is characterized by a frequency, intensity, and severity outside a predictable range, and that disrupts ecosystem, community, or population structure and changes resources or the physical environment” (Resh et al., 1988)

When considered in terms of stream regulation, disturbance can be used to predict or explain biotic response. For example, biotic diversity has been theorised to be maximised at an intermediate disturbance intensity (Connell, 1978); stream regulation can modify the intensity of disturbance events such as floods, thereby causing a predictable biotic response (e.g. Townsend et al., 1997). Nonetheless, it is also important to note that such theories have received criticism (e.g. Kondoh, 2001; Fox, 2013) and it should therefore be highlighted that whilst useful in some instances, such concepts are simplifications of complex interactions between mechanisms and processes that determine biotic assemblage (e.g. Cardinale et al., 2006).

Biotic response to stream regulation has been intensively studied across a wide range of taxonomic groups (Table 2.3) and Baxter (1977) and Bunn & Arthington (2002) provide detailed reviews. One group that has received particular attention is macroinvertebrates (Metcalfe, 1989; Norris & Thoms, 1999). Stream macroinvertebrates are considered one of the best groups of organisms for study as they are differentially sensitive to environmental pressures, ubiquitous, abundant and easy to collect (Metcalfe, 1989). A vast literature also exists regarding these organisms' ecology and phenology allowing for informed assessment of environmental conditions to be made (Metcalfe, 1989). Suitability of these organisms for this role is reflected in their integration into contemporary freshwater management legislation (e.g. the EU WFD (EC, 2000) and the Clean Water Act (USC, 2002)) where they are used as key indicators of environmental impact and change.

Table 2.3: Range of taxonomic groups assessed for impact of upstream impoundment and example respective publications.

Taxonomic group	Publications
Algae	Jones & Barrington (1985)
Birds	Gill (1973)
Fish	Baran et al. (1995); Gehrke et al. (1995); Linnik et al. (1998); Agostinho et al. (2004); Korman et al. (2004)
Macroinvertebrates	Armitage (1978); Englund & Malmqvist (1996); Munn & Brusven (1991); Nichols et al. (2006); Spence & Hynes (1971)
Macrophytes	Garcia De Jalon et al. (1994); Bernez et al. (2004)
Mammals	Ballard et al. (1998)
Riparian vegetation	Nilsson et al. (1991); Johnson (1998); Jansson et al. (2000)

From a theoretical perspective, undisturbed stream macroinvertebrate communities are expected to be characterised by relatively high numbers of shredders in headwaters, grazers in mid-order and collectors in high-order streams, largely reflecting food and energy source availability (Vannote et al., 1980). The introduction of a dam into headwaters has been predicted to dramatically reduce numbers of shredders due to alteration of the food supply (i.e. reduction in CPOM due to dam-trapping effect) but have little impact if constructed on higher order streams due to the reduced reliance on allochthonous food sources (Ward & Stanford, 1983). However, it has also been acknowledged that more complex impacts reflecting the interaction of several key factors (Figure 2.7) (Karr & Dudley, 1981) are likely to be observed.

The impact of stream regulation on macroinvertebrates downstream of impoundments has been the focus of many studies worldwide, yet varied observations have been made leading to a lack of clarity in understanding. In response to regulation, total abundance has been recorded to increase at some sites (e.g. Armitage, 1978; Munn & Brusven, 1991) but decrease at others (Englund & Malmqvist, 1996). Additionally, responses of some taxonomic groups have been observed to vary, for example, no change or decrease in abundance of Coleoptera (Spence & Hynes, 1971 and Nichols et al., 2006, respectively). Furthermore, varied responses in diversity have also been observed (increased: Poole and Stewart, 1976; Penaz et al., 1968; Maynard and Lane, 2012; decreased: Pearson et al., 1968; Armitage 1978; Munn & Brusven, 1991). Further research is therefore required to both clarify and understand the mechanisms behind these differential responses.

Some assessments regarding the impact of regulation on stream macroinvertebrate assemblages have been of limited intensity (i.e. ≤ 2 samples per year (e.g. Englund & Malmqvist 1996; Maynard & Lane, 2012; Gillespie et al., in press (a))) potentially resulting in failure to identify intra-annual variation of impacts. This may explain variation in observations as impacts are likely to vary both intra- and inter-annually as identified by Armitage (1978). An assessment of the impact of regulation on macroinvertebrate community dynamics at a fine temporal scale (e.g. utilising a monthly sampling regime) may have the potential to reveal novel insights (e.g. temporal variation in magnitude of impact) into impacts of regulation.

2.4 Summary

Humans have regulated stream systems since neolithic times and the first known dam was built by the Romans to aid water supply. Dams are now ubiquitous across the world, bringing advantages such as reduced severity of floods and irrigation and recreational opportunities. However, dams can also have a detrimental effect on the environment. Humans now recognise this and in an attempt to mitigate these impacts, research is required to understand them and how they may be reduced.

There is a large literature on the impact of dams on downstream ecosystems and, whilst some consistency in impacts can be seen (e.g. dams typically reduce the transport of sediment downstream), there are many examples where consistency of impact is not evident (e.g. total abundance of macroinvertebrates has been observed to increase in some streams but decrease in others). A recognition of the importance of local scale factors (e.g. climate, geology, dam operation) has driven the requirement for detailed, site specific studies to be undertaken to inform stream managers and enable them to meet obligatory legislative targets under increasingly ecocentric water legislation (e.g. EU WFD).

3 MITIGATION OF IMPACTS OF REGULATION USING ENVIRONMENTAL FLOWS

3.1 Chapter overview

This chapter presents a strategic review of the global impacts of environmental flows on downstream ecosystems in an attempt to establish whether consensus in response exists. First, the rationale for this review is presented including identification of the aims. This is then followed by a description of the methods used to undertake the strategic review. Subsequent sections present and discuss the results and include recommendations for future research.

3.2 Introduction

A drive to mitigate impacts of stream regulation on hydrology, physical chemistry, geomorphology and biota through reservoir outflow modification has recently been stimulated. These interventions are commonly described as environmental flows and it has been suggested that their implementation will be vital to meet the aims of contemporary legislation (e.g. the Australian National Water Initiative (Connell & Grafton, 2008) and the EU WFD (Acreman & Ferguson, 2010)).

Environmental flows have been defined as:

> “the quantity, timing, duration, frequency and quality of water flows required to sustain freshwater, estuarine and near-shore ecosystems and the human livelihoods and well-being that depend on them" (Acreman & Ferguson, 2010).

More specifically, Acreman et al. (2009) suggested that environmental flows should

> "be based on ecological requirements of different communities/ species/ life stages, which may vary within and between streams even for the same biological elements or communities".

It is clear that to define environmental flows for regulated streams, identification of cause-response relationships between flow modification and ecosystem variables must be achieved. A synthesis of the global literature has the potential to identify and quantify reservoir outflow modification-downstream ecosystem response relationships and assess current research methods and topics to clarify and prioritise future research agendas. However, to date no such study has been undertaken. Thus, this review aims to (i) develop and employ an objective methodology to search the literature for relevant studies; (ii) synthesise data on qualitative and quantitative relationships between reservoir flow modification and downstream ecosystem responses, and the sampling methods and analytical approaches used; (iii) critically evaluate the existing knowledge base and propose recommendations for future research to advance the science of regulated stream management.

3.3 Methods

3.3.1 Literature search

Relevant published literature was located through computerised searches of ISI Web of Knowledge which includes the following databases: Web of Science (1990-present), BIOSYS Citation Index (1969-present), BIOSYS Previews (1969-present), Data Citation Index (1900-present), MEDLINE (1950-present) and Journal Citation Reports. Table 3.1 lists the search terms used and number of results returned. A total of 3,981 records were assessed for suitability through attainment of the following criteria: (i) reported primary data; (ii) assessed the impact of modification of the outflow regime of a reservoir; (iii) focused on impacts to instream ecosystems (biotic and abiotic elements) downstream of the reservoir; (iv) were published in academic journals and had thus undergone peer review.

3.3.2 Data extraction and quality assessment

First, the study location(s) (reservoir where flow modification was made) reported in each study was recorded and plotted to assess any spatial patterns or biases in the literature. Next, ecosystem responses to flow modification highlighted in each study were recorded and categorised as either biotic or abiotic. Biotic changes were assigned to either *reduced*, *no change* or *increased* response categories to allow for comparison of general trends (see Poff & Zimmerman, 2010). For example, increased macroinvertebrate diversity in response to flow modification was classified as an *increased* response. Conversely, a reduction in fish movement in response to flow modification was classified as a *reduced* response. Additionally, biotic responses were split into native or non-native/ invasive groups where detail was given as each group may respond differently to flow modification (e.g. Cross et al., 2011). Abiotic responses were assigned to either *change* or *no change* categories as reductions or increases in abiotic parameters may be less comparable than for biotic responses (e.g. increased temperature and electrical conductivity (EC) are less likely to both be either ecologically 'good' or 'bad' than increased fish and macroinvertebrate abundance). Ecosystem responses were assigned to either: (i) fish; (ii) macroinvertebrates; (iii) macrophytes; (iv) primary producers (benthic); (v) morphology; (vi) physical chemistry (including suspended sediment transport) and (vii) other categories to enable further insight to be made regarding the types of ecosystem components assessed.

Flow modification can often be classified as more than one type of response; for example, a change in flow from a reservoir may result in both an increase in flow magnitude and duration (Poff et al., 1997). Thus, to classify the type of flow modification each ecosystem response was associated with, the element of flow modification that was most emphasised by each study was recorded (Poff & Zimmerman, 2010). Ecosystem responses have been observed to vary whether they are as a result of a single, or a series of flow modifications (e.g. Uehlinger et al., 2003). Thus, to allow for separate analysis of these two modification types, responses were further classified by whether they were reported as a result of a *single* or series of *cumulative* flow modifications. Ecosystem responses within each category were then synthesised and commonly reported responses were tabulated. To allow for clear tabulation of results, a frequency of observation of at least four was selected to represent a 'common' observation.

Table 3.1: Search terms used in literature search and respective number of results.

Search term	No. results
"reservoir operation"	825
effects AND hydropower	749
"selective withdrawal"	202
"reservoir release*"	200
"varying flows"	182
"pulse release*"	154
"controlled flood*"	129
"artificial flood*"	124
"dam operation"	124
"environmental flow*" AND dam	112
"experimental drought*"	110
"artificial flow*" NOT flower*	97
"flushing flow*"	93
"experimental flood*"	89
"hydropeaking"	83
"environmental flow*" AND reservoir	67
"dam release*"	65
"managed flood*"	56
"e-flows"	50
"artificial release*"	42
"flow alteration*" AND dam	42
"artificial drought"	40
"hydrop* flow*"	34
"planned flood*"	22
"altered flow* regime"	21
"flow alteration*" AND reservoir	21
"reservoir flushing"	20
"peaking flow*"	20
"scour* flow*"	18
"flood program"	18
"hydro-peaking"	17
"test flood*"	16
"hydropower peaking"	15
"environmental flow*" AND impoundment	14
"altered flow*" AND reservoir	11
"spate flow*" NOT flower*	10
"environmental release*" AND reservoir	10
"fluctuating flow*" AND dam	10
"peaking discharge*"	10
"flow alteration*" AND impoundment	9
"scour* flood"	8
"regulated flood"	8
"modified flow* regime"	7
"experimental low flow*"	7
"fluctuating flow*" AND reservoir	6
"dam reoperation"	3
"fluctuating flow*" AND impoundment	3
"spate flood*"	2
"dam re-operation*"	2
"environmental release*" AND dam	2
"reservoir reoperation"	1
"artificial low flow*"	1
"spate release*"	0
"scour* release*"	0
"reservoir re-operation"	0
"impoundment reoperation"	0
"impoundment re-operation"	0
"modified flow* AND reservoir"	0
"environmental release*" AND impoundment	0

In an attempt to produce quantitative relationships between reservoir outflow modification and ecosystem responses, first, studies where a *single* flow modification and associated ecosystem response could be represented as percent change were identified. This was possible for 20 studies located in the literature search, but some studies reported on more than one flow modification or ecosystem response resulting in extraction of a total of 119 data points relating flow modification to ecosystem response. From initial analysis of data points, all observed ecosystem responses were a result of modification of flow magnitude. Thus, percent change for each flow modification was defined using Equation 3.1 where $x1$ was pre flow modification discharge magnitude and $x2$ was maximum (or minimum in the case of a reduction in magnitude) discharge magnitude of the flow modification. Equation 3.1 was also used for calculation of percent change in ecosystem response where, $x1$ was pre flow modification condition and $x2$ was either condition of maximum change from $x1$ (if sampling was undertaken during flow modification) or condition immediately after the flow modification (if sampling was undertaken after flow modification). If possible, data were extracted from the text/ tables and alternatively from figures. For response variables, where sampling was replicated, mean values were used and where non-significant responses were noted, percent change was recorded as zero.

$$Percent\ change = \left(\frac{x2 - x1}{x1}\right) X\ 100 \qquad \text{(Equation 3.1)}$$

To visualise flow-ecosystem response relationships, data points were organised by response type using the seven categories employed in qualitative data extraction and, where more than five data points reported on the same ecosystem response, plots of flow (percentage change) versus ecosystem response (percentage change) were created. For some ecosystem response types, visualisation revealed broadly linear relationships; the significance of these relationships was assessed using Generalized Linear Models (GLM) with appropriate error distribution and link functions specified. Statistical analysis on fewer than 10 data points has been regarded as invalid (Roscoe, 1975) and therefore modelling was only carried out where at least 10 data points had been extracted. Model validation was carried out to ensure approximate normal distribution, independence and homogeneity of residuals. Significance of relationships was assessed through consideration of t-statistics and associated p-values (e.g. Zuur et al., 2009). All visualisations and statistical analyses were undertaken in R v2.15.3 (2013) and relationships were considered significant where $p < 0.05$.

To allow assessment of current research standards and aid recommendation for future research strategies, the following were recorded: (i) the type(s) of sampling strategy used to detect ecosystem responses (quantitative, semi-quantitative or qualitative); (ii) whether randomisation or replication was stated as used in sampling; (iii) type(s) of control sites used (if any) (e.g. upstream of reservoir; nearby unregulated stream); and (iv) analytical approaches used for each study.

3.4 Results

Most studies were located within North America and western Europe and there was a dearth within equatorial regions, South America, north Africa, Asia and eastern Europe (Figure 3.1A). Two study locations had notably high densities of work: Lake Powell (Glen Canyon Dam), USA and Lago di Livigno (Punt dal Gall Dam), Switzerland/ Italy (Figure 3.1B).

Figure 3.1: Location (A) and density (B) of the 76 studies considered within this review.

3.4.1 Qualitative analysis of assembled datasets

The majority of studies (n = 69) focussed on modified flow magnitude, with very few studies reporting on changed reservoir draw-off valve (n = 1), modified flow duration (n = 2), range (n = 2) and rate of change (n = 2) (Figure 3.2). Studies reporting fish response were the most frequent (n = 28) and a relatively high number of studies reported physical-chemical and macroinvertebrate responses (n = 27 and 19 respectively). In contrast, few studies reported on macrophytes and primary producers (n = 3 and 12 respectively) (Figure 3.3). A total of 55 and 21 studies reported ecosystem responses as a result of *single* or *cumulative* modifications in flow magnitude, respectively. However, only seven studies reported ecosystem responses associated with either rate of flow change, duration, range and draw-off depth from the reservoir (Table 3.2).

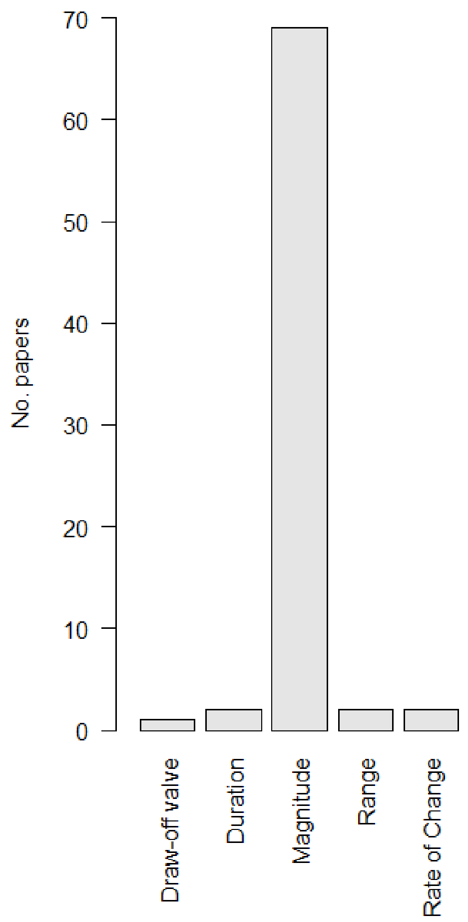

Figure 3.2: Number of studies (from a total of 76) that reported on each flow modification type.

Numerous studies detailing ecosystem responses to flow magnitude modification reported increased biotic responses (n = 35), although a similar number of studies reported decreased or no change in biotic response (n = 30 and 25 respectively). This trend was mirrored in ecosystem responses as a result of *single* flow magnitude modifications, but for *cumulative* modifications in flow magnitude, the majority of studies reported decreased biotic responses (Table 3.2). *Single* modifications of flow magnitude were commonly reported to result in: (i) both increased and no change in fish movement (during flow modification); (ii) no change in fish abundance (after flow modification), and (iii) increased macroinvertebrate drift (during flow modification) and reduced macroinvertebrate density (after flow modification). Similarly, *cumulative* modifications of flow magnitude were associated with reduced macroinvertebrate density and, additionally, reduced periphyton mass (after flow modification).

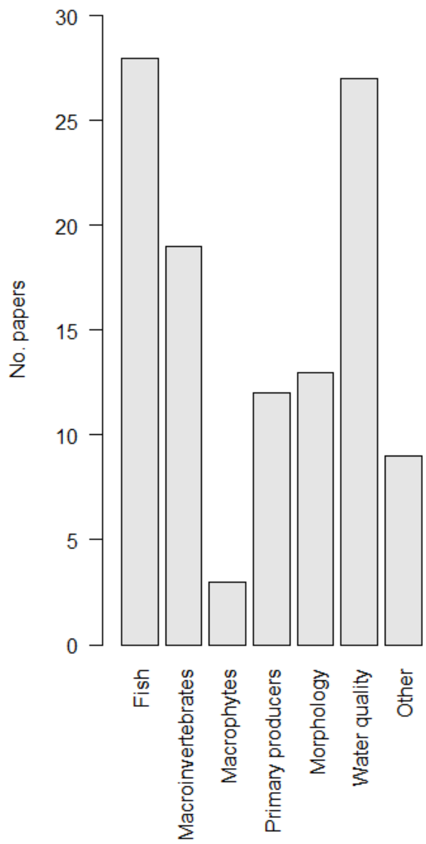

Figure 3.3: Number of studies (from a total of 76) that reported on each ecological response type. N.B. some studies reported on more than one category.

Table 3.2: Number of studies that reported on each flow modification, decreases, no changes or increases in biotic and abiotic flow modifications and the most common ecological responses reported from a literature review of 76 studies. Where possible, reports are split between impacts of single (S) and cumulative (C) flow modifications. Study reference numbers are shown in parentheses (see Appendix A for study details).

			Biotic responses				**Abiotic responses**		
Flow modification		**Total no. studies**	**No. studies reporting reduced ecological responses**	**No. studies reporting no changes**	**No. studies reporting increased ecological responses**	**Common ecological responses**	**No. studies reporting change**	**No. studies reporting no change**	**Common ecological responses**
Magnitude	S	55	12	14	21	No change in fish movement (10,18,31,35,37,60,75) Increased fish movement (15,18,27,35,37,57) No change in fish abundance (13,65,72,75) Increased macroinvertebrate drift (17,20,42,43,48,62) Reduced macroinvertebrate density (34,48,61,63,54)	32	9	Increased turbidity (6,7,34,49) Increased suspended solid concentration (14,25,32,34,56,63,68,73) Reduced electrical conductivity (19,34,56,73) Increased bedload transport (12,24,36,55,59,66,68) No change in temperature (34,37,45,63) Increased temperature (18,22,37,42,51)
	C	21	14	9	10	Reduced macroinvertebrate density (20,29,43,45,63,64) Reduced periphyton mass (21,26,30,45,74)	4	0	n/a
Rate of change	S	2	1	1	1	n/a	0	0	n/a
Duration	S	2	1	0	1	n/a	1	0	
Draw off depth	S	1	0	0	0	n/a	1	0	n/a
Range	C	2	1	1	2	n/a	0	0	n/a

The majority of studies reported changes in abiotic condition as a result of both single and cumulative modifications in flow magnitude. Common responses were identified as: (i) increased turbidity, suspended solid concentration and bedload transport (during flow modification); (ii) reduced EC (during flow modification); and (iii) both no change and an increase in stream temperature (during flow modification) (Table 3.2). Due to the limited number of studies reporting ecosystem changes as a result of other flow modification types, generalisations of ecosystem response associated with these flow modification types could not be made.

3.4.2 Quantitative analysis of assembled datasets

Periphyton AFDM and chlorophyll–a, benthic macroinvertebrate density and seston AFDM and chlorophyll-a either reduced or showed no change after increased flow magnitude (Figure 3.4A and C). Macroinvertebrate drift and concentrations of *Escherichia coli* either increased or did not change during increased flow magnitude (Figure 3.4B and D). No clear trends in response direction or flow thresholds could be identified for any biotic response.

Stream EC generally reduced during increased flow magnitude and a general negative linear relationship was observed (Figure 3.4E). Conversely, suspended solid concentration (SSC) increased with flow magnitude, but this trend was not significant ($t = -1.50$, $p = 0.16$). No clear trend was observed for turbidity (Figure 3.4F).

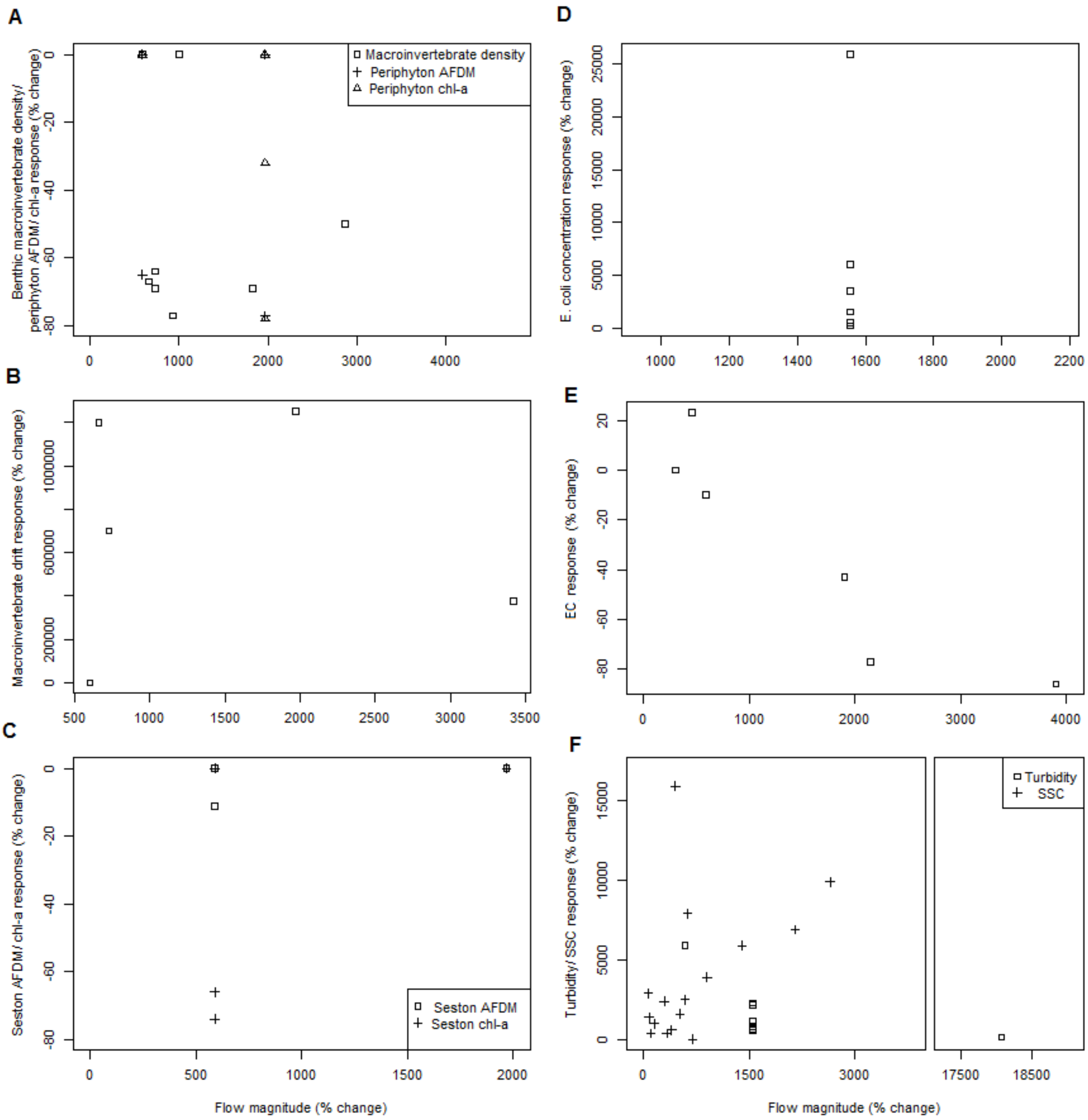

Figure 3.4: Biotic (A-D) and abiotic (E-F) ecological responses to flow magnitude percent change. N.B. Ecological responses are after- and during-flow modification for plots A-B and C-F respectively.

3.4.3 Quality assessment

Seventy-one, 14 and three studies used fully-, semi-quantitative, or qualitative methods respectively to assess ecosystem response to flow modification. Forty-seven studies described replication in sampling, whilst only 19 stated use of randomisation. Fully-quantitative methods of fish and macrophyte assessment were used in fewer than 60% of cases, whereas over 85% of assessments were fully-quantitative for all other ecosystem response types. Qualitative methods were only used for assessment of fish and macrophytes (six and 33% respectively). Whilst over 90% of assessments of physical-chemical response were fully-quantitative, fewer than five percent were stated as either replicated or randomised. Randomised sampling was stated in 50% of primary production assessments, whereas fewer than 25% of sampling for all other ecosystem responses was described as randomised. Over 50% of assessments of fish, macroinvertebrate and primary production response were defined as replicated, compared to fewer than 5% of assessments of physical chemistry (Figure 3.5).

Only 14 studies stated use of control sites, and of these 10 used nearby unregulated streams, five used sites upstream of the reservoir and one used a regulated (with unmodified flow) control (N.B. some studies used more than one control type). Thirty-four studies used descriptive or graphical methods to present results (i.e. no statistical testing) and 10 studies used correlation or regression between a metric of flow and ecosystem response. Twenty-eight studies assessed the impact of flow modification through comparison of ecosystem conditions either through time or between impact/ control sites using 1- or 2-way testing (e.g. Student's t-test; Mann-Whitney U; ANOVA; Kruskall-Wallis). Six studies used alternative methods: least linear squares/ polynomial regression, general linear/ additive/ generalised linear mixed modelling. Only three studies tested site:period interaction terms as part of Before-After-Control-Impact (BACI) (or derivations of) (Smith, 2002) designs and only eight studies used analytical methods that took account of temporal autocorrelation.

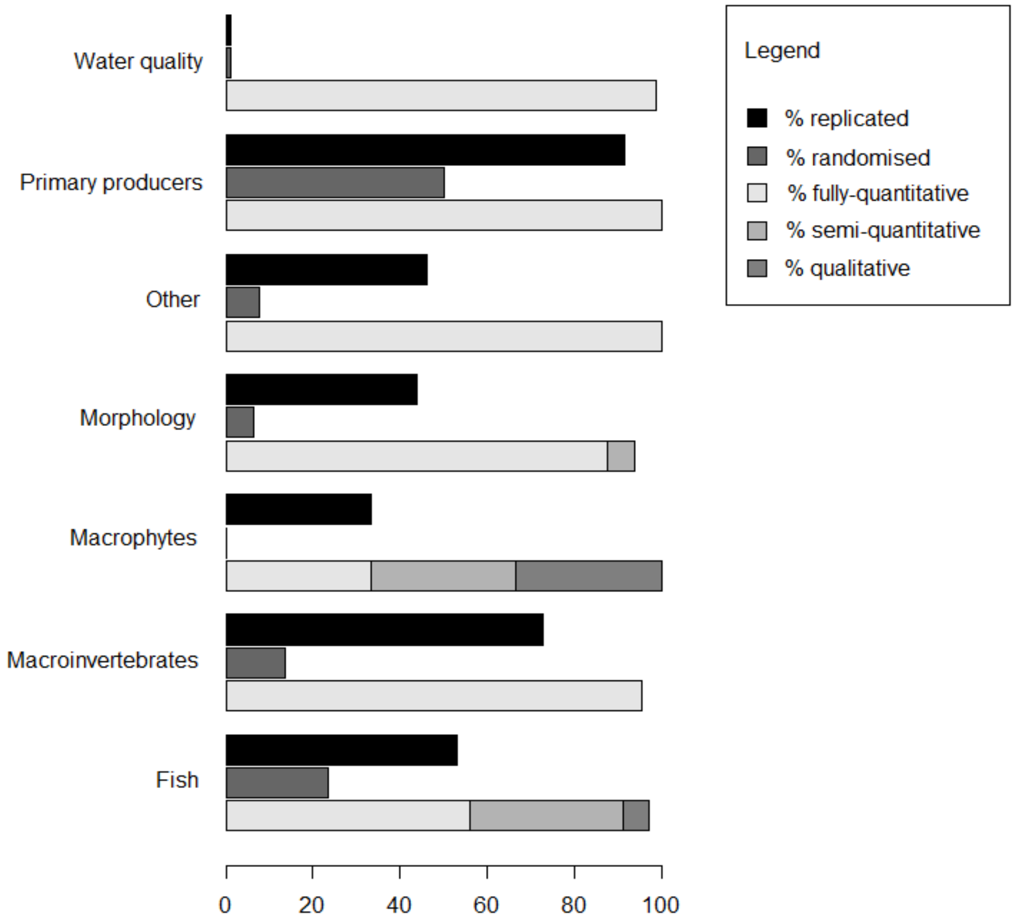

Figure 3.5: Barplot of quality assessment indices for each ecosystem response. Note: percentages were calculated based on the total number of reported ecosystem responses, therefore, the sum of quantitative and qualitative percentages is less than 100 where quality assessment indices could not be extracted from a study.

3.5 Discussion

3.5.1 Spatial distribution of studies

This assessment revealed that most research had been carried out within North America and western Europe and a dearth of studies was observed within equatorial regions, South America, north Africa, Asia and eastern Europe in agreement with a recent assessment by Olden et al. (2014). To assess whether this could be the result of spatial variation in reservoir densities, comparison with the Global Reservoir and Dam (GRanD) database (Lehner et al., 2008) was made. GRanD identifies the four most dense areas of reservoirs in the world as North America, western and eastern Europe and Asia, suggesting that that there is a genuine research bias within the aforementioned locations. Furthermore, central-eastern South America and areas of central and south Africa also have high densities of reservoirs (Lehner et al., 2008) but scant published research from these areas was found. This observed research bias should be taken into account when considering the global applicability and relevance of the findings of this review and it is suggested that the areas where reservoir density is high and published research is currently limited should be prioritised for future work.

3.5.2 Flow and ecosystem response types

The majority of studies reported flow modification as an expression of change in magnitude and only seven studies (< 10%) reported ecosystem responses associated with alternative flow modification types. This bias towards magnitude as the primary modified flow variable is reflected in previous reviews where similar methods of flow classification were used (e.g. Poff & Zimmerman, 2010). Future research should prioritise assessment of alternative flow modification types (e.g. rate of change, timing, frequency). Low flows are critical in determining ecosystem integrity in natural streams (Poff et al., 1997), but analysis of the dataset revealed that only one study (Saltveit et al., 2001) reported on a *reduction* in flow magnitude whilst all other studies concerning changes in magnitude reported on *increased* flow. Assessment of the impact of reduced flow magnitude in regulated streams is therefore a key priority for future research and this need is supported by the identification that typical compensation flows in the UK were set, on average, over 22% higher than pre-impoundment natural low flows and that an ecological benefit could be made with the introduction of lower flows in these systems (Gustard, 1989). This is of particular concern to water utilities in the UK in achieving supply/ demand balance, and may also be a concern elsewhere globally.

It was found that, in general, there was variation in ecosystem response types noted in the literature. However, fish and physical chemistry were most frequently researched and few studies examined impacts of flow modification on macrophytes. Additionally, no studies noted impacts on biotic groups such as aquatic fungi or Archaea, which both play important roles in lotic ecosystem processes (e.g. Manerkar et al., 2008). Furthermore, few studies examined impacts on ecosystem processes such as production and respiration or biotic community interactions (i.e. food webs). The lack of research of the aforementioned demonstrates a bias towards traditionally monitored taxa. The development and application of contemporary techniques in freshwater environments such as quantitative real-time polymerase chain reaction (Q-RT-PCR) based methods for measuring relative DNA contribution of taxa such as Archea, bacteria and fungi within ecosystems (e.g. Manerkar et al., 2008), has opened up new monitoring opportunities. There is a clear research gap for work of this nature and diversification of monitoring strategies to cover less traditionally monitored taxa and undertake more novel assessments of traditionally monitored taxa (which can provide unique insights into their responses) in future studies is recommend. It is also important to note that this research concentrated only on instream ecosystem impacts but, to enhance ecosystem understanding, streams should be viewed as elements within the surrounding environment. Further research should therefore assess the impact of flow modification upon both riparian zones and the wider terrestrial and aquatic environment and the processes and biotic interactions that link these systems.

3.5.3 Qualitative and quantitative flow-ecosystem response relationships

The main objective of this review was to extract, synthesise and evaluate ecosystem responses to reservoir outflow modification. It was expected that this would reveal general flow-ecosystem response relationships for regulated streams and highlight future research priorities, ultimately aiding the advancement of the science of regulated stream management.

Qualitative

The majority of flow magnitude modifications resulted in either increased or decreased ecosystem responses demonstrating that reservoir flow magnitude modification is a potentially useful option for regulated stream conservation or restoration. However, no clear trend in biotic response to all, single and cumulative flow magnitude modifications was identified, suggesting that site specific factors are important. For example, it was found that in response to single increased flow magnitude events, seven studies reported *no change* in, and six studies reported *increased* fish movement. These contradictory observations may be explained by a combination of factors, for example: the characteristics of the flow modification (e.g. the percentage increase, the rate of change etc.); the fish monitored (e.g. species, size, flow preference etc.); and additional abiotic factors such as season, antecedent flow conditions, instream habitat type, time since dam construction etc. To enable a more robust analysis of these relationships, detail on these potentially confounding factors must be considered in each study. Extraction of these data in this review was not possible due to limited availability and thus future publications should consider including detailed information on all potentially relevant factors.

Qualitative analysis revealed some general trends in macroinvertebrate response: increased drift (during flow modification) and reduced benthic densities were results of both single and cumulative increases in flow magnitude. Benthic macroinvertebrate density commonly increases post-impoundment (Petts, 1984a), suggesting that increased flow magnitude events have potential to mitigate for this impact. However, some studies have noted a quick recovery from single flow magnitude modifications (e.g. Jakob et al., 2003) suggesting that one-off flow modification events may not be viable long term mitigation methods. Understanding of long term responses of macroinvertebrates to reservoir flow modification is spatially limited (e.g. Robinson et al., 2004a; Mannes et al., 2008; Robinson & Uehlinger 2008) and is a topic that requires further research globally.

The vast majority of flow magnitude modifications resulted in abiotic changes, specifically, increased turbidity, suspended solid concentration and bedload transport. This suggests that flow magnitude modification has potential for use in mitigation of the effects of impoundment such as reduced sediment transport, which is commonly observed post-impoundment as sediment is trapped in the reservoir and resultant bed armouring occurs (Petts, 1984a). No studies were found that highlighted the long term impact of flow magnitude modification on sediment transport as all sampling was undertaken *during* each event. It is therefore recommend that future research aims to assess how stream sediment transport responds both during and after single and cumulative flow magnitude modifications.

Qualitative analysis of physical-chemical factors revealed that increased flow magnitude commonly resulted in reduced EC. Heterogeneity in concentrations of dissolved ions are typical in natural lotic systems (e.g. Glover & Johnson, 1974), thus, increased flow magnitude events have the potential to mitigate the reduction in temporal variability in EC observed post-impoundment (e.g. Palmer & O'Keeffe, 1990) where such an impact is deemed negative. It was also found that water temperature was commonly observed to decrease or not change as a result of increased flow magnitude. This is most likely due to the climatic and reservoir characteristics at each site and the vertical position of the draw-off valve used during flow modification. One study (Macdonald et al., 2012) found that draw-off level from the reservoir was a significant factor in determining downstream temperature. The potential for temperature modification through reservoir flow operation is evident and may be important given the crucial influence of temperature on biota in lotic ecosystems (Cummins, 1974) and the significant impact of reservoirs on downstream temperature regimes (Petts, 1984a; Dickson et al, 2012). Further research should be directed towards assessment of the relative importance of different flow modification types in controlling downstream temperature, especially the impact of reservoir draw-off level which, to date, has received little attention.

Quantitative

It was possible to extract 199 data points for quantitative analysis, but only 10 ecosystem response (seven biotic and three abiotic) types were reported on more than five times and were subsequently plotted. No clear trends were observed between flow magnitude modification and biotic responses, most likely reflecting the lack of data points and the importance of site specific factors. Approximately linear relationships were found between flow magnitude modification percent change and EC (negative relationship) and suspended solid concentration (SSC) (positive relationship) percent change. Statistical analysis was only viable for SSC and revealed that the relationship was not significant. No threshold flow changes (where abrupt changes in ecological response could be identified) were observed for these parameters. With the addition of further data points, a future review may be able to model ecological response and take into account confounding factors such as geology, local climate, antecedent flow conditions etc. allowing for robust statistical testing of relationships which is not currently possible.

This assessment used a similar method to Poff & Zimmerman (2010). These authors concluded that their focus on all stream types and all types of modification (e.g. dam construction, irrigation and urbanisation leading to increased run-off) may have limited their ability to find general flow-ecosystem response relationships. This review differed in that it focussed specifically on reservoir outflow modification post-impoundment in an attempt to reduce the impact of this limitation. Similarly to Poff & Zimmerman (2010), analysis is this review was restricted by both the small number of data points and the limited availability of information relating to potential confounders. As development of flow-ecosystem response relationships in reservoir regulated streams increases, it is suggested that future research would benefit by analysing these relationships collectively between areas of similar climatological and geological characteristics, as these factors are expected to influence ecosystem response to flow modification (Arthington et al., 2006; Poff et al., 2010). This would further the development of smaller scale, regional, or environment 'type' based relationships which are required for environmental flow setting frameworks such as ELOHA (Ecological Limits of Hydrological Alteration) (Poff et al., 2010) or the Building Block Methodology (BBM) (King & Louw, 1998).

3.5.4 Quality assessment

The majority of studies (> 90%) used fully-quantitative methods to assess at least one ecosystem response to flow modification, although method types used to assess each ecosystem response type varied. For example, fewer than 60% of methods were fully-quantitative for assessment of fish and macrophytes. A propensity for semi-quantitative electric fishing techniques (32% of all fish response assessments) and the limited number of assessments of macrophytes (n = 3) explain this observation. Research has suggested that semi-quantitative methods of fish sampling can be up to 95% accurate (Klein-Breteler et al., 1990), thus the high proportion of semi-quantitative methods for assessment of fish response is unlikely to be of concern.

Johnson (2002) describes replication and randomisation as two cornerstones of experimentation and states that they are integral to successful ecological research, yet ecologists often commit replication errors (Hurlbert, 1984) and rarely select study areas or sampling locations randomly (Johnson, 2002). This review identified similar trends as 47 studies (62%) stated that replication was used in sampling, whilst only 19 (25%) stated randomisation was applied. Further analysis revealed the distribution of use of these techniques was unequal among ecosystem response types. In particular, fewer than five percent of assessments of physical-chemical responses were stated as either replicated or randomised, whereas all other ecosystem elements were stated as being assessed using either replicated or randomised methods in at least 30% of cases. No reasons could be extracted from the studies to explain why replication or randomisation had not been carried out for physical-chemical assessment. However, the approaches used may reflect the consensus among literature outlining sampling protocols where replication (Hauer & Hill, 1996; USEPA, 2004) and randomisation (Hauer & Hill, 1996) are not highlighted as important. This assessment has identified a lack of use of these cornerstones and it is suggested that future research incorporates both facets. Furthermore, Johnson (2002) suggests that at a more holistic scale, the replication of studies (metareplication) is more important than carrying out individual studies and has the power to yield greater confidence that any identified relationships are general, and not specific to the prevailing conditions within a particular study. It is therefore recommend that if possible, research into the impact of flow manipulation on downstream ecology is replicated at different times, at different sites, and by different researchers (see Johnson (2002) for discussion of metareplication).

The majority of studies used one-way comparisons of sample periods (e.g. before/ after flow modification) or between control/ impact sites over sample periods. One of the limitations of these approaches is that they fail to take account of temporal autocorrelation (only eight studies (11%) took temporal autocorrelation into account) and can result in less robust analysis (Zuur et al., 2009). BACI designs have been suggested as useful methodological frameworks for use in impact assessment of anthropologically driven disturbance events (Underwood, 1991) such as flow modifications from reservoirs. BACI designed experiments allow for significance testing of site:period interaction terms (see Underwood, 1991) which takes variation that is assumed to have occurred if the impact (e.g. flow modification) had not been undertaken into account. Nevertheless, only three studies used this approach and it is suggested that future researchers consider this approach when planning assessment of reservoir flow modification. Selection of a control site is necessary when applying BACI approaches but this assessment revealed that only 14 studies (< 20%) used control sites. Within these approaches, considerable variability in the type of control was identified. Currently, research is lacking as to which type provides the most robust method, but given that ideal control sites should be both independent of and as similar as possible in abiotic and biotic characteristics to an impacted site (McMahon, 2000), it is probable that an independent, regulated control site has the potential to act as the most effective control. Further research is required to test this hypothesis.

3.6 Summary

This chapter synthesised the global literature concerning reservoir flow modification and associated downstream ecosystem response. Biases within both the location of studies and research topics were identified and flow-ecosystem response relationships were quantified. For example, as a result of increased flow magnitude, macroinvertebrate density and drift was commonly identified to decrease and increase respectively and periphyton mass was commonly observed to decrease. Further, during increased flow magnitude, reduced electrical conductivity and increased suspended solid concentration, turbidity and bedload transport was commonly observed. However, clear relationships between the majority of ecosystem response types and reservoir flow manipulation were unclear, potentially due to the importance of site specifc factors. Therefore, detailed studies of the impact of reservoir flow modification on downstream ecosystems are now required. The strategic review also identified methodological and analytical improvements that could be made in future assessments of the impact of environmental flows (e.g. using control sites and performing assessments of interaction terms under a BACI framework) and their implementation is recommended for future research. Overall, the findings of this literature review should redirect and focus regulated stream research in a concerted manner.

4 STUDY AREA

4.1 Chapter overview

This chapter firstly details the method used to select a study area. It then describes the area as a whole and each site in terms of surface geology, land cover and aspects such as catchment area and altitude. Some background physical-chemical information is also provided for sites where biological monitoring was undertaken. Finally, reservoirs within the study area, including their operation, are described.

4.2 Study area selection method

The majority of chapters within this thesis use the study area described in the following text to meet their aims, but Chapter 7 takes a wider (regional) approach and the sites used are described in that chapter rather than here.

This study had two broad aims: (i) to assess the impact of regulation and (ii) to assess the impact of Artificial Floods on downstream ecosystem characteristics in an upland area. A focus on an upland area was chosen as approximately 80% of large UK dams are situated in upland areas (Petts, 1988). Aim (i) could potentially be met through comparison of regulated and unregulated streams and aim (ii) could potentially be met through comparison of *impact* and *control* streams (e.g. Chester & Norris, 2006; Brooks et al., 2011). The optimum study area would therefore include all of these stream types. In addition, at least one of the reservoirs was required to be able to make releases of water and, to reduce the potential impact of confounding factors, all reservoirs and streams needed to be as similar as possible in aspects independent of regulation. Furthermore, safe access and permission were required to both visit and undertake all proposed monitoring.

Initial map based searches (sources: OS, 2012; USGS, 1998) combined with discussion with reservoir operators and landowners (YW) identified the chosen study area as potentially meeting all necessary criteria. Desk-based assessment of hydrological, climatological, physical, chemical and biological factors (sources: BGS, 2012; CORINE, 2010; EA, 2009; EA, 2013) and a subsequent site visit in August 2011 confirmed the suitability of the study area. Permission was then obtained from landowners and heath and safety methodological statements were prepared and authorised prior to commencement of monitoring.

4.3 Study area characteristics

For reasons of commercial sensitivity, the exact location of study area and sites have been anonymised throughout this thesis. The study area was located within the south Pennies of central-northern England, UK (Figure 4.1). Surface geology was 59% mud-/silt-/sand-stone and ~39% peat with minor areas of diamicton (Table 4.1). The majority of the study area was classed as wetland (e.g. peat bog), but large areas of forest/ semi-natural (~36%) and agricultural (~22%) land were also present. None of the study area land cover was classed as artificial (Table 4.1). Average annual rainfall for the south Pennies is approximately 1,200mm (Evans et al., 2006).

4.4 Site-scale characteristics

4.4.1 Locations

Eighteen sites were monitored during the duration of this project (Figure 4.2). Sites 1, 3, 5 and 8-10 were located on unregulated stream segments. Sites 6, 11 and 14 were located within 100m downstream of reservoirs and sites 2, 4, 7, 12, 13, 15, 16, 17 and 18 were all influenced by upstream regulated streamflow. Sites 2, 3 and 4 were located between 100 and 199m aod and sites 6, 8, 9 and 10 were located above 300m aod. All other sites were located between 200 and 299m aod. Most sites were between 4 and 7km from source (based on OS, 2012), but sites 1, 5-10 and 14 were located within 4km, and sites 2 and 4 were more than 10km from source (Table 4.1).

4.4.2 Geology

Surface geology of the drainage basins of most sites was dominated by peat. Conversely, the drainage basins of sites 4 and 7 were dominated by mud-/silt-/sand-stone, but both had >44% peat cover (Table 4.1). Sites 8, 9 and 10 drainage basin surface geology was exclusively peat and site 5's drainage basin had a high (>30%) proportion of diamicton (BGS, 2012).

4.4.3 Land cover

Most of the drainage basins within the study area were dominated by wetland or forest/ semi natural land cover. These classifications include areas of peat bog and shrub/ herbacious vegetation. Site 3 was the only site with a drainage basin dominated by agricultural land. Drainage basins of sites 1-4, 11-13 and 15-18 also had areas of agricultural land (Table 4.1).

Percentage waterbody land cover for each drainage basin are reflective of reservoir distribution within the study area (CORINE, 2010).

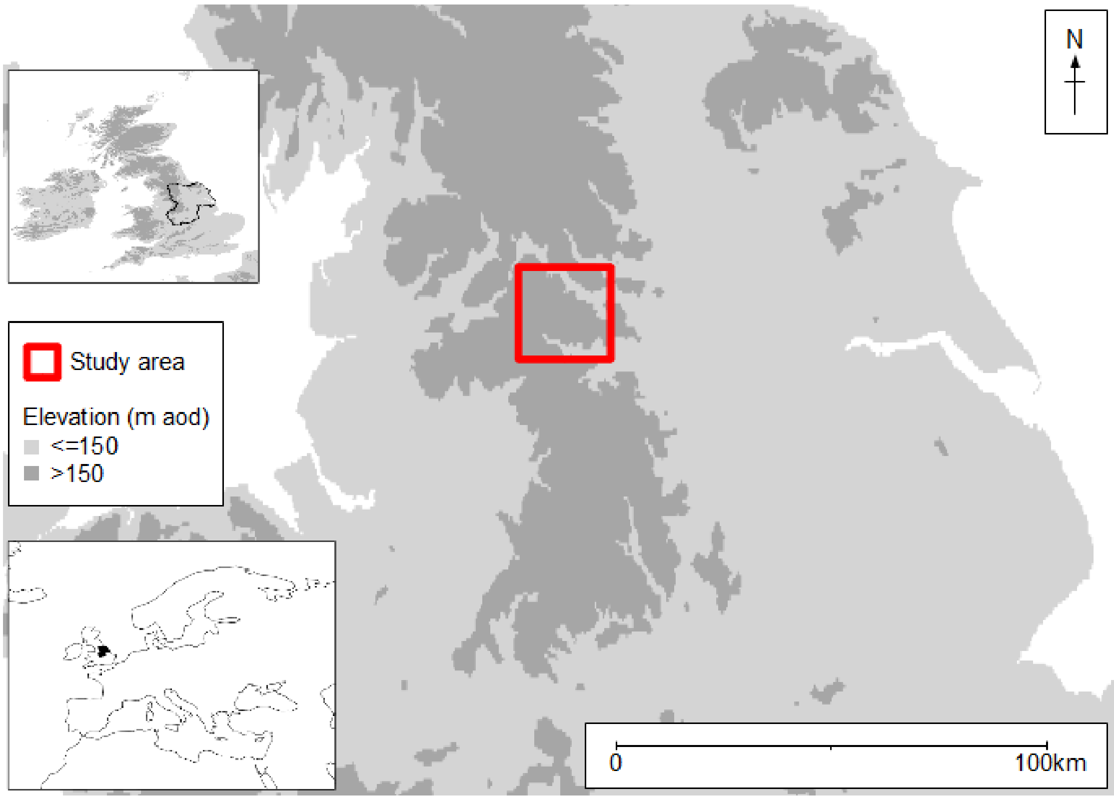

Figure 4.1: Study area location within Europe. Sources: USGS, 1998; NE, 2012.

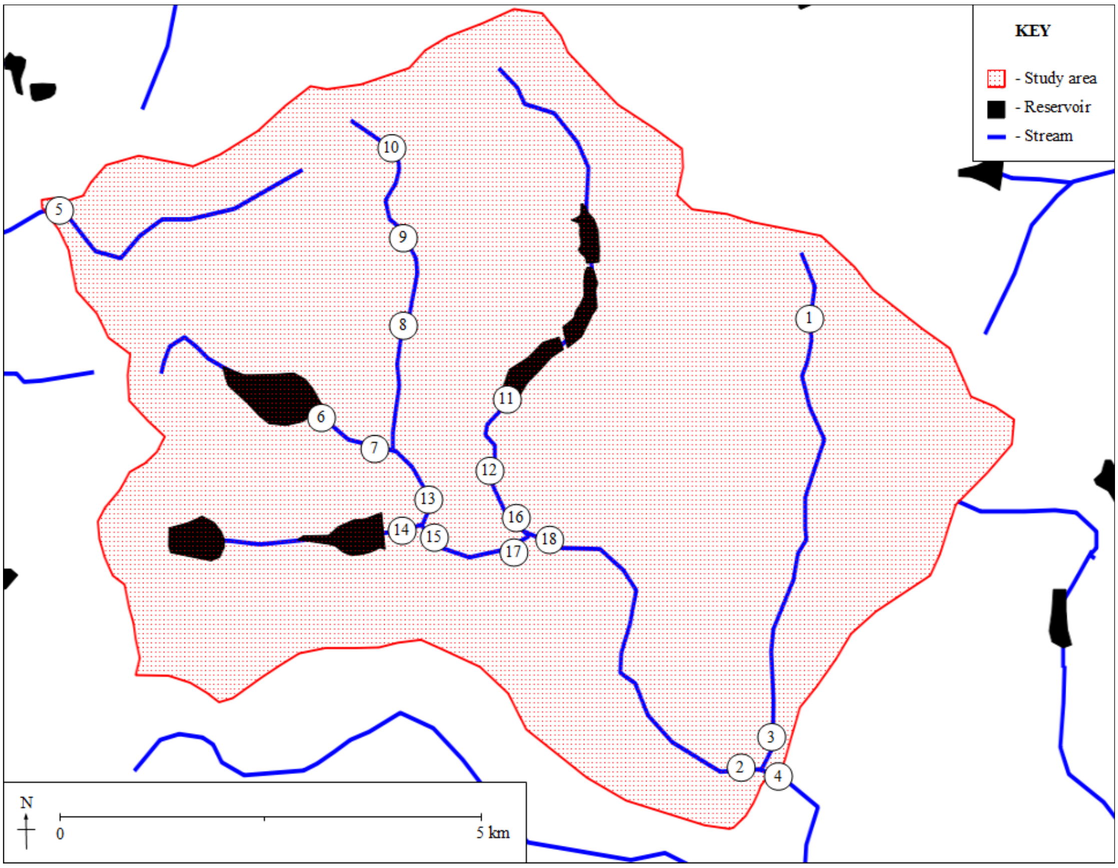

Figure 4.2: Site locations within the study area (derived from OS, 2012).

Table 4.1: Percentage surface geology and land cover composition of site and study area drainage basins (sources: BGS, 2012 & CORINE, 2010 respectively); altitude, catchment size and longest (instream) distance to source of each site and study area (source: OS, 2012).

	Surface geology			**Land cover**				**Site specific factors**		
Site	**% Mud-/silt-/sand-stone**	**% Peat**	**% Diamicton**	**% Agricultural**	**% Forests and semi-natural**	**% Wetlands**	**% Waterbodies**	**Altitude (m aod)**	**Catchment size (km^2)**	**Longest distance to source (km)**
1	43.90	56.10	0.00	17.80	1.40	80.80	0.00	294	3.03	2.6
2	42.90	57.10	0.00	16.10	41.20	40.70	2.00	156	40.66	10.9
3	31.70	68.30	0.00	48.90	13.30	37.80	0.00	160	11.89	6.0
4	58.60	41.30	0.20	23.70	35.00	39.70	1.60	137	52.10	11.6
5	0.70	65.10	34.10	0.00	55.00	45.00	0.00	283	3.62	3.4
6	48.00	52.00	0.00	0.00	62.30	25.90	11.80	307	3.09	3.0
7	56.00	44.00	0.00	0.00	66.10	23.80	10.10	292	3.63	3.7
8	0.00	100.00	0.00	0.00	13.20	86.80	0.00	346	3.10	2.4
9	0.00	100.00	0.00	0.00	14.40	85.60	0.00	368	2.04	1.9
10	0.00	100.00	0.00	0.00	25.60	74.40	0.00	401	1.13	1.3
11	20.20	79.80	0.00	1.80	26.00	67.30	4.90	289	9.05	4.7
12	27.80	72.20	0.00	8.20	27.90	59.50	4.30	264	10.32	5.6
13	34.90	65.10	0.00	4.40	40.50	51.70	3.50	271	10.50	5.0
14	8.50	91.40	0.10	0.00	47.50	52.50	0.00	274	7.14	3.8
15	23.70	76.30	0.00	2.20	43.10	52.70	2.00	266	18.48	5.4
16	31.20	68.80	0.00	11.60	28.30	56.10	4.00	253	11.09	6.2
17	31.40	68.60	0.00	4.60	45.40	48.10	1.80	245	20.18	6.4
18	31.70	68.30	0.00	7.00	40.10	50.20	2.60	236	30.73	6.7
Study area	59.00	38.60	2.40	22.20	36.30	40.10	1.50	n/a	55.72	n/a

4.4.4 Background physical chemistry

Monthly grab water samples were taken throughout the study period (Jan 2012 to August 2013) from sites where biological sampling was undertaken (7, 9 and 12) to provide an appreciation of background levels of dissolved nutrients and selected metal ions. Samples were collected using 125ml aseptic polypropylene containers, immediately refrigerated, filtered (0.45µm) within 48hrs of collection and refrigerated prior to analysis by ICP (metals) and autoanalyser (nutrients). The ICP (Perkin Elmer 5300DV ICP-OES) was calibrated with commercially prepared 1000ppm standards and checked by analysing Certified Reference Material (ERM CA011a) from the Laboratory of the Government Chemist prior to analysis and used a sample and nebuliser flow rate of 1.5 and 750ml/ minute respectively at 1400W. Autoanalyser (Skalar SAN++ continuous flow) analysis was undertaken according to Kamphake et al. (1967); USEPA (1974); Krom (1980); Searle (1984); Kempers & Luft (1988) and APHA (1989) and calibrated with commercially prepared standards. Mean, standard deviation, minimum and maximum concentrations are shown in Table 4.2 and demonstrate that there were no exceptionally extreme concentrations at any site throughout the study period.

Table 4.2: Dissolved nutrient and metal concentrations (µg/ l) calculated from monthly samples at sites where biological sampling occurred.

		Nutrients					**Metals**								
Site		**NH4**	**Σ(NO2 + NO3)**	**NO3**	**NO2**	**PO4**	**Al**	**Ca**	**Fe**	**K**	**Mg**	**Mn**	**Na**	**Si**	**Zn**
7	**Mean**	0.03	0.34	0.33	0.01	0.00	0.17	1.90	0.67	0.35	1.14	0.04	4.93	2.06	0.01
	St.dev.	0.02	0.10	0.10	0.00	0.00	0.06	0.41	0.29	0.07	0.24	0.01	0.96	0.36	0.00
	Max	0.06	0.54	0.53	0.01	0.01	0.26	2.84	1.19	0.50	1.63	0.05	6.21	2.74	0.01
	Min	0.00	0.18	0.17	0.00	0.00	0.07	1.30	0.29	0.25	0.81	0.02	3.35	1.50	0.00
9	**Mean**	0.04	0.30	0.29	0.01	0.01	0.17	1.20	0.80	0.28	0.86	0.06	4.15	2.24	0.01
	St.dev.	0.05	0.15	0.15	0.00	0.01	0.09	0.47	0.54	0.11	0.33	0.02	1.35	1.42	0.00
	Max	0.15	0.57	0.57	0.01	0.02	0.33	1.98	1.84	0.43	1.36	0.10	5.59	4.58	0.02
	Min	0.00	0.03	0.01	0.00	0.00	0.05	0.56	0.20	0.08	0.39	0.03	1.90	0.57	0.00
12	**Mean**	0.05	0.40	0.40	0.01	0.00	0.13	3.36	0.55	0.37	1.53	0.05	4.38	2.30	0.00
	St.dev.	0.09	0.11	0.11	0.00	0.00	0.07	0.88	0.22	0.10	0.42	0.03	0.83	0.54	0.00
	Max	0.39	0.61	0.61	0.01	0.01	0.26	4.59	0.93	0.57	2.11	0.17	5.45	3.05	0.01
	Min	0.00	0.22	0.22	0.00	0.00	0.03	2.02	0.26	0.24	0.93	0.01	2.78	1.57	0.00

4.5 Reservoir characteristics

Six reservoirs were within the study area (Figure 4.3 A-F). Reservoir C was constructed in the late 19th century and construction of D, E and F was finalised in 1907, followed by completion of reservoirs A and B in 1934 (Tedd & Hoton, 1994). The dam of reservoir B was the tallest at 30m, and reservoir C had the smallest dam height of 22m. Reservoir C had the largest maximum capacity (c. 2.9 million m^3) and the smallest was at reservoir D (c. 0.7 million m^3) (Tedd & Hoton, 1994). Reservoirs C and D had seasonally altering minimum compensation flow agreements set in place, whereas reservoir B had a constant annual minimum compensation flow (Table 4.3). The compensation flow from reservoir B was typically drawn from the bottom of the water column cf. middle of the water column from reservoir D throughout the duration of the study. However, compensation flow from reservoir C was typically drawn from the bottom of the water column, apart from between 02.05.2013 to 02.07.2013 where water was drawn from the middle of the water column for operational reasons (S. Firth (YW) pers. comm. 3rd March 2013 and 8th October 2013). Each reservoir was primarily used for water supply and no recreational or hydropower activities were undertaken.

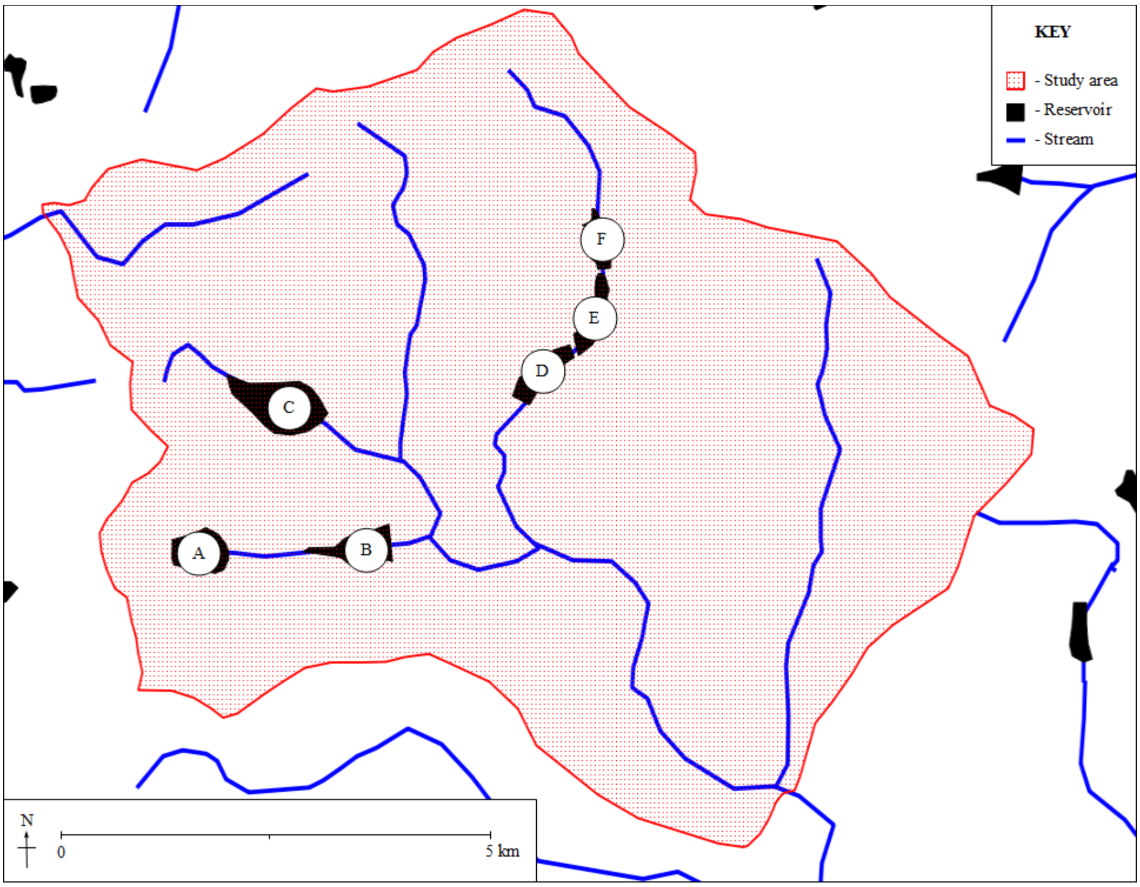

Figure 4.3: Reservoir locations within study area (derived from OS, 2012).

