Assessing how flow regime changes affect biotic indicators of ecosystem health in UK Chalk streams

Jennifer Anne Garbe

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"To see fish dying in the river is a bit disconcerting. The fish are part of the ecosystem. If that goes, you have had it"

Jim Smith

To my parents

ABSTRACT

The overall aim of this thesis was to assess the effects of changes in low flows on biotic indicators of environmental quality on chalk streams in the context of water trading. Water trading is likely to go ahead in England in the upcoming years with little assessment of the likely impacts on the freshwater environment and subsequently the ecosystem services provided. Using the river Nar in Norfolk (UK) as a case study, a method using habitat models and data collection was used to investigate this.

The methods were derived into three research questions: Firstly data collected in the field and EA data were used to investigate the impact of low flows on the three indicator species: Fish (brown trout), Macrophytes (Crowfoot) and benthic macro-invertebrate (Mayfly) (RQ1). Secondly hydraulic and habitat models (Flood modeller, TUFLOW, CASiMiR) were built to show how flows affected habitat availability (RQ2). Finally a trading model was developed by a team at Manchester University to show how water trading influences flow. These flow scenarios were run through the habitat models to show how water trading affects the habitat availability of the indicator species (RQ3).

The results showed how the indicator species are generally more affected by the antecedent flow conditions as opposed to the daily flows. Furthermore the difference in habitat between the typologies was highlighted during the first research question where the fen reach generally provided poorer habitat than the chalk reach. A key finding in RQ1 was that brown trout numbers increase when BMI have better quality and therefore when the brown trout have more food sources. This led onto a key area of research which investigated the interconnectedness of species. It was found that in order to use habitat models to fully assess habitat availability, not only do the hydraulic components need to be addressed but a species' biotic dependants i.e. food sources and refugia also need to be taken into account. RQ2 described a novel approach to do this and showed how spawning brown trout's habitat availability changes when their biotic dependants are included. Finally RQ3 showed how water trading does affect habitat availability but the small impact should be weighed up against the benefits to water resources.

Three main conclusions could be drawn from this research; firstly how there is a site specific nature of habitats and management should reflect these differences, secondly there is much uncertainty around habitat model and finally water trading does not impact habitat availability if HOF is activated.

These findings and methods can be taken forward with the increase in water trading in England which would enable environmentally efficient water trades whilst being beneficial to effective water resource management.

DECLARATION STATEMENT

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ABBREVIATIONS

ASPT	Average Score Per Taxon
BAP	Biodiversity Action Plan
BFI	Base Flow Index
BMI	Benthic macro-invertebrate
BMWP	Biological Monitoring Working Party
CAMS	Catchment Abstraction Management Strategy
CASiMiR	Computer Aided Simulation System for Instream Flow Requirements
DRIFT	Downstream Response to Imposed Flow Transformations
EA	Environment Agency
EF	Environmental Flow
EFI	Environmental Flow Indicator
ES	Ecosystem Services
GES	Good Ecological Status
GP	Generalised Parent approach
HHS	Hydraulic Habitat Suitability index
HMC	Habitat Modification Class
HMM	Hidden Markov Approach
HMS	Habitat Modification Score
HOF	Hands off Flow
HQA	Habitat Quality Assessment
HSC	Habitat Suitability Curves
HSI	Habitat Suitability Indices
IFIM	Instream Flow Incremental Methodology
LIFE	Lotic Invertebrate Index for Flow Evaluation
MEA	Millennium Ecosystem Assessment
NEA	National Ecosystem Assessment
PHABSIM	Physical Habitat Simulation System
RHS	River Habitat Survey
RQ	Research question
RS	RiverSurveyor
RSA	Restoring Sustainable Abstraction
SAC	Special Areas of Conservation
SI	Suitability Index
SSSI	Site of Special Scientific Interest

TWSTT	Transforming Water Scarcity Through Trading
WEI	Water Exploitation Index
WFD	Water Framework Directive
WUA	Weighted Usable Area
XS	Cross section

Chapter 1- Introduction

1.1 Background

Water scarcity resulting from hydrological and anthropogenic pressure not only causes direct implications to the human population but can also be the driver of many stressors on river ecosystems. Over- abstraction and hydrological alteration is a significant influence on this water scarcity, and is likely to increase the vulnerability of river ecosystems to extreme events and has impaired ecological health in many rivers by disrupting natural flows which sustain freshwater ecosystems (Garrick et al., 2009; Ledger and Milner 2015). Low or intermittent flows are often the result of these pressures which has direct effects on hydrological connectivity, biodiversity, water quality, pollution, and river ecosystem functioning (Blasco et al., 2015). These issues are heightened during hydrological drought periods. In Australia, for example, historic flow regulation and impoundments have been attributed to many ecological changes, for instance loss of wetlands, decline of riparian forests, changes in aquatic plant communities, population and species diversity declines of invertebrates, fish and waterbirds, including some invertebrate extinctions (Arthington and Pusey 2003). The need to quantify and predict this degradation is of vital importance in order to protect the ecosystems services provided by freshwater resources. These are the fundamental services which ecosystems provide to support and maintain human life, extreme events such as drought can have a major influence on these ecosystem services with potentially significant economic effects (Salles 2011; Ledger and Milner 2015). Due to major reports predicting a decline in cold extremes and an increase in warm extremes such as heat waves, it can be argued that the need to understand the ecological effects of extreme events such as drought, has never been greater (Ledger and Milner 2015).

In response to this water scarcity, water trading measures are being promoted to reallocate water rights and resources more fairly whilst simultaneously protecting water for the environment and restoring freshwater ecosystem health (Le Quesne et al., 2007; Garrick et al., 2009). Water trading involves the transfer of rights of the license to abstract water from one user to another thus allowing more efficient water abstractions whilst also enforcing environmental protections by preventing unnecessary abstractions (Erfani et al., 2015).

Achieving equilibrium between human resource needs and environmental protection is an important area of research. In order to address the environmental conditions required to sustain species and habitats whilst also enabling a freshwater resource for human use, the discipline of Eco-hydrology was developed, and is defined as: ‘the linking of knowledge from hydrological, hydraulic, geomorphological and ecological sciences to predict the response of freshwater biota and ecosystems to a variation of abiotic factors over a range of spatial and temporal scales’ (Hannah et al., 2004). By assessing the needs of freshwater dependant habitats and ecosystems, sustainable volumes of water can be abstracted whilst protecting the needs of the environment.

1.2 Problem/purpose

The current abstraction regime used in England will come under increasing pressure due to climate change, increasing water demand from an increasing population and stricter environmental standards to meet the aims of the Water Framework Directive (WFD) (Lumbroso et al., 2014). In 2008, 18% of management units in England were over-licensed, and 15% were over-abstracted, most of these being in the South-East of England (Stern 2013). Many water abstraction licensees have licenses which they do not exploit and as there are no incentives to relinquish them and they are often kept for use in times of drought (Acreman and Ferguson 2010). The abstraction regime is therefore unlikely to efficiently deal with extended periods of water scarcity and is therefore under reform in order to better protect the environment.

Water trading is being promoted in England as a way to alleviate water scarcity issues, particularly in drought prone areas such as the South East. The Environment Agency (EA) encourages this trading as it enables water resource management that meets the needs of humans whilst protecting the environment by preventing unnecessary abstractions (EA 2010). Water trading has been permitted in England for around the past 10 years, however barriers to this trading has limited the trades (Lumbroso et al., 2014). The Water Act 2014, derived from the Water White Paper, aims to implement a more efficient use of the water that is abstracted. Due to the significant push for water trading from the Water Act 2014 and the aims of the WFD, trading measures are likely to go ahead with little investigation into how this impacts upon the environment. The need to address the consequences to the species of the new water trading measures encouraged by the EA is of great importance in order to protect the vital ecosystem services provided by the species. The purpose of this thesis was therefore to assess the effects of changes in low flows caused by water trading on biotic indicators of environmental quality on chalk streams.

1.3 Transforming Water Scarcity Through Trading (TWSTT)

This research was part of a wider project entitled Transforming Water Scarcity Through Trading (TWSTT) which consisted of a team of 10 academics from Cranfield University, Manchester University, the University of Leeds and Heriot-Watt University. The project consisted of various work converging to develop a Market Simulator to investigate the economic impacts of water trading on the case study catchment (Section 1.4). Theory work on social and political opposition to water trading was completed by the team at Cranfield whilst the impact of agricultural water abstraction and trading was completed by the team at Leeds. The final optimization model was built by the team at Manchester (TWSTT 2012). Heriot-Watt aimed to show how these water trading measures impacted upon the environment.

1.4 Study catchment

The study focussed on the River Nar, a chalk stream in Norfolk in the South-East of England (Figure 1.1). Its distinctive progression from a chalk to fen stretch of the river gives the river a Site of Special Scientific Interest (SSSI) designation, with the chalk reach being particularly sensitive to low flows. Despite its status of high conservation value, abstraction, diffuse pollution and the legacy of channel modifications all contribute to pressures on the ecology of the river, around 90% of the total length being modified to some degree (Norfolk Rivers Trust 2013). Abstraction is a significant problem in the river; the lower river (downstream of Narborough) is classified as 'over-licensed', whilst the upper river is classified as 'over-abstracted' (EA 2005a). During the most extreme hydrological drought year on record at Marham (1991) the river failed its flow targets as set for the WFD to reflect the sensitivity of ecology in the river (Norfolk Rivers Trust 2013).

In response to the water scarcity issues, water trading has been proposed on the river; however the subsequent effects of this on species have not been explored. Abstraction is more safely guarded on the river due to its SSSI status, this could provide more limitation and constraints to buying/selling licenses (EA 2013b).

The river is approximately 42km in length with a catchment area of 260km^2 and one gauging station at Marham, situated around the dividing point between chalk and fen section (Figure 1.1).

The river was chosen as the projects case study catchment predominantly due to its abstraction sensitivity and ecological value (i.e. SSSI). Furthermore its geographical location means it is sensitive drought.

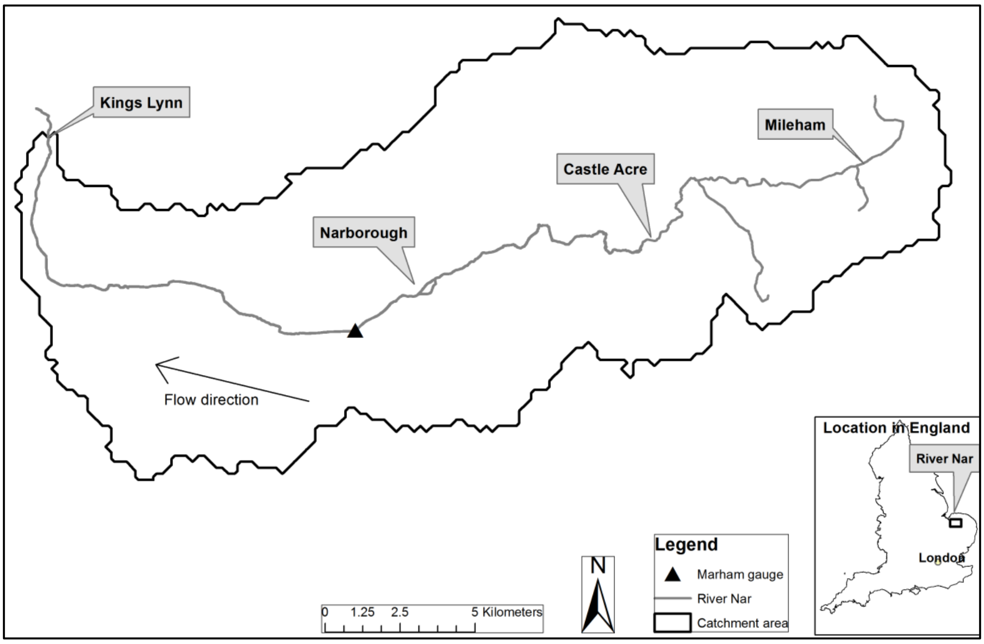


Figure 1.1- The River Nar catchment and location in the UK

1.4.1 Chalk streams

The chalk stream river provides a very specific ecological environment supporting a diverse and rich community of plant, invertebrate and fish assemblages (Berrie 1992). Chalk streams rise in cretaceous chalk aquifers through springs which makes them rich in calcium carbonate. Around 80% of the annual discharge of chalk streams comes from the aquifer, the remaining 20% comes from rainfall and runoff, thus moderate rainfall events create little difference to stream flow or turbidity. For the same reason, the temperature of the stream remains fairly constant throughout the year. The combination of factors in the stream results in a high primary production which creates a rich abundance of water Crowfoot (*Ranunculus*) and many other nutrient rich macrophytes. The high growth of macrophytes create concentrations of dissolved oxygen which are ideal for the resident trout requirements, and also provide shelter (Mann et al., 1989).

In a study by Wright and Berrie (1987), the ecological impacts of drought on a chalk stream were studied. Drought and abstraction were proven to have severe effects on the invertebrate and fish. Macrophytes however recovered rapidly after the return of flow as in intermittent sections of the river, flora had adapted to seasonal drought. The impacts were more severe in the upper perennial reaches where the impact of siltation on macrophytes and macroinvertebrate were still apparent a year after the drought had ceased. Thus, different species react in distinctive ways to low flows and drought.

1.4.2 Water abstraction and drought

Abstraction is a key issue threatening the ecology of the river, the upper Nar (above Narborough) is classified as 'over-licensed' for groundwater and 'over-abstracted' for surface water. The lower Nar (below Narborough) is classified as; 'no water available' for groundwater and 'over licensed' for surface water (Norfolk Rivers Trust 2013). Many local people and local experts remarked that if everyone abstracted their full amount of licensed abstractions, there would be no water left in the river. Over-abstraction has large impacts to the ecology of the river; pressures on the river are heightened if flows are artificially reduced to a great extent. The river warms up quicker in hot weather, the upper reaches dry more quickly, rare and vital macrophytes such as Crowfoot suffer from lack of oxygen and the volume of available habitat for fish and benthic-macroinvertebrate diminish (Norfolk Rivers Trust 2013). During drought periods, abstraction intensifies the impacts of drought; this is when the ecology is most severely threatened.

In an attempt to protect the freshwater ecology in the river from low flows, a condition is put onto abstraction licenses which require abstraction to stop or be limited when the river level falls below a specific point. These are known as Hands-off-Flows (HOF) (EA 2010). A HOF of Q_{33} (99.2ml/d) and Q_{54} (131.5ml/d) are given to abstraction licenses in the upper and lower Nar respectively (EA 2005a).

1.4.3 Geology and land use

The River Nar flows over a mixed geology overlain by quaternary sediments of mixed origin. Substrates found on the bed of the river are a relic of historical geomorphological processes; this makes the river sensitive to alterations in morphology from anthropogenic changes. There is little natural sediment supply of coarser materials and therefore little scope for natural adjustments to channel modifications. Fine sediment from roads, fields and drainage ditches accumulates within the river bed and can be detrimental to natural ecological functions due to a lack of natural sediment flushing (Sear et al., 2006).

There are two main channel units on the river: the upper river (above Narborough) is a freshwater fluvial catchment draining the plateau and chalk scarp to Narborough. The lower river is a low gradient alluvial, formerly tidal river section (Figure 1.2). The two sections are distinctive due to physical differences and the history and type of modification. The main difference however is gradient, the average gradient of the upper Nar is 0.002 (m/m) (typical of a chalk stream (Mann et al., 1989)) whilst in the lower river it is exceptionally flat with a gradient of only 0.00003 (m/m) (typical of a lowland river (Sear et al., 2006)).

Land uses within the catchment varies, with arable land dominating around 75% of the catchment, sheep and cattle grazing is found in the riparian land in the upper Nar, and pig farming is also present which provides a threat to the ecology of the river due to creating a significant input of sediment (6 tons per hour recorded during one summer storm). Sand and gravel pits, forestry and fish farms are also found on the catchment (Norfolk Rivers Trust 2013).

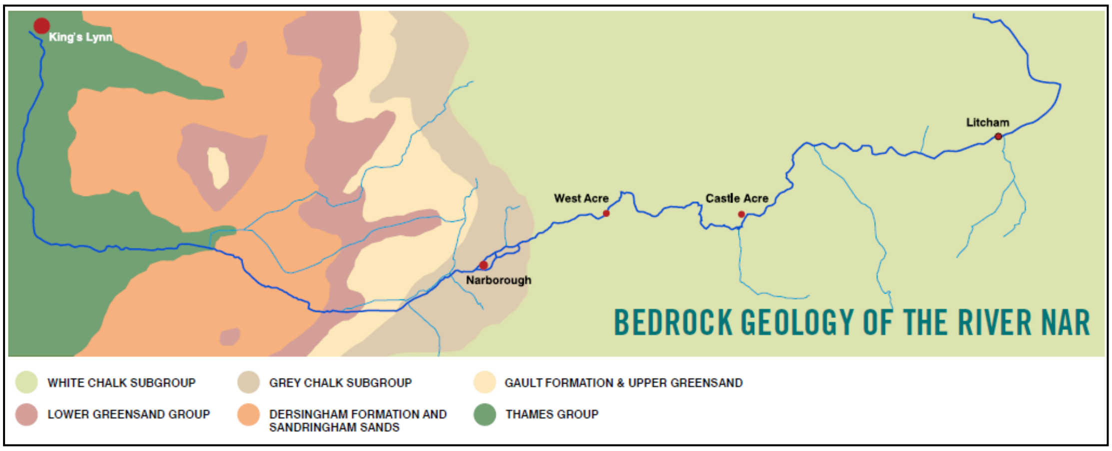


Figure 1.2- Bedrock geology of the River Nar (Norfolk Rivers Trust 2013)

1.4.4 Hydrology and river flows

The river has two natural hydrological controls on flow processes. Firstly a ground water dominated flow regime which creates a high Base Flow Index (BFI) of 0.9 which is an indication of groundwater contribution to surface water. This high BFI leads to a flow regime with seasonal rise in water levels, peaking in March and April when flooding occurs. In the past this was managed by water meadows allowing inundation of fields near Castle Acre. Aquifer recharge occurs in autumn and thus low flows are a function of low autumn rainfall in the preceding year (Sear et al., 2006).

The second hydrological control is a tidal control at the mouth of the river. Tidal ponding occurs up to 13.5km upstream of the confluence with the Ouse during spring tides. There are further modifications by hydrological controls including abstractions and discharges from the perennial channel network and aquifer, a tidal flap, a flood diversion channel, a flood storage reservoir, field drainage systems, and pumped drainage systems which all contribute to the modified flow network in the river.

There is a relatively small mean flow at the Marham gauge (Figure 1.1) of 1.14m^3/s with the highest recorded flow of 7.8m^3/s and lowest of 0.14m^3/s (between 1953 and 2014).

1.4.5 Habitats

The River Nar has a rich diversity of habitats and species and has SSSI status. Recent and continuous restoration works have helped improve the habitats in and around the river. Invasive species are present on the river; in 2013 American Signal crayfish were found which could be devastating to native white-clawed crayfish (BBC 2013). Specific information about fish, macrophytes and Benthic macro-invertebrate (BMI) in the river is provided in Section 1.5.

1.5 Indicator species and case study river environment

Indicator species were determined in order to assess the impacts of water trading and flow regime change on the aquatic environment. The species were chosen primarily based on the major report by SNIFFER (2012) which reviewed waterbodies for the Water Framework Directive (WFD), in which there are six ecological indicators of the effects of abstraction and flow regulation on river health:

- Physical- 19 indicators, e.g. loss of riffles/runs, absence of gravel substrate
- Fish- 8 indicators e.g. absence of trout and salmon
- Macroinvertebrate- 7 indicators e.g. reduction in taxon richness, absence of Baetid Mayfly
- Macrophytes, bryophytes and diatom- 14 indicators e.g. algae covering submerged macrophytes, dominance of emergent plants in relation to submerged
- Amphibian- 2 indicators e.g. presence of tadpoles in late summer/spring
- Riparian vegetation- 3 indicators e.g. loss of wetland species

The indicator species were also determined based on the case study river environment. Scoping exercises (October 2012 and January 2013) were carried out on the river to identify what species were present and establish their ecological niche; data availability was also taken into account. One indicator species became immediately apparent during scoping exercises due to the nature of the Chalk stream. The large abundances of Crowfoot (*Ranunculus*) were present due to the conditions in the chalk stream (see section 1.4.1). The large abundances of macrophytes in chalk streams provide food and shelter for dense and diverse populations of BMI (Mann et al., 1989). Subsequent to assessing EA data, the significant abundance of BMI in the river and long recorded data sets became apparent. The EA had current and historical data on BMI abundances and on macrophytes. Therefore macrophytes and BMI were chosen to represent the instream environment of the River Nar.

Ensuring concentrations of dissolved oxygen, the low spring and summer water temperatures in chalk streams provide ideal habitat requirements for brown trout. The abundance and fluctuation in populations of brown trout in the River Nar became apparent after assessing the EA's historical records.

The riparian environment was not incorporated into analysis because as the study seeks to assess flow stresses on the environment, instream habitats directly affected by flow were considered more important. Amphibians were not chosen for the same reasons. Whilst the physical environment was not directly assessed, the hydraulic models picked up on some of the physical factors affected during different flow regimes. The specific species from each of the indicators were chosen and are justified below.

1.5.1 Fish

Fish are excellent indicators of aquatic ecosystem health (Benejam et al., 2009). Fourteen species of fish can be found on the River Nar, two of which are Biodiversity Action Plan (BAP) protected species; brown trout (*Salmo trutta*) and European eel (*Anguilla Anguilla*). Numbers of brown trout have steadily increased since 1985 whilst European eel species continued to be the dominant species. Chubb (*Leuciscus cephalus*), Tench (*Tinca tinca*), Gudgeon (*Gobio gobio*), Rudd (*Scardinius erythrophthalamus*), Bullhead (*Cottus gobio*), Rainbow trout (*Salmo gairdneri*), Spined loach (*Cobitis taenia*), Roach and Bream are also found on the River Nar.

Brown trout were chosen as the specific species to focus on due to their importance socially, economically and ecologically both within Europe and the UK, and are often used as fish indicator species for many EU countries (Conallin et al., 2014). Chalk streams provide a pristine habitat for Brown Trout and they are highly sensitive to their surrounding environment (Berrie 1992), this makes the species appropriate for assessing how changing flows impact upon them. Many studies have been carried out on brown trout habitat requirements and the flow requirements of salmonoids are relatively well understood due to their economic importance (Acreman et al., 2008b). Thus this information can be used to assess how water trading impacts upon them.

1.5.2 Macrophytes

Macrophytes are considered fundamental to the structure and functioning of the freshwater ecosystem and are important controls of the ecological stability of the systems. There is limited analysis on macrophyte flow requirements, therefore understanding the key processes and factors that control and regulate the dynamics of river macrophytes is important in furthering knowledge on the structure and functioning of river communities (Acreman et al., 2008b; Franklin et al., 2008). Furthermore macrophytes are required to be used in the WFD to facilitate the establishment of good ecological status (Dawson 2002).

The scoping study revealed how rich and diverse the macrophytes are in the river where the first 10km from the source to West Lexham is typical of a calcareous lowland ditch area with a large amount of starwort (*Callitriche spp*) and reed sweet grass (*Glyceria maxima*). The following 12km to Narborough Mill exhibits faster flows over large gravel substrate, the rich chalk stream plants in this section include narrow-leaved-water parsnip (*Berula erecta*), mare's-tail (*Hippuris vulgaris*), greater tussock-sedge (*Carex paniculata*), water Crowfoot (*Ranunculus pseudofluitans*) and opposite leaved pondweed (*Groenlandia densa*). The final 18.5km has several species not found in the upper reaches which include pondweeds (*Potamogeton spp*), water Crowfoots (*Ranunculus spp*), hornwort (*Ceratophyllum demersum*), water milfoil (*Myriophyllum spicatum*) and river water dropwort (*Oenanthe aquatic*).

Macrophyte type and abundance is greatly affected by the nature of the river i.e. chalk and fen. Water Crowfoot (*Ranunculus*) is a rare chalk stream macrophyte, which the River Nar has an abundance of, furthermore research has suggested that *Ranunculus* in chalk streams has been in decline over the last 10 years, low flows due to changing rainfall patterns, siltation, over- abstraction, enrichment, and channel management have been attributed to this decline (Cranston and Darby 2004). Therefore this species was chosen due to its distinctive environmental requirements.

1.5.3 Benthic macro-invertebrate (BMI)

Benthic macroinvertebrate (BMI) were used as the final indicator species. BMI provide a good indicator of biological water quality dependant on abundance and type of species present. Furthermore BMI are a valuable early warning system to potential stresses in lotic environments (Rylands 2012). Many studies have been carried out on how BMI are affected by flow regimes particularly in chalk streams (e.g. Wilby 2010). This said however, there is little knowledge on the ecological requirements of individual species; river flow, temperature and substrate are the dominant variables controlling their distribution and survival (Acreman et al., 2008b). Therefore BMI provide a good species to relate to the flow of the river and demonstrate linkage between hydrological variables and system response.

The WFD classifies invertebrates in the River Nar as ‘good’ in all river sections. Parts of the upper Nar have recently undergone restoration works (2012), where pre-restoration it was found that BMWP (British Monitoring Working Party) scores exceeded the benchmark threshold which indicated a relative ecosystem health. Furthermore the species found were indicative of pristine and optimal habitat for a headwater stream. One meadow however exhibited results which indicated the reach was being subject to limitations in its ecological functioning (Mandley 2013).

Mayfly (*Baetis Spp.*) were chosen as a focus due to their abundance in the river and their importance as a food source for brown trout. Furthermore Mayfly have a unique lifecycle and are selective in their choice locations.

1.6 Research aims

In the context of water trading, the overall aim of this thesis was to assess the effects of changes in low flows on biotic indicators of environmental quality on chalk stream. Three main research questions were used in order to investigate this; these are introduced in subsequent sections.

1.6.1 Research question 1 (RQ1)

How are the ecosystem indicators affected during low flows?

This research question aimed to establish a baseline of quality and abundance of species in the river and to investigate linkages and relationships between these species and flows. This leads on to the findings in RQ2 and RQ3.

Measured BMI and macrophyte data collected in the field alongside Environment Agency BMI data, macrophyte data, electro- fishing and River Habitat Survey (RHS) were used to assess each of the indicator species. These were used to show how habitats change along the river and according to different measured flows. 5 recordings over a 15 month period of macrophyte and BMI data showed the natural cycle of the species in accordance with the different daily, seasonal and antecedent flow conditions. Statistical analysis was also used to investigate whether species abundances are statistically correlated to daily and antecedent flows.

1.6.2 Research question 2 (RQ2)

How useful are numerical models in investigating how low flow periods impact upon the ecosystem indicators?

Using the baseline findings from RQ1, this research question investigated what we can predict and understand by using habitat models. In order to understand the non-linear relationship between species' habitat availability and flow, hydraulic and habitat models were built based on collected data. This showed how habitat availability is altered based on different flow scenarios for the indicator species.

A common criticism of habitat models is that only hydraulic components are used to assess habitat availability (Orth 1987; Maddock 1999). However fish, for example, are affected by many more abiotic and biotic factors such as food availability, refugia and water quality. The analysis completed in this thesis introduces a novel approach integrating these considerations. Using the data and analysis from RQ1, brown trout habitat availability throughout the 32 year modelled period was determined incorporating food source (Mayfly) and refugia (Crowfoot) habitat availability. Thereby showing the critical low flows for all the species and the total habitat availability for brown trout, including their dependant biotic variables.

1.6.3 Research question 3 (RQ3)

How does water trading at a catchment scale impact upon the ecosystem indicators?

The Water Act 2014 promoted the need to use water trading as a measure to reduce water scarcity. However this water trading is accelerating with little investigation as to how this impacts the environment. This research question therefore aimed to use habitat models built for RQ2 to explore this likely impact. The output from the final trading model, developed by the team at Manchester University (Harou and Erfani 2014), determined different flows according to three different trading scenarios. These resulting flows were then input to the habitat models built for RQ2 to show the effect of the trading scenarios on the indicator species.

1.7 Layout of thesis

Chapter 2 describes relevant literature and a background to modelling. A review of relevant software is presented with a justification of the chosen software for modelling.

Chapter 3 presents the methods of achieving each research question. And furthermore describes the analysis that took place in each.

Chapter 4 provides all information regarding the hydraulic model development for both 1D and 2D. The habitat model builds are also described, explaining the data requirements.

Chapter 5, 6 and 7 gives the results for RQ1, RQ2 and RQ3 respectively. Finally *chapters 8 and 9* bring the work together in the discussion and conclusions section.

Chapter 2- Literature review and modelling background

2.1 Introduction

This chapter provides background on the key topics of this study. Firstly water scarcity is discussed focusing on threats to ecosystems of over-abstraction and drought. Incentives to reduce the impacts of over-abstraction (e.g. Hands-off-flows) to achieve the Good Ecological Status (GES) set by the Water Framework Directive (WFD) are described. Subsequently water markets and trading are discussed in response to the water scarcity issues.

Ecosystem services are then introduced with a focus on the three indicator species used in this study; Fish (Brown trout, *Salmo Trutta*), Benthic- macro invertebrate (Mayfly, *Ephemeroptera*) and macrophytes (Water- Crowfoot, *Ranunculus*). The reasons these species were chosen are detailed in Section 1.5. This section describes their background and ecology and the main threats they face and how these threats affect their habitat requirements.

Finally, a background on hydraulic and habitat modelling is considered with a review and justification of the software chosen for this study.

2.2 Water scarcity

Water scarcity is a key threat to many river ecosystems which can be caused by many different factors and has negative impacts on the river and ecosystem environment. Anthropogenic influences which exploit rivers for the benefit of ever increasing human populations have impacted ecosystem functions around the world; in many parts of the world water consumption exceeds water availability thus causing stress to the environment (Navarro-Ortega et al., 2015). Furthermore, traditional command and control approaches to engineering, such as large dams and river diversions, have resulted in a change to the natural flow regime and natural state of rivers. This has resulted in ecosystem degradation and aquatic habitat loss leading to the destruction of vital ecosystem services (Section 2.4) upon which humans rely (Poff et al., 1997; Postel 1998; Palmer 2010). Freshwater ecosystems are particularly vulnerable to flow regime shifts, in the UK this has contributed to populations of trout and salmon declining by 50-60% (Posthumus et al., 2010; NEA 2011).

The most significant long term threat to freshwater ecosystems is climate change, which is forecast to cause an increase in water scarcity and in the frequency of extreme events such as floods and droughts (Navarro-Ortega et al., 2015). Impacts of this climate change are predicted to create: higher water temperatures, longer ice free seasons, increased water body stratification, earlier snowmelt, increased sediment and nutrient transport, lower dissolved oxygen and increased salinity (Kundzewicz et al., 2007). These impacts along with an ever increasing population creating higher demands for water resources put huge stress on freshwater ecosystems.

Worldwide it is estimated that over 50% of the accessible surface water is already appropriated by humans, this is projected to increase to 70% by 2025 (Postel et al., 1996, cited in Postel 1998). Predictions indicate that by 2025 around 1.8 billion people will be living in water scarcity conditions and around two thirds of the global population will be under serious conditions of water stress (Navarro-Ortega et al., 2015). Water scarcity caused by anthropogenic pressures not only causes direct implications to the human population but can also be the driver of many stressors on river ecosystems. It can cause intermittent flows which has direct effects on: hydrological connectivity, biodiversity, water quality, pollution, and river ecosystem functioning (Blasco et al., 2015).

The Murray- Darling Basin in Australia suffers from threats on ecosystems from water scarcity. Flood events in Northern Victoria are known to maintain ecological connectivity on the floodplain and between major rivers, therefore supporting unique biota and playing a crucial role in the landscape ecology of the region (Ballinger & Mac Nally, 2006; cited in Peake et al., 2011). Most rivers in the region are highly regulated and much of the flows are diverted for irrigated agriculture, resulting in a decline of the natural frequency of flooding by around a third. Consequently biodiversity in the floodplains is declining (Peake et al., 2011). These issues called for a change in management regimes in order to restore appropriate flow patterns.

Changes in restoring seasonal flow patterns have proven successful for some degraded rivers where there have been dramatic improvements in floodplain forests, which provide rich wildlife habitats. For example the St Mary River in Alberta was dammed in 1900, this created an insufficient summer and spring flows which lead to riparian woodlands collapsing and 90% of cottonwoods dying between 1951 and 2000. After base flows were restored and extensive seedlings introduction many new trees and shrubs survived (Rood et al., 2005). Therefore there is need for regulations on natural flows and abstraction allowances.

Acreman et al.,(2000) notes how determining the degree of degradation of ecosystems is often subjective in nature and further discusses that other pressures such as: channelisation, sedimentation, reduced rainfall, poor site management and land use management can all contribute to the degradation, rather than only anthropogenic pressures. This has led to significant research and policy development within this field.

2.2.1 Water abstraction and drought

Abstraction is the removal of water either temporarily or permanently from water bodies such as rivers and lakes predominantly for use in agriculture, for drinking water and industries. In England and Wales the Environment Agency (EA) ensures the amounts of water abstracted are sustainable in order to protect the ever threatened water resources and environments which depend on them. The main challenge to water managers is how much water can be abstracted before the ability to meet social, ecological and economic needs is affected (Kashaigili et al., 2007; EA 2010). Many rivers in England are now classified as ‘under stress’ due to the change to the natural flow regime and physical diversity within rivers. Furthermore, due to pollution and over-abstraction only 27% of rivers in England are fully functioning ecosystems (DEFRA 2011). Additionally a third of river catchments in England are threatened by excessively high abstraction levels (Ecologist 2010).

The over-abstraction of freshwater is affecting ecologically dependent water ecosystem services such as fish and nutrient cycling (more in section 2.4) both in England and around the world. Rivers affected by over-abstraction have a decrease in channel maintenance flow and a decrease in water quality and ultimately have a degradation in river systems (Acreman and Ferguson 2010; Schinegger et al., 2012).

Freshwater abstractions create one of the most significant pressures on the quantity of freshwater resources, in Europe 353km^3 of water is abstracted per year, which amounts to 10% of Europe’s total freshwater resources. ‘Water- Stressed’ countries are measured through a Water Exploitation Index (WEI) which is the total water abstraction divided by the long term available annual resource i.e. how the total water abstraction puts pressure on water resources. In Europe, 4 countries, representing 18% of the population, are classified as ‘water-stressed’ (WEI>20%), namely; Italy, Malta, Spain and Cyprus. A WEI between 10-20% indicates ‘low water stress’ and a WEI less than 10% indicated ‘non water stressed’. The UK has a WEI of 12% and is therefore classed as low water stressed (Marcuello et al., 2003). Therefore there is a key need to adequately protect the UK’s freshwater resources through prevention.

Poff and Zimmerman, (2010) discovered that 87% of studies on environmental flows documented changes in the relationships between hydrologic change and ecological or geomorphological change in response to reduced flow volumes.

Furthermore, whilst aquatic and water dependant organisms are directly affected by water abstraction, there are also significant indirect effects, for example, terrestrial ecosystems downstream of the location of water use may suffer from water stress through reduced natural water ability (Pfister et al., 2011). These factors must be taken into account in decision making.

Through the Catchment Abstraction Management Strategy (CAMS) process, many catchments in England and Wales have been classed as over-abstracted or over-licensed (EA 2010), a third of river catchments in England are threatened by excessively high abstraction levels (Ecologist 2010). Figure 2.1 demonstrates the amount of water abstracted by different sectors in England and Wales between 2000 and 2013, showing that public water supply is the largest sector of abstraction. In 2013 abstractions were at their highest levels since 2002, this was followed by a 3% increase on total abstractions in 2013 (DEFRA 2015).

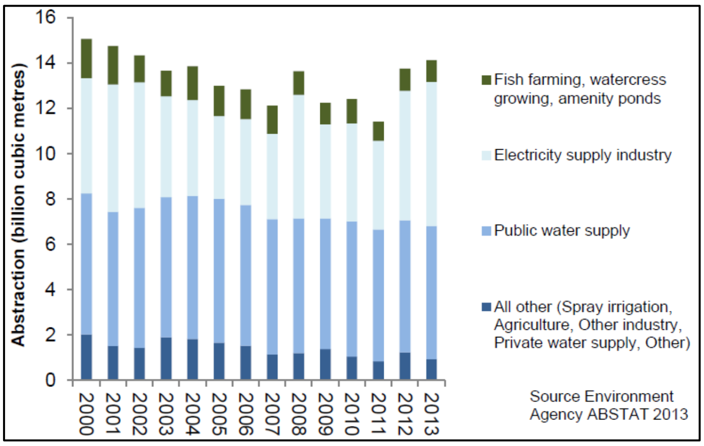


Figure 2.1- Amount of water abstracted in different sectors in England and Wales between 2000-2013 (DEFRA 2015)

Moreover, over-abstraction can exaggerate the effects of hydrological drought. In freshwater ecology droughts occur in two different forms; the predictable and the periodic seasonal droughts (Lake 2003). During the 2010-2012 droughts in England and Wales, low oxygen levels, limited effluent dilution, the development of algal blooms and desiccation of wetlands provided environmental and ecological stresses to many UK rivers (Kendon et al., 2013).

Arid or semi-arid regions of the world suffer from the effects of water abstraction more than in England. For example in Mediterranean regions water abstraction can increase the duration and magnitude of droughts, which in turn reduces flows and water quality, increases fish density and alters food resources (Benejam et al., 2009).

Abstraction licenses

Any abstraction over 20 cubic meters per day requires a license from the environment agency (EA). The EA determine how much is allowed per abstractor taking into account the needs of the environment, through the Environmental Flow Indicator (see following sections) (EA 2010). The Water Act 2014, derived from the Water White Paper, aims to implement a more efficient use of the water that is abstracted and discusses in great detail how there is a need to reform the current unsustainable abstraction regimes, the main issue is how to fairly give incentives to water companies to abstract sustainably (DEFRA 2011). In England and Wales, licenses are given to abstractors from any length of time between 6 to 24 years. If the licensee does not need to abstract they can trade their license in exchange for money to another abstractor and this can be a temporary or permanent trade (See section 1.3 for more detail).

The abstraction regime currently used in England and Wales was developed in the 1960's and was designed to manage human demand for water rather than protecting the environment (DEFRA 2011). Due to this and the damaging effects to water ecosystems from over abstraction, the EA are aiming to reform abstraction licenses using the Restoring Sustainable Abstraction (RSA) programme. This programme will affect all current and new license holders and aims to provide more protection to the environment (DEFRA 2011). Between 2012 and 2013 the EA reviewed thousands of abstraction licenses and altered many of the most damaging. Furthermore the scheme has returned water equivalent to the annual usage of water from Leeds to the environment (Solak 2013). The River Itchen in Hampshire provides an example of how the RSA has been effective, where water habitats have been protected and improved (EA 2012).

Hands off flows (HOF)

The 1995 Environment Act requires the EA to consider reasonable needs for water abstraction, the impact of abstraction on the environment and the rights of other users (Dunbar et al., 2007) and therefore they develop site-specific rules for managing water abstraction from surface and groundwater sources. These rules known as Hands off Flows (HOF) require abstraction to stop or be reduced when a river flow or level falls below a specific point. The HOF is a different amount per river, and is generally set as a percentile value, for example Q_{80}= HOF, therefore when the flow reaches Q_{80}, abstraction must stop. As more available water is allocated to abstraction licenses, the licenses are provided with an increasingly restrictive tier of HOF to ensure sufficient water is available for the environment (EA 2010). Whilst HOF are used predominantly for species protection, HOF can also be used as a protection of antecedent conditions in effect providing retention of volumes for upcoming droughts (SNIFFER 2006)..

A key issue with defining HOF is that different species have different low flow thresholds. In a study by SNIFFER (2006), experts were asked to define thresholds that would guarantee Good Ecological Status (GES) based on abstraction and impoundments. Macrophytes and macro-invertebrate experts determined thresholds for different waterbody types whilst fish experts defined thresholds for different fish species. There was much uncertainty in thresholds determined; they were broadly 10-20% allowable abstraction above Q_{95} with a HOF of Q_{95}. Furthermore, it emerged in this study that seasonal HOF values may be more appropriate, for example Q_{95} during summer and Q_{30} during winter higher flows i.e. more flexible HOF conditions. This theory is what the new Environmental Flow Indicator (EFI) criteria aims to achieve.

<u>Environmental Flow Indicator (EFI)</u>

The static thresholds set by the HOF do not always achieve ecologically or economically efficient results (Erfani et al., 2015). It is for this reason the Environmental Flow Indicator's (EFI) are now used by the EA to assess the environmental flow needs of a river with respect to the WFD requirements for rivers (EA 2010).

The EFI is a percentage deviation from the natural river flow which supports GES set for the WFD. This percentage deviation varies for different flows. It is also dependant on the ecological sensitivity of the river to changes in flow based on abstraction sensitivity bands (Table 2.1) (EA 2013a).

Table 2.1- Sensitivity bands used in EFI calculations (EA 2010)

Abstraction sensitivity	**Q30**	**Q50**	**Q70**	**Q95**
ASB3. high sensitivity	24%	20%	15%	10%
ASB2. moderate sensitivity	26%	24%	20%	15%
ASB1. low sensitivity	30%	26%	24%	20%

Figure 2.2- presents the EFI for the River Nar

- If the flow is higher than EFI at Q_{95} it should support GES: 'compliant'

- If flow falls below EFI at Q_{95} it may not support GES: 'non-compliant' (EA 2010)

At Q_{95} the recent actual flow is 63.5ml/d, the EFI is 68.6ml/d therefore the lower river Nar does not support GES. EFI's play a crucial role in the management of water allocation in England and Wales by indicating where abstraction is causing the greatest concern to the environment.

The EFI is used as a support of the WFD, to identify where hydrological alteration may be contributing to the failure of GES. If under the WFD a waterbody has an ecological status of moderate, poor or bad, it is an indication that the EFI is not sufficiently supporting the GES and therefore action must be taken on the waterbody. The EFI was mainly developed based on expert opinion, it is for this reason that the EA aims to improve the evidence base linking flow alteration and ecological response, furthermore this evidence can be used to develop and improve environmental flows (EA 2010; Klarr et al., 2014).

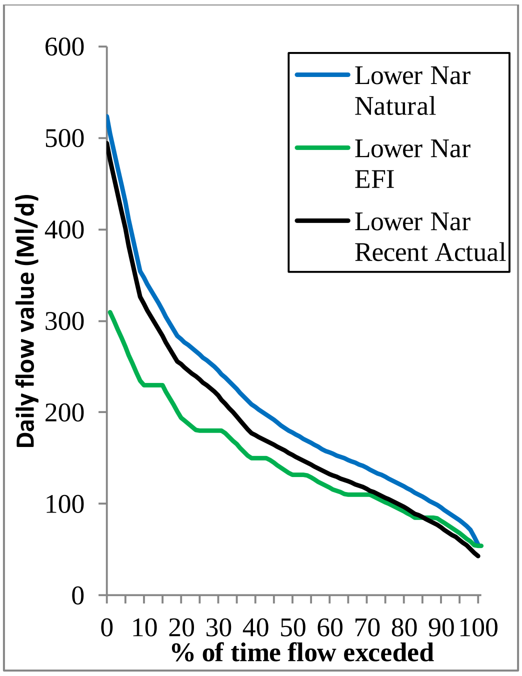


Figure 2.2- EFI, natural flow and recent actual flow for the lower River Nar

2.2.2 Water Framework Directive (WFD)

The EU Water Framework Directive 2000/60/ EC (WFD herein) is one of the most comprehensive legislations enacted in Europe for addressing the integrity of freshwaters for conservation and management needs (Radulovic et al., 2010). The main purpose of the WFD is for the sustainable management of water resources, taking into account environmental, economic and social considerations and ultimately to enhance and prevent further deterioration to freshwater ecology and ecosystems (Vlachopoulou et al., 2014). To achieve this, the WFD requires all member states of the European Union to assess and report on the ecological status of all rivers exceeding a catchment area of 10km^2. This status is determined by biological quality elements: phytoplankton, macrophytes, benthic invertebrate, fauna and fish (Schaumburg et al., 2004).

The WFD requires all surface waterbodies to achieve or maintain a status of good ecological status (GES) by 2027 and to maintain waterbodies designated as high ecological status (HES). The GES is classified based on a variety of aspects such as temperature and the level of nutrients which supports biological diversity (NIEA 2015).

Any waterbodies designated as 'heavily modified' should achieve good ecological potential (GEP) by 2015 (Acreman et al., 2008b). The CAMS legislation set by the EA supports the WFD by: providing a resource assessment for waterbodies, identifying waterbodies at risk of not achieving GES and preventing further deterioration of waterbodies for excess abstraction (EA 2010).

Figure 2.3 demonstrates that around 300 water bodies in England and Wales are at high risk of not achieving GES set by the WFD (DEFRA 2011), these are therefore areas which require initiatives to improve ecological health on the waterbodies.

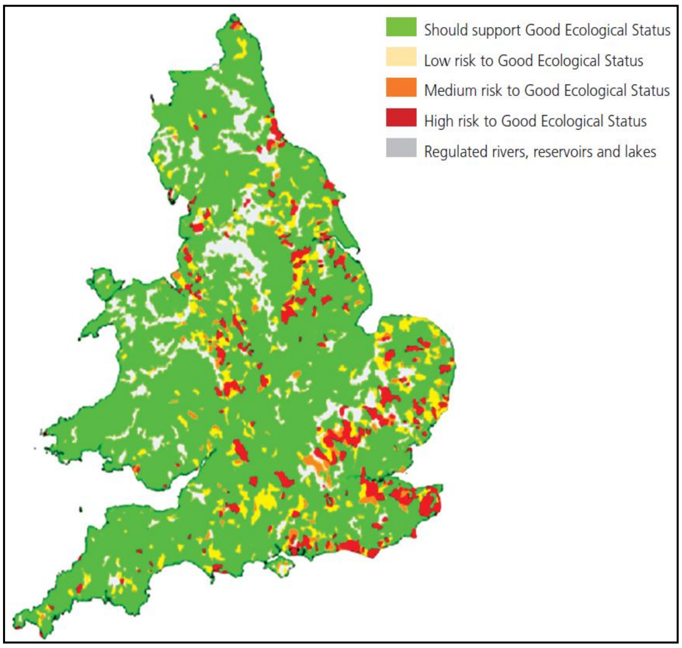


Figure 2.3- Risk to ecology from current abstraction in England and Wales (DEFRA 2011)

2.2.3 Environmental flows

In response to water scarcity issues and to help maintain the ecological processes generated from rivers, environmental flows (EF) were developed. EF cover the quantity, timing, duration, frequency and quality of water flows required to sustain water ecosystems and the human livelihoods and well-being that depend on them (Acreman and Ferguson 2010). In other words, EF determine the amount of water in a river vital to support the ecosystems related to it.

The concept of setting flows to sustain ecosystems and protect them from over-abstraction is not a new phenomenon. In the UK compensation flows have been set downstream of dams for over 100 years, similarly in the USA, minimum flows to sustain fish populations occurred in the late 1940's (Arthington et al., 2003; Dunbar et al., 2007). In the UK, these minimum standards (HOF- see section 2.2.1) were introduced under the 1963 Water Act, and with it created a primary tool for water resource management; the regulation of water abstraction licenses (Dunbar et al., 2007).

Despite this advancement in management however, it was discovered that merely setting a ‘minimum flow’ was inadequate as structures and functions of the river environment are dictated by temporal variations in flows. Setting rules to mimic the natural flow regime of a river to vary seasonally maximises effectiveness in protecting environments and habitats (Hendry et al., 2003; Arthington et al., 2006) i.e. both high and low flows are of importance to sustain freshwater ecosystems;

- High flows provide channel maintenance, wetland/floodplain flooding and connectivity on floodplains,
- Low flows are of importance for algae control and fish spawning habitat availability (Smakhtin and Eriyagama 2008; Peake et al., 2011).

Therefore floods and droughts along with regular flows are important in sustaining ecosystems. Accounting for this natural difference in flow variability provides a great challenge for water managers. EF can and have been used as a measure to help achieve GES (Acreman and Ferguson 2010) and work towards the goals of the WFD.

An example of how EF has been effective is in Tanzania where river degradation posed a serious threat to peoples livelihoods in terms of: irrigation, hydropower generation, fisheries, transportation and livestock, (Kashaigili et al., 2003). Poorer people in particular depend on these environments for a livelihood and since the rivers were drying up, action had to be taken. The government in Tanzania set a strategy to give priority for water to ecosystems once basic human needs for water were satisfied. In order to achieve this, EF assessments were carried out to determine how much water should be kept in the rivers to support vital water dependant ecosystems which people depend on for livelihood and survival (Acreman et al., 2006). This example shows how ecosystem services (see Section 2.4) can be protected and improved through the establishment of EF (Arthington et al., 2010). Similarly, King and Brown (2006) discussed how despite only half the people in South Africa having adequate water, it is a more stringent form of water management to give water allocations to ecosystem maintenance. The underlying thinking in this situation is that by allocating water to ecosystems, the services from these ecosystems will thrive, in turn creating more food and water and other goods for human use and health (Mazvimavi et al., 2007).

This study aims to investigate how flows impact on freshwater habitats and therefore the flows required to sustain species, particularly during lows flows, are of importance.

Determining environmental flows

There are numerous methods used to determine appropriate EF, 207 individual methodologies recorded for 44 countries within six world regions have been recorded (Tharme 2003). These methodologies can be applied to five different approaches:

- Hydrological methodologies- these are the most commonly used approach using the assumption that maintaining some percentage of the natural flow will provide for the environment.
- Habitat simulation methodologies- e.g. IFIM and PHABSIM (see Section 2.7), these have been considered the most scientifically and legally defensible methodologies available for assessing EF requirements. *N.b. This is the method used in this study*.
- Hydraulic rating methodology- this is also a commonly used approach which measure changes in the hydraulic habitat available based on one cross section as opposed to the habitat simulation methodologies which use multiple cross sections.
- Holistic methodologies- this is a less widely used approach involving multiple specialists providing a view on the appropriate EF. For example, the Downstream Response to Imposed Flow Transformations (DRIFT) methodology, this is a scenario based approach involving many specialists from different fields providing advice on flow requirements necessary for habitats such as fish.
- Extrapolation approach- this approach uses results of existing studies to model relationships between flow levels and environmental outcomes (Arthington et al., 2003; Tharme 2003; O'Keeffe and LeQuesne 2009).

In terms of legislation there are two approaches to setting EF’s, either objective based or scenario based. Scenario based approaches are used for example for abstraction management where a series of flow scenarios is considered. Objective based approaches are used when a desired ecological state is required, for example if the WFD dictate a desired condition or Biodiversity Action Plan (BAP) species are present (Dunbar et al., 2007).

The method, application and implementation of EF’s is site specific and depends on a number of factors such as the urgency of the problem, the resources available, the importance of the river, the difficulty of implementation and the complexity of the system (O'Keeffe and LeQuesne 2009). Thoms and Sheldon (2002) discuss determining EF’s for dryland rivers in Australia, it is made clear how management of these rivers must differ from those in more temperate regions if ecosystems are to maintain their ecological integrity. Therefore there is no one solution to environmental flow management and each river catchment must be treated independently. Despite this however, it is clear that there is an ever pressing need to recognise the value of the river, and environmental managers must find some trade-off between simple rules of thumb and full scale environmental flow assessments in order to be successful in their management and in protecting the environment (Arthington et al., 2006).

2.3 Water markets and trading

Water markets and water trading were developed in response to global water scarcity issues (Erfani et al., 2015). In this section an introduction to water markets and trading is given focusing on; the benefits they provide, areas where they have been unsuccessful and reasons for this. Both international and UK examples of water trading and markets are provided.

2.3.1 Introduction to water trading

Many water abstraction licensees have licenses which they do not exploit and are often kept for use in times of drought as there are no incentives to give up their licenses, further restricting other users from gaining new licenses (Acreman and Ferguson 2010). For this reason water trading measures were introduced, water trading involves the transfer of rights of the license to abstract water from one user to another thus allowing more efficient water abstractions whilst also promoting environmental protections by preventing unnecessary abstractions (Erfani et al., 2015). The Environment Agency (EA) encourages water trading as it enables water resource management that meets the needs of humans whilst protecting the environment by preventing unnecessary abstractions (EA 2010).

Around the world various trading initiatives have been developed to improve water quality and endangered ecosystems (Luo et al., 2007). California, Mexico, Australia and Chile are examples of areas where successful water trading occurs (Luo et al., 2007). In East Anglia (England) where water is often scarce, four farmers joined their abstraction licenses together to create a ‘Water Abstraction Group’. This group enabled the farmers to fill up a winter storage reservoir when water was plentiful instead of each being short of water in summer (EA 2011). This shows an example where water trading measures can be used on a more local scale. It is clear that without the rights to trade freely, areas with abundant water supply will overuse the supply whilst the areas with scarce water resources could suffer in drought. In the Eastern United States water is allocated according to ‘reasonableness’, meaning the deserving user may get all the water allocation. As the area usually has abundant water supply, disputes are rare in this case (Nieuwoudt and Armitage 2004). This water resource measure would be likely to fail however in areas where water is scarce and thus conflicts are common.

The current abstraction regime used in England will come under increasing pressure due to climate change, increasing water demand from an increasing population and stricter environmental standards to meet the aims of the WFD (Lumbroso et al., 2014). The abstraction regime is therefore unlikely to efficiently deal with extended periods of water scarcity and is therefore under reform in order to better protect the environment. In England water trading is being promoted as a way to alleviate water scarcity issues, particularly in drought prone areas such as the South East. The EA encourages this trading as it enables water resource management that meets the needs of humans whilst protecting the environment by preventing unnecessary abstractions (EA 2010). Water trading has been permitted in England for around the past 10 years however administrative and political barriers to this trading (Qureshi et al., 2009), has limited the trades that have occurred and when trading does occur, environmental considerations are given priority (Williams et al., 2012; Lumbroso et al., 2014).

2.3.2 Introduction to water markets

In order to understand sustainable development it is key to acknowledge that the economy is not separate to the environment (Pearce et al., 1994). This concept has become common practice in recent years, particularly after the Dublin water principles which stated the four main issues around water management, one of which is ‘Water has an economic value in all its competing uses and should be recognised as an economic and social good’ (Solanes and Gonzalez-Villarreal 2000). With the rapidly increasing shortage of water resources and often uneven supply of water in various countries and regions, water markets attempt to help alleviate these issues. Water markets have arisen all over the world in both developed and developing countries providing a more efficient mechanism for allocating scarce water supplies (Easter et al., 1999).

One of the most common areas where water markets have been used is with farmers and irrigation schemes, some being successful whilst others cause issues (Bjornlund 2003). In India for example, due to positioning of farms and added hydraulic structures to watercourses, some farmers received water for irrigation whilst others did not, consequently this led to conflict amongst farmers. Water market schemes can ease these disputes by creating a strong governmental run infrastructure (i.e. to act as a stock market) (Kerr, cited in; Pagiola et al., 2002). Amongst others, Australia, Chile, South Africa have been very successful in implementing water markets, however in some countries such as Pakistan for example, water markets are illegal which leads to illegal trading of water (Ahmad 2000).

In England water markets have not yet been established and many barriers to trading and water markets exist which can be lifted by encouraging trading by creating:

- Mechanisms to provide more information to the market.
- Mechanisms to provide more information to the traders within the market. Traders believe a high level of constant information is required in order to have a successful water market/water trading.
- Measures for streamlining the administrative process.
- A clear, transparent and consistent process of trading nationally.
- A clearer and more concise administration process for the process of trading (i.e. developing a market for trading).
- Clearer information around the nature of licence conditions, as the process of trading can be complex and hard to understand.
- Measures which help move towards more sustainable abstraction.
- Fewer constraints being added to licenses when traded, this would allow a freer market to exist (EA and OFWAT 2009).

One of the main outcomes of The Water Act 2014 was to implement more efficient use of the water that is abstracted. This is limited to trading on the same river between two tributaries so as to not remove water where not allowed under the catchment abstraction management strategy (CAMS) legislation, whilst allowing existing environmental policies such as the Hands-off-Flow (HOF) to remain the same (EA 2013b).

Due to the significant push for water trading from The Water Act 2014 and the aims of the WFD, trading measures will go ahead with little investigation into how this impacts upon the environment. The need to address the consequences to the species of the new water trading measures encouraged by the EA is of great importance in order to protect the vital ecosystem services provided by the species.

2.3.3 International water markets and their environmental influence

Internationally there is a growing understanding that water rights are important and that lack of formal water markets can create major conflicts and management problems in times of scarce water supply (Rajabu and Mahoo 2008). Most countries acknowledge the environment in their water markets in some way and market purchases for environmental use have increased in recent years (Howitt and Hansen 2005). Examples of some of these countries where water markets are used are detailed below.

South Africa:

In South Africa the Water Act of 1998 provides protection to the environment by granting water rights only to basic human needs and to environmental sustainability, the rights of farmers and irrigators are seen as a secondary importance. Legally, this must be considered in water market transactions (Nieuwoudt and Armitage 2004). In some areas of South Africa this has led to a successful and robust water market where the transfer of water from lower to higher valued crops occurs whilst also saving through conservation practices. The success of water trades is due to large land area availability, good irrigation and cropping practices (Gillitt et al., 2005). In other areas of the country no water markets exist and farmers generally use their full amount of water allowance, storing surplus for times of drought, this creates unsustainable and often unnecessary water abstractions (Nieuwoudt and Armitage 2004).

Chile:

Chile has one of the earliest and most well developed water markets in the world, where water rights have been freely traded for over two decades (Saleth and Dinar 2000) and the regulatory regime has encouraged investment leading to economic growth. This has occurred however with little protection of the environment, consequently the water market is criticised on the basis of its environmental protection (de la luz Domper 2009). This has been attributed to allowing water transfers across hydrological boundaries and inter-sectoral trades which has reduced return flows and impacted water quality through increased waste discharge (Le Quesne et al., 2007). Only since amendments to the Water Code in 2005 has there been ‘minimum ecological flows’ placed on new water rights (Williams et al., 2012).

Mexico:

The key to a successful water market does not necessarily relate to how ‘environmentally friendly’ it is, more how well it is managed through the government, legislation and stakeholder participation. Mexico for example has a well-developed water market which successfully regulates the distribution of water whilst having a conscious need to protect the environment (Salman and Bradlow 2006). The key to this success is a dual system of market and state control. Water User Associations regulate water abstraction but in times of water stress the government can step in to take control (Easter and Hearne, 1994; cited in; Takaya and Fleskens 2012).

Australia:

Australia has a highly developed water trading and water market scheme which was developed in response to environmental concerns and over allocation of water. Now the environment is viewed as a legitimate user of water in the water market through legally binding plans (Arthington and Pusey 2003; Williams et al., 2012). There was initial public reluctance until severe environmental consequences and water shortages occurred (Meinzen-Dick 2007). Now water markets exist as a key water demand strategy and to facilitate exchange of water allocations. The main aim of the water market (Watermove) is to reallocate water from low to high valued uses to promote efficiency gains in the sector. The implementation of Watermove has had great success and has generated substantial economic benefits (Brooks and Harris 2008).

California:

In California market growth in the aftermath of droughts has been driven by environmental concerns and now environmental purchases for instream uses and wildlife refuges account for around 1/3rd of total trades in the water market (Hanak 2002; Howitt and Hansen 2005).

Even in economically successful markets such as Chile, uncertainty exists around the impacts of water trading on the environment, for example, changing hydrological regimes and underestimation of sustainable environmental flows (Kiem 2013). Cross-sector trades can also impact the amount of water being returned to the river and therefore create an impact on the environment (EA 2013b). Consequently, it is necessary to quantify these impacts as the process of water trading does affect the natural flow dynamics of the river. However using international examples of water trading and water markets as a guide for the design of water markets in other countries must be considered cautiously as comparing water markets can suffer from inter-country differences i.e. in historical and legal contexts. (Brooks and Harris 2008).

Dinar et al., (1997) examined some of the positive and negative effects of water markets on the environment. If trades occur between the agricultural and urban sectors, the environment can benefit by improved management and efficiency in agriculture, therefore reducing irrigation-water related pollution. Furthermore, farmers could afford to internalise externality cost or pay higher pollution-related social cost. However it is likely that environmental degradation could be a more significant issue within water markets as transfer of water from agriculture to urban use may reduce return flows

(Dinar et al., 1997). This could have many knock-on effects for ecosystems within a catchment. Return flows and pollution can have adverse and often irreversible environmental effects of water markets (Dinar et al., 1997). Further to this, issues such as: increased load on existing drainage facilities, increased soil and water salinity, reduced reliability of irrigation water supply and reduced irrigated crop production in the area of origin, which are all environmental problems which could occur due to water markets (Dinar et al., 1997).

2.3.4 Advantages and disadvantages of water markets and trading

A summary of the advantages of water trading and markets include:

- Increased net benefit from water use.
- Creates incentives for water conservation.
- Allocation and re-allocation is achieved without political involvement.
- Purchasing water rights can be the only way to ensure water is secured for environmental needs.
- Increased efficiency means reduced environmental pollution.

However water trading and markets also come with some disadvantages:

- Differences in income levels can lead to disadvantages to poorer communities and furthermore the sale of water licenses may mean the sale of their means of livelihoods.
- Negative effects can occur for third parties not involved in the transfers.
- Trading means water is moved around the catchment or between catchments, this can have ecological impacts particularly when a downstream user sells to an upstream user.
- Trading can affect the amount of water used at different times of the year therefore impacts the natural flow regime and natural floods and droughts.
- The return- flow could change if transfers occur between sectors.

Therefore, like with any other mechanism, water trading only works under particular conditions, and the environmental impacts of water trading must be understood (Le Quesne et al., 2007).

2.4 Ecosystem services and indicator species

In recent years ecosystem services (ES) have become a subject of great interest from both a management and environmental perspective. The subject of defining ES is itself a well discussed and debated topic, however a general definition is 'the benefits human populations derive, directly or indirectly, from ecosystem functions' (Costanza et al., 1997). In other words, ES are the vital services which ecosystems provide to support and maintain human life (Salles 2011). In this section firstly the different classifications of ES are described explaining the categories which ES are split into, following on from this the ES approach is introduced which is an important management technique. Subsequently the three indicator species used in this study are described in terms of the ES they provide and their habitat and flow requirements, see section 1.5 for justification of indicator species.

2.4.1 Ecosystem Services Classifications

ES can be categorised in various ways, the original and most well-known categorisation form was created by the National Ecosystem Assessment (NEA), the following list provides the classifications described by the NEA (2011):

- *'Provisioning services* are manifested in the goods people obtain from ecosystems, such as food and fibre, fuel in the form of peat, wood or non-woody biomass, and water from rivers, lakes and aquifers...'
- *'Regulating services* provided by ecosystems are extremely diverse and include the impacts of pollination and regulation of pests and diseases on provision of ecosystem goods such as food, fuel and fibre. Other regulating services, including climate and hazard regulation, may act as final ecosystem services, or contribute significantly to final ecosystem services, such as the amount and quality of available freshwater...'
- *'Supporting services* provide the basic infrastructure of life. They include primary production, (the capture of energy from the sun to produce complex organic compounds), soil formation and the cycling of water and nutrients in terrestrial and aquatic ecosystems. All other ecosystem services – regulating, provisioning and cultural – ultimately depend on them...'
- *'Cultural services* are derived from environmental settings (places where humans interact with each other and with nature) that give rise to cultural goods and benefits... They comprise an enormous range of so-called 'green' and 'blue' spaces... Such places provide opportunities for outdoor learning and many kinds of recreation; exposure to them can have benefits including aesthetic satisfaction and improvements in health and fitness and an enhanced sense of spiritual well-being...'

Table 2.2 presents a list of the ES from freshwater resources. Criticisms around this form of classification are apparent, for example carbon sequestration could be classified as both a regulating and supporting service (de Groot et al., 2002). Furthermore there are criticisms surrounding double counting these ES, for example forests provide the carbon sequestration can also provide flood hazard mitigation and recreation (Boyd and Banzhaf 2007).

Table 2.2- Ecosystem services derived from freshwater resources

Provisioning services	**Regulating services**	**Supporting services**	**Cultural services**
- Fish: Salmon, trout, Crayfish - Reeds, osiers and watercress - Water: Drinking supply, irrigation, fish farming - Peat (compost) Navigation Health products	- Carbon regulation - Flood regulation/ water storage - Flow regulation - Water quality regulation - Local climate regulation - Fire regulation - Human health regulation - Refugia	- Biodiversity - Soil retention - Nutrient regulation - Nutrient cycling	- Science and education - Tourism and recreation - Sense of place

It can be argued that because these services are freely available with no markets and values, the protection of them has been limited which has contributed to the degradation of these ES (EC 2009). This has led to the valuation of ecosystem services which have been valued at an average of US$33 trillion per year worldwide (Costanza et al., 1997). The valuation of ES is outwith the scope of this study, however its importance remains: degrading ES through anthropogenic pressures could not only impact on human wellbeing but also impact widely on the earth's natural capital.

2.4.2 Ecosystems approach

In response to the degradation of vital ES, the ecosystems approach has been proposed at international, national, regional and local scales to ensure the environment is recognised at all stages in policy making and to acknowledge that developments must work alongside environmental protection rather than against it (Niu et al., 1993; DEFRA 2010). The approach sets a socio-economic context into which ES can be integrated into decision making, thus making the ecosystems approach a vital tool working towards the aims of the WFD (Vlachopoulou et al., 2014).

Many definitions have been determined, however most often the ecosystems approach is described as 'integrating and managing the range of demands placed on the natural environment in such a way that it can indefinitely support essential services and provides benefits for all' (DEFRA 2007b). Using the ecosystems approach and recognising that ES reflect different benefits of ecosystem management supports a more inclusive approach to river management and allows for decision making to benefit many stakeholders. Furthermore the ecosystems approach allows for more effective delivery of environmental outcomes and more informed decisions ensuring more sustainable development, and also provides a greater awareness of the value of the natural environment and more efficient use of resources (DEFRA 2007b; Everard 2012).

The UK NEA, was developed based on the ecosystems approach. The aims of this are that management of the natural environment should be focused away from sector- specific or habitat- specific decision making and instead use an integrated approach based on entire ecosystems which ensures the value of ES is adequately addressed (NEA 2011).

Frameworks have been determined for decision makers to use as a guide as to how to use the ecosystems approach, these are based on some main principles which may not be relevant to all stakeholders but are intended to be observed in all decision making strategies. These are: taking a more holistic approach to decision and policy making, ensuring the value of ES are reflected in decisions, ensuring environmental limits are respected, promoting adaptive management and making decisions at appropriate spatial scales (DEFRA 2007b).

Valuation is promoted as a way to work towards the ecosystems approach, as ultimately humans will not protect something that they do not value (Daily 1997). Valuing ES provides many benefits. For example, it can determine whether a policy that alters an ecosystem gives economic benefits or losses, it can help in choosing between competing uses for land and it can be used to create markets for ES (DEFRA 2007a).

The use of the ecosystems approach is relevant to this study as the indicator species used represent the ES in the river. Therefore by assessing how flows impact upon these species helps work towards the aims of the ecosystems approach by enabling management decisions based on enhancing and protecting the ES within the river.

2.4.3 Fish

Introduction and key threats to fish

Fish are of great importance both environmentally and economically. In terms of ES, they provide:

- Provisioning services as a source of food.
- Regulating services such as regulation of food web dynamics and transport of nutrients. Removal of fish from a river could therefore result in the ecosystem changing from one state of equilibrium to another.

- Cultural services such as the supply of recreational activities and aesthetic values. In England alone, the fisheries industry provides around £20.4 billion to the economy (Sen et al., 2011), thus reducing this would influence the economy.
- Transport of nutrients and energy between spatial boundaries when migrating and spawning. This increases primary production in areas of poor nutrient availability.
- Linkage with aquatic, aerial and terrestrial ecosystems when fed upon which provides a key component to many other species' food webs.
- Indication of ecosystem stress as they are sensitive to many factors (Holmlund and Hammer 1999).

It is therefore evident how important fish species are and how they are a key component of most aquatic ecosystems (Lake 2011). Salmonid fish in particular (e.g. brown trout and salmon) are considered of great economic and cultural importance in river ecosystems (Pennell and Prouzet 2003). Salmonid fish are however threatened by habitat loss from anthropogenic pressures. Furthermore, absence of salmonid fish in a river is considered as an indicator of high impact from over-abstraction (Pennell and Prouzet 2003; SNIFFER 2012).

Both good quality and appropriate quantity of water are of key importance for fish species. Fish require basic water quality requirements which include:

- Well oxygenated water with natural nutrient content and temperature range.
- Suitable volume and quality of water to prevent sustained variations in Ph value.
- Water without chemical contaminants and naturally low silt/fines content (Hendry et al., 2003).

Water quantity requirements for salmonid fish are described in detail in sections 4.8. 4.10 and 4.11, however generally salmonid fish require:

- Adequate flows at appropriate times of the year.
- Appropriate water depths, velocities and volumes for all lifestages.
- Regular occurrence of 'flushing flows' to ensure spawning gravels are maintained (Hendry et al., 2003).

Throughout a river system, natural changes in water velocities, volumes, depths and geometry provide a diverse range of habitats for fish including the factors noted above. Average and extreme conditions such as floods and droughts provide the diverse environments required (Whitton 1975). This extreme hydrological variation leads to fish having the skills to persist under highly variable and harsh conditions (Balcombe and Arthington 2009).

The main threats from over-abstraction and low flows to fish species are detailed below:

- Point and non-point pollution, this impacts upon water quality locally through agricultural practices (Hendry et al., 2003).
- Low flows caused by water abstraction cause sediment deposition and therefore smothering of plant life used as refugia and food sources and smothering of vial spawning grounds for fish (Hendry et al., 2003; S&TA 2014).
- Over-abstraction can also indirectly influence water quality: decreased dilution of pollutants results in increased nutrients, eutrophication, and low dissolved oxygen levels.
- Over-abstraction of freshwater has had a damaging influence on fish habitat in the UK. Resulting low flows from abstraction create a loss in wetted area and modified water velocities and depths along the river. In turn, this creates a loss in fish habitat and changes in river morphology.
- Water abstraction also increases the duration and magnitude of droughts. Drought results in a drying process which lowers water levels, depths and velocity, in turn increasing fish density and reducing food resources, thereby reducing habitat availability for fish. During drought, fish use a wide range of refuges as water levels drop, such as small springs or deep pools, migration is often required however for effective recovery (Armstrong et al., 2003; Benejam et al., 2009; Lake 2011).
- Species interactions can be altered as a result of drought, Elliott (2006) for example found that during drought habitat availability of brown trout (*Salmo Trutta*) reduced however habitat availability for bullhead (*Cottus Gobio*) increased making them the dominant species.
- The influx of large structures for flow regulation such as hydro-electric power generation create a hugely modified flow regime that results in temporal and spatial changes.
- Land use change and canalisation can create significant changes to the rivers natural flow regime. Some studies however show that changing morphology of rivers can be favourable to some habitat conditions. In a study by Millidine et al., (2012), it was discovered that a canalised reach was well suited to fry however lack of coarser substrate and high depth, low velocity areas made it unfavourable for salmon and trout parr respectively.
- The process of altering the natural flow dynamics of a river can entice more invasive species which could reduce pioneering species by using their resources. Reservoir management can often be adapted to the benefit of fisheries as they can provide a degree of peak flow control. For example increases in water releases during the

autumn and winter months can provide optimum conditions for spawning fish to ensure redds are not left dry (Hendry et al., 2003).

Ultimately the effects of anthropogenic change affect specific environmental conditions which influence fish at any one time, from discharge to terrestrial cover, as shown in figure 2.4. It is therefore difficult to depict which factors are affecting the species at any one time. Furthermore, changes in the physical or chemical habitat may not have direct effect of the fish, moreover a direct impact on the invertebrates upon which the fish depend for food. Figure 2.4 presents some of the many different factors influencing fish habitat, there are further biotic factors such as season, terrestrial cover and food availability also influencing the fish habitat. It is therefore clear that determining environmental flow requirements for fish species becomes a complex process, particularly as different species and lifestages have varying requirements.

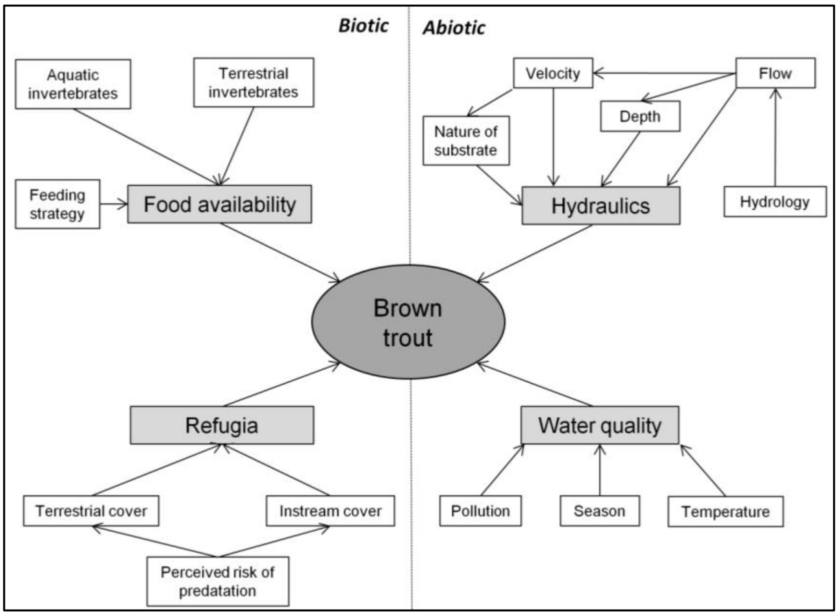


Figure 2.4- Abiotic and biotic factors affecting brown trout

Clearly then the protection of adequate flows for fish is a key issue. Generally very little or no abstraction is allowed when flows are less than Q_{95} (Acreman et al., 2008b). In order to adequately protect flows for fish, different hands- off- flows (HOF) should be set for different lifestages and species. Acreman et al.,(2008b) determines minimum flows of Q_{80} for spawning salmonids during autumn and winter and Q_{99} for rheophilic cyprinids during summer and autumn.

Whilst attempts have been made to link habitat requirements to fish species and lifestages, Hughes et al., (2001) notes how research is still a ‘patchy’ progress, and is a difficult process to model or predict.

Brown trout ecology and lifecycle

Species such as Barbel, Chub, Trout, Salmon, Carp, and Roach are a few of the common fish found in British rivers. Brown trout are commonly found in chalk stream rivers and as they have BAP protection, are highly valued fish (Berrie 1992). However, due to anthropogenic impacts and drought conditions, brown trout populations are under threat nationwide (Milner et al., 2003).

The lifecycle of brown trout (Figure 2.5) is similar to that of any other salmonoid species. Spawning adult females lay their eggs in redds in Autumn (October to December), in most UK rivers, the alveins stay in the gravel redd, feeding on their yolk sacs then emerge as fry when they begin feeding on invertebrates. Fry develop swimming behaviours to maintain position and feed in flowing water, this phase is often aggressive and territorial where high mortality rates occur. This critical period for brown trout can last for several months after emergence. Surviving fry, known as Parr then spend between 1 and 3 years in streams in Britain. Anadromous brown trout then undergo physiological changes that pre-adapt them to life in the sea and migrate as smolts in April-May (Armstrong et al., 2003; Milner et al., 2003). Those that do not migrate remain in the river as adult fish.

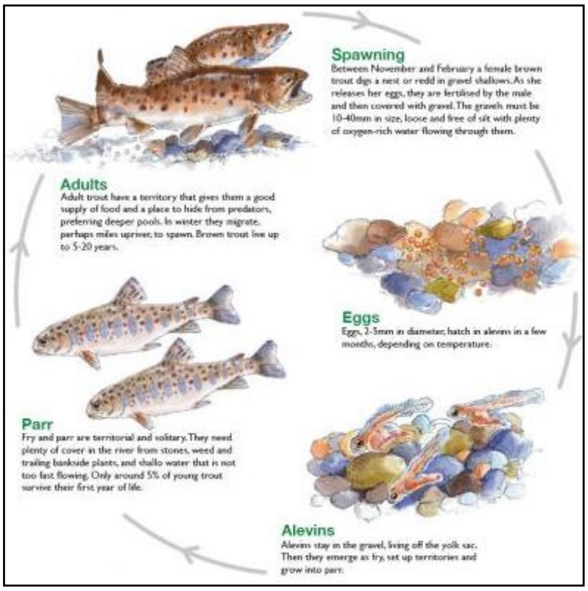


Figure 2.5- Brown trout lifecycle (The wild trout trust 2014)

Brown trout habitat and flow requirements

Note- a full literature search and synthesis of finding on flow requirements of brown trout (adult, juvenile and spawning) is found in chapter 4- Model build).

Many studies have aimed to quantify the habitat requirements of brown trout and their flow requirements are relatively well understood due to their economic importance

(Acreman et al., 2008b). The various lifestages of brown trout have different habitat requirements, all of which must be taken into account when determining environmental flows for specific rivers. Spawning brown trout have preferences of mean depths and velocities of 0.31m and 0.39m/s respectively, whilst adult brown trout have preferences of mean depths of 0.65m and velocity of 0.26m/s (Armstrong et al., 2003). Different authors report slightly different flow requirements and preferences for brown trout. This is related to different river catchments having different requirements. There are many factors which affect different stages of lifecycles and ultimately different depths and velocities depend on the river conditions and location (Verspoor et al., 2007). This is why in any situation the flow needs of brown trout are specific and must therefore be related to local management techniques (Hendry et al., 2003).

Drought has been proved to have a large impact on brown trout due to their sensitivity to flows. The anthropogenic impacts discussed in section 2.4.3 exacerbate the impacts of drought. In a study by James et al., (2010), adult brown trout populations were higher in early drought than in late drought, 238 and 69 Kilograms (KG)/ha respectively. Juvenile brown trout populations were also higher in early drought than in late drought, 43 and 23 KG/ha respectively. However fish can survive during drought and gain condition during flows and floods due to their physiological capacity to metabolise and catabolise lipids (Balcombe et al., 2012).

Depth, water velocity and substrate size are considered by most authors as the most important instream variables affecting brown trout, both directly and indirectly (Armstrong et al., 2003; Louhi et al., 2008). However in a study by Jowett (1992), abundances of adult brown trout were tested against fifty independent environmental variables. Significant positive correlations were found between trout abundance and: invertebrate biomass, temperature, cobble substrate and available habitat in the river. Invertebrate biomass was found to be the single most important factor in determining brown trout abundance, while suitable living space was deemed the second most important factor, including cover. Further to this, Eklov (1999) correlated densities of brown trout with various environmental variables and it was found that trout were more likely to be found in streams with oxygen-rich water and medium sized substrate. Higher densities of trout were also found in streams with few predators. Finally, variation in length of brown trout of the same age were explained by temperature. This shows how flow and related depth and velocity is of importance to directly provide habitat for the fish, but many more factors need to be taken into account such as food sources and refugia when determining habitat requirements of brown trout.

<u>Effect of antecedent flow conditions on fish</u>

Very few direct studies have been carried out on how antecedent flow conditions impact upon fish. In a study by Kitsios et al., (2012) on streams in Western Australia, it was discovered that there was no relationship between antecedent low flows and fish abundance, however this was attributed to dataset limitations, stream adaptation to low flow stress, sensitive species been filtered out due to recent extreme conditions and other factors than only hydrology affecting these species.

In a further study in Australia, Balcombe and Arthington (2009) found antecedent flows had a large influence on fish species abundance. Following high summer flows, water bodies supported rich and abundant fish species. However fewer species and lower numbers were recorded after periods of zero channel flow which were related to a less diverse food web and limited food resources. Furthermore, seasons were found to have a strong effect on fish abundances, in the summer-autumn period, abundances were much higher than in the autumn-winter or spring-summer seasons. Combinations of high water temperature and flow events were strong drivers of patterns found (Balcombe and Arthington 2009). Balcombe et al.,(2012) found that fish were in a better condition after recent flows and poorer condition when there had been no flows. This study however investigated only fish condition and not abundances; fish condition is a measure to determine fish well-being and robustness. Three months after a period of drying, numbers and condition of fish had being retained, this suggested algal productivity, which they feed on, was maintained to sustain the fish (Balcombe et al., 2012). This shows the difficulty of proving effect of antecedent flow conditions, as other factors must be taken into account.

2.4.4 Macrophytes

<u>Introduction and key threats to macrophytes</u>

Aquatic plants can be categorised into four groups: Submerged plants, floating leaved plants, emergent plants and algae (DEFRA 2014). Many submerged and floating macrophytes in UK rivers are of international importance and listed under Annex II of the habitats directive (92/42/EEC). The reason for this importance is that macrophytes provide many important ecosystem services, see Table 2.3.

Table 2.3- Ecosystem services provided by macrophytes (Cranston and Darby 2004)

Provisioning services	Regulating services	Supporting services	Cultural services
- Air-Water links to enable invertebrates to complete life cycles - Supply surfaces colonised by algae and diatoms	- Refugia for fish and invertebrate - Breeding sites for fish and invertebrate - Stabilises the substrate - Maintains water levels during low flows	- Nutrient cycling - Oxygenate the water - Creates structural diversity to the waterbody	- Amenity benefits - Aesthetic value - Recreational benefits

Macrophytes provide a unique species in that they are influenced by the flow regime, and the flow regime is also influenced by them, many studies have shown how macrophytes influence the structure of the instream habitat and the processes that occur (Clarke 2002). Franklin et al., (2008) noted how aquatic macrophytes act as 'biological engineers', altering the stream environment, affecting velocities, water depth, sediment patterns and water quality and providing structural habitat diversity. Hearne and Armitage (1993), observed that increased water abstraction may be possible with minimal reduction in habitats provided that macrophytes are not cut and removed. This indicates that macrophytes can be manipulated to aid management in watercourses. Likewise excessive growth of macrophytes can lead to an increased risk of flooding therefore weed cutting often occurs in rivers where there are excessive amounts of macrophytes (Bentley et al., 2014b).

Despite this importance however macrophytes, like many other aquatic species, are facing increasing pressures. Anthropogenically induced changes have contributed to an overall decline in macrophyte species richness and diversity and a homogenisation of communities (Franklin et al., 2008). Many authors have determined increasing abstraction as exacerbating the impacts of low flows (Cranston and Darby 2004). The hydrological regime has a large influence on macrophyte abundance, directly through mechanical damage and uprooting, or indirectly due to changes in sediment characteristics, nutrient and gas exchanges and competitive interactions (Franklin et al., 2008). The natural flow regime is therefore highly important for macrophytes. Research by Holmes (1999) discovered that there are very distinctive communities in different rivers even with the same characteristics, and that they can be correlated with different flows and physical habitat characteristics. Therefore this indicates that the same species can have varying responses to flows in different locations on both the same river and on different rivers. For this reason it is difficult to determine generic flow requirements for species.

As with other aquatic species, the effects of anthropogenic changes on macrophytes are exaggerated during drought periods. During drought conditions a widespread loss of aquatic habitats is apparent, with water dependant biotic communities being simplified or replaced by terrestrial communities. Changes to channel substrates provide the biggest threat to aquatic macrophytes which depend on clean gravel or pebble beds. Decreased velocities replace these substrates with silt, which in turn supports wetland rather than aquatic plants. Holmes (1999) found macrophytes to have a typical recovery period of two years, whilst Wright and Berrie (1987) found after the 1975 drought macrophytes recovered rapidly after the return of flow however invertebrate and fish took longer to recover. It was also discovered that in upper perennial reaches where loss of macrophytes had been severe, re-growth was slow and the effects of drought on macrophytes were still apparent in autumn of 1977, this again indicates site and flow characteristics influence how macrophytes respond.

Droughts reveal the exacerbated effects of water abstraction, balancing low flow and no flow in the usual perennial rivers clearly provides negative effects to all habitats. Studies have shown that environmental parameters mostly associated with positive macrophyte diversity are high local water stages, wide channels, and a high degree of semi-natural land use. On the other hand, steep channel gradients are strongly associated with negative diversity (Westwood et al., 2006). Over-abstraction would negatively impact upon these environmental parameters and therefore management techniques should aim to improve these factors.

***Ranunculus* ecology**

The *Ranunculaceae* family is a large family of 59 genera and approximately 2,500 species which occur mostly in temperate and boreal areas. Only two of the genera contain aquatic species; *Caltha* and *Ranunculus*. There are many taxa of aquatic *Ranunculus*; in the case study river (River Nar) *Ranunculus Aquatilis L* (Common water-Crowfoot) and *Ranunculus Fluitans* (River Water-Crowfoot) (Figure 2.6) are found. Common Water-Crowfoot prefers shallow water in marshes, ponds and ditches and the sheltered edges of lakes and margins of slow-flowing streams. It is widespread in Europe and is found in North Africa, North and West South America and scattered localities in Asia. River Water-Crowfoot on the other hand is found in rapidly flowing water and is the largest of the *Ranunculus* species. Its distribution is mainly controlled by its requirement for a stable substrate. In recent years the species has decreased in abundance in some rivers and is more sparsely found than Common Water-Crowfoot.

The species is endemic to Europe and found in rivers throughout England, it is however virtually absent from Ireland and Western Scotland (Spink 1992; Preston and Craft 1997). *Ranunculus* can have either submerged, finely divided (capillary) leaves or broad, floating (laminar) leaves or a combination of both (Bentley et al., 2014a).

Figure 2.6- *Ranunculus Fluitans* in the River Nar

Under the Habitats and Species Directive (92/43/EEC) the proposed Special Areas of Conservation (SAC) include the protection of floating vegetation of *Ranunculus* (Cranston and Darby 2004). Neither *Ranunculus Fluitans* nor *Ranunculus Aquatilis* are however listed as a UK BAP species.

Typically the lifecycle of *Ranunculus* starts in autumn or winter with the biomass increasing rapidly in the later winter and spring, reaching its maximum in spring or summer when flowering also occurs (Dawson 1979). Thus low abundances in autumn and winter are not necessarily related to the flow volumes and moreover related to the natural die back and growth patterns (Dawson 2002).

The successful growth of *Ranunculus* in UK chalk streams has been linked to chalk streams providing high level of nutrients and stable flows. During higher discharge in spring and summer where *Ranunculus* has a very rapid growth rate it is able to flower but not root, and the summer flushing events are also of great importance to remove algae cover. In the winter it is slower growing but more able to root. This provides a seasonal pattern of growth and recession of macrophyte lifecycles (Berrie 1992; Spink 1992; Franklin et al., 2008). In the last decade following a series of low flow years *Ranunculus* in chalk streams has suffered from a decline (Cranston and Darby 2004).

***Ranunculus* habitat and flow requirements**

Note- a full literature search and synthesis of finding on flow requirements of Ranunculus Fluitans is found in chapter 4- Model build.

Franklin et al., (2008) determined discharge and velocity, light availability, substrate and nutrient availability to be amongst the most important physical variables controlling macrophyte abundance and location. In a key study by Cranston and Darby (2004), many factors and drivers were considered as major aspects affecting *Ranunculu*s growth, shown in table 2.4.

Table 2.4- Factors and drivers affecting *Ranunculus* growth (Cranston and Darby 2004)

Factors	**Drivers**
Competition/Interaction/ Life cycle/ Colonisation	Channel management
Discharge/ Seasonal annual changes	Enrichment from point sources
Light/ shade/ Temperature	Natural climate cycles
Substrate/ Siltation	Shading by algae
Velocity/ Depths/ levels	Vegetation management
Water quality/ Enrichment/ Suspended solids	Abstraction/ Catchment water use
Grazing	Land use/ Diffuse enrichment
Physical dimensions	Rehabilitation/ Augmentation/ Fencing etc.

Over abstraction of freshwater is a driver that can have a significant impact on discharge and therefore velocity and sediment movement. Additionally low flows due to: changing rainfall patterns, enrichment, siltation and channel management all impact upon *Ranunculus* (Cranston and Darby 2004). *Ranunculus* abundance was found to have positive correlations with velocity in two chalk streams in South-East England (Wilby et al., 1998), thereby indicating daily flow conditions do have an effect on the species. This said however, studies have shown that variability in macrophyte abundance cannot always be directly attributed to variations in stream flow, and other non-flow related variables at different spatial scales, such as geology of the catchment and catchment rainfall are influencing macrophyte abundance (Westwood et al., 2006).

The natural flow variation in a river is extremely important for macrophyte survival and growth, the plant species present in any particular reach will be those that can tolerate the full range of natural river discharges, therefore implying that extreme conditions of both floods and droughts are important for macrophytes. Furthermore, Westwood et al., (2006) found a positive correlation coefficient of *Ranunculus* with stream flow. However low flow events caused by drought can have a negative impact on macrophytes (Franklin et al., 2008). In 1989 and 1990 *Ranunculus* was reported to have suffered during drought conditions in most chalk streams in England yet after particularly high flows in 1999 and 2000 *Ranunculus* is now reported to be dominant on many rivers which can be related to flow conditions (Cranston and Darby 2004).

Many studies researched the relationship between *Ranunculus* growth and environmental variables, notably, substrate, velocity and depth; the resounding conclusions were that the most important variables controlling macrophyte growth are substrate and velocity. *Ranunculus* is generally found in fast flowing waters with non-silted coarse substrate (Cranston and Darby 2004).

It has been found that there is a detrimental impact on *Ranunculus* growth due to silt accumulation as a result of low flows. However, silt is easily removed by increases in flow, therefore implying short periods of low flow would not necessarily impact on them (Cranston and Darby 2004).

<u>Effect of antecedent flow on *Ranunculus*</u>

Limited studies have been carried out on the direct effect of antecedent conditions of *Ranunculus* growth. However studies have determined that there is a link between preceding summer and winter flows during drought conditions and *Ranunculus* growth. In many rivers in England, *Ranunculus* suffered in the drought of 1976 and had subsequent recovery following higher flows. Very low flows in summer can weaken *Ranunculus* growth therefore making it more vulnerable to being removed during subsequent high flows. Similar effects occurred in 1989 and the early 1990's when there was widespread drought in the UK. However after two years of higher flows in 1999 and 2000 *Ranunculus* was reported to be dominant in many rivers. This pattern indicates poor growth in low flow years and then recovery following higher flows (Cranston and Darby 2004). Newbold (1996, cited in Cranston and Darby 2004) shows how important antecedent conditions are by giving winter and summer flows required to prevent sedimentation and promote healthy *Ranunculus* growth in rivers Kennet and Axford: 0.929-1.427 m^3/s in winter and 1.291 m^3/s in summer. Whilst this is river specific so cannot be transferred to other rivers, it shows the importance of antecedent flow conditions in *Ranunculus* growth.

2.4.5 Benthic macroinvertebrate (BMI)

<u>Introduction and key threats to BMI</u>

BMI are functionally important in many aquatic ecosystems providing ES such as nutrient cycling, sediment mixing, and energy flow through food webs (Covich et al., 1999). Equally, a significant increase in BMI can cause negative effects to ES by spreading disease to fish (Covich et al., 1999). BMI are particularly abundant in chalk streams due to high habitat diversity, large food supplies and stable flow conditions. Droughts have a detrimental impact on chalk stream BMI due to large volumes of fine sediment being deposited therefore creating unfavourable substrates (Berrie 1992; Wood and Petts 1999). River regulation can have damaging impacts on the species composition of BMI. Temperature and flow regime are two of the major factors altered by many river regulation schemes. A reduction in flow can lead to a higher abundance of species typical of slow flowing waters and likewise extreme conditions favour opportunistic species (Brittain and Saltveit 1989). This can impact on the species composition in a river which could subsequently encourage non- native species and/ or make certain species extinct.

Mayfly were chosen to represent BMI in the river predominantly due to their abundance in the river and their importance as a food source for brown trout. The following sections detail background information on Mayfly, this is followed by section 2.5 which details the scoring indices used for analysis of the BMI data.

Mayfly ecology and lifecycle

Mayfly (*Ephemeroptera*) are a small insect order, containing over 2000 species, grouped into approximately 200 genera and 19 families. Mayfly are often the most abundant taxa in BMI communities and are herbivore collector- grazers, feeding on detritus (Brittain and Saltveit 1989). They are unique amongst insect species as they have two winged adult stages, the subimago and imago. Mayfly provide important links in the food chain from primary production to secondary consumers for fish (Brittain 1982). There are four main stages of a Mayfly's lifecycle (Figure 2.7):

1) Eggs:

After swarming and mating, female Mayfly drop their eggs into the water with little regard to the conditions, however some species of female Mayfly are known to go underwater to select suitable stones to lay their eggs on, this involves testing the water quality first. Length of hatching time differs depending on species and area, for example in temperatures over 5^0C most eggs of *Baetis Rhodani* hatch in less than 10 days, however other species can differ (Brittain 1982).

2) Nymph:

Mayfly spend the majority of their lives in the aquatic environment, this can range from 3 or 4 weeks to around 2 years. Nymph development is largely influenced by environmental conditions such as food resources and temperature. During nymph stage, all Mayfly populations will experience movement which may be random, directional, daily or seasonal. During the final stages of nymph life, movement to shallower areas is common (Brittain 1982).

3) Emerger:

Emergence is a critical period for Mayfly; this is the stage between being an aquatic species to being a terrestrial one. Water temperature is one of the biggest factors influencing emergence time. Mayfly in temperate and cold regions emerge during summer (Brittain 1982).

4) Adult (Submago and imago):

Adult Mayfly live terrestrially from 1 or 2 hours to a few days where they have two functions; mating and oviposition. (Brittain 1982).

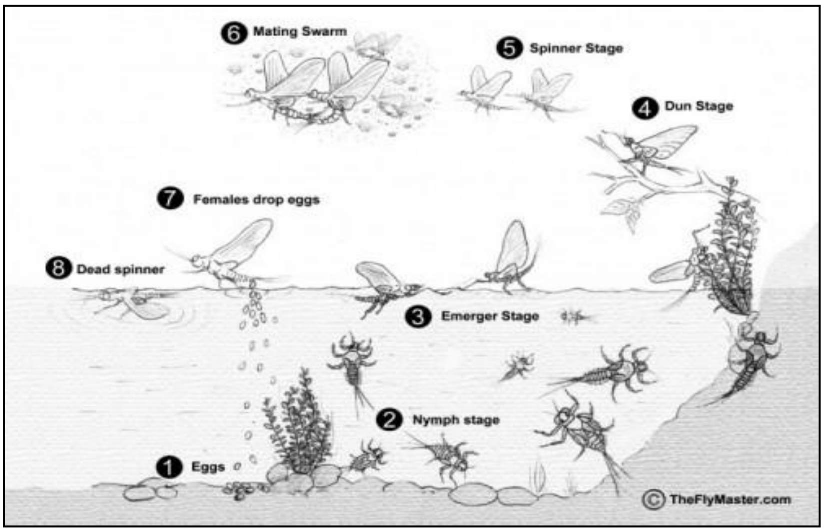


Figure 2.7- Lifecycle of Mayfly (McKelvie 2014)

<u>Mayfly habitat and flow requirements</u>

Note- a full literature search and synthesis of finding on flow requirements of Mayfly is found in chapter 4- Model build).

As the majority of their life is spent in the aquatic environment, pristine habitat requirements are of the utmost importance during this key stage. Studies have shown that BMI differ in their environmental tolerances and loss of habitat area or alteration of food sources from a decreased flow can influence behaviour and biotic interactions (Dewson et al., 2007).

At a larger catchment scale, BMI communities are influenced by a variety of factors such as land use, discharge, alkalinity, vegetation and altitude, whereas on a smaller spatial scale benthos respond to differences in habitat heterogeneity such as substrate composition, differences in water chemistry reflecting underlying geology and changes in hydrological regimes such as magnitude, and duration and timing of flows (Milner et al., 2015). Therefore the most ideal habitat required for benthos, including Mayfly, would be heterogeneous on a smaller scale.

The effects of low flows on BMI have been researched and reported in literature. BMI abundances can both increases or decrease in response to a decreased flow. When there is a reduction in flow, BMI abundances can decrease due to changes in competition and predation due to habitat area decreasing and food quantity and quality being altered. On the other hand however abundances have been known to increase during decreased flows; this can be related to a reduction in wetted area causing species to concentrate into a smaller area. Responses to food resources such as algae and organic matter can also strongly influence invertebrate density (Dewson et al., 2007).

Drought has a significant impact on BMI, particularly in chalk streams which are sensitive to extended periods of drought and abstraction. As the width of the stream reduces during drought, sediment is deposited at the stream margins, on the bed and within interstitial spaces of the substrate, this can lead to considerable changes in the BMI community. Studies have shown however that BMI recover rapidly after the return of normal flow (Wood and Petts 1999). Thus BMI can recover from periods of drought but any extended period of drought, exacerbated by abstraction, could negatively impact the BMI communities within a stream.

Effect of antecedent conditions on Mayfly

With the development of the Lotic Invertebrate Index for Flow Evaluation (LIFE) scoring system for BMI, many statistical attempts have been made to link antecedent flow conditions to LIFE scores. Section 2.5.3 describes the LIFE scoring system followed by details on the studies completed on linking this score to the antecedent flow.

2.5 Benthic macro-invertebrate (BMI) scoring indices

In the UK scoring systems for BMI have been developed to facilitate the interpretation of large quantities of data resulting from the biological monitoring of water quality (Armitage et al., 1983). Here, the three most commonly used scoring systems, which will be used in this study, are reviewed.

2.5.1 Biological Monitoring Working Party

The Biological Monitoring Working Party (BMWP) is the first well known scoring indices to assess biological quality in freshwater bodies. It is a scoring system designed primarily to summarise the effects of organic pollution on BMI communities from a simple numerical index (SNIFFER 2011). The BMWP was set up in March 1976 by the

Department of the Environment Standing Technical Advisory Committee on Water Quality (STACQA) to recommend a classification system for river pollution surveys. Stemming from the development of the Water Act of 1963, a series of attempts were made previous to this to monitor pollution in rivers. Previous attempts had failed to account for different river types and furthermore scepticism surrounded expressing biological communities in numerical form. The BMWP devised a method based on a scoring system of BMI which is unique as it allowed biologists from the Water Authorities from across England and Wales to input into the scoring system through questionnaires and consultations with experts (Armitage et al., 1983; Hawkes 1997).

