Characterising Historic Ecological Conditions in Lowland Rivers: Applying Palaeoecological Techniques to River Restoration Planning

By

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ABSTRACT

It is widely recognized that anthropogenic activities have resulted in significant changes to the ecology and hydromorphology of riverine ecosystems globally. Across much of lowland England a number of riverine habitats and the flora and fauna inhabiting them, have been lost or disadvantaged by historic channel modifications. Many of the most significant modifications took place in the decades following World War II, in a drive to increase food security through improved land drainage and associated flood management. A better understanding of the geomorphological, hydrological and biodiversity elements that have been compromised or lost is required in order to characterise the benefits of planned measures to restore and reinstate channel form and function for EU WFD, Habitats Directive and other conservation designations (e.g. SSSI condition assessments). To generate this understanding in an environment where natural processes have been impacted over large spatial scales, an innovative palaeoecological approach is employed in this thesis that provides a window on historic riverine ecology and habitat conditions so that the contemporary channel and community inhabiting it can be gauged prior to the implementation of river restoration programmes. The analysis of historic archival material (maps, photographs, local authority and management records), and the detailed investigation of sedimentary records and sub-fossil insect remains (Trichoptera, Coleoptera and Gastropoda) associated with in-channel bars, weirs, bridges and palaeochannels is used in this thesis in order to achieve this. These records will provide data regarding changes to the aquatic macroinvertebrate community and instream hydromorphology within specific reaches/biotopes/habitats subjected to historical physical modification. The historic data (documents and palaeoenvironmental data) is analysed in parallel with contemporary data on instream habitats and faunal community composition to define benchmark conditions on three Site of Special Scientific Interest rivers. This approach enables a comparison between past and present channel hydromorphology and the instream faunal communities. The characterisation of benchmarks provides a baseline for future conservation and restoration policies within riverine ecosystems that can be used to help define pre-impacted or ‘reference’ conditions. The research presented in this thesis has relevance to the conservation objectives of rivers with special designations (e.g. SSSI and Habitats Directive) for wildlife and to meeting the wider requirements of the Water Framework Directive.

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CHAPTER ONE: INTRODUCTION

1.1 Research Context

Water is arguably one of the most precious commodities of our world and yet is also one of the most degraded (Aronson et al., 2006). The majority of people in the developed western world have taken water for granted, paying scant regard to its sustainable utilization. The impact the human race has had on riverine environments has been extensive throughout history, with the last 200 years bringing about transformations on an unprecedented scale (England et al., 2008).

Across much of Europe, many lowland rivers have been substantially modified to facilitate the ever-increasing demands and needs of growing human populations. This has been achieved through channelization for flood control, to aid land drainage and to support the intensification of agriculture. In the UK some of the most significant river modifications have taken place since 1945 in a drive to increase food security (Mainstone, 2008). The majority of contemporary rivers are the product of the interactions between natural and anthropogenic processes, however the physical modifications associated with river engineering have unquestionably impacted the biodiversity of river ecosystems (Gregory, 2006).

Faced with the recognition of the increasing effects that anthropogenic activities have had on global water resources, there has been a growing realisation for the need to balance the needs of human water demands, economic benefits and the environmental concerns for the water needs of riverine ecosystems (Zalewski, 2002; Acreman & Ferguson, 2010; Poff et al., 2010). In response to this growing awareness, sustainable approaches to water resources are now being developed to manage, rehabilitate and restore historically heavily modified rivers and alleviate the widespread degradation they are facing (Janes et al., 2005; Linke et al., 2011).

Restoration and rehabilitation has become a fundamental element of ecosystem management and is at the forefront of river science (Wohl et al., 2005; Naiman et al., 2012). However, given the crisis facing the biodiversity of lotic systems there is an urgent need to re-examine restoration schemes, incorporating broader, more holistic approaches to aquatic ecosystem management (Giller, 2005; Arthington et al., 2010). River restoration efforts have rarely been based on inputs from ecological theory. Due to the realisation that physical and chemical objectives alone are no longer sufficient for the protection of aquatic ecosystems, restoration schemes should now be supplemented with clear biological objectives (Boon, 2000; Boavida et al., 2012). The practice of river restoration also requires a more detailed understanding of a rivers 'reference condition', and the changes that have historically occurred in order to inform and successfully implement future schemes (Darby & Sear, 2008). This is primarily due to a number of previous restoration schemes and related activities failing to benefit river ecology and in some cases have been responsible for causing serious damage to flora and fauna (Bannister et al., 2005).

River restoration is not an exact science and it may help to explain why there are multiple terms used to illustrate interventions taken by river managers to help 'improve' riverine environments. There is no single definition of river restoration or even an agreement as to how appropriate the term 'restoration' is when compared to rehabilitation or enhancement (Boon, 1998; Downs & Gregory, 2004; Wheaton et al., 2008). However, among the numerous proposed definitions one of the most widely used is that of Cairns (1991: p. 186) who describes restoration as 'the complete structural and functional return to a pre-disturbance state'. Unfortunately this level of restoration is rarely practiced or achieved, due to both a lack of knowledge regarding what constitutes as a 'reference condition', and shifting ecological baselines (Papworth et al., 2009). This prospect however may become an issue of the past as river restoration is becoming a higher priority for river and water resource managers globally. Broadly, river restoration can be considered as a generic term for activities aimed at improving the physical and ecological characteristics of a river (Wheaton et al., 2006). In the UK to date, nearly 1200 river restoration projects have been completed, with a further 500 proposed projects waiting to commence (The River Restoration Centre, 2013). Until recently the majority of these projects would have been centered at habitat scale, however now, river restoration projects are directed at reach scale especially along Sites of Scientific Interest (Bannister et al., 2005).

Although there has been a large increase in the number of river restoration projects around the world (Ormerod, 2004; Skinner & Bruce-Burgess, 2007) due to government legislation and agency requirements, there is a perception that the scientific foundations of river restoration are weak, especially when compared to the commercial implications of proceeding with river restoration schemes (Downs & Kondolf, 2002; Palmer et al., 2005; Skinner et al., 2008). A lack of systematic monitoring and project evaluation (pre and post restoration) restricts the ability to learn lessons from good practice in many instances (Clark, 2002; Bernhardt et al., 2005). This has increased the uncertainty surrounding river restoration and risks undermining the confidence of both the public and those who fund river restoration projects (Palmer et al., 2005; Raven, 2011). However, since 2003 increased attention has been focused on the management of the UK's rivers, with local and regional river restoration initiatives being developed to address the requirements of the EU Water Framework Directive (WFD) of achieving 'good ecological status' by 2015 (Commission of European Community, 2000). The WFD is probably the most important piece of water legislation produced by the European Commission as it identifies the need to protect and enhance the status of aquatic ecosystems (England *et al*. 2007). In order to achieve these goals all EU member states need to establish ecological targets through the implementation of an assessment procedure for the river's current water qualities and its 'reference conditions' (Adrianssens et al., 2006). In order to assess the current ecological status's of rivers, a number of ecological indicators are used including fish, macroinvertebrates, macrophytes and diatoms. The largely sedentary nature of macroinvertebrates, combined with their measurable response to changing environmental conditions, favour their use as important bio-indicators of water chemistry (Metcalfe-Smith, 1994). However, a major hurdle encountered with the Directive (as with river restoration) is defining what is meant by 'reference conditions'. One method, which has been investigated to help overcome this, is to establish pre-impact ecological conditions of the water-body through the application of palaeoecological techniques (Seddon et al., 2012). Traditionally, palaeoecological techniques have been extensively used within lake studies (Anderson et al., 2006; Birks & Birks, 2006) and until recently little research has been undertaken on reference conditions in rivers due to a rivers more dynamic nature and more appropriate depositional events for enhancing sedimentation.

In recent years palaeoecology has increasingly taken a more quantitative route to environmental reconstruction, with the aim of quantifying the relationships between biological parameters and past environments (Brown, 2002). The dynamic environment of river floodplains has been widely identified as a source of evidence in palaeoecology, through providing archives of change. Floodplains can be regarded as artifacts of human activities as well as preserved records of the rivers biological and physical history (Greenwood *et al*. 2006). This can be utilised to determine the range of natural variability experienced, hence providing a baseline or a series of reference points, which anthropogenic influences can be measured against (England *et al*. 2007). Ecological reference points are an essential element in the biological evaluation of rivers for the WFD. Another benefit of using reference conditions is that more detailed and accurate river restoration endpoints can also be defined with a 'reference' river condition in mind.

This project will use an innovative palaeoecological approach that enables the reconstruction of past riverine environments so that the present and historic community inhabiting the river can be compared and subsequently be used to frame 'reference conditions' for addressing WFD objectives and creating a series of targets for reaching good ecological status. Through gaining an understanding of the geomorphological, hydrological and biodiversity elements that have been affected/modified/changed, it will allow the effects of past anthropogenic activities to be gauged, helping to improve the development and implementation of suitable methods to restore the degraded channel and measure any resulting ecological benefits. There is a wide range of terminology is used within this thesis and the working definitions are presented in Table 1.1.

Table 1.1 Working definitions of the commonly used terms within this thesis.

Term	**Definition**	**References**
Biotope	An ecological system or habitat that provides a living place for a specific assemblage of plants and animals. e.g. a riffle within a river.	Newson & Newson, 2000; Harvey *et al.*, 2008.
Ecohydromorphology	The integration of ecology with hydrology and geomorphology.	Clarke *et al*., 2003; Naughan *et al.,* 2007.
Good Ecological Potential	A WFD target set for designated artificial water bodies (AWBs - canals and docks) and heavily modified water bodies (HMWBs - modified to the extent that it will not be possible for them to meet the WFD targets). This target will be derived from the best ecological condition achievable for that water body, taking into account both the physical modifications made to the water body, and its current use.	European Commission, 2000
Good Ecological Status	A WFD target indicating that human activities have had only slight impacts	European Commission, 2000

	on the ecological characteristics of aquatic plants and animal communities. In practice, this means that the ecological target may be a slight reduction in quality when compared to the pristine water body.	
Hydromorphology	A term used in river basin management to describe the hydrological and geomorphological processes and attributes of rivers, lakes, estuaries and coastal waters.	Newson & Large, 2006; Haase *et al*., 2012.
Reference condition	An insight into the past environment. The desired endpoint to river restoration and provides river managers with a greater sense of predictability of restoration outcomes	Hawkins *et al.*, 2006;
River restoration	The process of recovering a rivers physical (geomorphological and hydrological) and ecological characteristics that have been damaged, degraded, or destroyed. Restoration of a riverine ecosystem is an attempt to return the natural diversity of flows and channel geomorphology.	Sear, 1994; Kondolf, 1995; Brookes and Shields, 1996; Palmer *et al*., 2010.
River rehabilitation	The reinstatement of ecosystem processes, services, and productivity but it does not necessarily mean to restore the ecosystem to its pre-existing condition.	Large and Petts, 1994; Pretty *et al*., 2003; Janes *et al*., 2005.
Multiproxy Studies	A combination of a number of floral or faunal groups or environmental characteristics. Due to the complex network of interactions throughout an ecosystem, it is desirable to study multiple lines of evidence. This provides a wider overview of historical conditions than that possible from a single proxy.	Mann, 2002; Birks & Birks, 2006; Whitehouse *et al*., 2008.
Subfossil	A preserved organism that has not fully fossilised due to the conditions in which the remains were deposited not being optimal for fossilization.	Brown, 2002; Howard et al., 2009.
Transfer Functions	Used for the quantitative reconstruction of historic environmental and biological variables from biological proxy data. One of the most widely used transfer functions is between diatoms and lake-water pH, salinity and total phosphorus.	Sayer and Roberts, 2001; Thorp et al., 2006.

1.2 Thesis Aims and Objectives

The primary aim of this thesis is to determine the potential of palaeoecological techniques (via examination of instream and floodplain deposits) to help define reference conditions for river management and restoration planning. This will be achieved through the use of contemporary river morphology and instream macroinvertebrate ecological data in direct association with environmental palaeoecological data from adjacent sites. A multiproxy approach using contemporary samples and sub-fossil remains of Coleoptera, Trichoptera and Gastropoda from cores/sections collected from palaeochannel deposits, will be used to interpret the palaeoenvironmental setting of each case study river.

This thesis will directly contribute to existing knowledge through the definition of 'reference condition' states for lowland rivers where natural analogues do not exist in many instances. It is hypothesized that:

1. Using historic information (historic maps and documents) it is possible to identify instream morphological and habitat features that may have been significantly degraded by previous management operations.
2. Elements of the instream faunal community have been significantly compromised or may have become locally extinct as a result of historic channel management operations and that examination of palaeoecological communities; aquatic beetle (Coleoptera), caddisfly larvae (Trichoptera) and snails (Gastropods), will enable identification of this and other changes to in-stream communities.
3. Through detailed examination of the contemporary riverine communities and those within former channels (palaeochannels and floodplain deposits) it is possible to determine historic environments that may form the basis for future river restoration schemes and inform the definition of 'reference conditions' for WFD purposes.

Addressing these hypotheses represents a fundamental part of the evidence base for developing river restoration programmes (and delivering the EU WFD, the EU Habitats Directive and local conservation legislation for SSSI's and SAC's) where knowledge regarding historic instream channel form and its associated ecology (biodiversity) is currently limited or absent. A range of methods and data analysis techniques will be employed and the hypotheses identified above will be addressed by the following objectives to:

1. Examine the national and international drivers for contemporary river management, conservation and restoration (Chapter 2).
2. Examine the use of sub-fossil macroinvertebrates in riverine palaeoecological analysis in relation to the identification of a historic reference condition, through detailed examination of the literature and put into practice the multiproxy methods researched (Chapter 2).
3. Identify study sites from a list of potential rivers (provided by Natural England) and map the position of historic palaeochannels at each study site using GIS. Use this information to locate suitable areas to undertake contemporary and palaeo channel sampling (Chapters 4, 5 and 6).
4. i) Characterise the contemporary riverine environment of three lowland river reaches via sampling and examination of the biotic and abiotic elements of the rivers and;

   ii) Characterise the palaeoecology of each river at the time the palaeochannel was cut-off using multi-proxy approaches in order to aid definition of ‘reference conditions’ (Chapters 4, 5 and 6).
5. Compare the results from the contemporary analysis to the palaeoecological analysis to understand the changes that have taken place within each river over time (Chapters 4, 5 and 6).

1.3 Thesis Structure

The overall thesis structure, including the primary content of each chapter is shown schematically in Figure 1.1. Chapter 2 begins by presenting a review of previous research centred on river restoration and the ecological theory that supports it. The legislation and policies that have been put in place to drive and underpin river restoration schemes are also highlighted (e.g. WFD, Habitats Directive, SSSI, SAC). The chapter examines the different methods that have been used in river restoration, how successful they have been and the necessity of post restoration monitoring for future projects. The chapter explores the use and benefits of palaeoecology in riverine environments through the detailed examination of biological proxies and the factors affecting the validity of interpretations derived from them.

Chapter 3 outlines the methodology, fieldwork protocols and data analysis techniques used to explore the aims and objectives of this thesis. A detailed explanation of the methods used is provided for both the contemporary sampling and palaeoenvironmental studies. These methods include the extraction, preparation, identification and preservation methods carried out and the confidence that can be placed on the results.

Chapters 4, 5 and 6 present detailed case studies of the contemporary and palaeoecological results for each chosen river study site; the River Eye (Melton Mowbray, Leicestershire) (Chapter 4), the River Hull (Driffield, East Riding of Yorkshire) (Chapter 5) and the River Wensum (Fakenham, Norfolk) (Chapter 6), respectively (see Chapter 3.2 for site selection details).

Chapter 7 synthesises the results presented in Chapters 4, 5 and 6, through discussion of the concept of 'reference conditions' and makes recommendations with regards to attaining the WFDs 'good ecological status' and fulfilling the condition assessment for the Habitats Directive or SSSI status, for each river study site. In addition, this chapter also highlights the implications of scaling up results from river reaches to obtain a landscape perspective and an understanding of the linkages between the appropriate spatio-temporal scale and relevant ecological processes.

Chapter 8 summarises the findings of the research through combining the over arching themes in relation to the aims and objectives presented in this chapter (1). It addresses the wider applications of this research with regards to the development of future river restoration and management strategies in relation to the European Water Framework Directive. This chapter ends with recommendations for future research.

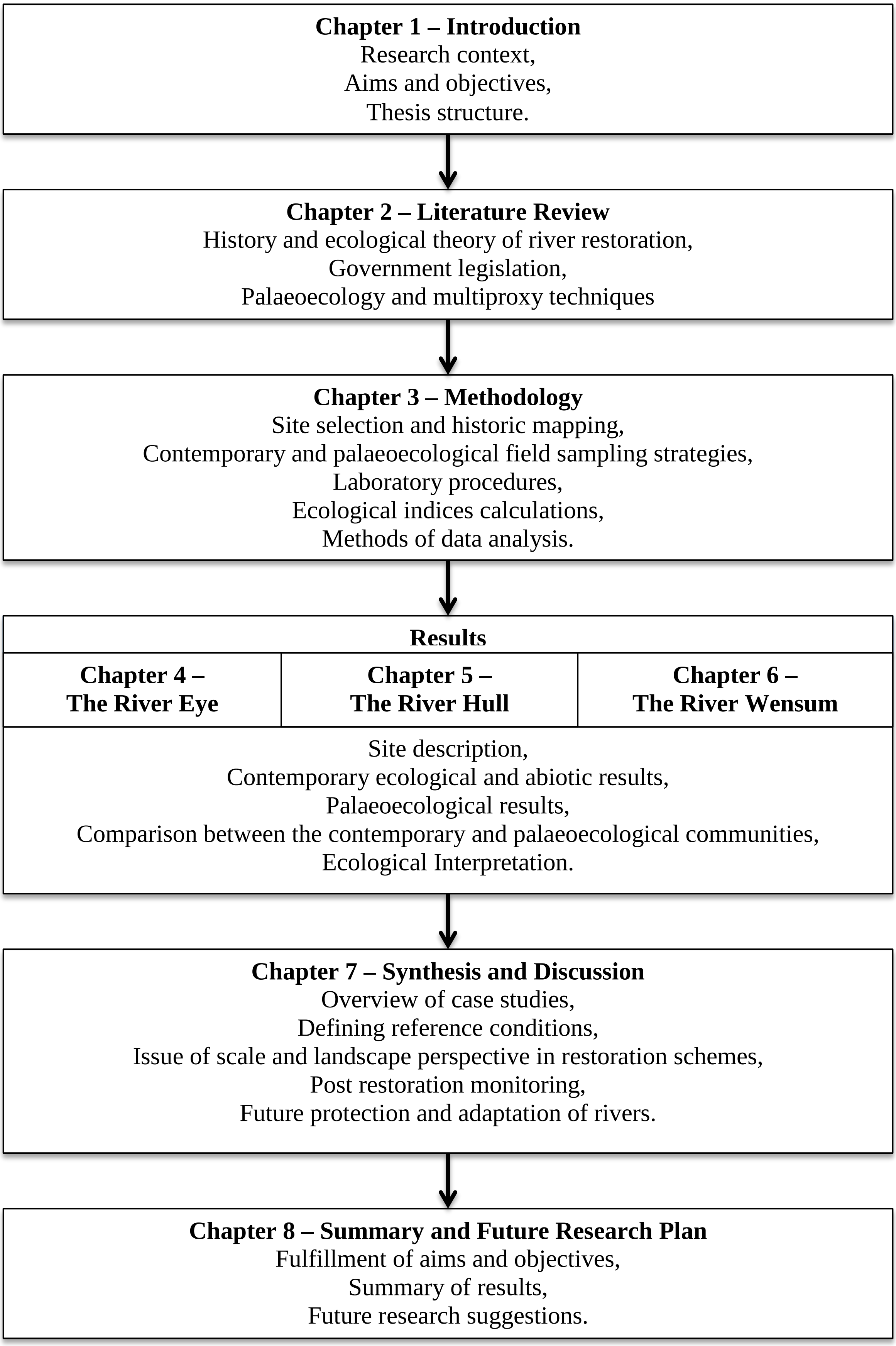

Figure 1.1 Schematic diagram of the thesis structure.

1.4 Summary

This chapter has provided an introduction to the key themes and concepts relating to river restoration, the concepts of ‘reference conditions’ and how the use of palaeoecological techniques can be used to inform these. The aims and objectives of the research are presented and the thesis structure is clearly outlined. The following chapter reviews the published research literature pertaining to river restoration and the use of palaeoecology, providing a theoretical and applied background to the research contained within the thesis.

CHAPTER TWO: LITERATURE REVIEW

2.1 Introduction

This chapter provides an overview of the literature addressing river restoration along with the associated legislation in the UK, EU and internationally. There is an overview of the current UK/EU government legislation driving river restoration and the methods that are used to determine if the outcomes are successful. This is followed by an insight into the use of biological proxies as indicators of past environments as a means to identify baseline conditions to inform the characterisation of ‘reference conditions’. The benefits of reconstructing past environments through the use of aquatic macroinvertebrate proxies are discussed. The proxies used in this thesis (Coleoptera, Trichoptera and Gastropoda) are considered separately and the potential information gained from multiproxy studies is also examined. The objectives of this chapter are to:

1. Examine the national and international drivers for contemporary river management, conservation and restoration.
2. Examine the use of sub-fossil macroinvertebrates in riverine palaeoecological analysis in relation to the identification of historical reference conditions, through detailed examination of the literature.

Freshwater within rivers and the diverse ecosystems that it supports presents a major challenge for water resource management, due to the increasing anthropogenic pressures placed on ecosystem services and products it provides. By way of example, freshwater ecosystems are of critical importance in underpinning global networks of food production in the form of fisheries and arable and pastoral agriculture (Petts et al., 2006; Petts, 2009). However, they are probably the most impacted ecosystems on the planet (Malmqvist & Rundle, 2002). Riverine floodplains and their valleys remain the focus of human settlement and consequently lotic systems have been over exploited historically and very few river catchments remain unaffected by anthropogenic pressures (Allan & Flecker, 2007).

The impact that anthropogenic activity has had on riverine environments has been extensive throughout history, with the last 200 years witnessing transformations on an unprecedented scale (England et al., 2008). This increasing human pressure and exploitation of rivers worldwide has seen them canalised for navigation purposes (Davis & Kidd, 2012), regulated by weirs and sluices for flood defences (Feld et al., 2011), treated as transportation sewers (Stapleton et al., 2008), used to drain urban areas and the surrounding floodplains reclaimed for agricultural and urban use (Pretty et al., 2003). The dramatic changes to the physical structures of rivers, brought about by human activities, have significantly altered their ecological functioning. Rivers and streams have the ability to withstand significant levels of exploitation and through their natural cleansing ability, recovery from some disturbances can occur with minimal anthropogenic intervention (Leopold et al., 1964; Lake, 2003; Li et al., 2012). However, in the face of continuing anthropogenic demands on rivers, these pressures can have a profound negative ecological effect causing the natural resilience and resistance of these freshwater ecosystems to come under significant threat (Giller, 2005; Barbour & Paul, 2010).

The realisation of the potential economic, social and ecological losses that result from river degradation is now becoming apparent, due to the rapidly growing drive to restore freshwater ecosystem functionality globally (Wohl et al., 2005; Linke et al., 2011). However, restoration poses major challenges to physical and ecological science as it tests the feasibility of recreating and reshaping complex natural environments from their current degraded state (White & Walker, 1997; Geist, 2011). The real and apparent conflicts and demands between ecosystem functioning and anthropogenic needs must be balanced. This requires new partnerships to be created between scientists, practitioners and other stakeholders allowing ecological goals to be identified, incorporating collective visions into river management and conservation (Arthington, et al., 2010). Close cooperation between all parties is necessary and would be highly beneficial in beginning to reverse the damage that has been inflicted upon many riverine systems and reduce the apparent failure rates of river restoration projects (Woolsey et al., 2007). It has been claimed that the majority of river restoration projects are currently undertaken with limited scientific input (Jansson et al., 2007). As a consequence many restoration schemes have not been self-maintaining and therefore have required continued management (Mika et al., 2010). Many schemes have been species- or habitat-based and thus have sought to recreate channel forms believed to favour these. However, there is now a growing recognition of the problems associated with this type of restoration project which has led to a demand for ecological-based restoration processes to help encourage more self-sustaining schemes (Clarke et al., 2003; Barmuta et al., 2011).

2.2 River restoration – its history and methods

Throughout history rivers have provided the foundation to socioeconomic development and despite the alterations they have faced, there have been numerous efforts made to restore them (Petts, 1989; Geist, 2011). Within Europe, modification of rivers accelerated during the early twentieth century, largely due to the intensification of agriculture, when rivers were channelized to make them straighter, deeper and wider in order to facilitate drainage of land and flood control (Hughes et al., 2005). Many streams were dredged and instream gravel deposits and woody debris removed, reducing their habitat heterogeneity (Harrison et al., 2004). Hard, physical engineering such as channelization, is now known to significantly reduce the diversity and abundance of flora and fauna, and such effects are increasingly seen to be unacceptable (Harrison & Keller, 2007). Channelization also significantly modifies a rivers geomorphology, which subsequently plays a very influential role on the ecological assemblages found within a river (Dunbar et al., 2010). The anthropogenically altered riverine environment can cause large changes in the flow regime and sedimentology; both of which are essential in maintaining a heterogeneous habitat for instream ecology. The restoration of running waters has now grown to become a sub-discipline in its own right and attracts interest from a variety of disciplines. It wasn't until the publication of '*The Restoration of Rivers and Streams: Theories and Experience*' (Gore, 1985) that attention was drawn towards the restoration of rivers and streams with biological outcomes being desirable. Prior to this point in time considerations and efforts were focused overwhelmingly on terrestrial restoration (Ormerod, 2003). Dobson *et al.* (1997) continued to explore restoration initiatives which helped to cement the idea that restored ecosystems should help to complement biodiversity conservation. The aims of river restoration are now multifunctional due to the growing responsibility being placed on those undertaking restoration projects to conserve biodiversity, whilst also safeguarding the ecological goods, ecosystem services and functions that rivers provide (Ormerod, 2004; Mainstone et al., 2011).

The restoration of rivers can either be 'passive', allowing natural processes to guide the outcomes, or 'active', where measures are applied to a channel and results of this change become evident more quickly (Stanford et al., 1996; Jähnig et al., 2010). The majority of rehabilitation projects that occur within the UK fall into the active category (Harrison *et al*., 2004), of which installation of artificial gravel riffles, large boulders or wood acting as flow deflectors, are the most common methods. Downs & Gregory (2004) classified restoration projects into five groups (Table 2.1), which describe the appropriate techniques for varying circumstances. One of the most visually striking and active methods of restoring rivers is the reconstruction of a channel (Vivash, 1999).

Table 2.1 Five types of river restoration based on Downs & Gregory (2004).

Type	**Description**	**Techniques**
Non-structural	Measures addressing underlying causes of river degradation in the catchment or corridor.	Land use planning, benign neglect, land-water management, buffer strips, fencing, tree planting on bank.
Network connectivity	Restore natural hydrological and sedimentological processes. Increase lateral and longitudinal connectivity.	Environmental flows, removing weirs and obstructions, setting back embankments (allowing floodplain inundation), reconnecting side channels and backwaters.
Prompted recovery	Instream measures to manipulate flow and sediment transport and create local diversity in flow and habitat hydraulics. Encouraging natural hydrological processes.	Deflectors, low weirs, sills, vanes, large woody debris, sediment traps. Recreating pools/riffles, introducing coarse sediment.
Morphological reconstruction	Direct reconstruction of channel form	Restore meanders and 'natural' channel form such as bank asymmetry, two-stage channels, island creation, dredging.
Erosion protection	Techniques to protect infrastructure and development.	Willow spiling, geotechnical fabrics, rip-rap, bed protection (grade control structures).

Most active river restoration projects have objectives that usually involve the creation of a new aligned, single thread, stable, meandering channel (Kondolf, 2006). The recreation of meanders is an obvious goal in rivers where historical courses have been straightened through the process of channelization. However, channel and meander reconstruction projects have also been undertaken on rivers that historically, never had a meandering form (Kondolf et al., 2001). Although most projects of this nature usually end in failure, with either the meanders being washed out, or if they remain stable then they do not successfully provide a habitat that would naturally exist within the environment. Nevertheless, they continue to be viewed as a popular restoration choice and there are a number of reasons for this (Kondolf, 2006). These include the ease with which they can be applied by people untrained in the science of fluvial geomorphology, and also because of the cultural preference for single thread meandering channels (Nassauer, 1995; Buijs, 2009). The notion that we find meander bends to be more aesthetically pleasing than 'messier' braided or anastomosing rivers is deeply rooted within our culture. Nassauer (1995) believes that even though this evidence may be largely unstated and unrecognised, it imposes itself subconsciously onto restoration designers and managers. However, this means that river reconstruction may not be based on objective ecological restoration goals alone. There is also a need for restoration projects to obtain a better understanding of the historical nature of the river's hydromorphology and its influences on ecology, in order to recreate a self-maintaining river.

The term hydromorphology was first used in the European Water Framework Directive (WFD) to refer to the combination of a rivers hydrology and geomorphology (European Commission, 2000) and is now rapidly becoming a prominent descriptor of a rivers habitat (Newson & Large, 2006). The hydromorphology of a river is not one of the key assessment criteria with the WFD. However it is considered as a reason why failure may occur and why a river may not achieve a 'high ecological status' (Raven et al., 2010). Hydromorphology is being viewed as an essential element of river conservation due to it providing a template of physical habitat, upon which all ecological structures and functions are based (Vaughan & Ormerod, 2010). A greater understanding of the relationships and the role that 'eco-hydromorphology' (the integration of ecology with hydromorphology) plays is essential for river restoration to ensure the current aims of the WFD are met, however this is still beyond current scientific practice (Boon et al., 2010; Vaughan & Ormerod, 2010).

River rehabilitation projects have been undertaken widely throughout the UK and many techniques have been employed to enhance a rivers physical, hydraulic and ecological environment (Bannister *et al*., 2005). The importance of pool-riffle sequences has been widely recognised and accepted, and is increasingly becoming the standard form of river habitat enhancement (Sear & Newson, 2004). Riffles and pools reflect the hydraulic forces acting upon the stream bed and are the result of complex interactions between the river's sediment and hydrology (Harrison et al., 2004). If artificial riffles are constructed correctly, they are believed to help improve and enhance riverine habitats and increase floral and faunal biodiversity (Radspinner et al., 2010). A number of rivers in the UK have been subjected to the construction of artificial riffles within their channels and yet until recently there have been few detailed studies on the geomorphological and biological consequences of these projects and therefore their success cannot be readily assessed (Vaughan et al., 2007). Ebrahimnezhad and Harper (1997) conducted a study of artificial riffles constructed in the channelized Harper's Brook (Northamptonshire, England) via the assessment of the macroinvertebrate community. They found that the effectiveness of the artificial riffles in increasing the abundance and diversity of macroinvertebrates was similar to those of the natural riffles. Their investigations also demonstrated that through the installation of riffles and pools, the physical habitat diversity of the study reach was also increased and resulted in a wider variety of water depths, velocities and substrate heterogeneity. They therefore supported and advocated this habitat restoration/enhancing technique. In contrast, Pretty *et al.* (2003) and Harrison *et al.* (2004) found that the construction of artificial riffles in number of lowland UK rivers, provided little evidence of the improvement in the conservation value (abundance, species richness and diversity) of instream fish and macroinvertebrate assemblages, but did increase flow heterogeneity. They suggested that this was because the artificial riffles were primarily created in low gradient systems that did not have the same dynamic nature as high gradient systems and therefore did not have the hydraulic power to transport the larger gravel particles found on the riffle. Overall, if the fluvial geomorphology of the river is understood, if there is historical evidence that the river has previously supported riffle environments and if the artificial riffles are correctly installed, then the addition of this alternative riverine environment will only help to increase the ecohydromorphology diversity and improve the rivers WFD status.

A strong active riffle-pool bedform is characteristic of high gradient rivers with high stream power and an adequate supply of coarse sediment (Brookes, 1990; Harrison *et al.,* 2004). This information needs to be taken into consideration during the planning and designing of the restoration schemes. The design procedure used for the reinstatement of pools and riffles depends upon whether the channel is lined or unlined, and whether or not additional material such as gravel will be added to the channel in order to create the bedforms (Harper et al., 1995). For unlined channels, an average of 5 to 7 channel widths has been found to be sufficient to imitate natural conditions, however the spacing of the pool-riffle sequence can range from 3 to 10 channel widths (Brookes & Shields Jr, 1996; Caamaño et al., 2012). Experience shows that excessively large pools may become a trap for silts and sediments enhancing the problems associated with high sedimentation rates (Jones et al., 2011). Therefore in order to prevent this problem from occurring, pools should have a minimum low-water depth of 0.3m and riffles should not exceed 0.3-0.5m in their projection from the river bed (Clarke et al., 2003). However, when using these dimensions within a river, caution needs to be exercised in order to avoid over-rigid application of this geomorphological spacing of the riffles and pools; as suggested by Brookes and Sheilds Jr, (1996). Sear & Newson (2004) demonstrated that correctly installed gravel bedforms increased the habitat diversity; however, their results also indicated that there was a need to apply more rigorous performance criteria when designing artificial riffles to ensure they are self maintaining, as failure to do so may lead to higher water elevations at increased discharges, thus increasing flood risk (Harrison & Keller, 2007).

Many physical modifications have caused channels to undergo channel incision, experience high nutrient and sediment inputs and loss of aquatic flora and fauna (Lester & Boulton, 2008). Introduction of wood into rivers and streams has helped to stabilise riverbeds and banks, increase biodiversity and increase the sediment storage capacity of a reach. Despite these improvements and the capabilities of this technique, Lester and Boulton (2008) believe that the reintroduction of wood should only be a short term transitory step that would subsequently lead the way for a larger, long term restoration solutions. This is because it is not a self-sustaining strategy and over time the wood will decay, be eroded and lost downstream. Another widely accepted restoration technique is the removal of dams and weirs. Removal of instream structures and barriers allows for uninterrupted biotic exchange along the river system (Hart & Poff, 2002; Verdonschot et al., 2012). The physical change also reduces the amount of fine sediment stored within the reach and encourages a more natural temperature regime, plant colonisation and longitudinal ecological connectivity (Stanley & Doyle, 2003; Jahnig et al., 2010).

The decision of how and why to restore a river is fundamentally linked to prevailing societal values and perceptions of nature (Wohl et al., 2005). As such, reasons for restoration can be expected to change according to the way in which anthropogenic values change (Newson & Clark, 2008). From early restoration projects primarily focusing on fisheries, there has been a shift to more widely ecologically driven restoration (Palmer et al., 2005). Early habitat enhancement projects tended to concentrate on improving habitat for one species (Clarke et al., 2003), but there has been a move towards restoration projects focused on improving biodiversity to safeguard the ecological goods, services and functions that rivers provide (Ormerod, 2004; Linke et al., 2011). River restoration aims are now multifunctional, however there is still inadequate knowledge regarding landscape level processes and the riverine environment, causing a large number of river restoration projects to fail (Mika et al., 2010). The universal assumption that by increasing a rivers biodiversity, the ecosystem functioning should also increase, is very relevant to restoration, and the majority of river restoration projects are undertaken with this in mind, despite it being largely unquantified (Lake, 2001; Craig et al., 2008). Restoration must seek to reinstate habitat heterogeneity as well as considering a range of natural spatial scales, ensuring improvements at both the catchment-reach and sub-reach scales are viable (Palmer *et al.,* 2010). The connectivity of rivers means that rivers encompass longitudinal, vertical and lateral dimensions through which the flow of water and sediment occur. Therefore, although most restoration projects occur at the reach scale, the effects of these projects can potentially cause changes to occur throughout the catchment (Kondolf, 2006). Thus, restored channels will only be fully self sustaining when undertaken within a catchment context, demonstrating the need for thorough monitoring throughout the restoration process and post appraisal checks; since there is no guarantee that restoration measures will be impact free (Clarke et al., 2003; Hering et al., 2010).

2.3 Ecological theory and river restoration

Despite efforts to restore degraded freshwater habitats, many are still subjected to pressures, causing further degradation. Many believe that these pressures are, in part, due to the insufficient role that ecological science has played in shaping restoration efforts (Palmer et al., 2005; Song & Frostell, 2012). Undertaking ecological restoration tests the feasibility of recreating complex ecosystems from degraded states, which presents a major challenge to ecological science (Jansson et al., 2007). Progress towards defining ecologically successful restoration programmes is constrained by the lack of common science-based frameworks that integrate both physical and biological processes with an accurate understanding of ecosystem dynamics (Petts et al., 2006; Naiman et al., 2012). With the increase in the number of river restoration schemes, largely due to the WFD, there is a need to move away from 'trial and error' approaches and instead move towards enhancing natural riverine processes. Subsequently embracing more coherent approaches where restoration combined with sound ecological principles and evidence, can be developed as a respected and proven scientific discipline (Cairns, 1991; Boavida et al., 2012).

In some form or other, ecological restoration has been practiced for decades. It is seen as an attempt to return the system to a previous (historical) state, although the difficulty of achieving this goal is widely recognised (Palmer et al., 2010). Instead, a more realistic (alternative) goal may be to return a damaged system back to an ecologically less damaged state, thus attempting to encourage recovery of a natural range of ecosystem functions and conditions (Jähnig et al., 2011). Restoration ecology is now seen to be the science behind the practice of ecological restoration and can be considered radical in scientific terms due to its challenge to traditional 'hard engineering' solutions to river restoration (Bernhardt & Palmer, 2007). Ecologically successful restoration should incorporate measurable changes in physiochemical and biological components within the river (Sundermann et al., 2011). Attributes of success include improved water clarity and quality, the re-establishment of aquatic flora and fauna historically excluded, and an increased ecosystem resilience so that the river has a better capacity to recover following perturbation and therefore, requires minimal on-going anthropogenic intervention (Norris & Thoms, 1999). However, it is essential for restoration practitioners to recognise that there is no universally applicable restoration endpoint due to regional differences in land use, climate, geology and species distribution (Bernhardt & Palmer, 2007). The unpredictable nature of the environment can cause restoration to follow multiple pathways, thus creating more difficulties when predicting the outcomes of projects.

Due to the recognised growing importance of restoration that has arisen from attempts to address the problems from our ‘misuse’ of freshwater habitats and resources, restoration has attracted large financial investment in recent years (Vaughan et al., 2007). However, there is still little or no consensus as to what constitutes successful ecological restoration even though there have been many schemes and attempts at rehabilitation/restoration (approximately 1200 in the UK alone) (Jansson et al., 2007). The success of restoration depends on setting appropriate objectives and the subsequent use of suitable criteria to help evaluate their outcomes (Cairns & Heckman, 1996). Therefore in order for this to occur, restoration requires an accurate understanding of ecosystem dynamics (Papworth et al., 2009; Pottier et al., 2009). Young et al. (2001) believe that two of the most relevant ecological concepts for ecological restoration are community succession (predictable turnover of species composition towards a pre-disturbance state) and community assembly rules (formation of communities after a site is cleared of species, which is determined by random variation colonisation). This concept relies upon the idea that the community structure can be predicted from knowledge of organisms’ traits, such as a species response to climate, providing useful ecosystem characteristics for evaluating the success of restoration attempts on riverine ecosystems (Ehrenfeld & Toth, 1997).

Ecological assessment and continued monitoring of restored environments is essential, not only for management purposes but also to help improve our understanding of how ecosystems function (Bradshaw, 1993). Palmer *et al.* (2005) believes that the ecological success of restoration activities depends on measurable changes in the restored river that move towards a desired endpoint, such as better water clarity. These desired endpoints are often defined within a community ecology perspective, using ‘reference conditions’ to classify the original biological community, which helps to underpin the WFD (Bernhardt & Palmer, 2011). Bernhardt & Palmer (2011) promote the adoption of standards for ecologically successful river restoration through proposing five assessment criteria:

1. In the design stages of the restoration project there should be a known specified ‘guiding image’ for a dynamic endpoint,
2. The ecological conditions of the river must be measurably enhanced,
3. The river ecosystem must become more self-sustaining thus increasing its adaptive capacity,

4. During restoration no lasting harm should be caused to the ecosystem,
5. Both pre and post-project assessment need to be conducted and the information must be shared to allow other river managers and ecologists to learn from the successes or failures (Palmer *et al.* 2005, pg. 214).

Jansson *et al.* (2005) among others strongly support the criteria proposed by Palmer *et al.* (2005) and share their desire to see more robust ecological assessments of restoration projects. It is believed by Jansson *et al.* (2005) that the criteria will help to clarify what constitutes a successful restoration project. However, measuring self-sustainability and resilience following restoration could potentially be problematic as it would require long-term data, including measurements from before the restoration commenced to judge the scale of improvement (Jansson *et al.,* 2005). In order to address this it has been suggested that strengthening the framework of restoration projects through the formulation of a conceptual model and the advancement of a sixth criterion; that ecological mechanisms help underpin restoration activities. Research undertaken by both Hering *et al.* (2013) and Pander & Geist (2013) suggests that, until there is consistency in ecological assessments of river restoration projects, opportunities to further understand their outcomes and to address the major gaps in our scientific knowledge regarding ecological restoration will continue to be lost and will constrain scientific progress.

Even though the benefits of restoration are widely accepted, it is still perceived by many conservationists and economists as a delusion and a waste of money (Aronson et al., 2006). Opponents of ecological restoration argue that restoration is using up the already limited funding for conservation and rural development. Instead the funding should be concentrated on sustainable development and preservation, and not on expensive restoration programmes that take too long to have any significant influence on economic development programmes (Moore et al., 2003; Mitsch, 2012; Convertino et al., 2013). However, conserving what is currently left of our natural environment and improving the degraded state of some ecosystems is seen as a sound investment (Eden & Tunstall, 2006); not only because restoration is complementary to conservation and sustainable development, but it also generates jobs, thus improving livelihoods and the economy (Gilvear et al., 2013) . It is also a necessity due to the relationship between human society and natural ecosystems not being mutualistic. It has also been suggested that we should have an ethical responsibility for the well being of other life forms in riverine ecosystems (Bernhardt & Palmer, 2011).

2.4 Legislative Drivers in River Restoration

A major challenge facing twenty-first century river managers is to try and balance the ever increasing anthropogenic water resource requirements with those needed to sustain riverine ecosystems (Naiman et al., 2002; Raven, 2011). The fears over failure to meet these demands coupled with the threats of large-scale flooding and the uncertainties brought about via climate change have become a high priority on a global basis within political agendas. This environmental awareness increased throughout the twenty-first century, subsequently providing political space for the introduction of a range of legislation enabling river restoration to take place (Wharton & Gilvear, 2006). Changes in both the UK and EU legislation fuelled the development and enforcement of a number of European Directives that called for an integrated approach to the protection, improvement and sustainable use of freshwaters including rivers. These key policy drivers relating to river restoration are summarised in Table 2.2.

The UK has experienced a large increase in the number of river restoration schemes. In 1992 the European Community adopted the Habitats Directive (European Commission, 1992) and this statutory development sought to promote protection and conservation of habitats and species considered to be important in a European context (Clarke et al., 2003). In the UK, the Directive passed into law in 1994 and saw rivers that were previously designated as Special Areas of Conservation (SACs), become future candidates for restoration. Through this directive, recognition was gained that restoration and subsequent conservation management should rely on self-maintenance of specific physical and ecosystem processes (Clarke et al., 2003; Newson & Large, 2006).

Since 1992 there has been a growing recognition that restoration principles needed to be developed based on scientific principles and founded upon an understanding of the complex interactions between the ecological, physical and chemical components within rivers. Based on these principles the Water Framework Directive (WFD) was developed (Community of European Community, 2000) and represents a radical approach to the management of water resources and aquatic ecosystems across Europe.

Table 2.2 Key policy drivers relating to river restoration in the UK. Adapted from England et al. (2008) and Mainstone & Holmes (2010).

Act/Report	**Year**	**Activity**
Wildlife and Conservation Act	1981	The need to 'further and promote the conservation and enhancement of natural beauty'.
Statutory Instruments 1199 (Town and Country Planning) and 1217 (Land Drainage Improvement Works)	1988	Environmental Assessment required if developments likely to significantly impact river environment.
Water Act	1989	Created the National Rivers Authority (NRA). Required to identify opportunities to increase catchment water retention and storage.
Water Resources Act	1991	First piece of legislation allowing conservation and enhancement as goals in their own right.
EU Habitats Directive	1992	The main aim of is to promote the maintenance of biodiversity by maintaining or restoring natural habitats and wild species listed on the Annexes to the Directive at a favourable conservation status.
Environment Act	1995	NRA merged with Pollution and Waste Regulation Authorities to create the Environment Agency of England and Wales, charged with achieving Sustainable Development.
ICE Commission on Flood Risk Management in England and Wales	2001	Identified catchment-based measures to reduce flood risk. This saw the beginning of recommendations for a soft approach to flood risk management.
Transposition of WFD into national legislation	2003	Requirement of 'good ecological status/potential' in all designated surface water bodies by 2015, with time derogations allowed on an exceptional basis until 2027.
Making Space for Water	2004	Holistic approach to flood management, including methods to increase temporary flood water storage.

This ground-breaking Directive integrates water resource management and ecosystem conservation, providing the greatest potential to radically increase the number of river restoration projects due to the central aim of achieving 'Good Ecological Status' (GES) or 'Good Ecological Potential' (GEP) in heavily modified water bodies by 2015 (Hatton-Ellis, 2008), or to develop a programme of measures to attempt to achieve the appropriate status thereafter (Commission of European Community, 2000). In order to reach these targets, the principle aims of the Directive are to:

- Promote sustainable water use based on long term protection of water resources;
- Reduce pollution output through reducing discharges and emissions and prevent further pollution;
- Mitigate the effects of extreme hydrological events such as flooding and droughts; and
- Prevent further deterioration, protect and enhance the status of aquatic ecosystems (Commission of European Community, 2000).

In order for these aims to be delivered they need to be supported by an advanced ecological understanding of rivers and to consider riverine issues at the scale of the river basin instead of administrative or political boundaries (Petts et al., 1995; Arthington et al., 2010). Ecological classification serves as a basis for river management and, through prescribing river assessments based on ecological typologies using biological reference conditions from each river, a starting point for restoration schemes (Adrianssens et al., 2006).

The adoption of the WFD is widely considered to represent a shift in the way that water bodies are managed (Dudgeon et al., 2006). It developed from the realisation that there is a need to develop an integrated catchment scale approach to managing rivers and their floodplains. Catchment scale plans for the restoration and protection of riverine ecosystems needs to be supported by new scientific research; for example Lake et al. (2007) sought to identify ecological theories to help enhance the scientific underpinnings of river restoration. Drawing on a wider range of academic disciplines (beyond river engineering and water resource management) and refocusing on larger spatial and longer temporal scales will also help in the development of catchment focused management techniques and policies.

Following the ratification of the WFD the UK Department for Environment, Food and Rural Affairs (DEFRA) commissioned a cross government consultation programme 'Making Space for Water' (DEFRA, 2004). This strategy proposed a more holistic approach to managing flood risks in England. It is anticipated that this strategy will deliver greater environmental, social and economic benefits and at the same time be consistent with the Governments sustainable development principles (DEFRA, 2004). By considering the effects that climate change may bring, river restoration became an integral mechanism within risk management for improving resistance and resilience of both society and riverine ecosystems (England et al., 2008).