Table 4.3: Characteristics of reservoirs B-D. [1] – January 1st to October 31st. [2] – November 1st to December 31st. Sources: Tedd & Hoton, 1994; S. Firth (YW) pers. comm. 3rd March 2013.

Reservoir	Dam construction date	Dam height (m)	Maximum capacity (m^3)	Minimum compensation flow (m^3/s)
A	1934	26	1,731,000	n/a
B	1934	30	1,261,000	0.03
C	1878	22	2,912,000	0.04 [1], 0.15 [2]
D	1907	24	727,000	0.04 [1], 0.17 [2]
E	1907	24	1,111,000	n/a
F	1907	24	932,000	n/a

5 IMPACT OF REGULATION AND ARTIFICIAL FLOODS ON STREAM DISCHARGE AND SELECTED PHYSICAL-CHEMICAL PARAMETERS

5.1 Chapter overview

This chapter presents an assessment of the impact of stream regulation and Artificial Floods on downstream hydrology and selected physical-chemical parameters. First, the importance of understanding these issues is presented followed by an identification of current gaps in research and identification of aims of the study. Next, the methods and analytical techniques used to undertake the assessment are detailed. This is followed by sections presenting and discussing the results, including recommendations for further research.

5.2 Introduction

The flow regime of a stream is fundamental in determining its ecological characteristics (Allan & Castillo, 2007). Over a wide range of temporal scales, ecosystems are shaped both directly and indirectly by stream flow (Poff et al., 1997) which has been described as a "master variable" (Power et al., 1995) that limits both the abundance and distribution of lotic species (Allan & Castillo, 2007). Electrical conductivity (EC), dissolved oxygen (DO) and pH are important stream physical-chemical parameters. Dissolved ions have a direct influence on stream ecology (e.g. aluminium can be toxic to fish (Driscoll et al., 1980)). EC reflects general ionic content and measures the capacity of ions to carry electrical current in water (Hach Company, 1997) demonstrating the potential importance of recording EC in stream systems. DO is a key driver of respiration in aquatic ecosystems and is essential for aquatic life (Hach Company, 1997). Extreme levels of DO are known to have deleterious effects on aquatic ecology (Edwards, 1964; Franklin, 2013), justifying its measurement in aquatic ecosystems. pH is a measure of hydrogen ion activity in a solution (Hach Company, 1997) and is commonly measured in aquatic ecosystems as it has an indirect influence on aquatic ecology by affecting the availability of ions for uptake by aquatic organisms. For example, aluminium becomes available for fish uptake in water with low pH (Driscoll et al., 1980), thus pH is a crucial parameter for measurement in aquatic systems.

Anthropogenic activity can have a profound effect on hydrology, EC, DO and pH in stream ecosystems (hydrology: see Poff & Zimmerman (2010); EC and DO: e.g. sewerage effluent:

Daniel et al. (2002); Gammons et al. (2011); pH: e.g. deforestation: Ågren et al. (2010) and acid mine drainage: Equeenuddin et al. (2013)) and it is therefore important to understand the impact of such activity to enable successful management to reduce its impact. Seventy-seven percent of total stream discharge in the northern hemisphere is either strongly or moderately affected by fragmentation of stream channels resulting from reservoir operation, inter basin diversion and irrigation (Dynesius and Nilsson, 1994). An understanding of any impact that these activities have on downstream hydrology, EC, DO and pH is therefore vital.

The impact of regulation on discharge is generally well understood. Typically, minimum flow magnitude is increased (Higgs & Petts, 1988; Gustard, 1989), median flow magnitude is reduced (Higgs & Petts, 1988) and high flows are reduced in magnitude and frequency (Higgs & Petts, 1988; Gustard, 1989) resulting in an overall reduction in flow variation (Baxter, 1977). However, the majority of assessments of the impact of regulation on discharge have been conducted using daily values (e.g. daily mean (Gustard, 1989)), resulting in a lack of understanding of event based impacts (e.g. during floods). Additionally, impacts have typically been reported at long term (e.g. annual (Higgs & Petts, 1988)) time scales, resulting in a lack of understanding of seasonal variation of impacts. The impact of regulation on EC, DO and pH is less well understood. A reduction in annual and seasonal range (Soja & Wiejaczka (2013) and Palmer & O'Keeffe (1990) respectively) and reduction in monthly mean EC (Soja & Wiejaczka, 2013) has been identified. DO has been studied most comprehensively, but both a reduction (Walker, 1985; Palmer & O'Keeffe, 1990; Bunn & Arthington, 2002) and no impact (Crisp, 1977) due to regulation have been observed. Similarly, variation in the impact of regulation on pH has been identified: Soja & Wiejaczka (2013) identified no impact and Palmer & O'Keeffe (1990) identified no impact downstream of headwater reservoirs but observed an increase in pH downstream of a mid-catchment reservoir. These inferences have typically been based on sampling strategies of short duration (i.e. sub-seasonal) (e.g. Crisp, 1977) or low frequency (i.e. daily measurements) (e.g. Soja & Wiejaczka, 2013) resulting in a lack of understanding of physical-chemical impacts over a variety of time scales (e.g. diurnal, weekly and seasonal) and during particular events (e.g. floods).

Contemporary environmental legislation such as the EU Water Framework Directive (EU WFD) (EC, 2000) requires that regulated streams meet an ecological standard (Good Ecological Potential (GEP)) based on abiotic and biotic aspects of stream ecology. Artificial Floods (AFs) have been suggested as a potential tool to enable the achievement of GEP through mitigation of impacts associated with regulation (Acreman & Ferguson, 2010). Thus, it is important that the relationships between AFs and stream EC, DO and pH are understood to allow for an appraisal of whether AFs can be used successfully as mitigation.

Globally, 74 publications have reported ecosystem responses to reservoir outflow modification such as AFs; of these, physical-chemical responses have regularly been assessed (Gillespie et al., in press (b)). EC has been reported on in seven publications with increases (Foulger & Petts, 1984a; Shannon et al., 2001; Bruno et al., 2010), decreases (Petts et al., 1985; Jakob et al., 2003; Cánovas et al., 2012) and no change (Cambray et al., 1997) reported as a result of AFs resulting in a lack of consensus. DO response to AFs has also varied between publications (increased: Shannon et al., 2001; Bednarek & Hart, 2005, Naliato et al., 2009 and decreased: Chung et al., 2008; Naliato et al., 2009) and pH response to AFs has not been published resulting in a requirement for research. In addition to a global perspective of physical-chemical response to AFs, a regional, or stream 'type' (i.e. of similar geological, climatological and land cover characteristics) based understanding of hydrology-ecosystem response has been recommended (Poff & Zimmerman, 2010; Gillespie et al., in press (b)). It is therefore crucial that the establishment of understanding in responses of EC, DO and pH to AFs occurs at the site scale to enable larger regional or 'type' based inferences to be made. The development of general relationships between hydrological parameters (e.g. magnitude) and ecosystem response variables has been used to identify quantitative relationships and potential threshold levels of hydrological parameters that invoke an ecosystem response (e.g. Poff & Zimmerman, 2010; Gillespie et al., in press (b)). The assessment of these relationships during AFs with respect to any impacts of regulation *per se* has the potential to reveal whether AFs could be used to mitigate against these impacts. To date, this has not been conducted for any regulated stream globally.

This chapter reports on a detailed study of hydrology, EC, DO and pH in a catchment in upland UK over a two year period. An assessment of the impact of regulation on each of these parameters is made through comparison with unregulated conditions and a series of AFs were also conducted. Given the gaps in understanding identified above, the aims of the study were to: (i) identify any impacts of regulation on hydrology at seasonal and event based (i.e. flood) scales; (ii) identify any impacts of regulation on EC, DO and pH at diurnal, weekly, seasonal and event based (i.e. flood) scales; (iii) identify any impacts of AFs on EC, DO and pH and (iv) assess the potential for the use of AFs as a mitigation technique. It was hypothesised that (H1) regulation would reduce flood frequency, magnitude and duration and the impact would vary temporally, (H2) regulation would impact downstream physical chemistry and these impacts would vary temporally, and (H3) AFs would impact downstream physical chemistry thereby demonstrating control of downstream physical chemistry and potential for use as mitigation.

5.3 Methods

5.3.1 Discharge measurement

Discharge was measured at six sites (regulated sites: 6 and 11, unregulated: 1, 5, 8 and 10) using three methods: (i) discharge data (recording frequency: 15 minutes) were provided by the reservoir operator (YW) for site 6 (YW recorded discharge calibrated water level data at a weir approximately 20m upstream of site 6, directly downstream of reservoir C). (ii) SEBA Hydrometrie MDS Dipper-3(T3) vented water level data loggers (accuracy ±1mm) were installed at sites 1, 5, 8, 10 and 11. Data loggers at sites 1, 5, 8 and 10 were mounted in perforated high-density polyethylene piping (diameter: 3cm) and secured in stream locations where flow was typically well mixed and non-turbulent (Figure 5.1). (iii) A further SEBA Hydrometrie MDS Dipper-3(T3) vented water level data logger was installed in a stilling basin that had previously been used by YW to record discharge via a float operated 'Palatine' recorder. The basin was fed via a pipe located ~5m upstream of a weir located ~20m upstream of site 11. Data loggers were calibrated prior to installation and set to record at 15 minute intervals. Once deployed, loggers were inspected for problems and data downloaded approximately monthly. Logger internal clocks were reset at each download interval to ensure between-logger temporal consistency.

To calculate discharge at sites 1, 5, 8 and 10, water level-discharge rating curves were constructed. The 'dry-salt' (Hudson & Fraser, 2005) method was used to gauge discharge as the velocity-area method (Gore, 1996) was unsuitable due to high bed roughness and the 'slug-injection' method (Moore, 2005) required more equipment to be transported to site (which was typically over rough terrain). Discharge measurements were made at each site at a range of flows. Time series discharge estimates were then calculated from the water level time series using coefficients obtained by regression analysis of water level-discharge rating curves. Water level-discharge rating curves and water levels recorded during the period of study for sites 1, 5, 8 and 10 are detailed in Appendix B. Coefficients were obtained from a water level-discharge rating curve constructed from a rating table for water level and discharge at site 11 (source: YW, n.d.). These coefficients were then used to calculate discharge from logged water level. The water level-discharge rating curve and regression coefficients calculated for site 11 are detailed in Appendix B.


Figure 5.1: Water level data logger secured to bank at site 8.

5.3.2 Precipitation monitoring

A tipping bucket rain gauge (Global Water RG200 20cm) was positioned approximately 50m north of site 11 on relatively flat, exposed ground, free from overhead obstructions. During installation, the gauge was bolted to a secure platform and the top of the gauge made horizontal with a spirit level. The gauge was fitted with a Hobo event datalogger which logged each tipping event. Prior to installation, laboratory calibration revealed that each tip was triggered at 4.9ml. Data were downloaded at approximately monthly intervals, converted to ml/cm^2 and internal logger clock reset to reduce drift.

5.3.3 Sondes

EC (specific conductivity at 25°C), pH and DO (% saturation) were recorded at 15 minute intervals using a 6560 conductivity probe (accuracy: ±0.5% of reading + 1µS/cm (YSI, 2005)), a 6-series pH probe (accuracy: ±0.2 units (YSI, 2011)) and a 6150 ROX ® Optical Dissolved Oxygen 6-series probe (accuracy: ±1% of reading or 1% air saturation (whichever is greatest) (YSI, 2008)) respectively. Probes were installed on a YSI 6600 V2-2 extended deployment sonde fitted with a protective shield and automated wiping brush to reduce fouling at sites 6, 10 and 11. Each sonde was located in a well-mixed location of the stream to enable comparison between sites. Data were transmitted using GRPS systems to a dedicated website maintained by Meteor Communications (Europe) Limited.

Two sondes were rotated at each site at approximately monthly intervals. Due to the extended deployment of each sonde, quality control of data was important to ensure comparability between sites. Prior to each deployment, each probe was calibrated by the EA against known standards. Once deployed, data were checked to ensure sound functioning. In the event that a probe had failed, a replacement sonde was deployed and installed by the EA typically within 1-week. To estimate instrument drift during deployment, prior to removal of each sonde, measurement of each parameter was made using a Hach HQ40d portable multi-parameter meter. Each probe installed on the HQ40d was calibrated prior to use against known standards using manufacturer protocols (Hach company, 2013a, b & c (EC, DO and pH respectively)). Once complete, sondes were swapped and consistency of position ensured.

5.3.4 Sites

To assess whether differences between unregulated and regulated discharge regimes could be identified, data from unregulated sites 1, 5 and 8 and regulated sites 6 and 11 were selected for analysis as selection of these sites maximised the number of datapoints available. To assess whether differences in unregulated and regulated physical-chemical regimes could be identified, sonde datasets from sites 10 (unregulated), 6 and 11 (regulated) were analysed.

5.3.5 Time scales

Hydrology

Discharge data were selected for analysis for the period 30th July 2012 to 19th August 2013 inclusive. These data were prior to the introduction of planned AFs and were therefore representative of unaltered, regulated conditions (for sites 6 and 11). Due to datalogger malfunction, precipitation data were only available for the period 1st December 2012 to 3rd July 2013. However, precipitation events during this period were frequent and of varying magnitude and duration, thus data were deemed suitable for testing for differences between regulated and unregulated discharge response to precipitation.

Physical chemistry

To assess the impact of regulation on downstream physical chemistry, EC and pH data were selected for analysis for the period 5th April 2012 to 19th August 2013 inclusive and DO data were selected for the period 29th November 2012 to 19th August 2013 inclusive (DO data prior to the 29th November 2012 had been recorded using an alternative probe type to that described above and instrument drift was high and data quality was deemed too poor for use). These data were prior to the introduction of planned AFs and are therefore representative of unaltered, regulated conditions (for sites 6 and 11). These sonde data were combined with discharge data from 30th July 2012 to 19th August 2013 inclusive for assessment of physical-chemical dynamics during floods.

Artificial Floods

AFs were carried out by YW on the 20th August, 19th September , 3rd October and 5th November 2013 (AFs 1-4 respectively) from reservoir C. The characteristics of each AF represented a balance between the practical restrictions placed on YW by the EA, local stakeholders, water resource availability and the capability of reservoir infrastructure to implement a flood. For these reasons, each AF differed in characteristics. However, during all AFs, water was drawn from the bottom of the reservoir water column.

To assess the impact of AFs on stream hydrology and physical chemistry, discharge and sonde data were collected at 15 minute intervals on each AF day from sites 6 and 11. Site 6 was chosen to represent the impacted hydrological and physical-chemical regime as it was directly downstream of reservoir C. Site 11 was designated a control site.

5.4 Data analysis

5.4.1 Quality control procedures

Hydrology

First, data were removed where problems had been identified in the field (e.g. logger lifted out of protective piping by animal). Data were then removed where suspected of being unreliable due to probe failure; these periods were typically characterised either by a sharp change in discharge over short periods of time (i.e. doubled or halved in 15 minutes) or by stochastic records that were assumed not to be realistic (see Appendix C for details of removed data). To ensure comparability of datasets between each site, where data were removed during quality control from one site, data for the corresponding time period at all other sites were also removed. This process was repeated prior to analysis of discharge-rainfall relationships.

Electrical conductivity, dissolved oxygen and pH

Sonde data were combined into a single spreadsheet containing all data at 15 minute intervals. Data quality was first ensured using a similar approach to Jones & Graziano (2013) whereby removal of data occurred if any of the following standards were met (which were assumed to indicate probe malfunction): EC: < 10 or > 300 µS/cm; doubled or halved in a 15 minute period; pH: < 3 or > 8; doubled or halved in a 15 minute period ; DO: < 20 or > 150 % saturation; doubled or halved in a 15 minute period. In addition, data were both visually assessed and cross-checked against calibration readings for drift and probe failure/ malfunction and removed if detected; during this process a conservative approach was taken so that if there was any doubt in the quality of data, they were removed (see Appendix C for details of all removed data). Some data were lost due to telemetry failure and a detailed record of these can also be found in Appendix C. To ensure comparability of datasets between each site, where data were removed during quality control from one site, data for the corresponding time period at all other sites were also removed.

5.4.2 Impact of regulation

Calculation of indices

To assess whether differences in hydrological regimes between unregulated and regulated site types could be identified, the following hydrological indices were extracted from each dataset: (i) flood frequency (number of floods per 28 day period); (ii) flood duration; (iii) flood magnitude (percentage increase from minimum to maximum flood discharge (Equation 5.1a x and y respectively)); (iv) rate of change (rising limb); (v) rate of change (falling limb). Rate of change (ROC) was calculated according to Equation 5.1b where M_{max} and M_{min} were maximum and minimum flood magnitude respectively and t_y and t_x were time at flood maximum discharge and flood start time (rising limb ROC) and flood end time and time at flood maximum discharge (falling limb ROC) respectively. This calculation ensured that ROC was comparable between floods and sites. These indices were chosen in accordance with Poff et al. (1997), where flood frequency, duration, magnitude and rate of change were identified as crucial elements of floods for stream ecological integrity.

$$Percent\ change = \frac{y - x}{x} X\, 100$$ Equation 5.1a

$$ROC = \frac{M_{max} - M_{min}}{t_y - t_x}$$ Equation 5.1b

Floods were defined as periods of at least 1 hour where Q_{25} discharge was exceeded. Floods were deemed to have ended when discharge fell below Q_{25}. Q_{25} was chosen to represent floods as it appeared to encapsulate flood events in hydrographs for all sites (Figure 5.2 black/ grey lines) and could thus be used to characterise the general nature of floods at each site.

To assess whether unregulated and regulated hydrological regimes differed temporally, indices as described above were calculated for autumn (1st September 2012 00:00:00 to 30th November 2012 23:59:59), winter (1st December 2012 00:00:00 to 28th February 2013 23:59:59), spring (1st March 2013 00:00:00 to 31st May 2013 23:59:59) and summer (1st June 2013 00:00:00 to 19th August 2013 23:59:59). In addition, *a priori* visualisation of discharge data revealed a period of relatively high flood frequency (Figure 5.6), F*h*, (1st September 2012 00:00:00 to 28th February 2013 23:59:59) and a period of relatively low flood frequency, F*l*, (1st March 2013 00:00:00 to 19th August 2013 23:59:59). Hydrological indices were also calculated for these periods.

Electrical conductivity, dissolved oxygen and pH

To assess whether differences in physical-chemical regimes between unregulated and regulated sites over varying time scales and during flood events could be identified, *mean* and *range* indices were extracted from each dataset for 1- and 7-day periods and for each flood as identified using the method detailed previously. Sufficient data to undertake a full seasonal assessment of physical-chemical dynamics were not available due to data removal during quality control resulting in several periods of missing data (Figures 5.3-5.5) However, data were available to compare periods F*h* and F*l* and thus 1- and 7-day indices were calculated for these periods.

Statistical testing

Hydrology and electrical conductivity, dissolved oxygen and pH

Hydrological and physical-chemical indices were pooled by site type (unregulated or regulated) and mean and standard deviation calculated. Indices were then modelled as response variables using either GLMM or GAMM (Wood, 2011) (depending on whether a linear or non-linear temporal response in indices was evident (assessed through the significance of temporal smoother terms)) as a function of site type. Where models were improved (based on Akaike Information Criterion (AIC)), either time of peak discharge during flood (for hydrology (excluding flood frequency) and physical chemistry during hydrological indices) or day or week number (1:*n* where *n* is total number of days/ weeks) (for 1- and 7-day physical-chemical indices) were also included as explanatory variables. Since indices were extracted from time series, data from each site were classed as repeated measures and thus mixed modelling was appropriate (Zuur et al., 2009). Sites were nested within site type and appropriate error distribution, link functions and correlation structures were specified to ensure approximate normal distribution, homogeneity and independence of residuals. Finally, significance of site type was assessed using t-statistics and associated p-values.

Discharge-precipitation relationship

Correlation (Spearman's rank) between precipitation and discharge at each site was carried out to assess whether differences in precipitation-discharge relationships could be identified between unregulated and regulated site types. The strength of the correlation was then assessed through interpretation of Spearman's rank rho and associated p-values. Prior to analysis, precipitation data were binned to 15 minute periods to ensure synchronicity with discharge.

5.4.3 Impact of Artificial Floods

Hydrology

To assess the impact of each AF on stream hydrology, hydrographs from sites 6 and 11 during each AF were plotted. Further, magnitude, duration, and ROC indices for each AF were extracted according to the method stated above. Additionally, the maximum discharge of each AF was calculated.

Electrical conductivity, dissolved oxygen and pH

To assess the impact of each AF on EC, DO and pH, a modified paired Before-After-Control-Impact (BACIP) (Stewart-Oaten et al., 1986; Smith, 2002) analysis was undertaken which replaced 'After' with 'During' (e.g. Dinger & Marks, 2007) to allow the during-flood impact to be statistically assessed through interrogation of a modelled interaction term (Smith, 2002). For each flood, response variables (raw sonde physical chemistry data) were modelled using either

GLMM or GAMM (Wood, 2011) as a function of *time* (fitted as a smoother where non-linear response in parameters were observed through time), *site* (two levels: site 6 (impact); site 11 (control)) and *period* (two levels: before; during) and the interaction between these two factors (*site:period*). *Period: before* was defined as all data from 00:00:00 until the start of each flood and period: *during* was defined by the start and end times of each flood. As data were collected through time at each site, *site* was treated as a random effect (Zuur et al., 2009). Appropriate error distributions, link functions and correlation structures were specified to ensure approximate normal distribution, homogeneity and independence of residuals. Finally, significance of the *site:period* term was assessed using t-statistics and associated p-values.

5.4.4 Correlation between hydrological and physical-chemical indices

To gain an understanding of general correlation between hydrological and physical-chemical indices for unregulated and regulated site types during a period of normal reservoir operation (i.e. prior to the implementation of AFs), first, plots of hydrological and physical-chemical indices were created for unregulated and regulated site types. Second, an analysis of covariance (ANCOVA) between site types was undertaken to assess whether statistically significant differences in correlations could be identified. ANCOVA was undertaken for each relationship using Generalized Linear Modelling (GLM) with physical-chemical indices as response variables and hydrological indices and site *type* as fixed effects (Rutherford, 2012). Appropriate error distributions, link functions and correlation structures were specified to ensure approximate normal distribution, homogeneity and independence of residuals. Finally significance of the site *type* term was assessed using t-values and associated p-values.

Hydrology-physical chemistry response relationships were then compared between pre-AF flood and AF periods through plotting. Due to the low number of AFs carried out, statistical analysis between the two periods was not possible.

Statistical tests were deemed significant at $p < 0.05$ and all plotting and analysis was undertaken using R v 2.15.3 (2013).

5.5 Results

5.5.1 Impact of regulation

Discharge regime

Discharge at all sites was generally highest from late August 2012 to late February 2013. From March to August 2013, frequent flooding was observed at sites 1,5 and 8. Site 6 also experienced several high discharge events, whereas site 11 experienced little variation in discharge (Figure 5.2).

For all discharge data, mean flood frequency and flood magnitude was highest for unregulated sites, whereas mean flood duration and rising and falling limb ROC was highest for regulated sites. Statistical testing revealed significantly lower flood frequency, and higher ROC (rising and falling limbs) in regulated cf. unregulated sites (Table 5.1). Differences between seasons were apparent: no statistically significant differences between site types were identified during autumn and winter, but significantly lower flood frequency, and ROC (rising and falling limbs) in regulated sites during spring and summer was observed. Additionally, during summer, regulated sites had significantly lower flood duration (Tables 5.1). Assessment of periods F*h* and F*l* revealed significantly higher ROC (falling limb) and significantly lower flood frequency and higher ROC (rising and falling limbs) in regulated sites respectively (Table 5.1).

Graphically, precipitation and discharge appeared generally aligned for all sites apart from site 11 (Figure 5.2). Significant positive correlations were observed between precipitation and discharge for all sites, but relationships were stronger for unregulated sites (Spearman's rho > 0.2) compared to regulated sites 6 and 11 (Spearman's rho 0.16 and 0.13 respectively) (Table 5.2).

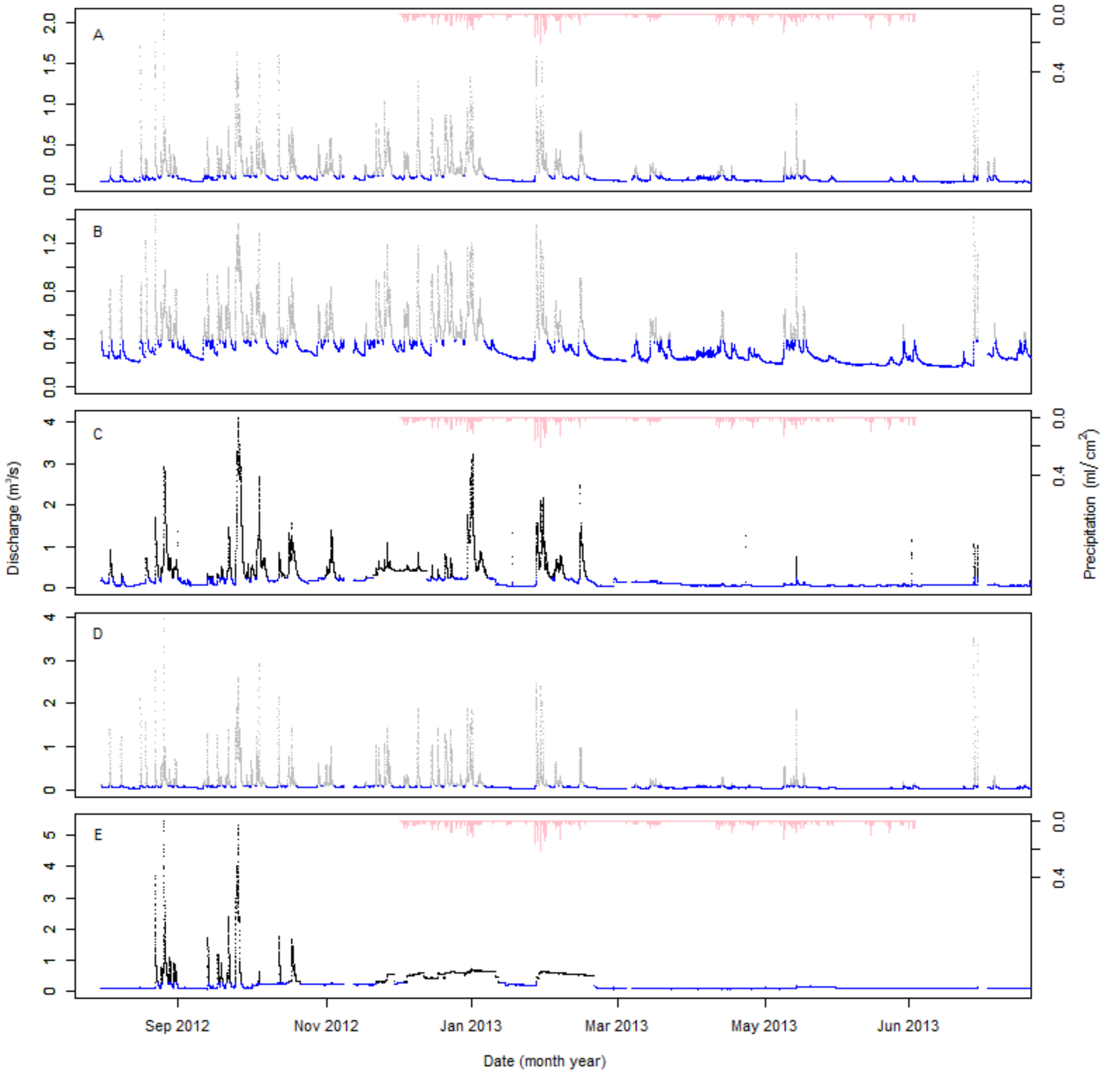

Figure 5.2: Discharge (left hand y-axis) for sites 1, 5, 6, 8 and 11 (A-E respectively) from 20th July 2012 to 19th August 2013. Discharge > Q_{25} is shaded in grey (unregulated) and black (regulated). Precipitation is shown in light red from December 2012 to June 2013 (right hand y-axis).

Table 5.1: Mean, standard deviation and test statistics for hydrological indices for unregulated and regulated sites calculated for various time periods. Model type is also delineated.

	Flood frequency		**Flood duration (days)**		**Flood magnitude (% increase)**		**ROC (rising limb) ($m^3/s/day$)**		**ROC (falling limb) ($m^3/s/day$)**	
	Mean (St. dev.)	**T-value Model type**	**Mean (St. dev.)**	**T-value Model type**	**Mean (St. dev.)**	**T-value Model type**	**Mean (St. dev.)**	**T-value Model type**	**Mean (St. dev.)**	**T-value Model type**
ALL DATA										
Unregulated	4.55 (2.66)	-2.29 *	1.45 (1.48)	1.56	531.00 (862.93)	-0.47	3.00 (5.98)	-5.24 **	0.68 (1.33)	2.34 *
Regulated	3.18 (3.99)	GLMM	2.31 (5.78)	GAMM	299.00 (438.86)	GAMM	7.13 (19.50)	GAMM	3.95 (14.96)	GLMM
AUTUMN										
Unregulated	5.92 (1.72)	-1.86	1.51 (1.31)	0.69	505.03 (751.78)	1.51	2.18 (3.91)	-3.09	0.48 (0.45)	1.57
Regulated	4.13 (2.58)	GLMM	1.91 (2.51)	GAMM	323.67 (466.53)	GAMM	2.55 (4.64)	GLMM	0.79 (0.96)	GAMM
WINTER										
Unregulated	3.17 (1.70)	1.49	2.72 (2.19)	-0.10	610.48 (748.98)	-1.36	2.02 (2.38)	0.68	0.37 (0.31)	-0.63
Regulated	5.13 (4.45)	GLMM	2.59 (7.45)	GLMM	125.78 (279.55)	GAMM	4.06 (18.53)	GLMM	1.68 (8.41)	GAMM
SPRING										
Unregulated	3.42 (2.15)	-3.35 **	0.78 (0.53)	1.59	175.82 (368.10)	0.45	0.90 (1.57)	-6.21 **	0.34 (0.33)	-9.32 **
Regulated	0.25 (0.46)	GLMM	0.17 (0.20)	GLMM	332.61 (152.40)	GAMM	52.83 (64.55)	GAMM	25.71 (33.27)	GAMM
SUMMER										
Unregulated	2.22 (1.39)	-2.23 **	0.84 (1.32)	2.33 *	586.43 (1161.05)	-0.32	4.30 (8.27)	4.56 **	1.68 (3.38)	3.66 **
Regulated	0.50 (0.84)	GLMM	0.17 (0.12)	GAMM	351.96 (25.83)	GAMM	43.06 (36.67)	GAMM	16.87 (22.32)	GLMM
F*h*										
Unregulated	3.17 (3.49)	0.18	2.03 (2.77)	-0.28	639.80 (907.02)	-0.60	3.32 (5.96)	-1.77	0.59 (0.61)	2.13 *
Regulated	3.00 (4.10)	GLMM	2.25 (5.56)	GLMM	306.35 (502.08)	GLMM	3.84 (12.40)	GLMM	2.65 (12.78)	GLMM
F*l*										
Unregulated	1.45 (1.99)	-3.08 **	0.80 (0.90)	-1.94	331.35 (788.29)	-0.32	2.21 (5.47)	-2.23 *	0.82 (2.12)	7.29 **
Regulated	0.18 (0.48)	GLMM	0.17 (0.13)	GAMM	344.22 (79.07)	GAMM	46.97 (41.75)	GLMM	20.41 (23.44)	GLMM

$*p < 0.05$, $**p < 0.01$.

Table 5.2: Spearman's rank rho and significance for precipitation and discharge correlations at sites 1, 2, 6, 8 and 11.

Site	Type	Rho
1	Unregulated	0.20 *
2	Unregulated	0.21 *
6	Regulated	0.16 *
8	Unregulated	0.22 *
11	Regulated	0.13 *

$*p < 0.05$, $**p < 0.01$.

Electrical Conductivity

Mean EC was similar at all sites throughout the study period, whereas diurnal EC range was similar at all sites during August – December 2012 but more variable during 2013, particularly at site 6 (Figure 5.3). For the entire study and F*h* and *Fl* periods, mean EC was highest for regulated sites at both 1- and 7-day time scales and EC range was lowest for regulated sites at both 1- and 7-day time scales, but these differences were not statistically significant (Table 5.3). EC range during floods was highest for regulated sites and mean flood EC was lower in regulated sites; the latter being statistically significant (Table 5.4).

Dissolved oxygen

Daily mean DO was similar at all sites and generally between 90 and 100% throughout the study period. Daily DO range at all sites was also similar, but generally increased from January to August 2013 (Figure 5.4). Mean DO was highest for regulated sites at both 1- and 7-day time scales for the entire study, F*h* and *Fl* periods, but these differences were not statistically significant. Conversely, DO range was lowest for regulated sites at both 1- and 7-day time scales and for all periods. These differences were statistically significant for all periods and time scales, apart from during F*h* at the 7-day time scale (Table 5.3). During flooding, DO was significantly higher at regulated sites, but there was no significant difference in range between site types (Table 5.4).

pH

The temporal trend of daily mean pH at regulated sites was similar, but pH at site 10 (unregulated) appeared more variable. This variability was also reflected in daily pH range which was generally higher at site 10 (Figure 5.5). Mean pH was highest and range was lowest in regulated sites at both 1- and 7-day time scales for all periods, but the difference in mean pH was not significant. The difference in 1-day pH range was statistically significant for the entire study and F*l* periods, and at a 7-day time scale only during F*h* (Table 5.3). During floods, mean pH was significantly higher at regulated sites and range was lower (but not significantly) for regulated sites (Table 5.4).

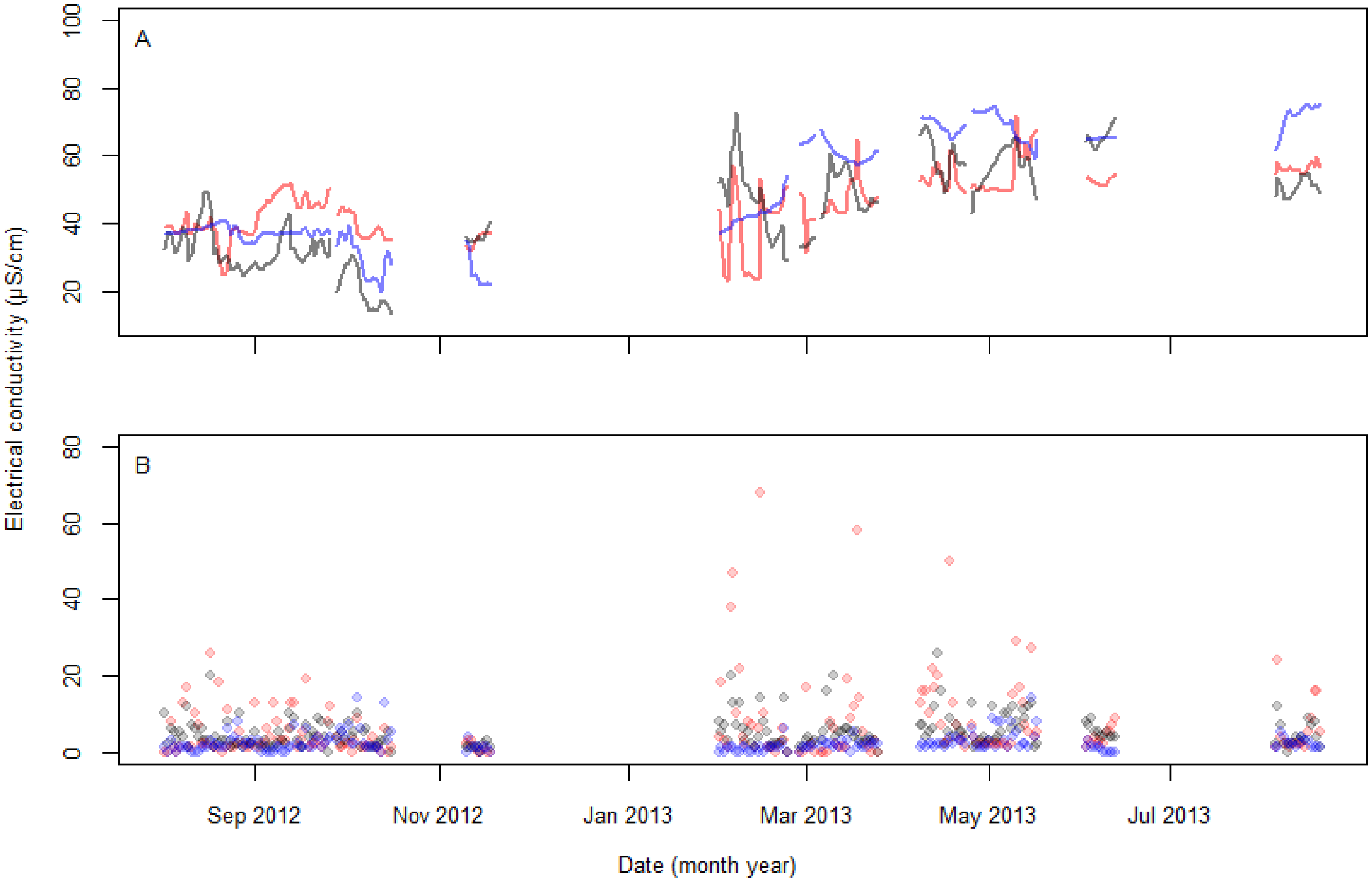

Figure 5.3: Electrical conductivity daily mean and range (panels A and B respectively) at sites 6, 10 and 11 (red, black and blue respectively) for the study period.

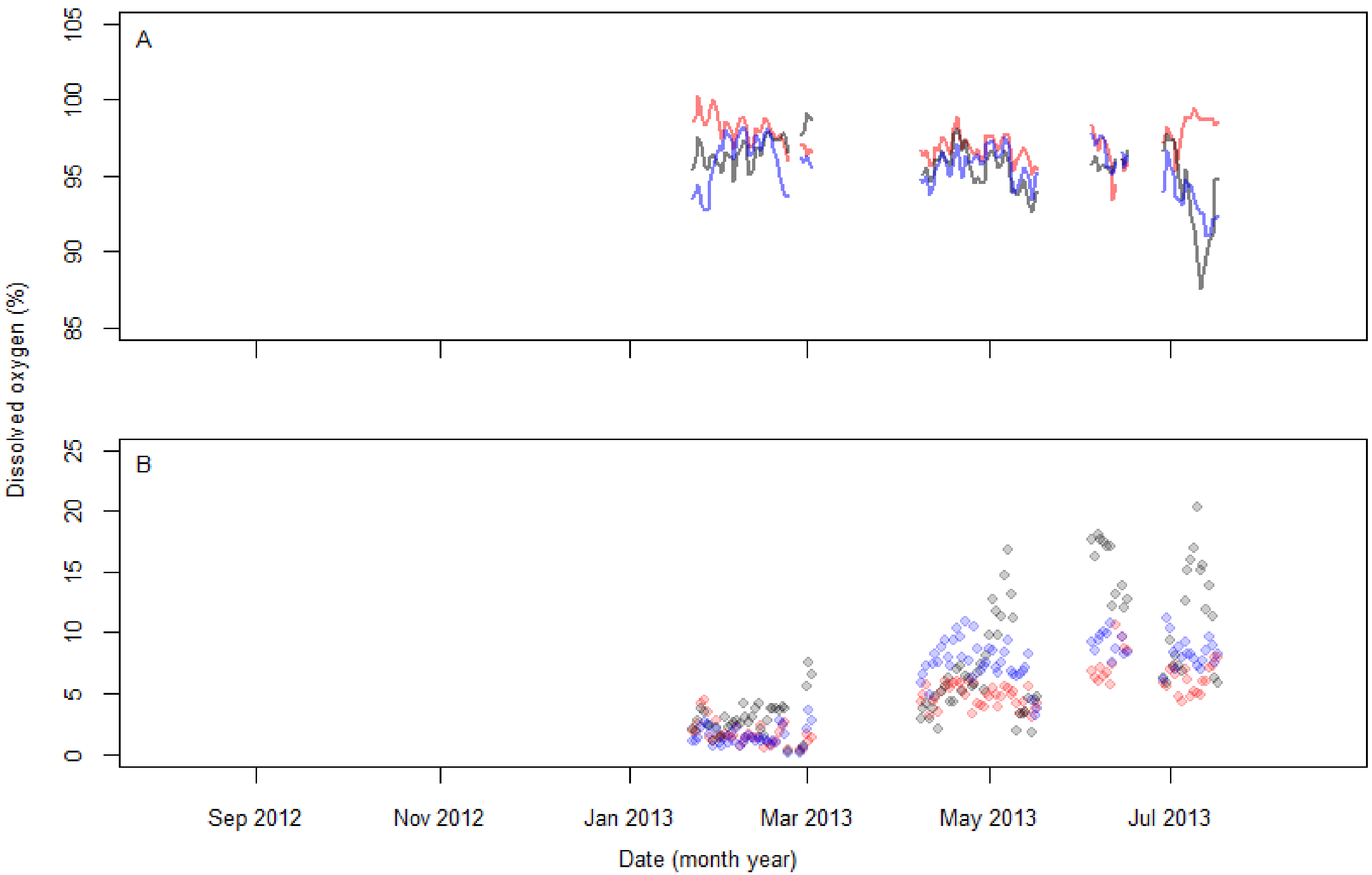

Figure 5.4: Dissolved oxygen daily mean and range (panels A and B respectively) at sites 6, 10 and 11 (red, black and blue respectively) for the study period.

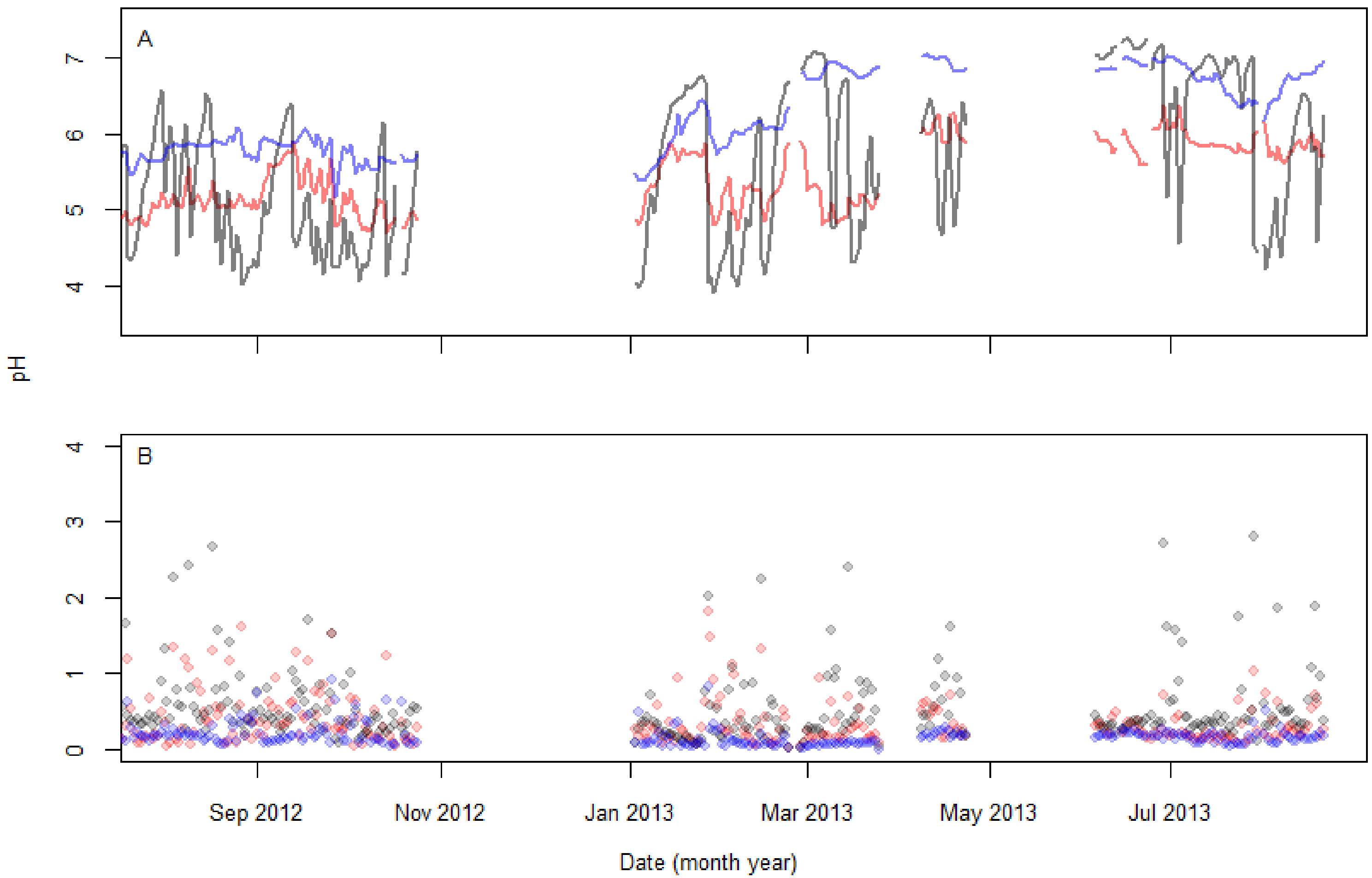

Figure 5.5: pH daily mean and range (panels A and B respectively) at sites 6, 10 and 11 (red, black and blue respectively) for the study period.

*Table 5.3: Mean, standard deviation and test statistics for 1- and 7-day EC (µS/cm), DO (%) and pH indices for unregulated and regulated site types for entire study, F*h and F*l* periods*. Model type is also delineated.*

	Entire study				**F*h***				**F*l***			
	Mean		**Range**		**Mean**		**Range**		**Mean**		**Range**	
	Mean (St. dev.)	**T-value**	**Mean (St. dev.)**	**T-value**	**Mean (St. dev.)**	**T-value**	**Mean (St. dev.)**	**T-value**	**Mean (St. dev.)**	**T-value**	**Mean (St. dev.)**	**T-value**
		Model type		**Model type**		**Model type**		**Model type**		**Model type**		**Model type**
1-day												
EC												
Unregulated	43.75 (14.01)	1.64	5.06 (4.31)	-0.44	35.93 (11.78)	0.93	4.50 (4.21)	-0.63	54.69 (8.46)	0.76	5.84 (4.34)	-0.53
Regulated	47.68 (13.21)	GAMM	4.14 (7.10)	GAMM	39.18 (7.96)	GAMM	3.63 (6.77)	GAMM	59.57 (9.34)	GAMM	4.86 (7.49)	GAMM
DO												
Unregulated	95.59 (2.03)	0.90	7.03 (5.15)	-2.09 **	96.62 (0.92)	0.19	2.60 (1.09)	-6.06 **	95.10 (2.22)	0.61	9.13 (4.99)	-2.36 *
Regulated	96.45 (1.74)	GAMM	4.94 (3.02)	GAMM	97.09 (1.73)	GAMM	1.46 (0.85)	GAMM	96.14 (1.66)	GAMM	6.59 (2.14)	GAMM
pH												
Unregulated	5.53 (1.03)	-0.63	0.55 (0.49)	-3.29 **	5.14 (0.88)	1.11	0.51 (0.47)	-1.43	6.18 (0.93)	0.09	0.59 (0.51)	-3.11 **
Regulated	5.79 (0.67)	GAMM	0.27 (0.26)	GAMM	5.56 (0.43)	GAMM	0.30 (0.32)	GAMM	6.26 (0.61)	0.61	0.22 (0.16)	GLMM
7-day												
EC												
Unregulated	43.57 (13.90)	1.41	14.50 (7.90)	-0.20	36.31 (12.31)	-0.02	14.09 (8.09)	-0.43	53.55 (9.00)	0.80	15.06 (7.86)	-0.12
Regulated	47.31 (13.02)	GAMM	13.07 (15.18)	GAMM	38.63 (6.98)	GAMM	12.32 (15.55)	GAMM	59.24 (9.44)	GAMM	14.09 (14.84)	GAMM
DO												
Unregulated	95.86 (1.92)	0.55	10.42 (5.29)	-3.05 **	96.56 (0.61)	-1.85	5.36 (1.02)	1.78	95.63 (2.15)	0.36	12.11 (5.05)	2.36 *
Regulated	96.38 (1.57)	GAMM	7.28 (3.29)	GAMM	97.11 (1.69)	GAMM	4.05 (1.35)	GLMM	96.15 (1.47)	GAMM	8.35 (3.04)	GAMM
pH												
Unregulated	5.52 (0.92)	0.61	1.67 (0.82)	1.86	5.07 (0.74)	1.35	1.66 (0.63)	-2.21 *	6.16 (0.82)	0.11	1.59 (0.99)	1.89
Regulated	5.80 (0.65)	GAMM	0.67 (0.42)	GAMM	5.52 (0.41)	GLMM	0.80 (0.47)	GLMM	6.26 (0.62)	GLMM	0.47 (0.27)	GLMM

$^*p < 0.05$, $^{**}p < 0.01$.

Table 5.4: Mean, standard deviation and test statistics for EC (µS/cm), DO (%) and pH indices for unregulated and regulated site types during floods. Model type is also delineated.

	Flood mean		Flood Range	
	Mean (St. dev.)	**T-value** **Model type**	**Mean (St. dev.)**	**T-value** **Model type**
EC				
Unregulated	41.17 (16.29)	2.85 *	6.38 (5.94)	1.36
Regulated	38.26 (7.68)	GAMM	8.92 (15.61)	GLMM
DO				
Unregulated	95.83 (1.09)	2.57 *	2.74 (1.95)	-0.16
Regulated	97.16 (1.32)	GAMM	2.92 (2.35)	GAMM
pH				
Unregulated	4.59 (0.64)	9.08 *	0.81 (0.80)	-0.76
Regulated	5.36 (0.34)	GAMM	0.56 (0.48)	GAMM

$^*p < 0.05$, $^{**}p < 0.01$.

5.5.2 Impact of Artificial Floods

Hydrology

The magnitudes of AFs 1 and 2 were below the threshold required to produce flood statistics using the method described in section 5.3. The following statistics are therefore provided from graphical analysis of each AF and are not comparable with pre-AF flood statistics. AF 1 was the longest AF (0.2 days) – AFs 2-4 were all approximately 0.1 days in length. Each AF was progressively larger in maximum discharge, but due to the increased compensation flow released from reservoir C between AFs 2 and 3, AF 2 had the largest % increase in magnitude. Each flood had a progressively larger rising and falling ROC (Table 5.5; Figure 5.6).

Physical chemistry

The largest range in EC was observed during AFs 3 and 4 (6 µS/cm), whereas EC range was only 1 µS/cm during AF 1 (Table 5.6). Mean EC decreased at site 6 during all AFs, but significant impacts were only observed for AFs 1, 2 and 4 (Table 5.7; Figure 5.7). The largest range in DO was observed during AF 1 (6.8 % saturation), whereas the lowest DO range was seen during AF 3 (1.5 % saturation) (Table 5.6). Mean DO response to each AF varied: significant impacts were only observed during AFs 1 and 2 (Table 5.7) where clear reductions in DO occurred at site 6 when compared to site 11 (Figure 5.8). An apparent reduction in DO during AFs 3 and 4 was not significant likely due to variance in DO at site 11 and pre AF at site 6 being observed (Figure 5.8). The largest range in pH was observed during AF 2 (0.41), whereas the smallest range was observed during AF 3 (0.06) (Table 5.6). pH was significantly impacted by each AF, reflected by the reduction in pH during each flood at site 6 (Table 5.7; Figure 5.9).

Table 5.5: hydrological indices for Artificial Floods (AFs) – note, these statistics were extracted from graphical analysis of each AF.

AF	**Date**	**Duration (days)**	**Magnitude (% increase)**	**ROC (rising limb) (m^3/s/day)**	**ROC (falling limb) (m^3/s/day)**	**Max. discharge (m^3/s)**
1	**20/08/13**	0.2	287.1	1.61	1.07	0.17
2	**19/09/13**	0.11	424.39	8.39	3.15	0.31
3	**03/10/13**	0.12	294.02	9.95	7.13	0.67
4	**05/11/13**	0.13	423.85	11.39	13.24	0.99

Table 5.6: EC, DO and pH range during each Artificial Flood (AF) – note, these statistics were extracted from graphical analysis of each AF.

AF	**EC range (µS/cm)**	**DO range (% saturation)**	**pH range (pH units)**
1	1	6.8	0.18
2	4	3.1	0.41
3	6	1.5	0.06
4	6	2.5	0.31

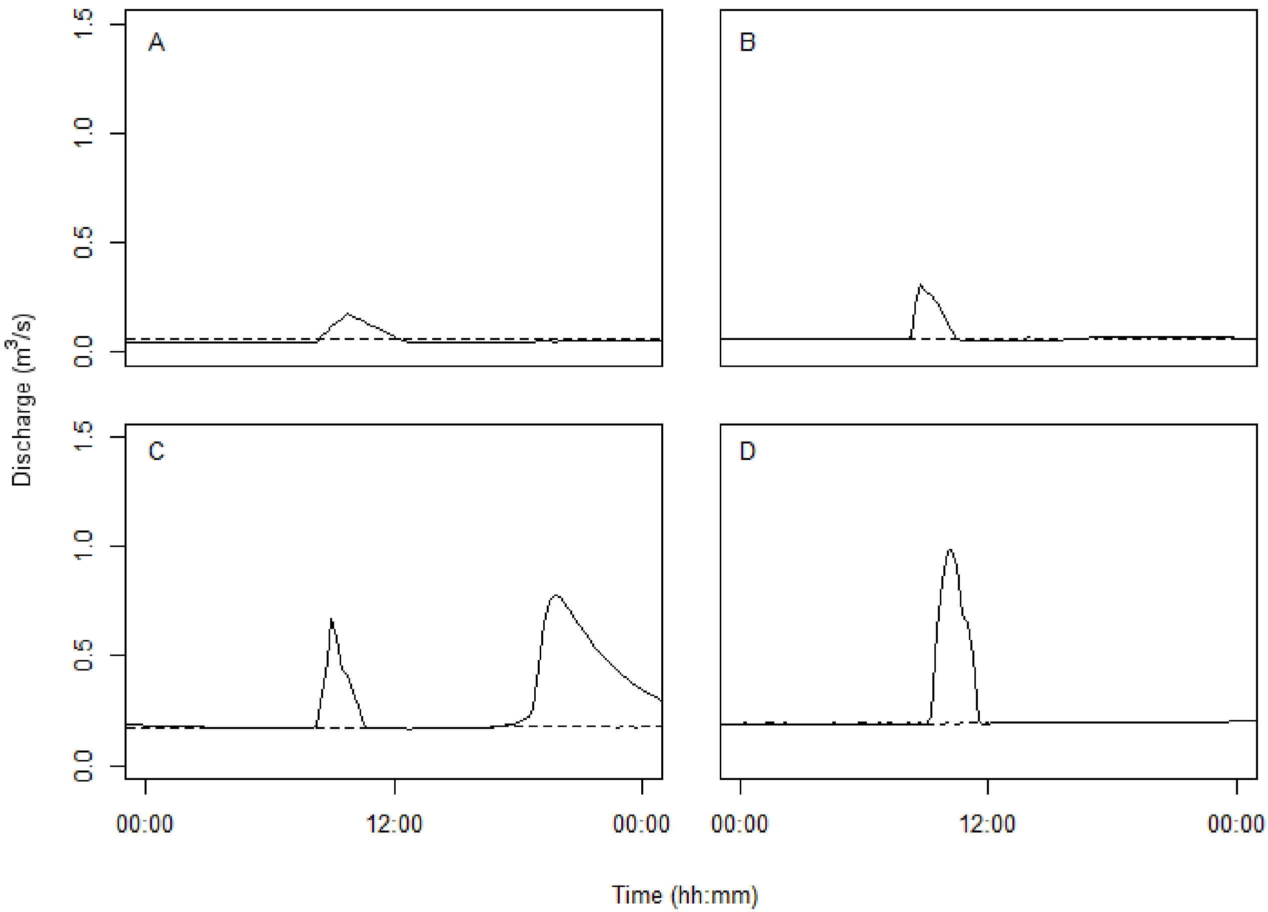

Figure 5.6: Hydrographs at site 6 during Artificial Floods (AFs) 1-4 (panels A-D respectively) (full lines). Dashed lines are synchronous hydrographs at site 11. Note: an overspill event occurred after AF 3.

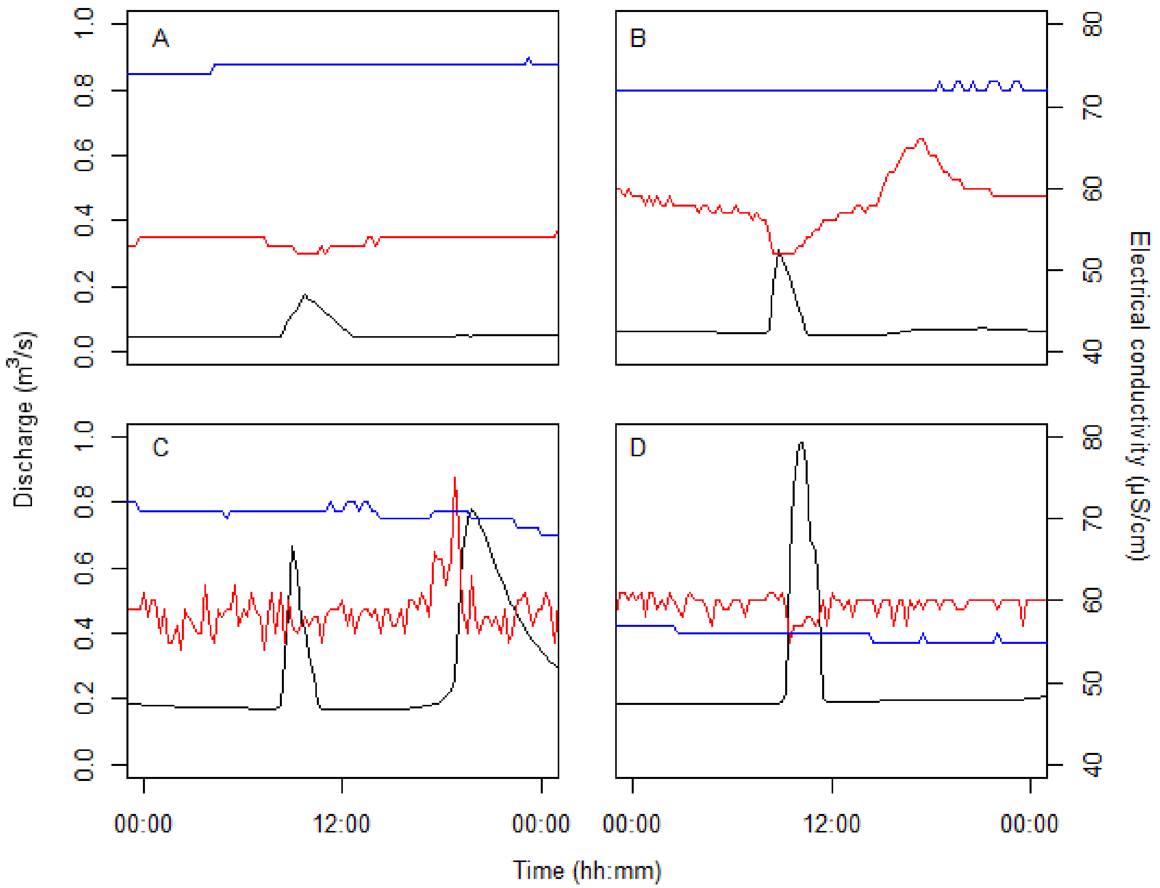

Figure 5.7: Electrical conductivity at sites 6 and 11 (red and blue lines respectively) on the day of Artificial Floods (AFs) 1-4 (panels A-D respectively). Black lines are hydrographs at site 6.