The BMWP scoring system is used in England and Wales and uses 82 scoring taxa. Each family is given a score between 1 and 10 dependent on its score given by the BMWP of perceived susceptibility to pollution. High scores indicate the taxa are intolerant to pollution and vice-versa. A final BMWP score in excess of 100 provides an indicator of good biological water quality (see table 2.5) (SNIFFER 2011; Rylands 2012). Armitage et al., (1983) noted how there is seasonal variability within BMWP scores however there is more variation between sites than within sites at different seasons.

The BMWP is calculated by summing up the total of the scores (Appendix A) of each invertebrate found. The BMWP scores were however not analysed as part of this thesis due to the ASPT scores being derived from the BMWP and therefore the same trends occurred. Furthermore the BMWP is no longer used by the EA for this reason.

Table 2.5- BMWP scoring interpretations (Smithers 2009)

BMWP score	Category	Interpretation
0-10	Very poor	Heavily polluted
11-40	Poor	Polluted or impacted
41-70	Moderate	Moderately impacted
71-100	Good	Clean but slightly impacted
›100	Very good	Unpolluted/un-impacted

2.5.2 Average Score Per Taxon

Currently UK agencies do not use the BMWP directly and instead use the Average Score Per Taxon (ASPT) which consists of the BMWP divided by the number of sampling taxa in the BMWP (SNIFFER 2011). Using the ASPT accounts for potential variation in the sampling effort, for example, a prolonged sampling effort could produce a higher score than a sample taken quickly (NRT, 2012). Score systems are largely influenced by the number of taxa in the sample, using the ASPT takes this influence into account and the method is preferred by many biologists. Use of the ASPT also takes into account seasonal variations (Hawkes 1998). Armitage et al.,(1983) discovered that the proportion of variation within sites in BMWP was approximately twice that of ASPT, thereby indicating that ASPT is more accurate and consistent on a seasonal basis than BMWP is. ASPT is used as a representation of organic pollution within freshwater bodies and is widely used internationally. Furthermore ASPT is now the most predominant (most highly weighted) biotic index in Europe (SNIFFER 2011). The interpretation of ASPT scores are shown in Table 2.6.

Table 2.6-ASPT scoring interpretations (Wenn 2008)

ASPT score	Category	Interpretation
3-3.6	Poor	Heavily polluted
3.6-4.3	Moderate	Polluted or impacted
4.3-4.8	Good	Moderately impacted
4.8-5.4	Very good	Clean but slightly impacted
›5.4	Excellent	Unpolluted/unimpacted

2.5.3 Lotic Invertebrate Index for Flow Evaluation

The Lotic- Invertebrate Index for Flow Evaluation (LIFE) index is a method linking qualitative and semi-quantitative change in BMI communities to prevailing flow regimes and is said to be the most useful index for assessing the effects of drought and abstraction (Rylands 2012). Developed by scientists in the EA, LIFE was designed to assess the effects of stresses on flow for example from over- abstraction and flow augmentation, and also to set benchmarks for flows suitable for protecting and maintaining ecological integrity, i.e. for setting environmental flows (SNIFFER 2011).

LIFE scores are developed by assigning taxa into one of six flow groups to represent their main ecological affiliation with respect to flow, (Appendix A). Groups I to V represent taxa preferring rapid flow to standing waters, group VI represents taxa tolerant to drought and low flow impacted sites. Then according to the abundance of taxa found, categories are given a final LIFE score (Table 2.7) (SNIFFER 2011). Higher flows should result in higher LIFE scores i.e. high abundances of group I BMI indicate the taxa are associated with rapid flows, however high abundances of group V BMI indicate taxa associated with standing water. Thus a healthy flowing river should aim for mid ranging scores.

Table 2.7- LIFE abundance scores (Rylands 2012)

		Abundance categories			
Group	Flow groups	A	B	C	D
I	Rapid	9	10	11	12
II	Moderate/fast	8	9	10	11
III	Slow/sluggish	7	7	7	7
IV	Flowing/standing	6	5	4	3
V	Standing	5	4	3	2
VI	Drought resistant	4	3	2	1

The final life scores are determined using Equation 2.1:

$$LIFE = \sum fs/n \qquad \text{Eq2.1}$$

Where $\sum$ *f/s*= sum of individual taxa flow scores

n = number of taxa used to calculate $\sum$ *f/s*

Limitations with this approach surround taxa which are found colonising a range of habitats and flows. However the method aims to find the primary ecological affiliation flow group and furthermore if species data is unavailable, it is possible to work from family level rather than species (Extence et al., 1999).

In a key paper by Extence et al., (1999), hydroecological links were investigated through correlating LIFE scores with flow variables. It was determined that summer flow variables are most influential in predicting BMI community structure in chalk and limestone streams, therefore showing how antecedent conditions have a huge impact on BMI communities (see section 2.4.5 for further details). Many taxa requiring fast velocities have a narrow niche of requirements and therefore habitat heterogeneity is essential for taxa requiring high velocities. On the other hand, taxa requiring low flow velocities are less selective in preferences, for example leeches which are usually associated with slow flowing water can still tolerate faster flowing water. Therefore taxa in lower LIFE flow groups are more abundant in more natural areas and taxa in higher LIFE flow groups are more abundant in modified areas (Dunbar et al., 2006).

Using the LIFE methodology to assess how current and antecedent flow conditions affect BMI communities is also used as a way of setting hydroecological objectives or environmental flows. One of the main ways to assess environmental flows is through models such as PHABSIM (see section 2.8.1 for further details) to show the impact of changing flow regimes on instream habitats. The main advantage of using the LIFE method over PHABSIM is that PHABSIM does not take into account the nature of a sites flow history and the impact of this variation on the structure of the invertebrate community, i.e. the antecedent conditions. This said however the LIFE method can be used to within or alongside the modelling methods (Extence et al., 1999). Wilby (2010) used the LIFE method to determine minimum flows below which BMI communities would suffer and therefore prove that the antecedent conditions are of the up most importance for BMI.

Effect of antecedent conditions on Mayfly/ BMI

Wilby (2010) attempted to determine environmental flows for the River Itchen in Hampshire (UK), this was the first known study to incorporate antecedent flow conditions with BMI abundances. Highly significant positive relationships (p-value= <0.0001) were found between antecedent summer Q_{95} flow conditions and BMI LIFE scores. When taking into account more preceding yearly conditions, highly significant relationships of p-value= <0.0001 and <0.0005 were found between the winter Q_{95} and summer Q_{95} respectively. Thus, according to this study, the antecedent flow conditions have a significant impact on BMI abundances (Wilby et al., 2010).

Demonstrating a generic response of LIFE to antecedent conditions, Dunbar et al., (2010) established how the LIFE index responded to both antecedent conditions and habitat modification. Antecedent low flow (summer Q_{95}), antecedent high flow (summer Q_{10}) and HMS (Habitat Modification Score from the River Habitat Survey (Section 3.2.1)) were significant predictors of LIFE score. The combined nature of these two factors was found to be of key importance. LIFE responded negatively to features associated with channel modification and positively to high and low flows in the preceding 4 and 6 months at two sites respectively. In the study it was found that more modified channels had lower LIFE scores. Modified channels generally have higher velocities for more time than natural channels and as flow groups are based on velocity preferences, the modification of the channel cannot be the mechanism determining the LIFE score. Thus it is more likely that less modified channels maintain greater substrate stability and more refugia for BMI during extreme high and low flow events. This shows the importance of the antecedent flow conditions and the site conditions but not necessarily the HMS score.

2.6 Hydraulic modelling

The habitat models used in this study required water levels (mAOD) according to different flows, for this a hydraulic model was required. Here a brief background and history of hydraulic modelling is presented with a review of software and a justification of the chosen software.

Flood inundation models are a useful tool for predicting and mitigating the effects of flooding by providing flood extent and depth measurements (Mason et al., 2011). To model the flow in rivers mathematically, the set up and solution of a series of mathematical relationships to convey the movement of water is required (Beevers 2003). Details of these underlying mathematical equations are given in the following 2 sections.

2.6.1 1D hydraulic modelling

1D hydraulic modelling first occurred in the 1950's in the USA. From 1960 onwards 1D modelling has had a wide commercial application and due to its simple and non-intensive computational nature, remains widely used in engineering in the current day (Beevers 2003).

For 1D hydraulic modelling two non-linear equations are used to describe the transition of a flood wave along a river channel, these are the St Venant equations (derived in 1871) which consist of the continuity (equation 2.2) and momentum (equation 2.3) equations:

Continuity equation:

$$\frac{\partial Q}{\partial x} + \frac{\partial A}{\partial t} = 0 \qquad \text{Eq2.2}$$

Momentum equation:

$$\frac{1}{A}\frac{\partial Q}{\partial x} + \frac{1}{A}\frac{\partial}{\partial x}\left(\frac{Q^2}{A}\right) + g\frac{\partial h}{\partial x} - g\left(S_0 - S_f\right) = 0 \qquad \text{Eq2.3}$$

Where Q is the flow discharge, A is the cross sectional surface area, g is the acceleration due to gravity, h is the cross sectional averaged water depth, S_0 is the bed slope in the longitudinal direction and S_f is the frictional slope (Pender and Neelz 2011).

A main strength of 1D hydraulic models is their capacity to simulate flows over hydraulic structures such as sluices and weirs. Further advances include enhanced conveyance techniques and afflux estimation techniques which are implemented in many commercial packages such as ISIS and InfoWorks-RS (Pender and Neelz 2011). A main disadvantage of 1D models however is that flow predictions (i.e. velocity and water level) are averaged across a cross section, therefore leading to only one value for velocity and the assumption that transverse and vertical velocities are negligible (Beevers 2003). In reaches with small widths, this is not such a problem as the velocity would be unlikely to change by any significant level across the channel. It is for this reason why sites with larger widths are generally modelled in 2D.

2.6.2 Justification of 1D software

For this study two software packages were considered, Flood Modeller and HEC-RAS. The main reasons for this were to do with the requirements of the habitat models. The habitat models required information on water levels according to different flows. The need for complex hydraulic parameters was therefore not required.

HEC- RAS: HEC-RAS was developed by the US Army Corps of Engineers at the Hydrologic engineering centre. Four analysis tools are available in the package:
1) steady flow water surface profile computations, 2) unsteady flow simulations, 3) sediment transport simulations and 4) water quality analysis. HEC-RAS uses a graphical user interface (gui) to enable user friendly access. The steady slow computations calculate energy loss through mannings n and contraction/expansion, the momentum equation is used when the water surface profile varies rapidly (US Army Corps of Engineers 2013).

***Flood Modeller*:** (previously ISIS) has been developed over the past 40 years. The package includes full modelling of open channels, floodplains, embankments and structures. Steady, unsteady, supercritical, subcritical and transitional flows can be modelled with links to Flood Modeller 2D, Flood Modeller Mapper and third party software such as TUFLOW. Flood Modeller 1D has become one of the most widely used hydraulic modelling software packages globally (CH2M Hill 2015).

Both Flood Modeller and HEC-RAS solve the 1D St Venant equations and require cross sectional data input and therefore were both suitable for this study. Flood Modeller was chosen for this study as Heriot-Watt University has a long history of working with the software and as such there is high confidence in the modelling practise and procedures employed in this work.

2.6.3 2D hydraulic models

1D modelling techniques are appropriate for simulations where defined 1D flow pathways exist such as rivers and pipelines. 2D methods are used in situations where no clear path is defined (Pender and Neelz 2011).
2D hydraulic modelling uses the shallow water equations (equation 2.4 and 2.5):

$$\frac{\partial U}{\partial t}+\frac{\partial F}{\partial x}+\frac{\partial G}{\partial y}=H \qquad \text{Eq2.4}$$

Where x and y are the two spatial dimensions and the vectors U, F, G, H are defined as:

$$U=\begin{pmatrix} h \\ hu \\ hv \end{pmatrix}; F=\begin{pmatrix} hu \\ g\frac{h^2}{2}+hu^2 \\ huv \end{pmatrix}; G=\begin{pmatrix} hv \\ huv \\ g\frac{h^2}{2}+hu^2 \end{pmatrix}; H=\begin{pmatrix} 0 \\ gh(S_{0x}-S_{fx}) \\ gh(S_{0xy}-S_{fy}) \end{pmatrix} \qquad \text{Eq2.5}$$

u and v are the depth averaged velocities in the x and y direction, S_{ox} and S_{oy} are the bed slopes in the x and y directions.

A number of terms can be added to this equation to incorporate more physical processes such as viscosity, the Coriolis effect, inflow volume and momentum, wall friction stress and shear stress, these contribute to floodplain flow.

2.6.4 Justification of 2D chosen software

Due to University licensing constraints, only one 2D hydraulic modelling package was available for this project; TUFLOW. TUFLOW was developed in 1990 from a research project between WBM Pty Ltd and The University of Queensland. Until 1997 it was used only for estuarine and coastal projects, only occasionally being used for flood studies. Since 1997, improvements in flood modelling capabilities and GIS linkages have been developed, resulting in extensive and wide-ranging application to flood investigations worldwide. In 2001 TUFLOW was made commercially available. TUFLOW's 2D solution is based on the Stelling finite difference, alternating direction implicit (ADI) scheme that solves the full 2D free surface shallow water flow equations (BMT group Ltd 2014).

TUFLOW therefore provided suitable software for carrying out the 2D modelling aspect of this study. The output from the model would provide water levels (mAOD) both along the section and across the cross section. This would give detailed habitat availability data.

2.7 Habitat modelling

2.7.1 Introduction and background

Rivers and streams are complex systems providing a wide variety of biotic and abiotic components. Allowing for qualitative assessment for habitat conditions, habitat models are appropriate tools to investigate intertwining ecological functions of these systems. Such models serve three main purposes; to predict species occurrences based on abiotic and biotic variables, to improve understanding of species-habitat relationships and finally to quantify habitat requirements of aquatic and riparian species (Ahmadi-Nedushan et al., 2006). Ultimately habitat models provide a link between hydrological scenarios, hydraulic modelling and habitat assessments to assess what happens to the variable of interest (i.e. the aquatic species) when there is a change in flow (Stalnaker et al., 1995; Booker et al., 2004).

Two primary forms of input are used in the traditional and most used forms of habitat modelling: Habitat Suitability Curves (HSC) and fuzzy based logic. Two primary forms of output are also used: the Weighted Usable Area (WUA) and the Hydraulic Habitat Suitability (HHS). Each of these are discussed in the following sections. Ultimately the results from the HSC and fuzzy rules can yield different outcomes, and their interpretation in terms of WUA or HHS can vary. Therefore consideration of which to use in habitat management is an important decision as misleading results can be concluded (Boavida et al., 2014).

2.7.2 History and application

During the mid- twentieth century in North America there was a surge of large reservoir developments to boost the economy and water resources, consequently this created a loss in riverine habitats, notably fish habitat. In response to this habitat degradation, assessment methods were put in place to determine minimum flow standards for fish. These methods consisted of hydraulic analysis of water supply along with habitat and ecological assessments. From this a threshold was set below which water could not be abstracted in order to protect fish habitat (Stalnaker et al., 1995).

In the 1970's, the concept of setting 'minimum flows' to protect aquatic habitats changed towards using the concept of setting 'in stream flows'. This change acknowledged that simply allocating part of a water supply to habitats was not sufficient to resolve conflicts between human and environmental needs (Stalnaker et al., 1995). It was recognised that fish, along with other habitats, require a variety of flows rather than solely a minimum accepted flow. This is when in the late 1970's the US Fish and Wildlife Service (FWS) received funding from the Environmental Protection Agency (EPA) to establish the Cooperative Instream Flow Service Group who aimed to produce methods of assessing how changing flow regimes affected aquatic habitats (Milhouse and Waddle 2012). The main results of this group's method were the creation of the Instream Flow Incremental Methodology (IFIM). The IFIM was created over a period of 15 years into network analysis that incorporated fish habitat, recreational opportunity and woody vegetation response to water management schemes (Milhouse and Waddle 2012). This was the early beginnings of eco-hydrology and is what most habitat modelling theory is based on.

Then, in the 1980's, habitat models became an important tool for river management and are now the focus of ongoing research. Since the development of IFIM, habitat models have been used extensively around the world, particularly for fish habitat assessments (Dunbar et al., 1996 cited in, Spence and Hickley 2000). Today a wide variety of habitat models exist, encompassing nearly all types of aquatic organisms

(Noack 2013). Despite this however habitat models have been most successful with fish, particularly salmonoids, due to data availability and ease of collecting data. Application for invertebrates in habitat modelling has been limited due to difficulties associated with the collection methods and large numbers of samples required, taxa identification and the application of habitat suitability curves also provide limits to invertebrate assessment (Dewson et al., 2007). Very few habitat modelling studies have been carried out on macrophytes, this is likely related to the natural growth and die back being an important factor in macrophyte abundance, therefore the flow of the river cannot always be relied upon as an indicator of abundance (Dawson 2002). This has been taken into account in this study (see Section 4.12).

2.7.3 Input: Habitat Suitability Curves/Index (HSC)

The most common index to describe the response to abiotic scenarios is the use of univariate curves, more commonly known as Habitat Suitability Curves (HSC). HSC are a fundamental component of habitat models representing abiotic habitat variables of: depth, velocity, substrate and cover.

The original development of HSC were created as part of the IFIM (Milhouse and Waddle 2012). The Instream flow group developed the Washington method and the univariate curve concept. The Washington method involved applying binary suitability functions, usually by visual observations, for salmon spawning in streams dependant on depth and velocity conditions. The area suitable for spawning was evaluated at various measured discharges; suitable spawning areas at unmeasured discharges were interpolated. The use of the univariate curve created a function covering the entire range of depth and velocity from a value of 0 to 1 (0 being unsuitable and 1 being suitable) (Milhouse and Waddle 2012).

Univariate suitability curves (HSC) became a popular technique for habitat predictions and are still used in habitat modelling. Developments of HSC nowadays use a much more sophisticated technique. The following example describes the process of developing HSC for fish:

- Electro fishing is used to measure; the number of fish, the area available to the fish, and the depth, velocity and substrate at the point of catch.
- This information is then transferred to a graph (figure 2.8- green columns). And a line of best fit is drawn through it (red line).

- The preference curve (red line) is then 'smoothed' to finalise the preference. There is no clear method to carry this out, an experts opinion or some literary evidence to show for example whether fish prefer medium or low depths is generally required.
- The depth or velocity preference is then determined using equation 2.6.

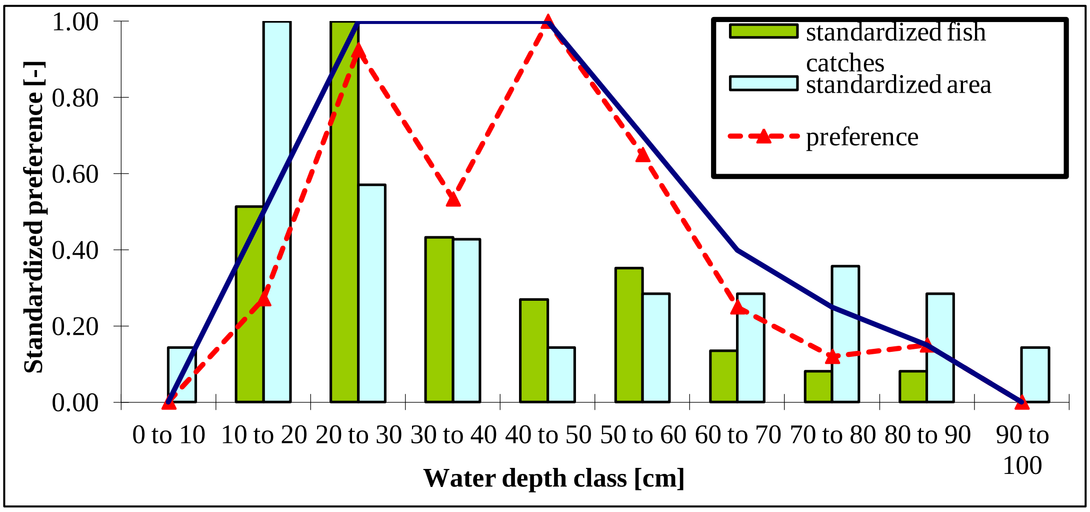


Figure 2.8- HSC development for water depth (Giesecke 2010)

$$Depth\ preference = \frac{\left(\frac{Standardised\ fish\ number}{Standardised\ area}\right)}{Max\left(\frac{Standardised\ fish\ no}{Standardised\ area}\right)} \qquad \text{Eq2.6}$$

There are areas of limitation to this method, mainly involving methods of electro-fishing:

- During electro-fishing, fish can detect movement in the water and therefore move away.
- Once stunned, high velocities can move the fish from the area they were originally.
- Fish species and ages require an experts judgment.
- Electro-fishing data captures only a 'snap-shot' of fish habitat, a higher number of studies would have to be carried out seasonally to get an accurate representation.
- The smoothing technique needs expert opinion, which can be highly subjective. (Giesecke 2010).

HSC are the source of much criticism surrounding habitat models. Firstly, as previously discussed, collection methods of this data are highly subjective. Secondly, HSC only take into account depth, velocity and substrate preferences, in reality aquatic organisms depend on other factors such as cover, food availability and habitat selection based on interspecies competition (see Figure 2.4) (Milhouse and Waddle 2012). It is for this reason why caution should be applied when determining HSC for any species, as there is more than one factor affecting the species at any one time (Armstrong et al., 2003). The results from using HSC as input are also of concern; see section 2.7.5 on Weighted

Usable Area for further information. Finally the habitat suitability indices do not account for the interrelation between habitat variables, for example, velocity can greatly influence the size and composition of particles in the substrate, likewise depth and velocity are highly dependent (Ahmadi-Nedushan et al., 2006).

Transferability of HSC

The transferability of HSC is a subject which has been greatly studied and tested, often with contradicting findings. The initial intention of creating HSC was to develop universal curves which could be transferred to diverse streams, however authors argue that derivation of suitability indices is best for individual study reaches due to all factors affecting a specific organism being present for that specific species and site (Maki-Petays et al., 1997; Slavik 1998). This leads researchers to choose between using HSC derived for a particular site, which is time and resource extensive (Boavida et al., 2014), or whether to use well established, generalised curves derived from literature.

Maki-Petays et al.,(2002) tested generalised HSC based on four rivers against a river specific HSC to assess how transferable HSC were across rivers. The river specific curves transferred well in 91% of the cases, whereas the generalised HSC transferred well in 82% of the cases. This study proved that whilst it is better to use site specific HSC, where resources are not available, generalised curves based on other studies can be used and will provide appropriate results. Furthermore, site specific HSC are based only a ‘snapshot’ of time, i.e. based on electro-fishing at one specific time, this could be open to bias, using generalisation curves eliminates this limitation (Maki-Petays et al., 1997; Maki-Petays et al., 2002).

In a study by Thomas and Bovee, (1993), microhabitat predicted by HSC developed for a the South Platte river, Colorado were compared with observed habitat use by adult and juvenile rainbow trout (*Oncorhynchus mykiss*) in the Cache la Poudre river, Colorado. It was determined that HSC could be transferable between two rivers for adult rainbow trout (*Oncorhynchus mykiss*) but not for juvenile rainbow trout (*Oncorhynchus mykiss*). Likewise Moir et al., (2005), discovered that HSC developed for the river Dee, Scotland catchment predicted spawning sites of Atlantic salmon (*Salmo Salar*) well in the river. However HSC based on streams in southern England did not correspond with patterns of spawning for the river Dee in Scotland. This finding indicated that HSC do not transfer well between rivers.

On the other hand, Louhi (2008) created generalised HSC for Atlantic salmon (*Salmo salar*) and brown trout (*Salmo trutta*) for rivers in different size classes. This provided key information for decision makers and because they are based on many different studies, they cover a large range of habitats used by the fish. Whilst determining HSC for individual study sites is preferable, decision makers often have to make quick decisions where it is not logically or economically feasible to develop HSC for all environmental conditions and for all species on that specific river. Therefore it is common practice to use HSC for streams which they were not originally developed for (Maki-Petays et al., 1997). Louhi (2008) achieved a transferable HSC for the species, but as the HSC were for rivers of specific sizes, the HSC were generally site specific.

To summarise, studies have shown both that HSC transfer well and equally that they do not transfer well between different rivers. Ideally site specific HSC would be used in order to encompass all abiotic and biotic variables influencing a species in one river. However due to time and resource constraints, HSC can be used based on literature and previous studies, with validation to specific river conditions.

2.7.4 Input: Fuzzy based logic

Recently the use of fuzzy based rules has become more favoured means of input to habitat modelling and are considered a more suitable approach than HSC (Mouton et al., 2011). Fuzzy based rules are based on linguistic variables of ‘high’, ‘medium’ or ‘low’ to describe the physical properties such as water depth and velocity. An example of this linguistic variable is given below:

> ‘IF the water depth is “high”, AND flow velocity is “medium”, AND substrate is “high”, THEN the suitability is “high” (Schneider et al., 2010).

The ‘if’ part of the rule (the antecedent) describes a situation which applies, while the ‘then’ part (the consequent) indicates whether the habitat in this situation is suitable or not (Mouton 2007).

The use of expert knowledge from biologists and ecologists is required for fuzzy rules which are translated into linguistic variables rather than definite numbers, as in HSC. Therefore fuzzy logic is able to overcome the issue of HSC not capturing the complexity of natural systems by representing the gradual transitions between predefined classes (Millidine et al., 2012; Noack 2013; Boavida et al., 2014). Furthermore, as the boundaries between the predefined classed are overlapping an element can partially belong to a fuzzy set and therefore provides transparent values (Mouton 2008).

Using expert knowledge in fuzzy rule creation has benefits and limitations. Due to the subjective nature of expert knowledge, inconsistencies can occur mainly due to different backgrounds, i.e. different research interests and geographical locations and different experiences in the field, (Boavida et al., 2014). Furthermore expert options are usually related to snapshot experiences. It is for this reason why fuzzy rules are more commonly being referred to as either 'expert knowledge fuzzy rules' or 'data driven fuzzy rules'. Mouton et al.,(2009) discovered that data driven fuzzy rules derived from a nearest ascent hill-climbing algorithm, outperformed expert knowledge based fuzzy rules based on literature. It was discovered complimenting expert based fuzzy rules with data driven techniques can improve model validation.

The output from fuzzy rules involves defuzzification where the final fuzzy rule is weighted with a degree of fulfilment and is transformed back to a crisp number between 0 (most unsuitable) and 1 (most suitable). These values are known as HSI (Habitat suitability index), referred to as SI (suitability index) from herein.

2.7.5 Output: Weighted Usable Area (WUA)

The Weighted Usable Area (WUA) is the most common output of habitat models. Expressed in units of microhabitat area per a distance along the stream, the WUA is calculated using the formula in Equation 2.7:

$$WUA = \sum_{i=1}^{n} Ai \times Ci \ \ (Units^2) \qquad \text{Eq2.7}$$

Where:

Ai= Surface area of cell i

C_i= Combined suitability of cell $_i$ (i.e. composite of depth, velocity and channel index individual suitability's

Figure 2.9 shows a typical WUA output, this example shows how available habitat area for each target species and life stage changes with an increasing discharge. This particular example shows that for adult brown trout a discharge of 1m^3/s is the optimum flow and for adult spawning salmon a discharge of 2m^3/s is the most preferred.

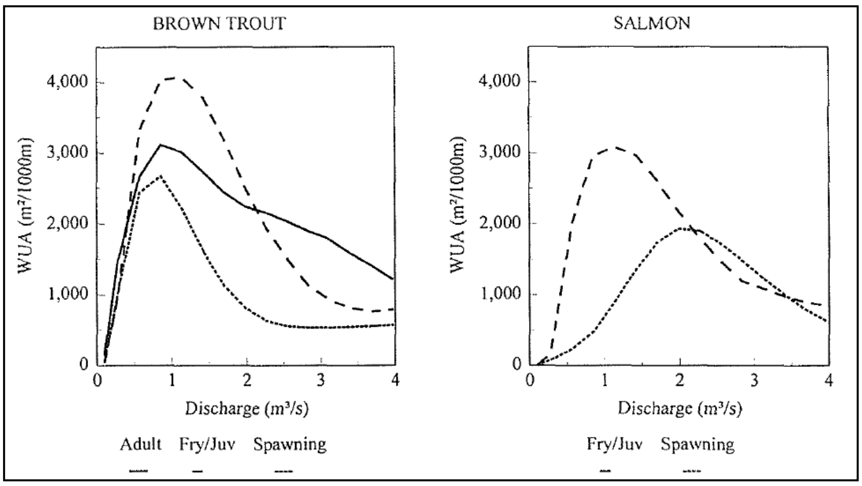


Figure 2.9- Example of WUA curves for brown trout and Salmon (Milhouse and Waddle 2012)

WUA criticism

Despite the WUA being the most commonly used output there are many criticisms of using this method. The WUA is determined by combining the results from HSC for depth, velocity, substrate and cover. The area of concern occurs when combining these results. There are three main approaches used to combine the results:

1) The most common approach is a multiplicative aggregation (or **product method**). This method is based on the assumption that species select variables independently of others. A negative of this approach is that a zero suitability is calculated for any unsuitable habitat variable (Noack 2013) (Equation 2.8).

$$C_i = V_i * D_i * S_i \qquad \text{Eq2.8}$$

2) A further approach is the **Geometric mean** which allows for a compensation effect between the component suitability values, however like the product method also leads to a zero suitability for any zero-valued HSI value (Noack 2013). (Equation 2.9).

$$C_i = \sqrt[3]{V_i} * D_i * S_i \qquad \text{Eq2.9}$$

3) The final approach is based on the **lowest HSI**. It is assumed the most limiting physical factor determines the upper limit of habitat suitability and high HSI cannot compensate for low HSI (Boavida 2012). (Equation 2.10).

$$C_i = Min(V_i, D_i, S_i) \qquad \text{Eq2.10}$$

Where:

$C_i = Composite\ suitability\ of\ cell\ I$

$V_i = suitability\ associated\ with\ velocity\ in\ cell\ I$

$D_i = suitability\ associated\ with\ depth\ in\ cell\ I$

$S_i = suitability\ associated\ with\ channel\ index\ in\ cell\ I$

From this, the following issues arise:

- The different approaches can yield different results therefore providing uncertainty to the output (Boavida et al., 2014).
- Further criticism surrounding the WUA is related to the shapes of WUA curves being highly uncertain. Some authors argue that estimates of WUA should be reported with standard errors or confidence intervals to make decision makers aware of the uncertainty (Ayllon et al., 2011).
- WUA curves are good at showing how suitable habitats are at different discharges, however do not account for how much area of the river is good or bad habitat (Schneider 2014). This can be solved by directly using SI values according to area

of percentage, this has been achieved in CASiMiR, see section 2.8.2 for further detail.

2.7.6 Output: Hydraulic habitat suitability (HHS)

The Hydraulic habitat index (HHS) is the second main output from habitat models; this is specifically used in CASiMiR (see section 2.8.2) and is determined by dividing WUA by the wetted area using equation 2.11.

$$HHS = \frac{1}{A_{ges}} \sum_{i=1}^{n} A_i \times SI_i = f(Q) \qquad (-) \qquad \text{Eq2.11}$$

Where:

$SI_i = habitat\ suitability\ index\ value\ for\ the\ i_{th}\ cell$

$A_i = area\ of\ the\ i_{th}\ cell$

The output from HHS is an index ranging from 0 to 1, which eliminates the influence of the wetted area to provide model comparisons between study-sites. Using WUA as output is dependent on the wetted area of the channel. Taking two channels for example:

- Channel 1 with a wetted area of $100m^2$ could have a WUA of $60m^2$.
- Channel 2 with a wetted area of $50m^2$ could have a WUA of $40m^2$.
- The results indicate channel 1 has a much higher WUA than channel 2.
- However in reality channel 2 has a much higher WUA in relation to the wetted area.

Use of HHS eliminates this limitation when comparing between study sites.

2.7.7 HSC versus fuzzy logic

To date, only a few studies have been completed on directly comparing HSC and fuzzy logic based models. Giesecke et al., (2012) reports on the main advantages of the fuzzy logic system over the traditional HSC.

- Expert knowledge can be numerically processed through fuzzy based rules.
- Fuzzy logic takes into account the interaction of parameters but do not require explicit assumptions regarding parameter independence.
- User-specified requirements can be easily included using this approach and finally the calculation approach is relatively straightforward and easy to understand.

Jorde et al., (2001, cited in Ahmadi-Nedushan et al., 2006) used fuzzy logic for fish evaluation in Switzerland, it was concluded that observed fish densities showed a higher correlation with fuzzy simulations than those based on traditional preference functions. Mouton (2008) also determined that habitat preferences derived from field observations were very similar to those predicted by the fuzzy based model. This therefore shows that fuzzy based rules provide a more realistic representation of habitat availability.

More recently Boavida et al., (2014) conducted a direct comparison between HSC and fuzzy rules. The outcomes of each were found to be very different. Figure 2.10 demonstrates the different outcomes of Fuzzy rules against HSC (product, arithmetic mean and geometric mean) for both WUA (black lines) and HHS (gray lines). The product combination revealed the lowest habitat values, whereas the arithmetic mean was closer to the fuzzy logic results. The flow thresholds which could be determined from this were very different between HSC and fuzzy rules. Clearly the results indicate the sensitivity of input and the misleading conclusions which can be adhered to.

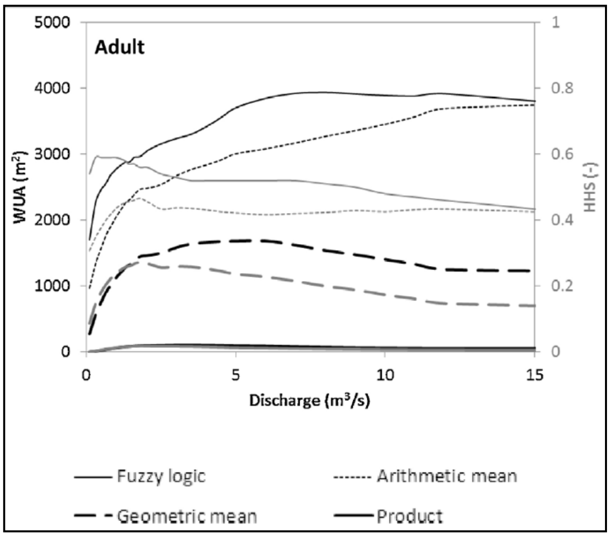


Figure 2.10- WUA (black lines) and HHS (gray lines) for adult Nase using fuzzy logic and arithmetic mean, geometric mean and product method based on the HSC (Westwood et al., 2006; Boavida et al., 2014)

2.7.8 Criticism of habitat modelling

The first and foremost criticism is that habitat models do not take into account any factors other than the hydraulic components (depth, velocity, substrate and cover). Habitat models predict changes in habitat resulting from changes in flow but ignores the dynamics of habitat through time i.e. focusing only on few variables affecting localised fish behaviour (Stalnaker et al., 1995). Figure 2.4 depicts some of the vast amount of variables impacting upon brown trout habitat availability.

Further criticism of habitat modelling is that there is little relationship between the output WUA and actual fish abundances, and the relationships between abundances and more than one HSC is rarely tested (Jowett 1992). Researchers have shown correlations between WUA and fish numbers particularly when the effects of flow are considered. Gallagher and Gard (1999) compared results from PHABSIM (Section 2.8.1) showing available habitat for spawning salmon with actual redd locations. WUA was significantly correlated with salmon spawning density and locations at a vast majority of sites, furthermore sites with higher numbers of redds had higher predicted WUA. However, in a study by Mouton et al., (2008), a fuzzy based model predicted spawning grayling (*Thymallus thymallus*) to be present for several instances where no spawning was observed. There are reasons why this may have occurred:

- The location of spawning might depend on other variables than only those included in the habitat suitability model, for example; competition, predation or immigration and emigration.
- The monitoring efficiency could be at fault where for example spawning grooves were present but not observed.

In a key study by Jowett (1992), abundances of adult brown trout were tested against 50 independent environmental variables. Significant positive correlations were found between trout abundance and: invertebrate biomass, temperature, cobble substrate and WUA. Invertebrate biomass was found to be the single most important factor in determining brown trout abundance. Furthermore suitable living space was deemed the second most important factor, including cover. Therefore invertebrate biomass, which cannot be taken into account by habitat model input, is actually shown to be a more important determinant for brown trout location than WUA arising from hydraulic variables; WUA is the second most important factor.

Despite these studies, the assumption that habitat use reflects habitat preference is rarely validated and there is still an unanswered question to whether fish species do actually select areas of model determine high preference. Boavida (2012) attempted to validate model results by comparing differences between fish habitat selection and simulated habitat availability. There was a large degree of discrepancy found for velocities when comparing values measured in the field with values predicted by the model. There are many reasons around modelling and measurement errors why this discrepancy could occur, such as misreading instruments, missing or wrong calibration, human errors while positioning equipment or turbulent water leading to poor measurements. However this does highlight the vast uncertainty surrounding habitat model prediction.

Finally the limitations of hydraulic modelling (section 2.6) as an input to these habitat models also needs to be taken into account i.e. it only gives a prediction of what water levels could be, factors such as climate change, river restoration and land use change could impact on the predictions.

2.8 Habitat modelling software and developments

PHABSIM was the original 1D habitat model; nowadays a suit of habitat modelling software exists for both 1D and 2D modelling. These software are reviewed below:

2.8.1 PHABSIM

The Physical Habitat Simulation System (PHABSIM) is arguably the most widespread and well known of habitat methods used worldwide to link habitat to inflow (Ayllon et al., 2011; Heggenes 2013). Use of this software has become a legal requirement for many impact studies in the USA (Acreman et al., 2008a). PHABSIM was developed subsequent to the IFIM framework (Milhouse and Waddle 2012) as an aid to in stream flow decision making:

> 'PHABSIM uses the hydraulic simulation models to predict depth and velocity at unmeasured flows... The resulting software suite multiplied surface area for a section of a stream by the univariate suitability curve values for depth, velocity and channel condition to arrive at a habitat index called Weighted Usable Area' (Milhouse and Waddle 2012)

The underpinning ecological concept of PHABSIM is that an interaction of environmental components determines the status of species populations in the stream. Therefore, observed populations and biomass of species is a function of environmental components operating on a variety of temporal and spatial scales (Milhouse and Waddle 2012). PHABSIM uses HSC for input and WUA for output.

Spence and Hickley (2000) review how PHABSIM has been used extensively in the UK for investigation into the following:

- Setting minimum flows for reservoirs to protect fish,
- Impact of surface and groundwater abstractions,
- Abstraction licensing,
- Impact of drought management
- Restoration work for habitat improvement schemes.

An example where PHABSIM has been used is on the River Worfe where investigations used PHABSIM to assess the impacts of baseflow reduction on brown trout populations. It was determined that over-abstraction led to a 20% loss of suitable habitat for spawning brown trout and a 60% loss of suitable habitat for juvenile brown trout (Spence at al 2000). Likewise results from a PHABSIM study on the river Allen in Dorset were used in negotiations between the National Rivers Authority (NRA) and the Bournemouth water company to form an action plan for the river including a proposal to reduce abstraction rates by 50% (Gustard and Elliott 1997). Studies such as this can be used to inform management decisions.

Criticisms of PHABSIM

There are criticisms of PHABSIM, mostly surrounding the use of the HSC, discussed in section 2.7.3 and 2.7.8. PHABSIM assumes the habitat available for fish is limited by the availability of physical habitat, whereas in reality this is not true. Production could be limited by anthropogenic impacts such as land use activities, or by food availability. Therefore, it is acknowledged that PHABSIM results must be viewed as an indicator of population potential in response to certain environmental changes i.e. increased or decreased flow (Milhouse and Waddle 2012). Furthermore it is recommended that PHABSIM studies should be used alongside complimentary techniques such as expert knowledge (Spence and Hickley 2000). In a study completed by Nagaya et al., (2008), PHABSIM was used to determine the effect of hydraulic structures on migratory fish in Japan. The results of the study showed that the PHABSIM results did not correspond to the field data collected and that the accuracy became lower with increased discharges, this therefore implied that PHABSIM cannot always be appropriate for use.

2.8.2 CASiMiR- fish 1D

Following on from the extensive developments of habitat models in the USA in the 1970’s (Stalnaker et al., 1995), in the early 1990’s the Institute of Hydraulic Engineering of Stuttgart University developed CASiMiR (Computer Aided Simulation System for Instream Flow Requirements), which unlike preceding models, focused on habitat conditions of benthic organisms rather than fish. Like many other habitat models though, the original incentive of CASiMiR was to study habitat issues related to hydropower and to setting minimum flow requirements. This idea was developed into a bespoke software named CASiMiR-Hydropower (Millidine et al., 2012). The first version of CASiMiR available to the public was developed in 1993 for habitat modelling of benthic macroinvertebrate based on a preference function approach using FST- hemispheres (see below). In 1996 CASiMiR-fish advanced to model fish preferences and finally in 1998, the multivariate fuzzy approach was implemented. The use of the multivariate analysis reduced criticism from previous univariate HSC as it determines responses to cumulative effect of a number of environmental factors rather than considering separate effects of individual parameters (Ahmadi-Nedushan et al., 2006). CASiMiR allows input through both fuzzy rules (Section 2.7.4) and the traditional HSC (Section 2.7.3).

Output: Velocity, HHS and SI

The CASiMiR- 1D approach is referred to more openly as the 1.5D approach as it includes a simple algorithm for calculating local flow velocities, for this reason rivers with uniform channels and no complex geometry can be reliably modelled in this version of CASiMiR. Equation 2.12 is used for this calculation.

$$V_i = f_{Aw} \times \frac{1}{\sqrt{\lambda}} \times \sqrt{g \times h_i \times I_E}\ (m/s) \qquad \text{Eq 2.12}$$

Where:

$v_i = flow\ velocity\ in\ the\ i_{th} cell\ (m/s)$

$f_{Aw} = conveyance\ factor\ (-)$

$\lambda = Darcy - Weishbach\ roughness\ coefficient\ (-)$

$g = acceleration\ due\ to\ gravity\ (m/s^2)$

$h_i = water\ depth\ in\ the\ i_{th} cell\ (m)$

$I_E = energy\ grade\ line\ slope$

There are two main outputs from CASiMiR, both of which are used for describing the relation between habitat quality and discharge for a specific river reach. Firstly the WUA (Section 2.7.5), and secondly the HHS (Section 2.7.6).

One of the disadvantages of using WUA and HHS is that they give no information about the distribution of low and high habitat for the study site. For example a 20m^2 area with a habitat suitability of 0.3 is shown to be as equally as important as the combination of two smaller areas of 5 and 15m^2 having suitability's of 0.9 and 0.1 respectively, both resulting in a WUA of 6m^2. To reduce these limitations, a further output provided by CASiMiR is the habitat suitability index (SI) separated into 10 ranges between 0 and 1. These are very important to show for example areas with SI over 0.7 determined as highly suitable habitats (Schneider et al., 2010). As both HHS and SI use ranges between 0 and 1, care should be taken not to confuse these with one another or with the HSC and fuzzy rule membership functions.

FST curves

The most recent version of CASiMiR allows for FST-hemisphere calculations. The abbreviation FST means Fließwasserstammtisch, which directly translated from German means 'regulars table of colleagues' working in Konstanz where discussions surrounding FST first took place. It was determined that hydraulic stress is the major factor determining benthic macro-invertebrate (BMI) distribution and abundance.

Therefore FST-hemispheres were developed for the assessment of the forces acting on BMI. Adaptations based on criticisms of this method were made due to firstly the placement method and the substrate upon which the hemisphere was placed (Kopecki 2008).

2.8.3 CASiMiR-GIS-Benthos

CASiMiR-GIS combines capabilities of both CASIMIR-fish and CASiMiR-benthos which integrates a calculation approach to obtain FST values with their spatial distribution. Likewise with CASiMiR-fish, this module allows for input from HSC or fuzzy based logic. The principle differences between this model and CASiMiR-fish is that FST values can be automatically calculated, allowing for multivariate capabilities and the calculation and visualisation features of GIS can be advantageous for some certain studies (Schneider and Kopecki 2011).

2.8.4 RAPHSA

Based around the concept of PHABSIM, RAPHSA (Rapid Assessment of Physical Habitat Sensitivity to Abstraction) was developed by the EA as an aid to assess the likely magnitude of change in a river ecosystem when water is abstracted. It could therefore be used to help manage water resources, to set abstraction licenses and to help meet the aims of the WFD. One of the main criticisms of previous habitat modelling software is that it is expensive and time consuming to collect data; this led to a need for more rapid physical habitat assessments which is why RAPHSA was developed. The principle output of RAPHSA is graphs showing flow against a range of physical variables:

-WUA- for target species based on depth and velocity

- WUA_d- for depth got a target species
- WUA_v- for velocity for a target species
- WW- wetted river width
- V- mean velocity
- D- mean depth

Thus the output is very similar to PHABSIM and such like software. The principle difference between RAPHSA and PHABSIM is the speed in which assessments can take place, RAPHSA uses a database of studies done using PHABSIM studies in the UK to determine the WUA versus flow curves, thereby reducing the need to undertake a full data collection and hydraulic and habitat analysis (Acreman et al., 2008a).

2.8.5 RHYHABSIM

River Hydraulic Habitat Simulation (RHYHABSIM) was developed in Denmark as a tool to model habitat responses to changing hydraulic conditions. It is a tool to assess ecosystem conditions as a way forward to the aims of the WFD. The principles of RHYHABSIM are based around the concepts of PHABSIM. The main difference between the models is that RHYHABSIM is simplified by limiting the number of input variables, thus making the model easier to use whilst still providing accurate results. A further benefit of this software is its ability to analyse data from different species and for different lifestages simultaneously (Thorn and Conallin 2006).

2.8.6 CASiMiR 2D

The principle components of CASiMiR 2D are the same as CASiMiR 1D. The only clear difference is that results from a 2D hydraulic model are used which incorporates velocity changes across the cross section. This can be beneficial to ecological modelling as niche habitats are often preferred by species which can occur at a fine spatial scale (Milner et al., 2015). Furthermore the substrate and cover values are input using a GIS shapefile rather than directly from the geometry data. Thus more detailed results are output about how habitat changes with depth and velocity across the cross section rather than only one value per cross section (Schneider et al., 2010).

2.8.7 River 2D

River2D can simulate both hydraulic and habitat 2D simulations. The software is a 2D depth- averaged finite element hydrodynamic model which has been customised for fish habitat studies. The normal modelling process would be:

- Create a topographic profile of the river from the field data
- River2D is then used to simulate water depths and velocities
- River2D is used to visualise and interpret the results using a PHABSIM type assessment adapted for a triangular irregular network (University of Alberta 2002).

More often than not however River2D is used only as the hydraulic input to CASiMiR 2D (e.g. Boavida et al., 2015).

2.8.8 Tailor made software

Many studies choose not to use readily available software for ecological modelling. The reasons behind this are mainly due to being able to incorporate many more variables into habitat suitability data. For example Mouton et al., (2009) incorporated fuzzy rules for ammonium, nitrate, phosphorus, conductivity and dissolved oxygen to habitat suitability data along with the traditional hydraulic components. Extensive coding would be necessary for these bespoke software's.

2.8.9 Comparison between 1D and 2D hydraulic- habitat modelling

Habitat modelling has primarily focused on 1D hydraulic approaches which uses the assumptions that mass and energy balance relations combined with a small number of transects can represent the conditions in the stream. Most natural streams do not adhere to these conditions and therefore 1D modelling has been criticised for use in these situations. 2D hydraulic-habitat modelling can improve studies where complex habitat conditions are encountered, i.e. 2D hydraulics can capture microhabitat features such as velocity shear zones, side channels, islands, bars and eddies. These features are important habitats for reproduction, growth and survival of many aquatic species (Waddle 1998; Gard 2009).

Very few studies have been carried out directly comparing 1D and 2D habitat models. Gard (2009) however, compared PHABSIM (1D) and River2D (2D) on the Sacramento River (California) using the same HSC for spawning Chinook salmon (*Oncorhynchus tschawytscha*). Different WUA results occurred between the two different models (Figure 2.11) which were related to the ability of the models to predict velocities and depths and also the different means of input of substrate values.

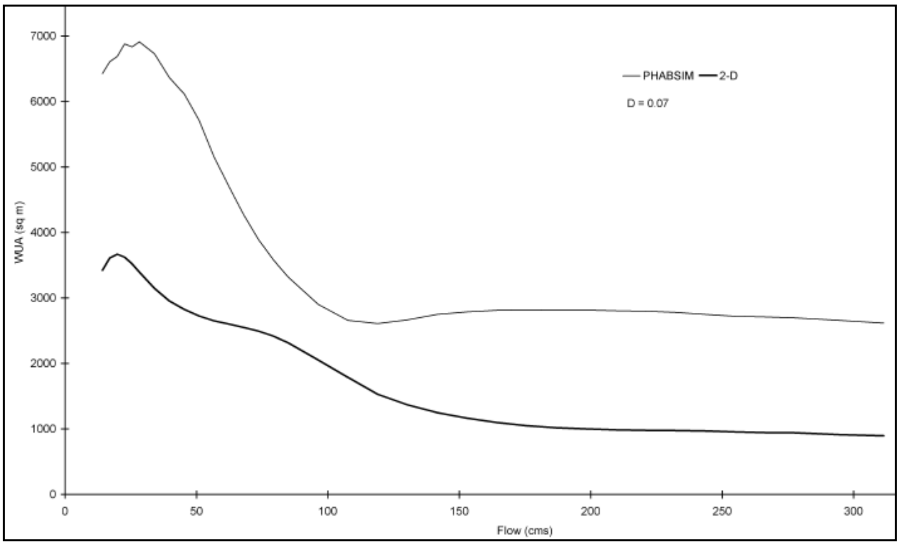


Figure 2.11- WUA curves predicted between 1D and 2D habitat model (Gard 2009)

2D habitat models tend to be more accurate than 1D models due to the smaller grids used compared to the larger grids used in 1D modelling (Gard 2009). Ultimately the main advantage of 2D habitat modelling is the ability to model complex conditions such as across channel variation in depth and velocity which 1D models cannot do. However in the study by Gard (2009) the 1D PHABSIM model accurately predicted redd locations as did the 2D model indicating that neither is better nor worse in predicting habitat availability.

2.8.10 Justification of chosen habitat model

After each of the software packages had been assessed the two clear choices were CASiMiR or PHABSIM. RAPHSA and RHYHABSIM were discounted due to simplistic methods used.

Both CASiMiR and PHABSIM were freely available however due to the criticisms of PHABSIM and the fact that CASiMiR allows for direct comparison between fuzzy logic rules and HSC, CASiMiR was the chosen software. It was also preferential to use an external hydraulic model to determine water levels to enable more detailed evaluations, CASiMiR allowed for this. Furthermore a training course was attended by the author in March 2014 to increase confidence in the modelling procedures and to confirm the methodology employed was fully sound (Appendix B). CASiMiR also had the option of use in 2D, thus the same processes and habitat suitability data could be used, only with 2D hydraulic input.

Chapter 3- Methodology

3.1 Introduction

This chapter sets out the methodology based on the three research questions. Data collection, methods and analysis are described in detail for each of the research questions. Inevitably there are overlaps between the different strands of research and analysis of each often requires results from another. These interdependencies are explained in the analysis section for each research question. The conceptual model of the research is presented in Figure 3.1. Firstly an explanation of data requirements and analysis which occurs throughout all of the research is described; how these are then used in each section is expressed in the subsequent analysis sections.

3.1.1 Available flow data

The Marham gauge (Figure 1.1) provided the only historical gauged flow data from the river. This was available in cubic meters per second (m^3/s) on a daily basis from September 1954 to July 2014. Habitat models use daily flow data which gives a habitat availability amount per daily flow; therefore the data available was suitable for this study. However the trading model output for research question 3 was in weekly flows which could not account for daily fluctuations in habitat. Therefore the weekly flows from the trading model were disaggregated into daily flow amounts (Section 3.4.1). Hourly data was not considered as it would be too fine a scale to show any significant changes in habitat availability.

3.1.2 Flow data- area weighting

Daily flow data from the river was used for the majority of analysis in this study and is therefore an important aspect of much of the subsequent analysis. The daily flows for each site used in the subsequent analysis were determined using an area weighting formula based on the Marham gauge historical recordings (Equation 3.1).

$$\frac{A_{site\ catchment}}{A_{catchment}} = \frac{Q_{Site}}{Q_{gauge}} = Area\ weighting \qquad \text{Eq.3.1}$$

Where:

A_{site}= catchment area at site, determined from Flood estimation handbook (Kjeldsen 2007)

$A_{catchment}$= whole area catchment

Q_{site}= flow to be determined at site

Q_{gauge}= flow at Marham gauge on day of flow to be determined

The areas for the nine macrophyte sites and area weighting factors used are given in Table 3.1 as an example (sites marked on Figure 3.4). This method approximates the actual flow at each site based on the known measurements at Marham. In the absence of further gauges along the river this represented the best option for flow estimations.

Table 3.1- Catchment areas at the 9 main sites with their area weighting factors, (sites downstream (1) to upstream (9))

Site number	Site name	A_ site_ catchment (km^2)	Area weighting factor
1	Highbridge	223.93	1.52
2	Marham	147.37	1.00
3	DS Nar	135.16	0.92
4	US Nar	120.79	0.82
5	West Acre	111.35	0.76
6	Castle	76.86	0.52
7	West	71.97	0.49
8	Litcham	43.09	0.29
9	Mileham	32.22	0.22

3.1.3 Flow data- antecedent conditions

In research question 1 (RQ1), antecedent flow conditions were used to investigate how the preceding flow conditions influence the indicator species. The antecedent conditions represent the conditions before the event; this can be the hour/day/month/season/year etc. before. In this case, the seasonal antecedent high, average and low flow conditions were examined to relate habitat data to.

The thresholds for 'low flow' and 'high flow' were determined as Q_{90} and Q_{10} respectively. Despite criticism surrounding using one mathematical definition of hydrological drought due to climatological factors (Lloyd-Hughes 2014), according to Lake (2011), droughts in perennial rivers are indicated when flows fall below Q_{90} or Q_{95}. Therefore, for this study, drought was defined as being the flow at or below Q_{90}.

Table 3.2- Months used for each season in subsequent analysis

	Months used in this study			**Typical hydrological months used**		
Winter	January	February	March	December	January	February
Spring	April	May	June	March	April	May
Summer	July	August	September	June	July	August
Autumn	October	November	December	September	October	November

For each measured site (i.e. kick sampling/macrophyte surveys) the Q_{90} (low flow), Q_{50} (average flow) and Q_{10} (high flow) were calculated from daily flow data for each site (details given in Section 3.1.2). These were determined on a seasonal basis for winter, spring, summer, and autumn. The months used in each season are shown in Table 3.2, these months are not the typical seasons used in hydrology. Rather they are based on the months used for the antecedent flows in the LIFE response curve project which much of the antecedent analysis is based around (EA 2005). In order to keep the data consistent, all analysis using seasonal data was based around these months.

3.1.4 Statistical calculations

Statistical analysis was conducted throughout the research (predominantly in RQ1) in order to investigate relationships between flow conditions and the ecosystem indicators. An explanation and meaning of the statistical values used is given in the subsequent section, more detail on how these are used is presented in the analysis of each research question. The antecedent flows for Environment Agency (EA) BMI and fish data were determined using a statistical model (Visser et al., 2016). This examined how the yearly antecedent flow conditions impacted upon fish (*Salmo Trutta*) and BMI scores which used the p-value and R^2 statistical tests, as described below.

Regression analysis: Correlation coefficient: the correlation coefficient is a standardised measure of relationship between two variables, for example, the relationships of numbers of fish to an increase in flow. Different values corresponded to different relationships as shown in Table 3.3.

Table 3.3- Correlation coefficient interpretations. Based on (Field 2009)

From	To	Meaning	Colour Code
-1		Perfect negative correlation	
-0.71	-0.9	Excellent negative correlation	
-0.51	-0.7	Good negative correlation	
-0.21	-0.5	Weak negative correlation	
-0.2	0.2	No correlation	
0.21	0.5	Weak positive correlation	
0.51	0.7	Good positive correlation	
0.71	0.9	Excellent positive correlation	
1		Perfect positive correlation	

A value of 1 or -1 indicates that as one variable increases or decreases, the other variable will also increase or decrease. For example if the relationship between flow and fish population was 1, this would indicate that as flow increases, fish populations would also increase. This would however require further investigation through R^2 and p-values, (explained below). A value of 0 however indicates as one variable changes the other does not (Field 2009).

Regression analysis: $\mathbf{R^2}$: the R^2 value is a statistical measure of strength of a relationship. The value indicates how close to the regression line the points are. Generally the higher the R^2, the better the model fits the data. Values >0.5 are considered as a strong relationship (Field 2009).

Regression analysis: p-value: the p-value is the estimated probability of rejecting the null hypothesis when that hypothesis is true. The closer the p-value is to 0, the higher the significance of the result is (Table 3.4) (Field 2009).

Table 3.4- p-value interpretations

p-value	Meaning	Colour code
P<0.05	Statistically significant	
P<0.01	Very statistically significant	
P<0.001	Highly statistically significant	

Mann-Whitney: the Mann-Whitney test was used to indicate if statistical differences ($p<0.05$) occurred between different yearly habitat results. This is the non-parametric equivalent of independent t-tests. Mann-Whitney was used as the assumption was that evenly distributed data did not exist within the results (Field 2009). The Mann-Whitney tests were calculated in the software, R-Studio.

3.1.5 Fieldwork and health and safety

Fieldwork for data collection was carried out on the River Nar in:

Spring: 5th-11th May 2013, 19th-22nd May 2014,

Summer: 6th-8th July 2013, 14th-16th July 2014,

Autumn: 20th- 23rd October 2013,

Winter: 29th-31st January 2014.

Note- benthic macro-invertebrate sampling and macrophyte surveys carried out in May 2013 were used as test data and were not included in the final analysis.

All fieldwork was carried out in accordance with the risk assessment approved by health and safety officers at Heriot-Watt University (see Appendix C). The following rules were abided by:

- Life jacket was worn at all times
- At least two people were present at all times
- Steel toe capped boots were worn
- High- visibility jackets were worn
- The river was not entered if over waist deep or too fast flowing (over 1m/s)

3.1.6 Conceptual model

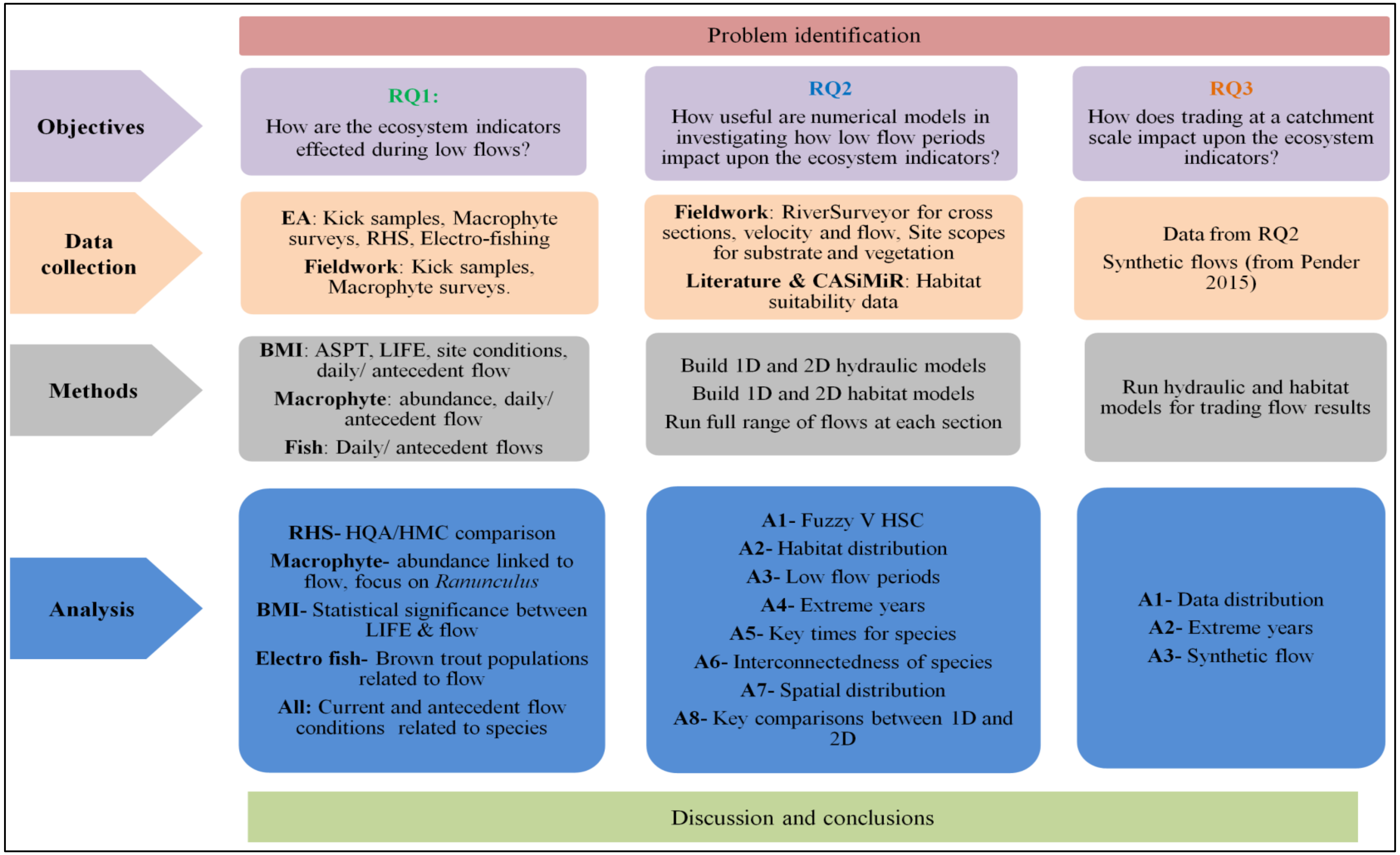


Figure 3.1- Conceptual model of thesis

3.2 Research question 1: How are the ecosystem indicators effected during low flows?

SNIFFER (2012) determined six ecological indicators to assess the implications of over-abstraction on freshwater ecosystems, this study uses three of these indicators (fish, macrophytes and benthic macro-invertebrate) to investigate the effects of flow regime change on them. This research question intended to show how the ecosystem indicators react during low flows. Data collected in the field and historical Environment Agency (EA) data was used to analyse the past and current status of the ecosystem indicators in the River Nar and to relate these to flow conditions.

3.2.1 Environment Agency River Habitat Survey data

The first area of analysis was using River Habitat Survey (RHS) data, provided by the EA to assess the character and habitat quality of the river based on its physical structure (Raven et al., 1998). The RHS method collects over 200 data entries for in-stream and banks for channel vegetation types, artificial features, and flow types for example. One of the ecological indicators determined by SNIFFER (2012) to assess the implications of over-abstraction, which is not directly assessed in this study, is the physical indicators. The RHS data however gives insight into the quality of some of the physical indicators at sites such as loss of riffles, runs and pools, lack of active channel bars and low width to depth ratio. The method results in three main outputs, described below:

Habitat Quality Assessment (HQA)- The HQA is a measure of ‘naturalness’ and diversity of a river corridor. Points are given for presence of features such as: point, side and mid-channel bars, eroding cliffs, large woody debris, waterfalls, backwaters and floodplain wetlands. Further points are given for variety of channel substrate, flow-types, in-channel vegetation and the extent of near natural land-use adjacent to the river. Therefore the higher the points awarded, the more natural and diverse the site is. HQA scores for UK rivers are usually between 10-80 (Naura 2012). A limitation to this scoring method is that it is based on expert opinion which is subjective in nature and can therefore only be used as a guide to habitat quality (Raven et al., 1998).

Habitat Modification Score HMS and Habitat Modification Class (HMC) - The HMS gives an indication of artificial modification to river channel morphology. Data from RHS and bank vegetation notes are used to provide an assessment of the ‘naturalness’ of the river banks with regard to the degree and extent of artificial modifications. HMS is calculated based on point allocation for the presence and extent of artificial features such as culverts and weirs and also modifications caused by re-profiling, canalisation and reinforcement of banks. In contrast to HQA, higher scores are given for greater and more severe modifications in the channel section (Table 3.5). The Habitat Modification Class (HMC) protocol allocates the condition of the channel at a site to one of five modification classes, based on the total HMS score (Table 3.5) (Naura 2012).

Table 3.5- HMS and HQA interpretations (Birkby 2015)

HMS score	**HMC score**	**Descriptive category of channel modification**
0-16	1	Pristine/semi natural
17-199	2	Predominantly unmodified
200-499	3	Obviously unmodified
500-1399	4	Significantly modified
1400+	5	Severely modified

HQA score	**Category**
0-20	Very poor
20-40	Poor
40-60	Fair
60-80	High
60-100	Very high

RHS data from 17 sites measured from downstream at Setchey to upstream at Mileham (Figure 3.6) were available from 1994 to 2013. Data that was collected pre and post 2008 was collected on different RHS survey forms and was interpreted in a different way, therefore it was not possible to make a direct comparison between the scores.

RHS results cannot be directly related to flows, as factors not directly related to in-stream flows are taken into account such as riparian vegetation and modification of the channel. The results were used to show how natural or unnatural different areas of the river were. This enabled an assessment into the likelihood of the indicator species being present or whether external factors are influencing habitat availability. For example if a RHS shows good quality habitat at a site but species are not present or have low abundances, it is likely that the flow conditions or external factors such as abstraction or pollution is affecting the species. This therefore allows investigations into how the flow is influencing the habitat availability.

<u>Analysis</u>

In order to assess the quality of habitat in terms of physical indicators at the sites, the HQA and HMC were used to show habitat quality at different areas of the river. For the reasons noted above, pre 2008, 2008 and post 2008 were compared separately. Graphs were drawn to show how habitat quality changes from upstream to downstream on the river which presented the quality of habitat at various sites.

3.2.2 Benthic Macro-Invertebrate sampling

Benthic macro-invertebrate (BMI) are commonly used as an indicator of river health. SNIFFER (2012) determined seven indicators of stress from over-abstraction and flow regime change, such as a major reduction in taxon richness, absence of baetid mayflies and dominance of *Gammarus spp*. In this study BMI data was collected to show the current quality of BMI in the river and to relate to flow conditions and physical processes through the RHS scores.

BMI sampling (commonly referred to as kick sampling) was carried out at 11 sites from downstream at Highbridge to upstream at Mileham (Figure 3.4) in accordance with the British Standard EN ISO 10870:2012 ‘Water quality- Guidelines for the selection of sampling methods and devices for BMI in fresh waters’ (European Standard 2003b). The sites were determined based on a spread of sites along the river to represent the whole river and to, where possible, correspond to EA sites and macrophytes sites to enable comparisons with historical data.

<u>Protocol</u>

The methods to collect BMI data in the field are detailed below:

- The intended site was located and details were noted such as adverse conditions and adjacent land use, a picture of the site was taken before entering the river.
- The sampling net (1mm mesh pond hand net, Figure 3.2) was immediately placed on the river bed facing downstream from where sampling occurred.
- The bed of the river was disturbed by kicking it and moving ones feet around for 3 minutes, this process collected eroded substrate and benthic organisms from the disturbed area (Figure 3.2).
- After three minutes the sample net was lifted out of the water ensuring to keep facing upwards to safeguard material being lost.
- The contents of the net were emptied into a white tray ensuring all material was removed which often required carefully picking contents off the net. Large stones or branches were removed from the sample after being thoroughly checked for invertebrates.
- The remaining contents were emptied into a sampling jar and filled with ethanol to preserve them for lab analysis. The jar was clearly marked where and when the sample was taken.

Figure 3.2- 1mm sampling net and kick sampling being carried out on the River Nar

The methods for analysis in the lab are described below:

- The contents of the jar collected in the field were emptied into a sediment tray and using tweezers, invertebrates were carefully taken out and placed in the white tray.
- The invertebrates were then identified to family level and counted, using a microscope for identification of smaller or more unusual invertebrate.
- Once this was complete, the scoring indices were determined based on the ASPT and LIFE scoring indices (described in section 2.5).

Analysis

Analysis of the measured kick samples intended to show seasonal changes to BMI populations and therefore how the species react to various flows during different seasons. This complements the EA data as they do not collect samples on a seasonal basis (Appendix D). Overall conclusions were drawn on how the lowest flows affect the habitats. ASPT and LIFE scoring methods were used to assess the condition of the BMI in the river.

Analysis 1- Seasonal change and trends: ASPT and LIFE scores were analysed separately to show how different seasons affect the scores. Scores were plotted against sites upstream to downstream for the different seasons. Any clear trends were noted. Average scores per season were also calculated to show the seasons which provided the highest and lowest scores more clearly and to give indication about the river condition, shown in tables 2.6 and 2.7. The differences between chalk and fen reaches of the river were also highlighted. Little has been done within literature to indicate whether there is a trend between scoring indices and flow. Chainho et al., (2007), however found the lowest numbers of invertebrate during winter and spring, whilst summer and autumn had large numbers on the Mondego river in Portugal.

In order to show any trends between seasons, scores were plotted against seasons at each site. By demonstrating if sites have similar patterns of highest to lowest scores, it could be seen which seasons provide the optimum and poorest conditions for BMI and also if different sites differ in trends.

Analysis 2- Daily and antecedent flows: Correlation coefficients were calculated to investigate the relationship between the flow on the day of the kick sample and the ASPT and LIFE scores. Regression analysis (p-value and R^2) was calculated to examine the significance and strength of this relationship. The same process was then used to show the relationship between the 3, 6 and 9 month antecedent Q_{50} flow (average flow) with the BMI scores. If the p-value and R^2 relationship was weak then this indicated other abiotic and biotic factors could be affecting BMI scores. However a strong relationship indicated flow has a strong relationship with the scores.

Analysis 3- Site conditions: Using the highest and lowest site scores for ASPT and LIFE, analysis showed if there were similar site conditions between sites with the lowest and highest scores, for example if a certain substrate was found in all lowest scoring sites. By also linking in RHS scores, this analysis showed which site conditions provided the optimum and poorest conditions for BMI.

3.2.3 Environment Agency Benthic Macro- Invertebrate sampling

The Environment Agency (EA) had collected benthic macro-invertebrate (BMI) data at 10 sites (Figure 3.4) along the river from 1985 to 2012 (see Appendix D for dates collected). The data included BMWP, ASPT and LIFE scores for each site. Only 9 sites were used in this analysis as one of the sites was located on a side channel and therefore has a different catchment area and river typology. Only ASPT and LIFE were used in the analysis as BMWP is mostly unused by the EA (SNIFFER 2011).

Analysis

The analysis firstly investigated relationships between the BMI scores and daily flow, followed by the effect of seasonal flows. Finally the influence of antecedent flows on BMI scores were investigated.

Analysis 1- Daily flow: Relationships were investigated between daily flow and ASPT and LIFE scores by calculating correlation coefficients for each of the nine sites. The strength and significance of these relationships were then determined through the R^2

and p-values. Where there were strong correlation coefficients backed up by significant p-value and R^2 values indicated that flow has a strong relationship with the scores.

Analysis 2- Seasonal flow: this analysis was split into 2 parts:

- **Firstly,** average ASPT and LIFE scores per site were plotted to demonstrate where the highest and lowest scores occurred. This was compared with collected data if available (Section 3.2.2).
- **Secondly,** ASPT and LIFE scores were analysed to investigate the relationship between scores in different seasons and daily flow. As no sites or years had data collected for all four seasons, only seasons with four or more years of collected data were considered to ensure a sufficient length of series. The results showed how BMI would potentially be affected if flows were increased or decreased.

Analysis 3- Antecedent conditions: Work reported in Visser (2014) was used to show how antecedent conditions and time lag flows affect BMI response in the River Nar. A method was developed based on work from the EA and further used in the DRIED-UP studies (Clarke and Dunbar 2005; Dunbar et al., 2006). Linear regression modelling was used to link antecedent flows to BMI response (LIFE), with a view to setting environmental flows. The methodology consisted of four scenarios relating LIFE (1993-2012) to antecedent flow (1993-2012):

Scenario A: Chalk River – Spring LIFE

Scenario B: Chalk River – Autumn LIFE

Scenario C: Fen River – Spring LIFE

Scenario D: Fen River – Autumn LIFE

Table 3.6- Models produced for each scenario for antecedent flow analysis (Visser et al., 2016)

	Model variables		**Model name**	**No. of models**
	EV1	**EV2**		
1	Summer Q10		S10	6
2	Summer Q95		S95	6
3	Winter Q10		W10	6
4	Winter Q95		W95	6
5	Summer Q10	Summer Q95	S10-S95	36
6	Winter Q10	Winter Q95	W10-W95	36
7	Winter Q10	Summer Q95	W10-S95	36
8	Winter Q95	Summer Q10	W95-S10	36
9	Winter Q95	Summer Q95	W95-S95	36
10	Winter Q10	Summer Q10	W10-S10	36

The flow indices represent high (Q_{10}) and low (Q_{95}) flows. Each scenario is made up of 10 models (Table 3.6). The explanatory variable (EV), flow (t), is time offset to account for the lag in response. This begins with t, the antecedent, immediately preceding flow (flow variables recorded 0-180 days before LIFE sampling) up to t_{-5} (5 years previous antecedent flow). All possible combinations were considered and all regression modelling was calculated using R-Studio (Visser et al., 2016).

No new work was be carried out by the author here, only comments on the results from (Visser 2014).

3.2.4 Macrophyte surveys

The second indicator species used was macrophytes; SNIFFER (2012) determined 14 macrophyte, bryophyte and diatom indicators of stress on river ecosystems such as dominance of emergent plants in relation to submerged plants, absence of submerged aquatic macrophytes and presence of non-rooted free-floating species such as Duckweed. Here the aim was to investigate which species were present on the river and to assess their abundance and natural yearly cycle. This shows how macrophytes respond to different flows and whether the response is within the natural growth cycle.

Macrophyte surveys were carried out at nine sites (Figure 3.5) in accordance with British Standard EN 14184; 'Water quality- Guidance for surveying of aquatic macrophytes in running waters' (European Standard 2003a). The specific method followed was based on the STAR project which is a macrophyte field survey procedure used to assess the ecological status of watercourses in Britain within the aims of the Water Framework Directive (WFD) (Dawson 2002). The results were used for seasonal analysis and to investigate if daily and antecedent flows affect macrophyte abundance.

Survey preparation

The steps involved in the survey preparation are detailed below:

- Nine representative sites were determined (Figure 3.5) which, as required by the British Standard, was based on the objectives of the survey, degree of confidence required from the data and resources and expertise available. An area was chosen based on a spread of sites along the river and which could be safely accessible during all seasons.
- Before fieldwork commenced, survey sheets were devised (Appendix E), which consisted of one sheet per site and included space for information on date and time of survey, site number and location, grid coordinates, adverse conditions, approximate depth(m) and width(m), main substrate type, artificial features, bank uses and pictures.

- The other side of the sheet had a pre-drawn map of the site which consisted of a grid marking the left bank, the right bank and 10m intervals. Here there was also a reference to the percentage cover scale for the vegetation (Appendix E).
- From information about the site and from local knowledge, the four main types of vegetation were determined (Narrow-leaved-water parsnip (*Berula erecta*), Crowfoot (*Ranunculus Fluitans*), Starwort (*Callitriche spp*) and common reeds). These were input to the sheet with an area to note the percentage covers of each.

The British Standard recommends to survey each site on two separate occasions during each sampling year to allow for different growth rates between species. However this study was interested in different seasonal changes to relate to flow, so surveys took place once each season for a full year.

Survey technique

The survey technique is detailed below. *Note*- if the river section was too deep to wade, the surveyor completed the assessment from the bank.

- The surveyor measured out a 100m distance (Dawson 2002) using measuring tapes in a downstream to upstream direction where possible.
- The 100m length was then walked, on the bank, for an initial scope and idea of the vegetation in the river, site information and any other relevant information was recorded.
- The surveyor entered the river at the most downstream point in order not to cover vegetation with disturbed substrate. The surveyor then waded up the river slowly in a zig-zag manor to map/draw the vegetation on the survey sheet and to estimate the species cover. The percentage scale is based on Dawson (2002) (Table 3.7).
- The surveyor then waded back downstream validating their mapping, note was taken that the surveyor’s movement in the water could have affected the visibility of the vegetation. Reference sheets were taken with pictures and information on all known macrophytes within the river for help identifying. If unknown vegetation was determined, it was photographed and marked as ‘unknown vegetation’.
- GPS coordinates were taken at the most upstream and downstream points of the survey site. Photographs were taken at the downstream, upstream and middle points, looking across, looking downstream and looking upstream at each point.

Limitations to the method are survey error and variation between errors in estimates of macrophyte cover and/or mis-identification of macrophytes. The STAR method notes that to reduce these errors, adequate training should take place and adoption of quality assurance measures. In this study however, there were no macrophyte experts involved. In order to reduce errors, the survey was done by the same surveyor each time and the percentage covers were within a scale rather than an exact percentage. Furthermore, only the four main macrophyte species (described above) were identified in stream. Studies have used similar recording methods previously due to difficulty in surveying to species level (e.g. Holmes 1999).

Figure 3.3- Macrophyte survey being carried out on the River Nar

1	≤0.1%
2	0.1-1%
3	1-2.5%
4	2.5-5%
5	5-10%
6	10-25%
7	25-50%
8	50-75%
9	≥75%

Table 3.7- Percentage cover estimates of macrophytes

Analysis

Three main areas of analysis were completed with the measured macrophyte survey data. During analysis, natural fluctuations in the occurrence of species were taken into account (Dawson 2002). The raw results were firstly plotted in a table to show the abundances per season and to indicate any areas where data was unusable.

Analysis 1- All species and daily flow: Relationships were investigated between daily flow and macrophyte abundance (Table 3.7) by calculating correlation coefficients for each of the sites. The strength and significance of these relationships were then determined through p-values and R^2. Times where there were strong correlation coefficients backed up by significant p-value and R^2 values, demonstrated that flow has a relationship with the macrophytes. From this it was determined how macrophytes would react to increasing or decreasing flows.

Analysis 2- All species and seasonal changes: Seasonal site changes were assessed noting any patterns. The graphs created for Analysis 1 presented abundances in each season and from this it could be seen which seasons had highest and lowest abundances of macrophytes. Macrophyte abundances were then linked to site conditions to determine whether they have preferences, for example for a certain type of substrate.

This aimed to show if abundances are more related to site conditions and/or seasons or rather daily flows from Analysis 1.

Analysis 3- *Ranunculus* daily and antecedent flow: Westwood et al., (2006) discovered that stream flow is statistically the biggest environmental variable influencing *Ranunculus*. Correlation coefficients were determined to investigate if there was any relationship between daily flow and *Ranunculus* abundance. The strength and significance of these relationships were then determined through the R^2 and p-values. Antecedent flow conditions (Section 3.1.3) were calculated to determine if there was any relationship (Correlation coefficient, R^2 and p-value) between the 3, 6 and 9 months antecedent Q_{10}, Q_{50} and Q_{90} flows with *Ranunculus* abundance. These results indicated whether *Ranunculus* had any relationship with current and/or antecedent flows.

3.2.5 Environment Agency Macrophyte data

The EA provided macrophyte survey data for five sites from downstream in Setchey to upstream in Mileham (Figure 3.5). The sites and dates recorded are shown in Table 3.8.

Table 3.8- Years where EA macrophyte surveys were carried out

ID	Name	Year a	Year b
1	Setchey RB	2003	
2	Narborough RB	2002	2003
3	West Acre RB	2002	2006
4	West Lexham	2002	2003
5	Mileham RB	2002	2003

All samples were taken in June or July of each recorded year; therefore no seasonal analysis could be investigated. Furthermore as only two years of data were available for each site, there was not enough data for statistical analysis. Due to this limited data, minimal analysis could take place.

Analysis 1: the total abundance of each species was plotted. These abundances were assessed to show which years provided the highest and lowest abundances. This was then linked to daily flow at each site to demonstrate whether higher or lower flows provide higher abundance. Thus providing an indication as to how low flows would impact on macrophytes.

3.2.6 Environment Agency electro-fishing data

The final indicator species was fish, with a focus on brown trout. The analysis aimed to show how brown trout populations respond to different flows in order to assess how low flows impacted upon them. The EA provided electro-fishing data from thirteen sites from downstream at Kings Lynn to upstream at Mileham (Figure 3.6) with 21 species of fish recorded. Only seven sites were used however, as these were the only sites where brown trout were found. The data spans from 1989 to 2014 (Appendix F).

Analysis

The analysis firstly investigated the relationship between daily flow and brown trout populations, followed by determining the relationship between antecedent flows and brown trout populations.

Analysis 1- Brown trout and daily flows: As the study focused on brown trout as an indicator species, changes in brown trout populations through recorded history were demonstrated indicating whether there had been an increase or decrease in population numbers. Fish numbers were then statistically correlated to flow on the day of measurement on both a site basis and for all flows combined. This indicated whether flow has an effect on the brown trout populations. Results were presented for all sites throughout recorded history.

Analysis 2- Brown trout and antecedent flows: The same process and methods employed by Visser et al.,(2016) was used to investigate if there was a relationship between brown trout populations and antecedent flow conditions. A multiple regression model was applied assessing combinations of variables (Table 3.10). The model pooled data from the chalk stream reach and used normalised flows. The models were based on the antecedent conditions shown in Table 3.9 in order to show low, medium and high flow conditions. All of the combinations of these antecedent conditions were run to create 10 different models.

Table 3.9- Antecedent conditions used in the 10 models with 1 or 2 variable combinations (M1 etc.)

	Variable 1	**Variable 2**
M1	Summer Q10	
M2	Summer Q95	
M3	Winter Q10	
M4	Winter Q95	
M5	Summer Q10	Summer Q95
M6	Winter Q10	Winter Q95
M7	Winter Q10	Summer Q95
M8	Winter Q95	Summer Q10
M9	Winter Q95	Summer Q95
M10	Winter Q10	Summer Q10

Table 3.10- Regression modelling combination for models 1-10. t is antecedent flow year(s)

	Variable 1 combination							Variable 2 combination					
M1, M2, M3, M4	T0												
	T0	T-1											
	T0	T-1	T-2										
	T0	T-1	T-2	T-3									
	T0	T-1	T-2	T-3	T-4								
	T0	T-1	T-2	T-3	T-4	T-5							
M5, M6, M7, M8, M9, M10	T0							T0					
	T0	T-1						T0					
	T0							T0	T-1				
	T0	T-1						T0	T-1				
	T0	T-1	T-2					T0					
	T0							T0	T-1	T-2			
	T0	T-1	T-2					T0	T-1				
	T0	T-1						T0	T-1	T-2			
	T0	T-1	T-2					T0	T-1	T-2			
	T0	T-1	T-2	T-3				T0					
	T0							T0	T-1	T-2	T-3		
	T0	T-1	T-2	T-3				T0	T-1				
	T0	T-1						T0	T-1	T-2	T-3		
	T0	T-1	T-2	T-3				T0	T-1	T-2			
	T0	T-1	T-2					T0	T-1	T-2	T-3		
	T0	T-1	T-2	T-3				T0	T-1	T-2	T-3		
	T0	T-1	T-2	T-3	T-4			T0					
	T0							T0	T-1	T-2	T-3	T-4	
	T0	T-1	T-2	T-3	T-4			T0	T-1				
	T0	T-1						T0	T-1	T-2	T-3	T-4	
	T0	T-1	T-2	T-3	T-4			T0	T-1	T-2			
	T0	T-1	T-2					T0	T-1	T-2	T-3	T-4	
	T0	T-1	T-2	T-3	T-4			T0	T-1	T-2	T-3		
	T0	T-1	T-2	T-3				T0	T-1	T-2	T-3	T-4	
	T0	T-1	T-2	T-3	T-4			T0	T-1	T-2	T-3	T-4	
	T0	T-1	T-2	T-3	T-4	T-5		T0					
	T0							T0	T-1	T-2	T-3	T-4	T-5
	T0	T-1	T-2	T-3	T-4	T-5		T0	T-1				
	T0	T-1						T0	T-1	T-2	T-3	T-4	T-5
	T0	T-1	T-2	T-3	T-4	T-5		T0	T-1	T-2			
	T0	T-1	T-2					T0	T-1	T-2	T-3	T-4	T-5
	T0	T-1	T-2	T-3	T-4	T-5		T0	T-1	T-2	T-3		
	T0	T-1	T-2	T-3				T0	T-1	T-2	T-3	T-4	T-5
	T0	T-1	T-2	T-3	T-4	T-5		T0	T-1	T-2	T-3	T-4	
	T0	T-1	T-2	T-3	T-4			T0	T-1	T-2	T-3	T-4	T-5
	T0	T-1	T-2	T-3	T-4	T-5		T0	T-1	T-2	T-3	T-4	T-5

Analysis 3- Combined analysis: The final analysis investigated if high numbers of brown trout correspond to high numbers of invertebrate and macrophytes. Brown trout numbers were plotted against ASPT and LIFE scores at Marham and Castle Acre, which were the only two sites that could be combined in this way. The location of the macrophyte sites did not correspond to these sites so could not be used (Table 3.11). It was investigated if there was any correlation between brown trout numbers and invertebrate scores by using correlation coefficients, p-values and R^2. This was important to investigate as it shows whether brown trout are influenced by food source availability. This can be used to inform the analysis in research question 2, particularly during the interconnectedness analysis (Section 3.3.7).

Table 3.11- Locations of all survey data

EA Electro-fishing		**EA BMI score**			**EA Macrophyte surveys**		
ID	Name	**ID**	**Name**	**Distance**	**ID**	**Name**	**Distance**
1	U/s Sluice Kings	1	Kings lynn	0m	n/a	n/a	n/a
2	D/s Setchey	2	Setchey	0m	5	Setchey	0m
3	Wormegay High	3	Highbridge	0m	n/a	n/a	n/a
4	Abbey Farm	n/a	n/a	n/a	n/a	n/a	n/a
5	Marham intake	4	Marham	0m	n/a	n/a	n/a
6	Narford Hall	n/a	n/a	n/a	n/a	n/a	n/a
7	Warren farm	n/a	n/a	n/a	3	West Acre RB	800m DS
8	Manor farm	n/a	n/a	n/a	n/a	n/a	n/a
9	Castle Acre	7	Castle acre	0m	n/a	n/a	n/a
10	West Lexham	n/a	n/a	n/a	2	West Lexham	300m DS
11	East Lexham	n/a	n/a	n/a	n/a	n/a	n/a

3.2.7 Location maps

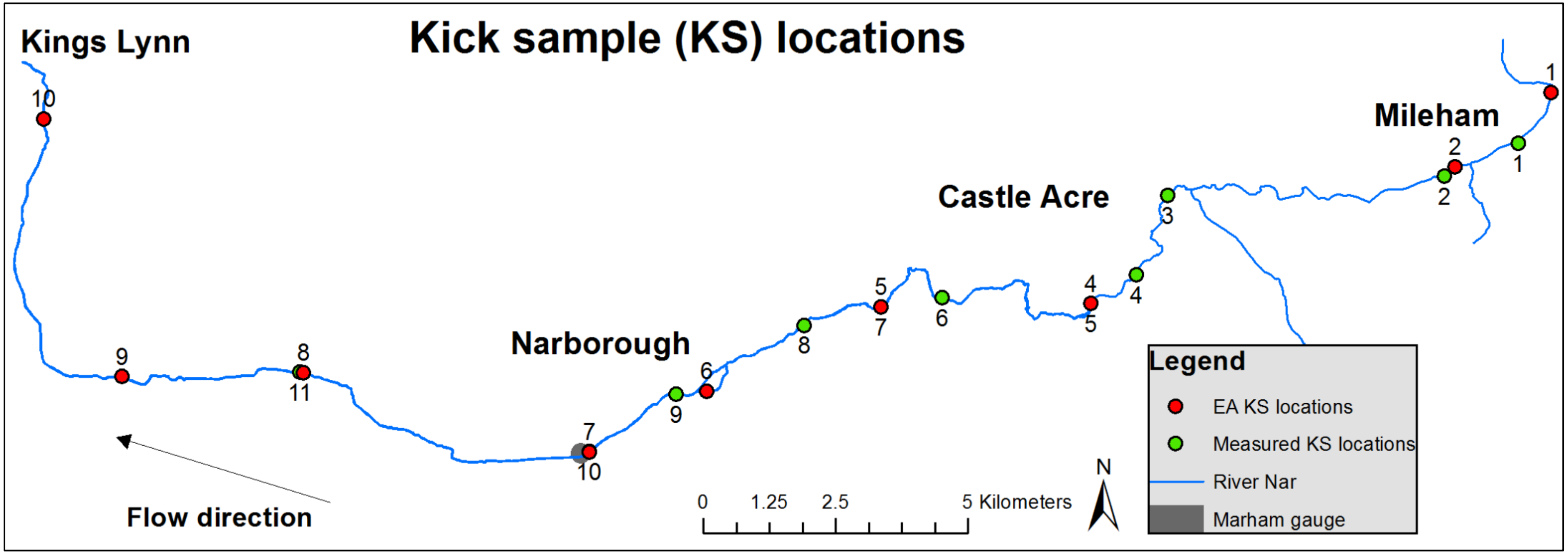


Figure 3.4- EA and measured kick sample survey locations

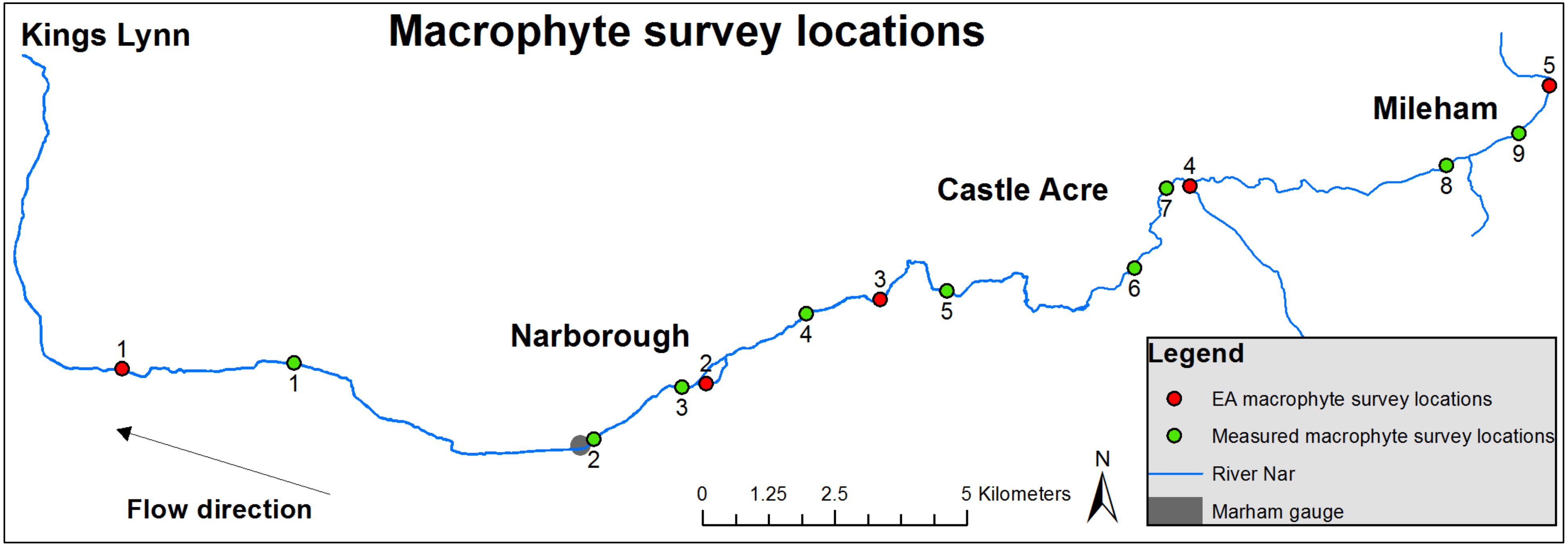


Figure 3.5- EA and measured macrophyte survey locations

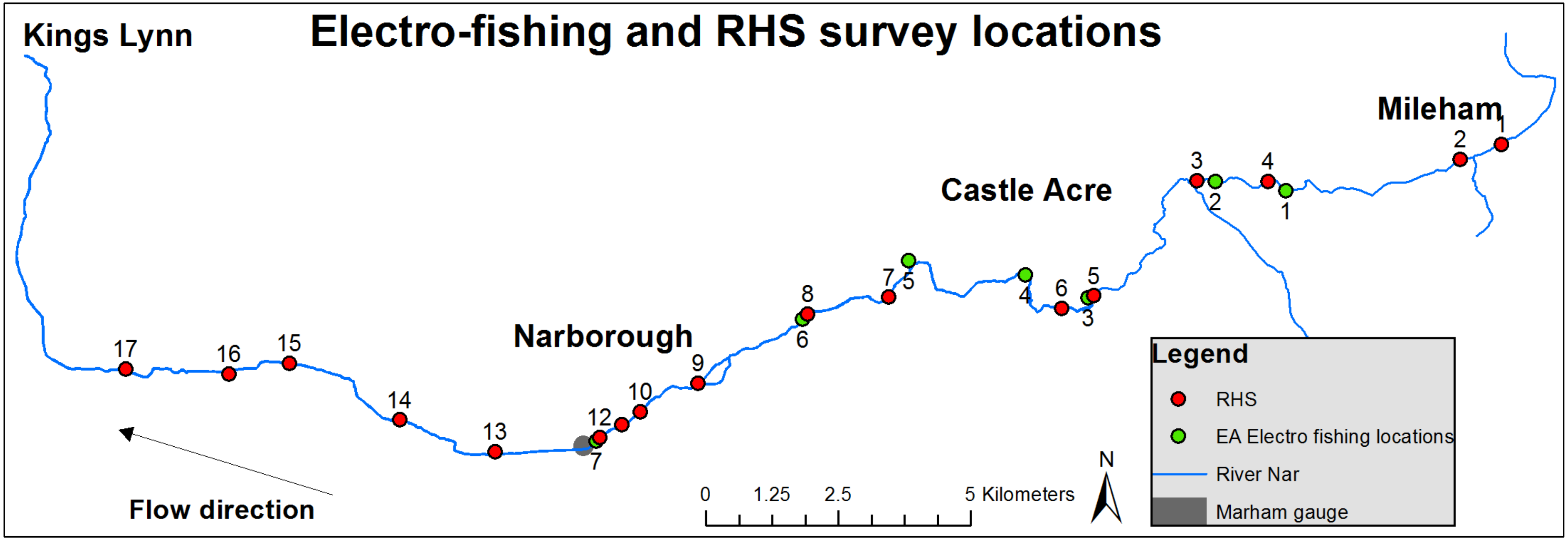


Figure 3.6- EA electro-fishing and RHS survey locations

3.3 Research question 2: How useful are numerical models in investigating how low flow periods impact upon the ecosystem indicators?

A methodology using habitat models at three sites on the river was used to investigate trends in habitat availability according to different flows through the 32 year period (1980-2011). Using the results from research question 1 as a baseline, the models aimed to show how low flows impact on the ecosystem indicators. Habitat models have come under much criticism for not incorporating all relevant abiotic and biotic factors and for being highly sensitive to input parameters (Mouton et al., 2008; Boavida et al., 2014). These criticisms were addressed in this research question showing how useful models are for investigating impacts of flow regime change on freshwater species.

The modelling aspect of the work did not aim to predict the future but rather to aid management decisions for the future by showing how low flows could impact the natural variance of the ecosystem indicators.

Please note- Chapter 4 (Model Build) explains how each of the 1D and 2D hydraulic and habitat models were built and using what data. Here the analysis carried out with the output is explained.

3.3.1 Site determination

Three representative sites, 1-2km in length, were used to represent the ecology of the river. It was ideal to incorporate as much ecological data both from the EA and measured as possible to allow comparisons between model outputs and ecological data. This was therefore taken into account when determining sites. Details and justification of the sites chosen are provided below and locations are shown in Figure 3.10. The analysis that takes place for each site is shown in Table 3.12.

<u>Site 1- Highbridge (990m)</u>

This is the most downstream site and represents the fen section of river (Figure 3.7). Downstream from Marham, the fen river is relatively uniform, thus habitats have similar traits along this reach. The site is highly canalised and has high embankments on both sides and water levels are above floodplain level. There is a footpath along the top of the embankments which allows easy access.

The EA Flood Modeller hydraulic model encompasses this site, therefore cross sections and flow data were available. The site also has data for: EA and measured kick samples, measured macrophytes, and River Habitat Surveys (RHS). Modelling was done in 2D due to the width of channel.

Figure 3.7- Site 1- Highbridge (most downstream site)

Site 2- DS Nar (827m)

This site is located in the middle of the river in the chalk stream reach and has historically been straightened (Figure 3.8). The site is an important location for fish spawning and has large macrophyte abundances. The Nar Valley Way footpath runs alongside both banks and therefore provides easy access.

The cross sections measured with the RiverSurveyor in May 2013 encompassed this area (see Chapter 4); therefore cross sections and flow data were available for this site. The site also has data for: Measured kick samples, EA and measured macrophytes, and RHS's. Due to the size of the channel (~5m), modelling was done in both 1D and 2D. This also enabled comparison between the two modelling techniques.