A further key driver, providing impetus for river restoration schemes to be undertaken is the Higher Level Stewardship (HLS) Scheme. This DEFRA scheme was introduced in 2005 and aimed to deliver significant environmental benefits to high priority areas through providing advice and support to environmental management projects (DEFRA, 2005). The scheme encourages effective environmental management across a range of farmland, providing funding to farmers and other land managers in the UK. However, a key challenge facing this scheme and others associated with restoration is the development of a suitable method of monitoring the wide range of interacting physical, chemical, geomorphic and ecological parameters that will all be playing a part in the shaping of the river system at any one point in time (Clarke et al., 2003)

The UK Government Agencies that are responsible for river conservation have been criticised historically due to the lack of attention paid to the protection of sites with special designations for wildlife (Mainstone, 2008). In response to this, conservation agencies, such as Natural England and the Environment Agency, created UK level guidance (Common Standards Guidance) for setting structured conservation objectives. Common Standards for conserving river habitats contain a range of abiotic and biological targets aimed at ensuring the creation of an environment that allows characteristic biological communities of a river channel to thrive (Mainstone et al., 2011; Mainstone et al., 2012). This approach looks at creating habitat-based objectives to guarantee that river management is focused on the conservation of the whole biological community and discourages 'habitat gardening' for individual species by focusing on the wider ecosystem (JNCC, 2005). However, each conservation agency is responsible for implementing the Common Standards in its respective part of the UK (Mainstone et al., 2011), which may cause difficulties in comparing overall outcomes of schemes if interpretations are different. Taking these advancements into consideration there is still a considerable lack of suitable, scientifically proven tools that river managers can employ to facilitate river restoration (Moss, 2004). More time and money are required to develop the tools that will subsequently help in successfully improving rivers that are currently degraded and yet ultimately need to achieve ‘Good Ecological Status’.

2.5 Palaeoecology – A Historical Perspective

Contemporary landscapes and ecosystems are the result of a range of natural and anthropogenic developments that have operated at a range of different timescales (Roberts, 1998). Palaeoecology is the study of an ecosystems history and provides a means to understand these unique, temporal patterns, potential drivers and rates of ecological change, thus providing an insight into past environments (Anderson et al., 2006). Traditionally, palaeoecological records provided a perspective on the timing and extent of the impacts caused by human activity (Willard & Cronin, 2007). However as ecosystems have become increasingly stressed and modified globally, palaeoecological studies are now more influential within international government and academic circles (Andersen et al., 2004; Saunders & Taffs, 2009). There is now a far greater need to understand and learn from past changes not only due to an increased public appreciation and awareness of environmental issues but in order to be better prepared for present and future environmental change (Rull, 2012).

Palaeoecological approaches help to increase understanding of the magnitude of landscape and ecosystem dynamism and also provides a frame of reference using proxy techniques, to assess modern processes and patterns (Swetnam et al., 1999). Written historical records are often brief (from 20 to 100 plus years) and only provide fragments of information regarding ecosystem change. Therefore plant and animal remains (fossils and sub-fossils) are helping to supplement the mapping of past environments. Knowledge regarding the factors that influence the abundance and distribution of modern organisms aids the interpretation of the proxies found in the fossil record (Lowe & Walker, 1997). Investigations involving environmental reconstructions using palaeoecology therefore requires an analysis of the physical attributes that have contributed towards the development of the record (e.g. taphonomy and sedimentology), combined with an interpretation of the modern environment represented by its biological content (Saunders & Taffs, 2009).

In order to obtain valid interpretations of past environments, certain assumptions need to be made about former plant and animal distributions. First, that environmental factors that govern the contemporary distributions of the flora and fauna populations are fully understood and that they are, and were, historically in equilibrium with their environmental controls (Pardo et al., 2012). Second, it is assumed that the fossil populations found have analogies in the modern biota whose ecological affinities have not changed through time (Whitehouse et al., 2008). Third, the origin of the deposit is known and unbiased and finally, the fossilized assemblage can be identified to a meaningful taxonomic resolution (Roberts, 1998). However, the validity of these assumptions will vary according to the type of fossil evidence under investigation. In reality organisms do not exist in isolation, therefore multi-proxy approaches help to increase the understanding of the inter-relationships not only between proxy populations but also with their past physical environments (Birks & Birks, 1980; Jackson & Hobbs, 2009).

Another method that has been widely used as a standard tool in palaeoecology is that of the transfer function. Over the last two decades transfer functions have been used to reconstruct a wide variety of biological variables. They have predominantly been used in lacustrine environments, however more recently have been applied lotic river environments (Sayer et al., 2010). This method of past reconstruction involves two stages: (1) calibrating surface sediment assemblages against modern environmental data and (2) determining the tolerances and optima of the taxa of interest. This data then provides an understanding of the changes that have taken place within the fossil population based on sediment cores (Sayer et al., 2010). However, problems and limitations have arisen that are associated with the use of transfer functions which relate to the complex ways in which species are connected to each other and the environment (Birks & Birks, 2006; Brodersen et al., 2008). Transfer functions that have been developed for palaeolimnology are known to exclude important, influential variables that effect community composition. An example of this is the research undertaken by Brooks et al. (2001) and Langdon et al. (2006), which saw the development of Chironomid calibration sets for a number of shallow lakes focused on physio-chemical variables such as dissolved oxygen and chlorophyll-a, however it ignored factors such as macrophyte abundance despite their known importance as food sources and habitat for macroinvertebrates. Sayer et al. (2010) suggests that palaeoecologists needs to rely less on transfer functions and instead consider more fully the mechanisms via which species, and subsequently fossils, are interconnected to other species and in turn how they can influence the environment. The integration of palaeoecology with contemporary studies will help with the interpretation of sediment core data and provide insights into changing ecological processes and patterns.

2.6 The use of Palaeoecology for Restoring Riverine Environments

Research centred on palaeoecological approaches provide the ability to interpret past environmental conditions and identify when natural thresholds may have been exceeded in the past (Murray, 2000). An integrated reflection of conditions can be pieced together to provide the temporal perspective required to identify a realistic 'reference condition' for environmental management. Additionally, in order to facilitate the understanding of how aquatic ecosystems have changed Pauly (1995) coined the term 'Shifty Baseline Syndrome' or SBS. SBS refers to changing human perceptions of biological systems due to loss of experience about past conditions (Papworth et al., 2009). Perceptions of the extent and causes of degradation can been affected due to SBS, which can subsequently influence targets set for river restoration (Pitcher, 2001). These false impressions result in the underestimation of ecosystem functions that freshwater populations historically performed (Humphries and Winemiller, 2009). Pauly (1995) found that in order for restoration goals to be reached past baselines must be identified and used to overcome SBS. The historical insights provided by palaeoecological methods can help provide the missing perspective that is key when designing river restoration projects.

Palaeoecological techniques have been extensively used within limnological studies (Anderson et al., 2006; Birks & Birks, 2006) and until recently these techniques were under-utilized within lotic environments due the dynamic nature of riverine environments. The potential benefits of applying of these techniques within riverine environments have become more apparent when developing management strategies, for river restoration studies. The use of palaeoecological approaches for defining reference conditions in lakes for management, are outlined in the European Union Water Framework Directive (European Union, 2000) and the United States Environment Protection Agency guidelines (USEPA, 2006), however not for rivers. The application of palaeoecological techniques in a management context has not always been accepted and was originally criticised as the methods were considered to be too qualitative, inaccurate, expensive and required too much individual expertise (Wills & Birks, 2006). Many policy guidelines rely on defining targets which require quantitative data assessments, therefore advances in palaeoecological research has facilitated the development of reconstruction techniques, such as transfer functions (Anderson et al., 2006). Palaeoecology and long term historical datasets continue to play an increasingly important role in conservation practice and policy, and the growing recognition of the benefits it can bring to the management of natural resources is becoming more widely acknowledged (Jackson, 2012).

River restoration targets were often based on routine monitoring data and modelling simulations, providing only a short-term response to a disturbance event or environmental variability (Andersen et al., 2004). Palaeoecological analyses now offers a way of extending the record of environmental observations through the use of biological or environmental proxies. This approach is essential for the development of adaptive management strategies. For instance, the long term (hundreds to thousands of years) perspective provided by palaeoecological techniques can provide environmental managers with the advantage of anticipating less predictable climate change (Willard & Cronin, 2007). However, even with these advances in palaeoecological and palaeoenvironmental techniques, attempts to restore rivers and their floodplains are still primarily based on inadequate monitoring data (Brown, 2002). In order to be successful and have longevity, riverine restoration schemes need to have an ecological basis rather than a 'cosmetic guestimate' (Brown, 2007). These needs can be addressed by employing palaeoecological methods to create baseline models, which can provide a reconstruction of the rivers past 'natural'/unimpacted state and act as a restoration goal (Davis et al., 2007). However in order for these templates of baseline conditions to be defined, fundamental questions need to be addressed such as: (1) What is the natural state?; (2) How can it be defined and modelled?; and (3) Can this state still be realistically created and sustained by natural processes today? (Millar & Woolfenden, 1999). Answering these questions is not only essential to ensure effective restoration, conservation and sustainable use of rivers and their catchments, but also to define goals and measures to evaluate the success of restoration schemes. Reference/baseline conditions help provide river managers with a greater sense of predictability of the restoration outcome as managers choose the level of physical and ecological variability to return a river to (Hughes et al., 2005).

Floodplains are distinctive, ecologically complex environments that are frequently subjected to change from both natural and anthropogenic sources (Davis et al., 2007). For example, sedimentation in riverine environments is highly episodic, and depositional units may vary in terms of extent and thickness due to the variation in the magnitude of sediment transport or depositional events. In addition, the sedimentary record of past/historic rivers may be biased by the erosion of pre-existing deposits (Brown, 2002). River channel changes may reflect high magnitude floods and in particular human channel management activities including dredging, channel straightening and widening. Due to potential episodic reworking of riverine sediments and channel migration, palaeochannels typically represent relatively limited periods of time (<500 years) compared to lake chronologies. However despite these limitations, fluvial sediments provide an archive of aquatic and terrestrial environmental change when evidence of short-term change is required (Lewin et al., 2005). Data for these records can be extracted from palaeochannels produced as a river moves across its floodplain by slower migration or relatively sudden avulsion (Figure 2.1). Avulsion has an important effect on the architecture of fluvial deposits as it is a primary control on channel location on a floodplain. Most avulsions occur when a triggering event, commonly a flood, forces a river across a stability threshold (Jones and Schumm, 2009). A river channel may become abandoned, isolated or cut off as a result of a range of processes (such as sediment accumulation forming a seal at each end of an abandoned reach) and anthropogenic straightening (Macklin & Lewin, 2008). Following isolation and cut off, many form backwaters which can persist for significant periods, but subsequently the channel is subjected to sedimentation and finally terrestrialisation (Greenwood et al., 2006). The sedimentary material contained within them are rich archives of environmental change and can contain sub-fossilized organic remains, such as those of Trichoptera, Coleoptera and Gastropoda, potentially providing an insight into a river's evolutionary history (Briant et al., 2005). Sub-fossil remains refer to any remains of a once living organism in which the fossilisation process is not complete, either due to a lack of time or because the conditions in which they were buried were not optimal for fossilisation (Howard et al., 2009). However over the years these natural archives may be subjected to filtering of past environmental information through physical and biological process. For instance periods of scour and extensive bioturbation (re-working of the sediment) can cause disturbance of the vertical sequencing of age zones or remove extensive parts of the record completely (Tockner & Stanford, 2002). These complexities can affect the palaeochannels potential for recording environmental signals as well as their ease of being located for sampling. Recent work performed by Howard et al. (2008) saw the use of a Lidar survey, which helped to produce a high-resolution topographic model and landform map for the River Trent Valley along which archaeological modelling was undertaken. This map provided a powerful tool to help locate palaeochannels, which could subsequently be used in the reconstruction of the rivers environmental history.

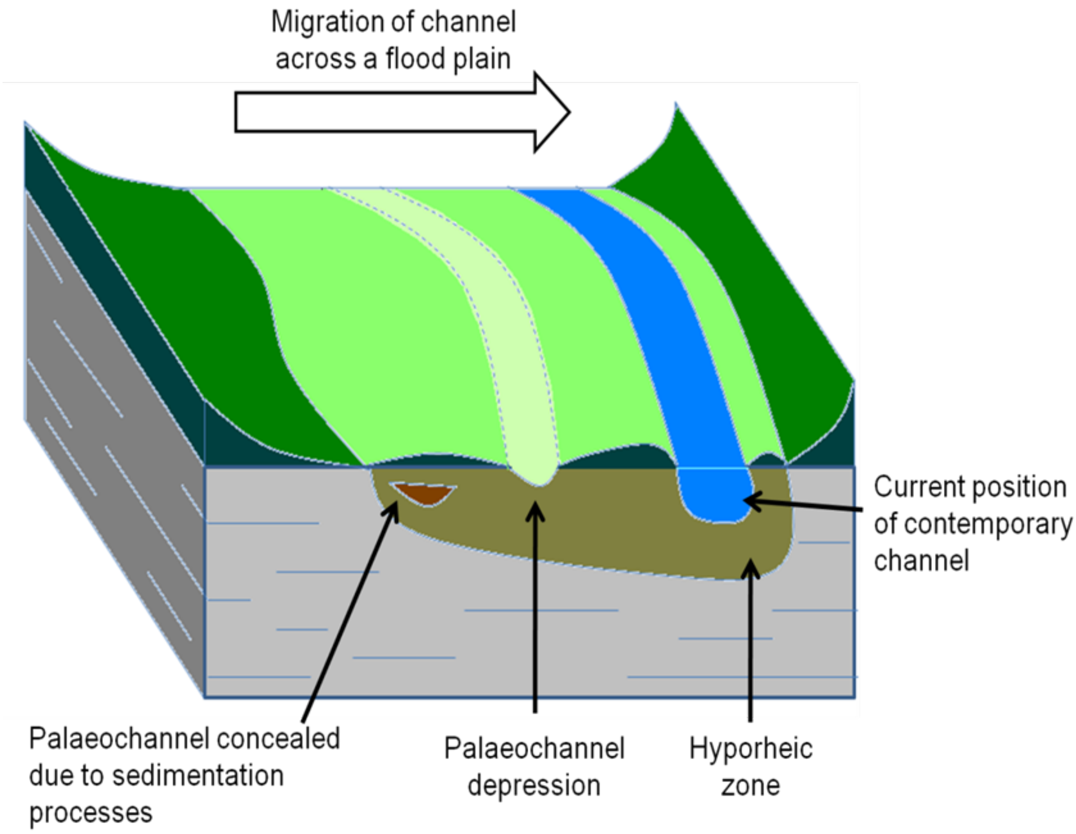

Figure 2.1 Cross section of an idealized river channel indicating its current position and relationship with the underlying alluvial aquifer, buried palaeochannel deposit and cut-off/isolated channel. Channel isolation and cut-off may be the result of channel evolution or human modification.

2.7 Sampling the Palaeoecology – A Multi-proxy Approach

Ecosystems are comprised of a network of interacting biotic and abiotic components constantly reacting to internal and external pressures (Birks & Birks, 2006; Bunn et al., 2010). These interactions can easily become unbalanced due to disturbances. When this occurs the character of the ecosystem will undergo change and in order to study and understand these past changes and the dynamics of the ecosystem, research into the changes in sub-fossil organisms (biological proxies) needs to be undertaken (Birks & Birks, 2006). Sub-fossil insects and molluscs can be found, often in high abundances, in a variety of sediments including fluvial sands and silts, peat bogs and lakes. In non-oxidizing aquatic environments disarticulated insect fragments or sclerites can be found, still displaying their original colours and delicate structures such as hairs (seatae) (Elias, 1994). This is due to the robustness of many insect chitinous exoskeletons. The sub-fossil insect record potentially facilitates the examination of past ecological processes (e.g. flow variability and the hydraulic environment) that may not be evident from other palaeoecological records, such as those based purely on plant macrofossils (Whitehouse et al., 2008).

Many historic studies have concentrated on the analysis of a single proxy such as pollen (Brown, 1999) or Coleoptera (Coope, 2000); however in order to gain a wider past perspective multi-proxy studies need to be undertaken (Bennion & Battarbee, 2007; Smol, 2002). The earliest multi-proxy studies, reviewed by Wright, (1966) and Birks & Birks (1980), utlised palaeolimnological records to examine ideas of lake ontogeny (the history of a lakes structural change) and biotic responses to internal and external processes. Although these studies contained little statistical analysis, they were able to provide insights into past limnological environments and how catchment changes affect lake dynamics (Birks & Birks, 2006). Apart from investigating ecosystem dynamics the use of multi-proxy techniques within palaeolimnology is now centred on reconstructing past climatic variability (Lotter, 2003; Sayer et al., 2010). This was exemplified by Lotter & Birks (2003) at Sagistalsee Lake in Switzerland, through a combination of abiotic proxies including pollen, plant macrofossils, Chironomids and Cladocera together with a number of physical parameters such as grain size, magnetics and geochemistry. Put together these provided an in depth Holocene environmental history of the lake.

Different proxies may reflect different environmental variables over a range of scales as they each have their own unique place in an ecosystem and their own strengths and weaknesses. If enough is known about the biological and the ecological sensitivities of a taxon, it can be used as an indicator species for the reconstruction of past environmental conditions (West et al., 1999). In addition, if an assemblage of taxa resembles a modern community that lives in a defined ecological range then this may be used to infer past conditions as well. These methods rely on modern analogies and assume that limiting conditions in the past were the same as those operating today (Birks & Birks, 1980; Birks, 2003). Through combining proxies, strengths such as the ability to provide insight into past flow variability can be exploited and weaknesses can also be identified (Mann, 2002; Linke et al., 2011). The advantages of using multi-proxy approaches are the clear reinforcement of evidence from different environmental signals to provide a more coherent set of evidence (Sayer, 1999; Saunders & Taffs, 2009). The value of this approach rests on the reliability of the proxies that have been adopted to reconstruct past environmental conditions. Ideally proxies should be complementary in terms of the information gained, and the evidence gained from each proxy should also be independently verifiable (Birks & Birks, 2006).

Birks (1998) highlighted that robust statistical techniques are needed to help distinguish between 'signal' and 'noise' effects associated with the use of fine resolution multi-proxy techniques for environmental reconstructions. Chaudhuri & Marron (1999) created a SiZer smoothing procedure which helps identify and assess which features, in a smoothed time series, are statistically significant and therefore which features represent 'signals'. This approach can be applied to palaeoecological reconstructions by considering stratigraphic records from multi-proxy studies (Korhola et al., 2000).

The majority of multi-proxy studies today are centred on palaeoecological questions in lakes; using ecological indicator species to create visions of the past and more detailed explanations of the possible underlying processes and driving factors behind change (Mann, 2002; Birks & Birks, 2006). Within this thesis the following proxies have been used: aquatic Coleoptera, Trichoptera and Gastropoda. This is due to their large abundances within palaeochannels, their easy extraction and the in-depth insights they can provide into the past riverine environments. They have been extracted from palaeochannel deposits and used in a multi-proxy line of evidence approach. This has the potential to significantly improve the characterisation of palaeochannel deposits and provide a higher resolution, quantitative environmental reconstruction of local instream palaeoecology. Each of the proxies used in this thesis are outlined below.

2.8 Coleoptera

Coleoptera (Beetles) are an order within the class Insecta, with approximately 300,000 known species worldwide (Elias, 1994). They are ubiquitous, inhabiting almost all terrestrial and aquatic environments (Lowe and Walker, 1997), with some species being specifically adapted to certain environmental niches, showing preferences for different temperature ranges (stenothermic) and substrates (Ashworth, 2001; Elias, 2006). This has meant that Coleotera has increasingly been used as a valuable indicator of biodiversity and palaeoecological conditions (Sánchez-Fernández et al., 2006).

2.8.1 The use of Coleoptera as Palaeoecological Indicators

The use of sub-fossilized insect remains found in riverine sediments has historically been centered on terrestrial Coleopteran communities due to their wide application as indicators of biodiversity (Coope, 2006). Beetles are ectotherms, making them dependent on environmental temperatures throughout their life cycle. Some species have been shown to shift their population distribution dramatically in response to changing climatic conditions (Coope, 1986); for example *Aphodius holdereri* was found in Britain during glacial intervals is now only found in the Himalayas (Coope et al. 1998). Due to the consistency of the species migrating rather than evolving, it has given them long term genetic stability and morphological consistency (Coope, 1978). This long-term genetic stability has been inferred from the highly comparable faunal assemblages seen over time and the consistency in their exoskeletal features. Coope (1978) suggested that the migration of species during times of climate change prevented populations from becoming genetically isolated long enough to allow for species evolution, therefore making them ideal candidates for palaeoecological and environmental reconstructions.

Beetle fauna have proven to be excellent indicators of riverine conditions. For example, Smith and Howard (2004) used groups of beetle fauna to characterise discharge rates of low gradient alluvial streams as Coleoptera are highly responsive to changing flow velocities. However, with the exception of Elmidae and a small number of Dytiscidae, most Coleoptera are associated with relatively slow flow velocities or still waters. This limits their practicality as indicators of contemporary and palaeohydrology on their own (Greenwood et al., 2006). Nevertheless, a study by Howard *et al.* (2009) successfully applied PalaeoLIFE to sub-fossil Coleoptera to reconstruct river flow conditions for a large palaeochannel of the River Trent. Through the use of freshwater beetles, the results provided strong evidence that a marked change in the flow environment had occurred within the river. Erwin (1997) believes that the information gained through the use of Coleoptera is critical for ecological restoration due to the interface between them and their environment being at such a small resolution. Results obtained through using beetles as biodiversity indicators can be used at both the local and regional scale and provide a rapid, inexpensive monitoring method (Sánchez-Fernández et al., 2006). This provides conservation and restoration managers with an effective tool for identifying priority areas for freshwater biodiversity conservation.

The majority of past research on sub-fossil Coleoptera has been focused on establishing a temperature scale for the Quaternary period. This has been achieved through the use of the Mutual Climatic Range method (MCR). This concept was developed in 1982 as a quantitative means of analysing palaeoclimatic information from sub-fossilized insect remains (Atkinson et al,. 1987). MCR uses temperature information for species that have known contemporary geographical ranges and are either scavengers or predators (therefore unaffected by changing vegetation) (Elias, 1998). Coleoptera are especially suited to this technique due to many species showing fairly well defined climatic tolerance ranges as well as fossil fragments being identifiable to species level. Calculation of overlapping temperature ranges of each species gives an indication of the maximum summer air temperature (T_{Max}), air temperature of the coldest month (T_{Min}) and the subsequent temperature range (T_{Range}) (Elias, 2000). This not only provides a measure of seasonality but also rapid temperature variations that are widely used in palaeoclimatic reconstructions (Coope et al., 1998; Krell & Schawaller, 2011).

The use of Coleoptera as palaeoecological proxies is possible due to adult sclerites, which include heads, pronota (thoraces) and elytra (wing cases) (see Plate 2.1) composed of chitin. Chitin is a nitrogenous polysaccharide that is insoluble in water, dilute acids and alkalis (Krell & Schawaller, 2011). This allows the sub fossilized sclerites to remain stable in sedimentary environments; however they can be broken down by bacteria if exposed to the atmosphere for an extended amount of time (Ashworth, 2001). They are easily extracted from sediment samples using simple and cost effective techniques (such as paraffin floatation) and can provide comprehensive palaeoenvironmental information (Coope, 1986). Elias (1994) described the paraffin floatation method used to isolate the fossils for their extraction and subsequent identification. This cheap and easy method relies on the adhesion of the paraffin to the beetle chitin, which subsequently floats in water. The extracted sclerites can be identified through the use of modern reference collections. They have an extensive fossil record as their remains have been studied since the late 19th Century, most notably through the work of Coope (1986), and can be readily identified to species level (Sadler et al., 2004).

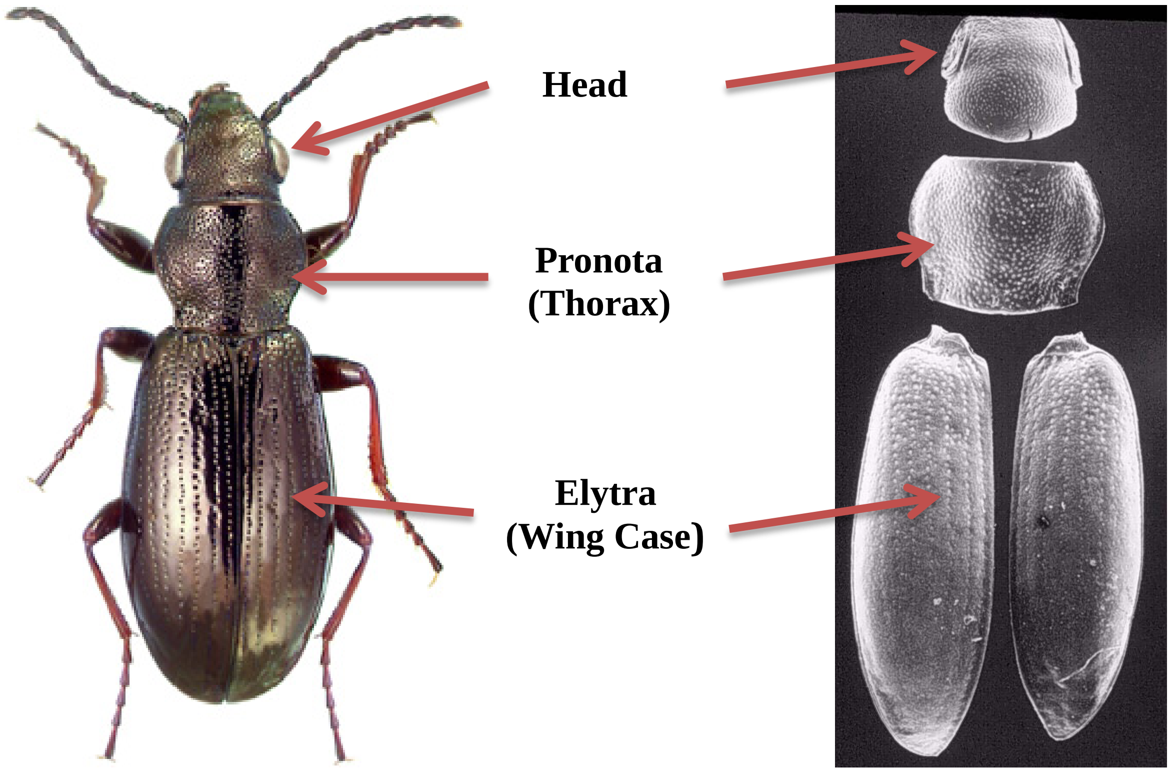

Plate 2.1 Contemporary (Left photo – Faldermann, 2012) and fossilized (Right photo - Schwert and Ashworth, 1995) comparison of *Diacheila polita.*

2.8 Trichoptera

Trichoptera (Caddisflies) are insects with a largely aquatic larval stage and are associated with almost all types of water bodies. They are one of the few orders that inhabit both standing (lentic) and flowing (lotic) waters and exploit the resources of a whole range of habitats including deep water lakes, shorelines, marshes, springs and rivers (Solem & Birks, 2000). Depending on the species/taxa, they are also able to live in eutrophic, dystrophic and oligotrophic waters and are highly adapted to specific temperatures, flow ranges and predation pressures (Feio et al., 2005). This wide diversification underlies the importance of caddisflies in freshwater biology as environmental indicators and their potential in palaeoecology (Kirill Yu et al., 2008; Extence et al., 2011). They are abundant and diverse in nature with over 9500 species having been recorded worldwide, consisting of 45 families and 600 genera (Morse, 1997). The modern distributions of Trichoptera within the UK have been reported in Wallace (1991) with 196 taxa being listed. There are two taxonomic groupings of caddisflies; families that build larval cases out of plant material and/or mineral grains and free-living families that live without cases and instead build silk nets to capture food (Greenwood *et al.*, 2003). In the UK the most common family of case building caddisflies are Limnephilidae which occur predominantly in still or slow flowing waters, whereas as caseless caddisflies such as Hydropsychidae are more generally associated with faster flowing waters (Becker, 1987).

Caddisflies have a four-stage life cycle from egg to larvae, through to pupa and finally metamorphosing into the adult form. The larval stage consists of multiple instars (usually five, with up to six or seven depending on family) and is considered to be the growth stage taking up to two years to complete (Greenwood et al., 2006). Moulting or ecdysis occurs between each instar stage as the insect grows. The head capsule and thoracic plates of each larval instar are sclerotized, therefore robust and have the ability to be preserved as sub-fossils in high abundance in aqueous, anaerobic environments, such as palaeochannels (Williams & Eyles, 1995) (Figure 2.2). The lifecycle of the caddisfly responds to two sets of conditions. Firstly the aquatic larval stage responds to conditions such as water temperature, flow variability, food availability and macroclimate changes. Secondly the adult stage which is essential for reproduction and dispersal, is highly responsive to changes in the thermal environment (Greenwood et al., 2003; Wallace, 1991). Adult caddisflies have a restricted range of dispersal and therefore usually lay their eggs on water or riparian vegetation close to water (Greenwood et al., 2006).

2.9.1 The use of Trichoptera as Palaeoecological Indicators

Caddisflies are an ecologically diverse group of aquatic insects and are readily well preserved within the riverine sediments of palaeochannels. However, unlike Coleoptera and Gastropoda, larval Trichoptera do not necessarily represent a death assemblage. This is due to their progression through their instar stages resulting in the shedding of their exoskeletons. Early work undertaken by Williams (1987; 1988) successfully saw the use of Trichoperan sub-fossil remains as indicators of instream conditions by relating the species found, by the use modern analogues, to variables such a vegetation, benthic habitat conditions and flow velocity. More recently Greenwood et al. (2003, 2006) used fossil caddisfly assemblages to reconstruct past flow and aquatic habitat conditions, to provide evidence for climatic variability. However it has been suggested that Trichoptera are still underutilised in the field of palaeoecology (Williams & Eyles, 1995; Greenwood et al., 2003), with the use of caddisflies for palaeoecological reconstructions receiving far less attention than that of Coleoptera and Chironomidae (Coope et al., 2002).

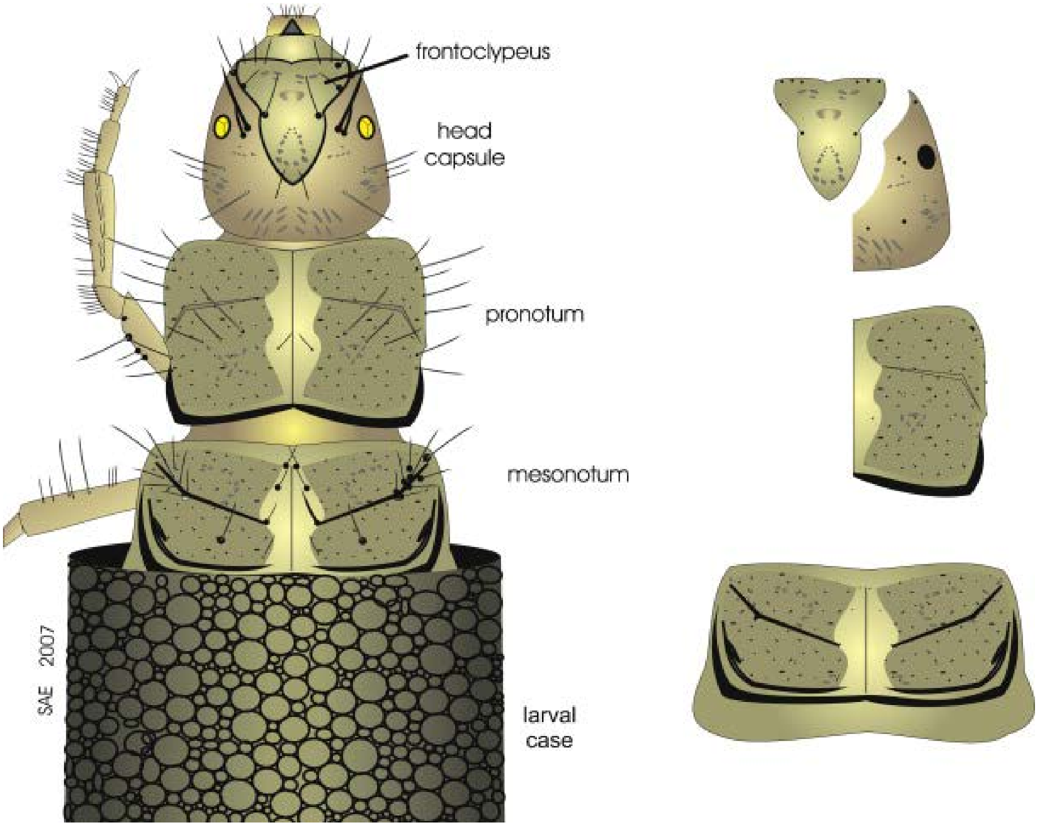

Figure 2.2 Illustration of the head and thorax of a caddisfly larvae showing the chitinous sclerites frequently preserved as fossils (Elias, 2010).

Although much of the recent research on insect-based palaeoecological reconstructions has been focused on Coleoptera and Chironomidae, Trichoptera have several advantages as palaeoecological indicators. These include their ability to act as aquatic signals due to the fact that their modern distributions and associations are well known and documented as well as their distinct sensitivities to variables such as flow, habitat characteristics (e.g.

substratum and vegetation) and water quality (Amoros et al., 1987). Trichoperan insect remains also provide indications of instream flora (Hughes, 2006). Trichoperan larvae have chitinous frontclypeal apotomes making them robust enough to be incorporated into the fossil record (Figure 2.3), however they are not resistant to erosion and if subjected to transportation, they can experience severe degradation (Williams and Eyles, 1995). This suggests that the sub-fossil recorded within this research have not been transported far. There is also evidence to demonstrate that caddisfly fauna react to seasonal changes and the associated change in flora (Greenwood et al., 2006). An additional benefit of their use is that sub-fossil extraction from sediments follows the standard procedure developed by Coope (1986) through the use of paraffin floatations. This can be undertaken at the same time simply through the use of a smaller mesh sieve. The characteristic patterning and setal pores on the frontoclypeal apotome of Trichoptera are visible on sub-fossil remains (Figure 2.3). This allows identification to be resolved to species level (in the majority of instances) by using the distinct characteristics based on both shape and size. A proposed system of setal nomenclature for Trichoptera is discussed in Wilkinson & Wiggins (1981a).

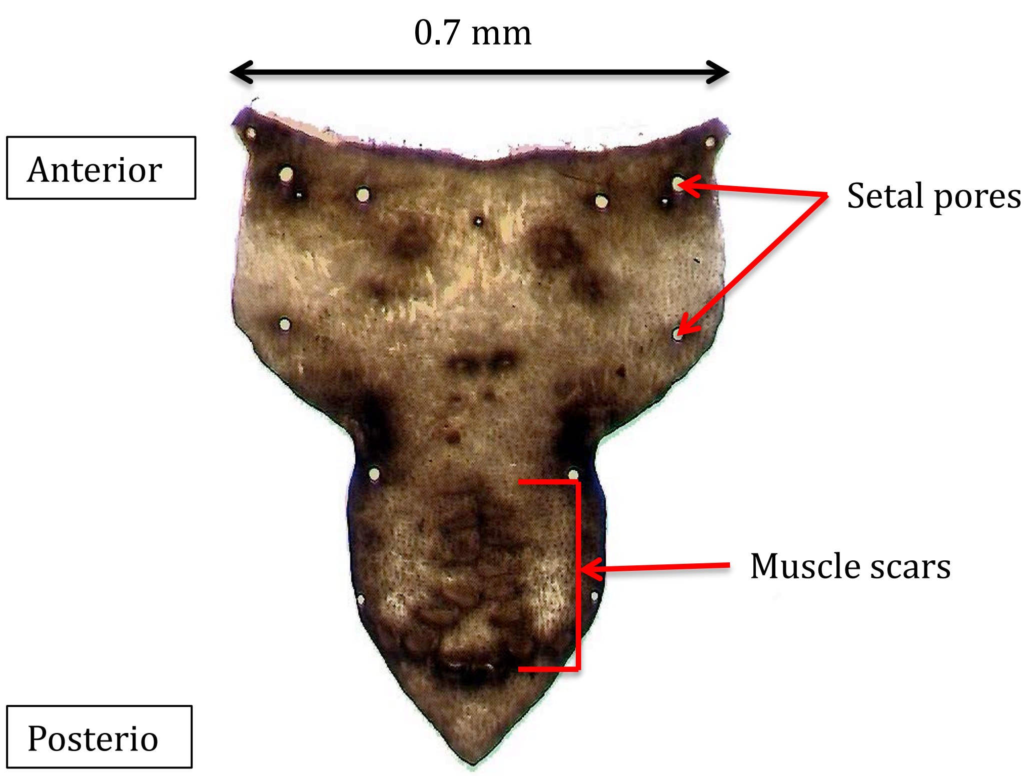

Figure 2.3 Frontoclypeal apotome of *Halesus radiatus*, extracted from sediments found in the palaeochannel of the River Wensum, Norfolk (E. Seddon).

In common with other palaeoenvironmental proxies Trichoptera have not only been used in reconstructing past riverine environments but they have also been employed to reconstruct lake ecosystems (Woodward & Shulmeister, 2005). However it has been found that sub-fossil assemblages representing lake and river ecosystems are unequally sensitive to environmental change (Williams & Eyles, 1995). Fossil assemblages found in riverine sediments reveal far more about climate change when compared to those extracted from lacustrine sediments. This is due to large lakes being thermally buffered and therefore providing relatively stable, cool water environments (Williams & Eyles, 1995).

Trichoptera offer a powerful tool for reconstructing former conditions of riverine environments. Two types of information can be derived from using Trichoptera as a palaeoenvironmental proxy. First, past aquatic habitats (substrate, flow velocity and presence of instream vegetation) can be reconstructed through knowledge of present day caddisfly ecology and second; when fossil data is combined with contemporary ecological distribution records, an insight into past macroclimates can be inferred (Greenwood et al., 2006).

2.10 Molluscs

Molluscs are a diverse group of invertebrates consisting of six classes of which riverine environments are represented by Bivalves and Gastropods. Both Bivalves and Gastropods have protective shells made of chitin, proteins and calcium carbonate and inhabit marine and freshwater environments. Bivalves are among the more diverse groups of molluscs with 9200 species and are notable for their two mirror-image shell halves (valves) (Dillon, 2000). There are four different life strategies that are displayed by bivalves: epifaunal, infaunal, boring and free moving. Epifaunal bivalves attach themselves to hard surfaces and remain in one position for the entirety of their life. Infaunal bivalves bury themselves in sands or sediments within the substrate (e.g. riverbeds). Boring bivalves bore into solid surfaces such as wood and remain there, and finally free-moving bivalves dig into sands and soft sediments and can also push themselves through water by the action of opening and closing their valves (Dillon, 2000).

Gastropods are the most diverse group of molluscs with more than 60,000 species making up 80% of all molluscs. They are highly diverse in terms of their shape, size, colour and shell morphology. Their feeding habits include grazing, predation, savaging, filter feeding and bottom feeding (Lysne et al., 2008). Both Bivalves and Gastropods are ubiquitous and are found in a variety of habitats including sinkholes, peat bogs, lacustrine and fluvial environments and the analysis of freshwater Mollusca has been firmly established within Great Britain (Ložek, 1986).

2.10.1 The use of Molluscs as Palaeoecological Indicators

Molluscs were one of the first fossil groups to be noted in scientific literature, appearing in text at the beginning of the 18th century and numerous publications in the 19th century contained large faunal lists of Molluscan taxonomies. However, there were few attempts at data interpretation (Keen, 2001). It wasn't until 1884, at Canadian archaeological sites, that Molluscan fauna studies were first used to recreate both cultural and natural environments (Miller & Bajc, 1989). Freshwater Molluscs attracted considerable attention in the early 20th century when it was thought that they were inherently bad for use as tools for dating in the classic geological sense (Keen, 2001). However this theory was disproved by research from the 1950s onwards, including that from Sparks, (1961), Sparks & West, (1972) and Kerney (1977) who found the use of quantitative sorting, counting and interpreting of Mollusc assemblages to be highly beneficial for palaeoenvironmental reconstruction.

Non marine molluscs are one of the most prominent groups of macrofossils to be found in palaeo-deposits and dominate fossil assemblages in freshwater tufas where they rival ostracods and plant macrofossil remains in their frequency (Ložek, 1986). Their preservation can be related to both biological and chemical attributes of the depositional environment. Mollusca shells can be preserved in a wide range of sediments where there is a sufficient amount of calcium carbonate and have the ability to withstand oxidation (Sparks, 1961). However, in acidic, non-calcareous sediments they are quickly leached out (Ložek, 1964).

Mollusca are usually found in great numbers within alluvial sediments, which provides easy analysis, when they are used as palaeoenvironmental proxies. However, these large assemblages can be caused by taxa that amalgamate together after death from a variety of habitats (Ložek, 1964). Once deceased, air becomes trapped within the shells of gastropods, permitting them to float long distances down stream. Therefore, species within a palaeochannel may represent a wide spectrum of aquatic and terrestrial habitats that have been brought together from upstream (Miller & Bajc, 1989). Miller and Bajc (1989) reported that shell size and shape also influenced at what point shells settled out within streams. As flow and stream size increases, more streamlined shells settle out first with smaller, lighter ones remaining in suspension, getting carried further downstream. Transport, sediment reworking and compaction from burial can cause large amounts of damage to the thin, hollow shells and due to the strength of the individual shell structure varying between species. Some species are more susceptible to corrosion than others (Keen et al., 1999), and a number of species have characteristic shells making them easily identifiable in the field (Lozek, 1986).

In the field of palaeoecology, one of the main advantages for the study of Molluscs is that they have been studied for a long period of time. Much is known about their past and present day ecology, providing comparable palaeoecological analysis opportunities. Molluscs have successfully been used as proxies in palaeoecological reconstruction in a number of different scenarios, from reconstructing former stream confluences to mapping local habitat to climate change (Beattie & Avery, 2012; Esu & Ghinassi, 2013). However it is argued by Ložek (1986) that molluscs are far from ideal indicators of climate change due to the large majority of freshwater species being able to tolerate a wide range of climates, and the fact that ecological changes which could have adverse effects on species numbers might be easily mistaken as climate change. Davies and Griffiths (2005) demonstrated the potential benefits of using ostracods alongside molluscs when interpreting environmental change. They found species to compliment each other, providing a more detailed picture of past changes. Whereas ostracods offered detailed hydrological information (Pfister et al., 2011), Molluscs provided an insight into general vegetation and habitat.

An additional method that has expanded the use of Molluscs as palaeoecological proxies is the use of isotopic data (Henderson & Price, 2012). Isotopic data obtained from the analysis of mollusc shells has provided environmental information with regards to where molluscs lived (Bemis & Geary, 1996; Demarchi et al., 2011). This combined with the relative ease with which large numbers of molluscs can be extracted from sites, guarantees that they will continue to be a fundamental component of palaeoecological reconstruction studies.

2.11 Taphonomy

Taphonomy is the science of the 'laws of burial' and is concerned with the information found in fossil records and the processes by which that record was formed (Martin, 1999). Although the term 'Taphonomy' was only coined in 1940 by Efremov, taphonomic studies have been carried out for centuries, beginning with Leonardo da Vinci (1452-1519) who used living and dead bivalves from local mountains to deduce that the Biblical flood had not transported them, but in fact they had lived and died *in situ* (Martin, 1999). Taphonomy can be divided into three areas: (1) necrolysis, referring to death/shedding and its causes (e.g. larval moulting); (2) biostratinomy, which involves the sedimentary history of the remains prior to burial including any post decay effects such as scavenging; and (3) diagenesis/lithification, comprising of physical or chemical modifications within the sediment (Martin, 1999; Martínez-Delclòs et al., 2004).

Fossil and sub-fossil records provide a rich source of palaeoenvironmental information at scales ranging from decades to centuries depending on rates of deposition (Lowe & Walker, 1997). The process of taphonomy (Figure 2.4) is a continuum of processes that are independent to one another and deals with the incorporation of organic remains into sediments or other contexts such as resin an its subsequent fate (Martínez-Delclòs et al., 2004). However, when combined they act together to filter out the 'noise' of short-term bio-fluctuations through reworking and mixing. Martin (1999) believed that this filtering process needed to be thoroughly understood to allow for unbiased interpretations to be made.

With regard to invertebrate and mollusc taphonomy, the preservation of invertebrate remains is only realistically possible where little microbial activity occurs or chemical agents are working and where rapid burial occurs within suitable anoxic conditions (Lowe & Walker, 1997). Invertebrate taphonomy studies a vital pre-requisite for the accurate reconstruction of fossil assemblages, to interpreting environmental conditions where invertebrates lived and died, and for the investigation of interactions between insects and other organisms (Martínez-Delclòs et al., 2004). However, little work has been preformed on the majority of invertebrate groups. One reason behind this is because the invertebrate carcasses that survive the biostratinomic processes are subsequently subjected to additional processes including mineralisation, flattening, deformation and reworking (Martínez-Delclòs & Martinell, 1993). This may lead to biases in the fossil invertebrate assemblages. The bulk of studies of the taphonomic representation of insects has been based on Coleoptera when associated with archaeological deposits

(Kenward, 1975; Kenward & Alison, 1994), but this however has not been extended to invertebrates in fluvial environments.