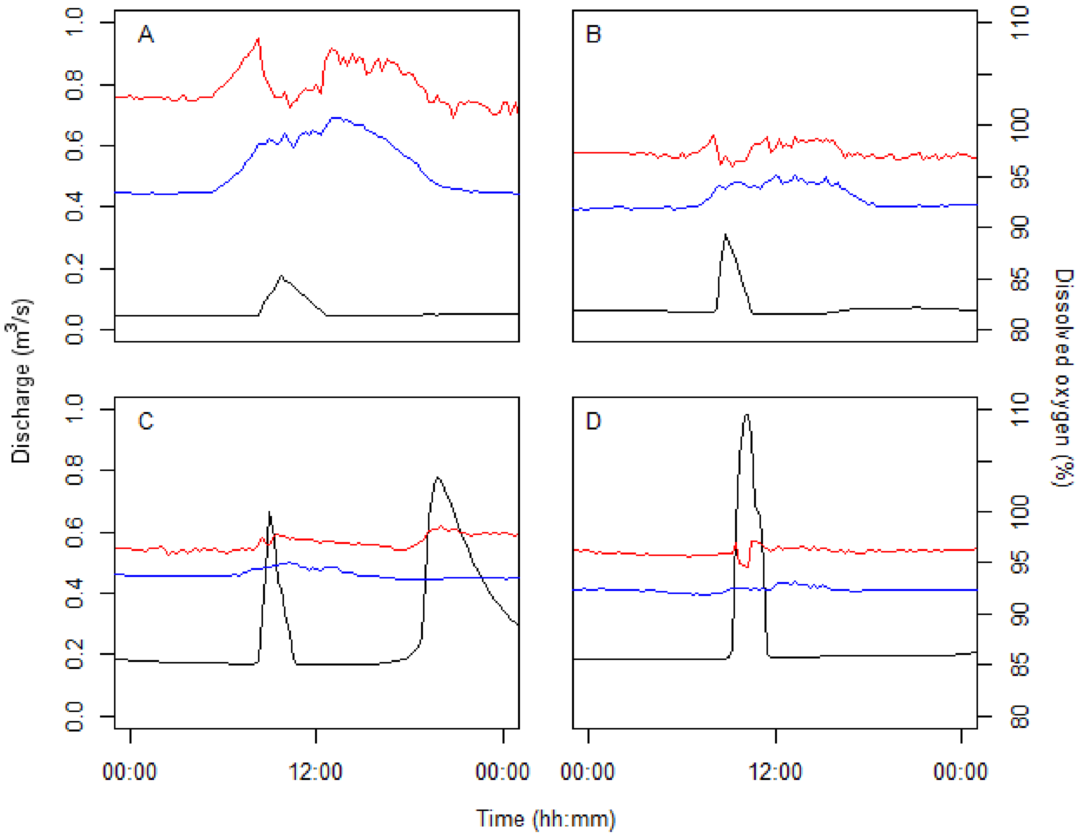

Figure 5.8: Dissolved oxygen at sites 6 and 11 (red and blue lines respectively) on the day of Artificial Floods (AFs) 1-4 (panels A-D respectively). Black lines are hydrographs at site 6.

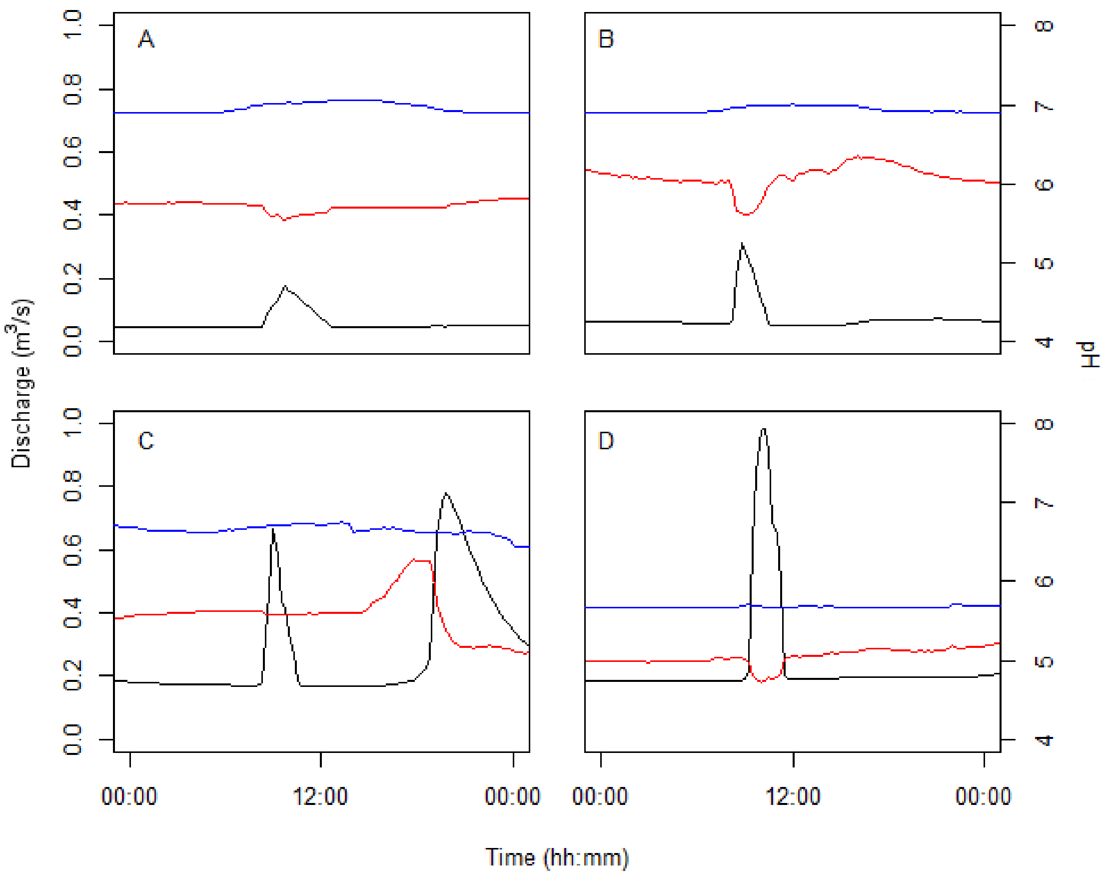

Figure 5.9: pH at sites 6 and 11 (red and blue lines respectively) on the day of Artificial Floods (AFs) 1-4 (panels A-D respectively). Black lines are hydrographs at site 6.

Table 5.7: Mean (& St. dev.) and test statistics for electrical conductivity (EC) (μS/cm), dissolved oxygen (DO) (%) and pH indices for impact (site 6) and control (site 11) sites before and during AFs 1-4. Model type is also delineated.

	AF 1		**AF 2**		**AF 3**		**AF 4**	
	Before	**During**	**Before**	**During**	**Before**	**During**	**Before**	**During**
EC								
Impact	53.90 (0.32)	52.63 (0.50)	57.90 (0.74)	52.91 (1.30)	57.90 (2.18)	57.30 (1.83)	59.76 (0.99)	58.00 (1.63)
Control	74.50 (0.53)	75.00 (0.00)	72.00 (0.00)	72.00 (0.00)	71.00 (0.00)	71.00 (0.00)	56.29 (0.46)	56.00 (0.00)
Test statistic	13.53 **	GAMM	15.91 **	GAMM	0.85	GLMM	3.59 **	GLMM
DO								
Impact	103.45 (1.39)	103.93 (1.80)	97.30 (0.30)	96.97 (0.94)	96.25 (0.18)	97.23 (0.46)	95.83 (0.13)	96.09 (0.89)
Control	93.92 (1.02)	98.65 (0.66)	91.99 (0.24)	94.07 (0.33)	93.80 (0.12)	94.71 (0.22)	92.19 (0.18)	92.44 (0.13)
Test statistic	13.83 **	GAMM	2.41 **	GAMM	-1.02	GAMM	-0.06	GAMM
pH								
Impact	5.75 (0.01)	5.62 (0.05)	6.06 (0.04)	5.76 (0.15)	5.61 (0.02)	5.58 (0.02)	4.99 (0.02)	4.87 (0.13)
Control	6.91 (0.02)	7.02 (0.02)	6.91 (0.01)	6.97 (0.01)	6.65 (0.02)	6.72 (0.01)	5.67 (0.01)	5.69 (0.01)
Test statistic	25.74 **	GAMM	12.42 **	GAMM	10.73 **	GAMM	6.58 **	GLMM

$*p < 0.05$, $**p < 0.01$.

5.5.3 General correlation between hydrological and physical-chemical indices

The general correlation between all hydrological indices and mean DO and pH and pH range prior to AFs significantly differed between unregulated and regulated site types. No significant differences in correlations between hydrological indices and mean EC and range of EC and DO were identified (Table 5.8). Graphical representation of correlations (Figures 5.10 - 5.13) revealed that mean DO was generally higher for regulated sites for all hydrological indices, apart from at high rates of change in the falling limb of floods. Mean pH was typically higher for regulated sites at all levels of all hydrological indices. Conversely, pH range was generally lower for regulated sites for all hydrological indices apart from magnitude, where at high flood magnitudes (> ~ 2000 % increase), this relationship reversed.

Statistical comparison of the correlation between hydrological and physical-chemical indices between normal reservoir operation and AF periods was not possible due to the small number of AFs implemented. The magnitudes of AFs 1 and 2 were below the threshold required to produce flood statistics using the method described in section 5.3, but indices for AFs 3 and 4 were produced and these datapoints were plotted on Figures 5.10 - 5.13. Visual assessment revealed that mean and range physical-chemical indices were within the range of indices extracted for pre-AF floods, but whilst both magnitude and duration indices of AFs were comparable with pre-AF floods, both AF3 and 4 had relatively large ROC (rising and falling limb) indices compared to pre-AF hydrological indices.

Table 5.8: ANCOVA test results (t-values and significance) for correlation between hydrological and physical-chemical indices between regulated and unregulated stream types.

	EC		**DO**		**pH**	
	Mean	**Range**	**Mean**	**Range**	**Mean**	**Range**
Magnitude	-0.75	-1.05	2.95 **	-0.27	9.14 **	-2.02 *
Duration	-1.08	-1.22	2.86 **	1.31	8.92 **	4.1 **
ROC rising	-0.41	-0.43	3.11 **	-0.20	2.07 *	3.19 **
ROC falling	0.21	-0.09	2.91 **	-0.41	2.05 *	2.61 *

$^*p < 0.05$, $^{**}p < 0.01$.

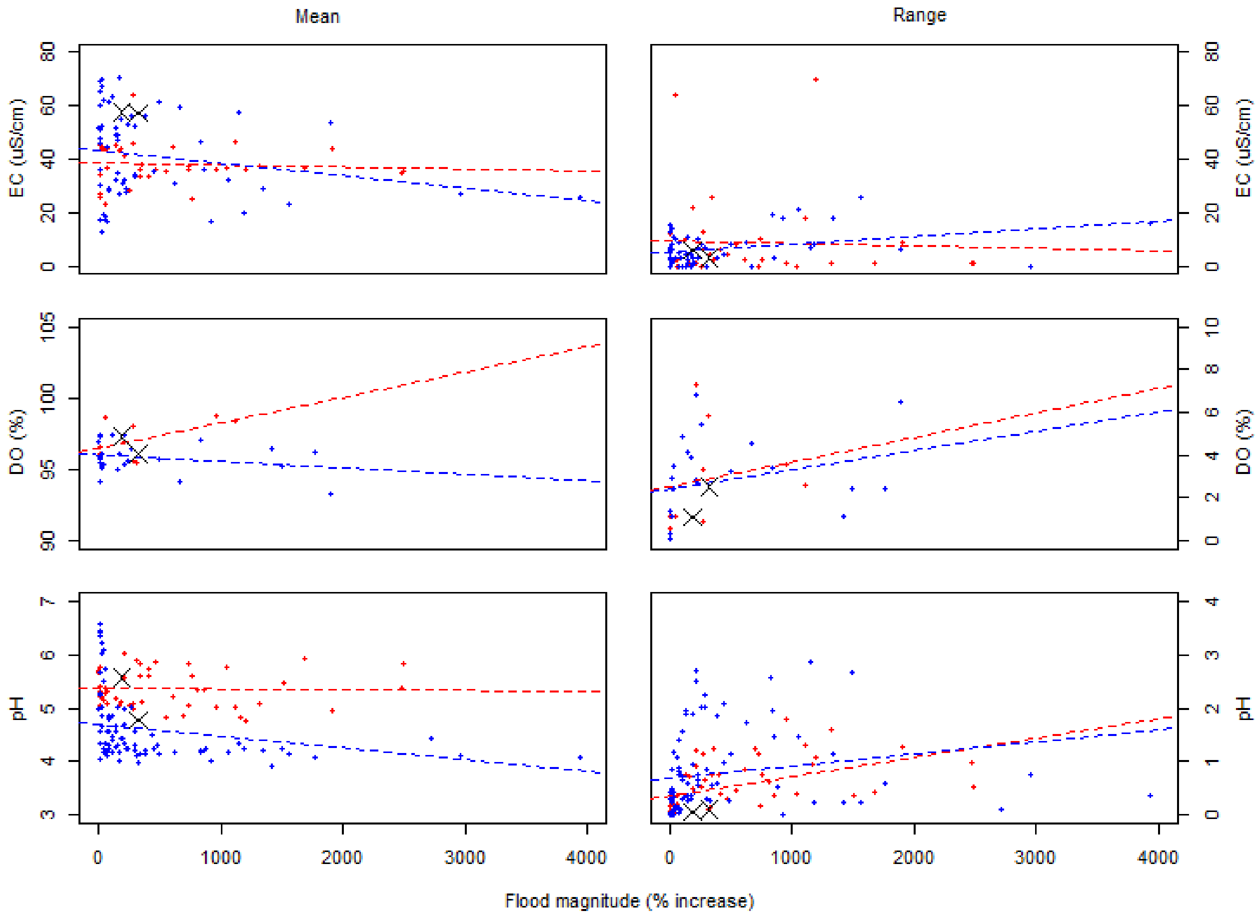

Figure 5.10: Correlation between flood magnitude and physical-chemical mean and range indices (left and right panels respectively) for unregulated site 10 (blue) and regulated sites 6 and 11 (red) during floods prior to Artificial Floods (AFs) and site 6 during AFs (black crosses). Linear regression trend lines for each correlation are displayed where possible.

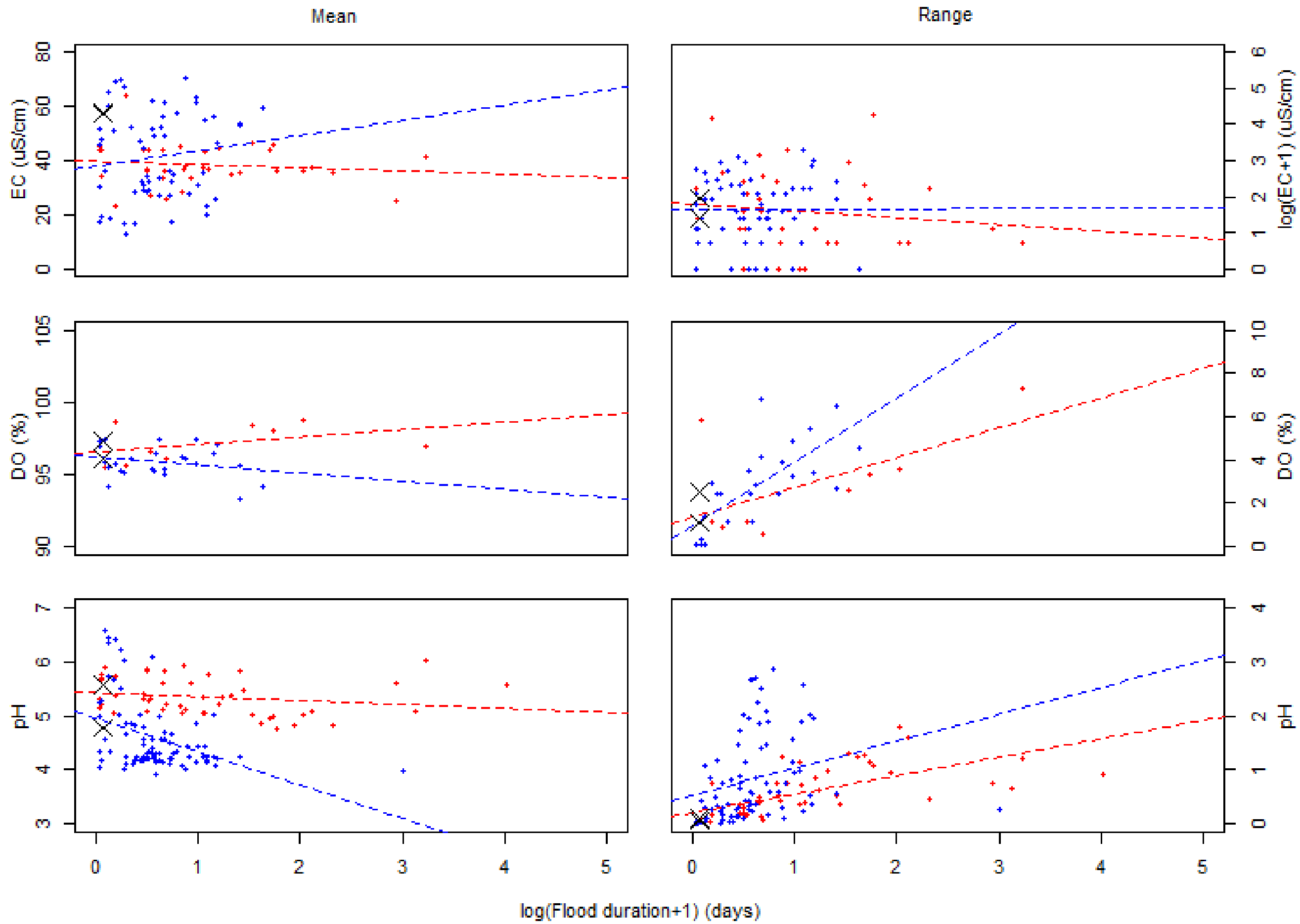

Figure 5.11: Correlation between flood duration and physical-chemical mean and range indices (left and right panels respectively) for unregulated site 10 (blue) and regulated sites 6 and 11 (red) during floods prior to Artificial Floods (AFs) and site 6 during AFs (black crosses). Linear regression trend lines for each correlation are displayed where possible.

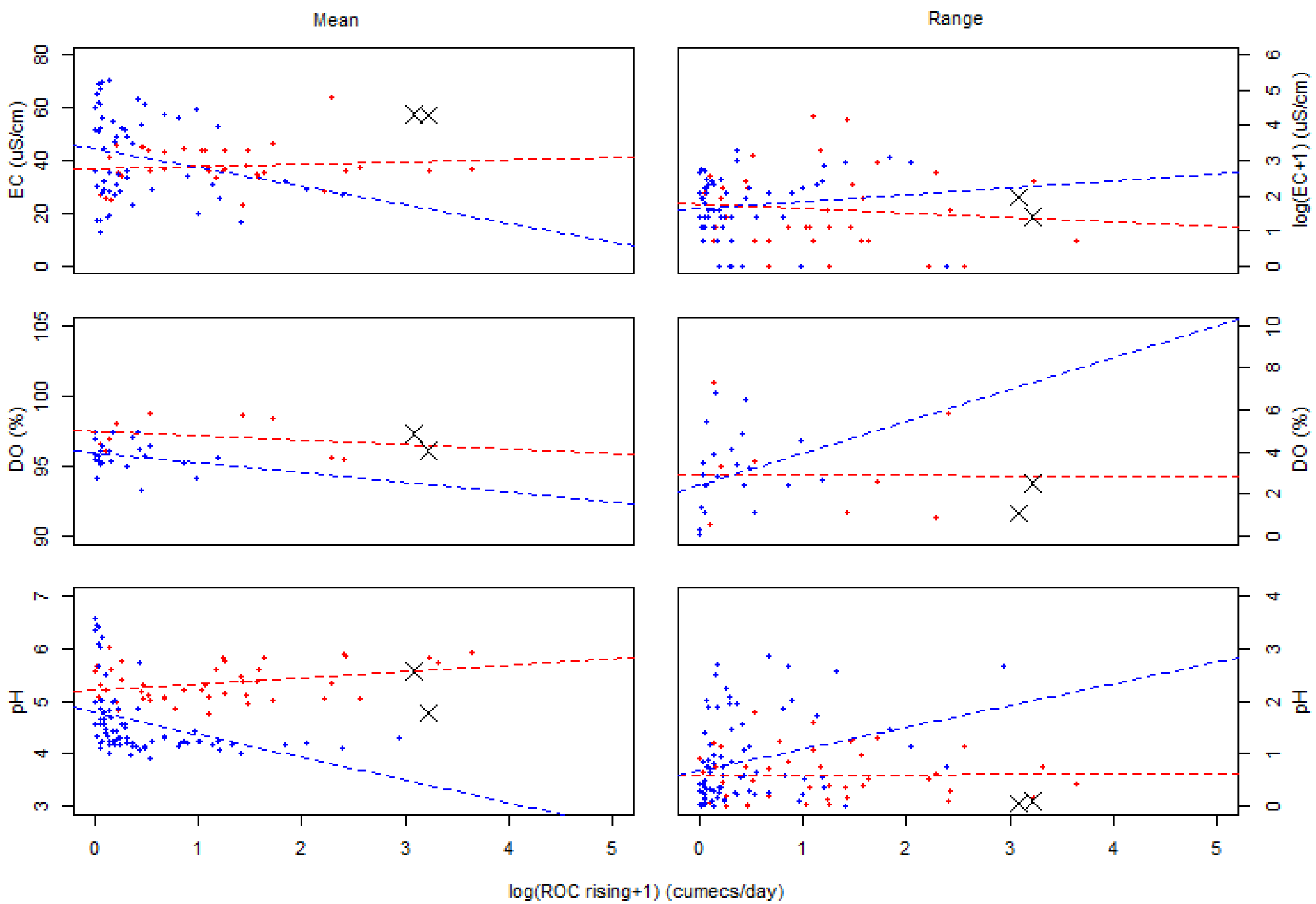

Figure 5.12: Correlation between flood ROC rising and physical-chemical mean and range indices (left and right panels respectively) for unregulated site 10 (blue) and regulated sites 6 and 11 (red) during floods prior to Artificial Floods (AFs) and site 6 during AFs (black crosses). Linear regression trend lines for each correlation are displayed where possible.

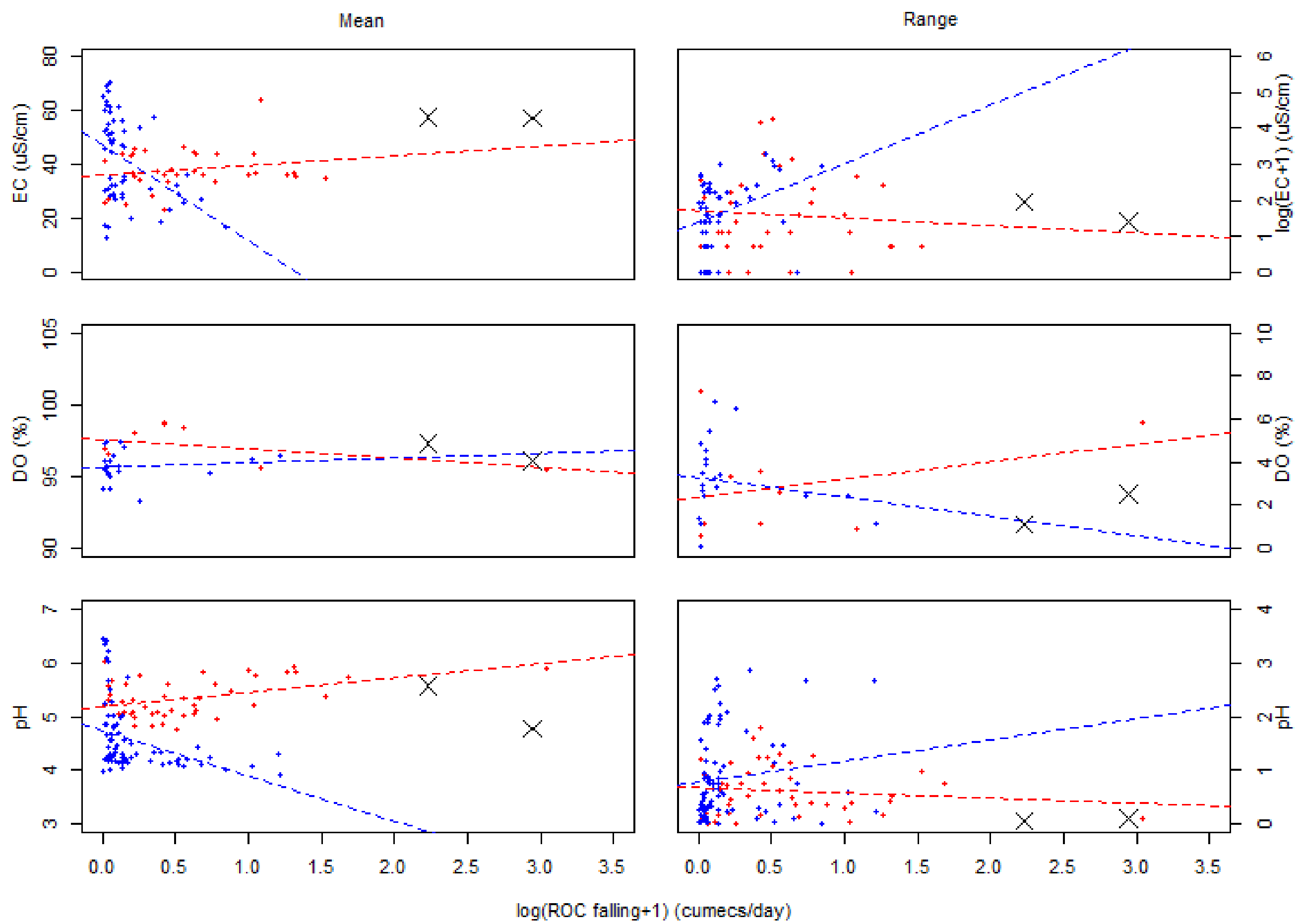

Figure 5.13: Correlation between flood ROC falling and physical-chemical mean and range indices (left and right panels respectively) for unregulated site 10 (blue) and regulated sites 6 and 11 (red) during floods prior to Artificial Floods (AFs) and site 6 during AFs (black crosses). Linear regression trend lines for each correlation are displayed where possible.

5.6 Discussion

This chapter reported on a detailed study of hydrology, EC, DO and pH in a catchment in upland UK over a two year period. An assessment of the impact of regulation on each of these parameters was undertaken through comparison with unregulated conditions at a variety of scales. A series of AFs were conducted which revealed significant impacts to EC, DO and pH. Finally, a comparison of the correlation between hydrological indices and physical-chemical responses during AFs was carried out. The following is a discussion of each of these themes in turn.

5.6.1 Impact of regulation

Hydrology

Marked differences in regulated and unregulated flow regimes have been reported globally. Typically, magnitude of minimum flows is increased (Higgs & Petts, 1988; Gustard, 1989), median flow magnitude is reduced (Higgs & Petts, 1988) and high flows are reduced in magnitude and frequency (Higgs & Petts, 1988; Gustard, 1989) resulting in an overall reduction in flow variation (Baxter, 1977). In agreement with these findings, this study identified significant differences between regulated and unregulated flow regimes. This impact was further evidenced by identification of significant positive correlations between precipitation and discharge for all sites, but a stronger correlation between precipitation and discharge for unregulated sites was identified. This reduction in the strength of correlation between precipitation rate and discharge in regulated streams is likely to be due to the attenuation of water within the reservoir if storage space is available (Petts, 1990) prior to overspilling.

It was hypothesised that temporal (seasonal and event based) differences in impacts of regulation on downstream hydrology would be identified (H1), and this hypothesis was upheld. Mid-late 2012 was characterised by relatively large volumes of precipitation in the study area, indeed, on the 22nd June 2012, the highest ever stream level was recorded by the EA at a weir located approximately 0.5km downstream of site 18 (EA, 2013). During this period, reservoir water levels were atypically high for this time of year, resulting in regular overspill events reflected by no significant differences between unregulated and regulated flow regime indices for autumn 2012 or winter 2012/13. Analysis of these two seasons combined (period: Fh) also failed to identify differences between regulated and unregulated flow regimes, apart from significantly higher ROC (falling limb) for regulated regimes.

Spring and summer 2013 were characterised by relatively low precipitation, resulting in lower reservoir water levels and fewer overspill events. This change in overspill frequency resulted in significant differences between unregulated and regulated flow regime indices and for this period (F*l*); flood frequency was significantly lower and ROC (rising and falling limb) significantly higher in regulated streams. Seasonal analysis also identified these significant differences in spring and summer, and in summer, flood duration was significantly shorter in regulated streams. High ROC has been identified in streams affected by hydro-electric power generation (e.g. Cushman, 1985) but not downstream of water supply reservoirs. The reason for these differences is unclear, but it may be due to the relatively large step-change in discharge caused by the start/ stop nature of overspill events, especially if the width of an overspill crest is large relative to the compensation discharge. Reduction in flood duration has previously been identified in regulated streams (e.g. Andrews, 1986), and is likely to be as a result of the attenuating ability of a reservoir (Petts, 1990). No significant differences in flood magnitude were observed between stream types for any season and this may be a result of the method used which meant that *all* floods rather than just *large* floods (which are typically reported to have reduced in magnitude post-impoundment (Higgs & Petts, 1988)) were selected for analysis resulting a large variance in mean flood magnitude.

The evidence for modification of the discharge regime identified for the headwater streams in this study could be viewed as contrary to a proven model of hydrological response to regulation in headwaters (the Serial Discontinuity Concept (SDC) (Ward & Stanford, 1983; 1995; Stanford & Ward, 2001). The SDC postulated that regulation of headwater streams would result in no change to annual discharge fluctuation (Ward & Stanford, 1983). Conversely, this study, in agreement with other studies (e.g. Crisp, 1977; Higgs & Petts, 1988), has identified evidence of reduced discharge fluctuation in the form of reduced flood frequency in an regulated headwater stream. The SDC was developed on streams with high groundwater contribution, and therefore relatively low annual discharge variation (Ward & Stanford, 1983). This may therefore explain the disparity in our conclusions, but serves to stress that generalisations across regulated stream 'types' (e.g. headwaters), should be made with care.

EC, DO and pH

Typically, a reduction in EC downstream of reservoirs has been observed (e.g. Soja & Wiejaczka, 2013; Palmer & O'Keeffe, 1990). This study found no significant differences in EC mean or range at either 1- or 7-day time scales for the entire study period or F*h* or F*l*. However, in support of H2 (that regulation would impact downstream physical chemistry), during flood events, mean regulated EC was significantly lower than during floods at the unregulated site.

These differences are potentially a result of water with low EC transferring into the downstream watercourse during overspill events, contrasting with relatively high EC water released throughout the year as compensation flow. Limnetic reservoir water with low EC may develop due to ionic uptake by phytoplankton (e.g. Atkins & Harris, 1925) and water with relatively high EC has been observed at the lower levels of reservoirs (e.g. Soares et al., 2008) potentially explaining both the lack of statistical difference between unregulated and regulated site types for mean EC at 1- and 7-day time scales and the significantly lower EC in regulated sites during floods.

In support of H2, this study found 1- and 7-day ranges in DO were significantly lower in regulated sites for the entire study period and during F*l*. No significant difference in mean DO was found between site types at these time scales. Previous research has identified a reduction in mean DO (Walker, 1985; Palmer & O'Keeffe, 1990; Bunn & Arthington, 2002), but a reduction in range had not previously been identified. These differences may potentially be explained by a reduction in variability in water temperature downstream of the reservoir (e.g. Lehmkhul, 1972; Tuch & Gasith, 1989; Webb & Walling, 1988; Webb & Walling, 1993) as the ability of water to dissolve oxygen varies with temperature (Imtiyaz et al., 2012). Alternatively, there may be less photosynthetic activity (a driver of DO concentration (Wang & Veizer, 2000)) in regulated streams due to hypolimnetic water supply, resulting in a dampening in diurnal DO variation. Significantly higher DO at regulated sites during floods was identified; this is likely to be due to oxygenation of water during overspill events. The spillways are typically 'stepped' and during overspill events, water cascades down the channel (Figure 5.14) likely resulting in oxygenation.

The impact of regulation on pH has received little attention and no consensus on impact has been reached (e.g. Soja & Wiejaczka, 2013; Palmer & O'Keeffe, 1990). In agreement with H2, this study identified that diurnal pH range was significantly lower in regulated streams for the entire study and F*l* periods. This inference may be linked to a reduction in DO variability in that photosynthetic activity can also drive changes in pH (Morrison et al., 2001). Compensation water is typically drawn from the bottom/ middle levels of reservoirs C and D respectively, which is likely to be relatively devoid of photosynthetically active organisms (e.g. phytoplankton, algae), thereby leading to a reduction in photosynthesis (and thus diurnal pH variation) immediately downstream of each reservoir. This study also identified significantly higher pH during flood events at regulated sites. Mean pH during flood events at regulated sites was similar to 1- and 7- day mean values, but at the unregulated site, mean pH at 1- and 7-day time scales was 5.52/ 5.53 whereas it was 4.59 during floods. This relatively substantial difference is likely to be due to the direct link between unregulated streams and their acidic peat catchments which readily supply acidic water during rainfall events (Åström, 2001). In regulated streams, it is likely that reservoir water acts as a buffer to incoming acid rich streamwater during flood events; this may then lead to minimal change in pH in reservoir tailwaters as identified in this study.

The research undertaken in this study concentrated on only three sites in total and only one unregulated control site was used. This raises two key priorities for future research: firstly, more control sites should be used in future studies to increase confidence in results, and second, the longitudinal aspect of impacts associated with regulation should be assessed. Both sondes in regulated sites were located within approximately 50m of the reservoir dam wall. As distance from the reservoir increases, it is likely that physical-chemical impacts will be reduced (e.g. Webb & Walling, 1993). It is also important to note that the statistically significant differences observed in physical-chemical parameters between stream types were small (EC: <5 μS/cm; DO: <5 % saturation and pH: <1 units). Given the low number of sites compared and the accuracy of probes used in the study, further studies are required to assess whether, firstly, the differences observed are accurate, and secondly, whether, if they are accurate, they are ecologically significant.

Figure 5.14: Water cascading down spillway at reservoir C.

5.6.2 Impact of Artificial Floods

EC, DO and pH

Previous studies have observed complex, spatially and temporally variable impacts of AFs on EC. For example, Petts et al. (1985) observed an initial reduction (prior to peak discharge) followed by a rise in EC after peak discharge. This relationship was found to vary longitudinally from the reservoir and was thought to depend on in-channel sources. Indeed, Foulger & Petts (1984) also highlighted the potential importance of in-channel sources in determining EC response to AFs. Site 6 in this study was relatively close to the outlet of reservoir C and was such that in-channel sources were unlikely to be a major influence on EC during AFs. In disagreement with H3 (that the impact of AFs would be dependent on reservoir physical chemistry or hydrological characteristics of the AFs), this potentially explains the apparent simple dilution effect observed in the majority of AFs. Under normal compensation flow conditions, after water enters the stilling basin, EC is likely to increase due to factors such as exposure to mineral substrate (Ramchunder et al., 2011). It is hypothesised that during AFs the influence of such processes was reduced due to decreased water travel time between the reservoir outflow and site 6 resulting in lower EC measurements. Future work should look to examine this hypothesis or determine whether alternative biogeochemical processes (e.g. groundwater interaction; photosynthesis/ respiration) are important.

Evidence that DO was impacted by AFs was found for AFs 1 and 2 but not for AFs 3 or 4. During AFs 1 and 2 a clear reduction in DO was observed. A reduction or no change in DO during AFs is in contrast to the findings of most other studies (e.g. Shannon et al., 2001; Bednarek & Hart, 2005, Naliato et al., 2009) which have all noted increased DO. Naliato et al. (2009) also observed reductions in DO during AFs and linked these instances to reservoir stratification. Stratification of a reservoir can result in the development of a hypolimnetic layer of water of relatively low DO (Petts, 1984a). If water is drawn from this layer during an AF but from higher in the water column before, DO may decline. Water prior to and during each AF reported on in this study was drawn from the same valve (which was located at the bottom of the dam wall of reservoir C) and therefore a modification in valves during reservoir stratification cannot explain the observations noted herein.

Reduced DO during AFs 1 and 2 may have been due to suspension of organic matter from the stream substrate as hypothesised by Chung et al. (2008) who also observed reduced DO. Alternatively, contrary to as hypothesised (H3), AFs 1 and 2 may have disrupted photosynthesis in the stilling basin of reservoir C resulting in a reduction in DO (Odum, 1956). This may have occurred through increased turbidity which was observed visually during each AF resulting in less light entering the stream water column to drive photosynthesis. Background photosynthesis at the time of AFs 3 and 4 (autumn) appeared to be much reduced cf. AFs 1 and 2 (summer). A disruption in photosynthesis during the former period therefore has less potential to cause an impact and may explain why no evidence for an impact of AFs 3 and 4 on DO was found.

Figure 5.15: Artificial Flood in progress at reservoir C.

According to the published literature, the impact of AFs on pH had not previously been assessed and therefore the observation of reduced pH in each AF make this study important. Higher levels of dissolved carbon dioxide in water can result in lower pH due to formation of carbonic acid (Harrison et al., 2000). Thus the reduction in pH during AFs may potentially be explained by dissolution of atmospheric carbon dioxide during the turbulent release of water during each AF (Figure 5.15). Buffering of carbonic acid would be likely in streams of high ionic content, but is less likely in upland streams of low EC such as those in this study indicating that the decreases in pH observed in this study may not be observed in higher order regulated streams. Further research is required to assess this hypotheses and consider the potential role of, and interactions between, alternative factors such as groundwater influence and photosynthesis/ respiration which can affect stream pH (Glaser et al., 1990; Morrison et al., 2001).

5.6.3 General hydrological-physical chemistry response correlation

The final aim of the study was to consider any observed impacts of AFs on EC, DO and pH with respect to any impacts identified as a result of regulation *per se* in an attempt to assess the potential for the use of AFs as a mitigation technique. First, the general correlation between hydrological and physical-chemical indices in unregulated and regulated sites were considered. These correlations were then considered with respect to the general correlations identified during AFs. A similar approach to Poff & Zimmerman (2010) and Gillespie et al. (in press (b)) was undertaken whereby plots of hydrological parameters and physical-chemical response were created. This approach had the potential to reveal any broad differences in trends between periods of normal reservoir operation and AFs and thus an assessment to be made on whether AFs had the potential to be used as mitigation for any impacts identified as a result of regulation *per se*.

Pre-Artificial Flood floods

A significant difference was identified between unregulated and regulated site types for the relationships between all hydrological indices and mean DO and pH and pH range. These observations are in support of the identification that during floods, mean DO and pH were significantly different between site types. However, a significant difference between site types for pH range was not originally identified, indicating that analysis of covariance should be used as a complimentary analytical method for identifying potential impacts of regulation. Of these significant differences, mean DO was generally higher for regulated sites for all hydrological indices, apart from at high rates of change in the falling limb of a flood. Mean pH was typically higher for regulated sites at all levels of all hydrological indices. Conversely, pH range was generally lower for regulated sites for all hydrological indices apart from magnitude, where, at high flood magnitudes (> ~ 2000 % increase), this correlation reversed. However, this finding may be a result of relatively few datapoints for large floods. Again, these insights are complementary to the outcomes of alternative modelling techniques detailed above.

5.6.4 Potential for Artificial Floods as mitigation technique

Reservoir flow modification is accepted as a potential tool for the mitigation of impacts associated with stream regulation *per se* (Acreman et al., 2009; Acreman & Ferguson, 2010). Understanding of relationships between reservoir flow modification and ecosystem response are limited and typically, the definition of flow requirements of a system are based on distinct elements of a flow regime which are known to be important for the survival of particular biological elements within that system e.g. the Building Block Methodology (BBM) (Tharme & King, 1998). To improve such methods, detailed understanding of the relationships between flow modification and ecosystem response is required (Acreman et al., 2009). An assessment of the ability of flow modification to successfully mitigate any impact associated with regulation is also required to assess the feasibility of any proposed mitigation. It was hypothesised that AFs would demonstrate potential for use as mitigation (H3) and this is discussed here.

Comparison of impacts of regulation with impacts of AFs identified in this study revealed potential for mitigation of impacts by AFs (Table 5.9). However, the extent of the potential for mitigation of each impact varies; this is evident in comparison of the hydrological characteristics of pre-AF floods and AFs. The duration of each AF was relatively short in comparison with the mean duration of pre-AF floods and was particularly constrained by reservoir water availability. This constraint is likely to be highest during summer which was when an impact on flood duration was identified. It is thus unlikely that AFs could act as mitigation unless this constraint can be overcome. However, the range of rising and falling limb ROC of AFs was lower than ROCs identified as significantly higher at regulated sites during spring and F*l* for pre-AF floods indicating that AFs have the ability to more closely replicate unregulated flood ROC during periods such as these.

This study identified that during pre-AF floods at regulated sites, mean EC was significantly lower than floods at unregulated sites. Assessment of the correlation between mean EC and hydrological indices during AFs revealed a negative correlation between AF magnitude and ROC (rising and falling limbs) and mean EC indicating that the potential for use of AFs as mitigation for this impact may be enhanced for AFs of low magnitude and ROC.

A reduction in 1- and 7-day range in DO was identified at regulated sites. Whilst DO range during one AF was higher than the mean 1-day range in DO at regulated sites prior to AFs (Table 5.9), analysis of each AF revealed that DO during each flood was not outside that of the diurnal range in DO that could have been expected if the AF was not implemented. This indicates that the introduction of AFs in an attempt to raise DO diurnal range is unlikely to be successful. Higher mean DO during pre-AF floods at regulated sites was also identified by this study, but, as ranges of DO during AFs was within expected ranges for each day, the success of AFs used to mitigate this impact are also likely to be low.

This study observed that diurnal pH range was significantly lower for regulated cf. unregulated sites. pH range during AFs 2 and 4 was higher than mean diurnal pH range prior to AFs and interpretation of the response in pH during AFs revealed that during AFs 1, 2 and 4, minimum pH values were lower than would have been expected if the AF had not occurred. Higher mean pH during pre-AF floods at regulated sites was also identified and mean pH during AF 4 was reduced to below mean pre-AF flood pH indicating, in agreement with H3, potential for AFs to mitigate both low diurnal pH range and high mean pH during flood events. Assessment of the relationship between pH range and hydrological indices during AFs revealed a positive and a negative relationship between flood magnitude, ROC (falling and rising limbs) and pH range and mean respectively, indicating that the potential for mitigation may be increased at higher AF magnitudes and ROC.

It is important to stress that assessments of potential for AFs to act as mitigation measures are based on a very limited number of AFs. Additionally, analysis of hydrological indices and ecosystem response was undertaken to act as a general indication of relationships only. Confidence in relationships is dependent upon the number of datapoints used to generate relationships and thus, with further future studies, the effectiveness of this technique can be enhanced. Nevertheless, the approach used in this study can a useful tool to enable prioritisation of future research direction.

Table 5.9: Summary of significant impacts of regulation identified and associated impacts during AFs 3 and 4 (where comparable hydrological and physical-chemical indices could be calculated).

Impact of regulation	Associated AF impact	Potential for use of AFs as mitigation?
Hydrology		
Reduced flood frequency	Not tested	n/a
Reduced duration (summer mean: 0.17 days)	Durations of <0.1 days*	Not demonstrated
Increased ROC (rising) (means: 46.97, 52.83 and 43.06 ($m^3/s/day$) during F*l*, spring and summer respectively)	Range of 20.76-24.02 (AF3 and 4 respectively)*	Yes - high
Increased ROC (falling) (means: 2.65, 20.41, 25.71 and 16.87 (cm^3/s /day) during F*h*, F*l*, spring and summer respectively)	Range of 8.30 – 18.02 (AF3 and 4 respectively)*	Yes - moderate
EC		
Lower mean during floods (38.26 µS/ cm)	Mean EC reduced during AF 1 and 2, however, negative correlation between EC and AF magnitude and ROC (rising and falling limb)	Yes - low
DO		
Lower 1- and 7-day range (means: 4.94 and 7.28 (% saturation) respectively) for entire study period	Ranges of 1.5 – 6.8 (% saturation) during AFs 1-4*	Yes - low
Higher mean during floods (97.16 % saturation)	No impact	Not demonstrated
pH		
Lower 1-day range (mean: 0.27) for entire study period	Ranges of 0.06 – 0.41 during AFs 1-4*	Yes - moderate
Higher mean during floods (5.36)	Reduced during AFs (means: 5.58 and 4.78 during AF3 and 4 respectively)*	Yes - moderate

*calculated using the method described in section 5.3 to ensure comparability with pre-AF indices.

5.7 Summary

This chapter has identified significant impacts on the frequency, duration and rate of change (ROC) of floods in regulated streams in an upland catchment in the UK. Additionally, significant reductions in DO and pH range were associated with regulation. Furthermore, during floods, mean EC was significantly lower and mean DO and pH were significantly higher in regulated sites. Assessment of physical-chemical impacts was limited to only one site per stream and thus expansion of the monitoring network both to other streams and longitudinally on each stream will both enhance the certainty of impacts observed and provide useful information regarding the persistence of impacts downstream. Furthermore, impacts were generally small in magnitude and further research would be required to assess their ecological significance.

The hydrological and physical-chemical impacts of a series of Artificial Floods (AFs) were assessed. It was found that EC was significantly affected by AFs 1, 3 and 4, DO by AFs 1 and 2 and pH by all AFs. In all cases, reductions in each parameter were observed. AFs were found to have some potential for use as mitigation for the impacts to hydrology and physical chemistry, in particular, the ROC of AFs was more similar to floods at unregulated sites cf. regulated sites. Additionally, AFs were identified as potential mitigation for impacts to EC and pH.

Consideration of the general relationships between hydrological indices and physical-chemical responses both for pre-AF floods and AFs was undertaken. Assessment of these relationships for pre-AF floods revealed further insight into the effects of regulation and provided context for assessment of relationships during AFs which revealed that the effectiveness of mitigation may be enhanced under certain flow conditions. The assessment of relationships between hydrological indices and physical-chemical response is useful, but more research on the impact of AFs is required to increase confidence in the assessment and to enable informed management decisions regarding the implementation of mitigation measures which are central to achieving legislative targets under legislation such as the EU Water Framework Directive.

6 IMPACT OF REGULATION AND ARTIFICIAL FLOODS ON STREAM TEMPERATURE

6.1 Chapter overview

This chapter presents an assessment of the impact of regulation and Artificial Floods on stream temperature. First, the importance of understanding this topic is presented followed by an identification of current gaps in research and aims of the study. Next, the methods and analytical techniques used to undertake the study are detailed. This is followed by sections presenting and discussing the results, including recommendations for further research.

6.2 Introduction

Water temperature affects the metabolism of aquatic organisms both directly (Beschta et al., 1987) and indirectly (Macan, 1963) (e.g. by influencing other physical-chemical parameters such as dissolved oxygen concentration (e.g. Hamor and Garside, 1976)). In aquatic ecosystems, water temperature can also control species distribution (Dickson et al., 2012) and influence life-cycle phases such as insect emergence timing (e.g. Harper and Peckarsky, 2006), drift (Brittain and Eikeland, 1988) and mortality (e.g. Sweeney et al., 1986) and fish spawning (Van Der Kraak and Pankhurst, 1997) and egg hatch timing (Mann, 1996). Identification and understanding of any potential impacts on stream water temperature is therefore crucial given the importance for aquatic biota.

Anthropogenic activity can have a profound effect on stream temperature including deforestation (Brown & Krygier, 1970), afforestation (Brown et al., 2010), urbanisation (Krause et al., 2004) and waste water treatment (Kinouchi et al., 2007). It is therefore important to understand the impact of such activity to enable successful implementation of mitigation measures. In the northern hemisphere 77% of total stream discharge is either strongly or moderately affected by fragmentation resulting from reservoir operation, inter basin diversion and irrigation (Dynesius and Nilsson, 1994). An understanding of any impact that these activities have on downstream temperature is therefore vital because of the large proportion of affected streams. Furthermore, contemporary legislation such as the EU Water Framework Directive (EU WFD) (EC, 2000) requires that regulated streams meet an ecological standard (Good Ecological Potential (GEP)) based on abiotic and biotic aspects of stream ecology.

Globally, the general downstream impacts of regulation on temperature have been identified. A reducuction in annual and diel range of stream temperature (e.g. Lehmkhul, 1972; O' Keeffe et al., 1990 and Webb and Walling, 1993 respectively) is typically observed. However, more recent studies have identified complex spatio-temporal impacts (e.g. Webb & Walling, 1997) attributed to factors such as reservoir operation and groundwater influence. A recent resurgence in the assessment of the impact of anthropogenic activity on stream temperature has been driven by the development of relatively inexpensive, standalone data-loggers capable of recording at relatively high frequencies (e.g. minute intervals) (Webb et al., 2008). This has enabled spatially and temporally intensive assessment of stream temperature which previous work on regulated streams has been limited by (e.g. Lavis & Smith, 1971; Lehmkhul, 1972; Crisp, 1977; Cowx et al., 1987; Tuch & Gasith, 1989; O'Keeffe et al., 1990). For example, whole catchments can now be instrumented with relative ease allowing for assessment of the relative impact of regulation from multiple reservoirs. Such an approach has the potential to significantly improve understanding of catchment-scale thermal effects of regulation as it is yet to be undertaken anywhere globally.

Advanced understanding of the key environmental factors that control stream temperature (see Chapter 2) combined with the use of new analytical techniques (Webb et al., 2008) has led to increases in assessments of stream temperature dynamics using empirical models that take account of potential confounding factors and correct for autocorrelated errors (e.g. Gomi et al., 2006). To date, this approach has only been undertaken on one regulated stream (Dickson et al., 2012) and all assessments of the impact of regulation in temperate environments have been limited in the account of potential confounding factors (e.g. Lavis & Smith, 1971; Lehmkhul, 1972; Crisp, 1977; Cowx et al., 1987; Webb & Walling, 1988, 1993, 1996 and 1997).

Artificial floods (AFs) have been suggested as a potential tool to enable the achievement of GEP through mitigation of impacts associated with regulation (Acreman & Ferguson, 2010). It is therefore important that relationships between AFs and stream temperature are understood to allow for appraisal of whether AFs can be used successfully as mitigation. To date few studies have reported on impacts of AFs on stream temperature (n=9) and have typically reported increases (Cambray et al., 1997; King et al., 1998; Ashby et al., 1999; Lagarrigue et al., 2002; Chung et al., 2008; Naliato et al 2009). However, no change (King et al., 1998; Robinson et al., 2004a) and decreases (Foulger & Petts, 1984; Lagarrigue et al., 2002; Bruno et al., 2010) have also been reported resulting in a lack of consensus. This is likely due to the importance of local factors (Gillespie et al., in press (b)) which has driven the requirement for understanding at regional, or stream 'type' (i.e. of similar characteristics (e.g. geology)) scales (Poff & Zimmerman, 2010; Gillespie et al., in press (b)).

It is therefore crucial that the establishment of understanding in responses of stream temperature to AFs occurs at nested spatial scales to enable upscaling from site specific to more general regional predictions. The development of general relationships between hydrological parameters (e.g. flood magnitude) and ecosystem response variables has been used to identify quantitative relationships and potential threshold levels of hydrological parameters that invoke ecosystem response (e.g. Poff & Zimmerman, 2010; Gillespie et al., in press (b)). The assessment of these relationships with respect to any impacts of regulation *per se* has the potential to reveal whether AFs could successfully be used as mitigation. To date, this has not been conducted for any regulated stream globally.

This chapter reports on a detailed study of stream temperature in an upland UK catchment over one year. A spatio-temporal assessment of the impact of regulation on stream temperature using contemporary analytical techniques is made through comparison of several regulated streams with unregulated conditions in a multi-reservoir catchment. A series of AFs were also conducted allowing for a spatial assessment of the impact of AFs on stream temperature. Specifically, the aims of the study were to: (i) identify any impacts of regulation on stream temperature at seasonal and event based (i.e. flood) scales in a multi-reservoir catchment; (ii) undertake a longitudinal assessment of the impacts of AFs on stream temperature and (iii) consider the utility of AFs for mitigation of any impacts identified as a result of regulation *per se*. It was hypothesised that (H1) regulation would reduce downstream temperature range and impact mean stream temperature according to season, (H2) impact would vary spatially, (H3) Change in downstream temperature would be observed during AFs, thereby demonstrating the potential for use of AFs as mitigation.

6.3 Methods

6.3.1 Data loggers

Temperature

Stream temperature was recorded at 15 minute intervals using a combination of Gemini Tinytag Aquatic 2 data loggers, SEBA Hydrometrie MDS Dipper-3(T3) and YSI 6560 temperature probes (manufacturer stated accuracies ± 0.5 (between 0 and 50ºC), 0.1 and 0.15ºC respectively). During deployment, calibration of the latter 2 probes to Gemini Tinytag Aquatic 2 data loggers was undertaken at a range of temperatures (see Appendix B for calibration curves and regression coefficients) to ensure comparable temperatures. Air temperature was recorded at 15 minute intervals using Gemini Tinytag Plus 2 data loggers (manufacturer stated accuracy: ± 0.4 - 0.6ºC between -10 and 50ºC). Each sensor was shielded from direct sunlight and internal clocks synchronised prior to deployment. Stream temperature sensors were secured in a well mixed location within the stream and air temperature sensors were secured at c. 1.5m above ground level adjacent to the stream. Once deployed, data were downloaded at approximately monthly intervals and internal clocks reset to reduce drift.

Discharge and water level

Water level was recorded at 15 minute intervals according to the method described in Chapter 5. Where possible, water level was calibrated to discharge according to the same method.

6.3.2 Sites

Spatio-temporal dynamics

To assess whether there were differences between unregulated and regulated stream temperature regimes, data from unregulated sites 1, 3, 5 and 8 - 10 and regulated sites 2, 4, 6, 7 and 11-18 were selected for analysis. To aid analysis of stream temperature, air temperature was recorded at sites 1, 5, 10 and 16.

Flood dynamics

To assess whether differences in unregulated and regulated stream temperature regimes during floods could be identified, stream temperature and paired discharge data were selected from sites 6, 10 and 11 for analysis. To aid assessment of the impact of AFs, water level data were selected from sites 7 and 13.

6.3.3 Time scale

Impact of regulation

To assess the impact of regulation on stream temperature, data were selected for analysis from the 1st September 2012 to 19th August 2013 inclusive. These temperature data were combined with discharge data for the same period for assessment of stream temperature dynamics during floods. This period was prior to the introduction of AFs and can therefore be considered representative of typical regulated conditions.

Artificial Floods

AFs were carried out by YW on the 20th August, 19th September, 3rd October and 5th November 2013 (AFs 1-4 respectively) from reservoir C. The characteristics of each AF represented a balance between the practical restrictions placed on YW by the EA, local stakeholders, water resource availability and the capability of reservoir infrastructure to implement an AF. For these reasons, each AF differed in characteristics. However, during all AFs, water was drawn from the lowest vertical valve within the water column.

To assess the impact of AFs on stream temperature, temperature data were collected at 15 minute intervals on each AF day from sites 6, 7, 11, 12, 13 and 16. Sites 6, 7 and 13 were selected as impact sites as they were downstream of reservoir C. Sites 11, 12 and 16 were chosen as respective control sites.

6.4 Data analysis

6.4.1 Quality control procedure

Temperature

Data quality was first evaluated using a similar approach to Jones & Graziano (2013) whereby removal of data occurred if the following standards (outside of which data were assumed to be incorrect) were met: stream temperature: < -1 or > 35 ºC; air temperature: < -10 or > 35ºC. In addition, data were visually assessed for drift and probe failure/ malfunction and removed if detected; during this process a conservative approach was taken so that if there was any doubt in the quality of data, they were removed (see Appendix C for details of all removed data). Some data were lost due to telemetry failure and a detailed record of these can also be found in Appendix C. To ensure comparability of datasets between each site, data were removed from all datasets for the same time period where data had been removed from one site.

Discharge and water level

The quality control procedure for discharge and water level data was as described in Chapter 5.

6.4.2 Impact of regulation

To assess whether differences in stream temperature regimes between unregulated and regulated sites could be identified, first, stream temperature data at each site were paired with air temperature data from the closest site. Next, diurnal *mean* and *range* indices were extracted from each stream (T*m* and T*r* respectively) and air temperature dataset for each site. Mean and standard deviation of each stream temperature index for each site was then calculated. Predictive models (multiple linear regression and multiple logistic regression (for T*m* and T*r* respectively)) using indices only from unregulated sites were then produced using a similar method to Gomi et al. (2006) and Dickson et al. (2012). Explanatory variables were initially included in each model based on *a priori* understanding of the key drivers of stream temperature (see Chapter 2 for discussion) and followed approaches similar to Dickson et al. (2012) and Moore et al. (2013). Final models were selected using AIC and took the form:

(i) $$Tm = \alpha + \beta\, y_a + \beta\, alt + \beta\, j + \beta \sin(2\pi\, j/d) + \beta \cos(2\pi\, j/d) + \varepsilon$$

(ii) $$Tr = \alpha + \beta\, y_a + \beta\, y_b + \beta\, y_a : y_b + \beta\, alt + \beta\, j + \beta \sin(2\pi\, j/d) + \beta \cos(2\pi\, j/d) + \varepsilon$$

where y_a = mean air temperature and y_b = air temperature range on day j , alt = altitude of each site, d = 365.25 (mean number of days in a year), α = intercept, β = coefficients estimated by regression and ε = error term. Sine and cosine variables were included to account for residual seasonality (e.g. Gomi et al., 2006) and ":" represents an interaction term between two variables. Site 3 was not included in the predictive model of T*r* as inclusion of this site resulted in a poor model fit; this prevented the statistical testing of regulated sites 2 and 4 as site 3 was a control for these sites.

Models were then used to predict stream temperature for each regulated site. Random disturbances (RD) were calculated according to Gomi et al. (2006). This method removed autocorrelation from the residuals resulting in independence between RD:

$$RD = (y_t - \hat{y}_t) - \acute{p}_1 (y_{\text{t-1}} - \hat{y}_{\text{t-1}}) \ldots - \acute{p}_k (y_{\text{t-k}} - \hat{y}_{\text{t-k}})$$

where y_t and $\hat{y}_t$ are the observed and predicted temperature indices on day t respectively and $\acute{p}_i$ is an estimate of the lag i autocorrelation coefficient from the model fit (obtained through calculation of partial autocorrelation factor). 95% prediction limits can then be estimated as $\pm 1.96\sigma\hat{u}_t$ (Gomi et al., 2006). If regulation had no effect on stream temperature, the distribution of unregulated and regulated RD should be equal. Therefore, two-sample Kolmogorov-Smirnov tests were used to statistically test for differences in distribution of RD between all unregulated sites combined and each regulated site separately.

To assess whether any impact of regulation on stream temperature indices varied temporally, RD were split into the following periods and tested separately using 2-sample Kolmogorov-Smirnov tests as described above: autumn (1st September 2012 00:00:00 to 30th November 2012

23:59:59), winter (1st December 2012 00:00:00 to 28th February 2013 23:59:59), spring (1st March 2013 00:00:00 to 31st May 2013 23:59:59) and summer (1st June 2013 00:00:00 to 19th August 2013 23:59:59).

To assess whether differences in stream temperature regimes occurred between unregulated and regulated sites during floods (i.e. natural floods and reservoir overspill/ operational release events respectively), first, stream temperature and discharge data were paired to ensure comparability. Next, floods were defined using the method described in Chapter 5. Mean (F*m*) and range (F*r*) indices were then calculated for stream temperature during each flood. Indices were modelled as dependent variables using GAMM as non-linear temporal trends in indices were apparent. Explanatory variables were: *site* (random effect due to repeated measures (Zuur et al., 2009)), *type* (unregulated or regulated) (fixed effect) and *time* of peak discharge of each flood (fixed effect). Sites were nested within site type and appropriate error distributions, link functions and correlation structures were specified to ensure approximate normal distribution, homogeneity and independence of residuals. Finally, significance of site *type* was assessed using t-statistics and associated p-values.

To assess whether associations between hydrological (magnitude, duration and rate of change (rising and falling limb)) and stream temperature indices (F*m* and F*r*) were different between unregulated and regulated site types, ANCOVA was performed using Generalized Linear Modelling (GLM) with stream temperature indices as response variables and hydrological indices and site *type* as fixed effects (Rutherford, 2012). Appropriate error distributions, link functions and correlation structures were specified to ensure approximate normal distribution, homogeneity and independence of residuals. Finally significance of site *type* was assessed using t-statistics and associated p-values.

6.4.3 Impact of Artificial Floods

To assess the impact of each AF on stream temperature, a modified paired Before-After-Control-Impact (BACIP) (Stewart-Oaten et al., 1986; Smith, 2002) analysis was undertaken which replaced 'After' with 'During' (e.g. Dinger & Marks, 2007) to allow the during-AF impact to be statistically assessed through interrogation of a modelled interaction term (Smith, 2002). For each AF, response variables (15 minute stream water temperature) were modelled as a function of *site* (2 levels: site 6 (impact); site 11 (control)) and *period* (2 levels: before; during) and the interaction between these two factors (*site:period*). *Period: before* was defined as all data from 00:00:00 until the start of each AF and *period: during* was defined by the start and end times of each AF. As data were collected through time at each site, *site* was treated as a random effect (Zuur et al., 2009). Either GLMM or GAMM (Wood, 2011) (depending on whether linear (GLMM) or non-linear (GAMM) temporal variation was evident in data (assessed through assessment of significance of temporal smoother terms)) models were used. Appropriate error distributions, link functions and correlation structures were specified to ensure approximate normal distribution, homogeneity and independence of residuals. Finally, significance of *site:period* terms were assessed using t-statistics and associated p-values.

Statistical tests were deemed significant at $p < 0.05$ and all plot creation and statistical analyses were undertaken using R v 2.15.3 (2013).

6.5 Results

6.5.1 Impact of regulation

For the period of assessment, sites 7-10 and 14 were typically coolest (mean stream temperature < 7.16ºC), and sites 2, 11, 12, 16 and 18 were typically warmest (mean stream temperature > 7.68ºC). Stream temperature range was smallest at site 14 and largest at sites 1, 5 and 10 and the coolest and warmest temperatures were recorded at sites 2 and 10 respectively (Table 6.1; Figure 6.1).