Figure 3.8- Site 2- DS Nar (mid-stream site)

Site 3- Castle Acre (512m)

This is the most upstream site which is also in the chalk stream however being towards the upper end of the river the river narrows and becomes much shallower, thus habitat is likely to be different to site 2. The site exhibits relatively unmodified features and is important for fish spawning (Figure 3.9). There are also large macrophyte abundances. A fisherman's path runs alongside the right bank which provides relatively easy access. The downstream end of the site is around 50m upstream of the waste water treatment outlet which provides excessive nutrient input to the river.

The cross sections measured with the RiverSurveyor in May 2013 started at this point (see Chapter 4); therefore cross sections and flow data were available for this site. The site also had data for: Measured kick samples, measured macrophytes, EA RHS's and EA electro-fishing. Modelling was carried out in 1D only due to small channel widths.

Figure 3.9- Site 3- Castle Acre (most downstream site)

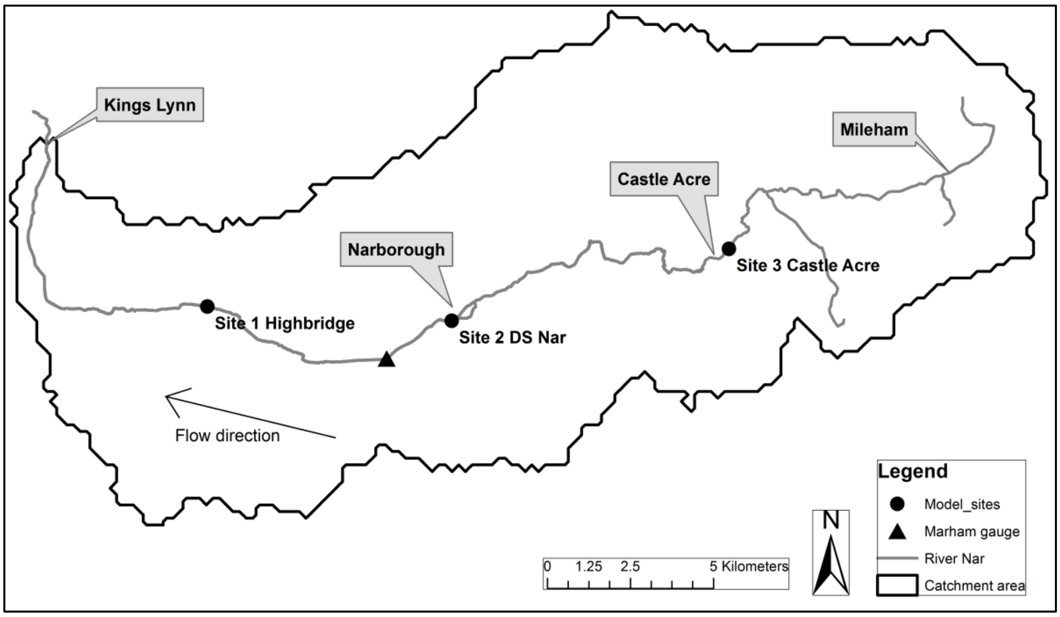


Figure 3.10- Model site locations on the River Nar

Table 3.12- Analysis undertaken for each site, gray indicates analysis was undertaken

		Site 1- Highbridge (2D)	Site 2- DS Nar (2D)	Site 2- DS Nar (1D)	Site 3- Castle Acre (1D)
A1	Fuzzy V HSC			gray	gray
A2	Habitat distribution	gray	gray		gray
A3	Low flow periods	gray	gray		gray
A4	Extreme year	gray	gray		gray
A5	Key times for species	gray	gray		gray
A6	Interconnectedness of species		gray		gray
A7	Spatial Distribution		gray		gray
A8	Key comparisons between 1D and 2D		gray	gray	

3.3.2 Analysis 1- Fuzzy V HSC

This analysis focused on comparing fuzzy rule results with HSC results for each indicator species for the 32 year period. It was important to show the sensitivity of results in order to investigate how useful habitat models are in investigating impacts of changing flow on species. Using literature and output results, reasons behind differences in output were analysed and discussed. Justification led to determining which input method was most appropriate for use in subsequent analysis. Only sites modelled in 1D (site 2 and 3) were used in this analysis as the aim was to compare the input methods, therefore it was not required to compare between 1D and 2D input methods.

3.3.3 Analysis 2- Habitat distribution

This analysis investigated how the available habitat changed throughout the 32 year period for each species. This begins to unravel how habitat availability would change during varying flow periods. All sites were analysed in this section, site 2 was however only analysed in 2D. This analysis was divided into two parts:

Part A- Total suitability: The results were firstly analysed on the HHS scale to determine the quality of habitat availability in each site. The average HHS for each species was calculated and a corresponding HHS suitability scale was determined, (Table 3.13). This informed how good/poor the area was for the species.

Table 3.13- Total habitat suitability interpretation using HHS values

Suitability scale	Corresponding HHS values (-)
Very good suitability	0.81-1
Good suitability	0.61-0.8
Moderate suitability	0.41-0.6
Poor suitability	0.21-0.4
Very poor suitability	0-0.2

Part B- Suitability dependant on species: The range of results from the HHS scores were then analysed by categorising the results into three even bins (upper, middle and lower) to inform the understanding of the spread of predicted habitat suitability. For example if a species had a total range of results between HHS 0.3 and 0.6 i.e. varying between poor suitability and moderate suitability, this was binned into:

- Upper habitat category (0.5-0.6),
- Middle habitat category (0.4-0.49),
- Lower habitat category (0.3-0.39).

This therefore showed what percentage of the total habitat suitability was in the upper, middle and lower regions of the overall HHS suitability range.

3.3.4 Analysis 3- Low flow periods

The HHS curves showing HHS results against an increasing flow for each species were presented, with lines depicting what classifies as a ‘low HHS’ for each species, this was determined from part B in Analysis 2. A line also showed what classified as a low flow (Q_{90} at the site). From this it could be determined if low flows and ‘low HHS’ are interconnected as thus if low flows are likely to cause low habitat availability.

3.3.5 Analysis 4- Extreme years

This analysis was devised to show how habitat availability was affected during low flow years in comparison to wet and average years. The analysis was carried out for all three indicator species at all three sites.

Using SI values (rather than HHS), the five wettest, five driest years and five most average years recorded (1980- 2011) were compared with habitat availability for all species using the scale shown in Table 3.14. The five driest and wettest years used were determined based on most days in the year below Q_{90} and above Q_{10} respectively. The five average years were determined by mean flow being closest to Q_{50}. Per year a percentage of days at each scale were derived therefore the years with the highest number of ‘wet’ and ‘dry’ days were the wettest and driest years. By comparison of wet to dry years the assessment shows how low flow years affect habitat availability in comparison to wet periods.

Table 3.14- Habitat suitability interpretation using SI values

Suitability scale	**Corresponding SI values (-)**
Highly suitable	0.8-1
Suitable	0.6-0.8
Moderately suitable	0.4-0.6
Unsuitable	0.2-0.4
Highly unsuitable	0-0.2

Mann- Whitney statistical tests were conducted to compare each year (broken down per season) for the five wettest years with five average years, and the five driest years with five average years. The purpose of this was to show if there was a significant change in habitat availability ($p<0.05$), during different yearly conditions. Table 3.15 shows the years used for each seasonal wet, dry and average condition.

Table 3.15- Years used for Mann- Whitney analysis ('Av'= average)

Winter			**Spring**			**Summer**			**Autumn**		
Dry	**Wet**	**Av**	**Dry**	**Wet**	**Av**	**Dry**	**Wet**	**Av**	**Dry**	**Wet**	**Av**
1990	1988	1986	1990	1981	1984	1990	1980	1982	1989	1987	1983
1991	1994	1987	1991	1988	2002	1991	1981	1994	1990	1993	1985
1992	1995	1993	1992	1994	2003	1992	1987	2000	1991	1998	1988
1996	2001	1998	1996	1998	2007	1996	2001	2004	2009	2000	1999
2006	2003	2005	2011	2001	2010	2011	2007	2008	2011	2002	2010

3.3.6 Analysis 5- Key times for species

This analysis aimed to determine if habitat availability was affected for key times for Crowfoot and spawning brown trout and therefore if more protection of flow is required at these important times.

Brown trout spawning: Brown trout spawn between October and December in the UK (Armstrong et al., 2003). Therefore historic flows were assessed to show how autumn flows affect spawning habitat availability. Ideally the best available habitat would be in these months, if the models show there is 'lower' habitat availability in these months then this could indicate more protection of flows in this month is needed.

Crowfoot growing season: The main growing season for Crowfoot is between April and August thus flows at this time are of high importance (Holmes 1999). Therefore habitats between April and August were focused on showing how available habitat was affected during these periods.

The assessment for both species involved determining percentage of days at or below 'lower' HHS between the specified months. These were then plotted against dry years to show if there was any connection to the dry years. Conclusions were drawn on whether low flows are negatively impacting upon these key times and therefore if the flows need more protection during these times.

3.3.7 Analysis 6- Interconnectedness of species

By incorporating other biotic parameters into the model results, this analysis aimed to reduce criticism surrounding habitat modelling that only hydraulic components (i.e. depth and velocity), substrate and cover are used to assess habitat availability (Orth 1987; Maddock 1999). Spawning brown trout rely not only on appropriate hydraulic parameters; depth, velocity and substrate, they also require good cover and food sources (Figure 2.4) (Bagenal 1969; Jowett 1992). Here, results were combined to show how much habitat availability spawning brown trout actually have including their biotic dependants of refugia (Crowfoot) and food sources (Mayfly) upon which they rely. This analysis was split into two areas:

- Firstly, critical flows were determined showing the minimum and maximum flows required to sustain 'middle' and/or 'upper' availability for spawning brown trout and their biotic dependants.
- Secondly, the seasonal scenario based analysis aimed to show how the habitat availability distribution differed for all three species between wet, dry and average

years (as determined in analysis 4). The binned time series of results were analysed for the three species to determine 27 combinations of ‘upper’, ‘middle’ and ‘lower’ habitat availability. For the purposes of this analysis the optimum habitat conditions are ‘upper’ for all species and likewise the worst habitat conditions are ‘lower’ availability for all species. The analysis focused on the interaction between the species by understanding the timing of the most suitable habitat for all three species.

The term ‘interconnectedness’ is used to indicate how species cannot be examined on an individual basis and instead should be assessed in relation to their ‘connected’ nature to other species. In this situation these species being food sources (BMI) and refugia (Crowfoot). This includes both spatially connected and biotically connected i.e. fish need food to survive as well as appropriate hydraulic conditions.

3.3.8 Analysis 7- Spatial distribution

This analysis was designed to show how the habitat availability varies spatially for the Q_{10}, Q_{50} and Q_{90} flow for each species. The habitat suitability plan maps were output for each species and discussed. This showed how habitat availability changed for all species spatially for each site. This analysis is linked to Analysis 6 showing the interconnectedness of the species focusing on spawning brown trout, therefore the analysis is only for site 2 (2D) and 3 (1D) due to importance for spawning brown trout.

3.3.9 Analysis 8- Key comparisons between 1D and 2D

The final analysis aimed to show the key differences between the 1D and 2D results for site 2 (DS Nar). Whilst it was not the aim of this project to determine these differences, it was of importance to show how different hydraulic dimensions alter results based on the same fuzzy rules and using the same geographical area. This also helps to investigate how useful the models are in investigating habitat availability as could highlight limitations with data input. Analysis 2 (habitat distribution) was carried out comparing the 1D and 2D results to show how the distribution changes dependent on the method used.

3.4 Research question 3: How does water trading at a catchment scale impact upon the ecosystem indicators?

Results from the trading model were used in the habitat models, developed for research question 2, to investigate how trading and abstraction affect the indicator species.

3.4.1 Trading model background (technical aspects)

The water trading model was built using GAMS (General Algebraic Modelling System) software which used economic optimisation to simulate and track pair-wise water market transactions between individual water users. The aim of the model was to predict hydrologic and economic implications of water trading. The results showed changes in flow regime patterns as a result of the water trades which could then be used to assess potential ecological changes in the river. The river network was modelled as a series of 27 nodes and conveyance links representing: demands, storage reservoirs, junctions, river reaches i.e. tributaries, these created a connection matrix for the river and abstractors (see figure 3.11 for schematic of nodes and figure 3.12 for representative locations on the Nar) (Harou and Erfani 2014).

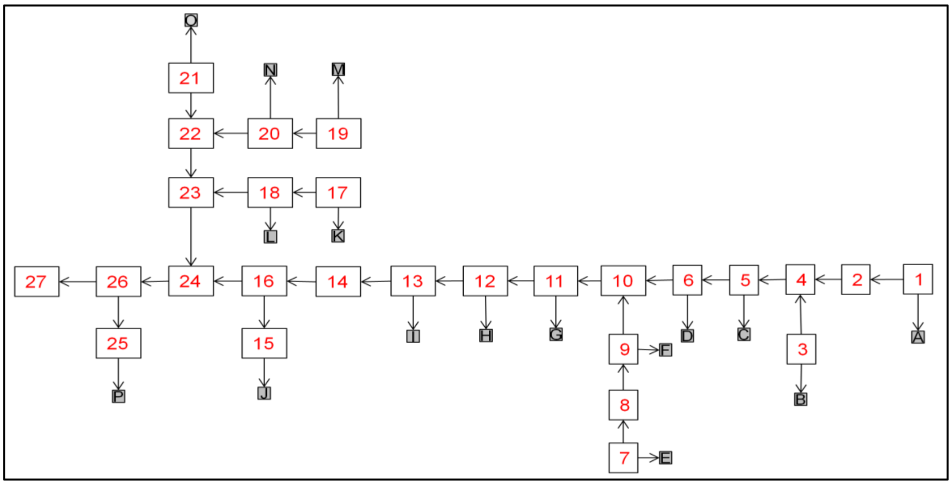


Figure 3.11- Schematic of trading node locations on the River Nar

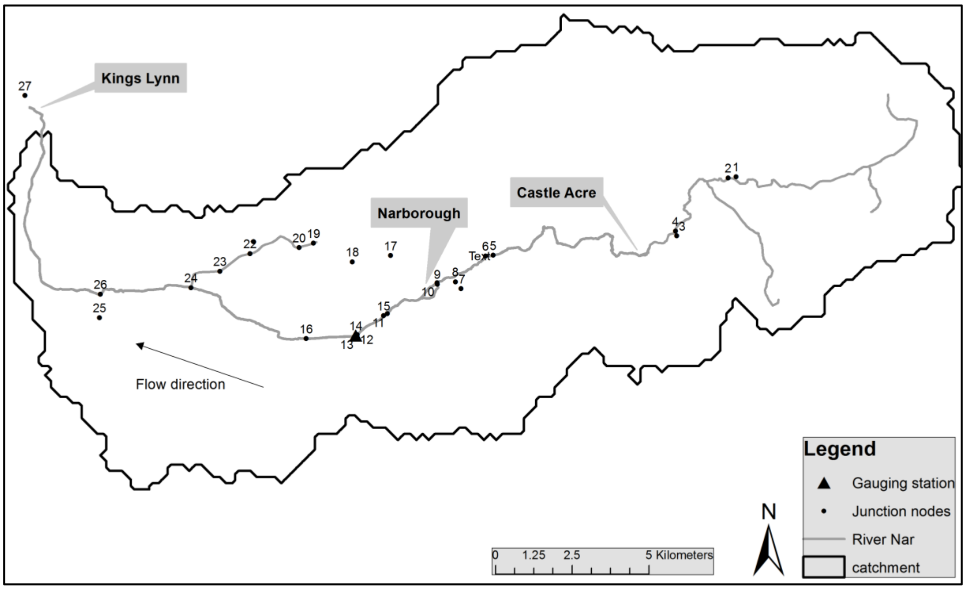


Figure 3.12- Trading node locations on the River Nar

Readers are advised to see Erfani et al.,(2014) for extensive information about how the model was developed and run, however a brief explanation of the model is given below:

- For each node at each time step, economic benefit functions that quantify economic gains are provided. The maximum object function is the sum of economic benefits from water use across all users in each individual time step, this object function identifies trades that make sense economically whilst adhering to constraints (i.e. HOF, see section 3.4.2).
- Trades are driven by economic demand curves that represent each abstractor's water demand, which is time varying.
- The model is used to simulate short-term (spot market) trading amongst individual water rights holders.
- A single-objective function means the model implements those trades which maximise regional economic benefits at each time step.
- The model assumes users with higher willingness to pay for water will buy from abstractors with lower marginal benefits if transaction costs do not discourage it.
- The individual preferences of specific abstractors to trade or not trade with other users are accounted for through detailed user-to-user transaction costs or rules imposed as constraints in the mathematical program (more details in Erfani et al., (2014).
- Most abstractors water use is not fully consumptive so some water is returned to the river as return flow. The sum, of volumes of water abstracted and sold cannot exceed their annual and weekly license allocations (Erfani et al., 2014; Erfani et al., 2015).

3.4.2 Trading model background (theoretical aspects)

A 32 year period from 1980-2011 was modelled in order to capture a variety of historical hydrological regimes from hydrological drought to higher flow periods.

The Hands- off- Flow (HOF) is a legal requirement for all abstractors and an important component in water trading, providing a limit to abstraction in periods of low flow (see Section 2.2). The Anglian Water (AW) abstraction has a HOF of 0.05m^3/s, meaning that when flows reduce to this level, abstraction must cease. Each abstractor on the river has its own HOF which is considered in the model. For example, abstractor G (Figure 3.11) cannot abstract if flows at Marham are less than or equal to 0.3m^3/s, whereas abstractor D has a much higher limit where they cannot abstract if the flow at Marham is less than or equal to 1.07m^3/s.

The model required weekly inflow data over 52 weeks in a given year. Week 1 always started on the 1st of January and years with more than 52 weeks were assumed to have 52 weeks only. The water trading model was simulated at a weekly time-step, while output for the habitat model required daily data. Consequently the work employed a systematic weighted disaggregation process shown in Equation 3.2:

$$\frac{Model\ weekly\ avearge\ (ml/w)}{Marham\ weekly\ average\ (m^3/s)} \times flow\ at\ Marham = Daily\ model\ flow\ (m^3/s) \quad \text{Eq3.2}$$

Outputs from three different trading scenarios from the water trading model were used:

- Scenario 1 (S1)- No trading with Hands off flow (HOF)
- Scenario 2 (S2)- Trading with HOF
- Scenario 3 (S3)- Trading without HOF

S1 is the baseline flow showing historical gauged flows and all results are compared to this baseline scenario. S1 is how the flows have been historically with no modifications. S2 and S3 represent the two trading scenarios, the model predicts the same trades in S2 and S3, with HOF flows being triggered in S2. This meant 5 trades occurred in S2 and 7 in S3 where there were no HOF restricting trades. The three different flow time series (S1, S2, S3) were used to drive the habitat models.

In S1 and S2 where HOF is applied, there are very few occurrences of HOF being activated. Between 1980 and 2011 a total of 13 weeks had HOF activation (1990= 3 weeks, 1991= 7 weeks, 1992= 1 week and 1996= 2 weeks). Interestingly, the influence of trading measures did not affect the HOF activation occurrences.

Once output flows from 1980-2011 were determined from the different scenarios in the trading model, these adapted flows were then run through the habitat models to show their effect on the indicator species.

The trading which occurred in these models were only downstream trades, which means abstracting water downstream as opposed to upstream thereby abstracting water where there is more available. Whilst this does still reduce flows locally, the impact should not be as significant as abstracting upstream.

3.4.3 Sites used

The same sites as used in RQ2 (Figure 3.10) were used however upon investigating initial flow results, site 3 (Castle Acre), which is the most upstream site resulted in very small and insignificant flow differences (maximum 0.06m^3/s). This is related to no significant trades happening and small flow values at this upstream site. Therefore the habitat results would not change at this site in any significant way and so this site was not used in analysis.

3.4.4 Analysis 1- Data distribution

This analysis aimed to determine the statistical differences between the distributions of HHS data in each of the trading scenarios. Summary statistics of the habitat distributions under the trading scenarios were calculated indicating how the scenarios affected overall habitat availability. These included: average, median, maximum, minimum, 95%ile, 50%ile, 5%ile, Standard deviation, Skew and Kurt values. Box plots were also determined for each species and trading scenario in order to visualise the results. Finally Mann- Whitney tests were used to determine if there was statistically different ($p<0.05$) habitat assemblages under the three trading scenarios. These statistical properties on the distributions enabled analysis into how the trading scenarios affect the habitat availability distribution of each species.

3.4.5 Analysis 2- Extreme years

Using the spatial distribution areas of SI, habitat availability for all species were compared for the wettest (2001), driest (1991) and average (1986) year recorded. These were determined based on the highest number of days in each year at below and above Q_{90} and Q_{10} respectively. The average year was determined as most days closest to the Q_{50}. For each of these years the percentage of area at each suitability scale was determined for each trading scenario. Mann- Whitney statistical tests were conducted to compare each year. This showed if there was a significant change in habitat availability ($p<0.05$), during different conditions.

3.4.6 Analysis 3- Synthetic flows

Changes in climate and land use impact on the hydro and biosphere at different spatial scales, this subsequently impacts on the hydrological processes within a catchment thus affecting hydraulics and habitat conditions (Guse et al., 2015). The purpose of this analysis therefore was to investigate if the variations in habitat availability caused by trading were within the limits of the natural flow variation. Thus indicating whether the uncertainty posed by climate change and changing hydrological regimes would affect natural habitat variations (Ledger and Milner 2015). The gauged flow data from Marham shows only one realisation of the flow regime, and therefore cannot accurately be used to predict how habitat availability varies in the future, as the past is not necessarily an accurate reflection of the future.

The ability to generate realistic daily streamflow sequences to help predict future flows taking into account these uncertainties has been shown to be useful for management decisions (Pender et al., 2015). In order to estimate realistic daily stream flow timeseries, stochastic modelling is often used to generate synthetic flow sequences. To generate synthetic flows for this analysis, a method determined by Pender et. al., (2015) was used to create 100 synthetic flows over a 32 year time series. This method used a combination of the Hidden-Markov Approach (HMM) and the generalised Parento approach (GP), this combination allowed a comprehensive estimation of flood flows therefore allowing the model to adequately encompass all flow conditions (Pender et al., 2015). These 100 time series have the same statistical attributes to the natural monitored flows at the site.

20 of these synthetic flows (determined from the 100 synthetic flows using random number generator in Excel) were then run through the habitat models. This determined the distributions of HHS habitat data in each of the 20 time series. The trading HHS predictions compared to these distributions to investigate if trading affects habitat distributions out with natural variation.

Chapter 4- Model development

4.1 Chapter introduction

This chapter sets out how the hydraulic and habitat models were built and calibrated for use in research question 2 and 3. Firstly details of how the 1D and 2D hydraulic models were built are described. Data requirements for the habitat models are detailed following on from this with an extensive review and description of habitat suitability data. Details of calibration and validation of the habitat suitability data are also provided. Justification of the software used is given in section 2.8.10.

Three sites were used to represent the river (see Section 3.3.1 for further details):

Site 1- Highbridge: Length of site= 990m, modelled in 2D

Site 2- DS Nar: Length of site= 827m, modelled in 1D and 2D

Site 3- Castle Acre: Length of site= 512m modelled in 1D

4.2 1D hydraulic model build

Hydraulic models were required to determine water levels at different flows for input to the habitat models.

Upstream model:

For the 1D sites (site 2 and 3) a Flood Modeller hydraulic model was built using cross section data collected in May 2013. The extent of the 1D hydraulic model covered the whole range of these two sections (Figure 4.1- upstream model).

Downstream model:

For the river below Marham (downstream model), a 1D Flood Modeller hydraulic model was provided by the EA. The resulting water levels from this model were used as calibration for the 2D model for Site 1 (Figure 4.1- downstream model).

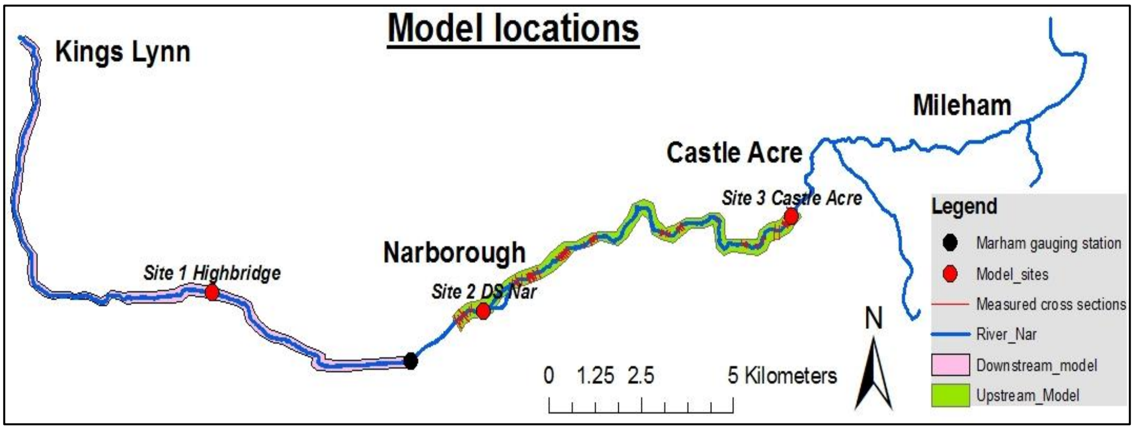


Figure 4.1- 1D model and site locations on the River Nar

4.2.1 Upstream model data requirements- Cross sections

The main data requirement for the upstream 1D hydraulic model was cross sections of the channel geometry. These were collected, measured and manipulated using the following process:

1) **<u>River surveyor (RS)</u>**

- Using an M9 River Surveyor (RS) provided by Sontek/ Xylem inc., 66 cross sections were measured between Castle Acre and Narborough (Figure 4.1) to determine location, bathymetry, velocity and flow rates. The RS collected in stream data only, river banks and floodplains were incorporated using LiDAR data.
- It was intended to take measurements approximately every 100m to ensure consistency and quality of data. However the equipment and time of year provided limitations to this. The RS was unable to measure accurate location coordinates if there were overhead trees or if there were depths less than 0.3m.
- The RS used laser beams to measure the bathymetry. Therefore as few macrophytes and large substrate (i.e. boulders) as possible was preferable in order to capture the bed of the river. Due to the time of year the survey was undertaken (spring 2013), there was a relatively large abundance of macrophytes, therefore this became problematic.
- 3 of the 66 cross sections were deemed unsuitable after recording had taken place where it was clear macrophyte elevation had been collected rather than the bathymetry.
- 4 transects were taken at each cross section as recommended by Sontek, to ensure the most accurate reading is taken. The one transect to use for the final cross section was determined based on accuracy of GPS data and the transect which picked up as few macrophytes as possible.

2) **<u>GIS and data manipulation</u>**

- The XY coordinate data from the cross section locations were imported to ArcMap and a line was drawn across the cross section including the land.
- LiDAR was available from the EA geometrics group to a 1m resolution, recorded in 2012. This was used to create elevation (Z) points which were connected to the XY data to create the full cross section coordinates.
- UTM (Universal Transverse Mercator) data coordinates were transformed to BNG (British National Grid) coordinates using Franson conversion software.
- Bank and channel locations were noted on the LiDAR data and the XYZ data from the RS was inserted into the channel section.

- During this process 15 out of the 64 cross sections were discarded due to insufficient or inaccurate data.

3) **Cross section interpolation**

- Further cross sections were interpolated in order to account for meanders. Locations of desired cross sections were drawn in ArcMap over LiDAR data and XYZ data was exported.
- Using the software HEC-RAS, interpolation was carried out at the distances apart from the original cross sections. These Z points were finally incorporated into the LiDAR lines.
- At each of the two 1D model sites, extensive interpolation occurred in this way to ensure the morphology of the river section was captured.

4.2.2 Calibration and validation

Flows and corresponding water levels were available for 49 cross sections using RS data. Due to the large number of different flow values, the measured flow values were grouped to the nearest 0.05m^3/s for calibration purposes, 18 flows ranging from 0.1m^3/s to 1.3m^3/s (Table 4.1) were used. It was not necessary to use this full range of data for calibration purposes, therefore some flow values were taken out, namely, 0.2, 0.5, 0.65, 0.7, 0.8, 0.9 and 1.15m^3/s, leaving 12 flows to calibrate with.

Calibration was carried out using manning's n of 0.035, 0.04 and 0.05 as the river fitted these n values according to Chow (1959). Figure 4.2 shows the results of these changes; n=0.05 gave the closest results and was therefore used. The upper reaches of the river however highlighted some issues where there were large differences from the measured values. During fieldwork these areas were investigated and it was discovered at cross sections 3600 and 4700 there were features which could potentially cause a hydraulic energy loss.

- At cross section 3600 there was a large surface water abstraction point along with groundwater abstractions.
- At cross section 4700 there was a bridge which narrowed the river and a pipe inlet/outlet from a sewage works.

Therefore at cross sections 3600 and 4700, the manning's n was increased to 0.5 to allow for this energy loss. Bridges and other structures were not incorporated due to time limitations and result requirements. Figure 4.3 and table 4.2 demonstrate the final calibration results.

Table 4.1- Calibration flows and related cross sections (measured on: 5th, 6th and 7th May 2013)

0.1 m^3/s			**0.2 m^3/s**			**0.4 m^3/s**			**0.45 m^3/s**			**0.5 m^3/s**		
XS number	**Q**	**Measured stage (m)**	**XS number**	**Q**	**Measured stage (m)**	**XS number**	**Q**	**Measured stage (m)**	**XS number**	**Q**	**Measured stage (m)**	**XS number**	**Q**	**Measured stage (m)**
5200	0.12	28.30	4800	0.19	27.77	5400	0.41	28.68	5300	0.43	28.13	3900	0.52	25.34
						4700	0.40	27.88	5000	0.47	27.85			
									4900	0.44	27.92			

0.55 m^3/s			**0.6 m^3/s**			**0.65 m^3/s**			**0.7 m^3/s**			**0.75 m^3/s**		
XS number	**Q**	**Measured stage (m)**	**XS number**	**Q**	**Measured stage (m)**	**XS number**	**Q**	**Measured stage (m)**	**XS number**	**Q**	**Measured stage (m)**	**XS number**	**Q**	**Measured stage (m)**
4600	0.57	26.51	4400	0.58	26.07	4200	0.65	25.72	2600	0.70	15.36	3500	0.75	22.43
4500	0.54	26.94	4100	0.60	25.67	4000	0.64	25.59	2400	0.71	14.50	3400	0.74	22.29
						3800	0.66	25.23				3000	0.75	20.33
												2300	0.74	14.80
												2200	0.76	14.75

0.8 m^3/s			**0.85 m^3/s**			**0.9 m^3/s**			**0.95 m^3/s**			**1 m^3/s**		
XS number	**Q**	**Measured stage (m)**	**XS number**	**Q**	**Measured stage (m)**	**XS number**	**Q**	**Measured stage (m)**	**XS number**	**Q**	**Measured stage (m)**	**XS number**	**Q**	**Measured stage (m)**
2900	0.81	19.30	3200	0.85	21.92	3100	0.88	20.39	2000	0.93	14.42	1900	1.01	12.80
2800	0.81	19.02	2500	0.84	14.91				1800	0.97	12.70	1300	1.08	10.73
1500	0.79	12.63	2100	0.85	14.54				1700	0.93	12.38	800	1.05	10.37

1.15 m^3/s			**1.2 m^3/s**			**1.25 m^3/s**			**1.3 m^3/s**		
XS number	**Q**	**Measured stage (m)**	**XS number**	**Q**	**Measured stage (m)**	**XS number**	**Q**	**Measured stage (m)**	**XS number**	**Q**	**Measured stage (m)**
1400	1.15	10.96	900	1.20	10.68	1000	1.27	11.01	300	1.31	7.31
1200	1.16	10.87	700	1.18	10.36	200	1.25	7.09	200	1.25	7.09
			600	1.22	10.22				100	1.28	7.13
			500	1.18	6.99						

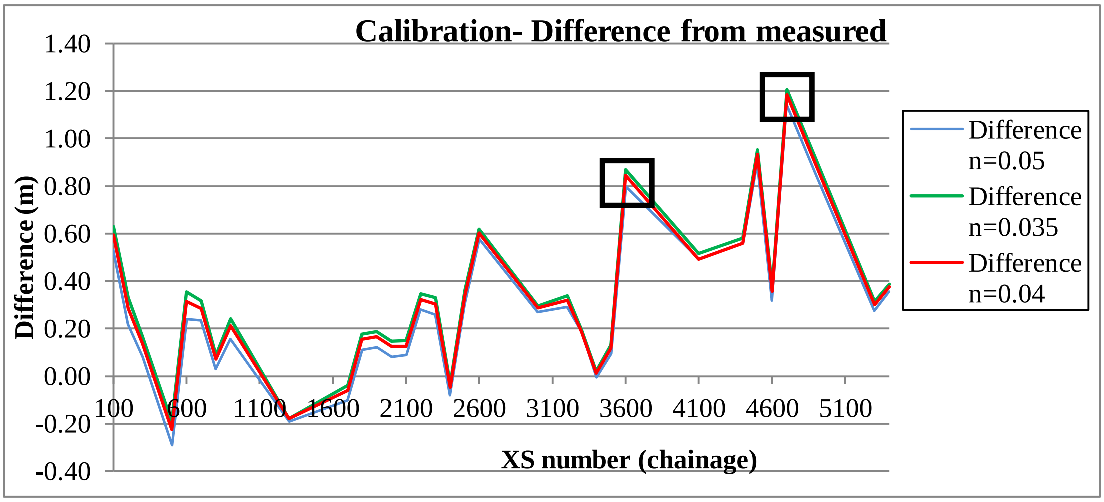


Figure 4.2- Calibration differences from measured for 1D model (0m= measured value)

Table 4.2- Final calibration for 1D model, XS= cross section, Red= Site 3, Green= Site 2

XS no	Measured Stage (m)	Model Stage (m)	Difference (m)
5400	28.68	28.32	0.36
5300	28.13	27.85	0.28
4900	27.92	27.68	0.25
4700	27.88	27.22	0.66
4600	26.51	26.19	0.32
4500	26.94	26.04	0.9
4400	26.07	25.55	0.53
4100	25.67	25.22	0.45
3600	23.75	23.31	0.45
3500	22.43	22.34	0.09
3400	22.29	22.3	-0.01
3300	22.18	21.99	0.19
3200	21.92	21.63	0.29
3000	20.33	20.06	0.27
2600	15.36	14.78	0.58
2500	14.91	14.61	0.3
2400	14.5	14.58	-0.08
2300	14.8	14.54	0.26
2200	14.75	14.47	0.28
2100	14.54	14.45	0.09
2000	14.42	14.34	0.08
1900	12.8	12.68	0.12
1800	12.7	12.59	0.11
1700	12.38	12.49	-0.1
1300	10.73	10.92	-0.19
900	10.68	10.53	0.16
800	10.37	10.34	0.03
700	10.36	10.13	0.24
600	10.22	9.98	0.24
500	6.99	7.29	-0.29
300	7.31	7.23	0.08
200	7.09	6.871	0.22
100	7.13	6.6	0.53

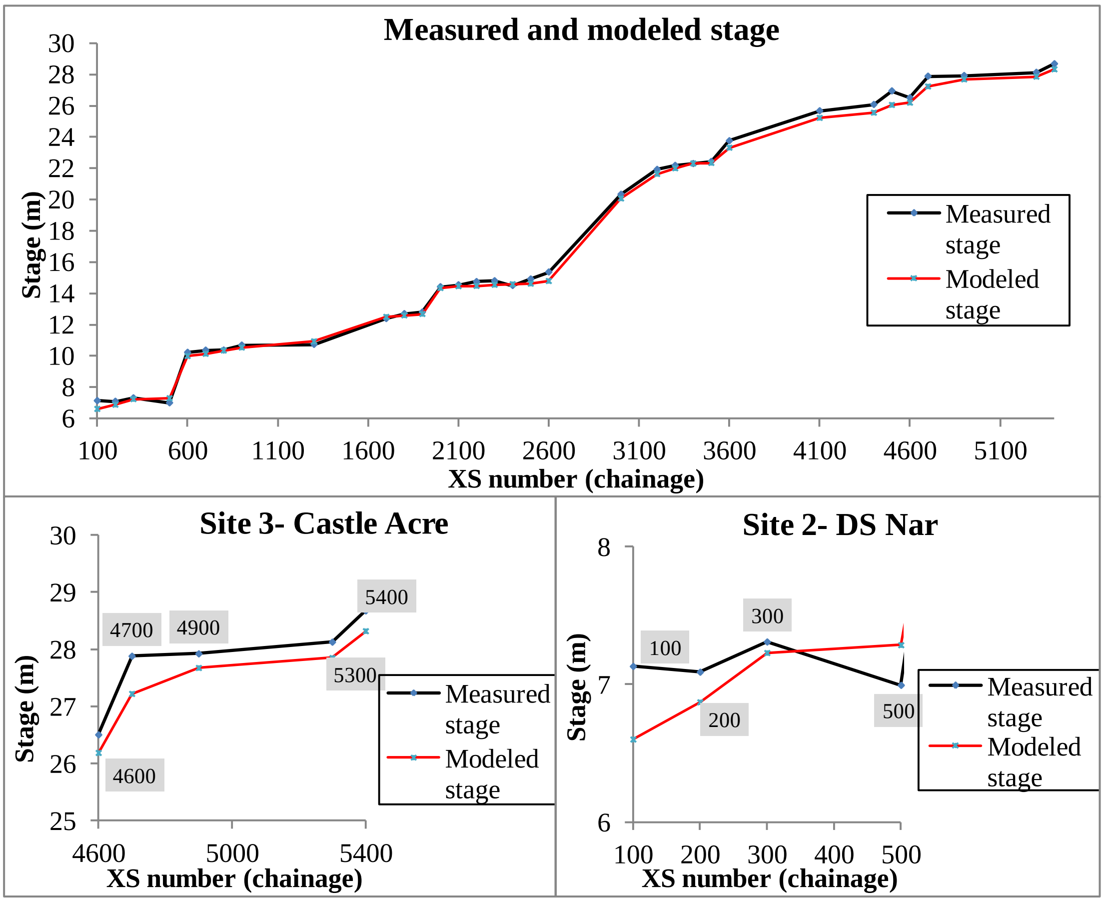


Figure 4.3- Final calibration of 1D model- whole model and individual sites

4.2.3 Inflows and output

The model was run in steady state with an unsteady timestep from the lowest recorded flow to the highest recorded flow in each section. Flows were area weighted from the Marham gauge (Section 3.1.2). Initial conditions were determined from pre-run snapshots at the point at which flows became steady state for each flow used. Resulting water levels at relevant cross sections for each site were recorded for input to the habitat models.

4.3 2D hydraulic model build

Sites 1 and 2 were modelled using the 2D hydraulic model TUFLOW. This provided changes in velocities across the cross sections at the sites which is required due to the larger width at the sites. The process of model build for each site is detailed below.

4.3.1 Site 2- DS Nar

Site 2 is the mid-stream site (~500m long) which was also modelled in 1D. The model was calibrated using the data collected by the RiverSurveyor in May 2013 (Table 4.1).

<u>Model setup</u>

The final model setup is shown in figure 4.4 and the five main GIS model components are described below:

- ***Grid***: four measured cross sections were used to create the grid, 20 further cross sections were interpolated using LIDAR data in GIS to account for meanders in the river section. The final grid was created from an extensive series of interpolated points between these 24 cross sections using HEC-RAS at a distance of 0.5m apart.
- ***Upstream boundary****:* one inflow was used for the whole model as no tributaries interact with the section. The upstream flow-time boundary therefore had only one attribute linking to inflow data (described in 'Inflow' section below).
- ***Downstream boundary****:* The downstream boundary was initially developed as a stage- flow (HQ) boundary specifying different downstream slopes for calibration. This however caused many warning messages that the water level at the HQ boundary had exceeded the top of the HQ boundary. Therefore a HQ boundary was defined using results from the 1D model. This resolved the warnings.
- ***Active area****:* the active area was determined at approximately 40-50m away from the main channel, therefore if flooding were to occur, glass walling (caused when flows interact with the outer extent of the active area) would not affect model results.
- ***Plot Output (PO) lines****:* five PO output lines were created to provide output results. 2 were placed at the exact location of the measured cross sections for calibration purposes.

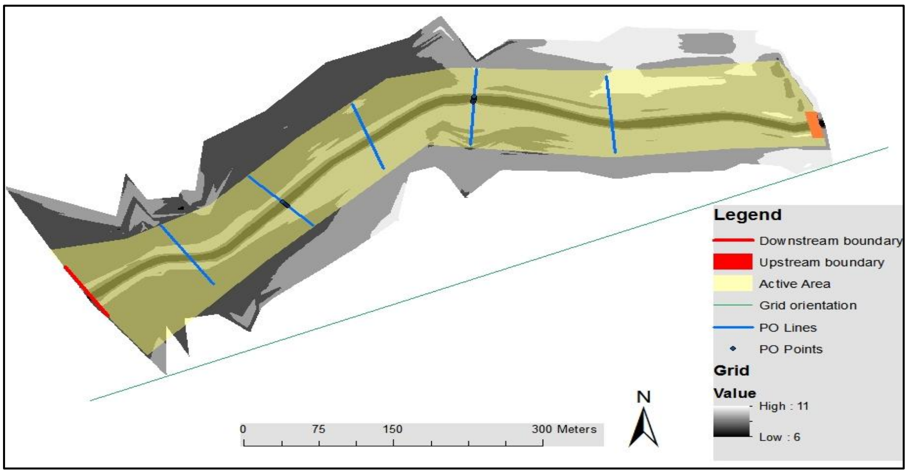


Figure 4.4- GIS model set up of site 2

<u>Calibration</u>

As the river is fairly weedy, calibration initially used a manning's n value of between 0.033 and 0.05 (Chow 1959). Changing the downstream boundary slope value had a significant impact upon results and therefore changes in slope value were also used for calibration. Different scenarios with varying mannings n values and downstream slope values were therefore used for calibration (Table 4.3). A manning's n of 0.05 with a downstream slope of 0.0025 (scenario 3) resulted in the closest water levels to the measured ones for both cross sections.

Table 4.3- Calibration for site 2. DS= downstream, WL= Water level

Scenario	Mannings n	DS slope	XS300 (PO2)- Measured WL=7.31	XS200 (PO4): Measured WL= 7.09
1	0.05	0.001	8.03	7.99
2	0.05	0.005	7.27	6.93
3	0.05	0.0025	7.30	7.10
4	0.05	0.00167	7.56	7.52
5	0.04	0.001	7.93	7.98
6	0.035	0.001	7.26	7.12
7	0.04	0.0025	7.23	6.91

Inflows and initial conditions

The whole series of recorded inflows at the model location were run through the model as steady flows with unsteady time conditions. Higher flows were also run for the synthetic flow analysis in RQ3. Specifically the flows were: 0.1, 0.2, 0.3, 0.4, 0.5, 0.6, 0.7, 0.8, 0.9, 1, 1.5, 2, 2.5, 3, 3.5, 4, 4.5, 5, 5.5, 6, 6.5, 7.5, 8.5 and 9.5m^3/s.

Initial conditions for each flow value were created using 'restart files', a pre-model was run to create these restart files. An example of the inflows for the restart file is shown in table 4.4, the restart file was written at 6 hours for each flow, allowing for the model to reach stable conditions. A timestep of 0.1 seconds was chosen for the run with a runtime of 10 hours in steady state. A grid size of 0.5m was used.

Table 4.4- Initial conditions inflow for site 2, flow of 0.4 m^3/s

Time (h)	Inflow (m^3/s)
0	0
0.5	0.1
1	0.2
1.5	0.3
2	0.4
2.5	0.4
3	0.4
3.5	0.4
7	0.4

Results and adaptation

Lower flows from 0.1 to 2m^3/s ran with no instability issues. Higher flows however from 2.5 to 9.5m^3/s had instability issues (Figure 4.5). Instability in hydraulic models can occur for a number of reasons such as inappropriate grid/cross section spacing, short computational timesteps, poor boundary conditions and stability weighting factors (Syme 2015). In this case the issues are related to the Courant number:

$$C_r = \frac{\Delta t\sqrt{2gH}}{\Delta x} \qquad \text{Eq 4.1}$$

Where:

$\Delta t = timestep$

$\Delta x = grid\ size$

g= acceleration due to gravity

H= depth of water (m)

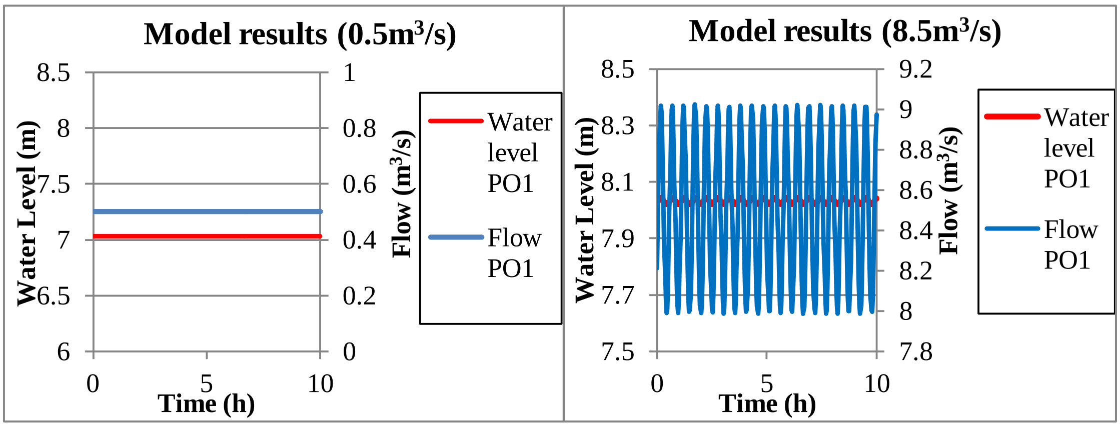


Figure 4.5- Instability issue with higher flows at site 2

TUFLOW uses an implicit scheme and therefore there is a target of C_r<5. If the Courant number is higher than 5, sensitivity tests should be run to determine an appropriate timestep and related grid size (Syme 2015). For the higher flows there are clearly higher water depths, therefore using a grid size of 0.5m and a timestep of 0.1seconds would cause the Courant number to be over 5 (equation 4.2). For this reason sensitivity runs were run to test various timesteps and grid sizes.

$$C_r = \frac{0.1\sqrt{2*9.81*1.43}}{0.1}$$

$$C_r = 5.29 \qquad \text{Eq 4.2}$$

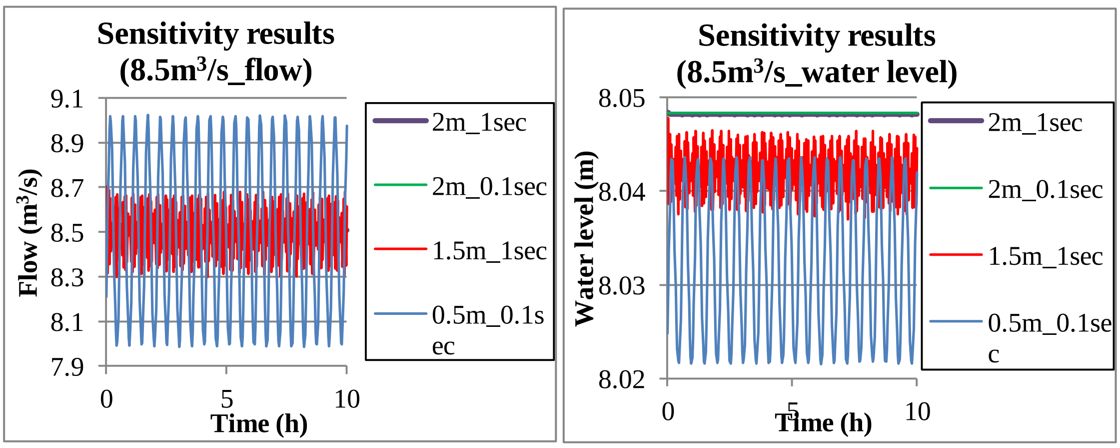


Figure 4.6- Sensitivity results for 8.5m³/s (flow and water level

The results from the sensitivity tests showed all scenarios apart from one to have both oscillating flows and water levels (Figure 4.6). Therefore for the higher flows, a grid size of 2m with a time step of 0.1 seconds was used which caused no oscillation in the results. The courant criteria had a number of 0.26 at this scenario and is therefore stable. The final grid sizes used are presented in table 4.5.

$$C_r = \frac{0.1\sqrt{2*9.81*1.43}}{2}$$

$$C_r = 0.26 \qquad \text{Eq 4.3}$$

Table 4.5- Final grid sizes used for site 2

Flow (m^3/s)	Grid size (m)
0.1	0.5
0.2	0.5
0.3	0.5
0.4	0.5
0.5	0.5
0.6	1.5
0.7	1.5
0.9	1.5
1	0.5
1.5	1.5

Flow (m^3/s)	Grid size (m)
2	1.5
3	2
3.5	2
4	2
4.5	2
5	2
5.5	2
6	1.5
6.5	1.5
7.5	2.5
8.5	2

4.3.2 Site 1- Highbridge

Site 1 is the most downstream site (1km in length) which was modelled in 2D due to the large width of the cross section (~8m) therefore having variations in velocity across the cross section. The model was calibrated using the Flood Modeller model provided by the EA.

Model setup

The final model setup is presented in figure 4.7 and the five main GIS model components are described below:

- ***Grid***: three measured cross sections were used in the creation of the grid, 13 further cross sections were interpolated using LIDAR data in GIS to account for meanders in the river section. The final grid was created from a series of interpolated points between these 14 cross sections using HEC-RAS at a distance of 0.5m apart.
- ***Upstream boundary:*** one inflow was used for the whole model as no tributaries interact with the section. The upstream flow-time boundary therefore had only one attribute linking to inflow data (see section on inflows).
- ***Downstream boundary:*** A downstream stage- flow (HQ) boundary was defined using output from the 1D Flood modeller model.

- ***Active area***: the active area was determined at approximately 40-50m away from the main channel, therefore if flooding were to occur; glass walling would not affect model results.
- ***Plot Output (PO) lines***: four PO output lines were created for output results. One was placed at the exact location of the measured cross section in the centre of the model for calibration purposes. The other two PO lines were placed at the upstream and downstream ends.

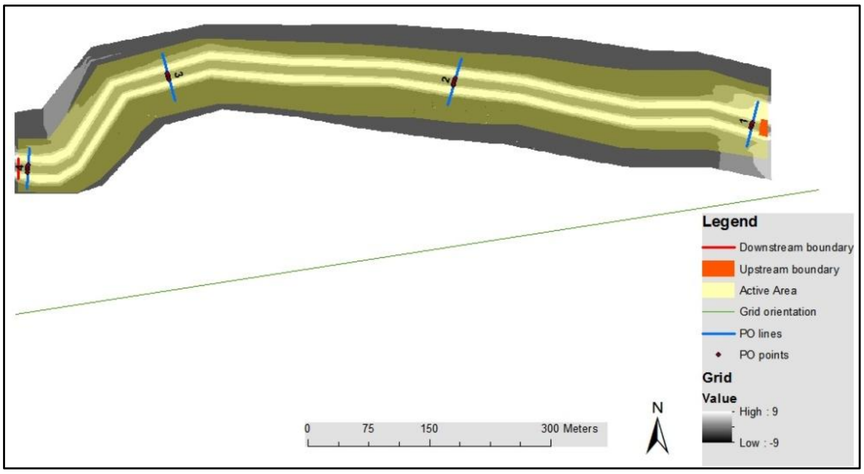


Figure 4.7- GIS model set up of site 1

Calibration

Flows of 0.5, 3.5 and 6.5m^3/s were used to calibrate low, average and high flows. This occurred at three cross sections where modelled data was available from the EA Flood modeller model.

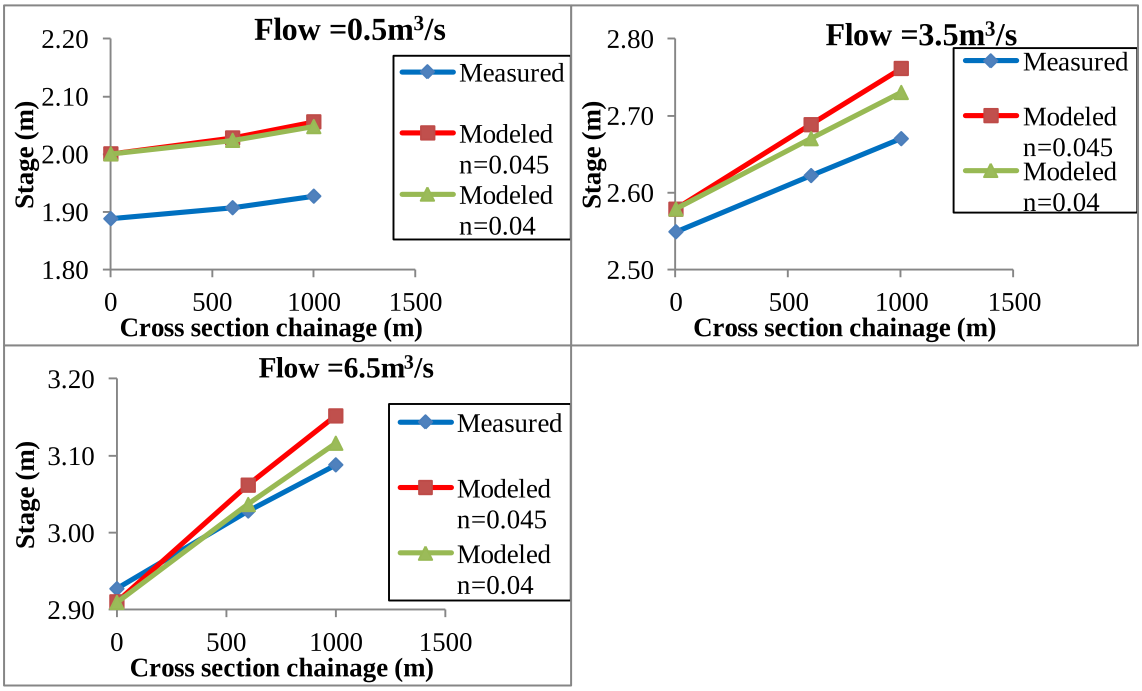


Figure 4.8- Calibration results for site 1

The initial mannings n used was 0.05 as this is what the 1D Flood modeller model specified, however this resulted in too high water levels so mannings n was reduced to 0.045 and 0.04. The calibration results are presented in figure 4.8. A final mannings n of 0.04 was chosen as this provided the closest results to the measured flows for each cross section and for each flow.

Inflows and initial conditions

The whole series of recorded inflows at the model location were run through the model as steady flows with unsteady time conditions. Higher flows were also run for the synthetic flow analysis in RQ3. Specifically these flows were: 0.14, 0.2, 0.4, 0.6, 0.8, 1.2, 1.4, 1.6, 1.8, 2, 2.5, 3, 3.5, 4, 5, 6, 7, 8, 9 and 10m^3/s. Initial conditions for each flow value were created using 'restart files', a pre-model was run to create these restart files. An example of the initial condition file for a flow of 0.4m^3/s is shown in table 4.4, the restart file was written at 6 hours for each flow, allowing for the model to reach stable conditions.

Results and adaptation

Runs at higher flows showed some instability due to the courant criteria and therefore the grid size was altered according to the flows, the final grid sizes used are shown in table 4.6.

Table 4.6- Final grid sizes used for site 1

Flow (m^3/s)	Grid size (m)
0.14	1
0.2	1
0.4	1
0.6	1.5
0.8	1
1.2	1
1.4	1.5
1.6	1.5
1.8	1.5
2	1.5

Flow (m^3/s)	Grid size (m)
2.5	2.5
3	1.5
3.5	1.5
4	2
5	2.5
6	2.5
7	2.5
8	3
9	3
10	2.5

4.4 1D habitat model build

The software CASiMiR-fish 1D was used for 1D habitat modelling of sites 2 and 3. The habitat model required geometry of the river bed and banks and water level data according to different flows, in this section, the development of these is firstly described. Secondly habitat models require habitat suitability data; a full explanation of these and the final data used is provided in sections 4.6 to 4.13.

4.4.1 Geometry data

CASiMiR-fish firstly required a .grd file, which is a geometry file of the modelled area; this consisted of cross sectional data, and cover and substrate across the cross sectional area. Each site was scoped during fieldwork in May 2014 to assess cover and substrate, (Appendix G). As cover can change seasonally, one month was chosen as input, and as the hydraulic models were calibrated using data from May 2013, the cover measured in spring was used for input. The whole cross section was included in analysis, giving non wetted areas a cover value of 0 (no cover) and a substrate value of 9 (rock) (Table 4.7).

4.4.2 Water surface profile data

The second input required for CASiMiR was a .wsp file, which is the water surface profile. The .wsp was determined from the output from the 1D hydraulic model. The whole range of recorded flows was included for each section (see section 4.2.3).

4.5 2D habitat model build

The habitat model used for the 2D analysis was Casimir-fish-2D. The input required was water surface profiles which were determined from output from the 2D TUFLOW models. The geometry data set was determined from the TUFLOW output grid (.2dm) and was transferred into a readable file in CASiMiR known as a SRNET file.

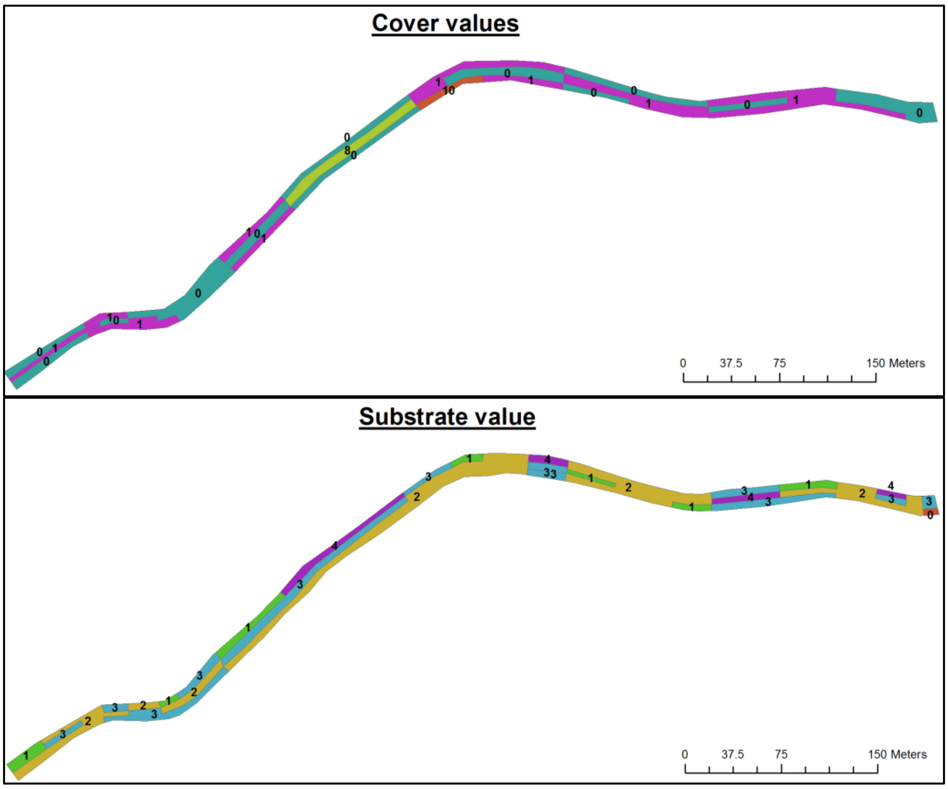


Figure 4.9- Substrate and cover values for 2D model at site 2

Substrate and cover parameters were created in a different way to the 1D model. For the 2D model, GIS polygons were created to assign different cover and substrate values to each individual section within the area. This could never be identical to the 1D inputs as it accounts for spatial changes rather than interpolation between cross sections, thus slight differences in results between 1D and 2D are likely. Efforts were made to ensure the values were as close as possible to the 1D input to ensure consistency. Cover and substrate polygons for site 2 are demonstrated in figure 4.9, this is not a direct example of the real world due to the 'blocky' nature of it, however as these are 500m-1km stretches; it would be extremely time consuming to record extensively detailed substrate values. Furthermore in the scale of habitat modelling this would not affect the results to any significant level.

4.6 Habitat suitability data introduction

Habitat Suitability Curves (HSC) and fuzzy logic rules are the two means of inputting habitat suitability data (see section 2.7). In this study both input variables were used to enable comparison between the two and to facilitate a choice in the most appropriate model based on quality of the available data and expert knowledge. Using both input methods also provides a means for investigating the sensitivity of the input methods in order to indicate how useful models are in assessing flow regime change on species. The same HSC and fuzzy rules were used for both 1D and 2D modelling. Generally fuzzy rules and HSC are determined in one of two ways:

1) Site specific suitability i.e. determined through electro-fishing
2) Data driven methods i.e. from literature or using expert knowledge

In this study, method 2 was used, basing habitat suitability on a combination of literature findings and expert knowledge, using data available for specific calibration/ validation to the River Nar.

This section is set out as follows: firstly an introduction to HSC, fuzzy rules and fuzzy sets followed by the calibration and validation methods. The fuzzy sets for brown trout (*Salmo Trutta*) are explained subsequently, followed by the final derivation of the habitat data (HSC, fuzzy rules and fuzzy sets) for each species.

4.6.1 Introduction to Habitat Suitability Curves (HSC)

Depth, water velocity, substrate size and cover are generally considered the most important microhabitat variables in determining habitat selection (Louhi et al., 2008). Therefore these variables were considered in habitat suitability creation.

The final HSC's were determined based on literature on flow requirements for the indicator species. It is preferable to determine individual HSC for selected sites however it has also been proven that the HSC can be successfully transferred between rivers (see Section 2.7.3) (Maki-Petays et al., 1997). Therefore for this study, preferences determined for similar rivers in terms of flow and typology were used.

To determine the HSC from literature, a comprehensive literature review was carried out, noting where the study was undertaken and therefore how relevant it was. Data on velocity (m/s), depth (m), substrate (mm) and cover preferences were recorded using the Wentworth substrate scale (Bovee 1986) and cover specifications from CASiMiR (Table 4.7). An average of relevant and most appropriate studies was used for the HSC for the different lifestages and microhabitat parameters.

Table 4.7- Substrate and cover index used in CASiMiR

Substrate types	Index
Organic material, detritus	0
Silt, clay, loam	1
Sand < 2mm	2
Fine gravel 2-6mm	3
Medium gravel 6-20mm	4
Large gravel 2-6cm	5
Small stones 6-12cm	6
Large stones 12-20cm	7
Boulders > 20cm	8
Rock	9

Cover types	Notes	Index
No cover		0
Aquatic plants		1
Stones/detritus		2
Roots		3
Deadwood	LWD	4
Wet branches	Removed	5
Dry branches	Overhanging branches	6
Floating macrophytes		7
Turbulence		8
Undercut banks		9
Overhanging grass		10

4.6.2 Introduction to fuzzy rules

Fuzzy rules provide the rules which determine the final suitability index, they state for example when water depth is 'high', and flow velocity is 'medium' and substrate is 'large' the suitability for the species is 'large'. Any number of these parameters can be used and are developed per species and per lifestage.

Fuzzy rules for adult and juvenile brown trout, Crowfoot and Mayfly were provided by CASiMiR. These rules were developed for a similar sized river with similar average flows and had been assessed by experts. In order to calibrate these to the River Nar the following process occurred:

- Start with fuzzy rules provided by CASiMiR.
- Broaden these back out to more generic findings in literature, thereby eliminating any site specific data for the ones determined by CASiMiR.
- Narrow the findings back down to the River Nar by calibrating/validating based on habitat data from the River Nar (see section 4.6.4).

Generally though, rules were kept to the original where possible in order to capture the expert knowledge. Fuzzy rules for spawning brown trout were not provided by

CASiMiR and were therefore developed purely based on literature. Detailed information on how fuzzy rules were developed for each species is provided in the following sections.

4.6.3 Introduction to fuzzy sets

Fuzzy sets determine what classifies as for example a ‘high’ depth and a ‘medium’ substrate. These are described by membership functions, which indicate the membership degree for each variable. As the boundaries can be overlapping, a variable can partially belong to a fuzzy set and therefore have a membership degree to this set ranging from 0 to 1 (Mouton et al., 2007). Generally information regarding preferences for specific species can be found in literature. The fuzzy sets also change dependant on the river hydraulics, for example a ‘high’ depth downstream is likely to be higher than a ‘high’ depth upstream.

Fuzzy sets were provided by CASiMiR which corresponded to the fuzzy rules also provided. The same general trends were kept as in the original fuzzy sets provided by CASiMiR, however were slightly modified based firstly on literature findings: for example if the rule was depicting high depths as least preferred for adult fish, the depth at which adult fish were no longer found based on literature was chosen. Secondly modifications occurred based on the river conditions. As there are three different sites used for modelling (upstream, middle and downstream), a ‘high’ depth in one section could be different to a ‘high’ depth in relation to another section and so on. Therefore, fuzzy sets were adjusted based on the conditions at different sections of the river. The substrate fuzzy sets remained the same for each site however, as a large substrate i.e. boulders, is the same at all points of the river. Detailed information on how fuzzy rules were developed for each species is provided in the following sections.

4.6.4 Calibration/ validation

Once the literature findings had been corresponded to the fuzzy rules determined by CASiMiR, the final stage was to narrow these findings back to species found in the River Nar. For this, calibration used existing ecological data for the river. The terms calibration and validation are used interchangeably as often the data is not directly used to calibrate with per se, and instead is used to validate findings from literature.

The existing habitat data available for calibration is explained below, highlighting any issues with using the data. The corresponding EA or collected sites which corresponded to the model sites could be used for final calibration to assess if high abundances were found in certain areas, however all data from the entire length of the river could be used in the calibration process to determine habitat preferences. Using the species data from the river was however used with caution as the antecedent flow conditions and site conditions impact on the species abundance. This is a finding from the RQ1 results which was then implemented to habitat models (see section 5.3-5.7). Therefore the literature findings were predominantly used, using the species specific data to validate the findings with.

Brown trout

- EA Electro-fishing:
- Not separated for adult and juvenile lifestages therefore can only be based on fish size, this can be inaccurate due to various environmental variables affecting fish size. To determine lifestage, fish scales would have to be analysed (Schneider et al., 2000) which was not recorded in the electro fishing data.
- No spawning brown trout data was available.
- The only model site which had electro-fishing data for was site 1, however there were no brown trout found at this site. Site 1 and 2 did not correspond with any electro fishing sites.
- Electro-fishing sites only provide a snapshot of time, as fish are mobile these findings could be inaccurate or unrealistic.

Thus calibration for brown trout data was limited. The main limitation however was that the electro-fishing data was not derived into adult and juvenile lifestages, therefore the data could not be used. General findings of where most brown trout were found in terms of substrate and cover were however taken into account. Otherwise calibration was carried out based on literature and fish sightings in the river.

Crowfoot

- EA macrophyte data
- Limited dates collected.
- The only corresponding site was for Site 2.
- Collected macrophyte data
- Corresponding macrophyte sites for each model site.
- Can easily relate to substrate types.

The macrophyte data could be used, but caution was taken due to different times of the year naturally having different abundances due to growth patterns (Dawson 2002). Thus general findings of locations in the river i.e. found in mid-stream or only in gravel were taken into account.

<u>Mayfly</u>

- EA BMI data
- Lots of historical data available
- Sites 1 and 2 have corresponding EA benthos data
- Collected BMI data
- Sites 1 and 2 have corresponding collected benthos data

BMI data could be used for each site. At sites where Baetidae were found, the depth and velocity on the day of measurement were taken into account. Substrates and cover values were also used.

4.7 Fuzzy sets for brown trout (all lifestages)

This section describes the process used to determine the fuzzy sets for all lifestages of brown trout. As previously described, slight adaptations were made to the fuzzy sets based on the site conditions. The hydraulic models were used to determine what classifies as a 'high', 'medium' and 'low' depth and velocity.

4.7.1 Hydraulic model data

As only one fuzzy set was provided for fish species; all lifestages of fish use the same fuzzy sets but different fuzzy rules. The fuzzy sets were however slightly altered for each site due to different hydraulic conditions.

In order to adjust the fuzzy sets per site, firstly the flows classified a low, medium and high at each of the sites was determined. The site percentiles were determined by area weighting from the Marham gauge. Q_{90} was used as a low flow, Q_{50} as an average flow and Q_{10} as a high flow. The flows used are shown in table 4.8.

Table 4.8- Flows (m^3/s) used for each site based on percentiles

Percentile	Marham	Site 1	Site 2	Site 3
Q_{90}	0.47	0.53	0.43	0.25
Q_{50}	0.93	1.11	0.91	0.51
Q_{10}	2.02	2.25	1.85	1.06

The flows shown in table 4.8 were run through the 1D hydraulic models to determine what classified as a 'low', 'medium' and 'high' depth and velocity at each site. An average was taken from the relevant cross sections in the study sites, as these are only average values they were used as a guide only, and were slightly adapted to fit in with rules from the other sites. For example a slightly higher velocity could be taken if it is known the section below has lower velocities. Changes were only made within 0.05m (depth) or 0.05m/s (velocity). The results from the hydraulic model run are presented in table 4.9.

Table 4.9- Depth and velocity results for all sites at each percentile flow

	Depth (m)			Velocity (m/s)		
	High	Medium	Low	High	Medium	Low
Site 1	1.23	0.95	0.74	0.22	0.15	0.1
Site 2	0.82	0.61	0.45	0.33	0.26	0.19
Site 3	0.86	0.62	0.44	0.34	0.29	0.24

4.7.2 Site 1- Highbridge- Depth and Velocity

Hydraulic model results for site 1 had the highest values for depth of all three sites and the lowest velocities; this would be expected as it is the most downstream site and in the fen section of river i.e. deeper with lower velocities. Basing the initial sets on the ones provided by CASiMiR for brown trout, the fuzzy sets in figure 4.10 were determined:

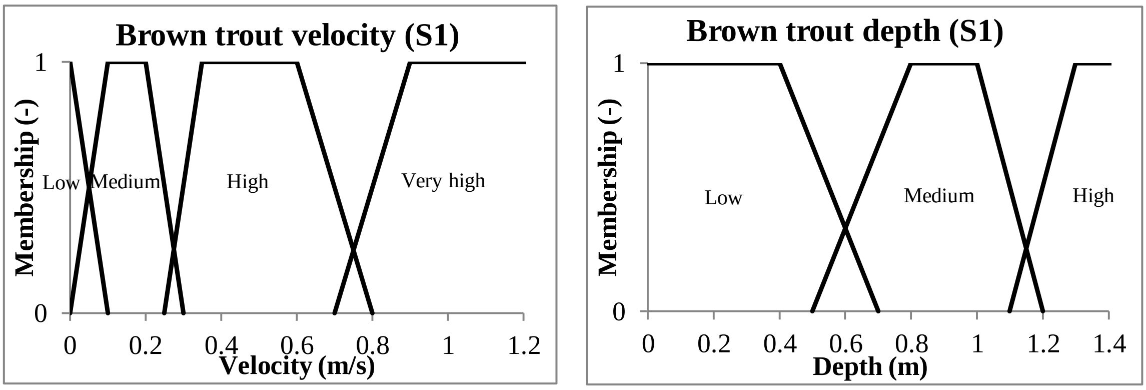


Figure 4.10- Depth and velocity fuzzy sets for site 1 for brown trout

For velocity, 'low' values are under 0.1m/s, 'medium' values are between 0m/s and 0.3m/s and 'high' values are between 0.25m/s and 0.8m/s. The 'high' values have a median of 0.15m/s as determined by the hydraulic model (Table 4.9). A 'very high' value is over 0.7m/s.

For depth, 'low' values are under 0.7m, 'medium' is 0.5m to 1.2m and a 'high' depth is above 1.1m. These fuzzy sets were based around the fuzzy sets provided by CASiMiR but also taking into account hydraulics at the site.

4.7.3 Site 2- DS Nar- Depth and Velocity

For velocity, 'low' values were classified below 0.2m/s, 'medium' between 0.1m/s and 0.55m/s, 'high' values are between 0.3m/s and 0.9m/s and 'very high' values are over 0.8m/s. For depth, 'low' values were specified under 0.45m, 'medium' values are between 0.25-0.95m (median 0.6m) and 'high' depths are over 0.8m. These remained to have the structure of those from Site 2, but were slightly adapted based on the site hydraulic conditions.

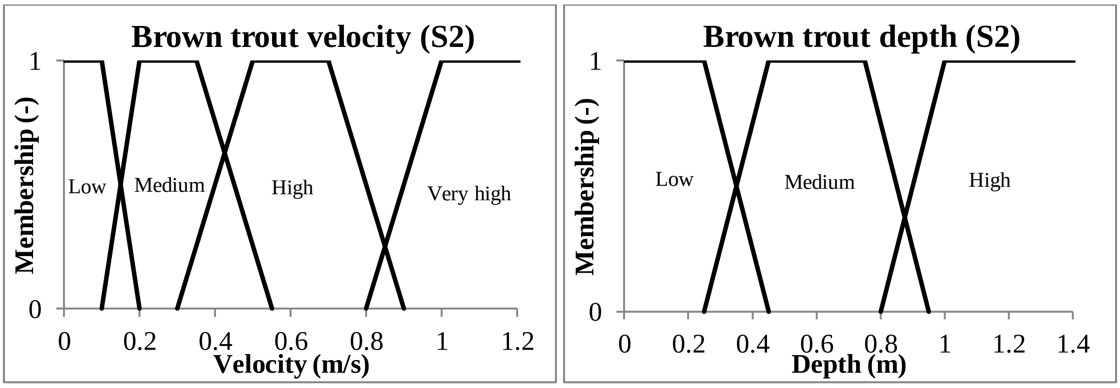


Figure 4.11- Depth and velocity fuzzy sets for site 2 for brown trout

4.7.4 Site 3- Castle Acre- Depth and Velocity

The velocity results were very similar to the velocity results for Site 2. However each variable is between 0.05m/s and 0.1m/s higher. This is expected as the most upstream site would have slightly higher velocities. The depth results were the same as Site 2 values apart from the high value being 0.05m/s higher. The river poses similar depth values here due to the fact that both sections are in the chalk stream reach. The fuzzy sets are shown in figure 4.12.

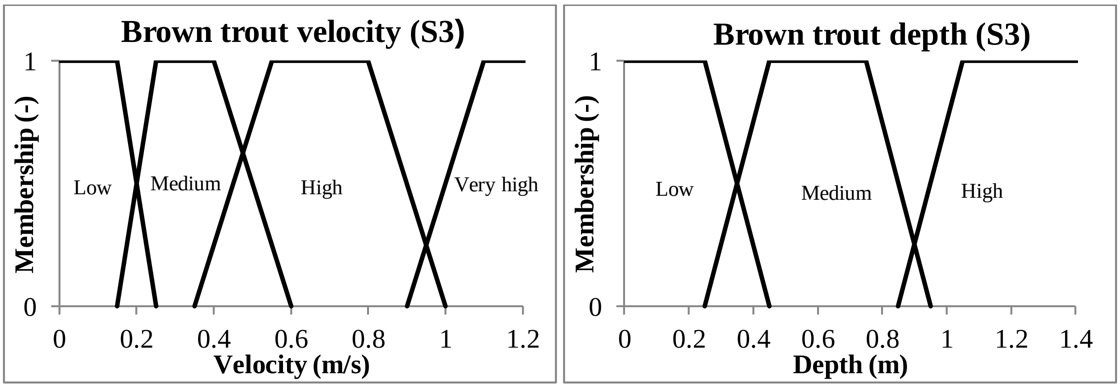


Figure 4.12- Depth and velocity fuzzy sets for site 3 for brown trout

4.7.5 Substrate fuzzy sets for all sites

A similar shape fuzzy set is used as the one CASiMiR provided, with a few alterations based on literature findings (Table 4.10, 4.14 and 4.17) and conditions in the river. It was intended to classify the smallest substrate smaller than ones CASiMiR provided, therefore an index of 6 was altered to an index of 3 for the ‘low’ substrate (See table 4.7 for index).

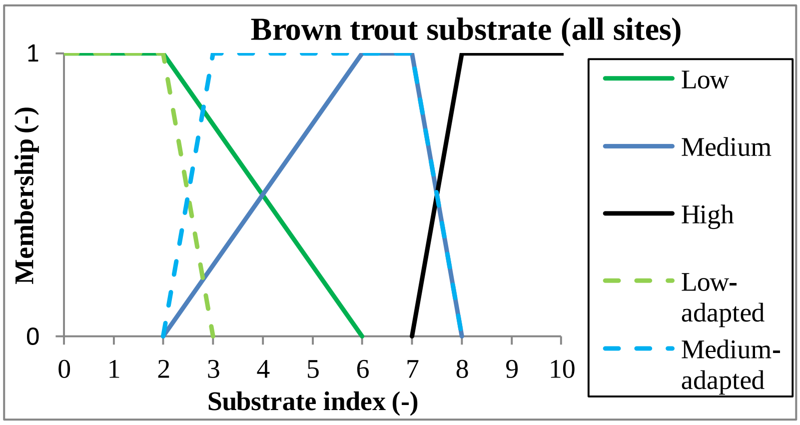


Figure 4.13- Substrate fuzzy sets for all sites for brown trout

4.7.6 Sensitivity analysis

Sensitivity analysis is not required by habitat models and many studies do not include it. Some researchers do however choose to carry out a sensitivity analysis depending on the nature of their study (e.g. Mouton et al., 2007; Ahmadi-nedushan et al., 2008; Hamilton et al., 2015). For example Mouton et al., (2007) was interested in showing how the substrate compared upstream to downstream, so a sensitivity analysis was run changing the lowest substrate class (silt) to the largest substrate class (boulders) to assess the impact. It was found that HHS only changed if substrate was suitable. If the hydraulic conditions were unfavourable for the species, substrate changes alone could account for a maximum increase of HHS of 16%.

For this study, a sensitivity analysis was run on the spawning brown trout fuzzy sets for site 3 by altering the factors around its nominal value to assess how the model responded. As much of the habitat suitability was developed as a result of literature searches, subjective views could be apparent, therefore the sensitivity test was important to carry out.

The original depth and velocity fuzzy sets derived in Section 4.7.2-4.7.4) were increased and decreased by 10% which is a standard approach taken from hydraulic modelling (Figure 4.14). The substrate values were not altered as they are not a linear scale, i.e. they are based on silt, gravel etc. Therefore cannot be increased by 10%.

The results of the sensitivity analysis (Figure 4.15) showed that changes did occur, which is expected. The most significant change was a 0.03 change to HHS which corresponds to a 7.5% change in habitat availability. This is therefore not a significant difference.

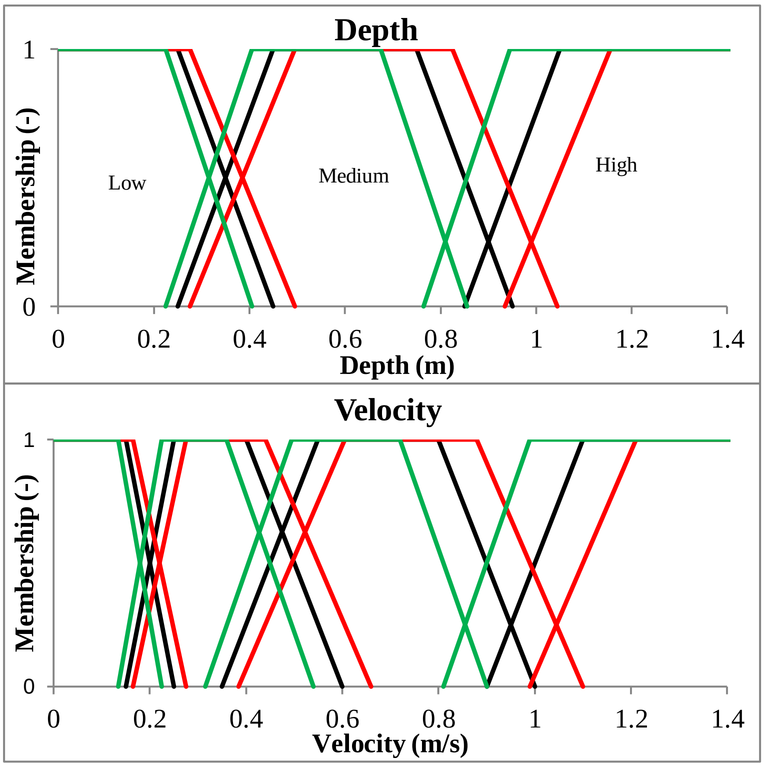


Figure 4.14- Fuzzy sets for sensitivity analysis (black line= original, red line= increase 10%, green line= decrease 10%)

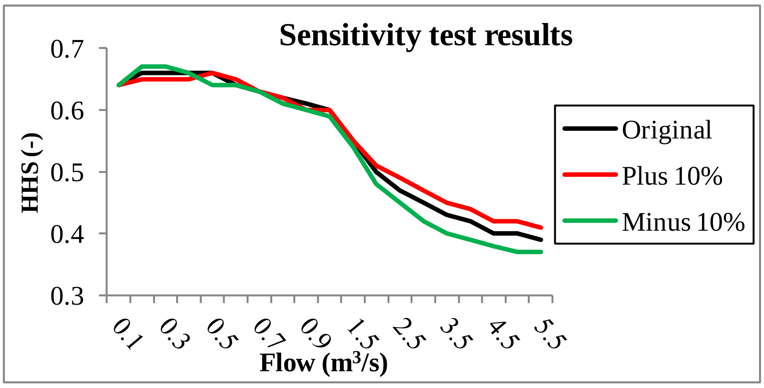


Figure 4.15- Results of sensitivity analysis

4.8 Spawning brown trout habitat suitability data (*Salmo Trutta*)

Many studies have been carried out on habitat requirements of spawning brown trout, for this thesis habitat suitability curves (HSC) and fuzzy rules were created from well-established literature (Table 4.10).

Many factors have been found to influence spawning locations other than the physical factors of depth, velocity, substrate and cover. Female fish size is one of these; the depth of egg burial and the velocity at the redd site has been found to be positively correlated to an increase in fish size (Crisp and Carling 1989). Furthermore competition for space is also a large factor in spawning location, likewise lower temperatures are preferred for spawning brown trout (Heggenes 2013). Despite this however, depth, velocity, substrate and cover are known to be the largest factors impacting upon spawning location (Armstrong et al., 2003). A review of literature on depth, velocity and substrate spawning preferences of brown trout is presented in table 4.10.

Table 4.10-Depth, velocity and substrate preferences for spawning brown trout from literature

	Reference	**Country/ notes**
Velocity (m/s)		
0.2- 0.55	(Louhi et al., 2008)	Combined
0.11-0.8 (mean= 0.46)	(Witzel and Maccrimmon 1983)	South western Ontario
0.15-0.75 (mean= 0.39)	(Shirvell and Dungey 1983)	6 rivers in New Zealand
0.02- 1.24 (mean= 0.47)	(Wollebaek et al., 2008)	Boreal rivers, Norway
0.3- 0.4	(Ottaway et al., 1981)	Teesdale, N England
Depth (m)		
0.15- 0.45	(Louhi et al., 2008)	Combined
0.06- 0.82 (mean= 0.31)	(Shirvell and Dungey 1983)	6 rivers in New Zealand
0.23- 2.15 (mean= 1.03)	(Wollebaek et al., 2008)	Boreal rivers, Norway
0.07-0.58 (mean=0.25)	(Witzel and Maccrimmon 1983)	South western Ontario
Substrate (mm)		
16- 64	(Louhi et al., 2008)	Combined
20- 370 (mean= 70)	(Wollebaek et al., 2008)	Boreal rivers, Norway
20-30	(Crisp and Carling 1989)	NE Eng, S Eng & SW
10.3- 112	(Ottaway et al., 1981)	Teesdale, N England
14	(Shirvell and Dungey 1983)	

4.8.1 HSC

Velocity and depth HSC

The range of velocities for spawning brown trout were predominantly 0.11 to 0.75m/s not including Wollebaek et al.,(2008) who discovered ranges of 0.02m/s to 1.24 m/s. Louhi et al., (2008) curve corresponded well with data from other studies and the mean values determined (0.39-0.47m/s) are all given a suitability of 0.9 or more. Louhi et al., (2008) created the most transferable HSC for spawning brown trout due to the comprehensive literature used based on 22 published articles and furthermore the study was done for small rivers with a discharge less than 10m^3/s on average.

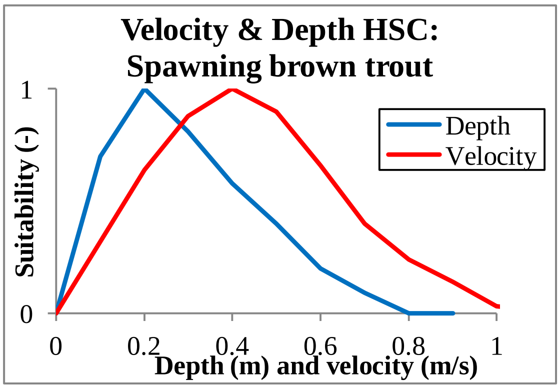


Figure 4.16- Velocity and depth HSC for spawning brown trout

Basing the main HSC results on Louhi et al., (2008), one alteration was made, 0m depth was altered to 0 suitability as realistically if there were no water depth, the brown trout could not spawn. 0.06m -0.82m was the range discovered in literature, with a mean of 0.31m. Wollebaek et al., (2008) discovered a large range of velocities from 0.23- to 2.15m, this is out of the scope of this curve however is due to the rivers used being of different typology. The final HSC is shown in figure 4.16.

Substrate HSC

According to Armstrong et al., (2003), substrate is the most important environmental factor for spawning brown trout, and the substrate type used can vary amongst rivers. Shirvell and Dungey (1983) noted how out of the three habitat criteria, (velocity, depth and substrate), substrate used was most consistent in all their six study rivers. Crisp and Carling (1989) determined that brown trout used gravel, sand and silt for spawning but mostly pebbles with a median grain size of 20-30mm. The main HSC was derived from Louhi et al.,(2008), these HSC were then altered to fit the CASiMiR substrate parameters (Table 4.7). The final HSC is shown in Figure 4.17.

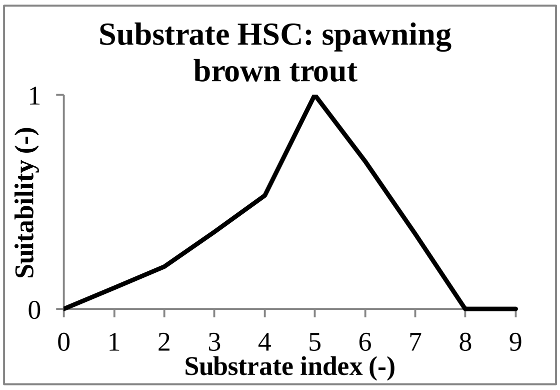


Figure 4.17- Velocity and depth HSC for spawning brown trout

4.8.2 Fuzzy rules

The fuzzy rules for spawning brown trout were created based on literature due to none being provided by CASiMiR. The fuzzy rules were determined by firstly overlaying the HSC (determined in Section 4.8.1) on the fuzzy sets (determined in Section 4.7), an example of this is shown in figure 4.18. The fuzzy sets used were for site 2 which provided the most average results (mid-stream site), one of the advantages of fuzzy logic it that it is not clear and crisp rules, and therefore by overlaying the HSC on the fuzzy sets it gives only an idea of preferences.

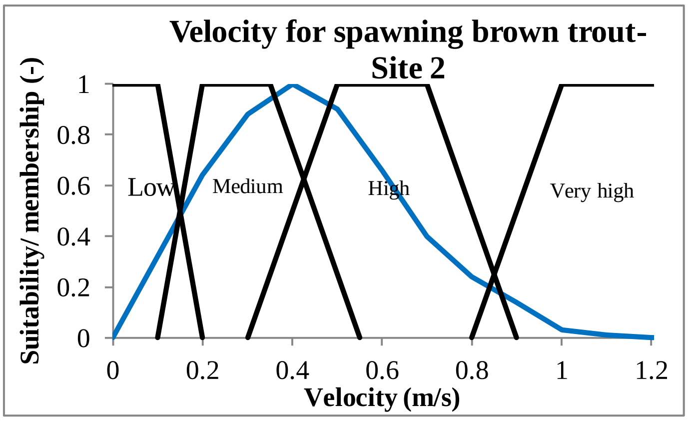


Figure 4.18- Velocity HSC and fuzzy sets for spawning brown trout to determine fuzzy rules

The fuzzy sets in figure 4.18 with HSC overlaid were interpreted as follows (example is for velocity at site 2):

- a low velocity has a low preference,
- a medium velocity has a medium preference
- a high velocity has a high preference
- a very high velocity has low preference

Table 4.11a was created from these statements for all variables in order to determine the SI values, and further to this the rules in table 4.11b were determined.

> **A worked example**, first column in fuzzy rules 1: A high velocity, high depth and high substrate. Using table 4.11 this gives one 'high' and two 'low' preferences which then using table 4.11b gives a Medium SI.

This was bias towards medium SI values as 4 out of the 10 combinations give medium values. It also does not account for any preferences. Therefore the following rules were applied based on literature:

Rule1- Increase if medium substrate

Rule 2- Decrease if high substrate

Rule 3- Decrease if high depth

Rule 4- Decrease if very high velocity

Rule 5- If rule 3 and 4 together then overrules rules 1 and 2

Final changes were made to 'Medium, Medium, Medium' and 'Medium, Low, Medium', these were reduced from 'Very high' SI to 'High' SI as it is more likely that these combinations provide 'high' availability based on literature findings. The final fuzzy rules are shown in table 4.11c.

Cover: Cover is regarded as important for spawning brown trout throughout literature. Armstrong et al., (2003) noted how instream cover such as logs and tree branches are of great importance. In a study by Witzel and Maccrimmon (1983) 84% of redds were recorded within 1.5m of cover. Despite these findings, data on cover used by spawning brown trout was deemed not viable enough to use. Furthermore the analysis of spawning brown trout (Section 3.3.7), aims to determine the interconnectedness of species incorporating food availability (benthos) and refugia (macrophyte cover), thus cover would be factored into spawning brown trout habitat availability. Therefore cover was not included as a habitat parameter in the HSC or fuzzy sets, assuming spawning brown trout have no preference either for or against cover.

Table 4.11 (a, b and c)- Fuzzy rules for spawning brown trout. V= velocity, D= depth, S= Substrate, SI= suitability, VH= very high, H= high, M= medium, L= low

Table 4.11a

	V	**D**	**S**
L	L	H	M
M	M	M	H
H	H	L	L
VH	L		

Table 4.11b

Combination	**Result**
3 L	L
3 M	H
3 H	VH
2 L, 1H	M
2 L, 1 M	L
2 M, 1 H	H
2 M, 1 L	M
2 H, 1 L	M
2 H, 1 M	H
1 H, 1 M, 1 L	M

Table 4.11c

V	**D**	**S**	**SI based on HSC**	**SI- HSC & pref**
H	H	H	M	L
H	H	M	M	M
H	H	L	M	L
H	M	H	M	L
H	M	M	H	VH
H	M	L	H	H
H	L	H	M	L
H	L	M	VH	VH
H	L	L	H	H
M	H	H	L	L
M	H	M	M	M
M	H	L	M	L
M	M	H	M	L
M	M	M	H	H
M	M	L	H	H
M	L	H	M	L
M	L	M	H	H
M	L	L	H	H
L	H	H	L	L
L	H	M	M	M
L	H	L	L	L
L	M	H	L	L
L	M	M	M	H
L	M	L	M	M
L	L	H	M	L
L	L	M	M	H
L	L	L	M	M
VH	H	H	L	L
VH	H	M	M	L
VH	H	L	L	L
VH	M	H	L	L
VH	M	M	M	M
VH	M	L	M	L
VH	L	H	M	L
VH	L	M	M	M
VH	L	L	M	L

4.9 Calibration/ validation data for brown trout (adult and juvenile)

The habitat data available for brown trout was not disaggregated into adult and juvenile lifestages, therefore general trends indicate general depth, velocity, substrate and cover preferences were picked up on for validation. The electro-fishing data from all sites were pooled together and plotted on a graph of the depths and velocities they were found in. Sites including ones where brown trout were not found were used, this is as the flow conditions at the site could be a reason why no brown trout were found there and therefore is important to include. The level classed as 'low', 'medium' and 'high' depth and velocity at site 2- DS Nar (mid stream and therefore has the most average conditions across the river) were then plotted on the graph to indicate where most/ least populations of brown trout were found (see section 4.7.1 for determining of depth and velocity categories). This is shown in figure 4.19 with table 4.12 presenting a summary of the findings. Table 4.13 then shows the substrate and cover calibration findings. Substrate and cover values at each site recorded in the field (May 2013) was combined with substrate and cover data provided with the electro-fishing to determine the average cover and substrate values at each site.

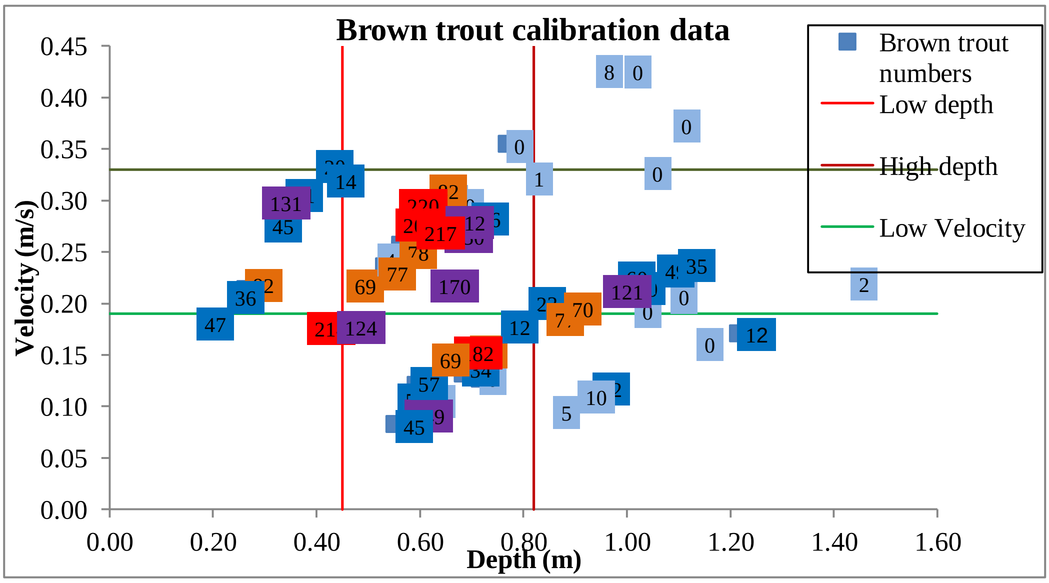


Figure 4.19- Brown trout depth and velocity calibration data

Table 4.12- Brown trout calibration data summary of results

Depth	Velocity	Total fish
Medium	Medium	1420
Medium	Low	664
Low	Low	385
Low	Medium	345
High	Medium	331
High	Low	186
Low	High	34
High	High	8
Medium	High	0

The following conclusions were drawn from the depth and velocity calibration results and were transferred into fuzzy rule and set determination:

- The vast majority of brown trout were found in medium depth and medium velocity.
- The least preferred conditions are high velocities, only 42 brown trout were found in high velocities.
- Low depths are more preferential than high depths

Table 4.13- Brown trout substrate and cover calibration data

	Dominant substrate	**In stream cover**	**Overhead Cover**	**Brown trout number**
Castle Acre	Medium/ large gravel	Some aquatic plants	None	1889
Manor farm	Medium gravel	Aquatic plants	Sparse	760
Narford Hall	Sand/fine gravel	None- some aquatic plants	Intermittent	379
Warren farm	Sand/ medium gravel	Aquatic plants	None	295
Marham intake	Silt/sand	None- some overhanging branches	None	41
Abbey Farm	Silt	None	None	8
D/s Setchey	Silt	Some overhanging	None	1
Highbridge	Silt	None	None	0

The following conclusions were drawn from the substrate calibration results and were transferred into fuzzy rule and set determination:

- Silt is disliked by brown trout
- The most preferential substrate is medium gravel
- Most fish were found where there was at least some aquatic vegetation.
- There was no preference or dislike of over head cover.

These findings were transferred into fuzzy rule and set determination

4.10 Juvenile brown trout habitat suitability data (*Salmo Trutta*)

Habitat usage by juvenile brown trout is the most varied of all lifestages. Many studies have been completed to determine habitat preferences of juvenile brown trout, using velocity, depth, substrate and cover as indices. Maki-Petays et al., (1997) noted how habitat selection by juvenile fish could also be affected by: fish size, time of day, mode of activity, food, competitors and predators, which the habitat models do not take account of. Table 4.14 gives a synthesis of findings.

Table 4.14- Depth, velocity and substrate preferences for juvenile brown trout from literature

	Reference	Country/ notes
Velocity (m/s)		
0-1.42 (mean 0.24)	(Heggenes 2002)	South West England
0.2-0.3	Maki-Petays et al., 1997)	Northern Finland, boreal
0.2-0.5	(Heggenes 1996)	Combination
0.33 (mean)	(Johnson and Douglass	Winter preference, New
Depth (m)		
0.55-0.65	(Maki-Petays et al., 1997)	Northern Finland, boreal
0.9-3.05 (mean	(Heggenes 2002)	South West England
0.2-0.3	(Heggenes 1996)	Combination
0.35-0.5	(Ayllon et al., 2009)	Northern Spain
0.31	(Johnson and Douglass	Winter preference, New
0.3-0.6	(Heggenes 1988b)	Combination
Substrate (mm)		
128-256	(Heggenes 2002)	South West England
32-64	(Maki-Petays et al., 1997)	Northern Finland, boreal
50-70	(Heggenes 1988c)	Artificial stream
Cover		
80-100% aquatic	(Maki-Petays et al., 1997)	Northern Finland, boreal
15%	(Johnson and Douglass	Winter preference, New

4.10.1 HSC

Velocity and depth HSC

The larger the fish the greater the depth they use for their habitat but the lower velocities they use (Heggenes 1996). Furthermore older and larger brown trout colonize all types of environment providing security by depth and cover (Heggenes 1988b). Three papers have defined HSC for juvenile brown trout; these curves are shown in figure 4.20 along with one determined using the requirements found in table 4.14.