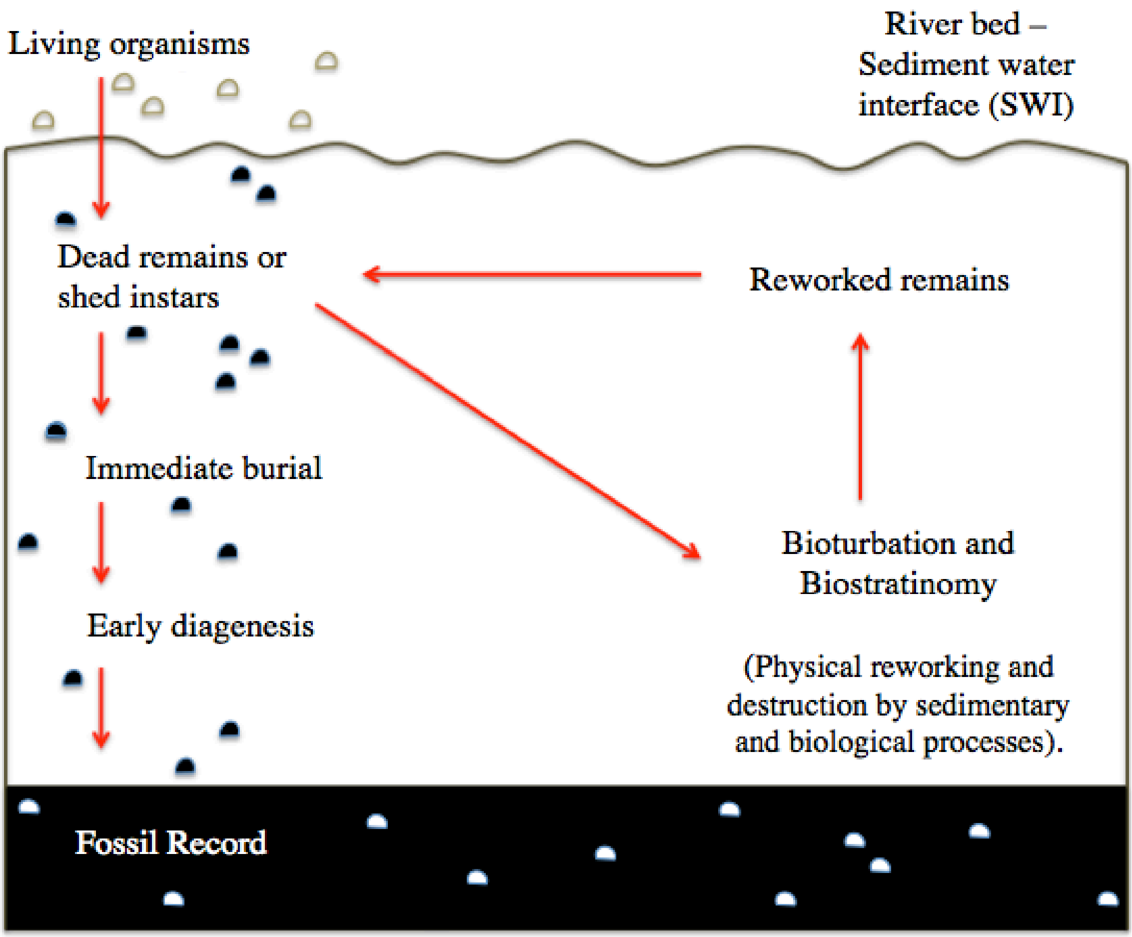

Figure 2.4 Diagram showing the process of fossilization and the interdependent and dynamic processes of taphonomy. (Based on Martin, 1999, p. 2).

A study undertaken by Rumes et al. (2005) of aquatic invertebrates within lakes found that recent death assemblages provided a more complete record of local invertebrate communities and their distribution when compared to live samples taken from a range of habitats. This has considerable bearing on the interpretation of palaeofossils, as the degree of representation of the living population by the taphonomic sample may be substantially higher again potentially resulting in a bias within the results. The environmental range also needs to be taken into consideration when interpreting taphonomic results. Fossil and sub-fossil assemblages extracted from within river channels will be a representation of wider environmental conditions (pool/riffle sequence to reach scale). However, this will be reduced in the case for oxbows or abandoned channels, as these environments would give a picture of the local ecological successions that have taken place within the local vicinity (Demko et al., 1998).

2.12 Conclusion

Anthropogenic manipulations of river form and function are not a new occurrence and rivers are now perceived as one of the most modified and degraded ecosystems (O'Donnell & Galat, 2008). Humans are by far the largest influence currently driving river system evolution, primarily due to society's dependency on their resources. Due to the degradation of river ecosystems, river restoration has now been accepted by governments worldwide as essential for the long-term conservation and management of natural resources (Dudgeon et al., 2006). River restoration tests the feasibility of recreating complex ecosystems from degraded states, thereby presenting a major challenge to ecological science.

Despite the considerable amount of river channel restoration activities that have been seen in the UK since the implementation of the WFD, there has been a growing realisation that there is a need to develop a coherent integrated catchment scale approach to managing both rivers and their floodplains. The connectivity of the rivers to their catchment which they drain encompasses longitudinal, lateral and vertical dimensions across the floodplains (Allan et al., 1997). The catchment should hence define the spatial boundaries of the river ecosystem instead of the focus being concentrated on the river channel. This will encourage future restoration practices to be developed based on integrated catchment scale information that will provide a more holistic overview for river restoration and may ultimately lead to increased rates of success for restoration schemes.

River restoration is not a very established science and is still largely disconnected from fundamental ecological concepts. The challenge for the future lies in protecting the ecological integrity and biodiversity of aquatic systems in the face of increasing pressures on our freshwater resources. Therefore, river restoration needs to have a multidisciplinary approach, taking into consideration; ecological, hydrological and geomorphological processes at all stages of the scheme. In order for this to be comprehensive, the range of spatial and temporal scales over which these variables operate and interact need to be more fully researched and understood. However, one of the most difficult aspects to restoration is knowing how much change and improvement to a river is enough. In order to gauge the answer to this question restoration managers must recognise that riverine ecosystems are dynamic and subject to naturally variable conditions and due to this, restoration success should not be viewed as a single endpoint, but rather as part of a trajectory towards attaining a more ecologically sound, self-sustaining state.

This literature review has identified the need for research into the use of palaeoecological techniques to define reference conditions for informing river restoration planning and has provided a foundation for this thesis. The research gaps identified in this chapter can be mapped onto the aims and objectives of the current research project (section 1.2). The subsequent chapters map out the methodological approaches (Chapter 3) and the results from the three study sites used (Chapters 4, 5 and 6) to address these gaps. Following description and discussion of the results obtained, key findings are integrated with existing literature (Chapter 7) and practical application of the knowledge gained is considered in the wider context (Chapter 8).

3.1 Introduction

This chapter outlines the methods adopted within this thesis, to achieve the aims and objectives of the research (Chapter 1.2). The techniques relating to data collection and its subsequent analysis are outlined. In order to achieve the aims and objectives of the thesis, a variety of field and laboratory methods, and analyses have been undertaken. Reaches on the River Eye (Melton Mowbray, Leicestershire), the River Hull (Driffield, Yorkshire) and the River Wensum (Fakenham, Norfolk), were visited and sampled to generate an understanding of the current and historic river (based on sampling palaeochannels and instream islands) and their ecological communities. The resulting data sets, and together with examination of current and historic maps, provided the basis of determining the nature of historic changes and allowed the contemporary instream macroinvertebrate community to be compared to the historic community recorded in the palaeo-deposits. This allowed the nature of changes to be quantified for the three study reaches examined.

3.2 Site Selection

Through consultation with Natural England, twenty candidate SSSI (Sites of Special Scientific Interest) rivers were reduced to six potential study sites for inclusion in this research. The candidate sites are lowland rivers in England and comprised: the River Beult (Kent), the River Eye (Leicestershire), the River Hull (East Yorkshire) and the River Wensum (Norfolk), all exhibiting straightening, impounding and widening to varying degrees; the River Mease (Staffordshire), with significantly less modification; and the River Blythe (West Midlands). As part of preliminary investigations each candidate site was visited to assess the presence of sedimentary deposits suitable for palaeoecological investigation. After a series of reconnaissance trips and consultation with site managers from Natural England, three rivers were selected for detailed examination; The River Wensum (Fakenham, Norfolk); the River Hull (Driffield, East Riding of Yorkshire) and the River Eye (Melton Mowbray, Leicestershire). Each of these three rivers had a clearly defined palaeochannel or in-stream sediment deposits, which was the key influencing factor for their selection.

The River Eye is an exceptional example of a semi-natural lowland clay stream. The River Wensum and the River Hull are both examples of lowland chalk streams and whilst England contains numerous examples of chalk rivers, they have limited distribution throughout the rest of Europe (The UK Biodiversity Steering Group, 1995). Due to their international importance, UK chalk rivers, such as the River Wensum and Hull, have been identified as a priority habitat and designated as such under the EU Habitats Directive (European Economic Community, 1992). Historically, they were extensively used to irrigate systems of water meadows and drive water mills (Berrie, 1992). Water derived from chalk aquifers is naturally high in quality and widely used for domestic supply (Hiscock et al., 2002). As a consequence they have experienced extensive water abstraction causing a reduction in flows in many chalk streams (Berrie, 1992; Mainstone, 1999). However, perennial chalk streams provide excellent conditions for salmonid fisheries and many support commercial fish farms. As a result few chalk streams can be considered to be in a natural condition (Newson and Large, 2006).

Each of the rivers selected and study sites contained one or more obvious palaeochannels or sediment deposits suitable for examination (e.g. instream island covered with mature vegetation). The knowledge provided on the history of the study sites, by local landowners and Natural England, proved extremely useful. The River Wensum was a particularly interesting study site because the palaeochannel was reattached/reinstated to the main channel as part of a Natural England and Environnment Agency restoration programme undertaken during the thesis study period. This allowed the contemporary and palaeochannel communities to be examined, in addition to allow monitoring of rates of recolonisation of the reinstated channel (see Chapter 4).

3.3 Historic Mapping

Contemporary riverine landscapes have been shaped and modified throughout history and via the use of historical sources such as maps, satellite imagery and photographs, temporal and spatial changes to river channels can be investigated (Downward, 1995; Milton et al., 1995; James et al., 2012). GIS-based approaches for handling historical maps have proved to be highly advantageous due to its ability to plot different map layers and differences in river channel length accurately (Gurnell, 1997). ArcGIS (ESRI, 2011) was used to examine historic river channels to highlight any natural and/or anthropogenic changes that occurred on each river by comparing historic Ordinance Survey maps, dated between 1849-1899, to contemporary maps dated 2012. The historic river paths were digitised along with the most recent OS maps for each area, allowing channel changes to be directly highlighted. However, through combining historic and modern maps confidence levels in final estimates of change can be limited (Zanoni et al., 2008). This is due to potential errors introduced when digitising maps and therefore this needs quantifying. The most direct source of error within this study is the accurate identification of the palaeochannels between images of variable scale and contrast.

3.4 Fieldwork Techniques

When undertaking any fieldwork it needs to be acknowledged that the sampling procedures employed may not necessarily represent the (sampled) population (Underwood, 2000). Obtaining a sample that is representative of natural community structure is extremely difficult due to natural macroinvertebrate community spatial and temporal variation (Quinn & Keough, 2002). Therefore, where possible, semi-structured sampling and sample replication is necessary in order to achieve appropriate representation and to establish levels of natural variability within a site (Blöschl & Sivapalanm, 1995). However, this can also prove difficult as semi-structured sampling has been shown to fail in detecting clumped or patchy distributions of fauna (Underwood, 1994). The standard fieldwork techniques and replication procedures described within this chapter were undertaken in order to achieve samples that provide good representation of the contemporary and palaeochannel fauna. For example, with regards to the palaeochannel sampling, replicate cores were extracted from across each palaeochannel in order to determine and quantify spatial variability.

3.4.1 Abiotic Sampling

Prior to the undertaking macroinvertebrate sampling in the field, abiotic measurements were collected. Flow velocity (ms^{-1}), depth (cm), electrical conductivity (uS), pH, temperature (^{0}C) using standard meters and substrate composition, were all recorded at each site. Flow velocity was measured using a SENSA RC-2 velocity meter (ADS Environmental Services, Huntsville, USA), which has the ability to accurately measure flow velocity from 0.000 m s^{-1} to 4.00 m s^{-1}. Velocity measurements were scanned every 2 seconds and averaged over 30 seconds. Measurements were taken at 0.6 times the water depth and were repeated three times at each site. Electrical conductivity ($\mu S\ cm^{-1}$), pH and temperature (^oC) were also logged at each sample site along each river. The conductivity was measured by using a Hanna Instrument HI-98303 Dist3 conductivity meter with automatic temperature correction (Hanna Instruments, Leighton Buzzard). Temperature and pH were both measured using a Hanna Instrument HI-98128 pH meter, which provides an accuracy resolution of 0.01 pH.

The nature of riverine sediment deposits play an important and influential role in determining the macroinvertebrates that inhabit them (Larsen et al., 2009). The substratum of a river comprises a number of individual mesohabitats/functional habitats that are recognisable from the river bank (Pardo & Armitage, 1997; Kemp et al., 1999). For instance, areas that are visually distinct based on their physical uniformity can be identified, such as pools containing leaf litter, riffles, boulder beds, exposed rock and marginal zones where high volumes of organic matter may be present (Allan, 1995). The percentage surface compositon of the substratum (cobbles, coarse gravel, fine gravel, sand, silt, woody debris and macrophytes) was visually assessed at each site using the mesohabitat methodology proposed by Pardo and Armitage (1997).

3.4.2 Biomonitors

Biomonitors are organisms that reflect the overall character or ecological health of an environmental system. The presence, absence or condition of an indicator organism or a community is now a well established tool for the assessment of water quality (Boothroyd and Stark, 2000; Solaun et al., 2013). The advantage of using biomonitors for water quality evaluation, in favour of physical and chemical tests, is that communities of living organisms provide continuous information over the period of their life cycle. Whereas physical and chemical methods provide data typically specific to the point in time the data was collected. Among the variety of organisms that can be studied in rivers, macroinvertebrates are by far the most widely used group for biological assessment in river management (Rosenberg and Resh, 1993; Hawkes, 1998; Bellucci et al., 2013).

Macroinvertebrates are diverse and represented by a range of different taxa. They display responses to a wide range of stressors, such as pollution, as each species possess well-known tolerance levels of varying sensitivity (Marzin et al., 2012). Their largely sedentary nature confines macroinvertebrates to relatively small areas geographically for a large part of their life cycle. They are therefore good indicators of local surrounding conditions providing an overview of the quality of water they have been subject to in antecedent periods (Smol, 2008).

3.4.3 Sampling the Contemporary Macroinvertebrate Fauna

Sampling the contemporary macroinvertebrate fauna inhabiting each study site provided information on general community structure and environmental quality. A wide range of macroinvertebrate sampling techniques have been developed, however sampling efficiency can vary considerably between each method. Through the comparison of sampling methods and consideration of factors such as sample size and replication, sampling techniques have been refined (Scarsbrook & Halliday, 2002). The method selected should allow a representative sample of macroinvertebrate fauna to be collected.

Using more than one sampling method on each study site potentially provided an approach to examine the entire community composition across all habitats and quantify its characteristics. Two methods were selected; i) Kick sampling, and ii) Surber sampling (see 3.4.4 and 3.4.5 for more details). Sampling was performed seasonally to capture any seasonal variability of the community coinciding with low/base, medium and high flow conditions and due to the fact that different taxa have different life cycles and emergence periods (for aquatic insects). Seasonal sampling also ensured that the entire river invertebrate fauna was characterised and any seasonal changes and trends could be considered. Table 3.1 Indicates when samples of each site was undertaken. The River Wensum was sampled more than the River Hull and Eye due to the fact that during the sampling period, the original/palaeo meander was restored/reinstated and provided an opportunity to monitor the recovery of the study reach.

Table 3.1 Seasonal sampling dates for each river site.

	River Sites		
	River Wensum	**River Hull**	**River Eye**
Sampling Dates	June 2010 November 2010 February 2011 June 2011 December 2011	March 2011 June 2011 September 2011	July 2009 May 2010 January 2011 September 2011

3.4.4 Kick Sampling

With all sampling methods it is important to maintain consistency as differences in the sampling technique may affect the density and diversity of the organisms recorded in the samples collected (Armitage et al., 1983). Kick-net samplers are the most commonly used devices in rapid bioassessment approaches in rivers (Resh & Jackson, 1993; Murray-Bligh, 1999). Advantages of the kick net approach include the ease of transport, low cost and ability to sample a variety of habitats, including deeper waters. This method is widely used for qualitative sampling of benthic macroinvertebrates, however it is generally regarded as a semi-quantitative way of sampling if a standardised technique and sampling period (time) is employed for each sample collected (Resh & McElravy, 1993).

In order to perform Kick sampling the operator faces downstream with a pond net held vertically in front, the rim against the substratum. The bed immediately upstream is vigorously disturbed with the feet, causing the dislodged organisms to flow into the net. The operator moves around the site for a predetermined amount of time ensuring all habitats are sampled equally in proportion to their occurrence at the site. The size of the net mesh determines the size limit of the organisms collected; however this also affects the efficiency of the sampler because if the mesh is too fine then backwash is created which can sweep organisms out of the net (Resh & Jackson, 1993). For this study a 1mm mesh was used. The length of the stream section sampled was based on the pool-riffle morphology. Sampling length influences both the number of species recorded and the abundance distribution (Carter et al., 2006). Therefore, attempts were made to sample across the entire riffle area to maximise representativeness of the benthic macroinvertebrate community. A standard effort was employed when undertaking the sampling, with 3 minutes being employed throughout (EA standard protocol) (Murray-Bligh, 1999). Additionally, stones in the net were examined for attached specimens. This technique has the advantage of covering a relatively large area of streambed, including the full range of habitats present. Samples usually contain a large proportion of the invertebrate taxa and therefore have a higher chance of containing rarer taxa than fully quantitative techniques (Chiasson, 2009). The samples were stored in 70% industrial methylated sprit (IMS) in sealable bags and refrigerated prior to processing.

3.4.5 Surber sampling

Surber sampling allows quantitative analysis of macroinvertebrate communities to be undertaken and is frequently used in routine biological monitoring programmes due to the approach allowing spatial, temporal and density differences to be directly compared (Resh & McElravy, 1993). The frame of the Surber sampler used measured 0.3m by 0.3m, providing a sample area of $0.09m^2$. The vertical frame had a <1mm mesh net attached to capture fauna. The Surber sampler is placed on the substrate surface with the net opening facing into the current and the riverbed substrate within the frame is disturbed manually using a metal trowel. This was performed five times for a duration of one minute at each site. The disturbed material was removed from the net (with specimens attached to the net removed by hand) and transferred into pre-labelled sealable bags with 70% IMS being added to the sample for preservation. Sample variation was minimised by using the same sampling operator for the collection of all ecological samples and the net was washed after each use to prevent cross contamination of samples.

Surber sampling has the principle advantage of being fully quantitative and is simple to perform. In addition to this, the technique causes little disturbance to the surrounding area, and recolonisation of the sampled area has been found to occur within 30 days (Matthaei et al., 1996), thus allowing repeat seasonal sampling to be undertaken. Limitations connected to the use of Surber sampling include the potential of underestimation of taxon richness and abundance, and the susceptibility of under recording instream patchiness (Dolédec et al., 2007). Representative samples are more likely to be gained through careful positioning of the Surber frame. However, this will be at the expense of randomization and the introduction of bias.

3.4.6 Processing and Identification

In the laboratory, all samples were washed through a 250μm mesh sieve in order to remove fine sediments and traces of preservatives. Samples were then transferred into white, flat-bottomed, plastic sorting trays and sufficient water was added to the sample to ensure it was fully submerged. This was done to dilute any preservative that remained within the sample and to reduce the amount of reflected light, which allowed macroinvertebrates to be more easily spotted (Murray-Bligh, 1999). For each sample the specimens were extracted using soft nosed tweezers and transferred into pre-labelled sealable plastic vials containing 70% IMS. Samples were subjected to full count analysis; see Section 3.6 for further information on the scoring systems used, in conjunction with the fauna recorded, to assess each rivers water quality and flow.

The level of identifications for studies using the benthic community as indicators of environmental impairment is highly contentious. On the one hand scholars believe that it is necessary to identify species to the highest possible resolution, whereas other views contest that lower levels of identification (e.g. to family or genus) are adequate (Resh & Jackson, 1993; Carter et al., 2006). For this thesis, all fauna were identified to the highest resolution possible (usually species or genus), exceptions being Diptera larvae and Oligochaeta, which were identified to family level (Diptera) and order (Oligochaeta). Species were identified using a Zeiss Stemi 1000 dissecting stereomicroscope with a Zeiss KL 200 light source and a range of lotic invertebrate taxonomic keys including; Holland 1972; ; Macan 1977; Elliot and Mann 1979; Hynes 1984; Elliot et al., 1988; Friday 1988; Savages 1989; Wallace et al., 1990; Edington and Hildrew 1995; Elliot 1996.

3.5 Sampling the Palaeo River Fauna

The location of the palaeochannels at each river site was determined using historic maps in addition to preliminary field visits (see Chapter 4: page 74, for The River Eye; Chapter 5: page 102, for The River Hull and Chapter 6: page 128, for The River Wensum).

This allowed suitable sites to be identified, for the extraction of material from sediment pits and sediment cores.

3.5.1 Sediment Core Extraction Technique

Once a suitable palaeochannel/deposits had been identified on each river, between 3 and 5 cores were extracted from it. The extraction position of each core was located to achieve a representative cross-section of the palaeochannel and to ensure the historical riverbed was sampled (Figure 3.1). For the purpose of this investigation only samples taken from the historical gravel riverbed were required. This provided an indication of the riverine environment when it was last actively flowing. The device used to extract the core from the palaeochannel was a Cobra Corer. This powerful piece of machinery allowed the collection of 1m sections of cores at each time. The equipment was heavy and relatively cumbersome, however vehicle access was possible at each river providing the opportunity to set up the equipment as close to the site as possible. Each of the site locations for palaeo core extraction can be seen in the following chapters; River Eye (Chapter 4.3), River Hull (Chapter 5.3) and River Wensum (Chapter 6.3).

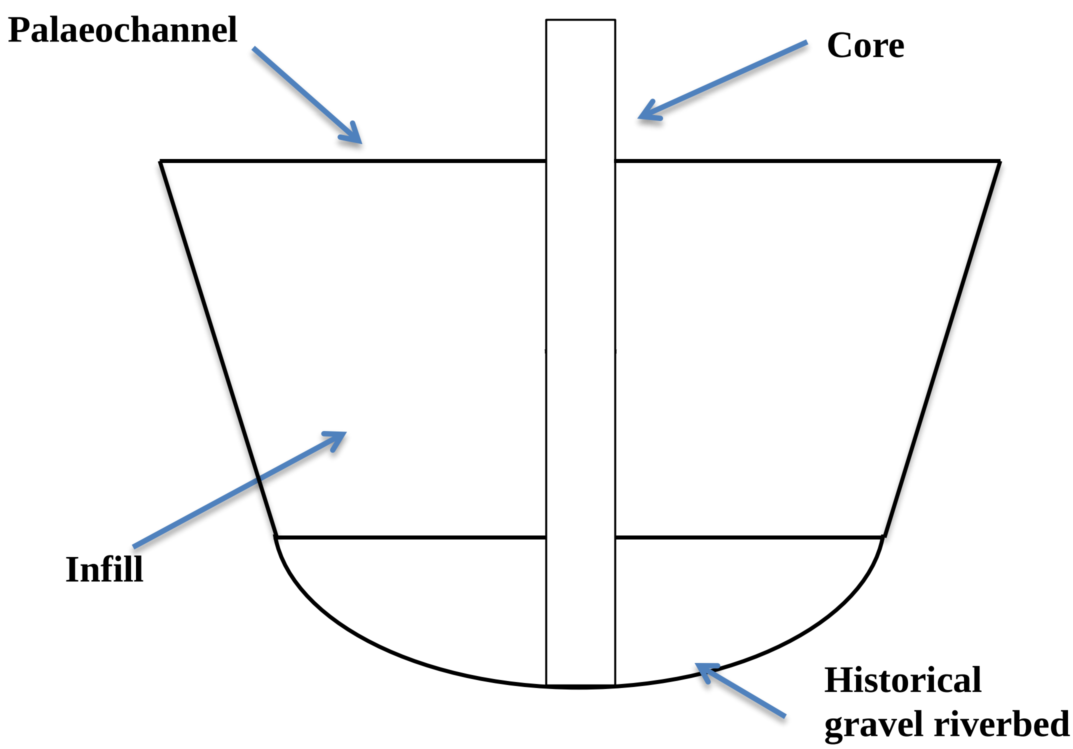

Figure 3.1 A conceptual diagram of a cross section of a palaeochannel and the relative positioning of the core in order to ensure the historical gravel riverbed is sampled.

3.5.2 Sediment Pits

A preliminary investigation was undertaken on the River Eye to determine if the instream island would prove beneficial as a site for palaeo sediment extraction. The depth of the first pit went to 45cm, as this is when gravel was reached. At each 10cm spacings, sediment was extracted and stored in labelled bags and the stratigraphy was noted. The area chosen proved fairly difficult to excavate due to the large amount of roots present from a nearby willow tree. A large volume of sub fossil material was extracted from the sediment pit. The subsequent visit to the site saw a deeper pit dug to a depth of 55cm, which was made substantially wider. The sediment was extracted at 5cm stages and double bagged.

At the River Wensum the sediment pit was dug by a JCB digger that was on site for the river restoration scheme (see Chapter 6.3). Sediment samples weighing around 5kg were extracted from the pit at 10cm intervals from 20cm to 200cm deep, following the procedure of Greenwood et al., (2003). These samples were stored in labelled containers and refrigerated to prevent decay before processing.

3.5.3 Dendrochronology

Dendrochronology (tree ring dating) studies the annual growth rings in trees (Schweingruber, 1988) and its practical applications are numerous. It can help to improve our understanding of environmental processes and conditions and provides insights into potential future environmental issues by putting the present into appropriate historical contexts (Jones, 1990). Tree ring widths are related to growing conditions, which in turn can provide a method for reconstructing stream flow series (Brown, 2007; Watson et al., 2009). This technique was employed in the River Eye study to help approximate the minimum age of the island on which the sediment pits were excavated, due to the presence of a large Willow tree growing on the in-stream island.

In order to extract the core a Mattson Three-Thread Increment Borer was used. The auger was pressed into the tree, perpendicular to the trunk. It was pushed and rotated clockwise until it had cored to the approximate centre of the tree. The core was extracted from the tree by pulling and turning the instrument in an anti-clockwise direction. It was then carefully bagged and transported back to the laboratory where the tree rings were counted. Finding a location on the tree to core proved difficult as the Willow tree was growing at an angle, which may have been caused by past flooding. In addition to this the tree trunk was fairly wide therefore it was difficult to accurately determine whether the corer had reached the centre of the tree. Due to these uncertainties only an approximate estimation of the age of the tree and therefore the island it grew on, can be made.

3.5.4 Processing, Identification and Preservation of Sub-fossil Fragments

Sub-fossilised insect remains are extremely susceptible to desiccation as they have not been through the mineralisation processes, either due to a lack of time or suboptimal burial conditions (Howard et al., 2009). This creates difficulties during processing and identification, therefore samples were carefully handled and kept in damp, cold, refrigerated conditions prior to analysis. The extraction of the Trichoptera, Coleoptera and Gastropoda followed the flotation method previously used by Coope (1986) with a minor alteration of the addition of a 125µm sieve and a 90µm sieve to remove smaller Trichoptera fragments (Greenwood et al., 2003). In order to facilitate the physical breakdown of the palaeo samples a method of mechanical sieving was introduced. The apparatus used consisted of a metal sieve clamped above a plastic bowl. The arm of the sieve was attached to a small electrical motor which when switched on, delivered variable amounts of vibration to the sieve, which could be controlled by adjusting the speed of the motor. During use, the mesh section of the sieve, containing either a 1kg sample from a sediment pit or a 10cm section of core, was fully submerged in luke warm water containing 2% sodium hexametaphosphate to aid disaggregation. Gentle sieving continued until the entire sample had passed through the sieve or larger particles were retained within the sieve (Howard, 2007). This process was then repeated for a further 1kg sample extracted from sediment pits only. This device was created and tested by Lynda Howard as part of her doctoral thesis. Howard (2007) tested the effectiveness of the sieve to disaggregate samples and to determine if it caused any adverse effects to the sub-fossilised material contained within it. Howard (2007) found that mechanical sieving of palaeo sediment had no significant effect, apart from speeding up the disaggregation process, and that break up of sclerites within samples was not due to the sieve but occurred randomly during deposition or processing.

After the palaeo material had been fully disaggregated, it underwent flotation (see Figure 3.2 for the step by step method used). The method of flotation was used as it separates organic matter from inorganic material allowing easy removal of sub-fossil material.

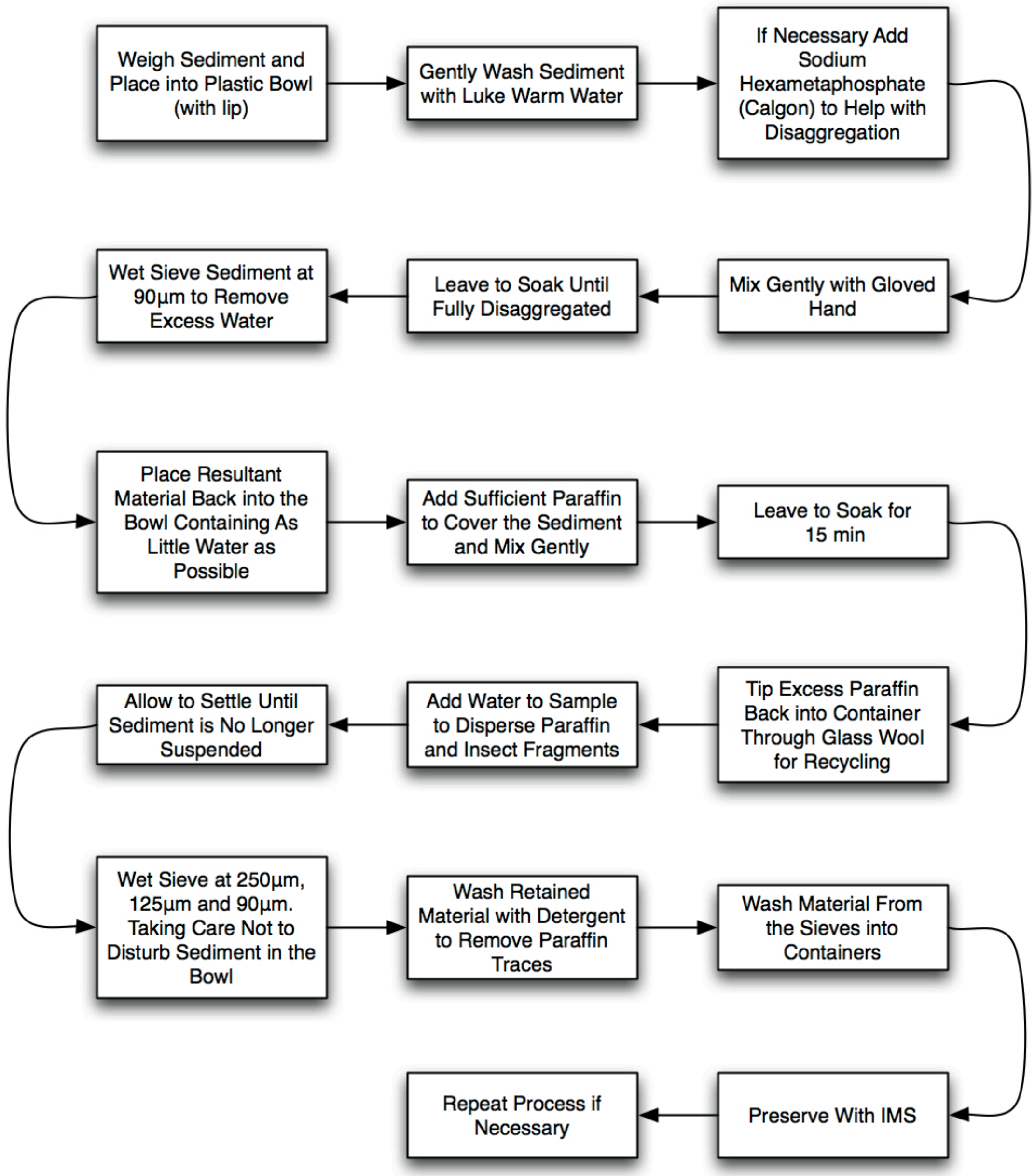

Figure 3.2 A flow diagram showing the step-by-step method used for the extraction and processing of the Trichoptera, Coleoptera and Chironomidae.

The flotation process was repeated twice to ensure as many fragments as possible were extracted from the material. However, sub-fossilised Gastropods do not float and they were therefore were picked out by hand using soft nosed tweezers once the sample had undergone flotation. Where possible all taxa were identified to species level, using a Zeiss microscope of up to 45 times magnification. Expert assistance was sourced from suitable specialists when faced with any identification uncertainties; Malcolm Greenwood (Loughborough University) for the Trichoptera and Paul Buckland (independent consultant) for the Coleoptera.

The preserved fragments of Trichoptera frontoclypeal apotomes were mounted onto labelled microscope slides with Hoyer's medium. This mountant allowed for recovery if the material was not flat or obscured, due to it being water-soluble. In order to identify the subfossil Trichoptera, comparisons were made with the Freshwater Biological Association Keys (Wallace et al. 1990; Edington and Hildrew 1995) and photographic reference collections created by Malcom Greenwood. The taxonomy used in this thesis follows Barnard (1985). Coleopteran identification was completed by using texts including Friday (1988) and Duff (2008) and through comparison with reference collections at the Leicestershire Museum at Barrow on Soar and consultation with experts (Prof. Paul Buckland). The Trichoptera frontoclypea and Coleoptera wing cases were photographed using a Zeiss microscope with a digital camera attachment, for future reference.

3.6 Macroinvertebrate environmental quality indices

Composition and abundance of taxa comprising the macroinvertebrate community of rivers can be used to characterise riverine systems and used to develop typologies relating to flow velocity. Biotic indices have been developed to consider the physiological response, sensitivity and tolerance of organisms to a range of factors including organic pollution and river flows. The biological indices used in this thesis are used by the Environment Agency for part of the screening process for WFD objectives to determine Good Ecological Status (GES). The indices used in this thesis were; The Biological Monitoring Working Party (BMWP) (Armitage et al., 1983) and The Average Score Per Taxon (ASPT) (Armitage et al., 1983) for water quality, The Lotic-invertebrate Index for Flow Evaluation (LIFE) (Extence et al., 1999), The Community Conservation Index (CCI) (Chadd & Extence, 2004) and Proportion of Sediment-sensitive Invertebrates (PSI) (Extence et al., 2011) (Table 3.2).

Table 3.2 Advantages and disadvantages of the biological indices used within this thesis.

Biological Indices	**Advantages**	**Disadvantages**	**References**
The Biological Monitoring Working Party Score (BMWP)	Provides a biological proxy of water quality and ecological changes. Both BWMP and ASPT provide a good indication of water quality. Can respond to inorganic pollutants.	Does not include an abundance element for individual taxa so not sensitive to low level enrichment. Influenced by habitat type (pool or riffle), sampling effort and season (ASPT addresses this to some extent).	Furse et al., 1981; Armitage et al., 1983; Chadd, 2010.
The Average Score Per Taxon (ASPT)	Compared to BMWP score, ASPT is subject to less variability between rivers and seasons allowing rapid and easy comparison between rivers/sites.	May be objected to variability when low numbers of taxa are present.	Armitage et al., 1983; Monk et al. 2006; Chadd, 2010.
The Lotic-invertebrate Index for Flow Evaluation (LIFE)	Helps identify river sites that have been subject to ecological pressures associated with river flow velocity. Can be applied to all river types. Operates at family and species level. Sensitive to abundance o taxa.	Output may be compromised by poor water quality. Generally cannot be applied to deep silty rivers. Scores may be reduced at sites subject to organic pollution.	Extence et al., 1999; Monk et al., 2008; Chadd, 2010.
The Community Conservation Index (CCI)	Uses species level identification to calculate a sites/samples conservation status. Can be adjusted to allow for local rarity of species and also weighted for taxon richness	Only directly applicable to Britian and Ireland. The larger the species dataset obtained, the better the resolution of the final score. Not applicable for palaeoecological investigations.	Chadd & Extence, 2004***;*** Chadd, 2010.
Proportion of Sediment-sensitive Invertebrates (PSI)	Helps identify site-specific fine sediment pressures (deposition). May be used to help identify and manage impacts associated with sedimentation.	Separating the effects of flow pressures and sedimentation is taxing.	Extence et al., 2013.

These indices (minus CCI due to its reliance on the presence of rare taxa) were also used to identify reference conditions based on the sub-fossil communities extracted from palaeochannel deposits. The use of palaeoecological approaches for setting reference conditions for management are outlined in the WFD (European Commission, 2000), therefore the same criteria were used within this thesis.

3.6.1 Biological Monitoring Working Party (BMWP) and Average Score Per Taxon (ASPT)

The Biological Monitoring Working Party (BMWP) and its derivative Average Score Per Taxon (ASPT) are biotic indices for measuring water quality and ecological changes (Furse et al., 1981). The system was devised for the Department of the Environment in 1976 as a means of classifying how organically polluted rivers were at a time when the need for biological surveillance in conservation was first becoming apparent. This was undertaken in response to the criticism following the first official biological monitoring method; the 1970 National River Pollution Survey undertaken by the Department of the Environment following increased environmental awareness in the 1960s (Hawkes, 1998). The score reflects the sensitivity of macroinvertebrates to oxygen depletion, providing a good indication of whether the river is impacted by organic pollution (Monk et al. 2006).

For calculation of the BMWP score, identification of taxa to family level is sufficient (Armitage et al., 1983). Each family is given a score from 1 to 10, which is dependent upon its perceived sensitivity to organic pollution (Rosenberg and Resh 1993). Pollution intolerant families are given high scores (e.g. Goeridae = 10), whereas pollution tolerant families are given low scores (e.g. Oligochaeta = 1) (Rosenberg and Resh 1993). The sum of the scores for all families present within the sample provides the BMWP score; therefore a higher score represents a higher water quality. By dividing the BMWP value by the number of families recorded in the sample, the ASPT score can be determined (Armitage et al., 1983). Previous research has found that the ASPT is less variable between river types and is less seasonally variable allowing rapid and easy comparison between rivers (Monk et al. 2006).

3.6.2 Lotic-invertebrate Index for Flow Evaluation (LIFE)

Flow dynamics within a river can have a large spatial variability and are a fundamental determinant of the physical structure of habitats and the distribution of biotic communities (Lytle & Poff, 2004; Armitage et al., 1997). Numerous species assemblages are documented to be associated with distinct flow conditions. For instance, *Goeridae*, a cased caddisfly, is typically associated with a range of flowing conditions (moderate to fast flow velocity) (Greenwood et al., 2006). The Lotic-invertebrate Index for Flow Evaluation (LIFE) technique was developed by Extence *et al.* (1999) by linking qualitative and semi-quantitative in stream macroinvertebrate community change to varying flow regimes. LIFE scores/flow groups are based on the known requirements and preferences of flow velocity of riverine benthic macroinvertebrate species and families (Extence et al., 1999). This method helps to identify river sites that have been subjected to ecological stress primarily associated with reduced flows, which may have resulted from anthropogenic abstraction or natural drought (Monk et al., 2008). Each macroinvertebrate species and family has been assigned a flow group from I to VI (I = Rapid flows; VI = Drought resistant). Taxa such as Oligochaeta and Chironomidae, which show no definitive to relationship to flow velocity with regards to abundance at the order of family level and have a ubiquitous distribution, have not been included in the LIFE methodology (Extence et al., 1999).

The calculation of the LIFE score involves dividing the sum of the flow scores of all scoring taxon present in the sample (*fs*), by the number of scoring taxon found in the sample (Equation I). Flow scores (*fs*) are obtained by the use of a matrix (Table 3.3), using taxon abundance, derived from standard Environment Agency macroinvertebrate abundance categories (Table 3.4). Higher LIFE scores typically reflect higher river flows (Extence et al., 1999).

$$LIFE = \frac{\sum fs}{n}$$

(I)

Where $\sum fs$ is the sum of the individual taxon flow scores for the whole sample, *n* is the number of taxa used to calculate $\sum fs$.

Equation I. LIFE equation taken from Extence *et al.* (1999).

Table 3.3 Flow scores (*fs*) for different abundance categories of taxa associated with flow groups I-IV and their mean current velocities (adapted from Extence et al., 1999).

	Flow Groups	Mean Current Velocity	Abundance Categories A	B	C	D/E
I	Rapid	Typically > 100 cm s-1	9	10	11	12
II	Moderate/fast	Typically 20-100 cm s-1	8	9	10	11
III	Slow/sluggish	Typically < 20 cm s-1	7	7	7	7
IV	Flowing/standing		6	5	4	3
V	Standing		5	4	3	2
VI	Drought resistant		4	3	2	1

Table 3.4 Standard Environment Agency macroinvertebrate abundance categories.

Category	Estimated Abundance
A	1-9
B	10-99
C	100-999
D	1000-9999
E	10,000+

The robustness of this technique has been investigated in a number of studies, for instance the research into using LIFE together with River InVertebrate Prediction And Classification System (Clarke et al., 2003), as well as the production of generalised LIFE response functions (Dunbar & Clarke, 2004). Both of these studies suggest that the LIFE scoring method has wide applicability for all river types as the LIFE data set combines long term river gauging records with routine biomonitoring for 291 sites across England and Wales (Extence et al., 1999). The data set also includes sites that are largely unaffected by water quality issues and other factors that may influence flow regimes on instream communities, such as impoundments (Monk et al., 2008). This comprehensive river evaluation technique was used in this study not only to assess the contemporary river characteristics but also due to its ability to allow palaeoflow environments to be reconstructed (Greenwood et al., 2006; Howard et al., 2009), thus providing an insight into how the flow regime may have varied historically. The Palaeo LIFE method was first applied to palaeoecological reconstruction of riverine channels by Greenwood *et al.*, (2006). The study demonstrated that sub-fossilised larval Trichoptera, extracted from palaeochannels of the River Trent, could be used in association with LIFE to reconstruct past flow signals and provide information on the historic aquatic habitat structure (Greenwood et al., 2006; Howard et al., 2010). The methodology has subsequently been extended to incorporate Coleoptera and Chironomidae (Howard et al., 2009; Howard et al., 2010).

3.6.3 Proportion of Sediment-sensitive Invertebrates (PSI)

Impacts of fine sediment in riverine systems and the effects it has on benthic macroinvertebrate communities has been recognized for over fifty years (Cordone and Kelley 1961; Ryan 1991; Wood and Armitage 1997; Waters 1995; Jones et al. 2011). The erosion and deposition of fine sediments are fundamental and natural components of the hydrogeomorphic processes within fluvial systems (Jones et al., 2012). However, within the UK there has been no generally accepted method to assess the impact of fine sediment deposition on riverine invertebrate communities and traditionally, physical and visual methods were adopted to measure the volumes of deposited sediment. However, Extence et al. (2013) devised an index that uses macroinvertebrates as biological indicators for fine sediment impacts called The Proportion of Sediment-sensitive Invertebrates (PSI). PSI uses macroinvertebrate communities as a proxy measure to determine and standardize site-specific impacts of fine sediment (defined as organic and inorganic particles of less than 2mm in diameter) deposition (Extence et al., 2013). This method follows similar principles to those set out in the formulation of the LIFE score (Extence et al., 1999) and CCI scores (Chadd & Extence, 2004) (see Chapter 3.6.4), with benthic macroinvertebrate species and families being assigned one of four Fine Sediment Sensitivity Ratings (FSSR) (Table 3.5).

Table 3.5 Fine Sediment Sensitivity Rating definitions and abundance weighted scores for PSI calculation (Extence et al., 2011).

Group	Fine Sediment Sensitivity Rating (FSSR)	Log Abundance			
		1-9	10-99	100-999	1000+
A	Highly sensitive	2	3	4	5
B	Moderately sensitive	1	2	3	4
C	Moderately insensitive	1	2	3	4
D	Highly insensitive	2	3	4	5

In a similar manner to the LIFE score, a number of taxa have been excluded from the FSSR due to them being unsuitable for fine sediment assessment, for instance Leptoceridae and Chironomidae are excluded, as they occur in a very wide range of habitat conditions (Extence et al., 2013). Once the FSSR has been allocated to all taxon found in a sample the following equation (Equation II) is applied:

$$PSI\ (\Psi) = \frac{\sum Scores\ for\ Sediment\ Sensitivity\ Groups\ A\&B}{\sum Scores\ for\ all\ Sediment\ Sensitivity\ Groups\ A;\ B; C\ \&\ D} \times 100$$

(II)

Equation II. PSI equation taken from Extence *et al.* (2011).

The PSI score created describes the percentage of fine sediment sensitive taxa recorded in a sample (Extence et al., 2013), and its interpretation is aided by the use of an interpretation table (Table 3.6). Suspended and deposited sediments have the potential to threaten the ecological integrity of water bodies, which may result in it not attaining good ecology status/potential under the WFD (Collins et al., 2011). This fast and effective method has the ability to help identify and manage sedimentation impacts (Extence et al., 2011) and can also be applied to palaeo-proxies in order to gauge potential historic effects.

Table 3.6 Interpretation of PSI scores (Extence et al., 2011).

PSI	River bed conditions
81-100	Minimally sedimented/unsedimented
61-80	Slightly sedimented
41-60	Moderatly sedimented
21-40	Sedimented
0-20	Heavily sedimented

3.6.4 Community Conservation Index (CCI)

The Community Conservation Index (CCI) is a conservation indexing protocol that takes into consideration the presence of macroinvertebrate taxa within aquatic communities and uses them to derive a sites/samples conservation status. Calculation of the CCI requires species level identification across the range of taxa found in the sample. Generally the larger the species dataset obtained, the better the resolution of the final score (Chadd & Extence, 2004). Species are given a Conservation Score (*CS*), which is in accordance with the nationally agreed scheme outlined by the Joint Nature Conservation Committee (JNCC, 2005). The sum of the *CS* scoring taxon is calculated and divided by the number of contributing species; as shown in the below equation (Equation III):

$$CCI = \frac{\sum CS}{n} \times CoS \tag{III}$$

Equation III. CCI Equation taken from Chadd and Extence (2004).

This score is then multiplied by a Community Score (*CoS*) which is derived from the rarest taxon present in the sample (CS_{max}) (Chadd & Extence, 2004). Interpretation of the calculated CCI output is aided by the use of an interpretation table (Table 3.7).

Table 3.7 Interpretation of CCI scores (Chadd & Extence, 2004).

CCI	Interpretation
0.0 - 5.0	Sites supporting only common species and/or a community of low taxon richness. *Low conservation value.*
> 5.0 - 10.0	Sites supporting at least one species of restricted distribution and/or a community of moderate taxon richness. *Moderate conservation value.*
> 10.0 - 15.0	Sites supporting at least one uncommon species and/or a community of high taxon richness. *Fairly high conservation value.*
> 15.0 – 20.0	Sites supporting several uncommon species, at least one of which may be nationally rare and/or a community of high taxon richness. *High conservation value.*
> 20.0	Sites supporting several rarities, including species of national importance, or at least one extreme rarity and/or a community of very high taxon richness. *Very high conservation value.*

3.7 Statistical Analysis Techniques

A variety of statistical techniques were utilised to reduce data into a more manageable format, calculate and identify statistical patterns within each dataset. This assisted in highlighting statistical relationships (similarities and differences) between the contemporary and palaeoecological communities. The data analysis techniques described in the following sections were chosen to address the aims and objectives outlined in Chapter 1.2.

3.7.1 Analysis of Variance (ANOVA)

The Statistical Package for Social Sciences (SPSS Inc., 2011) was used to run one-way Analysis of Variance (ANVOA) to obtain descriptive statistics such as the mean/average values of variables and to examine if these variables differ statistically between groups. AVOVA provides a powerful tool for determining deviations from the null hypothesis of no significant difference over space or time. The validity of these results relies on three main assumptions being met: 1) Independence of data within samples: 2) Homogeneity of variances for each population; and 3) Normality of data distribution (Underwood, 1997). Where ANOVA indicated that there was a significant difference between groups, Tukey's post-hoc tests (Tukey, 1953) were undertaken to determine where these differences occurred. The Kruskal-Wallis test, a nonparametric equivalent to one-way AVOVA was also applied to the data where homogeneity of variance could not be assumed.