6.5.2 Diurnal mean stream temperature

For the entire assessment period, T_m appeared to generally be higher in fully-regulated sites cf. unregulated sites (Figure 6.2a), but mean T_m was only higher for three fully-regulated cf. unregulated sites (Table 6.1). A clear difference between T_m for semi-regulated sites and unregulated sites was not visually apparent (Figure 6.2b), but mean T_m was higher for all semi-regulated cf. unregulated sites (Table 6.1). All explanatory factors included in the predictive model of T_m were significant ($p < 0.01$) and model r-squared was 0.96, RMSE was 1.00 and MAE was 0.78 indicating adequate predictive power (Table 6.2).

T_m RD for unregulated sites displayed approximately constant temporal variance throughout the assessment period. Regulated sites generally displayed more temporal variance than unregulated sites, particularly at sites 6 and 11 where many RD were outside the 95% confidence intervals between September 2012 and March 2013 which was not observed at other sites (Figure 6.3).

Table 6.1: Summary statistics for 15 minute and diurnal mean and range data for each site for the study period (prior to Artificial Floods (AFs)). U = unregulated, FR = fully-regulated (no tributary influence 1:50K OS map (OS, 2012)).

		15 – minute data					**Diurnal mean (*Tm*)**		**Diurnal range (*Tr*)**	
	Site	**Mean (°C)**	**Minimum (°C)**	**Maximum (°C)**	**St. dev. (°C)**	**Range (°C)**	**Mean (°C)**	**St. dev. (°C)**	**Mean (°C)**	**St. dev. (°C)**
U	**1**	7.16	-0.12	24.42	5.24	24.54	7.34	4.95	3.31	2.53
	2	7.77	0.10	18.63	4.38	18.53	7.97	4.27	1.53	0.87
U	**3**	7.36	-0.07	17.04	4.22	17.11	7.55	4.11	1.45	0.77
	4	7.55	0.20	17.04	4.12	16.84	7.75	4.04	1.29	0.68
U	**5**	7.28	-0.30	23.12	5.31	23.42	7.46	5.06	2.96	2.12
FR	**6**	7.25	0.41	17.65	4.82	17.24	7.55	4.80	0.89	0.68
FR	**7**	7.09	-0.16	18.64	4.82	18.80	7.38	4.72	2.07	1.73
U	**8**	6.96	-0.11	24.13	5.20	24.24	7.14	4.86	3.67	2.78
U	**9**	6.80	-0.11	24.54	5.58	24.65	6.99	5.29	3.28	2.60
U	**10**	6.64	-0.11	24.77	5.70	24.88	6.83	5.39	3.42	2.91
FR	**11**	7.70	0.17	19.32	4.93	19.16	7.95	4.85	1.04	0.77
FR	**12**	7.76	0.27	21.44	4.80	21.17	7.99	4.63	2.24	1.79
	13	7.50	-0.23	22.29	5.01	22.52	7.74	4.75	3.18	2.63
FR	**14**	7.16	0.26	15.86	4.39	15.60	7.46	4.38	1.15	0.74
	15	7.47	0.06	20.78	4.82	20.72	7.73	4.65	2.55	2.07
FR	**16**	7.72	0.62	19.92	4.61	19.30	7.94	4.49	1.54	1.21
	17	7.66	-0.06	21.57	4.84	21.63	7.90	4.62	2.88	2.34
	18	7.69	-0.90	21.69	4.93	22.59	7.93	4.71	2.78	2.18

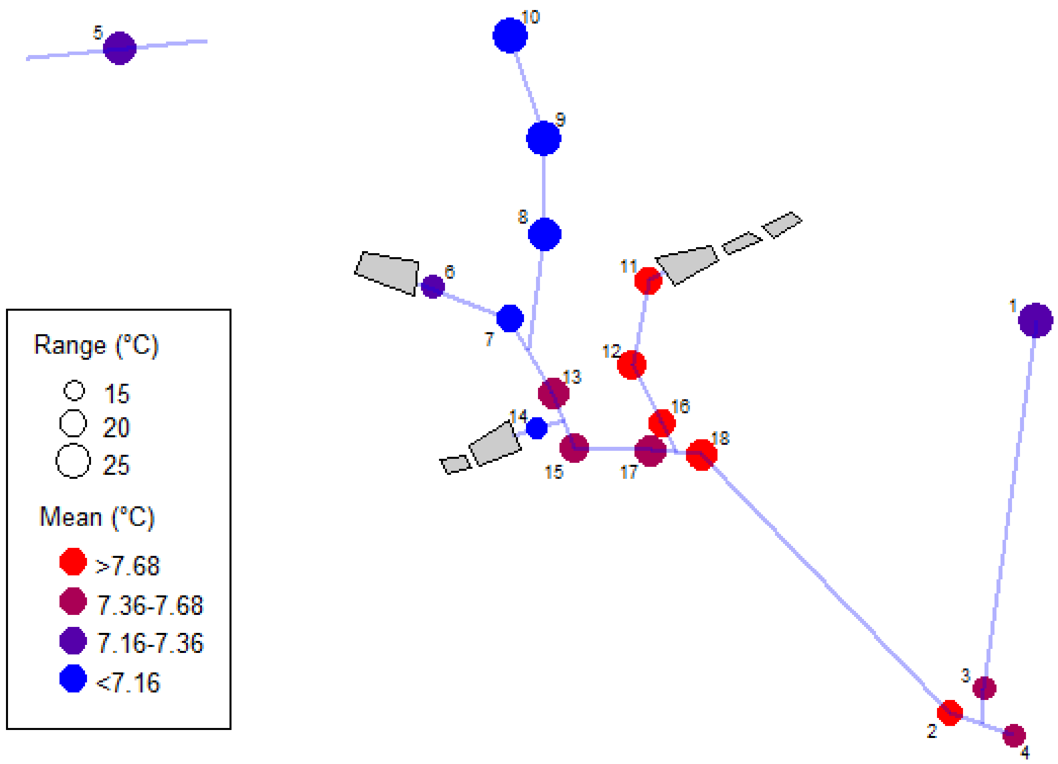

Figure 6.1: Stream temperature range and mean for the entire study period from 15 minute data at sites 1-18. Note: Map not to scale. Reservoirs are shown in grey and site numbers are displayed next to each point. Site 5 is in a different catchment to all other sites.

Two-sample Kolmogorov–Smirnov tests indicated significant differences between RD for all unregulated sites combined and RD for regulated sites 6, 11, 12, 14 and 16 for the entire study period (Table 6.3). Seasonal analysis identified significant differences in RD between all unregulated sites combined and regulated sites 6 and 14 during autumn, 6, 11, 12, 14 and 16 during winter, no sites during spring and sites 6 and 18 during summer (Table 6.3). Examination of season and regulated site combinations identified as significantly different from unregulated sites revealed that observed T*m* was approximately 2°C higher than predicted at sites 6 and 14 during the first half of autumn, but similar during the latter half. During winter, observed T*m* at sites 6, 11, 12, 14 and 16 was broadly similar to predicted, but during mid to late January 2013, observed T*m* was approximately 1-2°C warmer than predicted. At site 18 during summer, observed T*m* was generally warmer than predicted, but differences were typically <1°C. At site 6 during early June 2013, observed T*m* was up to ~3.5°C warmer than predicted for c. five days, but for c. 15 days between July and August 2013, observed T*m* was up to ~3.5°C cooler than predicted (Figure 6.4).

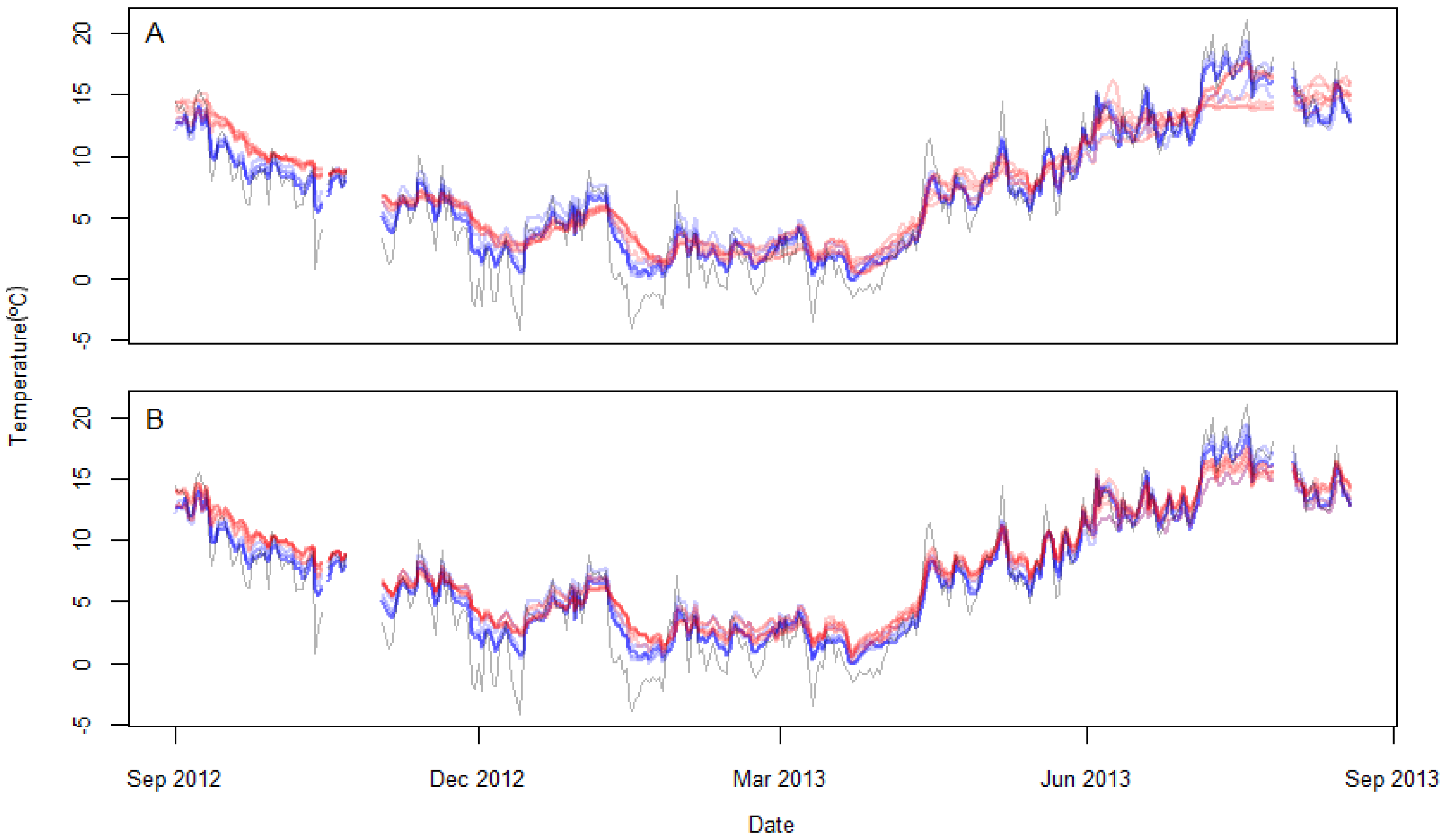

Figure 6.2: Mean diurnal stream temperature (Tm) for fully-regulated (no tributary influence 1:50K OS map (OS, 2012)) and semi-regulated sites (panels A and B respectively) (red lines) and unregulated sites (blue lines). Mean mean diurnal air temperature for all sites is shown in black..

Table 6.2: Summary statistics for explanatory variables included in models of mean and range diurnal stream temperature using unregulated site data for prediction of regulated site indices and 'goodness of fit' statistics: Root mean squared error (RMSE) and maximum absolute error (MAE). Significant factors are displayed in bold. Note: Adjusted R-squared value not available for logistic regression.

Mean		
	Estimate	**t**
Intercept	-211.7000	-12.88 **
Mean diurnal air temperature (y_a)	0.5696	72.36 **
Altitude	-0.0014	-4.78 **
Day of the year (j)	0.0052	13.11 **
sin(2πj/d)	-1.3910	-22.39 **
cos(2πj/d)	-1.6360	-30.99 **
Adjusted R-squared	0.96	
RMSE	1.00	
MAE	0.78	
Range		
Intercept	23.0260	5.30 **
Mean diurnal air temperature (y_a)	0.0896	19.25 **
Diurnal air temperature range (y_b)	0.0892	33.73 **
Mean diurnal air temperature : diurnal air temperature range interaction (y_a:y_b)	-0.0029	-14.62 **
Altitude	0.0003	2.52 *
Day of the year (j)	-0.0006	-5.40 **
sin(2πj/d)	0.3105	20.95 **
cos(2πj/d)	-0.2427	-9.30 **
Adjusted R-squared	n/a	
RMSE	0.38	
MAE	0.29	

**p < 0.05, **p < 0.01.*

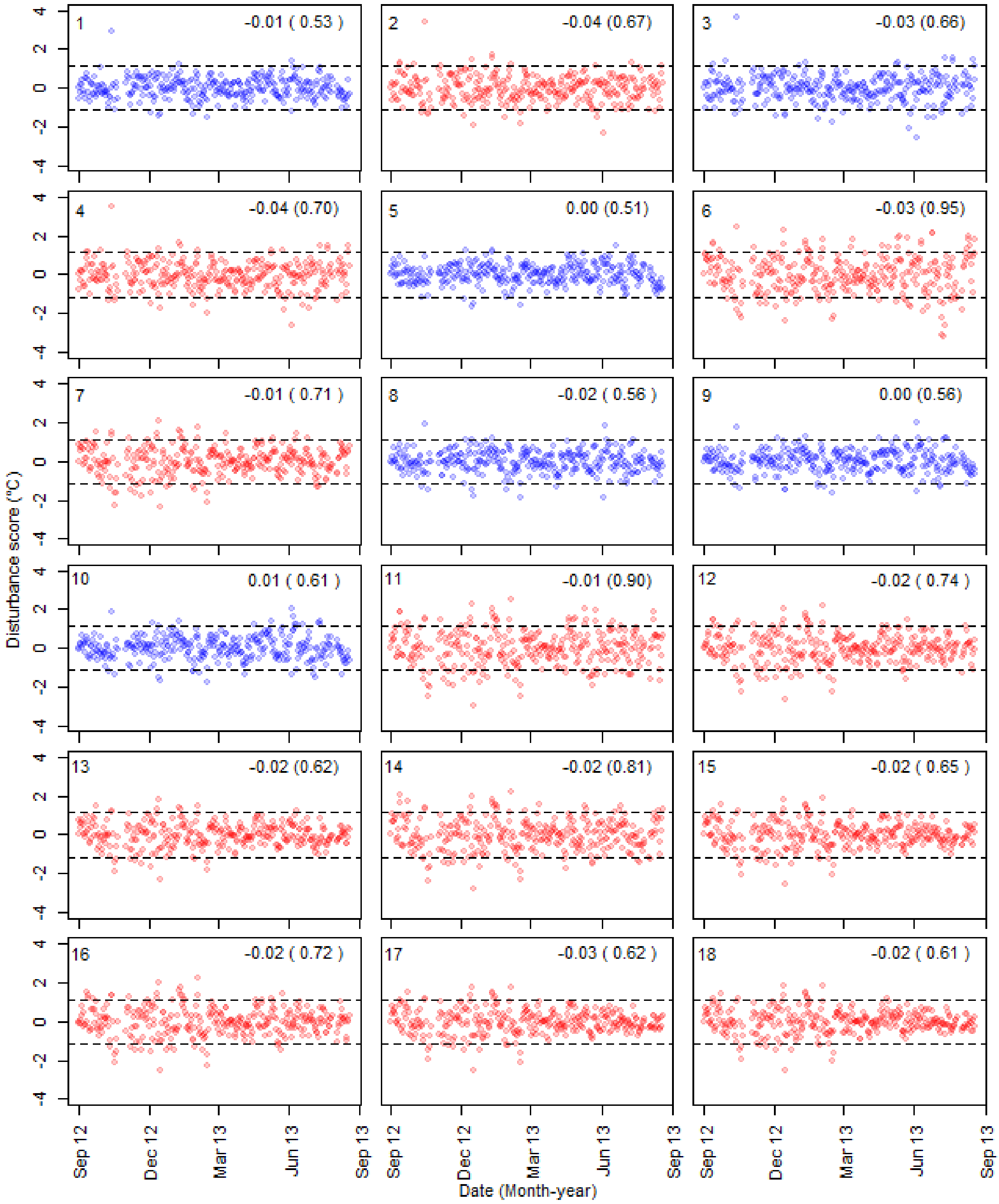

Figure 6.3: Diurnal mean stream temperature (Tm) random disturbances (RD) for sites 1-18 respectively. Red and blue points are used to distinguish between regulated and unregulated sites respectively. Dashed lines represent 95% confidence intervals (based on all unregulated sites combined). Mean (St. dev.) RD is shown in the top right of each plot.

Table 6.3: Mean diurnal (& St.dev.) stream temperature mean (Tm) (°C) and 2-sample Kolmogorov–Smirnov test statistic (D) for unregulated random disturbances (RD) vs RD for each regulated site.

	All data		**Autumn**		**Winter**		**Spring**		**Summer**	
Site	**Mean (St.dev.)**	**D**	**Mean (St.dev.)**	**D**	**Mean (St.dev.)**	**D**	**Mean (St.dev.)**	**D**	**Mean (St.dev.)**	**D**
2	7.97 (4.27)	0.07	9.03 (2.36)	0.13	3.93 (1.34)	0.14	6.19 (3.08)	0.08	13.87 (1.69)	0.07
4	7.75 (4.04)	0.07	8.95 (2.33)	0.09	3.97 (1.42)	0.11	6.04 (3.03)	0.10	13.10 (1.65)	0.10
6	7.55 (4.80)	0.13 **	9.41 (3.03)	0.18 *	2.99 (1.20)	0.16 *	5.15 (3.27)	0.10	13.96 (1.39)	0.20 *
7	7.38 (4.72)	0.06	9.18 (2.93)	0.16	2.83 (1.27)	0.13	5.11 (3.21)	0.08	13.72 (1.27)	0.10
11	7.95 (4.85)	0.11 **	9.34 (2.81)	0.16	3.25 (1.20)	0.16 *	5.86 (3.33)	0.12	14.66 (1.71)	0.13
12	7.99 (4.63)	0.08 *	9.24 (2.60)	0.13	3.50 (1.17)	0.16 *	6.00 (3.11)	0.10	14.48 (1.70)	0.12
13	7.74 (4.75)	0.04	9.13 (2.77)	0.13	3.18 (1.34)	0.10	5.64 (3.29)	0.09	14.33 (1.41)	0.10
14	7.46 (4.38)	0.11 **	9.39 (3.07)	0.16 *	3.14 (1.20)	0.16 *	5.35 (2.64)	0.09	13.18 (1.25)	0.12
15	7.73 (4.65)	0.05	9.25 (2.87)	0.15	3.23 (1.26)	0.13	5.66 (3.18)	0.07	14.07 (1.34)	0.14
16	7.94 (4.49)	0.11 **	9.19 (2.47)	0.12	3.62 (1.15)	0.17 *	5.94 (2.98)	0.09	14.24 (1.70)	0.12
17	7.90 (4.62)	0.05	9.21 (2.76)	0.14	3.40 (1.92)	0.12	5.96 (3.18)	0.07	14.26 (1.52)	0.15
18	7.93 (4.71)	0.06	9.22 (2.73)	0.12	3.42 (1.26)	0.12	5.89 (3.28)	0.10	14.47 (1.61)	0.18 *

*$p < 0.05$, **$p < 0.01$.

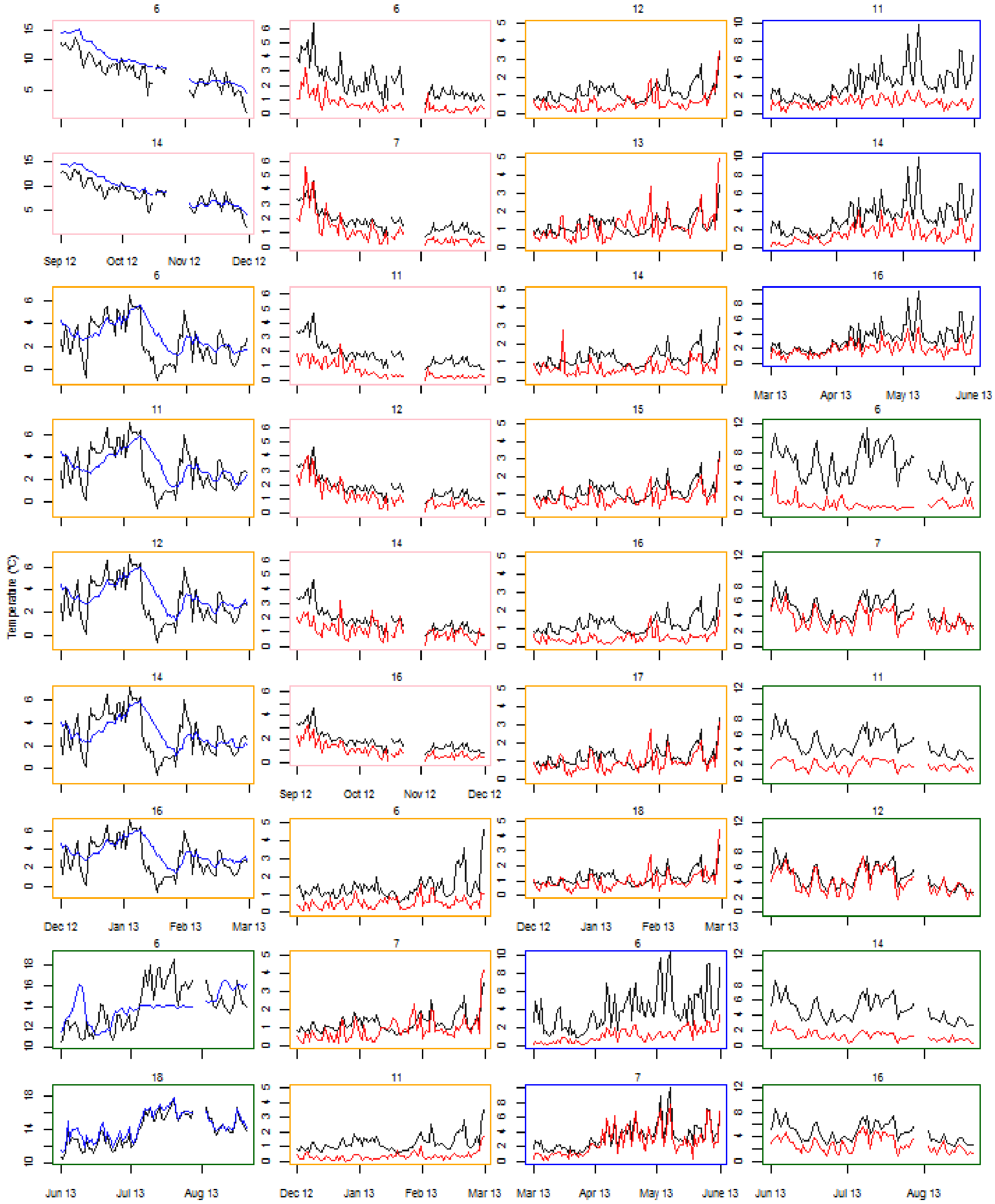

Figure 6.4: Observed diurnal stream temperature mean (Tm) (blue lines) and range (Tr) (red lines) with predicted values (black lines) for seasons where Kolmogorov–Smirnov tests indicated significant differences in random disturbances. Site numbers are displayed above each plot and season is delineated by surrounding box colour: pink – autumn, orange – winter, blue – spring & green – summer.

Diurnal stream temperature range (Tr)

For the entire assessment period, T*r* was generally lower in fully-regulated sites cf. unregulated sites (Figure 6.5a); these differences appeared to be smaller for semi-regulated sites (Figure 6.5b). Mean T*r* was lower at for three fully-regulated sites cf. all unregulated sites (Table 6.1); these three sites were directly downstream of reservoirs (Figure 6.1). All explanatory factors included in the final predictive model were significant at either $p < 0.01$ or 0.05. RMSE of the model was 0.38 and MAE was 0.29 indicating adequate predictive power (Table 6.2).

RD for all unregulated sites displayed similar temporal variance throughout the study period, whereas regulated sites displayed different patterns. In particular, all regulated sites displayed relatively little variance during winter and site 6 displayed relatively high RD between June and September 2013 (Figure 6.6).

Two-sample Kolmogorov–Smirnov tests indicated significant differences in RD between all unregulated sites combined and all tested regulated sites apart from sites 13 and 17 for the entire dataset. During autumn and summer, RD at regulated sites 6, 7, 11, 12, 14 and 16 were significantly different from those for all unregulated sites combined and during winter, RD for all tested sites were significantly different from those for all unregulated sites combined. During spring, RD at sites 6, 7, 11, 14 and 16 was significantly different from all unregulated sites combined (Table 6.4).

Of note, observed T*r* was approximately 1-2°C lower throughout the majority of autumn at sites 6, 11 and 14 and was generally lower than predicted at all of the aforementioned sites during winter, but these differences were typically less than 1°C. During spring, observed T*r* was generally lower than predicted at all of these sites where significant differences in RD were identified. However, differences at site 7 and 16 were small (<0.5°C and <1°C respectively). Notably, during May 2013, observed T*r* was up to approximately 5°C lower than observed at sites 6 and 11. Observed T*r* during summer was generally lower than predicted at all aforementioned sites, but differences at sites 7, 12 and 16 were typically <1°C. Notably, during July, observed T*r* was up to approximately 7°C (site 6) and 4°C (sites 11 and 14) lower than predicted (Figure 6.4).

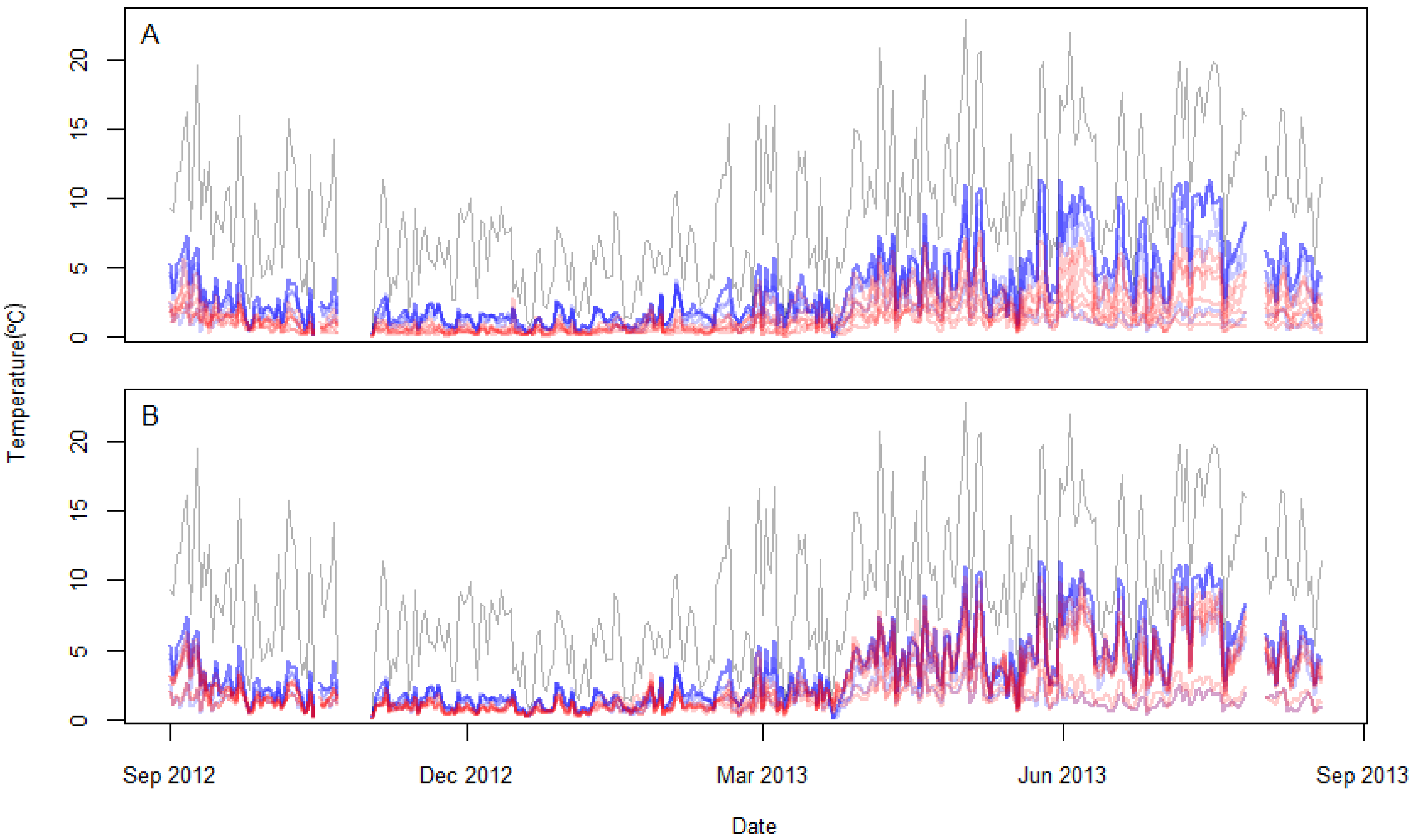

Figure 6.5: Diurnal stream temperature range (Tr) for fully-regulated (no tributary influence 1:50K OS map (OS, 2012)) and semi-regulated sites (panels A and B respectively) (red lines) and unregulated sites (blue lines). Mean diurnal air temperature range for all sites is shown in black.

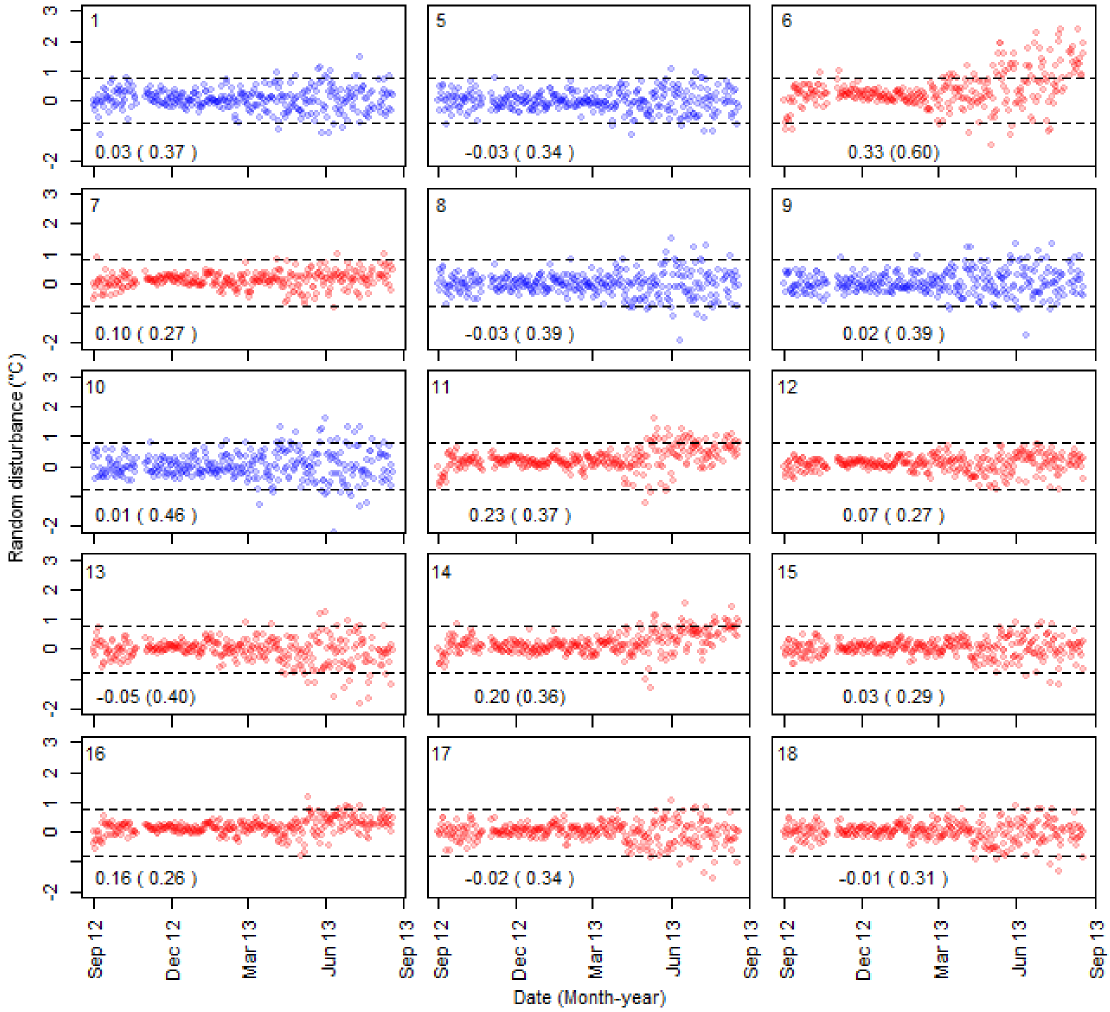

Figure 6.6: Diurnal stream temperature range (Tr) random disturbances (RD) for sites 1 and 5-18. Red and blue points are used to distinguish between regulated and unregulated sites respectively. Dashed lines represent 95% confidence intervals (based on all unregulated sites combined). Mean (St. dev.) RD is shown at the bottom of each plot.

Table 6.4: Mean (& St. dev.) diurnal stream temperature range (Tr) (°C) and 2-sample Kolmogorov–Smirnov test statistic (D) (where applicable) for unregulated random disturbances (RD) vs RD for each regulated site.

	All data		**Autumn**		**Winter**		**Spring**		**Summer**	
Site	**Mean (St.dev.)**	**D**	**Mean (St.dev.)**	**D**	**Mean (St.dev.)**	**D**	**Mean (St.dev.)**	**D**	**Mean (St.dev.)**	**D**
2	1.53 (0.87)	n/a	1.22 (0.54)	n/a	0.91 (0.50)	n/a	2.11 (0.97)	n/a	1.90 (0.74)	n/a
4	1.29 (0.68)	n/a	1.21 (0.53)	n/a	0.86 (0.48)	n/a	1.74 (0.81)	n/a	1.32 (0.51)	n/a
6	0.89 (0.68)	0.32 **	0.71 (0.58)	0.37 **	0.54 (0.30)	0.34 **	1.10 (0.73)	0.30 **	1.23 (0.78)	0.49 **
7	2.07 (1.73)	0.23 **	1.16 (1.09)	0.19 *	0.89 (0.65)	0.29 **	2.71 (1.83)	0.23 **	3.69 (1.36)	0.27 **
11	1.04 (0.77)	0.36 **	0.68 (0.56)	0.31 **	0.36 (0.26)	0.37 **	1.34 (0.54)	0.35 **	1.88 (0.62)	0.51 **
12	2.24 (1.79)	0.21 **	1.27 (0.89)	0.23 **	0.65 (0.53)	0.32 **	3.08 (1.39)	0.13	4.14 (1.51)	0.27 **
13	3.18 (2.63)	0.06	1.67 (1.34)	0.12	1.18 (0.81)	0.18 *	4.16 (2.38)	0.09	5.99 (2.31)	0.16
14	1.15 (0.74)	0.27 **	1.13 (0.60)	0.20 **	0.70 (0.41)	0.22 **	1.48 (0.96)	0.26 **	1.28 (0.59)	0.53 **
15	2.55 (2.07)	0.14 **	1.52 (1.08)	0.14	0.89 (0.52)	0.23 **	3.22 (1.83)	0.15	4.82 (1.85)	0.15
16	1.54 (1.21)	0.32 **	1.03 (0.68)	0.26 **	0.48 (0.32)	0.32 **	1.99 (0.94)	0.30 **	2.81 (1.22)	0.44 **
17	2.88 (2.34)	0.08	1.69 (1.21)	0.12	0.97 (0.54)	0.18 *	3.70 (2.03)	0.12	5.45 (2.08)	0.13
18	2.78 (2.18)	0.09 *	1.64 (1.13)	0.16	0.97 (0.63)	0.20 **	3.72 (1.95)	0.11	5.00 (1.90)	0.12

**p < 0.05, **p < 0.01.*

Stream temperature mean (Fm) and range (Fr) during flood events

During pre-AF floods, F*m* was significantly warmer (estimate: 1.57°C), and F*r* significantly lower (estimate: - 0.47°C) for regulated cf. unregulated site types (Table 6.5). Statistical assessment of the difference in relationships between hydrological and stream temperature indices for unregulated and regulated stream types revealed significant differences between site types for F*m* and all hydrological indices, but for F*r*, the relationships were only significantly different between site types for flood duration (Table 6.6). Plots of the relationships revealed weak positive associations between F*m* and flood magnitude and ROC (rising and falling limb) for both unregulated and regulated site types, but conversely, a weak negative association between F*m* and flood duration was observed (Figure 6.7). A weak negative association between all hydrological indices and F*r* for the unregulated site type was observed; this association was similar for ROC indices for regulated site types, but conversely, weak positive associations between flood magnitude and duration and F*r* for regulated site types were observed (Figure 6.7).

Table 6.5: Mean (St. dev.) stream temperature indices for unregulated and regulated site types and test statistics during floods.

	Mean (F*m*) (°C)	**Range (F*r*) (°C)**
Unregulated	5.59 (3.81)	1.58 (1.52)
Regulated	8.41 (4.09)	1.06 (1.03)
Est.	1.57	-0.47
t	3.81 **	-2.79 **

$*p < 0.05$, $**p < 0.01$.

Table 6.6: ANCOVA test statistics for relationship between flood and stream temperature indices between unregulated and regulated stream types.

	Magnitude	**Duration**	**ROC rising**	**ROC falling**
Mean (F*m*)	2.21 *	4.55 **	-2.11 *	-2.64 **
Range (F*r*)	1.91	2.51 *	0.78	0.88

$*p < 0.05$, $**p < 0.01$.

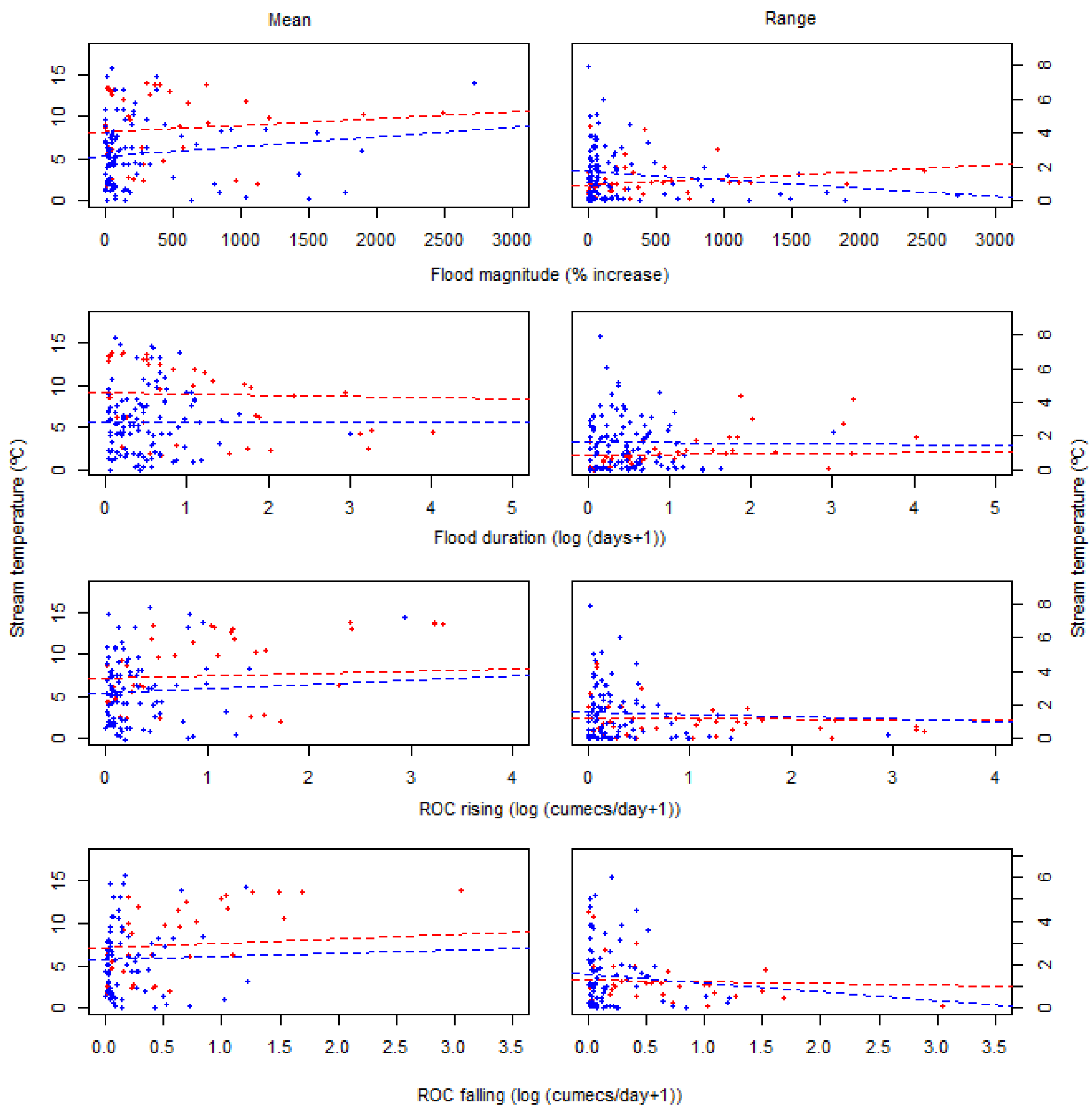

Figure 6.7: Association between flood and stream temperature indices for unregulated (blue) and regulated (red) site types.

6.5.3 Impact of Artificial Floods

Mean stream temperature before and during each AF was highest for AF1 and lowest for AF4 for all sites (Table 6.7). A general diurnal trend in stream temperature was observed on each AF day at all sites, although this trend appeared to be suppressed at sites 6 and 11 (immediately downstream of reservoirs C and D respectively) (Figure 6.8). Visually, there was no clear indication of any impact of any AF on stream temperature and, in support of this observation, statistical testing failed to reveal any significant impact of any AF (Figure 6.8; Table 6.7). Due to the absence of significant impacts of AFs on stream temperature, further analysis of the relationship between hydrological indices and stream temperature response during AFs was not undertaken.

Table 6.7: Mean (& St.dev.) and test statistics for stream temperature indices (°C) for impact and control sites before and during Artificial Floods (AFs) 1-4. All models were undertaken using GAMM.

	AF 1		**AF 2**		**AF 3**		**AF 4**	
	Before	**During**	**Before**	**During**	**Before**	**During**	**Before**	**During**
Sites 6 & 11								
Impact	15.86 (0.04)	16.13 (0.05)	10.41 (0.07)	10.95 (0.18)	11.80 (0.02)	11.85 (0.02)	7.36 (0.02)	7.47 (0.05)
Control	15.34 (0.09)	15.77 (0.10)	10.35 (0.03)	10.54 (0.07)	12.11 (0.02)	12.29 (0.07)	7.26 (0.01)	7.33 (0.02)
Test statistic	-0.29		1.57		1.4		1.62	
Sites 7 & 12								
Impact	14.34 (0.35)	16.03 (0.38)	9.91 (0.05)	10.74 (0.24)	11.80 (0.03)	11.99 (0.05)	7.13 (0.04)	7.36 (0.05)
Control	14.23 (0.50)	15.95 (0.54)	10.00 (0.07)	10.50 (0.17)	12.16 (0.05)	12.58 (0.09)	7.28 (0.08)	7.63 (0.04)
Test statistic	1.22		1.96		-1.63		0.69	
Sites 13 & 16								
Impact	13.22 (0.49)	15.93 (0.91)	9.70 (0.05)	10.46 (0.38)	11.83 (0.06)	12.23 (0.10)	6.97 (0.15)	7.52 (0.07)
Control	13.76 (0.13)	14.56 (0.30)	10.06 (0.04)	10.12 (0.07)	12.06 (0.01)	12.23 (0.09)	7.37 (0.06)	7.61 (0.01)
Test statistic	1.41		1.37		0.91		1.38	

**p < 0.05, **p < 0.01.*

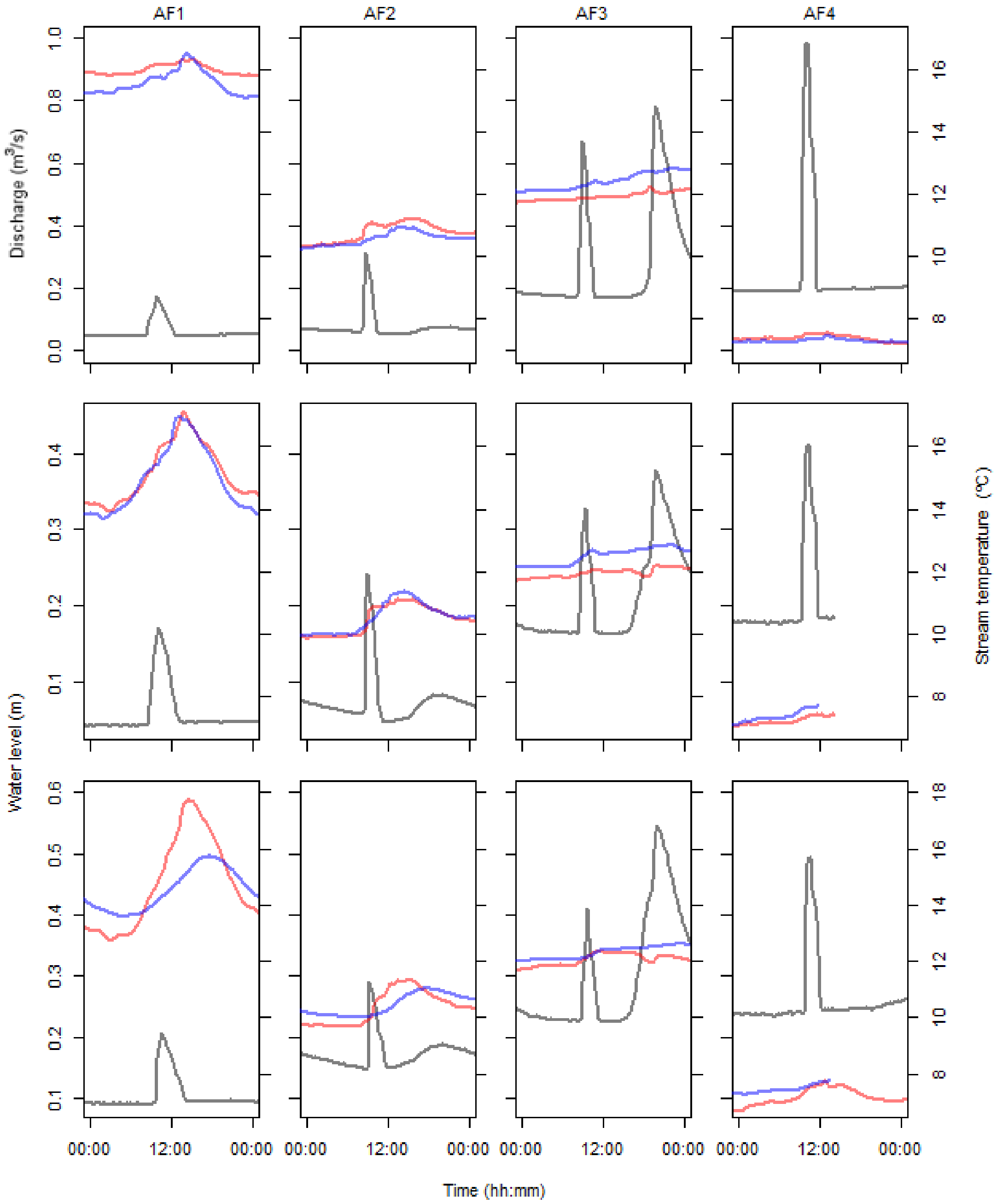

Figure 6.8: Stream temperature during Artificial Floods (AFs) 1 – 4 at sites 6 & 11 (top row), 7 & 12 (middle row) and 13 & 16 (bottom row) (impact and control sites are displayed in red and blue lines respectively). Black lines are discharge/ water level as labelled.

6.6 Discussion

This chapter reported on a spatial-temporal study of stream temperature in a catchment in upland UK over a two year period. An assessment of the impact of regulation on diurnal stream temperature mean (T*m*) and range (T*r*) was undertaken through comparison with regulated conditions revealing novel insights at a variety of scales. Additionally, a flood-focussed analysis of stream temperature mean (F*m*) and range (F*r*) was undertaken. A series of AFs were also conducted which revealed no significant impacts on stream temperature. The following is a discussion of each of these themes in turn.

6.6.1 Impact of regulation

In agreement with H1, this study found evidence that diurnal mean temperature was impacted by regulation for the entire study period at sites closest to the reservoirs (6, 11 and 14). However, for this period, no evidence of impact at sites further than approximately 1.5km downstream of either reservoir was observed suggesting mean stream temperature quickly equilibrated to unimpacted levels as distance from the reservoirs increased. This is in contrast to other findings globally where mean stream temperature has been modified to distances of up to 260km (Zhong & Power, 1996), indicating the importance of site specific factors such as stream discharge, reservoir operation and local climatic conditions.

The nature of the impacts was temporally and spatially complex: in broad agreement with other studies worldwide (e.g. Dickson et al., 2012), during winter, when rapid reduction in mean stream temperature was predicted, a lag in response in stream temperature was observed resulting in higher than predicted temperature (up to 2°C) at sites immediately downstream of reservoirs. Conversely, in broad agreement with Cowx et al (1987), during summer, cooler stream temperatures than predicted (up to ~3.5°C) were observed downstream of one reservoir, but no significant impacts were apparent at other sites. The identification of inconsistent impacts of regulation from reservoirs within the same catchment is novel for the UK and suggests that reservoir or reach specific factors (e.g. reservoir operation, design, stratification, riparian vegetation etc), rather than catchment-wide factors (e.g. weather) are important. This suggestion is similar to that of Webb & Walling (1997) who concluded that reservoir operation was an important factor in determining stream temperature in a multi-year study in the UK.

Previous publications have reported reductions in stream temperature range as a result of regulation (e.g. Lavis & Smith, 1972; Lehmkhul, 1972; Crisp, 1977; O'Keeffe et al., 1990). In agreement with these observations and H1, this study also identified significant reductions in diurnal range for the entire study period at most sites tested. But, in agreement with other literature and H2 (e.g. Webb & Walling, 1988; O'Keeffe et al., 1990) considerable spatial-temporal variability in impacts was also identified. In particular, during autumn and summer, taking logger accuracy into account, clear impacts were confined to within 1.5 and 0.2km downstream of reservoirs respectively. These observations are in contrast to Webb & Walling (1988) who identified impacts on stream temperature range up to 40km downstream of an upland UK reservoir, thus highlighting the importance of site specific factors.

The evidence for modification of stream temperature range identified for the headwater streams in this study could be viewed as contrary to a proven model of physical-chemical response to regulation in headwaters (the Serial Discontinuity Concept (SDC) (Ward & Stanford, 1983; 1995; 2001). Ward and Stanford explicitly argued that regulation of headwater streams would result in little or no change to diel stream temperature range (Ward & Stanford, 1983; 1995). Conversely, this study has identified evidence of significantly reduced diurnal stream temperature range in a regulated headwater stream. The SDC was developed on streams with high groundwater contribution, and therefore relatively low stream temperature variation (Ward & Stanford, 1983). This may therefore explain the disparity in our conclusions, but serves to stress that generalisations across regulated stream 'types' (e.g. headwaters), should be made with caution.

In agreement with other literature (e.g. Webb & Walling, 1993), this study identified temporal variation in the magnitude of impact on stream temperature range. Taking logger accuracy into account, minimal (i.e. < 0.5°C) reduction of diurnal range was typically observed during winter, but during spring and summer, large reductions were recorded (up to 5 and 7°C respectively) at sites immediately downstream of reservoirs. This is also in contrast to the findings of Webb & Walling (1993) who reported that effects were greatest during winter and gives weight to the argument that seasonal effects can be driven by site specific factors such as reservoir characteristics (Crisp, 1977).

In agreement with Webb & Walling (1993), it is proposed that the relatively large thermal capacity of the supplying reservoir water is the primary driver of the observed impacts on stream temperature identified in this study. It is not thought that thermal stratification occurred to any great extent during the period of study (as observed by Crisp, 1977) as typically, mean stream temperatures were followed a lagged response to rapid changes in air temperature rather than a temperature deviation independent of the predicted pattern. However, such a deviation occurred immediately downstream of reservoir C during summer and was assumed to be driven by stratification. Further research should be undertaken to examine this theory as, if confirmed, future management decisions would need to be based on sound knowledge of factors such as reservoir stratification.

Solar radiation and air temperature are key drivers of diurnal cycles in stream temperature in unimpacted streams (Sullivan & Adams, 1991; Caissie, 2006). It is proposed that these drivers were unable to operate on the hypolimnetic reservoir outflow water due to the relatively high thermal capacity of the reservoirs resulting in the observed reductions in stream temperature range downstream of reservoirs in this study. During summer, diurnal range in solar radiation and air temperature are typically greater than during winter (Caissie, 2006). During summer, there is therefore greater potential for impact of regulation on stream temperature range; this phenomena likely explains the peak in impact observed on stream temperature range during summer.

Vegetation cover is a key driver of stream temperature (Sullivan & Adams, 1991; Caissie, 2006). Riparian tree cover within the study area is minimal (pers. obs.) and this, combined with the relatively small reservoir outflow discharges (see Chapter 5) (and therefore small thermal capacity), is therefore likely to be a factor in explaining why the impacts to stream temperature mean and range identified within this study were negligible after relatively short distances downstream of the reservoirs. Further research to examine these hypotheses is recommended.

Further research is required to ascertain, firstly, whether the impacts identified in this study are significant within an ecological context and secondly, if ecologically significant, the magnitude of their significance. Other studies have attributed significant ecological impacts to stream temperature modification due to regulation (e.g. Zhong & Power (1996) stated that fish spawning was delayed by 20-60 days and Webb & Walling (1993) suggested that fish fry emergence would be advanced by up to 57 days due to stream temperature modification) and thus this should be a research priority.

Event based assessment

This study is the first to report on impacts of regulation on stream temperature during floods. It was found that mean stream temperature was estimated to be 1.57°C higher during pre-AF floods in regulated cf. unregulated stream types. This likely explained the significant difference observed between mean stream temperature and hydrological indices during floods between the two site types. Higher mean stream temperature during pre-AF floods downstream of reservoirs could potentially be explained by relatively warm surface epilimnial reservoir water overspilling, but temperature-depth profiles of the reservoirs in the study area are unknown and thus further research is required to test this theory.

The relatively minor difference in stream temperature range during pre-AF floods in regulated cf. unregulated sites and the limited differences in associations between stream temperature range and hydrological indices between the two site types, combined with the accuracies of the loggers used in this study, suggests that stream temperature range is little effected by regulation during floods. If further research was undertaken to assess this, use of loggers with higher accuracies would be recommended.

Impact of Artificial Floods

In contrast with the majority of published literature and H3, this study found no impact of AFs on stream temperature at any of three sites within 2km downstream of reservoir C, suggesting, in disagreement with H3, limited potential of AFs for use as mitigation. However, in agreement with the findings of this study, King et al. (1998) and Robinson et al. (2004) both reported no change in stream temperature during AFs. Both authors suggested that this was due to pre-AF and AF water being hypolimnetic in origin and this likely explains the observations in this study. It is therefore plausible that, for AFs to have an impact on stream water temperature in the study area, the draw off level of water from the water column must be modified from pre-AF conditions and further research is required to test this. Further research may enable comparison of pre-AF and AF relationships between stream temperature response and hydrological indices which was not possible in the current study. A full appraisal of the potential for AFs to mitigate impacts on downstream temperature was therefore not possible in this study; this is a future research priority.

6.7 Summary

This study is the first to report impacts of regulation from multiple reservoirs in a catchment. Mean diurnal stream temperature was identified to be up to 2°C warmer than predicted, but impacts were generally less than 2°C in magnitude, short lasting (<5days) and confined to sites within 1.5km downstream of reservoirs. Observed diurnal stream temperature range was identified as generally being lower than predicted throughout the study period at all sites tested and up to approximately 7°C lower than predicted at one site immediately downstream of one reservoir. Impacts were enhanced during spring and summer, but clear impacts during these periods were limited to within 200m downstream of reservoirs. For both stream temperature indices, substantial differences between sites immediately downstream of reservoirs were identified, indicating the potential importance of reservoir/ reach specific (e.g. reservoir operation, design, stratification, riparian vegetation) rather than catchment-wide factors (e.g.

weather). It was proposed that the key driver of these impacts was the relatively large thermal capacities of the reservoirs cf. streams and that cessation of impacts within short distances downstream of reservoirs was likely due to the combined effect of streams with relatively small discharge and lack of riparian tree cover. During pre-AF floods, mean stream temperature was identified as being significantly higher in regulated cf. unregulated site types, potentially due to overspill of relatively warm epilimnial reservoir water. AFs were found to have no significant impact on downstream temperature in line with other published studies where the draw-off level during AFs was unchanged from pre-AFs. Further research is required to ascertain whether the impacts identified in this study are likely to be significant in an ecological context and whether modification of water draw-off level during AFs invokes a change in downstream temperature.

7 COARSE SEDIMENT TRANSPORT DYNAMICS IN A REGULATED STREAM

7.1 Chapter overview

This chapter presents a temporal assessment of coarse sediment transport dynamics in a regulated stream, including during Artificial Floods (AFs). First, the importance of understanding this topic is presented followed by an appraisal of current gaps in research and identification of aims of the study. Next, the methods and analytical techniques used to undertake the study are detailed. This is followed by sections presenting and discussing the results, including recommendations for further research.

7.2 Introduction

Sediment transport in streams can be viewed as a key process in the control of channel form at a variety of scales, for example, interstices between clasts and patches such as riffles or pools (Bridge, 1993; Church, 1995; Pitlick & Wilcock, 2013). Channel form is intrinsically linked to biological processes (ASCE, 1992) (e.g. throughflow of water in gravel interstices is required for the successful gestation of some fish eggs (Sear, 1993)). Given these relationships, understanding the influence of anthropogenic pressures on sediment transport is important, especially given the biocentricism of contemporary freshwater legislation e.g. the EU Water Framework Directive (EU WFD) (EC, 2000) and the Clean Water Act (USC, 2002).

In the northern hemisphere 77% of total stream discharge is affected by fragmentation of stream channels resulting from reservoir operation, inter basin diversion and irrigation (Dynesius and Nilsson, 1994). An understanding of any impact that these activities have on sediment transport is therefore vital. Furthermore, contemporary legislation such as the EU WFD (EC, 2000) requires that regulated streams meet an ecological standard (Good Ecological Potential (GEP)) based on abiotic and biotic aspects of stream ecology. AFs have been suggested as a potential tool to enable the achievement of GEP through mitigation of impacts associated with regulation (Acreman & Ferguson, 2010). Thus, it is also important that the relationships between AFs and sediment transport are understood to allow for an appraisal of whether AFs could successfully be used as mitigation.

Numerous studies have assessed the impact of regulation on downstream geomorphology (see Chapter 2 for discussion). Reservoir outflows typically lack suspended and coarse sediment (e.g. Dendy & Cooper, 1984; Verstraeten & Poesen, 2000; Kummu, 2010) resulting in downstream bed armouring (coarsening) (Carling, 1988; Brandt, 2000) and/ or bed degradation (removal of all sediment clast sizes) (Brandt, 2000) respectively. Due to the reduction of extreme high flow events which would typically re-work gravels and erode banks, bed stabilisation has also been observed (Carling, 1988; Brandt, 2000). The extent to which these effects occur are highly variable and depend upon factors such as trapping efficiency of the reservoir, flow regime, and downstream geology (Carling, 1988). Nevertheless, it is likely that where these effects occur, downstream sediment transport regimes will be affected, potentially resulting in a reduction of overall transport (due to reduced supply) and a higher threshold discharge required to initiate transport (due to bed stabilisation). To assess these hypotheses, a process based approach is required.

Process based understanding of sediment transport in streams has been cited as a key research priority (Petts & Lewin, 1979; Carling, 1988), yet few studies have addressed this need and those that have (Gilvear, 1987; Lyons, 1992; Sear, 1993), have primarily focussed on transport of *suspended* sediment, most likely reflecting the limitations associated with monitoring the coarse component (Reid et al., 2007; Turowski & Rickenmann, 2011). This has resulted in a dearth of process based understanding of coarse sediment transport in regulated streams. A detailed understanding of elements of the relationship between discharge and sediment transport, including an appreciation of temporal variability and controlling factors (e.g. antecedent conditions), is therefore required to inform and enable successful management of these systems. For example, implementation of sediment transport specific mitigation measures under the EU WFD (UKTAG, 2008).

A recent global review found that only three publications have examined coarse sediment transport response to AFs (Gillespie et al., in press (b)). All of these studies employed pre- and post- flood (i.e. not during) assessments to infer whether coarse sediment transport occurred (Scullion & Sinton, 1983; Petts et al., 1985; Murle et al., 2003). The limited number of published observations concerning sediment transport *during* AFs is now driving the requirement for understanding at regional scales (Poff & Zimmerman, 2010; Gillespie et al., in press (b)). It is therefore crucial that the establishment of understanding in responses of coarse sediment transport to AFs occurs at the site scale to enable larger regional based inferences to be made.