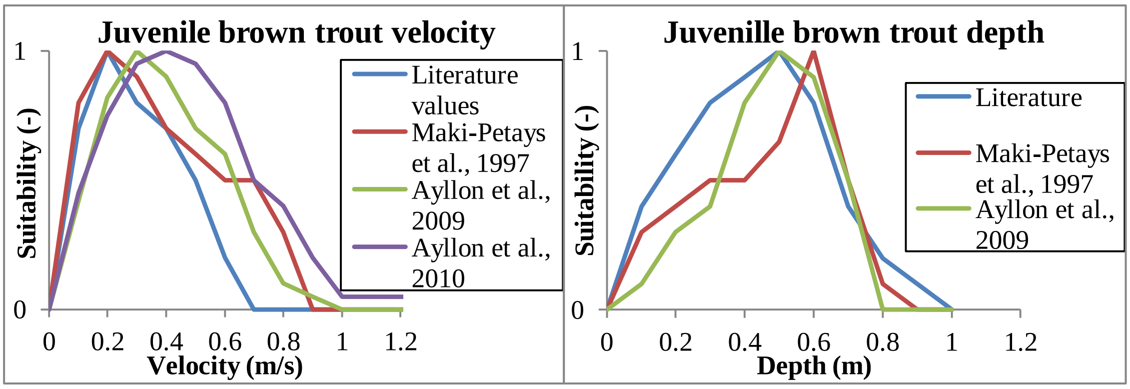


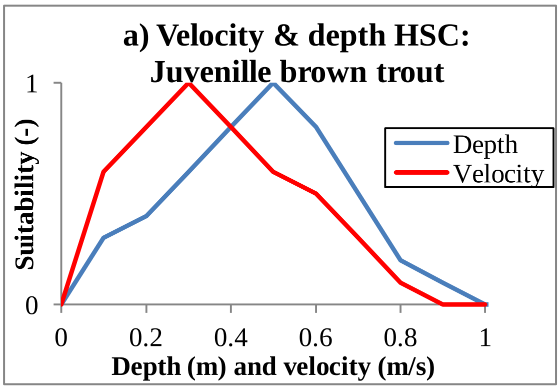
Figure 4.20- Velocity and depth HSC's for juvenile brown trout from literature

Ayllon et al.,(2010) had the most differing results from other studies, this can be attributed to the nature of the study which determined HSC for 7 different streams with specific stream attributes. Results were taken from the most similar stream to the case study in terms of flows and geometry however this curve was discounted due to the dissimilarity to the general curves. Ayllon et al., (2009) presented results from a fast flowing stream and a slow flowing stream. The slow flowing results were used as the average flow was most similar to the River Nar, along with Maki-Petays et al., (1997) to determine the final velocity HSC for juvenile brown trout (Figure 4.21a).

For the final depth and velocity HSC, the curve created from the literature was used as a base (Figure 4.20). Then the rising limb was lowered slightly to correspond with the other studies and the falling limb was raised a small amount for the same reason. Otherwise the results remained the same (Figure 4.21a). The calibration data indicated high velocities (<0.33m/s) were disliked, the HSC corresponds to this as the falling limb starts at 0.33m/s. The depth calibration data indicated low depths (<0.4m) are more preferential that high depths (>0.82m), this is in accordance with the HSC.

Substrate HSC

A key aspect taken from literature was that juvenile brown trout avoid smaller substrate such as silt, sand and fine gravel, and rockier substrate is preferred (Heggenes 1988b; Ayllon et al., 2009). In a study completed by Heggenes (1988c), there was a 40% increase in young brown trout when a sandy bedload was reduced by 86%. The calibration data also indicated that silty substrate is disliked and the most preferential substrate is medium gravel (i.e. index 4). Therefore a moderate substrate index of 5 was regarded most preferable, with substrates under 3 deemed as unsuitable in the final HSC (Figure 4.21b).



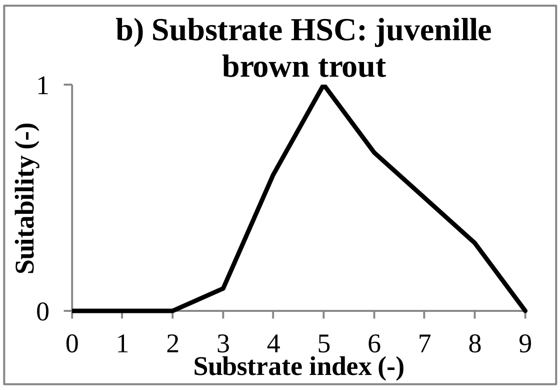


Figure 4.21- Final velocity and depth (a) and substrate (b) HSC's for juvenile brown trout

4.10.2 Fuzzy rules

The fuzzy rules for juvenile brown trout were provided by CASiMiR, as these had been checked by an expert, they remained largely the same, only adapting slightly based on literature and calibration data. In terms of determining which variables were most important for juvenile brown trout, Heggenes (1996) noted how there is correlation between all different variables i.e. depth influences velocity, velocity influences substrate and therefore it is difficult to say which is most important and non can be considered more or less important. The fuzzy rules are presented in table 4.16, these show the original fuzzy rules provided by CASiMiR along with the new adapted fuzzy rules determined below.

Cover: Heggenes (1996) stated how brown trout are very wary fish and therefore cover is very important. Cover is considered an important factor for juvenile brown trout by most authors (e.g.Heggenes 1988b; Armstrong et al., 2003). Studies have shown the amount of overhanging banks and macrophyte cover correlate with the densities of juvenile brown trout, furthermore debris, turbulence and overhanging grass has also been proven to be important (Heggenes 1988b). Whilst in the field, mid-sized brown trout (assumed juvenile) were often seen swimming out from aquatic vegetation and large woody debris, this corresponds with findings from Ayllon et al., (2009) who noted how juveniles mainly selected habitat characterised by presence of cover. The fuzzy rules provided by CASiMiR do not include any cover values therefore a change was made to incorporate cover values in (Table 4.15).

Table 4.15- Cover preferences for juvenile brown trout

Cover types	Index (-)	Preference
No cover	0	No
Aquatic plants	1	Yes
Stones/detritus	2	No
Roots	3	No
Deadwood (LWD)	4	Yes
Dry branches	6	No
Floating macrophytes	7	Yes
Turbulence	8	Yes
Undercut banks	9	No
Overhanging grass	10	Yes

Substrate: Substrate has been deemed less important for juvenile fish (Heggenes 1996), this however does not correspond with the rules provided where all combinations with a low substrate are given a low SI. Furthermore high numbers of fish were found in areas dominated by fine gravel. The highest numbers of juvenile were found in areas dominated by medium gravel and no fish were found in areas dominated by silt (Table 4.13). Therefore it can be said that small substrates with sand and silt are not preferred.

Depth and velocity: A combination of appropriate depth (relatively shallow) and velocity (relatively fast) is also of key importance for juveniles according to literature (Table 4.14). The original fuzzy rules however depicted low velocities to be more appropriate than high velocities. Therefore the resulting SI values for these were changed to switch the high and low velocities. All other values remained the same.

New fuzzy rules explained- refer to Table 4.16

Step 1: ‘low’ velocity SI was swapped with ‘high’ velocity SI results

Step 2: Duplicate for adding cover values and increase by one point i.e. ‘low’ to ‘medium’ for each with cover

Step 3: ‘medium’ and ‘high’ velocities were preferred therefore were kept the same apart from ‘high’ depth which was one level lower for each combination apart from ‘medium’ substrate which remained the same

Step 4: ‘low’ and ‘very high’ velocities are least preferred therefore ‘high’ depths were lowered

Cover values are as follows (refer to table 4.15)

Cover C= 1, 4, 7, 8, 10

Cover A=0, 2, 3, 6, 9

Table 4.16- Fuzzy rules for juvenile brown trout, V= velocity, D= depth, S= substrate, Co= cover, SI= suitability, VH= very high, H= high, M= medium, L- low

V	D	S	SI- original	SI (cover A)	SI (cover C)
H	H	H	L	L	L
H	H	M	L	L	M
H	H	L	L	L	L
H	M	H	L	L	M
H	M	M	L	M	H
H	M	L	L	L	M
H	L	H	L	M	H
H	L	M	L	H	VH
H	L	L	L	L	M
M	H	H	L	L	L
M	H	M	L	L	M
M	H	L	L	L	L
M	M	H	M	M	H
M	M	M	M	M	H
M	M	L	L	L	M
M	L	H	M	M	H
M	L	M	M	M	H
M	L	L	L	L	M
L	H	H	L	L	L
L	H	M	L	L	L
L	H	L	L	L	L
L	M	H	L	L	M
L	M	M	M	L	M
L	M	L	L	L	L
L	L	H	M	L	M
L	L	M	H	L	M
L	L	L	L	L	L
VH	H	H	L	L	L
VH	H	M	L	L	L
VH	H	L	L	L	L
VH	L	H	L	L	M
VH	L	M	L	L	M
VH	L	L	L	L	L
VH	M	H	L	L	M
VH	M	M	L	L	M
VH	M	L	L	L	L

4.11 Adult brown trout habitat suitability data (*Salmo Trutta*)

Adult brown trout tend to use deeper but slower flowing habitats than juvenile brown trout (Heggenes 1996). A large range of studies have been carried out on habitat requirements of adult brown trout. A summary of the studies and results is shown in table 4.17.

Table 4.17- Depth, velocity and substrate preferences for adult brown trout from literature

	Reference	**Country/ notes**
Velocity (m/s)		
0.1-0.7	(Heggenes 1988a)	Southeast Norway
0-0.65 (mean 0.26)	(Shirvell and Dungey 1983)	New Zealand
0-0.6	(Ayllon et al., 2010)	
Depth (m)		
0.14-1.2 (mean 0.65)	(Shirvell and Dungey 1983)	
More than 0.5	(Heggenes 1988a)	
Min of 0.05 (mean 0.12-	Baldes (Armstrong)	Artificial flume
0.9-3.05 (mean 0.69)	(Heggenes et al., 2002)	
0.5-0.75m	(Maki-Petays et al., 1997)	Finland
0.09-3.05 (mean 0.69)		
Substrate (mm)		
8-128	(Eklov et al., 1999)	Southern Sweden
50-70 (Max- 128)	(Heggenes 1988a)	
Cover		
Over 50%	(Heggenes 1988a)	Norway
Over 55%	(Heggenes 1996)	Wyoming

4.11.1 HSC

<u>**Velocity and depth HSC**</u>

The HSC for depth and velocity derived in literature for adult brown trout are shown in figure 4.22. The velocities used were generally between 0-0.6m/s. Thus the best suitability for adult brown trout was placed at 0.2m/s with a maximum preference of 0.8m/s; this is slightly lower than for juvenile brown trout. The curve determined from literature was generally used for depth preferences as the other studies fitted well with the curve. A highest preference of 0.6 to 0.8m was determined with a maximum tolerance of 1.3m; this is slightly higher than for juvenile brown trout. Ayllon et al., (2010) noted how juveniles and adults use similar depths. The final HSC are shown in figure 4.23a.

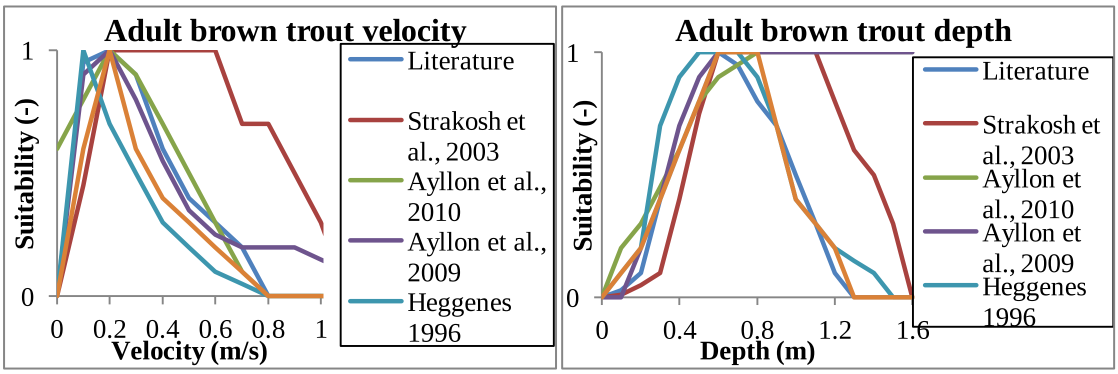


Figure 4.22- Depth and velocity HSC for adult brown trout from literature

Substrate

Adult brown trout use larger substrate than other lifestages. Eklov et al.,(1999) reports substrate sizes of 8-128mm being preferred whilst Heggenes (1988a) reports 50-80mm being used. Overall the HSC for substrate showed an index of 6 (6-12cm) being most preferred, with tolerances of indexes between 2-9. Therefore a wide variety of substrate is used by adult brown trout, however very fine organic material and are not preferable (Figure 4.23b). This is also in accordance with the calibration data which indicated small substrate, particularly silt was not liked by brown trout.

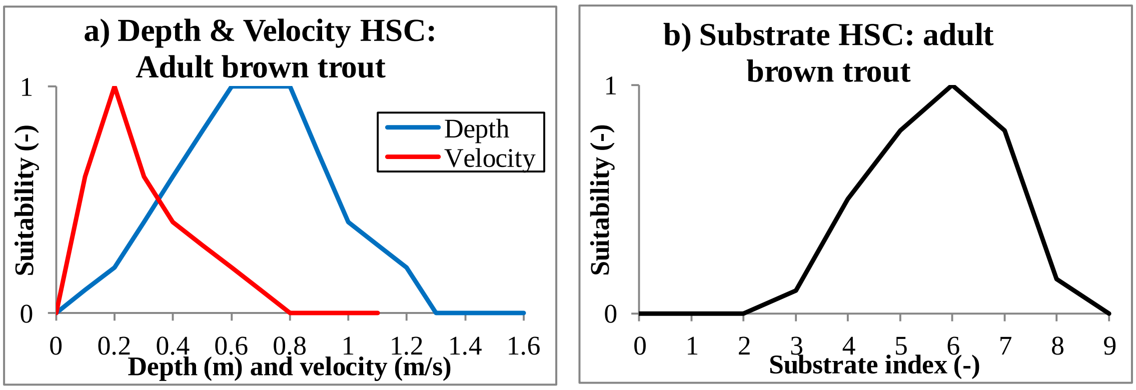


Figure 4.23- Depth and velocity (a) and substrate (b) HSC for adult brown trout

4.11.2 Fuzzy rules

The fuzzy rules provided by CASiMiR corresponded well with literature findings, for example, a key finding from literature was that adult brown trout prefer deep areas and velocities at around 0.2m/s (maximum of 0.7m/s). The calibration data indicated that lower depth are preferred to higher depths, however as this is for both adult and juvenile it cannot be fully relied upon. For this reason the fuzzy rules were left the same. Only the cover preferences were altered for adult brown trout fuzzy rules.

Cover: Adult brown trout favour areas with cover as it provides protection from predators. Studies done in Norway and Wyoming showed preferences of over 50% and over 55% cover respectively (Heggenes 1988b; Heggenes 1996). Unlike juveniles however adults avoid areas of turbulence (Heggenes 1988b). Adult brown trout have a maximum preference for either pool habitats or positions dominated by cover and the preference for submerged cover increases as depth decreases (Ayllon 2010), this was also found in the calibration data (Table 4.13). From these findings the preferences shown in Table 4.18 were determined for the fuzzy rules.

The preferences indicated in table 4.18 translate to:

Cover A= Index: 0, 1, 6, 7, 8 and 10

Cover C= Index: 2, 3, 4, 5 and 9

Suitability's were reduced when covers 'A' were included as literature and calibration data (Table 4.13) indicated that adult brown trout have a preference for cover 'C' i.e. aquatic plants and undercut banks. The final fuzzy rules are shown in Table 4.19.

Table 4.18- Cover preferences used for adult brown trout

Cover types	Index	Preference
No cover	0	No
Aquatic plants	1	Yes
Stones/detritus	2	No
Roots	3	Yes
Deadwood (LWD)	4	Yes
Overhanging	6	No
Floating macrophytes	7	Yes
Turbulence	8	No
Undercut banks	9	Yes
Overhanging grass	10	No

Table 4.19- Fuzzy rules for adult brown trout. V= velocity, D= depth, S= substrate, SI= suitability, VH= very high, H= high, M= medium, L- low

V	D	S	SI with cover A	SI with cover C
H	H	H	H	H
H	H	M	M	H
H	H	L	L	H
H	M	H	M	M
H	M	M	L	M
H	M	L	L	M
H	L	H	L	L
H	L	M	L	L
H	L	L	L	L
M	H	H	VH	VH
M	H	M	H	VH
M	H	L	M	VH
M	M	H	H	H
M	M	M	M	H
M	M	L	L	H
M	L	H	M	M
M	L	M	L	M
M	L	L	L	M
L	H	H	H	H
L	H	M	M	H
L	H	L	M	H
L	M	H	M	M
L	M	M	L	M
L	M	L	L	M
L	L	H	L	L
L	L	M	L	L
L	L	L	L	L
VH	H	H	L	L
VH	H	M	L	L
VH	H	L	L	L
VH	L	H	L	L
VH	L	M	L	L
VH	L	L	L	L
VH	M	H	L	L
VH	M	M	L	L
VH	M	L	L	L

4.12 Macrophytes Habitat Suitability Data (*Ranunculus Fluitans*)

CASiMiR provided HSC and fuzzy rules and sets for Crowfoot: *Ranunculus Fluitans*, these were calibrated based on literature findings and on data collected in the field.

4.12.1 Calibration data

The data used for calibration was based on the collected Crowfoot data collected between July 2013 and July 2014; this data is shown in table 4.20. The EA macrophyte data was unused as there were limited dates and sites collected. The collected data was used with caution as natural growth and die-back had to be taken into account, therefore flow is not the determining factor of abundance (Cranston and Darby 2004).

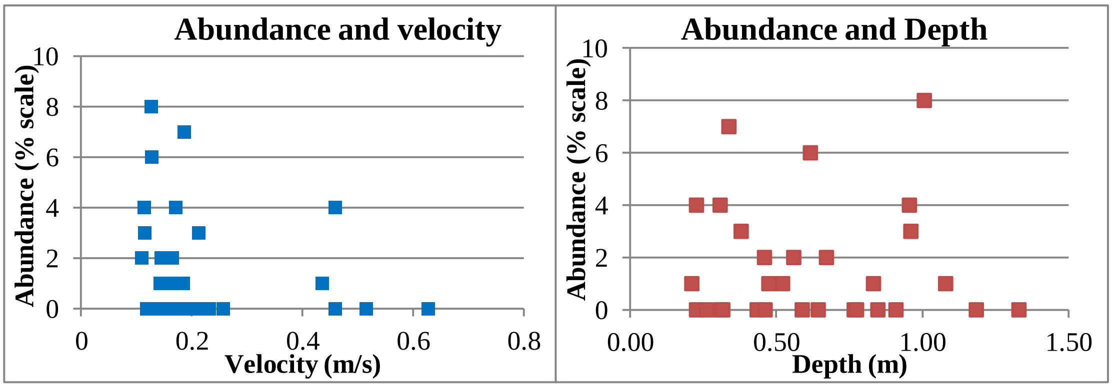


Figure 4.24- Crowfoot calibration data (see table 3.7 for % scale)

Sites 7, 8 and 9 had no hydraulic model data for them so did not have any velocity or depth data. Figure 4.24 shows velocity and depth in relation to macrophyte abundance. Generally higher abundances were found in lower velocities (<0.3m/s) however the depths were varied. The highest abundances were found where there was gravel substrate. These findings were used in determining the HSC and fuzzy rules for Crowfoot.

Table 4.20- Crowfoot calibration data (see table 3.7 for % scale)

		Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7	Site 8	Site 9
		Highbridge	**Marham**	**DS Nar**	**US Nar**	**W Acre**	**C Acre**	**Litcham**	**Lexham**	**Mileham**
	Substrate	Sand/silt	Gravel/ sand	Gravel	Gravel/ sand	M Gravel	Gravel	Gravel	Gravel	Silt
	Closest XS	10ds	16	320	1900	2900	5400	n/a	n/a	n/a
Jul-13	**Abundance (% scale)**	1	8	6	7	1	4	3	0	0
	Flow m^3/s	0.96	0.63	0.58	0.51	0.47	0.32	0.3	0.18	0.13
	Depth (m)	0.832	1.006	0.62	0.34	0.47	0.23	n/a	n/a	n/a
	Velocity (m/s)	0.151	0.126	0.127	0.186	0.169	0.459	n/a	n/a	n/a
Oct-13	**Abundance (% scale)**	0	3	0	0	2	0	0	0	0
	Flow m^3/s	0.81	0.53	0.49	0.45	0.42	0.29	0.27	0.16	0.12
	Depth (m)	0.774	0.96	0.59	0.32	0.46	0.23	n/a	n/a	n/a
	Velocity (m/s)	0.139	0.114	0.118	0.175	0.164	0.459	n/a	n/a	n/a
Jan-13	**Abundance (% scale)**	0	0	0	0	0	0	1	1	n/a
	Flow m^3/s	2.31	1.52	1.39	1.14	1.05	0.73	0.69	0.41	n/a
	Depth (m)	1.184	1.33	0.85	0.46	0.64	0.31	n/a	n/a	n/a
	Velocity (m/s)	0.22	0.194	0.2	0.256	0.231	0.627	n/a	n/a	n/a
May-14	**Abundance (% scale)**	0	1	2	3	1	0	2	0	0
	Flow m^3/s	1.21	0.8	0.75	0.67	0.6	0.45	0.42	0.24	0.19
	Depth (m)	0.909	1.079	0.67	0.38	0.52	0.26	n/a	n/a	n/a
	Velocity (m/s)	0.166	0.142	0.144	0.212	0.184	0.515	n/a	n/a	n/a
Jul-14	**Abundance (% scale)**	0	4	2	4	0	1	5	0	n/a
	Flow m^3/s	0.79	0.52	0.45	0.41	0.37	0.26	0.24	0.14	n/a
	Depth (m)	0.766	0.955	0.56	0.31	0.43	0.21			
	Velocity (m/s)	0.138	0.113	0.109	0.17	0.16	0.435			

4.12.2 HSC

The HSC for Crowfoot provided by CASiMiR had been assessed and approved by ecologists. Therefore it was intended that to use these directly as input to the models. Firstly however a literature search was carried out to reduce any site specific findings from the findings from CASiMiR and secondly, collected Crowfoot data was used to validate the HSC to the River Nar.

Table 4.21- Depth and velocity preferences for Crowfoot from literature

	Reference
Velocity (m/s)	
0.4-1	(Spink 1992)
1	(Cranston and Darby 2004)
Depth (m)	
Over 1	(Spink 1992)
0.5-1.5	(Newbold 1997)
0.1-0.95	(Husak 1998)
Max 2	(Dawson 1973)
0.35-2.75	(Dawson 1973)

Literature demonstrated that substrate is one of the most important determining factors in determining Crowfoot growth. Crowfoot requires stable substrate of coarse gravel and pebbles and moreover silt provides the least preferable habitat conditions (Cranston and Darby 2004). Furthermore highest abundances of Crowfoot were found in the River Nar in areas which had gravel as substrate i.e. sites 3 and 4 (Table 4.20).

According to literature findings (e.g. Dawson 1973; Spink 1992), the HSC provided by CASiMiR corresponded well for depth and velocity preferences and were therefore left the same. No clear trend occurred in the calibration data as to i.e. an increasing velocity provides an increasing abundance; this is mainly due to the limitations of using the data (section 4.6.4). However the highest abundances of Crowfoot were found in depths and velocities of 1m and 0.126m/s respectively (Figure 4.25). These findings are in accordance with the HSC provided. The lowest abundances i.e. 0, occurred in a variety of depths and velocities indicating that Crowfoot has a wide ranging habitat preference. Therefore the wide ranging preference depicted by the HSC (suitability of 1= 0.1-0.8m depth and 0.4-0.9m/s velocity) shows an accurate representation of suitability. Overall the HSC provided by CASiMiR corresponded well to both literature and calibration data and were therefore left the same (Figure 4.25).

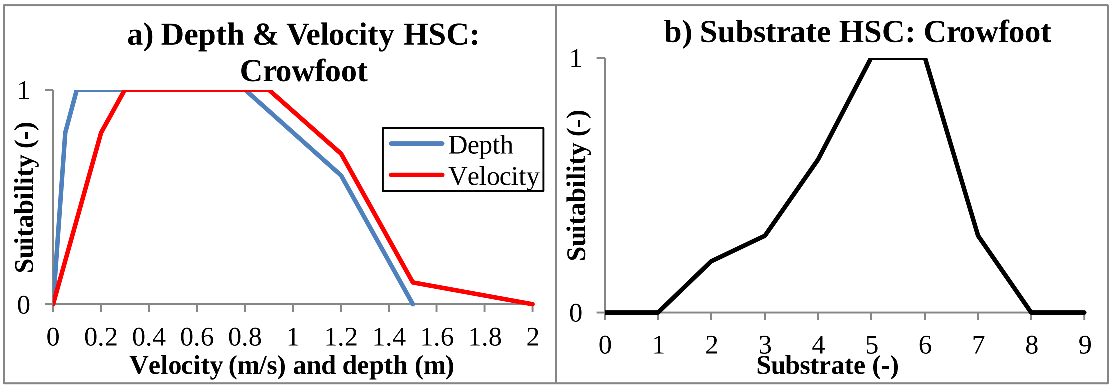


Figure 4.25- Final velocity and depth (a) and substrate (b) HSC for Crowfoot

4.12.3 Fuzzy sets

CASiMiR provided fuzzy rules and sets which were based on the HSC which were also provided. Therefore the fuzzy sets provided were directly used. However as with other species slight changes were made in accordance with literature findings and with calibration data.

The only change made to the original fuzzy sets was the substrate values. Crowfoot requires stable substrate generally consisting of pebbles, silted substrate is not preferable (Spink 1992; Cranston and Darby 2004) and furthermore low abundances of Crowfoot were found in the river when substrate was sand or silt. Therefore in the fuzzy rules it was intended to set a rule to determine anything under an index of 3 (fine gravel 2-6mm) as low suitability, the fuzzy set was altered accordingly (Figure 4.26).

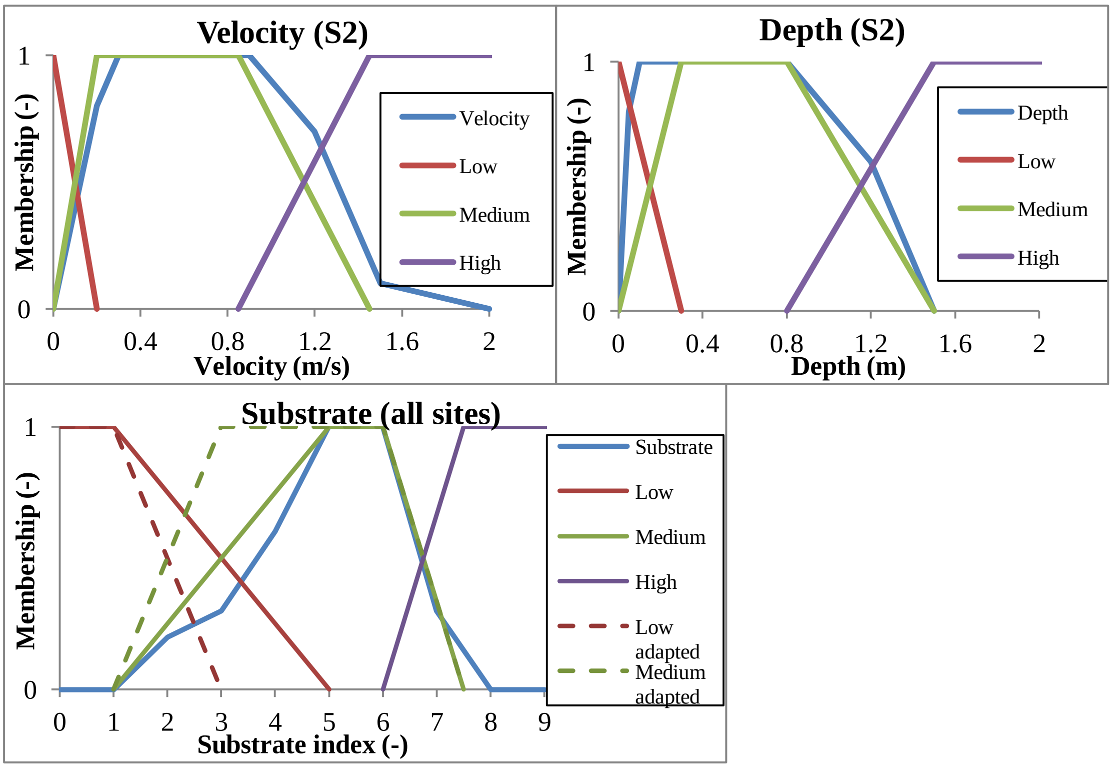


Figure 4.26- Fuzzy sets for Crowfoot

The fuzzy sets were also slightly adapted based on each river section. For example, as velocities tend to increase upstream, a high velocity at Site 1 (downstream site) was 0.75m/s, at Site 2 (mid site) it was 0.85m/s and at Site 3 (upstream site) it was 0.9m/s, these were based on model results (shown in Appendix H).

4.12.4 Fuzzy rules

Fuzzy rules were provided for Crowfoot by CASiMiR. Minor changes were made based on literature. The original and adapted fuzzy rules are demonstrated in table 4.22, the following list explains any changes made:

- Low velocities are not preferable for Crowfoot, the original fuzzy rules depicted this and were therefore were left the same.
- All medium substrate values with ‘low SI’ were increased to ‘medium SI’ as this was shown in literature and calibration data.
- ‘High velocity’ with a ‘medium depth’ was increased to ‘high SI’ as this combination is preferred (Newbold 1997; Cotton et al., 2006).

Table 4.22- Fuzzy rules for Crowfoot. V= velocity, D= depth, S= substrate, SI= suitability, VH= very high, , H= high, M= medium, L= low

V	D	S	SI
L	L	L	L
L	L	M	L
L	L	H	L
L	M	L	L
L	M	M	L
L	M	H	L
L	H	L	L
L	H	M	L
L	H	H	L
M	L	L	L
M	L	M	L
M	L	H	L
M	M	L	M
M	M	M	VH
M	M	H	M
M	H	L	L
M	H	M	M
M	H	H	L
H	L	L	L
H	L	M	L
H	L	H	L
H	M	L	L
H	M	M	L
H	M	H	L
H	H	L	L
H	H	M	L
H	H	H	L

Original

V	D	S	SI
L	L	L	L
L	L	M	L
L	L	H	L
L	M	L	L
L	M	M	L
L	M	H	L
L	H	L	L
L	H	M	L
L	H	H	L
M	L	L	L
M	L	M	M
M	L	H	L
M	M	L	M
M	M	M	VH
M	M	H	M
M	H	L	L
M	H	M	M
M	H	H	L
H	L	L	L
H	L	M	M
H	L	H	L
H	M	L	L
H	M	M	H
H	M	H	M
H	H	L	L
H	H	M	M
H	H	H	L

New

4.13 Benthos habitat suitability data, Baetidae (*Baetis Spp.)*

CASiMiR provided generic fuzzy rules for the family of Mayfly. These were then therefore adapted to be more specific to the species Baetidae (*Baetis Spp.*); this was primarily based on specific FST numbers and also on calibration data. The fuzzy rules were developed first and the HSC were developed as a result of these.

4.13.1 Calibration data

The data available for calibration was based on the EA BMI data. Sites 4 (Castle Acre road bridge) to 10 (Kings Lynn) (see figure 3.5) were the only sites which could be used for velocity and depth calibration as these are the only sites which are encompassed by the upstream and downstream hydraulic models. The flow on each day of recorded BMI data was run through the relevant 1D hydraulic model to determine the depth and velocity at the closest cross section. The depths and velocities were then categorised into low, medium and high related to the other recordings, these are presented in table 4.23a. The abundances of Baetidae (Table 4.23c) were then pooled into each category (Table 4.23b), for example, 22 recordings of category 'A' (1-9 Baetidae found) occurred in 'low' depths (0.18-0.59m) and 'medium' velocities (0.227-0.300m/s). Table 4.23c presents the abundance categories. A full list of velocity and depth calibration results is presented in Appendix I.

Table 4.23- Baetidae velocity and depth calibration data

a) Depth and velocity categories

Depth (m)	Low	0.18	0.59
	Medium	0.60	1.00
	High	1.01	1.41
Velocity (m/s)	Low	0.054	0.226
	Medium	0.227	0.399
	High	0.400	0.572

b) Baetidae abundances in categories

Depth (m)	Velocity (m/s)	A	B	C	D
Low	Low	1	0	1	0
Low	Medium	22	6	4	0
Low	High	9	2	1	0
Medium	Low	24	1	12	0
Medium	Medium	5	1	2	0
Medium	High	0	0	0	0
High	Low	6	0	11	1
High	Medium	7	0	0	0
High	High	0	0	0	0

c) Abundance category

A	1- 9
B	10-99
C	100-999
D	1,000- 9,999
E	10,000+

The highest abundances of Baetidae were found in high depths and low velocities. High abundances are also found in medium depths and low velocities however. Further to this the site conditions could also be influencing the available habitat; these findings were therefore transferred into the HSC and fuzzy rules and sets taking this into account.

The substrate and cover calibration data (Table 4.24) indicated that the highest abundances of Baetidae were found at sites 5, 9 and 2:

- Site 5 had sand and small gravel with no cover,
- Site 9 had fine gravel and some aquatic vegetation,
- Site 4 had large gravel and aquatic plants.

It is clear therefore that Baetidae use a variety of substrates and covers. It is unclear as to which the least preferable substrate for Baetidae are as some abundances of Baetidae were found at all sites, equally however 0 abundances were found at each site. Therefore it is more appropriate to use literature to determine preferences for the species using the calibration data as a guide.

Table 4.24- Baetidae cover and substrate calibration data

Site	Substrate	Cover	A	B	C	D
1	Fine gravel	Overhanging branches	5	0	0	0
2	Medium Gravel	Overhanging branches	14	1	1	0
4	Large gravel	Aquatic plants	8	5	2	0
5	Sand/ small gravel	Non	11	15	0	1
6	Medium gravel	Overhanging branches	8	5	1	0
7	Fine gravel	Non	15	5	1	0
8	Silt	Non	9	0	0	0
9	Fine gravel	Some aquatic	11	1	3	0
10	Silt/ sand	Non	12	2	1	0

4.13.2 Fuzzy sets

***FST*:** FST-hemispheres numbers are the assessment of the forces acting on BMI (see section 2.8.2 for further details). FST numbers were provided by CASiMiR for Baetidae. The FST values were categorised into ‘low’, ‘medium’, ‘high’ and ‘very high’, which were given ‘medium’, ‘very high’, ‘high’ and ‘low’ suitability’s respectively. All ‘very low’ FST values were removed and made ‘low’. The final fuzzy set for FST is shown in figure 4.27.

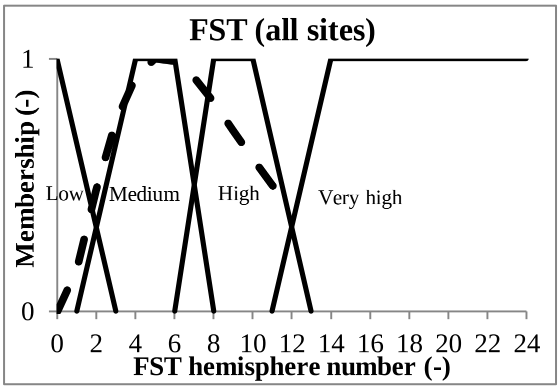


Figure 4.27- FST fuzzy sets for Baetidae. FST curve in dashed line.

Velocity and depth**:** Higher velocities were shown as preferred in the original fuzzy rules provided by CASiMiR; this is in accordance with literature where it was found species have a preference of velocities over 0.75m/s and 0.56m/s (Jowett 1990; Kopecki 2008). 'High' velocity was given 'high' preference, 'medium' was given 'medium' preference and 'low' was given 'low' preference.

Two depths were specified, a 'medium' which was established as 'high' preference and 'low' which was established as 'low' preference (figure 4.28).

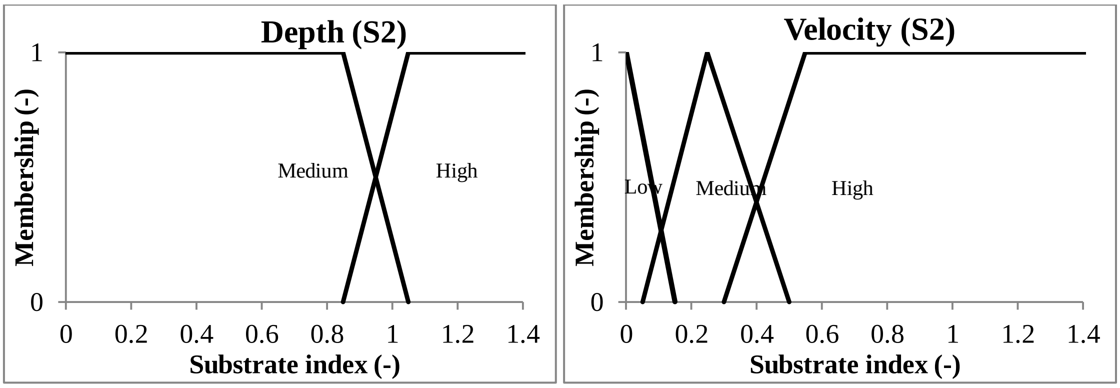


Figure 4.28- Depth and velocity fuzzy sets for Baetidae

Substrate**:** Invertebrate abundance has been shown to increase up to cobble size but decrease in boulders and bedrock (Jowett 1990). Only two values were set for depth; 'medium' providing the worst habitat and 'high' providing the best (Figure 4.29).

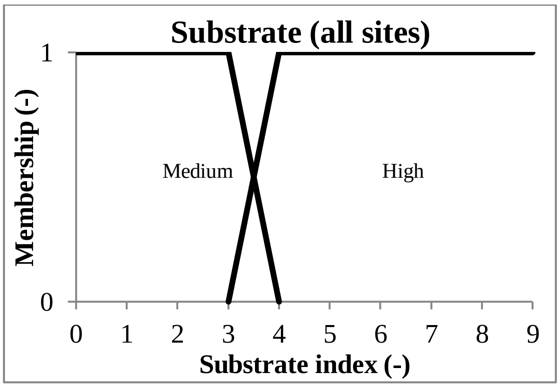


Figure 4.29- Substrate fuzzy sets for Baetidae

All the fuzzy sets for each site are presented in Appendix J.

4.13.3 Fuzzy rules

Generic rules for Mayfly were provided by CASiMiR (Figure 4.25 'original'). These were therefore adapted to be specific to Baetidae based on literature and calibration results. The original and final fuzzy rules are presented in table 4.25 followed by a description of the process used to determine them.

Table 4.25- Fuzzy rules for Baetidae. V= velocity, D= depth, S= substrate, SI= suitability, VH= very high, H= high, M= medium, L= low, VL= very low

V	D	S	FST	SI
L	M	M	VL	VL
L	M	M	L	VL
L	M	M	M	VL
L	M	M	H	VL
L	M	M	VH	VL
L	M	H	VL	VL
L	M	H	L	L
L	M	H	M	M
L	M	H	H	H
L	M	H	VH	VH
L	H	M	VL	VL
L	H	M	L	VL
L	H	M	M	VL
L	H	M	H	VL
L	H	M	VH	VL
L	H	H	VL	VL
L	H	H	L	VL
L	H	H	M	VL
L	H	H	H	VL
L	H	H	VH	VL
M	M	M	VL	VL
M	M	M	L	VL
M	M	M	M	VL
M	M	M	H	VL
M	M	M	VH	VL
M	M	H	VL	VL
M	M	H	L	L
M	M	H	M	M
M	M	H	H	H
M	M	H	VH	VH
M	H	M	VL	VL
M	H	M	L	VL
M	H	M	M	VL
M	H	M	H	VL
M	H	M	VH	VL
M	H	H	VL	VL
M	H	H	L	VL
M	H	H	M	VL
M	H	H	H	VL
M	H	H	VH	VL
H	M	M	VL	VL
H	M	M	L	VL
H	M	M	M	VL
H	M	M	H	VL
H	M	M	VH	VL
H	M	H	VL	VL
H	M	H	L	L
H	M	H	M	M
H	M	H	H	H
H	M	H	VH	VH
H	H	M	VL	VL
H	H	M	L	VL
H	H	M	M	VL
H	H	M	H	VL
H	H	M	VH	VL
H	H	H	VL	VL
H	H	H	L	VL
H	H	H	M	VL
H	H	H	H	VL
H	H	H	VH	VL

Original

V	D	S	FST	SI
L	M	M	L	L
L	M	M	M	M
L	M	M	H	L
L	M	M	VH	L
L	M	H	L	M
L	M	H	M	H
L	M	H	H	M
L	M	H	VH	L
L	H	M	L	L
L	H	M	M	L
L	H	M	H	L
L	H	M	VH	L
L	H	H	L	L
L	H	H	M	M
L	H	H	H	L
L	H	H	VH	L
M	M	M	L	L
M	M	M	M	H
M	M	M	H	M
M	M	M	VH	L
M	M	H	L	M
M	M	H	M	VH
M	M	H	H	H
M	M	H	VH	L
M	H	M	L	L
M	H	M	M	L
M	H	M	H	L
M	H	M	VH	L
M	H	H	L	L
M	H	H	M	H
M	H	H	H	M
M	H	H	VH	L
H	M	M	L	M
H	M	M	M	H
H	M	M	H	M
H	M	M	VH	L
H	M	H	L	M
H	M	H	M	VH
H	M	H	H	H
H	M	H	VH	L
H	H	M	L	L
H	H	M	M	M
H	H	M	H	L
H	H	M	VH	L
H	H	H	L	M
H	H	H	M	H
H	H	H	H	M
H	H	H	VH	L

New

The adaptations were based on the following changes:

- Original rules are the same for each velocity section

- ‘Very low’ SI was removed and changed to ‘low’; this is as no other species has ‘very low’ suitability option.
- Secondly rules were set in the following order:

Velocity	
Set	**Availability**
High	High
Medium	Medium
Low	**Low**

FST	
Set	**Availability**
Low	Medium
Medium	Very high
High	High
Very	Low

Depth	
Set	**Availability**
Medium	High
High	Low

Substrate	
Set	**Availability**
Medium	Low
High	High

- As FST values are the most important for Mayfly- all rules started with ‘low’ as medium availability, ‘medium’ as very high availability, ‘high’ as high availability and ‘very’ high as low availability. These rules are set when there are 3 high availabilities (high velocity, medium depth and high substrate)
- For each individual rule, these were used to set the availability. For example, ‘low’ velocity, ‘low’ depth and ‘medium’ substrate scores, 2 lows and 1 high availability. Therefore all apart from ‘medium’ FST score ‘low’. For ‘high’ velocity, ‘medium’ depth and ‘medium’ velocity, there were 2 ‘high’ scores and 1 ‘low’, therefore rules remain largely the same but dropped one level of availability for ‘medium’ and ‘high’ FST.

4.13.4 HSC

The HSC for Baetidae were developed based on the fuzzy sets. The HSC line was drawn to show where would be most preferable and least preferable. For example, a ‘high’ substrate determined by the fuzzy rules were generally given high suitability’s, therefore the highest point of the HSC (suitability 1) was placed at around an index of 6. The final HSC for Baetidae are shown in Figure 4.32.

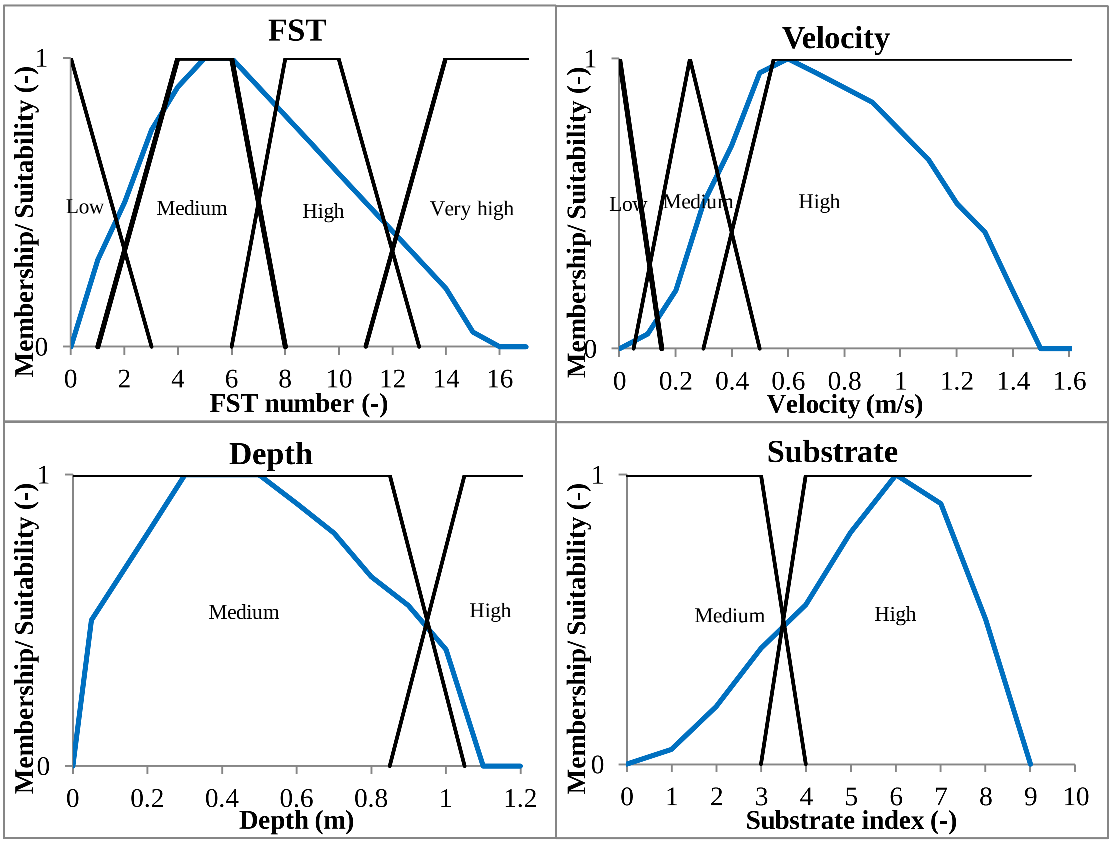


Figure 4.30- HSC's for Baetidae based on fuzzy sets

4.14 Transferability of HSC's and fuzzy rules

More information and background is provided in section 2.7.3 on the transferability of HSC's. However too summarise, HSC's can be transferred between rivers but generally it is more appropriate to determine species and site specific habitat preferences.

The HSC's and fuzzy rules derived in this project cannot be directly transferred to other rivers as they include specific data from the River Nar. The findings from the literature reviews and general curves and information could be used as a base but species specific information from different rivers would need to be used for calibration purposes.

Chapter 5- Research question 1 results

5.1 Chapter introduction

Research question 1 (RQ1): How are the ecosystem indicators effected during low flows? This section presents results for RQ1 including results from collected data and Environment Agency (EA) data. See section 3.2 for details on the analysis undertaken.

5.2 River Habitat Surveys (RHS)

The RHS gives an indication of habitat quality along the river, both how natural (Habitat Quality Assessment (HQA)) and how modified (Habitat Modification Class (HMC) and Habitat Modification Class (HMC)) the section is. These results were used to indicate the general condition for physical features on the river and are used during further analysis to show for example if high fish populations correspond with good or bad quality physical habitat as shown by the RHS scores. The HMS is of particular importance for the benthic macro-invertebrate (BMI) LIFE score analysis as correlations have been found between the two scores (Dunbar et al., 2010). The RHS locations are shown in Figure 3.6.

5.2.1 Habitat quality

As data was collected on different forms, the scores from pre-2008, 2008 and post-2008 were analysed separately. Figure 5.1 shows the results for HMS, HMC and HQA scores. Despite results from pre and post- 2008 being incomparable due to different capture methods, the trends remain the same:

- The HMS scores showed that the most downstream sites (10, 12, 13, 15 and 17) were classified as 'severely modified'. Sites 14 and 16 were however 'significantly modified'. It is likely that these sites had less modified features and more signs of recovery than the other downstream sites.
- Most points upstream were also classified as 'significantly modified', with anomalies in sites 5 and 7 scoring 'obviously unmodified' and site 6 scoring 'predominantly unmodified'.
- The HMC demonstrates a more 'pristine' river midstream gradually increasing to a 'highly modified' river downstream.
- Sites within the fen area all scored 4 or 5 on the HMC indicating they are severely modified. The highest scoring and therefore most natural sites were at Site 5, 6 and 7 which are all located mid- catchment. Site visits demonstrated there had been

minimal modification to this section of river particularly when compared to the fen sections which are highly canalised.

- All of the sites scored fair, poor or very poor on the HQA scoring, this indicates that overall the river does not exhibit very good habitat quality.
- Despite this however, results for the HQA exhibit a good measure of naturalness upstream to a low measure of naturalness downstream. The pre- 2008 data shows two anomalous points in that Sites 6 and 7 have very high scores compared to the others, thus indicating that this is where the river is at its most natural.
- The HQA scores were compared to a study by Raven (1998) which determined the number of chalk stream sites which had different HQA scores (Figure 5.2). The results indicated that all the HQA scores from the River Nar were within the average scores in this nationwide study.

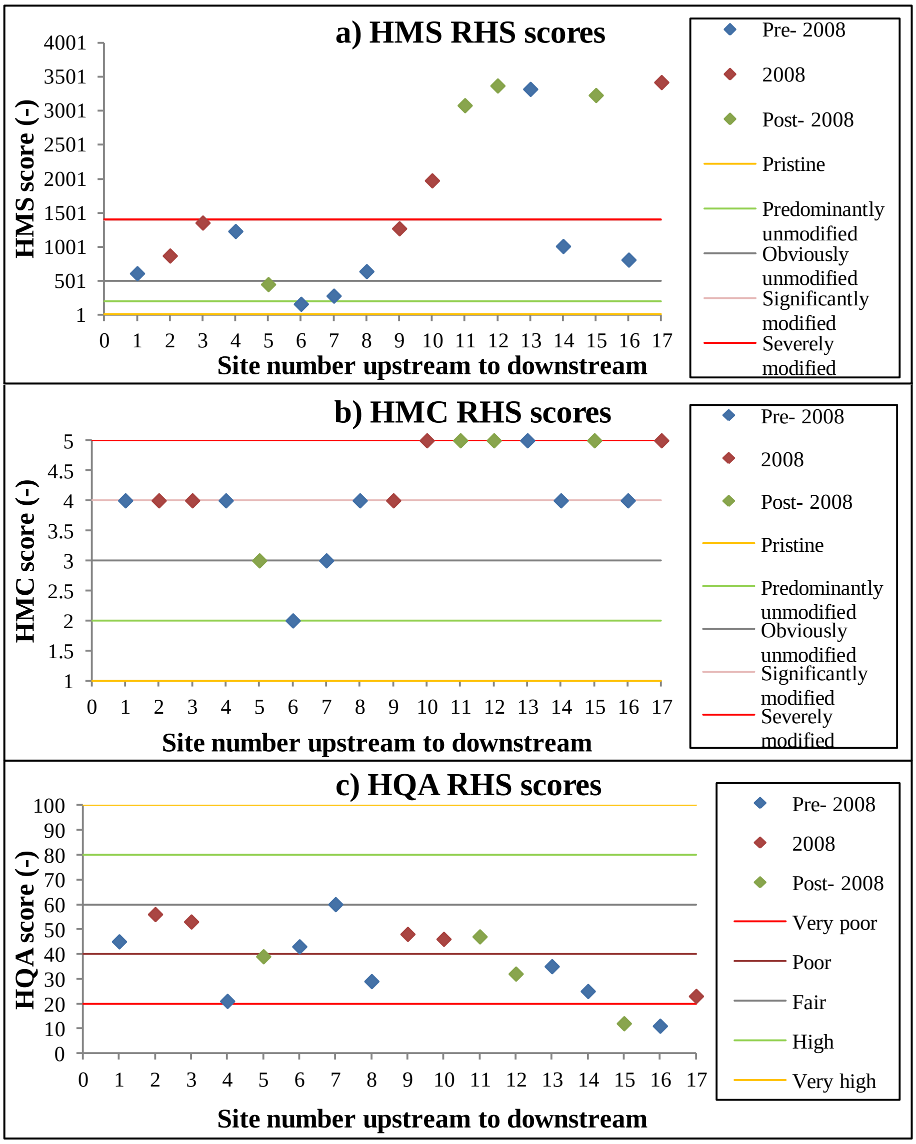


Figure 5.1- EA RHS results for each site

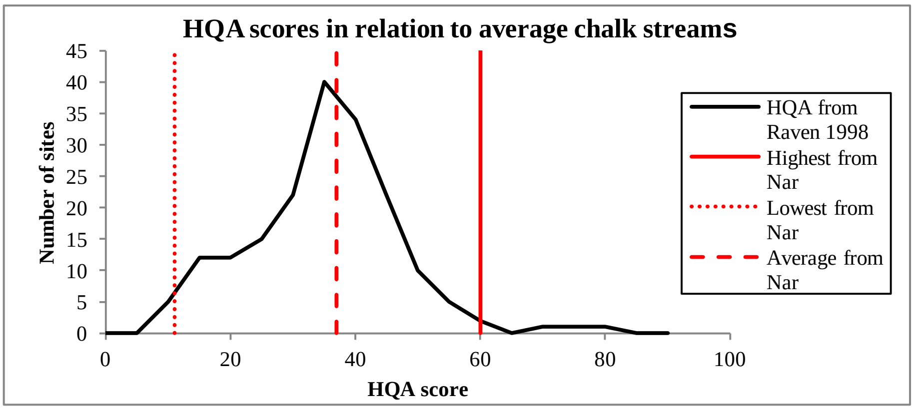


Figure 5.2- HQA scores from the River Nar in relation to average chalk stream scores

5.2.2 *Summary of analysis*

- Generally HQA scores decrease from upstream to downstream, showing how higher quality habitats are found upstream.
- Generally HMC and HMS scores increase from upstream to downstream showing how higher modified habitats are found downstream in terms of the physical indicators.

These findings indicate that the health or the River Nar is generally good and in line with other UK chalk stream rivers. The chalk stream reach (i.e. upstream) has better and a less modified physical habitat. Therefore a hypothesis that species have better quality habitat in the chalk stream reaches than in the fen reaches can be proposed.

5.3 Measured Benthic Macro Invertebrate (BMI)

During data collection for this project (July 2013 to July 2014), kick sampling took place on a seasonal basis at 11 locations along the river from Highbridge (downstream, KS11), to Mileham (upstream KS1), see Figure 3.5 for locations. These results were used to analyse the current status of BMI in the river and to show a 15 month cycle of BMI data incorporating seasonal patterns and effects of different flows.

Species in the following families were found within the river during the kick sampling:

-*Trichoptera* (Caddisflies)
-*Crustacea* (Shrimps and Hog lice)
-*Ephemeroptera* (Mayflies)
-*Plecoperta* (Stoneflies)
-*Odonata* (Dragonflies)
-*Hemiptera* (Waterbugs)
-*Coleoptera* (Waterbeetles)

-*Tricladida* (Flatworms)
-*Hirudinea* (Leeches)
-*Gastropoda* (Snails)
-*Bilvalvia* (Mussles)
-*Cladocera* (Water flea)
-*Simuliidae* (Blackflies)
-*Tipulidae* (Craneflies)
-*Chironomidae* (Midges)

A total of 41 taxa were recorded throughout all seasons:

- 37 in July 2013

- 18 in October 2013
- 15 in January 2013
- 24 in May 2014
- 23 in July 2014

The results are organised according to ASPT and LIFE scores (see Section 2.5 for definitions). Each section has two areas of analysis:

- Seasonal analysis, investigating seasonal changes and trends.

- Relationships between BMI scores with daily and antecedent conditions.

This is followed by analysis on site conditions and conclusions on measured BMI data.

5.3.1 Raw results and data removal

Raw results of all seasons and sites are presented in Table 5.1

Table 5.1- Kick sampling score results from measured kick samples per season (KS1 most upstream site, KS11 most downstream site)

	Jul-13		Oct-13		Jan-14		May-14		Jul-14	
	ASPT	LIFE	ASPT	LIFE	ASPT	LIFE	ASPT	LIFE	ASPT	LIFE
KS1	5.8	7	6.5	7	n/a		7.6	7.2	n/a	
KS2	7.19	7.63	5.71	7.86	5.25	7.42	5.22	7.11	6.5	7
KS3	n/a		7.75	8	6.25	7.75	6.33	7.17	7.25	7.5
KS4	6.65	7.7	5.56	7.33	5	7.63	7.8	8.4	7.11	7.67
KS5	6.78	7.52	6.14	8	6.6	8.4	6.86	7.21	6.71	8
KS6	6.83	7.82	6.13	7.75	5.4	7.4	8.5	8.17	6.11	7.11
KS7	6.35	7.71	5.57	7.86	7	8.67	7.63	7.25	7.82	7.73
KS8	6.85	7.92	7.83	8.5	6.67	8.33	7.22	8	7.2	8
KS9	6.57	7.65	6.4	7.2	5.5	8	6.14	7.86	5.83	6.67
KS10	6.22	6.89	4.75	7.25	6	7.5	3	5	5.4	6.8
KS11	5.25	6.75	5.33	6.67	5.5	7.5	3	5	6.25	6.5

- There are no scores for January 2014 and July 2014 for KS1 as:

- KS1 could not be accessed in January 2014 due to buffalo and calves grazing on site therefore being too dangerous to enter.

- In July 2014, the river had been diverted to a different course as part of the Norfolk Rivers Trust restoration plans, in June 2014.

- KS3 in July 2013 recorded anomalous scores with much lower score (ASPT= 5.2) than the other sites and seasons. The site conditions here are very natural scoring highly on the HQA and hence a higher ASPT score would be expected. According to Scherer (2013), who carried out the kick sample, this can be explained by the conditions at the site being a deep, slow flowing pool. BMI could have been lost from the sample due to these conditions and therefore the actual location of the kick sample was relocated slightly closer towards the bank for the subsequent seasonal kick samples. **Therefore KS3 (July 2013) has been removed for all scores.**

- Approximately 4km upstream of KS5 is a wastewater treatment works (WWTW) discharge point which discharges excessive nutrients into the river. This eutrophication creates large summer overgrowth, dominating the river (Figure 5.3). Results from KS5 capture the effect of this outfall. Results from this site took this into account.

Figure 5.3- River reach 4km upstream of KS5 in summer 2013 (left picture) and in winter 2014 (right picture- picture courtesy of Anne Visser)

5.3.2 ASPT

The ASPT scores are the BMWP divided by the amount of taxa found which attempts to neglect the sample size (see section 2.5 for further details), and is therefore said to give a more accurate representation of the sample.

Seasonal trends

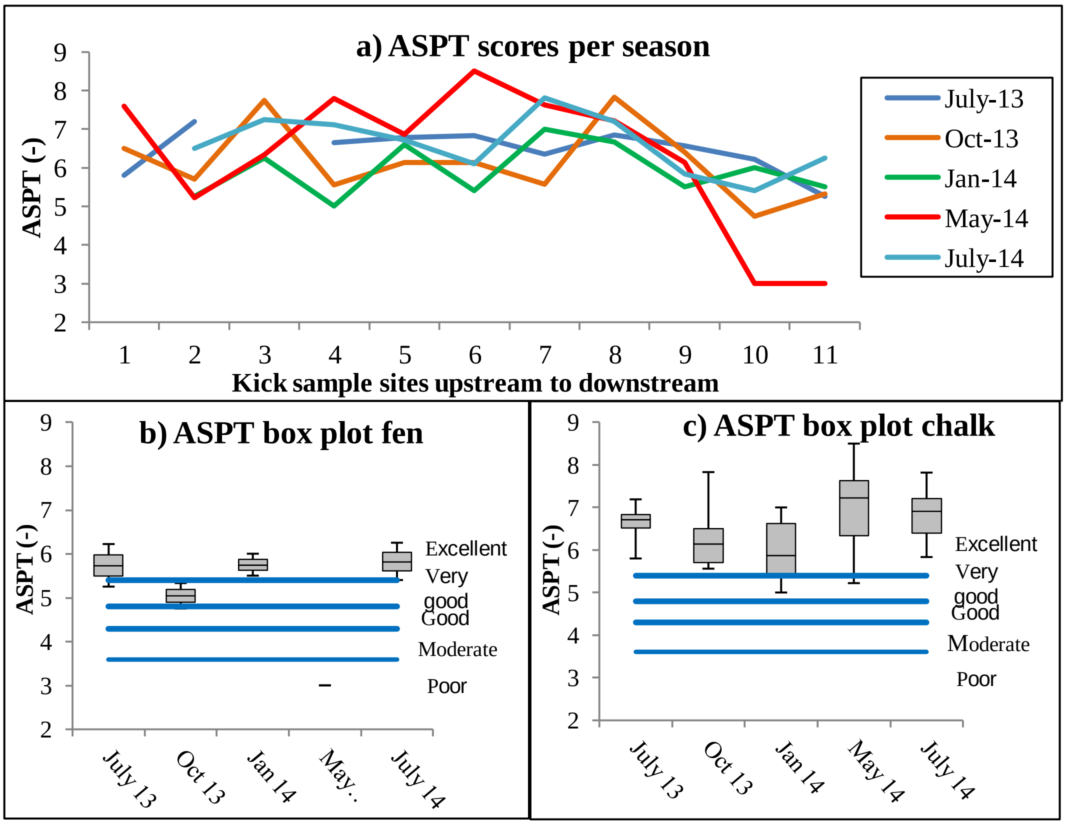


Figure 5.4- ASPT scores for all sites and seasons, box plots shown separately for fen and chalk sites

Table 5.2- Average ASPT scores per season and interpretations

Date	Average ASPT score	Water quality determined
Jul-13	6.45	Excellent
Oct-13	6.15	Excellent
Jan-14	5.92	Excellent
May-14	6.3	Excellent
Jul-14	6.62	Excellent

The results are demonstrated in Figure 5.4 and Table 5.2.

- ASPT scores in all seasons result in an excellent water quality score. Such results are usually found in relatively pristine river system where water quality and channel structure do not suppress the aquatic species (Norfolk Rivers Trust 2013).
- Summer scores are highest, followed by May 2014, then by October 2013 and finally January 2014 has the lowest score.
- Mid- stream sites have slightly higher ASPT scores whilst the most upstream and downstream sites have the lowest.
- KS 10 and 11 have significantly lower ASPT scores in May 2014; this could be attributed to its location in the fen river. All samples at these sites recorded low scores which indicate that these are related to the site rather than the flow conditions.
- Discounting KS 1 and KS 3 (see section 5.3.1), 8 out of the 9 KS have their highest or second highest score in July of either year. Again showing that the summer months provide the best habitat for BMI. 4 out of the 9 KS have their lowest scores in January indicating that winter provides the worst habitat for BMI.
- Summer months therefore provide the best habitat for BMI however individual sites do pose different conditions so it cannot be generalised.

Daily and antecedent flows

This analysis aimed to show if the flow conditions on the day of the sample and/ or the antecedent flow conditions influences the ASPT score. Table 5.3 demonstrates the results of this analysis.

Relationship to daily flow: The ASPT scores have no statistically significant relationships with daily flow for this river.

Relationship to antecedent 3 month flows: KS5 had strong positive relationships with the 3 month antecedent flow. Other sites indicated both strong positive and negative relationships with the antecedent 3 month flow however this was not supported by strong R^2 or significant p-value scores.

Relationship to antecedent 6 month flows: KS2 had statistically significant relationships with the 6 month antecedent flow for Q_{10}, Q_{50} and Q_{90} flows; however this was the only site which had this trend.

Relationship to antecedent 9 month flows: KS5 and 7 had strong negative relationships with the 9 month antecedent flow. These all had negative correlations indicating higher ASPT values are found when there is a lower flow in the 9 months previous.

Table 5.3- ASPT daily and antecedent flow relationships- see Table 3.3 for colour descriptions

		KS1	KS2	KS3	KS4	KS5	KS6	KS7	KS8	KS9	KS10	KS11
Daily flow	**Correl**	0.86	-0.61	-0.81	-0.54	0.17	-0.29	0.19	-0.64	-0.70	0.22	-0.06
	p-value	0.336	0.276	0.185	0.343	0.786	0.639	0.762	0.243	0.191	0.726	0.930
	R^2	0.75	0.37	0.66	0.30	0.03	0.08	0.04	0.41	0.49	0.05	0.00
3 months Q_{10}	**Correl**	0.53	0.01	-0.56	0.85	0.82	0.92	0.55	-0.24	0.22	-0.57	-0.78
	p-value	0.641	0.986	0.436	0.066	0.092	0.029	0.340	0.700	0.718	0.312	0.121
	R^2	0.29	0.00	0.32	0.73	0.67	0.84	0.30	0.06	0.05	0.33	0.61
3 months Q_{50}	**Correl**	0.37	0.17	-0.57	0.86	0.89	0.84	0.57	-0.34	0.23	-0.42	-0.65
	p-value	0.174	0.784	0.426	0.064	0.045	0.072	0.314	0.573	0.715	0.479	0.230
	R^2	0.14	0.03	0.33	0.73	0.79	0.71	0.33	0.12	0.05	0.18	0.43
3 months Q_{90}	**Correl**	0.65	-0.08	-0.52	0.86	0.75	0.95	0.54	-0.14	0.22	-0.68	-0.83
	p-value	0.549	0.893	0.481	0.061	0.142	0.014	0.345	0.828	0.727	0.210	0.082
	R^2	0.42	0.01	0.27	0.74	0.57	0.90	0.29	0.02	0.05	0.46	0.69
6 months Q_{10}	**Correl**	-0.97	0.97	0.77	0.28	0.09	-0.07	-0.17	0.09	0.57	0.45	0.49
	p-value	0.153	0.007	0.227	0.644	0.890	0.910	0.790	0.890	0.316	0.451	0.404
	R^2	0.94	0.94	0.60	0.08	0.01	0.01	0.03	0.01	0.32	0.20	0.24
6 months Q_{50}	**Correl**	-0.96	0.97	0.86	0.25	0.07	-0.01	-0.29	0.07	0.69	0.44	0.39
	p-value	0.174	0.006	0.143	0.689	0.905	0.990	0.635	0.917	0.198	0.453	0.516
	R^2	0.93	0.94	0.73	0.06	0.01	0.00	0.08	0.00	0.48	0.20	0.15
6 months Q_{90}	**Correl**	-0.91	1.00	0.70	0.24	0.22	-0.04	-0.18	-0.13	0.58	0.54	0.41
	p-value	0.277	0.000	0.303	0.693	0.726	0.950	0.770	0.835	0.304	0.352	0.488
	R^2	0.82	0.99	0.49	0.06	0.05	0.00	0.03	0.02	0.34	0.29	0.17
9 months Q_{10}	**Correl**	-0.94	0.44	0.61	-0.56	-0.56	-0.37	-0.93	0.16	0.60	0.54	0.35
	p-value	0.230	0.459	0.394	0.330	0.324	0.535	0.021	0.801	0.284	0.344	0.569
	R^2	0.88	0.19	0.37	0.31	0.32	0.14	0.87	0.02	0.36	0.29	0.12
9 months Q_{50}	**Correl**	-0.47	-0.07	0.63	-0.77	-0.95	-0.55	-0.91	0.51	0.27	0.28	0.37
	p-value	0.690	0.912	0.374	0.131	0.012	0.339	0.033	0.378	0.665	0.650	0.542
	R^2	0.22	0.00	0.39	0.59	0.91	0.30	0.83	0.26	0.07	0.08	0.14
9 months Q_{90}	**Correl**	-0.30	-0.15	0.70	-0.63	-0.97	-0.39	-0.88	0.68	0.34	0.07	0.24
	p-value	0.806	0.816	0.305	0.256	0.005	0.511	0.050	0.202	0.580	0.911	0.702
	R^2	0.09	0.02	0.48	0.40	0.95	0.16	0.77	0.47	0.11	0.00	0.06

It can be concluded that ASPT is not statistically significantly related to antecedent flow. There are however more relationships in mid-stream reaches and many of the sites which were not statistically correlated were close to being so and therefore with more data points it is likely that they would be statistically related. No previous studies have been carried out on assessing ASPT relationships with daily and/ or antecedent flow conditions therefore it is unknown whether there is a link. This study has however shown that there is little statistical significance between flow conditions and ASPT values, this indicates that the scoring system is more related to the water quality rather than quantity, as the LIFE scores are.

5.3.3 LIFE

LIFE scores generally range from 1-12; with low LIFE scores indicating a high abundance of drought resistant BMI therefore found frequently in low velocities. High scores indicate a high abundance of BMI preferring high velocities and therefore found frequently in high velocities (SNIFFER 2011).

Seasonal trends

The seasonal results for LIFE scores are demonstrated in Figure 5.5.

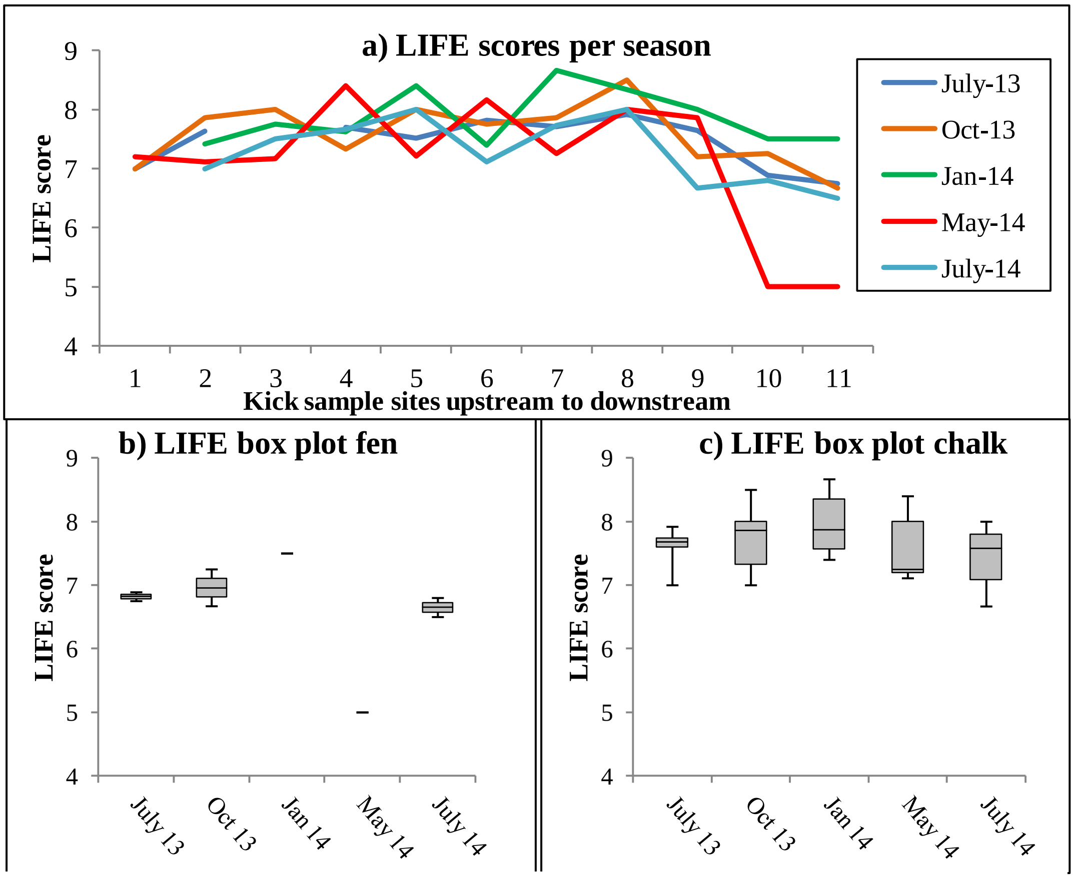


Figure 5.5- LIFE scores for all sites and seasons, box plots shown separately for chalk and fen sites

Most of the LIFE scores were between 7 and 8, indicating that BMI in the river are generally exposed to relatively high flows throughout all sections of the river (SNIFFER 2011), however the BMI will also be adaptable to lower flows.

As the seasons all scored within a similar medium to high range (7.3-7.86), there is not much seasonal variance which indicates a natural flow regime is present throughout the year i.e. high flows in winter and low flows in summer and therefore BMI are adaptable to extreme events. As Dunbar et al.,(2010) describes, taxa requiring fast velocities have a narrow niche of requirements and therefore habitat heterogeneity is essential for taxa requiring high velocities. Thus indicating that habitat heterogeneity is present in the River Nar.

The lowest scores were found at KS10 and 11 in May. These sites are in the fen reach which is highly modified and canalised. The box plots promote this finding showing both much lower scores and a smaller range in all seasons in the fen reach in comparison to the chalk reach. Figure 5.5b shows how for January and May 2014 there was just one result in the box plots, this is as the two fen sites had the same score so no range occurred. The fen reach (KS 10 and 11) has the highest flows but similar if not lower velocities in comparison to the rest of the river, the hydraulic models showed the highest velocities occur in the middle reaches i.e. KS 4-7 with the lowest velocities in the fen reaches. Dunbar et al., (2010) discovered that whilst modified reaches often portray higher velocities due to the straightened channels, they usually also have the lowest LIFE scores, this shows that the site conditions in more natural reaches provide better quality habitat in terms of substrate and refugia. This promotes the need to assess site conditions.

Table 5.4 highlights the highest recorded LIFE scores with a score of 8 or above (orange). These all occur between KS3 and KS9 with KS8 having 4 out of 5 seasons at or above a LIFE score of 8. All of these sites occur in the mid reaches which are in the chalk sections and are the most unmodified according to the HMS scores.

Table 5.4- Highest LIFE scores per site and season

	KS1	**KS2**	**KS3**	**KS4**	**KS5**	**KS6**	**KS7**	**KS8**	**KS9**	**KS10**	**KS11**
Jul-13	7.00	7.63	n/a	7.70	7.52	7.82	7.71	7.92	7.65	6.89	6.75
Oct-13	7.00	7.86	8.00	7.33	8.00	7.75	7.86	8.50	7.20	7.25	6.67
Jan-14	n/a	7.42	7.75	7.63	8.40	7.40	8.67	8.33	8.00	7.50	7.50
May-14	7.2	7.11	7.17	8.4	7.21	8.17	7.25	8.00	7.86	5.00	5.00
Jul-14	n/a	7	7.5	7.7	8.00	7.10	7.70	8.00	6.7	6.80	6.50
Average	7.07	7.40	7.61	7.74	7.83	7.65	7.84	8.15	7.47	6.69	6.48
RHS HMC	4	4	4	n/a	3	n/a	3	4	4	5	5

Daily and antecedent flows

This analysis aimed to show if the flow conditions on the day of the sample and/ or the antecedent flow conditions influences the LIFE score. Table 5.5 demonstrates the results of this analysis.

The flow conditions for LIFE scores exhibited widely varying results with no clear trends. Wilby (2010) discovered highly significant p-value relationships between the antecedent Q_{95} flow and LIFE scores, therefore it is expected that some relationships would be seen in this data.

Relationship to daily flows: there was no significant relationship to daily flows. Most sites had weak or good positive correlations however this was not supported by strong R^2 or significant p-value scores. Many studies report that the antecedent flow conditions have more of an influence on LIFE scores than the daily conditions (e.g. Extence et al., 1999; Dunbar et al., 2010), therefore this finding was not unusual.

Relationship to antecedent 3 month flows: the 3 month antecedent flow correlation coefficients had negative or positive correlations at every site. The majority were negative correlations which show that generally LIFE scores decrease with and increasing flow 3 months previous. KS 3, 5 and 10 were supported by both strong R^2 and significant p-values. KS 3 and 10 had negative correlations whilst KS 4 had positive correlations. This indicates the findings are site specific and cannot be generalised for the 3 month antecedent flow.

Relationship to antecedent 6 month flows: There were no significant relationships for 6 month antecedent conditions. This shows the 6 month antecedent flows are not related to the LIFE scores.

Relationship to antecedent 9 month flows: 9 month antecedent conditions had one positive correlation at KS2 in both the Q_{10} and Q_{50} which was supported by significant R^2 and p-values. KS1 also had a significant relationship in the 9 month Q_{10} however the perfect negative correlation indicates a potential anomaly in the results as it is very unlikely that a perfect correlation would occur.

Table 5.5- LIFE daily and antecedent flow relationships- see Table 3.3 for colour descriptions

		KS1	KS2	KS3	KS4	KS5	KS6	KS7	KS8	KS9	KS10	KS11
Daily flow	**Correl**	0.99	-0.05	-0.01	0.09	0.43	-0.06	0.74	0.28	0.74	0.22	0.42
	p-value	0.084	0.939	0.995	0.886	0.474	0.919	0.153	0.645	0.152	0.718	0.483
	R^2	0.98	0.00	0.00	0.01	0.18	0.00	0.55	0.08	0.55	0.05	0.18
3 months Q_{10}	**Correl**	0.82	-0.53	-0.96	0.95	-0.89	0.62	-0.72	-0.78	0.34	-0.91	-0.82
	p-value	0.389	0.354	0.042	0.015	0.753	0.265	0.169	0.122	0.577	0.030	0.091
	R^2	0.67	0.29	0.92	0.89	0.04	0.38	0.52	0.60	0.12	0.83	0.67
3 months Q_{50}	**Correl**	0.70	-0.55	-0.99	0.89	-0.85	0.52	-0.69	-0.88	0.28	-0.83	-0.73
	p-value	0.504	0.333	0.011	0.043	0.065	0.374	0.201	0.051	0.642	0.080	0.166
	R^2	0.49	0.31	0.98	0.79	0.73	0.27	0.47	0.77	0.08	0.69	0.53
3 months Q_{90}	**Correl**	0.89	-0.54	-0.94	0.96	-0.89	0.65	-0.76	-0.71	0.32	-0.96	-0.88
	p-value	0.297	0.349	0.064	0.009	0.042	0.232	0.139	0.182	0.602	0.010	0.049
	R^2	0.80	0.29	0.88	0.93	0.79	0.43	0.57	0.50	0.10	0.92	0.77
6 months Q_{10}	**Correl**	-0.80	0.12	0.12	-0.33	-0.20	-0.23	-0.33	-0.48	-0.62	0.21	0.11
	p-value	0.406	0.850	0.884	0.591	0.753	0.706	0.585	0.416	0.269	0.741	0.855
	R^2	0.65	0.01	0.01	0.11	0.04	0.05	0.11	0.23	0.38	0.04	0.01
6 months Q_{50}	**Correl**	-0.78	0.25	0.18	-0.31	-0.27	-0.08	-0.34	-0.46	-0.47	0.19	0.11
	p-value	0.426	0.679	0.816	0.616	0.655	0.904	0.575	0.434	0.429	0.760	0.863
	R^2	0.62	0.06	0.03	0.09	0.08	0.01	0.12	0.21	0.22	0.04	0.01
6 months Q_{90}	**Correl**	-0.67	0.17	0.08	-0.25	-0.24	-0.14	-0.26	-0.56	-0.38	0.20	0.17
	p-value	0.529	0.790	0.922	0.680	0.693	0.827	0.673	0.329	0.523	0.742	0.788
	R^2	0.45	0.03	0.01	0.06	0.06	0.02	0.07	0.31	0.15	0.04	0.03
9 months Q_{10}	**Correl**	-1.00	0.94	0.95	-0.69	0.17	0.11	0.26	0.37	0.05	0.62	0.54
	p-value	0.022	0.016	0.053	0.195	0.786	0.856	0.673	0.546	0.934	0.266	0.348
	R^2	1.00	0.89	0.90	0.48	0.03	0.01	0.07	0.13	0.00	0.38	0.29
9 months Q_{50}	**Correl**	-0.77	0.89	0.95	-0.81	0.49	-0.05	0.42	0.85	-0.10	0.67	0.53
	p-value	0.438	0.045	0.054	0.093	0.398	0.940	0.482	0.068	0.873	0.212	0.355
	R^2	0.60	0.79	0.90	0.66	0.24	0.00	0.18	0.72	0.01	0.45	0.28
9 months Q_{90}	**Correl**	-0.64	0.84	0.87	-0.71	0.36	0.06	0.24	0.85	-0.16	0.51	0.34
	p-value	0.554	0.078	0.131	0.176	0.546	0.921	0.695	0.069	0.794	0.378	0.577
	R^2	0.42	0.70	0.76	0.51	0.13	0.00	0.06	0.72	0.03	0.26	0.11

Unlike studies by Wilby (2010) and Dunbar et al.,(2010), the results here found very little statistically significant relationships with the antecedent flow conditions. Wilby (2010) discovered strong significance with the antecedent summer flows, the results here demonstrated little significance with the antecedent 3, 6 or 9 month flows. This finding shows that results found in one river cannot necessarily be transferred to another river. However more data may increase the strength and significance of the results as strong positive and negative correlations were found.

5.3.4 Site conditions

This analysis showed which site conditions provided the optimum and poorest conditions for BMI. The site characteristics are presented in Table 5.6.

Highest scores:

- The highest scoring sites for ASPT were: KS8, 7 and 1.
- The highest scoring sites for LIFE were: KS8, 7 and 5.
- Therefore sites: 8 and 7 provide the best quality habitats for BMI. On the RHS, site 7 was classified ‘obviously modified’ scoring ‘fair’ on the HQA. Site 8 was ‘significantly modified’ with a HQA of ‘poor’. Therefore the top scoring sites do not necessarily correspond to the best quality or most natural sites.
- The top two sites providing the best quality habitat had medium gravel substrate and tree cover.

Lowest scores:

- The lowest scoring sites for ASPT were: KS2, 10 and 11.
- The lowest scoring sites for LIFE were: KS1, 10 and 11.
- Therefore KS 10 and 11 provide the worst quality habitats for BMI in relation to the other sites.
- KS 10 and 11 are the only sites in the fen section of the river and are highly modified according to the HMC and furthermore score very low on the HQA.
- The lowest scoring sites also had no in stream cover indicating cover may be beneficial for BMI populations. There is little information in literature which states whether in stream cover is positive or negative for BMI communities, however habitat heterogeneity is known to positively influence BMI (Milner et al., 2015). Therefore a variety of areas some with and some without cover would be beneficial for BMI communities.

Table 5.6- Site characteristics. Red= highest scores, Blue= lowest scores

KS no.	Substrate	Cover	Artificial features	L bank use	R bank use	Comments	RHS HQA	RHS HMC	RHS HMS	ASPT average ± SD	ASPT Order	LIFE average ± SD	LIFE Order
KS1	Silt	Non	Re-meandered	Buffalow grazing	Buffalow grazing	Heavily poached	45	4	600	6.63 ± 0.91	3	7.07 ± 0.12	9
KS2	Gravel	All trees	Non	Gardens	Farms		56	4	860	5.97 ± 0.85	9	7.4 ± 0.36	7
KS3	Gravel	Some trees	Non	Wetland	Wetland		53	4	1345	6.9 ± 0.73	6	7.61 ± 0.35	8
KS4	Gravel	All trees	Non	Wetland	Wetland		n/a	n/a	n/a	6.42 ± 1.14	7	7.74 ± 0.39	4
KS5	Medium gravel	Non	Non	Wetland	Wetland	Used for recreation	39	3	440	6.62 ± 0.28	4	7.83 ± 0.46	3
KS6	Medium gravel	All trees	Non	Forest	Forest		n/a	n/a	n/a	6.59 ± 1.18	5	7.65 ± 0.41	5
KS7	Medium gravel	All trees	Non	Gardens	Gardens/ forest		60	3	270	6.87 ± 0.93	2	7.84 ± 0.52	2
KS8	Gravel/ some sand	All trees	Non	Forest	Forest		29	4	630	7.15 ± 0.45	1	8.15 ± 0.25	1
KS9	Gravel	Some trees & overhanging veg	Non	Farms	Farms		48	4	1260	6.09 ± 0.43	8	7.48 ± 0.54	6
KS10	Sand/gravel	Non	Non	Farms	Farms		32	5	3360	5.07 ± 1.29	10	6.69 ± 0.98	10
KS11	Sand/silt	Non	DS of bridge	Cattle grazing	Open fields		12	5	3220	5.07 ± 1.22	11	6.48 ± 0.91	11

5.3.5 Summary of analysis

A summary of the measured BMI data is presented below:

ASPT: The ASPT records similar information to the BMWP which assesses biological quality in freshwater bodies and was designed primarily to summarise the effects of organic pollution on BMI communities (Section 2.5). The ASPT however accounts for potential variation in the sampling time, for example, a prolonged sampling time could produce a higher score than a sample taken quickly (NRT, 2012).

- ASPT indicates excellent water quality.
- The ASPT scores indicate BMI are not influenced by daily flows.
- There are little statistically significant relationships between ASPT and antecedent flows.

LIFE: LIFE scores are used to assess the effects of flow on the indicator species.

- No seasonal trends occur.
- BMI are exposed to relatively high flows all year, this is as the River Nar has a high base flow index (BMI) (Sear et al., 2006).
- The fastest flowing reaches and therefore best BMI communities (determined through LIFE scores) are in the middle reaches.
- The highest scores correspond with least modified sites.

<u>Conclusions</u>

- BMI quality in the River Nar is excellent.
- Summer provides the best conditions, followed by spring and autumn equally and then winter (this is the natural pattern).
- BMI do not have a strong relationship with daily flows.
- ASPT had little statistical relationship to the antecedent flows.
- LIFE exhibited some statistical relationships to the antecedent flows; however these were not similar to those found in past studies.
- Fen sections of the river provide the lowest scores of all indices.
- Natural seasonal changes of BMI populations are shown in the River Nar.
- The middle reaches have the best BMI habitat.
- Cover and medium gravel substrate provide the best environment for BMI.
- No cover, silty substrate and channel modification provide poor habitat for BMI.

5.4 Environment Agency Benthic Macro Invertebrate (BMI)

Nine sites of collected BMI data from the EA were analysed to show how the species interact to different flows.

5.4.1 *Analysis 1: Daily flow*

This analysis investigated the relationship between ASPT and LIFE with daily flow. Table 5.7 shows the correlation coefficient, R^2 and p-value results from the nine sites.

Table 5.7- Statistics results for ASPT and LIFE daily flow analysis- see Table 3.3 for colour descriptions

		ASPT			LIFE		
Site number	**Site name**	**Correlation coefficient**	**R^2**	**p-value**	**Correlation coefficient**	**R^2**	**p-value**
10	Kings Lynn	-0.08	0.01	0.657	0.00	0.00	0.998
9	Setchey	0.10	0.01	0.613	0.49	0.26	0.005
8	Highbridge	0.05	0.00	0.819	-0.20	0.04	0.342
7	Marham	0.58	0.33	0.001	0.39	0.15	0.033
6	Nar RB	0.24	0.06	0.123	0.26	0.07	0.088
5	West Acre RB	0.20	0.04	0.204	0.08	0.01	0.641
4	Castle Acre	0.23	0.05	0.241	0.21	0.04	0.291
2	Litcham	0.57	0.33	0.001	0.73	0.54	0.000
1	Mileham	0.38	0.15	0.088	0.39	0.15	0.083

- Correlation coefficients indicate the vast majority of sites have positive correlations, indicating BMI scores increase with higher flows.
- For LIFE, 6 sites out of 9 have positive correlation coefficient relationships with daily flow however for ASPT there are only 5 sites with positive correlation coefficients.
- The sites which have significant p-values for both ASPT and LIFE are Site 2 and 7. Site 2 scores very highly on the HQA and has a HMC of 4. Site 7 scored low on the HQA and also very low on the HMC. This therefore indicates that site conditions may not affect the scores.
- Only site 2 for LIFE had relationships which also had a significant p-value and a strong R^2 score.
- It is likely that for other sites the antecedent flow conditions influence the results (see section 5.4.4 for further analysis on antecedent conditions).

To conclude it cannot be proved that BMI are influenced by daily flow conditions however generally higher scores are found when there are higher daily flows.

5.4.2 *Analysis 2: Seasonal flow*

Part A

The average ASPT and LIFE scores are shown for each site in Figure 5.6.

- Sites in the mid-reaches of the river have the highest BMI index scores.
- Sites in the fen reaches have lower scores apart from Setchey which shows an anomalous result for LIFE.
- The two sites in the upper reaches of the river have extremely low scores for LIFE and are unusually lower than the fen reaches. This is likely to be as a result of individual site conditions.

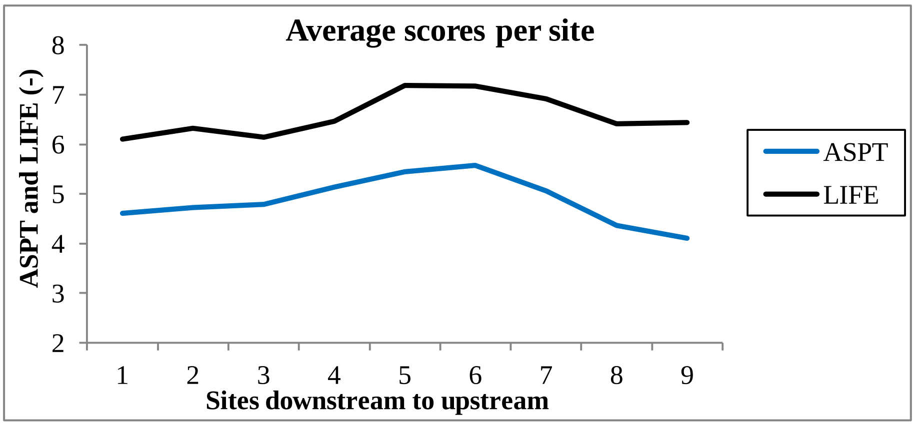


Figure 5.6- Average BMI scores per site

Part B

In this analysis scores were analysed to investigate the relationship between scores in different seasons and daily flow. The seasons were separated based on the months stated in Table 3.2. The vast majority of samples were taken in autumn and spring, with only a few in summer and winter, for this reason if a season had only three or less samples taken it was discounted from the analysis to ensure a sufficient length of series.

The trends which occurred from the results (Table 5.8) are demonstrated below:

- There is little relationship between the flow on the day of the sample and seasonal scores.
- There is predominantly positive correlation indicating scores increase with an increase in flow, however there were two ‘excellent’ negative correlations supported by significant p-values and strong R^2 values. This indicates scores decrease with an increasing flow.
- For ASPT a positive correlation was found in all seasons except summer. For LIFE scores relationships are found in all seasons.
- Site 2 had positive correlation between LIFE and flow in all seasons, the same site had correlation in all but one season for ASPT and flow. This suggests site conditions could be an important factor in determining scores. Site 2 is 45m from RHS site 2, and 270m from measured KS site 2. RHS site 2 scored highly on the

HQA however had a relatively high HMC score of 4. The measured KS at Litcham (KS2) did not score in either the top three or bottom three scoring sites and therefore exhibits average conditions.

Table 5.8- Seasonal ASPT and LIFE daily flow analysis- see Table 3.3 for colour descriptions

			ASPT			**LIFE**		
Site no	**Site name**		**Corr-Coeff**	**R^2**	**p-value**	**Corr-Coeff**	**R^2**	**p-value**
10	Kings Lynn	Winter	0.68	0.47	0.318	-0.45	0.21	0.547
		Spring	0.38	0.14	0.225	0.37	0.14	0.239
		Summer	-0.08	0.01	0.820	-0.71	0.50	0.015
		Autumn	-0.41	0.17	0.244	0.28	0.08	0.432
9	Setchey	Winter	-0.36	0.13	0.636	-0.50	0.25	0.498
		Spring	0.14	0.02	0.702	0.64	0.41	0.047
		Summer	0.21	0.04	0.792	0.40	0.16	0.600
		Autumn	-0.14	0.02	0.689	0.30	0.09	0.378
8	Highbridge	Spring	0.39	0.15	0.263	-0.30	0.09	0.404
		Autumn	-0.18	0.03	0.590	0.35	0.12	0.290
7	Marham	Spring	0.61	0.37	0.035	-0.01	0.00	0.967
		Autumn	0.54	0.29	0.071	0.75	0.56	0.005
6	Nar road bridge	Winter	-0.19	0.04	0.685	-0.47	0.22	0.288
		Spring	0.21	0.04	0.488	0.02	0.00	0.936
		Summer	0.18	0.03	0.611	0.67	0.45	0.033
		Autumn	-0.01	0.00	0.967	0.27	0.07	0.352
5	West Acre Road bridge	Winter	-0.85	0.72	0.149	-0.93	0.86	0.075
		Spring	-0.20	0.04	0.485	0.02	0.00	0.943
		Summer	-0.09	0.01	0.828	-0.02	0.00	0.955
		Autumn	0.33	0.11	0.246	0.39	0.15	0.168
4	Castle acre road bridge	Spring	0.11	0.01	0.758	0.03	0.00	0.922
		Autumn	0.02	0.00	0.964	-0.13	0.02	0.708
2	Litcham	Winter	0.04	0.00	0.956	0.39	0.21	0.099
		Spring	0.55	0.31	0.050	0.72	0.52	0.006
		Autumn	0.46	0.21	0.099	0.63	0.40	0.015
1	Mileham	Spring	0.23	0.05	0.520	0.11	0.01	0.760
		Autumn	0.32	0.10	0.395	0.89	0.80	0.001

5.4.3 Analysis 3: Antecedent flow

The results presented here are a summary of Visser's (2014) work which has subsequently been published in Visser (2016). The conclusions drawn are then used for further analysis in the project. Preliminary analysis indicated there was a time lagged effect of LIFE scores with flow therefore the yearly conditions did not influence the LIFE scores, see Figure 5.7. The work by Visser (2014) therefore aimed to investigate this time lagged effect by 'Developing a model relating antecedent low flows and macro-invertebrate health (using LIFE) in the River Nar'. Methods were developed using two other major studies on antecedent flow-LIFE relationships as a basis: Generalised LIFE Response Curves (EA 2005b) and DRIED-UP (Dunbar et al., 2006; Dunbar and Mould 2009). See section 3.2.3 for further explanation of the methods used.

Previous methods (e.g. Wilby 2010 on the River Itchen) used linear regression modelling, including winter flows, to show relationships between LIFE scores in spring and low flows in preceding summer. An empirical relationship was found between invertebrate status and antecedent flows. This method was applied to the River Nar, see Section 3.2.3 for full details on the method carried out.

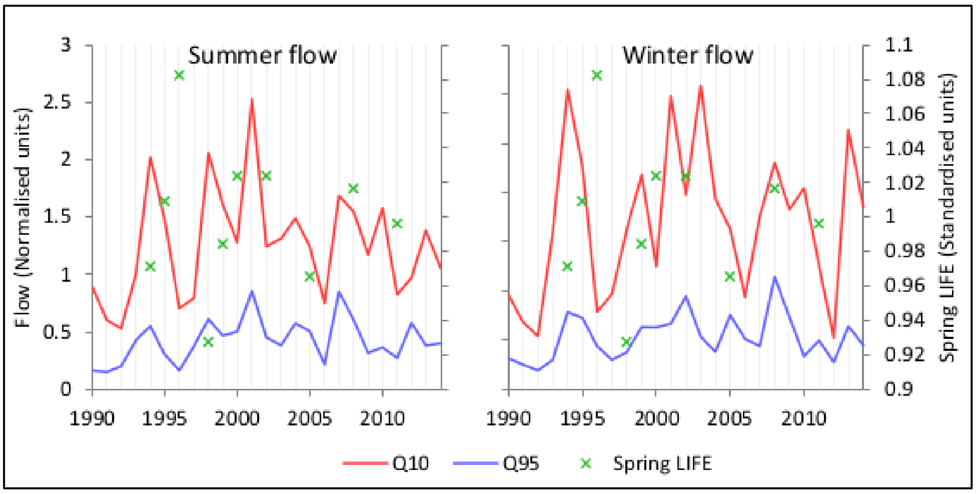


Figure 5.7- Time lagged effect of LIFE scores (Visser et al., 2016)

Key findings from the assessment of the chalk river are detailed below; refer to table 3.9 and 3.10 on pages 84 and 85 respectively for details on the models and variable combinations used:

<u>Spring LIFE</u>

- Models 1-4 which included only 1 variable showed that the summer flows had the strongest relationships with the LIFE scores, i.e. summer $Q_{95/10}$ had the largest influence on spring LIFE scores.
- Models 1 and 2 investigating summer flows had significant p-value (p=<0.05) and strong R^2 values (>0.5) for variables including the previous 4 years.
- Models 3 and 4 investigating winter flows had much weaker relationships.
- Models 5-10 included combinations of variables. Any variable which included summer $Q_{10/95}$ had stronger R^2 and p-value results.
- Models 5, 7, 8, 9 and 10 which included more than 4 years of antecedent flows were not significant but did show an increasing R^2 value up to the 4 years of antecedent flow.

Overall for Spring LIFE, the combinations which had the most significant results were: Summer Q_{10} and Winter Q_{95} with 1 variable and Winter Q_{95} plus Summer Q_{10} with 2 variables. The models showing most significance had two years of antecedent flows, the analysis carried out for this thesis only included up to 9 months antecedent flow, this could be a reason why the LIFE scores did not show as much significance.