SPSS was also used to generate clustered error bar plots to graphically illustrate patterns of change. The advantage of using clustered error bar plots is that they have the ability to recognise both site and seasonal specific differences as well as summarising general patterns in the dataset.

3.7.2 Ordination

Ordination techniques were used to investigate the underlying structure and trends in the data, gradients in species composition and seasonal relationships through the use of direct (e.g. Canonical Correspondence Analysis) and indirect gradient analysis (e.g. Detrended Correspondence Analysis). The technique condenses large numbers of variables into a lower number of indices (axes) while still being able to retain the original meaning of the data set (Randerson, 1993). Ordination techniques have been widely used within ecological investigations (Birks, 1995) and were performed using the programme Canoco (ter Braak & Smilauer, 2002).

3.7.3 Detrended Correspondence Analysis (DCA)

Detrended Correspondence Analysis (DCA) was used to explore the spatial and temporal patterns within the ecological communities of the rivers and to extract any latent gradient in the palaeoecological data. It is a method based on eigenvalue analysis of indirect gradient analysis and uses weighted averages applied to the data matrix, deriving sample scores from the species scores and weights, or visa versa (Kent & Coker, 1992). Therefore the analysis devises theoretical variables (axes) to maximise the dispersion of the species and sample scores on multiple, independent axes (ter Braak, 1995). DCA was developed by Hill & Gauch (1980) to overcome two major faults observed in ordination techniques: 1) the arch (horseshoe) effect, in which the second axis is an arched function of the first axis and 2) a compression of the gradient at each end (Jackson & Somers, 1991; Leps & Smilauer, 2003). The arch effect is caused by monotonic species distribution (i.e. species which either increase or decrease, but not both, as a function of environmental factors). Detrending removes this ‘arch effect’ through the division of the first axis and the centering of the second data axis (Beh & Lombardo, 2012). These errors are also corrected through rescaling the correspondence output. Rescaling ensures that the distance or space in the ordination diagram is constant (Daley & Barber, 2012). However this does mean that the interpretive relevance of DCA relies on a judgement of whether the axes represent realistic environmental variables as it performs best for species that have one optimal environmental condition (Manjarrés-Martínez et al., 2012).

The output of DCA can help with the interpretation of temporal and spatial variation within a community. The ordination plots created allow the differences between samples from different locations or from different seasons to be compared and any similarity/dissimilarity between groups can be shown. The taxa driving these patterns can also be plotted and indicator taxa highlighted. The ordination diagrams were plotted using the programme CanoDraw and Excel (Braak & Smilauer, 1996).

3.7.4 Detrended Canonical Correspondence Analysis (DCCA)

DCCA is a direct gradient analysis method which explores the relationships between the distribution of biotic communities and environmental variables (Perez-Quintero, 2012). This technique detects the patterns of variation within communities and identifies the environmental variables that are directly accountable (Li et al., 2012). Therefore a choice of meaningful variables in DCCA is essential for a significant output. DCCA combines two data sets (biotic and environmental) and performs weighted averaging to the simultaneous analysis of many species and environmental variables, through the incorporation of regression and correlation within the ordination analysis (Von Bertrab et al., 2013).

3.8 Summary

This chapter outlines the fieldwork methods employed and the identification procedures used for both the contemporary and palaeo samples extracted. The statistical analysis techniques and ecological indices that were employed in this research to analyse the resulting data have also been described, with the subsequent chapters presenting the results found. The methods and techniques that have been explained have been specifically chosen and designed to address the aims and objectives of this thesis and ultimately explore how palaeoecological multiproxy techniques can help define baseline/benchmarks to help characterise reference conditions for river restoration.

CHAPTER FOUR: THE RIVER EYE

4.1 Introduction

This chapter provides an overview of the contemporary aquatic macroinvertebrate community characteristics and in-stream habitats of a reach of the River Eye (Leicestershire) as well as the historic (benchmark) conditions present around 1956, based on palaeoecological analysis. Sites along a reach of the River Eye (SSSI) were sampled to investigate the contemporary macroinvertebrate community composition and distribution within the study reach. The modern assemblage recorded was directly compared with palaeoecological evidence from an in-channel island, that was a minimum of 57 years old (based on dendrochronological evidence). The chapter includes an introductory description of the River Eye SSSI, its geology and in-stream/bankside habitats as well as the anthropogenic influences on it. The results obtained from the contemporary, palaeoecological and palaeohydrological analysis are presented, discussed and used to interpret the changing state of the River Eye.

The objectives of this chapter are to (refer to Chapter 1.2, page 5 for relevant objective):

- Explore the contemporary riverine environment of the River Eye and derive the macroinvertebrate community biotic indices;
- Identify the River Eye's historic palaeochannel and characterise the palaeoecology through extraction of sediment profiles and mulitproxy palaeoecological techniques to define a historical benchmark.
- Compare the contemporary and palaeoecological results to characterise the changes that have taken place within the River Eye study reach.

4.2 The River Eye

The River Eye was first notified as a SSSI in 1983 under the Wildlife and Countryside Act. The site was designated as one of the best examples of a semi-natural lowland clay river that included natural structural features such as riffles, pools, small cliffs and meanders. However, along with most lowland rivers in England it suffers from a range of pressures including enrichment, siltation and channel modification (Natural England, 2012). As a result of both historical and recent physical modifications, plans for future river restoration are actively being considered. The river rises at Bescaby, approximately 10km north east of Melton Mowbray in Leicestershire (Figure 4.1) and is characterised by a clay catchment. The course of the SSSI is 8km long, starting at Stapleford and ending just upstream of the town of Melton Mowbray, where the name changes to the River Wreake, and forms a tributary of the River Soar (Figure 4.2).

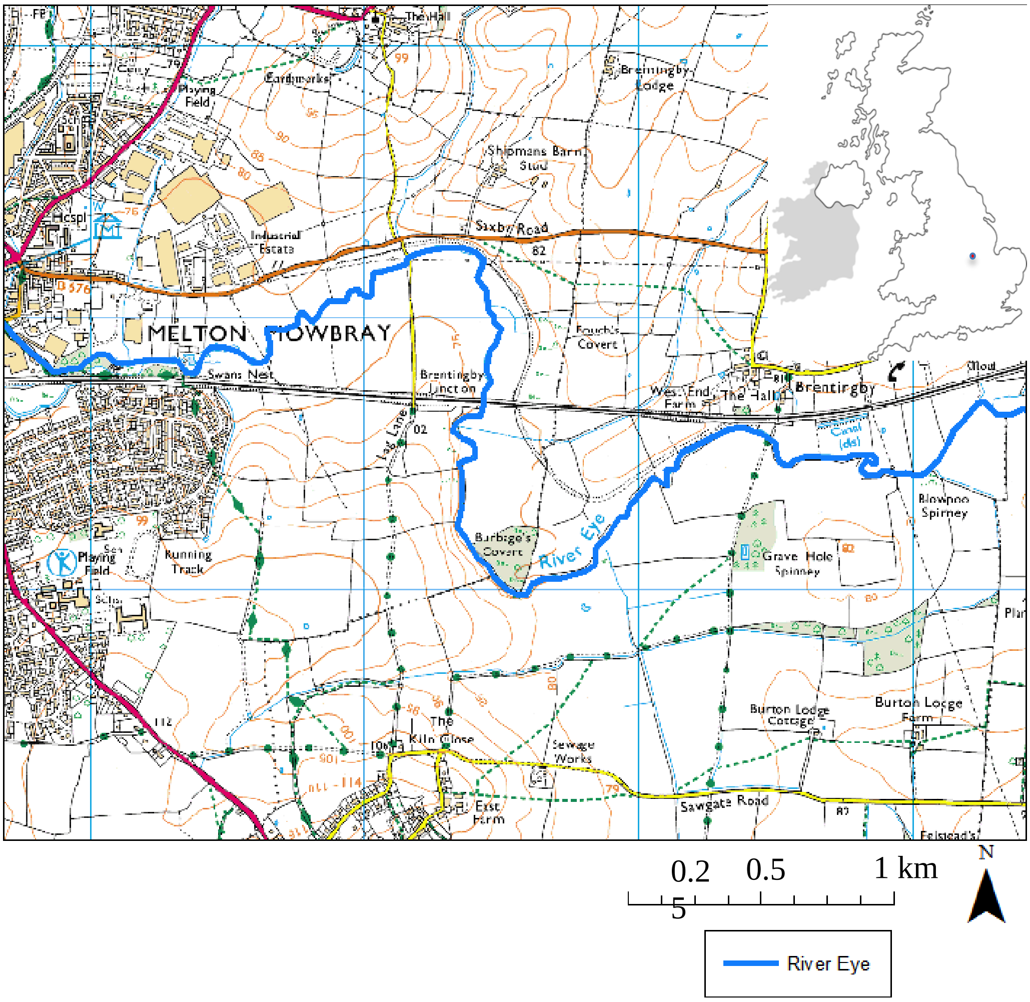

Figure 4.1 Location map of the River Eye study area (EDINA Digimap, 2013).

The watercourse is a single thread low gradient clay stream and currently suffering from insufficient flow volumes, over deepening and widening of the channel, dredging and impoundment effects (Natural England, 2013b). The entire length of the SSSI is deemed to be ‘Unfavourable No Change’ when compared against favourable condition targets set as part of the site’s conservation objectives established by Natural England (2013a). A number of structures are also present along the SSSI, which may be detrimental to the ecological status of the river. A summary of the features, modifications and impacts currently affecting the River Eye are presented in Table 4.1.

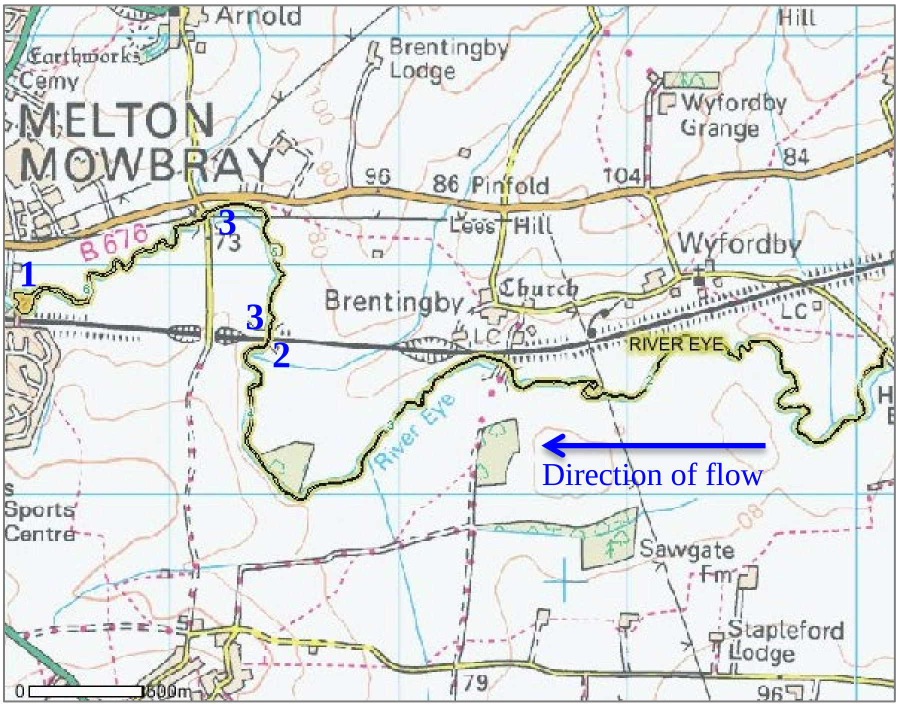

Figure 4.2 Map showing the full extent of the River Eye SSSI (Natural England, 2012).

Table 4.1 Summary of structural modifications and pressures and their impacts on the River Eye SSSI. Based on APEM (2010). Key structures are numbered and their locations are highlighted on Figure 4.2.

Structural Modifications/Pressures	Impacts
Two weirs (1) and one flood alleviation dam/culvert adjacent to railway bridge (2)	The impounded water results in a ponding effect causing flow velocity reduction and increased fine sediment deposition.
Bank reinforcement	Habitat structure is lost through the removal of the niche riparian habitat zone and marginal silts and vegetation.
Two bridges (3)	Possible scour downstream of bridge due to the constriction of flow, resulting in over deepening and over widening of the channel.
Dredging/over-deepening	Channel becomes wider and deeper. Coarse substrates are reduced and gravel habitats for spawning are lost
Overgrazing of river banks along much of the SSSI length	Bankside poaching by pastoral animals can reduce riparian vegetation and habitat quality and increase sedimentation from adjacent agricultural land.

The geology of the River Eye catchment consists primarily of Jurassic and Glacial Diamicton (Boulder Clay) but is also influenced by Jurassic limestone (Figure 4.3). The study was undertaken on a reach of the river approximately 1.4km in length, which flowed through a pastoral floodplain and arable farmlands. The river displayed a range of morphological features such as series of pools, gravel riffles and runs. Downstream sections of the river have been widened due to cattle/sheep poaching and in some areas sedimentation has led to the development of low in-channel berms. These features, together with bank slips, which are now exploited by livestock for easier access to the river, suggests that in places the river is over-deepened. Channel realignment, historic dredging and a series of weirs and impoundments downstream of the SSSI have significantly modified the hydromorphology of the River Eye. These effects have been recognised in the draft Natural England report 'River Eye SSSI Hydromorphic Audit: Implications for Weir Removal' (Natural England, 2012).

Vegetation is abundant within the river with the upper reaches of the study section being dominated by marginal vegetation including bulrush (*Scirpus lacustris*), greater pond sedge (*Carex riparia*) and reed canary grass (*Phalaris arundinacea*), interspersed by Willow trees (*Salix spp*.) in the riparian zone. Within the deeper pool sections, the river supports beds of yellow water lily (*Nuphar letea*), but the diversity of submerged plants is generally low. Natural England (2013a) describe the invertebrate community as being typical for a small, unpolluted but diverse river system with its key note species being the white-legged damselfly (*Platycnemis pennipes).* Records show that the white-legged damselfly (*Platycnemis pennipes)* is close to the northerly limit of its distribution in the UK (Garrison et al., 2010).

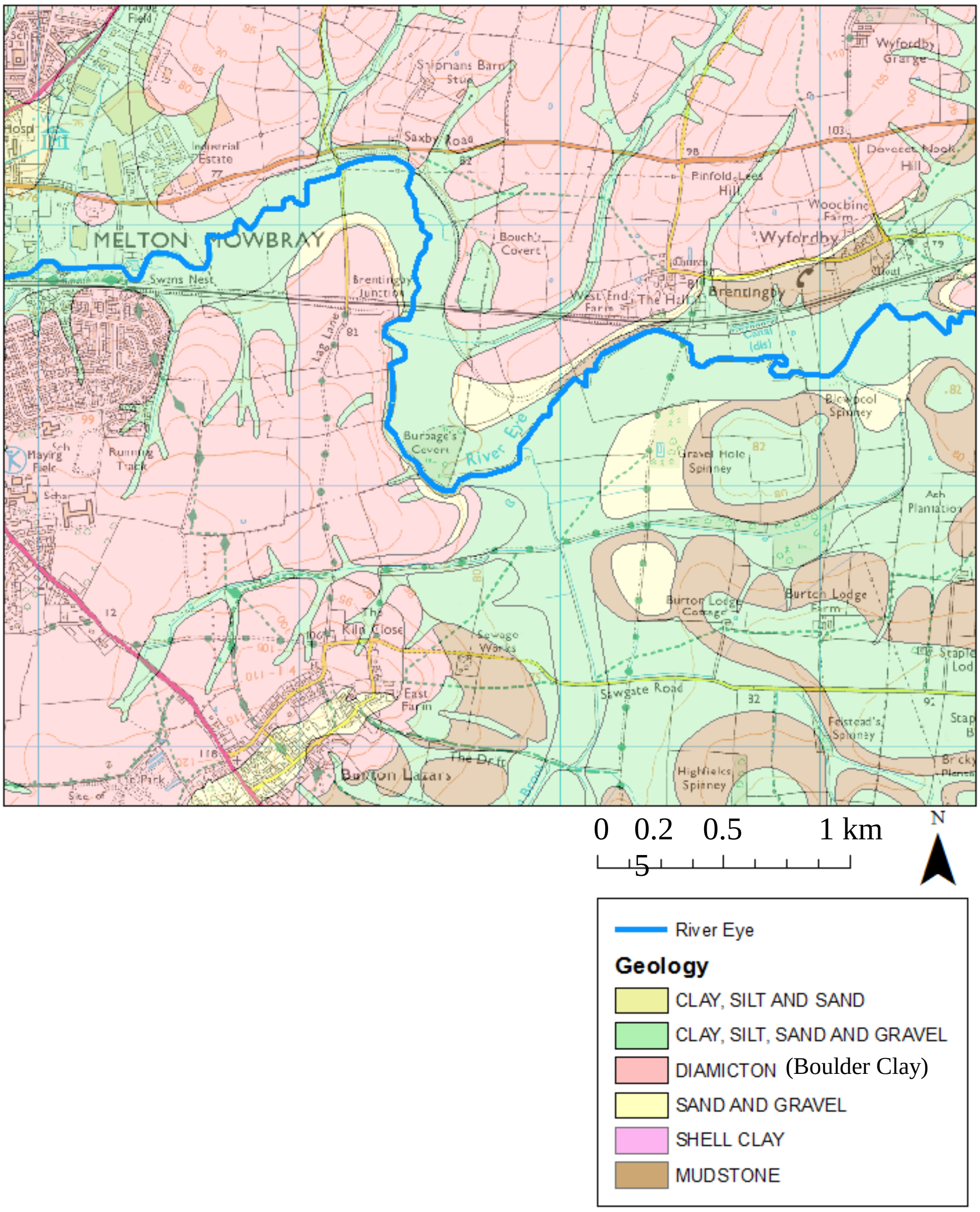

Figure 4.3 Map showing the local drift geology of the River Eye SSSI (EDINA Geological Digimap, 2012).

4.3 Study Sites

During preliminary visits to the River Eye, study sites were established for sampling the contemporary aquatic macroinvertebrate community (Figure 4.4). Three of the sites were chosen upstream of the flood defence structure and railway bridge, and two downstream. The river is bordered by meadow and grazing land, with some areas supporting large trees at the margins (Plate 4.1).

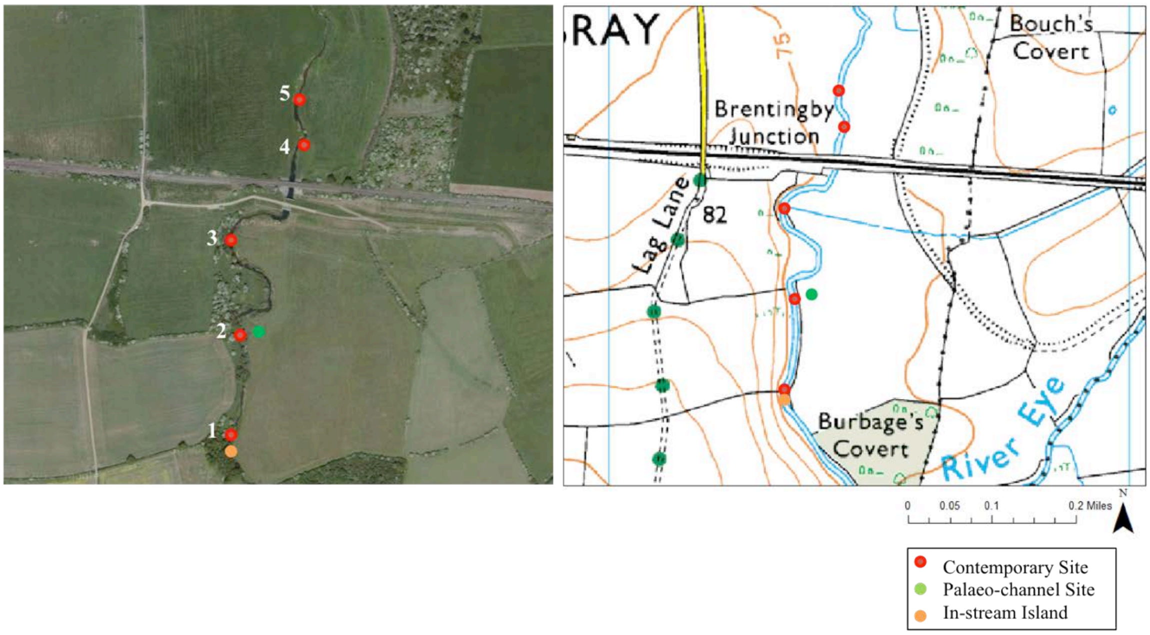

Figure 4.4 Aerial photograph and map showing the locations of the contemporary sampling sites and landuse, the palaeo-channel and the in-stream island (Tele Altas, 2012).

Plate 4.1 Study reaches facing downstream along the River Eye in May 2010. Pictures reflect the numbered sites in Figure 4.4. (GPS Coordinates for Site 1: 52.756461, -0.855378).

Two locations were identified for the extraction of palaeoecological samples (Figure 4.4). Cores were extracted using a Cobra Corer from a palaeochannel running parallel to the river (thought to be associated with an old mill) and a sediment pit was also excavated on a small mid-stream island located, at the top end of the study reach (Plate 4.2 and see Chapter 3.5 for details of extraction methods used). The stratigraphy of the sediment pit showed that the top 10cm was comprised of soil, with a band of clay running from 10-30cm. The key historic river sample zone was found between 35-45cm. Below the historic riverbed the gravel material reverted back into clay.

The in-stream island was selected as a suitable site due to the presence of a large willow tree growing on the island. This indicated that the sediment associated with it must have been *in situ* some time. The tree provided a means of obtaining a minimum age for the deposits beneath. The use of dendrochronology (counting the tree rings) indicated a minimum age of the island (Plate 4.3). The tree was 55 years old (See Methodology, Chapter 3.5.3) and the latest estimated date the in-stream island could have been established was 1956.

Plate 4.2 The inside of one the sediment pits dug on the in-stream island, measuring a depth of 50cm.

Plate 4.3 Photograph showing the core obtained from the willow tree located on the in-stream island.

4.4 The Historical River Eye

Examination of historic Ordinance Survey maps allowed any river channel movements to be clearly identified. The earliest map (Country Series) available for the river is dated 1849-1899 (EDINA Historic Digimap, 2012). The map indicates that the river has experienced very little morphological change and that it has been very stable. The map indicates the presence of a side channel dating back to 1849 and through communication with the local landowner (Julia Hawley), the channel was determined to be a bypass channel, probably used by a historic mill operating in the area (Plate 4.4).

4.5 Results of the Contemporary Ecological Community Composition

In order to gain a greater understanding of the contemporary abiotic environment within the River Eye, water velocity, pH, conductivity and temperature, where measured seasonally, *in-situ* at the sample sites (see Methodology Chapter 3.4.1). A summary of the results recorded is presented in Table 4.2. Flow velocity displays little variation between the seasons, with only a marginal reduction in autumn. The pH ranged from 7.5 – 8.5 indicating the river was alkaline in nature. The conductivity levels were at their highest in autumn and winter (695 µS cm^{-1}). This may be a result of the local geology as the river runs through areas of clay soils which possess a higher conductivity because of the presence of minerals that ionize when washed into the water (USEPA, 2013). The seasonal temperature range of the river (4.2 – 16.7°C) are not considered wide enough to have a significant influence on the rates of chemical reactions or the solubility of gasses (Chapman, 1996).

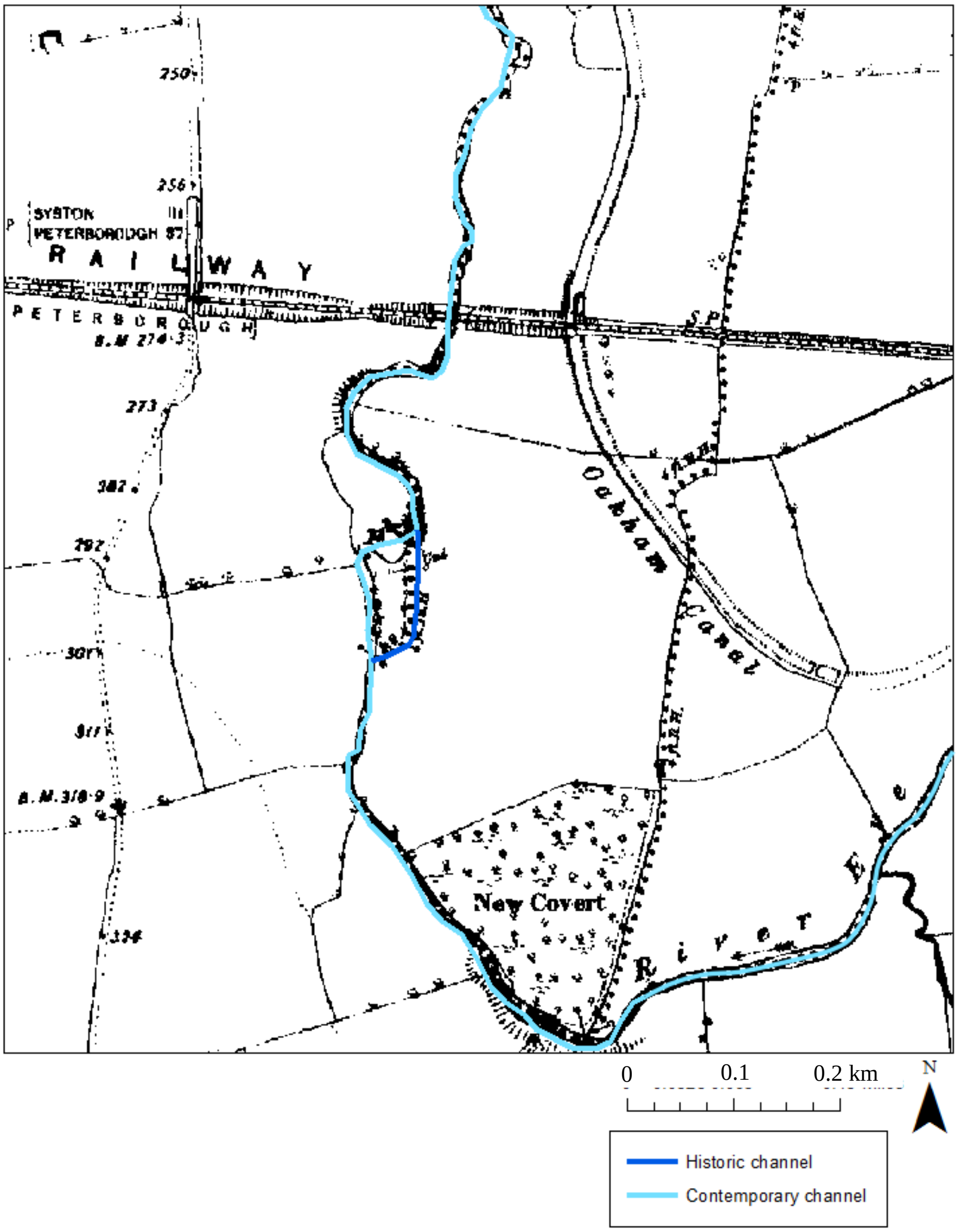

Figure 4.5 Country Series (1:10560) OS map, dated 1849-1899, of the River Eye depicting the contrast between the historic and contemporary channel (EDINA Historical Digimap, 2012).

Plate 4.4 Photograph of the bypass palaeochannel, facing downstream, probably used by the mill located downstream of this point on the River Eye SSSI (GPS: 52.75797, -0.854702).

Table 4.2 Table highlighting the range of the abiotic measurements and their averages indicated in brackets, for each seasonal sampling undertaken along the River Eye SSSI.

		Sampling Season		
		Spring 2010	**Winter 2011**	**Autumn 2011**
Abiotic Measurement (Range)	**Depth** (m)	0.26 – 0.36 (0.32)	0.18 – 0.3 (0.25)	0.19 – 0.39 (0.3)
	Flow Velocity (ms^{-1})	0.23 – 0.36 (0.295)	0.19 – 0.36 (0.275)	0.11 – 0.29 (0.20)
	pH	7.9 – 8 (7.94)	7.9 – 8.5 (8.18)	7.5 – 8.4 (7.86)
	Conductivity ($\mu S\ m^{-1}$)	0.56 – 0.58 (0.57)	0.64 –0.69 (0.67)	0.65 – 0.69 (0.67)
	Temperature (°C)	15.4 – 16.7 (15.82)	4.2 – 5 (4.5)	11.3 – 14.2 (12.56)

Macroinvertebrate sampling was undertaken in spring (May) 2010, winter (January) 2011 and autumn (September) 2011. Samples were collected using timed (3 minute) kick samples and Surber samples (see Chapter 3.4.3 for full details on methods used). Seasonal samples were collected to ensure the full variability of the community was represented. A total of 64 taxa, representing 44 families were identified within the contemporary river (see full species abundance lists in Appendix 1). The fauna was dominated by Trichoptera, accounting for 42% of the taxa, of which *Lepidostoma hirtum* (22%) and *Goera pilosa* (26%) were the most abundant species.

4.5.1 Detrended Correspondence Analysis (DCA)

DCA was undertaken in the programme Canoco (ter Braak & Šmilauer, 2002) using the River Eye kick sample data to explore the structure of the contemporary macroinvertebrate community over the three surveying seasons. Eigenvalues for this analysis are given in Table 4.3 and provide a measure of the relative importance of each axis in the analysis. The first four DCA axes explained 46.4% of the cumulative variance in the species data with Axis 1 accounting for 24.3% and axis 2 a further 14.3%.

Table 4.3 Summary of eigenvalues and cumulative variance of species data for DCA from kick samples collected from the River Eye.

Axes	1	2	3	4	Total inertia
Eigenvalues	0.343	0.201	0.087	0.022	1.411
Lengths of gradient	2.502	2.125	1.412	1.101	
Cumulative percentage variance of species data (%)	24.3	38.6	44.8	46.4	
Sum of all eigenvalues					1.411

The results indicate an overlap in the community composition between seasons for kick samples on both axes, although seasonal variability is clearly evident. Variability in the community was greatest (as indicated by the wider spread of samples on axis 1 and 2) in the autumn and lowest in spring. Spring and winter sample scores were closer to the origin of the bi-plot but still highlighted variability in the macroinvertebrate community (Figure 4.6).

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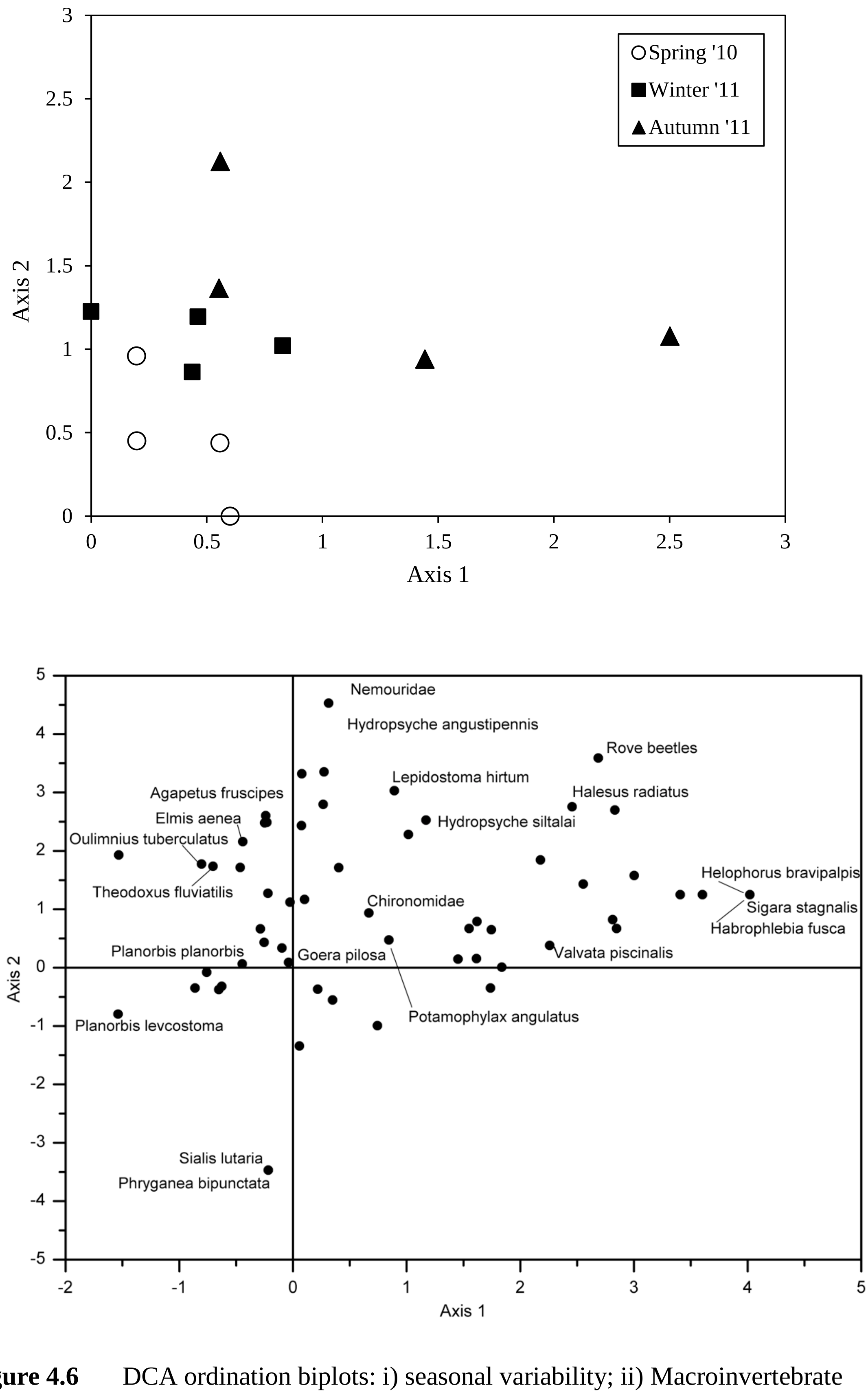

Figure 4.6 DCA ordination biplots: i) seasonal variability; ii) Macroinvertebrate community data.

The location of some taxa on axis 1 and 2 reflected seasonal changes in abundance, for example *Habrophlebia fusca* (mayfly larvae), *Helophorus brevipalpis* (water beetle) and *Sigara stagnalis* (corixid) were only recorded during the Autumn and have high scores on axis 1, reflecting low flow velocities. Taxa such as *Phryganea bipunctata* (caddisfly) were only recorded during spring and others such as *Hydropsyche angustipennis* (caddisfly) and *Nemouridae* (stonefly) were only recorded during the winter survey.

4.5.2 Detrended Canonical Correspondence Analysis (DCCA)

Canonical Correspondence Analysis was undertaken using the contemporary community Surber sample data in association with environmental data to identify the major sources of statistical variation within the combined abiotic and ecological data sets. However, this identified a clear arc effect (horseshoe) in the output and as a result Detrended Canonical Correspondence Analysis (DCCA) was used in Canoco (ter Braak & Šmilauer, 2002). A summary of results for the River Eye Surber Sampler data is presented in Table 4.4.

Table 4.4 Results of DCCA **A**: Summary of the eigenvalues and percentage of variance of species data and species environment relationship explained on the first four canonical axes for Surber samples collected from the River Eye. **B**; The significance of the first canonical axis and environmental variables using forward selection procedure in Canoco and the Monte Carlo random permutations test (999 permutations).

A	**Axes**	**1**	**2**	**3**	**4**	**Total inertia**
	Eigenvalues	0.258	0.069	0.011	0.003	2.365
	Lengths of gradient	1.593	1.156	0.580	0.403	
	Cumulative percentage variance of species data (%)	10.9	13.8	14.3	14.4	
	Cumulative percentage variance of species-environment relation (%)	41.3	57.1	57.1	57.1	
	Sum of all eigenvalues					2.365
B		F ratio	P value			
	Significance of first canonical axis	8.443	< 0.005			
	Significance of all canonical axes	4.515	< 0.005			
	Significance of Env. Variable:					
	1) pH	6.92	< 0.005			
	2) Temperature	4.94	< 0.005			
	3) Conductivity	4.46	< 0.005			
	4) Depth	3.53	< 0.005			
	5) Flow velocity	1.11	Not Significant			

The results indicated that a total of 14.4% of the variance in the species data and 57.1% of the species-environment relationship could be accounted for on the first four axes. Axis 1 accounted for 10.9% of the variance in the species data and 41.3% of the variance in the species-environment relationship. The first canonical axis represented a significant seasonal gradient ($p < 0.005$). When the individual variables were considered, the following variables were found to be statistically significant and influential over the aquatic macroinvertebrate community distribution on the axes; pH ($p < 0.005$), temperature ($p < 0.005$), conductivity ($p < 0.005$) and depth ($p < 0.005$) (see Table 4.4 B) Axis 2 explained an additional 2.9% of the variation in species data and 15.8% of the species-environment relationship. Axis 2 reflected variability in flow velocity with reduced flows during autumn associated with higher conductivities (Figure 4.7 i); although flow velocity was not statistically significant in the analysis.

The faunal bi-plot indicated common taxa, such as *Elmis aenea* (riffle beetle), *Goera pilosa* (caddislfy larvae) and *Chironomidae*, plotted towards the centre of the diagram (Figure 4.7 ii). Seasonal changes in abundance are apparent on axis 1 with *Haliplidae* larvae, for example being more abundant in the winter and as a result have a negative loading on axis 1. In contrast Caddisfly larvae, *Hydropsyche siltalai* occurred most frequently and resulted in higher loading on axis 1. *Helophorus brevipalpis*, were only recorded during the Autumn, indicating a preference for warmer, shallower waters with slower velocities. In contrast, taxa with preferences for cooler temperatures such as *Physa fontinalis* were only recorded in riffles during the winter samples and have a negative axis 1 score.

4.5.3 Ecological Indices

The following biological indices were derived using the macroinvertebrate kick sample data (BMWP, ASPT, LIFE, PSI and CCI) and the seasonal means plotted as error bars. One way Analysis of Variance (ANOVA) was used to examine if any statistical differences in the biological indices occurred between seasons. The results indicate that there is no statistical difference ($p > 0.05$) among any of the indices examined on the River Eye over the three seasons suggesting a stable community (Table 4.5 and Figure 4.8).

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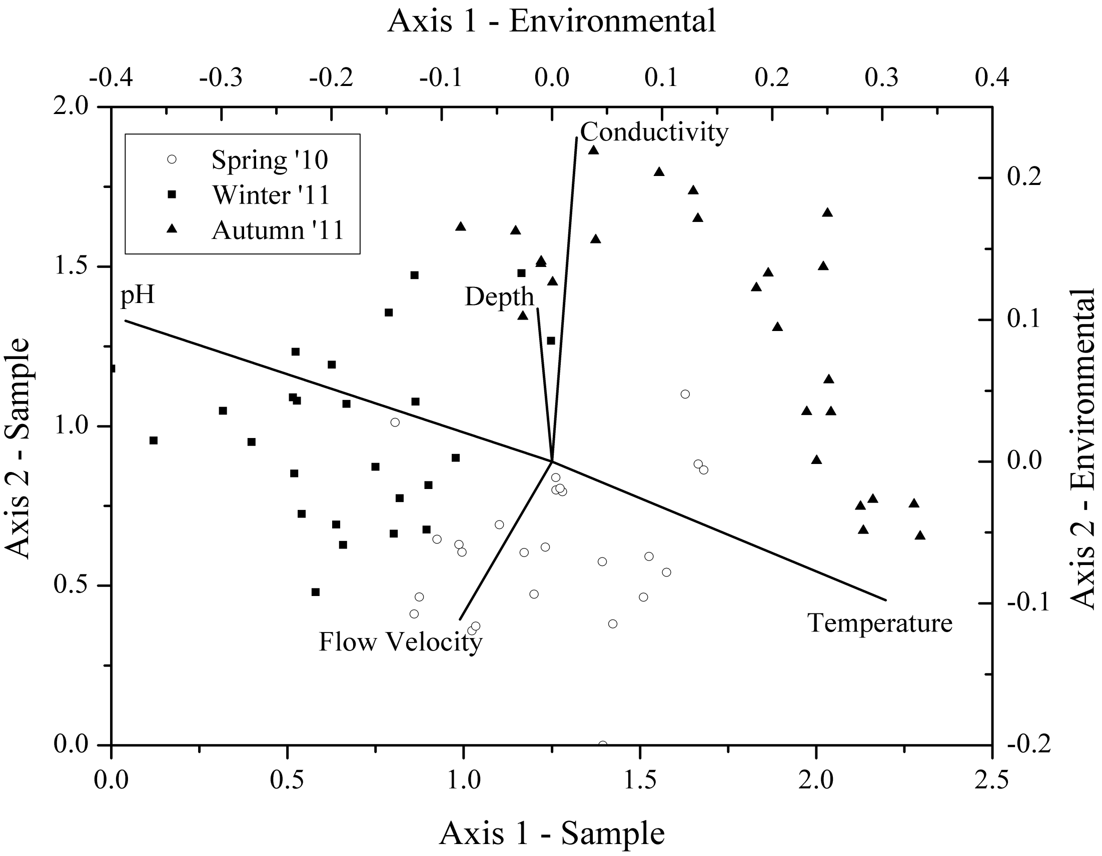

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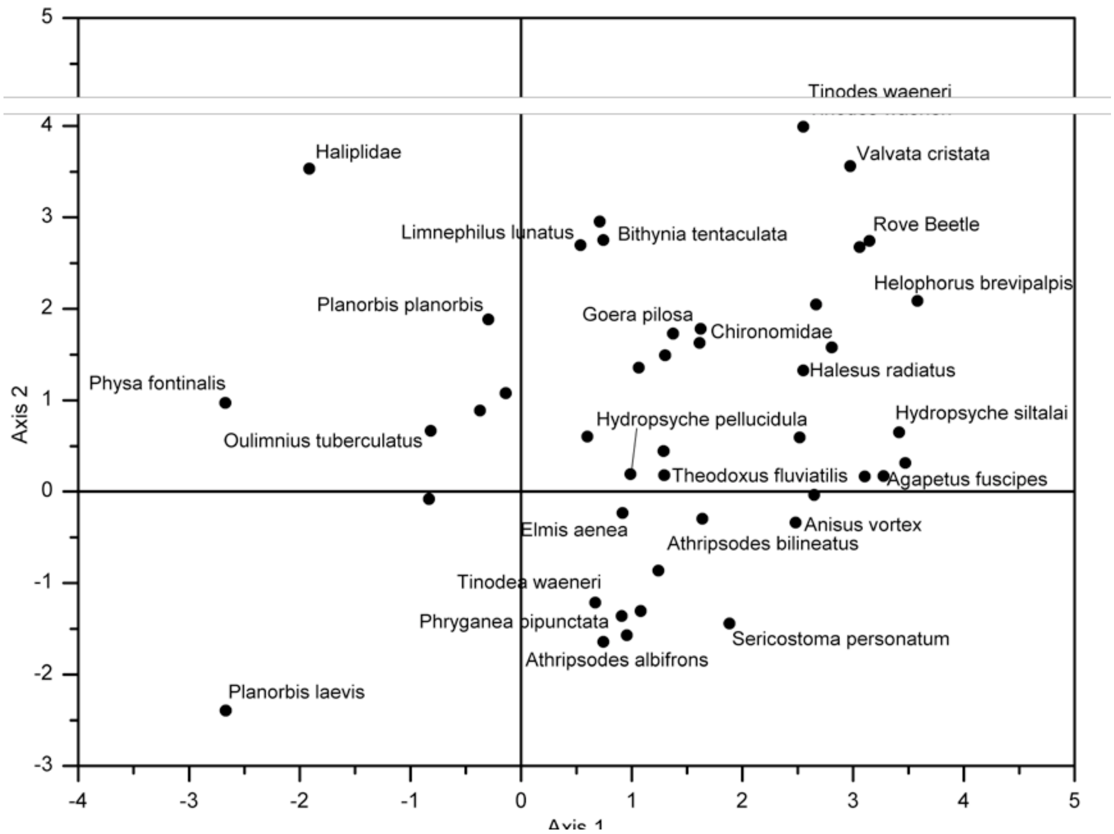


Figure 4.7 DCCA ordination of macroinvertebrate and abiotic data, with season as a covariable: i) Season-environment biplot; ii) Species-environment biplot.

Table 4.5 One-Way ANOVA results between each of the ecological indices.

		Degrees of Freedom	Sum of Squares	F-ratio
BMWP	Between groups	2	248.357	0.702
ASPT	Between groups	2	0.150	0.573
LIFE	Between groups	2	0.067	0.392
PSI	Between groups	2	243.949	0.695
CCI	Between groups	2	3.486	3.594

Examination of the raw scores indicated that the PSI score (Extence *et al.*, 2013) was low and that the river may be subject to sedimentation pressures. A PSI score of 21-40 indicates a river to be ‘sedimented’. The scores recorded during autumn (41-60), indicate the river to be ‘moderately sedimented’. Within the River Eye’s Condition Assessment (Natural England, 2013) fine sediment was identified as a major problem on the river, affecting water quality and covering river gravels used by fish for spawning. There are a number of sources of fine sediment such as bankside poaching, side ditches and tributaries, which bring sediments into the river from the surrounding arable fields. Impoundment structures such as weirs are also potentially detrimental with regards to the sediment problem owing to the reduced flushing flows, causing sediments to build up behind them.

The CCI scores have the clearest seasonal trend and provide an indication of a rivers potential conservation status and a basis for monitoring restoration programmes (Chadd & Extence, 2004). The scores for each season fall within the 5-10 range (Figure 4.8 v), indicating that the river supports at least one species of restricted distribution and/or a community of moderate taxon richness (Chadd & Extence, 2004). The highest scoring species included *Platycnemis pennipes, Athripsodes albifrons, Athripsodes bilineatus* and *Sigara stagnalis*, each of which has a Conservation Score (CS) of 5. This indicates that they are ‘local’ species with the remainder of the species being ‘very common’ to ‘occasional’ in occurrence.