The development of general relationships between hydrological parameters (e.g. flood magnitude) and ecosystem response variables has been used to identify quantitative relationships and potential threshold levels of hydrological parameters that invoke an ecosystem response (e.g. Poff & Zimmerman, 2010; Gillespie et al., in press (b)). The assessment of these relationships during AFs with respect to conditions prior to the implementation of AFs could be used to examine the potential for use AFs in management of regulated streams. To date, this has not been conducted for any regulated stream globally for coarse sediment transport.

The development of sediment impact sensors has enabled the monitoring of coarse sediment transport in streams (e.g. Carling, 2002; Reid et al., 2007; Turowski et al., 2011). Sediment impacts sensors are formed of impact-sensitive loggers attached to steel plates which are installed flush with the streambed. Any impact on the plate (of a clast of diameter > c.12mm (P. Downs (Plymouth University) *pers. comm*. 14th December 2013)) is logged and thus relative numbers of impacts per unit time can be used as a proxy for sediment transport. These sensors have been shown to negate some of the limitations associated with traditional techniques and provide results in line with contemporary theory (Carling et al., 2002). However, there are two key limitations of the devices as outlined by Reid et al., 2007: first, the response of the sensor is non-linear (Carling et al., 2002) resulting in a conservative estimate at high impact rates and second, representative data for whole channel width impact rates are hard to obtain as sediment is often routed along defined pathways (Reid et al., 2007). Nevertheless, the sensors have proved useful for the identification of the timing of sediment transport events and relative transport intensities through time (Reid et al., 2007). To date, these instruments have not been deployed in a regulated stream and therefore have the potential to reveal novel insights into coarse sediment transport in these systems.

This chapter reports on a detailed one year study of coarse sediment transport in a regulated upland UK stream using sediment impact sensors. A series of AFs were also conducted allowing for an assessment of the impact of AFs on coarse sediment transport. Finally, a comparison of the relationship between hydrological parameters and ecosystem responses during AFs is made with respect to conditions prior to implementation of AFs. Specifically, the aims of the study were to: (i) quantify the relationship between discharge and coarse sediment transport in a regulated stream, including the identification of threshold discharges required to invoke transport; (ii) carry out an appraisal of this relationship through time and assess the importance of flood characteristics and antecedent conditions on transport; (iii) identify any impacts of AFs on coarse sediment transport, and, (iv) consider the potential for use of AFs to manage downstream morphology. It was hypothesised that (H1) the sediment transport-discharge relationship and threshold discharge to invoke sediment transport would vary temporally reflecting antecedent flow conditions and (H2) AFs would invoke sediment transport and therefore demonstrate potential for use as a morphological management tool.

7.3 Methods

7.3.1 Sensor description and installation

Sediment impact sensors were custom built using a piezioelectric bimorph vibration element and a Tinytag TGPR-1201 count logger (set to not record more than one count per 0.2s) and were similar to those used in previous studies (e.g. Reid et al., 2007; Rickenmann & McArdell, 2007; Turowski & Rickenmann, 2010; Turowski et al., 2011) The impact plate of each sensor was 150 x 130 x 6mm and formed of 304 grade stainless steel (Figure 7.1). Three impact sensors were mounted in a row on a custom built steel frame at c. 300mm intervals (Figure 7.2). The frame was then installed flush with the streambed, perpendicular to the bank, in riffle habitat typical of site 7. The position of the sensors was determined so that data could be obtained at the channel centre and laterally towards the right hand bank (looking downstream) to provide a representation of sediment transport over a variety of morphological conditions. Dataloggers were set to record the total number of impacts in each two-minute period and were downloaded (and internal clocks reset to reduce drift) approximately every eight weeks.

7.3.2 Discharge

To inform the assessment of coarse sediment transport within the stream, water level at site 7 was recorded at 15 minute intervals and converted to discharge according to the method described in Chapter 5. Water level-discharge rating curves and water levels recorded during the period of study for site 7 are detailed in Appendix B.

7.3.3 Time scale

To assess the relationship between discharge and coarse sediment transport prior to the introduction of AFs, data were selected for analysis for the period 19th August 2012 to 19th August 2013 inclusive. This was after the first major flood event post-installation to allow for sediment in the vicinity of the sensors to settle after disruption during installation.

To assess the impact of AFs on coarse sediment transport, data were selected for analysis for each AF day. AFs were carried out by YW on the 20th August, 19th September , 3rd October and 5th November 2013 (AFs 1-4 respectively) from reservoir C. The characteristics of each AF represented a balance between the practical restrictions placed on YW by the EA, local stakeholders, water resource availability and the capability of reservoir infrastructure to implement a flood. For these reasons, each AF differed in characteristics. However, during all

AFs, water was drawn from the lowest vertical valve within the water column.

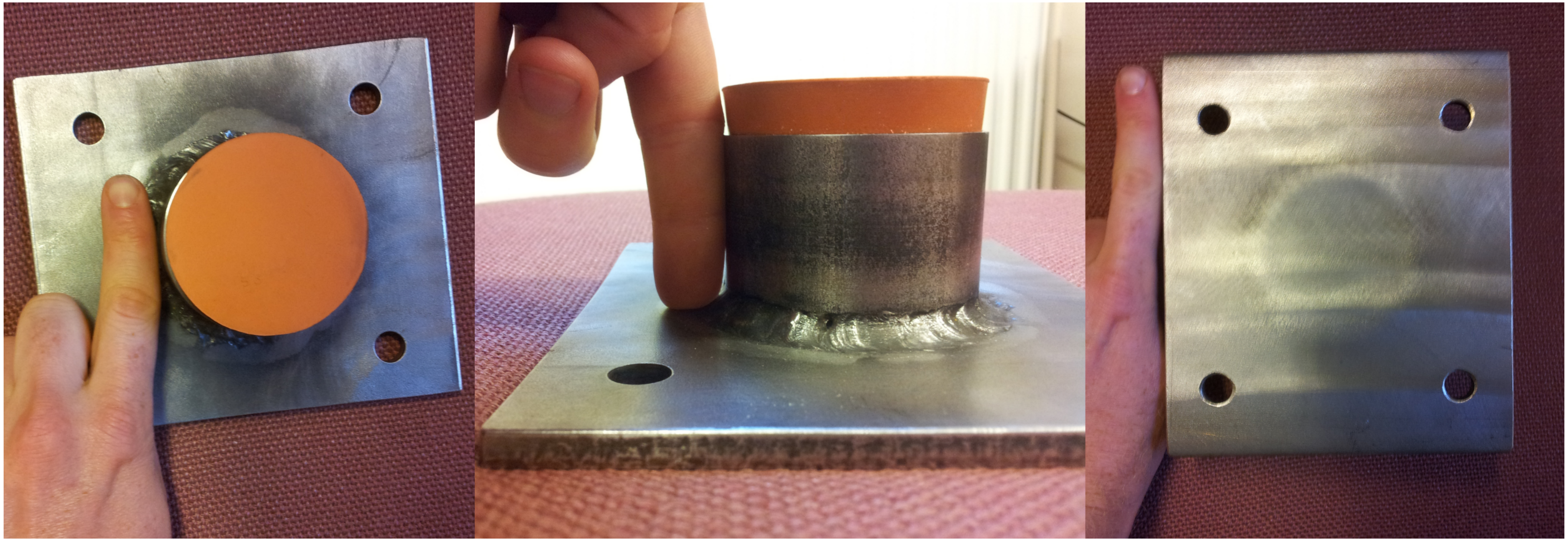

Figure 7.1: Sediment impact sensor: below, side and above (L-R).

Figure 7.2: Impact sensors mounted on frame (top) and installed in stream (bottom).

7.4 Data analysis

7.4.1 Quality control procedure

To ensure quality of impact sensor data, first, as sensors had to be disturbed during download, data were removed for these periods. Data were then scanned by eye for irregular or stochastic records that were assumed not to be realistic (see Appendix C for details of removed data). This resulted in removal of data from one logger between 19:12 and 21:36 on the 26th April 2013 where impacts were logged and presumed to be caused by human/ animal disturbance as discharge was relatively low and no impacts were recorded for either other sensor. Quality control of discharge data was carried out as detailed in Chapter 5. Prior to pairing of impact sensor and discharge data, impact sensor data were binned to 15 minute periods, then the mean number of impacts per m^2 per 15 minute period (I) was calculated across all three sensors. To ensure comparability of sensor and discharge datasets, where data were removed during quality control from one sensor, data for the corresponding time period for all other sensors were also removed.

7.4.2 Coarse sediment transport dynamics

To assess temporal variation in the relationship between discharge and number of impacts, datasets was split based on *a priori* identification of frequently and infrequently flooded periods (prior to and post- March 2013 respectively). Plots of instantaneous discharge and I were then produced. To test the hypothesis that during frequently and infrequently flooded periods, the relationship would differ (H1), ANCOVA between the two time periods using Generalized Additive Modelling (GAM) (Wood, 2011) was performed according to the model:
$I = \alpha + \beta T + s(Q) + \varepsilon$ where T = time period, $s(Q)$ = instantaneous discharge (Q) fitted as a smooth term, α = regression intercept, β = regression coefficient and ε = error term. GAM was used as it provided a better fit (according to AIC) than a Generalised Linear Model (GLM). Approximate normal distribution, independence and homogeneity of residuals was ensured prior to assessment of significance of T using t-statistics and associated p-values. Next, identification of threshold discharge required to mobilise sediment transport was undertaken through graphical analysis of discharge categories and mean I ($\bar{I}$) that occurred at each category. Datasets were split by March 2013 to test the hypothesis that mobilisation threshold discharge would differ in frequently and infrequently flooded periods (H1).

To identify the relationship between coarse sediment transport and discharge indices during pre-AF floods, first, discharge indices were calculated for each flood as described in Chapter 5.

Next, total mean number of impacts per m^2 per 15 minutes ($\sum I$) that occurred during each flood were calculated and summary statistics presented and plots of each relationship drawn.

To identify the relative importance of both hydrological indices and antecedent conditions a similar method to Angert et al. (2011) was followed: GLMs provided the best fit (cf. GAMs) and were therefore used to model $\sum I$ during each flood using the indices listed in Table 7.1 as explanatory variables. Explanatory variables were chosen based on *a priori* understanding of potential influencing factors on sediment transport and were similar to those used by Sidle (1988). Inter-variable correlations were all between -0.7 and 0.7 (e.g. Angert et al., 2011). A *gamma* error distribution with *log* link function was specified in each model to ensure approximate normal distribution, independence and homogeneity of residuals. The best subset of models was identified using AIC_c (AIC adjusted for small sample sizes as n/K < 40 where n= number of observations and K = number of explanatory factors (Burnham & Anderson, 2002)) and all models with AIC_c differences ($\Delta_i = AIC_i - AIC_{min}$) ≤ 2 were reported. To account for model uncertainty, model averaging was then performed for all coefficients of explanatory factors included in reported models. All statistical analyses were performed in RStudio (version 0.97.551) using packages MuMin (Barton, 2013) and glmulti (Calcagno, 2013).

Table 7.1: Descriptions of explanatory variables included in GLMs.

Explanatory variable	Description
Magnitude (F_i)	Magnitude of flood *i*
ROC falling (F_i)	Rate of change (falling limb) of flood *i*
Magnitude (F_{i-1})	Magnitude of previous flood
ROC falling (F_{i-1})	Rate of change (falling limb) of previous flood
Duration (F_{i-1})	Duration of previous flood
ROC rising (F_i)	Rate of change (rising limb) of flood *i*
ROC rising (F_{i-1})	Rate of change (rising limb) of previous flood
Duration (F_i)	Duration of flood *i*
Time between peak discharge (F_i - F_{i-1})	Time between peak discharge of flood i and previous flood
Time between start of F_i and end of F_{i-1}.	Time between end of previous flood and start of flood *i*

Impact of Artificial Floods

Discharge indices and $\sum I$ were calculated for each AF using the methods described above. Graphical representation was then used to assess the relationship between AF indices and $\sum I$

with respect to the relationship defined for pre-AF floods. This would allow for assessment of the hypothesis that AFs would demonstrate potential for use as a morphological management tool (H2).

7.5 Results

7.5.1 Sediment transport dynamics

$\bar{I}$ for the assessment period was 129 (st. dev. 815) and the maximum was 26,820. A duration frequency curve of *I* displayed relatively few impacts for c. 30% of the study duration and an approximate logarithmic increase in *I* occurring below this exceedance frequency (Figure 7.3). Visually, *I* appeared to be broadly correlated to discharge and the majority of impacts occurred prior to March 2013 (Figure 7.4). For the entire assessment period, a general power relationship between instantaneous discharge and *I* was apparent, but relatively large numbers of impacts at low discharges were observed resulting in a poor model fit (R^2 : 0.64) (Figure 7.5). This relationship post-March 2013 was not significantly different from pre-March 2013 (GAM model: R^2 : 0.71, t : -1.53, p = 0.13). For the entire assessment period, a relatively large increase in $\bar{I}$ was observed at discharges between 0.5 and 0.8 m^3/s; pre-March 2013 also displayed this trend, but this large increase was not seen post-March 2013 evident in a difference of c. 0.2 m^3/s associated with an $\bar{I}$ of c. 200 for pre- and post-March 2013 (Figure 7.6).

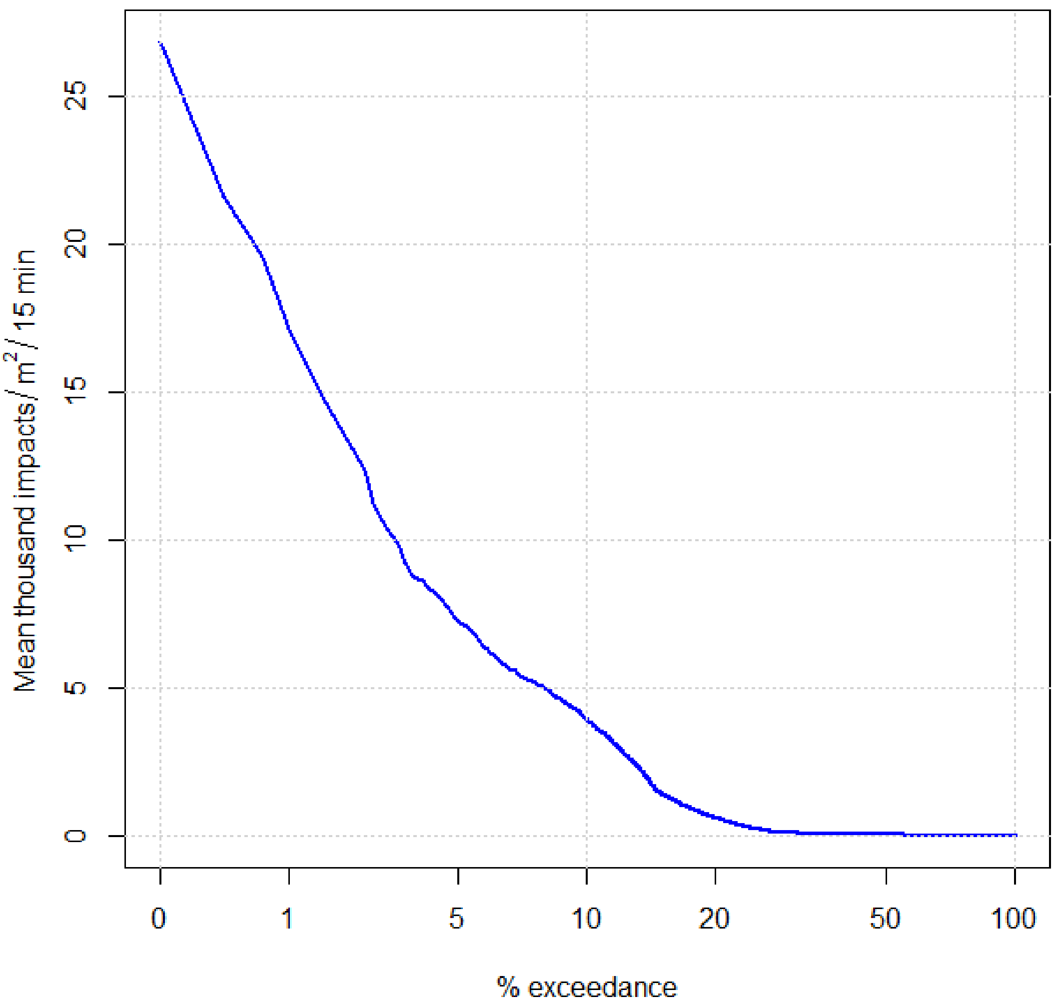

Figure 7.3: Exceedance curve for mean number of impacts (I).

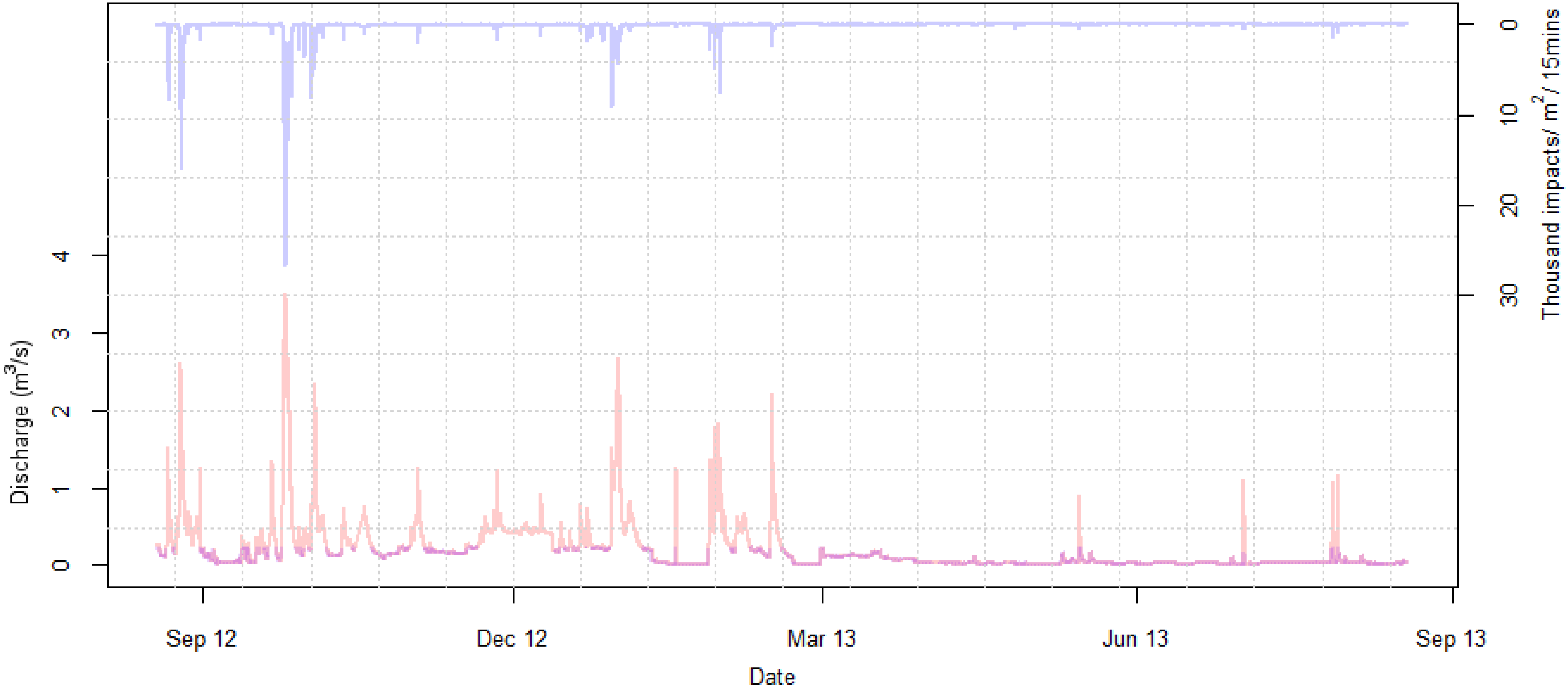

Figure 7.4: Temporal dynamics of mean number of impacts (I) (blue) in relation to discharge (orange (>Q_{25}) / pink (<Q_{25})).

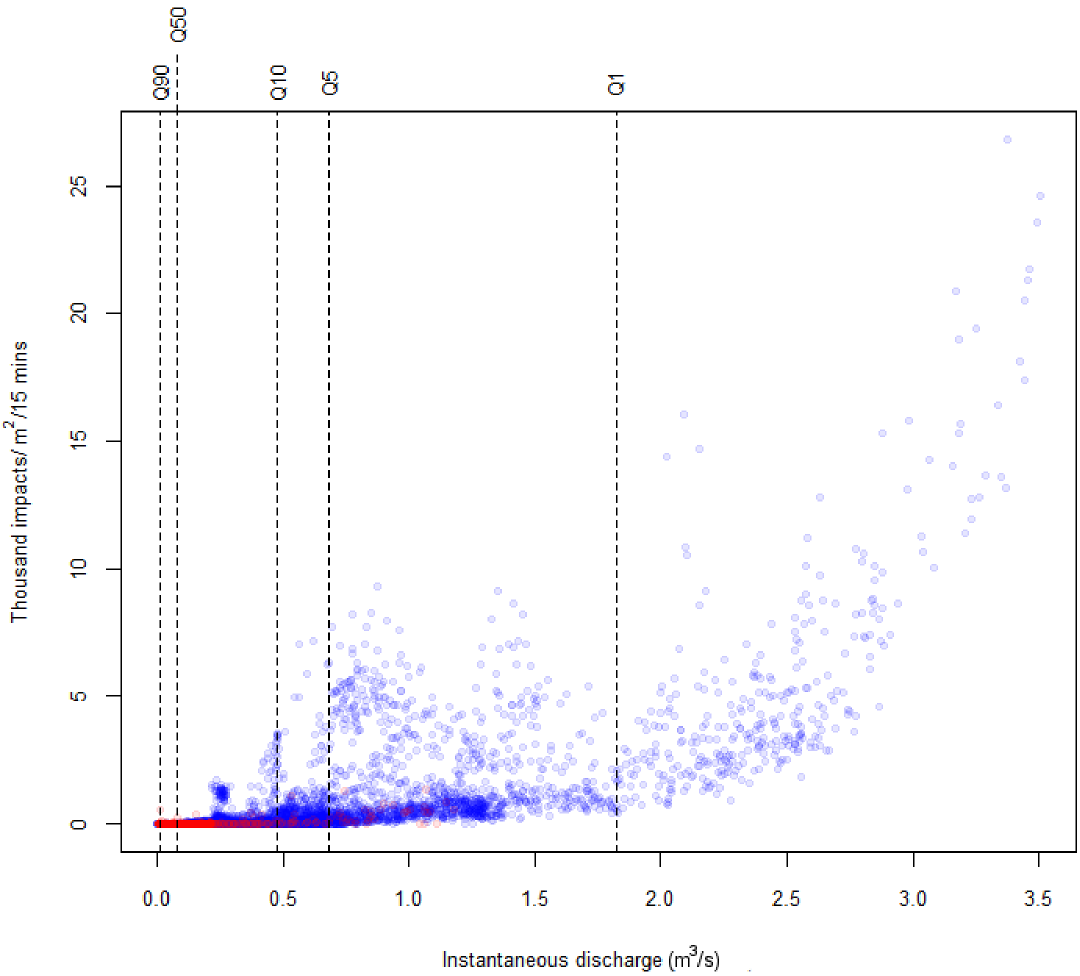

Figure 7.5: Mean number of impacts (I) against instantaneous discharge <March 2013 (blue) and >March 2013 (red).

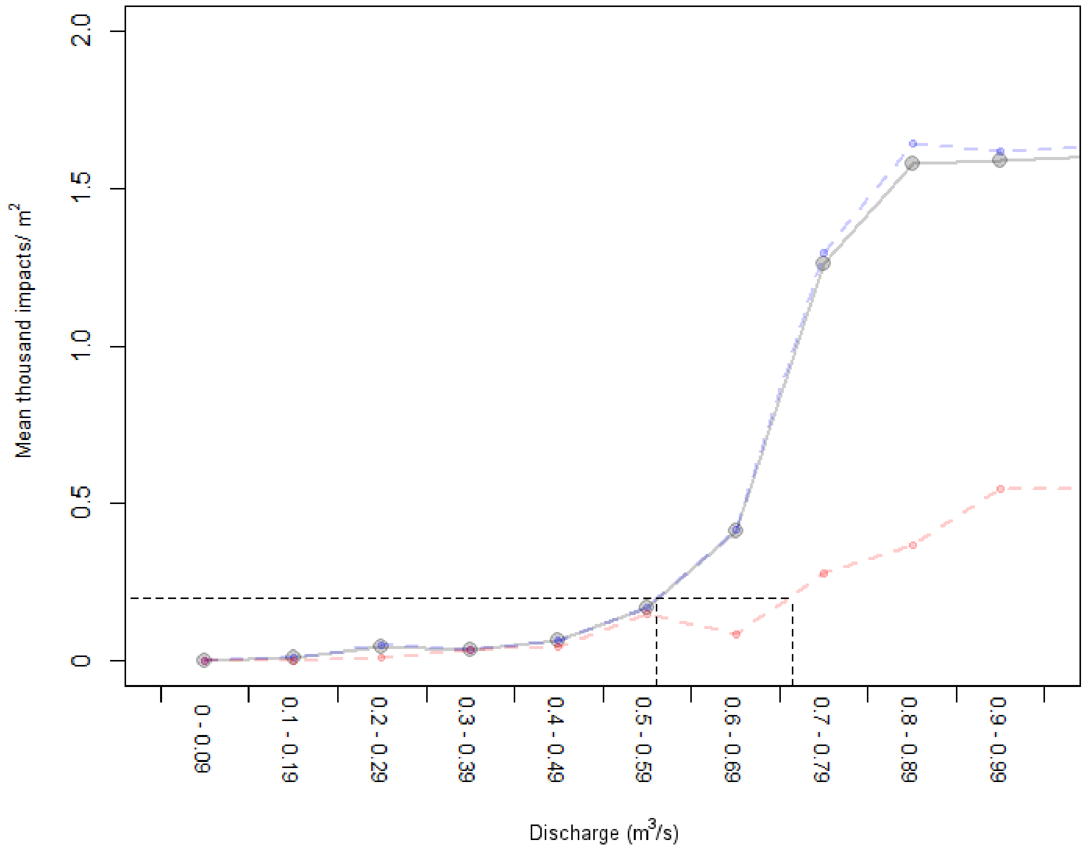

Figure 7.6: Plot displaying mean mean number of impacts ($\bar{I}$) at discharge <1 cumec for entire study period (full black line), <March 2013 (blue dashed) and > March 2013 (red dashed).

7.5.2 Event based analysis

Mean $\sum I$ was 88,897 (st. dev. 255,521) and a maximum $\sum I$ of 1,542,667 was recorded (Table 7.2). Mean hydrological indices were 233.27, 1.81, 3.13, 1.29 for magnitude (% increase), duration (days) and ROC rising and falling (m^3/s/ day) respectively (Table 7.2).

During floods of up to a 400% increase in magnitude, relatively few impacts occurred. However, a broadly positive association between flood magnitude and $\sum I$ was observed for floods of magnitude > 400% increase. A broadly positive association between flood duration and $\sum I$ was observed, but no clear association between flood ROC for either falling or rising limbs and $\sum I$ was observed (Figure 7.7).

*Table 7.2: Hydrological indices and total mean number of impacts per m² per flood (∑**I**) for pre-Artificial Flood floods.*

	∑*I*	Magnitude (% increase)	Duration (days)	ROC rising (m^3/s/day)	ROC falling (m^3/s/day)
Mean	88892	233.27	1.81	3.13	1.29
St. dev.	255521	336.66	3.59	7.74	4.03
Min.	0	1.69	0.03	0.05	0.06
Max.	1542667	1,421.45	22.02	51.08	28.11

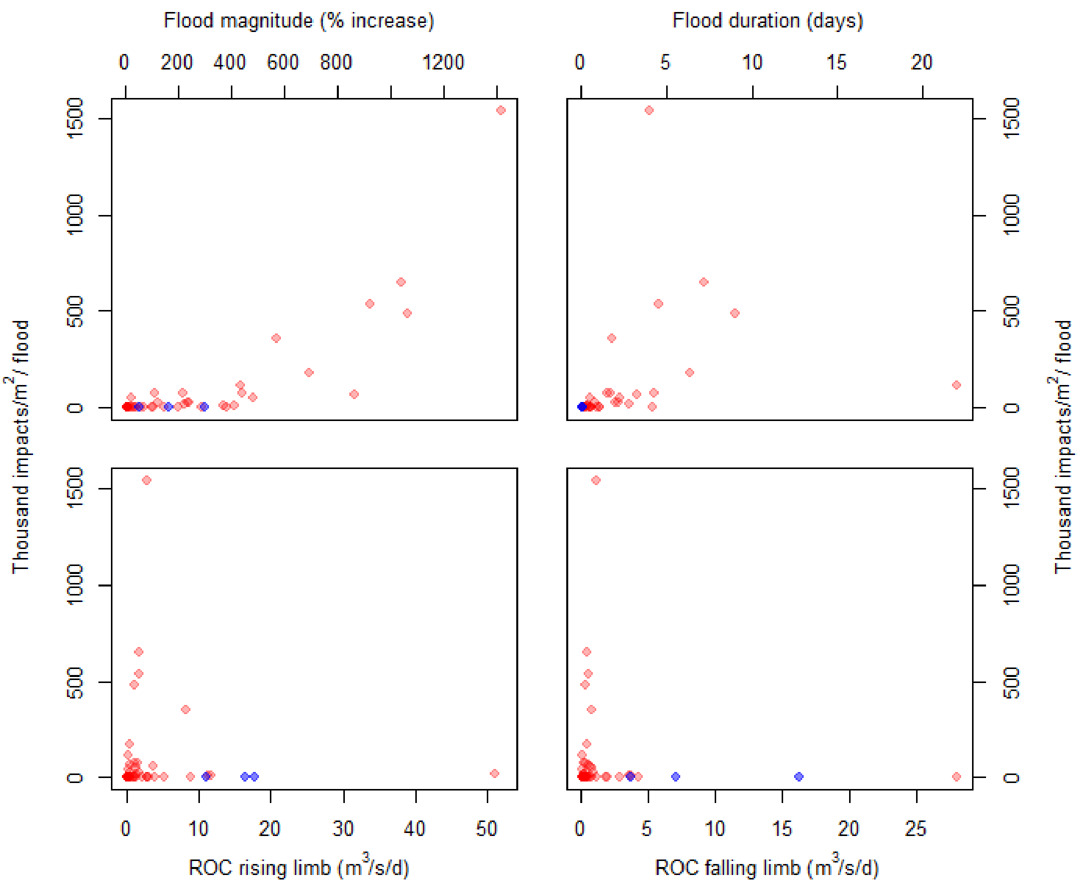

Figure 7.7: Plots of total mean number of impacts per m² per flood (∑I) against hydrological indices for pre-Artificial Flood (AF) floods (red) and AFs (blue).

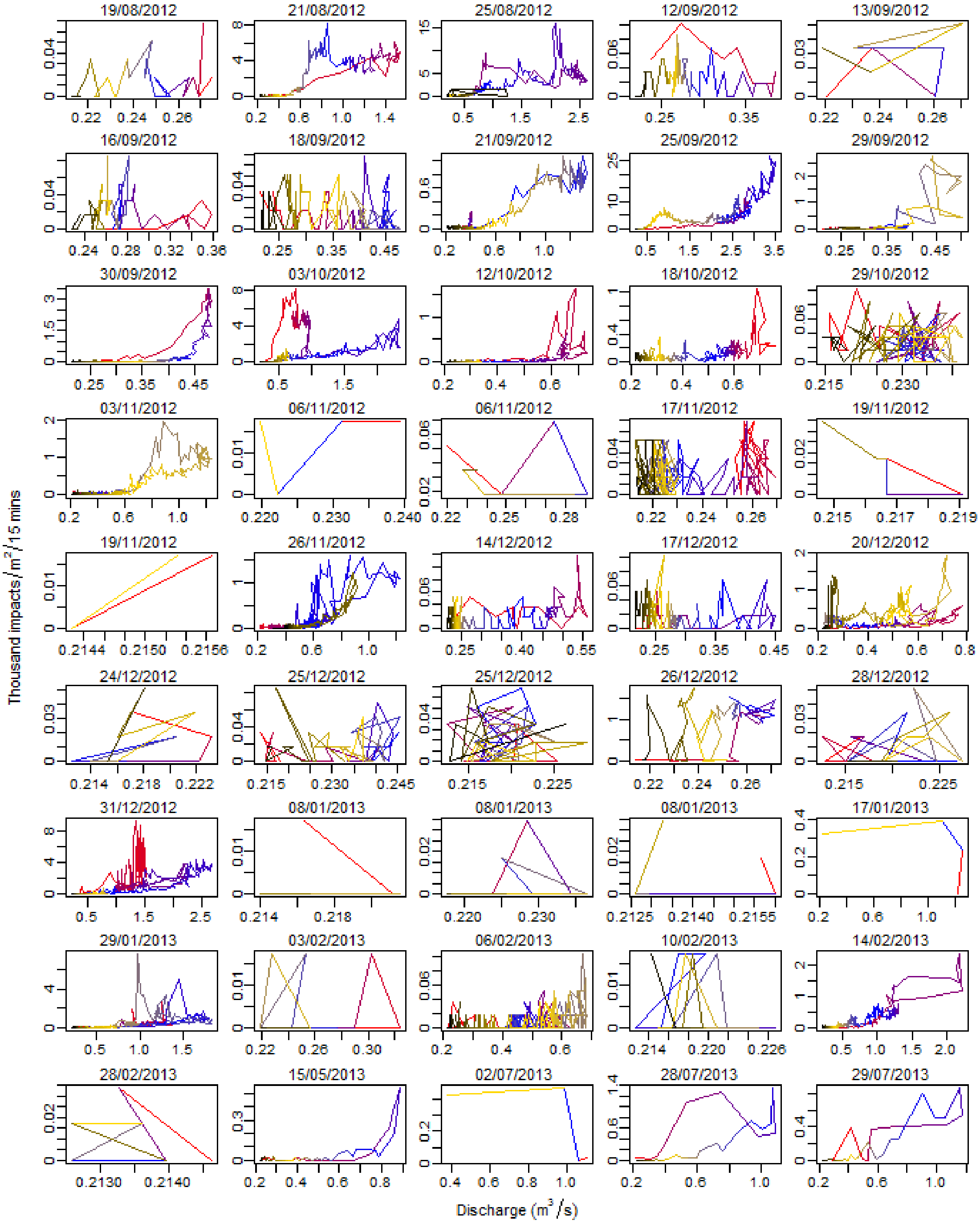

Figure 7.8: Hysteresis curves of mean number of impacts/m^2/15 minutes (I) against discharge for each flood during the study period. Lines are coloured to follow progression from the start to end of each flood (red ➔ blue ➔ gold ➔ black respectively).

A clear trend in *I* throughout all floods was not evident, but four distinct types of response could be identified: (i) a general positive relationship between discharge and *I* was identified for nine floods (21st, 25th, 29th & 30th September 2012, 12th & 18th October 2012, 26th November 2012, 14th February 2013 & 15th May 2013). (ii) a clear peak in *I* prior to maximum discharge was evident in floods on two occasions (3rd October 2012 & 31st December 2012), whereas the opposite (iii) was observed in four floods (21st August 2012, 16th September 2012, 3rd November 2012 & 29th January 2013). The relationship between discharge and I in remaining floods was either complex (e.g. 29th October 2012) or unclear due to short flood duration (e.g. 2nd July 2013) (iv) (Figure 7.8).

Assessment of association of explanatory variables with $\sum I$ identified nine models with $\mathbf{\Delta_i} \leq 2$. Of these models, none contained variables: *duration* (F_i), *time between peak discharge* ($F_i - F_{i-1}$) or *time between start of F_i and end of F_{i-1}*. All models contained variables *magnitude* and *ROC falling* (F_i) significant at $p < 0.05$. Model averaging revealed that these latter two variables shared equal Akaike weights of 1.00. However, of these two variables, only coefficients for *magnitude* (F_i) had 95% CI that did not encompass 0 (1.37 to 2.53, estimate: 1.95) (Table 7.3).

Table 7.3: Results of model selection and averaging for models relating flood (F_i) and antecedent condition indices to ∑I. Variables included within each model are delineated with the symbol ● and models are ranked in order of increasing AIC$_c$ differences (Δ_i). Akaike weights (w_i) indicate the relative likelihood of each model, given the set of models considered (Burnham & Anderson, 2002) (models were considered if $\Delta i \leq 2$). Model-averaged regression coefficients (β) are averages of β_i of all considered models weighted by each model's w_i and $\beta = 0$ where a variable was not selected for inclusion within a model. Model average β 95% CI that do not encompass 0 are delineated in bold. Relative variable importance (w_{ip}) is the sum of all w_i across all models including that variable (Burnham & Anderson, 2002).

	Model rank									**Model average**		
Variable	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**β**	**95% CI**	**w_{ip}**
Magnitude (F_i)	● **	● **	● **	● **	● **	● **	● **	● **	● **	1.95	**1.37 to 2.53**	1.00
ROC falling (F_i)	● **	● *	● *	● **	● **	● *	● **	● **	● *	-0.59	-1.19 to 0.01	1.00
Magnitude (F_{i-1})	● *	● *					● *	●	●	-0.87	**-1.56 to -0.17**	0.57
ROC falling (F_{i-1})	●			●	●			●		0.47	-0.20 to 1.14	0.47
Duration (F_{i-1})			●		●				●	-0.40	-1.00 to 0.20	0.29
ROC rising (F_i)							●			0.48	-0.25 to 1.21	0.08
ROC rising (F_{i-1})								●		-0.34	-0.92 to 0.24	0.07
Duration (F_i)										-	-	-
Time between peak discharge (F_i - F_{i-1})										-	-	-
Time between start of F_i and end of F_{i-1}.										-	-	-
Δ_i	0.0	0.4	0.8	1.0	1.3	1.3	1.6	1.9	2.0			
w_i	0.19	0.16	0.12	0.11	0.10	0.10	0.08	0.07	0.07			

$*p < 0.05$, $**p < 0.01$.

7.5.3 Artificial Flood analysis

Peak magnitude of AF1 was < Q_{25} discharge and therefore hydrological indices could not be calculated for this AF according to the method described in section 5.3. The highest $\sum I$ occurred during AF4 (649.57). Peak I during this AF was 290.60 which occurred prior to peak discharge. $\sum I$ during AFs 2 and 3 was 34.19 and 136.75 respectively, but values of I observed during AFs were not dissimilar to those prior to each AF (Figure 7.9).

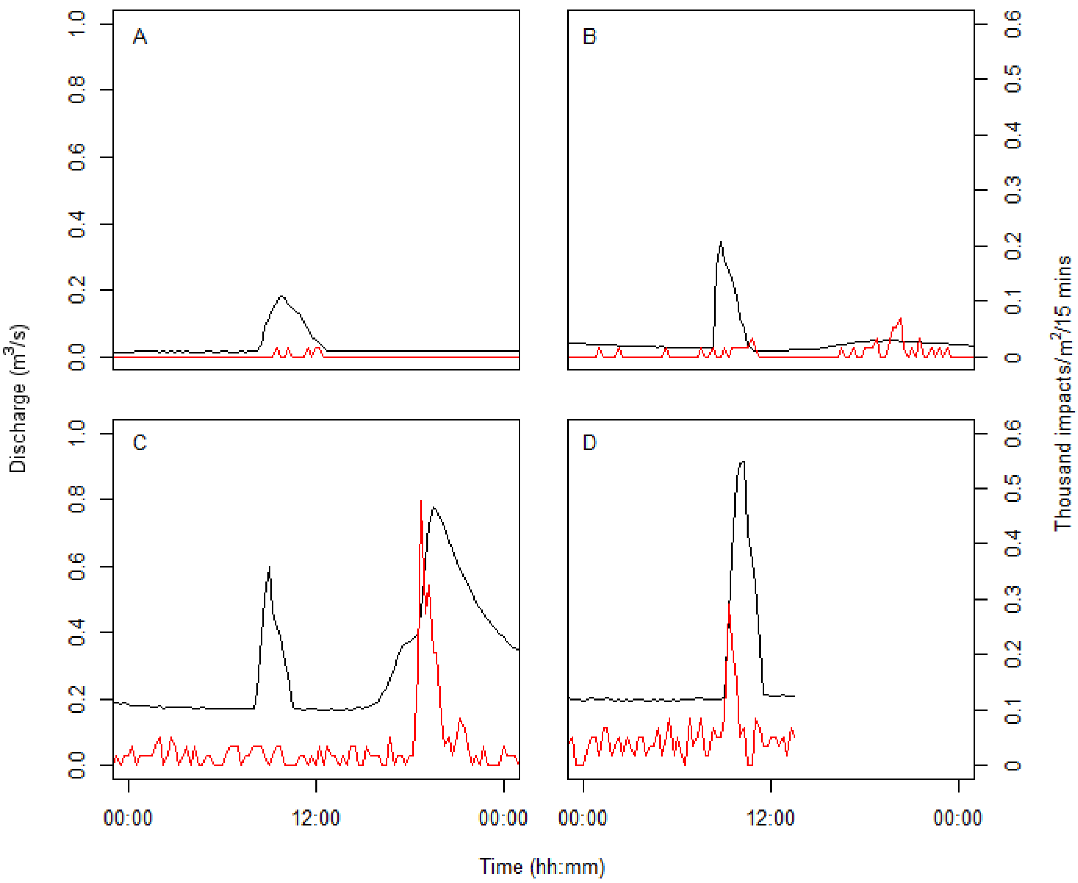

Figure 7.9: Temporal dynamics of mean number of impacts (I) (red lines) during Artificial Floods 1-4 (panels A-D respectively). Discharge during each AF is shown in black.

Comparison of the relationship between hydrological indices and $\sum I$ between pre-AF floods and AFs revealed that relationships were similar for flood magnitude and duration. ROC (rising limb) of two AFs were larger than observed for any of the pre-AF floods, but the relationships were not dissimilar to those observed for pre-AF floods. The relationships observed for ROC (falling limb) during AFs were also not visually dissimilar to those observed for pre-AF floods (Figure 7.7).

7.6 Discussion

A process based understanding of coarse sediment transport in regulated streams is currently lacking. This chapter has reported on a detailed study of coarse sediment transport dynamics in an upland UK regulated stream over a year of contrasting flows. A holistic and event focussed, process based assessment of coarse sediment transport dynamics was undertaken revealing novel results. Finally, a comparison of the relationship between hydrological indices and coarse sediment transport during AFs was carried out with respect to these relationships identified during pre-AF floods. The following is a discussion of each of these themes in turn.

7.6.1 Sediment transport dynamics

Mean number of impacts was generally positively associated with discharge as observed by other authors for unregulated streams (e.g. Reid et al., 2007; Turowski & Rickenmann, 2011). However, the relationship did not follow a clear trend due to 'noise' at low discharges where relatively high numbers of impacts were recorded. It is proposed that these records are potentially due to bank collapse or bed destabilisation due to preceding high flow (e.g. Klingeman & Emmett, 1982). Alternatively, the relative lack of datapoints at higher discharges may be due to saturation of the sensors (Reid et al., 2007) or it may be that clast saltation length at high discharges is larger than the impact plate dimensions of each sensor resulting in an underestimate of sediment transport at these discharges. Further research is therefore required to test these hypotheses. Nevertheless, this is the first study to demonstrate that coarse sediment transport in a regulated stream is broadly positively correlated with discharge as observed in unregulated streams.

Discharge during the assessment period of this study was classified into frequently and infrequently flooded periods (prior to and post-March 2013 respectively). It was hypothesised that the relationship between discharge and number of impacts would differ between these periods (H1), but this study found no statistical evidence for a difference suggesting that the holistic relationship did not differ between relatively dynamic and banal discharge regimes. However, examination of threshold discharges required to stimulate sediment transport revealed that higher discharges (c. 0.2 m^3/s) were required to elicit c. 200 impacts/ m^2/ 15 minutes during the infrequently flooded period cf. the frequently flooded period. This was in agreement with the hypothesis that different threshold discharges would be observed between periods (H1) and was potentially due to changes in sediment supply or differences in the characteristics of floods between periods. Bed armouring during periods of infrequent flooding is commonly observed in gravel-bed regulated streams (e.g. Sear, 1993; Vericat et al., 2006) and it may be that the development of an armoured, more stable bed contributed to this change in threshold. Further assessment would be required to confirm the drivers of this change.

7.6.2 Event based analysis

Flood based analysis of sediment transport revealed that relatively low gross sediment transport occurred in small floods (< 400% increase from the Q_{25}), but during larger floods, a positive relationship between flood magnitude and gross sediment transport was observed suggesting that c. 400% flood magnitude increase over Q_{25} acts as a threshold for sediment transport, potentially relating to physical change, for example, bank collapse or mobilisation of a relatively abundant clast size. The sensors utilised by this study do not allow for assessment of clast sizes involved in transport and this is a key priority for future research. Flood duration appeared to be generally positively associated with gross sediment transport, but several extreme outliers reduced the validity of the general relationship. No clear association between either ROC rising or falling could be identified, suggesting that the key driver of gross sediment transport is flood magnitude.

Considerable variation in discharge-number of impacts hysteresis curves was evident indicating a temporally complex relationship as observed in other gravel bed streams (e.g. Moog & Whiting, 1998; Ryan et al., 2005). In the majority of floods, no clear pattern was evident, but a general positive relationship between number of impacts and discharge was observed in nine floods, while peaks in impacts were observed post-peak discharge (anticlockwise hysteresis) on four occasions and the opposite (clockwise hysteresis) in two floods. Asynchronous peaks in discharge and number of impacts in *some*, but not *all* floods suggests that these observations are driven by factors specific to each flood, rather than associated with sensor capability. This gives weight to the earlier hypothesis that bank collapse or bed destabilisation due to preceding high flow (e.g. Klingeman & Emmett, 1982) may have driven relatively high numbers of impacts at low discharges. Alternatively, in the case of clockwise hysteresis, exhaustion of sediment supply prior to peak discharge is likely (Dunne & Leopold, 1978; Moog & Whiting, 1998). Further research is therefore required to assess these hypotheses.

An assessment of the relative importance of flood and antecedent indices revealed that flood magnitude was the most important factor in prediction of gross sediment transport in agreement with other studies (e.g. Bogen et al., 2003). A β of 1.95 represented the positive association between the two variables. ROC falling also had a high importance, but 95% CI for the coefficient encompassed zero and therefore the effect of this factor is uncertain. Magnitude of the previous flood was the third most important factor in prediction of gross sediment transport (β: - 0.87) and 95% CI for this factor did not encompass zero, suggesting a negative relationship between these two variables. This gives support to the argument that previous floods of high magnitude can reduce gross sediment transport of a following flood, potentially through reduction of sediment available for transport (e.g. Sidle, 1988). Of the antecedent indices tested, *time between peak discharges of a flood and the previous flood* and *time between the end of the previous and the start of the current flood* were not included in any of the best models and it is therefore suggested that these factors are not significantly related to gross coarse sediment transport. Remaining antecedent factors were identified as important to some extent, but uncertainty due to the coefficient 95% CI for all of these factors encompassing zero suggests that the importance of these factors should be treated with caution. This is the first study to identify the importance of antecedent flow conditions in predicting coarse sediment transport in a regulated stream and this observation is likely to be an important consideration in the management of such streams.

7.6.3 Artificial Flood analysis

Prior to this study, few publications had reported on coarse sediment transport response to AFs. Scullion & Sinton (1983) inspected the stream bed pre- and post-AF and stated that coarse sediment transport did not occur and Petts et al. (1985) noted the transport of small diameter (< 35mm) gravel. More recently, Murle et al. (2003) described channel morphological changes after a series of AFs and suggested that coarse sediment transport had occurred. This study observed that coarse sediment transport (above that of background transport) occurred only during one (the largest in magnitude) out of four successively larger magnitude AFs (peak magnitudes of 0.18, 0.35, 0.60 and 0.91 m^3/s respectively representing c. Q_{33}, Q_{19}, Q_6 and Q_4 flows prior to AFs) suggesting that the threshold discharge for coarse sediment transport during these AFs was between 0.60 and 0.91 m^3/s. This was consistent with observations made in pre-AF floods post March 2013. Notably, peak coarse sediment transport in AF4 occurred prior to peak discharge indicating limitation of sediment supply. Further research is required to assess whether this is a consistent observation for AFs, as this may have important implications for management of regulated streams.

Comparison of the relationship between hydrological indices and gross sediment transport between pre-AFs and AFs highlighted that apart from ROC rising, relationships were comparable for both flood types. AF rising limb ROC were higher than those observed in pre-AF floods and responses were therefore not comparable. It is important to note however, that gross sediment transport during each AF was relatively small compared to observed during pre-AFs (likely due to the relatively small magnitudes of each AF) and there is therefore a need to assess coarse sediment transport during AFs of larger magnitude in order to better understand the relationships between hydrological indices and coarse sediment transport. Although limited data were available for comparison between AFs and pre-AF floods, in agreement with findings elsewhere (e.g. Petts et al., 1985; Murle et al., 2003), the data suggest that, in agreement with H2, AFs could potentially be used as a tool for the management of the coarse sediment regimes in regulated streams. There is however a clear need for a detailed understanding of the relationship between AF characteristics and coarse sediment transport to enable informed management decisions to be taken.

7.7 Summary

This study is the first to take a process-based approach to examination of coarse sediment transport in a regulated stream. The ability to relate discharge to coarse sediment transport and identify threshold discharges and relative transport rates during distinct flood events has been demonstrated. Sediment transport was generally positively correlated with discharge. It was proposed that deviances from this association were due to discrete events (e.g. bank collapse) rather than an artefact of the sensors used. This assumption was further evidenced through assessment of hysteresis curves which demonstrated considerable variability of sediment transport dynamics during each flood. Threshold discharges required to stimulate sediment transport appeared to be temporally variable, demonstrating the importance of antecedent conditions.

Flood based analysis of coarse sediment transport revealed that a discharge increase of ~400% above Q_{25} appeared to act as a general threshold for significant sediment transport, potentially linked to physical change (e.g. mobilisation of a particularly abundant clast size). Total sediment transport during floods was most associated with flood magnitude (positive relationship), but interestingly, magnitude of the previous flood was negatively associated with total sediment transport indicating that sediment supply after large floods may be limited for transport in subsequent floods.

Significant coarse sediment transport only occurred in the final, largest magnitude, AF. This finding was consistent with the relationship between discharge and sediment transport prior to AFs and gives weight to the argument for the use of AFs as a morphological management tool.

Whilst this study has improved the basic understanding of coarse sediment transport dynamics in a regulated upland stream, the following remain key research priorities: (i) a spatio-temporal assessment of the impact of stream regulation on coarse sediment transport dynamics, and (ii) a spatio-temporal assessment of the impact of AFs on coarse sediment transport under a range of flow and antecedent conditions. It is suggested that the impact sensors utilised in this study could be used to, in part, successfully achieve these aims. It is suggested that without the achievement of these aims, the management of regulated upland stream sediment regimes is likely to be unsuccessful due to the complex interaction of factors which can determine sediment transport.

8 MACRO-SPATIAL IMPACTS OF REGULATION ON MACROINVERTEBRATE ASSEMBLAGES

8.1 Chapter overview

This chapter presents a macro-scale (regional) assessment of the impact of regulation on downstream benthic macroinvertebrate assemblages. First, the importance of understanding this topic is presented followed by an appraisal of current gaps in research and identification of aims and hypotheses of the study. Next, the methods and analytical techniques used to undertake the study are detailed. This is followed by sections presenting and discussing the results, including recommendations for further research.

8.2 Introduction

Approximately 15% of the world's stream flow can be impounded by dams (Nilsson et al., 2005). In the northern hemisphere 77% of total stream discharge is either strongly or moderately affected by fragmentation of stream channels resulting from reservoir operation, inter-basin diversion and irrigation (Dynesius and Nilsson, 1994). In the UK, compensation flows from many reservoirs have historically been set in excess of pre-reservoir Q_{95} resulting in elimination of extreme low flows (Higgs & Petts, 1988; Gustard, 1989). This effect is particularly prevalent at sites within the Humber catchment; for example, discharge downstream of Scout Dike reservoir was estimated to have increased from 6 to 21% of the pre-regulation mean discharge (Gustard et al., 1987). Additionally, across the UK, average post-impoundment peak discharges have been reduced to 74% of the maximum level observed prior to regulation (Gustard, 1989). For example, in the Humber catchment, 67% of large floods were recorded to be retained by Ladybower reservoir. These changes in flow are likely to have resulted in modification of downstream macroinvertebrate assemblages, and the understanding of such impacts is crucial for effective management of these systems under contemporary ecocentric legislation (e.g. the Water Framework Directive (EU WFD) (EC, 2000)).

The EU WFD requires member states to ensure that water bodies classed as heavily modified meet Good Ecological Potential (GEP) unless derogation is appropriate (UKTAG, 2008). Water bodies are classed as heavily modified when they are likely to fail to meet Good Ecological Status (GES) (see Acreman and Ferguson, 2010) due to physical alteration, leading to modification of the flow regime (Acreman et al., 2009). Environmental flows (Acreman and

Ferguson, 2010) have been suggested as a potential tool to enable the achievement of GEP. However, without sound understanding of the impact of reservoirs on macroinvertebrates, the necessity of introducing environmental flows may be questioned.

The impact of stream regulation on benthic macroinvertebrates downstream of impoundments has been the focus of many studies worldwide, yet varied observations have been made leading to a lack of clarity in understanding. In response to regulation, total abundance has been recorded to increase at some sites (e.g. Armitage, 1978; Munn & Brusven, 1991) but decrease at others (Englund & Malmqvist, 1996). Additionally, responses of some taxonomic groups have been observed to vary, for example, no change or decrease in abundance of Coleoptera (Spence & Hynes, 1971 and Nichols et al., 2006, respectively). Furthermore, varied responses in diversity have also been observed (increased: Poole and Stewart, 1976; Penaz et al., 1968; Maynard and Lane, 2012; decreased: Pearson et al., 1968; Armitage 1978; Munn & Brusven, 1991). Further research is required to both clarify and understand the mechanisms behind these differential responses.

Research into the impact of regulation on downstream macroinvertebrates has typically taken a two-tier ‘binary’ approach to classify streams (i.e. regulated or unregulated) (e.g. Armitage, 1989) rather than more detailed ‘continuous’ descriptors of the extent of regulation. The scale at which research has been undertaken may also be important: research conducted on single streams (Petts et al., 1993; Maynard and Lane, 2012) has identified clear impacts but has the shortcoming of failing to identify patterns over larger scales. Regional-scale studies (e.g. Growns and Growns, 2001) have the potential to yield a better understanding of how regulation drives changes in biotic pattern and process. Moreover, this approach has the additional benefit of being aligned with current water legislation (e.g. EU WFD (EC, 2000)) which utilises regional management plans (i.e. River Basin Management Plans).

Recently developed indices such as the Lotic Invertebrate index for Flow Evaluation (LIFE) (Extence et al., 1999) and Proportion of Sediment-sensitive Invertebrates (PSI) (Extence et al., 2013) may be useful in identification of impacts of stream regulation on macroinvertebrate assemblages. LIFE has been developed to link macroinvertebrate community composition to hydrological dynamics and thus may be particularly useful in regulated stream research as hydrology can be significantly modified by impoundment as discussed above. PSI scores have been shown to reflect the extent to which stream surface sediment is either composed of, or covered by, fine sediment (Extence et al., 2013) and, due to geomorphological modifications associated with stream regulation such as bed armouring (reduction of fine sediment) (Carling, 1988; Brandt, 2000), the use of PSI in these systems is potentially appropriate. Both LIFE and PSI are abundance-weighted and are calculated based on taxon-specific flow (Extence et al.,

1999) and habitat (Extence et al., 2013) associations, respectively. Higher LIFE scores indicate higher prevailing antecedal flows (Extence et al., 1999) whereas higher PSI scores indicate minimally/ unsedimented substrate (Extence et al., 2013). While these relationships have been quantified and substantiated (Extence et al., 1999; Monk et al., 2008; Dunbar et al., 2010(a & b); Extence et al., 2013; Worrall et al., 2013; Glendell et al., 2014) for streams in England and Wales, neither index has been used to assess the impacts of upstream impoundment *per se*.

In light of the identified research gaps this chapter aimed to identify relationships between the extent of stream regulation and macroinvertebrate communities. It takes a multi-site, regional-scale approach and advances a new index representing the extent of stream regulation. It also evaluates two recently developed indices (LIFE and PSI) alongside established biomonitoring indices to consider their relative performance for assessment of the impacts of regulation. It was hypothesised that: (H1) macroinvertebrate indices and community composition would both be affected by upstream impoundment with some taxa increasing and others decreasing in abundance relative to their sensitivity to changes in flow; (H2) A continuous index representing extent of regulation would be more sensitive to differences in community composition than categorical classifications and, (H3) LIFE and PSI would decrease as the extent of stream regulation increases, and demonstrate superior sensitivity to alternative indices such as diversity, dominance, BMWP and ASPT in detecting any impacts.

8.3 Methods

8.3.1 Macroinvertebrate data

Data for 19 fully-regulated, 28 semi-regulated and 17 unregulated sites (see definitions below) were obtained from Yorkshire Water (YW; n=47), the Environment Agency (EA; n=15), and Severn Trent Water (STW; n=2). All samples were taken using a standardised 3-minute kick method, supplemented with a 1-minute hand search following EA (1997) procedures during the period March to May 2011. Samples were then sorted, macroinvertebrates counted and identified to species level where possible and subsequently subjected to an Analytical Quality Control procedure (EA, 1997) to ensure correct identification. Only upland sites (>150m aod) were sampled because, in the UK, 80% of large dams are in upland areas (Petts, 1988) and macroinvertebrate response to regulation has been found to vary between the uplands and lowlands (e.g. Armitage, 1978). Prior to data analyses, macroinvertebrate communities from each site were summarised using the indices listed in Table 8.1.

8.3.2 Study sites

All sites were within the drainage basin of the River Humber in north-east England at altitudes of between 150 and 600 m above ordnance datum (aod) (Figure 8.1). The sites had drainage areas ranging from 0.8 to 221.7 km^2 and were, on average, predominantly covered by forest/ semi-natural vegetation (CORINE, 2010) and had sand/ silt/ mud-stone surface geology (BGS, 2012) (Table 8.2). Under EU WFD classification, none of the water bodies in which sites were located were classed as *bad* or *poor* for either macroinvertebrates or overall physical chemistry and the percentage of *high*, *good* and *moderate* classifications were similar between fully-, semi- and un-regulated site types (Table 8.3) (EA, 2014).

Table 8.1: Macroinvertebrate indices calculated for each site.

Index	Notes
Taxonomic richness	Total number of taxa per sample
1/ Simpson's diversity index	$1/S=1/\frac{\sum n_i(n_i-1)}{N(N-1)}$ where n is the number of individuals of taxon i and N is total number of individuals in a sample (e.g. Ramchunder et al., 2012)
Taxonomic dominance	Berger-Parker index (D): $D=N_{max}/N$ where N_{max} is the number of individuals and N is total abundance in each sample (e.g. Ramchunder et al., 2012)
LIFE	Species level – calculated using ASTERICS (v 3.3.1) (AQEM, 2011)
PSI	Species level according to Extence et al. (2013)
BMWP	Calculated using ASTERICS (v 3.3.1) (AQEM, 2011)
ASPT	Calculated using ASTERICS (v 3.3.1) (AQEM, 2011)
% relative abundance of Coleoptera; Diptera; Ephemeroptera; Plecoptera; Trichoptera and other	Selected to be comparable with previous research (e.g. Armitage, 1978)

Table 8.2: Mean % land cover and geology types for drainage basins of all sites. Sources: CORINE (2010) and BGS (2012).

	Mean % for drainage basins of all sites (St. dev.)
Land cover type	
Artificial surfaces	1.2 (3.7)
Agriculture	35.9 (31.5)
Forests and semi-natural	45.3 (27.5)
Wetlands	15.9 (19.0)
Water bodies	1.8 (3.0)
Geology type	
Sand/ silt/ mud-stone	64.6 (28.9)
Limestone	5.5 (17.7)
Clay	0.3 (1.0)
Peat	29.2 (27.7)
Igneous	0.1 (0.5)
Unknown	0.4 (1.5)

Table 8.3: Percent of EU WFD macroinvertebrate and physical chemistry (PC) classifications for water bodies of sites within this study (source: EA, 2014). Note: where data were not available (n=9 sites), data from the adjacent downstream water body were used.