Autumn LIFE

Similar results were found for the autumn LIFE scores as for the Spring LIFE score analysis:

- Models 1-4 suggested the summer flows had the strongest relationships and there was no significance with the winter flows in any of the models.
- Model 6 which included two winter variables had no significance.
- Models 5, 7, 8, 9 and 10 showed no significance when more than 4 years antecedent flow were taken into account.
- Models including up to 4 years antecedent flow conditions gave the largest R^2 results but models which included the preceding 3 years had no significance. This indicates there may be issues with the models including the antecedent 4 year conditions.

Overall for Autumn LIFE, the combinations which had the most significant results were: Summer Q_{10} and Summer Q_{95} with 1 variable and Summer Q_{95} plus Summer Q_{10} with 2 variables.

Overall the results indicated that BMI had a lagged response to flows. This suggested that flows would be best considered continuously rather than as individual events in management and planning. The models showed that summer high and low flows are the most critical in sustaining BMI health. This shows that any management decisions should take into account this lagged response and should provide protection for summer low flows and moreover that multiannual relationships should be considered in habitat models. The results have shown that consideration of a broader temporal scale is likely to result in a more accurate approach to environmental flows. This said however the results also indicated that a unique modelling procedure is required for individual rivers for adequate management and that results cannot necessarily be transferred across rivers (Visser et al., 2016).

5.4.4 Summary

- Mid-stream reaches provide the best habitats for BMI.
- Daily flow and ASPT and LIFE are not related.
- Seasonal flows and ASPT and LIFE are related.
- Low flows do not result in low scores.
- The results indicated that BMI had a lagged response to flows.
- Summer high and low flows are the most important for sustaining BMI health.

5.5 Measured macrophytes

Macrophyte surveys took place at nine locations along the river from Highbridge (downstream, Site 1), to Mileham (upstream Site 9), (Figure 3.5). These results were used to analyse the current status of macrophytes in the river and to show a year's cycle of *Ranunculus*. Furthermore the results show how *Ranunculus* responds to different flows. An example of a time series of photos from the same site is shown in Figure 5.8.

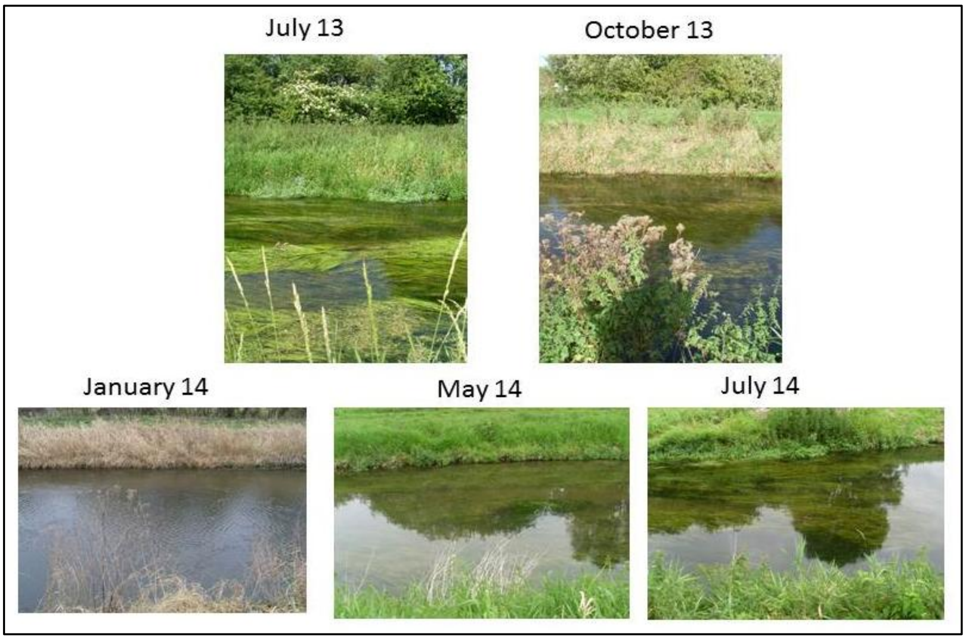


Figure 5.8- Time series of macrophyte photos at site 2

5.5.1 Data removal and raw results

Raw results of all seasons and sites are presented in Table 5.9.

Table 5.9- Macrophyte abundance results per site and season

		S1	S2	S3	S4	S5	S6	S7	S8	S9
Jul-13	**Parsnip**	6	0	1	1	2	1	5	2	0
	Reeds	2	2	8	0	1	5	6	0	0
	Crowfoot	1	8	6	7	1	4	3	0	0
	Starwort	0	0	1	1	1	1	2	2	0
Oct-13	**Parsnip**	1	5	3	0	0	3	3	2	1
	Reeds	1	1	6	0	2	4	2	0	0
	Crowfoot	0	3	0	0	2	0	0	0	0
	Starwort	1	0	0	0	2	2	3	1	0
Jan-14	**Parsnip**	1	2	2	0	0	1	1	2	n/a
	Reeds	1	1	0	0	0	0	0	0	n/a
	Crowfoot	0	0	0	0	0	0	1	1	n/a
	Starwort	0	0	0	1	2	1	1	0	n/a
May-14	**Parsnip**	2	2	2	0	0	1	3	3	6
	Reeds	2	3	5	2	1	2	2	0	0
	Crowfoot	0	1	2	3	1	0	2	0	0
	Starwort	0	0	1	0	2	0	0	0	0
Jul-14	**Parsnip**	2	5	3	4	2	2	4	2	n/a
	Reeds	4	4	4	0	1	4	2	0	n/a
	Crowfoot	0	4	2	4	0	1	5	0	n/a
	Starwort	3	0	0	2	1	1	3	0	n/a

% categories	
1	≤0.1%
2	0.1-1%
3	1-2.5%
4	2.5-5%
5	5-10%
6	10-25%
7	25-50%
8	50-75%
9	≥75%

- Site 9 could not be accessed in January due to due to buffalo and calves grazing on site. therefore being too dangerous to enter.
- Upon inspection of site 9 in July 2014, the river had been diverted to a different course as part of the Norfolk Rivers Trust restoration plans during June 2014.
- Table 5.13 demonstrates the raw findings from the macrophyte surveys with a key to the numbers. For example in July 2013 at site 1, Narrow-leaved-water parsnip (*Berula erecta*) occurred in 10-25% of the 100m channel and Crowfoot (*Ranunculus Fluitans)* occurred in less than 0.1% of the channel.

5.5.2 Analysis 1- All species daily flow

The first analysis carried out showed total abundance of all recorded macrophytes per season, per site and relating these abundance levels to daily flow levels. The results are described; refer to Figure 5.9 and Table 5.10:

- There were very small amounts of macrophytes in January at all sites.
- The highest flows correspond with the lowest abundances in winter (Figure 5.9), despite this there is little relationship between macrophyte abundance and daily flow (Table 5.10).
- 84% of the results for all sites and species had negative correlations between flow and abundance, 77% of which were ‘good’ or ‘excellent’ negative correlation, this indicated there was a trend of higher abundances associated with drier daily conditions. However there is low R^2 and insignificant p-values to support the accuracy of this correlation. The reasons for this are likely related to the antecedent flows having a greater impact on them than the daily flows (Section 5.5.4) (Cranston and Darby 2004).
- There was one strong positive correlation with strong R^2 and significant p-values to prove it; this was for Crowfoot in site 8. This can however be discounted as the regression is only based on one data point as at this site Crowfoot was only found in one month (January).

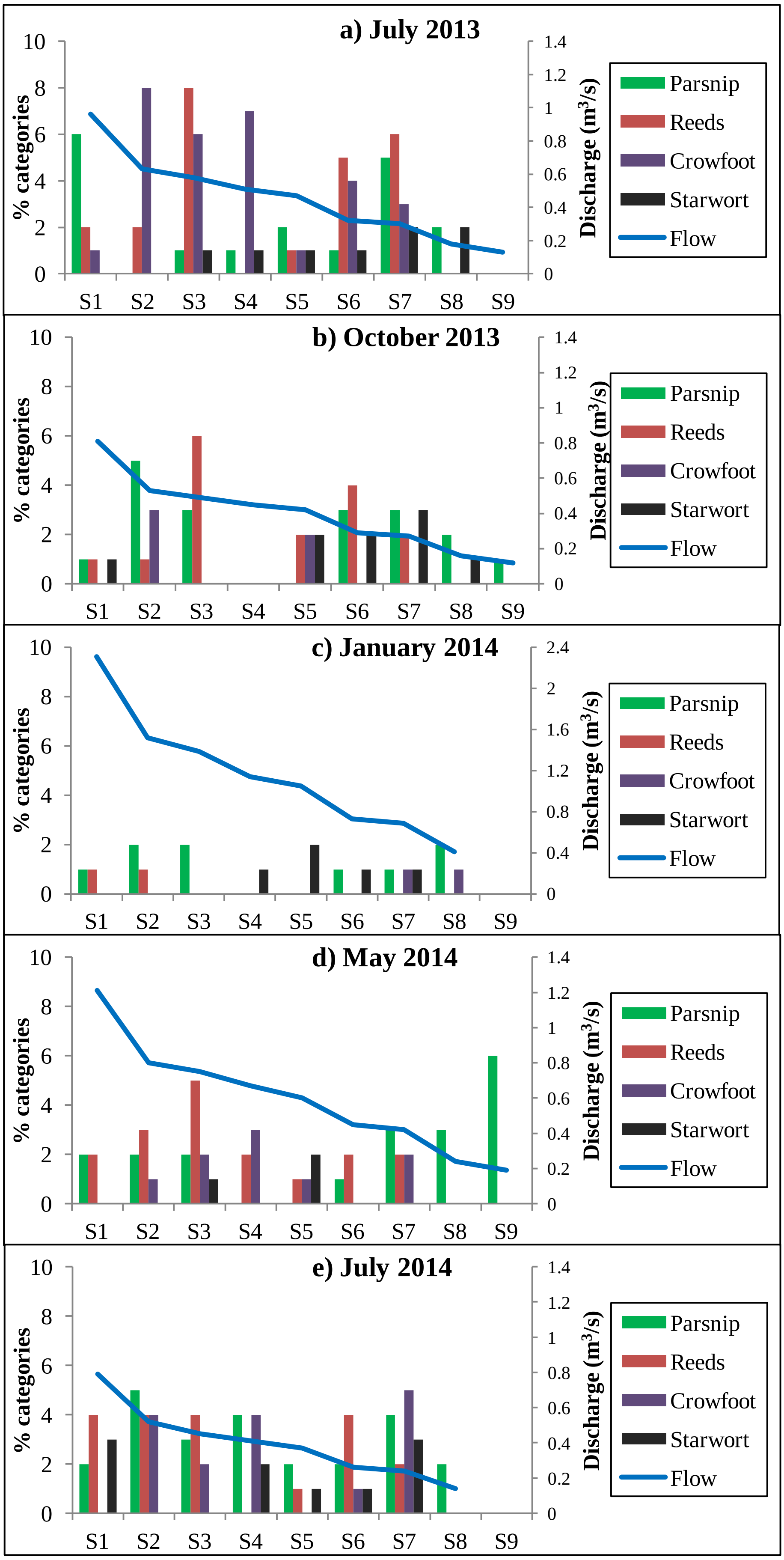


Figure 5.9- Seasonal macrophyte abundances related to flow

Table 5.10- Statistical regression analysis for macrophyte abundances

Site	Species	Correlation coefficient	R^2	p- value
1	**Parsnip**	-0.33	0.11	0.593
	Reeds	-0.50	0.25	0.396
	Crowfoot	-0.23	0.05	0.715
	Starwort	-0.51	0.26	0.381
2	**Parsnip**	-0.36	0.13	0.550
	Reeds	-0.46	0.21	0.432
	Crowfoot	-0.63	0.40	0.253
3	**Parsnip**	-0.29	0.08	0.638
	Reeds	-0.81	0.66	0.096
	Crowfoot	-0.38	0.15	0.527
	Starwort	-0.16	0.03	0.799
4	**Parsnip**	-0.50	0.25	0.393
	Reeds	0.06	0.00	0.919
	Crowfoot	-0.48	0.23	0.415
	Starwort	-0.07	0.01	0.906
5	**Parsnip**	-0.54	0.29	0.350
	Reeds	-0.81	0.65	0.097
	Crowfoot	-0.45	0.21	0.443
	Starwort	0.54	0.29	0.350
6	**Parsnip**	-0.56	0.32	0.321
	Reeds	-0.94	0.88	0.018
	Crowfoot	-0.38	0.15	0.526
	Starwort	-0.29	0.09	0.632
7	**Parsnip**	-0.85	0.71	0.071
	Reeds	-0.59	0.34	0.298
	Crowfoot	-0.42	0.18	0.482
	Starwort	-0.66	0.44	0.224
8	**Parsnip**	0.07	0.01	0.909
	Crowfoot	0.94	0.88	0.018
	Starwort	-0.40	0.16	0.501
9	**Parsnip**	0.96	0.92	0.184

5.5.3 Analysis 2- All species seasonal change and site conditions

This analysis aimed to show how macrophyte abundance changed during seasons and to relate this to site conditions. The results are presented in Table 5.11 and the findings are discussed below:

- For many sites, macrophyte abundance was significantly reduced during winter, for example at site 2; Crowfoot is highly abundant in all months apart from January where there is 0% Crowfoot.
- Sites 5, 8 and 9 had the lowest macrophyte abundances (shown in blue in Table 5.11).
    - Site 9 has the least abundance through all seasons. This is likely due to the channel being recently re-meandered and it is also heavily poached therefore limiting macrophyte growth. Furthermore the channel width is very small and wide channels are associated with high macrophyte growth (Westwood et al., 2006).

  - At site 8 the channel is very shallow which restricts growth. However site 8 did score very highly on the HQA indicating good habitat quality, therefore low macrophyte abundances would not be expected.

- Sites 2, 3 and 7 had the highest abundances (shown in red in Table 5.13):

- Site 2 and 3 are very similar in characteristics, with clearly realigned wide channels and both with gravel substrate
- Site 7's channel width is relatively small and therefore high abundances are unusual, however this is one of the most natural sections with very little modification or anthropogenic influence, therefore allowing natural flows and processes to occur.

Table 5.11- Site conditions for macrophytes. Blue= lowest abundances, Red= highest abundances

Site no	Name	Substrate	Cover	Artificial features	L bank use	R bank use	Channel width (m)	Average depth (m)	Comments	RHS HMC	RHS HQA
1	Highbridge	Sand/silt	Non	DS of bridge	Cattle grazing	Open fields	10.5	0.93	High canalisation	5	12
2	Marham	Sand/ gravel	Non	Non	Farms	Farms	9	1.16		5	32
3	DS Nar	Gravel	Some trees & overhanging veg	Non	Farms	Farms	8	0.62		4	48
4	US Nar	Gravel/ some sand	All trees	Non	Forest	Forest	9	0.3		4	29
5	W acre	Medium gravel	All trees	Non	Forest	Forest	7	0.54		n/a	n/a
6	C acre	Gravel	All trees	Non	Wetland	Wetland	5	0.27		n/a	n/a
7	Lexham	Gravel	Some trees	Non	Wetland	Wetland	4	0.5		4	53
8	Litcham	Gravel	All trees	Non	Gardens	Farms	5	0.15		4	56
9	Mileham	Silt	Non	Re meandered	Buffalo grazing	Buffalo grazing	0.5	0.1	Heavily poached	4	45

5.5.4 Analysis 3- Ranunculus daily and antecedent flow

This analysis assessed *Ranunculus* abundance in relation to different sites, seasons and flows, the results are presented in Figure 5.10.

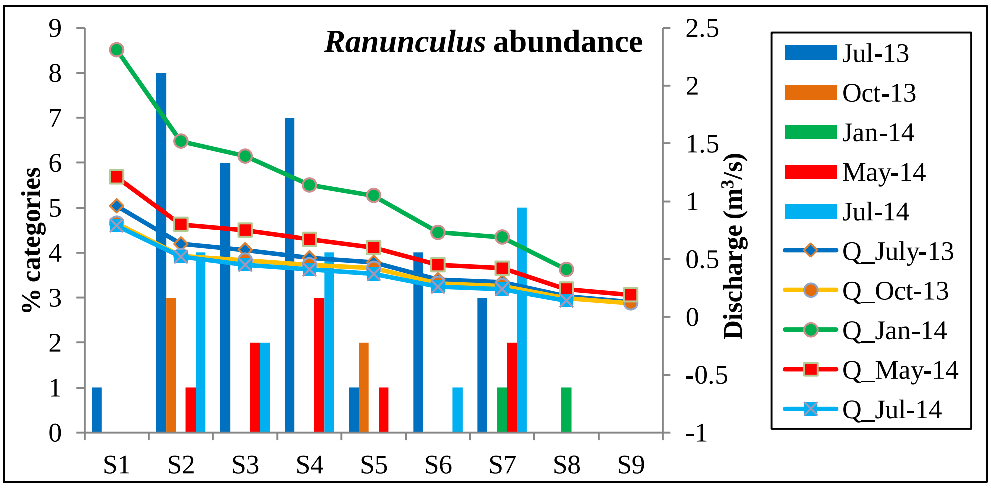


Figure 5.10- *Ranunculus* abundance at each site in each season related to flow

- The highest flows are associated with the lowest abundance. The low abundance cannot necessarily be related to the flow however due to natural growth and die back (Dawson 2002). Furthermore due to growth rates being relatively slow there is often a lag effect and periods of hydrological stability are required to sustain macrophyte levels. (Franklin et al., 2008). Thus the higher flows in winter sustain the macrophyte abundances throughout the year. However if flows are too high, this can also be damaging to macrophytes.
- The highest abundances of *Ranunculus* in all sites apart from site 5 and 7 were recorded in July 2013 which exhibited fairly average flow conditions. Low flows during summer have been shown to be detrimental to *Ranunculus* growth (Wilby et al., 1998)
- The lowest flows were recorded in July 2014; however these had the second highest abundance levels. This finding backs up the knowledge that once flows are too low, it becomes negative for macrophyte growth (Franklin et al., 2008).
- All results gave negative correlation between daily flow and *Ranunculus* abundance (Table 5.12), which indicates higher abundances are associated with lower flows.
- Site 2 in particular showed high negative correlation between flow and *Ranunculus*. Despite this however it cannot be proved by a significant p-value or strong R^2 value.

Table 5.12- *Ranunculus* ststistics for daily and antecedent conditions. Site 9 removed (no *Ranunculus*), sites 8 and 1 removed (only 1 month with *Ranunculus*)

		Site 7	Site 6	Site 5	Site 4	Site 3	Site 2
Daily flow	C- coeff	-0.42	-0.38	-0.45	-0.48	-0.38	-0.63
	p-value	0.482	0.526	0.443	0.415	0.527	0.253
	R^2	0.18	0.15	0.21	0.23	0.15	0.40
3 months Q_{10}	C- coeff	0.31	0.21	-0.06	0.56	0.52	0.07
	p-value	0.695	0.733	0.922	0.324	0.369	0.910
	R^2	0.09	0.04	0.00	0.32	0.27	0.01
3 months Q_{50}	C- coeff	0.43	0.36	-0.16	0.69	0.64	0.20
	p-Value	0.570	0.553	0.801	0.196	0.242	0.743
	R^2	0.19	0.13	0.02	0.48	0.41	0.04
3 months Q_{90}	C- coeff	0.26	0.10	0.00	0.47	0.42	-0.01
	p-value	0.737	0.875	0.997	0.423	0.479	0.983
	R^2	0.07	0.01	0.00	0.22	0.18	0.00
6 months Q_{10}	C- coeff	0.63	0.80	0.09	0.76	0.72	0.94
	p-value	0.366	0.104	0.882	0.135	0.171	0.015
	R^2	0.40	0.64	0.01	0.58	0.52	0.89
6 months Q_{50}	C- coeff	0.51	0.87	0.21	0.79	0.79	0.99
	p-value	0.488	0.054	0.733	0.115	0.111	0.001
	R^2	0.26	0.76	0.04	0.62	0.63	0.98
6 months Q_{90}	C- coeff	0.71	0.93	0.04	0.85	0.85	0.98
	p-value	0.290	0.021	0.946	0.069	0.067	0.002
	R^2	0.50	0.87	0.00	0.72	0.73	0.97
9 months Q_{10}	C- coeff	-0.72	0.49	0.57	0.69	0.26	0.56
	p-value	0.277	0.399	0.312	0.913	0.677	0.324
	R^2	0.52	0.24	0.33	0.00	0.07	0.32
9 months Q_{50}	C- coeff	-0.71	-0.15	0.61	-0.53	-0.40	0.03
	p-value	0.290	0.814	0.270	0.354	0.509	0.961
	R^2	0.50	0.02	0.38	0.29	0.16	0.00
9 months Q_{90}	C- coeff	-0.71	-0.25	0.73	-0.56	-0.45	-0.02
	p-value	0.293	0.686	0.162	0.324	0.451	0.974
	R^2	0.50	0.06	0.53	0.32	0.20	0.00

- There is a strong negative correlation between antecedent Q_{50} and abundance however the relationship between antecedent Q_{50} and *Ranunculus* abundance is very weak (evidenced by R^2 and p-value).
- The highest relationship is with the 6 month antecedent flow where all sites except Site 5 had strong correlation coefficients and strong R^2 values, however only Site 2 had p-value significance. It is likely that with more data points the p-value would show a higher significance.
- The highest abundances were in summer 2013 which had average flows, and the second highest abundances were in summer 2014 which had the lowest flows.

- Summer 2013 had a higher 6 month antecedent Q_{50} flow.
- Summer 2014 had a lower 6 month antecedent Q_{50} flow.
- This would explain the large differences in the abundance amounts, and indicates that the 6 month Q_{50} antecedent conditions do impact upon abundance amount.

- The lowest abundances were in winter where the 6 month antecedent flow conditions were very dry, this is related to the natural lifecycle (Dawson 2002).
- Ultimately the natural fluctuations and growth patterns of macrophytes are demonstrated, however it is clear that lower abundances are found when there are drier 6 month antecedent flows. This is proven by the fact that summer 2014 had much lower abundances than summer 2013.

5.5.5 Summary

The highest abundances were found during the lowest flow period, with a preceding high flow in winter. The lowest abundances were found during a high flow period with low flow preceding 6 months. Therefore whilst dry periods provide the best conditions, the whole cycle needs to be taken into account. For example, the winter of 2013/14 had slightly drier conditions than winter 2012/13; this resulted in lower abundances in the summer time.

Analysis 1:

- Macrophyte abundance is not related to daily flow conditions despite a trend of higher abundances during drier conditions and vice versa.
- Change in abundance is more related to natural fluctuations and growth throughout the year.

Analysis 2:

- The lowest abundances were found in: highly modified areas with narrow channels, silt substrate and heavy poaching
- The highest abundances were found in: wide channels (even if artificially modified), gravel substrate with little anthropogenic use. These sites scored 5 and 4 on HMC which indicates modified areas actually create good habitat for macrophytes.

Analysis 3:

- *Ranunculus* abundance is not related to daily flows
- *Ranunculus* has the best growth when there are drier conditions with wetter conditions 6 months previous which is as would be expected during summer months.
- 6 month antecedent flow conditions affect *Ranunculus* growth, wetter conditions 6 months previous create higher abundances
- In order for these results to be reliable however, more values would need to be accounted for as this is only a snapshot year.
- Site conditions play a large part in *Ranunculus* growth.

- The importance of natural flow regime has been presented in these finding.

5.6 Environment Agency macrophyte data

The EA provided a limited amount of macrophyte data therefore detailed historical analysis as with BMI data was not possible. The data was analysed to show how different flows correspond to different abundance amounts. See Figure 3.5 for location of the sites. Results are shown in Figure 5.11.

5.6.1 Total abundance and flow

- There was a large variety of macrophytes found at the four sites.
- At the West Acre road bridge more varieties and a higher abundance were found in 2006 compared to 2002.
- At the West Lexham Road Bridge more species and higher abundances were found in 2003 than in 2002
- At the Narborough road bridge and Setchey road bridge more species were found in 2002 however higher abundances were found in 2003
- The EA have not classified *Ranunculus Fluitans* as a separate species, and therefore it was assumed '*Ranunculus*' encompassed this species. This was only found at the West Acre road bridge and the west Lexham road bridge which are the 2 most upstream sites.
- *Ranunculus* was not found at the Narborough road bridge which is approximately 500 meters upstream of the measured site 3 where *Ranunculus* was found in July 2013, May 2014 and July 2014, with 7, 2 and 2 categories of abundance (see Table 3.7) respectively. This indicates the localised conditions could be different despite it only being 500m away, for example the substrate and cover could be unsupportive of the species.
- 2003 had on average higher abundances than 2002. 2003 was drier than 2002, both had preceding wet winters however 2003 was wetter than 2002. This shows wetter antecedent conditions provide higher abundances, which is as was found in the measured macrophyte data.

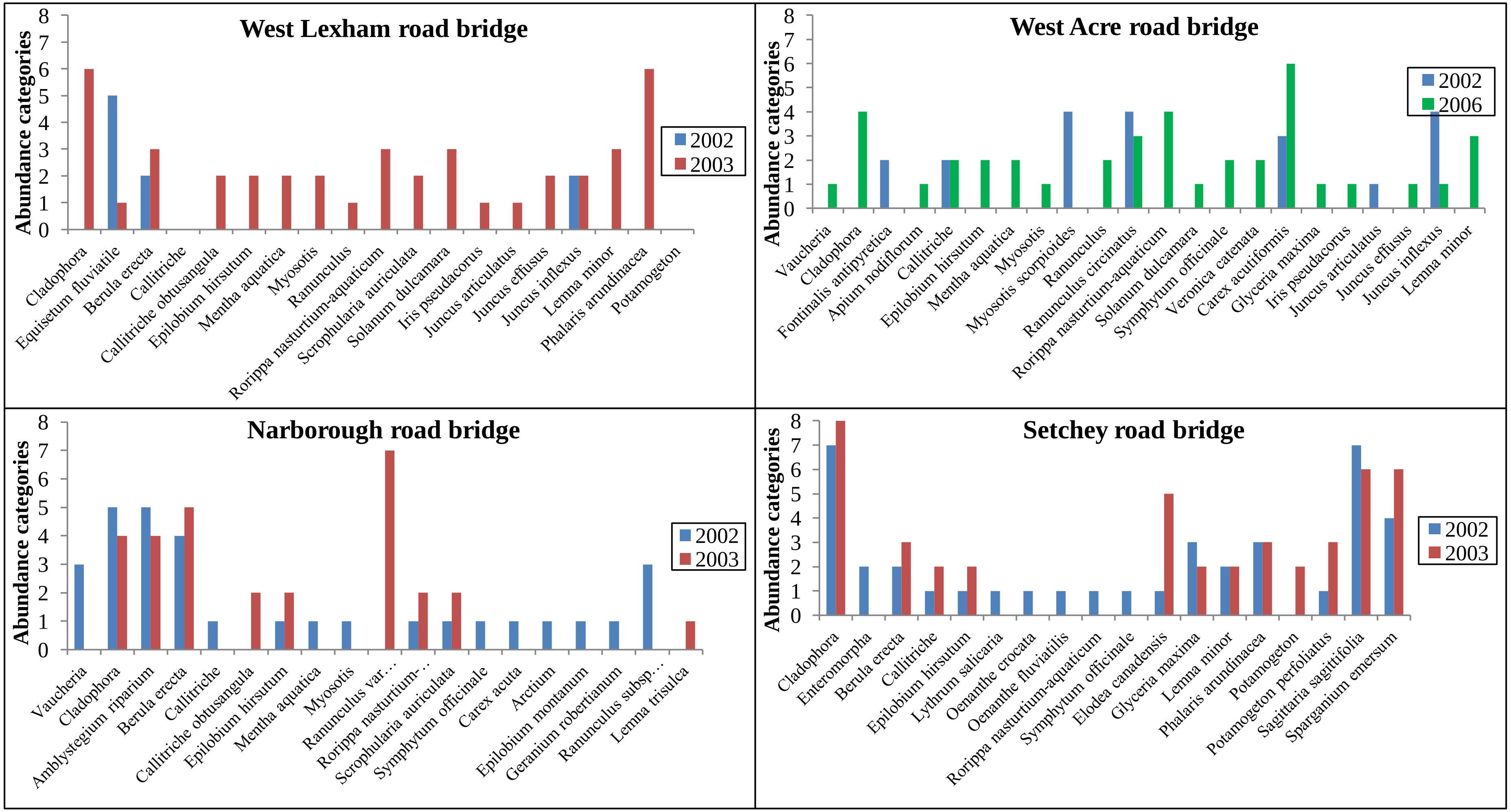


Figure 5.11- EA *Ranunculus* abundance results (see Table 3.7 for abundance scale)

5.6.2 Summary of analysis

- Localised conditions are very important for macrophytes, particularly *Ranunculus*
- Wetter antecedent conditions for summer growth provide higher abundances.

5.7 Brown trout

The final indicator species is brown trout (*Salmo Trutta*). In order to investigate a relationship between brown trout and low flows, electro-fishing data from the EA was used. Two main areas of analysis took place, firstly correlation analysis with daily flow was calculated and secondly correlation analysis with antecedent flow condition was calculated. This showed how brown trout populations are influenced by different flows.

5.7.1 Brown trout population data

The data provided by the EA is demonstrated in Table 5.13. Brown trout were found at seven sites on the river; generally the most downstream sites in the fen reach had no recorded brown trout population. This is due to the chalk stream reaches providing good habitat quality for the species (Berrie 1992). Figure 5.12 shows the change in population of brown trout from the beginning of records in 1989.

Table 5.13- Brown trout electro-fishing raw data

Site no	Site name	Date	Brown trout no	Site no	Site name	Date	Brown trout no
1	East Lexham	(12/03/2012)	4	4	Manor farm	(13/11/1989)	57
		(22/03/2007)	7			(15/03/1993)	77
		(06/04/2010)	5			(22/01/1996)	182
2	West Lexham	(09/11/1989)	91			(22/03/2007)	40
		(11/03/1993)	3			(26/03/2010)	35
		(18/01/1996)	279			(23/03/2011)	70
		(22/03/2007)	21			(14/03/2012)	69
		(06/04/2010)	20			(05/04/2013)	60
		(12/03/2012)	7	5	Warren farm	(14/11/1989)	47
3	Castle Acre	(10/11/1989)	69			(12/03/1993)	45
		(10/03/1993)	4			(23/01/1996)	82
		(19/01/1996)	214			(29/03/2007)	51
		(12/03/2003)	82			(07/04/2010)	14
		(18/03/2004)	56			(29/03/2012)	36
		(18/03/2005)	78	6	Narford Hall	(19/01/1990)	53
		(10/03/2006)	170			(17/03/1993)	34
		(19/03/2007)	130			(31/01/1996)	149
		(13/03/2008)	220			(02/04/2007)	63
		(17/03/2009)	205			(09/04/2010)	12
		(24/03/2010)	131			(21/03/2012)	45
		(22/03/2011)	77	7	Marham intake	(15/11/1989)	5
		(13/03/2012)	124			(24/03/1993)	0
		(25/03/2013)	112			(25/01/1996)	12
						(12/04/2010)	12
						(02/04/2012)	10

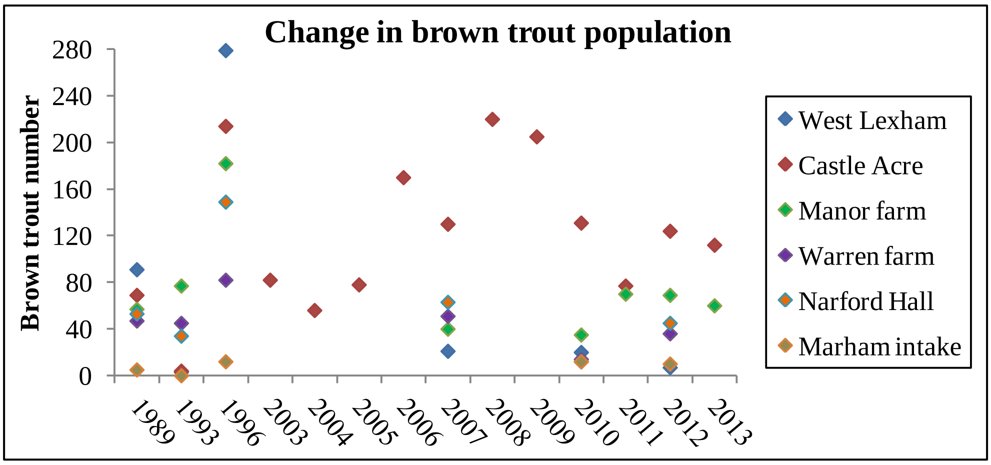


Figure 5.12- Change in brown trout population from 1989-2013 at each recorded sites

The initial findings indicate that:

- Brown trout are highly abundant in the river with a maximum finding of 278 brown trout found in 1996 at West Lexham.
- 1996 had the largest populations.
- 1993 and 2010 had the lowest populations.
- At 4 sites (out of 6), populations of brown trout have decreased since first recorded in 1989.

Overall however there is a rich abundance of brown trout in the river. Furthermore brown trout are not stocked in the river and are therefore all wild brown trout, this indicates habitat conditions are good for the species, particularly in the chalk stream reaches.

5.7.2 Analysis 1: Brown trout and daily flow

This analysis aimed to show if brown trout abundance has any relationship with the daily flow conditions. Correlation coefficients and regression analysis was carried out to investigate this.

Limited studies have been carried out on relationships of flow with fish abundance despite velocity and depth preferences being well understood. Therefore it is unknown whether there is likely to be any relationship between daily flows and brown trout abundances. Flow, velocity and depth are all known to influence habitat availability of brown trout (Armstrong et al., 2003), it is important however to investigate whether this is a direct effect or whether there is a timed lag effect of habitat availability i.e. the antecedent conditions influencing the species more than the daily.

Table 5.14- Brown trout daily flow analysis

Site number	Site name	Correlation coefficient	R^2	P-value
1	**East Lexham**	0.66	0.43	0.545
2	**West Lexham**	-0.45	0.20	0.370
3	**Castle Acre**	0.05	0.00	0.853
4	**Manor farm**	-0.51	0.26	0.193
5	**Warren farm**	-0.53	0.28	0.278
6	**Narford Hall**	-0.52	0.27	0.278
7	**Marham**	0.11	0.01	0.857
	Combined	-0.32	0.10	0.013

Table 5.14 demonstrates the results from the regression analysis, (see Tables 3.3 and 3.4 for colour descriptions). The main findings are detailed below:

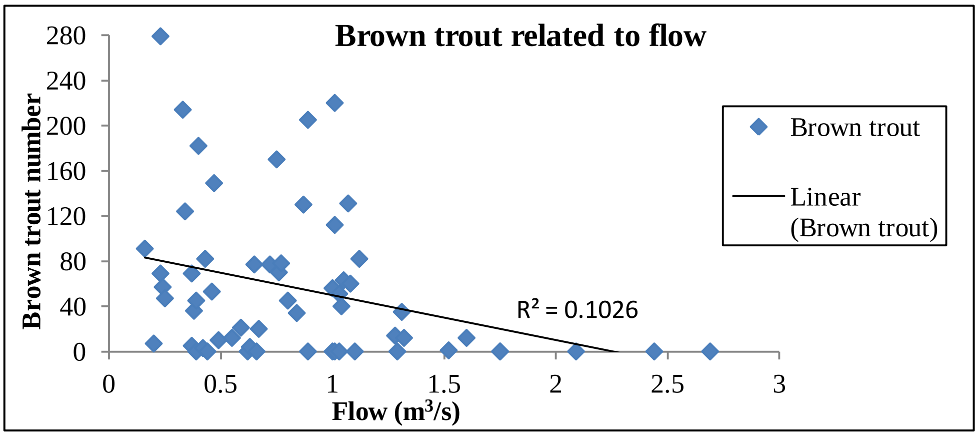


Figure 5.13- Brown trout populations corresponding to increasing flow

- Three sites had 'good negative correlation' and 1 had 'weak negative correlation', this finding indicates higher populations of brown trout are found during lower flows.
- However this relationship between daily flow and brown trout populations is not significant and is a poor strength (evidenced by R^2 and p-value).
- Interestingly when the results from each site were pooled, significant relationships were determined (p-value= 0.013), this finding suggests that if more data were available, more significant relationships would be found.
- Figure 5.12 further demonstrates how when pooled from all sites, higher populations are found in lower flows, particularly in flows under 1m^3/s.
- Site 1 was the only site with positive correlation, this site however only had 3 years of recorded data and thus data was limited.

5.7.3 Analysis 2: Brown trout and antecedent flow

This analysis aimed to investigate relationships between brown trout populations and antecedent flow conditions. The analysis is split into regression analysis (part A) and multiple regression analysis (part B).

Part A- Regression analysis

The first area of analysis investigated relationships between brown trout and antecedent flow conditions based on 1 variable i.e. antecedent summer Q_{10} flows. Six combinations (A-F) of antecedent flows were determined i.e. *T-5* (5 years previous antecedent flow). The combinations, model variables and results are demonstrated in Table 5.15.

Table 5.15- Regression analysis for brown trout

							M1		M2		M3		M4	
							Summer Q_{10}		Summer Q_{95}		Winter Q_{10}		Winter Q_{95}	
	Combinations						R^2	p-val	R^2	p-val	R^2	p-val	R^2	p-val
A	T0						0.09	0.063	0.14	0.017	0.02	0.393	0.17	0.008
B	T0	T-1					0.18	0.026	0.20	0.014	0.09	0.161	0.26	0.004
C	T0	T-1	T-2				0.18	0.066	0.22	0.029	0.14	0.149	0.27	0.009
D	T0	T-1	T-2	T-3			0.51	0.000	0.46	0.001	0.22	0.087	0.30	0.020
E	T0	T-1	T-2	T-3	T-4		0.52	0.000	0.48	0.001	0.22	0.178	0.32	0.039
F	T0	T-1	T-2	T-3	T-4	T-5	0.57	0.002	0.56	0.002	0.25	0.310	0.46	0.018

- ***Model 3*** showed there are no significant relationships between brown trout populations and antecedent winter Q_{10} (high) flows.
- ***Model 4*** indicated significant relationships between brown trout populations and antecedent winter Q_{95} (low) flows; however these were not strong relationships according to the R^2 values.
- Like model 4, ***model 2*** showed significant relationships between antecedent summer Q_{95} flows and brown trout populations, one of the combinations (F) was a strong relationship. This combination had the most amounts of antecedent conditions i.e. *T-5*; this therefore suggests stronger relationships are found when more antecedent conditions are taken into account.
- ***Model 1*** had strong significant relationships in 3 of the combinations (D, E and F); likewise with the finding from model 3, this indicates that stronger and more significant relationships are found between antecedent flows and brown trout populations when more antecedent conditions are taken into account.
- Overall the most antecedent significant flows influencing brown trout populations are summer Q_{95} and Q_{10} flows. And the most significant combinations occur when *T-5* conditions are taken into account. The winter Q_{95} also had significant relationships however were not supported by strong R^2 values.
- Including more antecedent flow values can also increase the likelihood of error which should be taken into account (Visser et al., 2016).

Part B- Multiple regression analysis

In addition to investigating relationships with 1 antecedent variable, relationships were investigated between brown trout populations and 2 antecedent flow variables i.e. Summer Q_{10} and Summer Q_{95}. Six models (M5-M10) were used with different variables (Table 5.16). 36 combinations of different yearly antecedent conditions were investigated (Table 5.17).

Table 5.16- Variables used in multiple regression analysis for brown trout

	Variable 1	Variable 2
M5	Summer Q10	Summer Q95
M6	Winter Q10	Winter Q95
M7	Winter Q10	Summer Q95
M8	Winter Q95	Summer Q10
M9	Winter Q95	Summer Q95
M10	Winter Q10	Summer Q10

Table 5.17- Combinations used in multiple regression analysis for brown trout

	Variable 1 combination						Variable 2 combination					
A	T0						T0					
B	T0	T-1					T0					
C	T0						T0	T-1				
D	T0	T-1					T0	T-1				
E	T0	T-1	T-2				T0					
F	T0						T0	T-1	T-2			
G	T0	T-1	T-2				T0	T-1				
H	T0	T-1					T0	T-1	T-2			
I	T0	T-1	T-2				T0	T-1	T-2			
J	T0	T-1	T-2	T-3			T0					
K	T0						T0	T-1	T-2	T-3		
L	T0	T-1	T-2	T-3			T0	T-1				
M	T0	T-1					T0	T-1	T-2	T-3		
N	T0	T-1	T-2	T-3			T0	T-1	T-2			
O	T0	T-1	T-2				T0	T-1	T-2	T-3		
P	T0	T-1	T-2	T-3			T0	T-1	T-2	T-3		
Q	T0	T-1	T-2	T-3	T-4		T0					
R	T0						T0	T-1	T-2	T-3	T-4	
S	T0	T-1	T-2	T-3	T-4		T0	T-1				
T	T0	T-1					T0	T-1	T-2	T-3	T-4	
U	T0	T-1	T-2	T-3	T-4		T0	T-1	T-2			
V	T0	T-1	T-2				T0	T-1	T-2	T-3	T-4	
W	T0	T-1	T-2	T-3	T-4		T0	T-1	T-2	T-3		
X	T0	T-1	T-2	T-3			T0	T-1	T-2	T-3	T-4	
Y	T0	T-1	T-2	T-3	T-4		T0	T-1	T-2	T-3	T-4	
Z	T0	T-1	T-2	T-3	T-4	T-5	T0					
AA	T0						T0	T-1	T-2	T-3	T-4	T-5
BB	T0	T-1	T-2	T-3	T-4	T-5	T0	T-1				
CC	T0	T-1					T0	T-1	T-2	T-3	T-4	T-5
DD	T0	T-1	T-2	T-3	T-4	T-5	T0	T-1	T-2			
EE	T0	T-1	T-2				T0	T-1	T-2	T-3	T-4	T-5
FF	T0	T-1	T-2	T-3	T-4	T-5	T0	T-1	T-2	T-3		
GG	T0	T-1	T-2	T-3			T0	T-1	T-2	T-3	T-4	T-5
HH	T0	T-1	T-2	T-3	T-4	T-5	T0	T-1	T-2	T-3	T-4	
II	T0	T-1	T-2	T-3	T-4		T0	T-1	T-2	T-3	T-4	T-5
JJ	T0	T-1	T-2	T-3	T-4	T-5	T0	T-1	T-2	T-3	T-4	T-5

Table 5.18- Multiple regression analysis results for brown trout

	M5		M6		M7		M8		M9		M10	
	R^2	p-value	R^2	p-value	R^2	p-value	R^2	p-value	R^2	p-value	R^2	p-value
A	0.17	0.032	0.32	0.001	0.15	0.052	0.37	0.000	0.37	0.000	0.10	0.142
B	0.21	0.034	0.35	0.001	0.17	0.073	0.46	0.000	0.39	0.000	0.16	0.096
C	0.23	0.022	0.39	0.000	0.22	0.028	0.41	0.000	0.38	0.001	0.19	0.056
D	0.23	0.049	0.39	0.001	0.22	0.062	0.47	0.000	0.40	0.001	0.19	0.103
E	0.21	0.071	0.38	0.002	0.19	0.117	0.43	0.000	0.43	0.000	0.20	0.086
F	0.24	0.044	0.39	0.001	0.24	0.046	0.47	0.000	0.38	0.002	0.19	0.113
G	0.23	0.094	0.39	0.003	0.26	0.054	0.47	0.000	0.43	0.001	0.26	0.064
H	0.24	0.084	0.39	0.004	0.24	0.082	0.47	0.000	0.41	0.002	0.19	0.176
I	0.26	0.097	0.40	0.007	0.31	0.045	0.48	0.001	0.43	0.003	0.29	0.060
J	0.52	0.000	0.45	0.002	0.31	0.043	0.48	0.001	0.44	0.003	0.27	0.079
K	0.47	0.001	0.47	0.001	0.52	0.000	0.55	0.000	0.50	0.001	0.53	0.000
L	0.52	0.001	0.46	0.004	0.52	0.001	0.48	0.002	0.45	0.005	0.44	0.007
M	0.55	0.000	0.49	0.002	0.55	0.000	0.55	0.000	0.50	0.001	0.53	0.001
N	0.56	0.001	0.52	0.002	0.54	0.001	0.48	0.006	0.45	0.010	0.44	0.015
O	0.57	0.001	0.50	0.004	0.56	0.001	0.65	0.000	0.53	0.002	0.53	0.002
P	0.57	0.002	0.55	0.002	0.56	0.002	0.65	0.000	0.53	0.004	0.57	0.002
Q	0.52	0.001	0.48	0.003	0.33	0.062	0.48	0.003	0.49	0.003	0.29	0.122
R	0.52	0.001	0.48	0.003	0.59	0.000	0.61	0.000	0.50	0.002	0.54	0.001
S	0.54	0.002	0.52	0.003	0.55	0.001	0.48	0.007	0.50	0.005	0.44	0.016
T	0.59	0.001	0.51	0.004	0.60	0.000	0.61	0.000	0.51	0.004	0.54	0.002
U	0.56	0.002	0.56	0.003	0.57	0.002	0.49	0.013	0.50	0.011	0.44	0.031
V	0.59	0.001	0.51	0.008	0.61	0.001	0.66	0.000	0.57	0.002	0.55	0.004
W	0.60	0.002	0.57	0.005	0.57	0.004	0.66	0.000	0.63	0.001	0.63	0.001
X	0.59	0.003	0.55	0.007	0.61	0.002	0.68	0.000	0.57	0.005	0.66	0.000
Y	0.60	0.005	0.57	0.010	0.66	0.001	0.69	0.000	0.64	0.002	0.67	0.001
Z	0.57	0.005	0.49	0.021	0.49	0.021	0.49	0.020	0.55	0.007	0.39	0.094
AA	0.59	0.003	0.50	0.017	0.64	0.001	0.63	0.001	0.56	0.005	0.58	0.004
BB	0.59	0.008	0.53	0.023	0.58	0.009	0.50	0.039	0.55	0.015	0.48	0.050
CC	0.60	0.006	0.55	0.016	0.64	0.002	0.64	0.002	0.57	0.010	0.61	0.005
DD	0.60	0.011	0.54	0.035	0.60	0.011	0.52	0.050	0.57	0.022	0.54	0.036
EE	0.62	0.009	0.65	0.004	0.66	0.004	0.65	0.004	0.68	0.002	0.61	0.010
FF	0.68	0.005	0.54	0.063	0.67	0.006	0.65	0.009	0.64	0.011	0.63	0.014
GG	0.62	0.017	0.66	0.006	0.68	0.005	0.67	0.005	0.68	0.005	0.65	0.009
HH	0.68	0.010	0.68	0.010	0.68	0.010	0.68	0.010	0.68	0.010	0.68	0.010
II	0.68	0.010	0.68	0.010	0.68	0.010	0.68	0.010	0.68	0.010	0.68	0.010
JJ	0.68	0.010	0.68	0.010	0.68	0.010	0.68	0.010	0.68	0.010	0.68	0.010

The results are demonstrated in Table 5.18. The following trends occur:

- Model 6, 8 and 9 had significant relationships between antecedent flow variables and brown trout populations for all 36 combinations.
- A common antecedent variable in each of these models was the Winter Q_{95} flow. This was also found in the regression analysis (Part A) to have significant relationship.
- However only some of the combinations from these models had strong significances according to the R^2values, these were:

- N onwards, excluding Q, R and Z in Model 6
- K onwards, excluding L, N, Q, S, U and Z in Model 8
- K onwards, excluding L, N and Q in Model 9

Combinations Q, R and Z all had a *T-0* antecedent condition in one of their variables; this could therefore be a reason for not having strong significance as the antecedent flow is not fully represented.

- M5 had strong significant relationships from combination L onwards, combinations A-K had one variable at *T-0* antecedent thus not fully representing antecedent flows.
- M7 had strong significant values from K onwards, excluding model Q
- M10 generally had strong significant relationships from K onwards, with 7 combinations not having a strong relationship.
- Generally strong significant relationships were found when more antecedent years were taken into account and fewer relationships were found when 1 variable had a *T-0* combination. This finding indicated that brown trout have a more significant relationship with the antecedent flows than the flows in that year.
- The most significant variable affecting brown trout is the antecedent Winter Q_{95} flow. Therefore low flows in winter have a significant impact on brown trout
- As this study seeks to understand the relationships species have with low flows, it is important to assess the summer Q_{95} flows in detail. Models 5, 7 and 9 include Summer Q_{95} as one of the variables. These models generally have strong significant relationship from combinations K onwards.
- Model 9 shows the winter Q_{95} with the summer Q_{95}. This model has the highest amount of strong significant relationships between antecedent flows and brown trout populations. This indicates that these variables are the most important flows to protect for brown trout as they provide the most significant conditions.

5.7.4 Combined results

The final analysis of brown trout data aimed to inform whether brown trout populations are influenced by food source availability i.e. BMI through ASPT and LIFE scores. Brown trout populations were correlated with ASPT and LIFE scores, the results are presented in Figure 5.13 and Table 5.19.

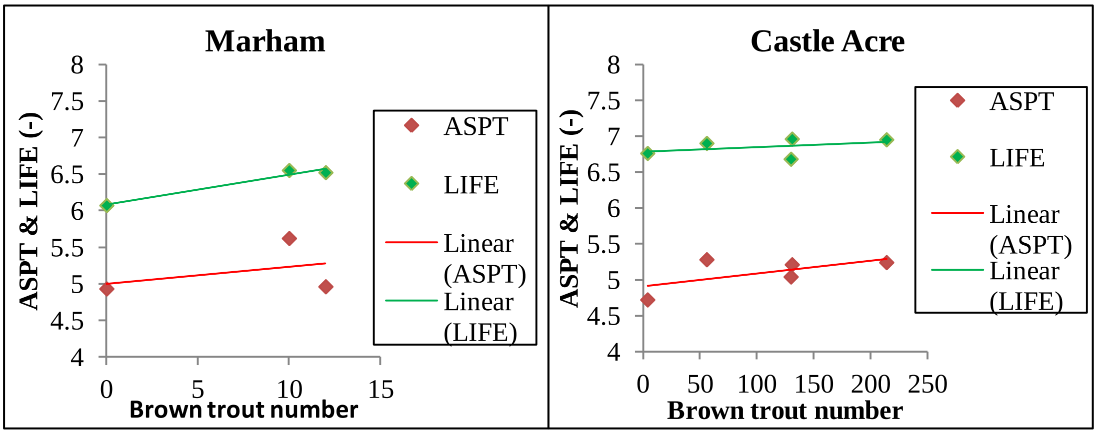


Figure 5.14- Combined results for brown trout and BMI

Table 5.19- Statistical analysis for combined results

	ASPT			**LIFE**		
	C- coeff	**R^2**	**p-value**	**Co-coeff**	**R^2**	**p-value**
Marham	0.39	0.16	0.742	0.98	0.96	0.135
Castle Acre	0.63	0.39	0.260	0.41	0.16	0.498

The following observations were determined from the results (Figure 5.14, Table 5.19):

- Brown trout numbers increase when there are higher ASPT and LIFE scores.
- Positive correlations between brown trout number and ASPT and LIFE scores occurred at both sites. However none had significant relationships and only Marham for LIFE values had a strong relationship. Due to the minimal data available, it is likely that with more data values the p-values would show stronger significance.
- From the data available it can be concluded that there are indications that brown trout numbers increase when BMI abundance is higher (i.e. higher scoring indices) therefore providing more food sources.

Jowett (1992) discovered that invertebrate biomass was found to be the single most important factor in determining brown trout abundance. The results found here suggest that in the River Nar brown trout abundances are influenced by invertebrate quality but not necessarily abundance.

5.7.5 Summary

- Brown trout populations have generally decreased in the River Nar since 1989
- Brown trout populations do not have any statistical relationship with daily flows.
- The most significant antecedent conditions are the winter and summer Q_{95} (low) flows.
- Brown trout numbers increase when BMI have better quality and therefore more food sources.

5.8 Chapter summary

This research question aimed to determine the impacts of different flows, particularly low flows on the indicator species. A brief overview of the results are provided here. Discussion of the results can be found in chapter 8.

5.8.1 Benthic macro-invertebrate (BMI)

According to the ASPT scores, BMI quality in the River Nar is excellent with summer providing the best conditions, followed by spring and autumn equally and then winter, this therefore shows the natural hydrological process. Furthermore, mid-stream reaches provide the best habitats for BMI and fen sections of the river provide the lowest ASPT and LIFE scores.

BMI generally do not have a strong relationship with daily flows according to both the collected and EA data. The collected data showed that ASPT had little statistical relationship to the antecedent flows and LIFE exhibited some statistical relationships to the antecedent flows. The results from the EA data indicated that BMI had a lagged response to flows and that summer flows are the most critical in sustaining BMI health, to summarise the antecedent flow relationships:

- For Spring LIFE, the combinations which had the most significant results were: Summer Q_{10} and Winter Q_{95} with 1 variable and Winter Q_{95} plus Summer Q_{10} with 2 variables
- For Autumn LIFE, the combinations which had the most significant results were: Summer Q_{10} and Summer Q_{95} with 1 variable and Summer Q_{95} plus Summer Q_{10} with 2 variables.

Site conditions are very important for BMI, cover and medium gravel substrate provide the best environment for BMI whilst no cover, silty substrate and channel modification provide poor habitat for BMI.

5.8.2 Macrophytes (Ranunculus)

The highest abundances of macrophytes were found during the driest period, with a preceding wet winter. The lowest abundances were found during a wet period with dry preceding 6 months. Therefore whilst dry periods provide the best conditions, the whole cycle needs to be taken into account. For example, the winter of 2013/14 had slightly drier conditions than winter 2012/13; this resulted in lower abundances in the subsequent summer.

A further finding in this research was that macrophyte abundance is not related to daily flow conditions despite a trend of higher abundances during drier conditions and vice versa. Any change in abundance is more related to natural fluctuations and growth throughout the year. The lowest abundances were found in: highly modified areas with narrow channels, silt substrate and heavy poaching. The highest abundances were found in: wide channels (even if artificially modified) and gravel substrate with little anthropogenic use. These sites scored 5 and 4 on HMC which indicates modified areas may create good habitat for macrophytes.

Ranunculus abundance is not related to daily flows and is moreover related to the antecedent flows. Ranunculus had the best growth when there are drier conditions with wetter conditions 6 months previous which is as would be expected during summer months. The 6 month antecedent flow conditions affect *Ranunculus* growth, wetter conditions 6 months previous create higher abundances. In order for these results to be reliable however, more data would need to be collected as this is only a snapshot year. Historical abundance data collected every three months would be ideal for this analysis. Furthermore, site conditions play a large part in *Ranunculus* growth.

5.8.3 Fish (Brown trout)

Brown trout populations have generally decreased in the River Nar since 1989, their lowest numbers were in 1993, 2 years after the driest recorded year. Brown trout populations do not have any statistical relationship with daily flows but are influenced by antecedent flows. The most significant antecedent conditions found were the winter and summer Q_{95} (low) flows. A small but key finding was that brown trout numbers increase when BMI have better quality and therefore the brown trout have more food sources.

5.9 Errors associated with fieldwork data collection

All attempts were made during data collection to avoid error and uncertainty. The macrophyte data collection is open to subjectivity as it is based on personal opinions, in order to reduce any uncertainty around this data collection the same surveyor always carried out the macrophyte mapping. Likewise with the kick sample data, the British Standard methods were always used which is the most reliable and well used method available. Furthermore EA data was used where possible.

With regards to using electro-fishing data, there is always uncertainty surrounding data collection, for example, fish can move before the sample is taken, smaller fish may not be detected, the scaling and therefore age of the fish was not recorded, however the data gathered was done do through official EA methods and are the best methods available therefore the data is reliable enough to use for analysis. Overall, despite any errors which could occur, the most appropriate and robust methods were used at all stages and therefore the data is as accurate as possible.

Chapter 6- Research question 2 results

6.1 Chapter introduction

Research question 2: How useful are numerical models in investigating how low flow periods impact upon the ecosystem indicators?

This chapter demonstrates results and analysis from research question 2 (RQ2). Two main areas were investigated with these results, firstly the habitat models were used to assess how different flows, particularly low flows affect habitat availability of the ecosystem indicators. Secondly the results were used to investigate the sensitivity of input therefore determining how useful the models are in investigating the impacts of flow on ecosystem indicators. The results are presented in each of the 7 analysis sections:

- Analysis 1: Fuzzy V HSC
- Analysis 2: Habitat distribution
- Analysis 3: Low flow periods
- Analysis 4: Extreme years
- Analysis 5: Key times for species
- Analysis 6: Interconnectedness of species
- Analysis 7: Spatial distribution
- Analysis 8: Key comparisons between 1D and 2D

Further information about each analysis is given in Chapter 3. This chapter solely presents the results of each.

6.2 Analysis 1- Fuzzy V HSC

There are two means of determining habitat suitability preferences. This analysis compared the results from these two input methods: fuzzy rules and HSC, with an aim of determining which was most appropriate for use in subsequent analysis. Graphs are presented for adult and spawning brown trout and Crowfoot. Graphs for juvenile brown trout and Mayfly are in Appendix K.

6.2.1 Site 2- DS Nar (1D)

The results (Figure 6.1) show there are differences between the HSC and fuzzy rules. These differences are not unusual and has been found in previous studies, for example Munoz-Mas et al. (2012), found different results from fuzzy logic rules and HSC data.

Furthermore Boavida et al., (2014) found differences between the two input methods and also between the two output methods of WUA and HHS. Therefore the results imply that the results from habitat models should be used with caution.

For spawning brown trout and Crowfoot when there is a large increase in flow, there is a large decrease in habitat availability (see box highlighted in Figure 6.1a) indicating lower flows provide better habitat than high flows. These differences will be explored in further sections however what is important to note is that the same pattern occurs for HSC and fuzzy rules for these two species, i.e. an increase in flow causes a decrease in habitat. Similar trends occur for juvenile brown trout and Mayfly (see Appendix K), where both HSC and fuzzy rules provide the same trends i.e. increase in flow creates an increase in habitat availability.

For adult brown trout the fuzzy rules generally follow the pattern of the flow regime i.e. when there is an increase in flow there is an increase in habitat availability, and vice versa. However the opposite is true for HSC which decrease very slightly when there is an increase in flow. This could be related to the HSC's and fuzzy rules not corresponding to the same outcomes due to them being developed in different ways.

Mann-Whitney tests (Table 6.1) revealed statistically significant differences between the fuzzy rules and HSC for all species. This shows that even with rules based around the same information and data, different results occur promoting how sensitive habitat models are to input methods.

A similar trend which occurs throughout all habitat results for site 2 is fuzzy rules providing a wider range of results, whilst HSC have a smaller range. Furthermore HSC's generally provide lower habitat availability results than fuzzy rules.

Overall these results demonstrate the high sensitivity and importance of the input to the habitat models. However for fish particularly they would not be so selective with their habitat as HSC depicts, and for example if there were high velocities which they did not prefer, with low velocities and low substrates which they did prefer, then they would still prefer this over other combinations. The fuzzy rules allow for this natural selective behaviour.

For site 2, fuzzy rules are more appropriate as they provide a more diverse set of results. Species are more likely to use a diverse range of habitats and therefore fuzzy results are more appropriate in this respect.

6.2.2 Site 3- Castle Acre (1D)

Likewise with site 2 the fuzzy rules and HSC give different results and the Mann-Whitney tests revealed statistically significant differences between the fuzzy rules and HSC for all species (Table 6.1).

As opposed to results for site 2 however, the HSC results generally gave higher values than for fuzzy rules for adult and juvenile fish (Figure 6.1). For other species the fuzzy results were either higher or very similar. However as with site 2, fuzzy rules provide a larger range of results than HSC. This supports the site specific nature of the results.

For spawning and juvenile brown trout large troughs in available habitat occur at the same time for both fuzzy rules and HSC. However for adult lifestages, differing results occur with large peaks occurring at certain times for fuzzy but large troughs occurring at the same time with the HSC. When related to flow, the same trends occur as for those in site 2, this is as would be expected as the same fuzzy rules and HSC were used for each site. However the same issue occurred with the adult brown trout in that HSC and fuzzy rules exhibited differing results.

6.2.3 Presentation of results

This section presents the graphs and tables from each site for this analysis

Table 6.1- Mann- Whitney results for HSC and fuzzy rules for both sites

	Site 2	**Site 3**
	p-value	
Adult brown trout	0.000	0.000
Juvenile brown trout	0.000	0.000
Spawning brown	0.000	0.000
Mayfly	0.000	0.000
Crowfoot	0.000	0.000

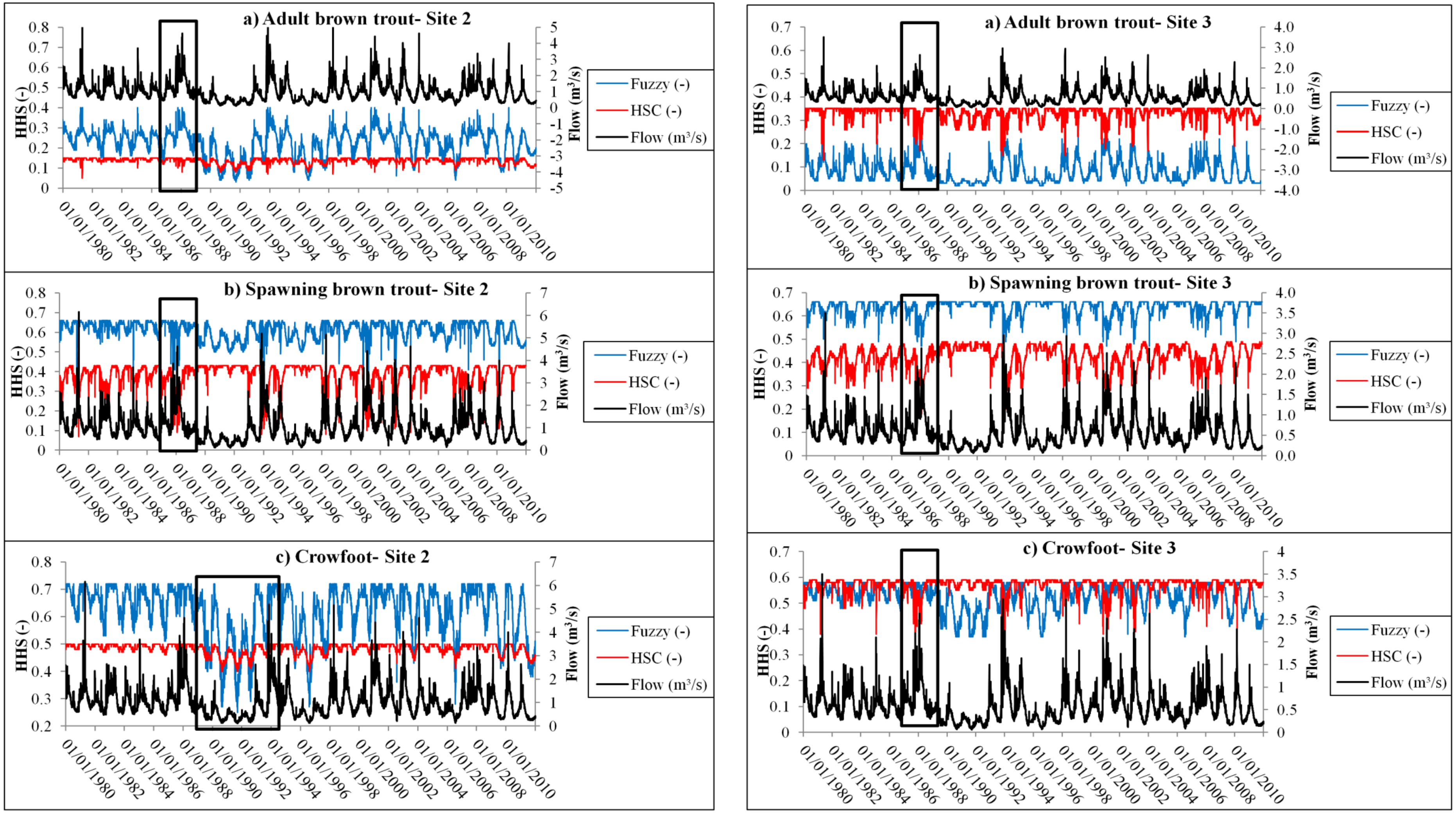


Figure 6.1- Fuzzy rules and HSC output for site 2 (left) and 3 (right) for adult and spawning brown trout and Crowfoot

6.2.4 Summary of analysis

Some of the differences shown between HSC's and fuzzy rules can be attributed to HSC being derived from literature, whilst fuzzy rules were derived from CASiMiR which incorporated expert knowledge and imprecise information. Thus it could be argued that more detailed information was present in the fuzzy rules than in the HSC from literature. However a main criticism in habitat modelling is related to combining the results of the independent physical variables i.e. depth, velocity, substrate and cover used in HSC. For example, it has been found that using the arithmetic mean method of combining results produces the highest habitat availability values, followed by the geometric mean then by the product method which results in the lowest habitat availability values (Boavida et al., 2014). This analysis used the geometric mean. Thus using fuzzy rules generates weightings between variables which is more likely in reality.

Fuzzy rules are accepted as a suitable approach in habitat suitability assessments due to them taking into account inherent uncertainty of ecological variables (Ahmadi-Nedushan et al., 2006). As the HSC's were derived from literature it was decided that the fuzzy logic results were most appropriate to use as not only are the based on expert knowledge, they are closer to the human way of thinking and communicating allowing for imprecise information i.e. if the depth is low and the velocity is high then the suitability is high, this allows for a combination of variables to be assessed. Furthermore as the results show a wider range of availability for fuzzy rules, this is more likely particularly for fish which would move and adapt to different situations and use a diverse range of habitats. Thus subsequent analysis focuses on fuzzy rule results.

6.3 Analysis 2- Habitat distribution

This analysis investigated how the available habitat distribution changes throughout the 32 year period for each species. The section presents results and discussion of the analysis. Full details of all descriptive statistics and box plots of the distribution of habitat data (Figure 6.3) for each site and species are presented in section 6.3.4.

6.3.1 Site 1- Highbridge (2D)

Figure 6.3 and 6.4 demonstrate the results for site 1.

Adult brown trout:

- The average HHS is 0.04, which means 'very poor' suitability.

- The distribution of HHS between its maximum and minimum values was binned as: 9% ‘upper’ availability, 29.1% ‘middle’ availability 61.9% ‘lower’ availability. The data is positively skewed towards ‘lower’ availability.
- The low range is unusual as the site is the furthest downstream and will therefore have highest flows which adult brown trout should prefer.

Juvenile brown trout:

- The average HHS is 0.05, which means ‘very poor’ suitability.
- The distribution of HHS between its maximum and minimum values was binned as: 3.6% ‘upper’ availability, 78% ‘middle’ availability 16.5% ‘lower’ availability. The data is a normal distribution, as the mean equals the median.
- The low HHS is similar to adult brown trout. However juvenile brown trout have slightly better availability than adult brown trout as the vast majority of time has ‘middle’ availability.

Spawning brown trout:

- The average HHS is 0.59, which means ‘moderate’ suitability;
- The distribution of HHS between its maximum and minimum values was binned as: 90.1% ‘upper’ availability, 8.4% ‘middle’ availability 1.5% ‘lower’ availability. The data is negatively skewed towards ‘upper’ availability.
- The relatively high HHS is unusual as brown trout usually spawn in the upper reaches. Studies have however shown that brown trout can spawn in different places to maximise their survival of their offspring (Dolben 2014).

Crowfoot:

- The average HHS is 0.46, which means ‘moderate’ suitability.
- The distribution of HHS between its maximum and minimum values was binned as: 73.3% ‘upper’ availability, 19.7% ‘middle’ availability 7.1% ‘lower’ availability. The data is highly negatively skewed towards ‘upper’ availability.
- This species provides one of the widest ranging results throughout the 32 year period with values from 0.1 to 0.57 HHS. This shows how the flow has a large influence on Crowfoot at this site.
- The ‘moderate, suitability is representative of fen typology as little Crowfoot would grow in this typology.

Mayfly:

- The average HHS is 0.2, which means ‘poor’ suitability.

- The distribution of HHS between its maximum and minimum values was binned as: 43.4% ‘upper’ availability, 20.3% ‘middle’ availability 36.4% ‘lower’ availability. The data is fairly evenly distributed, skewed to the extreme ‘upper’ and ‘lower’ availabilities. However as the mean is slightly smaller than the mean, this indicates a negative Skew.
- Small numbers of Mayfly were found in this section of the river in all seasons, this is therefore indicative of the ‘poor’ suitability.

6.3.2 Site 2- DS Nar (2D)

Figure 6.5 and 6.6 demonstrate the results for site 1

Adult brown trout:

- The average HHS is 0.18, which means ‘very poor suitability’.
- The distribution of HHS between its maximum and minimum values was binned as: 50.2% ‘upper’ availability, 38.2% ‘middle’ availability 11.6% ‘lower’ availability. The data is negatively skewed towards ‘upper’ availability.

Juvenile brown trout:

- The average HHS is 0.27, which means ‘poor suitability’.
- The distribution of the HHS between its maximum and minimum values has been binned as follows: 75.6% ‘upper’ availability, 20.2% ‘middle’ availability 4.2% ‘lower’ availability. The data is negatively skewed towards ‘upper’ availability.

Spawning brown trout:

- The average HHS is 0.63, which means ‘good suitability’.
- The distribution of HHS between its maximum and minimum values was binned as: 90.7% ‘upper’ availability, 9.7% ‘middle’ availability 0.1% ‘lower’ availability. The data is negatively skewed towards ‘upper’ availability.

Crowfoot:

- The average HHS is 0.6, which means ‘moderate suitability’.
- The distribution of HHS between its maximum and minimum values was binned as: 83.8% ‘upper’ availability, 13.7% ‘middle’ availability 2.5% ‘lower’ availability. The data is negatively skewed towards ‘upper’ availability.

Mayfly:

- The average HHS is 0.44, which means ‘moderate suitability’.
- The distribution of HHS between its maximum and minimum values was binned as: 82% ‘upper’ availability, 18% ‘middle’ availability 0.03% ‘lower’ availability. The data is negatively skewed towards ‘upper’ availability.

6.3.3 Site 3- Castle Acre (1D)

Figure 6.7 and 6.8 demonstrate the results for site 1.

Adult brown trout:

- The average HHS is 0.08, which means 'very poor suitability'.
- The distribution of HHS between its maximum and minimum values was binned as: 1.3% 'upper' availability, 20.4% 'middle' availability 78.3% 'lower' availability. The data is positively skewed towards 'lower' availability.
- Due to the nature of the river at this site i.e. narrow and shallow, the 'very poor' suitability classification is not unexpected. It would be expected that adult brown trout have more available habitat further downstream, at site 2 for example, there is an average HHS of 0.2, therefore indicating there is more availability further downstream for adult brown trout.

Juvenile brown trout:

- The average HHS is 0.24, which means 'poor suitability'.
- The distribution of HHS between its maximum and minimum values was binned as: 95% 'upper' availability, 5.6% 'middle' availability 0.4% 'lower' availability. The data is negatively skewed towards 'upper' availability.

Spawning brown trout:

- The average HHS is 0.64, which means 'good suitability'.
- The distribution of HHS between its maximum and minimum values was binned as: 98.3% 'upper' availability, 1.4% 'middle' availability 0.3% 'lower' availability. The data is negatively skewed towards 'upper' availability.
- Spawning brown trout have the highest availability out of the three lifestages, due to site conditions here having relatively low depths, low channel widths and predominantly gravel substrate; this is as would be expected.

Crowfoot:

- The average HHS is 0.53, which means 'moderate suitability'.
- The distribution of HHS between its maximum and minimum values was binned as: 71.1% 'upper' availability, 20.4% 'middle' availability 8% 'lower' availability. The data is negatively skewed towards 'upper' availability.
- A large abundance of Crowfoot was found at this site therefore the 'moderate suitability is indicative of this.

Mayfly:

- The average HHS is 0.48, which means 'moderate suitability'.

- The distribution of HHS between its maximum and minimum values was binned as: 84.3% ‘upper’ availability, 13.1% ‘middle’ availability 2.6% ‘lower’ availability. The data is negatively skewed towards ‘upper’ availability.
- Unlike site 2, results demonstrated here do not depict major reductions in habitat availability when there is a major reduction in flow. This along with a small range in results of 0.43 to 0.49 indicates that Mayfly at this site are not greatly influenced by the change in flows.

6.3.4 Presentation of results and statistical properties

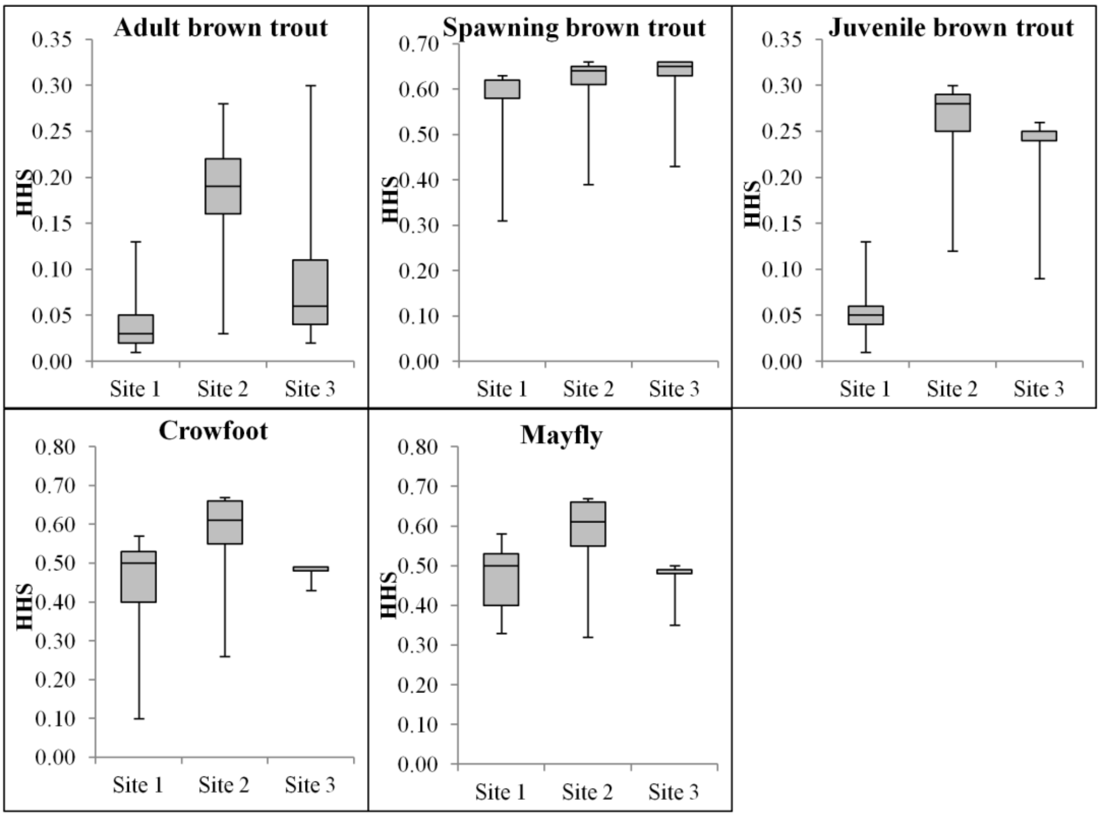


Figure 6.2- Box plot distributions of habitat availability (HHS) at each site for each species

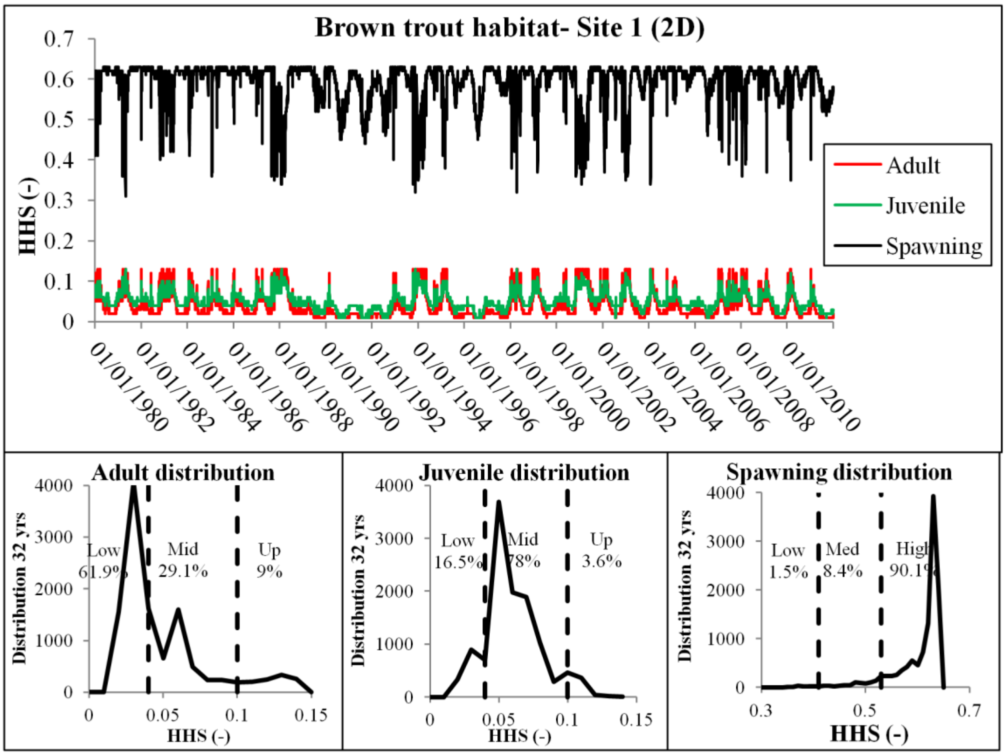


Figure 6.3- Brown trout habitat availability results (HHS) and distributions for site 1

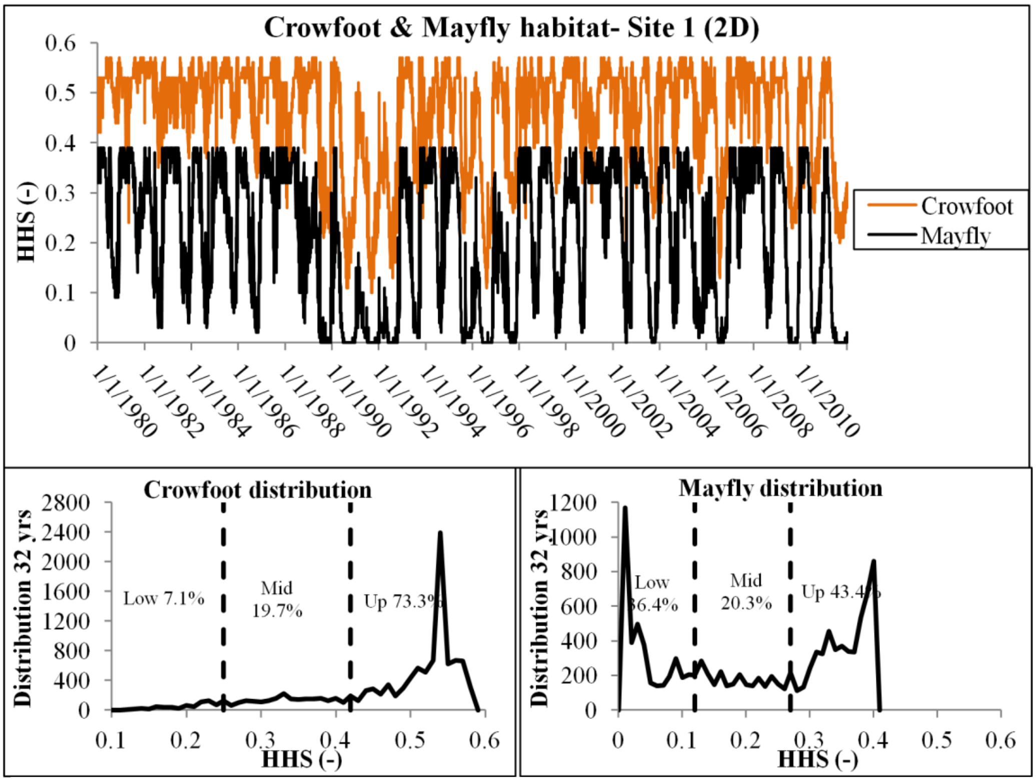


Figure 6.4- Crowfoot and Mayfly habitat availability results (HHS) and distributions for site 1

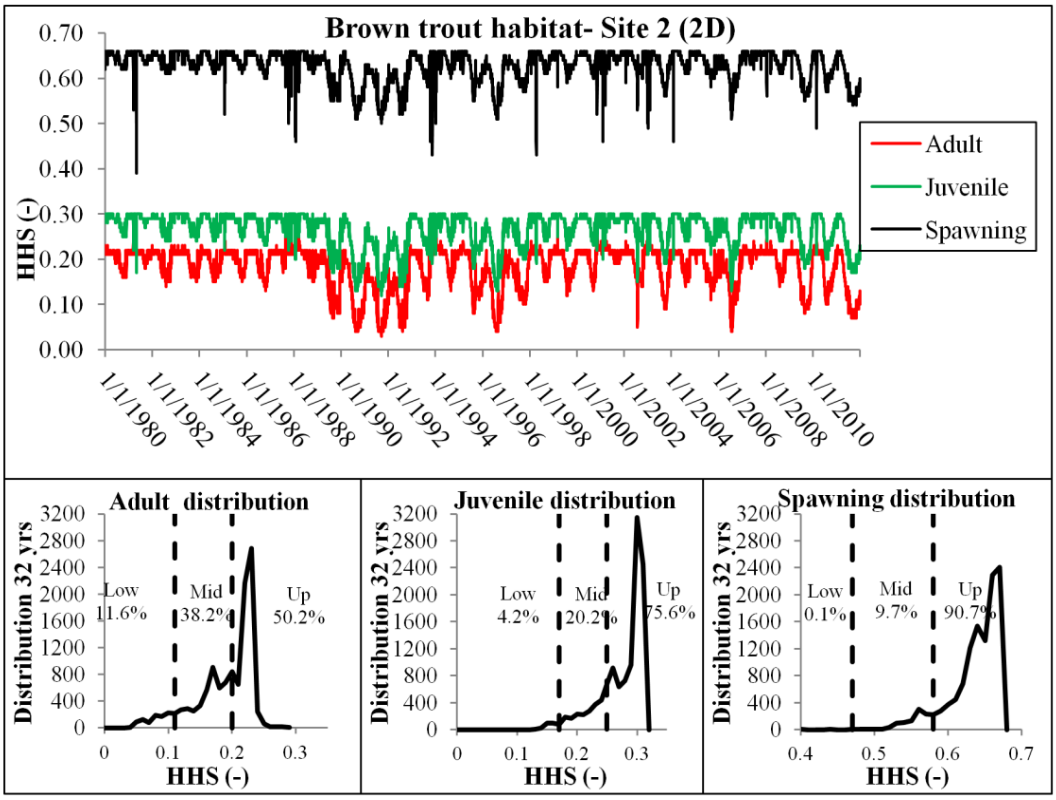


Figure 6.5- Brown trout habitat availability results (HHS) and distributions for site 2

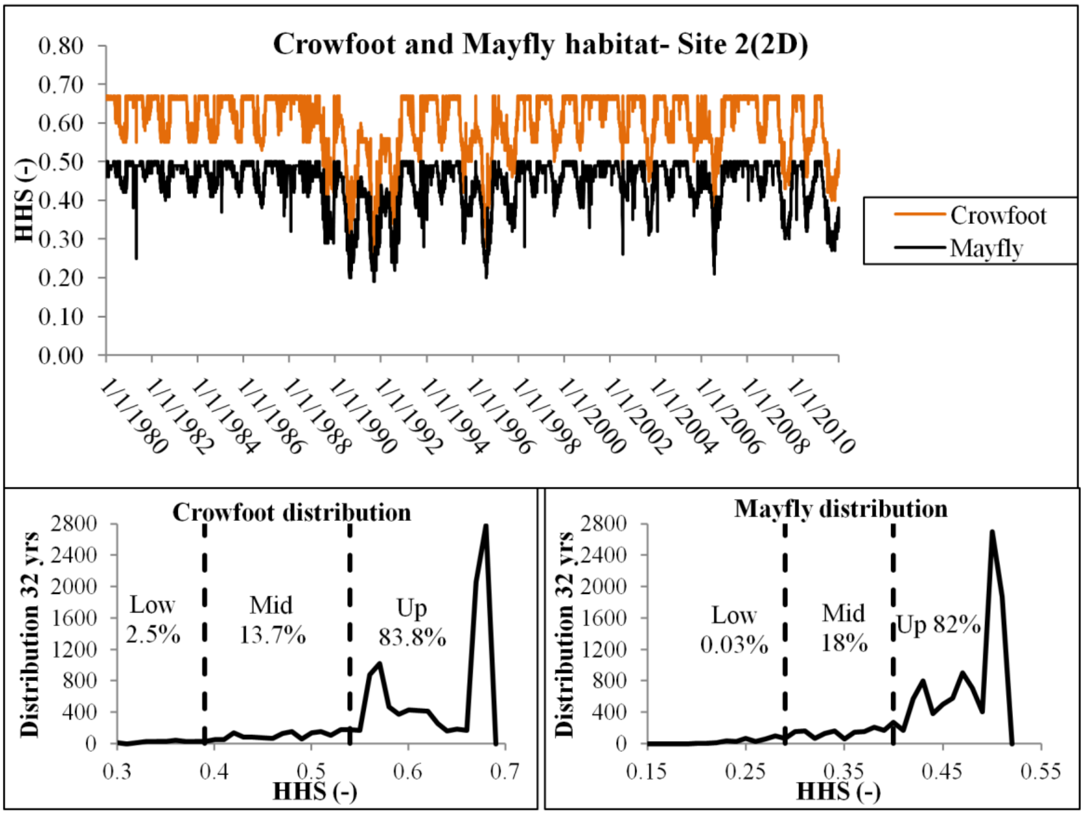


Figure 6.6- Crowfoot and Mayfly habitat availability results (HHS) and distributions for site 2

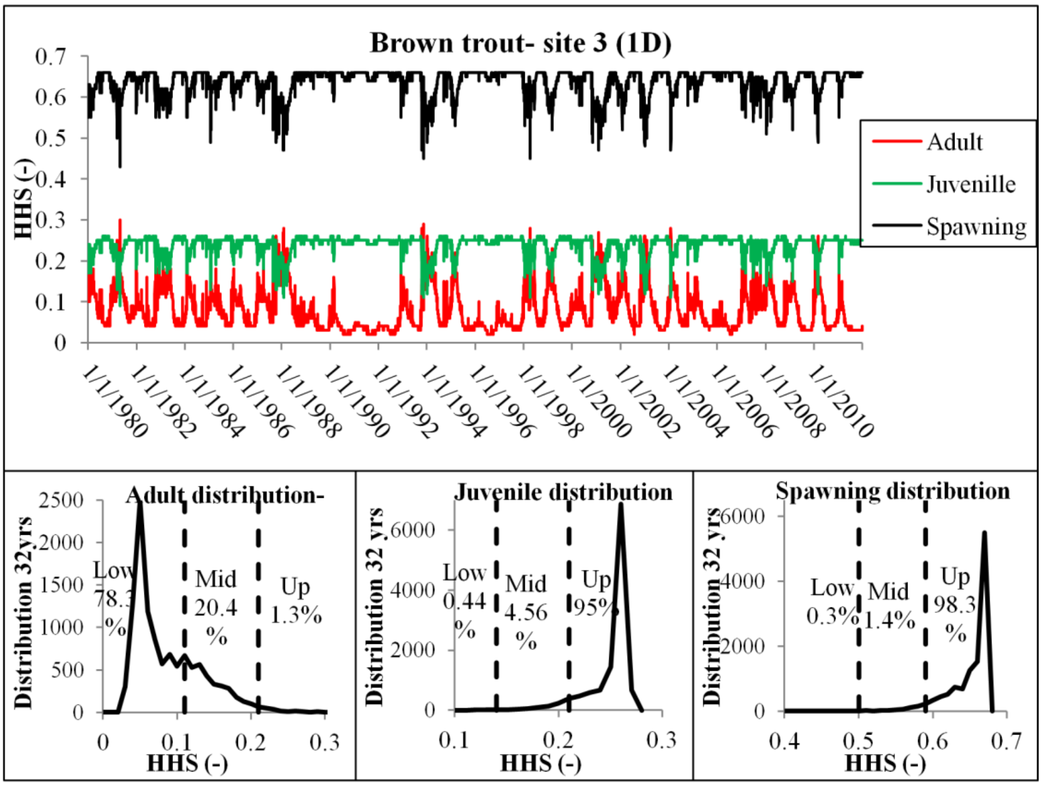


Figure 6.7- Brown trout habitat availability results (HHS) and distributions for site 3

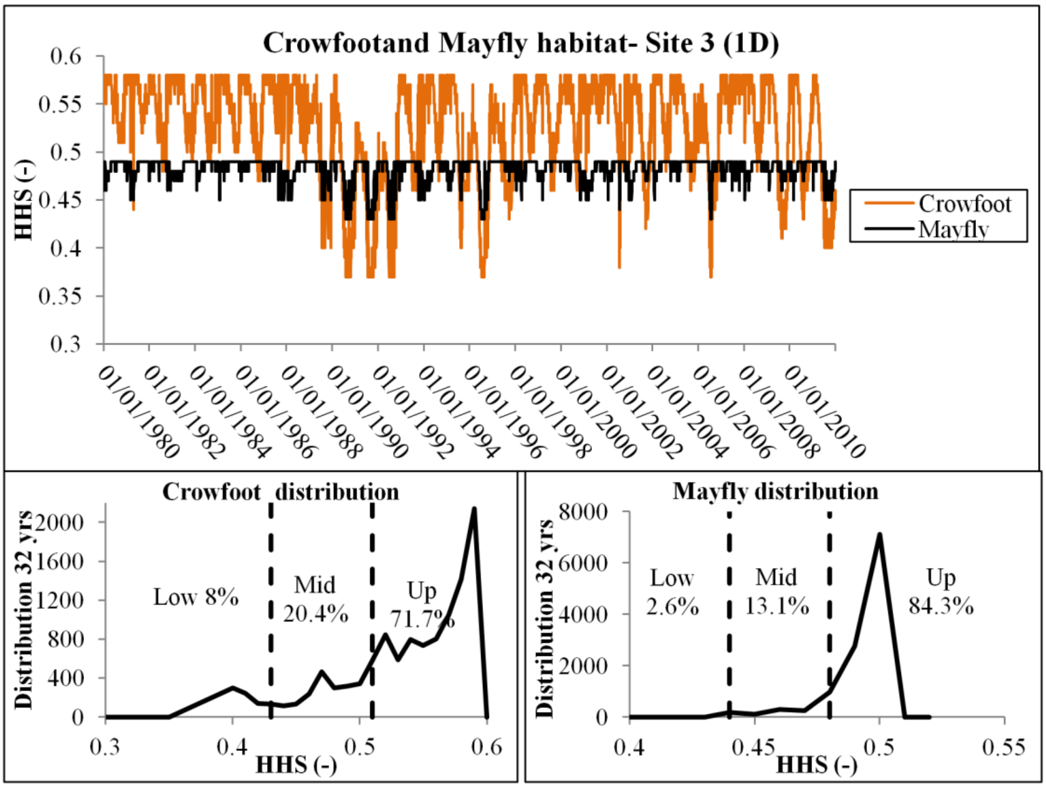


Figure 6.8- Crowfoot and Mayfly habitat availability results (HHS) and distributions for site 3

Table 6.2- Descriptive statistics for the distribution of each species at each site

		Adult BT	**Juvenile BT**	**Spawning BT**	**Crowfoot**	**Mayfly**
Site 1- Highbridge	Average	0.04	0.05	0.59	0.46	0.20
	Median	0.03	0.05	0.62	0.50	0.22
	Maximum	0.13	0.13	0.63	0.57	0.39
	Minimum	0.01	0.01	0.31	0.10	0.00
	95 percentile	0.01	0.02	0.49	0.23	0.00
	50 percentile	0.03	0.05	0.62	0.50	0.22
	5 percentile	0.12	0.09	0.63	0.56	0.39
	Standard	0.03	0.02	0.05	0.11	0.14
	Skew	1.52	0.60	-2.22	-1.21	-0.15
	Kurt	1.57	0.51	5.42	0.45	-1.52
Site 2- DS Nar	Average	0.18	0.27	0.63	0.44	0.60
	Median	0.20	0.28	0.64	0.46	0.61
	Maximum	0.28	0.30	0.66	0.50	0.67
	Minimum	0.03	0.12	0.39	0.19	0.26
	95 percentile	0.08	0.18	0.55	0.30	0.43
	50 percentile	0.20	0.28	0.64	0.46	0.61
	5 percentile	0.22	0.30	0.66	0.50	0.67
	Standard	0.05	0.04	0.03	0.06	0.08
	Skew	-1.13	-1.34	-1.42	-1.44	-1.19
	Kurt	0.56	1.15	1.85	1.55	1.19
Site 3- Castle Acre	Average	0.08	0.24	0.64	0.53	0.48
	Median	0.06	0.25	0.65	0.54	0.49
	Maximum	0.30	0.26	0.66	0.58	0.49
	Minimum	0.02	0.09	0.43	0.37	0.43
	95 percentile	0.03	0.20	0.58	0.41	0.45
	50 percentile	0.06	0.25	0.65	0.54	0.49
	5 percentile	0.17	0.26	0.66	0.58	0.49
	Standard	0.05	0.02	0.03	0.05	0.01
	Skew	1.04	-2.25	-1.91	-1.06	-2.35
	Kurt	0.55	5.86	4.39	0.41	5.94

6.3.5 Summary and discussion of analysis

Table 6.3 presents a synthesis of the habitat availability at each site for each species.

Table 6.3- Synthesis of habitat availability averages

	Site 1 (2D)	Site 2 (2D)	Site 3 (1D)
	Downstream to Upstream		
Adult brown trout	Very low	Low	Very low
Juvenile brown	Very low	Low	Low
Spawning brown	Moderate	High	High
Crowfoot	Moderate	Moderate	Moderate
Mayfly	Low	Moderate	Moderate

Overall the three sites of the river provided different habitat availabilities for each species. Crowfoot was the exception where 'moderate' suitability was found in each of the sites. However upon assessing the box plots for Crowfoot, the distribution of the data still differs for the species. In site 1 there is a wide range from 0.1 to 0.57, whereas in site 3 there is a small range from 0.37 to 0.58.

For the other species, the suitability categories remained fairly similar, for example adult brown trout had 'very low', 'low' and 'very low' habitat availability in Site 1, Site 2 and Site 3 respectively. There was a trend of a slight increase in habitat availability from downstream to upstream i.e. Mayfly habitat increases from 'Low' at site 1, (downstream) to 'moderate' at site 3 (upstream). This concurs with the finding from the RHS analysis in RQ1 where an increase in the quality of habitat was found further upstream. This said however, adult brown trout had 'very low' habitat availability in both the most upstream and downstream sites which indicates that for certain species this trend does not occur, so the best available habitats are species specific. Furthermore, site conditions are very important for habitat availability and localised conditions do impact on the available habitat.

6.4 Analysis 3- Low flow periods

This analysis aimed to determine if low flows result in low HHS.

6.4.1 Site 1- Highbridge (2D)

Figure 6.9 presents the results for site 1. The figures show the HHS curve i.e. the HHS scores related to an increasing flow. The green line shows the low flow (Q_{90}) at this section of river ($\leq 0.53 m^3/s$). The dashed lines represent the 'low' HHS values, these were determined in analysis 2, part B by binning the data into upper, middle and lower components.

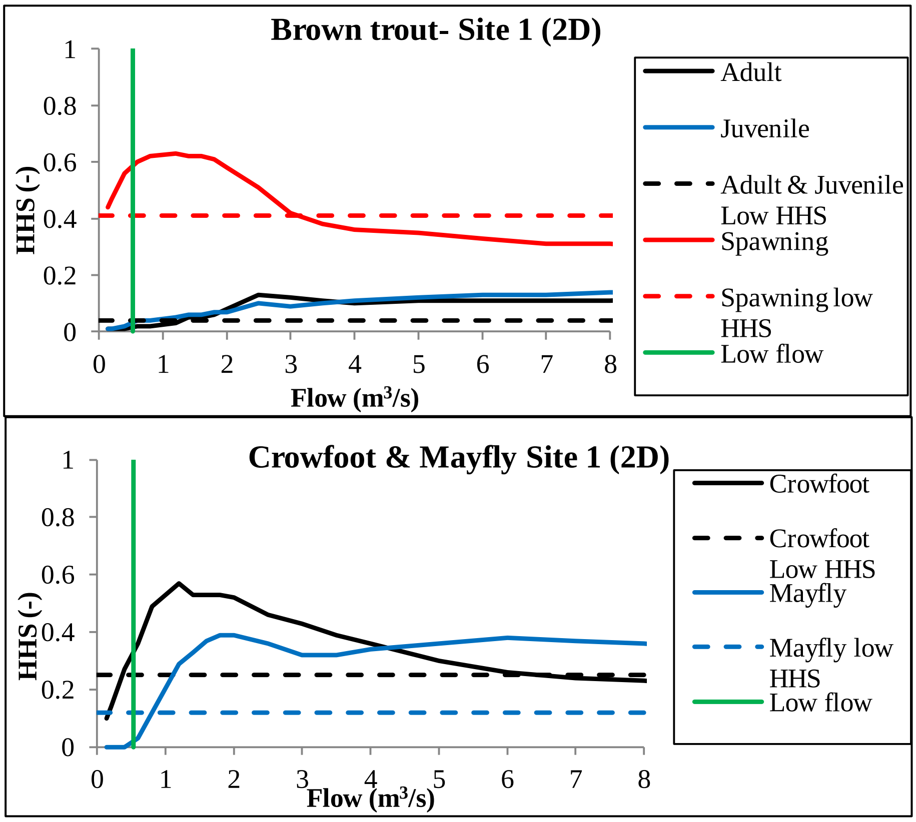


Figure 6.9- Low flows/ low HHS results for site 1

Adult brown trout:

- Very low overall habitat availability, HHS peaks at 2.5m^3/s then remains fairly constant as the flow increases.
- The flow at which the HHS becomes ‘low’ is below 1.24m^3/s therefore in order to maintain good habitat availability for adult brown trout, flows should be above this.
- HHS at low flow= 0.02. Low HHS= 0.04. As HHS at low flow is smaller than low HHS, low flows **do** result in low HHS.