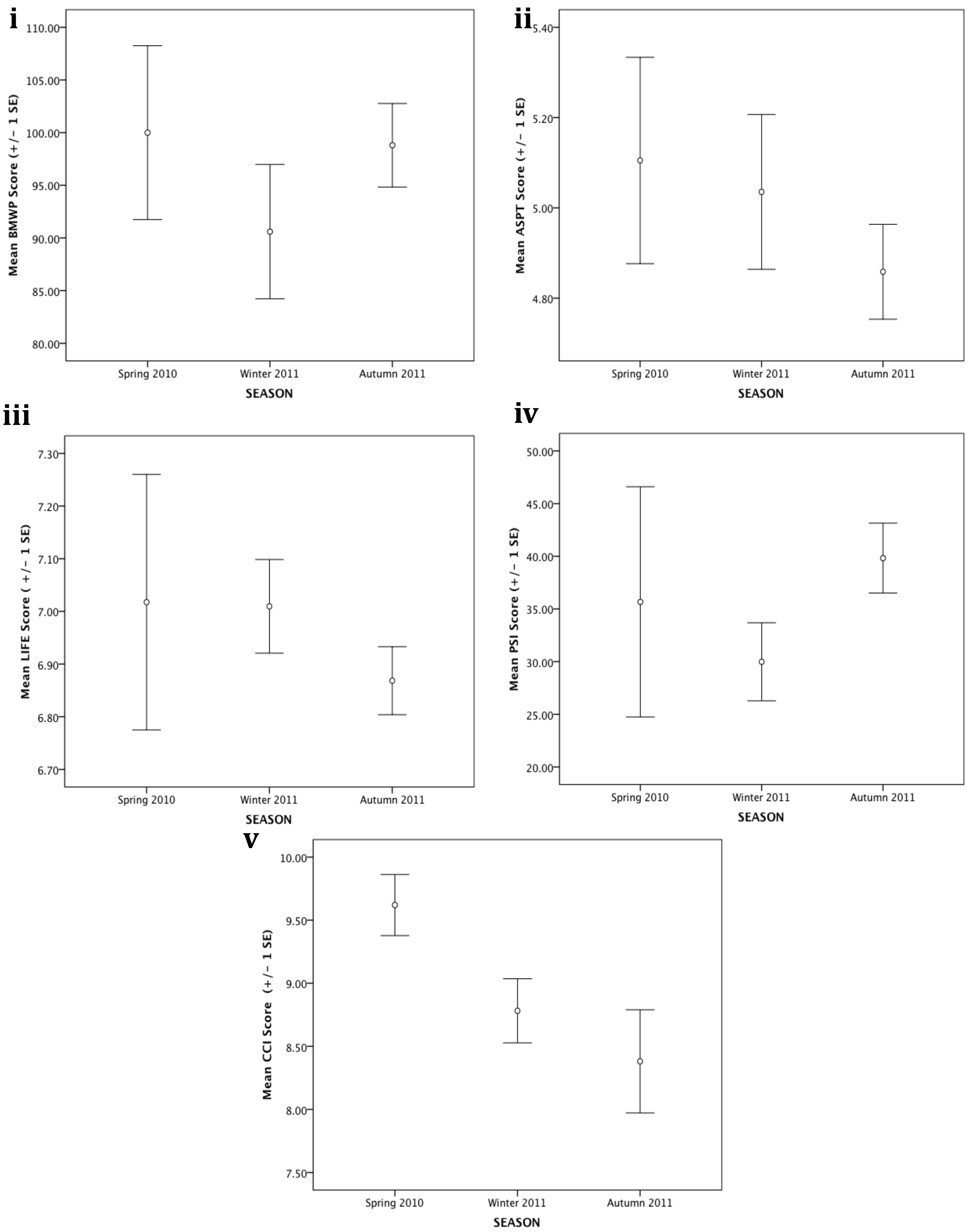


Figure 4.8 Error bar plots showing the mean i). BWMP score, ii) ASPT score, iii) LIFE score. iv) PSI score and v) CCI score (plus or minus one standard error) for each sampling season.

4.6 Results Comparing Contemporary River Ecology and Palaeoecology: River Eye

In order to facilitate a direct comparison between the contemporary and historic community, sub-fossil material was collected to compare with the contemporary Gastropoda, Coleoptera and Trichoptera communities. These three groups are well represented in the sub-fossil record and allow a quantitative comparison of the types of community recorded. These three groups comprise in excess of 60% of the total freshwater invertebrate taxa included in biomonitor metrics (Extence et al., 1999; Howard, 2010).

4.6.1 Detrended Correspondence Analysis (DCA)

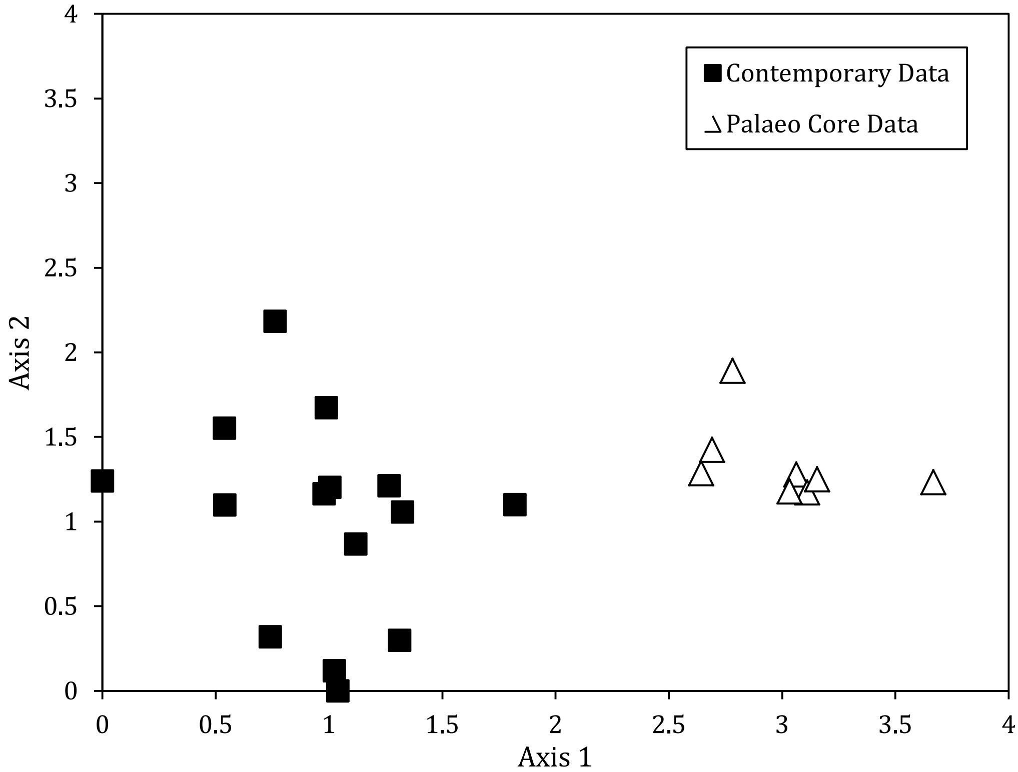
DCA was used to examine the similarities and differences in the contemporary and palaeoecological macroinvertebrate community structure. This analysis allows the investigation into similarities in communities over time (palaeoecological and contemporary samples) and particularly those taxa unique to any one set of samples. The first four axes accounted for a total of 39.1% of the variance in the faunal community data. Axis 1 accounted for 23.9 % of the variance, with axis 2 accounting for an additional 9.4% within the faunal community data (Table 4.6).

There is clear difference in community composition between the contemporary samples and the palaeoecological sample. (Figure 4.9 i). Both the contemporary and palaeoecological samples form two separate clusters indicating the presence of distinct taxa in each time period.

Table 4.6 Summary of the eigenvalues and cumulative variance of contemporary and palaeoecological data for DCA of kick and sediment pit samples from the River Eye.

Axes	1	2	3	4	Total inertia
Eigenvalues	0.500	0.197	0.075	0.048	2.095
Lengths of gradient	3.668	2.185	1.412	1.554	
Cumulative percentage variance of species data (%)	23.9	33.3	36.9	39.1	
Sum of all eigenvalues					2.095

i



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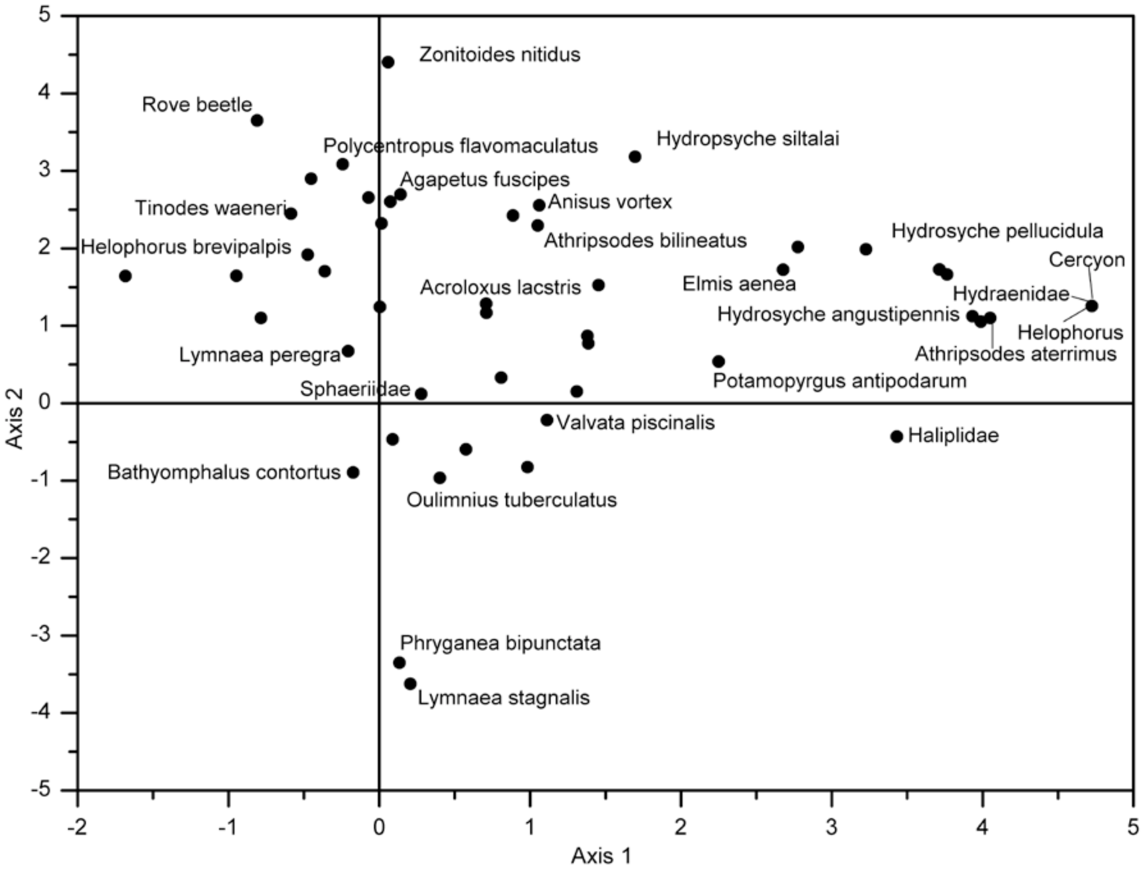


Figure 4.9 DCA faunal plot of presence/absence data: i) Different temporal data sets used ii) Faunal biplot.

Positioning of some taxa on the species ordination plot relate to their temporal occurrence (Figure 4.9 ii). For instance, *Zonitoides nitidus* and Rove beetles, which have the highest scores on axis 2, and *Lymnaea stagnalis* and *Phryganea bipunctata*, which have the lowest negative score on axis 2 are all only found in the contemporary biological samples. All taxa with a negative score on axis 1 were only recorded within the contemporary samples (e.g. *Lymnaea peregra, Helophorus brevipalpis, Polycentropus flavomaculatus* and *Tinodes waeneri*). Taxa unique to the palaeoecological samples (*Helophorus, Cercyon, Hydraenidae* and *Athripsodes aterrimus*) were clustered together at the positive end of axis 1. The taxa that were present in both contemporary and palaeoecological samples plotted near the centre of the ordination. To determine the potential, significant influence that species unique to one set of samples had on the results, the unique taxa were removed from the subsequent analysis and only taxa common to both the contemporary and palaeoecological samples were used in the following analysis (Table 4.7). In some instances species were merged to higher taxa as it is not always possible to resolve some sub-fossil material to species level during identification.

Table 4.7 List of species that were either removed or combined from the merged list of contemporary and palaeo species.

Removed Species	Combined Species
Lymnaea truncatula *Lymnaea stagnalis* *Lymnaea peregra* *Zonitoides nitidus* *Potamonectes depressus elegans* *Helophorus brevipalpis* *Helophorus* *Cercyon* *Hydraenidae* *Rove beetle* *Agapetus fuscipes* *Polycentropus flavomaculatus* *Tinodes waeneri* *Hydropsyche sp.* *Hydroptila sp.* *Phryganea bipunctata* *Limephilus lunatus* *Potamophylax angulatus* *Halesus radiatus* *Lepidostoma hirtum* *Tinodes waeneri*	*Valvata cristata* and *Valvata piscinalis.* *Planorbis carinatus, Planorbis leucostoma, Planorbis planorbis, Gyrautus albus, Bathyomphalus contortus, Anisus vortex* and *Acroloxus lacustris.* *Haliplus ruficollis* and *Haliplidae.* *Athripsodes aterrimus, Athripsodes albifrons* and *Athripsodes bilineatus.*

In addition, taxa that were identified to species level in contemporary samples, but could only be resolved to genus level in palaeoecological samples to facilitate analysis, were combined in a presence absence data set (Table 4.7). The DCA of common taxa in presence/absence format explained 51.3% of the variance across the first four axes. Axis 1 explained 30.4%, with axis 2 explaining a further 12.6% (Table 4.8).

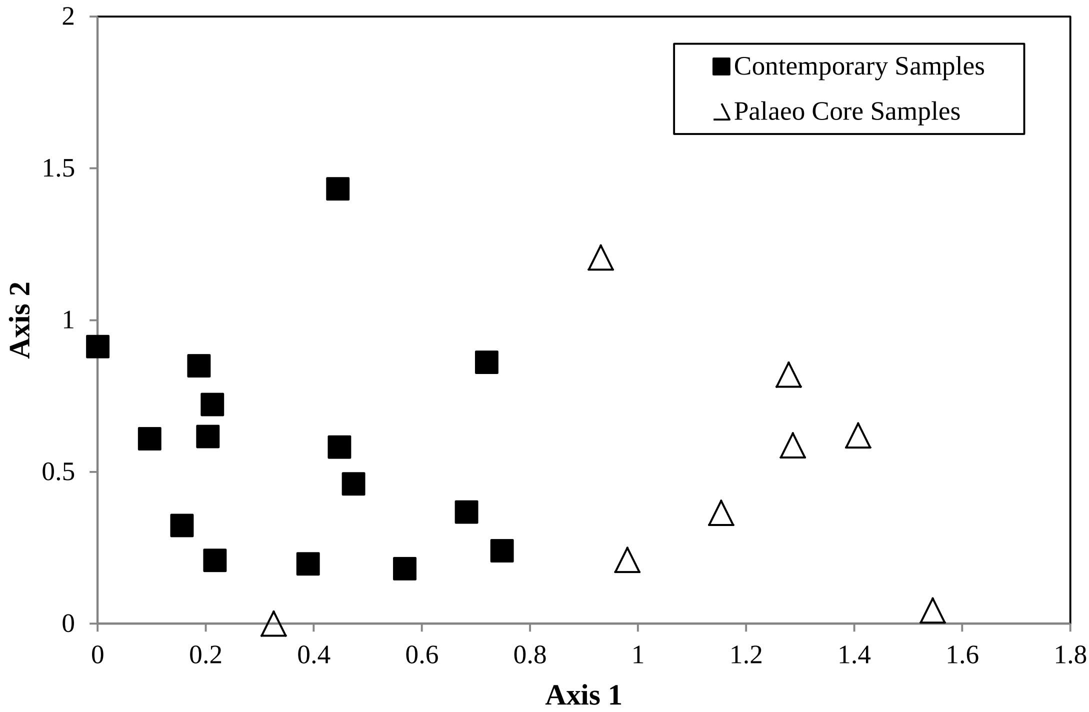
Table 4.8 Summary of the eigenvalues and cumulative variance of the presence/absence contemporary and palaeoecological data for the DCA of kick and sediment pit samples from the River Eye.

Axes	**1**	**2**	**3**	**4**	**Total inertia**
Eigenvalues	0.190	0.078	0.039	0.013	0.624
Lengths of gradient	1.546	1.432	0.966	1.097	
Cumulative percentage variance of species data (%)	30.4	43.0	49.2	51.3	
Sum of all eigenvalues					0.624

The analysis of common taxa across the contemporary and palaeoecological samples resulted in less separation of samples in ordination space. However, there was still a marked separation between the two sets of samples (Figure 4.10 i) indicating that community composition of these taxa has changed. Both sets of samples were less clustered however (than Figure 4.9 i), but differences in community composition are still apparent.

The species presented on Figure 4.10 ii are found within both the contemporary and palaeoecological samples. When observing the presence of individual taxa (Appendix 1), species with a negative axis 1 score, such as *Valvata piscinalis* or *Bithynia tentaculata*, occur more frequently in the contemporary samples compared to the palaeoecological samples. For instance, the snail *Valvata piscinalis* was recorded in 8 of the contemporary samples but was only found once in the palaeoecological sample. In contrast, species with a high, positive axis 1 score, such as *Hydropsyche angustipennis* and *Hyropsyche instabilis*, occured more frequently in the palaeoecological samples than in contemporary samples.

i



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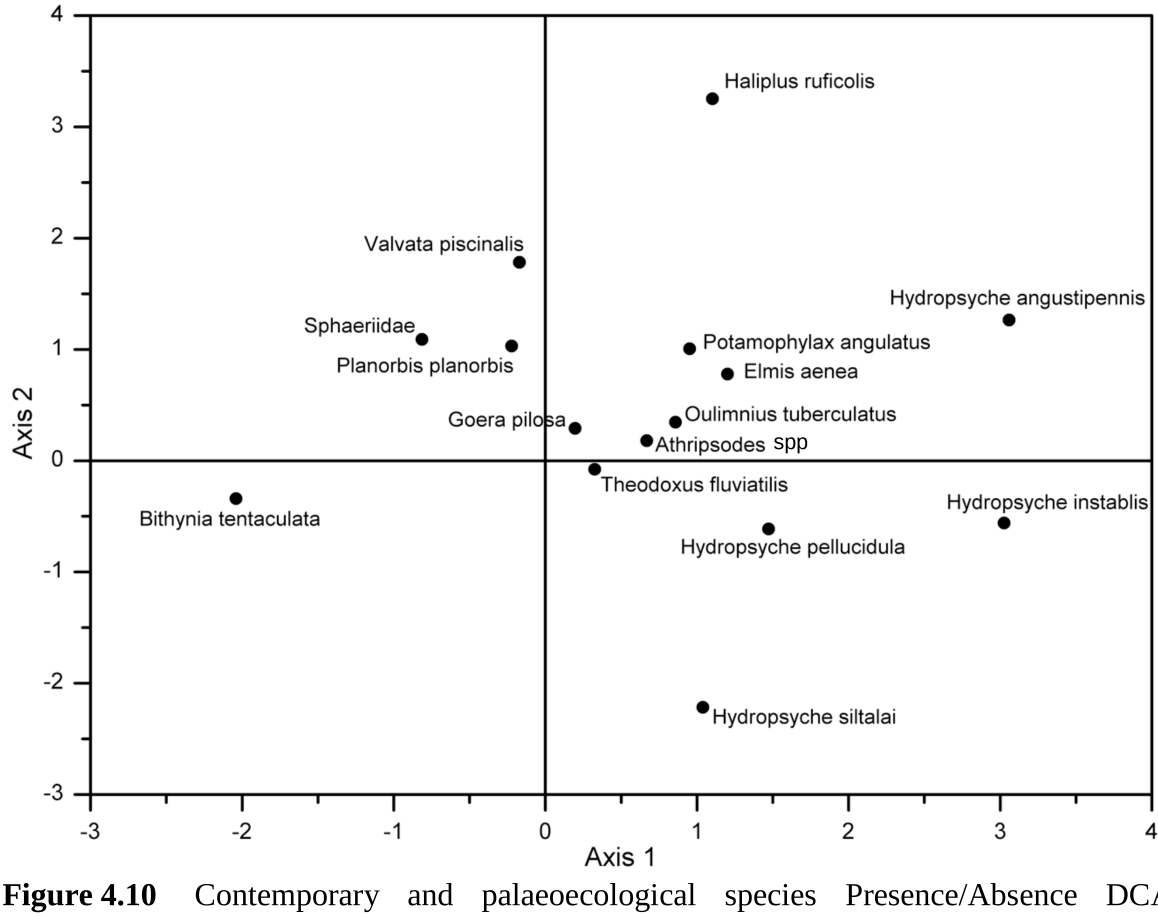


Figure 4.10 Contemporary and palaeoecological species Presence/Absence DCA ordination biplots: i) temporal variability; ii) Macroinvertebrate community data.

4.6.2 Ecological Indices

When the contemporary and palaeoecological macroinvertebrate indices were compared (Figures 4.11) it shows that the ASPT is less sensitive to seasonal variability than the BMWP scores. The palaeoecological samples (based on the combination of Gastropods, Coleoptera and Trichoptera) indicate that the historic River Eye had a significantly lower ($P < 0.001$ – using the Kruskal-Wallis test) BMWP score (48) compared to the contemporary river (71-96). However the ASPT results for the palaeoecological samples were not significantly different to that of the contemporary samples. It has been demonstrated in studies of biological water quality that the ASPT score is seasonally more robust temporally than the BMWP score (Armitage et al., 1983; Rodríguez & Wright, 1991). Examination of the number of taxa and LIFE flow groups indicated taxa associated with the full range of flow velocities, from fast to rapid flows (LIFE flow groups I and II), through to slow flowing and standing water habitats (LIFE flow groups IV and V) (Extence *et al.* 1999) in both contemporary and palaeoecological samples. Plotting the contemporary and the palaeoecological LIFE scores indicated that there was not a significant difference between the historic and the contemporary community (Figure 4.11 iii). The PSI scores for the palaeoecological results (Figure 4.11 iv) are slightly higher than that of the contemporary samples and were statistically different ($P < 0.005$ KW) to the contemporary winter samples. This palaeoecological PSI score indicates that the river (at least at the locations sampled), was ‘moderately sedimented’ even in the 1950s.

Table 4.9 Kruskal-Wallis pair-wise comparison between seasons for macroinvertebrate community indices along the River Eye. *$P < 0.05$, ** $P < 0.01$, *** $P < 0.001$.

		Spring	Winter	Autumn
BMWP	Winter	NS	-	-
	Autumn	NS	*	-
	Palaeo	***	***	***
ASPT	Winter	NS	-	-
	Autumn	NS	NS	-
	Palaeo	NS	NS	NS
LIFE	Winter	NS	-	-
	Autumn	NS	NS	-
	Palaeo	NS	NS	NS
PSI	Winter	NS	-	-
	Autumn	NS	*	-
	Palaeo	*	**	*

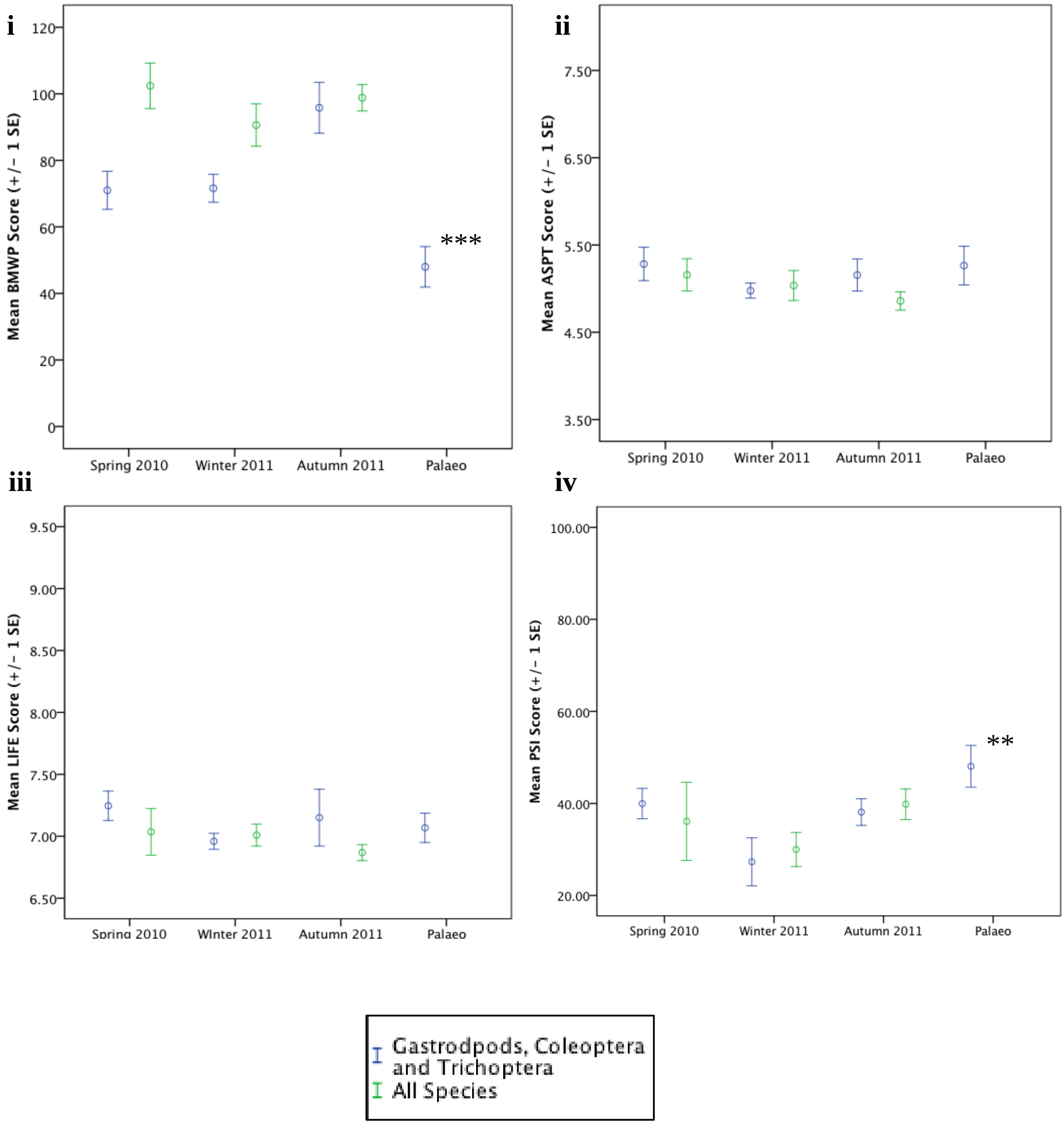


Figure 4.11 Error bar plot showing the comparisons between the i) mean BMWP score, ii) mean ASPT score, iii) LIFE score and iv) PSI score (plus or minus one standard error) for the contemporary and palaeoecological samples taken from the River Eye, and the full contemporary species list for each sampling season. *** P < 0.001, ** P < 0.005 using Krukal-Wallis.

4.7 Ecological interpretation

A total of 17 taxa were common to both the contemporary and the palaeoecological samples, many of which are described as widespread and common in the contemporary landscape (Wallace et al. 1990; Edington & Hildrew, 1995). However, when comparing the combined totals of Gastropods, Coleoptera and Trichoptera, there was a 54% increase in the number of species found in the contemporary sample, compared to the palaeoecological sample and there are a number of possible reasons for this. This could be due to taphonomic issues, where the sub-fossil fragments may be eroded by in-stream sediments, washed away or completely destroyed preventing their preservation. This may be an issue for both *Athripsodes sp* and *Lepidostoma hirtum.* However there was no evidence of this in the material examined and no evidence of degradation. *Lepidostoma hirtum* live on the surface of the substrate, when they either pupate or die, they may be washed away preventing them from settling within the riverine sediments and their subsequent fossilisation. However, they have been recorded in previous studies and their absence from the palaeoecological samples may therefore reflect recent colonization of the site. The taxa unique to the palaeoecological sample include; *Helophorus, Cercyon, Hydraenidae* and *Athripsodes aterrimus*. *Athripsodes aterrimus* is widespread and common and proved to be the most abundant unique palaeo-species, found in 7 out of the 8 palaeoecological samples. It is indicative of slow to moderately flowing waters and usually found among plants and on muddy sand (Eutaxa, 2013).

Through close examination of the ecological preferences of the contemporary and sub-fossil species, a reconstruction of the past riverine environment is possible. Hydropsychidae are net spinning caseless caddisflies that catch their food in flowing water. Brunke *et al.* (2001) found that the lower and upper tolerances of flow velocity for this species is 8.3cm s^{-1} and 38.3cm s^{-1} in the River Spree, Germany. In British rivers there are well known marked downstream sequences of species within the families of Hydropsychidae and Polycentropodidae that have been linked to physical gradients along watercourses and are associated with flow velocity (Edington & Hildrew, 1995). For example *Hydropsyche pellucidula* (recorded in both the contemporary and the palaeoecological community of the River Eye) is usually found in the middle reaches of rivers (Edington & Hildrew, 1995). However, *Lepidostoma hirtum* which is also indicative of relatively fast flow velocities and *Polycentropus flavomaculatus*, which is a species commonly found in slower flowing water or marginal areas of the lower reaches of a river system (Wallace et al., 2003), were only recorded within the contemporary samples.

As well as providing a picture of the flow regime, other species create an insight into the composition of the substrate. For instance *Goera pilosa* is associated with fast flowing water and is classified as LIFE Flow Group 1 by Extence *et al.* (1999). However, it also has an affinity for gravel substrates and this was found in both the contemporary and palaeoecological communities. *Athripsodes aterrimus,* which are only found in the palaeo samples, is often associated with mud and sandy substrates and found within more eutrophic, stagnant waters. However, *Athripsodes albifrons* which is associated with stony substrata (Wallace et al., 2003) and *Valvata cristata*, which is largely restricted to well-oxygenated, slowly flowing or still water, with a strong preference for richly vegetated places on muddy substrates were only found within the contemporary community (Kerney, 1999). *Potamonectes depressus elegans*, a predatory diving beetle and the robust riffle beetles; *Elmis aenea* and *Limnius volkmari,* are typically associated with unpolluted water and gravelly, sandy river beds (Nilsson & Holmen, 1995). These species were found in abundance in both sets of samples. This suggests there was and remains to be a heterogeneous set of substrates and habitat patches within the River Eye study reach. However, the installation of weirs downstream of the SSSI and the recent flood defences may impact on the relative proportions of riverine substrates. Due to the influences these structures have on the River Eye, it has created significant ponding behind them. This may be the reason for the significantly reduced the total area of riffle habitat and reduced heterogeneity of mesohabitats present on the River Eye compared to the historic conditions.

Sedimentation is a naturally occurring phenomenon within rivers (Wood & Armitage, 1997) although human activities, primarily agriculture in the case of the River Eye, have greatly increased this natural process. The increasing quantities of eroded sediment inputs sourced from agricultural runoff may be one of the factors influencing the natural faunal assemblage of the river (Natural England, 2013b). High sedimentation rates can have a marked adverse impact on primary productivity and faunal diversity through the reduction of light penetration which, as a result, reduces photosynthesis (Richards et al.,1993). Aquatic macrophytes however, have an important role in helping to reduce sedimentation problems as they effectively act like a sieve, trapping settling sediment particles (Carpenter and Lodge, 1986). There are high abundances of aquatic fauna present within the river showing that primary and secondary production is high and suggesting that sedimentation has not degraded the system to the extent that macrophytes are damaged or excluded. Sedimentation accompanied by excessive algal growth within the River Eye (Natural England, 2012), has been identified as a potential problem (Plate 4.5) and this is clear when looking at its past and contmporary macroinvertebrate community. The presence of *Potamopyrgus antipodarum*, an invasive mud snail, provides an indication of the presence of fine sediments within a river due to their tolerance of high siltation rates and preference for fine sediment deposits (Elder & Collins, 1991). As well as providing evidence for the presence of fine sediment within the river system, *Potamopyrgus antipodarum* thrives in eutrophic waters as it feeds on epiphytic algae (Elder and Collins, 1991). *Hydropsyche instabilis* is another species that is tolerant to eutrophic waters, providing an indication of eutrophic water. Both of these taxa were found in both the contemporary and palaeoecological samples, suggesting that water quality has not changed significantly since the late 1960s and that agricultural intensification following World War II may have led to nutrient enrichment even by the mid 1950s.

Plate 4.5 A River Eye sediment trap *in-situ* after being in the river for a period of 21 days. This clearly shows sedimentation and excessive algal growth to be a significant problem within the river.

4.8 Summary

In this chapter the River Eye SSSI study reach has been introduced and historic channel changes have been outlined through the use of historical maps. The results of this investigation indicate the following:

- Through the use of DCCA and DCA, it has demonstrated that there is a strong seasonal environmental gradient within the contemporary data obtained.
- Using WFD metrics, palaeoecological BMWP and PSI scores were found to be significantly different from the contemporary scores indicating that the river in the 1950s had slightly higher sedimentation levels.
- The historic and current macroinvertebrate communities of the River Eye are broadly comparable in structure and biotic scores.
- Man-made structures located within and directly downstream of the SSSI (e.g. weirs and flood defences), are probably having an adverse effect on river sediments due to ponding, thus reducing riffle habitats and biotope mesohabitat diversity. However, while it is not possible to evaluate the loss of riffle (or any other biotope) area with the palaeoecological methods used, the approach does allow an evaluation of the changes in proportion and total habitat area present within individual reaches (in the reach upstream of the area where the sediment cores/samples were collected).
- Moderate to high sedimentation rates have been identified as the major underlying problem causing the river to be of an ‘Unfavourable no change’ condition.

CHAPTER FIVE: THE RIVER HULL

5.1 Introduction

This chapter examines the macroinvertebrate community of a reach of the River Hull headwaters (East Riding of Yorkshire). In particular it explores how the community has changed between 1849 and 2012 by comparison between the contemporary macroinvertebrates and those from the sub-fossil community preserved in the historic palaeochannel. The River Hull provided a valuable opportunity to examine the concept of multiple reference conditions primarily due to the availability of data from two historic studies by Whitehead (1935) and Pearson and Jones (1984), whose sites were in close proximity to the ones chosen for this thesis. This provided two additional reference points to compare both contemporary and palaeoecological samples to help define 'reference conditions' and allowed a more detailed insight into the changing river. The chapter provides a detailed site description of the River Hull SSSI and highlights the four sites that were chosen along the contemporary channel from which seasonal macroinvertebrate sampling was performed. The results obtained from the contemporary and palaeoecological analysis are used to interpret the changing nature of the macroinvertebrate community and habitat characteristics in the River Hull. The objectives of this chapter are to (refer to Chapter 1.2 for relevant objective):

- Explore the contemporary riverine environment of the River Hull and derive macroinvertebrate community biotic indices;
- Identify the River Hull's historic palaeochannel and characterise the palaeoecology through sediment core extraction and mulitproxy palaeoecological techniques to define a historic reference condition.
- Compare the contemporary and palaeoecological results to understand the changes that have taken place within the River Hull study reach.

5.2 The River Hull

The headwaters of the River Hull have been recognized as nationally important by Natural England, being designated as a SSSI in 1988 (Natural England, 2009) as the most northerly chalk stream system in Britain. The River Hull SSSI headwaters are located north of Kingston-Upon-Hull close to the town of Driffield (see Figure 5.1). The River Hull SSSI runs for approximately 30km and comprises two main tributaries West Beck and Frodington Beck (see Figure 5.2). The Hull continues downstream of the SSSI and eventually discharges into the Humber Estuary. Tributaries of the West Beck include Driffield Beck, Driffield Trout stream and Eastburn Beck (where the primary study sites for this research were located).

The River Hull catchment is low lying, with the vast majority of land elevation only 10m above the Ordnance Datum and as a consequence it has limited natural containment features for flood water (Environment Agency, 2007). The land use surrounding Eastburn Beck has remained predominantly agricultural, including both grazed and arable land, since the 1850s except for the expansion and urbanization of the town of Driffield.

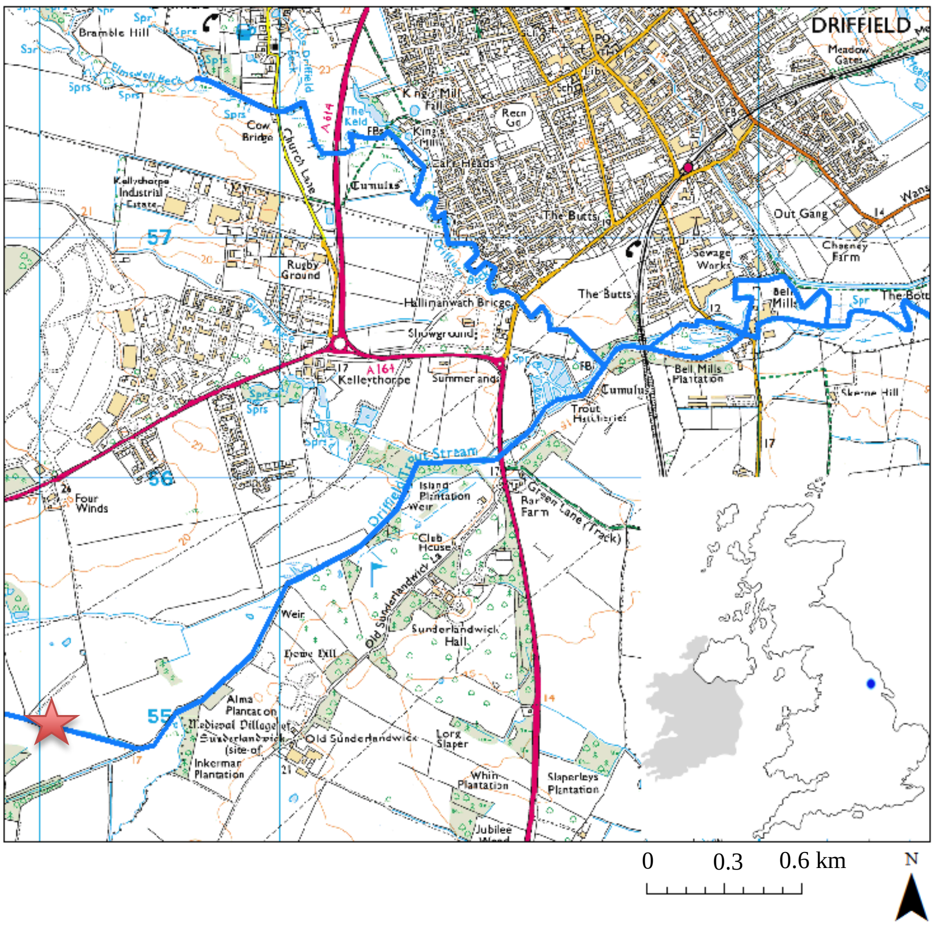


Figure 5.1 Location map of the River Hull study area (star indicating site location), in relation to the rest of the United Kingdom (EDINA Digimap, 2013).

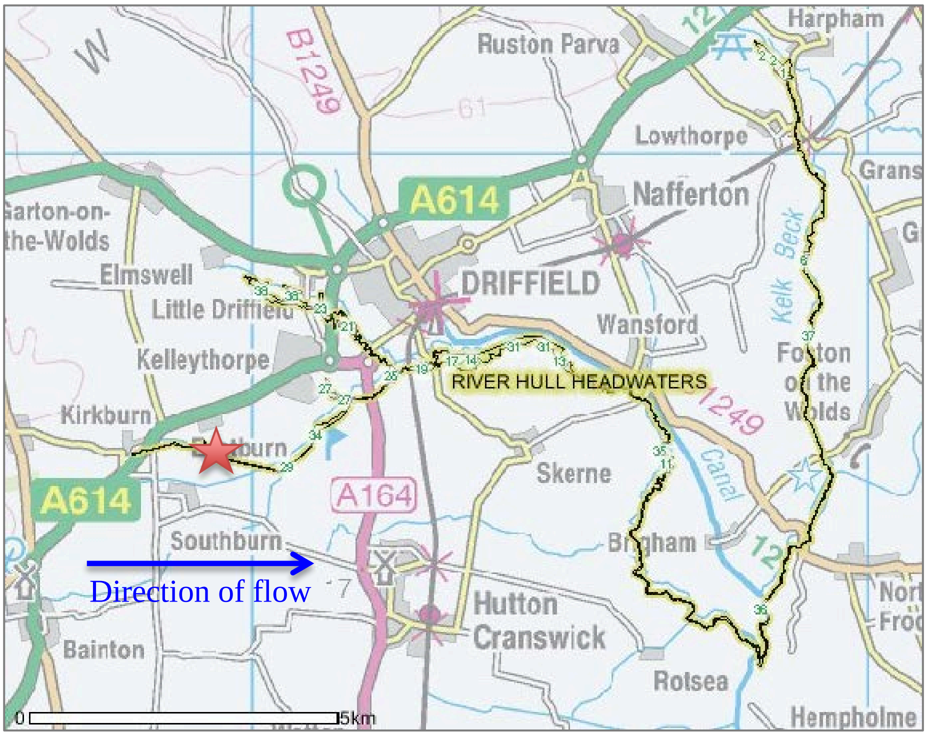


Figure 5.2 Map showing the full extent of the River Hull SSSI (Natural England, 2012a).

The underlying Cretaceous Chalk geology of the catchment has a strong influence on the character of the river. The headwaters of the River Hull are predominantly fed by base flow from the underlying aquifer (Natural England, 2012b). Chalk aquifers are highly permeable and fractured, typically with thin soil cover (Smith et al., 2003). The fractured nature of chalk aquifers and limited protection offered by the thin overlying soils, mean that agricultural nitrate fertilizers are potentially able to enter ground water relatively easily (Berrie, 1992). As a result, the study area has been designated as a Nitrate Vulnerable Zone (Environment Agency, 2007). The river is typical of chalk rivers in the UK, characterized by stable, clear flowing water with a distinct seasonal flow regime (Environment Agency, 2003). The superficial geology (Figure 5.3) has a strong influence on the character of the SSSI with gravel, sand and silts dominating the substratum of the riverbed. The headwaters have been modified via anthropogenic management, such as weirs and the development of a trout farm downstream of Driffield. Despite these historic modifications, the riverbed remains morphologically diverse with deep pools interspersed with shallow riffles, allowing macrophytes to thrive. In particular, macrophytes that prefer slower flow velocities are common including Shining pond weed (*Potamogeton lucens*) and Water crow-foot (*Ranunculaceae*), which is recognized to be characteristic of chalk streams (Mainstone, 1999; Environment Agency, 2008). According to the site condition assessment information compiled in May 2012, 72% of the River Hull Headwaters SSSI is currently in an unfavourable recovering state (Figure 5.4) (Natural England, 2012b). The presence of numerous weirs and sluices throughout the catchment has resulted in localised ponding of water upstream of the structures, and silt deposition on the bed. The large sections of artificially straightened river channel lacks morphological diversity and is subject to excessive siltation. Table 5.1 describes the key conservation and management issues currently facing the River Hull.

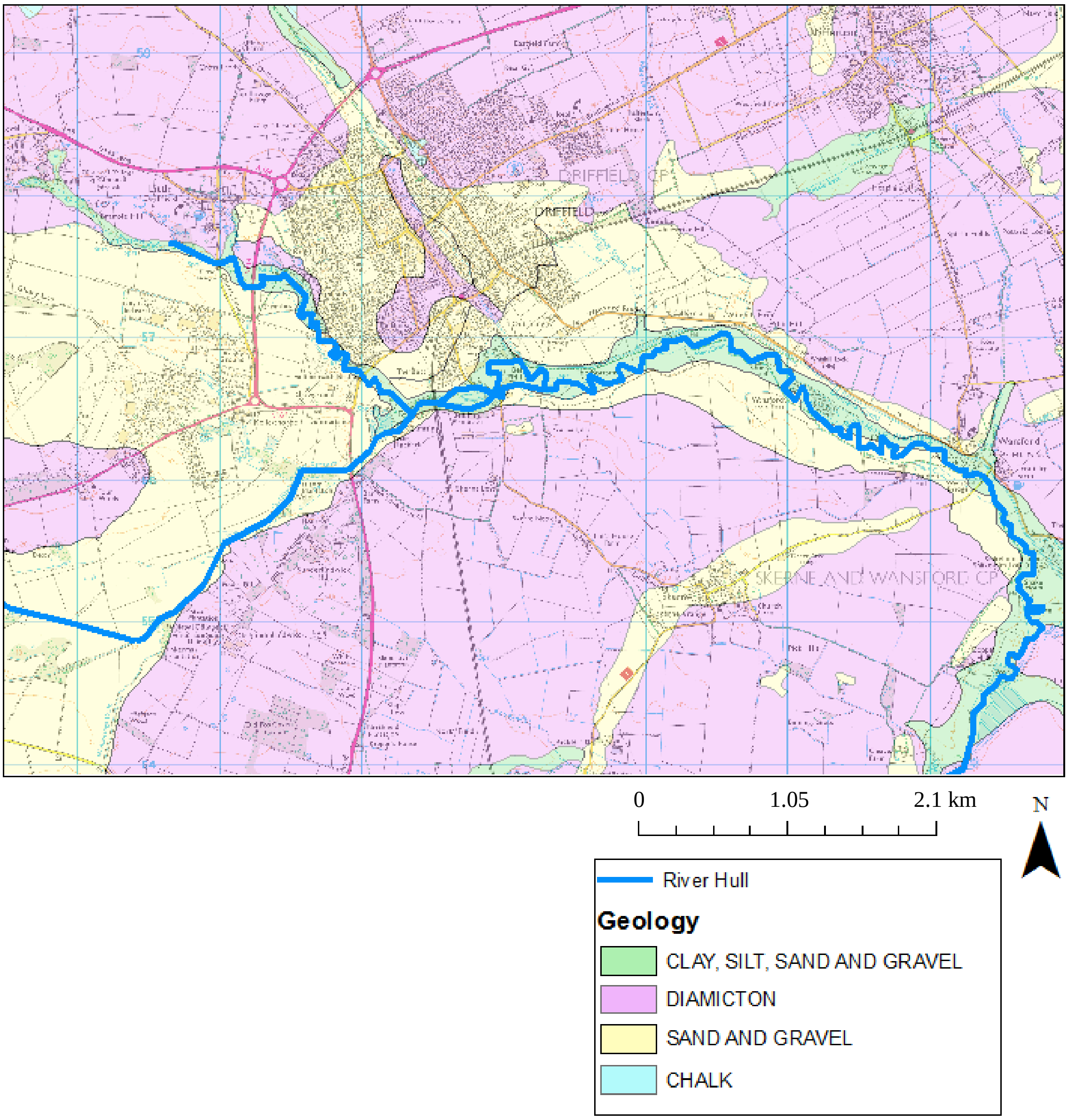


Figure 5.3 Map showing the surface geology of the River Hull SSSI (EDINA Geological Digimap, 2012).

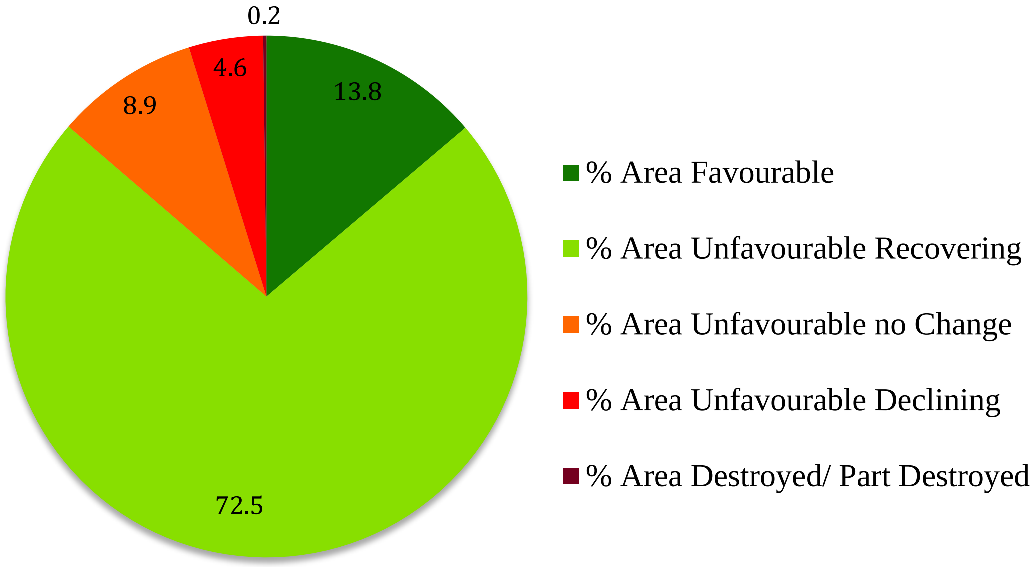


Figure 5.4 The condition summary for the River Hull complied by Natural England on 1st May 2012 (Natural England, 2012b).