	% High		% Good		% Moderate	
	Macroinvertebrates	PC	Macroinvertebrates	PC	Macroinvertebrates	PC
Fully-regulated	21.05	47.37	52.63	36.84	26.32	15.79
Semi-regulated	32.14	35.71	46.43	57.14	21.43	7.14
Unregulated	29.41	47.06	58.82	35.29	11.76	17.65

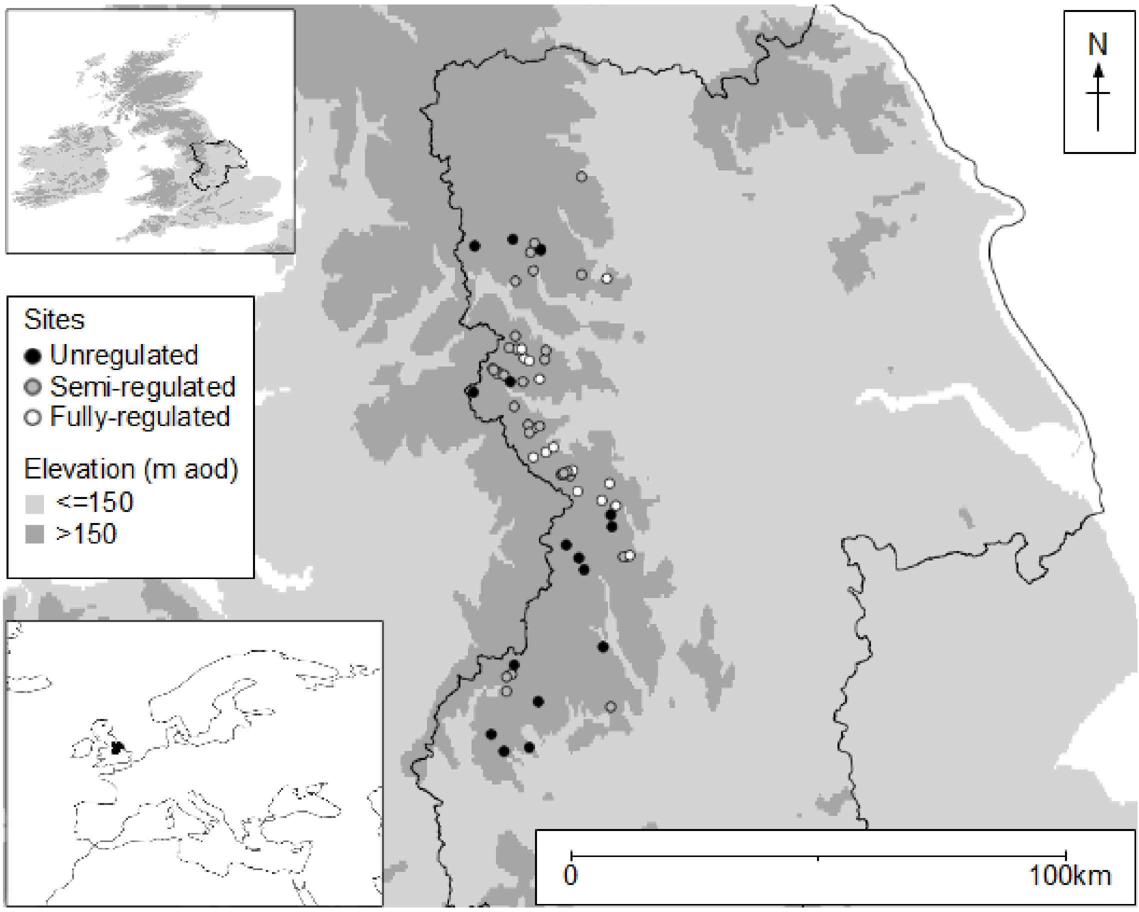

Figure 8.1: Location of study sites used within this chapter. Sources: (USGS, 1998; EA, 2011; NE, 2012).

8.3.3 Descriptive data

To aid explanation of any differences observed in macroinvertebrate communities, the following information was collated for each site:

(1) Quantification of the extent of regulation (ER) was undertaken using three methods of varying resolution (low, medium and high): (a) ER_{LOW}: differentiation between regulated (at least one reservoir upstream of site) and unregulated (no upstream reservoirs) – i.e. a binary approach; (b) ER_{MED}: three categorical groups: (i) fully-regulated (reservoir upstream of site; no tributary influence), (ii) semi-regulated (reservoir upstream of site; unregulated tributary influence) or (iii) unregulated (no reservoir upstream of site); (c) ER_{HIGH}: continuous score, Index of Regulation (IR) from 0 (fully-unregulated) to 1 (fully-regulated) based on both the number and size (Strahler, 1957) of tributaries. If a stream on which a site was located was neither fully-unregulated or fully-regulated (IR of 0 or 1, respectively), IR was calculated according to Equation 7.1 where *k* is the stream segment downstream of a confluence, *SO* is the stream order of each segment, *i* is the segment originating from a reservoir and *j* is the adjoining

tributary. Calculation of IR was repeated for each confluence between a site and the nearest upstream reservoir and the final IR_k used to classify each site (see Figure 8.2 for worked example of IR calculation).

$$IR_k = \frac{SO_i}{SO_{i+j}} \times IR_i + \frac{SO_j}{SO_{i+j}} \times IR_j \qquad \text{(Equation 7.1)}$$

Tributary influence was defined as any tributary joining the stream segment between any reservoir and the site location on a 1:50,000 Ordnance Survey map (OS, 2012) and a reservoir was defined as a body of water held back by a dam listed in the British Register of Dams (Tedd and Hoton, 1994).

(2) Percentage land cover, surface geology and catchment size for each site were determined using a Geographic Information System (GIS) (CORINE, 2010; BGS, 2012; OS, 2012). Prior to calculation of percentage cover, surface geology classification types were grouped into six categories (see Table 8.2).

(3) Altitude was calculated from a Digital Elevation Model (DEM) (USGS, 1998).

8.4 Data analysis

Prior to analyses, due to a high degree of colinearity between environmental data (points (2) and (3) above), Principal Component Analysis (PCA) was conducted to condense the variables based on their correlative structure (e.g. Malmqvist & Hoffsten, 1999). In addition to the environmental factors detailed, survey date and data source were both included in the PCA to mitigate for these potential confounding effects. Principal components (PCs) were retained which together explained at least 80 % of the cumulative variance.

Univariate analyses

First, to assess the potential impact of regulation on macroinvertebrate indices and the relative sensitivity of each ER resolution, least squares regression (Pinheiro et al., 2013) was undertaken with indices as response variables and ER and PCs as explanatory variables:
$I = \alpha + \beta\, ER + \beta\, PC + \varepsilon$ where I = univariate biotic index, ER = extent of regulation, PC = principal component (the number included in each model was determined as described above), α = regression intercept, β = regression coefficient and ε = model error. For each index and ER resolution combination, Ordinary Least Squares (OLS) models were compared against Generalized Least Squares (GLS) models with spatial correlation structures using Akaike

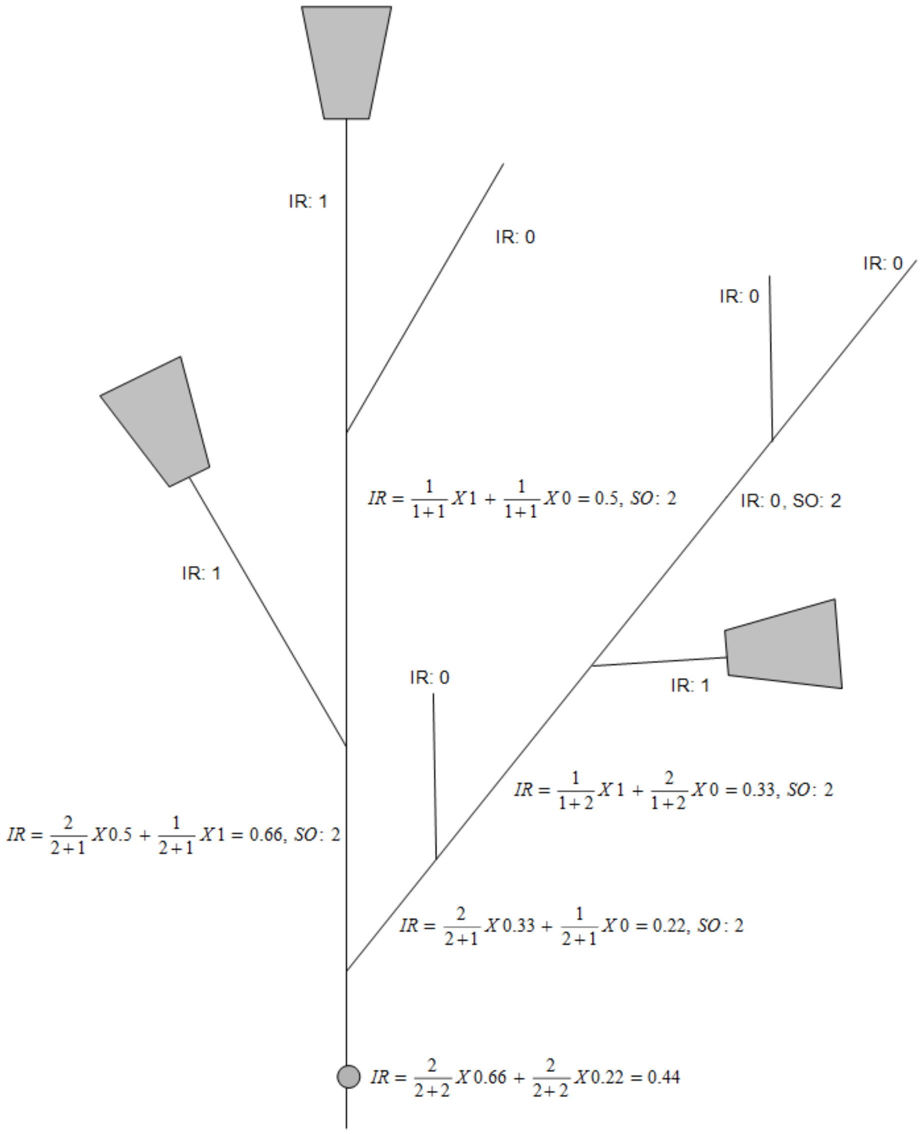

Figure 8.2: Worked example for calculation of IR for site marked by filled circle. Reservoirs are represented by filled polygons and stream reaches are marked by lines. IR = Index of Regulation; SO = stream order (only shown if SO>1).

Information Criterion (AIC) and variograms (Zuur et al., 2009). Next, optimal models were checked to ensure approximate normality, independence and homogeneity of residuals (Zuur et al., 2009) prior to implementation of ANOVA F-tests to assess the significance of *ER*.

Multivariate analyses

To further assess how macroinvertebrate community composition varied with regulation, multivariate analyses were undertaken as follows: First, to test for differences in taxonomic composition between the classification categories at both ER_{LOW} and ER_{MED} resolutions, ANOSIM was conducted on a Bray-Curtis dissimilarity matrix of arcsin square-root transformed relative abundance data (Oksanen et al., 2013). ANOSIM requires categorical grouping of variables and therefore could not be undertaken for ER_{HIGH}. Second, to test for associations between ER resolutions and specific taxa, Redundancy Analysis (RDA) on arcsin square-root transformed relative abundance data was undertaken. RDA was appropriate as initial Detrended Correspondence Analysis (DCA) revealed that community variation was within 3.5 standard deviations and thus a linear ordination method (e.g. RDA) should be used (Lepš and Šmilauer, 2003; Borcard et al., 2011). Initially, PCs that explained at least 80% of cumulative variance in environmental variables were included in each model and final selection of explanatory variables was undertaken using a stepwise algorithm based on AIC (Zuur et al., 2009). All statistical analyses were undertaken in R (v 2.15.3) (R, 2012) and results deemed significant where $p < 0.05$.

8.5 Results

Three PCs explained >80 % of cumulative variance in environmental factors and were retained for use in subsequent models. PC1, 2 and 3 accounted for 46, 21 and 13 % of total variance respectively (Table 8.4). PC1 was associated most strongly with altitude and reflected the transition from sand/ silt/ mud-stone geology to peat/ wetland. PC2 and PC3 were most strongly associated with catchment size and forest/ semi-natural land cover respectively.

Univariate analyses

LIFE scores and % Coleoptera both had significant negative relationships with $ER_{LOW,\ MED\ \&\ HIGH}$ (Table 8.5). Percent Ephemeroptera was observed to have a significant negative relationship with $ER_{LOW\ \&\ HIGH}$. Conversely, % others and % Trichoptera had significant positive relationships with $ER_{LOW,\ MED\ \&\ HIGH}$ and ER_{HIGH}, respectively. Mean taxonomic richness, Simpson's Diversity Index (1/S), PSI, BMWP and ASPT scores were highest at unregulated sites and mean taxonomic dominance was highest at fully-regulated sites, but none of these differences were statistically significant.

Multivariate analyses

ANOSIM revealed significant differences in macroinvertebrate community composition between ER categories using both ER_{LOW} ($R=0.32$, $p<0.01$) and ER_{MED} ($R=0.17$, $p<0.01$) resolutions. $ER_{LOW,\ MED\ \&\ HIGH}$ were significant factors within the RDA models but explained only 3.39, 3.62 and 4.07 % of total variance, respectively (Table 8.6). RDA axis 2 was negatively associated with ER_{HIGH} (Figure 8.3) which had a positive association with *Hydropsyche siltalai,* Chironomidae*, Amphinemura sulcicollis,* Oligochaeta and *Potamopyrgus antipodarum* and a negative association with *Rhithrogena semicolorata, Brachyptera risi, Limnius volckmari* and *Elmis aenea* (Figure 8.4).

Table 8.4: Loading scores for each environmental variable explaining >80% of total variance in PCA.

	PC1	PC2	PC3
Variable			
Altitude	-0.56	0.52	-0.20
Catchment size	0.08	-0.56	0.23
Geology:			
Sand/ silt/ mud-stone	0.40	0.25	-0.41
Limestone	0.01	-0.23	0.02
Clay	0.01	-0.01	0.01
Peat	-0.42	-0.01	0.38
Igneous	0.00	0.00	0.00
Unknown	0.00	-0.01	0.00
Land cover:			
Artificial surfaces	0.02	0.01	0.02
Agriculture	0.47	0.35	0.28
Forests and semi-natural	-0.23	-0.39	-0.63
Wetlands	-0.26	0.03	0.33
Water bodies	0.00	0.00	0.00
Data source	-0.01	0.01	-0.01
Date	-0.08	0.16	-0.05
Standard deviation	55.43	37.77	30.07
Eigenvalues	3072.92	1426.25	904.44
Proportion of variance explained	0.46	0.21	0.13
Cumulative % variance explained	45.82	67.08	80.57

Table 8.5: Summary statistics for macroinvertebrate indices and test results. Where significant results were found, the direction of the relationship between each index and extent of regulation (ER) is indicated in parentheses.

	Total abundance (N)	**Taxonomic richness**	**1/Simpson's diversity index (1/S)**	**Taxonomic dominance (D)**	**LIFE**	**PSI**	**BMWP**	**ASPT**	**% Coleoptera**	**% Diptera**	**% Ephemeroptera**	**% Plecoptera**	**% Trichoptera**	**% other**
All sites														
Mean (St. dev.)	1200 (862)	42 (11)	6.70 (3.30)	0.34 (0.14)	7.70 (0.35)	79.71 (10.39)	148 (32)	6.39 (0.50)	5.60 (6.25)	29.17 (17.66)	24.07 (18.32)	15.78 (16.93)	10.45 (8.47)	14.94
Min	146	20	1.40	0.13	6.95	51.06	81	4.50	0.11	1.97	0.38	0.00	0.46	0.92
Max	4001	80	18.10	0.86	8.40	92.41	212	7.38	26.75	86.03	79.27	88.06	45.68	63.18
Fully-regulated														
Mean (St. dev.)	1209 (842)	40 (14)	5.90 (2.40)	0.36 (0.16)	7.54 (0.36)	77.87 (8.99)	141 (33)	6.27 (0.46)	2.79 (3.62)	31.73 (20.02)	17.61 (17.24)	20.18 (19.87)	12.59 (11.95)	15.09 (13.78)
Min	152	21	1.40	0.21	7.00	59.52	85	5.62	0.11	6.58	0.38	0.43	0.46	1.06
Max	3465	80	11.50	0.86	8.33	88.89	205	7.38	14.47	86.03	57.28	57.89	45.68	53.51
Semi-regulated														
Mean (St. dev.)	1024 (620)	42 (8)	7.00 (2.70)	0.31 (0.10)	7.71 (0.30)	79.75 (11.27)	150 (30)	6.40 (0.56)	5.52 (5.23)	29.92 (15.16)	24.33 (14.82)	11.66 (9.98)	10.88 (6.13)	17.69 (14.16)
Min	226	30	3.30	0.17	6.95	51.06	81	4.50	0.16	6.21	0.44	0.00	3.19	2.11
Max	2960	56	13.80	0.52	8.11	92.41	207	7.08	19.44	57.79	56.18	35.86	29.02	48.77
Unregulated														
Mean (St. dev.)	1480 (1157)	43 (13)	7.20 (4.80)	0.35 (0.17)	7.83 (0.36)	81.69 (10.57)	151 (36)	6.51 (0.44)	8.84 (8.52)	25.05 (19.05)	30.87 (22.75)	17.63 (21.44)	7.36 (6.53)	10.24 (14.96)
Min	146	20	2.50	0.13	7.00	58.82	98	5.23	0.50	1.97	3.98	1.05	1.74	0.92
Max	4001	61	18.10	0.62	8.40	91.55	212	6.94	26.75	64.78	79.27	88.06	23.63	63.18
ER_{LOW}	0.65	0.80	0.55	1.16	6.63 * (-)	3.28	0.05	1.51	4.92 * (-)	1.04	12.05 ** (-)	0.00	3.19	48.74 ** (+)
ER_{MED}	2.00	0.58	0.54	0.35	9.05 ** (-)	2.67	0.49	2.44	7.79 ** (-)	0.05	2.95	1.08	3.52	30.52 ** (+)
ER_{HIGH}	0.40	0.54	1.11	0.83	8.22 ** (-)	2.15	0.15	2.42	7.70 ** (-)	0.26	4.05 * (-)	0.83	5.42 * (+)	19.62 ** (+)

$^*p < 0.05$, $^{**}p < 0.01$.

Table 8.6: Summary of RDA results for all ER resolutions.

ER	Total model variance	Total constrained variance	ER explained variance	% of total	% of constrained	F
LOW	0.50	0.05	0.02	3.39	33.23	2.38 **
MED	0.50	0.05	0.02	3.62	35.08	2.46 **
HIGH	0.50	0.05	0.02	4.07	37.72	2.79 **

**p < 0.05, **p < 0.01.*

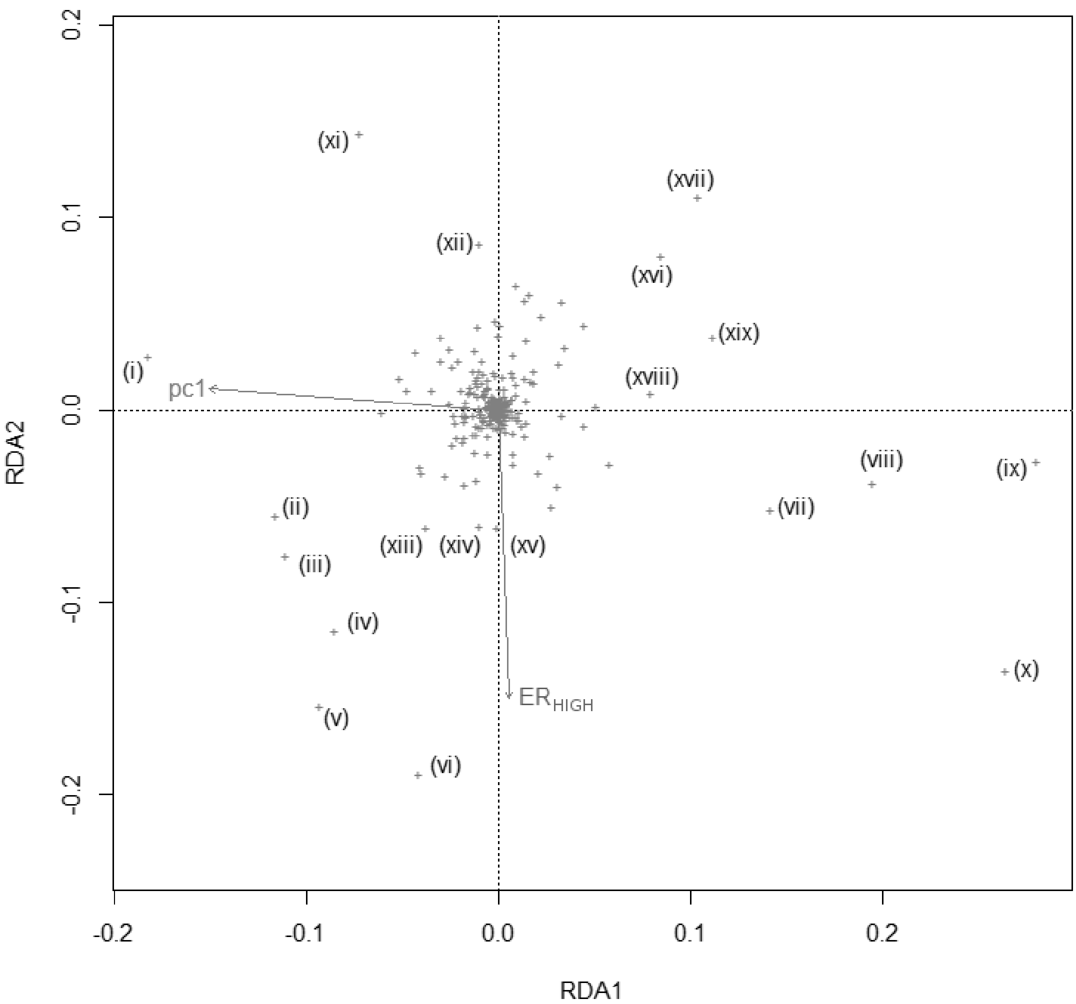

Figure 8.3: RDA biplot of taxa and significant factors (pc1 (principal component 1) and ER_{HIGH}). Selected taxa are labelled as follows: (i) Gammarus pulex (ii) Baetis rhodani (iii) Potamopyrgus antipodarum (iv) Oligochaeta (v) Chironomidae (vi) Hydropsyche siltalai (vii) Chloroperla torrentium (viii) Isoperla grammatica (ix) Leuctra inermis (x) Amphinemura sulcicollis (xi) Rhithrogena semicolorata (xii) Limnius volckmari (xiii) Pisidium sp. (xiv) Polycentropus flavomaculatus (xv) Polycelis felina (xvi) Elmis aenea (xvii) Brachyptera risi (xviii) Baetis scambus/fuscatus group (xix) Simuliidae.

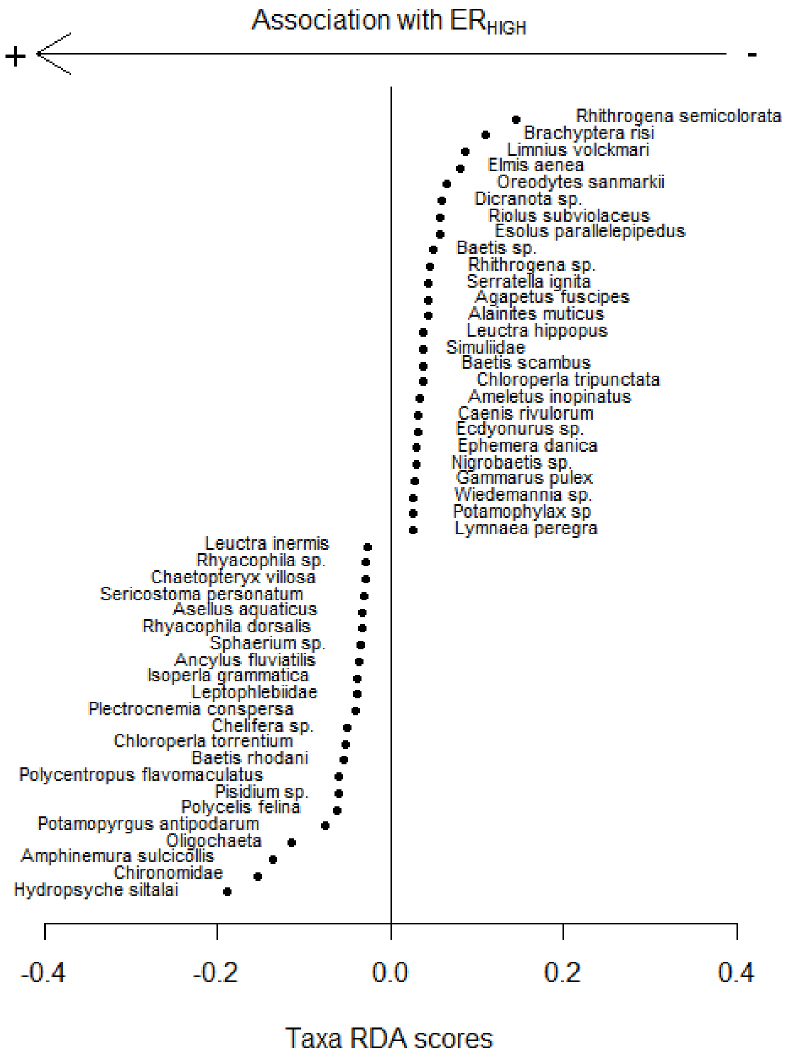

Figure 8.4: Taxa RDA scores highlighting association with ER_{HIGH}. Only scores > ±1 St. dev. from the mean are displayed for clarity.

8.6 Discussion

This chapter has identified a significant negative relationship between LIFE scores and regulation, whereas no significant relationships were found between diversity, dominance, PSI, BMWP or ASPT indices and regulation. ER_{HIGH} was identified as superior to both ER_{LOW} and $_{MED}$ in detecting that regulation was significantly associated with reduced relative abundance of Coleoptera and Ephemeroptera and enhanced relative abundance of Trichoptera, Chironomidae and Oligochaeta. The following discussion further explores these observations and is structured according to the three hypotheses.

It was hypothesised that macroinvertebrate indices and community composition would be affected by upstream impoundment and that responses would vary by taxon (H1). The findings of this study were in agreement with this hypothesis and are supported by those of Boon (1988), with both Coleoptera and Ephemeroptera negatively associated with regulation. Although Boon (1988) and Armitage (1989) found that responses of Trichoptera to regulation were highly variable, this study found a consistent significant increase in their relative abundance at regulated sites. This study also found that traditional macroinvertebrate indices (i.e. BMWP; ASPT) and other measures of community structure were not sensitive to the effects of regulation which is supported by Armitage (1989). These findings suggest that the use of traditional biomonitoring approaches may be unsuitable for studying the impacts of stream regulation and that a targeted flow sensitive focus on community composition is potentially a more effective method of assessment.

Both ANOSIM and RDA revealed significant differences in taxonomic composition between sites of differing ER designations and elucidated taxa which were positively or negatively associated with extent of regulation. This study found that two Coleoptera (*E. aenea* and *L. volckmari*) were negatively associated with regulation; an observation that has been made elsewhere in the UK (e.g. Inverarity et al., 1983). The drivers of this relationship remain unverified although it has been suggested that elevated concentrations of iron and manganese, often observed downstream of reservoirs (Petts, 1984a), may result in fewer Coleoptera (Inverarity et al., 1983). Alternatively, a reduction of high velocities associated with stream regulation (Petts, 1984a) may explain the observed reduction of rheophilic species such as *E. aenea* and *L. volckmari* (Schmedtje & Colling, 1996). The mayfly *R. semicolorata* was also identified as being negatively associated with extent of regulation; previous research has also identified this impact and linked it to both regulation-driven changes to flow and temperature (Brittain and Saltveit, 1989) and siltation (Boon, 1988).

This assessment identified a strong positive association between the net-spinning caddis, *Hydropsyche siltalai* and extent of regulation. Filter-feeding caddis such as *H. siltalai* have previously been found to proliferate downstream of reservoirs and have been associated with regulation-driven changes in flow regime and substrate stability (Boon, 1987). Conversely, reduced numbers of Hydropsychidae have been observed downstream of reservoirs (e.g. Inverarity et al., 1983), suggesting that site specific factors are key. The depth at which water is drawn from the reservoir may also be important for filter feeders through modification of plankton availability (Boon, 1988). In agreement with Boon (1988) and Armitage (1989) this study found a positive association between extent of regulation and both Chironomidae and Oligochaeta, potentially due to enhanced siltation (Armitage et al., 1987).

This study identified a positive association between extent of regulation and the invasive snail

Potamopyrgus antipodarum. Upstream impoundment has been cited as a key factor in allowing for the establishment of invasive and non-native species globally (e.g. Stromberg et al., 2007), including *P. antipodarum* (e.g. Cross et al., 2010). However, association between *P. antipodarum* and upstream impoundment in the UK has not previously been documented. *P. antipodarum* are associated with "slow/ sluggish" prevailing flow conditions (Extence et al., 1999) thus, the observations made in this assessment may be explained by a reduction in high flow events associated with upstream impoundment (Petts, 1984a).

Negative impacts to Perlodidae and Chloroperlodidae have previously been highlighted by Boon (1988) but not identified by Armitage (1989). In agreement with Armitage (1989), no evidence of these impacts was found in this assessment. However, in agreement with Boon (1988) a negative association between extent of regulation and *Brachyptera risi* was found and a positive association between extent of regulation and *Amphinemura sulcicollis* was also identified, but this has not previously been observed across sites in the UK. *A. sulcicollis* had a strong negative association with PC1 (which was negatively correlated with altitude) suggesting that the impact of reservoirs on *A. sulcicollis* is associated with sites of higher altitude. This may potentially be due to regulation driven reduction in temperature variation (Petts, 1984a), thus allowing for range expansion of stenothermic species such as *A. sulcicollis* (Graf et al., 2009). *A. sulcicollis* is also associated with algae presence (Clifford et al., 1992) which may also explain this observation as increased algae as a result of regulation in some UK streams has been observed (Bass & Armitage, 1987). However, further research would be required to confirm this theory for the study area.

In the absence of discharge data to directly quantify the impact of regulation, assumptions must be made to estimate this effect. This assessment compared three site classification approaches of varying resolution of regulation: two categorical classifications (ER_{LOW} and $_{MED}$) and a novel continuous index (ER_{HIGH}). It was hypothesised that the continuous index would be more sensitive to detecting impacts of regulation compared to the categorical classifications (H2). ER_{LOW} and $_{HIGH}$ revealed significant impacts to % Ephemeroptera, whereas ER_{MED} failed to identify this impact. In addition, ER_{HIGH} was the only resolution to reveal significant impacts to % Trichoptera suggesting it was the most effective approach for identification of impacts. This was further evidenced through multivariate modelling of community composition where all resolution types were significant but ER_{HIGH} marginally explained the most variance cf. $ER_{LOW\ \&\ MED}$. Thus ER_{HIGH}, or, Index of Regulation (IR), in agreement with the hypothesis, appears to be more sensitive to detecting impacts of regulation and thus an improvement on classifications used in previous research. However, it is important to note that only 4% of variance in community composition was explained by ER_{HIGH} highlighting the limited magnitude of impact associated with this measure of extent of regulation. This may reflect minimal impact of regulation, or insensitivity of IR; thus, development of IR (such as catchment area or discharge weighting of scores) should be a priority for similar research to ensure optimal sensitivity.

IR used the Strahler (1957) system to define the relative size of the extent to which a regulated or unregulated stream influenced a site downstream (hereinafter termed IR_S). Strahler classification was calculated using a 1:50,000 Ordnance Survey map (OS, 2012) and an obvious potential improvement would be to use a finer scale map (e.g. 1:10,000). Additionally, relative size of influence of a tributary could also be defined by drainage area so that:

$$IR_{D(k)} = \frac{DA_i}{DA_{i+j}}$$

where IR_D is the Index of Regulation calculated by drainage area for site k, DA = sum of upstream regulated (i) and unregulated (j) drainage areas. See Figure 8.5 and Table 8.7 for examples of calculations for different sites within a catchment.

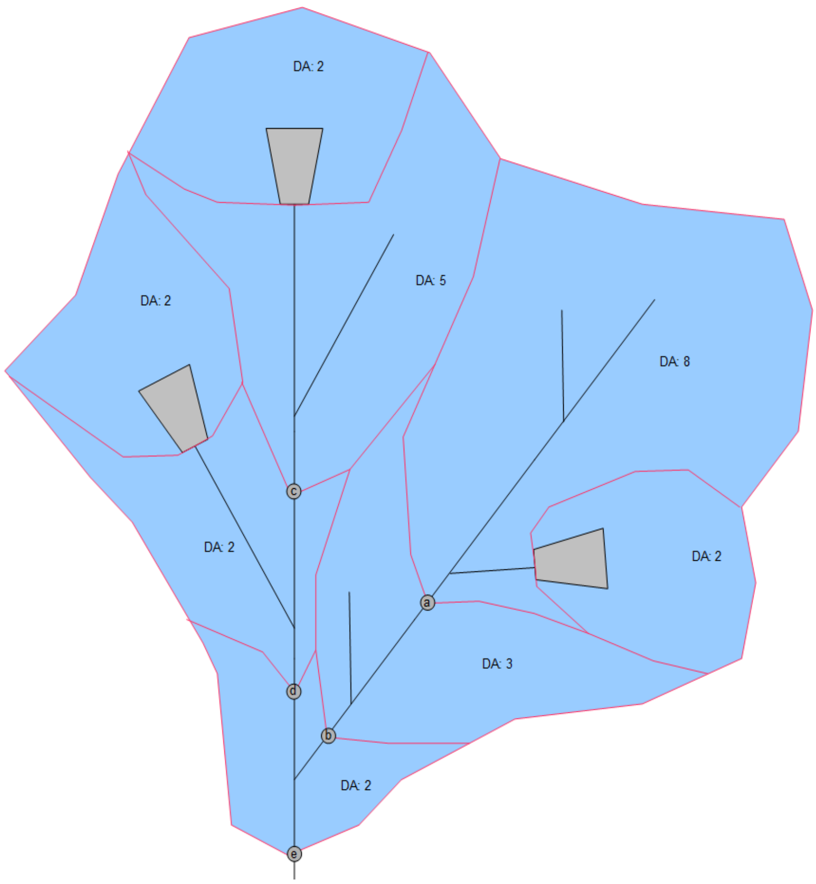

Figure 8.5: Example calculation of IR_D. Sites are marked in circles, reservoirs as polygons, streams as black lines and drainage areas (DA) in red. The size of each DA is also noted. Note: drawing not to scale.

Table 8.7: Calculation of IR_D based on example sites delineated in Figure 8.5.

Site	Calculation	IR_D
a	$\frac{2}{2+8}$	0.20
b	$\frac{2}{2+8+3}$	0.15
c	$\frac{2}{2+5}$	0.29
d	$\frac{2+2}{2+2+5+2}$	0.36
e	$\frac{2+2+2}{2+2+2+2+8+3+5+2+2}$	0.21

IR_D has the benefit of being simple to calculate and does not require detailed information regarding tributaries. Additionally, as it is calculated from drainage areas rather than tributary influence, it can potentially reflect accretion between a site and a reservoir, even if no tributary is delineated on a map. In this sense it may be more ecologically relevant than $IR_{S,}$ but further research is required to test this approach.

In the event that discharge data for a regulated stream network are available, it may be possible to calculate an IR score to more accurately reflect the extent to which a site is affected by regulation. This method is hereinafter termed IR_Q. This method first involves calculation of an unaffected, or 'natural' stream discharge for the site, of which there are many potential methods to choose (e.g. Mattikalli et al., 1996; Nijssen et al., 2001; Coe & Burkett, 2004). Second, an index is calculated comparing unaffected and regulated stream discharge data which is then used as the IR_Q for that site.

Figure 8.6 shows example hydrographs for four scenarios of differing extent of regulation at one site. In panel A, both unaffected and regulated discharge are equal; panel B: peak discharges have been reduced throughout the year, particularly during summer; panel C: peak discharges have been reduced throughout the year and floods have been eliminated during spring and summer and, panel D: floods have been eliminated throughout the year. Essentially, the extent of regulation can be regarded to increase from panels A-D and IR_Q should therefore reflect this gradient.

Table 8.8 lists four potential indices that could be used to calculate IR_Q. Each index results in values between zero (no difference between unaffected and regulated hydrological regimes) and one (maximum difference between unaffected and regulated hydrological regimes). Prior to use of these indices, their utility and ecological relevance will need to be tested. In particular, how the indices respond to specific changes in hydrological regime of ecological significance, such as flood magnitude, duration, rate of change and timing (Poff et al., 1997). It is also important to note that IR_Q can only be calculated retrospectively and that it does not take factors such as height at which water is drawn from the reservoir (which is known to be ecologically significant (e.g. Macdonald et al., 2012)) into account.

However, IR_Q has three key attributes of note: firstly, IR_Q could be built into either IR_S or IR_D to potentially improve accuracy of gradation, and secondly, IR_Q could be used by stream managers to prioritise resources. For example, a high IR_Q represents a large deviation from an unaffected regime, it may therefore be prudent to prioritise such sites for future management. Finally, IR_Q has the potential for temporal dynamism (i.e. it can be updated as new discharge data become available). This attribute will enable informed adaptive management through periods of change, for example, under scenarios of future climate change where stream flows are likely to change (Arnell & Reynard, 1996) or prescribed regulated stream flow management. It is recommended that the application and utility of the IR methods described above are tested in future research and considered for use in applied, management situations.

It was hypothesised there would be a negative relationship between LIFE and extent of stream regulation (H3) and this was upheld. LIFE scores are primarily based on a taxon's flow association and have been shown to correlate with a number of flow regime descriptors, the strength of which depends on many site specific factors such as geology and altitude (Extence et al., 1999). Thus, without knowledge of these site specific factors, it is not possible to accurately infer the specific facets of a discharge regime with which LIFE scores are associated. Therefore, further research is required to enable elucidation of the key flow drivers of the observed association between LIFE and extent of regulation in this research. As hypothesised, LIFE was sensitive to observed associations between taxonomic composition and extent of regulation, whereas traditional biomonitoring indices (e.g. BMWP and ASPT) were not. BMWP and ASPT are primarily associated with organic chemical pollution of streams (Armitage et al., 1983) potentially explaining their apparent insensitivity to macroinvertebrate community changes associated with upland stream regulation. Physical-chemical changes are key responses to stream regulation, that can be both driven by, and independent of, changes to the flow regime (Petts, 1984a). Thus, the use of alternative indices (e.g. Walley Hawes Paisley and Trigg

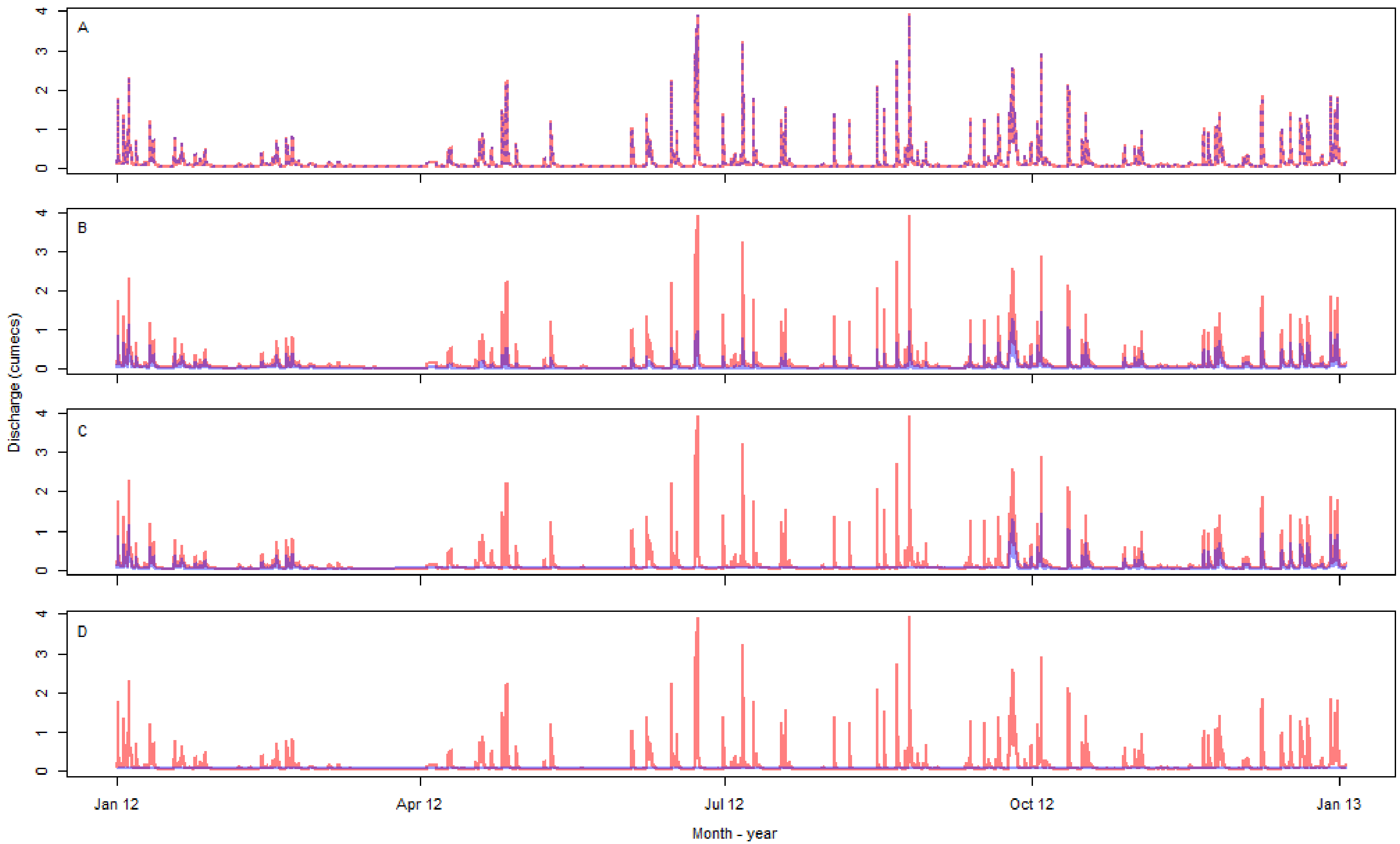

Figure 8.6: Simulated unaffected ('natural') (red) and regulated (blue) stream discharges for four scenarios (A-D) for Jan 2012 to Jan 2013 (panels A-D respectively) of increasing extent of regulation.

Table 8.8: Potential indices for calculation of IR_Q with respective formulas and results when tested on scenarios A-D.

Index	**Formula**	**Results**			
		A	**B**	**C**	**D**
1 - Ratio of Standard deviations (1 - rSD)	$rSD = \frac{\sigma_u}{\sigma_r}$	0.00	0.60	0.66	1.00
1- sample Pearson product-moment correlation coefficient (1 - r)	$r = \frac{\sum_{i=1}^{n}(u_i - \bar{u})(r_i - \bar{r})}{\sqrt{\sum_{i=1}^{n}(u_i - \bar{u})^2}\sqrt{\sum_{i=1}^{n}(r_i - \bar{r})^2}}$	0.00	0.06	0.33	1.00
1- sample Pearson product-moment correlation coefficient squared (1 - r^2)	r^2	0.00	0.12	0.55	1.00
1 – weighted r^2 (Krause et al., 2005) (1 - br^2)	$br^2 = \lvert b \rvert . r^2 \; for \; b \leq 1$ or $\lvert b \rvert^{-1} . r^2 \; for \; b > 1$	0.00	0.67	0.88	1.00

u = unaffected & *r* = regulated discharge data; *b* = gradient coefficient from linear regression model of *u~r*.

(WHPT); Davy-Bowker et al. (2008)) and potentially the development of a water-quality sensitive index for use specifically in regulated streams may be complimentary to the use of LIFE scores for future monitoring and management of these environments.

It was hypothesised that there would be a negative relationship between PSI and extent of stream regulation (H3), yet no evidence for this was found suggesting no clear effect of regulation on downstream fine bed sediment within the study area. This hypothesis is supported by Carling (1988) who noted inconsistent morphological effects of upstream regulation in UK streams due to site specific factors (e.g. sediment supply, geology). It is therefore suggest that the primary application of PSI may be in long term monitoring of regulated streams where a change in bed fine sediment over time may be detected, rather than assessing the impact of regulation *per se*. The relationship between PSI and fine sediment in regulated streams has been confirmed elsewhere (e.g. Extence et al., 2013), but evaluation of PSI performance in upland streams (where fine sediment abundance is typically low (Carling & Reader, 1982)) is lacking. Further evaluation of the sensitivity of PSI in upland streams is therefore recommended.

Macroinvertebrates respond quickly to environmental change, justifying their choice as environmental indicators (Metcalfe, 1989). However, this characteristic limits the temporal extrapolation of observations made over short time periods. Indeed, evidence of temporally dynamic response of macroinvertebrates to regulation has been found (Armitage, 2006). Whilst this research has identified impacts of regulation on macroinvertebrate communities, the analyses were conducted on data that were collected over a few months and thus extrapolation to longer time scales is likely to be confounded by temporal variation in environmental drivers and biota phenology. Future research should be carried out over longer time scales to address this limitation.

8.7 Summary

This study has identified key impacts of regulation on macroinvertebrate community composition in upland sites within a large British catchment, specifically: reduced relative abundance of Coleoptera (primarily driven by changes in relative abundance of *Elmis aenea* and *Limnius volckmari*) and Ephemeroptera (particularly *Rhithrogena semicolorata*) and enhanced relative numbers of Trichoptera (including *Hydropsyche siltalai*), Chironomidae and Oligochaeta. Positive associations between regulation and *Potamopyrgus antipodarum* and *Amphinemura sulcicollis* were identified which is a novel finding for the UK. These observations can be integrated into regional management plans concerned with management of regulated streams (e.g. the introduction of environmental flows) and should also direct future research towards further understanding the mechanisms behind the associations identified.

A continuous method of defining extent of regulation (IR), as opposed to categorical classifications, was most sensitive to identification of changes in community composition and thus appeared to be superior. IR is a useful approach for assessment of the impact of upstream impoundment on macroinvertebrates and its value should be tested in additional regions, both within the UK and globally, both in its current form and with additional catchment area (IR_D) and/or discharge weighting (IR_Q). Additionally, the potential for the application of IR in understanding the effect of regulation on stream physical chemistry could be explored. Finally, several established biomonitoring indices failed to detect differences in community composition, whereas a significant negative relationship between LIFE scores (Extence et al., 1999) and extent of regulation was identified. LIFE therefore appears to have potential for use in future regulated stream research and management.

9 TEMPORAL IMPACTS OF REGULATION AND ARTIFICIAL FLOODS ON MACROINVERTEBRATE ASSEMBLAGES

9.1 Chapter overview

This chapter presents a temporal (13 month) assessment of the impact of stream regulation on macroinvertebrate assemblages at a sub-catchment scale. An assessment of the impact of Artificial Floods on macroinvertebrate assemblage is also undertaken. First, the importance of understanding these issues is presented followed by an identification of current gaps in research and outline of the study aims. Next, the methods and analytical techniques used to undertake the study are detailed. This is followed by sections presenting and discussing the results, including recommendations for further research.

9.2 Introduction

Organisms have long been used to monitor environmental quality (Cairns & Pratt, 1993) and a preference for monitoring macroinvertebrates to achieve this aim in streams has long been established (Metcalfe, 1989). Macroinvertebrates are differentially sensitive to environmental pressures, ubiquitous, abundant and easy to collect and identify (Metcalfe, 1989) resulting in incorporation of their monitoring into contemporary freshwater legislation (e.g. the EU WFD (EC, 2000) and the US Clean Water Act (USC, 2002)). The monitoring of stream macroinvertebrates thus has the potential to assess impacts of anthropogenic pressures (e.g. Blasius & Merritt, 2002; Fritz et al., 2011; Mercer et al., 2013) and management interventions (e.g. Friberg et al., 1998; Nakano & Nakamura, 2006).

One such potential pressure is stream regulation, and because approximately 15% of the world's stream flow can be impounded by dams (Nilsson et al., 2005), the monitoring of macroinvertebrates in regulated streams is particularly important. The impact of regulation on downstream macroinvertebrates has been the focus of many studies worldwide, yet varied observations have been made leading to a lack of clarity in understanding. Total abundance has been recorded to increase at some sites (e.g. Armitage, 1978; Munn & Brusven, 1991; Benítez-Mora & Carmargo, 2014) but decrease at others (Englund & Malmqvist, 1996). Additionally, responses of some taxonomic groups have been observed to vary; for example, no change or decrease in abundance of Coleoptera (Spence & Hynes, 1971 and Nichols et al., 2006, respectively). Furthermore, varied responses in measures of α-diversity (Whittaker, 1972) have also been observed (Table 9.1). However, some of these studies have been of limited sampling intensity (i.e. ≤ 2 samples per year (e.g. Englund & Malmqvist (1996); Maynard & Lane (2012); Gillespie et al., in press (a))) potentially resulting in failure to identify fine-scale temporal variability of impacts. This may partly explain the wide variation in observations because impacts are likely to vary both intra- and inter-annually (Armitage, 1978). An assessment of the impact of regulation on intra-annual macroinvertebrate community dynamics at a sub-seasonal scale is yet to be undertaken globally; such a study may therefore have the potential to reveal novel insights into impacts of regulation on downstream macroinvertebrate assemblages.

Table 9.1: Varied response in α-diversity of stream macroinvertebrates to upstream regulation.

Response type	Publications
Increase	Poole and Stewart, 1976; Penaz et al., 1968; Maynard and Lane, 2012
No change	Gillespie et al., in press (a)
Decrease	Pearson et al., 1968; Armitage 1978; Munn & Brusven, 1991; Benítez-Mora & Carmargo, 2014

Artificial Floods (AFs) have been suggested as a tool to mitigate impacts associated with regulation (Acreman & Ferguson, 2010). To date, 18 publications have reported on macroinvertebrate responses to AFs (Gillespie et al., in press (b)) and this literature has revealed a general consensus in response: macroinvertebrate abundance, richness and diversity typically decrease post-flood (e.g. Pardo et al., 1998; Harby et al., 2001; Cereghino et al., 2004; Robinson et al., 2004a; Mannes et al., 2008) with reductions in certain taxonomic groups such as Ephemeroptera (Harby et al., 2001; Mannes et al., 2008). Furthermore, Mannes et al. (2008) stated that multiple AFs resulted in a more disturbance resistant assemblage. An assessment of the impact of AFs on stream macroinvertebrates in the UK is yet to be published despite the potential for AFs to contribute towards attainment of EU WFD targets in the EU (Acreman & Ferguson, 2010).

This study aimed to (i) conduct an assessment of intra-annual temporal dynamics (monthly sampling) of the impact of regulation and, (ii) examine the impact of AFs on downstream benthic macroinvertebrates in the UK. It was hypothesised that (H1) a difference in macroinvertebrate assemblages between regulated and unregulated sites could be observed in line with previous studies, (H2) the difference would vary intra-annually reflecting taxon life-cycle attributes and environmental preferences, (H3) macroinvertebrate abundance, richness and diversity would decrease as a result of AFs and (H4) taxa would respond to AFs based on their specific environmental preferences resulting in a more disturbance resilient assemblage.

9.3 Methods

9.3.1 Impact of regulation

To assess the impact of regulation on intra-annual dynamics of benthic macroinvertebrate populations, five replicate $0.05m^2$ Surber samples (mesh size: 250µm) were collected monthly from January 2012 to January 2013 inclusive from one unregulated reference site (U) and two regulated sites (R1 & R2) (sites 9, 7 and 12 respectively). Samples were taken randomly from riffle/ run habitat to ensure inter-site comparability. All samples were immediately preserved in 70% methanol. Samples were sieved (mesh size: 250µm) to remove fine particles to aid sorting which was undertaken using Protocol P3 without subsampling as described by Stark et al. (2001). Next, where possible, individuals were identified to species level using a light microscope (x50 magnification). However, the following taxa were identified to the following levels: Chironomidae: family; Oligochaeta: sub-class and Sphaeriidae: genus which enabled comparability with the majority of relevant literature. As a means of quality control, taxa identification was confirmed by UKAS accredited staff at APEM Ltd aquatic science consultancy. A list of keys used and taxa identified during this study can be found in Appendix D.

To provide hydrological context for the study period, discharge at sites 6, 9 and 11 (recorded and calculated according to the methods described in Chapter 5) was used to represent discharge dynamics at sites R1, U and R2 respectively.

9.3.2 Impact of Artificial Floods

To assess the impact of AFs on benthic macroinvertebrate populations, five replicate $0.05m^2$ Surber samples (mesh size: 250µm) were taken within 48 hours before and after each AF (described in Chapter 5). The first AF was carried out on the 20^{th} August 2013 and thus these AFs did not confound the temporal assessment described in section 9.3.1. Samples were collected randomly within riffle/ run habitat at sites R1 and R2. All samples were subsequently analysed using the same method as described in section 9.3.1. This sampling methodology followed a Paired Before-After-Control-Impact (BACIP) framework (Smith, 2002). AFs were conducted from reservoir C, thus, site R1 was classed as 'Impact' and site R2 used as 'Control'.

9.4 Data analysis

Prior to data analysis, macroinvertebrate taxonomic richness and dominance, 1/Simpson's diversity index, BMWP, ASPT and LIFE were calculated as described in Chapter 8 for each Surber sample. In addition, 'global' β-diversity (Whittaker, 1972) was calculated according to Brown et al. (2007): average taxonomic richness calculated from replicate samples divided by pooled sample taxonomic richness; lower values represent higher β-diversity. Furthermore, total number of individuals, Coleoptera, Diptera, Ephemeroptera, Plecoptera and Trichoptera were estimated per m^2.

To test hypothesis (a), community indices were pooled by site type: *regulated* and *unregulated*. GLMM or GAMM (Wood, 2011) (selection based on AIC) was then used to test for statistical differences in indices between site types. GLMMs and GAMMs were formulated according to equations 8.1 and 8.2 respectively where I = macroinvertebrate community index, *type* = site type (regulated or unregulated), *s(day)* = sample date fitted as a smoother, α = regression intercept, β = regression coefficient and ε = error term.

$$I = \alpha + \beta_{type} + \varepsilon \qquad \text{Equation 8.1}$$

$$I = \alpha + \beta_{type} + s(day) + \varepsilon \qquad \text{Equation 8.2}$$

Samples were taken repeatedly from sites and replicates were taken from within each site on each sampling occasion. To avoid problems associated with lack of independence between replicates and sampling dates, both *site* and *replicate* were included within each model as random factors. *replicate* was nested within *site* and *site* was nested within *type* to reflect the experimental design (after Crawley, 2002 and Zuur et al., 2009). Appropriate error distribution, link functions and correlation structures were specified to ensure approximate normal distribution, homogeneity and independence of residuals. Finally, significance of *type* was assessed using t-statistics and associated p-values. To test for differences in macroinvertebrate community structure at each site, one-way analysis of similarity (ANOSIM) was conducted on a taxon abundance matrix constructed from means of Surber replicates taken each month at each site. Sites were grouped by site type and ANOSIM was undertaken using Bray-Curtis dissimilarities with 999 permutations (Oksansen et al., 2013).

To test hypothesis (b), plots of each index against sample date were produced to enable visualisation of temporal dynamics. Non-metric Multi-Dimensional Scaling (NMDS) was then undertaken on means of replicates taken each month at each site using Bray-Curtis dissimilarities of log(n+1) taxa abundance (Oksansen et al., 2013) to allow visualisation of the relative importance of taxa in defining populations throughout the study period at each site. Additionally, ANOSIM between site types for each month was undertaken on taxa abundance matrices of replicates as described above.

To test hypotheses (c) and (d), macroinvertebrate community indices for Surber samples collected at sites R1 and R2 for each sampling date before and after each AF were modelled using GLMM (Wood, 2013) according to the following formula: $I=\alpha+\beta\, site : period+\varepsilon$ where I , α , β and ε were as described above and *site:period* was the interaction term between factors *site* (R1 or R2) and *period* (before or after). As described previously, both *site* and *replicate* were included within each model as random factors with *replicate* nested within *site*. Appropriate error distribution, link functions and correlation structures were specified to ensure approximate normal distribution, homogeneity and independence of residuals. Finally, significance of the *site:period* interaction term was examined using t-statistics and p-values to provide an indication of whether an impact had occurred. To further assess the validity of hypotheses (c) and (d), composition of macroinvertebrate assemblages before and after each AF were visualised using the results of NMDS undertaken as described above. Moreover, to objectively test similarity of macroinvertebrate community composition before and after each AF under a BACI framework, *site:period* interaction terms were tested using multivariate analysis of variance (MANOVA) on taxon abundance matrices of replicates from each site using 999 permutations and Bray-Curtis dissimilarities (Oksansen et al., 2013).

9.5 Results

9.5.1 Impact of regulation

Discharge throughout the study period was generally similar in magnitude at sites representing R1 and U, but higher for R2. Floods at site U generally appeared to be of shorter duration and more 'flashy' that at R1 and 2. During the latter part of the study period in particular, floods that occurred at sites R1 and U did not occur at R2 (Figure 9.1).

Mean total, Coleoptera, Ephemeroptera and Trichoptera densities were highest for the regulated site type, and apart from total density, these differences were statistically significant. Mean total Diptera and Plecoptera densities were higher at the unregulated site, but these differences were not statistically significant. Similarly, non-significant differences between site types for mean β-diversity and ASPT were also observed. Mean values for taxonomic richness, 1/Simpson's diversity index and BMWP for the regulated site type were almost double ($p<0.05$) those observed for the unregulated site type. Mean taxonomic dominance and LIFE scores were significantly higher and lower respectively, for the unregulated cf. regulated site type (Table 9.2).

Mean Chironomidae density was the highest of any taxon at all sites for the study duration. The second and third most abundant taxa for sites R1 and U were *Leuctridae* spp. and *Baetidae* spp. for site R2. Notably, high mean densities of *Polycentropus flavomaculatus* and *Oulimnius* sp. were present at site R1, whereas none of these taxa were in the top ten most abundant taxa of sites U or R2. Site U had high mean densities of *Protonemura meyeri* which was not in the top ten most abundant taxa at either regulated site and, similarly, R2 had high numbers of *Isoperla grammatica* which was not present in the top ten most abundant taxa of the other two sites (Table 9.3).

Temporally, all sites displayed similar reductions in taxonomic dominance during winter and Plecoptera density and ASPT during mid-summer (Figures 9.2 & 9.3). During spring/ summer, LIFE scores and total Diptera and Ephemeroptera density peaked at all sites, but variation was observed between sites as, for example, total density at sites R1 and U peaked during April but not until June at site R2. β-diversity remained relatively constant throughout the study period at all sites. Conversely, dissimilarities in temporal trends between regulated and unregulated sites for taxonomic richness, 1/Simpson's diversity index and BMWP were observed. For regulated sites, clear reductions in these indices during summer occurred that were not evident at site U and these changes were reflected by non-significant community differences between site types during July and August (Table 9.4). Coleoptera and Trichoptera densities at site U were too low throughout the study period to compare to trends observed at sites R1 and R2 (Figures 9.2 & 9.3).

Axis 1 of the NMDS broadly revealed dissimilarity between each site: site U samples generally scored the lowest (range: -1 – 0.2), site R2 the highest (range: 0 – 0.75) and site R1 samples were generally between sites U and R2 (Figure 9.5). In January 2012, all sites had relatively low axis 1 scores, suggesting populations dominated by Leuctridae (site R1), Nemouridae (site U) and taxa such as *Amphinemura sulcicollis* and *Ecdyonurus* sp. (site R2). The highest axis 1 score for sites R1 and U was observed in July, but was highest at R2 two months later in September. Notably, at these times of the year, Ephemeroptera such as *Rhithrogena semicolorata* appeared to be an important descriptor of macroinvertebrate assemblage at site R2 and Simulidae and Psychodidae at sites U and R1 respectively. For each site, NMDS scores were approximately similar during January 2012 and 2013 (Figure 9.5). Holistically, no overlap of site U pathways (lines drawn between consecutive sample dates to interpolate temporal change in community composition) between sampling dates was observed, but overlap was observed between pathways of sites R1 and R2 (Figure 9.5), indeed, ANOSIM revealed a significant difference between taxonomic composition of the two site types (r statistic: 0.36, $p < 0.01$).

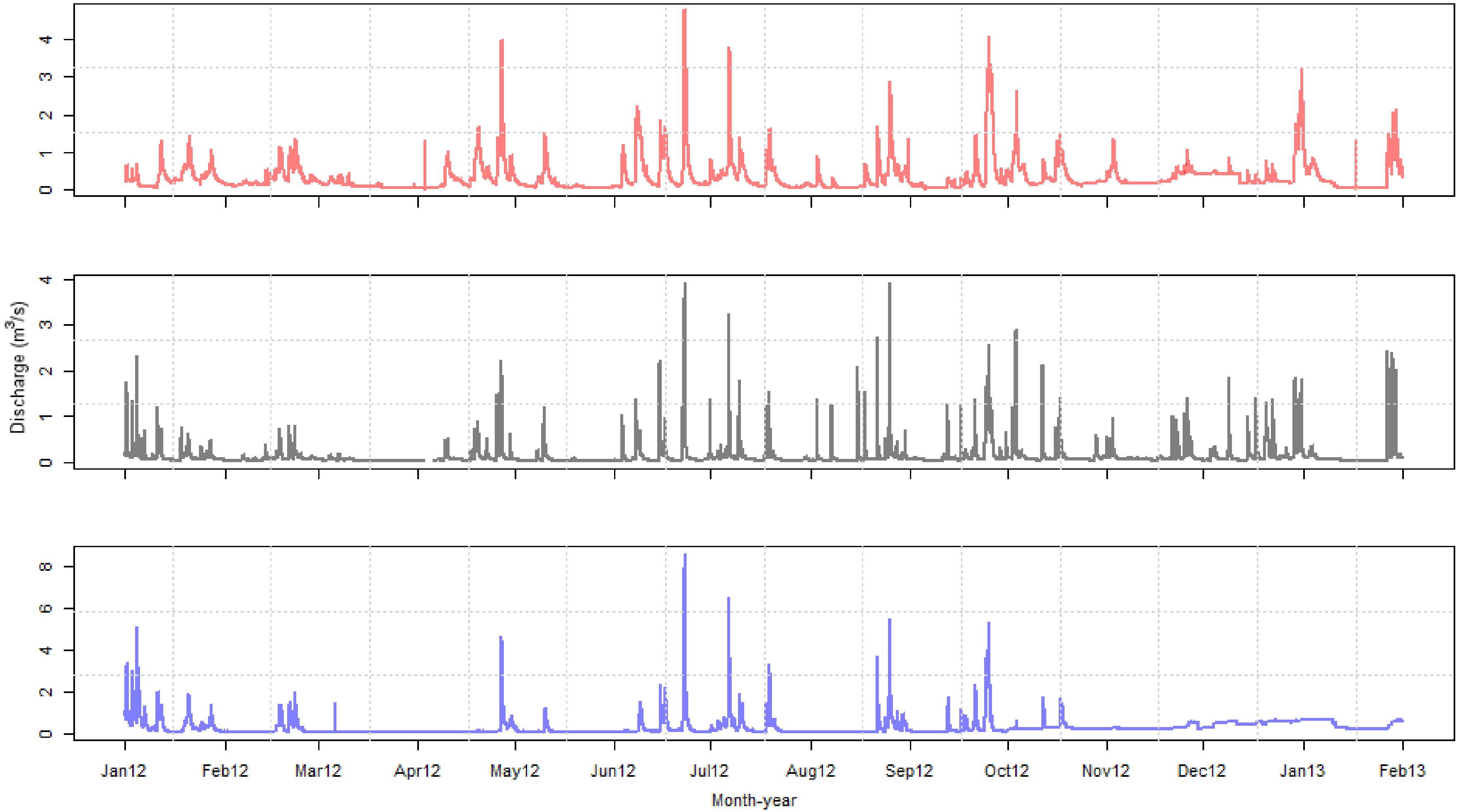

Figure 9.1: Discharge at sites R1, U and R2 (red, black and blue lines respectively) for the study period prior to Artificial Floods.