Juvenile brown trout:

- Very low overall habitat availability, HHS progressively increases as flow increases.
- The flow at which the HHS becomes ‘low’ is below 0.99m^3/s therefore in order to maintain good habitat availability for juvenile brown trout, flows should be above this.
- HHS at low flow= 0.02. Low HHS= 0.03. As HHS at low flow is smaller than low HHS, low flows **do** result in low HHS.

Spawning brown trout:

- Typical HHS curve which increases to a peak of HHS 0.63 at 1.2m^3/s, then progressively decreases with increasing flows.

- The flow at which the HHS becomes 'low' is over 3.06m^3/s; therefore in order to maintain good habitat availability for spawning brown trout, flows should be below this.
- HHS at low flow= 0.59. Low HHS= 0.41. As HHS at low flow is larger than low HHS, low flows **do not** result in low HHS.

Crowfoot:

- Typical HHS curve which increases to a peak of HHS 0.57 at 1.2m^3/s, then progressively decreases with increasing flows.
- The flow at which the HHS becomes 'low' is below 0.37m^3/s and above 6.26m^3/s therefore in order to maintain good habitat availability for Crowfoot, flows should be between these volumes.
- HHS at low flow= 0.33. Low HHS= 0.25. As HHS at low flow is larger than low HHS, low flows **do not** result in low HHS.

Mayfly:

- HHS peak of 0.39 at 1.9m^3/s, then slightly decreases before increasing again with increasing flows.
- The flow at which the HHS becomes 'low' is below 0.82m^3/s therefore in order to maintain good habitat availability for Mayfly, flows should be above this.
- HHS at low flow= 0.02. Low HHS= 0.12. As HHS at low flow is smaller than low HHS, low flows **do** result in low HHS.

6.4.2 Site 2- DS Nar (2D)

Figure 6.10 presents the results for site 2. The figures show the HHS curve i.e. the HHS scores related to an increasing flow. The green line shows the low flow (Q_{90}) at this section of river (≤0.43m^3/s). The dashed lines represent the 'low' HHS values, these were determined in analysis 2, part B by binning the data into upper, middle and lower components.

Adult brown trout:

- Progressively increases until peak of HHS 0.28 at 4.5m^3/s, then progressively decreases.
- The flow at which the HHS becomes 'low' is below 0.39m^3/s therefore in order to maintain good habitat availability for adult brown trout, flows should be above this.
- HHS at low flow= 0.13. Low HHS= 0.11. As HHS at low flow is larger than low HHS, low flows **do not** result in low HHS.

Juvenile brown trout:

- Peak of HHS 0.3 at 1.5-2m^3/s, then progressively decreases.
- The flow at which the HHS becomes 'low' is below 1.24m^3/s and above 5.75m^3/s therefore in order to maintain good habitat availability for juvenile brown trout, flows should be between these volumes.
- HHS at low flow= 0.23. Low HHS= 0.17. As HHS at low flow is larger than low HHS, low flows **do *not result in low HHS.***

Spawning brown trout:

- Peak of HHS 0.66 at 1.5-2m^3/s, then dramatically decreases with increasing flows.
- The flow at which the HHS becomes 'low' is above 4.25m^3/s therefore in order to maintain good habitat availability for spawning brown trout, flows should be below this.
- HHS at low flow= 0.6. Low HHS= 0.47. As HHS at low flow is larger than low HHS, low flows **do not** result in low HHS.

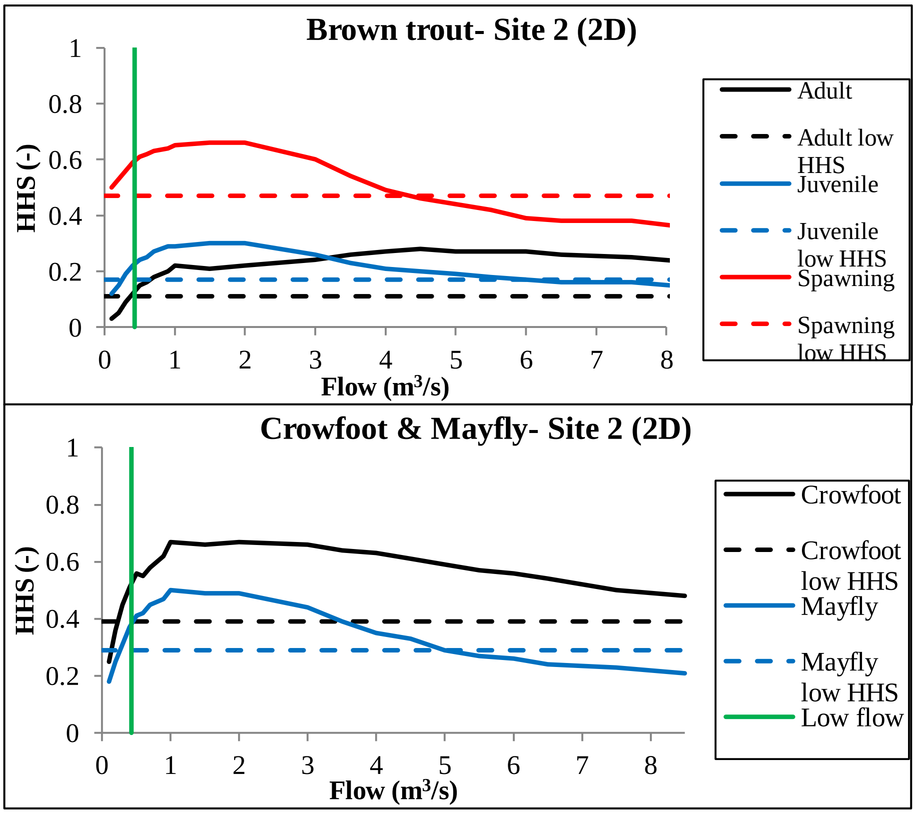


Figure 6.10- Low flows/ low HHS results for site 2

Crowfoot:

- Sudden increase to a peak of HHS 0.67 at 2m^3/s, then gradually decreases with increasing flows.
- The flow at which the HHS becomes 'low' is below 0.24m^3/s therefore in order to maintain good habitat availability for Crowfoot, flows should be above this.

- HHS at low flow= 0.53. Low HHS= 0.39. As HHS at low flow is larger than low HHS, low flows **do not** result in low HHS.

Mayfly:

- Increase to a peak of HHS 0.5 at 1m^3/s, then gradually decreases with increasing flows.
- The flow at which the HHS becomes 'low' is below 0.28m^3/s and above 4.93m^3/s therefore in order to maintain good habitat availability for Mayfly, flows should be between these volumes.
- HHS at low flow= 0.38. Low HHS= 0.29. As HHS at low flow is larger than low HHS, low flows **do not** result in low HHS.

6.4.3 Site 3- Castle Acre (1D)

Figure 6.11 presents the results for site 3. The figures show the HHS curve i.e. the HHS scores related to an increasing flow. The green line shows the low flow (Q_{90}) at this section of river ($\leq$0.25m^3/s). The dashed lines represent the 'low' HHS values, these were determined in analysis 2, part B by binning the data into upper, middle and lower components.

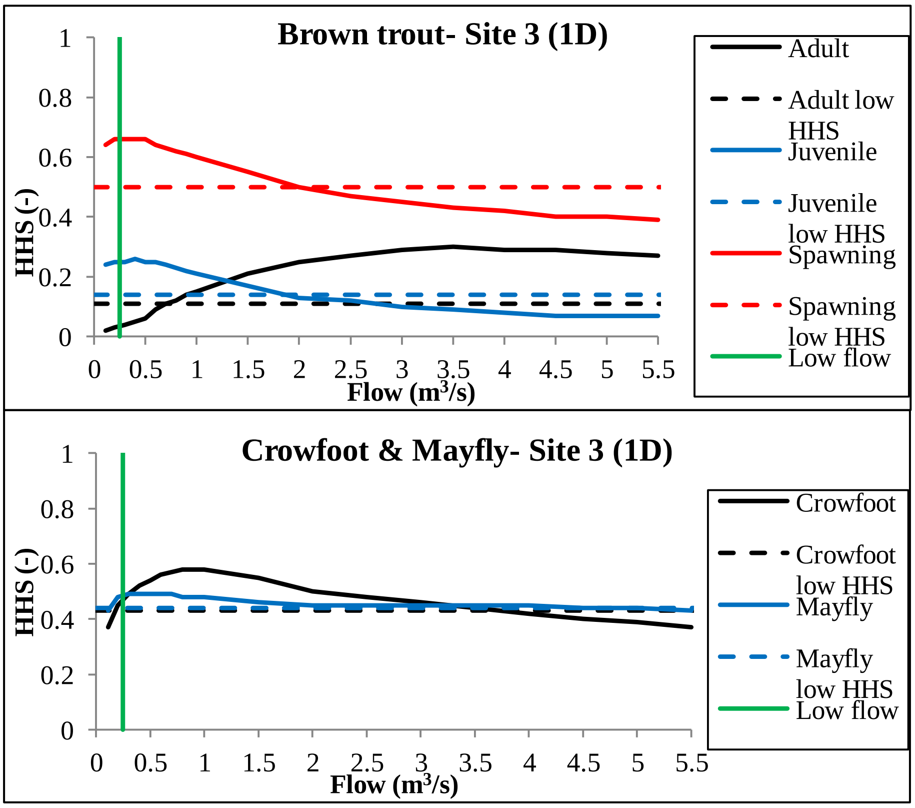


Figure 6.11- Low flows/ low HHS results for site 3

Adult brown trout:

- Increase to a peak of HHS 0.3 at 3.5m^3/s, lots of area available.

- The flow at which the HHS becomes 'low' is below 0.75m^3/s therefore in order to maintain good habitat availability for adult brown trout, flows should be above this.
- HHS at low flow= 0.04. Low HHS= 0.11. As HHS at low flow is smaller than low HHS, low flows **do** result in low HHS.

Juvenile brown trout:

- Quickly increases to a peak of HHS 0.26 at 0.4m^3/s, then rapidly decreases as flow increases.
- The flow at which the HHS becomes 'low' is below 1.82m^3/s therefore in order to maintain good habitat availability for juvenile brown trout, flows should be above this.
- HHS at low flow= 0.25. Low HHS= 0.14. As HHS at low flow is larger than low HHS, low flows **do not** result in low HHS.

Spawning brown trout:

- Starts almost at the peak of HHS 0.66 at 0.2-0.5m^3/s, then rapidly decreases as flow increases.
- The flow at which the HHS becomes 'low' is above 1.95m^3/s therefore in order to maintain good habitat availability for spawning brown trout, flows should be below this.
- HHS at low flow= 0.66. Low HHS= 0.5. As HHS at low flow is larger than low HHS, low flows **do not** result in low HHS.

Crowfoot:

- Typical HHS curve, peaks at HHS 0.58 at 0.8-1m^3/s, then gradually decreases as flow increases.
- The flow at which the HHS becomes 'low' is below 0.18m^3/s and above 3.62m^3/s therefore in order to maintain good habitat availability for Crowfoot, flows should be between these volumes.
- HHS at low flow= 0.47. Low HHS= 0.43. As HHS at low flow is larger than low HHS, low flows **do not** result in low HHS.

Mayfly:

- Very flat HHS curve, peaks at HHS 0.49 at 0.3-0.7m^3/s.
- The flow at which the HHS becomes 'low' is below 0.14m^3/s and above 4.25m^3/s therefore in order to maintain good habitat availability for Mayfly, flows should be between these volumes.

- HHS at low flow= 0.49. Low HHS= 0.44. As HHS at low flow is larger than low HHS, low flows **do not** result in low HHS.

6.4.4 Summary of analysis

Table 6.4 provides a synthesis of the findings from this analysis. In summary, different flows are required by different species, this therefore makes managing the flows difficult for decision makers. In order to protect all the species, flows would have to be between 1.24-3.06m^3/s in site 1, between 0.39-4.25m^3/s at site 2 and between 0.75-1.82m^3/s at site 3 (see Table 6.4). It would be difficult to ensure flows always remain between these boundaries particularly when natural seasonal flows are taken into account. However it is clear that Hands- off- Flow (HOF) limits can be set to protect the different species' requirements rather than only establishing the HOF limit on one species' requirement.

Table 6.4- Summary of analysis 3

	Species	Ideal flow(s) (m^3/s)	Do low flows cause low HHS?
S1 (2D)	**Adult brown trout**	>1.24	Yes
	Juvenile brown trout	>0.99	Yes
	Spawning brown trout	<3.06	No
	Crowfoot	0.37-6.26	No
	Mayfly	>0.82	Yes
S2 (2D)	**Adult brown trout**	>0.39	No
	Juvenile brown trout	0.27-5.75	No
	Spawning brown trout	<4.25	No
	Crowfoot	>0.24	No
	Mayfly	0.28-4.93	No
S3 (1D)	**Adult brown trout**	>0.75	Yes
	Juvenile brown trout	<1.82	No
	Spawning brown trout	<1.95	No
	Crowfoot	0.18-3.62	No
	Mayfly	0.14-4.25	No

6.5 Analysis 4- Extreme years

In order to understand the distribution of habitat availability spatially, the distributions of SI values, were used to investigate the influence of low flows on habitat availability by comparing wet, dry and average years. Mann- Whitney statistical tests compared habitat availability between years and seasons to investigate whether there were statistically significant differences between wet and dry years and seasons. See Table 3.15 (in methodology) for the years used for the analysis.

6.5.1 Mann-Whitney

Due to the large volume of Mann-Whitney tests carried out for this analysis, only results for adult brown trout for site 1 are presented in Appendix L. The same trends occurred for all species at all sites which are discussed here.

Generally for all species the Mann-Whitney tests revealed that for spring and summer seasons the predicted habitat availability was statistically similar ($p<0.05$) between hydrologically similar years and statistically different ($p>0.05$) between dry and wet years. Consequently for spring and summer season habitat availability in wet and dry years provide statistically different habitats. However, for autumn and winter seasons the predicted habitat availability was statistically different between hydrologically similar years according to the Mann-Whitney tests. Consequently there were no clear differences resulting from dry or wet conditions during these seasons. These trends occurred throughout all species and sites, any anomalies are discussed in the relevant sections.

6.5.2 Site 1- Highbridge (2D)

The results for site 1 are presented in Figure 6.12. These graphs present the amount of available habitat (e.g. highly suitable, moderate etc), for each species in the five wettest, five driest and five average years.

Adult brown trout:

Figure 6.13a demonstrates that for adult brown trout in both wet and dry years the vast majority of time is 'highly unsuitable'. Thus indicating this section is not good for adult brown trout. Overall wet years provide slightly more suitable habitats than dry years.

Mann- Whitney tests revealed statistically significant differences between wet and dry years in all seasons, however in autumn and winter many statistically similar results were also found between hydrologically similar years i.e. between 2 wet years. This indicates that, whilst the graphs show differences, these are not statistically significant differences.

Juvenile brown trout:

For juvenile brown trout, the habitat availability is 'highly unsuitable' for the majority of time in both wet and dry years indicating this section is not good for juvenile brown trout**.** Overall wet years provide slightly more suitable habitats than dry years.

Mann- Whitney tests indicated statistical similarities between hydrologically similar years in winter and autumn; therefore the statistical differences also seen between wet and dry years in these seasons are not accurate. Categories 'moderately suitable', 'unsuitable' and 'highly unsuitable' all resulted in non-applicable Mann-Whitney scores; this is as all these categories had 0 SI.

Spawning brown trout:

For the majority of time the habitat availability is 'suitable' in wet years and 'moderately suitable' in dry years for spawning brown trout. Overall the dry years provide more suitable habitats despite wet years having a high percentage of habitat at 'suitable', the wet year also has some time at 'highly unsuitable' which the dry year does not.

The 'highly unsuitable' category resulted in N/A scores from the Mann-Whitney tests as all SI results were 0. The same general trends occurred as with the other brown trout lifestages, autumn and winter results did not have statistical differences between wet and dry years as statistically different results were found between hydrologically similar years. However this was not the case for spring and summer.

Crowfoot:

The vast majority of time, the habitat availability is 'suitable' in wet years and 'highly unsuitable' in dry years. The wet years provide more suitable habitats than dry years.

Mann- Whitney tests revealed statistically significant differences between wet and dry years in all seasons, however in autumn and winter many statistically similar results were found between hydrologically similar years i.e. between 2 wet years. Furthermore the statistical differences between wet and dry years were most predominant in categories 'suitable', 'moderately suitable' and 'unsuitable'.

Mayfly:

For the majority of the time, the habitat availability is 'highly unsuitable' in dry years and in wet years however they have different percentages, e.g. 95% of time is 'highly unsuitable; in the driest year (1991) and 34% of time is 'highly unsuitable' in the wettest year (2001). The wet years provide more suitable habitats than dry years.

Non- applicable results were determined from the Mann-Whitney tests in the 'highly unsuitable' category due to all SI results being 0. The same general trends occurred as with the other species, autumn and winter results did not have statistical differences between wet and dry years as statistically different results were found between hydrologically similar years. However this was not the case for spring and summer where statistically different results were found between wet and dry years in all suitability categories.

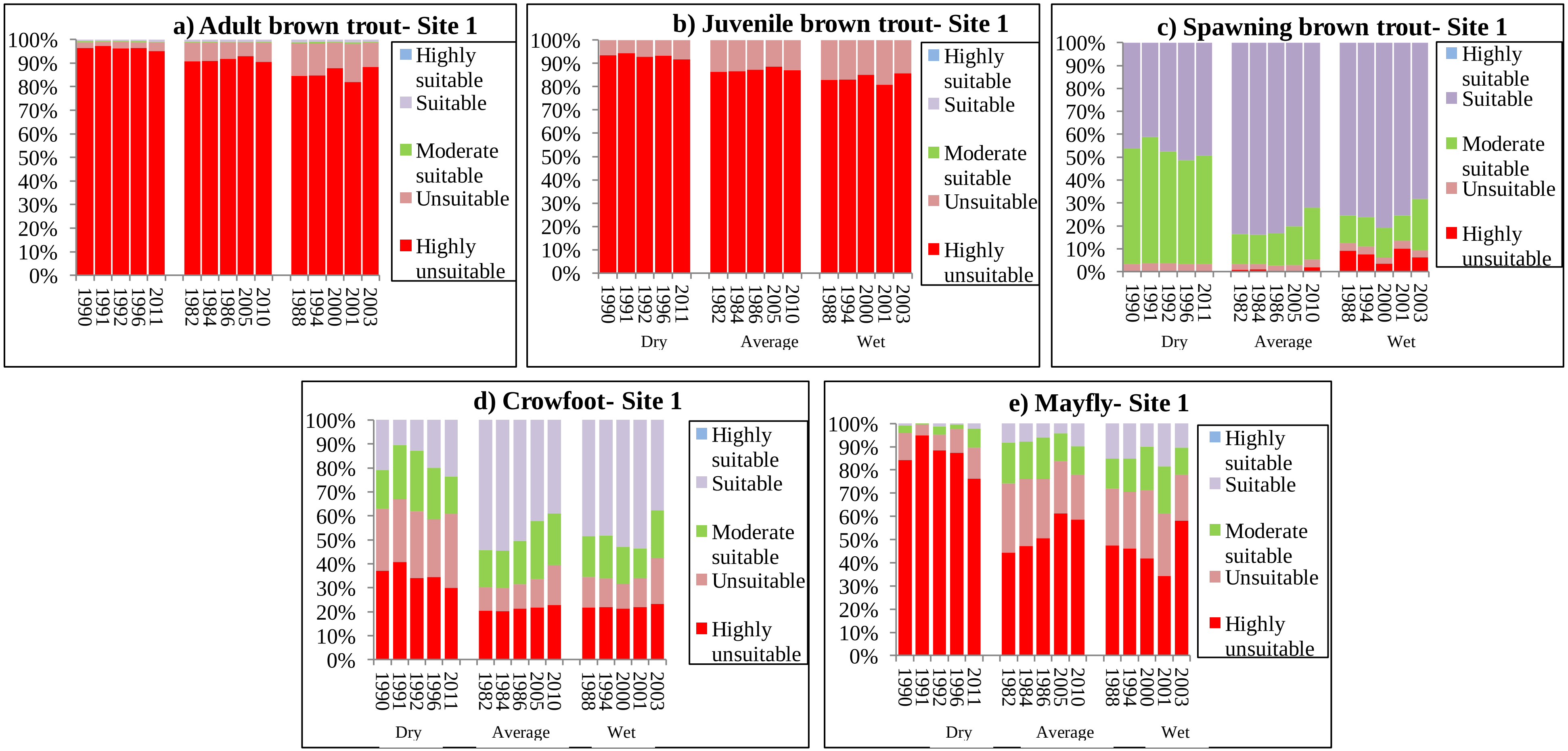


Figure 6.12- Extreme year results for site 1

6.5.3 Site 2- DS Nar (2D)

The results for site 2 are presented in Figure 6.13. These graphs present the amount of available habitat (e.g. highly suitable, moderate etc), for each species in the five wettest, five driest and five average years.

Adult brown trout:

For the majority of time, habitat availability is 'highly unsuitable' in the dry years and wet years however they have different percentages, e.g. 75% in the driest year (1991) and only 57% in the wettest year (2001). Overall the wet years are more suitable as higher percentages of both 'highly unsuitable' and 'suitable' habitat availability for the dry years, and higher percentages for 'suitable' and highly suitable' habitat availability for the wet years.

Mann- Whitney tests revealed statistically different results in all seasons between wet and dry years in all categories apart from some in the 'highly unsuitable' category. There were mostly statistical differences between the same hydrological years in all seasons, however less in summer and spring. This indicates differences shown in the figures between the wet and dry years are not statistically different for autumn and winter months.

Juvenile brown trout:

For the majority of time habitat availability is 'highly unsuitable' in dry years and 'unsuitable' in wet years for juvenile brown trout. The wet years provide more suitable habitat conditions that wet years. Unlike adult fish there are no clear trends in the results for juveniles. It is unclear whether wet conditions or dry conditions are more suited to the species as for 'highly unsuitable' areas, dry years are worse; however for 'unsuitable' areas wet years are worse. The same trend occurs for 'suitable' and 'highly suitable' habitat.

The same general trends occurred as with the other species, autumn and winter results did not have statistical differences between wet and dry years as statistically different results were found between hydrologically similar years. However this was not the case for spring and summer.

Spawning brown trout:

For the majority of time, habitat availability is 'highly unsuitable' in the dry years and 'unsuitable' in the wet years for spawning brown trout. The wet years provide more suitable habitat conditions.

The results are unusual as it is shown how the wet years are most suited with the wet years having more available ‘suitable’ and ‘highly suitable’ habitat. And the dry years have more ‘unsuitable’ habitat than the wet years. This is unusual as spawning brown trout have preferences for shallower depths (Louhi et al., 2008).

Statistical differences were found between wet and dry years in all seasons, however in winter and spring statistical differences were also found between hydrologically similar years, therefore there are not necessarily statistical differences between the wet and dry years which can be seen in the Figure 6.14c.

Crowfoot:

For the majority of time, habitat availability is ‘suitable’ in both wet and dry years for Crowfoot. Overall the wet years provide more suitable habitat than dry years.

The same general trends occurred as with the other species, autumn and winter results did not have statistical differences between wet and dry years as statistically different results were found between hydrologically similar years. However this was not the case for spring and summer.

Mayfly:

For Mayfly, for the majority of time, habitat availability is ‘suitable’ in wet years and ‘highly unsuitable’ in dry years. Overall the wet years provide more suitable habitats than dry years.

The same general trends occurred as with the other species, autumn and winter results did not have statistical differences between wet and dry years as statistically different results were found between hydrologically similar years. However this was not the case for spring and summer.

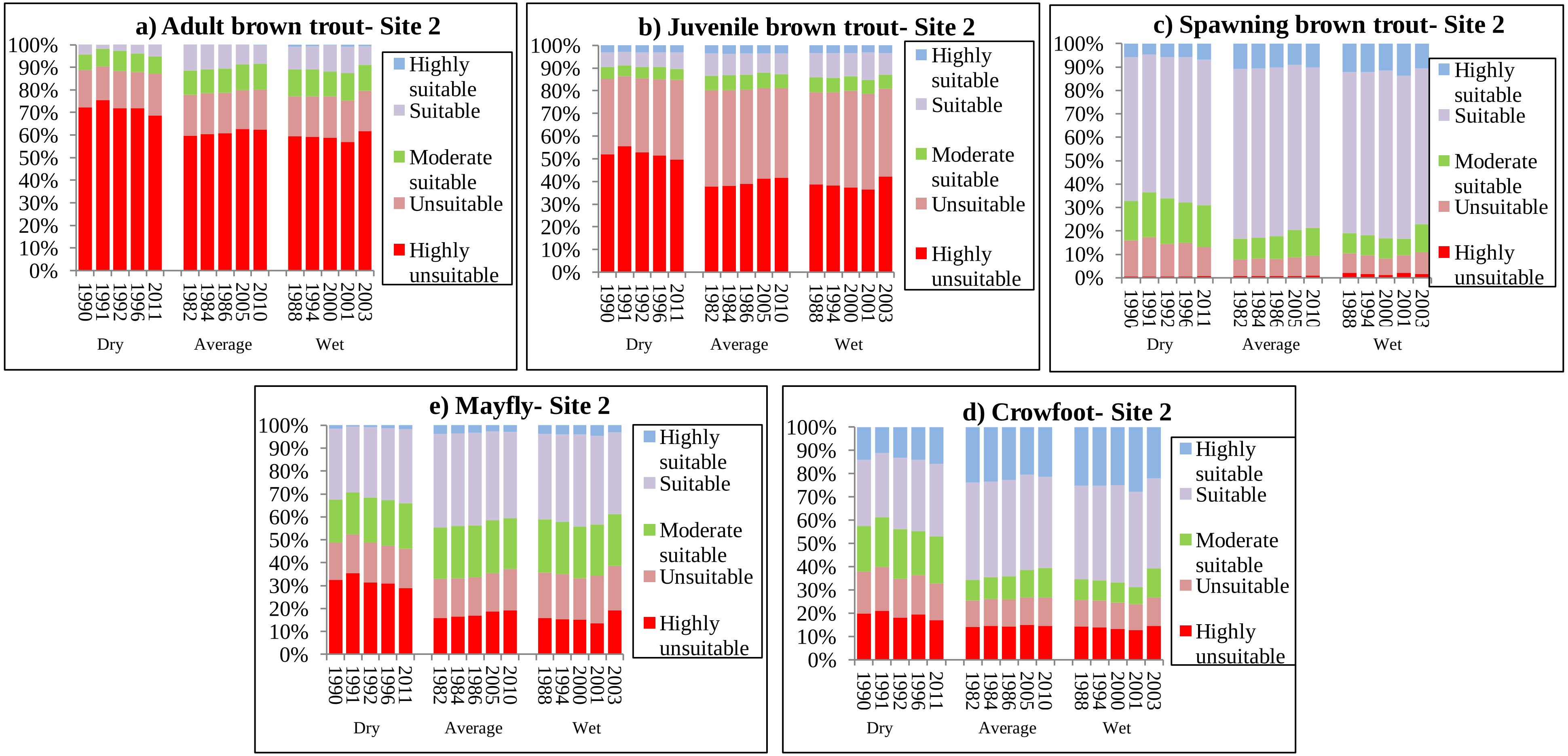


Figure 6.13- Extreme year analysis results for site 2

6.5.4 Site 3- Castle Acre (1D)

The results for site 3 are presented in Figure 6.14. These graphs present the amount of available habitat (e.g. highly suitable, moderate etc), for each species in the five wettest, five driest and five average years.

Adult brown trout:

The most predominant available habitat for adult brown trout is 'highly unsuitable' with dry years providing slightly worse habitat (highly unsuitable) than wet years.

Mann- Whitney tests revealed statistically different results between wet and dry years in all seasons however in autumn and winter generally statistical differences were also found between hydrologically similar years (i.e. between 2 wet years), therefore the differences between the wet and dry years seen in Figure 6.15a are not statistically significantly different in autumn and winter.

Juvenile brown trout:

Fairly even results were shown between wet, dry and average years for juvenile brown trout with no large differences in results between wet, dry or average years. This indicates that the yearly flow changes do not have much of an impact on the species.

The same general trends occurred in the Mann-Whitney tests for juvenile brown trout that wet and dry years were generally statistically different however for autumn and winter, statistically different flows were also found between hydrologically similar years (i.e. between 2 dry years). During spring however some occurrences of statistically similar results between dry and wet years occurred, particularly for the highly unsuitable category. Figure 6.15a however demonstrates different results. This therefore shows how the graphs can present different findings and that the habitat availability amount can still be statistically different.

Spawning brown trout:

For spawning brown trout the Mann-Whitney trend is specifically seen for the unsuitable, suitable and highly suitable categories. During wet years a higher proportion of time is spent in the unsuitable category and during drier years, the results tended towards suitable or highly suitable habitat suitability.

Crowfoot:

For Crowfoot the wet years provided more suitable habitat than the dry years, this can be seen by having more 'highly suitable' habitat in the wet years. During dry years the results tended to lower classes while wetter years had a greater proportion of highly suitable habitat. The difference between different hydrological years was particularly pronounced during the summer season for Crowfoot, whilst the trend was much weaker during spring. The same general trends occurred for Crowfoot as with the other species in the Mann-Whitney results.

Mayfly:

For Mayfly, the habitat suitability tended to be poorer (unsuitable) during dry years, whilst wet years offered better (suitable) habitat. A good range of all suitability's i.e. highly suitable, unsuitable etc. was available throughout all years as opposed to the fish species which had their majority of time in 1 suitability class. The same general trends occurred for Crowfoot as with the other species in the Mann-Whitney results.

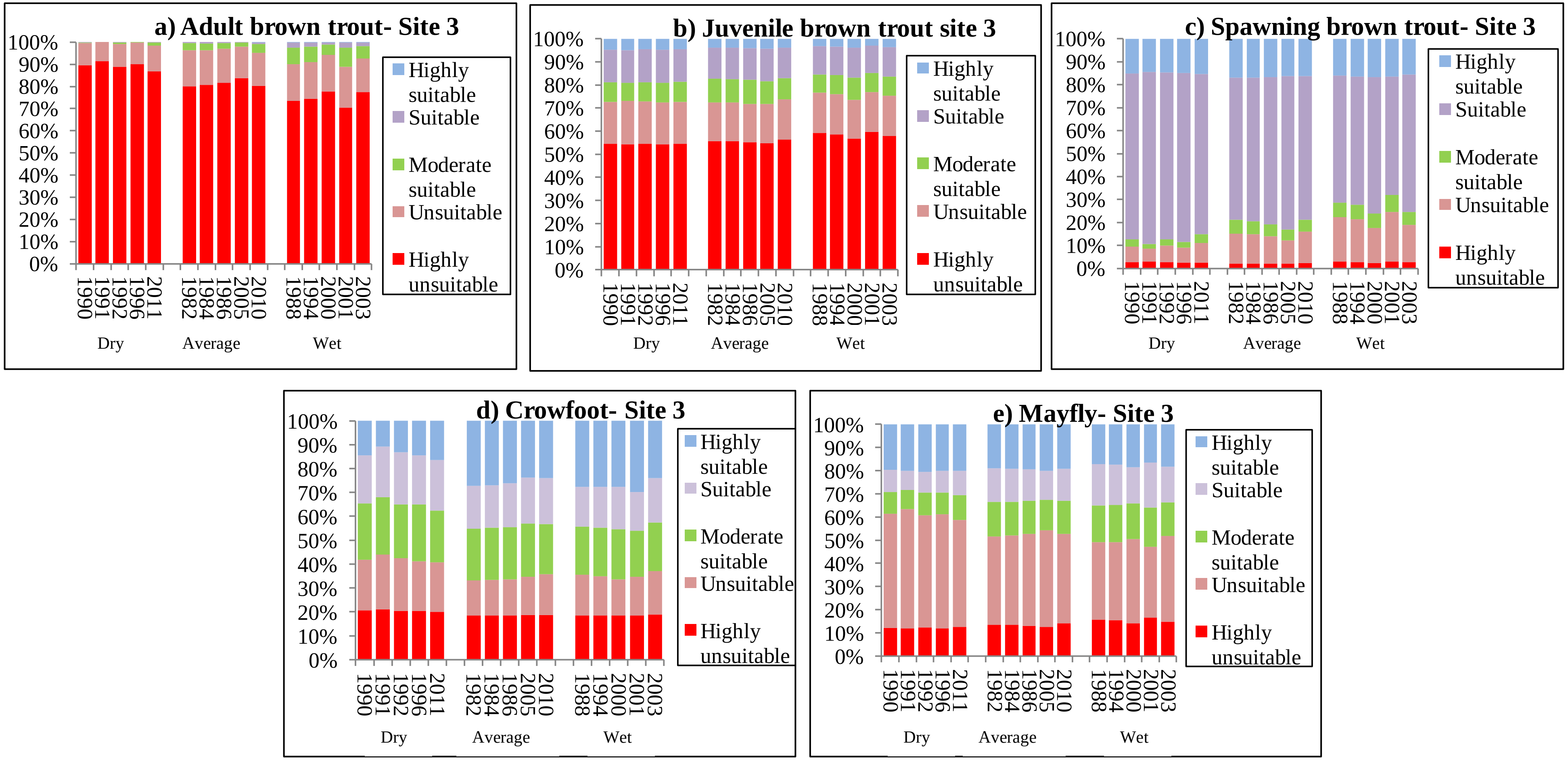


Figure 6.14- Extreme year results for site 3

6.5.5 Summary of analysis

Overall this analysis suggests that different conditions are preferred by different species and that low flows during dry years are good for spawning fish however these conditions provide less habitat availability for Crowfoot and Mayfly. The Mann-Whitney tests reveal that these observations are robust for spring and summer results however are less evident for winter and autumn.

6.6 Analysis 5- Key times for species

This analysis focused on showing how key times of the year for spawning brown trout and Crowfoot are affected by the hydraulic conditions.

6.6.1 Brown trout spawning (October-December)

Results are demonstrated in Table 6.5 for each site showing percentage of days between October and December at or below 'low' HHS in each year.

The results demonstrated that the key times for spawning are not affected by 'low' HHS to any great extent. Only 7 years out of 32 had any days below the 'low' HHS classification. Of these 7 years the maximum was 11% of the time for site 1 in 1993. Historical flows indicated that between 1980 and 2011, the highest Autumn Q_{10} was recorded in 1993 (2.77m^3/s at Marham). The years 1987 and 2000 had 9% and 5% respectively of their time at low HHS; these sites also had very high Q_{10} autumn flows (2.45 and 2.69m^3/s respectively). This therefore explains why these years experienced 'low' HHS during this time and that it is not the low flow causing the 'low' HHS but instead the high flow.

There was no connection between the driest years being affected and the percentage of time at low habitat availability, this is due to spawning brown trout having a preference for lower depths and velocities meaning that the conditions in low flow years would be more preferential to conditions in high flow years (Louhi et al., 2008). Furthermore the spawning period is not in a time of usual high or low flows i.e. winter or summer. The years in which low HHS did occur during October to December had conditions which were not preferable to the species i.e. higher flows.

Table 6.5- Results for key times for spawning brown trout

Year	Order: Driest to wettest	Site 1-(2D)	Site 2-(2D)	Site 3-(1D)
1991	1	0	0	0
2011	2	0	0	0
1990	3	0	0	0
1992	4	1	0	0
1996	5	0	0	0
1989	6	0	0	0
2009	7	0	0	0
2006	8	0	0	0
1995	9	0	0	0
1997	10	0	0	0
2003	11	0	0	0
2010	12	0	0	0
1993	13	11	2	3
1999	14	0	0	0
2002	15	3	0	2
2005	16	0	0	0
2000	17	5	0	1
1986	18	0	0	0
1980	n/a	0	0	0
1981	n/a	0	0	0
1982	n/a	2	0	0
1983	n/a	0	0	0
1984	n/a	0	0	0
1985	n/a	0	0	0
1987	n/a	9	0	3
1988	n/a	0	0	0
1994	n/a	0	0	0
1998	n/a	0	0	0
2001	n/a	0	0	0
2004	n/a	0	0	0
2007	n/a	1	0	0
2008	n/a	0	0	0

6.6.2 Crowfoot growing season (April- August)

Results are demonstrated in Table 6.6 for each site showing percentage of days between April and August at or below ‘low’ HHS in each year.

The Crowfoot growing season (April-August) was much more affected by times of ‘low’ HHS availability than spawning brown trout were. For site 1 and 3, 12 years out of 32 had ‘low’ HHS periods. For site 2, 6 years out of 32 had ‘low’ HHS periods.

There was a strong relationship between percentage of time affected and dry years. Of the 12 years that were affected for site 3, 9 of these were in the top 10 driest years. As the time period (April-August) coincides with the seasonal times of lowest flows, it is clear that Crowfoot does not prefer times of low flow. The top 9 driest years also had the lowest summer flows. 1991 had an extremely low summer Q_{90} of $0.15 m^3/s$ (at Marham), this explains why the 9 years had so much of their time classified as ‘low’ HHS.

Three anomalies occurred in the results, 1998, 1981 and 2002. Historical flows indicated that the summer flows in these years were not particularly low, however for 1998 and 1996 the Q_{10} was reasonably high (1.05 and 1.27m^3/s respectively). Therefore this indicates that the flow was high enough in these years to cause low HHS. The year 2002 however had relatively average flow conditions; it is therefore unknown why this year had some percentage at low HHS.

Table 6.6- Results for key times for Crowfoot growing

Year	Order: Driest-wettest	Site 1 (2D)	Site 2 (2D)	Site 3 (1D)
1991	1	34	22	37
2011	2	24	0	35
1990	3	35	21	40
1992	4	51	24	56
1996	5	52	38	54
1989	6	8	0	15
2009	7	8	0	12
2006	8	25	11	30
1995	9	14	0	20
1997	10	0	0	0
2003	11	0	0	0
2010	12	0	0	8
1993	13	0	0	0
1999	14	0	0	0
2002	15	5	1	5
2005	16	0	0	0
2000	17	0	0	0
1986	18	0	0	0
1980	n/a	0	0	0
1981	n/a	1	0	1
1982	n/a	0	0	0
1983	n/a	0	0	0
1984	n/a	0	0	0
1985	n/a	0	0	0
1987	n/a	0	0	0
1988	n/a	0	0	0
1994	n/a	0	0	0
1998	n/a	1	0	0
2001	n/a	0	0	0
2004	n/a	0	0	0
2007	n/a	0	0	0
2008	n/a	0	0	0

6.6.3 Summary of analysis

The main finding from this analysis is that the habitat availability for spawning brown trout is affected by autumn high flows whilst Crowfoot are affected by summer low flows. Whilst it is reasonably easy to protect low flows by using HOF's, it is more difficult to protect against high flows.

6.7 Analysis 6- Interconnectedness of species

This analysis aimed to reduce criticism surrounding habitat modelling by incorporating biotic parameters other than the traditional depth, velocity and substrate into the model results. Results were combined to show how much habitat availability spawning brown trout have including their biotic dependants of food sources (BMI; Mayfly) and refugia (macrophytes; Crowfoot). Part A determined the critical flows below which spawning habitat availability would be compromised. Part B examines the different scenarios at which a habitat could be at any 1 time, i.e. 'upper' habitat availability for spawning brown trout whilst refugia and food sources have 'lower' availability. This analysis was only carried out on sites 2 and 3, as site 1 is not an important area for spawning brown trout. Table 6.7 demonstrates the 27 different scenarios used and Appendix M presents the full tables of results for each site.

Table 6.7- Scenarios used for analysis 6, colours correspond to colours used in results

Scenario	Spawning brown trout	Refugia (Crowfoot)	Food (Mayfly)
1	Upper	Upper	Upper
2	Upper	Upper	Middle
3	Upper	Upper	Lower
4	Upper	Middle	Upper
5	Upper	Middle	Middle
6	Upper	Middle	Lower
7	Upper	Lower	Upper
8	Upper	Lower	Middle
9	Upper	Lower	Lower
10	Middle	Upper	Upper
11	Middle	Upper	Middle
12	Middle	Upper	Lower
13	Middle	Middle	Upper
14	Middle	Middle	Middle
15	Middle	Middle	Lower
16	Middle	Lower	Upper
17	Middle	Lower	Middle
18	Middle	Lower	Lower
19	Lower	Upper	Upper
20	Lower	Upper	Middle
21	Lower	Upper	Lower
22	Lower	Middle	Upper
23	Lower	Middle	Middle
24	Lower	Middle	Lower
25	Lower	Lower	Upper
26	Lower	Lower	Middle
27	Lower	Lower	Lower

6.7.1 Site 2- DS Nar (2D)

Part A) Critical flows

Figure 6.15 presents the results for the critical flows at site 2.

- Low flows associated with low availability for spawning brown trout, food sources and refugia are $0.1m^3/s$, $0.27m^3/s$ and $0.23m^3/s$ respectively. Thus flows should not fall below $0.27m^3/s$ in order to protect overall habitat for spawning brown trout.
- The HOF limit at Marham is a fairly high Q_{33}, this corresponds to a flow of $1.11m^3/s$. Under the new abstraction reform EFI, there is a minimum flow of $1.05m^3/s$. Therefore HOF and EFI limits do adequately protect spawning brown trout at this site.
- The upper limits of the low HHS also have to be taken into account. When flows get to $4.26m^3/s$ and $4.94m^3/s$, the available habitat becomes low for fish and food respectively. Therefore when flows get to $4.26m^3/s$ overall habitat for spawning brown trout becomes low.
- The ideal flow for spawning brown trout is between $0.27m^3/s$ and $4.26m^3/s$ including the biotic parameters they require. Whilst ensuring the upper limit of this is unrealistic due to the natural flow regime. It can be seen the vital importance that flows should not fall below $0.27m^3/s$.

<u>Part B) Seasonal scenario analysis</u>

Figure 6.16 presents the results for the critical flows at site 2. Scenarios (see Table 6.7): 1, 2, 5, 10, 11, 13, 14, 15, 18 and 20 occurred at this site.

Winter:

- The best available habitats occurs in average years where the majority of time is S1.
- Wet winters provide less preferable habitat than dry winters due to higher occurrence of lower scenarios such as S20.
- Dry winters provide predominantly S1 scenarios.

Spring:

- Wet and average springs have only S1 scenarios which is the best case scenario.
- Dry springs are however more varied, with some occurrences of S18

Summer:

- A clear trend occurs for summer with wet years providing best habitat, followd by average years and finally with dry years providing the worst habitat
- Very little S1 or S2 habitat occurs in dry summers, they are dominated by S18 and S15.

Autumn:

- Autumn results are very similar to summer results in that a clear trend occurs with wet providing best, followed by average and then dry years providing the worst habitat availability.

Overall average years in all seasons provide the best habitat (i.e. predominantly S1 scenarios). Dry summers and autumns provide the worst habitat (i.e. predominantly S18 and S15 scenarios). This finding shows that low flows do cause an overall decline in habitat availability.

6.7.2 Site 3- Castle Acre (1D)

<u>Part A) Critical flows</u>

Figure 6.15 presents the results for the critical flows at site 3.

- Low flows associated with low availability for spawning brown trout, food sources and refugia are $0.1m^3/s$, $0.13m^3/s$ and $0.18m^3/s$ respectively. Thus flows should not fall below $0.18m^3/s$ in order to protect overall habitat for spawning brown trout.
- The HOF limit at Marham is a fairly high Q_{33}, this corresponds to a flow of $0.63m^3/s$. Therefore HOF limits do adequately protect spawning brown trout here.
- The upper limits of the low HHS also have to be taken into account. When flows get to $2m^3/s$, $3.7m^3/s$ and $4.3m^3/s$, the available habitat becomes low for fish, refugia and food respectively. Therefore when flows get to $2m^3/s$ overall habitat for spawning brown trout becomes low.
- The ideal flow for spawning brown trout is between $0.18m^3/s$ and $2m^3/s$ including the biotic parameters they require. Whilst ensuring the upper limit of this is unrealistic due to the natural flow regime. It can be seen the vital importance that flows should not fall below $0.18m^3/s$.

<u>Part B) Seasonal scenario analysis</u>

Figure 6.17 presents the results for the critical flows at site 2. Scenarios: 1, 2, 4, 6, 9, 13 and 23 occurred at this site (see Table 6.7).

Winter:

- The best available habitats occurs in average years
- Wet winters have the worst scenarios of S13 and S23. These are the highest amounts of the lower scenarios throughout all seasons.
- S13 and S23 only occur in one of the dry years and this is a very small amount
- All other dry years have only S1 and S2, showing the habitat availability in dry winters is generally good.

Spring:

- The vast majority of time is S1 and S2 for all dry, wet and average years indicating that spring provides good habitat in all years.
- Average springs provide the best habitat which are all S1 or S2.
- Wet springs have some S13 which indicates wet years provide the least habitat availability for all species combined.
- Dry springs are mostly S1 or S2 with some S9 and S13.

Summer:

- Mostly S1 and S2 in wet and average summers
- Dry summers have mostly S9 which is the worst of all the scenarios present in summer but is still a relatively good scenario (Upper, lower, lower)

Autumn:

- Mainly S1 and S2 in wet and average autumns
- Dry autumns have S6, S9 and S13

Overall scenario 1 and 2 occurs most frequently during average years and scenario 23 occurs most frequently during wet winters, thus indicating that wet winters provide the worst habitat for all species combined. Dry years predomninantly had scenarios 9, and 13 indicating the habitat is less preferable for all the species together.

6.7.3 Presentation of results

<u>Part A- Critical flows</u>

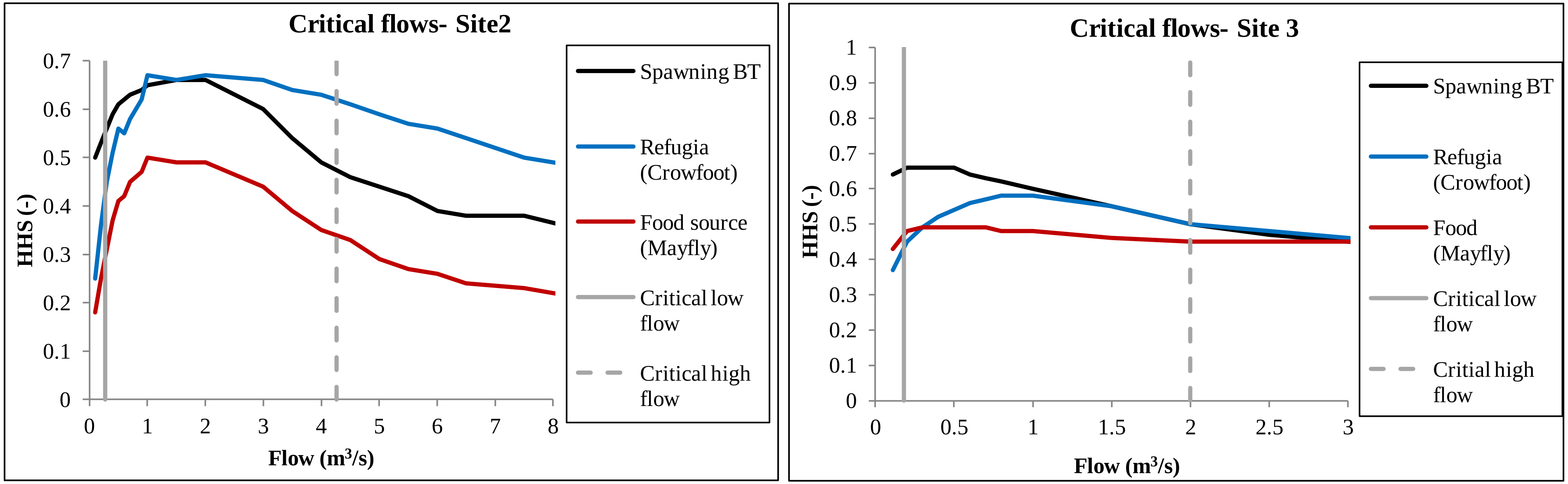


Figure 6.15- Critical flow for spawning brown trout, site 2 and site 3

<u>Part B) Seasonal scenario analysis</u>

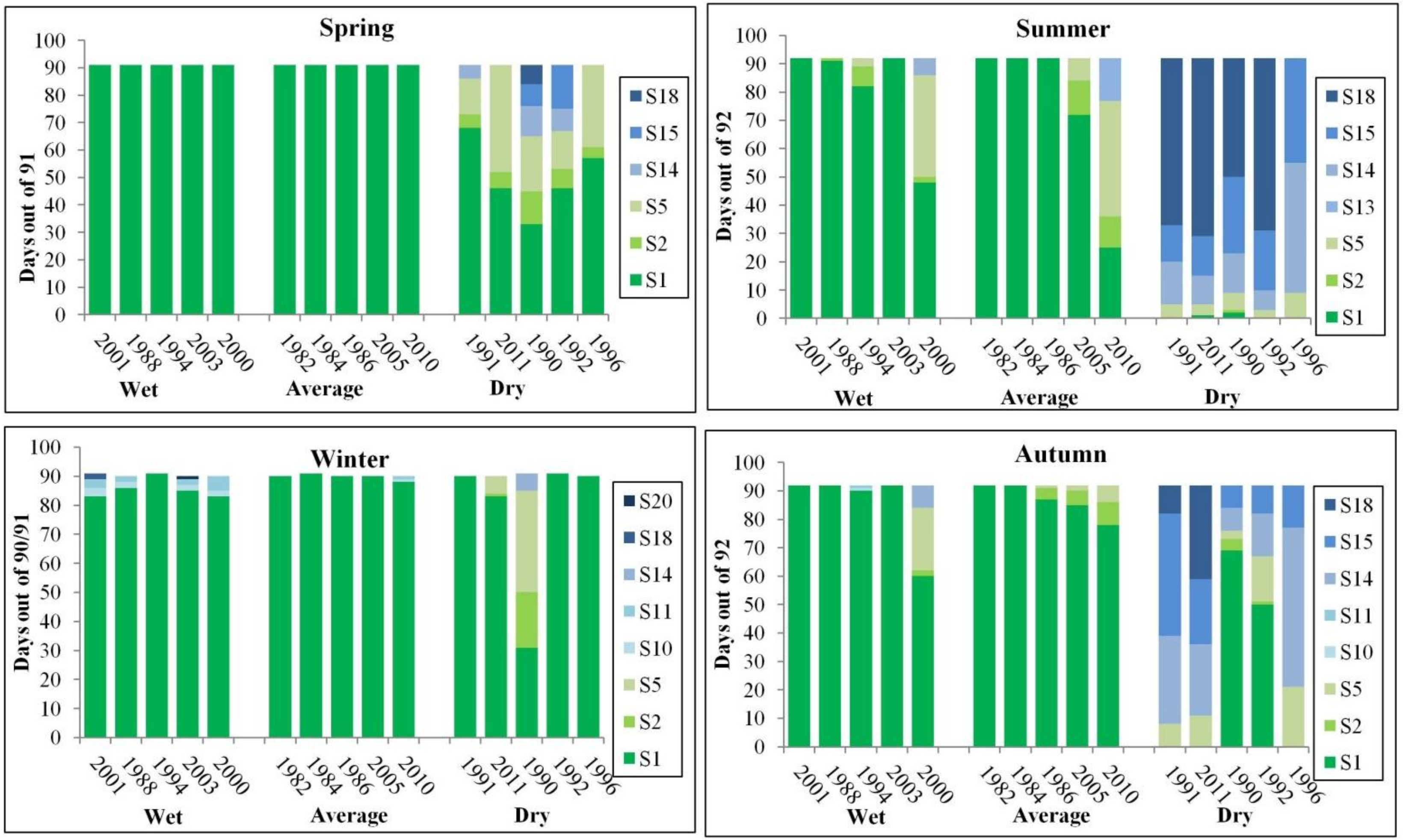


Figure 6.16- Seasonal scenario analysis for site 2

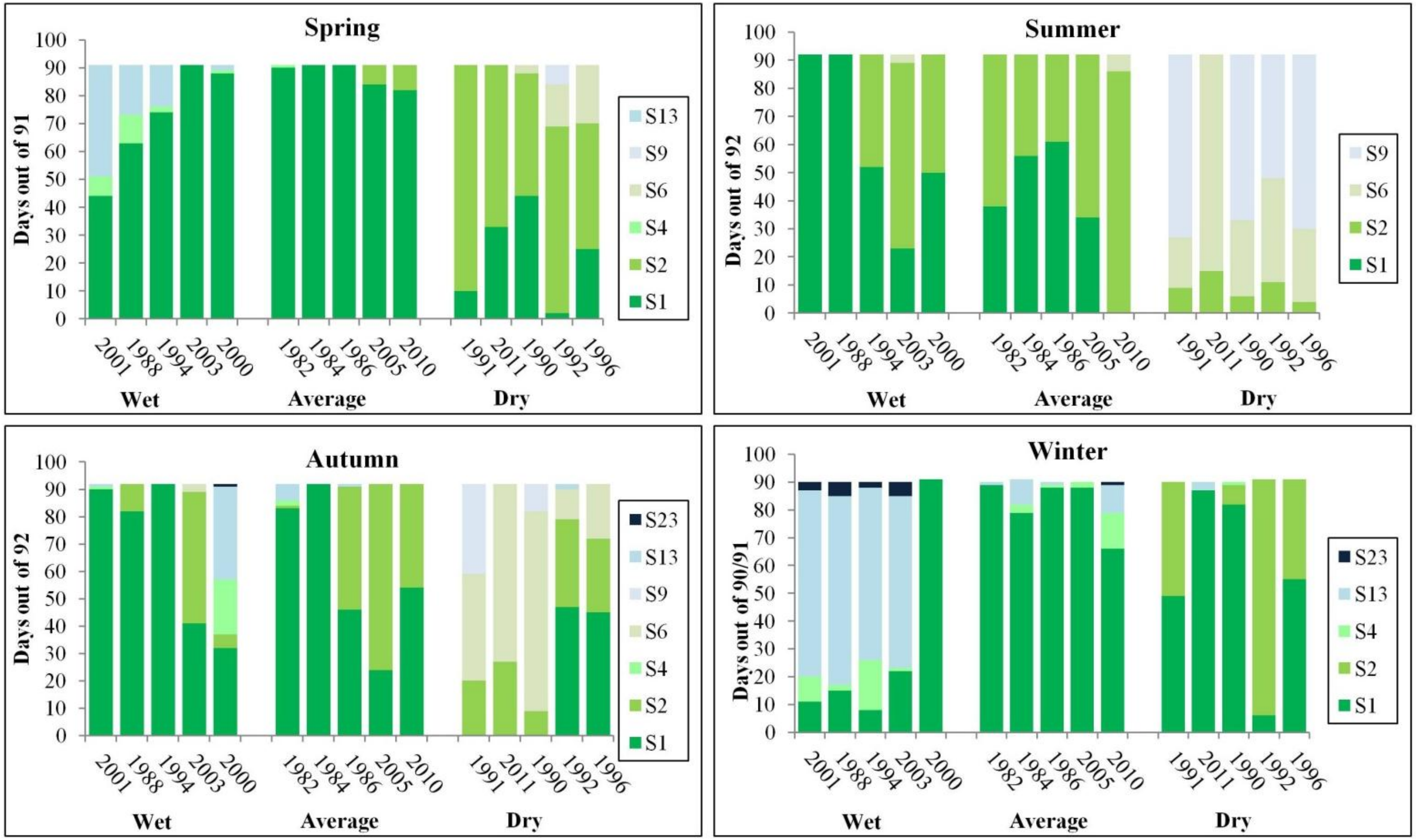


Figure 6.17- Seasonal scenario analysis for site 3

6.7.4 Summary of analysis

For both sites, average flow years provided the best overall habitat. Dry summers provided the worst overall habitat. Wet winters provide the worst habitat in site 3 but not in site 2, this indicates site conditions are very important and influential on habitat availability rather than only flow and season.

The results show that low flows do have a negative impact on spawning brown trout and their biotic components, this is an important finding as when habitat availability for spawning brown trout alone is assessed, the dry year conditions are preferable (analysis 4 and 5 shows dry years are most preferable for spawning brown trout when assessed individually). This shows the importance of incorporating biotic dependants into decision making, rather than focusing on individual species requirements.

6.8 Analysis 7: Spatial distribution

This analysis was linked to analysis 6 investigating how habitat availability is affected for spawning brown trout and their biotic dependants. Spatial analysis of different significant flow conditions (Q_{10}, Q_{50} and Q_{90}) was undertaken to investigate areas of potential increase or decrease in predicted available habitat for spawning brown trout based on their biotic dependants. By assessing where the best locations were for each species, an assessment of optimum habitat overlap could be undertaken.

6.8.1 Site 2- DS Nar (2D)

Figure 6.18 presents the findings from this analysis. The whole site shows relatively good habitat availability for spawning brown trout, with SI values of around 0.5 along the whole length. However as already shown in analysis 6, the available habitat for refugia and food sources are of importance for spawning brown trout habitat availability. Whilst the whole area is relatively favourable, the output must be looked at in more detail as the areas of good suitability for refugia and food sources are more likely to be used by the spawning fish. Therefore the habitat model output (e.g. HHS and SI) for spawning brown trout alone gives limited results as higher SI values would actually be found in the areas where refugia and food sources are higher. For example are the areas shown in black boxes in Figure 6.20 where spawning brown trout have a SI of 0.5, in the same area refugia and food sources have an SI of 0.8 and 0.9 respectively, this would increase the predicted habitat availability as depicted solely by spawning brown trout. Therefore this highlights the importance of looking in more detail at the output from habitat modelling and the importance of incorporating other biotic parameters to habitat models.

6.8.2 Site 3- Castle Acre (1D)

The middle of the reach tends to provide the best availability for all species (SI=0.8/0.9), while the downstream reaches provide SI's of around 0.3/0.4 for food (Mayfly) and refugia (Crowfoot) however slightly higher availability for spawning brown trout.

There is a small area towards the bottom of the reach (shown in the black box) where the SI increases to 0.9 for food (Mayfly) and refugia (Crowfoot) indicating a important area for these species, but interestingly not for spawning brown trout. This is due to a combination of hydraulic conditions (depth and velocity) being present which is preferred by food (Mayfly) and refugia (Crowfoot) but not by spawning brown trout. This hotspot is of importance as whilst the habitat model for spawning fish predicts that area to have low suitability, the fish are still likely to use the area in a transient manner due the presence of the biotic parameters. Analysis of the spatial distribution of habitat varies for reach flow conditions as a function of the species requirements. However what is important to note is that an overlap of hotspots for habitat for all three species does occur within the reach.

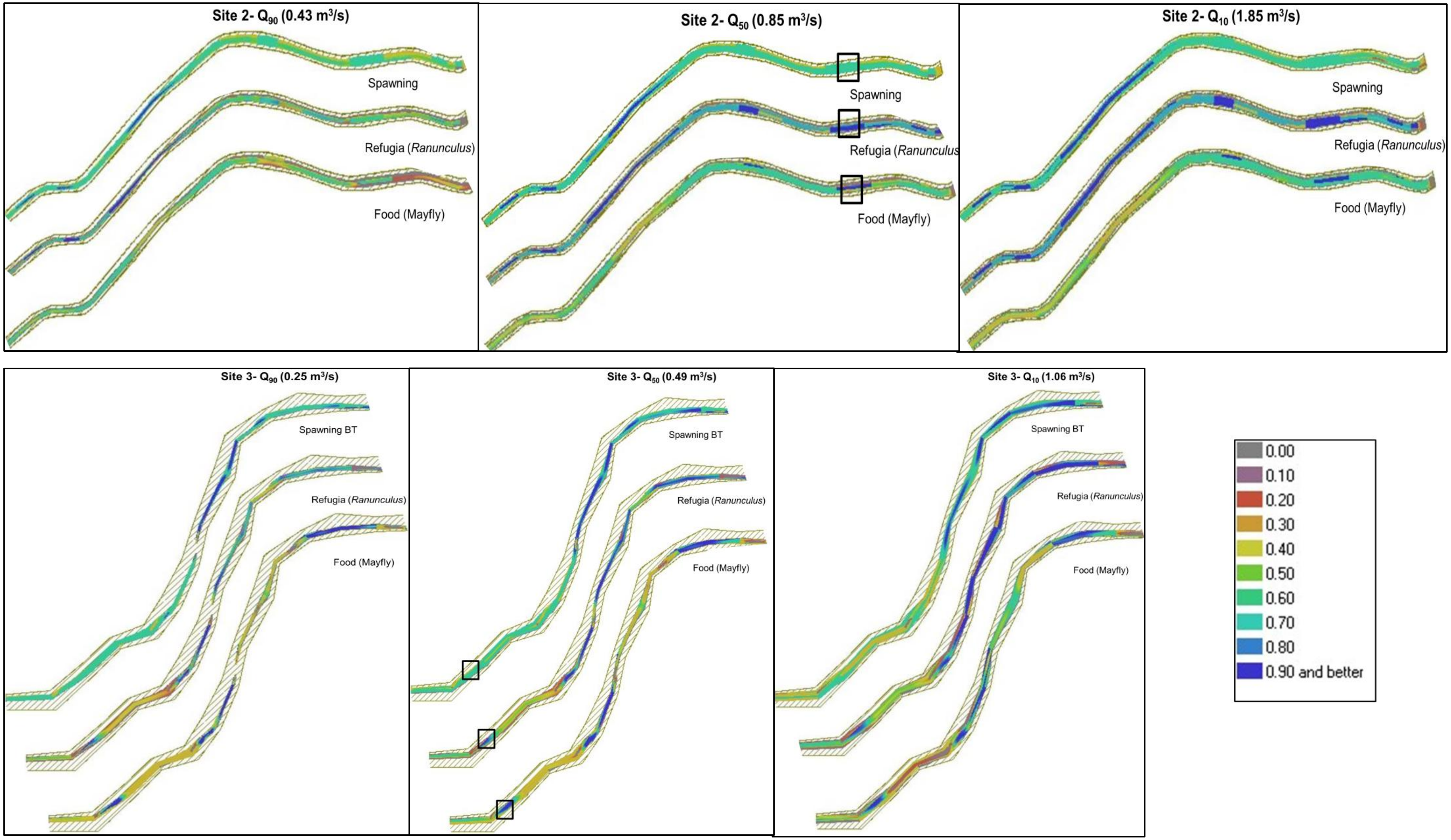


Figure 6.18- Spatial distribution analysis of spawning brown trout and their biotic dependants

6.8.3 Summary of analysis

The results have demonstrated the importance of assessing spatial distributions in addition to the interconnectedness investigation in analysis 6. The main output from habitat models is the HHS which provides 1 number to represent the available habitat. This HHS value can be disaggregated into more detailed SI values. These fail to incorporate other biotic parameters i.e. food sources and refugia. Important areas do occur where spawning brown trout are predicted to have relatively medium available habitat whereas the refugia and food sources have quite high available habitat, this means that spawning brown trout maybe more likely to use the area than the individual spawning brown trout model predicts.

6.9 Analysis 8- Key comparisons between 1D and 2D for site 2

This analysis aimed to show the key differences between the 1D and 2D results for site 2 (DS Nar). Analysis 2 (habitat distribution) was carried out comparing the 1D and 2D results to show how the distribution changes dependent on the method used. Furthermore Mann-Whitney tests were carried out to show if there was statistically significant differences between 1D and 2D results.

Table 6.8- 1D and 2D statistics of 1D and 2D results comparison

	Adult brown trout		**Juvenile brown trout**		**Spawning brown trout**		**Mayfly**		**Crowfoot**	
	1D	**2D**	**1D**	**2D**	**1D**	**2D**	**1D**	**2D**	**1D**	**2D**
Average/ mean	0.22	0.18	0.25	0.27	0.61	0.63	0.44	0.44	0.62	0.60
Median	0.24	0.20	0.26	0.28	0.62	0.64	0.47	0.46	0.65	0.61
Maximum	0.40	0.28	0.30	0.30	0.66	0.66	0.56	0.50	0.72	0.67
Minimum	0.03	0.03	0.10	0.12	0.25	0.39	0.18	0.19	0.25	0.26
95 percentile	0.08	0.08	0.16	0.18	0.52	0.55	0.25	0.30	0.41	0.43
50 percentile	0.24	0.20	0.26	0.28	0.62	0.64	0.47	0.46	0.65	0.61
5 percentile	0.30	0.22	0.29	0.30	0.66	0.66	0.56	0.50	0.72	0.67
Standard Deviation	0.07	0.05	0.04	0.04	0.05	0.03	0.10	0.06	0.10	0.08
Skew	-0.64	-1.13	-0.85	-1.34	-1.31	-1.42	-0.66	-1.44	-1.08	-1.19
Kurt	-0.14	0.56	-0.29	1.15	2.92	1.85	-0.68	1.55	0.45	1.19

Table 6.8 demonstrates that all species have different results based on 1D or 2D outputs. Adult brown trout had the largest difference; the average for 1D was HHS 0.22, whereas the average for 2D was HHS 0.18. Mayfly on the other hand resulted in the same average for 1D and 2D but had different maximum and minimums. The Mann-Whitney tests (Table 6.9) revealed statistically significant differences between the 1D and 2D results. The results are discussed per species below and the results are presented in figures and tables in section 6.9.6. These results are discussed in more detail in the sections below for each species.

Table 6.9- Mann-Whitney results for 1D and 2D comparison

	p- value
Adult brown trout	0.000
Juvenile brown trout	0.000
Spawning brown trout	0.000
Crowfoot	0.000
Mayfly	0.000

6.9.1 Adult brown trout

Adult brown trout had the largest difference in results between 1D and 2D simulations for all species. Figure 6.21a shows how the same pattern occurs throughout the 32 year period i.e. peaks and troughs at the same time. This is due to the same fuzzy rules being used; however the 1D results are generally larger than the 2D results. This is further demonstrated in that the average HHS is higher by 0.04 in the 1D result. Figure 6.21aa also demonstrates how the data is more evenly distributed for 1D and that there is an increased negative Skew in the 2D results. There is a positive Kurt value for 2D (0.56) which indicates the data is peaked, however there is a negative Kurt value (-0.14) for the 1D results which indicates the data distribution is flat. The maximum HHS reports the largest difference, of 0.4 in 1D and of 0.28 in 2D results.

Overall the 1D and 2D methods resulted in very different data distributions for adult brown trout, the Mann-Whitney results further supported this by showing statistically significant differences between the 1D and 2D results. This implies that the 1D and 2D results give statistically different habitat availability results therefore showing the sensitivity of results and promoting the importance of the different methods used. This is discussed further in Section 8.5.1.

6.9.2 Juvenile brown trout

Figure 6.21b and 6.21bb presents the results for juvenile brown trout. The 1D and 2D results for juvenile brown trout were generally fairly similar. The 1D HHS average is only 0.02 lower than the 2D. Furthermore the maximum values are the same for 1D and 2D and the minimum value is lower by only 0.02 in the 1D results. The distribution is more highly negatively skewed in the 2D results. This however the data distributions for juvenile brown trout are not affected much by the 1D and 2D methods. However the Mann-Whitney tests proved there were statistical differences in the results.

6.9.3 Spawning brown trout

The differences in the 1D and 2D results for spawning brown trout are similar to those of juvenile brown trout. Figure 6.21c and 6.21cc presents the results for spawning brown trout. The 1D results have a lower average by 0.02, the maximum values are the same but the minimum is lower by 0.14 which is a significant difference. The distribution of data is very similar for 1D and 2D results, with a very similar negative Skew (-1.31 and -1.42 respectively). Like the other fish species however, the Mann-Whitney tests revealed there were statistical differences between the 1D and 2D results

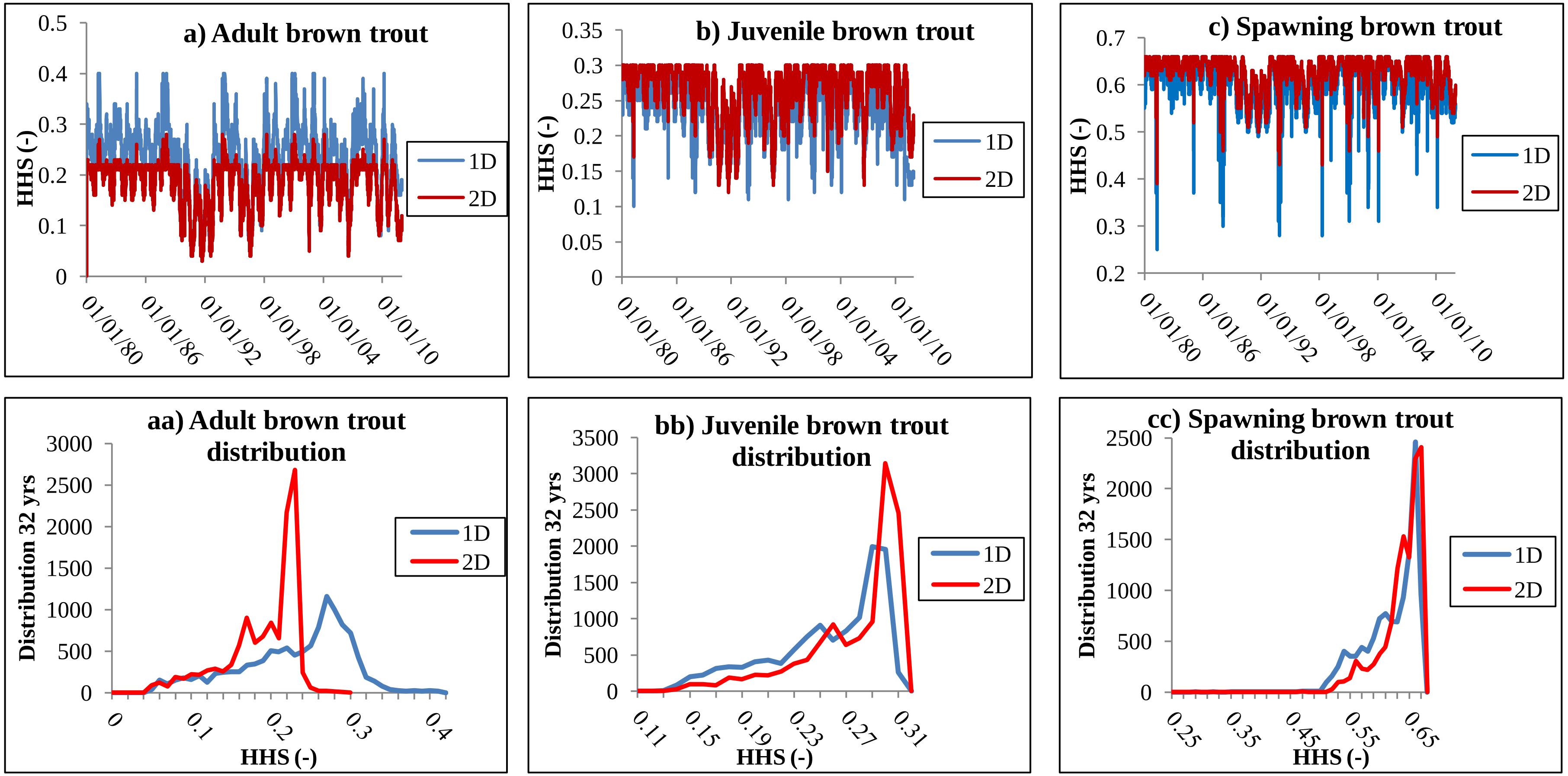
6.9.4 Crowfoot

Statistically significant differences were found between the 1D and 2D results for Crowfoot. Figure 6.21d demonstrates how the 1D results are slightly higher than the 2D results for Crowfoot, the same pattern is followed however (i.e. peaks and troughs at the same time). The distribution appears very different between the two results while the Skew value is similar for both. The Kurt value however exhibits differences, with a value of 0.45 for 1D and 1.19 for 2D, this indicated 2D results have a much lower peak.

6.9.5 Mayfly

There is a high positive Kurt value for 2D (1.55) which indicates the data is peaked, for 1D however there is a high negative Kurt value (-0.68) which indicates the data is flat. This demonstrates a large difference in the data distribution for Mayfly. Furthermore Figure 6.21e shows how the 1D results are generally higher for 1D results. For all species, Mann-Whitney tests revealed statistically significant different results between 1D and 2D input methods.

6.9.6 Presentation of results



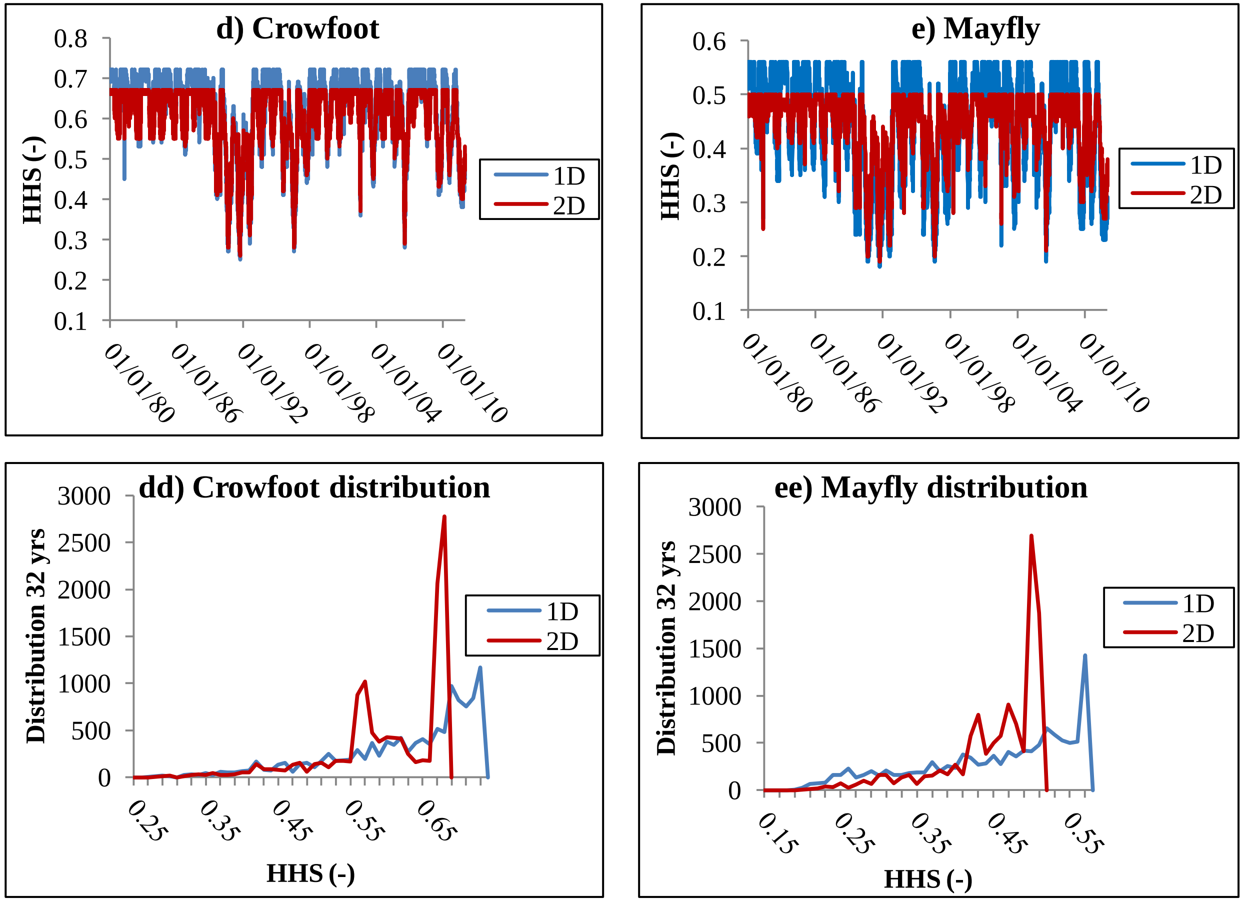


Figure 6.19- 1D and 2D comparison results for each species

6.9.7 Summary of analysis

The 1D and 2D inputs depicted different results for each species, the Mann-Whitney tests proved this by revealing statistically significant differences in the results. Neither 1D nor 2D results were consistently lower or higher for all species, generally however 1D results had much higher maximum HHS values. This indicates the importance and sensitivity of the methods used in habitat models.

6.10 Errors associated with model data

The models provide perhaps the largest level of uncertainly in the project. Barry and Elith (2006) discuss how errors are almost certain in habitat models and understanding the source and magnitude of these errors is essential if the models are to be used for decision making. From data collection (cross section determination), to calibration to habitat data input, areas of uncertainty are likely to occur. Likewise with the data collection, all efforts were made to avoid this. For example, in the cross section collection, any transects/ cross sections were removed from use which had any potential issues (see section 4.2.1).

Sensitivity tests were carried out on habitat data input to determine what would happen if slightly different scores were used (see section 4.7.6). Futhermore the model results were not used in isolation and instead were combined with findings from RQ1 where species specific information was determined, for example the model showed adult brown trout to have low habitat availability during low flows, this finding was corresponded to the electro-fishing data analysis in RQ1 which clarified these were the least preferred times and that antecedent flow conditions were more important than daily flow conditions.

6.11 Chapter summary

This chapter has investigated two main areas, firstly to assess how different flows, particularly low flows affect habitat availability of the ecosystem indicators. Secondly to investigate the sensitivity of input and therefore determine how useful the models are in investigating the impacts of flow on ecosystem indicators.

Overall it has become clear that there is a site specific nature to the habitat availability at each site and that the localised site conditions have a large impact on availability. Furthermore generally low flows do not cause ‘low HHS’, again this is species and site specific. The extreme year analysis showed that low flows during dry years are good for spawning fish however these conditions provide less habitat availability for Crowfoot and Mayfly. However when other biotic factors i.e. refugia and food sources are taken into account, it has been shown that low flows do have a negative impact on spawning brown trout. This shows the importance of including further biotic parameters.

Analysis 1 and 8 aimed to show the sensitivity surrounding the input to habitat models. The difference between fuzzy rules and HSC was investigated as was the difference between 1D and 2D input methods. Statistically different results ($p<0.05$) were found between each combination for each species at each site. This shows how the input can give large differences in the results and that this needs to be taken into account when making management decisions based on these models.

Chapter 7- Research question 3 results

7.1 Chapter introduction

This chapter presents the results from research question 3 (RQ3) investigating how water trading impacts upon the indicator species. The chapter is organised by results from the two sites used, site 1 and 2 both used 2D habitat analysis. Site 3 was not used for analysis due to such small and insignificant results occurring for the site. Finally a summary of findings is given at the end of the chapter.

Three distinct analyses were carried out for each site:

1) Data distributions are presented to show how the baseline distribution of each species is affected under the two trading scenarios.
2) The extreme year analysis, as carried out in RQ2, is completed but with a focus on how the trading scenarios affect the habitat availability in each of the 'extreme' years.
3) Finally, synthetic flow analysis investigates whether the change in flows from the trading scenarios affects the habitat within the natural variation.

The three trading scenarios used were:

- Scenario 1 (S1)- No trading with Hands off flow (HOF) (Baseline)
- Scenario 2 (S2)- Trading with HOF
- Scenario 3 (S3)- Trading without HOF

As very few trades occurred (5 trades in S2 and 7 in S3), minimum changes were expected. Section 3.4.1 and 3.4.2 provide details on the water trading model.

7.2 Site 1- Highbridge

The following section presents the results and discussion for site 1 (Highbridge); all results are from the 2D analysis.

7.2.1 Data distribution

Figure 7.1 demonstrates the box plot distributions of each species for each trading scenario for site 1. Table 7.1 presents this data in numerical format in order to assess how the statistical properties are affected in the trading scenarios.

As determined in RQ2, site 1 has little habitat availability for adult and juvenile brown trout and for Mayfly. Due to the small change in flow between the trading scenarios at this site, the habitat availability is not changed in any significant detail. The average HHS value only falls by 0.01 in S3 for Crowfoot and Mayfly in comparison to

S1 (baseline). For all other species it remains the same as do the maximum and minimum values for all species.

Slight changes can be seen in the Skew and Kurt values for all species between S1 and S3 this upper indicates that whilst the averages and minimum and maximum values do not change, the distributions are slightly altered.

The Skew value measures symmetry within a distribution of data, a 0 value indicates perfect symmetry, a negative value indicates the data is Skewed to the left and a positive value indicates the data is Skewed to the right (i.e. the bulk of the data is right of the peak). Adult and juvenile brown trout have positive Skew values indicating their distribution is Skewed to the right, i.e. the majority of data is within the upper portion of results i.e. higher HHS values. For juvenile brown trout this number decreases from 0.4 in S1 (baseline) to 0.33 in S3 (trading without HOF), this finding shows that for juvenile brown trout trading without HOF would decrease the portion of HHS values in the upper HHS values, thus, decreasing overall habitat availability.

Spawning brown trout, Crowfoot and Mayfly have negative Skew values indicating that the majority of their distributions are in the lower HHS values. For Crowfoot, S1 has a Skew of -1.3, S2 of -1.31 and S3 of -1.48, this shows that trading has negative effects on habitat availability for Crowfoot as S3 (trading without HOF) has a higher proportion of time in the lower HHS values. For Mayfly and spawning brown trout however the opposite is true and the trading scenarios actually cause a higher proportion of results to be in the lower HHS values i.e. for spawning brown trout S1 and S2 are -1.43 and S3 is -1.39.

The Kurt values describe a measure of peakedness or flatness. A positive value indicates a relatively peaked distribution and a negative value means a relatively flat distribution. Spawning brown trout and Crowfoot had positive distributions whilst adult and juvenile brown trout and Mayfly had negative distributions. The biggest change in Kurt values between trading scenarios was for Crowfoot where S1 was 1.14, S2 was 1.21 and S3 was 2.09, this showed that the trading scenarios increased the peakedness of results indicating a larger cluster of HHS results in 1 area, thereby reducing fluctuation in the HHS availability. This is not good for the natural diversity required by the species.

The Mann-Whitney tests revealed statistically different scores in summer for all species (Table 7.2), however statistically similar results were found in all other season and between all seasons combined. This shows that despite differences being shown, these small differences are not statistically different and therefore are not robust.

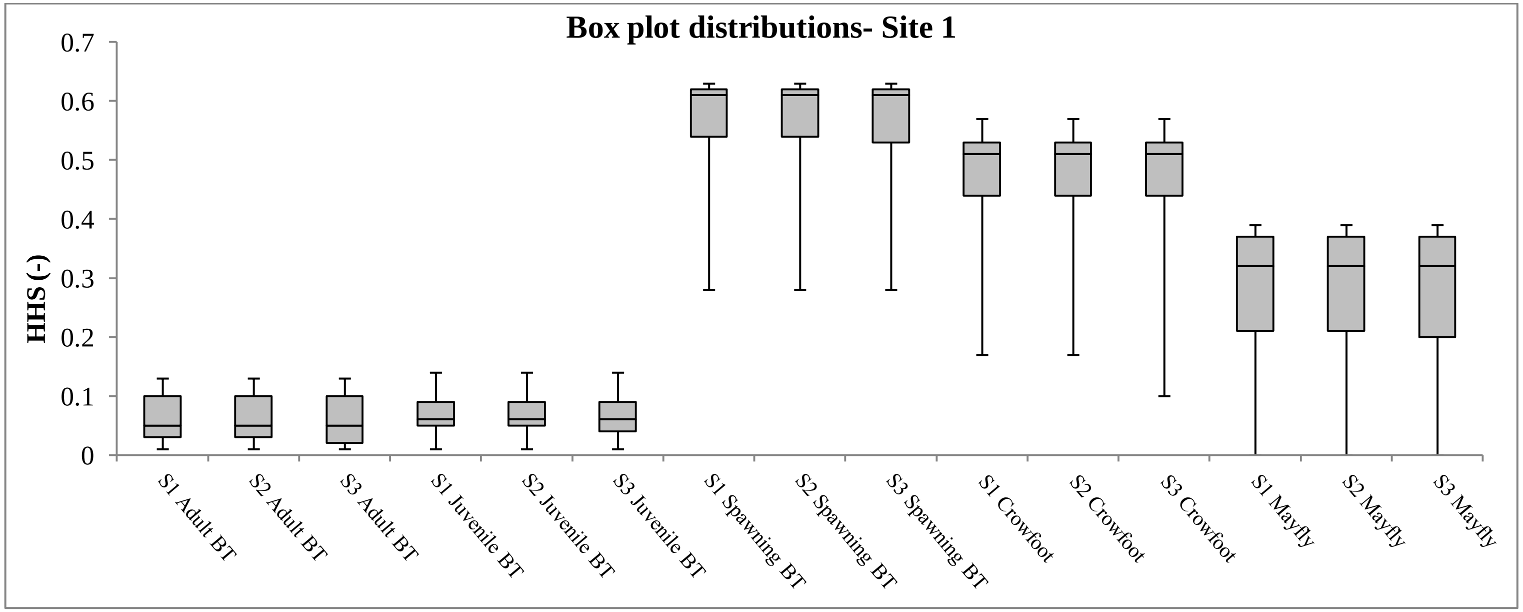


Figure 7.1- Box plot distributions of HHS values at site 1

Table 7.1- Statistical factors of HHS values for site 1

	Adult			**Juvenile**			**Spawning**			**Macrophyte**			**Benthos**		
	S1	**S2**	**S3**	**S1**	**S2**	**S3**	**S1**	**S2**	**S3**	**S1**	**S2**	**S3**	**S1**	**S2**	**S3**
Average	0.06	0.06	0.06	0.06	0.06	0.06	0.57	0.57	0.57	0.48	0.48	0.47	0.28	0.28	0.27
Median	0.05	0.05	0.05	0.06	0.06	0.06	0.61	0.61	0.61	0.51	0.51	0.51	0.32	0.32	0.32
Maximum	0.13	0.13	0.13	0.14	0.14	0.14	0.63	0.63	0.63	0.57	0.57	0.57	0.39	0.39	0.39
Minimum	0.01	0.01	0.01	0.01	0.01	0.01	0.28	0.28	0.28	0.17	0.17	0.10	0.00	0.00	0.00
95 %ile	0.02	0.02	0.02	0.03	0.03	0.03	0.37	0.37	0.37	0.32	0.31	0.31	0.02	0.02	0.02
50 %ile	0.05	0.05	0.05	0.06	0.06	0.06	0.61	0.61	0.61	0.51	0.51	0.51	0.32	0.32	0.32
5 %ile	0.13	0.13	0.13	0.10	0.10	0.10	0.63	0.63	0.63	0.56	0.56	0.56	0.39	0.39	0.39
SD	0.04	0.04	0.04	0.02	0.02	0.02	0.08	0.08	0.08	0.08	0.08	0.08	0.12	0.12	0.12
Skew	0.46	0.47	0.46	0.40	0.40	0.33	-1.43	-1.43	-1.39	-1.30	-1.31	-1.48	-1.10	-1.08	-1.05
Kurt	-1.26	-1.26	-1.27	-0.60	-0.61	-0.55	0.83	0.82	0.70	1.14	1.21	2.09	-0.07	-0.13	-0.23

Table 7.2- Mann-Whitney results for HHS values- Site 3. Grey indicates statistically different results

		S1 to S2 p-value	S1 to S3 p-value
Adult brown trout	**All**	0.585	0.215
	Winter	1.000	1.000
	Spring	0.720	0.662
	Summer	0.273	0.001
	Autumn	1.000	1.000
Juvenile brown trout	**All**	0.593	0.187
	Winter	1.000	1.000
	Spring	0.750	0.689
	Summer	0.265	0.000
	Autumn	1.000	1.000
Spawning brown trout	**All**	0.708	0.115
	Winter	1.000	1.000
	Spring	0.802	0.535
	Summer	0.552	0.004
	Autumn	1.000	1.000

		S1 to S2 p-value	S1 to S3 p-value
Mayfly	**All**	0.400	0.091
	Winter	1.000	1.000
	Spring	0.453	0.335
	Summer	0.344	0.002
	Autumn	1.000	1.000
Crowfoot	**All**	0.534	0.024
	Winter	1.000	1.000
	Spring	0.598	0.395
	Summer	0.492	0.001
	Autumn	1.000	1.000

7.2.2 Extreme years

The graphs shown in figure 7.2 show the percentage of available habitat in each year, further split into changes in the trading scenario, this demonstrates how available habitat changes in wet, dry and average years through the trading scenarios.

The results (Figure 7.2) generally show wet years provide better quality habitat for; adult and juvenile brown trout, Mayfly and Crowfoot, as wet years provide more highly suitable/ suitable habitat and less highly unsuitable/ unsuitable habitat. E.g. for adult brown trout in S1:

- the amount of 'suitable' habitat is 0.89% in the dry year and 1.18% in the wet year,
- the amount of 'highly unsuitable' habitat is: 95.41% in the dry year and 77.76% in the wet year.

Spawning brown trout however have a preference for the dry or average conditions. The amount of 'highly unsuitable' habitat is: 0.03% in the dry year and 27.47% in the wet year. The amount of 'suitable' habitat is: 63.62% in the dry year, 64.19% in the wet year and 79.73% in the average year (all results from S1). No clear trend occurred for Crowfoot, i.e. more 'highly suitable' and more 'highly unsuitable' habitat occurred in the wet year in comparison to the dry year. This indicates that the species have a wide preference for flows and generally similar results occurred in each of the years.

The trading scenarios did have an impact on the results for all species, for adult and juvenile brown trout and Mayfly a clear trend of S2 providing slightly worse habitat i.e. more 'highly unsuitable/ unsuitable' habitat, followed by S3 providing a further decrease in habitat availability occurred e.g. for juvenile brown trout the amount of 'highly unsuitable habitat in the dry year was: 91.33% in S1, 91.45% in S2 and 91.64% in S3, and in the wet year: 75.7% in S1, 75.3% in S2 and 75.75% in S3. These results therefore suggest that the trading scenarios negatively impact on habitat availability. Trading with HOF (S2) however does not affect the habitat availability as much as trading without HOF (S3) does, thus promoting the importance of HOF limits. For Crowfoot the S2 scenario did not affect the habitat availability to any extent and percentages remained the same, the S3 scenario however decreased habitat availability e.g. in the dry year the amount of 'highly suitable' habitat was 31.3% in S1 and 30.92% in S3, the amount of 'highly unsuitable' habitat was 25.29% in S1 and 28.59% in S3.

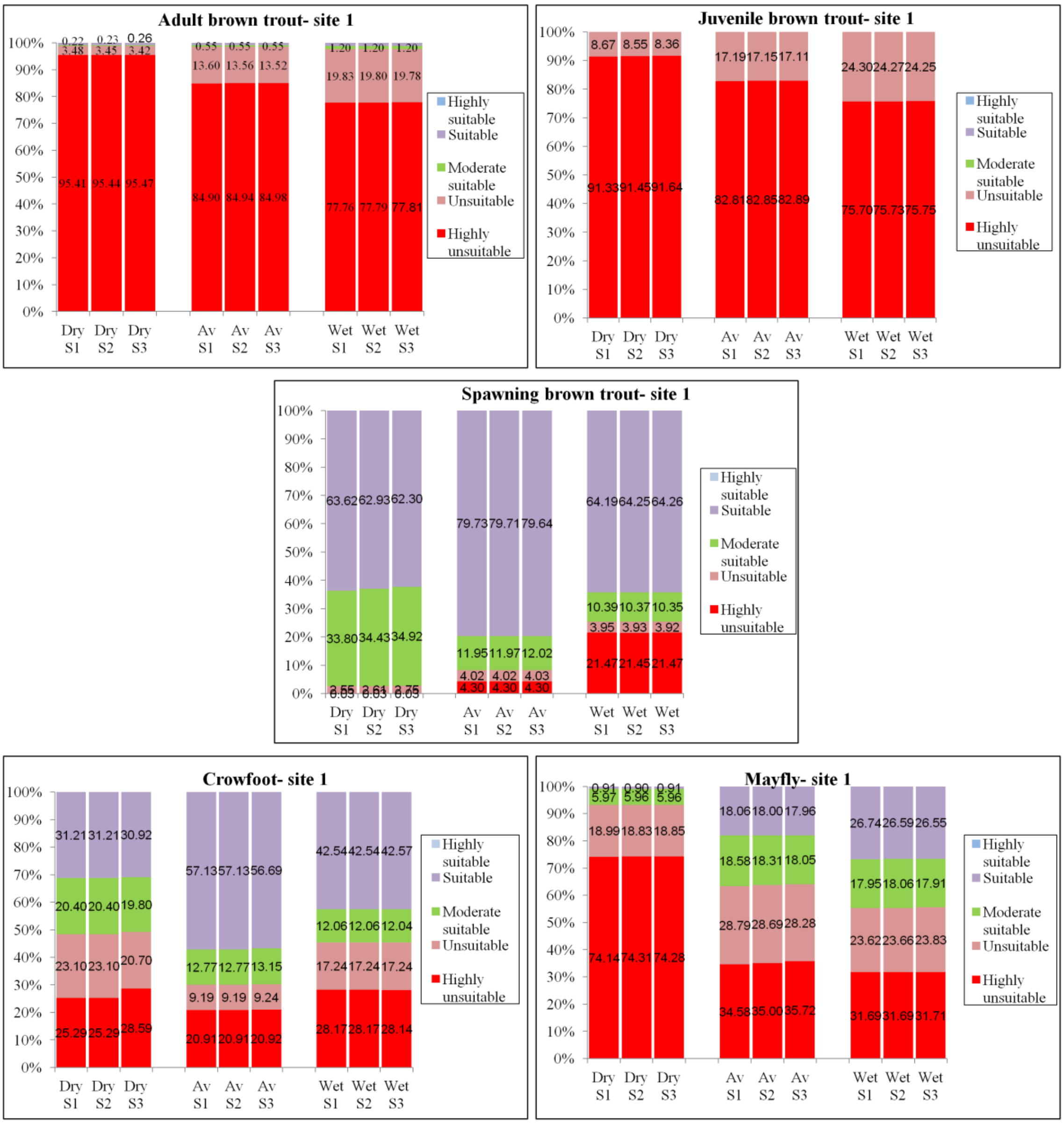


Figure 7.2- Extreme year analysis for site 1

For spawning brown trout however the trading scenarios can be seen to improve the habitat availability, in the wet year the amount of 'highly suitable' habitat increases from 64.19% to 64.25% to 64.26% in S1, S2 and S3 respectively. The opposite occurs in dry years however where the amount of 'highly suitable' habitat decreased from 63.62% to 62.93% to 62.3% in S1, S2 and S3 respectively. In summary:

- Dry years; decreases in flow through trading causes suitable habitat to **decrease** i.e. flows are too low for spawning brown trout.
- Wet years; decreases in flow through trading causes suitable habitat to **increase** i.e. brown trout prefer lower flows during wetter conditions.

This finding highlights species- specific preferences. Reducing flows in wet year's habitat improves habitat for spawning brown trout, however for adult and juvenile brown trout, habitat availability is reduced. In dry years however, reducing flows reduces habitat availability for spawning brown trout as it does for adult and juvenile brown trout, indicating that a threshold of accessibility is being breached.

Very little statistically significant differences between the trading scenarios occurred in site 1. Table 7.3 demonstrates that for the 3 years presented (1986, 1991 and 2001) no statistically significant differences occurred. Furthermore throughout the 32 year period (see Appendix N) there were only 3 occurrences of statistically different results, there were all between S1 and S3 for 'moderate' suitability in dry years (for adult brown trout and Crowfoot). Thus whilst differences are observed, the results are not supported by positive Mann-Whitney results. This therefore means that the trading scenarios do create a change in habitat availability, but this is not a significant change and the species would therefore cope with the change.