Table 5.1 Summary of the current management and conservation issues identified as affecting the River Hull Headwaters modified from Natural England (2009).

Key Issues	Impacts
Fine sediment deposition	Largely attributable to inputs from field drains and tributaries. Accumulations upstream of weirs and mills is having a detrimental effect on main stream habitat requirements of key interest species including *Ranunculus spp.*, *Potamogeton spp.*, Brown Trout and Grayling. Fine sediment smothers clean gravels and reduces diversity of bed features.
Channelisation and disconnection of the river from the floodplain	Historical modifications have altered the hydrological regime through the creation of long stretches of uniform flow, potentially reducing habitat heterogeneity.
Lack of bankside shelter and over shading	Lower reaches of the headwaters have limited tree and shrub cover causing a lack in morphological diversity. Upper reaches suffer from overshading, preventing light reaching the channel. This restricts the growth of the submerged plant communities.
In-channel structures	In-channel weirs are disrupting the rivers continuity and causing fine sediment accumulation.

5.3 Study Sites

Four sites were selected for contemporary macroinvertebrate sampling (Figure 5.5). The banks lining each of the four sampling sites were steep and approximately 1.5m high. Scattered light shading occurred from trees lining the river and both banks were fenced for the majority of the reach. Dominant land use in the area is pastoral grazing (Plate 5.1). Four sediment cores were extracted from a palaeochannel that was identified following a preliminary site visit and the use of historical maps. Plate 5.2 shows the extraction of the cores from the palaeo channel and three sections of the cores extracted (see Chapter 3.5.1 for extraction methods used).

Figure 5.5 Aerial photograph and map showing the locations of the contemporary and palaeoecological sampling sites (Tele Altas, 2012).

Plate 5.1 Study reaches along the River Hull in Summer 2011 (GPS coordinates: 53.981923, -0.481821).

Core 1 Core 2 Core 3

50 cm deep

150 cm deep

Plate 5.2 Core extraction from the River Hull palaeochannel and photographs of Core 1 – 3 (GPS coordinates: 53.983563, -0.465353).

5.4 The Historical River Hull

Historically, the River Hull headwaters were comprised of extensive areas of marshland. However due to a number of anthropogenic interventions during the 18th and 19th centuries, including the construction of a major land drainage scheme to enhance agricultural productivity, coupled with the straightening of river channel, the landscape has changed dramatically (Environment Agency, 2003). The results of this land drainage channelization process resulted in the majority of the catchment being used for agriculture with a reduction in flood frequency due to regular dredging. This also resulted in steeply graded river banks, which increased bank-full capacity (Environment Agency, 2006). Dredging was also undertaken prior to 1900 to maintain commercial navigation, resulting in some of the lower reaches of the SSSI being embanked (Natural England, 2009). Weir maintenance, renovation and installation occurred in several parts of the river. However, these structures are currently having a detrimental effect on the instream habitat, through impounding water and acting as river bed controls, which limit the adjustment of the channel bed gradient (Natural England, 2009). Through the examination of historical maps of the River Hull SSSI (Figure 5.6) there is clear evidence of anthropogenic channel movement between 1849-1899. The historic path of the river (dark blue) indicates a meandering channel prior to 1845 and the current contemporary channel (light blue) shows an anthropogenically-straightened river. The current river is more than 2m below the flood plain, whereas in comparison, through sampling the palaeochannel, the historical riverbed was no more than 50cm below the base of the current palaeochannel surface depression. This indicates that through the processes of dredging and incision, the contemporary channel has become more disconnected from the surrounding floodplain.

With regards to the stratigraphy to the cores extracted the top 25-30cm comprised of top soils with a band of clay going down to a depth of 50-55cm. The clay gradually became sandier, turning into a band of chalk. The key historic river sample zone was found between 60-80cm deep. Below the historic river bed the material returned to clay.

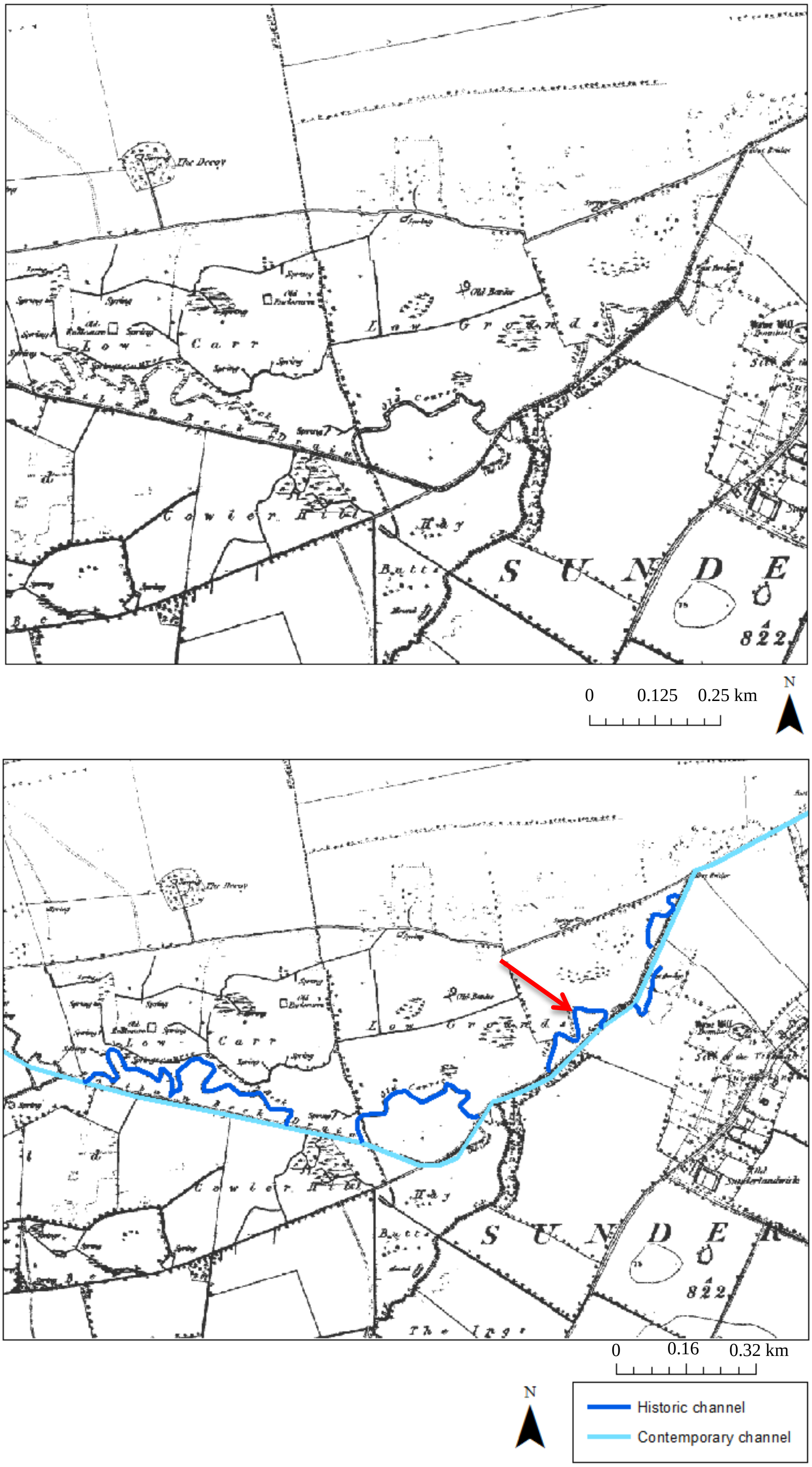


Figure 5.6 Country Series (1:10560) OS map, dated 1849-1899, of the River Hull depicting the contrast between the historic and contemporary channel (EDINA Digimap, 2013). Arrow indicates palaeo sample extraction location.

5.5 Results of the Contemporary Ecological Community Composition

The abiotic variables of the contemporary sampling site were measured (see Methodology 3.4.1) and are summarised in Table 5.2. Flow velocity varied significantly from 0.09 sm^{-1} in autumn to 0.79 sm^{-1} in spring. The pH for the river ranged from 7.4 to 8.5, which shows the river to be alkaline in nature. The rivers conductivity ranged from 420 µS cm^{-1} to 564 µS cm^{-1}.

Table 5.2 Table highlighting the range of the abiotic measurements and their averages indicated in brackets, for each seasonal sampling undertaken along the River Hull SSSI.

		Sampling Season		
		Spring 2011	**Summer 2011**	**Autumn 2011**
Abiotic Measurement (Range)	**Depth** (m)	0.29 – 0.49 (0.364)	0.26 – 0.40 (0.334)	0.6 – 0.81 (0.212)
	Flow Velocity (m s^{-1})	0.51 – 0.79 (0.63)	0.32 – 0.79 (0.58)	0.09 – 0.37 (0.21)
	pH	8 – 8.5 (8.3)	7.4 – 7.7 (7.6)	8.3 – 8.5 (8.4)
	Conductivity (µS m^{-1})	0.56 – 0.56 (0.56)	0.42 – 0.43 (0.425)	0.55 – 0.56 (0.555)
	Temperature (°C)	11.2 – 11.6 (11.45)	10.2 – 11.6 (10.68)	14.3 – 14.6 (14.43)

Macroinvertebrate sampling was undertaken in spring (March) 2011, summer (June) 2011 and autumn (September) 2011 (refer to Chapter 3.4.3 for methods used). This helped to ensure that the full range of species inhabiting the river was sampled. The seasonal sampling programme was essential as some taxa such as the fish leech *Piscicola geometra; Oulimnius tuberculatus* (Coleoptera) and the caddis fly; *Potamophylax cingulatus* (Trichoptera) were only recorded in a single season reflecting natural seasonal variability. A total of 22 taxa was recorded in the contemporary River Hull. The fauna was dominated by the fresh water shrimp, *Gammarus pulex* and the caddisfly larvae, *Agapetus fuscipes* (Trichoptera), which accounted for 45% and 33% of the total number of individuals recorded respectively.

Table 5.3 Macroinvertebrate taxa recorded from the contemporary kick and Surber samples collected from the River Hull over three seasons (Spring, Summer and Autumn 2011).

	Spring 2011	Summer 2011	Autumn 2011
Potamopyrgus jenkinsi	x	x	x
Zonitoides nitidus	x	x	-
OLIGOCHAETA	x	x	x
Piscicola geometra	-	x	-
Glossophonia complenata	x	-	x
Erpobdella octoculata	x	x	x
Asellus meridianus	-	-	x
Gammarus pulex	x	x	x
Baetis rhodoni	x	-	-
Ephemeralla ignita	x	x	x
Elmis aenea	x	x	x
Elmis aenea (larvae)	x	x	x
Oulimnius tuberculatus	-	-	x
Rhyacophila dorsalis	-	x	x
Agapetus fuscipes	x	x	x
Drusus annulatus	x	x	x
Potamophylax cingulatus	x	-	-
Silo nigricornis	x	x	x
Sericostoma personatum	x	x	x
Chironomidae	x	x	x
Simuliidae	x	x	-
Psychodidae	x	-	-
Dinocrota	x	x	x

5.5.1 Detrended Correspondence Analysis (DCA)

DCA was undertaken using Canoco (ter Braak & Šmilauer, 2002) on the River Hull kick sample data to explore the contemporary macroinvertebrate community structure and variability over spring, summer and autumn 2011. Eigenvalues for this analysis are displayed in Table 5.5 and provide a measure of the relative importance of each axis in the analysis. The cumulative percentage of variance explained by the four canonical axes was 59.7%. Axis 1 explains 37.4% of the total variation within the species data with axis 2 explaining an additional 15.1%.

Table 5.4 Summary of eigenvalues and variance of species data for DCA of kick samples collected from the River Hull.

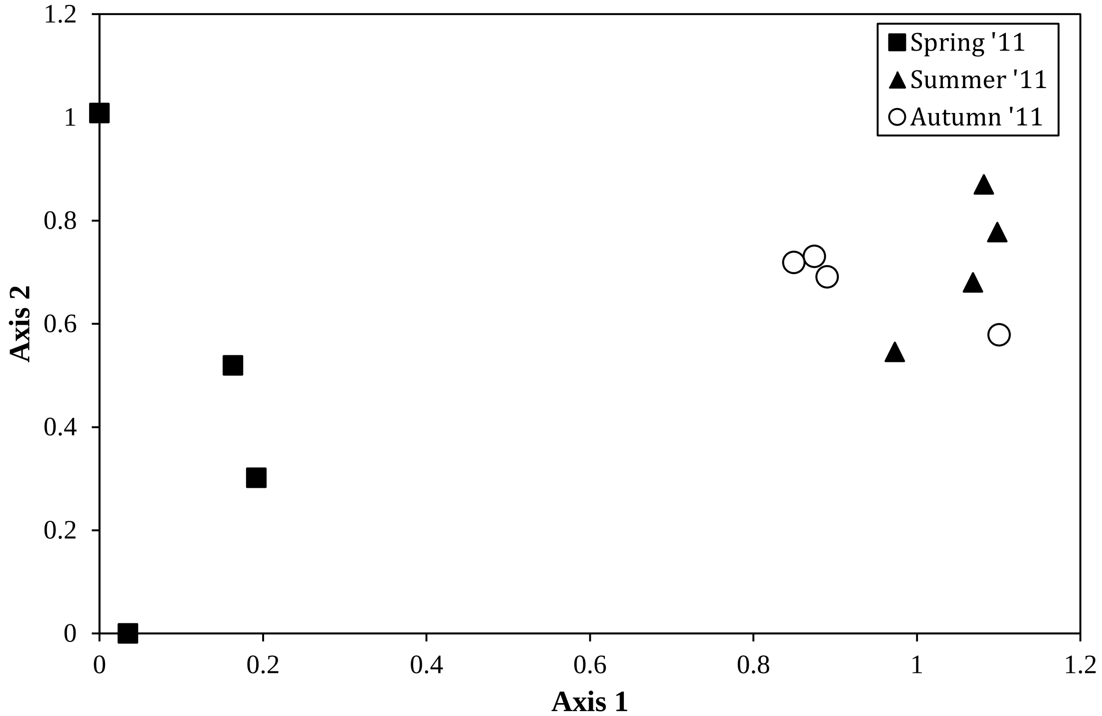
Axes	1	2	3	4	Total inertia
Eigenvalues	0.182	0.074	0.029	0.005	0.486
Lengths of gradient	1.101	1.008	0.725	0.686	
Cumulative percentage variance of species data (%)	37.4	52.5	58.6	59.7	
Sum of all eigenvalues					0.486

There is a slight overlap between the summer and autumn samples, however seasonal change is evident between the three seasons (Figure 5.7.i). Summer and autumn samples were loaded positive on axis 1 while the spring samples all had low axis 1 scores (but highly variable axis 2 scores). The location of some taxa on axis 1 and 2 (Figure 5.7.ii) reflects the seasonal patterns in abundance, for instance the mayfly, *Baetis rhodani* and Psychodidae (Diptera) that were recorded during spring, both have low axis 1 scores. *Piscicola geometra* (fish leech) was only found in summer and *Oulimnius tuberculatus* (Coleoptera) was only found in autumn, both having high axis 1 scores. Species such as *Elmis aenea* (Coleoptera), *Agapetus fuscipes* (Trichoptera) and *Silo nigricornis* (Trichoptera) were clustered around the centre of the ordination, reflecting the fact that they were recorded during all sampling seasons.

5.5.2 Detrended Canonical Correspondence Analysis (DCCA)

DCCA was undertaken using the contemporary Surber sample data in Canoco (ter Braak & Šmilauer, 2002). This allowed analysis of the abiotic indices against taxa abundance to be undertaken and to determine how influential the environmental variables were (the eigenvalues are displayed in Table 5.4 A). The first four axes accounts for a total of 15.9% of the variance in the species data and 64% of the species-environment relationship. Axis 1 accounted for 13.1% of the species data variance and 49.1% of the species-environment data with conductivity ($p < 0.005$) and pH ($p < 0.01$) having a significant influence on the ordination (Table 5.4 B).

i



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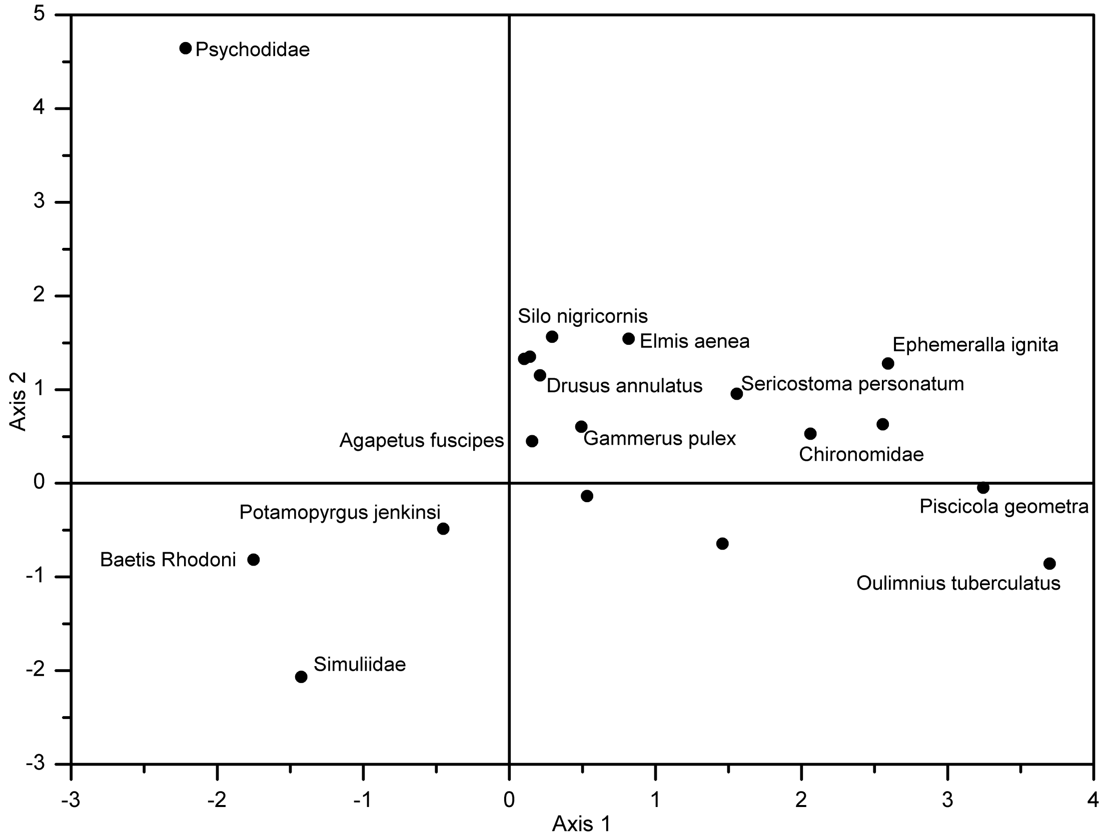


Figure 5.7 DCA ordination biplots for the River Hull: i) seasonal variability; ii) Macroinvertebrate community data.

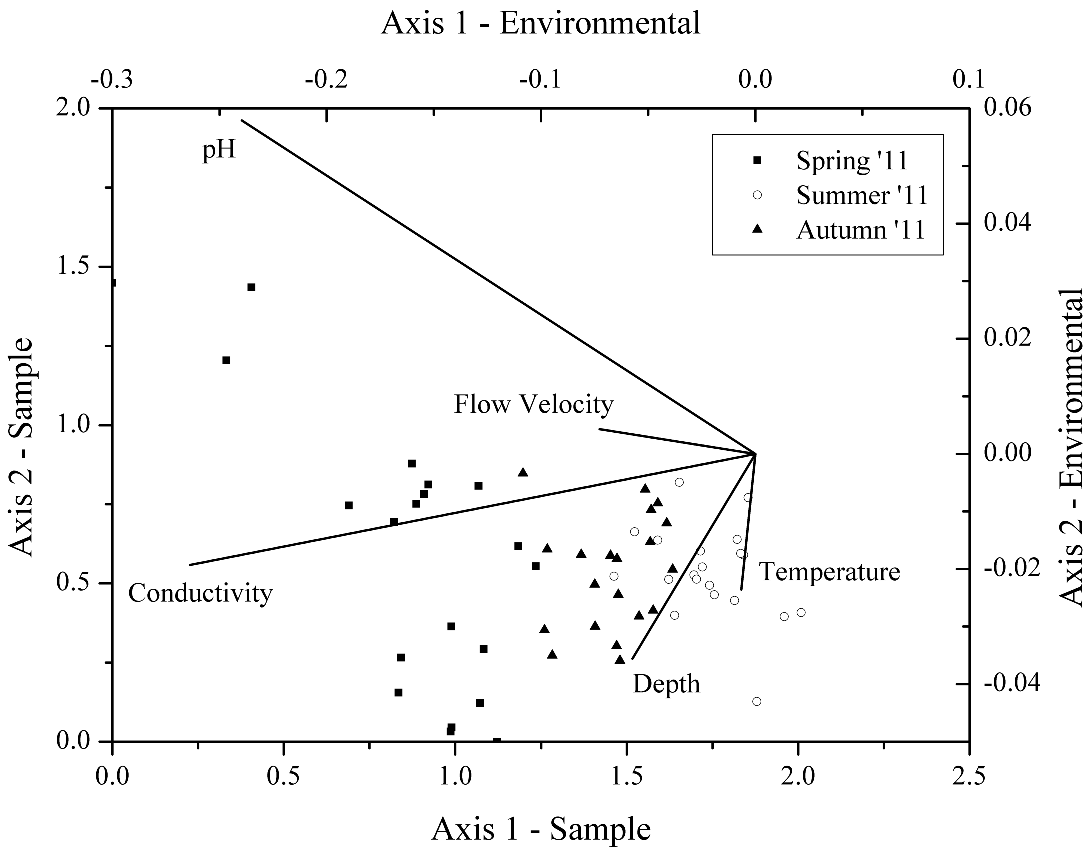
Axis 2 explained an additional 2.1% of the species data variance and 14.9% of the species-environment data. Despite overlap between the samples from each season a clear gradient reflecting seasonal variability is apparent (Figure 5.8 i).

Table 5.5 Results of DCCA **A**; Summary of the eigenvalues and percentage of variance of species data and species environment relationship explained on the first four canonical axes for Surber samples collected from the River Hull. **B**; The significance of the first canonical axis and environmental variables using forward selection procedure in Canoco and the Monte Carlo random permutations test (999 permutations).

A	**Axes**	**1**	**2**	**3**	**4**	**Total inertia**
	Eigenvalues	0.130	0.020	0.011	0.004	0.989
	Lengths of gradient	1.148	0.632	0.457	0.585	
	Cumulative percentage variance of species data (%)	13.1	15.2	15.6	15.9	
	Cumulative percentage variance of species-environment relation (%)	49.1	64.0	64.0	64.0	
	Sum of all eigenvalues					0.989
B		F ratio	P value			
	Significance of first canonical axis	8.162	< 0.01			
	Significance of all canonical axes	3.781	< 0.005			
	Significance of En. Variable					
	1) Conductivity	6.36	< 0.005			
	2) Temperature	6.87	< 0.005			
	3) pH	3.35	< 0.01			
	4) Flow velocity	0.93	Not Significant			
	5) Depth	0.40	Not Significant			

Common taxa recorded during all seasons plotted towards the centre of the species plot (Figure 5.8 ii), indicating that they were not significantly influenced by the variability of the environmental parameters included in the analysis. Some of the most widespread taxa, such as *Potamopyrgus jenkinsi* (Gastropoda), *Elmis aenea* (Coleoptera) and *Agapetus fuscipes* (Trichoptera), plotted centrally despite temporal variability in abundance. Some taxa's seasonal occurrence are apparent on Figure 5.8 ii, for example *Potamophylax cingulatus* (Trichoptera) was only recorded in spring and had a negative axis 1 score but a high axis 2 score. An additional example of this is the high abundance of *Ephemeralla ignita* recorded during the summer, which has a high axis 1 score.

i



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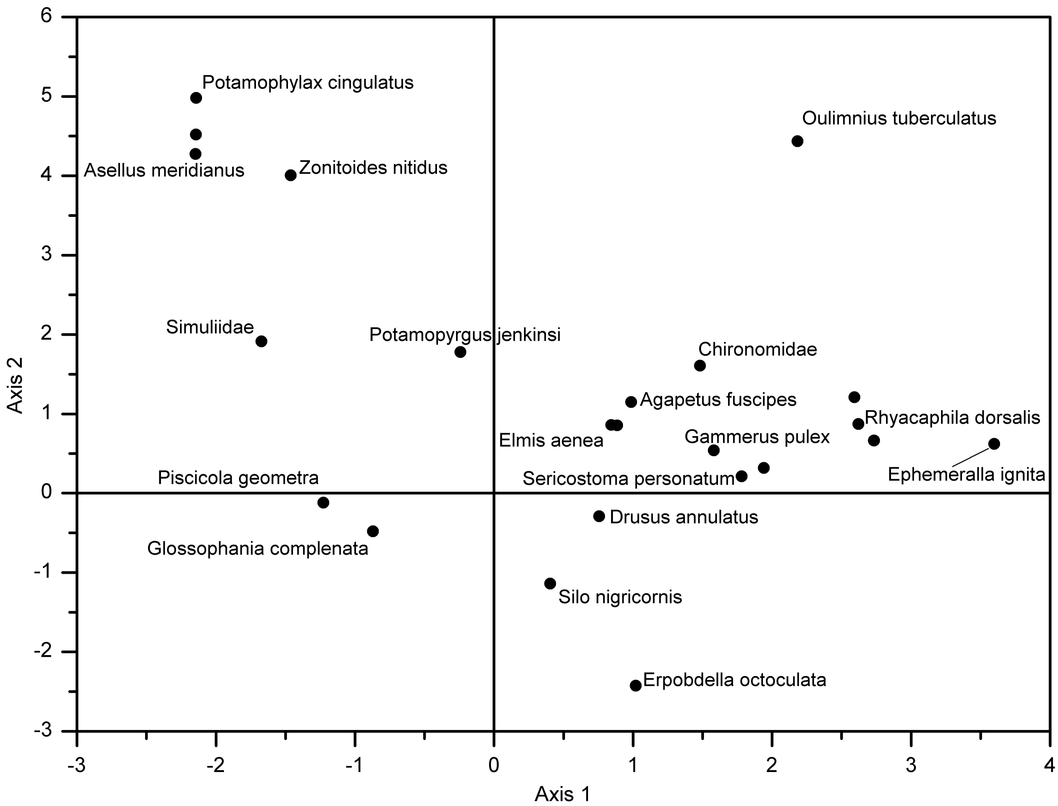


Figure 5.8 DCCA ordination of macroinvertebrate and abiotic data for the River Hull, with season as a covariable: i) Season-environment biplot; ii) Species-environment biplot.

5.5.3 Ecological Indices

Seasonal ecological indices (BMWP, ASPT, LIFE, PSI and CCI) were plotted using error bars in the SPSS software (Figure 5.9). One way Analysis of Variance (Anova) indicated that there was no significant difference between seasons for the indices examined, with the exception of ASPT scores (Table 5.6). The BMWP scores for spring and autumn are similar with the score rising slightly in summer (Figure 5.9 i). There is a significant increase in ASPT scores, with spring having the lowest score and autumn the highest (Figure 5.9 ii). The LIFE score did not show any significant variation over time. PSI scores did not vary significantly and the scores (81-100 PSI) indicate the river to be minimally sedimented/unsedimented (Extence *et al.*, 2011). When referring to the River Hull's restoration plan (Natural England, 2009) fine sediment is highlighted to be one of the main problems occurring within the river. These results suggest that this is not a problem on the river's headwaters but downstream as well. The CCI scores did not vary significantly among seasons. The scores (all within the 5-10 bracket) indicates that the river supports at least one species of restricted distribution and/or a community of moderate taxon richness, giving it a 'moderate conservation value' (Chadd and Extence, 2004). The highest scoring species found within the river was *Silo nigricornis*, which has a score of 5, defining it to be a 'local' species and *Zonitoides nitidus*, which possess a score of 4 and classified as being 'occasional' (Chadd and Extence, 2004).

Table 5.6 One way ANOVA between the ecological indices for the River Hull.

		df	Sum of Squares	F-ratio
BMWP	Between groups	2	3.167	0.083
ASPT	Between groups	2	1.330	10.658**
LIFE	Between groups	2	0.254	1.908
PSI	Between groups	2	94.911	1.156
CCI	Between groups	2	0.630	1.073

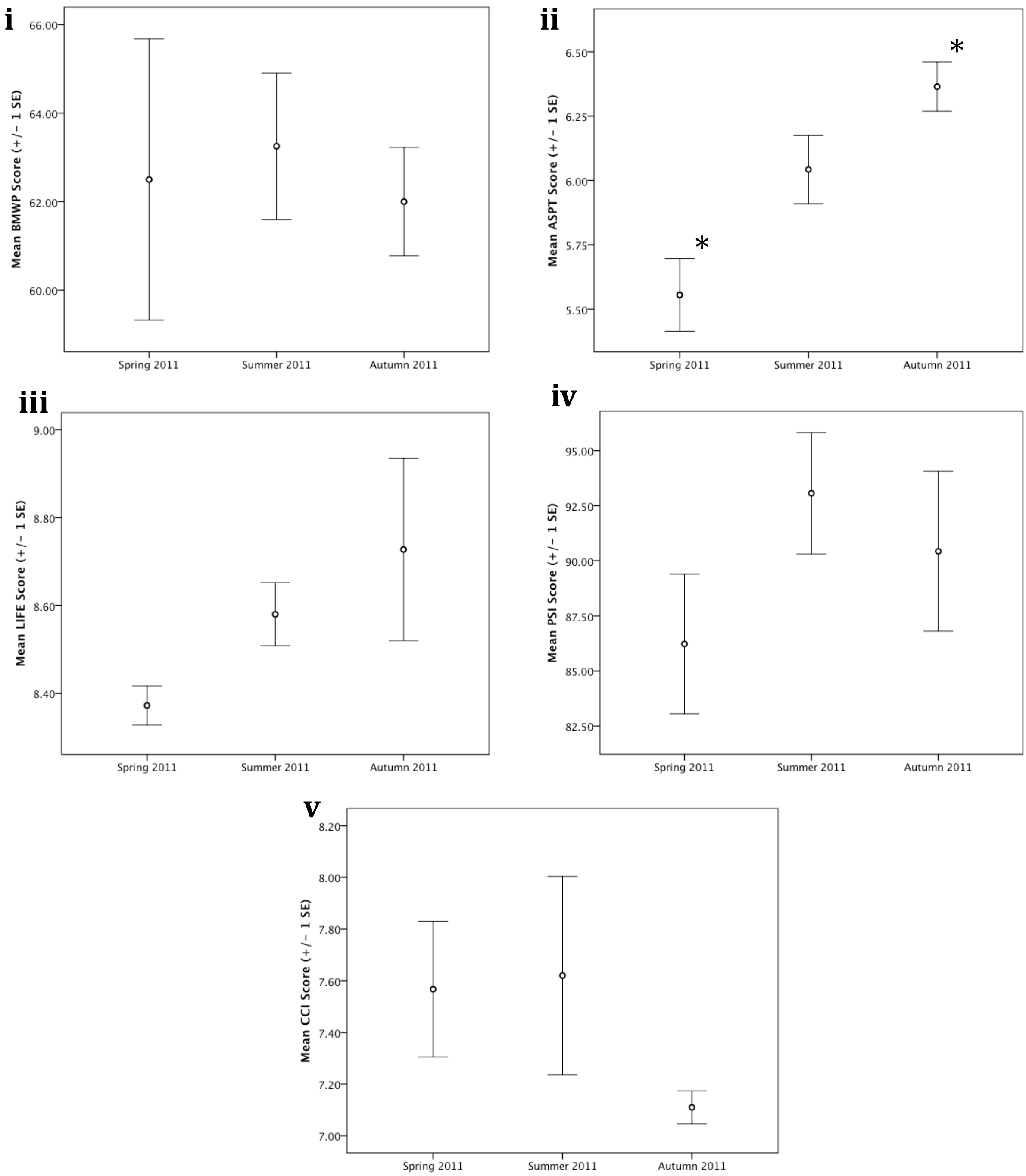


Figure 5.9 Error bar plots showing the mean i). BWMP score, ii) ASPT score, iii) LIFE score. iv) PSI score and v) CCI score (plus or minus one standard error) for each sampling season. * Indicates samples that are significantly different (Anova $p < 0.05$).

5.6 Comparison of Contemporary River Ecology and Palaeoecology: River Hull

In order to facilitate a direct comparison between the contemporary and palaeoecological communities, only the results comprising of Gastropoda, Coleoptera and

Trichoptera from the contemporary species list were used in association with palaeoecological data. This allowed for a quantitative comparison of the types of community recorded due to the proxies being well represented in the sub-fossil record. In addition the data collected by Whitehead (1935) and Pearson and Jones (1984) (located approximatly 500m downstream from the contemporary sampling sites) were used in the analysis as taxa counts were available in these published papers.

5.6.1 Detrended Correspondence Analysis (DCA)

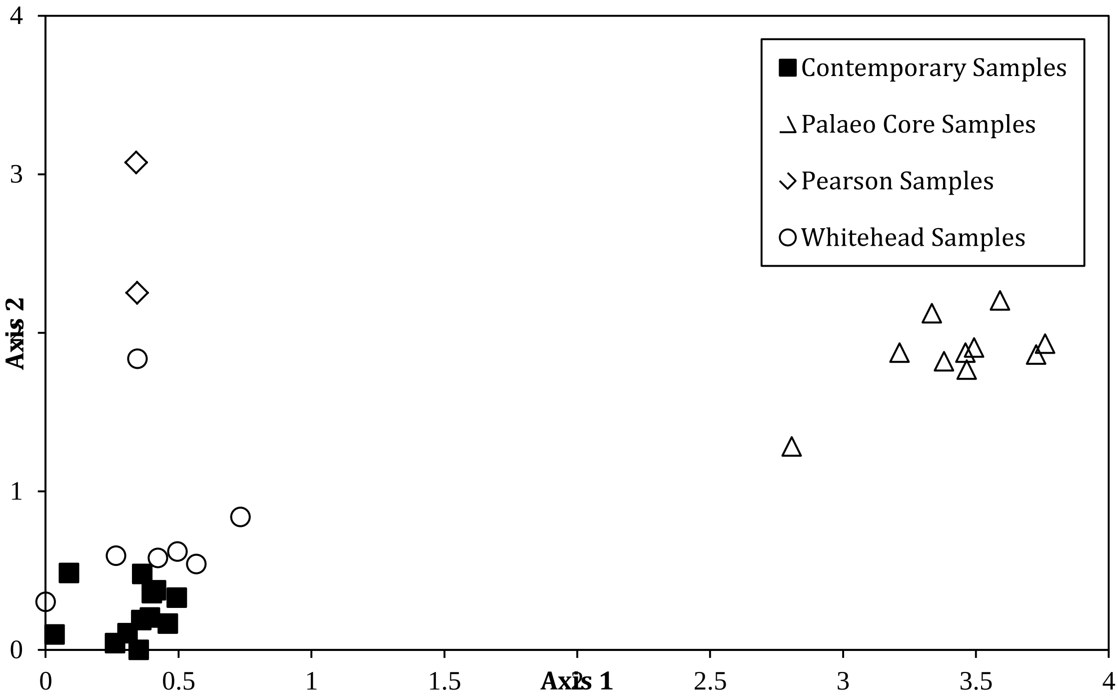
DCA was used in order to investigate the variability between the contemporary and palaeoecological macroinvertebrate community composition. This allowed for the examination of any temporal patterns and the fauna associated with the four sampling time periods. The first four axes accounted for a total of 56.2% of the variance in the faunal community data. Axis 1 explains 25.9% of the variance within the species data and axis 2 explains a further 18% (Table 5.7).

Table 5.7 Summary of the eigenvalues and cumulative variance of contemporary and palaeoecological data for DCA of kick and palaeo-core samples from the River Hull.

Axes	1	2	3	4	Total inertia
Eigenvalues	0.784	0.544	0.240	0.133	3.026
Lengths of gradient	3.761	3.074	2.721	2.332	
Cumulative percentage variance of species data (%)	25.9	43.9	51.8	56.2	
Sum of all eigenvalues					3.026

The differences between each sampling time period were more obvious when the sample scores were plotted (Figure 5.10 i) and there is a distinct difference between the contemporary and palaeoecological samples. Both form separate clusters at each end of axis 1, thus indicating differences in community composition for each sampling period. The contemporary samples and those of Whitehead (1935) samples, cluster closely together at the lower end of each axis. The samples from Pearson and Jones (1984) have similar on axis 1 scores but higher axis 2 scores.

i



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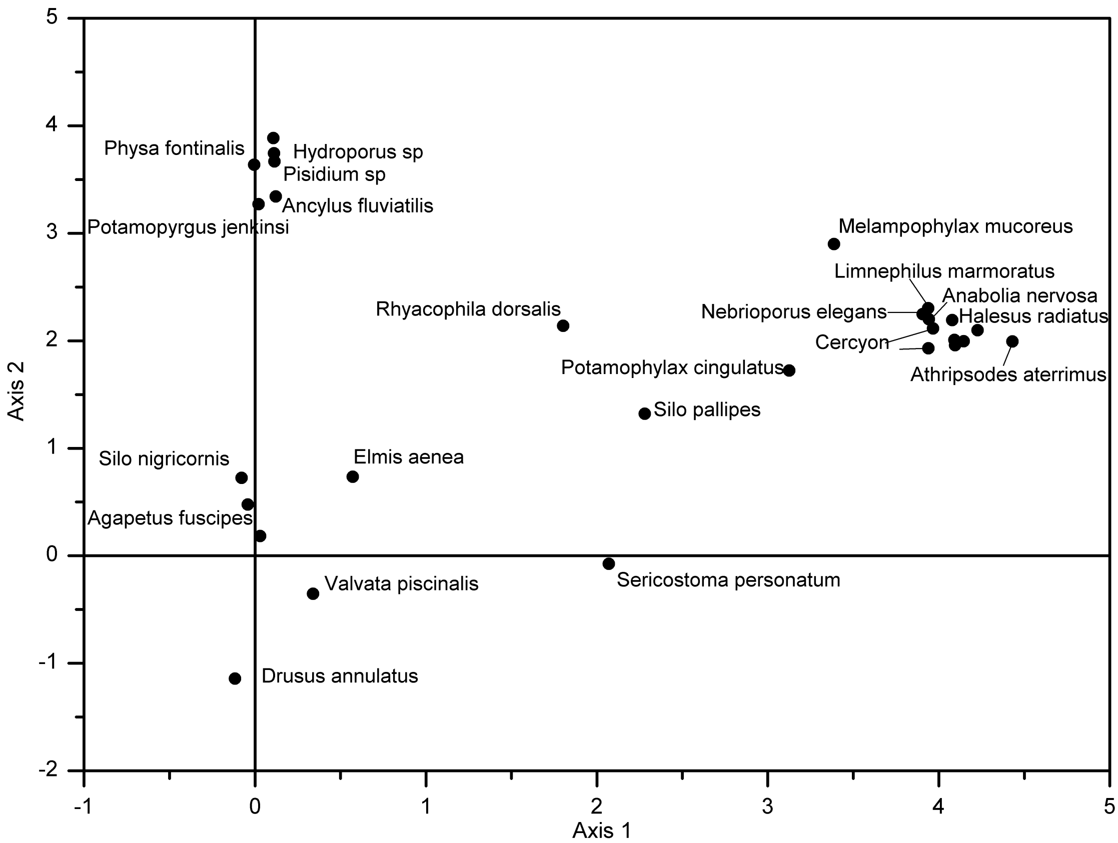


Figure 5.10 DCA faunal plot of presence/absence data from the River Hull: i) Different data sets used ii) Faunal biplot.

The taxa are displayed in the species plot (Figure 5.10 ii) and indicate the more frequent occurance of some taxa within some time periods. For example the caddisfly larvae, *Drusus anulatus*, which has a negative species biplot score on both axis 1 and 2, was only recorded in the contemporary samples. *Limnephilus marmoratus, Anabolia nervosa* and *Halesus radiatus* (Trichoptera), all have high axis 1 scores and were only recorded in the palaeoecological samples; all are typically associated with slower flow velocities. Other patterns were also recorded for other historic samples, for example *Ancylus fluviatilis* (limpet) and *Pisidium sp* were taxa only recorded in the Pearson and Jones (1984) samples from 1972 and had low axis 1 scores and a high axis 2 scores. *Valvata piscinalis*, was only recorded in the Whitehead (1935) samples from 1930 and have low axis 1 scores and a negative axis 2 scores.

To compare each sampling period in an unbiased manner only those taxa common to more than one sampling period were used in subsequent analysis. In addition to this species abundance was replaced with records of presence/absence to remove the influence of relative abundance on the results. This is due to abundance distorting the output as a higher abundance means higher loadings on results (Table 5.8). Some species were merged to higher taxa due to inability to resolve some sub-fossil material to species level during identification.

Table 5.8 List of taxa that were removed as they occurred in only one data set or were combined from the merged list of contemporary and palaeo species due to differences in the taxonomic resolution.

Species Removed	Species Combined
Valvata piscinalis	*Drusus annulatus* and *Melampophylax mucoreus.*
Ancylus fluviatilis	
Pisidium sp.	*Silo pallipes* and *Silo nigricornis.*
Nebrioporus elegans	
Haliplus sp.	*Sericostomatidae* and *Sericostoma personatum.*
Hydroporus sp.	
Helophorus	
Cercyon	
Agapetus fuscipes	
Hydropsyche pellucidula	
Hydropsyche angustipennis	
Limnophilidae	

DCA was performed with the common presence/absence species list and the summary of the eigenvalues are presented in Table 5.9. This DCA explained 50.9% of the variance across the first four axes. Axis 1 accounted for 28.8% of the variance within the faunal data, with axis 2 explaining an additional 13.7%. This indicates a slight reduction of the variance explained when compared to the original DCA (Table 5.7)

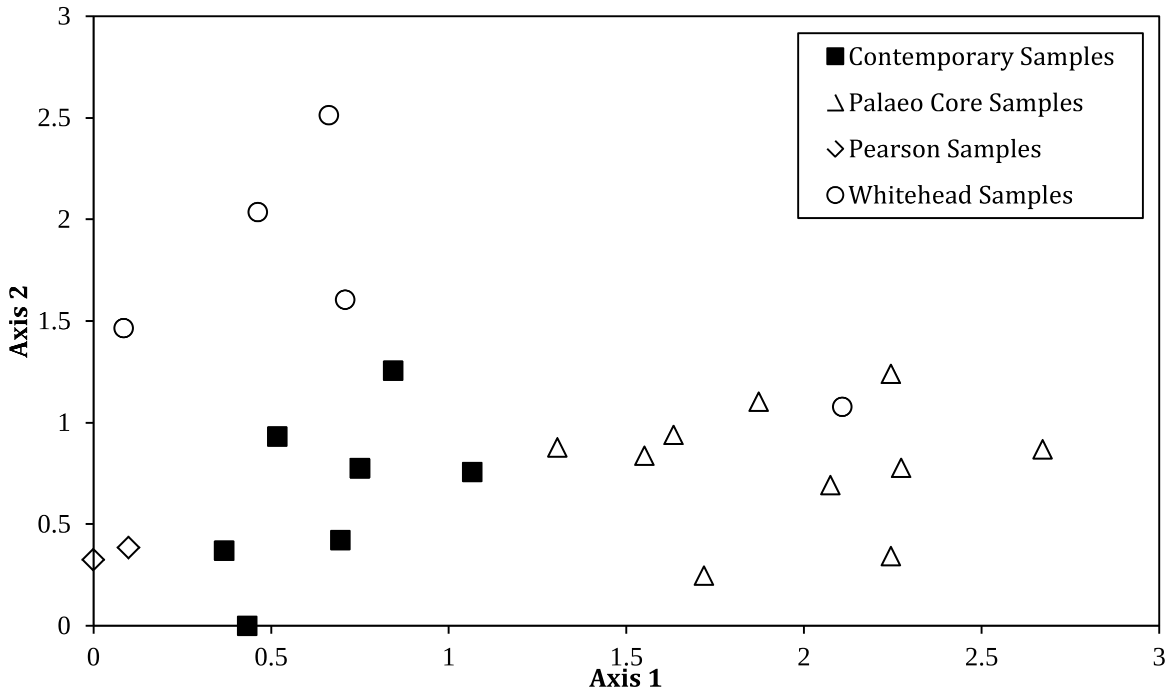
Table 5.9 Summary of the eigenvalues and cumulative variance of the presence/absence contemporary and palaeoecological data for the DCA of kick and palaeo-core samples from the River Hull.

Axes	1	2	3	4	Total inertia
Eigenvalues	0.476	0.227	0.094	0.045	1.654
Lengths of gradient	2.673	2.513	1.757	1.231	
Cumulative percentage variance of species data (%)	28.8	42.5	48.2	50.9	
Sum of all eigenvalues					1.654

The presence/absence DCA (Figure 5.11 i) indicates a similar pattern to that of the whole Gastropoda, Coleoptera and Trichoptera community (Figure 5.10 i). However the difference between data sets on axis 1 has been reduced, becoming less dispersed. Axis 2 also shows a reduction in the separation of datasets in the ordination space. The grouping of the contemporary and palaeoecological samples remains distinct indicating that community composition of these core taxa has changed.