Table 9.2: Mean macroinvertebrate indices for unregulated and regulated site types for the study period. Model type and statistics are also delineated.

	Mean	St. dev.	Est.	t
Total density (ind. / m²)				
Unregulated	796	558	0.91	0.91
Regulated	949	630		GLMM
Taxonomic richness				
Unregulated	6	1	2.00	530 **
Regulated	10	3		GAMM
1/Simpson's diversity index (1/S)				
Unregulated	2.83	0.78	2.59	5.87 **
Regulated	5.39	1.73		GAMM
β -diversity				
Unregulated	0.46	0.07	0.04	1.68
Regulated	0.49	0.07		GLMM
Taxonomic dominance (D)				
Unregulated	0.55	0.12	-0.62	-5.53 **
Regulated	0.35	0.13		GAMM
BMWP				
Unregulated	26	4	23	4.61 **
Regulated	48	16		GAMM
ASPT				
Unregulated	5.91	0.51	0.35	1.72
Regulated	6.18	0.75		GAMM
LIFE				
Unregulated	7.54	0.25	0.42	3.39 **
Regulated	7.95	0.26		GLMM
Coleoptera (ind. / m²)				
Unregulated	1	3	25	2.68 **
Regulated	56	84		GLMM
Diptera (ind. / m²)				
Unregulated	370	524	-1	-1.16
Regulated	212	251		GLMM
Ephemeroptera (ind. / m²)				
Unregulated	5	9	19	2.56 *
Regulated	201	319		GLMM
Plecoptera (ind. / m²)				
Unregulated	392	238	1	-1.14
Regulated	332	263		GAMM
Trichoptera (ind. / m²)				
Unregulated	13	11	8	4.57 **
Regulated	122	123		GLMM

$^*p < 0.05$, $^{**}p < 0.01$.

Table 9.3: Mean density of the ten most abundant taxa ('top ten') at each site for the study duration.

Site R1		**Site U**		**Site R2**	
Taxa	**Mean density (ind. / m^2)**	**Taxa**	**Mean density (ind. / m^2)**	**Taxa**	**Mean density (ind. / m^2)**
Chironomidae	163	Chironomidae	320	Chironomidae	194
Leuctra inermis	131	*Leuctra hippopus*	146	*Baetis rhodani*	155
Leuctra hippopus	90	*Leuctra inermis*	123	*Baetis* sp.	135
Polycentropus flavomaculatus	82	*Protonemura meyeri*	66	*Isoperla grammatica*	82
Oulimnius sp.	69	Simuliidae	18	*Amphinemura sulcicollis*	69
Amphinemura sulcicollis	65	*Amphinemura sulcicollis*	16	*Leuctra inermis*	52
Hydropsyche siltalai	62	*Nemoura cinerea*	16	*Siphonoperla torrentium*	37
Baetis rhodani	41	Oligochaeta	14	*Rhyacophila dorsalis*	32
Elmis aenea	29	*Dicranota* sp.	12	Simuliidae	32
Baetis sp.	27	Clinocerinae	10	*Leuctra hippopus*	31

Table 9.4: ANOSIM test result (R) and significance for similarity of macroinvertebrate communities at regulated and unregulated site types by month.

Month	**R**
January 2012	0.53 *
February	0.53 **
March	0.44 *
April	0.36 *
May	0.34 *
June	0.55 **
July	0.24
August	0.23
September	0.67 **
October	0.28 *
November	0.34 *
December	0.80 **
January 2013	0.30 *

*p < 0.05, **p < 0.01.

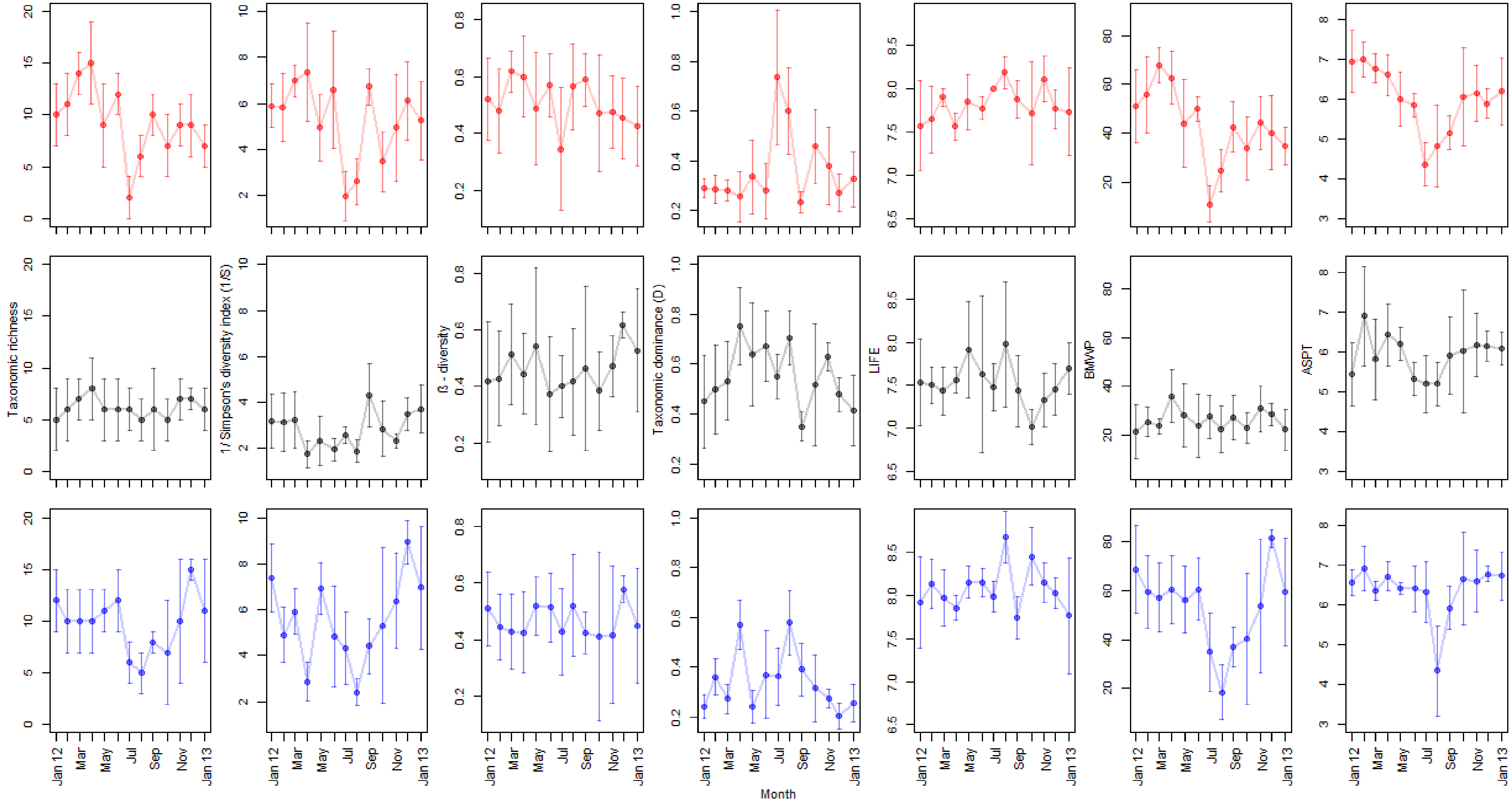

Figure 9.2: Temporal variation of macroinvertebrate diversity and biomonitoring indices at sites R1, U and R2 (red, black and blue respectively). Error bars represent ±1 standard deviation.

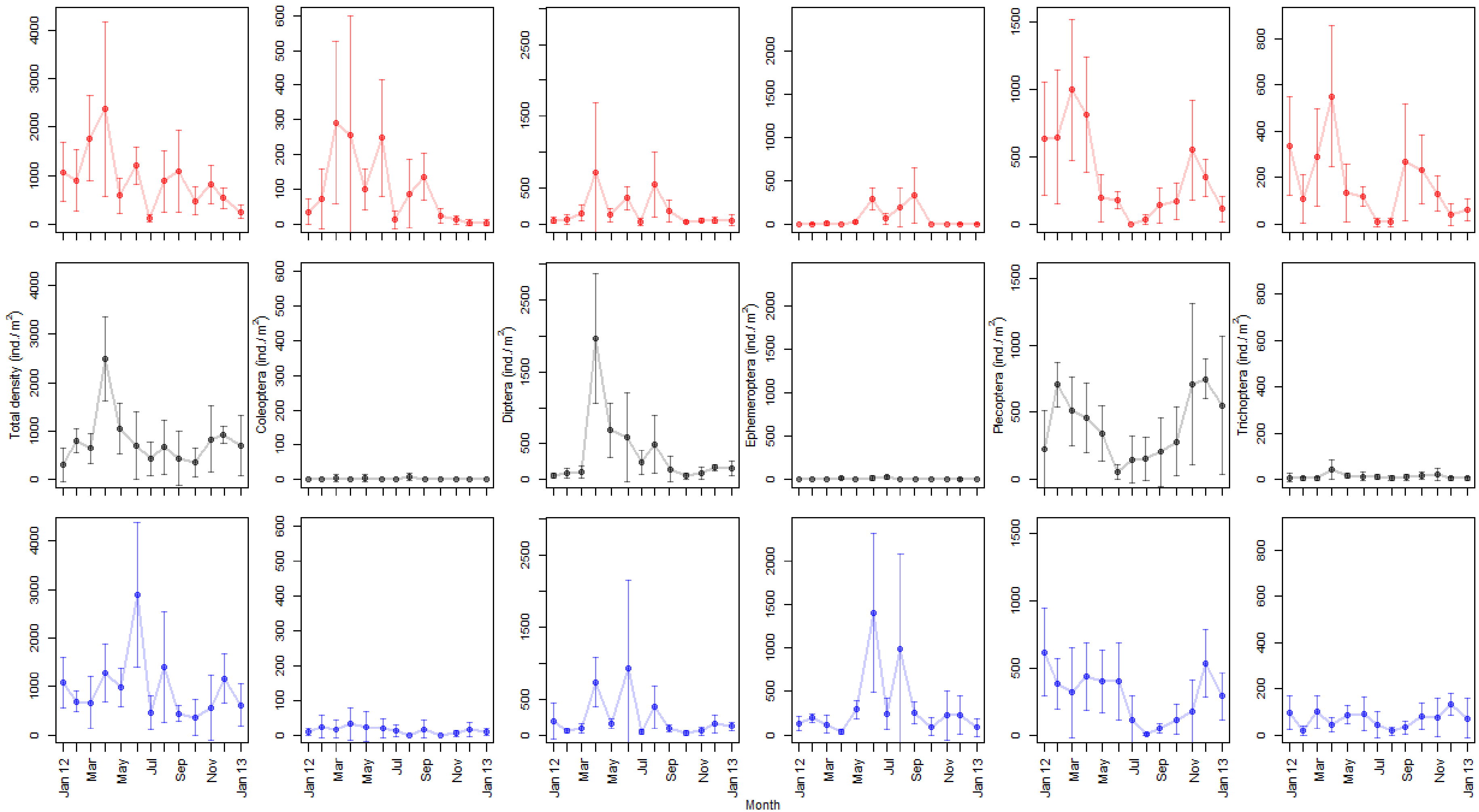

Figure 9.3: Temporal variation of macroinvertebrate density indices at sites R1, U and R2 (red, black and blue respectively). Error bars represent ±1 standard deviation.

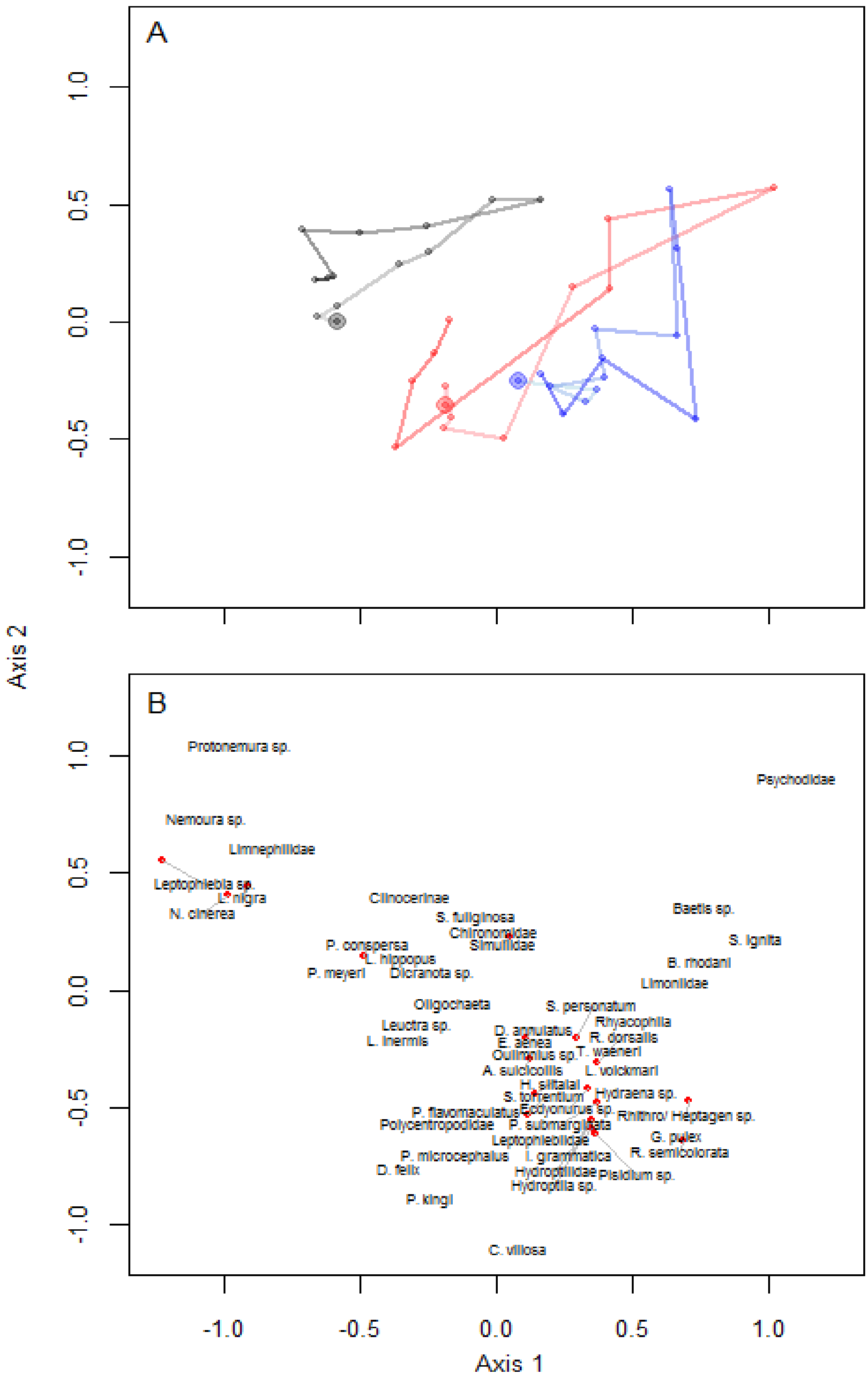

Figure 9.4: NMDS plot of mean monthly macroinvertebrate samples and temporal transition (panel A) for sites R1, U and R2 (red, black and blue respectively). Enlarged points indicate Jan 2012 and lines darken throughout the study period. Panel B: species NMDS scores. Note: taxa labels are abbreviated; full names can be found in Appendix D.

9.5.2 Impact of Artificial Floods

Non-significant site:period interaction terms were identified for models testing macroinvertebrate density, taxonomic richness, β-diversity, BMWP, ASPT, LIFE indices and densities of Coleoptera, Diptera and Trichoptera for all AFs. Conversely, the interaction term testing Plecoptera density was significant for AF1, 1/Simpson's diversity index was significant for AF2, dominance was significant for AF2 and Ephemeroptera density was significant for AF2 and 4 (Table 9.5). A reduction in mean Plecoptera density (-108 individuals/m^2) was observed after AF1 at site R1 but an increase (+232 individuals/m^2) was identified at site R2 for the same period. A similar pattern was seen for mean 1/Simpson's diversity index after AF2 (reduction (-1.56) at R1 and increase (+2.21) at R2). Conversely, mean taxonomic dominance increased (+0.09) at R1 and decreased (-0.14) at R2 after AF2. This pattern was repeated for mean Ephemeroptera density (+16 individuals/m^2 at R1 and -108 individuals/m^2 at R2). After AF4, mean taxonomic dominance was observed to decrease at both sites (-0.04 (R1), -0.02 (R2)) (Figures 9.5 & 9.6).

Throughout the period of AF implementation, NMDS axis 1 and 2 scores for site R1 were generally lower and higher respectively than for site R2 (Figure 9.7). Temporally, NMDS scores at both sites were characterised by generally increasing axis 1 scores throughout the series of AFs, but this increase was delayed at site R1 until after AF3. Magnitude of changes in composition after each AF were similar at both sites (Figure 9.7). Changes in NMDS scores at site R1 represented shifts from populations characterised by taxa such as *Rhyacophila dorsalis* and *Amphinemura sulcicollis* to Isoperla grammatica to *Plectrocnemia conspersa*. At site R2 macroinvertebrate assemblage was initially characterised by taxa such as Simuliidae and *Leuctra* sp.. Subsequently, species such as *Elmis aenea* and *Leuctra inermis* became increasingly prevalent (Figure 9.7). However, for all AFs, non-significant MANOVA site:period interaction terms were observed (Table 9.5).

Table 9.5: GLMM and MANOVA test statistics for impact of Artificial Flood (AF) 1-4 on macroinvertebrate indices and community composition respectively.

	AF1	AF2	AF3	AF4
GLMMs				
Total density (ind. / m²)	-0.78	-1.77	-0.81	-0.29
Taxonomic richness	-0.79	-0.57	-0.46	1.17
1/Simpson's diversity index (1/S)	0.88	-3.72 **	1.25	1.68
β -diversity	0.63	0.31	0.07	-0.11
Taxonomic dominance (D)	1.14	-3.40 **	1.21	0.45
BMWP	-0.41	-0.45	-1.95	1.21
ASPT	-0.57	0.68	-1.90	1.20
LIFE	1.07	-1.18	-1.52	-1.59
Coleoptera (ind. / m²)	0.32	0.30	1.51	-1.65
Diptera (ind. / m²)	-1.03	-1.35	-0.96	-0.27
Ephemeroptera (ind. / m²)	1.53	-3.71 **	-0.27	-2.52 *
Plecoptera (ind. / m²)	-2.32 *	0.22	0.22	-0.76
Trichoptera (ind. / m²)	1.67	0.59	-0.62	-0.99
MANOVA F	0.07	0.07	0.06	0.03

**p < 0.05, **p < 0.01.*

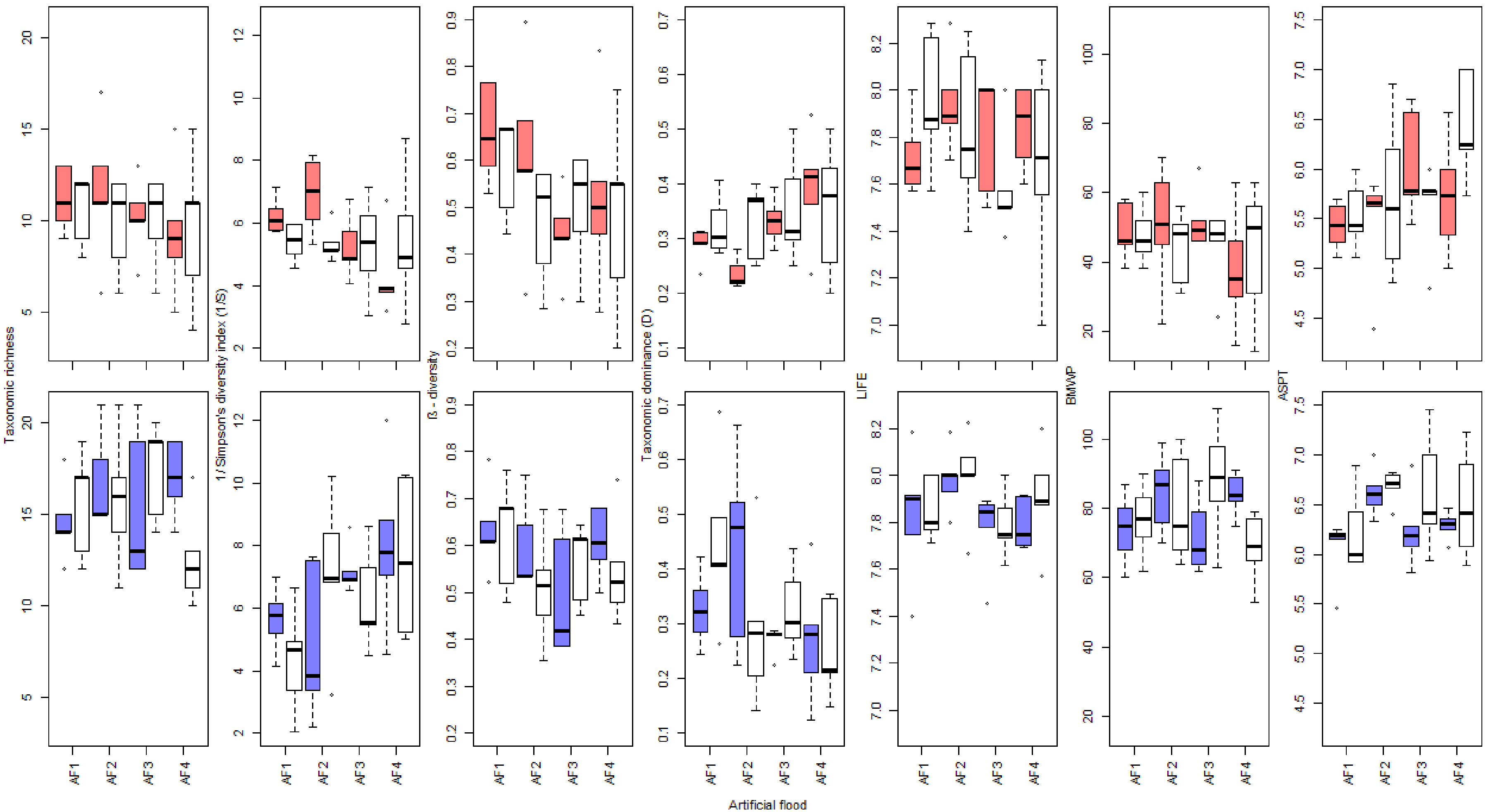

Figure 9.5: Boxplots of macroinvertebrate diversity and biomonitoring indices at sites R1 (impact, red) and R2 (control, blue) before (coloured) and after (white) each Artificial Flood (AF).

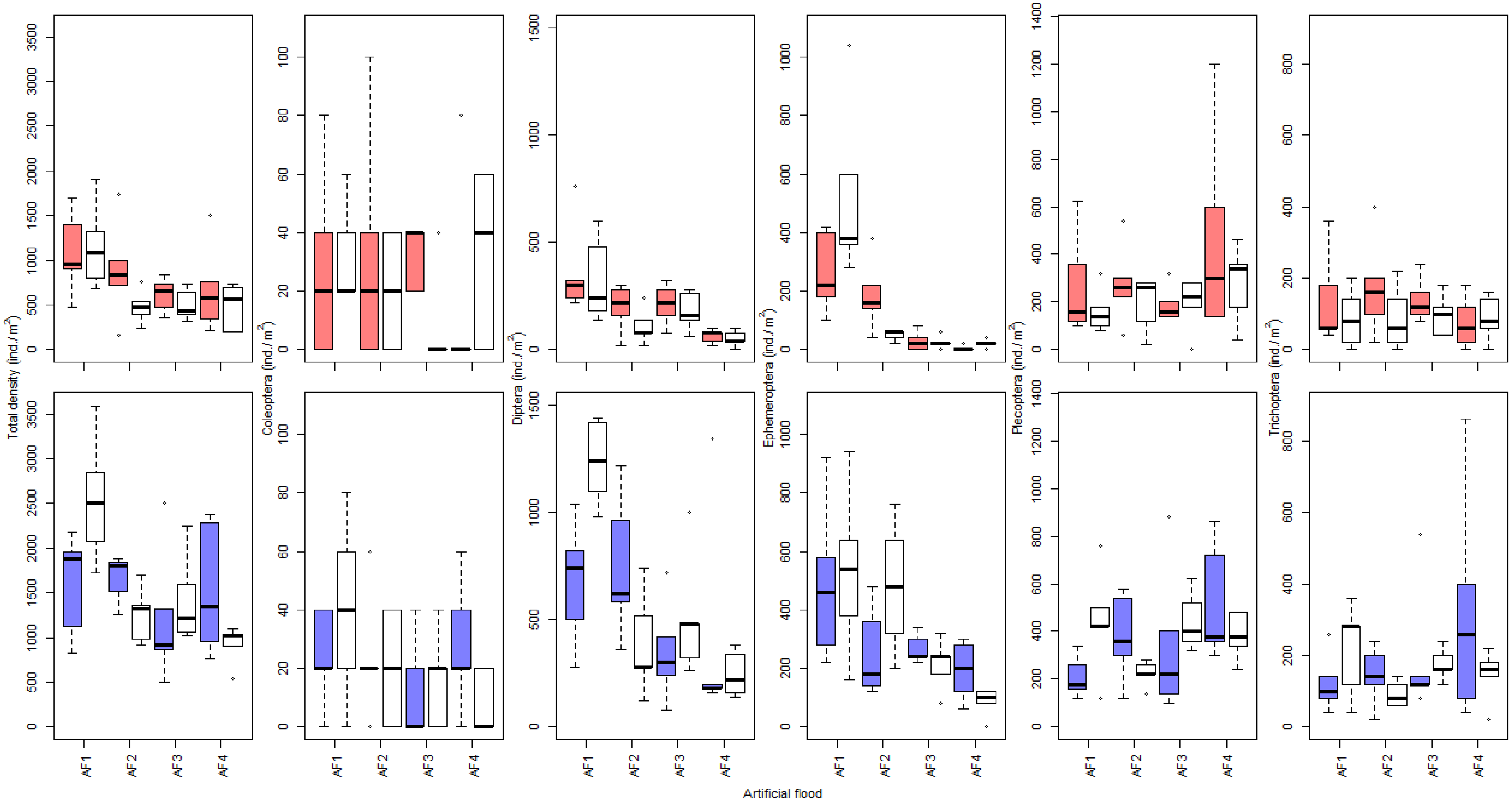

Figure 9.6: Boxplots of macroinvertebrate density indices at sites R1 (impact, red) and R2 (control, blue) before (coloured) and after (white) each Artificial Flood (AF).

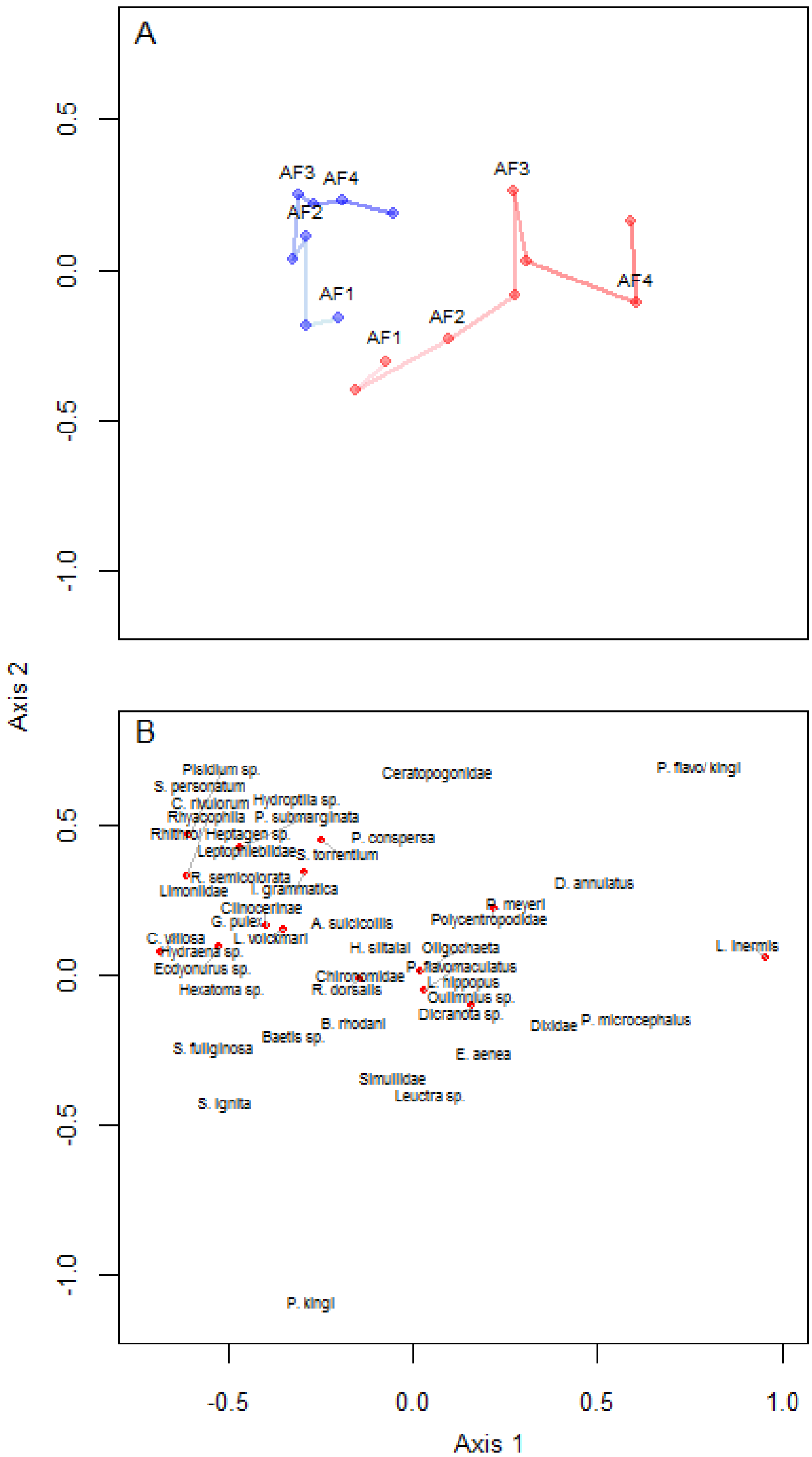

Figure 9.7: NMDS plot for mean macroinvertebrate samples and temporal transition (panel A) for sites R1 and R2 (red and blue respectively) before and after each Artificial Flood. Note: samples taken before each AF are labelled, for example, AF1 = NMDS score for sample prior to Artificial Flood 1. Panel B: species NMDS scores. Note: taxa labels are abbreviated; full names can be found in Appendix D.

9.6 Discussion

A sub-seasonal understanding of the impact of regulation on macroinvertebrate populations and their response to Artificial Floods (AFs) is currently lacking. This chapter has reported on an intensive survey of macroinvertebrate populations in one unregulated and two regulated streams in an upland UK catchment over the course of a year. Subsequently, the impact of four AFs on a benthic macroinvertebrate community in the same catchment was examined. The following is a discussion of each of these two themes in turn.

9.6.1 Impact of regulation

Significantly higher taxonomic richness and 1/Simpson's diversity index and lower taxonomic dominance for the regulated site type was both in agreement (e.g. Penaz et al., 1968; Pool & Stewart, 1976; Maynard & Lane, 2012) and disagreement (e.g. Armitage, 1978; Munn & Brusven, 1991; Gillespie et al., in press (a)) with H1. Maximum diversity has been theorised to occur when disturbance occurs at an intermediate intensity (Connell, 1978); this theory can be applied to streams in that extreme hydrological episodes (e.g. droughts; floods) can be perceived as disturbance events (Poff et al., 1997). Regulation, through reduction of extreme hydrological episodes, has been theorised to shift disturbance intensity towards an intermediate level (Maynard & Lane, 2012) and may explain the higher diversity indices observed at regulated sites in this study.

The evidence for higher diversity as a result of regulation identified for the headwater streams in this study could be viewed as contrary to a proven model of biotic response to regulation in headwaters (the Serial Discontinuity Concept (SDC) (Ward & Stanford, 1983; 1995; Stanford & Ward, 2001). The SDC postulated that regulation of headwater streams would result in a reduction in biodiversity, driven primarily by changes to nutrient and organic matter supply (Ward & Stanford, 1983; 1995). The SDC was developed on streams characterised by high groundwater contribution and dense canopy cover and dominated by heterotrophy (Ward & Stanford, 1983; 1995) which were in contract to the surface water dominated streams with little riparian vegetation canopy cover assessed in this study. This may therefore explain the disparity in our conclusions, but serves to stress that generalisations across regulated stream 'types' (e.g. headwaters), should be made with care.

BMWP and ASPT were primarily developed to reflect gradients of organic stream pollution (Armitage et al., 1983). Significantly higher BMWP scores were observed for regulated sites, likely reflecting higher richness observed in both regulated streams. This is further evidenced as non-significant differences between site types for ASPT (which is an average rather than additive index) were found. This indicated, in agreement with Gillespie et al. (in press (a)), that regulation was not affiliated with effects typically associated with organic pollution. This is therefore further evidence that use of these indices for identification of impacts associated with upland regulation in the UK may be inappropriate.

In contrast to Gillespie et al. (in press (a)), who found that LIFE was significantly lower in upland UK regulated (cf. unregulated) streams, LIFE was significantly higher for the regulated site type in this study suggesting higher richness and/ or abundance of taxa in association with high flows. In the UK, and particularly within the Pennine region, compensation flows from many reservoirs have historically been set in excess of pre-reservoir Q_{95} resulting in elimination of extreme low flows (Higgs & Petts, 1988; Gustard, 1989). Low flows have been observed to modify upland macroinvertebrate assemblages, favouring taxa adapted to such events (e.g. Cowx et al., 1984; Wood & Petts, 1994). It is possible that similar processes operated at the unregulated site within this study, reducing density of rheophilic taxa resulting in lower LIFE scores. In support of this hypothesis, taxa assigned the LIFE flow group (see Extence et al., 1999) 'flowing/ sluggish' or 'standing' were present in the 'top ten' taxa (mean most abundant throughout the study period) for site U, but not at either R1 or 2. Whereas, only 'moderate/ fast' or 'rapid' LIFE flow group taxa were present in the 'top ten' found at either sites R1 or 2 but not at site U (Table 9.6). The elimination of extreme low flow events through the introduction of compensation flows at the regulated sites may have therefore favoured rheophilic taxa resulting in higher LIFE scores. However, the number of sites assessed by Gillespie et al. (in press (a)) was far greater than in this study and further research into the generality of the findings detailed here is therefore required.

Assessment of the effect of stream regulation on β-diversity (i.e. patchiness) is novel, but this study found no evidence for any impact, indicating that patch-scale diversity is unaffected by regulation at the study sites. The reasons behind this remain unclear, but it suggests that variation in factors which affect patch diversity (e.g. bed morphology, erosion and deposition (Pringle et al., 1988)) is similar between site types. Further patch-scale studies to assess this suggestion are therefore recommended. Similarly, no evidence for differences between site types was observed for total density. This observation is in contrast to much of the published literature (e.g. Armitage, 1978; Munn & Brusven, 1991; Englund & Malmqvist, 1996) where, typically, higher total densities were observed and explained by increased availability of food for macroinvertebrates (e.g. Armitage, 1978; Munn & Brusven, 1991; Hoffsten, 1999). This suggests that food availability was not significantly modified by reservoir presence in this study. Further insight into this unusual finding could be gained from assessment of ecosystem biomass and production which have been hypothesised to change in response to modified food supply

Table 9.6: LIFE flow group classification (Extence et al., 1999) for 'top ten' taxa found at each site type but not at the other.

Taxa	LIFE flow group (Extence et al., 1999)
Found at site U, but neither R1 or 2	
Protonemura meyeri	Rapid
Nemoura cinerea	Flowing/ sluggish
Oligochaeta	Standing
***Dicranota* sp.**	Moderate/ fast
Clinocerinae	*
Found at either sites R1 or 2, but not U	
Polycentropus flavomaculatus	Moderate/ fast
***Oulimnius* sp.**	*
Hydropsyche siltalai	Moderate/ fast
Baetis rhodani	Moderate/ fast
Elmis aenea	Moderate/ fast
***Baetis* sp.**	*
Isoperla grammatica	Rapid
Siphonoperla torrentium	Rapid
Rhyacophila dorsalis	Rapid

** - flow group not assigned (Extence et al., 1999)*

downstream of reservoirs (e.g. Armitage, 1978).

In agreement with H1, significant dissimilarities in community composition were identified (e.g. higher Coleoptera and Ephemeroptera densities in both regulated sites). Higher Coleoptera density is a surprising finding as, typically, reduced numbers are found in regulated streams (e.g. Nichols et al., 2006; Armitage et al., 1987, Gillespie et al., in press (a)). Reduced macrophyte/ moss cover or high levels of metals such as iron and manganese have been cited as reasons for reductions in Coleoptera in regulated streams (Boon et al., 1988), but clear difference in macrophyte cover between sites in this study was not visually apparent (*pers. obs.*) and levels of dissolved iron and manganese were similar (see Chapter 4). *Elmis aenea* and *Limnius volckmari* were both associated with regulated sites but not with the unregulated site and both taxa are sensitive to low pH (Eyre et al., 1993). During floods, significantly lower pH was observed for the unregulated stream (see Chapter 5) and is therefore suggested as a potential reason for this disparity. This difference in pH regime may also explain the reduced numbers of Ephemeroptera which are, in general, very sensitive to low pH (Elliot et al., 2010).

Significantly higher numbers of Trichoptera, particularly the net-spinning *Polycentropus flavomaculatus* and *Hydropsyche siltalai*, were observed for the regulated stream type. Enhanced numbers of net-spinning Trichoptera have previously been observed downstream of other UK reservoirs (e.g. Spence & Hynes, 1971; Boon, 1987) and reduced incidence of droughts and spate flows has been cited as a key driver (Boon, 1987). These reasons are therefore likely to explain the relatively high abundance of Trichoptera found in this study given the hydrological impacts association with the reservoirs in the study area identified in Chapter 5.

Coleoptera, Ephemeroptera and Trichoptera reduced in density throughout the study period, resulting in reductions in taxonomic richness, 1/Simpson's diversity index and BMWP at both regulated sites during summer, but not at the unregulated site. In support of H2, it is hypothesised that these changes were in part due to the high frequency and magnitude of discharge events (e.g. Scullion & Sinton, 1983; Molles, 1985; Jakob et al., 2003; Robinson et al., 2004a) during this period. From a theoretical perspective, disturbance intensity could be seen to have shifted to a higher level resulting in lower diversity in accordance with the Intermediate Disturbance Hypothesis (Connell, 1978). Alternatively, reductions in Ephemeroptera and Trichoptera density could potentially be explained by adult emergence (e.g. Edington et al., 1995), but further data on specific emergence timing of such taxa in the streams assessed would be required to evaluate this suggestion. Future research should look to incorporate such strategies into data collection programmes.

An important point to note is that upland stream systems, such as those studied herein, can display significant spatial variation in physical-chemical variables (Ramchunder et al., 2011). For example, pH is influenced by exposure of water to underlying bedrock and an increase in mean pH has been observed to positively correlate with stream order where first-order streams are peat dominated (Ramchunder et al., 2011). Such gradients are likely to affect biological communities (Eyre et al., 1993; Ramchunder et al., 2011) and it is therefore prudent to bear this in mind when interpreting the results of this study which may be confounded by such variables as only one unregulated stream was used as a reference. However, the general approach used is consistent with that of previous studies (e.g. Englund & Malmqvist, 1996; Cortes et al., 2002) and is therefore arguably valid where no pre-regulation data exist. However, future studies should either seek to increase the number of sites (both unregulated and regulated) to increase confidence and statistical power, or compare biotic communities affected by regulation both with unregulated reference sites and predicted assemblages, although the latter may be limited to semi-quantitative comparisons (e.g. Armitage, 1989).

9.6.2 Impact of Artificial Floods

AFs have been suggested as a potential tool to mitigate the impacts of upstream regulation (Acreman & Ferguson, 2010) and have been shown to affect downstream biotic assemblages in some streams worldwide (e.g. Agostinho et al., 2004; Mannes et al., 2008). This study is the first to assess the impact of AFs on macroinvertebrate assemblage in an upland regulated UK stream. AFs were hypothesised to reduce macroinvertebrate abundance, richness and diversity in alignment with published studies (e.g. Pardo et al., 1998; Harby et al., 2001; Cereghino et al., 2004; Robinson et al., 2004a; Mannes et al., 2008; Benítez-Mora & Carmargo, 2014) (H3). However, this research did not reveal any significant changes for AF1, 3 or 4. The method of statistical testing used in this study (examination of significance of site:period interaction term under a BACI design) is commonly used to indicate whether a change has occurred at an 'impact' site (e.g. Zimmer et al., 2001; Solazzi et al., 2011; Fraser & Lemphere, 2013). However, interpretation is limited as the approach can be confounded (Underwood, 1994) if changes unrelated to the impact under assessment occur between 'before' and 'after' samples (Hurlbert, 1984). Samples were taken within 48 hours before and after each AF to reduce the potential for this and no obvious confounding events were noted during the study period. Nevertheless, in addition to examination of significance of the interaction term, careful assessment of the data is required (Conquest, 2000).

Statistically significant interaction terms were identified only for total Plecoptera and Ephemeroptera densities and Simpson's diversity index. A significant interaction term was also observed for taxonomic dominance after AF4, but only very minor reductions were observed at both impact and control sites (0.04 and 0.02 respectively). Contrary to these findings, MANOVA did not present any evidence of significant change in taxonomic composition due to either AF. It is therefore prudent to remain sceptical as to whether an impact due to an AF occurred and use of more control sites to improve statistical power (Underwood, 1994) is recommended for future research of macroinvertebrate response to AFs in upland UK streams. Both control and impact sites displayed similar magnitude in change in taxonomic composition over the whole period of AFs. Comparison of pre- and post- assemblages at the impact site revealed little change in comparison with the control site indicating no clear impact of AFs on taxonomic composition which was supported by MANOVA test results contrary to hypothesis (H4). These findings conflict sharply with assessments of macroinvertebrate response to AFs globally, which to date have all revealed clear shifts in taxonomic composition post-AF. For example, reductions in Ephemeroptera (Lauters et al., 1996; Harby et al., 2001; Mannes et al., 2008) and Chironomidae (Robinson et al., 2004b) have been observed resulting in the development of a more disturbance-resilient assemblage (Mannes et al., 2008).

The lack of clear macroinvertebrate response in this study is potentially due to the hydrological characteristics of each AF. Percentage increase in discharge of the AFs in this study (up to c. 435%) were comparable with AFs assessed globally where change in macroinvertebrate assemblage has been observed (e.g. Robinson et al., 2004a (c. 300%); Cross et al., 2010 (c. 400%)), but durations of AFs in this study (up to c. 4hours) were lower (e.g. Robinson et al., 2004a (7 to 8 hours); Cross et al., 2010 (60 hours)) suggesting that the relatively short AFs implemented in this study may be restricting change in macroinvertebrate response. It is therefore recommended that AFs of longer periods are implemented to explore this hypothesis. Whilst magnitude of percentage change is commonly used as an ecologically relevant flood descriptor (e.g. Poff & Zimmerman, 2010; Gillespie et al., in press (b)), it is limited, as for example, flow velocity (which has been linked to macroinvertebrate response (e.g. Brooks et al., 2005)) change is not taken into account. Future research should therefore aim to develop methods to adequately compare floods between studies and sites.

Alternatively, lack of clear evidence of change may be a reflection of the characteristics (e.g. behaviour) of the macroinvertebrate assemblage within this stream cf. others where change has been observed. Hydrologically, the stream assessed in this study could be classed as 'dynamic' due to repeated overspill events causing disturbances in addition to AFs in the months prior to experimental manipulations cf. other regulated streams globally where antecedent hydrological conditions were more 'stable' (e.g. Robinson et al., 2004a). The prevailing hydrology may have resulted in a disturbance resilient assemblage (i.e. taxa that can resist high flows) in the stream assessed (cf. others worldwide) which can potentially explain the observations of this study. Further research to compare assemblages from differing hydrological regimes and their responses to AFs is therefore required to test this assumption. To formalise this model, the *Regulated Stream Disturbance Hypothesis* (RSDH) has been developed.

The RSDH builds on the relationship between diversity and disturbance conceptualised as the Intermediate Disturbance Hypothesis (IDH) by Connell (1978). Connell (1978) theorised biotic diversity would be maximised at an intermediate disturbance intensity. This was based on observations of tropical rainforests and coral reefs and since this time, evidence in support of this relationship in streams has been identified (Townsend et al., 1997). Nonetheless, it is also important to note that such theories have received criticism (e.g. Kondoh, 2001; Fox, 2013) based on identification of complex relationships between diversity and productivity and empirical evidence. However, current literature also supports the IDH and suggests that such a concept can be useful under some circumstances taking into account its underlying assumptions (Huston, in press).

The RSDH has four scenarios (*a-d*) where diversity is driven by various levels of hydrological disturbance intensity. Scenario *a* is the hypothesised unaffected or reference condition: flood frequency and magnitude (hydrological disturbance intensity) is relatively high (cf. regulated conditions) resulting in a relatively narrow temporal range in biotic diversity (Figure 9.8). Scenario *b* can be described as the 'dynamic' regulated condition and is characterised by a wide range in hydrological disturbance intensity driven by periods of reservoir overspill (where disturbance is similar to experienced in scenario *a*) and where regulation reduces flood frequency as observed for the sites in this thesis (Chapter 5). This results in a wide temporal range in diversity (that overlaps with that seen in scenario *a*), but has a higher overall mean diversity than all other scenarios (Figure 9.8) as observed in this thesis (Chapter 9) (Table 9.7).

Scenario *c* can be described as the 'stable' regulated condition and is characterised by the lowest hydrological disturbance intensity of all scenarios. In this scenario, the upstream reservoir(s) exert a strong regulating effect on the hydrological regime allowing few floods to occur downstream (e.g. Robinson & Uehlinger, 2008). This results in a small temporal range in and lower mean biotic diversity than unaffected conditions (Figure 9.8; Table 9.7) (e.g. Armitage, 1978; Munn & Brusven, 1991). Scenario *d* applies at a macro-scale (e.g. regional) for regulated streams where a large range in hydrological disturbance intensity occurs (due to site specific factors such as rainfall intensity and reservoir operation). This results in large variation in impacts of regulation on biotic diversity but, on average, there is no difference in diversity to unaltered conditions (Figure 9.8) as observed in this thesis (Chapter 8) (Table 9.7). However, it is important to note that although diversity is equal, biotic community dissimilarity between the two scenarios (*a* and *d*) is evident (see Chapter 8).

While the RSDH is based on observations of benthic macroinvertebrate response to stream regulation, it may apply to alternative biotic groups, as for example, there is evidence that hydrological driven disturbance plays an important role in determining stream bryophyte (e.g. Downes et al., 2003) and fish (e.g. Kinsolving & Bain, 1993; Gehrke et al., 1995) diversity. Further research is therefore recommended to test this hypothesis and it is hoped that the RSDH can be critically applied to results from future biotic surveys of the impact of regulation and its underlying assumptions tested.

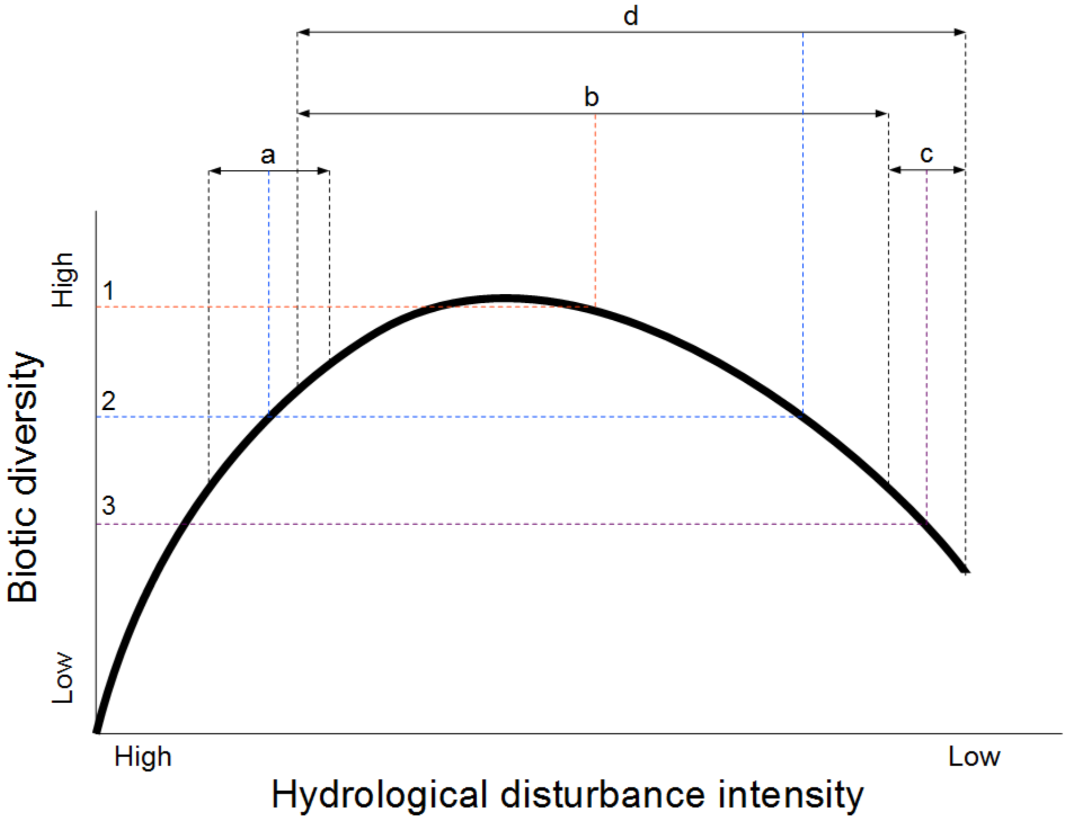

Figure 9.8: Graphical representation of the four scenarios (a-d) of the Regulated Stream Disturbance Hypothesis: interaction between hydrological disturbance intensity and biotic diversity. Temporal variation is denoted by arrows and mean disturbance/ diversity levels are shows by coloured lines. Scenarios: a - 'natural reference condition', mean biotic diversity level 2; b – 'dynamic' regulated condition, mean biotic diversity level 1; c – 'stable' regulated condition, mean biotic diversity level 3; d – macro-scale regulated condition, mean biotic diversity level 2.

Table 9.7: Evidence in support of the Regulated Stream Disturbance Hypothesis. Diversity indices for scenarios b-d and respective reference condition (scenario a) are shown. Indication of the direction of difference (regulated cf. reference) between indices is also shown where possible.

Scenario	Source	Index type	Index (st. dev.)	Reference condition index (st. dev.) (scenario a)	Direction of difference
b	Chapter 9	Taxonomic richness	10 (3)	6 (1)	+
b	Chapter 9	1/ Simpson's Diversity Index	5.39 (1.73)	2.83 (0.78)	+
c	Armitage (1978)	Shannon & Weaver diversity	1.7 (*)	3.4 (*)	-
c	Munn & Brusven (1991)	Taxonomic richness	2 (*) (summer), 5 (*) (autumn)	30 (*) (summer), 38 (*) (autumn)	- (ǂ)
d	Chapter 8	Taxonomic richness	40 (14)	43 (13)	None
d	Chapter 8	1/ Simpson's Diversity Index	5.9 (2.4)	7.20 (4.80)	None

* - not given, ǂ – not tested statistically

9.7 Summary

This study has identified significantly higher macroinvertebrate diversity and LIFE scores in regulated sites when compared to a nearby unregulated site. These differences were driven by significantly higher abundances of Coleoptera, Ephemeroptera and Trichoptera potentially reflecting a combination of a shift towards a more intermediate intensity in hydrological disturbance events, elimination of extreme low flows and suppression of natural variation in pH downstream of the reservoirs within the study area.

A novel aspect of this study was to assess impacts of regulation over a sub-seasonal temporal scale. This assessment revealed summer reductions in Coleoptera, Ephemeroptera and Trichoptera resulting in reduced diversity at both regulated sites but not the unregulated site. It was hypothesised that these observations could be explained by unusually high rainfall during June 2012 resulting in frequent reservoir overspill events shifting the hydrological disturbance regime towards a higher (more similar to experienced at the unregulated site) intensity. This concept was formalised as the Regulated Stream Disturbance Hypothesis which can aid holistic thinking around the role of disturbance in controlling regulated stream biota and direct future research. Further research is required to test this hypothesis.

The impact of a series of AFs on a downstream macroinvertebrate assemblage were assessed. In contrast to published literature and the hypotheses, no clear impacts were observed. It was theorised that, although percentage change in discharge during AFs were similar to those undertaken in other studies where shifts in assemblage were observed, AF duration was lower and potentially explained the lack of observed impact. Alternatively, lack of clear impact could have been explained by characteristics (e.g. resilience) of the macroinvertebrate assemblage at the study site. Further research is required to test these theories. It was also proposed that future research undertaking similar assessments could be improved by incorporating methodological changes such as using more than one control site.

10 RESEARCH SYNOPSIS

10.1 Chapter overview

This chapter presents a synopsis of the key findings of this chapters 5 to 9 of this thesis. It relates these findings to the initial aims and objectives of the study and details whether hypotheses made were accepted or rejected. To direct future research, key research needs identified in each chapter are also presented.

10.2 Research synopsis

This thesis has achieved the following aims as set out in Chapter 1:

1. Assess the impact of regulation on downstream ecosystems, including hydrological, physical-chemical, morphological and biotic elements;

2. Assess the impact of environmental flows (Artificial Floods (AFs) specifically) on downstream ecosystems.

10.2.1 Aims

This thesis is driven by a requirement to better understand regulated stream ecosystems. The primary aims are to:

3. Assess the impact of regulation on downstream ecosystems, including hydrological, physical-chemical, morphological and biotic elements;

4. Assess the impact of environmental flows (AFs specifically) on downstream ecosystems.

10.2.2 Hydrology

Table 10.1 details the objective and hypothesis associated with this element. It also details whether the hypothesis was accepted or rejected and the following text further details the conclusions made regarding this element.

Table 10.1: Objectives and hypotheses associated with Chapter 5. An indication of whether hypotheses were either accepted (A) or rejected (R) is made.

Objective		**Respective hypothesis**		**A/R**
1	Undertake an assessment of the impact of regulation on hydrology, EC, DO and pH through comparison with unregulated conditions	1	Regulation would reduce flood frequency, magnitude and duration and the impact would vary temporally	A
		2	Regulation would impact downstream physical chemistry and these impacts would vary temporally	A
2	Conduct a series of AFs to assess the potential for use of AFs as mitigation for any impacts identified	3	AFs would impact downstream physical chemistry thereby demonstrating control of downstream physical chemistry and potential for use as mitigation	A

In agreement with other research from across the globe (e.g. Baxter, 1977; Higgs & Petts, 1988; Gustard, 1989), stream regulation was associated with reduced flood frequency and duration. A novel observation was that regulation (from a non-hydropower reservoir) was associated with increased flood rate of change (both rising and falling limbs) cf. floods on a nearby unregulated stream. These observations are salient as flood frequency, duration and rate of change have all been noted as key drivers of stream ecological integrity (Poff et al., 1997).

It is important to note that these associations were not temporally consistent: impacts were generally confined to within spring and summer where rainfall was relatively low, resulting in periods where reservoirs had capacity to store runoff rather than overspill. The impact of regulation on discharge rate of change during floods was thought to reflect the relatively large overspill channel crest width cf. the small downstream receiving channel capacity. It is thought that such observations have not previously been made in combination before and are driven by the relatively small capacity (from a global perspective) of the reservoirs within the study area

(Lehner et al., 2011). This point should be taken into account when assessing the global applicability of the impacts identified by this study.

Previous research (e.g. Higgs & Petts, 1988) has identified an association between regulation and reduced flood magnitude, but evidence for this was not identified in this study. It was theorised that this may have been due to the method used to define flood magnitude which was based on a percentage increase from Q_{25} flows. Q_{25} was selected for use prior to analysis as it appeared to adequately distinguish floods from the hydrographs at all sites, but alternative Q values may have yielded different results. Further research is therefore required to address this potential methodological limitation by using alternative approaches that either test alternative Q values for defining floods, or alternatively, compare actual with predicted discharge rather than with an unregulated reference.

It was concluded that the use of AFs as mitigation for reduced flood frequency and duration identified as associated with regulation is unlikely to be successful given that impacts were only identified during periods of low rainfall. Such periods are likely to coincide with increased demand for water from humans (Bond et al., 2008) resulting in a drive to store as much water in reservoirs as possible and therefore reducing the likelihood of AF implementation. The rate of change of AFs was demonstrated to be controllable and there is therefore potential to use such methods to mitigate the relatively high rates of change during floods identified to be associated with regulation.

Research highlights:

- Evidence for association between regulation and reduced flood frequency and duration;
- Evidence for association between regulation and increased discharge rate of change during floods;
- No evidence for impact of regulation on flood magnitude;
- Limited potential demonstrated for use of AFs as mitigation of flood frequency and duration impacts; better potential for rate of change impacts.

10.2.3 Physical-chemical variables

Tables 10.1 and 10.2 detail the objectives and hypotheses associated with this element. They also detail whether the hypotheses were accepted or rejected and the following text further details the conclusions made regarding this element.

Table 10.2: Objectives and hypotheses associated with Chapter 6. An indication of whether hypotheses were either accepted (A) or rejected (R) is made.

	Objective		**Respective hypothesis**	**A/R**
1	Undertake a spatio-temporal assessment of the impact of regulation on stream temperature using contemporary analytical techniques through comparison of several regulated streams with unregulated conditions in a multi-reservoir catchment	1	Regulation would reduce downstream temperature range and impact mean stream temperature according to season	A
		2	Impact would vary spatially	A
2	Conduct an assessment of the impact of a series of AFs on downstream temperature	3	Change in downstream temperature would be observed during AFs, thereby demonstrating the potential for use of AFs as mitigation.	R

No evidence for impact of reservoirs on downstream electrical conductivity (EC) during the routine release of compensation water was found, but during overspill events, EC was found to be significantly lower downstream of reservoirs (cf. during floods in an unregulated stream) potentially reflecting ionic uptake by phytoplankton in the reservoir epilimnion (e.g. Atkins & Harris, 1925). Regulation was found to be associated with reduced 1- and 7-day range in dissolved oxygen (DO). Temperature can influence the amount of oxygen water can hold (cooler water can hold more oxygen (USGS, 2014)) and this observation may therefore reflect reduced temperature range which was identified to be associated with regulation (see below). Alternatively, reduced photosynthetic activity (which drives dissolved oxygen levels (USGS, 2014)) of the receiving waters may have occurred, resulting in reduced DO range.

Another novel finding of this study was that during overspills, significantly higher DO was found in reservoir tailwaters cf. during flood events in a nearby unregulated reference stream, likely reflecting oxygenation of water cascading down overspill channels. Such an effect may be important in other locations where reservoirs are of relatively small capacity cf. precipitation levels and overspill events are common.

The impact of stream regulation on downstream pH has received little attention and this thesis has developed understanding in this area. Evidence for an association between regulation and reduced pH diurnal range was identified. Photosynthesis and respiration can drive diurnal pH range in water through consumption and production of carbon dioxide (Morrison et al., 2001; Soares et al., 2008). It was hypothesised that such processes would be reduced due to hypolimnetic supply of compensation flow which was likely devoid of photosynthetically active organisms (e.g. phytoplankton) resulting in reduced pH range.

During overspill events, pH was significantly higher downstream of reservoirs cf. during floods in an unregulated reference stream, potentially reflecting the ability of reservoirs to increase pH through buffering due to prolonged interaction between water and reservoir substrates and photosynthetic activity (Soares et al., 2008). Alternatively, differences in land cover/ geology may explain this observation, but no evidence for differences in mean pH between sites was identified and this latter theory is therefore unlikely.

Globally, the impact of regulation on stream temperature has been well studied. Site specific complexities in impact have been identified and this thesis has advanced understanding of some of these complexities. Changes in diurnal mean stream temperature were identified to be associated with regulation in broad agreement with previous research (e.g. Cowx et al., 1987; Dickson et al., 2012). During winter, diurnal mean stream temperature was up to 2°C warmer than predicted although during summer, negligible impact on mean stream temperature was apparent downstream of most reservoirs. Two key points emerged: (i) impacts were generally confined to within ~1.5km downstream of reservoirs indicating rapid equilibration with atmospheric conditions and, (ii) impacts were spatially diverse, as for example, during summer, diurnal mean stream temperature directly downstream of one reservoir was up to ~3.5°C lower than predicted.