Table 7.3- Mann- Whitney results for SI values- Site 1. Grey indicates statistically different results. N/A when all values are 1

		Highly unsuitable		**Unsuitable**		**Moderate**		**Suitable**		**Highly suitable**	
		S1 to S2	**S1 to S3**	**S1 to S2**	**S1 to S3**	**S1 to S2**	**S1 to S3**	**S1 to S2**	**S1 to S3**	**S1 to S2**	**S1 to S3**
		p-values									
Adult brown trout	**1986**	0.756	0.531	0.808	0.583	0.808	0.577	0.680	0.238	0.909	0.908
	1991	0.902	0.708	0.698	0.523	0.312	0.152	0.697	0.523	NA	NA
	2001	0.960	0.793	0.922	0.903	0.914	0.895	0.923	0.910	0.859	0.859
Juvenile brown trout	**1986**	0.805	0.583	0.808	0.583	N/A	N/A	N/A	N/A	N/A	N/A
	1991	0.439	0.120	0.698	0.523	N/A	N/A	N/A	N/A	N/A	N/A
	2001	0.962	0.614	0.923	0.904	N/A	N/A	N/A	N/A	N/A	N/A
Spawning brown trout	**1986**	0.739	0.489	0.934	0.826	0.948	0.694	0.661	0.310	N/A	N/A
	1991	0.479	0.338	0.507	0.283	0.696	0.521	0.698	0.523	N/A	N/A
	2001	0.960	0.981	0.860	0.773	0.864	0.748	0.976	0.925	N/A	N/A
Crowfoot	**1986**	0.861	0.935	0.734	0.378	0.737	0.365	0.666	0.344	N/A	N/A
	1991	0.516	0.367	0.870	0.139	0.581	0.282	0.813	0.813	N/A	N/A
	2001	0.921	0.902	0.954	0.871	0.941	0.990	0.988	0.982	N/A	N/A
Mayfly	**1986**	0.639	0.342	0.854	0.343	0.548	0.327	0.808	0.583	N/A	N/A
	1991	0.832	0.625	0.813	0.813	0.905	0.905	0.889	0.889	N/A	N/A
	2001	0.997	0.978	0.894	0.824	0.903	0.911	0.912	0.892	N/A	N/A

7.2.3 Synthetic flows

Throughout RQ2 the limitations of the habitat models became apparent i.e. not taking into account other species (Jowett 1992; Garbe et al., 2016), differences in input methods i.e. HSC/ fuzzy rules (Boavida et al., 2014) and 1D or 2D hydraulic inputs (Gard 2009). More recently Beven and Alcock (2012) have described the uncertainty surrounding any kind of modelling for environmental purposes. For these reasons it was important to show the distributions according to the synthetic flows in order to take into account uncertainty surrounding both models and flows. Overall the results demonstrate that the predicted HHS values are always within this distribution which helps quantify the results in respect of uncertainty. The purpose of this analysis therefore was to understand the changes which result from trading in the context of synthetic distributions. Figure 7.3 presents the habitat distributions of the 2 trading scenarios in the black and red lines. The grey shaded section shows the maximum and minimum distribution of each HHS predicted by the trading scenarios.

The results (Figure 7.3) demonstrate that the two trading scenarios (S2 and S3) are always within the distribution of HHS, as predicted by the synthetic flows, for each species. Occasionally the trading scenarios are on the upper or lower limit of the variability, this indicates that there would be the potential for trading to affect habitat availability, however in this situation this is not the case. Therefore whilst trading does have an effect on flow and subsequently habitat availability, the changes are not to a significant extent.

This is an important finding as whilst the habitat models do not aim to predict the future, they give an indication of the natural variation in flow and subsequent habitat availability. Thus with the uncertainty of future impacts of climate change (Ledger and Milner 2015) it is important to assess the range in natural conditions. The grey areas on the graphs demonstrate that the habitat availability throughout the past 32 years in not necessarily a prediction of the next 32 years and the distributions are likely to fluctuate.

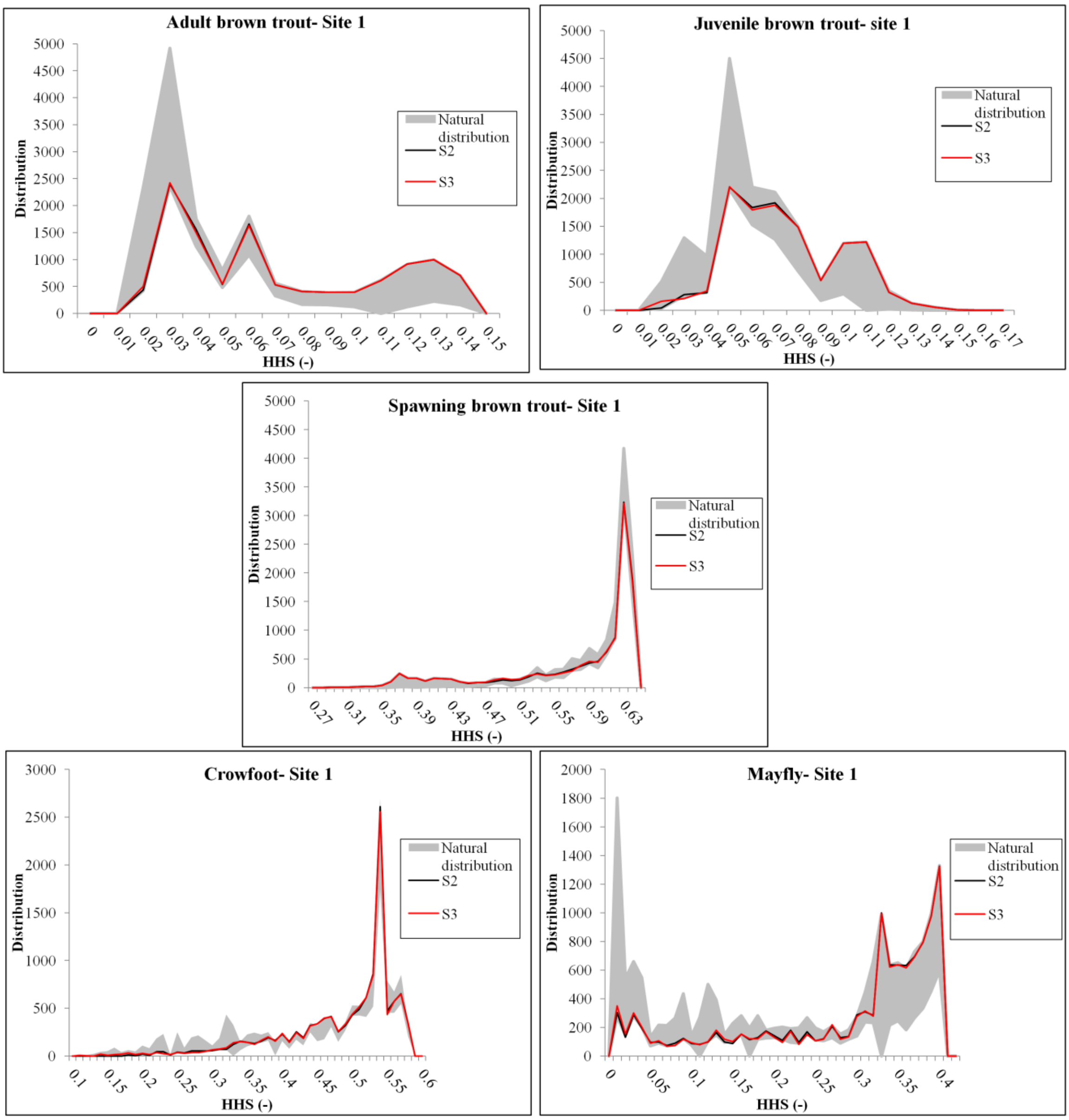


Figure 7.3- Synthetic flow analysis for site 1

7.3 Site 2- DS Nar

The following section presents the results and discussion for site 2 (DS Nar); all results are from the 2D analysis.

7.3.1 Data distribution

Figure 7.4 demonstrates the box plot distributions for each species for each of the trading scenarios. The distribution between S1 and S2 is predominantly the same. Visually the box plots and statistics show little change, however the statistical factors (table 7.5) demonstrate the Skew and Kurt values having a small change, thus indicating a different distribution (see section 7.2.1 for definition of properties). All average and maximum HHS values remain the same for all 3 trading scenarios for each species, indicating the trading scenarios do not impact the general habitat availability; the minimum value is reduced for all species apart from spawning brown trout in S3. This shows that whilst the average remains the same, some lower HHS values are created when trading without HOF occurs. The minimum value does however remain the same for S2 showing the HOF protects the habitat availability.

The Skew value is negative for every species, this indicates that the distribution is negatively Skewed and the majority of data is in the lower HHS values. The Skew slightly decreases between S1 and S2 for all species (e.g. juvenile brown trout S1= -1.22, S2= -1.19, Mayfly S1=-1.34, S2=-1.3) showing that trading actually decreases the amount of data in the lower HHS sector. Interestingly the opposite occurs for S3 (trading without HOF) (e.g. juvenile brown trout S1= -1.22, S2= -1.4, Mayfly S1=-1.34, S2=-1.52). This is a key finding showing that trading with HOF slightly improves habitat availability, whilst trading without HOF decreases habitat availability in terms of the distribution of time within HHS values.

The Kurt values are positive for each species showing that they all have a relatively peaked distribution and are thus clustered around one area of HHS rather than being more evenly spread. The Kurt value decreases from S1 to S2 for each species (e.g. juvenile brown trout S1= 0.9, S2= 0.77, Mayfly S1= 1.42, S2=1.29) indicating that trading with HOF creates a slightly more even distribution of HHS results. Furthermore the Mann-Whitney tests revealed statistically similar results between S1 and S2 in all seasons and throughout the whole 32 year period (Table 7.4).

Likewise with the Skew values, the Kurt values increase in S3, showing that the S3 (trading without HOF) creates a more peaked distribution and data is clustered around one area. This is not favourable for any of the species and natural diversity is a positive aspect providing variety in physical habitat throughout time.

Overall the distribution between S1 and S3 is different for all species. For all species apart from spawning brown trout the minimum HHS value reduces, the box plots show this change in distribution more clearly. Despite this however the mean and median remain the same. According to the Mann-Whitney results however, statistically different habitat availabilities (Table 7.5) were found for each species between S1 and S3 results for the whole 32 year period and for the summer season. This therefore shows that trading without HOF creates significantly different habitat results and according to the data distribution statistics, this difference is a negative difference creating worse habitat availability.

In summary the S3 trading scenario negatively affects habitat availability, however S2 does not statistically affect habitat availability as little change occurs as a result of trading when the HOF is activated (S2). Once the HOF is removed (S3), statistically lower HHS values occur.

Table 7.4- Mann- Whitney results for HHS values- Site 2. Grey indicates statistically different results

		S1 to S2 p-value	**S1 to S3 p-value**
Adult brown trout	**All**	0.516	0.026
	Winter	1.000	1.000
	Spring	0.509	0.338
	Summer	0.543	0.000
	Autumn	1.000	1.000
Juvenile brown trout	**All**	0.483	0.017
	Winter	1.000	1.000
	Spring	0.413	0.247
	Summer	0.552	0.000
	Autumn	1.000	1.000
Spawning brown trout	**All**	0.472	0.015
	Winter	1.000	1.000
	Spring	0.376	0.238
	Summer	0.575	0.000
	Autumn	1.000	1.000

		S1 to S2 p-value	**S1 to S3 p-value**
Mayfly	**All**	0.493	0.021
	Winter	0.493	0.328
	Spring	0.493	0.328
	Summer	0.567	0.000
	Autumn	1.000	1.000
Crowfoot	**All**	0.506	0.040
	Winter	1.000	1.000
	Spring	0.460	0.298
	Summer	0.585	0.000
	Autumn	1.000	1.000

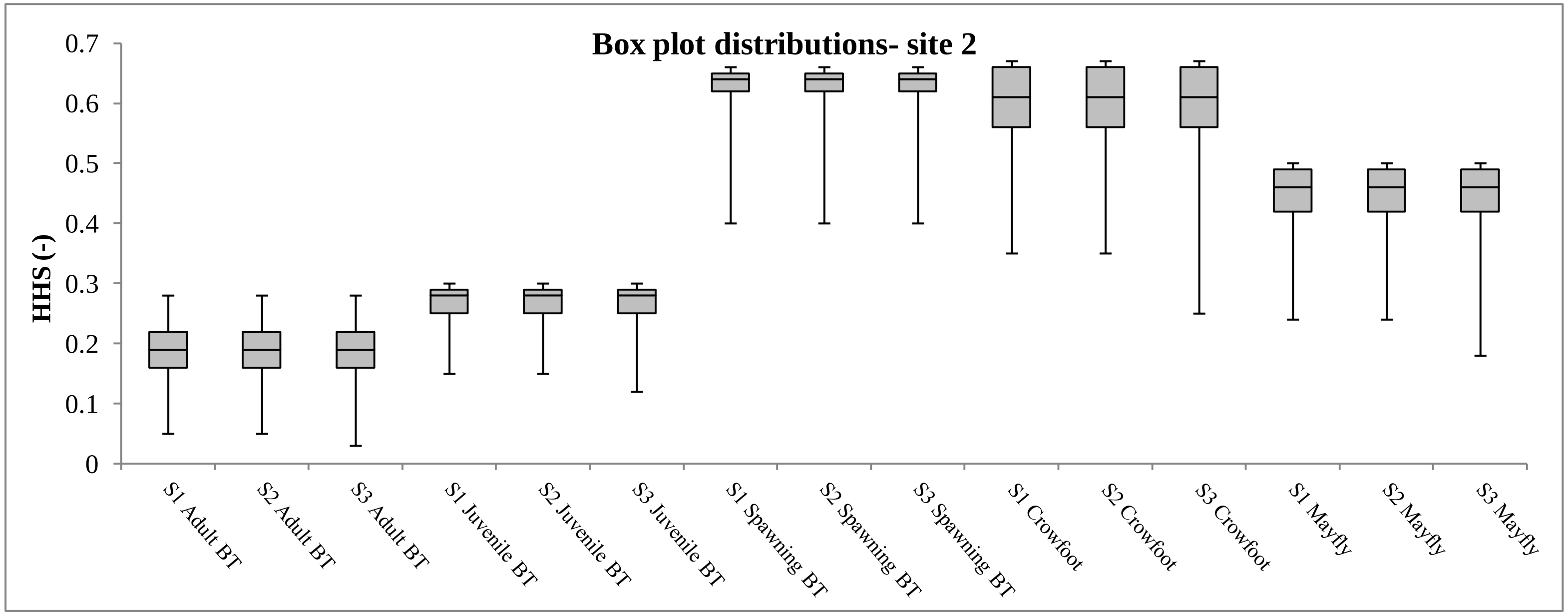


Figure 7.4- Box plot distributions of HHS values for Site 2

Table 7.5- Statistical factors of HHS values for site 2

	Adult			**Juvenile**			**Spawning**			**Crowfoot**			**Mayfly**		
	S1	**S2**	**S3**	**S1**	**S2**	**S3**	**S1**	**S2**	**S3**	**S1**	**S2**	**S3**	**S1**	**S2**	**S3**
Average	0.18	0.18	0.18	0.27	0.27	0.27	0.63	0.63	0.63	0.60	0.60	0.60	0.45	0.45	0.45
Median	0.19	0.19	0.19	0.28	0.28	0.28	0.64	0.64	0.64	0.61	0.61	0.61	0.46	0.46	0.46
Maximum	0.28	0.28	0.28	0.30	0.30	0.30	0.66	0.66	0.66	0.67	0.67	0.67	0.50	0.50	0.50
Minimum	0.05	0.05	0.03	0.15	0.15	0.12	0.40	0.40	0.40	0.35	0.35	0.25	0.24	0.24	0.18
95 %ile	0.10	0.10	0.10	0.20	0.20	0.20	0.57	0.57	0.57	0.47	0.47	0.46	0.33	0.33	0.32
50 %ile	0.19	0.19	0.19	0.28	0.28	0.28	0.64	0.64	0.64	0.61	0.61	0.61	0.46	0.46	0.46
5 %ile	0.22	0.22	0.22	0.30	0.30	0.30	0.66	0.66	0.66	0.67	0.67	0.67	0.50	0.50	0.50
SD	0.04	0.04	0.04	0.03	0.03	0.04	0.03	0.03	0.03	0.07	0.07	0.07	0.05	0.05	0.06
Skew	-1.04	-1.01	-1.16	-1.22	-1.19	-1.40	-1.38	-1.35	-1.49	-0.88	-0.85	-1.22	-1.34	-1.30	-1.52
Kurt	0.50	0.40	0.93	0.90	0.77	1.70	2.38	2.25	2.55	0.35	0.24	1.75	1.42	1.29	2.28

7.3.2 Extreme years

The graphs shown in figure 7.5 show the percentage of available habitat in each year, further split into changes in the trading scenario. This demonstrates how available habitat changes in wet, dry and average years through the trading scenarios.

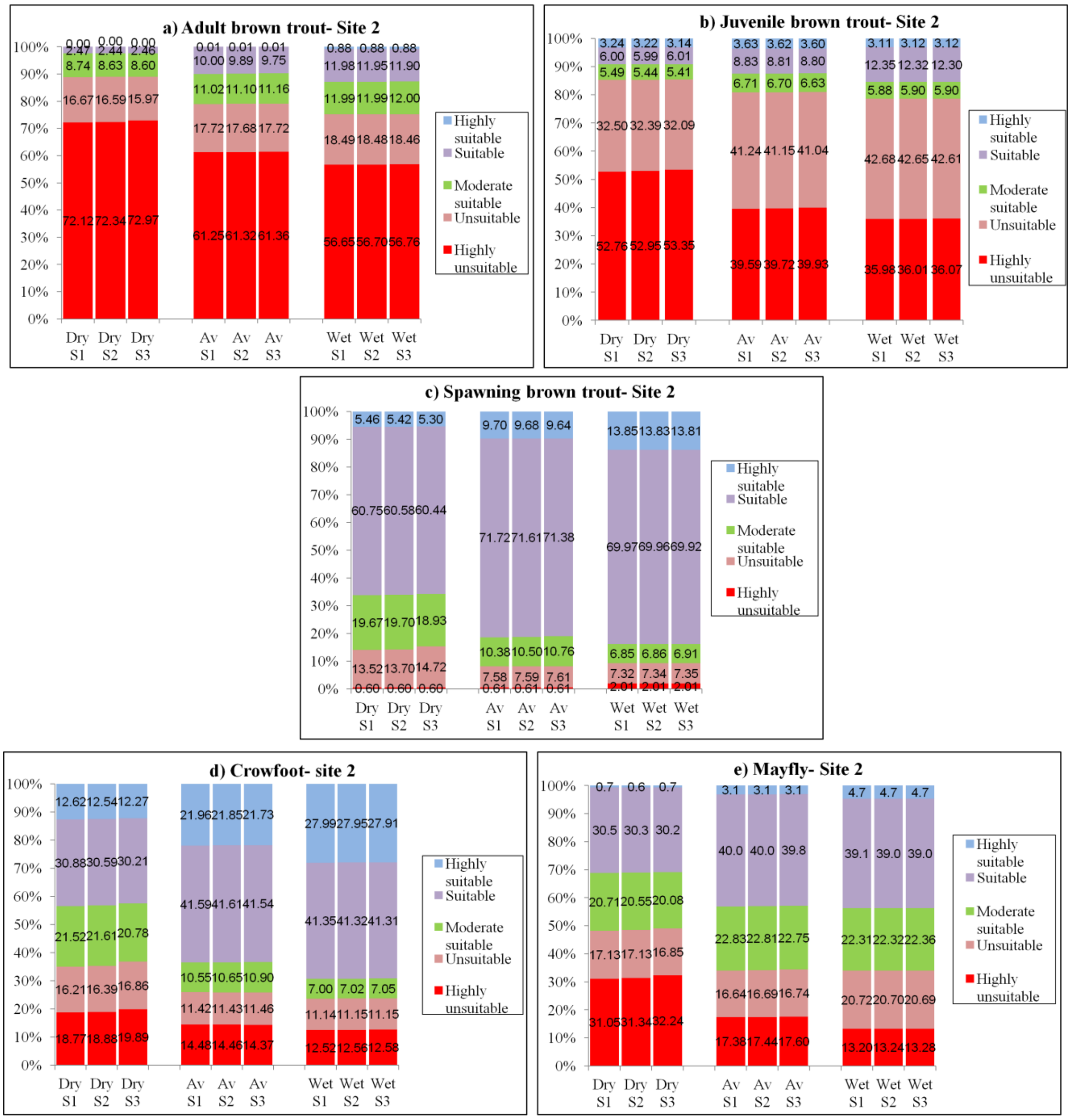


Figure 7.5- Extreme year analysis for site 2

The results (Figure 7.5) demonstrate similar trends to those in site 2: wet years provide better habitat for all species apart from spawning brown trout, followed by average years, with dry years providing the least amount of suitable habitat. For example for 4 species out of 5, the ‘highly suitable’ and ‘suitable’ habitat availability was highest in the wet year, and lowest in the dry year, likewise for 4 out of 5 species, the amount of highly unsuitable habitat was highest in the dry year and lowest in the wet year. Spawning brown trout provided the anomaly in the trend where the amount of ‘suitable’ habitat was highest in the average year, and the amount of ‘highly unsuitable’ habitat was highest in the wet year. These anomalies are related to spawning brown trout having a preference for lower flows and other biotic dependants i.e. food and refugia (Garbe et al., 2015).

For the trading scenarios the following general trends were observed for all species:

- ‘Highly suitable’ and ‘suitable’ habitat was reduced slightly in S2 and by slightly more in S3 e.g. spawning brown trout, highly suitable, wet year: S1=13.85%, S2=13.83%, S3=13.81%
- ‘Unsuitable’ and for most cases ‘highly unsuitable’ habitat increases in S2 and increases by more in S3 e.g. Crowfoot, unsuitable, average year: S1=11.42% S2= 11.43%, S3=11.46%.

Therefore S3 provides the least habitat availability and due to the constraints of the HOF S2 provides only slightly worst habitat than S1. Mann-Whitney tests revealed little statistically different results between the trading scenarios (Table 7.6 (see Appendix O for all 32 years)). No statistical differences were found between S1 and S2 due to the small changes, this could be due to the small amount of trades. And only a few statistical differences were found between S1 and S3, furthermore these were only found in average and dry years. This indicates that whilst changes are seen between S1 and S2, this is not a statistically significant difference and therefore the results cannot be relied upon. The results for S1 to S3 however can be relied on for dry and average years in all habitat availability classes except ‘suitable’ and ‘highly suitable’.

Table 7.6- Mann- Whitney results for SI values- Site 2. Grey= statistically different results. N/A= all values 1

		Highly unsuitable		**Unsuitable**		**Moderate**		**Suitable**		**Highly suitable**	
		S1 to S2	**S1 to S3**	**S1 to S2**	**S1 to S3**	**S1 to S2**	**S1 to S3**	**S1 to S2**	**S1 to S3**	**S1 to S2**	**S1 to S3**
		p-values									
Adult brown trout	**1986**	0.854	0.301	0.772	0.506	0.652	0.648	0.775	0.406	1.000	1.000
	1991	0.805	0.505	0.448	0.127	0.540	0.263	0.879	0.879	NA	NA
	2001	0.857	0.788	0.928	0.742	0.958	0.869	0.821	0.480	0.929	0.929
Juvenile brown trout	**1986**	0.868	0.973	0.788	0.430	0.880	0.084	0.787	0.420	0.693	0.094
	1991	0.662	0.501	0.732	0.395	0.732	0.395	0.732	0.395	0.732	0.395
	2001	0.831	0.677	0.983	0.761	0.759	0.810	0.902	0.850	0.756	0.807
Spawning brown trout	**1986**	0.624	0.138	0.979	0.654	0.787	0.420	0.683	0.355	0.787	0.420
	1991	0.732	0.395	0.667	0.348	0.892	0.016	0.732	0.395	0.732	0.395
	2001	0.942	0.855	0.792	0.723	0.834	0.658	0.875	0.841	0.903	0.850
Crowfoot	**1986**	0.852	0.371	0.754	0.344	0.777	0.412	0.791	0.428	0.787	0.420
	1991	0.604	0.084	0.736	0.472	0.925	0.038	0.732	0.395	0.732	0.395
	2001	0.707	0.656	0.831	0.936	0.733	0.646	0.888	0.801	0.903	0.850
Mayfly	**1986**	0.905	0.749	0.758	0.590	0.541	0.031	0.796	0.434	0.796	0.434
	1991	0.461	0.213	0.815	0.459	0.732	0.395	0.732	0.395	0.879	0.879
	2001	0.739	0.687	0.995	0.805	0.840	0.780	0.945	0.716	0.918	0.811

7.3.3 Synthetic flows

Likewise with site 1, it was important to assess the trading results in respect to uncertainty in habitat models (Jowett 1992; Beven and Alcock 2012; Garbe et al., 2016). Figure 7.6 presents the habitat distributions the 2 trading scenarios black and red lines. The grey shaded section shows the maximum and minimum distribution of each HHS predicted by the trading scenarios.

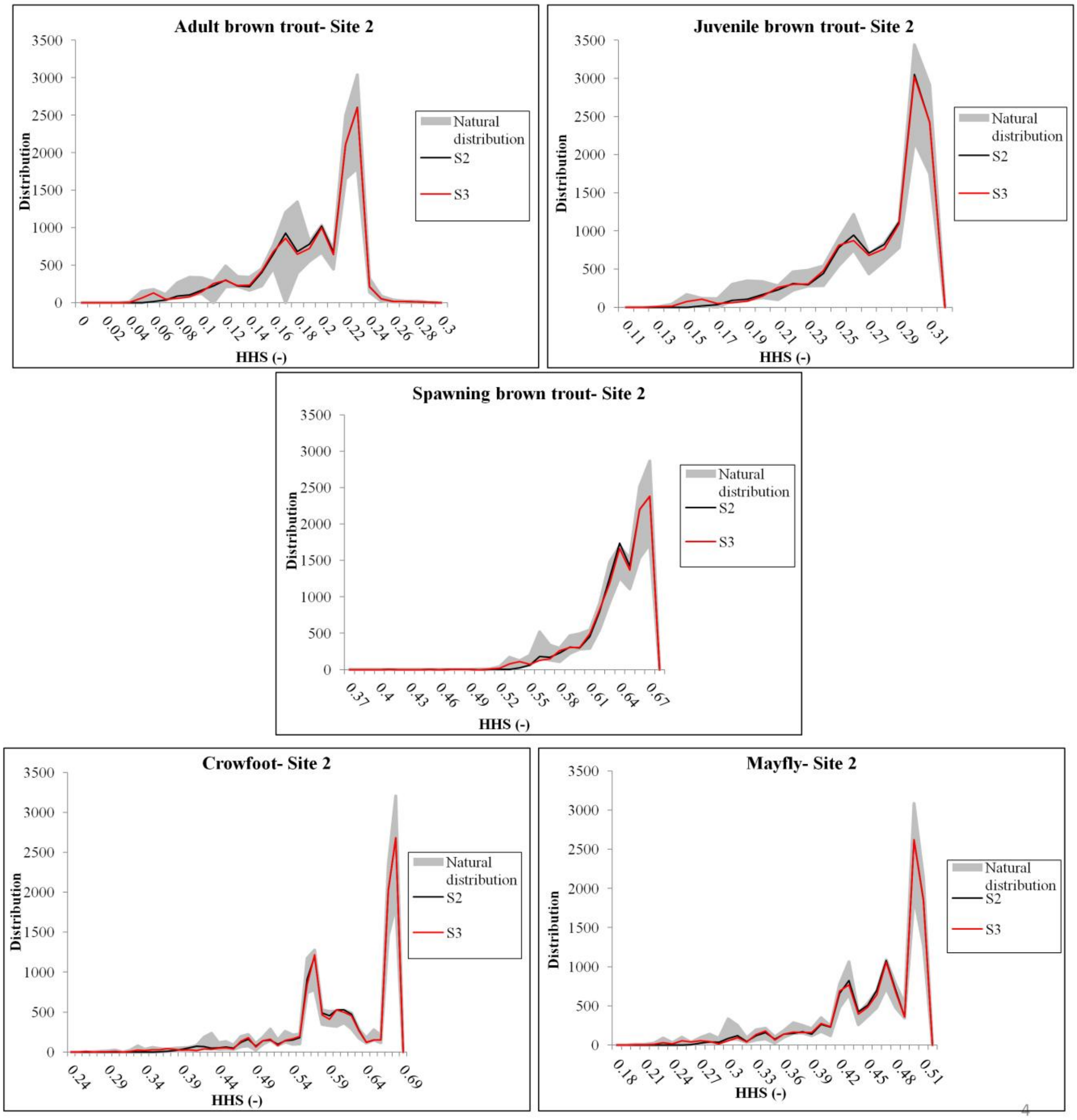


Figure 7.6- Synthetic flow analysis for site 2

The results (Figure 7.6) demonstrate that the two trading scenarios (S2 and S3) are always within the natural distribution of HHS for each species. Occasionally the trading scenarios are on the upper or lower limit of the natural variability, i.e. juvenile brown trout at around 0.27 HHS, this indicates that there would be the potential for trading to affect habitat availability, however in this situation this is not the case. Therefore whilst trading does have an effect on flow and subsequently habitat availability, the changes are not to a significant extent.

7.4 Chapter summary

This chapter has highlighted the impacts of trading on the indicator species. The key finding was that trading only has a small impact on the species; however this could be related to the low level of trading that was predicted in the catchment. The importance of the HOF has been highlighted as much smaller differences occurred when the HOF was activated. See section 8.6 in the discussion chapter for further details.

Chapter 8- Discussion

8.1 Chapter introduction

The overall aim of this thesis was to assess the effects of changes in low flows caused by water trading on biotic indicators of environmental quality on chalk stream and used three research questions to achieve this:

1) How are the ecosystem indicators affected during low flows?

Research question 1 (RQ1) aimed to investigate how the ecosystem indicators are related to and affected by low flows. Throughout a variety of methods and investigation the analysis and subsequent findings facilitated the determination of the factors influencing the species with a particular focus on flow. See Chapter 5 for the results.

2) How useful are numerical models in investigating how low flow periods impact upon the ecosystem indicators?

Research question 2 (RQ2) aimed to investigate two areas of research, firstly to assess how different flows, particularly low flows, affect habitat availability of the ecosystem indicators. Secondly to investigate the sensitivity of input methods to habitat models and therefore determine how useful the models are in investigating the impacts of flow on ecosystem indicators. See Chapter 6 for the results.

3) How does trading at a catchment scale impact upon the ecosystem indicators?

Research question 3 (RQ3) investigated how water trading scenarios impact upon the indicator species. See Chapter 7 for the results.

This chapter synthesises the findings from each of these research questions, bringing the work together to show the overall aim of assessing the impacts of low flow on the three indicator species and how water trading may affect the species. The chapter is organised firstly by a discussion of each indicator species, giving a background, the main findings and a discussion of what these findings mean in both a UK and international context. This section focuses on the results from RQ1 and RQ2. The subsequent section discusses the novel approach proposed which investigates how species can be interconnected in habitat models to show the full picture of biotic factors affecting one species. Finally the water trading results from RQ3 are discussed with a focus on what this means in both a UK and international perspective.

8.2 Fish: Brown trout (*Salmo Trutta*)

8.2.1 Background

Fish were chosen as an indicator species due to their importance socially, economically and ecologically in the UK and in Europe (Conallin et al., 2014). Chalk streams provide pristine habitat for brown trout (Berrie 1992) which have Biodiversity Action Plan (BAP) protection and populations in the River Nar are wild as oppose to being stocked. Brown trout provide many important ecosystem services such as food, transport of nutrients and recreational activities (Holmlund and Hammer 1999) and are a key component in many aquatic ecosystems. However brown trout, and many other fish species, are threatened by anthropogenic pressures, particularly over-abstraction of freshwater (Hendry et al., 2003). The research carried out in this study helped quantify the effect of this over-abstraction by assessing how low flows, caused by over-abstraction, are impacting on brown trout and investigating if numerical models can aid in ecologically- based decision making. The data analysis used historical electro-fishing data and flow data to show how populations of brown trout are affected during different flows. The model analysis showed how different flows impact habitat availability. Figure 8.1, 8.2 and 8.3 presents the main findings from each of the research questions for adult, juvenile and spawning brown trout respectively.

8.2.2 Main findings

Brown trout are highly abundant in the River Nar, this is related to the conditions in a chalk stream river creating pristine habitat for the species (Berrie 1992), however generally a decline in numbers has been observed since records began in 1989. This could be related to a variety of factors such as water quality, food sources and land use change but ultimately the in channel flow has the largest influence on the species as this is the driver of many of the other changes (Armstrong et al., 2003).

In order to understand the interactions of brown trout with flow it is firstly important to determine the differences in river typologies. The most downstream sites in the fen reach, which is highly canalised, had no recorded brown trout populations over the sampled period (Figure 8.1). This finding corresponds with literature, where studies have shown that in canalised reaches the lack of coarser substrate, high depth and low velocities made it unfavourable for trout (Millidine et al., 2012). Furthermore according the major report on over-abstraction by SNIFFER (2011), the absence of salmonid species provide an indication of over-abstraction. The lower Nar (below Narborough) is classified as ‘over licensed’ for surface water (Norfolk Rivers Trust 2013). The fact that no salmonoid species are found in this area could be correlated to it being 'over-abstracted' as well as 'over-licensed', however the flow conditions may also influence their habitat availability. Higher abundances of brown trout were found in the chalk reaches than in the fen reach (Figure 8.1), this is however not unexpected as chalk streams are renowned for their trout abundances (Mann et al., 1989). Only small amounts of silt are washed into chalk streams during normal conditions, consequently the substratum consists predominantly of clean and compact gravel, these beds provide ideal environments for spawning brown trout and thus other lifestages of brown trout (Berrie 1992). Fen reaches however have more build-up of fine sediment due to their flat gradient and slower flows; this does not create an ideal environment for brown trout. For these reasons lower habitat availability would be expected in site 1 (downstream, fen site) than in the two upper stream sites in the chalk reach.

A main finding from the model analysis (RQ2) however was that juvenile and spawning brown trout had a trend of the same availability in site 2 (mid- stream) and site 3 (upper stream), with a lower availability in site 1 (downstream). For example; spawning brown trout have 'moderate habitat availability' in site 1, with 'high habitat availability' in both site 2 and site 3 (Table 8.1, Figure 8.1, 8.2). Thus for juvenile and spawning brown trout the conditions and canalisation in site 1 do have an impact on habitat availability, which corresponds to the electro- fishing data in that no brown trout were found at this site. For adult brown trout, site 1 had 'very low habitat availability' which whilst it does correspond to the electro-fishing data cannot be solely put down to being in the canalised reach as site 3 (the most natural site) also had 'very low habitat availability'. This provides a weakness with habitat models as they appear to not be accurately representing the habitat as the predicted habitat does not match up to the abundances recorded. Limited studies have tested this and those that have result in conflicting findings, this has mainly been tested with spawning locations. Gallagher and Gard (1999) found predicted habitat availability (WUA) was significantly correlated with salmon spawning locations, and sites with higher numbers of redds had higher predicted WUA. However, in a study by Mouton et al., (2008), a fuzzy based model predicted spawning grayling (*Thymallus thymallus*) to be present for several instances where no spawning was observed. This study therefore corresponds to that of Mouton et al., (2008).

Table 8.1- Overall fish habitat availability at each site

	Downstream to upstream		
	Site 1 (2D)	**Site 2 (2D)**	**Site 3 (1D)**
Adult brown trout	Very low	Low	Very low
Juvenile brown	Very low	Low	Low
Spawning brown	Moderate	High	High

Flow, velocity and depth are all known to influence habitat availability of brown trout (Armstrong et al., 2003), it was important however to investigate whether this is a direct effect or whether there is a timed lag effect in order to show how flows affect habitat availability. One of the most significant findings from the data analysis for brown trout was that the species are little affected by daily flow conditions and are moreover affected by the antecedent flow conditions. Regression analysis showed how the antecedent winter and summer Q_{95} (low) flows had the most significant relationships with brown trout populations. Low flows in winter were shown to have a particularly significant impact on brown trout. This finding indicates the management measures should take into account the difference in response rates throughout the year. Moreover more significance was noted when more antecedent conditions were applied, i.e. using the 5 year antecedent flow conditions. Past studies have indicated that antecedent flows had a large influence on fish species abundance (Balcombe and Arthington 2009), however limited studies have been completed on assessing the effect of antecedent flow on fish abundances. Preliminary investigations have indicated that this may be worth further analysis.

When investigating the impact of low flow on fish, it can be concluded that low flows do have an impact on the species but it is not a direct effect and instead there is a time lagged impact. Whilst more data would be needed to prove this, the lowest abundances of brown trout were found in 1993, the driest year on record was 1991, which suggests a lagged impact. However there were no records for 1991, so 1991 may have had lower fish numbers i.e. more data and sampling would be required to prove this.

The extreme year analysis (analysis 4 in the model analysis (RQ2)) suggested that different conditions are preferred by different species and that low flows during dry years are preferable for spawning fish however these conditions provide less habitat availability for Crowfoot and Mayfly. This is important for the interconnectedness analysis (see section 8.5). The Mann-Whitney tests reveal that these model trends are robust for spring and summer results however the same trends are less evident for winter and autumn. This suggests that environmental policies should take into account the different flow requirements and moreover consider the food web as a whole rather than focus on one species or one environmental consideration (i.e. EFI, see Section 8.2.3).

Analysis 3 of the model analysis (RQ2) investigated if low flows result in low HHS availability, Table 8.2 presents the result of this analysis for fish along with their ideal flows below or above which habitat becomes 'low':

Spawning brown trout

For spawning brown trout low flows did not cause low HHS at any site, and there is no lower flow limit at which HHS becomes 'low'. This is related to spawning brown trout generally having preferences for lower flows in order to spawn. Instead spawning brown trout were more affected by the upper flow limits having an upper flow limit of 3.06, 4.25 and 1.95m^3/s for site 1, site 2 and site 3 respectively. This however does not mean that there is no lower flow limit for spawning brown trout, as if there were no flow then the species could not spawn. The trading analysis (RQ3) highlighted this further where in dry years the suitable habitat decreases when trading was taken into account as flows were too low. However in wet years, suitable habitat increased through trading as spawning brown trout preferred the lower flows. This was a key finding in that the assessment is sensitive to the flow conditions i.e. wet or dry (see section 8.6 for further details).

Following on from this, it was discovered that the key times for spawning (Autumn- October to December) had very few occurrences of 'low HHS'. Only 7 years (out of 32 years) had a period of low HHS during this key time, these 7 years did not correspond to dry years and moreover corresponded to wet years e.g. in 1993, which recorded the highest Autumn Q_{10} (2.77m^3/s) of the analysed period, 11% of the time in October- December was at 'low HHS'. All of the above findings show that high flows create the issue for spawning brown trout as opposed to low flows. High flows are however important to sustain spawning grounds by reworking the gravel to remove any build up of silt (Hendry et al., 2003). So whilst the model predicts low availability, during higher flows in 7 years out of 32, these higher flows are necessary. Habitat models do not take into account sediment movements throughout the year. Fine sediment deposition and build up during low flow years and drought periods are not modelled. These would cause lower habitat availability for spawning brown trout as it would smother vital plant life used as refugia and cover vital spawning grounds (Hendry et al., 2003; S&TA 2014). Therefore it is important to include this in analysis of habitat availability throughout time. Habitat models show only a snapshot of time, however using the model results alongside the results from RQ1, it is clear that low flows do not create the most significant threat to spawning brown trout.

Adult and juvenile brown trout

For adult brown trout low flows do cause ‘low HHS’ at 2 sites, and for juveniles at 1 site. Furthermore as opposed to spawning brown trout, adult brown trout have lower limits rather than upper limits, showing that generally adult brown trout prefer higher flows. These results show that low flows do not necessarily mean low HHS, however as determined in RQ1, there is more of a relationship between brown trout populations and antecedent flows than there is between daily flows.

Table 8.2- Summary of low flow analysis

Species	Site	Ideal flow(s) (m^3/s)	Do low flows cause low HHS?
Adult brown trout	1	>1.24	Yes
	2	>0.39	No
	3	>0.75	Yes
Juvenile brown trout	1	>0.99	Yes
	2	0.27-5.75	No
	3	<1.82	No
Spawning brown trout	1	<3.06	No
	2	<4.25	No
	3	<1.95	No

8.2.3 UK context

Brown trout populations in the River Nar fit the well evidenced hypothesis that abundances are high in chalk streams due to the pristine conditions available (Berrie 1992) and furthermore that canalised reaches do not provide good conditions for brown trout (Millidine et al., 2012). This analysis has however discovered that low flows do impact on them, not through short periods, but through the longer term antecedent flow conditions. Low flows in winter and summer were shown to have a particularly significant impact on brown trout. This shows that flow policies should ideally take the seasonality into account, providing more protection during periods of low flow.

The low flow policies currently used in England are based on the Environmental Flow Indicator (EFI), which is a percentage deviation from the natural river flow which supports GES set for the WFD. This percentage deviation varies for different flows. It is dependent on the ecological sensitivity of the river to changes in flow based on abstraction sensitivity bands (EA 2013a). The HOF is then based on this. This is general for all river catchments; the only difference is the abstraction sensitivity band of the river. The research carried out here has shown that for brown trout in particular there are differences between surveyed sites in one river alone, and therefore EFI's cannot necessarily be transferred across rivers. EFI are beneficial as they are environmentally considerate and flexible taking into account different sensitivity bands. Ideally however, different sections of the river would have different low flow policies and this would be based on entire ecosystem needs rather than on one specific species. Management occurs on different spatial scales: European, national, regional and catchment level. This study has highlighted the importance of management on an even smaller scale than this. The meso-habitat communities within different reach types respond to differences in habitat heterogeneity such as substrate composition and differences in water chemistry which reflects underlying geology (Milner et al., 2015). Therefore having one overall management technique for a river or catchment which encompasses more than one reach type (i.e. plane- bed, step- pool, chalk, fen), is not necessarily the most appropriate management technique. Here it has been shown that differences occur on such scales and that management decisions should reflect this.

8.2.4 International context

An important finding from this research which can be applied at an international scale was that generally low flows do not directly cause 'low HHS' values for spawning brown trout, and instead it is the upper limit which must be addressed. HOF limits can be set to adequately protect the low flows however ensuring the upper limit is not breached would be a difficult task due to the natural flow cycles. Poor habitat is predicted during high flows for spawning brown trout, but occasionally high flows are necessary to flush out silt build-up (Hendry et al., 2003).

Previous studies have highlighted the effect of antecedent flow conditions on fish abundances. In a study by Kitsios et al., (2012) on streams in Western Australia, it was discovered that there was no relationship between antecedent low flows and fish abundance, however this was attributed to dataset limitations, stream adaptation to low flow stress, sensitive species been filtered out due to recent extreme conditions and other factors than only hydrology affecting these species. In another study in Australia, Balcombe and Arthington (2009) found antecedent flows had a large influence on fish species abundance. Following high summer flows, water bodies supported rich and abundant fish species. However fewer species and lower numbers were recorded after periods of zero channel flow which were related to a less diverse food web and limited food resources (Balcombe and Arthington 2009). The study carried out for this thesis concurs with that of Balcombe and Arthington (2009) where a time lagged effect of low flows on brown trout species was found. Thus ultimately brown trout are influenced by low flows but this is not a direct effect and the effect occurs a few years after the low flow/ drought event. On an international level this finding could be implemented in low flow policies where the antecedent flows should be taken better account of in environmental flow determination. By assessing a rivers natural flow regime, it can be shown what the impact of a drought is likely to have on brown trout and therefore measures can be taken accordingly when a drought occurs, furthermore the duration and magnitude of a drought is important, if the drought occurs for a relatively short period of time for example, this could be beneficial to spawning brown trout, for a longer period of time, or during the spawning season, this could have devastating effects. In the latter scenario, stricter environmental controls (i.e. abstraction bans) would be necessary.

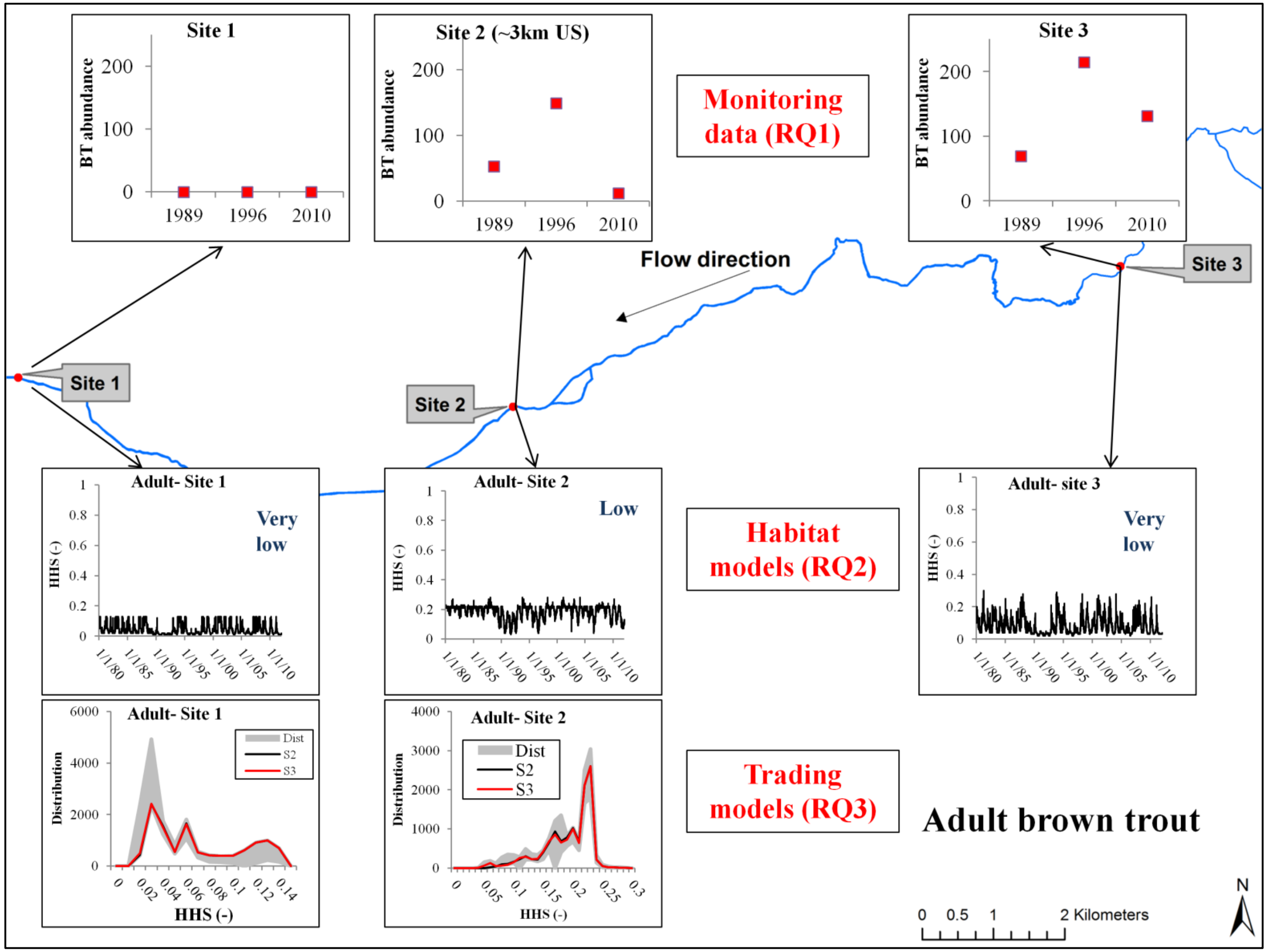


Figure 8.1- Adult brown trout results map

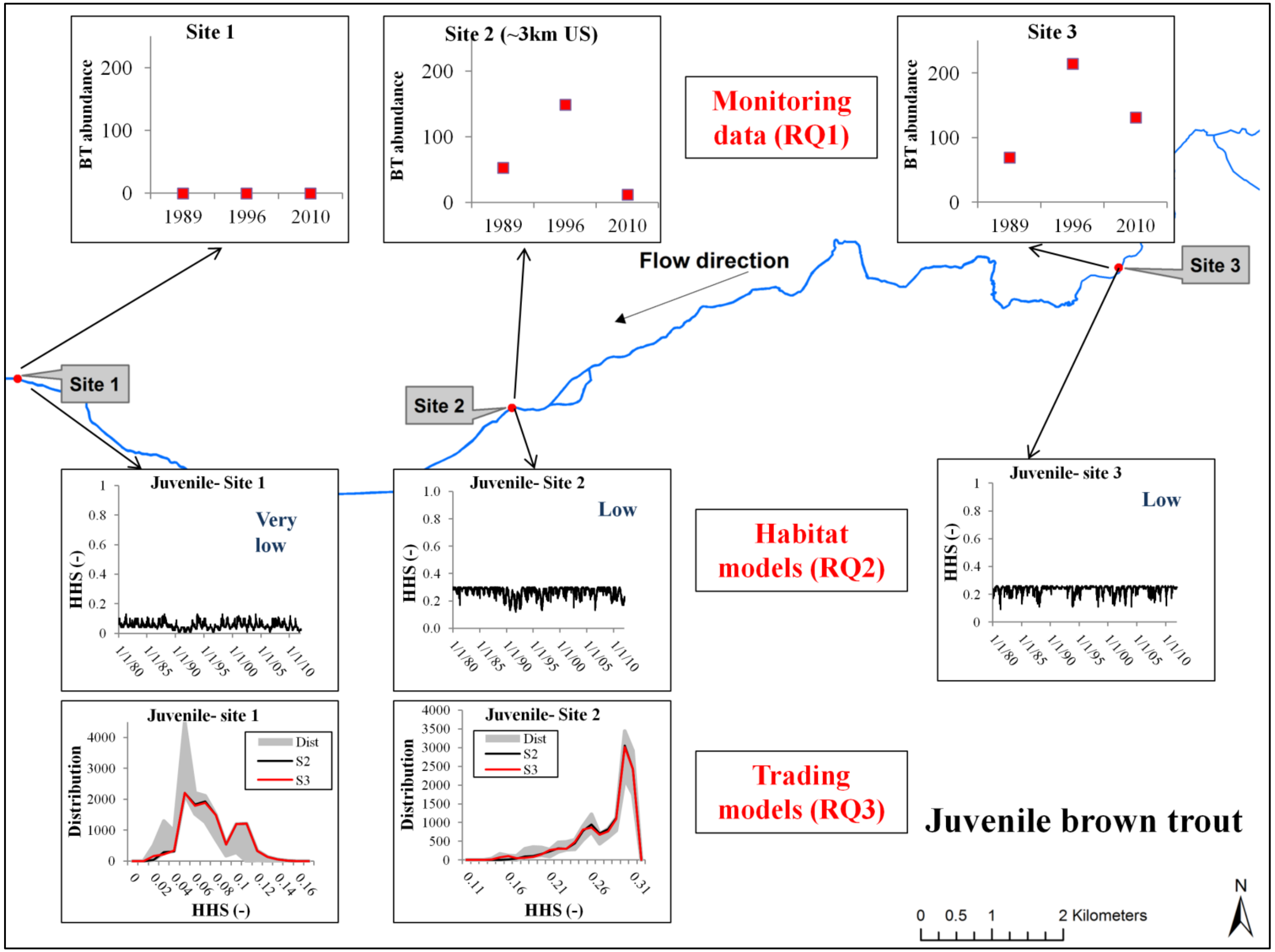


Figure 8.2- Juvenile brown trout results map

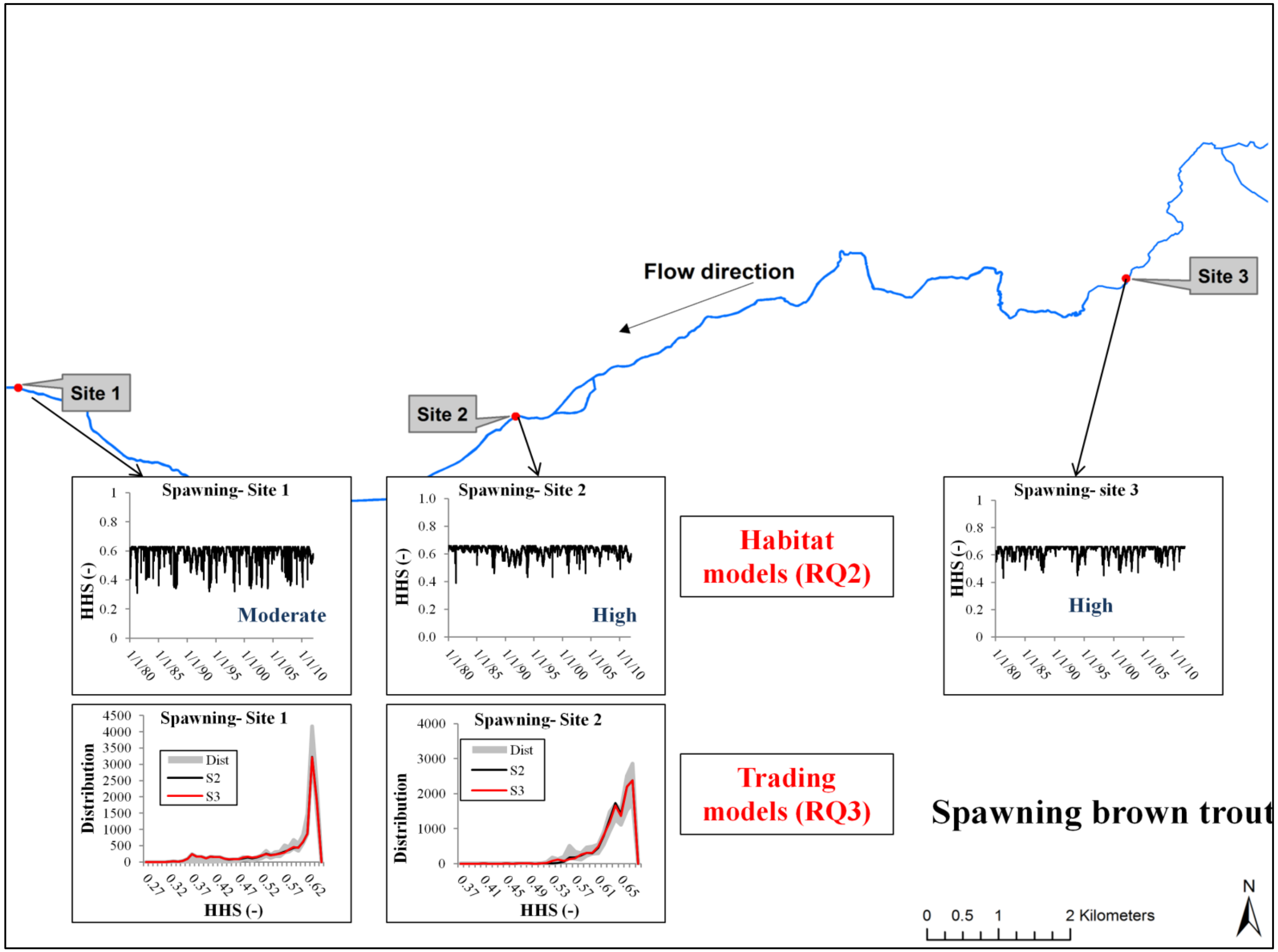


Figure 8.3- Spawning brown trout results map

8.3 Macrophytes: Crowfoot (*Ranunculus*)

8.3.1 Background

Macrophytes were investigated predominantly due to their importance as a driver the structure and functioning of a freshwater ecosystem. SNIFFER (2012) determined macrophytes as one of the indicators of effects of abstraction and flow regulation on river health. Therefore assessing the interactions of macrophytes and flow would provide information on how macrophytes would be affected as a result of low flows caused by over-abstraction.

Macrophytes provide many important ecosystem services which include; refugia for fish and invertebrate, oxygenation of the water and aesthetic values (Westwood et al., 2006). Their lifecycle starts in autumn or winter with the biomass increasing rapidly in the later winter and spring, reaching its maximum in spring or summer when flowering also occurs (Dawson 1979). Summer flushing is also required to remove any build up of silt and algae thus natural high and low flows are required. The biggest threat to macrophytes is drought where changes to channel substrates can occur. Macrophytes depend on clean gravel or pebble beds, decreased velocities replace these substrates with silt, which in turn supports wetland rather than aquatic plants (Holmes 1999). Furthermore droughts can be exacerbated by over-abstraction. Figure 8.4 presents the main findings from each of the research questions for macrophytes.

8.3.2 Main findings

The chalk stream reaches of the River Nar support a wide range of aquatic macrophytes, particularly Crowfoot. This is expected as chalk streams providing pristine conditions for Crowfoot (Berrie 1992). Higher abundances of Crowfoot were found in the fen reach however. The results from RQ1 illustrated that natural fluctuations and growth patterns occur in the River Nar. However it is clear that the natural fluctuations are governed by natural flows i.e. high flows in winter followed by low flows in summer.

Likewise with brown trout, in order to assess how the species interact with low flows, the site conditions firstly had to be assessed. The site conditions had a significant effect on macrophyte abundance as higher abundances were found in wider channels (even if they were artificially modified), gravel substrates with little anthropogenic uses (i.e. farming/ recreation). These findings are in accordance with those of Westwood et al., (2006) who found positive diversity associated with wide channels, semi-natural land use and high water stages. The lowest abundances were found in highly modified channels with narrower channels, silt substrate and heavy poached by livestock. Many authors (e.g. Wilby et al., 1998; Cranston and Darby 2004; Westwood et al., 2006; Franklin et al., 2008) demonstrate that other physical factors should be considered in management alongside that of discharge and velocity. Light availability is key as macrophytes require light for photosynthesis therefore if light availability is limited, a negative impact on growth rates is likely (Franklin et al., 2008). This research has indicated the difference between the chalk and fen reaches; the fen reaches provided slightly better physical habitat for the species than the chalk reach did. Both reach types reported high abundances of the species however generally the fen reach was higher (Figure 8.4). On the River Nar higher abundances were found where there was some overhead cover, however these were mainly found in the centre of the river where light could access them. High abundances of macrophyte species such as Crowfoot and reeds were found where there was no overhead cover, this could however cause eutrophication (i.e. algae growth causing a reduction in the oxygen content of the water) and therefore the right balance of overhead cover is required.

From these findings it would be expected that site 2 (DS Nar) would have the highest habitat availability from the model results; however the model results (RQ2) showed there to be ‘moderate habitat availability’ at all three sites. Therefore initial findings indicate that the model results do not accurately match the abundances found in the river. This is an area of further work to compare the findings of the model to the actual river, for example compare Crowfoot locations to the highest availability for Crowfoot predicted by the model.

During the data analysis, statistical correlation analysis indicated that Crowfoot abundance was not related to daily flows. Having low macrophyte abundance in some sites is not necessarily due to flow conditions and could be related to site conditions, as discussed above. Many other factors other than flow influence the abundance and location of macrophyte growth. Relationships were found between abundance and the antecedent 6 month flow conditions. Crowfoot had the best growth when there were lower flow conditions with a higher flow conditions in the preceding 6 months. This replicated the abundances in winter and summer, i.e. low abundances are found in winter, then 6 months later in the summer during lower flows, higher abundances of Crowfoot were found. Similar findings have occurred in literature, from a 10 year study on the river Lambourn, Ham et al., (1981, cited in Cranston and Darby 2004), determined a positive correlation between spring (March, April and May) discharge and Crowfoot growth, this indicated that higher antecedent flows during spring have a positive effect on Crowfoot growth.

The highest abundances of Crowfoot were found during the lowest flows, using this finding alone it could be concluded that low flows are beneficial to Crowfoot abundances. However as only 1 years' worth of data was collected, the natural fluctuations in growth were the predominant factor to determine abundance and furthermore there was only limited historical data available. Therefore low abundances in winter could not necessarily only be related to the flow, as natural growth and die back was occurring (Dawson 2002). Any prolonged period of low flow would significantly impact Crowfoot growth as higher flows are required for the species, not only due to the antecedent conditions but also to flush out any build-up of silty sediment. These finding have been common throughout literature, Franklin et al., (2008) described how the successful colonisation of macrophytes is controlled by flood frequency as macrophyte growth rates are very slow therefore prolonged periods of hydrological stability are required for macrophytes to develop. Likewise however low flow events caused by drought can have negative effects upon macrophyte growth creating a silt build up. Thus the seasonal wetting and drying is of high importance for macrophyte success. The highest abundances discovered on the River Nar were in the summer months, i.e. the lowest flow conditions; however these were not necessarily 'low' flow conditions in relation to historical flows. Indeed summer 2013 and 2014 had 68 and 12 low flow days (days under Q_{95}) respectively (out of 123 days of summer). Thus summer 2013 was a relatively low flowing summer whereas summer 2014 was not. Cranston and Darby (2004) noted how in the low flows of the late 1980's and early 1990's, major changes occurred in Crowfoot populations in chalk streams due to lack of rain, summer droughts and inadequate winter recharge. Therefore if historical data were available for this analysis, it would be expected that in the years prior to drought conditions i.e. after 1991, macrophyte abundances would be very low.

The main growing season for Crowfoot is April to August, this generally coincides with the lowest flows during the year and therefore the habitat models predicted that during this key period:

- ***For site 1 and 3***, 12 years out of 32 had 'low' HHS periods.
- ***For site 2***, 6 years out of 32 had 'low' HHS periods.

This suggested that during these times the flow conditions would create low HHS for Crowfoot. Whereas in reality this time is probably the best for growth, the habitat models cannot take this into account and instead views the hydraulic conditions to provide low habitat availability. This finding indicated that habitat models cannot be used in isolation and human judgment and analysis needs to be included.

The importance of the natural flow regime for Crowfoot has been highlighted throughout this research, it is clear that Crowfoot requires natural flows including high, low and average flows in order to thrive, this has been determined both in literature (e.g. Dawson 1979; Berrie 1992; Franklin et al., 2008) and in this study. In all three sites Crowfoot had a wide range of results:

-***Site 1***: 0.10-0.57 HHS,

-***Site 2***: 0.19-0.5 HHS,

-***Site 3***: 0.37-0.58 HHS,

All sites were however highly skewed to the upper range of HHS. This shows that the natural flow regime, as exists in the River Nar, does create a range of different habitat availabilities for Crowfoot, however for the majority of the time this is in the ‘upper’ proportion, indicating that the range of habitat required by the species does not exist to its full extent.

Ultimately the findings have demonstrated that *Ranunculus* would survive during low flow periods in drought; however the range of flows is required to sustain their abundances. This is why Crowfoot is so predominant in chalk streams due to the high Base Flow Index (BFI) of 0.9 which sustains a high base flow all year round. Thus management should aim to ensure the natural fluctuations in flow are protected. Furthermore in order to sustain and enhance *Ranunculus* populations, management should deal efficiently with site conditions enhancement. Land use change and narrow channel widths negatively impact on *Ranunculus* growth, enhancing and maintaining the physical features around *Ranunculus* habitat i.e. increasing HQA and decreasing HMS should be important management aims as the subsequent impacts on *Ranunculus* would be beneficial.

To conclude, Crowfoot abundance is not related to daily flow and therefore it cannot be concluded that macrophytes have preference for higher or lower flows. It is more appropriate to examine the entire natural cycle related to flows i.e. high flow in winter, low flow in summer.

8.3.3 UK context

The most key finding for Crowfoot was how important the natural flow regime is for the species. The natural higher flows in winter correspond to the natural die back of Crowfoot and the natural lower flows in summer provide good conditions for growth. Therefore management decisions should take into account a fluctuating flow regime rather than continuous sustained flows.

A further key finding which could be used at national level is how Crowfoot are influenced by site conditions. For example, if there was a certain lack of Crowfoot in a river and restoration measures aimed to reintroduce/increase them, the site conditions could be adapted to account for their preferences as described in section 1.3.2. Old et al., (2014) notes how for 2000 years, chalk streams have been modified; low flows, high nutrient concentrations and deep accumulations of fine sediments have resulted in few macrophytes. Furthermore clearing of riparian woodlands has created less natural shade which increased productivity. This study confers with these findings, and it is clear that whilst flows are the driving factor in macrophyte abundance, the site conditions are key to macrophyte growth.

8.3.4 International context

There are around 210 chalk streams worldwide and 160 of these are in England (Pearce 2014) therefore the findings from this study cannot necessarily be used at international level. A finding that could be transferred however is that habitat models are not the best way to assess habitat availability for macrophytes generally due to their natural growth and die back patterns. The results in this study showed periods of low habitat availability during the summer months whereas in reality there would be more available habitat due to their natural growth. This therefore implies that habitat modelling is not the most appropriate means of making management decisions and instead expert judgment should be used. Studies have shown that variability in macrophyte abundance cannot always be directly attributed to variations in stream flow, and other non-flow related variables at different spatial scales, such as geology of the catchment and catchment rainfall are influencing macrophyte abundance (Westwood et al., 2006). This study also found that site conditions are of great importance for macrophyte growth and therefore showing how habitat models which focus only on hydraulic conditions cannot be used in isolation for management decisions.

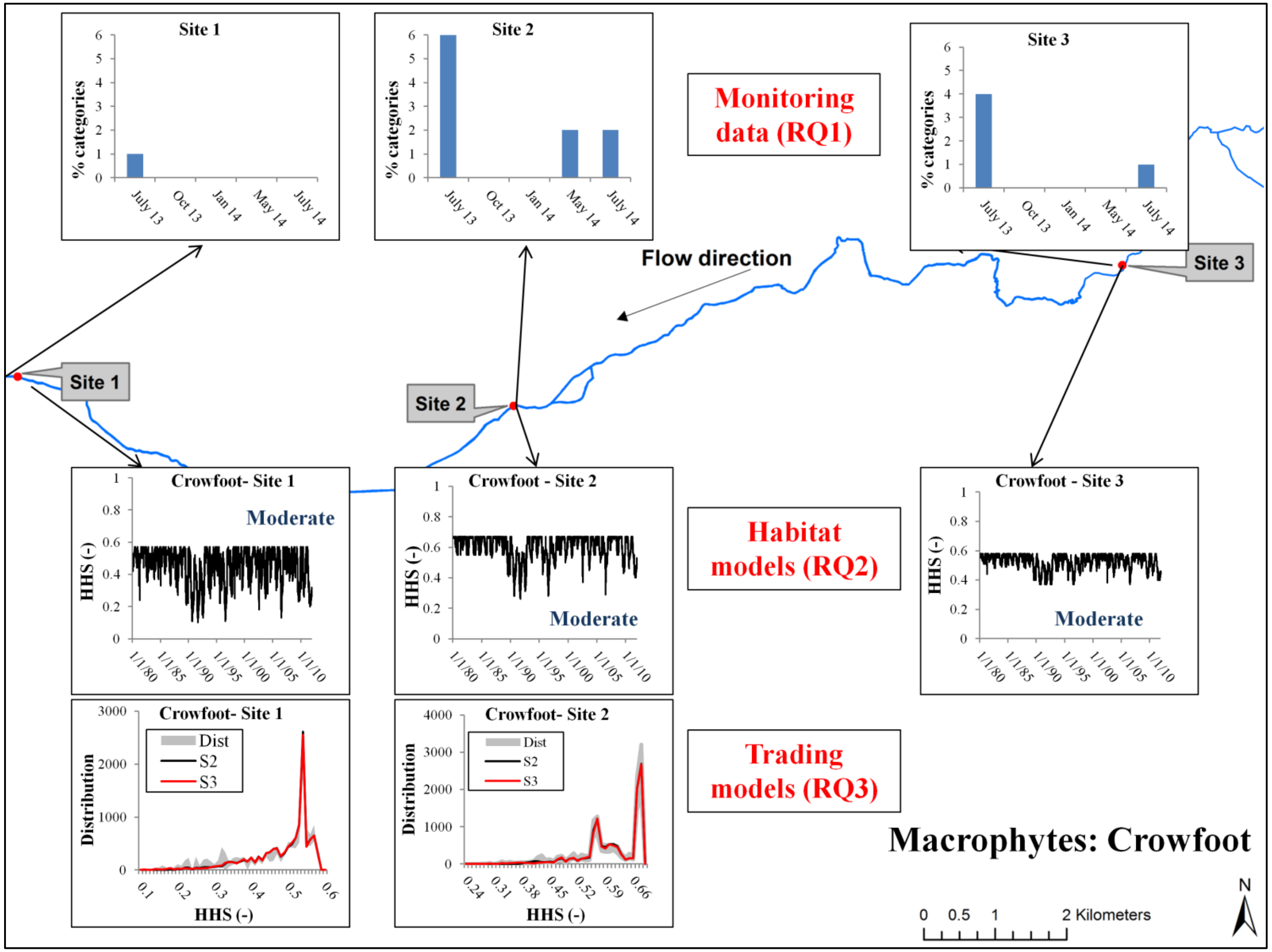


Figure 8.4- Macrophyte results map

8.4 Benthic macro- invertebrate: Mayfly (*Baetid*)

8.4.1 Background

Benthic macro-invertebrate (BMI) were used as an indicator species as they are a good indicator of water quality, and more recently with the development of the LIFE index (see Section 5.2.3), they can be correlated to the flow in a river and used to demonstrate the linkage between hydrological variables. Furthermore BMI provide nutrient cycling, sediment mixing, and energy flow through the food web and therefore provide many ecosystem services (Covich et al., 1999). Mayfly (*Baetid*) were chosen due to their importance as a food source to brown trout. Droughts have a significant impact on BMI, predominantly due to a build up of fine sediment which creates unfavourable substrates. This can reduce the diversity of stream invertebrates and the impacts can be species specific (Berrie 1992; Wood and Petts 1999; Lancaster and Ledger 2015), this makes BMI a good indicator species to use to assess the effects of low flow. It was important to determine how low flows impact on the BMI in the river as the species are functionally important in many aquatic ecosystems whilst also providing an important food source for fish. Therefore the pressures from low flow and over-abstraction that BMI face could be detrimental to their ecological functioning. Figure 8.5 presents the main findings from each of the research questions for BMI.

8.4.2 Main findings

Overall the BMI quality in the river is excellent with the fen reach providing much lower scores than the chalk stream reach. Higher ASPT and LIFE scores were found in the chalk reach. The ASPT is an indication of the quality of the water; therefore the results show that BMI has much better quality habitat in the chalk reach than the fen. Low LIFE scores indicate a high abundance of drought resistant BMI which are found frequently in low velocities. High LIFE scores indicate a large abundance of BMI preferring high velocities and therefore are found frequently in high velocities (SNIFFER 2011). The results reflect this with higher velocities occuring in the chalk reach. In a study by Milner et al., (2015) a significant difference was found in BMI community composition at reach scale which suggested that fluvial geomorphology affects BMI distributions at reach scale. The findings in this study are therefore in line with the study by Milner et al., (2015).

Site 1 (Highbridge) in the fen reach scored ‘low habitat availability’ whilst site 2 (DS Nar) and site 3 (Castle Acre) in the chalk reach, scored ‘moderate habitat availability’. Additionally low flows caused ‘low HHS’ at site 1 but not at site 2 or 3, showing that site 1 was sensitive to the low flows. Natural river systems such as the River Nar, provide habitat heterogeneity across multiple spatio-temporal scales (Milner et al., 2015), therefore differences in BMI communities between chalk and fen reaches provide an indication of natural processes. The site conditions proved to impact upon BMI scores, high habitat quality measured on the RHS provides good correlations between scores and flow. For example, no overhead or instream cover, silty substrate and channel modification provide poor habitat conditions for BMI. These factors should therefore be taken into account in management decisions.

The results from both the collected BMI data and the Environment Agency (EA) BMI data indicated that BMI had little, or no, relationship with daily flow and instead were impacted by the antecedent flow conditions, as discovered by Wilby (2010). A further finding was that low flows do not necessarily result in low scores, and instead they have a lagged response to flows. The ASPT scores had very little significance to antecedent flow conditions; this is as the scores are more related to water quality than quantity. The measured LIFE scores also had little statistical significance to the antecedent flow conditions. However the measured LIFE scores had more statistical significance to flow than the ASPT scores had. When the historical BMI data from the EA was taken into account, stronger statistical relationships with antecedent flow conditions were found. The main findings suggested that summer flows are the most critical in sustaining BMI health and that a very high summer Q_{10} flows negatively affect LIFE scores. Similar findings have occurred in past studies. Extence et al.,(1999) discovered that summer flow variables are the most influential in predicting BMI community structure in chalk and limestone streams. Wilby (2010) found highly significant relationships between LIFE and antecedent summer Q_{95} flow conditions. The findings from the River Nar therefore correspond to the findings.

In response to these findings it is clear that the yearly conditions required to sustain BMI species should be taken into account from a management and environmental flow perspective.

8.4.3 UK and International context

This section has been combined for both UK and international contexts as the findings are relevant to both.

It is well known that the natural flow regime with natural hydrological extremes (floods and droughts) are required to sustain any freshwater ecosystem (Ledger and Milner 2015). However it is essential to understand how BMI respond to stream drying and how they recover, or fail to recover from the events and thus how a prolonged drought would impact on the species. The ability to interpret the resilience however depends on knowledge of the frequency and severity of antecedent droughts (Bogan et al., 2015). This therefore poses a limitation of habitat models in that they can only show how habitat availability changes after an extreme event rather than the necessary post-recovery process of the species.

The most significant and relevant finding from this research was that in the River Nar, BMI have a lagged response to flow and therefore in order to protect BMI species, management policies should take into account the preceding yearly conditions. Due to chalk streams having a high BFI the surface flow is of vital importance as small changes in the surface water flow can have large implications on species compositions. It is for this reason why BMI are related to antecedent conditions. Furthermore likewise with Crowfoot, site conditions can be tailored to improve BMI habitat.

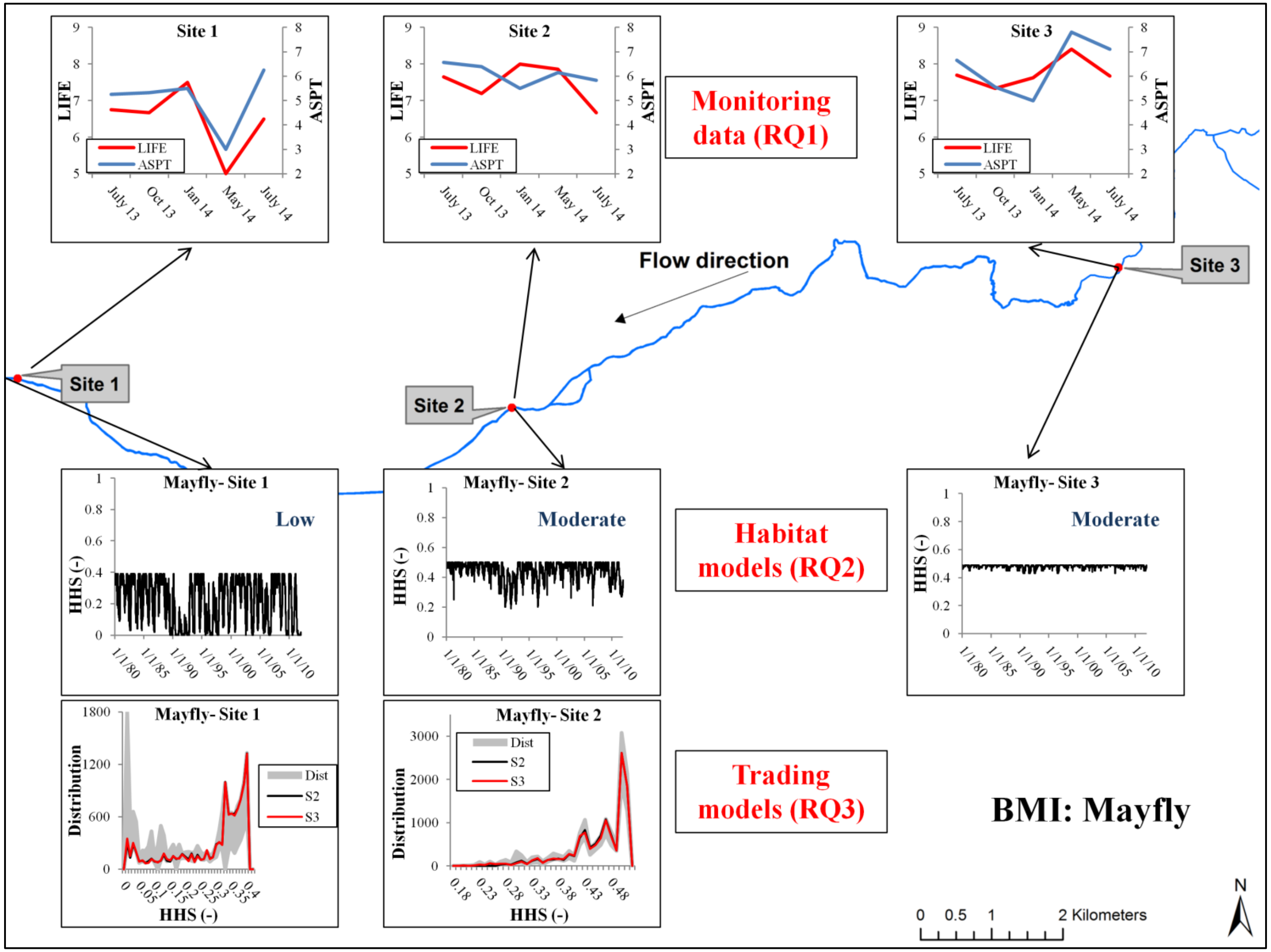


Figure 8.5- BMI results map

8.5 Interconnectedness and limitations with model

A small but significant finding from RQ1 was that brown trout are positively correlated to an increase in food source availability (i.e. BMI), this was also found by Jowett (1992). Other authors also determined that more factors i.e. temperature, cobble substrate, light availability, than only hydraulic factors influence habitat availability (e.g. Stalnaker et al., 1995; Armstrong et al., 2003; Milhouse and Waddle 2012). The analysis carried out in RQ1 was based on a small amount of data however the data which was available provides a vital finding in that brown trout are influenced by other factors other than the widely used depth, velocity, substrate and cover. This finding draws on an important area of research that more than just the hydraulic components influence the habitat availability for any species. This research has been published in Garbe et al., (2016).

Depth, water velocity and substrate size are considered as the most important instream variables affecting brown trout (Armstrong et al., 2003; Louhi et al., 2008). Subsequently habitat models focus on these aspects when determining habitat availability. In reality more factors affect the species and related habitat, as found in RQ1 where increases in brown trout corresponded to increases in BMI scores. This has led to criticism of such approaches as the results represent an incomplete analysis of potential impacts to species of flow changes (Orth 1987). A key challenge for the future and development of habitat modelling is to understand, and integrate the numerous spatial and temporal, abiotic and biotic factors affecting fish and then translate these into models (Maddock 1999).

For these reasons a novel approach was taken to investigate the wider biotic controls on brown trout habitat, alongside the standard abiotic variables. The method used the results from the habitat models for spawning brown trout in conjunction with habitat models investigating refugia (macrophytes: *Ranunculus Fluitans*) and food source (Macro-invertebrates: Mayfly: *Ephemeroptera Beraeidae*) habitat availability.

The interconnectedness of species analysis (analysis 6 in RQ2) indicated that low flows have a negative impact on spawning brown trout when habitat availability is assessed including its biotic dependants of food sources and refugia. This finding is contradictory to the findings in the extreme year analysis (analysis 4, RQ2) where it was found that low flows are preferable for the species. Results indicated that in order to adequately protect all the species, flows would have to be between 1.24-3.06m^3/s at site 1, between 0.39-4.25m^3/s at site 2 and between 0.75-1.82m^3/s at site 3.

Ultimately the interconnectedness research found that understanding the habitat availability of spawning brown trout in isolation does not provide a full picture of the potential interactions associated with its resilience to low flow periods. Consequently, the work has highlighted the importance of combining the biotic dependents of particular species in any investigation, as where there is high available habitat for one species there may be low availability for its dependents. It is understanding these that allows scientists to appreciate the flow requirements of any river reach. This has furthered the research surrounding the criticism of habitat modelling not incorporating more factors.

The spatial analysis (analysis 7 in RQ2) demonstrated the importance of assessing spatial distributions in addition to the interconnectedness investigation. The main output from habitat models is the HHS which provides one number to represent the available habitat. This HHS value can be disaggregated into more detailed SI values but these fail to incorporate other biotic parameters i.e. food sources and refugia. When assessing the spatial variance of habitat availability it becomes clear that whilst the habitat model may predict low available habitat for spawning brown trout, the actual availability may be much higher due to good habitat availability in the same area for food sources and refugia. Particularly when studies, including this study (see section 5.7.4), have shown high correlation between brown trout abundance and invertebrate biomass and cover (Jowett 1992).

Flow has been shown as an important factor in habitat availability for spawning brown trout; this was demonstrated particularly during the hydrological drought of 1991-92 where availability for spawning brown trout remained relatively high whilst availability for food (*Ephemeroptera Beraeidae*) and refugia (*Ranunculus Fluitans*) decreased.

Ultimately low flow conditions do have an impact on habitat availability for spawning brown trout, as whilst habitat models may predict the available habitat for spawning brown trout to remain relatively stable, the habitat availability for their biotic dependants reduces, indicating that the overall available habitat would decrease. This highlights the importance of protecting flows for a wide range of species rather than only one species.

These key findings are important at both UK and international scales as it is clear that habitat modelling requires more data and information than solely hydraulic components to adequately make decisions based on the results.

8.5.1 Limitations of model results

In RQ2, analysis 1 and 8 were the direct sources of determining the sensitivity of input to habitat models. A main area of criticism in habitat modelling is the means of input (e.g. Boavida et al., 2014), here both fuzzy rules and HSC were used and investigated. Ultimately statistically different results were found for all species between HSC and fuzzy rules despite being derived from the same sources, this promotes the sensitivity of inputs and how very different results can be determined. Analysis 8 investigated the differences between results from 1D and 2D models, but using the same habitat suitability data. Again statistically significant differences were found between 1D and 2D results. This could be related to the different means of input of substrate and cover however this does not deter from the fact that differences are found.

Overall these findings show that habitat modelling cannot be fully relied upon for assessing habitat availability and instead should be used as a guide alongside expert knowledge as to how certain flows alter habitat availability. The sensitivity analysis carried out in section 4.7.6 showed that altering the fuzzy rules by ±10% affected the results by a maximum of 7.5% (see section 4.7.6). Therefore the input methods are sensitive to small changes. The results therefore should be used alongside expert knowledge on a site specific basis. Clearly there are limitations with this as time and resources would not allow for this at each site. Therefore this promotes the use of the EFI in low flow policies as using a more generalised method based on the current status of the river both utilises environmental conditions and is time and resource efficient.

8.6 Water trading

8.6.1 Background

The aims of water trading are to create a more sustainable abstraction regime whilst providing more protection to the environment. Water trading involves the transfer of rights of the water abstraction license form one user to another thus allowing more efficient water abstractions whilst also enforcing environmental protections by preventing unnecessary abstractions (Erfani et al., 2015). RQ3 has however demonstrated that the short term effects of decrease habitat availability for all three species.

8.6.2 Main findings

For both sites, trading did not cause habitat availability to change in any significant detail, but the small changes that did occur provide important information. The data distributions showed minimal change therefore throughout the 32 years, trading did not affect the overall distributions of habitat availability. Furthermore the Mann-Whitney tests revealed statistically different results between S1 and S3 in the summer seasons for site 1 and 2 and also for the whole period for site 2.

The Skew and Kurt values for changes in habitat availability due to trading did however reveal changes on a more detailed scale. In site 1 the Skew values revealed trading without HOF to decrease the proportion of habitat availability in the upper HHS for juvenile brown trout and Crowfoot. However the same scenario increases the proportion of habitat availability in the upper HHS for Mayfly and spawning brown trout. For adult brown trout the trading results remained fairly similar to the baseline. For site 2 the skew values revealed trading with HOF to slightly improve habitat but trading without HOF slightly decreased habitat availability. These findings showed that trading does have an impact on habitat distribution. For some species this is providing better habitat availability however for others this provides worse habitat availability.

Thus change does occur as a result of trading and it could be argued that any change is negative in relation to the baseline habitat availability. Providing a decrease in habitat availability for species has a clear negative effect in terms of loss of ecosystem services and knock on effects such as a loss of BMI creates a loss of food resources for fish therefore hindering fish habitat. Likewise providing an increase in habitat availability for species could cause an excessive abundance of Crowfoot which therefore chokes the river or an increase in habitat for brown trout could cause a decrease in smaller fish due to brown trout becoming the top predator.

The proposed abstraction regime reform and increase in water trading does decrease habitat availability, but this small and often insignificant change must be weighed up against the benefits of trading to humans and the overall environment in terms of unsustainable abstraction amounts.

The purpose of the synthetic flow analysis (Section 7.2.3 and 7.3.3) was to investigate if the variations in habitat availability caused by trading were within the limits of the natural flow variation and associated habitat. Thus indicating whether the uncertainty posed by climate change and changing hydrological regimes would affect natural habitat variations (Ledger and Milner 2015). The results from this analysis demonstrated that the changes in habitat availability that do occur as a result of water trading do so within the natural variability. Therefore species would likely adapt and move according to the change in flow created by water trading.

Water trading measures are being promoted as a way of more efficiently distributing water resources. The findings from this analysis are very important in terms of assessing the impacts of water trading, as it shows how the trading measures do not significantly impact upon the habitats within the river. In terms of management of the water trading measures this does not imply water trading can go ahead without any assessment of this impact on a catchment scale due to the site and species specific nature of the results. Moreover it suggests water trading can be used but the HOF is of great importance in order to protect species within the river.

The data distribution analysis (section 7.2.1 and 7.3.1) shows different impacts at different sites, therefore the site conditions are very important. If cover and substrate are the most important habitat determinants for a species then the small change in flow would have little impact on available habitat predicted. Due to the site specific conditions, in order to clarify these results, more sites and rivers would need to be analysed to show the effects of trading in different geographical areas. More trading scenarios would have to be analysed to show how more or less trades affect habitat results and also to show how factors such as the EFI and HOF impact on them.

8.6.3 UK context

Ultimately it has been discovered that water trading only impacts on habitats and ecosystems to a small and insignificant degree, however the important aspect to note is that changes do occur and therefore water trading on a larger scale has the potential to impact on habitats and species.

It is likely that water trading will go ahead in the UK in upcoming years in response to the Water Act 2014 of which one of the main outcomes of was to implement more efficient use of the water that is abstracted (EA 2013b). With this in mind the findings here suggest that so long as the HOF is strictly implemented, the associated habitat on rivers should be protected. It must be remembered however that water is finite and the water that is abstracted must also protect the vital freshwater ecosystems and the services they provide.

Many countries worldwide have implemented water trading measures some more successful than others. The UK has the benefit of being able to create a water trading and market system from scratch, and can therefore tailor its policies and measures on a bottom- up basis in order to maximise efficiency. Different sites on the same river have been shown to impact species in different ways as a result of water trading, this implies that the results found here cannot be generalised and ideally there would be a testing measure as to how water trading is likely to impact species on a river specific basis. Clearly this would be time and cost efficient, however habitat models have been useful in determining the impacts on habitat availability, thus this method could be developed to aid in trading decisions.

8.6.4 International context

Water trading and markets have occurred worldwide in many different forms, ultimately is it clear before doing any research that not one trading system can be appropriate for all countries and catchments. From large country scale, different climates, policies, governments, populations, and water scarcity issues influence how the water trading can physically and administratively work. To smaller scale where changes in flow in different river typologies on the same river cause different reactions per species. This means that it is not a one solution fits all situation and water trading must be assessed case by case.

Chile has one of the earliest and well developed water markets in the world, where water rights have been freely traded for over two decades (Saleth and Dinar 2000). However the water market has been criticised on its lack of environmental protection; by allowing water transfers across hydrological boundaries and inter-sectoral trades, return flows have been reduced and water quality has been impacted through increased waste discharge (Le Quesne et al., 2007). The results from this study can be applied at this scale as the most predominant finding was how important the HOF limit is. In 2005 an ‘ecological’ limit was set on the water trading in Chile which has provided more protection to the environment (Williams et al., 2012). It would be interesting to discover whether the water trading has been impeded due to this newly determined limit or whether this remains an example of a successful water trading scheme. In the River Nar trading scenario the addition of the HOF meant only 5 trades took place as opposed to the 7 trades when the HOF was not implemented, this shows how the HOF does impede water trades, however ecologically this is beneficial and should therefore be a requirement.

8.7 Inherent errors in model and data collection

In this work data analysis (RQ1) was carried out in conjunction with model analysis (RQ2), so the impacts could be assessed on both basis and not relying on the model results. Sensitivity analysis and calibration was carried out during all model stages i.e. in the hydraulic and habitat models, with reasonable results. In RQ3 synthetic flow analysis was carried out in order to show the habitat changes within the scope of natural variation, as the flow data used was only one possible flow scenario and could not be used to predict the future. During data collection errors could occur which are discussed in section 5.9. Despite this however the best methods were used with what was available and all attempts were taken to mitigate against any errors occurring. For example a whole years worth of data, including 2 years worth of summer months were recorded, this enabled comparison between the 2 years so it was know that it wasn't an anomalous year. Additionally during cross section data collection, 4 transects were recorded so the most accurate reading could be used.

Chapter 9- Conclusions

9.1 Introduction

The purpose of this thesis was to assess the impacts of low flow on the three indicator species: Fish (brown trout), macrophytes (Crowfoot) and benthic macro-invertebrate (BMI) (Mayfly) and to investigate the how useful numerical models are in investigating this. These developments were then put into the context of water trading to address how the proposed and forthcoming water trading measures in England would impact the species. This final chapter presents a discussion on the advances in research determined from the work, some concluding remarks on the scientific and applied aspects of the work and finally areas of further work.

9.2 Advances in research

Two major advances in research and additions to science were developed throughout this research:

- Firstly, the interconnectedness of other species developments. The research investigated how more factors that only hydraulic components should be included in habitat models in order to adequately represent habitat availability. This research has been published in Garbe et al., (2016). This is a novel approach as the dynamic nature of habitats could be assessed rather than being included as a variable. Therefore seasonality of other species i.e. refugia and food sources could also be included. This research could be extended. For example BMI are influenced by stream bed composition, predictability of drying/ wetting, extent of drying, amount of woody debris and bank vegetation (Black 2009). Macrophytes are influenced by light and nutrient availability (Franklin et al., 2008). An extension of adding these variables into habitat models would enhance the reliability of the results.
- The second advance in research is the analysis of how trading impacts on habitat. The method used presents a unique and novel approach to this investigation. This research is particularly pertinent given that water trading is being promoted in England and this can improve our understanding of potential impacts. There are undoubtedly ways forward with this research; this is discussed in section 8.6.

9.3 Key scientific findings

This section discusses the key scientific findings from the work, the aim of the thesis was to assess the effects of low flows on biotic indicators of environmental quality:

- Flow is important but cannot be used in isolation to determine impacts on species. Extreme events (i.e. floods and droughts) have impacts on species but under normal flow conditions, site conditions are more influential to species than flow. This leads onto the key finding that of the **site specific nature of habitats**. A large difference in habitat exists between chalk and fen reaches. Subsequently management strategies need to factor in these differences, if minimum flows and other management techniques are set for an entire catchment based on one area/ finding or species this may negatively affect species in other areas. Catchment based plans may not be the best option for management; this study has shown that different typologies provide different habitat availabilities and characteristics which should be taken into account in decision making.
- Any degradation caused by low flows can be partly offset by good channel physical habitat. This was shown in how important the site conditions are for specific species, as well as suitable flow conditions.
- Likewise, multi-species linkages (interconnectedness) can also improve any degradation caused by low flows in terms of the role of macrophytes and BMI as part of the physical habitat template. Food webs are also important and species recovery pathways trajectories following stress and disturbance are important.

9.4 Key applied findings

This section discusses the key applied findings from the work:

- Habitat models can show low habitat availability during low flow but fail to show recovery and the effect of antecedent flow conditions. Furthermore habitat models fail to incorporate important biotic parameters such as refugia, water quality and food sources. Thus the secondary key finding is that of **habitat model uncertainty**. This study has presented a way forward with habitat modelling by using different species results in combination in order to include for example refugia and food sources into available habitat calculations (Garbe et al., 2016).
- Trading in the UK, specifically the south east of England is likely to be encouraged in the near future through policy change. This study has investigated the impacts of this trading on the freshwater environment and found that **trading does not impact species if HOF is activated**. Furthermore, whilst trading does impact on habitat availability, it is only by a relatively small amount which whilst potentially due to

the low amount of trades, shows how this must be weighed up against the benefits of water trading to water resources.

9.5 Further work

As with any research project this research has highlighted areas which could be expanded upon in which were not in the scope of this study.

- As water trading is relatively new to England, there is the benefit that the policies, management and administration has a blank slate and therefore can be adapted as most appropriate. Moving forward from this research, a system could be developed that links habitat models with trading models to create an online coupled system where traders could buy and sell their licenses with the environmental limitations. A key finding from this research was how that site specific conditions impact results. If there were a coupled system which incorporates habitat models, and therefore hydraulic models, the site conditions (i.e. specific width and depths), could be included showing the impacts a particular trade could have. For example, if a trader took out Xml/d at X location, this would reduce flows by Xml/d amount and therefore impact habitat availability by X HHS'. The specific details of this system would need refining; this is however a potential way forward for water trading.
- It would be beneficial to apply the methods used in this study to other rivers in order to validate the findings. Site conditions were found to be critical to findings and hence it would be useful to assess how habitat is affected and how sensitive it is to change in different rivers.
- An area of further work which would be beneficial would be to compare the model findings to the actual river, for example to compare *Ranunculus* locations to the highest availability for *Ranunculus* predicted by the model. This has scarcely been done in literature and would be beneficial to validate model findings.
- If modelling could incorporate antecedent flows and sediment movement this would improve the basis around habitat models as if there were a drought, it would be important to show how long habitats would take to recover. This would be necessary for agent based modelling.
- Water quality is an area which has not been covered in this study which does have a potentially important implication on habitat availability. This could be both an a localised scale i.e. spatial analysis (analysis 7 of RQ2, see section 6.8) and also on greater scale for example, showing if or how water trading affects water quality and

return flows. Mackintosh (2015) investigated if water quality could be a determining factor in habitat availability, the following conclusions were drawn:

- Crowfoot are limited by light intensity which also limits Dissolved Oxygen (DO) production.
- Spawning Brown Trout and BMI are limited by DO and Oxygen reduction potential (ORP).
- These linkages suggest light intensity directly or indirectly limits habitat availability of all indicator species.

It was discovered therefore that species are impacted by water quality parameters and this should be taken forward in further research.

9.6 Concluding remarks

To open the concluding remarks I put forward a question raised by Beven and Alcock (2012), due to inherent uncertainty in knowledge, and the natural randomness of environmental forcing, can predictions made by models really be useful in informing management decision? The fact is that modelling occurs on many different scales: hydraulic, hydrological, habitat, climate change, sediment movement etc. and furthermore there are debates and conflicts amongst scientists regarding habitat preferences and how ecosystems and the services they provide will be impacted in the future (Lake 2003; Louhi et al., 2008; Bogan et al., 2015). These conflicts and uncertainties occur as there is no one-size-fits-all global solution to the uncertain future of the environment, moreover the global heterogeneity of species and environmental makes generalisations obsolete. Regardless of uncertainty however, decisions must be made, the question is what are the best decisions to make based on the information available? This study has shown that whilst limitations and weaknesses occur in habitat modelling, they are useful to aid in decision making so long as expert knowledge is used alongside them. What is clear is that there is a need to protect the freshwater environment for both humans and species through ecosystem services.

This research has unpicked how species are influenced by low flows in the context of water trading. Future climate change and observed global trends for drought are uncertain, but evidence suggests that drought has increased in some regions (e.g. Mediterranean) and decreased in others (e.g. central North America) since the 1950’s (Ledger and Milner 2015). Effects of single events are highly context dependant, from damaging to beneficial. Here it has been shown that drought and low flows impact different species in different ways i.e. beneficial for spawning brown trout habitat but detrimental for Crowfoot habitat.

Chapter 10-References

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Chapter 11-Appendix

11.1 Appendix A- LIFE and BMWP scores

BMWP and LIFE scoring taxa

All taxa on this list should be routinely identified for "BMWP family level" analyses. Those in bold are the additional taxa required by the LIFE score system and will be included in 2006-2011 CEH audit.