The species highlighted on Figure 5.11 ii are present within all data sets and the differences in community composition reflect the differences in sampling periods (Appendix 3). Taxa such as *Elmis aenea* (Coleoptera) and *Silo pallipes* (Trichoptera), have a low scores on both axes and occurred more frequently within the contemporary samples. In contrast, *Athripsodes aterrimus* and *Halesus radiatus* (Trichoptera) have high axis 1 scores and were more common within the Pearson and Jones (1984), Whitehead (1935) and palaeoecological samples from the cores collected.

i



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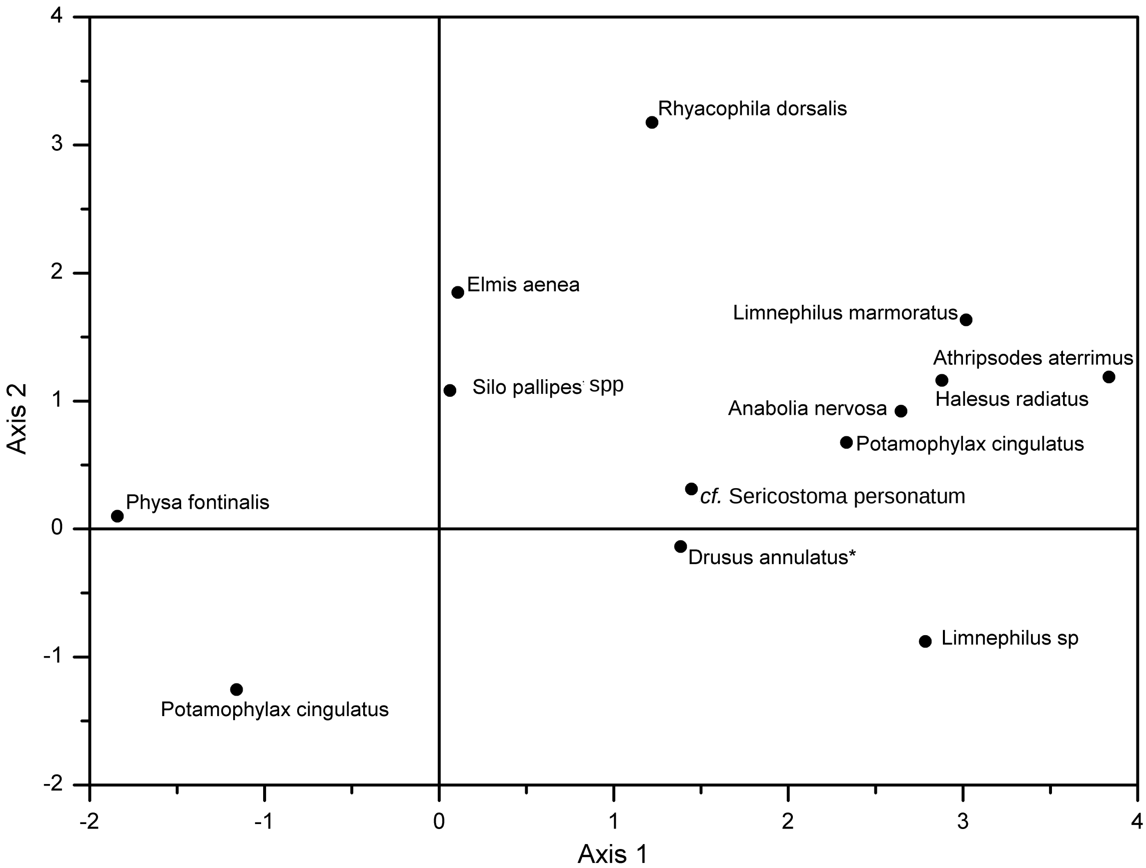

Figure 5.11 Contemporary and palaeoecological species Presence/Absence DCA ordination biplots: i) temporal variability; ii) Macroinvertebrate community data (Species marked with a * indicates taxa combined due to difference in identification resolution).

5.6.2 Ecological Indices

When the contemporary and historic cores, (Whitehead, 1935 & Pearson and Jones, 1984) macroinvertebrate indices were compared no significant differences were recorded for the BMWP scores (Figure 5.12 i). There were a number of differences among the data sets for the ASPT score (Figure 5.12 ii and Table 5.10). The majority of these were between the three contemporary season and all other biotic samples.

The Lotic Index for Flow Evaluation (LIFE Index) indicated that contemporary samples supported an assemblage associated with faster flow velocities, as the majority of species were from LIFE flow groups 1 or 2 (indicating fast or rapid flows) (Extence *et al.* 1999), compared to historic samples. There was a significant difference between all contemporary and historic samples (Figure 5.12 iii and Table 5.10). This can be explained through the examination of the rivers history and sediment cores. Before the headwaters of the river were dredged and channelized, the majority of the area consisted of marshland (Environment Agency, 2003), thus accounting for the fine grained sediments and low flow velocities. When examining the sub-fossil taxa found within the samples the community contains taxa that belong to a range of LIFE groups (1 to 4) thus characterizing a morphologically dynamic environment of slow flowing and standing waters to fast flowing waters (Extence *et al.* 1999). However LIFE flow group 4 contained the highest abundances of species indicating a dominance of slow flowing waters.

The PSI results (Figure 5.12 iv) show significant difference ($P < 0.01$) between the contemporary samples compared to the Pearson and Jones (1984) and palaeoecological samples. Results obtained from the Whitehead (1935) were not significantly different to the contemporary results. Both the contemporary and the Whitehead (1935) results were between 81-100 PSI indicating the river was minimally sedimented/unsedimented (Extence *et al.*, 2011). The palaeoecological samples were in the range of 20-40 PSI, and the Pearson and Jones (1984) samples were between 41- 60 PSI. This indicates the river may have been naturally siltier and therefore historical PSI scores were lower. Results from Figure 5.12 indicates similar patterns for both the whole contemporary community and the combination of the three proxies (Gastropoda, Coleoptera and Trichoptera). This clearly demonstrates that the use of the three chosen proxies does not strongly influence the nature of the score, although the scores have been reduced.

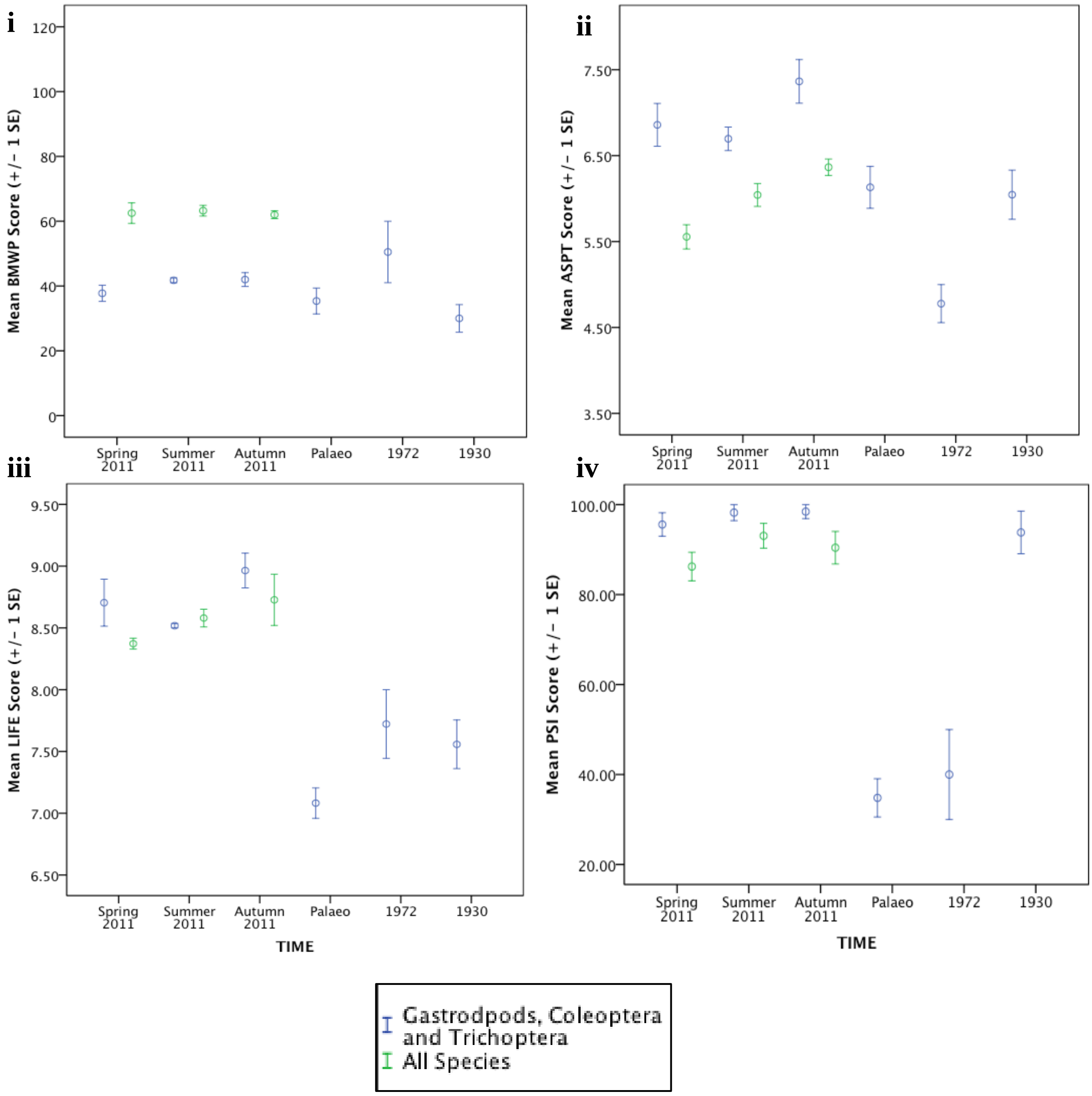

Figure 5.12 Error bar plot showing the comparisons between the i) mean BMWP score, ii) mean ASPT score, iii) LIFE score and iv) PSI score (plus or minus one standard error) for the contemporary and palaeoecological samples taken from the River Hull, and the full contemporary species list for each sampling season.

Table 5.10 Kruskal-Wallis pair-wise comparison between seasons for macroinvertebrate community indices along the River Hull. *P < 0.05, ** P < 0.01, *** P < 0.001.

		Spring	Summer	Autumn	Palaeo	1972
BMWP	Summer	NS	-	-	-	-
	Autumn	NS	NS	-	-	-
	Palaeo	NS	NS	NS	-	-
	1972	NS	NS	NS	NS	-
	1930	NS	NS	NS	NS	NS
ASPT	Summer	NS	-	-	-	-
	Autumn	NS	*	-	-	-
	Palaeo	*	**	**	-	-
	1972	**	*	**	NS	-
	1930	*	**	*	NS	NS
LIFE	Summer	NS	-	-	-	-
	Autumn	NS	*	-	-	-
	Palaeo	***	**	**	-	-
	1972	***	**	**	*	-
	1930	***	***	**	*	NS
PSI	Summer	NS	-	-	-	-
	Autumn	NS	NS	-	-	-
	Palaeo	**	**	**	-	-
	1972	**	**	*	NS	-
	1930	NS	***	***	**	*

5.7 Ecological interpretation

Through comparison between the different sets of samples and particularly the three proxies used in all data sets (Gastropoda, Coleoptera and Trichoptera) a number of patterns can be observed. Only three species are common to every sample, these are: *Elmis aenea* (Coleoptera), *Rhyacophila dorsalis* and *Sericostoma personatum* (Trichoptera). *Elmis aenea* is associated with unpolluted water and sandy river beds (Nilsson and Holmen, 1995), *Rhyacophila dorsalis* shows preferences towards medium to fast paced rivers with stony substratum (Edington and Hildrew, 1995) and *Sericostoma personatum* is a burrowing caddisfly, associated with silts and sand (Wallace et al., 2003). The palaeoecological samples contain 19 species not recorded in the contemporary samples. Species found solely within the palaeoecological samples are; Coleoptera; *Nebrioporus depressus elegans*, *Helophorus*, *Cercyon,* and Trichoptera; *Hydropsyche pellucidula, Hydropsyche angustipennis, Limnephilus marmoratus, Anabolia nervosa, Potamophylax cingulatus, Halesus radiatus* and *Athripsodes aterrimus*. This large increase in the number of taxa in the palaeoecological sample was unexpected and by examining the abundance of each species found and their habitat preferences, a picture can be built of what the riverine habitats were probably like historically. Apart from *Hydropsyche pellucidula,* which is a taxa typical of fast flowing water, the majority of the species were indicative of lentic to slow flowing waters. *Limnephilus marmoratus* and *Anabolia nervosa* (the most abundant taxa recorded) are found in still and slow flowing waters, often among vegetation, with *Athripsodes aterrimus* showing a preference towards stagnant habitats that have slow to moderately flowing waters and are also usually found among plants and on muddy sand (Wallace et al., 2003). *Potamophylax cingulatus* and *Halesus radiatus,* part of the family Limnephilidae, are both commonly found in streams and small to medium sized rivers on stony substratum (Wallace et al., 2003). This provides an insight into a heterogeneous environment that was slow to fast flowing with bed material made up of patches of silty sand and stony substratum. The palaeoecology results indicate that the river at this headwater location contained much more fine sediment historically than today with a high abundance of species showing preference for silt and sand.

The macroinvertebrate survey of the Driffield Trout Stream, performed by Whitehead (1935) shows a number of similarities when compared to the contemporary data. Whitehead (1935) found that the streambed consisted of chalk pieces and flint flakes combined with varying amounts of silt and that the aquatic vegetation (*Ranunculus*) grew in patches of varying sizes. The abiotic measurements taken throughout the year showed flow rate on the surface to be on average 1.5 m s^{-1} which is higher than that recorded in the contemporary river, however pH varied from 7.6 in the winter months to 8.4 in the summer months with temperature ranging from $7^{o}C$ to $15^{o}C$ which is similar to that of the contemporary abiotic measurements. Whitehead (1935) noted that the water was very clear and displayed a bluish tinge and assumed that this was due to the fact the stream was running freely, clear from pollution and had abundant vegetation, and that the oxygen content was not a limiting factor. Whitehead (1935) reported that *Agapetus fuscipes* was one of the most common taxa recorded during late summer and autumn with *Gammarus pulex* and various species of Oligochaeta found in large numbers throughout the year. When compared to the contemporary, palaeoecological and Pearson and Jones (1978) results only one taxa was unique to Whitehead's sample; *Valvata piscinalis*. This species of Gastropod was found in large quantities during spring, predominantly in silt and mud streambeds. *Valvata piscinalis* has been associated with oligotrophic environments, however they prefer clear water habitats with high siltation rates (Grigorovich et al., 2005). When compared to the more recent macroinvertebrate samples collected for this thesis, *Physa fontinalis* (Gastropoda) and *Silo pallipes* (Trichoptera) were only found within the Whitehead (1935) samples. *Physa fontinalis* displays a preference towards slow flowing waters and is found on aquatic plants or muddy substrates (Dillon, 2000) and *Silo pallipes* are widespread and common, however show preferences for, and are ecologically adapted towards, running/faster flowing waters with hard stony substrates (Viðinskienë, 2005).

Pearson and Jones (1984) described the headwaters of the River Hull to be of low gradient, high conductivity and pH, moderate to swift velocities and contain relatively mobile substratum made up largely of gravel, with some patches of sand and silt. Abiotic measurements taken by Pearson and Jones (1984) are similar to the results obtained from the contemporary survey (Table 5.2); water depth (0.2 – 0.5m), flow velocity (0.25 – 0.5 m s^{-1}). Seasonal samples (one Kick and five Surber samples) were taken from the upper, middle and lower zones of the River Hull and most species found had either restricted distributions and/or peaks of abundance in particular zones. Species common to just the upper zone of the river included: *Ancylus fluviatilis* (Gastropoda), *Limnius volckmari* (Coleoptera), *Silo nigricornis*, *Melampophylax mucoreus* and *Rhyacophila dorsalis* (Trichoptera). *Ancylus fluviatilis* inhabits quick flowing water, adhering firmly to stones and requires clean water free from suspended matter and avoids muddy substrates (Kerney, 1999). Species such as *Gammarus pulex (Shrimp)* and *Baetis rhodani* (Mayfly) were widely distributed and found in all zones but most abundant in the upper zone. Pearson and Jones (1984) described the upper zone to support a community which was characteristic of chalk streams. Comparison of Pearson and Jones (1984) results to the entire contemporary community resulted in an additional two species present in Pearson and Jones (1984) samples; *Sialis lutaria,* and *Melampophylax mucoreus.* The presence of *Sialis lutaria* (Alder fly larvae) suggests the presence of slow flows and silt deposits (Derbyshire Wildlife Trust, 2013) and *Melampophylax mucoreus* (Trichoptera) is found in streams and rivers with stony substratum and commonest in alkaline waters (Wallace et al., 2003).

Pearson and Jones (1978) reported that parts of the headwaters of the River Hull suffered from prolific growths of instream vegetation that impeded adequate drainage of lowland areas. They undertook an investigation into the effects of weed-cutting with regards to macroinvertebrate disturbance and their recolonisation. Results showed that the re-establishment of the invertebrate populations after weed cutting was rapid, demonstrating the capability of the rivers fauna to recover rapidly from disturbances. There were little changes to community compositon and this displayed similar results to the impacts that dredging had also caused along the river, however recovery times increased due to the disturbance being of greater magnitude (Pearson and Jones, 1975). Pearson and Jones (1975, 1978) also reported that the majority of the taxa found within the river, especially *Gammarus pulex* which was the dominant species in all samples, were active up and downstream migrants following disturbances (weed cutting or dredging).

5.8 Summary

This chapter has focused on the historical channel movement and environmental changes experienced within the River Hull headwater SSSI study reach. Key findings include:

- Through the use of DCCA and DCA, clear seasonal, environmental and temporal gradients between the contemporary and palaeoecological results have been highlighted.
- Based on WFD metrics, results have demonstrated that the river experienced higher sedimentation rates and slower flow velocities compared to the contemporary river. This is especially evident when looking at the results from the Pearson and Jones (1984) surveys from 1972.
- The data suggest an impoverishment of biotopes caused by channelization, restricting the biological community to rheophilic components and causing a loss of more limnophilic components compared to historical conditions. This has been caused by habitat simplification and also an increase in stream gradient generated by loss of river length.

CHAPTER SIX: THE RIVER WENSUM

6.1 Introduction

This chapter examines the changes experienced on the River Wensum since part of its course was straightened in 1946. The current status of the river, the geology of the catchment and history of the site, including reasons for its SSSI and SAC designation are outlined. The historic movement of the river channel study reach will be examined via the use of historical maps. Contemporary and palaeoecological sampling was undertaken to explore and compare the two macroinvertebrate assemblages and to gain an understanding of how the instream ecology has changed over time. Due to restoration measures being implemented on the study reach during the study period, through the reconnection of the palaeomeander, the recovery and recolonisation of the restored channel is also assessed. These results will be used to define a ‘reference condition’ and provide an insight into the rivers historic biotope composition, water quality, flow characteristics and fine sediment regime. The objectives of this chapter are to (see Chapter 1.2 for relevant objective):

- Explore the contemporary riverine environment of the study reach and to evaluate its characteristics with a range of macroinvertebrate community indices;
- Identify the River Wensum’s historic palaeochannel and characterise the palaeoecological community recorded from the sediment sequence (sediment pit) to define a historic reference community.
- Compare the contemporary and palaeoecological community structure and composition to examine and describe the changes that have taken place.

6.2 The River Wensum

The River Wensum rises near South Raynham at Pear Tree Corner in Norfolk and flows easterly towards Norwich (Figure 6.1). It incorporates the lower part of the River Tat and the downstream reaches of the Langor Drain and Guist Drain. The 71km stretch of river upstream of Norwich is one of 31 rivers in England to be designated a ‘whole river Site of Special Scientific Interest’ and is currently one of a series of Demonstration Test Catchments (Figure 6.2).

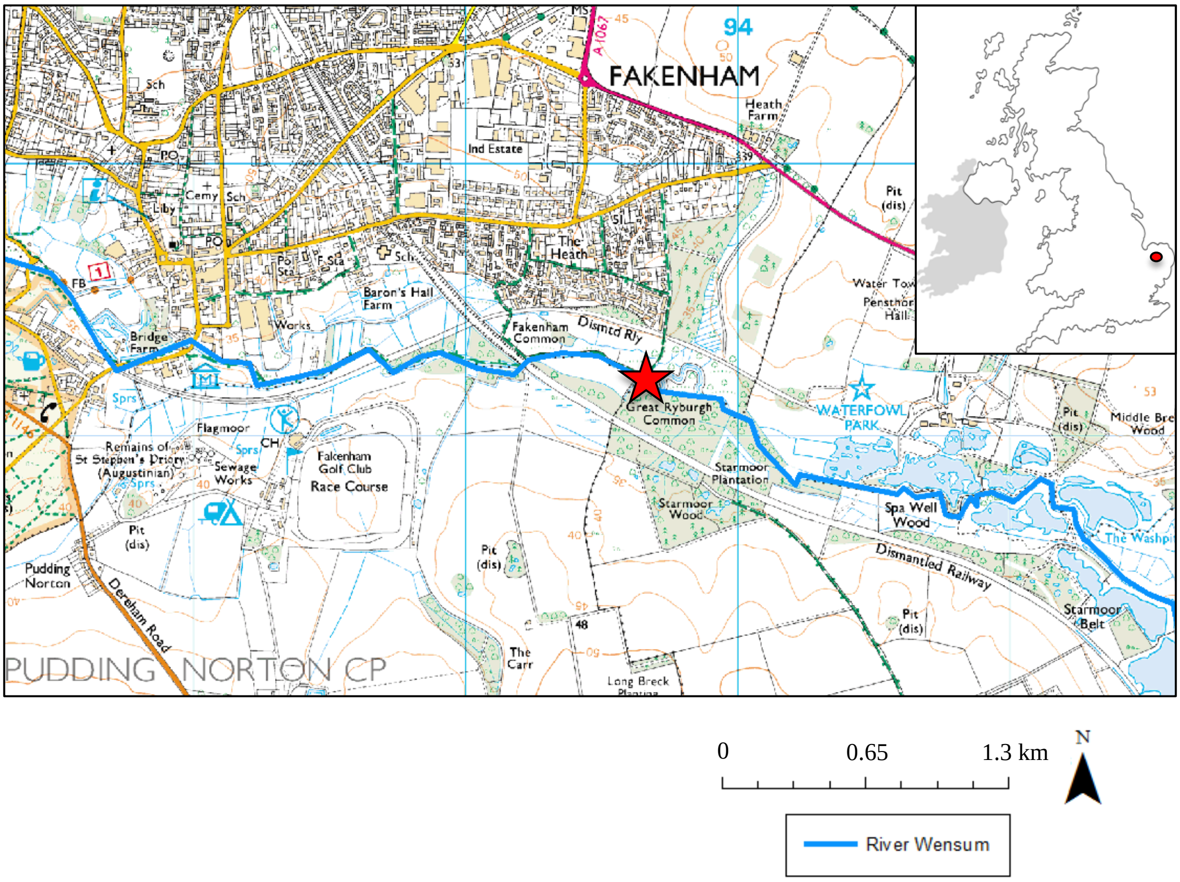

Figure 6.1 Map showing the entire length of the River Wensum SSSI, in relation to the rest of the United Kingdom/UK/Britain. Study reach highlighted by red star (EDINA Digimap, 2013).

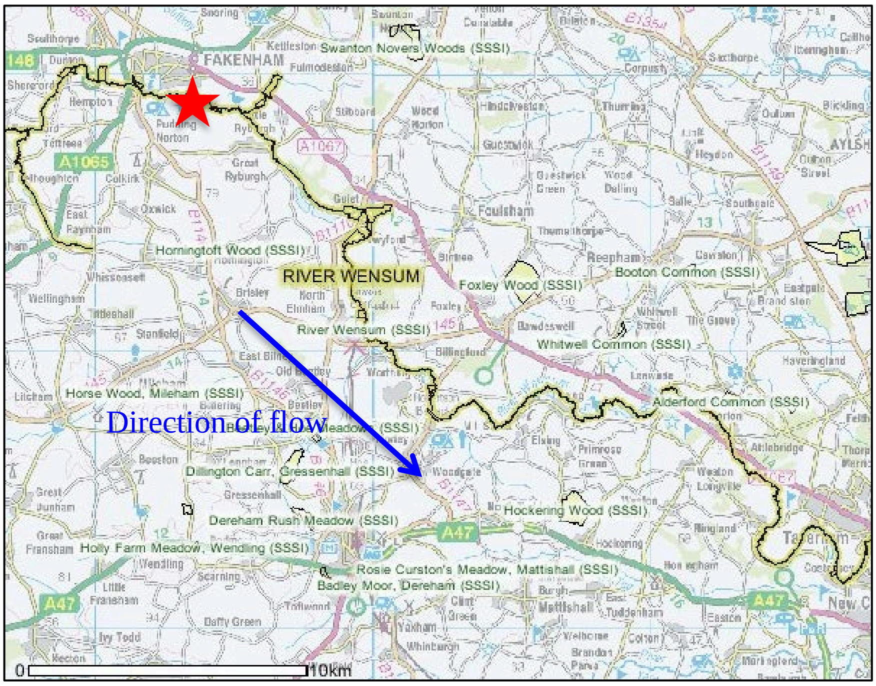

Figure 6.2 Map showing the full extent of the River Wensum SSSI (Natural England, 2012a). Study reach highlighted by red star.

The Demonstration Test Catchment (DTC) program was launched in England to provide underpinning research to help inform both policy and practical approaches for reducing diffuse pollution and improving the ecological status of freshwater (Collins et al., 2013). The River Wensum is an example of a lowland, calcareous, chalk river and each of its eleven sub-catchments is part of the DTC program and it is also one of 16 rivers in England to be notified as a European Special Area of Conservation (SAC) under the EC 'Habitats and Species' Directive. It was designated as an eastern example of the Annex I river habitat watercourses of plain to submontane levels with *Ranunculion fluitantis* and Batrachion vegetation. The River Wensum has a predominantly rural catchment (650 km^2), with intensive arable farmland dominating the landscape and grazing marsh, fen, scrub and scattered woodlands characterizing the floodplains (Hiscock et al., 2001). Unusually for a lowland river, much of the floodplain of the River Wensum is still traditionally managed, although there are a series of flooded gravel pits in the vicinity of Costessey, Lenwade, Lyng, Fakenham and Great Ryburgh. The hydrological regime of the Wensum is dominated by groundwater, however water management and artificial drainage significantly affects water levels and flow in the catchment. The river flows over a catchment underlain by Senonian Chalk, that has been overlain by a complex sequence of glacial drift, sands and gravels (Figure 6.3), this often separates the river from the chalk aquifer by considerable depths of superficial material (JBA, 2007).

The condition assessment undertaken by English Nature (now Natural England) in 2002 concluded that the river was in an unfavourable condition. The current river channel is considered to be the product of a long history of modification and management thus limiting its ecological potential to support the classic chalk river habitat (English Nature, 2002a) (Figure 6.4). The principle reasons for the unfavourable condition are due to poor water quality, high phosphate levels, siltation and physical modification of the channel impeding natural hydrological and geomorphological functioning (see Figure 6.4 and Table 6.1 for details). These modifications have included extensive dredging which has over-deepened the channel, straightening of the channel and the presence of instream structures (e.g. mills and weirs). This has resulted in sluggish flows and the accumulation of fine sediment deposits within the channel (English Nature, 2002b).

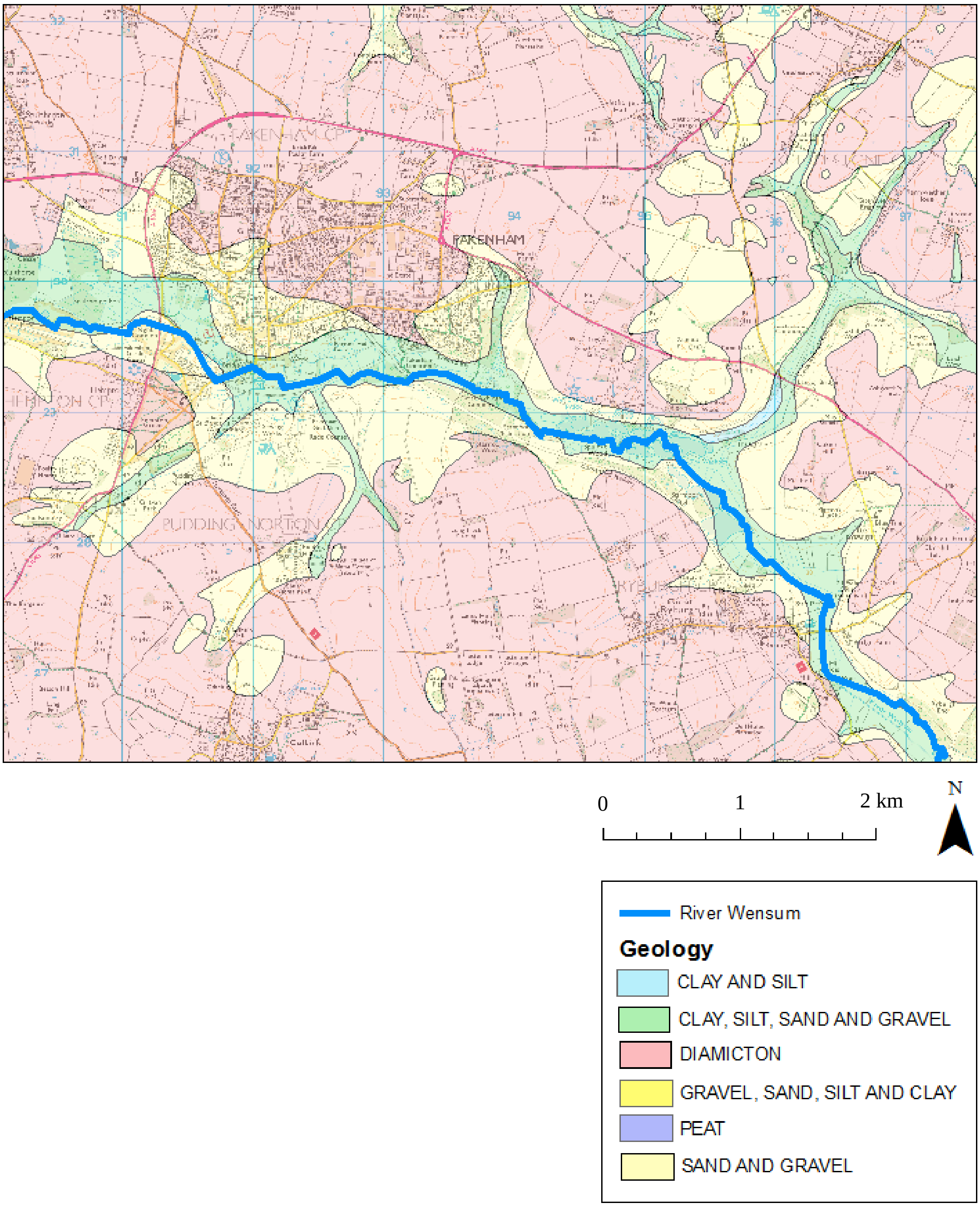

Figure 6.3 Map showing the surface geology of the River Wensum SSSI (EDINA Geological Digimap, 2012).

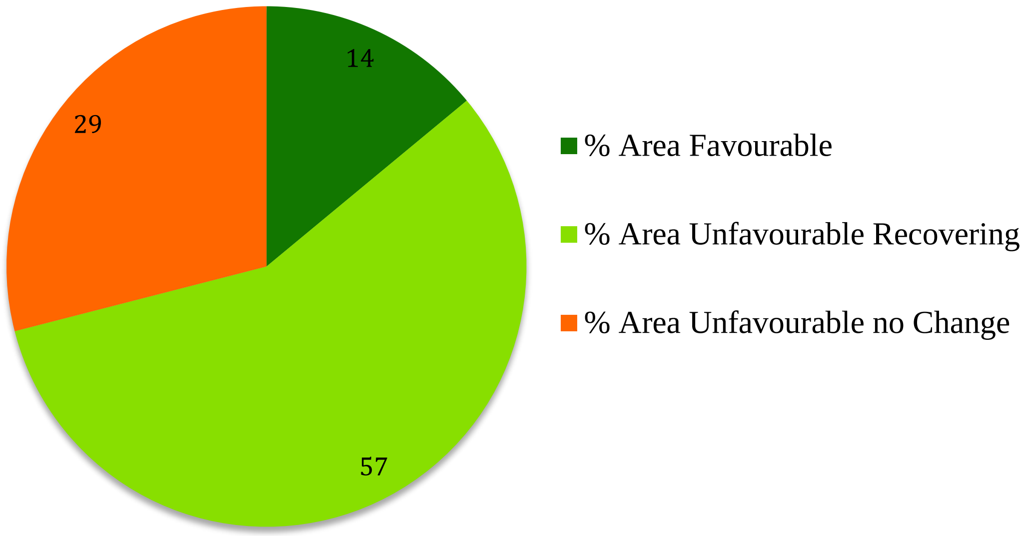

Figure 6.4 The condition summary for the River Wensum complied by Natural England on 1st May 2012 (Natural England, 2012b).

Table 6.1 Summary of key issues affecting the River Wensum (adapted from JBA, 2007).

Key Pressures	Impacts
The over-wide and over-deep channelized channel	Channelization has caused the channel to become straightened and embanked along much of its length. There is a lack of diverse velocities, flow structure, habitat and loss of gravel bed material.
Historical land drainage	Historical modifications have altered the hydrological regime and increased silt ingress, potentially reducing habitat heterogeneity.
Invasion of non native species	Increasingly problematic downstream and there is noticeable lack of self-sustaining fish populations.
In-channel structures	In-channel weirs and mill structures are disrupting the rivers continuity, are acting as barriers to fish passage and causing fine sediment accumulation especially upstream of mills.

6.3 The Historical River Wensum

Since clearance of the floodplain forests for settlement and agriculture approximately 4,500 years ago, sections of the post-glacial meandering channel have been straightened, dredged, diverted, impounded and embanked (JBA, 2007). Since around 1200 AD the waters of the River Wensum have been harnessed to provide power for water mills and although the last mill ceased operating at the end of the 1960s, there are 14 mill structures still present along the course of the river (JBA, 2007). These mill structures modify the morphology of the channel and have had significant impounding effects. The mill sluices and their millponds generate a stepped bed and water surface profile that has caused up to 70% of the river to be ponded behind these structures, resulting in excessive fine sediment acumulation (English Nature, 2002b). In order to examine changes to the river channel, contemporary and historic maps were studied. This allowed changes to the channel course to be identified. The channel at Great Ryburgh (the main study site in this chapter) was anthropogenically straightened in 1946 (Morrissey, *pers comm*. 2011) (Figure 6.6).

6.4 Study Sites

Preliminary site visits identified 3 contemporary sampling sites (Figure 6.5). During the sampling period, Natural England undertook a restoration project, which involved the reconnection of the historic meander (palaeochannel). Plate 6.1 i and ii show the distinct depression of the palaeochannel before its reinstatment. The reinstated channel follows the original 1946 course and Plate 6.1 iii shows the sediment pit where the palaeoecological samples were collected. The channel reinstatement did not occur until late November 2010, therefore sites 4 and 5 were not sampled during the initial survey. However the restoration work provided an opportunity to monitor recolonisation and recovery rates following the reinstatement of the meander. Plate 6.2 i and ii show Sites 1 to 3. This section of the river was deep and contained large volumes of instream macrophytes and bank side vegetation. Plate 6.2 iii and iv show Sites 4 and 5 respectively following the reinstatement of the channel and are of one of the pools and riffles created.

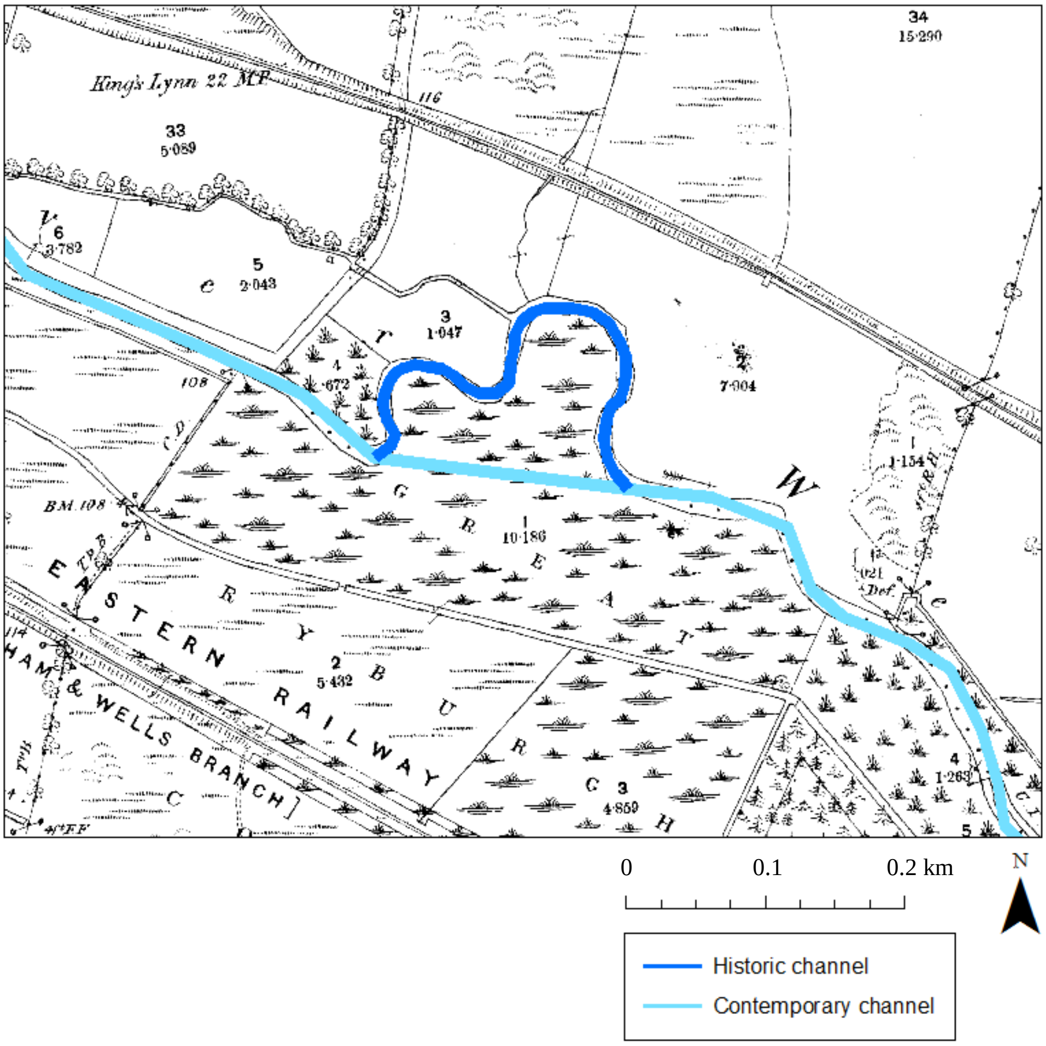

Figure 6.5 Country Series (1:10560) OS map, dated 1849-1899, of the River Wensum indicating the historic and contemporary channel at Great Ryburgh (EDINA Digimap).

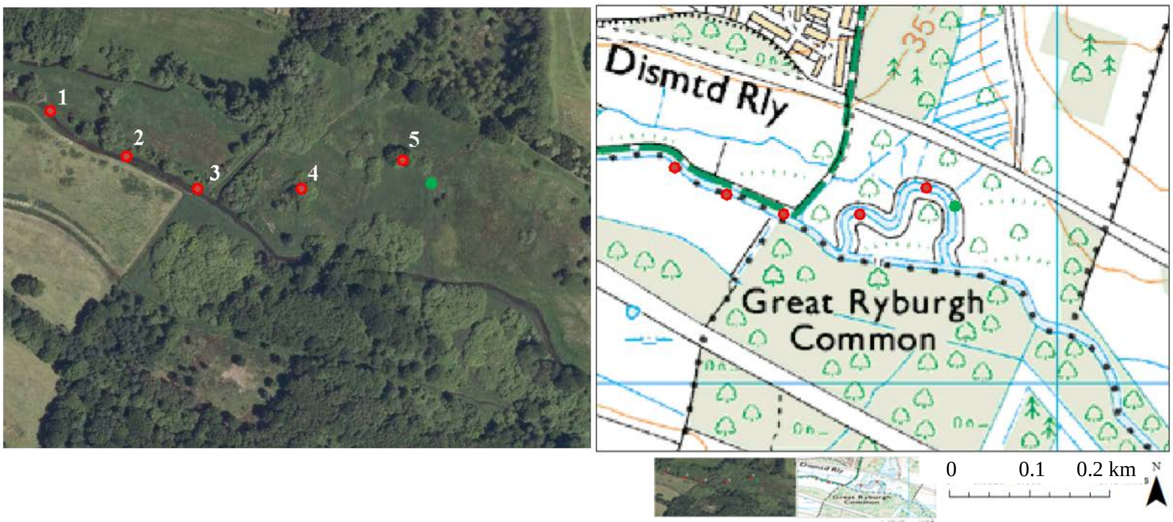

Figure 6.6 Aerial photograph and map showing the locations of the contemporary and palaeoecological sampling sites (Tele Altas, 2012).

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Plate 6.1 The River Wensum palaeochannel before reinstatement. i) and ii) show the palaeochannel before its excavation and reconnection and iii) shows the sediment pit (coordinates: 52.825817, 0.875875).

Two gravel layers were found within the sediment pit providing an interesting stratigraphy. The smaller of the gravel layers lay between 60-70cm, with the larger gravel layer (the historic river bed) found between 110-130cm. From the surface to 50cm deep the material comprised of a mixture of soil and decomposed vegetation. This mix graduated into sand between 50-60cm. Between the two gravel layers were alternating bands of sand and soil.

Plate 6.2 Contemporary study reaches on the River Wensum: i and ii; the contemporary channel, iii and iv; sections of the reinstated/restored river following the reconnection of the palaeochannel.

6.5 Results of the Contemporary Ecological Community Composition

The abiotic variables from each contemporary sampling site were measured *in-situ* during each site visit (see Chapter 3.4.1) to obtain an understanding of the contemporary riverine environment under a range of conditions. The average values recorded during each sampling season are presented in Table 6.2. Following the reinstatement of the meander (November 2010), a riffle was created where the highest flow velocity readings were recorded. Flow velocity at Sites 1, 2 and 3 did not display significant variability between sites and seasons and ranged from 0.114 m s^{-1} to 0.435 m s^{-1} (highest flows at the centre of the channel and lowest at the margins). The pH of the river ranged from 7.6 to 8.7, conductivity ranged between 719 to 894 µS cm^{-1} over the year and water temperature reflected seasonal changes by being highest in summer and lowest during the winter.

Table 6.2 Table highlighting the range of the abiotic measurements and their averages indicated in brackets, for each seasonal sampling undertaken along the River Wensum SSSI.

		Sampling Season				
		Summer 2010	**Autumn 2010**	**Winter 2011**	**Summer 2011**	**Winter 2011**
Abiotic Measurement (Range)	**Depth** (m)	0.71 – 0.72 (0.715)	0.17 – 0.99 (0.69)	26 – 0.96 (0.59)	0.21 – 1.02 (0.533)	0.15 – 0.98 (0.576)
	Flow Velocity (m s^{-1})	0.24 –0.30 (0.27)	0.19 – 1.09 (0.45)	0.36 – 1.04 (0.54)	0.11 – 0.99 (0.37)	0.16 – 1.04 (0.6)
	pH	7.9 – 8 (7.95)	7.8 – 8 (7.87)	8.2 – 8.7 (8.36)	7.6 – 8 (7.9)	7.8 – 8 (7.9)
	Conductivity (µS m^{-1})	0.73 – 0.77 (0.75)	0.73– 0.75 (0.74)	0.72 – 0.73 (0.725)	0.74 – 0.78 (0.76)	89 – 90 (0.895)
	Temperature (°C)	17.8 – 19.8 (18.73)	4.6 – 5.1 (4.8)	7.3 – 7.5 (7.4)	14.3 – 15 (14.6)	4.7 – 5.1 (4.93)

6.5.1 Detrended Correspondence Analysis (DCA)

DCA was undertaken in Canoco (ter Braak & Šmilauer, 2002) using kick sample data from the reinstated meander and contemporary River Wensum channel at Great Ryburgh. This allowed the current macroinvertebrate community structure to be explored over five sampling seasons. The cumulative percentage of variance explained by the four canonical axes was 38.1%, with 21.4% of the species data variance explained by axis 1 and an additional 10.4% explained by axis 2 (Table 6.3).

Table 6.3 Summary of eigenvalues and variance of species data for the DCA of kick samples collected from the River Wensum.

Axes	**1**	**2**	**3**	**4**	**Total inertia**
Eigenvalues	0.266	0.129	0.058	0.021	1.241
Lengths of gradient	2.316	1.425	1.316	0.930	
Cumulative percentage variance of species data (%)	21.4	31.8	36.4	38.1	
Sum of all eigenvalues					1.241

The sample biplot indicates a relatively high degree of variability within the data, but also indicated a degree of seasonal overlap in the community composition (Figure 6.7 i). However, seasonal variability over the five sampling seasons is apparent. Axis 1 sample scores were highest in winter (February) 2011 and were lowest in summer 2010. Summer 2010 samples were collected before the meander reinstatement, so only consists of three samples. Autumn 2010 samples include the samples from the reinstated meander, however one sampling site, (the side of the meander) was not included due to macroinvertebrate colonization having not occurred. The remaining sampling seasons included two sets of samples from both the contemporary and reinstated channel. Axis 2 scores are highest in summer 2010 and 2011. The data suggests that within six months of the channel being reinstated, there was limited difference in the community at sites 4 and 5 (the reinstated meander) compared to the community at sites 1, 2 and 3, showing that recovery and colonization by macroinvertebrates occurred quickly after its reinstatement.

The DCA of the River Wensum's contemporary macroinvertebrate community is presented in Figure 6.7 ii. The positioning of taxa reflects the seasonal patterns highlighted Figure 6.7 i. The caseless caddis fly larvae, *Rhyacophila dorsalis*, displays high axis 1 scores due to its high abundance at all sites during winter 2011. In addition to this the water beetle, *Nebrioporus depressus elegans* and the caddisfly *Athripsodes albifrons,* also have high axis 1 scores due to their great abundance in winter 2011.