This thesis found, in agreement with previous research (e.g. Lavis & Smith, 1972; Lehmkhul, 1972; Crisp, 1977; O'Keeffe et al., 1990), that regulation was association with reduced diurnal stream temperature range. However, considerable spatio-temporal complexities were identified. For example, during autumn and summer, clear impacts were confined to within 1.5 and 0.2km downstream of reservoirs respectively. Additionally, during winter, reductions in diurnal stream temperature range were typically < 0.5°C, but during summer, reductions were up to ~ 7°C. It was proposed that the range in spatial-temporal impacts identified in this study were due to a number of interacting factors including: (i) the relatively large thermal capacity of the reservoirs cf. the small size (and therefore thermal capacity) of receiving waters; (ii) reservoir specific factors such as volume, operation and stratification status and, (iii) general lack of riparian vegetative cover of receiving streams.

AFs appeared to drive reductions in EC, DO and pH, but such responses were not observed in all AFs. Changes in EC were hypothesised to be a result of reduced water residence time in the reservoir stilling basin where, under normal compensation flows, EC would likely increase due to contact with mineral substrate. Additionally, alternative processes (e.g. photosynthesis/ respiration) were also noted to potentially have played a role and further research is required to assess these hypotheses. Reduced DO was only observed during AFs 1 & 2; it was hypothesised that such changes may have occurred due to disruption to photosynthetic activity. The impact of AFs on pH had not been assessed prior to this study and the driver behind the reductions is unknown and it was proposed that it may have been due to dissolution of atmospheric carbon dioxide during the turbulent release (Raymond & Cole, 2001). Conversely, stream temperature was not affected by any AF; this finding was consistent with other studies where the AF supply valves were not changed from pre-flood water supply.

It was concluded that the potential for mitigation of the physical-chemical impacts of regulation identified above was greatest for pH and may be enhanced in AFs of higher magnitude and rates of change. Assessment of AFs supplied by valves at different (or mixed) vertical heights within the dam may yield alternative observations, particularly with regards to stream temperature (e.g. Macdonald et al., 2012) and should therefore be carried out in future research.

Research highlights:

- Evidence for reduced electrical conductivity and increased dissolved oxygen and pH cf. unregulated conditions during floods due to stream regulation;
- Evidence for lower dissolved oxygen (1- and 7-day) and pH (1-day) range due to regulation;
- Evidence for warming effect downstream of reservoir during winter and cooling during summer under some circumstances;
- Evidence for reduction in diurnal stream temperature range due to reservoirs; impacts greatest during summer but limited to within short distances downstream;
- Some Artificial Floods resulted in reductions in electrical conductivity, pH and DO, but no change in temperature during any Artificial Flood;
- Potential for use of Artificial Floods to mitigate impacts of regulation greatest for pH.

10.2.4 Coarse sediment transport

Table 10.3 details the objectives and hypotheses associated with this element. It also details whether the hypotheses were accepted or rejected and the following text further details the conclusions made in this study.

Table 10.3: Objectives and hypotheses associated with Chapter 7. An indication of whether hypotheses were either accepted (A) or rejected (R) is made.

	Objective		**Respective hypothesis**	**A/R**
1	Undertake a detailed study of coarse sediment transport in a regulated upland UK stream to better understand the sediment transport-discharge relationship and threshold discharges	1	The sediment transport-discharge relationship and threshold discharge to invoke sediment transport would vary temporally reflecting antecedent flow conditions	A
2	Assess the impact of a series of AFs of varying characteristics on coarse sediment transport and potential for use of AFs as morphological management tools	2	AFs would invoke sediment transport and therefore demonstrate potential for use as a morphological management tool	A

This thesis reported on the first process-based assessment of coarse sediment transport dynamics in a regulated stream. Similar to unregulated streams (e.g. Reid et al., 2007; Turowski & Rickenmann, 2011), sediment transport was generally positively correlated with discharge. It was proposed that deviances from this association were due to discrete events (e.g. bank collapse) rather than an artefact of the sensors used. This assumption was further evidenced through assessment of hysteresis curves which demonstrated considerable variability of sediment transport dynamics during each flood in agreement with other studies (e.g. Moog & Whiting, 1998; Ryan et al., 2005). Whilst this general association did not significantly differ between periods characterised by relatively high and low discharges, threshold discharges required to stimulate sediment transport appeared to be higher during the latter, demonstrating the importance of antecedent conditions (e.g. Sear, 1993; Vericat et al., 2006).

Flood based analysis of coarse sediment transport revealed that a discharge increase of ~400% above Q_{25} appeared to act as a general threshold for significant sediment transport, potentially linked to physical change (e.g. mobilisation of a particular clast size). Total sediment transport during floods was most associated with flood magnitude (positive relationship) in agreement with other studies in unregulated streams (e,g. Bogen et al., 2003) but interestingly, magnitude of the previous flood was negatively associated with total sediment transport indicating that sediment supply after large floods may be limited for transport in subsequent floods.

Significant coarse sediment transport only occurred in the final, largest magnitude, AF. This finding was consistent with the relationship between discharge and sediment transport prior to AFs and gives weight to the argument for the use of high magnitude AFs as a morphological management tool. However, the requirement for such mitigation should be carefully assessed, especially in regulated streams where overspill events occur, given the ability of such events to transport coarse sediment.

Research highlights:

- General positive relationship between coarse sediment transport and discharge identified for a regulated stream;
- Threshold discharges required to invoke sediment transport differed between periods of relatively high and low overspills;
- Flood magnitude appeared to be the greatest driver of total sediment transported during each flood, however, a negative correlation with magnitude of the previous flood was also identified indicating the importance of antecedent conditions;
- Use of Artificial Floods to manage sediment transport was demonstrated.

10.2.5 Macroinvertebrates

Tables 10.4 and 10.5 detail the objectives and hypotheses associated with this element. They also detail whether the hypotheses were accepted or rejected and the following text further details the conclusions made in this study.

Table 10.4: Objectives and hypotheses associated with Chapter 8. An indication of whether hypotheses were either accepted (A) or rejected (R) is made.

Objective		Respective hypothesis		A/R
1	Identify relationships between the extent of stream regulation and macroinvertebrate communities using a multi-site, regional-scale approach	1	Macroinvertebrate indices and community composition would both be affected by upstream impoundment with some taxa increasing and others decreasing in abundance relative to their sensitivity to changes in flow	A
2	Examine the utility of new, continuous, index representing the extent of stream regulation	2	A continuous index representing extent of regulation would be more sensitive to differences in community composition than categorical classifications	A
3	Evaluate two recently developed indices (LIFE and PSI) alongside established biomonitoring indices to consider their relative performance for assessment of the impacts of regulation	3	LIFE and PSI would decrease as the extent of stream regulation increases, and demonstrate superior sensitivity to alternative indices such as diversity, dominance, BMWP and ASPT in detecting any impacts	A & R respectively

Table 10.5: Objectives and hypotheses associated with Chapter 9. An indication of whether hypotheses were either accepted (A) or rejected (R) is made.

Objective		Respective hypothesis		A/R
1	Conduct an assessment of intra-annual temporal dynamics of the impact of regulation on stream macroinvertebrates	1	A difference in macroinvertebrate assemblages between regulated and unregulated sites could be observed in line with previous studies	A
		2	The difference would vary intra-annually reflecting taxon life-cycle attributes and environmental preferences	A
2	Examine the impact of AFs on downstream benthic macroinvertebrates in the UK	3	Macroinvertebrate abundance, richness and diversity would decrease as a result of AFs	R
		4	Taxa would respond to AFs based on their specific environmental preferences resulting in a more disturbance resilient assemblage	R

This thesis enhanced understanding of the impact of regulation on downstream macroinvertebrate assemblages in upland UK systems. At a regional scale, Coleoptera and Ephemeroptera were found to be negatively affected and Trichoptera positively so. At a sub-catchment scale, Coleoptera and Ephemeroptera were found in greater densities downstream of reservoirs (cf. an unregulated stream) indicating the importance of spatial scale in assessments (Wiens, 1989).

The regional scale study reported in this thesis found associations between upstream regulation and the invasive snail *Potamopyrgus antipodarum* and the stonefly *Amphinemura sulcicollis*. These associations were hypothesised to be a result of changes in flow and temperature regimes associated with regulation and had not previously been identified for the UK. The importance of *P. antipodarum* as an ecosystem engineer has recently been highlighted (Moore et al., 2012) and further monitoring of this apparent association between this species and regulation in the UK is recommended.

Both spatial studies concluded that traditional biomonitoring indices (i.e. BMWP and ASPT) appeared to be ineffective at detecting community scale changes associated with regulation. Conversely, the recently developed flow-specific LIFE index (Extence et al., 1999) did appear to detect these changes. BMWP and ASPT are sensitive to pollution not typically associated with regulation, but LIFE has been developed to detect flow specific pressures and is therefore more suited to use in scenarios where flows are impacted (e.g. regulated streams). The continued use and development of LIFE is recommended as it has potential to identify where regulation-specific pressures are evident and could potentially be used to assess long term dynamics in such pressures and therefore the effectiveness of mitigation measures such as AFs.

The regional scale study tested three methods of describing the extent to which a site is affected by regulation in the absence of detailed hydrological and water quality information. It was found that a continuous score, Index of Regulation, rather than two categorical classifications, appeared to be the more effective at allowing for changes associated with regulation to be detected. Further testing and development of this method and proposed extensions (IR_D and IR_Q) is therefore recommended.

The sub-catchment scale study found that higher levels of macroinvertebrate diversity (taxonomic richness and 1/Simpson's Diversity Index) were associated with regulated sites both in agreement (e.g. Penaz et al., 1968; Pool & Stewart, 1976; Maynard & Lane 2012) and disagreement (e.g. Armitage, 1978; Munn & Brusven; 1991; Gillespie et al., in press (a)) with other studies. It was hypothesised that a reduction in intensity/ occurrence of extreme high and low flow events may have shifted disturbance intensity towards an intermediate level, resulting in higher diversity in accordance with Intermediate Disturbance theories (Connell, 1978; Townsend et al., 1997; Huston, in press). The sub-catchment scale study revealed novel intra-annual changes in macroinvertebrate assemblages affected by regulation. The primary changes of note were a reduction in diversity indices (driven by reduced densities of Ephemeroptera, Trichoptera and Coleoptera) and asynchronous removal of significant differences in community assemblages between regulated and unregulated sites. These changes were theorised to have occurred due to a series of large floods which shifted the balance of disturbance towards a higher level, resulting in lower diversity (Connell, 1978). The Regulated Stream Disturbance Hypothesis was developed to formalise these concepts and stimulate and direct future research.

The impact of four successively larger magnitude AFs on a downstream benthic macroinvertebrate community assemblage was assessed although, in contrast to other studies (e.g. Pardo et al., 1998; Harby et al., 2001; Cereghino et al.,2004; Robinson et al., 2004a; Mannes et al., 2008; Benitez-Mora & Carmargo, 2014), little evidence of change was identified. The AFs were of similar magnitude change to other studies where impacts were observed (e.g. Robinson et al., 2004a; Cross et al., 2010), but of considerably shorter duration and it is hypothesised that this may be the reason for little evidence of change, although flood characteristics between studies may not be directly comparable. Additionally, factors specific to the macroinvertebrate assemblage studied (e.g. resilience to floods) may also have been important. Further research is therefore required to test these theories.

Research highlights:

- At a regional scale, Coleoptera and Ephemeroptera were found to be negatively affected and Trichoptera positively so. Conversely, at a sub-catchment scale, Coleoptera and Ephemeroptera were positively associated with regulation;
- Evidence that the invasive snail *Potamopyrgus antipodarum* and the stonefly *Amphinemura sulcicollis* were associated with regulation was found;
- Traditional biomonitoring indices (BMWP and ASPT) appeared ineffective in detecting impacts associated with regulation, but LIFE appeared sensitive to such community differences;
- A continuous score, Index of Regulation, rather than two categorical classifications, appeared to be the most effective at allowing for changes associated with regulation to be detected;
- Evidence for higher levels of diversity associated with regulation was identified at a sub-catchment scale. Temporal variation in community assemblage was identified and hypothesised to be a reflection of hydrological disturbance events;
- Artificial Floods demonstrated little evidence for use as management tools of downstream macroinvertebrate assemblages.

10.3 Summary

This chapter has detailed the findings of this thesis on the impact of (i) stream regulation and (ii) Artificial Floods on downstream hydrology, physical chemistry, coarse sediment transport and benthic macroinvertebrate assemblages.

Stream regulation was found to impact elements of discharge, electrical conductivity, dissolved oxygen, pH and temperature regimes. Additionally, regulation was found to be associated with impacts on downstream macroinvertebrate communities.

Control of hydrological characteristics was demonstrated during Artificial Floods which generally resulted in reductions of electrical conductivity, dissolved oxygen and pH and no change in stream temperature, although variability in response was observed. Evidence for coarse sediment transport in line with overspill events prior to Artificial Floods was identified, but little evidence for change in macroinvertebrate assemblage as a result of Artificial Floods was found. Evidence for the use of Artificial Floods as management tools was greatest for coarse sediment transport and pH, but overall, limited potential was demonstrated.

It is hoped that through conceptual developments such as the *Index of Regulation* and the *Regulated Stream Disturbance Hypothesis*, a theoretical baseline for future studies assessing the impact of regulation on downstream biota has been established thereby presenting key research themes and hypotheses for testing. Such ideas are important for driving forward the science of regulated streams in an novel, and potentially scientifically rewarding, directions.

The findings of this thesis and conceptual advances are likely to prove useful in a world where (i) reservoir construction is continuing at a fast pace in some countries (Olden et al., 2014) and, (ii) contemporary legislation increasingly requires understanding of such systems to enable attainment of their targets.

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Wood PJ, Petts GE (1994) Low flows and recovery of macroinvertebrates in a small regulated chalk stream. Regulated Rivers: Research and Management 9:303–316.

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Worrall T, Dunbar M, Extence C, et al. (2013) The identification of hydrological indices for the characterisation of macroinvertebrate community response to flow regime variability. Hydrological Sciences Journal 0:null. doi: 10.1080/02626667.2013.825722

YSI (2005) Linearity and Accuracy of Conductivity Sensors on YSI 6-Series Sondes.

YSI (2008) ROX Optical Dissolved Oxygen Sensor.

YSI (2011) Fast-Response pH Sensor.

YW, Yorkshire Water (n.d.) Reservoir D weir inspection and rating curve calculation [unpublished report].

Zhong Y, Power G (1996) Environmental impacts of hydroelectric projects on fish resources in China. Regulated Rivers: Research & Management 12:81–98. doi: 10.1002/(SICI)1099-1646(199601)12:1<81::AID-RRR378>3.0.CO;2-9

Zimmer KD, Hanson MA, Butler MG (2001) Effects of Fathead Minnow Colonization and Removal on a Prairie Wetland Ecosystem. Ecosystems 4:346–357. doi: 10.1007/s10021-001-0016-1

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12 APPENDIX A: PUBLICATIONS USED IN SYSTEMATIC REVIEW

Table 12.1a: Publications included in systematic review undertaken in Chapter 3.

Publication reference number	Publication
1	Agostinho AA, Gomes LC, Verissimo S, Okada EK. (2004). Flood regime, dam regulation and fish in the Upper Parana River: effects on assemblage attributes, reproduction and recruitment. Reviews in Fish Biology and Fisheries 14: 11–19.
2	Andersen DC, Shafroth PB. (2010). Beaver dams, hydrological thresholds, and controlled floods as a management tool in a desert riverine ecosystem, Bill Williams River, Arizona. Ecohydrology 3: 325–338. DOI:10.1002/eco.113.
3	Angradi TR, Kubly DM. (1993). Effects of atmospheric exposure on chlorophyll a, biomass and productivity of the epilithon of a tailwater river. Regulated Rivers: Research & Management 8: 345–358. DOI:10.1002/rrr.3450080405.
4	Arntzen EV, Geist DR, Dresel PE. (2006). Effects of fluctuating river flow on groundwater/surface water mixing in the hyporheic zone of a regulated, large cobble bed river. River Research and Applications 22: 937–946. DOI:10.1002/rra.947.
5	Asaeda T, Rashid MH. (2012). The impacts of sediment released from dams on downstream sediment bar vegetation. Journal of Hydrology 430–431: 25–38. DOI:10.1016/j.jhydrol.2012.01.040.
6	Ashby SL, Myers JL, Laney E, Honnell D, Owens C. (1999). The Effects of Hydropower Releases from Lake Texoma on Downstream Water Quality. Journal of Freshwater Ecology 14: 103–112. DOI:10.1080/02705060.1999.9663659.
7	Batalla RJ, Vericat D. (2009). Hydrological and sediment transport dynamics of flushing flows: implications for management in large Mediterranean Rivers. River Research and Applications 25: 297–314. DOI:10.1002/rra.1160.
8	Bednarek AT, Hart DD. (2005). MODIFYING DAM OPERATIONS TO RESTORE RIVERS: ECOLOGICAL RESPONSES TO TENNESSEE RIVER DAM MITIGATION. Ecological Applications 15: 997–1008. DOI:10.1890/04-0586.
9	Benenati PL, Shannon JP, Blinn DW. (1998). Desiccation and recolonization of phytobenthos in a regulated desert river: Colorado River at Lees Ferry, Arizona, USA. Regulated Rivers: Research & Management 14: 519–532. DOI:10.1002/(SICI)1099-1646(1998110)14:6<519::AID-RRR518>3.0.CO;2-H.
10	Berland G, Nickelsen T, Heggenes J, Økland F, Thorstad EB, Halleraker J. (2004). Movements of wild atlantic salmon parr in relation to peaking flows below a hydropower station. River Research and Applications 20: 957–966. DOI:10.1002/rra.802.
11	Bernez I, Haury J, Ferreira MT. (2002). Downstream effects of a hydroelectric reservoir on aquatic plant assemblages. TheScientificWorldJournal 2: 740–750. DOI:10.1100/tsw.2002.142.
12	Beschta RL, Jackson WL, Knoop KD. (1981). Sediment Transport During a Controlled Reservoir Release1. JAWRA Journal of the American Water Resources Association 17: 635–641. DOI:10.1111/j.1752-1688.1981.tb01270.x.
13	Bradford MJ, Higgins PS, Korman J, Sneep J. (2011). Test of an environmental flow release in a British Columbia river: does more water mean more fish? Freshwater Biology 56: 2119–2134. DOI:10.1111/j.1365-2427.2011.02633.x.
14	Brandt SA, Swenning J. (1999). Sedimentological and geomorphological effects of reservoir flushing: The Cachi Reservoir, Costa Rica, 1996. Geografiska Annaler. Series A, Physical Geography 81: 391–407.
15	Brenden TO, Murphy BR, Hallerman EM. (2006). Effect of Discharge on Daytime Habitat Use and Selection by Muskellunge in the New River, Virginia. Transactions of the American Fisheries Society 135: 1546–1558. DOI:10.1577/T05-256.1.
16	Brooks AJ, Russell M, Bevitt R, Dasey M. (2011). Constraints on the recovery of invertebrate assemblages in a regulated snowmelt river during a tributary-sourced environmental flow regime. Marine and Freshwater Research 62: 1407–1420. . Accessed 16 Oct 2013
17	Bruno M, Maiolini B, Carolli M, Silveri L. (2010). Short time-scale impacts of hydropeaking on benthic invertebrates in an Alpine stream (Trentino, Italy). Limnologica - Ecology and Management of Inland Waters 40: 281–290. DOI:10.1016/j.limno.2009.11.012.

Table 12.2b: Publications included in systematic review undertaken in Chapter 3.

Publication reference number	Publication
18	Cambray JA, King JM, Bruwer C. (1997). Spawning behaviour and early development of the Clanwilliam yellowfish (Barbus capensis; Cyprinidae), linked to experimental dam releases in the Olifants River, South Africa. Regulated Rivers: Research & Management 13: 579–602. DOI:10.1002/(SICI)1099-1646(199711/12)13:6<579::AID-RRR486>3.0.CO;2-F.
19	Cánovas CR, Olias M, Vazquez-Suñé E, Ayora C, Nieto JM. (2012). Influence of releases from a fresh water reservoir on the hydrochemistry of the Tinto River (SW Spain). Science of The Total Environment 416: 418–428. DOI:10.1016/j.scitotenv.2011.11.079.
20	Céréghino R, Legalle M, Lavandier P. (2004). Drift and benthic population structure of the mayfly Rhithrogena semicolorata (Heptageniidae) under natural and hydropeaking conditions. Hydrobiologia 519: 127–133. DOI:10.1023/B:HYDR.0000026499.53979.69.
21	Chester H, Norris R. (2006). Dams and Flow in the Cotter River, Australia: Effects on Instream Trophic Structure and Benthic Metabolism. Hydrobiologia 572: 275–286. DOI:10.1007/s10750-006-0219-8.
22	Chung SW, Ko IH, Kim YK. (2008). Effect of reservoir flushing on downstream river water quality. Journal of Environmental Management 86: 139–147. DOI:10.1016/j.jenvman.2006.11.031.
23	Cowx IG, O'Grady KT, Parasiewicz P, Schmutz S, Moog O. (1998). The effect of managed hydropower peaking on the physical habitat, benthos and fish fauna in the River Bregenzerach in Austria. Fisheries Management and Ecology 5: 403–417. DOI:10.1046/j.1365-2400.1998.550403.x.
24	Cross WF, Baxter CV, Donner KC, Rosi-Marshall EJ, Kennedy TA, Jr ROH, Kelly HAW, Rogers RS. (2011). Ecosystem ecology meets adaptive management: food web response to a controlled flood on the Colorado River, Glen Canyon. Ecological Applications 21(6): 2016–2033.
25	Elliott JG, Hammack LA. (2000). Entrainment of riparian gravel and cobbles in an alluvial reach of a regulated canyon river. Regulated Rivers: Research & Management 16: 37–50. DOI:10.1002/(SICI)1099-1646(200001/02)16:1<37::AID-RRR564>3.0.CO;2-V.
26	Foulger TR, Petts GE. (1984). Water quality implications of artificial flow fluctuations in regulated rivers. Science of the Total Environment 37 (2-3): 177–185.
27	Fuller RL, Doyle S, Levy L, Owens J, Shope E, Vo L, Wolyniak E, Small MJ, Doyle MW. (2011). Impact of regulated releases on periphyton and macroinvertebrate communities: The dynamic relationship between hydrology and geomorphology in frequently flooded rivers. River Research and Applications 27: 630–645. DOI:10.1002/rra.1385.
28	Gido KB, Larson RD, Ahlm LA. (2000). Stream-Channel Position of Adult Rainbow Trout Downstream of Navajo Reservoir, New Mexico, Following Changes inReservoir Release. North American Journal of Fisheries Management 20: 250–258. DOI:10.1577/1548-8675(2000)020<0250:SCPOAR>2.0.CO;2.
29	Gilvear DJ. (1989). Experimental analysis of reservoir release wave routing in upland boulder bed rivers. Hydrological Processes 3: 261–276. DOI:10.1002/hyp.3360030306.
30	Harby A, Alfredsen KT, Fjeldstad HP, Halleraker JH, Arnekleiv JV, Borsanyi P, Flodmark LEW, Saltveit SJ, Johansen SW, Vehanen T, Huusko A, Clarke K, Scruton DA. (2001). Ecological impacts of hydro peaking in rivers (B Honningsvag, GH Midttomme, K Repp, KA Vaskinn, and T Westeren, Eds.). A a Balkema Publishers, Leiden
31	Hardwick GG, Blinn DW, Usher HD. (1992). Epiphytic Diatoms on Cladophora glomerata in the Colorado River, Arizona: Longitudinal and Vertical Distribution in a Regulated River. The Southwestern Naturalist 37: 148. DOI:10.2307/3671663.
32	Heggenes J, Omholt PK, Kristiansen JR, Sageie J, Økland F, Dokk JG, Beere MC. (2007). Movements by wild brown trout in a boreal river: response to habitat and flow contrasts. Fisheries Management and Ecology 14: 333–342. DOI:10.1111/j.1365-2400.2007.00559.x.
33	Henson SS, Ahearn DS, Dahlgren RA, Nieuwenhuyse EV, Tate KW, Fleenor WE. (2007). Water quality response to a pulsed-flow event on the Mokelumne river, California. River Research and Applications 23: 185–200. DOI:10.1002/rra.973.
34	Irvine RL, Oussoren T, Baxter JS, Schmidt DC. (2009). The effects of flow reduction rates on fish stranding in British Columbia, Canada. River Research and Applications 25: 405–415. DOI:10.1002/rra.1172.
35	Jakob C, Robinson CT, Uehlinger U. (2003). Longitudinal effects of experimental floods on stream benthos downstream of a large dam. Aquatic Science 65 (3): 223–231.
36	Jeffres CA, Klimley AP, Merz JE, Jr JJC. (2006). Movement of Sacramento Sucker, Catostomus occidentalis, and Hitch, Lavinia exilicauda, during a Spring Release of Water from Camanche Dam in the Mokelumne River, California. Environmental Biology of Fishes 75: 365–373. DOI:10.1007/s10641-005-2924-y.

Table 12.3c: Publications included in systematic review undertaken in Chapter 3.

Publication reference number	Publication
37	Jodeau M, Paquier A, Hauet A, Coz J, Thollet F, Fournier T. (2007). Effect of a reservoir release on the morphology of a gravel bar: Field observations and 2Dh modeling. In: Dohmen-Janssen C, Hulscher S (eds) River, Coastal and Estuarine Morphodynamics: RCEM 2007, Two Volume Set. Taylor & Francis, pp. 1029–1036. http://www.crcnetbase.com/doi/abs/10.1201/NOE0415453639-c130. Accessed 16 Oct 2013
38	King J, Cambray JA, Impson ND. (1998). Linked effects of dam-released floods and water temperature on spawning of the Clanwilliam yellowfish Barbus capensis. Hydrobiologia 384: 245–265. DOI:10.1023/A:1003481524320.
39	King AJ, Tonkin Z, Mahoney J. (2009). Environmental flow enhances native fish spawning and recruitment in the Murray River, Australia. River Research and Applications 25: 1205–1218. DOI:10.1002/rra.1209.
40	Korman J, Campana SE. (2009). Effects of Hydropeaking on Nearshore Habitat Use and Growth of Age-0 Rainbow Trout in a Large Regulated River. Transactions of the American Fisheries Society 138: 76–87. DOI:10.1577/T08-026.1.
41	Korman J, Kaplinski M, Melis TS. (2011). Effects of Fluctuating Flows and a Controlled Flood on Incubation Success and Early Survival Rates and Growth of Age-0 Rainbow Trout in a Large Regulated River. Transactions of the American Fisheries Society 140: 487–505. DOI:10.1080/00028487.2011.572015.
42	Krein A, Symader W. (2000). Pollutant sources and transport patterns during natural and artificial flood events in the Olewiger Bach and Kartelbornsbach basins, Germany. In: IAHS Press, pp. 167–173
43	Lagarrigue T, Céréghino R, Lim P, Reyes-Marchant P, Chappaz R, Lavandier P, Belaud A. (2002). Diel and seasonal variations in brown trout (Salmo trutta) feeding patterns and relationship with invertebrate drift under natural and hydropeaking conditions in a mountain stream. Aquatic Living Resources 15: 129–137. DOI:10.1016/S0990-7440(02)01152-X.
44	Lauters F, Lavandier P, Lim P, Sabaton C, Belaud A. (1996). Influence of Hydropeaking on Invertebrates and Their Relationship with Fish Feeding Habits in a Pyrenean River. Regulated Rivers: Research & Management 12: 563–573. DOI:10.1002/(SICI)1099-1646(199611)12:6<563::AID-RRR380>3.0.CO;2-M.
45	Macdonald JS, Morrison J, Patterson DA. (2012). The Efficacy of Reservoir Flow Regulation for Cooling Migration Temperature for Sockeye Salmon in the Nechako River Watershed of British Columbia. North American Journal of Fisheries Management 32: 415–427. DOI:10.1080/02755947.2012.675946.
46	Mannes S, Robinson C-T, Uehlinger U, Scheurer T, Ortlepp J, Mürle U, Molinari P. (2008). Ecological effects of a long-term flood program in a flow-regulated river. Revue de géographie alpine/Journal of Alpine Research: 125–134. DOI:10.4000/rga.450.
47	Marty J, Smokorowski K, Power M. (2009). The influence of fluctuating ramping rates on the food web of boreal rivers. River Research and Applications 25: 962–974. DOI:10.1002/rra.1194.
48	McKinney T, Speas D, Rogers R, Persons W. (2001). Rainbow trout in a regulated river below Glen Canyon Dam, Arizona, following increased minimum flows and reduced discharge variability. Journal Articles 21. http://scholarworks.umass.edu/fishpassage_journal_articles/240
49	Meissner K, Muotka T, Kananen I. (2002). Drift responses of larval blackflies and their invertebrate predators to short-term flow regulation. Archiv für Hydrobiologie 154: 529–542. . Accessed 16 Oct 2013
50	Mürle U, Ortlepp J, Zahner M. (2003). Effects of experimental flooding on riverine morphology, structure and riparian vegetation: The River Spöl, Swiss National Park. Aquatic Sciences 65: 191–198. DOI:10.1007/s00027-003-0665-6.
51	Muirhead RW, Davies-Colley RJ, Donnison AM, Nagels JW. (2004). Faecal bacteria yields in artificial flood events: quantifying in-stream stores. Water research 38: 1215–1224. DOI:10.1016/j.watres.2003.12.010.
52	Naliato DA, Nogueira MG, Perbiche-Neves G. (2009). Discharge pulses of hydroelectric dams and their effects in the downstream limnological conditions: a case study in a large tropical river (SE Brazil). Lakes & Reservoirs: Research & Management 14: 301–314. DOI:10.1111/j.1440-1770.2009.00414.x.
53	Ortlepp J, Mürle U. (2003). Effects of experimental flooding on brown trout: The River Spol, Swiss National Park. Aquatic Sciences 65: 232–238.
54	Pardo I, Campbell IC, Brittain JE. (1998). Influence of dam operation on mayfly assemblage structure and life histories in two south-eastern Australian streams. Regulated Rivers: Research & Management 14: 285–295. DOI:10.1002/(SICI)1099-1646(199805/06)14:3<285::AID-RRR502>3.0.CO;2-6.
55	Pert EJ, Erman DC. (1994). Habitat Use by Adult Rainbow Trout under Moderate Artificial Fluctuations in Flow. Transactions of the American Fisheries Society 123: 913–923. DOI:10.1577/1548-8659(1994)123<0913:HUBART>2.3.CO;2.

Table 12.4d: Publications included in systematic review undertaken in Chapter 3.

Publication reference number	**Publication**
56	Petticrew EL, Krein A, Walling DE. (2007). Evaluating fine sediment mobilization and storage in a gravel-bed river using controlled reservoir releases. Hydrological Processes 21: 198.
57	Petts GE, Foulger TR, Gilvear DJ, Pratts JD, Thoms MC. (1985). Wave-movement and water-quality variations during a controlled release from Kielder Reservoir, North Tyne River, U. K. Journal of Hydrology 80: 371–389.
58	Propst DL, Gido KB. (2004). Responses of Native and Nonnative Fishes to Natural Flow Regime Mimicry in the San Juan River. Transactions of the American Fisheries Society 133: 922–931. DOI:10.1577/T03-057.1.
59	Rickenmann D, Weber D, Stepanov B. (2003). Erosion by debris flows in field and laboratory experiments. In: Rickenmann D, Chen CL (eds) Debris-Flow Hazards Mitigation: Mechanics, Prediction and Assessment., pp. 883–894
60	Robertson MJ, Pennell CJ, Scruton DA, Robertson GJ, Brown JA. (2004). Effect of increased flow on the behaviour of Atlantic salmon parr in winter. Journal of Fish Biology 65: 1070–1079. DOI:10.1111/j.0022-1112.2004.00516.x.
61	Robinson CT, Aebischer S, Uehlinger U. (2004). Immediate and habitat-specific responses of macroinvertebrates to sequential, experimental floods. Journal of the North American Benthological Society 23 (4): 853–867.
62	Robinson CT, Uehlinger U, Monaghan MT. (2003). Effects of a multi-year experimental flood regime on macroinvertebrates downstream of a reservoir. Aquatic Sciences - Research Across Boundaries 65: 210–222.
63	Robinson CT, Uehlinger U, Monaghan MT. (2004). Stream ecosystem response to multiple experimental floods from a reservoir. River Research and Applications 20: 359–377.
64	Robinson CT, Uehlinger U. (2008). Experimental Floods Cause Ecosystem Regime Shift in a Regulated River. Ecological Applications 18: 511–526.
65	Rolls RJ, Boulton AJ, Growns IO, Maxwell SE. (2011). Response by fish assemblages to an environmental flow release in a temperate coastal Australian river: A paired catchment analysis. River Research and Applications 27: 867–880. DOI:10.1002/rra.1402.
66	Rubin DM, Nelson JM, Topping DJ. (1998). Relation of inversely graded deposits to suspended-sediment grain-size evolution during the 1996 flood experiment in Grand Canyon. Geology 26: 99–102. DOI:10.1130/0091-7613(1998)026<0099:ROIGDT>2.3.CO;2.
67	Saltveit SJ, Halleraker JH, Arnekleiv JV, Harby A. (2001). Field experiments on stranding in juvenile atlantic salmon (Salmo salar) and brown trout (Salmo trutta) during rapid flow decreases caused by hydropeaking. Regulated Rivers: Research & Management 17: 609–622. DOI:10.1002/rrr.652.
68	Schmidt JC, Parnell RA, Grams PE, Hazel JE, Kaplinski MA, Stevens LE, Hoffnagle TL. (2001). THE 1996 CONTROLLED FLOOD IN GRAND CANYON: FLOW, SEDIMENT TRANSPORT, AND GEOMORPHIC CHANGE. Ecological Applications 11: 657–671. DOI:10.1890/1051-0761(2001)011[0657:TCFIGC]2.0.CO;2.
69	Sear DA. (1993). Fine sediment infiltration into gravel spawning beds within a regulated river experiencing floods: Ecological implications for salmonids. Regulated Rivers: Research and Management 8: 373–390.
70	Shannon JP, Blinn DW, McKinney T, Benenati EP, Wilson KP, O'Brien C. (2001). Aquatic Food Base Response to the 1996 Test Flood below Glen Canyon Dam, Colorado River, Arizona. Ecological Applications 11 (3): 672–685.
71	Speas DW. (2000). Zooplankton density and community composition following an experimental flood in the Colorado River, Grand Canyon, Arizona. Regulated Rivers: Research & Management 16: 73–81. DOI:10.1002/(SICI)1099-1646(200001/02)16:1<73::AID-RRR565>3.0.CO;2-#.
72	Strydom NA, Whitfield AK. (2000). The effects of a single freshwater release into the Kromme Estuary. 4 : Larval fish response : The effects of a single freshwater release into the Kromme Estuary. Water S.A. 26: 319–328. . Accessed 16 Oct 2013
73	De Sutter R, Krein A, Van Poucke L. (2000). Simulation of sediment transport in a small basin with artificial flood events. ADVANCES IN FLUID MECHANICS III 26: 549–558. . Accessed 16 Oct 2013
74	Uehlinger U, Kawecka B, Robinson CT. (2003). Effects of experimental floods on periphyton and stream metabolism below a high dam in the Swiss Alps (River Spöl). Aquatic Sciences 65: 199–209.
75	Valdez RA, Hoffnagle TL, McIvor CC, McKinney T, Leibfried WC. (2001). EFFECTS OF A TEST FLOOD ON FISHES OF THE COLORADO RIVER IN GRAND CANYON, ARIZONA. Ecological Applications 11: 686–700. DOI:10.1890/1051-0761(2001)011[0686:EOATFO]2.0.CO;2.
76	Wright SA, Kaplinski M. (2011). Flow structures and sandbar dynamics in a canyon river during a controlled flood, Colorado River, Arizona. Journal of Geophysical Research: Earth Surface 116: n/a–n/a. DOI:10.1029/2009JF001442.

13 APPENDIX B: CALIBRATIONS

13.1 Water level - discharge

Water level – discharge rating curves developed from salt discharge gauging for sites 1, 5, 7, 8 and 10 are shown in Figure 13.1. Figure 13.2 is the discharge - water level rating curve and coefficients developed from a weir rating table provided by YW for site 11. Figure 13.3 shows the water levels recorded at sites 1, 5, 7, 8 and 10 for the periods that analyses were undertaken.

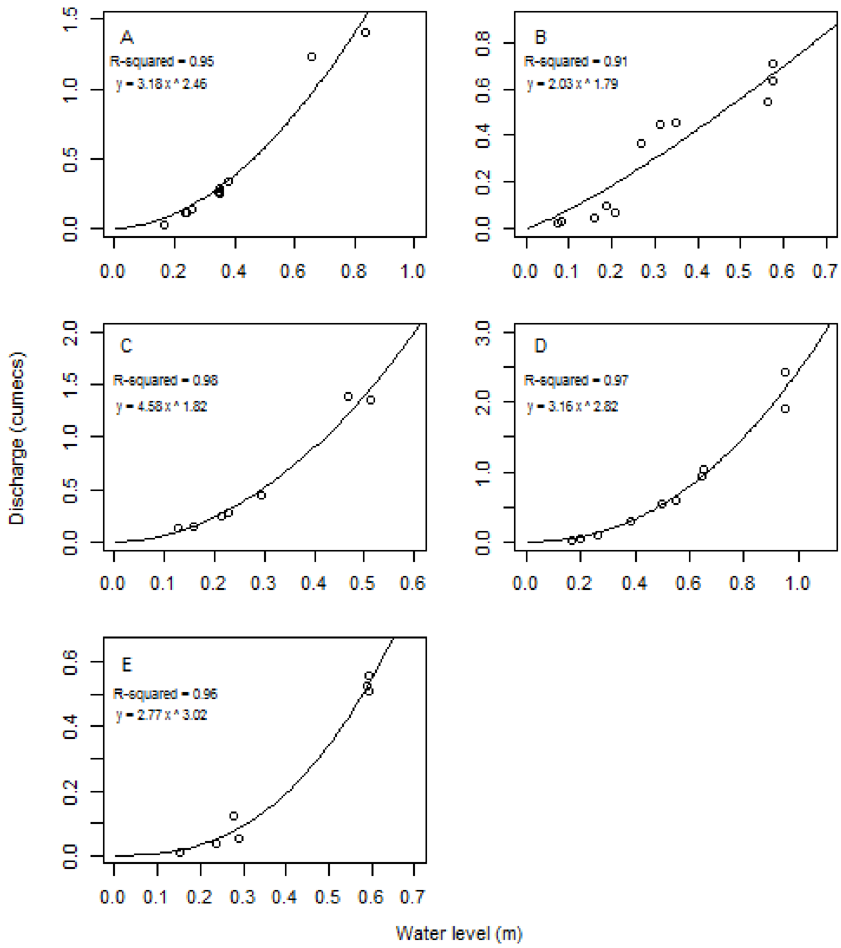

Figure 13.1: Water level – discharge rating curves for sites 1, 5, 7, 8 and 10 (panels A – E respectively) developed from salt – discharge gauging.

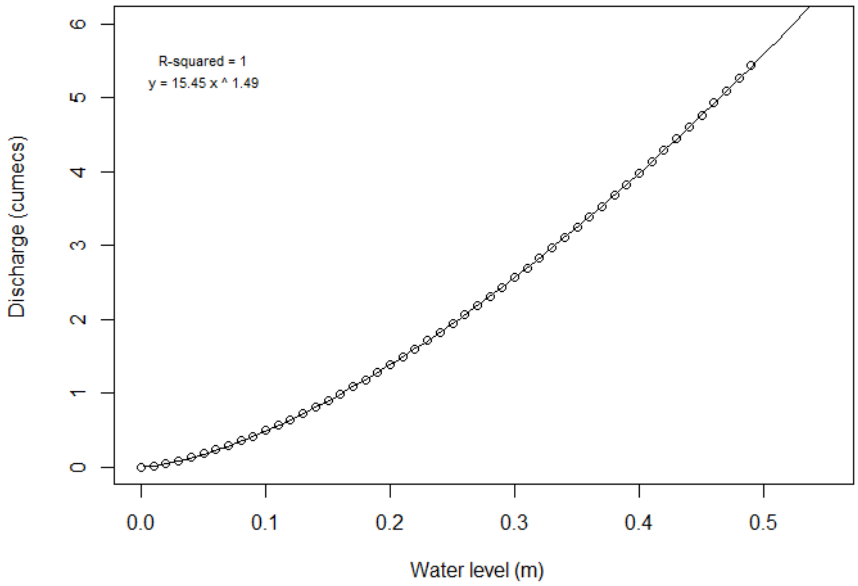

Figure 13.2: Water level – discharge rating curve for site 11 developed from YW (n.d.).

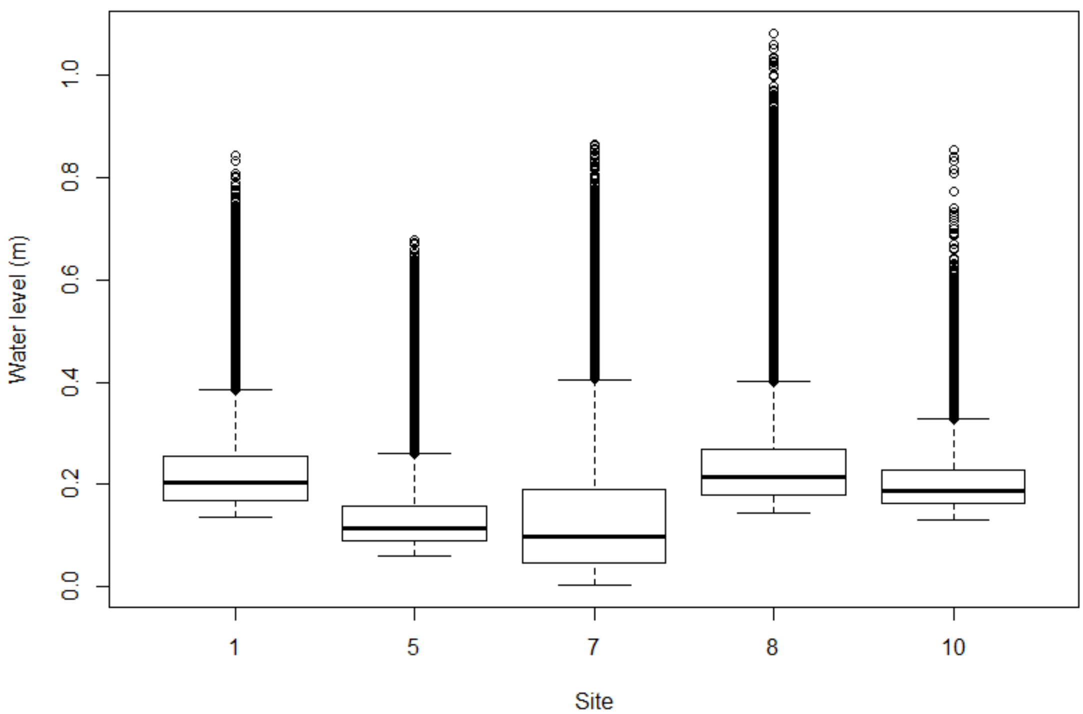

Figure 13.3: Boxplots of water levels recorded at sites 1, 5, 7, 8 and 10 during the assessment periods.

13.2 Stream temperature

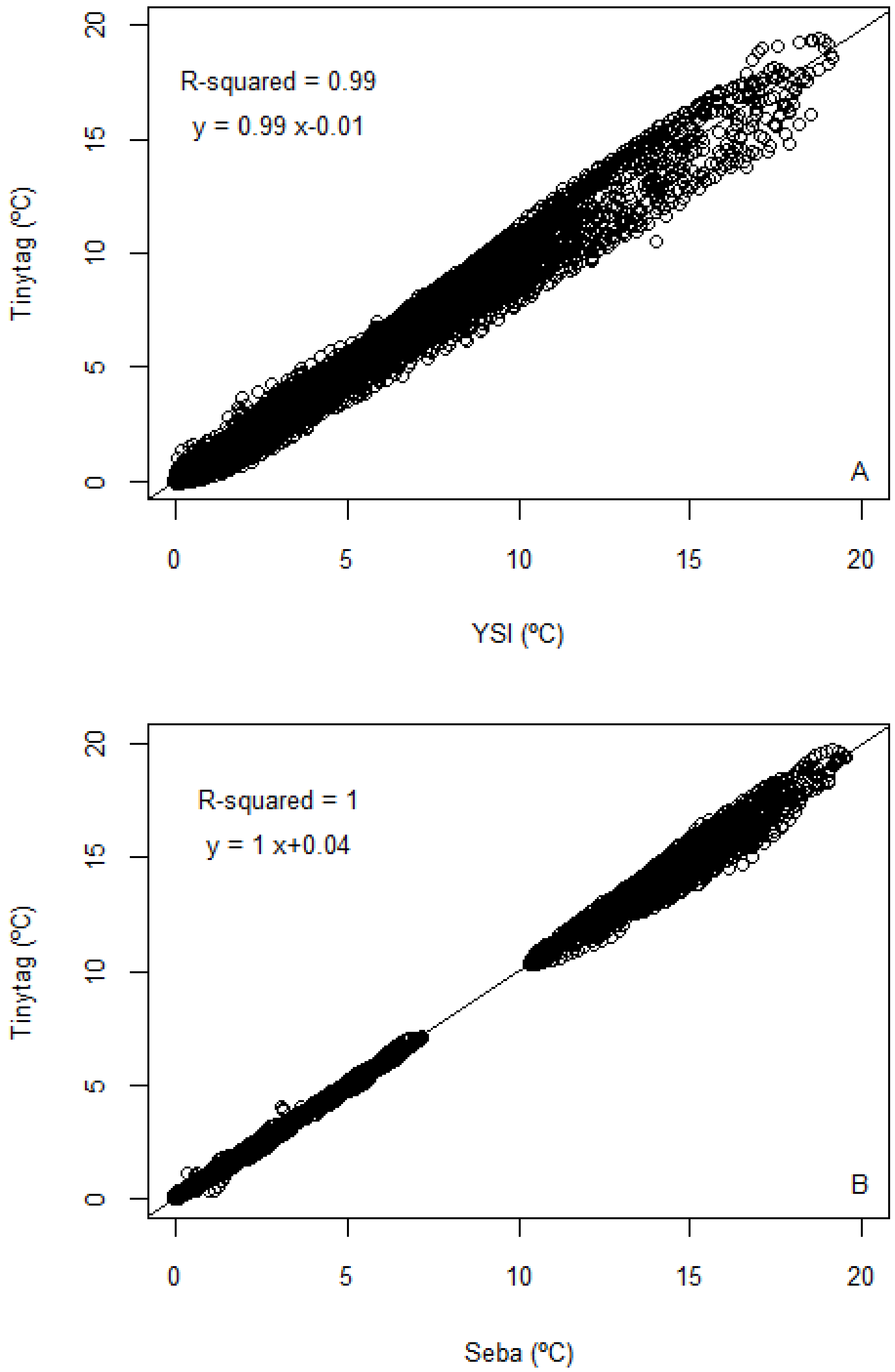

Figure 13.4: Stream temperature calibration curves and coefficients for Tinytag and YSI and Seba dataloggers (panels A and B respectively).

14 APPENDIX C: REMOVED DATA

14.1 Discharge

Table 14.1: Discharge data removed during quality control.

Site	Start	End	Reason
1	22/05/2012 13:15:00	04/06/2012 07:45:00	Probe out of position
1	10/07/2012 00:00:00	30/07/2012 08:30:00	Probe out of position
5	09/12/2011 20:00:00	02/07/2012 10:45:00	Probe out of position
5	10/09/2013 05:00:00	25/09/2013 07:30:00	Probe malfunction
6	25/11/2012 09:30:00	25/11/2012 14:00:00	Probe malfunction
6	05/03/2013 16:15:00	08/03/2013 02:00:00	Probe malfunction
7	14/12/2011 11:45:00	05/01/2012 11:30:00	Probe out of position
7	22/06/2012 17:15:00	02/07/2012 15:30:00	Probe out of position
7	17/01/2013 14:30:00	17/01/2013 14:30:00	Probe malfunction
8	08/12/2011 20:00:00	15/12/2011 23:45:00	Probe out of position
8	03/04/2012 00:00:00	05/04/2012 23:45:00	Probe malfunction
8	30/07/2013 00:00:00	02/08/2013 23:45:00	Probe malfunction
10	05/06/2012 12:15:00	06/07/2012 23:45:00	Probe out of position
10	17/12/2012 08:45:00	04/01/2013 10:15:00	Probe out of position

14.2 Physical-chemical variables

The following detail physical-chemical data manually removed during quality control. A record of missing data due to telemetry failure can be found in the supplementary documents folder on the accompanying CD.

Table 14.2: Electrical conductivity data removed during quality control.

Site	Start	End	Reason
6	22/01/2012 00:00:00	28/01/2012 00:00:00	Probe malfunction
6	02/06/2012 00:00:00	30/07/2012 00:00:00	Probe malfunction
6	24/09/2012 12:00:00	26/09/2012 04:48:00	Probe malfunction
6	15/10/2012 00:00:00	21/10/2012 00:00:00	Probe malfunction
6	29/12/2012 00:00:00	06/01/2013 00:00:00	Probe malfunction
6	24/04/2013 00:00:00	25/04/2013 00:00:00	Probe malfunction
6	27/07/2013 00:00:00	02/08/2013 00:00:00	Probe malfunction
6	19/08/2013 12:00:00	20/08/2013 00:00:00	Probe malfunction
10	18/10/2012 00:00:00	07/11/2012 00:00:00	Probe malfunction
10	05/03/2013 00:00:00	06/03/2013 00:00:00	Probe malfunction
10	15/06/2013 00:00:00	04/08/2013 00:00:00	Probe malfunction
10	20/08/2013 00:00:00	21/08/2013 00:00:00	Probe malfunction
11	23/12/2011 16:48:00	24/12/2011 00:00:00	Probe malfunction
11	25/12/2011 00:00:00	10/01/2012 00:00:00	Probe malfunction
11	16/05/2012 00:00:00	25/05/2012 00:00:00	Probe malfunction
11	23/06/2012 00:00:00	24/06/2012 00:00:00	Probe malfunction
11	04/07/2012 00:00:00	31/07/2012 00:00:00	Probe malfunction
11	03/01/2013 00:00:00	31/01/2013 00:00:00	Probe malfunction
11	13/06/2013 00:00:00	21/06/2013 00:00:00	Probe malfunction
11	19/08/2013 12:00:00	20/08/2013 00:00:00	Probe malfunction
11	11/09/2013 00:00:00	20/09/2013 00:00:00	Probe malfunction

Table 14.3: Dissolved oxygen data removed during quality control.

Site	Start	End	Reason
6	01/01/2013 00:00:00	22/01/2013 00:00:00	Probe malfunction
6	04/03/2013 00:00:00	27/03/2013 00:00:00	Probe faulty when received
6	19/06/2013 00:00:00	24/06/2013 00:00:00	Probe malfunction
6	27/07/2013 12:00:00	02/08/2013 00:00:00	Probe malfunction
6	30/07/2013 00:00:00	13/09/2013 00:00:00	Probe faulty when received
6	31/08/2013 00:00:00	20/09/2013 00:00:00	Probe malfunction
11	01/01/2013 00:00:00	11/01/2013 00:00:00	Probe malfunction
11	05/03/2013 14:24:00	05/03/2013 14:52:48	Probe malfunction
11	02/06/2013 00:00:00	04/06/2013 00:00:00	Probe malfunction
11	17/06/2013 00:00:00	28/06/2013 00:00:00	Probe malfunction
11	17/07/2013 00:00:00	30/07/2013 00:00:00	Probe malfunction

Table 14.4: pH data removed during quality control.

Site	Start	End	Reason
6	02/11/2012 00:00:00	02/12/2012 00:00:00	Probe malfunction
6	05/03/2013 10:33:36	05/03/2013 11:02:24	Probe malfunction
6	24/04/2013 00:00:00	05/06/2013 00:00:00	Probe malfunction
10	02/10/2012 09:36:00	02/10/2012 14:24:00	Probe malfunction
10	05/03/2013 12:00:00	06/03/2013 00:00:00	Probe malfunction
10	03/04/2013 12:00:00	03/04/2013 14:24:00	Probe malfunction
11	23/12/2011 00:00:00	12/01/2012 00:00:00	Probe malfunction
11	03/10/2012 12:00:00	04/10/2012 12:00:00	Probe malfunction
11	05/03/2013 14:24:00	05/03/2013 14:52:48	Probe in air while swapping
11	02/04/2013 12:57:36	02/04/2013 13:26:24	Probe malfunction
11	16/05/2013 00:00:00	05/06/2013 00:00:00	Probe malfunction

Table 14.5: Stream temperature data removed during quality control.

Site	Start	End	Reason
2	03/06/2013 11:30:00	03/06/2013 11:30:00	Logger in air while swapping
3	03/06/2013 11:15:00	03/06/2013 11:15:00	Logger in air while swapping
9	30/01/2013 13:30:00	30/01/2013 14:15:00	Logger in air while swapping
9	04/06/2013 09:45:00	04/06/2013 09:45:00	Logger in air while swapping
9	28/07/2013 00:00:00	02/08/2013 00:00:00	Logger in air while swapping

14.3 Impact sensors

Table 14.6: Impact sensor data removed during quality control

Sensor	Start	End	Reason
Furthest from bank	26/04/2013 19:12:00	26/04/2013 21:36:00	Presumed human/ animal interference
Furthest from bank	29/08/2012 10:13:00	29/08/2012 11:47:00	Data download
Furthest from bank	19/10/2012 08:19:00	19/10/2012 11:47:00	Data download
Furthest from bank	28/10/2012 01:59:00	28/10/2012 01:31:00	Data download
Furthest from bank	04/01/2013 12:43:00	04/01/2013 13:47:00	Data download
Furthest from bank	02/04/2013 10:33:00	03/04/2013 17:47:00	Data download
Furthest from bank	02/07/2013 14:29:00	03/07/2013 17:47:00	Data download
Furthest from bank	02/10/2013 09:01:00	02/10/2013 16:47:00	Data download
Middle	29/08/2012 09:15:00	29/08/2012 10:47:00	Data download
Middle	19/10/2012 07:25:00	19/10/2012 10:47:00	Data download
Middle	28/10/2012 01:59:00	28/10/2012 01:31:00	Data download
Middle	04/01/2013 11:47:00	04/01/2013 13:47:00	Data download
Middle	02/04/2013 10:45:00	03/04/2013 16:47:00	Data download
Middle	02/07/2013 14:03:00	03/07/2013 16:47:00	Data download
Middle	02/10/2013 08:01:00	02/10/2013 16:47:00	Data download
Closest to bank	29/08/2012 09:15:00	29/08/2012 10:47:00	Data download
Closest to bank	19/10/2012 07:25:00	19/10/2012 10:47:00	Data download
Closest to bank	28/10/2012 01:59:00	28/10/2012 01:31:00	Data download
Closest to bank	04/01/2013 11:45:00	04/01/2013 13:47:00	Data download
Closest to bank	02/04/2013 10:43:00	03/04/2013 16:47:00	Data download
Closest to bank	02/07/2013 14:03:00	03/07/2013 16:47:00	Data download
Closest to bank	02/10/2013 08:01:00	02/10/2013 16:47:00	Data download

15 APPENDIX D: MACROINVERTEBRATES

15.1 Identification literature

Table 15.1: Literature used to identify macroinvertebrates.

Taxonomic group	Literature used
Coleoptera	Foster et al., 2011; Friday, 1988
Diptera	Smith, 1989
Ephemeroptera	Macan, 1979; Elliot et al., 2010
Megaloptera	Elliot, 1996
Plecoptera	Hynes, 1993
Sphaeriidae	Fitter & Manuel, 1994
Trichoptera	Edington et al., 1995; Wallace et al., 2003

15.2 Taxa list

Table 15.2a: Taxa identified in the study described in Chapter 9 – note: taxa are sorted by order.

Full taxa name	Abbreviated taxa name	Taxa ID
Gammaridae	Gammaridae	1
Gammarus sp.	Gammarus sp.	2
Gammarus pulex	G. pulex	3
Dytiscidae	Dytiscidae	4
Agabus sp.	Agabus sp.	5
Elmidae	Elmidae	6
Elmis sp.	Elmis sp.	7
Elmis aenea	E. aenea	8
Limnius sp.	Limnius sp.	9
Limnius volckmari	L. volckmari	10
Oulimnius sp.	Oulimnius sp.	11
Hydraenidae	Hydraenidae	12
Hydraena sp.	Hydraena sp.	13
Hydrophilidae	Hydrophilidae	14
Anacaena sp.	Anacaena sp.	15
Anacaena globulus	A. globulus	16
Hydroporinae	Hydroporinae	17
Ceratopogonidae	Ceratopogonidae	18
Chironomidae	Chironomidae	19
Clinocerinae	Clinocerinae	20
Dixidae	Dixidae	21
Hexatoma sp.	Hexatoma sp.	22
Limoniidae	Limoniidae	23
Pediciidae	Pediciidae	24
Dicranota sp.	Dicranota sp.	25
Psychodidae	Psychodidae	26
Simuliidae	Simuliidae	27
Ameletidae	Ameletidae	28
Ameletus sp.	Ameletus sp.	29
Ameletus inopinatu	A. inopinatus	30
Baetidae	Baetidae	31
Baetis sp.	Baetis sp.	32
Baetis rhodani	B. rhodani	33
Caenidae	Caenidae	34
Caenis sp.	Caenis sp.	35
Caenis rivulorum	C. rivulorum	36

Table 15.2b: Taxa identified in the study described in Chapter 9 – note: taxa are ordered by order.

Full taxa name	Abbreviated taxa name	Taxa ID
Ephemerellidae	Ephemerellidae	37
Serratella sp.	Serratella sp.	38
Serratella ignita	S. ignita	39
Heptageniidae	Heptageniidae	40
Ecdyonurus sp.	Ecdyonurus sp.	41
Rhithrogena sp.	Rhithrogena sp.	42
Rhithrogena semicolorata	R. semicolorata	43
Rhithrogena/ Heptagenia sp.	Rhithro/ Heptagen sp.	44
Leptophlebiidae	Leptophlebiidae	45
Leptophlebia sp.	Leptophlebia sp.	46
Leptophlebia marginata	L. marginata	47
Paraleptophlebia sp.	Paraleptophlebia sp.	48
Paraleptophlebia submarginata	P. submarginata	49
Sialidae	Sialidae	50
Sialis sp.	Sialis sp.	51
Sialis fuliginosa	S. fuliginosa	52
Leuctridae	Leuctridae	53
Leuctra sp.	Leuctra sp.	54
Leuctra inermis	L. inermis	55
Leuctra hippopus	L. hippopus	56
Leuctra nigra	L. nigra	57
Nemouridae	Nemouridae	58
Amphinemura sp.	Amphinemura sp.	59
Amphinemura sulcicollis	A. sulcicollis	60
Nemoura sp.	Nemoura sp.	61
Nemoura cinerea	N. cinerea	62
Protonemura sp.	Protonemura sp.	63
Protonemura meyeri	P. meyeri	64
Chloroperlidae	Chloroperlidae	65
Siphonoperla sp.	Siphonoperla sp.	66
Siphonoperla torrentium	S. torrentium	67
Perlodidae	Perlodidae	68
Isoperla sp.	Isoperla sp.	69
Isoperla grammatica	I. grammatica	70
Perlodes sp.	Perlodes sp.	71
Perlodes microcephalus	P. microcephalus	72
Taeniopterygidae	Taeniopterygidae	73
Rhabdiopteryx sp.	Rhabdiopteryx sp.	74
Rhabdiopteryx acuminata	R. acuminata	75
Hydropsychidae	Hydropsychidae	76
Diplectrona sp.	Diplectrona sp.	77
Diplectrona felix	D. felix	78

Table 15.2c: Taxa identified in the study described in Chapter 9 – note: taxa are ordered by order.

Full taxa name	Abbreviated taxa name	Taxa ID
Hydropsyche sp.	Hydropsyche sp.	79
Hydropsyche siltalai	H. siltalai	80
Hydroptilidae	Hydroptilidae	81
Hydroptila sp.	Hydroptila sp.	82
Leptoceridae	Leptoceridae	83
Limnephilidae	Limnephilidae	84
Chaetopteryx sp.	Chaetopteryx sp.	85
Chaetopteryx villosa	C. villosa	86
Drusus sp.	Drusus sp.	87
Drusus annulatus	D. annulatus	88
Polycentropodidae	Polycentropodidae	89
Plectrocnemia sp.	Plectrocnemia sp.	90
Plectrocnemia conspersa	P. conspersa	91
Polycentropus	Polycentropus	92
Polycentropus flavomaculatus	P. flavomaculatus	93
Polycentropus kingi	P. kingi	94
Polycentropus flavomaculatus/ kingi	P. flavo/ kingi	95
Psychomyiidae	Psychomyiidae	96
Tinodes	Tinodes	97
Tinodes waeneri	T. waeneri	98
Rhyacophilidae	Rhyacophilidae	99
Rhyacophila	Rhyacophila	100
Rhyacophila dorsalis	R. dorsalis	101
Sericostomatidae	Sericostomatidae	102
Sericostoma sp.	Sericostoma sp.	103
Sericostoma personatum	S. personatum	104
Oligochaeta	Oligochaeta	105
Sphaeriidae	Sphaeriidae	106
Pisidium sp.	Pisidium sp.	107