FLATWORMS	BMWP	LIFE
Planariidae	5	4
-Dugesiidae	5	4
Dendrocoelidae	5	4
MOLLUSCS		
Neritidae	6	2
Viviparidae	6	3
Valvatidae	3	4
Hydrobiidae	3	4
-Bithyniidae	3	4
Physidae	3	4
Lymnaeidae	3	4
Planorbidae	3	4
Ancylidae	6	2
-Acroloxidae	6	4
Margaritiferidae	-	2
Unionidae	6	4
Sphaeriidae	3	4
Dreissenidae		4
LEECHES		
Piscicolidae	4	2
Glossiphoniidae	3	4
Hirudinidae	3	4
Erpobdellidae	3	4
SPIDERS		
Agelenidae *	-	5
CRUSTACEA		
Artemiidae	-	6
Chirocephalidae	-	6
Triopsidae	-	6
Astacidae	8	2
Mysidae	-	5
Asellidae	3	4
Corophiidae	6	3
Talitridae	-	6
Gammaridae	6	2
-Crangonyctidae	6	4
-Niphargidae	6	
MAYFLIES		
Siphlonuridae	10	4
Baetidae	4	2
Heptageniidae	10	1
Leptophlebiidae	10	2
Potamanthidae	10	3
Ephemeridae	10	2
Ephemerellidae	10	2
Caenidae	7	4

STONEFLIES	BMWP	LIFE
Taeniopterygidae	10	2
Nemouridae	7	4
Leuctridae	10	2
Capniidae	10	1
Perlodidae		
Perlidae	10	1
Chloroperlidae	10	1
	10	1
DRAGONFLIES/DAMSELFLIES		
Platycnemididae	6	4
Coenagriidae	6	4
Lestidae	8	4
Calopterygidae	8	3
Gomphidae	8	3
Cordulegasteridae	8	2
Aeshnidae	8	4
Corduliidae	8	4
Libellulidae	8	4
BUGS		
Mesoveliidae	5	5
Hebn'dae	-	4
Hydrometridae	5	4
Veliidae	-	4
Gerridae	5	4
Nepidae	5	5
Naucoridae	5	4
Aphelocheiridae	10	2
Notonectidae	5	4
Pleidae	5	4
Corixidae	5	4
BEETLES		
Haliplidae	5	4
Hygrobiidae	5	5
Dytiscidae	5	4
-Noteridae	5	4
Gyrinidae	5	4
Hydrophilidae	5	4
-Hydraenidae	5	4
Scirtidae	5	4
Dryopidae	5	4
Elmidae	5	2

ALDERFLIES/LACEWINGS	BMWP	LIFE
Sialidae	4	4
OsmyNdae	-	2
Sisyridae	-	4
CADDISFLIES		
Rhyacophilidae	7	1
-Glossosomatidae	7	2
Hydroptilidae	6	4
Philopotamidae	8	1
Psychomyiidae	8	2
-Ecnomidae	8	3
Polycentropodidae	7	4
Hydropsychidae	5	2
Phryganeidae	10	4
Brachycentridae	10	2
Lepidostomatidae	10	2
Limnephilidae	7	4
Goeridae	10	1
Beraeidae	10	2
Sericostomatidae	10	2
Odontoceridae	10	1
Molannidae	10	4
Leptoceridae	10	4
TRUEFLIES		
Tipulidae	5	4
-Limoniidae	5	2
-Pediciidae	5	2
Ptychopteridae	5	2
Chaoboridae	-	5
Culicidae	-	5
Simuliidae	5	2
Chironomidae	2	-
Syrphidae	-	5

*Agelenidae will change 'to Cybaeidae with the next taxon dictionary update c. March 2006

11.2 Appendix B- CASiMiR Course certificate

sje

Ecohydraulic Engineering GmbH

sje Ecohydraulic Engineering GmbH
Viereichenweg 12 · 70569 Stuttgart

Fon: +49-(0)711-677-3435, -68-3436
Fax: +49-(0)711-677-3436

Dr.-Ing. Matthias Schneider

Stuttgart, 22.3.2014

Certificate of Participation

Ms. Jennifer Garbe

has successfully attended the

Introduction course for the CASiMiR habitat simulation system

The course took place on: **March 21 and 22, 2014**

In: **Stuttgart**

The main topics covered were:

- Physical Habitat modelling in surface waters and approaches
- Simulation model CASIMIR-Fish and range of application
- Fuzzy-rule based habitat modelling
- Exercises and structure of input data
- Simulation model CASiMiR-Hydropower
- Simulation model CASiMiR-GIS

Dr.-Ing. Matthias Schneider, Geschäftsführer der sje GmbH

sje
Ecohydraulic Engineering GmbH
Viereichenweg 12
D-70569 Stuttgart

Handelsregister Stuttgart
HRB 22240
Geschäftsführer:
Dr.-Ing. Matthias Schneider
Dr.-Ing. Klaus Jorde

Fon: +49-(0)711-677-3435
Fax: +49-(0)711-677-3436
e-mail: mailbox@sjeweb.de
URL: http://www.sjeweb.de

USt-IdNr. DE 216049057
Landesbank BW
BLZ 600 501 01
Konto 22 66 024

11.3 Appendix C- Risk assessment form

School/Institute/Directorate:	Location:	Reference:	Date:	Assessor:
School of the Built Environment	Edinburgh	Field trip to Norfolk	30/6/14	Jennifer Garbe, Dr Lindsay Beevers

Describe the task and equipment used:
3 day fieldwork trip on the River Nar in Norfolk. The fieldwork involves: Walking alongside the river, Carrying out kick samples in the river. Equipment: Sample net, white tray, sample jars, ethanol, waders, buoyancy aid, high-vis jacket. Carrying out vegetation surveys both in and alongside the river. Equipment: Waders, record sheets, buoyancy aid, high-vis jacket.
PPE required for task:

What are the Hazards?	Who might be harmed?	Control measures (What are you already doing?)	L	S	R	Additional control measures (What further action is necessary?)	Action by whom?	Action by date
Travel to and around study site (car and train)	PhD students	-Wear Seat belts -Take breaks -Spare tyre in car -Avoid driving at night time -Share driving if possible -Carry mobile phone -Use sat nav and plan routes	1	2	2			
Working in remote areas: Getting lost/	PhD students	-Take detailed map, plan routes -Carry phone, GPS and compass -Leave a plan and a phone number with hotel	2	2	4			

stranded, injury, risk from livestock, hypothermia, dehydration etc.		-Take suitable clothing for expected weather: waterproof, warm coat, sun hat, gloves -Ensure no-one is on their own at any time -Carry food, warm drink and first aid kit -Check weather forecast -Wear hi-vis clothing -Avoid boisterous cattle -Do not pass between cows and calves -Avoid fields with bulls -Adhere to outdoor access code						
Working alongside the river: slip on bank, injury	PhD students	-Wear boots with good tread and ankle support. -Avoid hazardous terrain wherever possible, steep slopes, rock, hidden ditches -Carry first aid kit -Wear hi-vis clothing -Use gate and hurdle points where feasible. -Wear buoyancy aid and hard hats						
Working in the river: loss of balance, contaminated water, drowning, hypothermia, high flows	PhD students	-Do not get into water flowing at or above 1m/s -Always wear buoyancy aid, hi- vis jacket and hard hat -Do not enter water more than waist deep -Always have access to the bank of the river -Do not enter river where not in walking distance of car -Use assistance when getting into river from banks -Use bridges to survey where possible -When entering river, ensure facing upstream to ensure knees are not easily exposed to current -Check weather forecast and river levels by the EA before going on site -Avoid areas with debris in the river to avoid falling	2	3	6			

		-Use hand sanitizer after working in the water						
Carrying equipment: Muscular injury, trip/ fall	PhD students	-Carry larger items on back/attached to rucksacks to keep hands free -Bend knees and have good posture when lifting heavy items -If item is too heavy, ask for assistance, do not attempt to carry -Carry a first aid kit	1	2	2			

Risk Analysis Matrix Level of Risk						Likelihood	Severity	Next review date:
Likelihood	4	4	8	12	16	Unlikely	Insignificant/No Injury	
	3	3	6	9	12	Possible	Minor Injury	
	2	2	4	6	8	Likely	Moderate Injury	
	1	1	2	3	4	Certain	Major Injury/Fatality	
	x	1	2	3	4	*Score likelihood*	*Score severity*	
	Severity							

11.4 Appendix D- EA BMI sampling dates

	KS	10	9	8	7	6	5	4	2	1
1985	Win									
	Sp									
	Sum									
	Aut									
1986	Win									
	Sp									
	Sum									
	Aut									
1987	Win									
	Sp									
	Sum									
	Aut									
1988	Win									
	Sp									
	Sum									
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2007	Win									
	Sp									
	Sum									
	Aut									
2008	Win									
	Sp									
	Sum									
	Aut									
2009	Win									
	Sp									
	Sum									
	Aut									
2010	Win									
	Sp									
	Sum									
	Aut									
2011	Win									
	Sp									
	Sum									
	Aut									
2012	Win									
	Sp									
	Sum									
	Aut									

11.5 Appendix E- Macrophyte survey sheets

US

Right Bank

Left Bank

100m

90m

80m

70m

60m

50m

40m

30m

20m

10m

0m

DS

Parsnip

Reeds

Ran

Algae

Starwort

Species	Abundance in 100m
Common water crowfoot	
River water crowfoot	
Water parsnip	
Mares tail	
Starwort	
Watercress	
Other	
Other	

Scale (for 100m survey length)	
1	<0.1%
2	0.1-1%
3	1-2.5%
4	2.5-5%
5	5-10%
6	10-25%
7	25-50%
8	50-75%
9	>75%

Site number..
Site location..
US Grid coordinates..
DS Grid coordinates...
Adverse conditions..
Depth..
Width..
Substrate type..
Artificial features in section..
Features of interest.................... ....
Left bank use................... ....
Right bank use..
Photo refs:

DS	XS		Look US		Look DS	
Middle	XS		Look US		Look DS	
US	XS		Look US		Look DS	

Other.....

Notes- surveyed from right/left bank
Date
Time

11.6 Appendix F- EA electro fishing dates

	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7
	East Lexham	**West Lexham**	**Castle Acre**	**Manor farm**	**Warren farm**	**Narford Hall**	**Marham intake**
1989							
1990							
1991							
1992							
1993							
1994							
1995							
1996							
1997							
1998							
1999							
2000							
2001							
2002							
2003							
2004							
2005							
2006							
2007							
2008							
2009							
2010							
2011							
2012							
2013							

11.7 Appendix G- Cover and substrate recording example (Site 3)

XS5400 (1)			
582655.7		315265.4	
Factor	**L bank**	**Centre**	**R Bank**
Substrate	4	5	4
Cover type	1	0	0

XS5350 (2)			
582627.3		315269.9	
Factor	**L bank**	**Centre**	**R Bank**
Substrate	4	5	4
Cover type	0	1	0

XS5300 (3)			
582593.9		315265.6	
Factor	**L bank**	**Centre**	**R Bank**
Substrate	4	4	4
Cover type	10	0	0

XS5250 (4)			
582560.7		315258.3	
Factor	**L bank**	**Centre**	**R Bank**
Substrate	2	3	4
Cover type	0	1	10

XS5200 (5)			
582532.4		315232.5	
Factor	**L bank**	**Centre**	**R Bank**
Substrate	2	3	3
Cover type	10	0	0

XS4960 (6)			
582522.1		315200.8	
Factor	**L bank**	**Centre**	**R Bank**
Substrate	2	3	3
Cover type	0	0	1

XS4940 (7)			
582499.1		315148.8	
Factor	**L bank**	**Centre**	**R Bank**
Substrate	5	3	5
Cover type	10	0	0

XS4900 (8)			
582493.3		315108.9	
Factor	**L bank**	**Centre**	**R Bank**
Substrate	2	3	3
Cover type	0	0	1

XS4800 (9)			
582477.2		315068.3	
Factor	**L bank**	**Centre**	**R Bank**
Substrate	5	4	4
Cover type	0	0	10

XS4760 (10)			
582452.3		315043.4	
Factor	**L bank**	**Centre**	**R Bank**
Substrate	4	4	5
Cover type	10	0	0

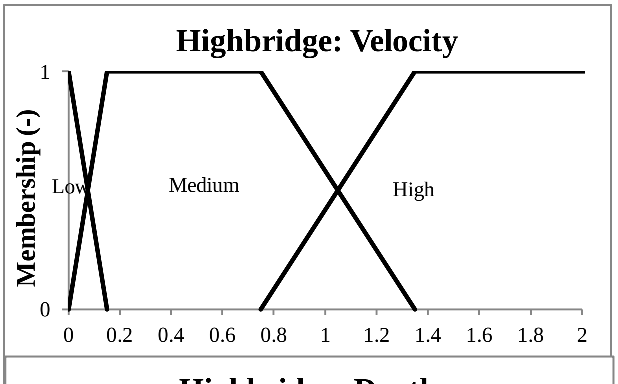
XS4745 (11)			
582420.9		315030.2	
Factor	**L bank**	**Centre**	**R Bank**
Substrate	4	4	4
Cover type	7	0	0

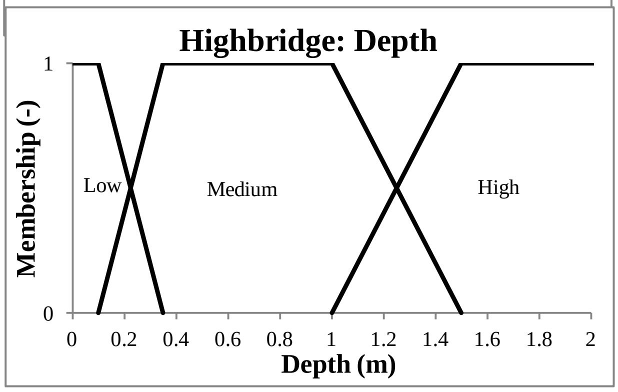
XS4740 (12)			
582397.6		315005.2	
Factor	**L bank**	**Centre**	**R Bank**
Substrate	4	4	4
Cover type	7	0	0

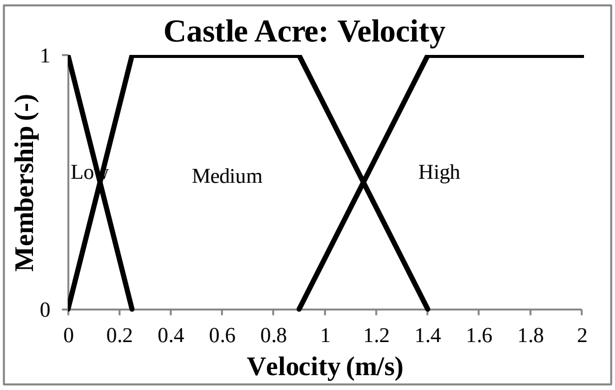
XS4720 (13)			
582363.1		314975.9	
Factor	**L bank**	**Centre**	**R Bank**
Substrate	2	4	4
Cover type	0	0	0

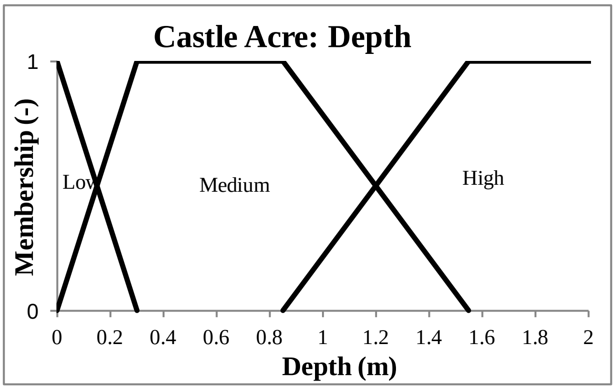
XS4700 (14)			
582320.2		314974.3	
Factor	**L bank**	**Centre**	**R Bank**
Substrate	4	4	4
Cover type	0	0	0

11.8 Appendix H- Fuzzy sets for Crowfoot









11.9 Appendix I- Velocity and depth validation results for Mayfly

Upstream model

Site no.	Flow (m^3/s)	Vel (m/s)	Depth (m)	Baetidae
4	0.75	0.185	0.933	C
	1.03	0.211	1.003	C
	0.59	0.165	0.883	A
	0.13	0.072	0.652	A
	0.31	0.112	0.753	A
	0.62	0.165	0.883	*
	0.79	0.191	0.948	A
	0.55	0.158	0.864	*
	0.56	0.158	0.864	B
	0.64	0.172	0.9	B
	0.25	0.1	0.724	A
	0.43	0.143	0.824	B
	0.24	0.1	0.724	A
	0.31	0.112	0.753	B
	0.28	0.112	0.753	B
	0.29	0.112	0.753	*
	0.85	0.197	0.962	*
	0.73	0.185	0.933	A
	0.64	0.172	0.9	
	0.53	0.158	0.864	*
	0.67	0.179	0.917	*
	0.39	0.133	0.803	A
	1.07	0.215	1.017	
	0.66	0.172	0.9	
	0.5	0.151	0.844	
	0.77	0.185	0.933	
	0.97	0.207	0.99	
	0.3	0.112	0.753	
5	0.59	0.455	0.335	B
	0.63	0.468	0.347	B
	1.36	0.572	0.486	D
	0.7	0.475	0.36	A
	0.74	0.485	0.371	B
	0.32	0.362	0.246	A
	0.26	0.341	0.225	B
	0.39	0.398	0.279	B
	0.16	0.283	0.18	A
	0.19	0.314	0.25	A
	0.18	0.314	0.25	
	0.41	0.398	0.279	B
	0.82	0.495	0.382	B
	1.05	0.536	0.433	A
	0.72	0.475	0.36	B
	0.74	0.485	0.371	A
	0.84	0.504	0.392	A
	0.33	0.381	0.263	A
	0.57	0.443	0.321	B
	0.32	0.362	0.246	
	0.41	0.398	0.279	A
	0.32	0.362	0.246	B
	1.12	0.542	0.443	B
	0.96	0.522	0.413	A
	1.01	0.529	0.424	
	0.7	0.475	0.36	
	0.89	0.513	0.403	B
	0.51	0.402	0.321	B
	1.13	0.548	0.416	B
	0.53	0.443	0.321	A
	0.8	0.495	0.382	B

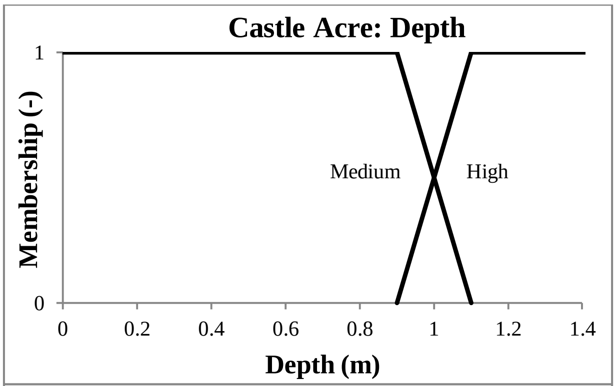
Downstream model

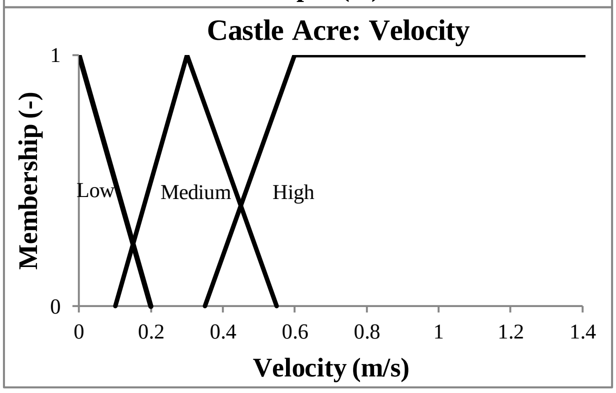
Site no.	Flow (m^3/s)	Vel (m/s)	Depth (m)	Baetidae
7	0.76	0.138	1.058	A
	1.32	0.18	1.261	C
	0.9	0.151	1.118	
	0.95	0.155	1.138	
	0.51	0.11	0.895	A
	0.27	0.071	0.81	B
	0.5	0.11	0.895	B
	0.21	0.06	0.78	A
	0.25	0.071	0.81	A
	0.26	0.071	0.81	A
	0.52	0.11	0.895	
	1.21	0.174	1.228	
	1.34	0.183	1.277	
	0.93	0.155	1.138	A
	0.95	0.155	1.138	A
	1.08	0.166	1.193	A
	0.43	0.104	0.92	B
	0.73	0.133	1.037	
	0.34	0.089	0.867	A
	0.53	0.117	0.969	A
	0.45	0.104	0.92	A
	0.76	0.138	1.058	A
	1.23	0.177	1.244	A
	0.9	0.151	1.118	A
	1.14	0.17	1.21	B
	0.96	0.155	1.138	A
	1.81	0.21	1.411	B
	1.11	0.166	1.193	
	0.42	0.097	0.894	
	0.81	0.142	1.079	
8	1.37	0.137	0.964	
	0.3	0.054	0.53	A
	0.8	0.099	0.77	A
	1.61	0.148	1.016	
	2.04	0.17	1.127	
	1.4	0.137	0.964	
	1.44	0.14	0.977	A
	1.63	0.151	1.029	A
	0.67	0.088	0.71	
	1.1	0.12	0.876	A
	0.45	0.07	0.617	A
	0.8	0.099	0.77	A
	0.69	0.092	0.731	A
	2.69	0.196	1.269	
	1.87	0.163	1.09	
	1.36	0.135	0.949	A
	1.74	0.156	1.054	
	2.21	0.177	1.159	
	2.74	0.196	1.278	
	1.69	0.153	1.042	
	2.21	0.177	1.159	
	1.7	0.153	1.042	
	2.51	0.189	1.226	
	0.76	0.096	0.751	
9	1.54	0.322	0.84	A
	0.94	0.291	0.643	A
	2.04	0.346	0.959	C
	1.4	0.315	0.794	
	1.46	0.317	0.809	A

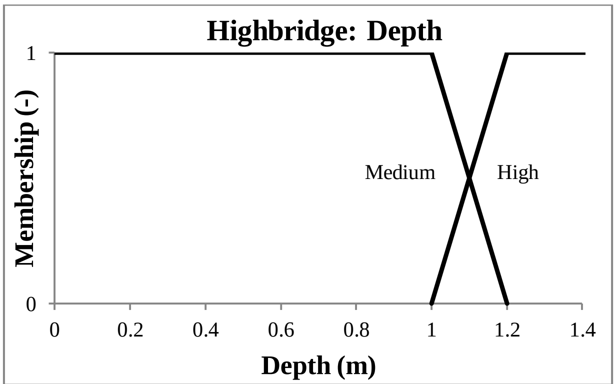
	0.3	0.362	0.246	
	0.56	0.443	0.321	
	0.34	0.381	0.263	
	0.98	0.529	0.424	
	0.29	0.362	0.246	
	0.21	0.314	0.25	
	0.4	0.398	0.279	
	0.36	0.381	0.263	
	0.64	0.468	0.347	
	0.79	0.495	0.382	
6	0.83	0.305	0.485	A
	0.86	0.305	0.485	B
	0.47	0.275	0.352	B
	0.3	0.257	0.29	
	0.46	0.275	0.352	A
	0.17	0.226	0.211	B
	0.23	0.251	0.265	A
	0.23	0.251	0.265	A
	0.48	0.28	0.37	A
	0.97	0.313	0.494	
	1.23	0.333	0.593	A
	0.85	0.305	0.485	
	0.87	0.305	0.485	A
	0.68	0.294	0.439	B
	0.76	0.298	0.455	A
	0.41	0.286	0.334	C
	1.33	0.34	0.618	
	0.62	0.287	0.405	B
	0.94	0.313	0.494	
	0.89	0.309	0.5	
	0.66	0.29	0.423	
	0.14	0.226	0.211	
	0.4	0.286	0.334	
	0.4	0.286	0.334	
	0.47	0.275	0.352	
	1.44	0.346	0.641	
	0.77	0.298	0.455	
	1.29	0.336	0.606	
	1.05	0.32	0.487	
	1.41	0.343	0.629	
	0.74	0.298	0.455	
	1.15	0.327	0.568	
	0.34	0.263	0.313	
	1.51	0.35	0.652	
	0.46	0.275	0.352	
	0.51	0.28	0.37	
	0.31	0.257	0.29	
	0.25	0.251	0.265	
	0.3	0.257	0.29	
	0.47	0.275	0.352	
	0.42	0.286	0.334	
	0.75	0.298	0.455	
	1.04	0.32	0.487	
	0.92	0.309	0.5	

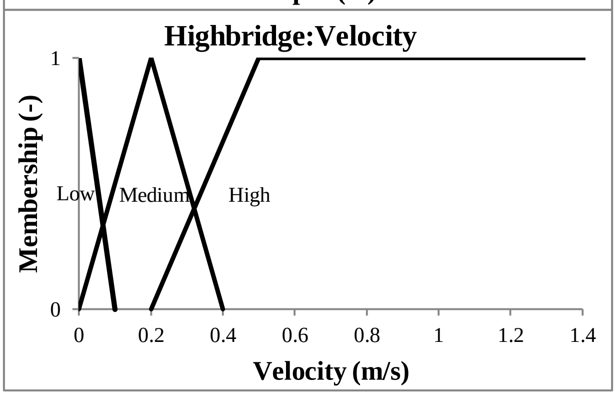
	0.71	0.276	0.54	A
	0.48	0.261	0.444	A
	0.75	0.279	0.562	A
	0.37	0.249	0.358	A
	0.4	0.254	0.387	A
	0.33	0.249	0.358	A
	1.02	0.294	0.662	C
	1.64	0.327	0.866	
	1.03	0.294	0.662	B
	1.57	0.325	0.853	A
	1.65	0.327	0.866	
	1.21	0.305	0.731	
	0.46	0.258	0.416	A
	0.72	0.276	0.54	C
	2.25	0.354	1.016	
	1.76	0.293	0.813	
	1.95	0.341	0.945	
	0.57	0.265	0.47	
	2.55	0.366	1.084	
	0.78	0.283	0.583	
	0.89	0.289	0.624	
	0.41	0.254	0.387	
	0.91	0.289	0.624	
	1.42	0.315	0.794	
10	1.6	0.424	0.667	A
	0.78	0.348	0.488	
	0.28	0.26	0.315	
	0.82	0.348	0.488	A
	0.32	0.26	0.315	A
	0.54	0.309	0.399	C
	0.43	0.294	0.376	B
	1.12	0.381	0.564	B
	1.26	0.395	0.596	A
	1.68	0.434	0.682	
	1.71	0.434	0.682	A
	1.81	0.444	0.697	A
	1.82	0.444	0.697	A
	0.75	0.342	0.474	
	1.22	0.391	0.585	
	1.41	0.408	0.628	
	0.9	0.359	0.516	A
	0.76	0.342	0.474	A
	3	0.534	0.864	
	1.78	0.444	0.697	
	1.55	0.419	0.661	A
	1.78	0.444	0.697	A
	1.62	0.424	0.667	A
	2.04	0.466	0.735	
	0.83	0.354	0.502	
	2.61	0.507	0.812	
	1.13	0.386	0.47	
	1.75	0.439	0.69	
	1.07	0.375	0.37	
	0.83	0.354	0.502	
	0.65	0.327	0.446	
	2.1	0.47	0.742	
	2.61	0.507	0.812	
	1.38	0.408	0.628	
	0.8	0.348	0.488	
	0.86	0.354	0.502	
	0.45	0.294	0.376	

11.10 Appendix J- Fuzzy sets for Mayfly

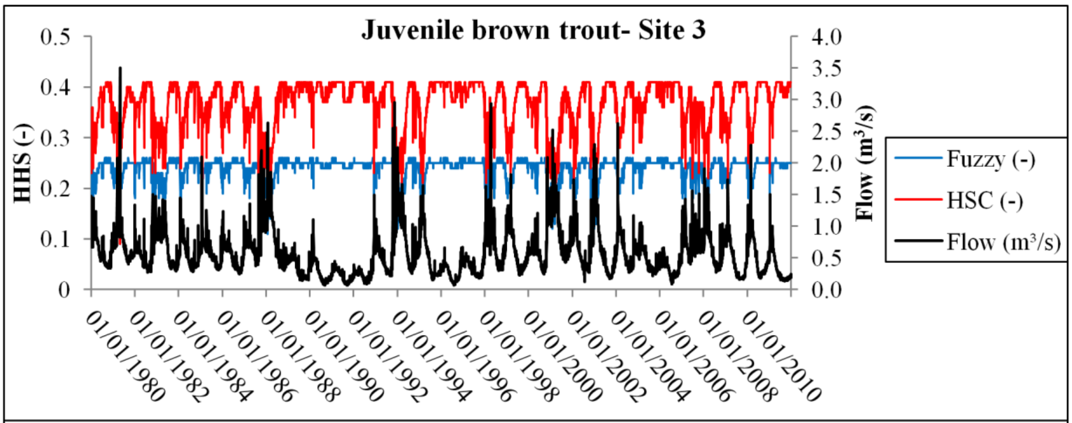


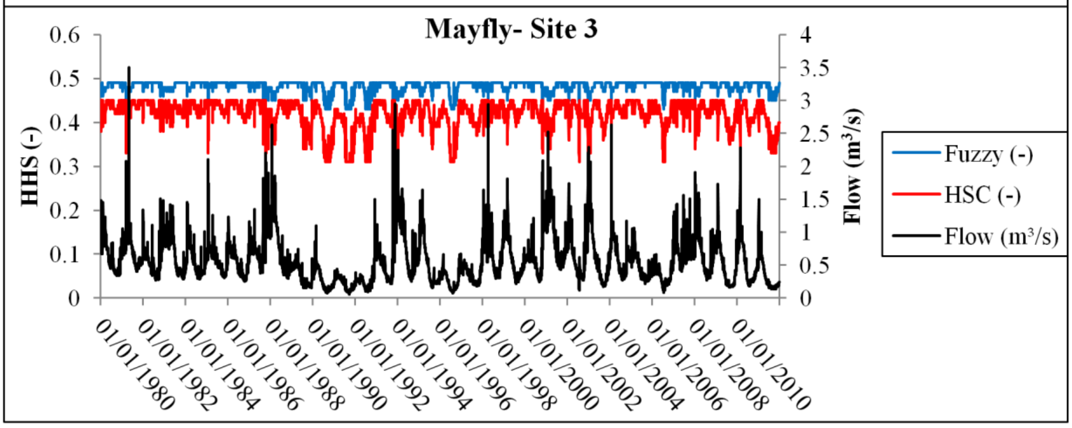


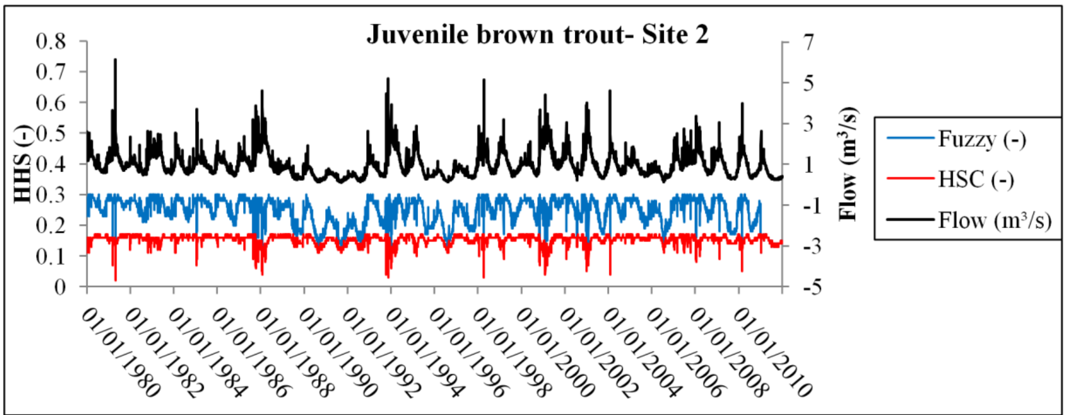


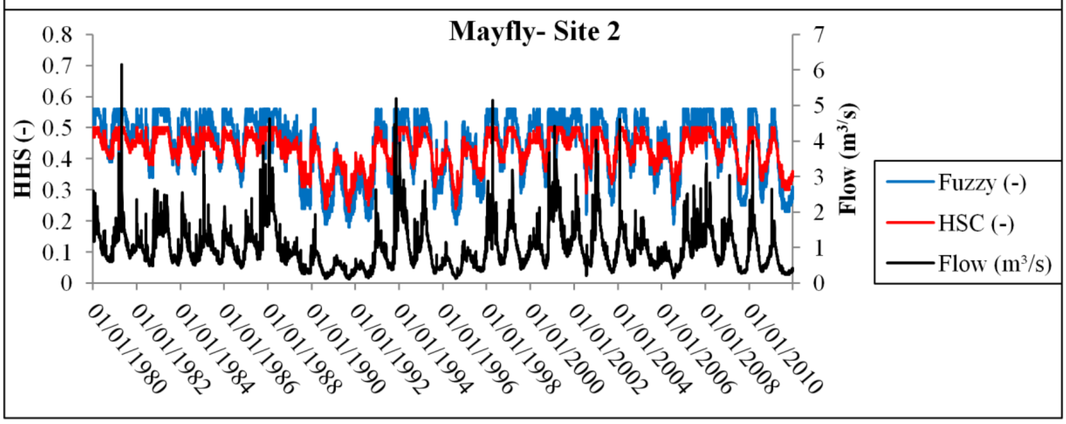


11.11 Appendix K- Analysis 1 Fuzzy V HSC extra graphs for site 2 and 3









11.12 Appendix L- Mann Whitney results for adult brown trout site 1 (analysis 4- RQ2)

This appendix presents an example of the results from analysis 4 of research question 2.

1= Highly unsuitable, 2= Unsuitable, 3= Moderate, 4= Suitable, 5= Highly suitable

Blue= Wet year, Yellow= Dry year, Orange= Average year, Green= Statistically similar

Winter

		1	2	3	4	5
1988	1994	0.001	0.002	0.007	0.000	0.000
1988	1995	0.770	0.154	0.000	0.000	0.001
1988	2001	0.002	0.005	0.055	0.008	0.001
1988	2003	0.793	0.696	0.005	0.000	0.145
1994	1995	0.008	0.001	0.000	0.001	0.524
1994	2001	0.976	0.994	0.238	0.165	0.587
1994	2003	0.034	0.071	0.992	0.909	0.045
1995	2001	0.013	0.000	0.000	0.000	0.929
1995	2003	0.704	0.043	0.000	0.001	0.231
2001	2003	0.032	0.074	0.282	0.214	0.152
1986	1987	0.021	0.041	0.041	0.044	0.570
1986	1993	0.000	0.000	0.000	0.106	0.007
1986	1998	0.584	0.053	0.051	0.132	0.333
1986	2005	0.000	0.001	0.001	0.955	0.780
1987	1993	0.000	0.000	0.000	0.000	0.001
1987	1998	0.458	0.465	0.472	0.918	0.702
1987	2005	0.000	0.000	0.000	0.059	0.747
1993	1998	0.000	0.000	0.000	0.002	0.000
1993	2005	0.817	0.146	0.152	0.082	0.007
1998	2005	0.000	0.000	0.000	0.291	0.488
1990	1991	0.000	0.000	0.000	0.000	0.083
1990	1992	0.000	0.000	0.000	0.000	0.083
1990	1996	0.044	0.000	0.000	0.000	0.083

		1	2	3	4	5
1988	1990	0.192	0.000	0.000	0.000	0.000
1988	1991	0.000	0.000	0.000	0.000	0.000
1988	1992	0.000	0.000	0.000	0.000	0.000
1988	1996	0.000	0.000	0.000	0.000	0.000
1988	2006	0.404	0.000	0.000	0.000	0.000
1994	1996	0.000	0.000	0.000	0.000	0.000
1994	2006	0.000	0.000	0.000	0.000	0.000
1995	1996	0.000	0.000	0.000	0.000	0.000
1995	2006	0.074	0.000	0.000	0.000	0.000
2001	2006	0.000	0.000	0.000	0.000	0.000
2003	2006	0.484	0.000	0.000	0.000	0.000
1994	1990	0.000	0.000	0.000	0.000	0.000
1995	1990	0.052	0.000	0.000	0.003	0.000
2001	1990	0.000	0.000	0.000	0.000	0.000
2003	1990	0.276	0.000	0.000	0.000	0.000
1994	1991	0.000	0.000	0.000	0.000	0.000
1995	1991	0.000	0.000	0.000	0.000	0.000
2001	1991	0.000	0.000	0.000	0.000	0.000
2003	1991	0.003	0.000	0.000	0.000	0.000
1994	1992	0.000	0.000	0.000	0.000	0.000
1995	1992	0.000	0.000	0.000	0.000	0.000
2001	1992	0.000	0.000	0.000	0.000	0.000
2003	1992	0.000	0.000	0.000	0.000	0.000

		1	2	3	4	5
1990	1993	0.000	0.000	0.000	0.000	0.029
1990	1998	0.000	0.000	0.000	0.000	0.000
1990	2005	0.000	0.000	0.000	0.000	0.000
1991	1993	0.000	0.000	0.000	0.000	0.001
1991	1998	0.000	0.000	0.000	0.000	0.000
1991	2005	0.000	0.000	0.000	0.000	0.000
1992	1993	0.083	0.000	0.000	0.000	0.001
1992	1998	0.000	0.000	0.000	0.000	0.000
1992	2005	0.156	0.000	0.000	0.000	0.000
1996	1998	0.000	0.000	0.000	0.000	0.000
1996	2005	0.000	0.000	0.000	0.000	0.000
1990	1986	0.000	0.000	0.000	0.000	0.000
1991	1986	0.000	0.000	0.000	0.000	0.000
1992	1986	0.000	0.000	0.000	0.000	0.000
1996	1986	0.000	0.000	0.000	0.000	0.000
2006	1986	0.000	0.000	0.000	0.000	0.000
1990	1987	0.000	0.000	0.000	0.000	0.000
1991	1987	0.000	0.000	0.000	0.000	0.000
1992	1987	0.000	0.000	0.000	0.000	0.000
1996	1987	0.000	0.000	0.000	0.000	0.000
2006	1987	0.000	0.000	0.000	0.000	0.000
1996	1993	0.000	0.000	0.000	0.000	0.001
2006	1993	0.000	0.000	0.000	0.000	0.001

1990	2006	0.664	0.054	0.054	0.165	0.083
1991	1992	0.000	0.000	0.000	0.000	NA
1991	1996	0.949	0.054	0.107	0.054	NA
1991	2006	0.000	0.000	0.000	0.000	NA
1992	1996	0.006	0.000	0.000	0.000	NA
1992	2006	0.000	0.000	0.000	0.000	NA
1996	2006	0.017	0.000	0.000	0.000	NA

2001	1996	0.000	0.000	0.000	0.000	0.000
2003	1996	0.001	0.000	0.000	0.000	0.000

2006	1998	0.000	0.000	0.000	0.000	0.000
2006	2005	0.000	0.000	0.000	0.000	0.000
1994	1986	0.000	0.000	0.000	0.039	0.000
1995	1986	0.000	0.000	0.000	0.006	0.000
1988	1986	0.000	0.000	0.000	0.000	0.000
1994	1987	0.000	0.000	0.000	0.066	0.000
1995	1987	0.000	0.000	0.000	0.015	0.000
2001	1987	0.000	0.000	0.000	0.000	0.000
2003	1987	0.000	0.000	0.000	0.010	0.000
1994	1993	0.000	0.000	0.000	0.391	0.000
1995	1993	0.000	0.000	0.000	0.000	0.000
2001	1993	0.000	0.000	0.000	0.007	0.000
2003	1993	0.000	0.000	0.000	0.098	0.000
2001	1998	0.000	0.000	0.000	0.001	0.000
2003	1998	0.000	0.000	0.000	0.036	0.000
2001	1986	0.000	0.000	0.000	0.000	0.000
2003	1986	0.000	0.000	0.000	0.012	0.000
1988	1987	0.000	0.000	0.000	0.000	0.000
1988	1993	0.000	0.000	0.000	0.000	0.000
1988	1998	0.000	0.000	0.000	0.000	0.000
1988	2005	0.000	0.000	0.000	0.000	0.000
1994	1998	0.000	0.000	0.000	0.175	0.000
1994	2005	0.000	0.000	0.000	0.056	0.000
1995	1998	0.000	0.000	0.000	0.003	0.000
1995	2005	0.000	0.000	0.000	0.004	0.000
2001	2005	0.000	0.000	0.000	0.000	0.000
2003	2005	0.000	0.000	0.000	0.016	0.000

Autumn

		1	2	3	4	5
1987	1993	0.018	0.000	0.000	0.145	0.000
1987	1998	0.674	0.000	0.000	0.007	0.000
1987	2000	0.000	0.249	0.879	0.001	0.478
1987	2002	0.000	0.000	0.000	0.743	0.000
1993	1998	0.067	0.000	0.000	0.000	0.000
1993	2000	0.000	0.015	0.378	0.007	0.000
1993	2002	0.004	0.001	0.002	0.006	0.014
1998	2000	0.000	0.000	0.000	0.140	0.000
1998	2002	0.000	0.002	0.001	0.175	0.008
2000	2002	0.000	0.000	0.000	0.303	0.000
1983	1985	0.018	0.329	0.329	0.778	0.044
1983	1988	0.026	0.000	0.000	0.000	NA
1983	1999	0.423	0.000	0.000	0.000	0.158
1983	2010	0.158	0.606	0.641	0.607	NA
1985	1988	0.000	0.006	0.006	0.000	0.044
1985	1999	0.000	0.001	0.001	0.000	0.390
1985	2010	0.275	0.050	0.050	0.192	0.044
1988	1999	0.102	0.130	0.130	0.117	0.158
1988	2010	0.001	0.004	0.004	0.004	NA
1999	2010	0.049	0.000	0.000	0.000	0.158
1989	1990	0.000	0.000	0.000	0.000	NA
1989	1991	0.000	0.000	0.000	0.000	NA
1989	2009	0.000	0.000	0.102	0.000	0.044
1989	2011	0.000	0.000	0.001	0.000	NA
1990	1991	0.031	0.031	0.031	0.031	NA
1990	2009	0.000	0.000	0.000	0.000	0.044
1990	2011	0.000	0.000	0.000	0.000	NA
1991	2009	0.000	0.000	0.000	0.000	0.044
1991	2011	0.000	0.000	0.000	0.000	NA
2009	2011	0.000	0.000	0.000	0.000	0.044

		1	2	3	4	5
2002	2011	0.000	0.000	0.000	0.000	0.000
1987	1989	0.000	0.000	0.000	0.000	0.000
1987	1990	0.000	0.000	0.000	0.000	0.000
1987	1991	0.000	0.000	0.000	0.000	0.000
1987	2009	0.000	0.000	0.000	0.000	0.000
1987	2011	0.000	0.000	0.000	0.000	0.000
1993	2009	0.000	0.000	0.000	0.000	0.000
1993	2011	0.000	0.000	0.000	0.000	0.000
1998	2009	0.000	0.000	0.000	0.000	0.000
1998	2011	0.000	0.000	0.297	0.000	0.000
2000	2009	0.078	0.000	0.000	0.000	0.000
2000	2011	0.011	0.000	0.000	0.000	0.000
2002	2009	0.000	0.000	0.000	0.000	0.000
1993	1989	0.000	0.000	0.000	0.000	0.000
1998	1989	0.000	0.000	0.000	0.000	0.000
2000	1989	0.309	0.000	0.000	0.000	0.000
2002	1989	0.000	0.000	0.000	0.000	0.000
1993	1990	0.000	0.000	0.000	0.000	0.000
1998	1990	0.000	0.000	0.000	0.000	0.000
2000	1990	0.000	0.000	0.000	0.000	0.000
2002	1990	0.000	0.000	0.006	0.000	0.000
1993	1991	0.000	0.000	0.000	0.000	0.000
1998	1991	0.000	0.000	0.007	0.000	0.000
2000	1991	0.000	0.000	0.000	0.000	0.000
2002	1991	0.000	0.000	0.227	0.000	0.000

		1	2	3	4	5
1987	1988	0.008	0.000	0.000	0.000	0.000
1987	1999	0.191	0.000	0.000	0.000	0.000
1987	2010	0.459	0.000	0.000	0.000	0.000
1993	1999	0.000	0.000	0.000	0.000	0.000
1993	2010	0.068	0.000	0.000	0.000	0.000
1998	1999	0.004	0.000	0.000	0.384	0.000
1998	2010	0.252	0.000	0.000	0.000	0.000
2000	2010	0.000	0.000	0.000	0.000	0.000
2002	2010	0.000	0.000	0.000	0.000	0.000
1987	1983	0.837	0.000	0.000	0.000	0.000
1993	1983	0.006	0.000	0.000	0.000	0.000
1998	1983	0.076	0.000	0.000	0.000	0.000
2000	1983	0.000	0.000	0.000	0.000	0.000
2002	1983	0.000	0.000	0.000	0.000	0.000
1987	1985	0.135	0.000	0.000	0.000	0.000
1993	1985	0.156	0.000	0.000	0.000	0.000
1998	1985	0.265	0.000	0.000	0.000	0.001
2000	1985	0.000	0.000	0.000	0.000	0.000
2002	1985	0.000	0.000	0.000	0.000	0.000
1993	1988	0.000	0.000	0.000	0.000	0.000
1998	1988	0.000	0.000	0.000	0.011	0.000
2000	1988	0.001	0.000	0.000	0.000	0.000
2002	1988	0.000	0.000	0.000	0.000	0.000
2000	1999	0.000	0.000	0.000	0.007	0.000
2002	1999	0.000	0.000	0.000	0.005	0.000
2009	1999	0.000	0.000	0.000	0.000	0.398
2011	1999	0.000	0.000	0.019	0.000	0.158
2011	2010	0.000	0.000	0.000	0.000	NA
1989	1983	0.000	0.000	0.848	0.000	NA
1990	1983	0.000	0.000	0.000	0.000	NA
1991	1983	0.000	0.000	0.000	0.000	NA
1989	1985	0.000	0.000	0.047	0.000	0.044
1990	1985	0.000	0.000	0.000	0.000	0.044
1991	1985	0.000	0.000	0.000	0.000	0.044

2009	1985	0.000	0.000	0.003	0.000	0.972
2011	1985	0.000	0.000	0.004	0.000	0.044
1989	1988	0.000	0.000	0.006	0.000	NA
1990	1988	0.000	0.000	0.000	0.000	NA
1991	1988	0.000	0.000	0.000	0.000	NA
2009	1988	0.006	0.000	0.000	0.000	0.044
2011	1988	0.000	0.000	0.001	0.000	NA
2009	1983	0.000	0.000	0.127	0.000	0.044
2011	1983	0.000	0.000	0.000	0.000	NA
1991	2010	0.000	0.000	0.000	0.000	NA
2009	2010	0.000	0.000	0.477	0.000	0.044
1989	1999	0.000	0.000	0.000	0.000	0.158
1989	2010	0.000	0.000	0.044	0.000	NA
1990	1999	0.000	0.000	0.000	0.000	0.158
1990	2010	0.000	0.000	0.000	0.000	NA
1991	1999	0.000	0.000	0.000	0.000	0.158

Spring

		1	2	3	4	5
1981	1988	0.022	0.096	0.075	0.001	0.231
1981	1994	0.253	0.000	0.000	0.000	0.000
1981	1998	0.175	0.001	0.002	0.002	0.000
1981	2001	0.019	0.173	0.034	0.000	0.413
1988	1994	0.197	0.073	0.102	0.270	0.032
1988	1998	0.001	0.468	0.766	0.462	0.058
1988	2001	0.732	0.010	0.000	0.001	0.654
1994	1998	0.020	0.208	0.151	0.945	0.740
1994	2001	0.122	0.000	0.000	0.009	0.004
1998	2001	0.001	0.000	0.000	0.009	0.008
1984	2002	0.058	0.000	0.000	0.066	0.001
1984	2003	0.178	0.294	0.294	0.135	0.763
1984	2007	0.009	0.085	0.085	0.323	0.493
1984	2010	0.527	0.227	0.227	0.000	0.000
2002	2003	0.556	0.017	0.017	0.716	0.003
2002	2007	0.500	0.064	0.064	0.366	0.009
2002	2010	0.005	0.000	0.000	0.001	0.000
2003	2007	0.173	0.561	0.560	0.665	0.792
2003	2010	0.052	0.076	0.076	0.000	0.000
2007	2010	0.000	0.077	0.076	0.000	0.000
1990	1991	0.000	0.000	0.000	0.000	NA
1990	1992	0.885	0.000	0.047	0.000	NA
1990	1996	0.018	0.000	0.857	0.000	NA
1990	2011	0.327	0.078	0.139	0.078	NA
1991	1992	0.011	0.000	0.202	0.000	NA
1991	1996	0.000	0.839	0.000	0.839	NA
1991	2011	0.000	0.004	0.000	0.004	NA
1992	1996	0.175	0.004	0.017	0.004	NA
1992	2011	0.843	0.000	0.240	0.000	NA
1996	2011	0.004	0.001	0.134	0.001	NA

		1	2	3	4	5
2001	2011	0.336	0.000	0.000	0.000	0.000
1981	1990	0.000	0.000	0.000	0.000	0.000
1981	1991	0.101	0.000	0.000	0.000	0.000
1981	1992	0.001	0.000	0.000	0.000	0.000
1981	1996	0.000	0.000	0.000	0.000	0.000
1981	2011	0.000	0.000	0.000	0.000	0.000
1988	1990	0.540	0.000	0.000	0.000	0.000
1988	1991	0.002	0.000	0.000	0.000	0.000
1988	1992	0.846	0.000	0.000	0.000	0.000
1988	1996	0.143	0.000	0.000	0.000	0.000
1988	2011	0.730	0.000	0.000	0.000	0.000
1994	1996	0.000	0.000	0.000	0.000	0.000
1998	2011	0.000	0.000	0.000	0.000	0.000
1994	2011	0.021	0.000	0.000	0.000	0.000
1994	1990	0.003	0.000	0.000	0.000	0.000
1998	1990	0.000	0.000	0.000	0.000	0.000
2001	1990	0.572	0.000	0.000	0.000	0.000
1994	1991	0.667	0.000	0.000	0.000	0.000
1998	1991	0.001	0.000	0.000	0.000	0.000
2001	1991	0.013	0.000	0.000	0.000	0.000
1994	1992	0.032	0.000	0.000	0.000	0.000
1998	1992	0.000	0.000	0.000	0.000	0.000
2001	1992	0.861	0.000	0.000	0.000	0.000
1998	1996	0.000	0.000	0.000	0.000	0.000
2001	1996	0.374	0.000	0.000	0.000	0.000

		1	2	3	4	5
1981	1984	0.748	0.000	0.000	0.000	0.000
1981	2002	0.044	0.000	0.000	0.000	0.000
1981	2003	0.249	0.000	0.000	0.000	0.000
1981	2007	0.003	0.000	0.000	0.000	0.000
1981	2010	0.685	0.000	0.000	0.040	0.000
1988	2002	0.577	0.000	0.000	0.003	0.000
1988	2003	0.277	0.000	0.000	0.000	0.000
1988	2007	0.883	0.000	0.000	0.000	0.000
1988	2010	0.006	0.000	0.000	0.395	0.000
1994	2002	0.170	0.000	0.000	0.106	0.000
1994	2003	0.639	0.000	0.000	0.037	0.000
1994	2007	0.037	0.000	0.000	0.009	0.000
1994	2010	0.157	0.000	0.000	0.079	0.000
1998	2002	0.001	0.000	0.000	0.128	0.000
1998	2003	0.015	0.000	0.000	0.064	0.000
1998	2007	0.000	0.000	0.000	0.013	0.000
1998	2010	0.391	0.000	0.000	0.101	0.000
2001	2002	0.093	0.000	0.000	0.083	0.000
2001	2003	0.025	0.000	0.000	0.168	0.000
2001	2007	0.249	0.000	0.000	0.396	0.000
2001	2010	0.001	0.000	0.000	0.000	0.000
1988	1984	0.026	0.000	0.000	0.000	0.000
1994	1984	0.560	0.000	0.000	0.001	0.000
1998	1984	0.126	0.000	0.000	0.003	0.000
2001	1984	0.003	0.000	0.000	0.682	0.000
2011	1984	0.035	0.000	0.000	0.000	0.000
2011	2002	0.729	0.000	0.000	0.000	0.323
2011	2003	0.195	0.000	0.000	0.000	0.001
2011	2007	0.758	0.000	0.000	0.000	0.002
2011	2010	0.000	0.000	0.000	0.000	0.000
1990	1984	0.001	0.000	0.000	0.000	0.000
1991	1984	0.101	0.000	0.000	0.000	0.000
1992	1984	0.009	0.000	0.000	0.000	0.000
1996	1984	0.000	0.000	0.000	0.000	0.000

1996	2002	0.000	0.000	0.000	0.000	0.323
1996	2003	0.000	0.000	0.000	0.000	0.001
1996	2007	0.003	0.000	0.000	0.000	0.002
1996	2010	0.000	0.000	0.000	0.000	0.000
1990	2002	0.076	0.000	0.000	0.000	0.323
1990	2003	0.011	0.000	0.000	0.000	0.001
1990	2007	0.258	0.000	0.000	0.000	0.002
1990	2010	0.000	0.000	0.000	0.000	0.000
1991	2002	0.000	0.000	0.000	0.000	0.323
1991	2003	0.020	0.000	0.000	0.000	0.001
1991	2007	0.000	0.000	0.000	0.000	0.002
1991	2010	0.448	0.000	0.000	0.000	0.000
1992	2002	0.280	0.000	0.000	0.000	0.323
1992	2003	0.041	0.000	0.000	0.000	0.001
1992	2007	0.575	0.000	0.000	0.000	0.002
1992	2010	0.000	0.000	0.000	0.000	0.000

Summer

		1	2	3	4	5
1980	1981	0.000	0.000	0.000	0.000	0.158
1980	1987	0.000	0.000	0.000	0.000	0.254
1980	2001	0.000	0.000	0.000	0.000	0.247
1980	2007	0.000	0.000	0.000	0.000	0.000
1981	1987	0.317	0.667	0.667	0.666	0.024
1981	2001	0.873	0.718	0.718	0.694	0.024
1981	2007	0.000	0.000	0.000	0.048	0.000
1987	2001	0.224	0.405	0.405	0.834	1.000
1987	2007	0.000	0.000	0.000	0.078	0.000
2001	2007	0.000	0.000	0.000	0.006	0.000
1982	1994	0.449	0.119	0.104	0.136	0.158
1982	2000	0.388	0.117	0.063	0.117	NA
1982	2004	0.027	0.044	0.043	0.044	NA
1982	2008	0.002	0.007	0.007	0.007	NA
1994	2000	0.889	0.950	0.762	0.804	0.158
1994	2004	0.166	0.878	0.898	0.805	0.158
1994	2008	0.068	0.821	0.830	0.787	0.158
2000	2004	0.095	0.779	0.677	0.779	NA
2000	2008	0.047	0.996	0.958	0.996	NA
2004	2008	0.528	0.806	0.806	0.806	NA
1990	1991	0.195	0.195	0.312	0.245	NA
1990	1992	0.001	0.001	0.002	0.001	NA
1990	1996	0.847	0.847	0.691	0.767	NA
1990	2011	0.000	0.000	0.000	0.000	NA
1991	1992	0.000	0.000	0.000	0.000	NA
1991	1996	0.111	0.111	0.096	0.102	NA
1991	2011	0.000	0.000	0.000	0.000	NA
1992	1996	0.000	0.000	0.000	0.000	NA
1992	2011	0.000	0.000	0.000	0.000	NA
1996	2011	0.000	0.000	0.000	0.000	NA

		1	2	3	4	5
2007	2011	0.000	0.000	0.000	0.000	0.000
1980	1990	0.000	0.000	0.000	0.000	0.158
1980	1991	0.000	0.000	0.000	0.000	0.158
1980	1992	0.000	0.000	0.000	0.000	0.158
1987	1996	0.000	0.000	0.000	0.000	0.024
1987	2011	0.000	0.000	0.021	0.000	0.024
1980	1996	0.000	0.000	0.000	0.000	0.158
1981	2011	0.000	0.000	0.008	0.000	NA
1987	1990	0.000	0.000	0.000	0.000	0.024
1987	1991	0.000	0.000	0.000	0.000	0.024
1987	1992	0.000	0.000	0.000	0.000	0.024
2001	2011	0.000	0.000	0.024	0.000	0.024
1981	1996	0.000	0.000	0.000	0.000	NA
1980	2011	0.000	0.000	0.000	0.000	0.158
1981	1990	0.000	0.000	0.000	0.000	NA
1981	1991	0.000	0.000	0.000	0.000	NA
1981	1992	0.000	0.000	0.000	0.000	NA
2001	1990	0.000	0.000	0.000	0.000	0.024
2007	1990	0.000	0.000	0.000	0.000	0.000
2001	1991	0.000	0.000	0.000	0.000	0.024
2007	1991	0.000	0.000	0.000	0.000	0.000
2001	1992	0.000	0.000	0.000	0.000	0.024
2007	1992	0.000	0.000	0.182	0.000	0.000
2001	1996	0.000	0.000	0.000	0.000	0.024
2007	1996	0.000	0.000	0.000	0.000	0.000

		1	2	3	4	5
1996	2008	0.000	0.000	0.000	0.000	NA
1990	1994	0.000	0.000	0.000	0.000	0.158
1990	2000	0.000	0.000	0.000	0.000	NA
1990	2004	0.000	0.000	0.000	0.000	NA
1990	2008	0.000	0.000	0.000	0.000	NA
1991	1994	0.000	0.000	0.000	0.000	0.158
1991	2000	0.000	0.000	0.000	0.000	NA
1991	2004	0.000	0.000	0.000	0.000	NA
1991	2008	0.000	0.000	0.000	0.000	NA
1992	1994	0.000	0.000	0.000	0.000	0.158
1992	2000	0.000	0.000	0.000	0.000	NA
1992	2004	0.000	0.000	0.000	0.000	NA
1992	2008	0.000	0.000	0.000	0.000	NA
1996	2000	0.000	0.000	0.000	0.000	NA
1996	2004	0.000	0.000	0.000	0.000	NA
2011	2004	0.000	0.000	0.000	0.000	NA
2011	2008	0.000	0.000	0.000	0.000	NA
1990	1982	0.000	0.000	0.000	0.000	NA
1991	1982	0.000	0.000	0.000	0.000	NA
1992	1982	0.000	0.000	0.000	0.000	NA
1996	1982	0.000	0.000	0.000	0.000	NA
2011	1982	0.000	0.000	0.000	0.000	NA
1996	1994	0.000	0.000	0.000	0.000	0.158
2011	1994	0.000	0.000	0.000	0.000	0.158
2011	2000	0.000	0.000	0.000	0.000	NA
1980	1982	0.000	0.000	0.000	0.000	0.158
1980	1994	0.000	0.000	0.000	0.000	0.987
1980	2000	0.000	0.000	0.000	0.000	0.158
1980	2004	0.000	0.000	0.000	0.000	0.158
1980	2008	0.000	0.000	0.000	0.000	0.158
1981	1982	0.086	0.000	0.000	0.000	NA
1981	1994	0.601	0.000	0.000	0.000	0.158
1981	2000	0.171	0.000	0.000	0.000	NA
1981	2004	0.513	0.000	0.000	0.000	NA

1981	2008	0.985	0.000	0.000	0.000	NA
1987	1994	0.094	0.000	0.000	0.000	0.259
1987	2000	0.010	0.000	0.000	0.000	0.024
1987	2004	0.627	0.000	0.000	0.000	0.024
1987	2008	0.289	0.000	0.000	0.000	0.024
2001	2004	0.687	0.000	0.000	0.000	0.024
2001	2008	0.793	0.000	0.000	0.000	0.024
2007	2008	0.000	0.000	0.000	0.000	0.000
1987	1982	0.004	0.000	0.000	0.000	0.024
2001	1982	0.030	0.000	0.000	0.000	0.024
2007	1982	0.000	0.000	0.000	0.000	0.000
2001	1994	0.395	0.000	0.000	0.000	0.261
2007	1994	0.000	0.000	0.000	0.000	0.000
2001	2000	0.097	0.000	0.000	0.000	0.024
2007	2000	0.000	0.000	0.000	0.000	0.000
2007	2004	0.000	0.000	0.000	0.000	0.000

11.13 Appendix M- Interconnectedness results

Site 2

Dry years																												
				1990					1991					1992					1996					2011				
Sc	**Sp**	**Ref**	**Food**	**Total**	**W**	**Sp**	**Su**	**A**	**Total**	**W**	**Sp**	**Su**	**A**	**Total**	**W**	**Sp**	**Su**	**A**	**Total**	**W**	**Sp**	**Su**	**A**	**Total**	**W**	**Sp**	**Su**	**A**
1	H	H	H	158	90	68			130	83	46	1		135	31	33	2	69	187	91	46		50	147	90	57		
2	H	H	M	5		5			7	1	6			36	19	12	1	4	8		7		1	4		4		
5	H	M	M	26		13	5	8	60	6	39	4	11	64	35	20	6	3	33		14	3	16	60		30	9	21
14	M	M	M	51		5	15	31	35			10	25	39	6	11	14	8	30		8	7	15	102			46	56
15	M	M	L	56			13	43	37			14	23	43		8	27	8	47		16	21	10	52			37	15
18	M	L	L	69			59	10	96			63	33	49		7	42		61			61						

Average years																												
				1982					1984					1986					2005					2010				
Sc	**Sp**	**Ref**	**Food**	**Total**	**W**	**Sp**	**Su**	**A**	**Total**	**W**	**Sp**	**Su**	**A**	**Total**	**W**	**Sp**	**Su**	**A**	**Total**	**W**	**Sp**	**Su**	**A**	**Total**	**W**	**Sp**	**Su**	**A**
1	H	H	H	365	90	91	92	92	366	91	91	92	92	360	90	91	92	87	338	90	91	72	85	282	88	91	25	78
2	H	H	M											4				4	17			12	5	19			11	8
5	H	M	M											1				1	19			8	2	47			41	6
10	M	H	H																					1	1			
11	M	H	M																					1	1			
13	M	M	H																					15			15	

Wet years																												
				1988					**1994**					**2000**					**2001**					**2003**				
Sc	**BT**	**Ref**	**Food**	**Total**	**W**	**Sp**	**Su**	**A**	**Total**	**W**	**Sp**	**Su**	**A**	**Total**	**W**	**Sp**	**Su**	**A**	**Total**	**W**	**Sp**	**Su**	**A**	**Total**	**W**	**Sp**	**Su**	**A**
1	H	H	H	358	83	91	92	92	360	86	91	91	92	354	91	91	82	90	360	85	91	92	92	282	83	91	48	60
2	H	H	M						1			1		7			7							4			2	2
5	H	M	M											3			3							58			36	22
10	M	H	H	3	3				2	2				1				1	2	2				2	2			
11	M	H	M	3	3				2	2				1				1	2	2				5	5			
14	M	M	M																					14			6	8
18	M	L	L	2	2																							
20	L	H	M																1	1								

Site 3

Dry years																												
				1991					**2011**					**1990**					**1992**					**1996**				
Sc	**Sp**	**Ref**	**Food**	**Total**	**W**	**Sp**	**Su**	**A**	**Total**	**W**	**Sp**	**Su**	**A**	**Total**	**W**	**Sp**	**Su**	**A**	**Total**	**W**	**Sp**	**Su**	**A**	**Total**	**W**	**Sp**	**Su**	**A**
1	H	H	H	59	49	10			120	87	33			126	82	44	0	0	55	6	2		47	125	55	25		45
2	H	H	M	151	41	81	9	20	100		58	15	27	66	7	44	6	9	195	85	67	11	32	112	36	45	4	27
4	H	M	H											1	1													
6	H	M	L	57			18	39	142			77	65	103	0	3	27	73	63		15	37	11	67		21	26	20
9	H	L	L	98			65	33						69	0	0	59	10	51		7	44		62			62	
13	M	M	H						3	3									2				2					
23	L	M	M																				1					

Average years																												
				1982					1984					1986					2005					2010				
	Sp	**Ref**	**Food**	**Total**	**W**	**Sp**	**Su**	**A**	**Total**	**W**	**Sp**	**Su**	**A**	**Total**	**W**	**Sp**	**Su**	**A**	**Total**	**W**	**Sp**	**Su**	**A**	**Total**	**W**	**Sp**	**Su**	**A**
1	H	H	H	300	89	90	38	83	318	79	91	56	92	286	88	91	61	46	230	88	84	34	24	202	66	82		54
2	H	H	M	55			54	1	36			36		76			31	45	133		7	58	68	133		9	86	38
4	H	M	H	3		1		2	3	3				1	1				2	2				13	13			
6	H	M	L																					6			6	
13	M	M	H	7	1			6	9	9				2	1			1						10	10			
23	L	M	M																					1	1			

Wet years																												
				2001					1988					1994					2003					2000				
	Sp	**Ref**	**Food**	**Total**	**W**	**Sp**	**Su**	**A**	**Total**	**W**	**Sp**	**Su**	**A**	**Total**	**W**	**Sp**	**Su**	**A**	**Total**	**W**	**Sp**	**Su**	**A**	**Total**	**W**	**Sp**	**Su**	**A**
1	H	H	H	237	11	44	92	90	252	15	63	92	82	226	8	74	52	92	177	22	91	23	41	261	91	88	50	32
2	H	H	M						10				10	40			40		114			66	48	47			42	5
4	H	M	H	17	9	7		1	12	2	10			20	18	2			1	1				21		1		20
13	M	M	H	108	67	40		1	86	68	18			77	62	15			62	62				36		2		34
22	L	M	H						1	1				0														
23	L	M	M	3	3				5	5				2	2				5	5								

11.14 Appendix N- Mann Whitney tests for extreme year analysis- site 1

		Highly unsuitable		Unsuitable		Moderate		Suitable		Highly suitable	
		S1 to S2	S1 to S3	S1 to S2	S1 to S3	S1 to S2	S1 to S3	S1 to S2	S1 to S3	S1 to S2	S1 to S3
		p-values									
Adult brown trout	1980	0.996	0.771	0.840	0.709	0.848	0.707	0.813	0.840	0.811	0.786
	1981	0.892	0.982	0.886	0.873	0.883	0.871	0.880	0.874	0.851	0.851
	1982	0.883	0.827	0.883	0.740	0.883	0.739	0.750	0.320	1.000	1.000
	1983	0.988	0.687	0.942	0.696	0.942	0.696	0.979	0.770	0.932	0.932
	1984	0.910	0.927	0.868	0.667	0.868	0.667	0.932	0.339	0.950	0.950
	1985	0.588	0.382	0.887	0.837	0.886	0.836	0.912	0.909	1.000	1.000
	1986	0.756	0.531	0.808	0.583	0.808	0.577	0.680	0.238	0.909	0.908
	1987	0.788	0.662	0.977	0.968	0.977	0.968	0.971	0.902	1.000	1.000
	1988	0.994	0.709	0.887	0.681	0.897	0.690	0.709	0.940	0.530	0.527
	1989	0.727	0.289	0.830	0.351	0.870	0.804	0.779	0.303	1.000	1.000
	1990	0.541	0.312	0.756	0.536	0.464	0.022	0.757	0.537	1.000	1.000
	1991	0.902	0.708	0.698	0.523	0.312	0.152	0.697	0.523	NA	NA
	1992	0.887	0.676	0.824	0.681	0.765	0.325	0.818	0.674	1.000	1.000
	1993	0.872	0.672	0.934	0.910	0.934	0.910	0.921	0.881	1.000	1.000
	1994	0.743	0.527	0.899	0.748	0.899	0.748	0.922	0.630	1.000	1.000
	1995	0.832	0.886	0.872	0.597	0.854	0.757	0.824	0.498	1.000	1.000
	1996	0.749	0.658	0.874	0.429	0.918	0.044	0.870	0.425	1.000	1.000
	1997	0.614	0.779	0.765	0.563	0.756	0.654	0.757	0.539	1.000	1.000
	1998	0.878	0.631	0.920	0.807	0.924	0.811	0.935	0.773	0.966	0.966
	1999	0.866	0.713	0.879	0.786	0.880	0.786	0.918	0.779	0.979	0.979
	2000	0.723	0.450	0.937	0.773	0.936	0.771	0.934	0.555	0.951	0.951
	2001	0.960	0.793	0.922	0.903	0.914	0.895	0.923	0.910	0.859	0.859
	2002	0.856	0.751	0.886	0.841	0.886	0.988	0.781	0.564	1.000	1.000
	2003	0.946	0.642	0.895	0.695	0.895	0.695	0.894	0.604	0.950	0.950
	2004	0.729	0.662	0.882	0.778	0.881	0.777	0.870	0.469	0.979	0.979
	2005	0.726	0.433	0.813	0.451	0.810	0.449	0.862	0.225	0.995	0.995
	2006	0.687	0.361	0.878	0.786	0.723	0.059	0.875	0.780	1.000	1.000
	2007	0.781	0.764	0.959	0.958	0.959	0.957	0.970	0.970	1.000	1.000
	2008	0.731	0.627	0.857	0.776	0.857	0.776	0.880	0.611	0.969	0.969
	2009	0.982	0.769	0.816	0.436	0.895	0.636	0.780	0.395	0.959	0.959
	2010	0.800	0.652	0.899	0.738	0.914	0.819	0.927	0.705	0.875	0.875
	2011	0.955	0.813	0.846	0.532	0.916	0.823	0.723	0.408	1.000	1.000
Juvenile brown trout	1980	0.874	0.757	0.851	0.709	N/A	N/A	N/A	N/A	N/A	N/A
	1981	0.910	0.966	0.886	0.873	N/A	N/A	N/A	N/A	N/A	N/A
	1982	0.914	0.840	0.883	0.740	N/A	N/A	N/A	N/A	N/A	N/A
	1983	0.895	0.586	0.942	0.696	N/A	N/A	N/A	N/A	N/A	N/A
	1984	0.915	0.998	0.868	0.667	N/A	N/A	N/A	N/A	N/A	N/A
	1985	0.664	0.526	0.887	0.837	N/A	N/A	N/A	N/A	N/A	N/A
	1986	0.805	0.583	0.808	0.583	N/A	N/A	N/A	N/A	N/A	N/A
	1987	0.871	0.859	0.977	0.968	N/A	N/A	N/A	N/A	N/A	N/A
	1988	0.961	0.813	0.895	0.688	N/A	N/A	N/A	N/A	N/A	N/A
	1989	0.738	0.432	0.830	0.351	N/A	N/A	N/A	N/A	N/A	N/A
	1990	0.617	0.813	0.756	0.536	N/A	N/A	N/A	N/A	N/A	N/A
	1991	0.439	0.120	0.698	0.523	N/A	N/A	N/A	N/A	N/A	N/A
	1992	0.984	0.375	0.824	0.681	N/A	N/A	N/A	N/A	N/A	N/A
	1993	0.937	0.640	0.934	0.910	N/A	N/A	N/A	N/A	N/A	N/A
	1994	0.613	0.436	0.899	0.748	N/A	N/A	N/A	N/A	N/A	N/A
	1995	0.906	0.911	0.872	0.597	N/A	N/A	N/A	N/A	N/A	N/A
	1996	0.551	0.057	0.874	0.429	N/A	N/A	N/A	N/A	N/A	N/A
	1997	0.567	0.790	0.765	0.563	N/A	N/A	N/A	N/A	N/A	N/A
	1998	0.817	0.576	0.920	0.807	N/A	N/A	N/A	N/A	N/A	N/A

	1999	0.861	0.679	0.879	0.786	N/A	N/A	N/A	N/A	N/A	N/A
	2000	0.507	0.363	0.937	0.773	N/A	N/A	N/A	N/A	N/A	N/A
	2001	0.962	0.614	0.923	0.904	N/A	N/A	N/A	N/A	N/A	N/A
	2002	0.906	0.902	0.886	0.841	N/A	N/A	N/A	N/A	N/A	N/A
	2003	0.953	0.604	0.895	0.695	N/A	N/A	N/A	N/A	N/A	N/A
	2004	0.848	0.838	0.882	0.778	N/A	N/A	N/A	N/A	N/A	N/A
	2005	0.715	0.473	0.813	0.451	N/A	N/A	N/A	N/A	N/A	N/A
	2006	0.703	0.917	0.878	0.786	N/A	N/A	N/A	N/A	N/A	N/A
	2007	0.804	0.801	0.959	0.958	N/A	N/A	N/A	N/A	N/A	N/A
	2008	0.867	0.808	0.857	0.776	N/A	N/A	N/A	N/A	N/A	N/A
	2009	0.936	0.712	0.816	0.436	N/A	N/A	N/A	N/A	N/A	N/A
	2010	0.932	0.836	0.899	0.738	N/A	N/A	N/A	N/A	N/A	N/A
	2011	0.987	0.946	0.846	0.532	N/A	N/A	N/A	N/A	N/A	N/A
Spawning brown trout	**1980**	0.987	0.802	0.953	0.866	0.806	0.718	0.965	0.728	N/A	N/A
	1981	0.884	0.892	0.793	0.771	0.792	0.755	0.880	0.921	N/A	N/A
	1982	0.951	0.703	0.899	0.553	0.854	0.122	0.791	0.583	N/A	N/A
	1983	0.985	0.866	0.929	0.933	0.946	0.933	0.986	0.649	N/A	N/A
	1984	0.821	0.652	0.909	0.660	0.909	0.394	0.789	0.405	N/A	N/A
	1985	0.811	0.797	0.850	0.658	0.824	0.563	0.816	0.675	N/A	N/A
	1986	0.739	0.489	0.934	0.826	0.948	0.694	0.661	0.310	N/A	N/A
	1987	0.991	0.968	0.967	0.958	0.915	0.899	0.876	0.804	N/A	N/A
	1988	0.736	0.609	0.871	0.831	0.763	0.756	0.979	0.739	N/A	N/A
	1989	0.985	0.343	0.959	0.874	0.877	0.357	0.808	0.332	N/A	N/A
	1990	0.783	0.560	0.827	0.190	0.767	0.546	0.754	0.534	N/A	N/A
	1991	0.479	0.338	0.507	0.283	0.696	0.521	0.698	0.523	N/A	N/A
	1992	0.783	0.641	0.635	0.302	0.812	0.669	0.811	0.666	N/A	N/A
	1993	0.819	0.683	0.897	0.788	0.827	0.682	0.900	0.860	N/A	N/A
	1994	0.964	0.886	0.980	0.968	0.947	0.798	0.822	0.587	N/A	N/A
	1995	0.906	0.566	0.892	0.728	0.771	0.286	0.758	0.387	N/A	N/A
	1996	0.657	0.288	0.653	0.087	0.764	0.349	0.874	0.429	N/A	N/A
	1997	0.535	0.294	0.556	0.447	0.569	0.382	0.763	0.561	N/A	N/A
	1998	0.874	0.937	0.951	0.918	0.897	0.908	0.915	0.662	N/A	N/A
	1999	0.931	0.634	0.992	0.767	0.966	0.509	0.869	0.706	N/A	N/A
	2000	0.877	0.936	0.940	0.912	0.854	0.932	0.857	0.512	N/A	N/A
	2001	0.960	0.981	0.860	0.773	0.864	0.748	0.976	0.925	N/A	N/A
	2002	0.963	0.995	0.873	0.816	0.886	0.628	0.784	0.655	N/A	N/A
	2003	0.943	0.817	0.886	0.596	0.845	0.373	0.849	0.617	N/A	N/A
	2004	0.911	0.779	0.858	0.918	0.868	0.770	0.767	0.554	N/A	N/A
	2005	0.843	0.458	0.908	0.502	0.990	0.481	0.673	0.274	N/A	N/A
	2006	0.890	0.796	0.994	0.860	0.899	0.806	0.877	0.785	N/A	N/A
	2007	0.959	0.959	0.992	0.991	0.926	0.923	0.921	0.914	N/A	N/A
	2008	0.858	0.733	0.867	0.900	0.860	0.946	0.796	0.580	N/A	N/A
	2009	0.830	0.227	0.952	0.530	0.615	0.128	0.768	0.312	N/A	N/A
	2010	0.870	0.536	0.895	0.674	0.677	0.421	0.847	0.438	N/A	N/A
	2011	0.912	0.579	0.912	0.458	0.717	0.390	0.806	0.490	N/A	N/A
Crowfoot	**1980**	0.850	0.812	0.959	0.882	0.976	0.902	0.932	0.648	N/A	N/A
	1981	0.886	0.873	0.899	0.876	0.895	0.931	0.870	0.897	N/A	N/A
	1982	0.949	0.746	0.719	0.421	0.748	0.317	0.785	0.499	N/A	N/A
	1983	0.908	0.991	0.981	0.694	1.000	0.740	0.993	0.580	N/A	N/A
	1984	0.953	0.760	0.739	0.452	0.871	0.403	0.760	0.379	N/A	N/A
	1985	0.887	0.803	0.895	0.766	0.857	0.779	0.810	0.672	N/A	N/A
	1986	0.861	0.935	0.734	0.378	0.737	0.365	0.666	0.344	N/A	N/A
	1987	0.976	0.965	0.981	0.940	0.945	0.874	0.925	0.814	N/A	N/A
	1988	0.877	0.773	0.882	0.877	0.897	0.789	0.946	0.659	N/A	N/A
	1989	0.911	0.481	0.776	0.359	0.929	0.646	0.823	0.350	N/A	N/A
	1990	0.589	0.240	0.949	0.500	0.491	0.089	0.767	0.697	N/A	N/A
	1991	0.516	0.367	0.870	0.139	0.581	0.282	0.813	0.813	N/A	N/A
	1992	0.538	0.344	0.858	0.401	0.769	0.483	0.835	0.780	N/A	N/A
	1993	0.786	0.577	0.962	0.941	0.873	0.839	0.874	0.819	N/A	N/A
	1994	0.995	0.974	0.941	0.674	0.793	0.334	0.850	0.570	N/A	N/A
	1995	0.838	0.740	0.755	0.232	0.647	0.787	0.752	0.326	N/A	N/A
	1996	0.755	0.163	0.913	0.304	0.796	0.028	0.887	0.733	N/A	N/A
	1997	0.500	0.239	0.657	0.486	0.748	0.891	0.762	0.560	N/A	N/A
	1998	0.875	0.841	0.976	0.832	0.927	0.687	0.873	0.603	N/A	N/A

	1999	0.929	0.721	0.919	0.734	0.762	0.472	0.825	0.520	N/A	N/A
	2000	0.938	0.927	0.900	0.588	0.922	0.338	0.816	0.541	N/A	N/A
	2001	0.921	0.902	0.954	0.871	0.941	0.990	0.988	0.982	N/A	N/A
	2002	0.925	0.689	0.798	0.642	0.834	0.752	0.775	0.619	N/A	N/A
	2003	0.920	0.470	0.814	0.572	0.786	0.376	0.818	0.493	N/A	N/A
	2004	0.845	0.875	0.921	0.588	0.817	0.355	0.745	0.410	N/A	N/A
	2005	0.897	0.496	0.798	0.295	0.783	0.259	0.746	0.335	N/A	N/A
	2006	0.941	0.845	0.884	0.970	0.840	0.164	0.867	0.821	N/A	N/A
	2007	0.959	0.958	0.998	0.998	0.948	0.944	0.907	0.903	N/A	N/A
	2008	0.901	0.886	0.894	0.685	0.827	0.583	0.778	0.539	N/A	N/A
	2009	0.927	0.454	0.715	0.241	0.771	0.714	0.723	0.330	N/A	N/A
	2010	0.888	0.489	0.846	0.530	0.826	0.745	0.802	0.594	N/A	N/A
	2011	0.919	0.139	0.781	0.474	0.832	0.772	0.831	0.516	N/A	N/A
Mayfly	**1980**	0.716	0.374	0.846	0.901	0.928	0.704	0.840	0.709	N/A	N/A
	1981	0.866	0.834	0.885	0.869	0.904	0.988	0.808	0.795	N/A	N/A
	1982	0.713	0.485	0.726	0.071	0.855	0.709	0.883	0.740	N/A	N/A
	1983	0.969	0.391	0.963	0.828	0.881	0.416	0.908	0.664	N/A	N/A
	1984	0.639	0.299	0.839	0.134	0.912	0.669	0.868	0.667	N/A	N/A
	1985	0.940	0.790	0.748	0.581	0.795	0.468	0.887	0.837	N/A	N/A
	1986	0.639	0.342	0.854	0.343	0.548	0.327	0.808	0.583	N/A	N/A
	1987	0.963	0.791	0.977	0.984	0.711	0.434	0.977	0.968	N/A	N/A
	1988	0.904	0.426	0.832	0.973	0.916	0.615	0.892	0.686	N/A	N/A
	1989	0.860	0.933	0.714	0.275	0.854	0.463	0.863	0.819	N/A	N/A
	1990	0.746	0.519	0.767	0.697	0.987	0.987	0.987	0.987	N/A	N/A
	1991	0.832	0.625	0.813	0.813	0.905	0.905	0.889	0.889	N/A	N/A
	1992	0.822	0.986	0.834	0.780	0.831	0.790	1.000	1.000	N/A	N/A
	1993	0.979	0.954	0.911	0.876	0.895	0.871	0.951	0.951	N/A	N/A
	1994	0.763	0.399	0.906	0.500	0.674	0.267	0.899	0.748	N/A	N/A
	1995	0.778	0.594	0.729	0.351	0.775	0.555	0.880	0.748	N/A	N/A
	1996	0.830	0.673	0.869	0.714	0.877	0.842	0.691	0.685	N/A	N/A
	1997	0.755	0.653	0.739	0.540	0.761	0.585	0.830	0.814	N/A	N/A
	1998	0.866	0.559	0.911	0.873	0.817	0.474	0.922	0.809	N/A	N/A
	1999	0.725	0.585	0.877	0.516	0.828	0.643	0.878	0.801	N/A	N/A
	2000	0.894	0.571	0.841	0.381	0.877	0.620	0.937	0.773	N/A	N/A
	2001	0.997	0.978	0.894	0.824	0.903	0.911	0.912	0.892	N/A	N/A
	2002	0.652	0.564	0.744	0.448	0.853	0.783	0.888	0.842	N/A	N/A
	2003	0.753	0.508	0.781	0.303	0.830	0.629	0.912	0.714	N/A	N/A
	2004	0.783	0.402	0.959	0.433	0.519	0.263	0.882	0.778	N/A	N/A
	2005	0.830	0.347	0.908	0.154	0.783	0.424	0.813	0.452	N/A	N/A
	2006	0.996	0.900	0.922	0.876	0.888	0.888	0.921	0.921	N/A	N/A
	2007	0.983	0.983	0.951	0.948	0.956	0.945	0.959	0.958	N/A	N/A
	2008	0.844	0.424	0.951	0.729	0.600	0.355	0.852	0.771	N/A	N/A
	2009	0.875	0.617	0.634	0.238	0.791	0.432	0.778	0.732	N/A	N/A
	2010	0.856	0.762	0.731	0.539	0.927	0.783	0.894	0.882	N/A	N/A
	2011	0.900	0.783	0.652	0.364	0.869	0.611	0.714	0.668	N/A	N/A

11.15 Appendix O- Mann Whitney tests for extreme year analysis- site 2

		Highly unsuitable		Unsuitable		Moderate		Suitable		Highly suitable	
		S1 to S2	S1 to S3	S1 to S2	S1 to S3	S1 to S2	S1 to S3	S1 to S2	S1 to S3	S1 to S2	S1 to S3
		p-value									
Adult brown trout	**1980**	0.785	0.769	0.631	0.432	0.772	0.320	0.784	0.396	1.000	1.000
	1981	0.769	0.655	0.878	0.921	0.883	0.802	0.657	0.622	0.913	0.913
	1982	0.913	0.117	0.757	0.590	0.909	0.579	0.872	0.567	1.000	1.000
	1983	0.981	0.612	0.910	0.536	0.787	0.362	0.964	0.482	0.992	0.992
	1984	0.956	0.107	0.744	0.620	0.671	0.415	0.836	0.460	1.000	1.000
	1985	0.783	0.622	0.648	0.572	0.768	0.578	0.785	0.703	1.000	1.000
	1986	0.854	0.301	0.772	0.506	0.652	0.648	0.775	0.406	1.000	1.000
	1987	0.897	0.854	0.954	0.938	0.845	0.695	0.955	0.938	1.000	1.000
	1988	0.744	0.946	0.842	0.447	0.917	0.452	0.666	0.240	0.998	0.998
	1989	0.234	0.181	0.723	0.305	0.527	0.153	0.914	0.689	1.000	1.000
	1990	0.718	0.027	0.532	0.009	0.778	0.367	0.944	0.944	1.000	1.000
	1991	0.805	0.505	0.448	0.127	0.540	0.263	0.879	0.879	NA	NA
	1992	0.816	0.286	0.753	0.253	0.642	0.340	0.691	0.665	1.000	1.000
	1993	0.976	0.926	0.976	0.854	0.847	0.342	0.880	0.835	1.000	1.000
	1994	0.725	0.753	0.826	0.507	0.883	0.540	0.724	0.344	1.000	1.000
	1995	0.839	0.914	0.753	0.807	0.643	0.451	0.974	0.422	1.000	1.000
	1996	0.276	0.010	0.358	0.017	0.832	0.107	0.686	0.629	1.000	1.000
	1997	0.931	0.502	0.949	0.185	0.786	0.004	0.765	0.445	1.000	1.000
	1998	0.707	0.872	0.749	0.447	0.934	0.470	0.835	0.570	0.998	0.998
	1999	0.797	0.762	0.808	0.792	0.771	0.595	0.849	0.658	1.000	1.000
	2000	0.570	0.603	0.684	0.309	0.826	0.474	0.865	0.531	1.000	1.000
	2001	0.857	0.788	0.928	0.742	0.958	0.869	0.821	0.480	0.929	0.929
	2002	0.837	0.278	0.793	0.652	0.714	0.851	0.782	0.684	0.999	1.000
	2003	0.983	0.342	0.829	0.667	0.655	0.962	0.902	0.497	1.000	1.000
	2004	0.702	0.610	0.770	0.579	0.826	0.445	0.808	0.612	1.000	1.000
	2005	0.871	0.224	0.715	0.184	0.827	0.168	0.857	0.253	0.992	0.992
	2006	0.808	0.070	0.740	0.322	0.740	0.475	0.820	0.815	1.000	1.000
	2007	0.836	0.828	0.909	0.880	0.890	0.879	0.848	0.837	1.000	1.000
	2008	0.626	0.848	0.807	0.614	0.772	0.417	0.797	0.599	1.000	1.000
	2009	0.851	0.814	0.708	0.579	0.609	0.084	0.921	0.350	0.998	0.998
	2010	0.644	0.991	0.760	0.446	0.693	0.067	0.834	0.531	1.000	1.000
	2011	0.322	0.939	0.494	0.384	0.806	0.176	0.897	0.504	1.000	1.000
Juvenile brown trout	**1980**	0.763	0.451	0.717	0.484	0.925	0.897	0.840	0.551	0.988	0.719
	1981	0.715	0.671	0.942	0.979	0.746	0.604	0.869	0.833	0.585	0.625
	1982	0.807	0.961	0.811	0.516	0.807	0.015	0.879	0.573	0.795	0.073
	1983	0.977	0.750	0.921	0.540	0.842	0.713	0.912	0.557	0.687	0.221
	1984	0.902	0.844	0.791	0.427	0.900	0.016	0.859	0.479	0.859	0.101
	1985	0.778	0.574	0.776	0.698	0.811	0.771	0.801	0.723	0.797	0.702
	1986	0.868	0.973	0.788	0.430	0.880	0.084	0.787	0.420	0.693	0.094
	1987	0.949	0.929	0.958	0.935	0.960	0.952	0.962	0.946	0.918	0.701
	1988	0.725	0.508	0.869	0.462	0.849	0.706	0.908	0.526	0.746	0.529
	1989	0.885	0.180	0.982	0.158	0.926	0.124	0.982	0.160	0.989	0.146
	1990	0.767	0.362	0.743	0.343	0.666	0.293	0.743	0.343	0.740	0.342
	1991	0.662	0.501	0.732	0.395	0.732	0.395	0.732	0.395	0.732	0.395
	1992	0.629	0.370	0.804	0.489	0.794	0.481	0.804	0.489	0.790	0.477
	1993	0.860	0.731	0.885	0.840	0.781	0.699	0.886	0.840	0.777	0.685
	1994	0.757	0.506	0.842	0.520	0.937	0.214	0.903	0.599	0.970	0.198
	1995	0.987	0.341	0.906	0.358	0.898	0.113	0.902	0.361	0.950	0.142
	1996	0.938	0.238	0.935	0.230	0.959	0.215	0.935	0.230	0.953	0.221
	1997	0.660	0.136	0.745	0.317	0.647	0.258	0.745	0.317	0.734	0.309
	1998	0.789	0.667	0.798	0.531	0.768	0.764	0.942	0.708	0.831	0.451
	1999	0.973	0.626	0.849	0.659	0.790	0.267	0.871	0.680	0.756	0.342
	2000	0.739	0.901	0.799	0.409	0.756	0.123	0.963	0.667	0.774	0.222
	2001	0.831	0.677	0.983	0.761	0.759	0.810	0.902	0.850	0.756	0.807
	2002	0.881	0.977	0.810	0.705	0.744	0.203	0.846	0.740	0.700	0.219

	2003	0.795	0.884	0.925	0.513	0.917	0.093	0.941	0.528	0.965	0.172
	2004	0.745	0.702	0.829	0.629	0.869	0.421	0.868	0.669	0.954	0.248
	2005	0.830	0.786	0.790	0.219	0.968	0.101	0.865	0.257	0.880	0.096
	2006	0.866	0.670	0.849	0.665	0.906	0.718	0.849	0.665	0.838	0.655
	2007	0.816	0.802	0.970	0.941	0.988	0.991	0.958	0.955	0.904	0.907
	2008	0.753	0.606	0.819	0.623	0.948	0.747	0.824	0.630	0.829	0.299
	2009	0.961	0.267	0.977	0.280	0.970	0.067	0.970	0.277	0.907	0.173
	2010	0.829	0.322	0.937	0.608	0.716	0.298	0.906	0.582	0.997	0.532
	2011	0.765	0.470	0.934	0.275	0.691	0.120	0.934	0.275	0.706	0.140
Spawning brown trout	**1980**	0.694	0.899	0.836	0.605	0.826	0.529	0.733	0.504	0.840	0.551
	1981	0.886	0.847	0.647	0.614	0.824	0.778	0.772	0.902	0.869	0.833
	1982	0.934	0.135	0.785	0.952	0.878	0.573	0.816	0.520	0.879	0.573
	1983	0.880	0.654	0.978	0.936	0.926	0.538	0.916	0.516	0.912	0.557
	1984	0.733	0.214	0.920	0.932	0.853	0.474	0.826	0.443	0.859	0.479
	1985	0.697	0.623	0.758	0.540	0.795	0.717	0.599	0.270	0.801	0.723
	1986	0.624	0.138	0.979	0.654	0.787	0.420	0.683	0.355	0.787	0.420
	1987	0.991	0.840	0.957	0.965	0.954	0.924	0.804	0.716	0.962	0.946
	1988	0.726	0.857	0.737	0.742	0.806	0.346	0.886	0.405	0.908	0.526
	1989	0.993	0.169	0.658	0.241	0.818	0.777	0.993	0.154	0.982	0.160
	1990	0.706	0.319	0.771	0.363	0.696	0.045	0.741	0.342	0.743	0.343
	1991	0.732	0.395	0.667	0.348	0.892	0.016	0.732	0.395	0.732	0.395
	1992	0.804	0.489	0.619	0.321	0.701	0.064	0.804	0.489	0.804	0.489
	1993	0.942	0.896	0.827	0.637	0.883	0.838	0.837	0.790	0.886	0.840
	1994	0.989	0.544	0.775	0.853	0.834	0.451	0.824	0.384	0.903	0.599
	1995	0.916	0.293	0.862	0.463	0.909	0.432	0.943	0.330	0.902	0.361
	1996	0.938	0.229	0.954	0.167	0.388	0.037	0.935	0.230	0.935	0.230
	1997	0.664	0.268	0.599	0.047	0.745	0.317	0.745	0.317	0.745	0.317
	1998	0.747	0.855	0.842	0.912	0.925	0.668	0.816	0.434	0.942	0.708
	1999	0.935	0.746	0.910	0.651	0.862	0.670	0.821	0.612	0.871	0.680
	2000	0.432	0.359	0.689	0.862	0.918	0.603	0.935	0.485	0.963	0.667
	2001	0.942	0.855	0.792	0.723	0.834	0.658	0.875	0.841	0.903	0.850
	2002	0.709	0.457	0.989	0.743	0.826	0.732	0.801	0.689	0.846	0.740
	2003	0.912	0.259	0.972	0.841	0.892	0.468	0.920	0.483	0.941	0.528
	2004	0.630	0.706	0.806	0.956	0.861	0.655	0.722	0.383	0.868	0.669
	2005	0.873	0.050	0.855	0.712	0.864	0.257	0.766	0.177	0.865	0.257
	2006	0.991	0.799	0.882	0.680	0.818	0.599	0.841	0.657	0.849	0.665
	2007	0.878	0.870	0.909	0.902	0.944	0.940	0.922	0.889	0.958	0.955
	2008	0.837	0.791	0.801	0.782	0.804	0.590	0.714	0.459	0.824	0.630
	2009	0.998	0.282	0.926	0.209	0.966	0.277	0.966	0.274	0.970	0.277
	2010	0.874	0.552	0.720	0.236	0.897	0.574	0.971	0.638	0.906	0.582
	2011	0.658	0.129	0.663	0.531	0.934	0.944	0.932	0.274	0.934	0.275
Crowfoot	**1980**	0.821	0.786	0.856	0.766	0.837	0.447	0.805	0.535	0.840	0.551
	1981	0.625	0.594	0.723	0.761	0.642	0.610	0.934	0.898	0.869	0.833
	1982	0.923	0.197	0.963	0.581	0.879	0.573	0.879	0.573	0.879	0.573
	1983	0.898	0.741	0.803	0.482	0.973	0.506	0.911	0.556	0.912	0.557
	1984	0.951	0.222	0.747	0.513	0.857	0.478	0.819	0.448	0.859	0.479
	1985	0.775	0.666	0.661	0.638	0.798	0.713	0.801	0.723	0.801	0.723
	1986	0.852	0.371	0.754	0.344	0.777	0.412	0.791	0.428	0.787	0.420
	1987	0.987	0.924	0.951	0.875	0.957	0.930	0.962	0.946	0.962	0.946
	1988	0.638	0.819	0.866	0.888	0.822	0.414	0.894	0.511	0.908	0.526
	1989	0.259	0.086	0.520	0.123	0.789	0.962	0.982	0.160	0.982	0.160
	1990	0.661	0.019	0.801	0.384	0.871	0.095	0.743	0.343	0.743	0.343
	1991	0.604	0.084	0.736	0.472	0.925	0.038	0.732	0.395	0.732	0.395
	1992	0.717	0.180	0.566	0.264	0.753	0.108	0.804	0.489	0.804	0.489
	1993	0.993	0.836	0.918	0.456	0.883	0.837	0.886	0.840	0.886	0.840
	1994	0.718	0.367	0.999	0.485	0.876	0.548	0.893	0.585	0.903	0.599
	1995	0.630	0.766	0.526	0.309	0.903	0.371	0.902	0.361	0.902	0.361
	1996	0.281	0.010	0.954	0.193	0.500	0.103	0.935	0.230	0.935	0.230
	1997	0.946	0.457	0.762	0.001	0.745	0.317	0.745	0.317	0.745	0.317
	1998	0.764	0.836	0.857	0.508	0.924	0.685	0.887	0.646	0.942	0.708
	1999	0.803	0.486	0.676	0.918	0.870	0.678	0.866	0.676	0.871	0.680
	2000	0.619	0.464	0.772	0.414	0.957	0.655	0.879	0.580	0.963	0.667
	2001	0.707	0.656	0.831	0.936	0.733	0.646	0.888	0.801	0.903	0.850
	2002	0.810	0.340	0.726	0.567	0.838	0.743	0.812	0.707	0.846	0.740

	2003	0.940	0.188	0.935	0.650	0.900	0.482	0.937	0.525	0.941	0.528
	2004	0.705	0.532	0.927	0.379	0.867	0.662	0.861	0.664	0.868	0.669
	2005	0.839	0.196	0.958	0.173	0.863	0.256	0.861	0.255	0.865	0.257
	2006	0.728	0.050	0.873	0.470	0.796	0.998	0.849	0.665	0.849	0.665
	2007	0.880	0.876	0.952	0.961	0.948	0.944	0.959	0.947	0.958	0.955
	2008	0.793	0.883	0.831	0.556	0.817	0.615	0.824	0.630	0.824	0.630
	2009	0.682	0.506	0.677	0.049	0.956	0.269	0.982	0.284	0.970	0.277
	2010	0.663	0.395	0.826	0.021	0.885	0.556	0.921	0.595	0.906	0.582
	2011	0.285	0.933	0.493	0.499	0.934	0.760	0.934	0.275	0.934	0.275
Mayfly	**1980**	0.845	0.544	0.641	0.860	0.633	0.944	0.771	0.515	0.776	0.508
	1981	0.637	0.605	0.907	0.825	0.703	0.692	0.983	0.978	0.954	0.915
	1982	0.730	0.705	0.937	0.152	0.947	0.038	0.851	0.549	0.879	0.573
	1983	0.993	0.829	0.765	0.888	0.714	0.433	0.924	0.536	0.915	0.550
	1984	0.925	0.450	0.955	0.176	0.954	0.093	0.804	0.437	0.796	0.430
	1985	0.801	0.606	0.602	0.631	0.737	0.844	0.795	0.715	0.801	0.723
	1986	0.905	0.749	0.758	0.590	0.541	0.031	0.796	0.434	0.796	0.434
	1987	0.963	0.946	0.928	0.698	0.977	0.699	0.958	0.933	0.962	0.946
	1988	0.800	0.653	0.787	0.883	0.593	0.838	0.826	0.367	0.867	0.486
	1989	0.883	0.186	0.991	0.060	0.906	0.141	0.982	0.158	0.914	0.687
	1990	0.772	0.363	0.712	0.320	0.727	0.333	0.743	0.343	0.944	0.944
	1991	0.461	0.213	0.815	0.459	0.732	0.395	0.732	0.395	0.879	0.879
	1992	0.759	0.453	0.879	0.471	0.804	0.489	0.804	0.489	0.691	0.665
	1993	0.876	0.798	0.899	0.626	0.884	0.839	0.878	0.831	0.884	0.839
	1994	0.863	0.680	0.851	0.907	0.931	0.427	0.823	0.433	0.881	0.568
	1995	0.915	0.329	0.709	0.898	0.950	0.220	0.911	0.352	0.960	0.432
	1996	0.957	0.156	0.926	0.202	0.952	0.222	0.935	0.230	0.686	0.629
	1997	0.543	0.194	0.973	0.081	0.674	0.274	0.745	0.317	0.765	0.445
	1998	0.898	0.840	0.921	0.732	0.759	0.618	0.821	0.548	0.859	0.614
	1999	0.984	0.655	0.802	0.638	0.904	0.511	0.856	0.661	0.865	0.675
	2000	0.819	0.954	0.888	0.307	0.704	0.135	0.825	0.462	0.716	0.420
	2001	0.739	0.687	0.995	0.805	0.840	0.780	0.945	0.716	0.918	0.811
	2002	0.936	0.950	0.928	0.544	0.743	0.194	0.820	0.703	0.796	0.695
	2003	0.915	0.681	0.669	0.759	0.882	0.152	0.913	0.443	0.930	0.519
	2004	0.855	0.814	0.803	0.650	0.637	0.346	0.846	0.643	0.841	0.645
	2005	0.861	0.761	0.930	0.867	0.846	0.086	0.834	0.241	0.856	0.253
	2006	0.843	0.659	0.699	0.323	0.962	0.771	0.849	0.665	0.820	0.815
	2007	0.953	0.950	0.961	0.950	0.907	0.899	0.961	0.943	0.935	0.910
	2008	0.819	0.675	0.978	0.706	0.849	0.519	0.811	0.596	0.824	0.630
	2009	0.994	0.247	0.826	0.342	0.843	0.164	0.975	0.279	0.960	0.376
	2010	0.931	0.447	0.747	0.537	0.808	0.481	0.931	0.595	0.921	0.607
	2011	0.933	0.356	0.690	0.098	0.629	0.127	0.932	0.274	0.897	0.504