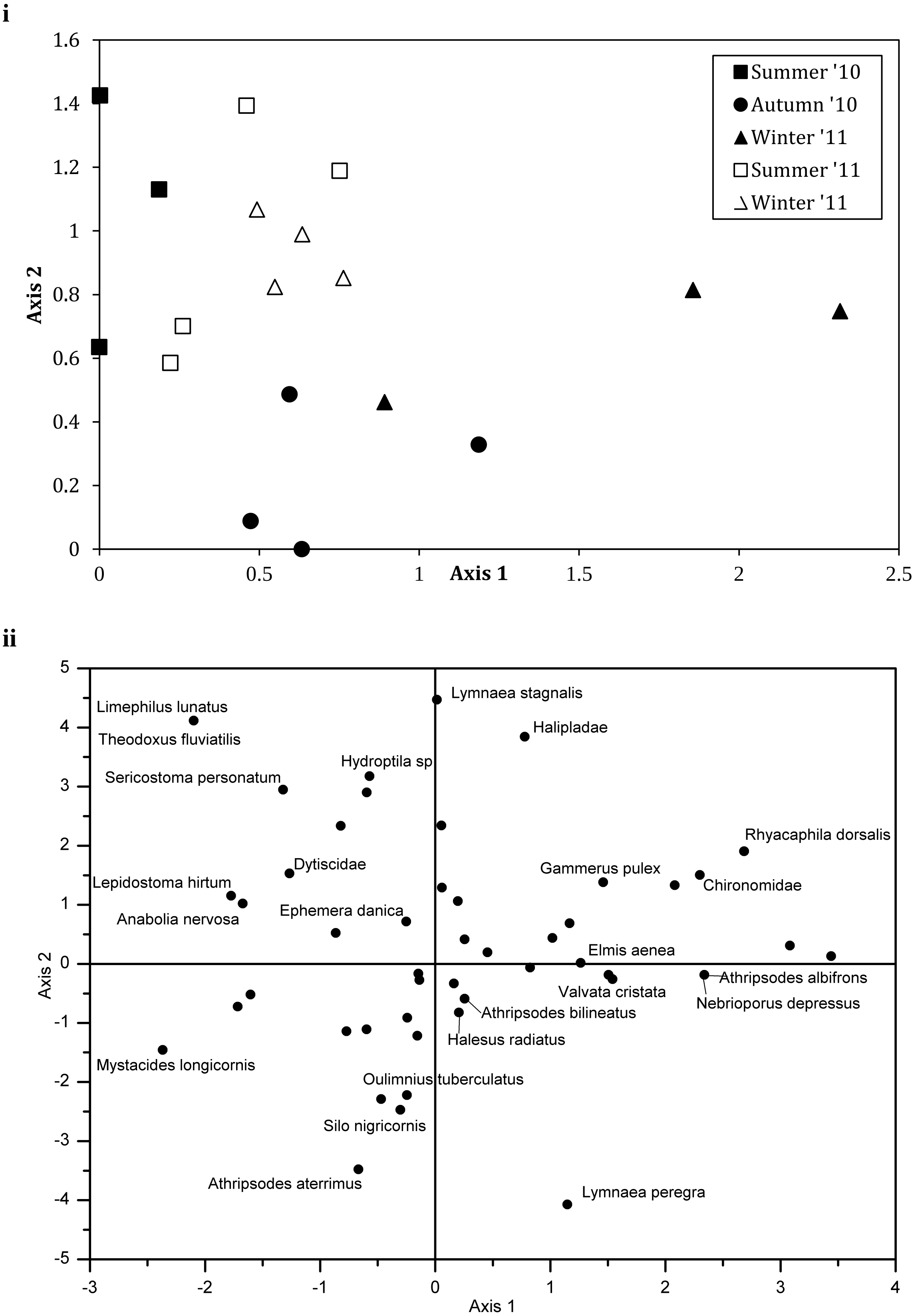

Figure 6.7 DCA ordination biplots: i) seasonal samples (summer 2010 – winter 2011); ii) Macroinvertebrate community taxa biplot.

Species with negative axis 1 scores but positive axis 2 scores such as the caddisfly larvae; *Limephilus lunatus, Anabolia nervosa, Lepidostoma hirtum* and *Sericostoma personatum*, were predominantly recorded in the summer sampling seasons (Figure 6.7 i). Species plotted around the origin of Figure 6.7 ii, such as *Gammerus pulex* (freshwater shrimp), *Elmis aenea* (water beetle) and *Ephemera danica* (mayfly) were recorded during all sampling seasons, although they displayed markedly different abundances over time.

6.5.2 Detrended Canonical Correspondence Analysis (DCCA)

DCCA was used to investigate the relationships between invertebrate community abundances and environmental variables over five sampling seasons. Quantitative Surber data and five environmental variables were included in the analysis; two hydrological variables (depth and flow) and three water chemistry variables (pH, conductivity and temperature). The results generated allowed temporal pattern variability to be examined and to explore how influential these five abiotic variables were in structuring the macroinvertebrate community.

A summary of the DCCA output for the axes and significant variables for the River Wensum Surber Sampler data is presented in Table 6.4. A total of 17.4% of the variance in the species data and 64.4% of the species-environment relationship could be accounted for on the first four axes. Axis 1 explained 10.4% of the variance in the species data and 37.3% of the variance within the species-environment data. Axis 1 has a significant gradient associated with pH ($p < 0.005$) (Figure 6.8 i). Scores for abiotic measurements varied between seasons and years, with pH decreasing from winter (February) 2011, as conductivity increased and peaked in summer and winter (December) 2011. Axis 2 accounts for a further 4.9% of the variation in species data and 27.1% of the species-environment relationship. Axis 2 scores increase from summer 2010 to winter 2011 when temperature ($p < 0.005$) was lowest and conductivity peaked. The seasonal macroinvertebrate community composition displays significant overlap, although seasonal clustering is still apparent.

Table 6.4 Results of DCCA for the River Wensum. **A**; Summary of the eigenvalues and percentage of variance of species data and species environment relationship explained on the first four canonical axes for Surber samples collected from the River Wensum. **B**; The significance of the first canonical axis and environmental variables using forward selection procedure in Canoco and the Monte Carlo random permutations test (999 permutations).

A	**Axes**	**1**	**2**	**3**	**4**	**Total inertia**
	Eigenvalues	0.264	0.123	0.035	0.019	2.540
	Lengths of gradient	2.133	1.345	1.006	1.141	
	Cumulative percentage variance of species data (%)	10.4	15.3	16.6	17.4	
	Cumulative percentage variance of species-environment relation (%)	37.3	64.4	64.4	64.4	
	Sum of all eigenvalues					2.540
B		F ratio	P value			
	Significance of first canonical axis	9.635	< 0.005			
	Significance of all canonical axes	4.972	< 0.005			
	Significance of Env. Variable					
	1) Conductivity	9.09	< 0.005			
	2) Temperature	8.60	< 0.005			
	3) pH	3.17	< 0.005			
	4) Flow velocity	1.41	Not Significant			
	5) Depth	1.16	Not Significant			

The DCCA species biplot (Figure 6.8 ii) indicated a number of associations between taxa and seasons. Species positioned around the centre of the biplot were typically recorded in all five sampling seasons. These species included *Potamopyrgus antipodarum* (snail), *Elmis aenea* (water beetle – both adult and larvae), *Polycentropus flavomaculatus* (caddisfly larvae) and *Chironomidae* (nonbiting midge). Seasonal occurrences of some taxa were apparent on both axis 1 and 2. Hydroptila sp (caddisfly larvae) was only recorded in winter (December) 2011 and displays a high score on each axis. Species such as the caddisfly larvae; *Anabolia nervosa*, *Halesus radiatus* and *Lepidostoma hirtum*, which have high axis 1 scores but a negative axis 2 had high abundances in summer 2011, and summer 2010 when temperature was elevated and flow velocities were lower.

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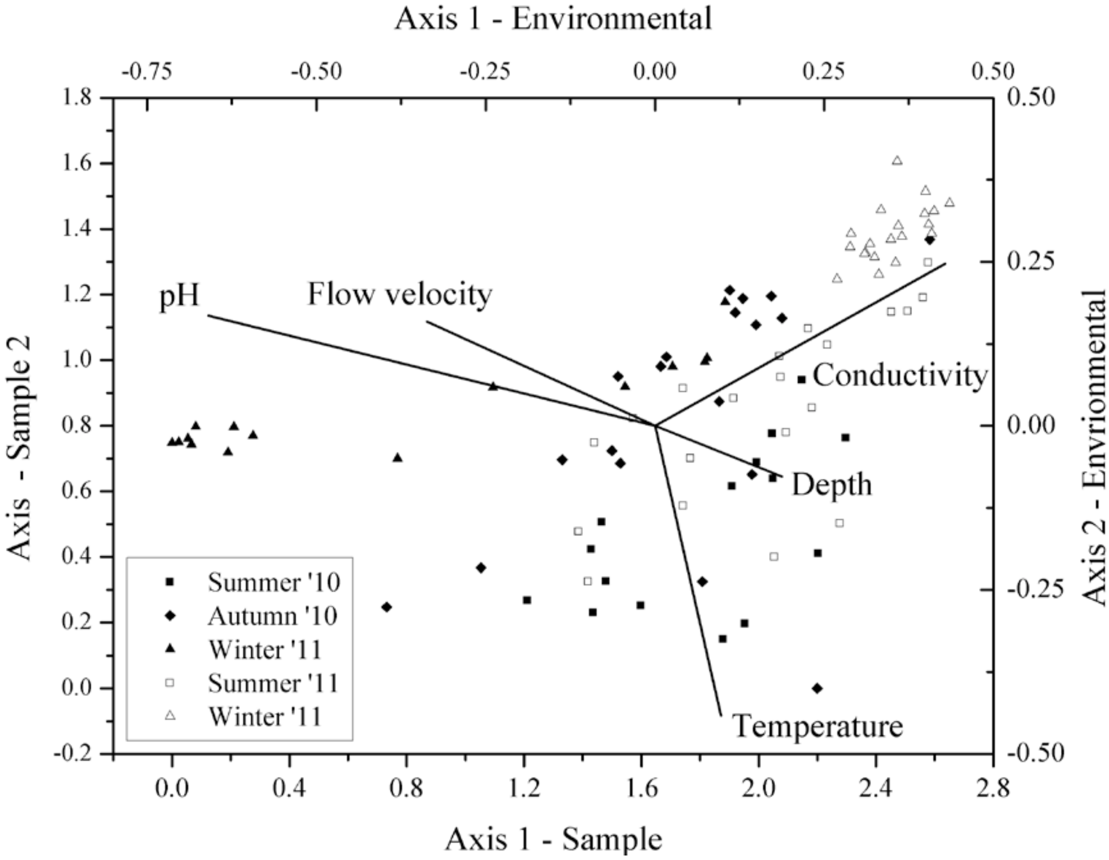

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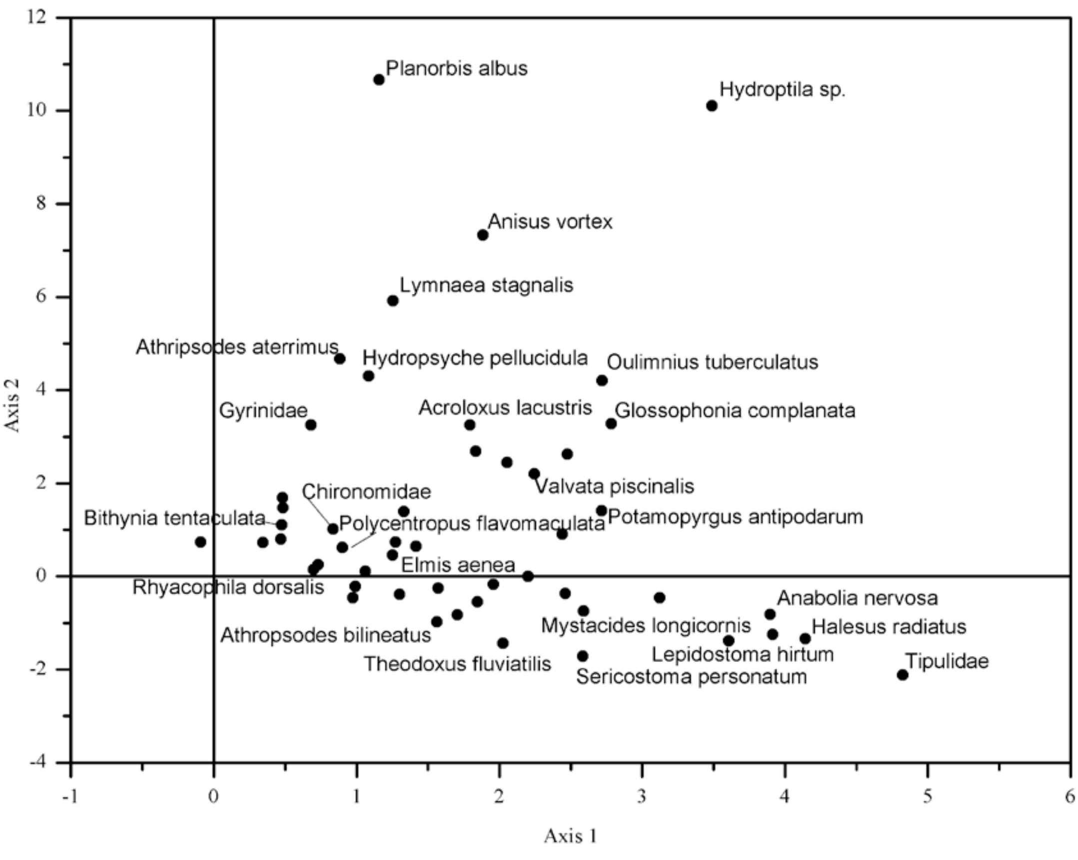

Figure 6.8 DCCA ordination of macroinvertebrate and abiotic data, with season as a covariable: i) environment biplot indicating sampling occasions; ii) Species biplot.

6.5.3 Ecological Indices

A range of ecological indices were used to characterise the community, comprising of BMWP, ASPT, LIFE, PSI and CCI. One Way Analysis of Variance (Anova) indicated that there are no significant differences between the indices over time (Table 6.5). The seasonal change in the BMWP and ASPT scores is shown in Figure 6.9 i and ii. The ASPT results indicate that both summer season and winter 2010 have the highest scores, with levels decreasing in winter 2011 (February and December) and were at their lowest in December 2011, but were not statistically significant.

Table 6.5 One-Way ANOVA results between each of the ecological indices

		df	Sum of Squares	F-ratio
BMWP	Between groups	2	14135.111	3.480
ASPT	Between groups	2	6.720	1.912
LIFE	Between groups	2	3.933	3.956
PSI	Between groups	2	2675.146	1.578
CCI	Between groups	2	24.495	0.247

Seasonal LIFE scores did not vary significantly between each season (Figure 6.9 iii), however highest scores were recorded in winter (February) 2011 and were lowest in winter (December) 2011. The majority of taxa were classified under LIFE flow group IV and flow group II indicating variable slow and fast flowing microhabitats across all sites (Extence *et al.,* 1999).

PSI scores display no significant variation overtime (Figure 6.9 iv), with scores indicating the river to be ‘sedimented’ to ‘moderately sedimented’ (Extence *et al*., 2011). The River Wensum Restoration Strategy highlights silt and sediment accumulation to be one of the major issues currently affecting the river (JBA, 2007).

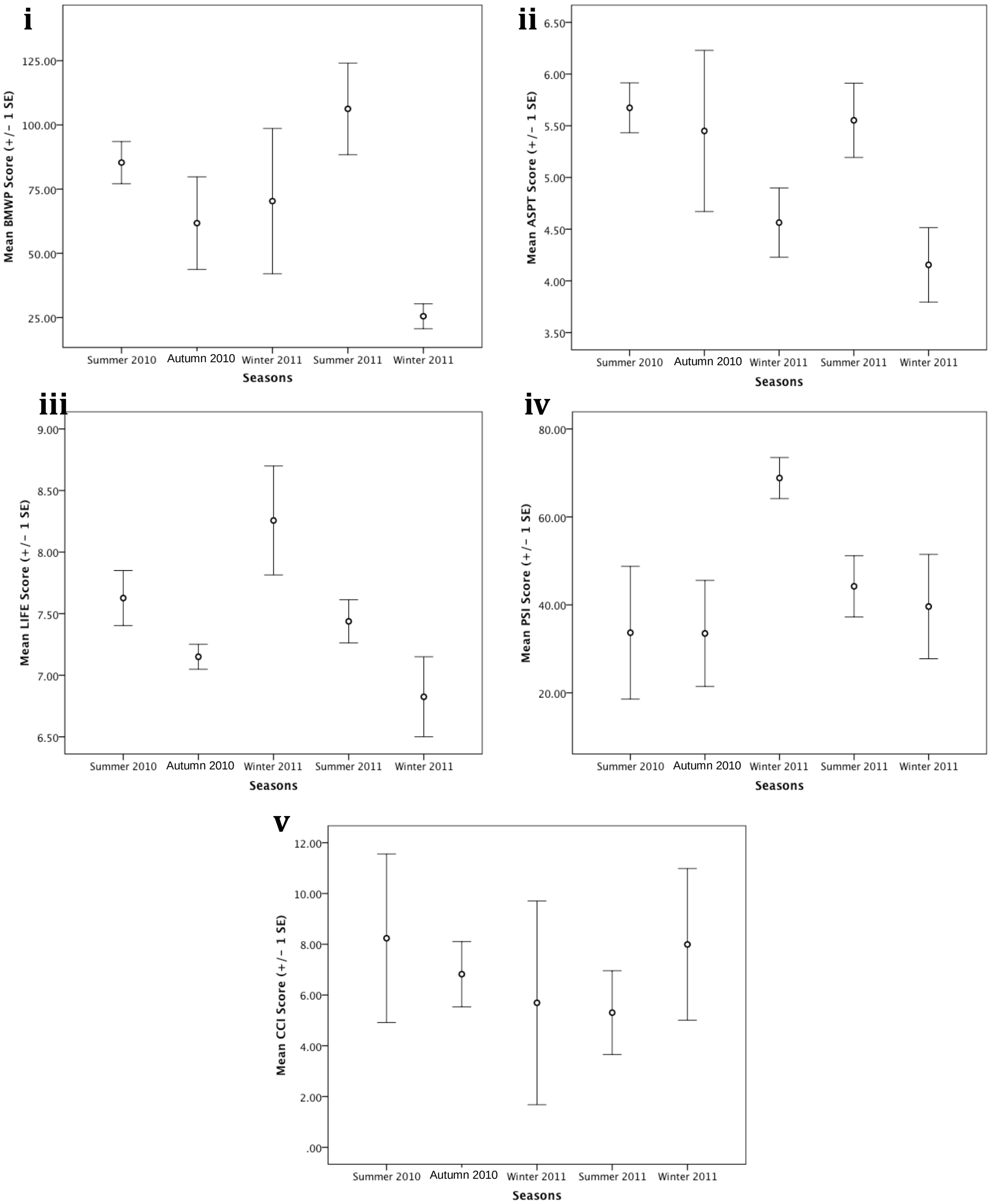

Figure 6.9 Error bar plots showing the mean i). BWMP score, ii) ASPT score, iii) LIFE score. iv) PSI score and v) CCI score (plus or minus one standard error) for each sampling season for the River Wensum (Sample removed from restored side of meander site in Autumn 2010 due to only 2 taxa being found).

CCI scores did not vary significantly over time (Figure 6.9 v) and indicate that the river supports at least one species of restricted distribution and/or a community of moderate taxon richness, thus giving it a 'moderate conservation value' (Chadd and Extence, 2004). *Nebrioporus depressus* was the highest scoring taxa with a value of 7, which gives it a status of 'Notable'. The remainder of the species ranged from being 'very common' to 'local' (JNCC, 2005).

6.6 Comparison of the Contemporary River Ecology and the Palaeoecology

The comparison of the ecology from the contemporary and palaeo channel of the River Wensum potentially provides an overview of how the riverine environment and macroinvertebrate community has changed since the channel was straightened in 1946. In order to ensure the two sets of results were fully comparable, contemporary results containing only the combined abundances of Gastropoda, Coleoptera and Trichoptera were used. These three proxies promise in excess of 60% of the total freshwater invertebrate taxa (Extence et al 1999) and are well represented in the sub-fossil record.

6.6.1 Detrended Correspondence Analysis (DCA)

DCA was used to examine the similarities and differences in the community composition between the contemporary and palaeoecological macroinvertebrate assemblages. This was specifically undertaken in order to examine the presence of any biotic and/or environmental gradients within the data. All faunal data were transformed ($\log_{10}$ +1) prior to analysis to reduce any clustering of abundant or common taxa at the centre of origin. The eigenvalues are presented in Table 6.6, and provide a measure of the relative importance of each axis in the analysis. The first four axes accounted for a total of 39.8% of the variance in the faunal community data; with axis 1 representing 24.9% of the variance within the species data and axis 2 a further 7.1%.

There is clear separation of the contemporary and palaeoecological samples with each sampling period forming 2 distinct clusters (Figure 6.10 i). This indicates that there is not a homogenous community composition across time periods.

Table 6.6 Summary of the eigenvalues and cumulative variance of contemporary and palaeoecological data for the DCA of kick and palaeo-core samples from the River Wensum.

Axes	1	2	3	4	Total inertia
Eigenvalues	0.661	0.189	0.142	0.064	2.658
Lengths of gradient	3.479	2.441	1.742	2.043	
Cumulative percentage variance of species data (%)	24.9	32.0	37.3	39.8	
Sum of all eigenvalues					2.658

The species biplot (Figure 6.10 ii) indicates a number of taxa that were associated with different time periods (contemporary verses palaeo community). Species with low and negative axis 1 values such as the snail, *Valvata cristata* (which is restricted to well oxygenated waters); the water beetle, *Oulimnius tuberculatus* and the caddisfly larvae, *Lepidostoma hirtum* (which is a shredder and lives in waters with high organic content/leaf litter) and *Sericostoma personatum*, (whose preferred habitat is sandy sediments), were exclusively recorded within the contemporary samples (Wagner, 1990; Azevedo-Pereira et al., 2006). In contrast, species with a high positive axis 1 score including *Anisus vortex* (snail), *Hydropsyche angustipennis*, *Molanna angustata* and *Goera pilosa* (caddisfly), were only recorded in the palaeoecological samples.

In order to determine if the results were significantly influenced by the presence of specific taxa that may be a function of differences in sampling, preservation and abundance associated with the contemporary and palaeoecological samples, only taxa common to both data sets were used in the subsequent analysis (Table 6.7). In addition, to ensure that species abundance differences in the contemporary and palaeoecological samples were not influencing the results, all abundances were converted into presence and absence. This process reduces the sensitivity of the rare taxa that can distort results. The palaeoecological samples had lower numbers of taxa, therefore this provided an objective, unbiased way of comparing the contemporary and palaeo ecologies. DCA was then repeated with the new presence absence taxa.

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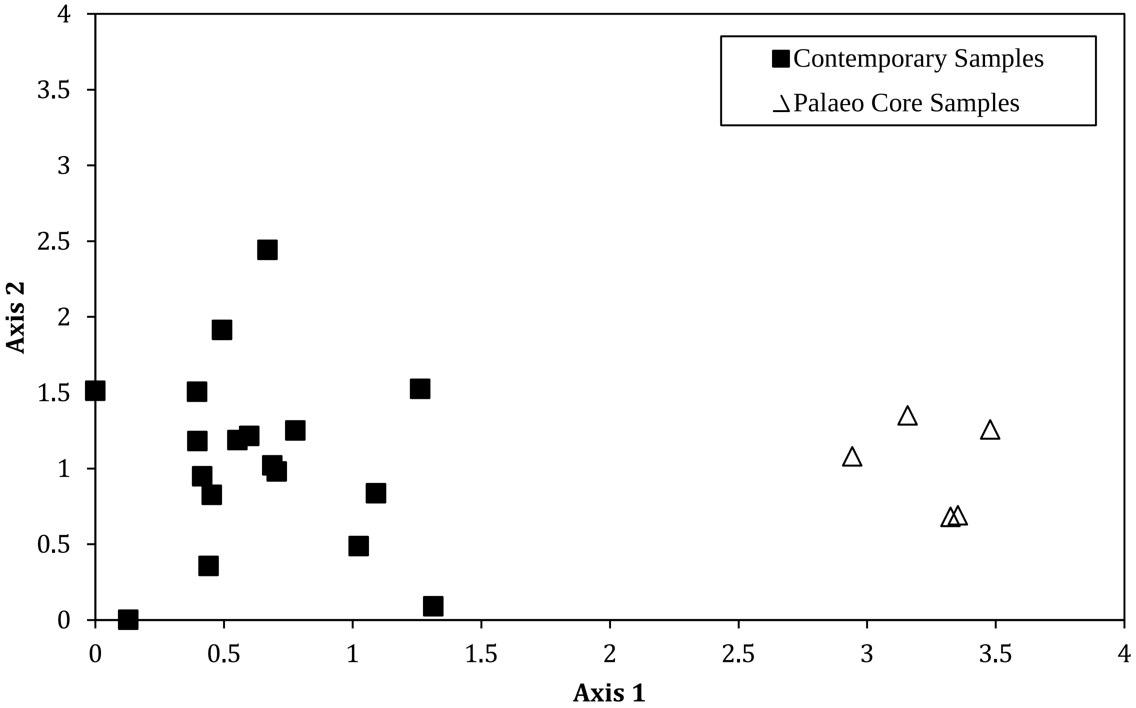

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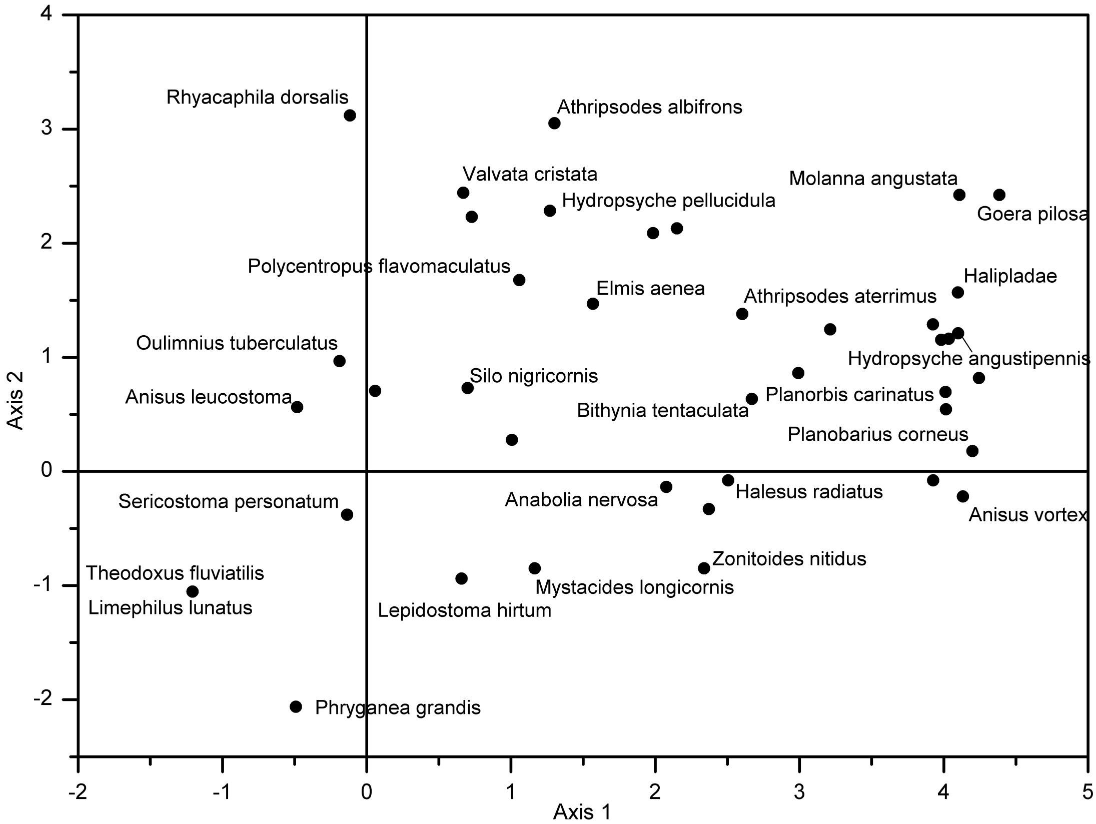


Figure 6.10 DCA faunal plot of presence/absence data from the River Wensum: i) Different temporal data sets used ii) Faunal biplot.

Table 6.7 List of taxa from the River Wensum that were removed as they occurred in only one data set or were combined from the merged list of contemporary and palaeo species due to differences in the taxonomic resolution.

Species Removed	Species Combined
Theodoxus fluviatilis	*Valvata cristata* and *Valvata piscinalis.*
Brychius elevates	*Lymnaea palustris, Lymnaea peregra* and *Lymnaea stagnalis.*
Halipladae	*Planorbarius corneus, Planorbis carinatus, Anisus vortex, Anisus leucostoma, Bathyomphalus contortus* and *Acroloxus lacustris.*
Helophorus	*Elmis aenea* and *Elmidae.*
Rhyacophila dorsalis	*Oulimnius* and *Oulimnius tuberculatus.*
Phryganea grandis	*Hydropsyche pellucidula* and *Hydropsyche angustipennis.*
Phryganea bipunctata	*Limephilus lunatus* and *Potamophylax cingulatus.*
Molanna angustata	*Athripsodes aterrimus, Athripsodes albifrons, Athripsodes bilineatus* and *Mystacides longicornis.*
Lepidostoma hirtum	*Goera pilosa* and *Silo nigricornis.*
Sericostoma personatum	

A summary of the eigenvalues for the presence absence data are presented in Table 6.8. The DCA explained 39.5% of the variance across the first four axes; with 19.1% of the variance in the faunal data being explained by axis 1 and a further 10.7% by axis 2.

Table 6.8 A summary of the eigenvalues and cumulative variance of the presence/absence contemporary and palaeoecological data for the DCA of kick and palaeo-core samples from the River Wensum.

Axes	1	2	3	4	Total inertia
Eigenvalues	0.300	0.168	0.103	0.050	1.574
Lengths of gradient	2.318	1.631	2.647	1.993	
Cumulative percentage variance of species data (%)	19.1	29.8	36.3	39.5	
Sum of all eigenvalues					1.574

The differences between the data sets observed in the original DCA (Figure 6.10 i) has been reduced markedly by the use of the taxa common to both sampling methods and presence absence data (Figure 6.11 i). The difference between the contemporary and palaeoecological samples along axis 1 has been largely removed and the communities now overlap significantly. However, there are still some differences. Species with a low and negative axis 1 score were recorded more frequently in contemporary than palaeoecological samples; *Potamopyrgus antipodarum* (snail) that is tolerant to a wide range of aquatic conditions, *Oulimnius* (riffle beetle) that is found in well oxygenated, running waters and *Hydropsyche pellucidula* (caseless caddisfly larvae) which are known to preferentially colonize gravel river beds with macrophyte patches (Elliott, 2008; Bálint & Ujvárosi, 2009). Species with a higher axis 1 score were recorded with similar frequency in both data sets.

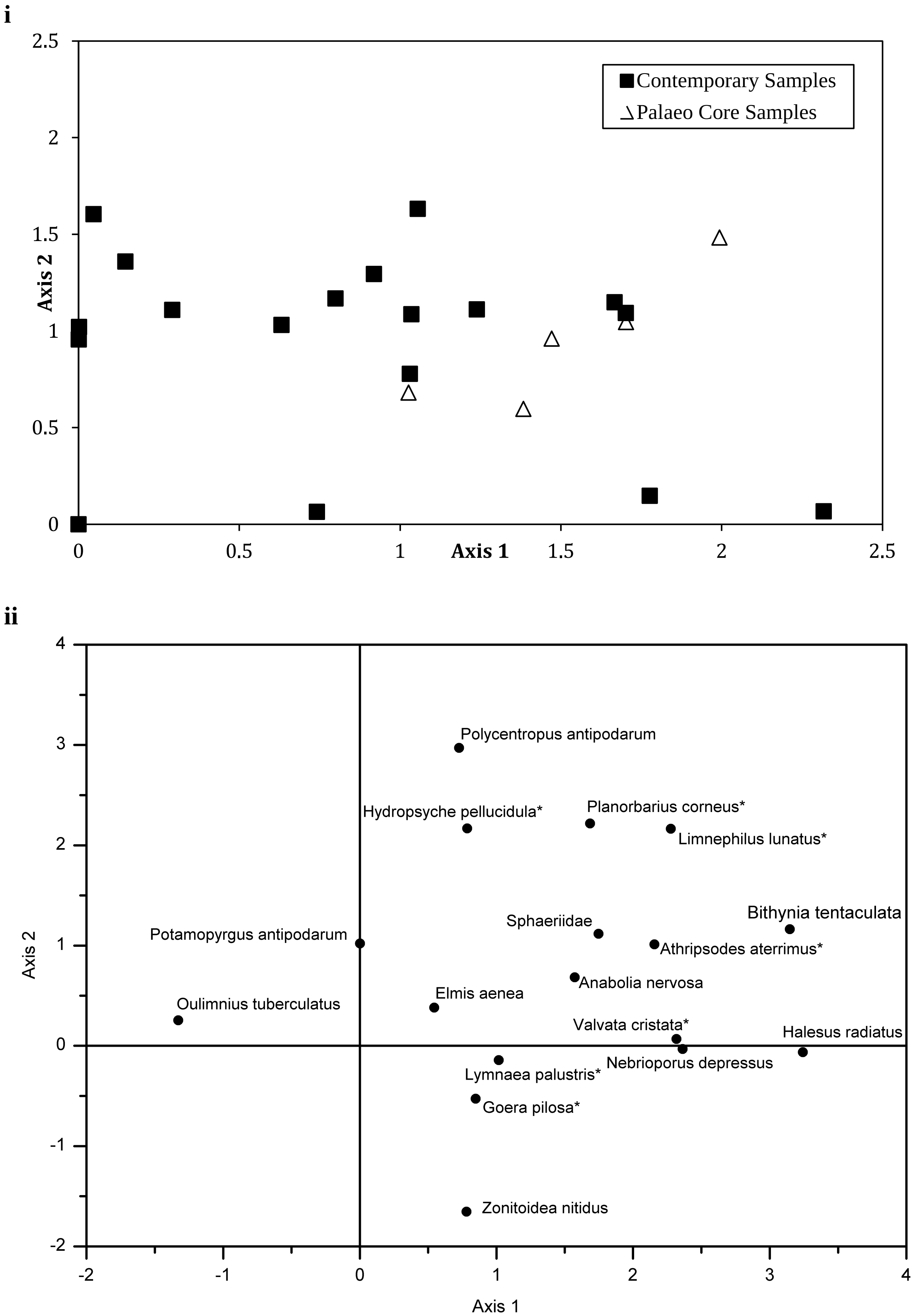


Figure 6.11 Contemporary and palaeoecological species Presence/Absence DCA ordination biplots: i) seasonal variability; ii) Macroinvertebrate community data from the River Wensum (Species marked with a * indicate a combination – see Table 6.7).

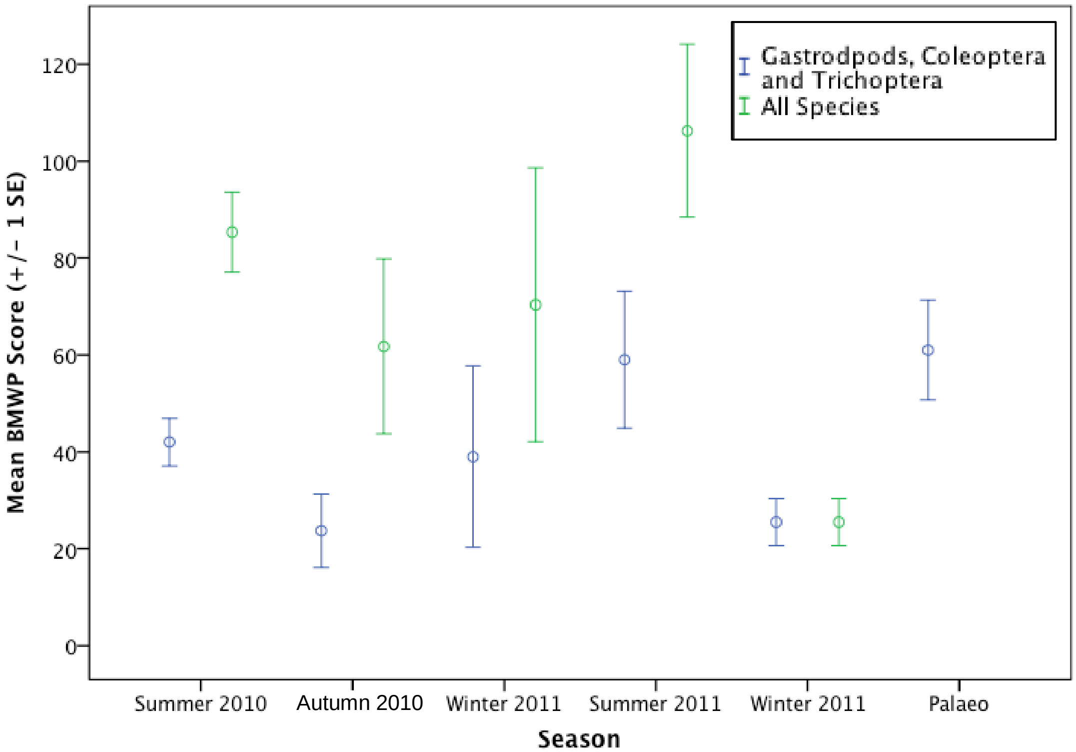
6.6.2 Ecological Indices

There were no significant differences between the contemporary and palaeoecological indices for any of the variables examined with the exception of the PSI score (Table 6.9). The results of the water quality indices, BMWP and ASPT are displayed in Figures 6.12 i and ii. There was a significant difference between summer 2011 and winter 2011 for the BMWP and ASPT score. Mean LIFE scores varied between 6.65 and 8.30 and were similar across all sites (Figure 6.13). There was only a significant difference between winter 2011 and summer 2011 (Table 6.9). The mean PSI results for the contemporary indicated that there was a significant difference between the palaecological and winter 2011 (Table 6.9). The PSI scores for the palaeoecological sample indicates that the PSI scores were at the lower end of the range recorded within the contemporary river and that historically there may have been more fine sediment available and present in the channel; although the results are not significantly different in most instances. These results may have been influenced by the sampling of the reconnected channel in winter 2011, as very few taxa were recorded during this season due to recolonisation having not fully occurred.

Table 6.9 Kruskal-Wallis pair-wise comparison between seasons for macroinvertebrate community indices along the River Wensum. $^*P < 0.05$, $^{**} P < 0.01$, $^{***} P < 0.001$.

		Summer 2010	Autumn 2010	Winter 2011	Summer 2011	Winter 2011
BMWP	Autumn 2010	NS	-	-	-	-
	Winter 2010	NS	NS	-	-	-
	Summer 2011	NS	NS	NS	-	-
	Winter 2011	NS	NS	NS	*	-
	Palaeo 2011	NS	NS	NS	NS	*
ASPT	Autumn 2010	*	-	-	-	-
	Winter 2010	NS	NS	-	-	-
	Summer 2011	NS	NS	NS	-	-
	Winter 2011	NS	NS	NS	NS	-
	Palaeo 2011	NS	NS	NS	NS	NS
LIFE	Autumn 2010	NS	-	-	-	-
	Winter 2010	NS	NS	-	-	-
	Summer 2011	NS	NS	*	-	-
	Winter 2011	NS	NS	NS	NS	-
	Palaeo 2011	NS	NS	NS	NS	NS
PSI	Autumn 2010	NS	-	-	-	-
	Winter 2010	NS	*	-	-	-
	Summer 2011	NS	*	*	-	-
	Winter 2011	NS	NS	NS	NS	-
	Palaeo 2011	NS	NS	*	NS	NS

I



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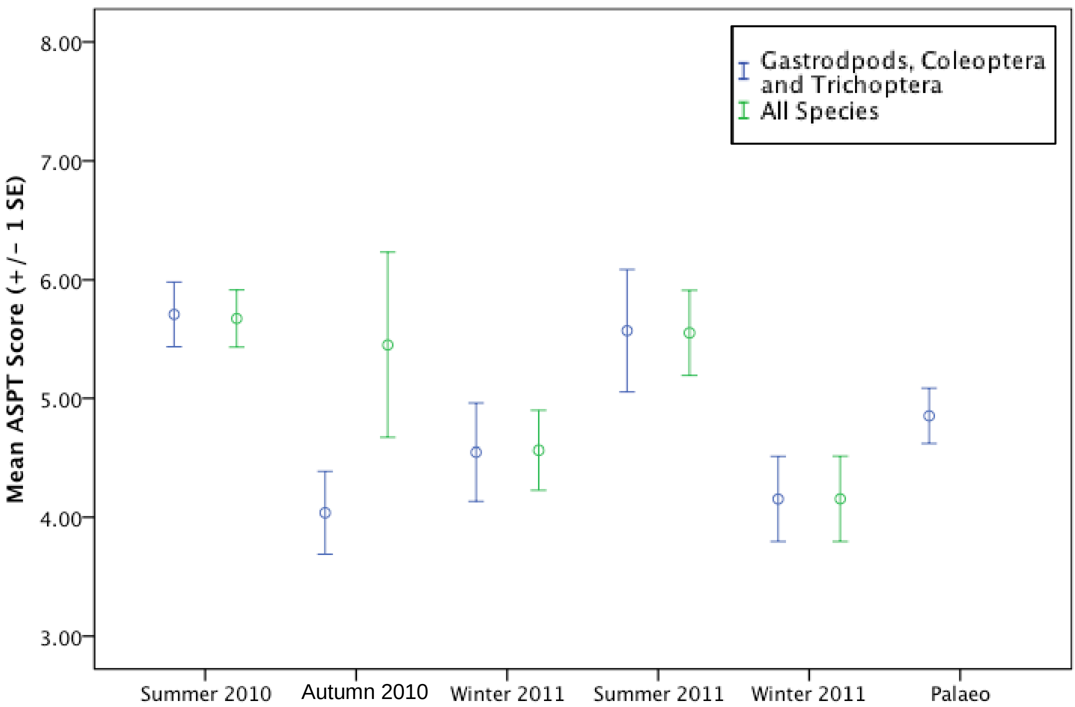

Figure 6.12 Error bar plot showing the comparisons between i) the mean BMWP score and ii) the mean ASPT score (plus or minus one standard error) for the contemporary and palaeoecological samples taken from the River Wensum, and the full contemporary species list for each sampling season.

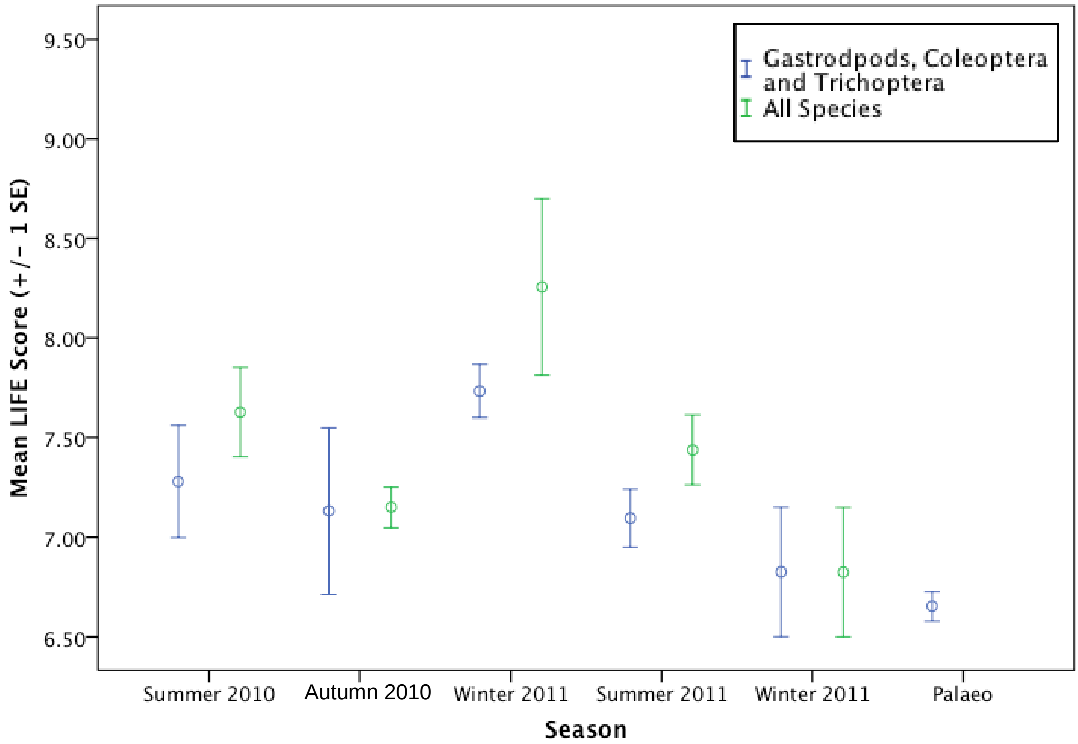

Figure 6.13 Error bar plot showing the comparisons between the mean LIFE score (plus or minus one standard error) for the contemporary and palaeoecological samples taken from the River Wensum, and the full contemporary species list for each sampling season.

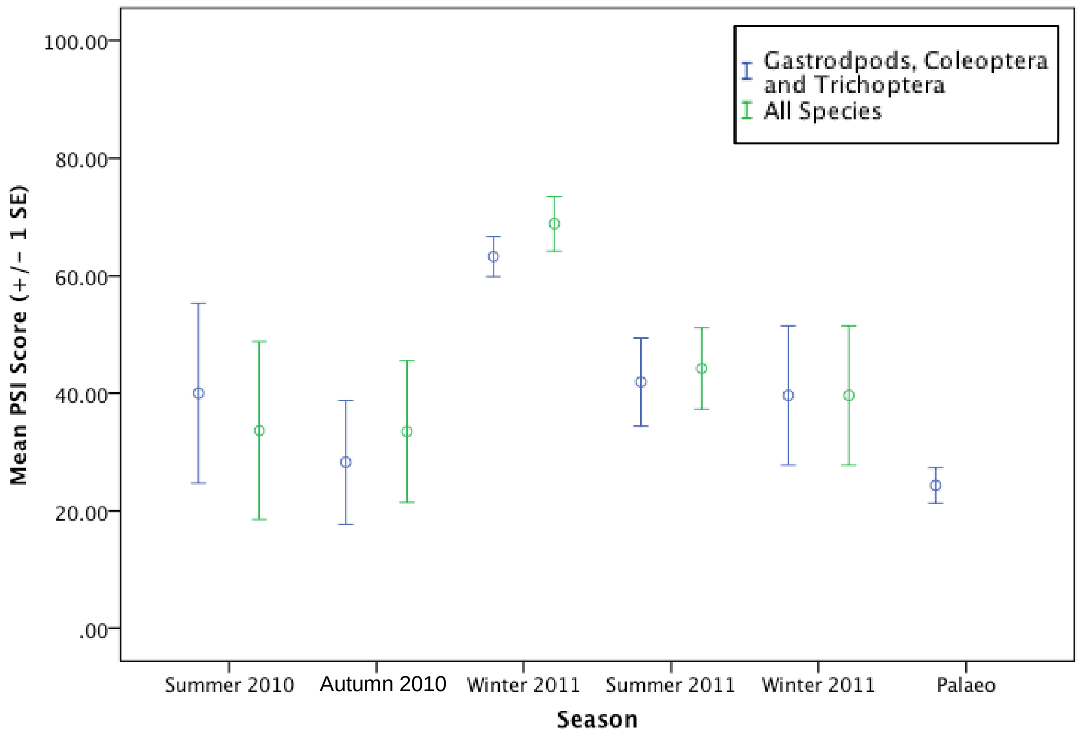

Figure 6.14 Error bar plot showing the comparisons between the mean PSI score (plus or minus one standard error) for the contemporary and palaeoecological samples taken from the River Wensum, and the full contemporary species list for each sampling season.

6.7 Ecological interpretation

Comparison between the contemporary and palaeoecological results for the River Wensum highlights a number of similarities and differences between the communities. By examining the ecological preferences associated with each species and the species common and unique to both time periods it helps to reconstruct the nature of riverine habitats present at the time (Table 6.10). Fifteen species were found to be common to both sampling periods, with fourteen species unique to the contemporary river and thirteen unique to the palaeoecological sample.

Table 6.10 Summary of the species found both common and unique to each temporal sample.

Common to both temporal samples	Found only in the contemporary samples	Found only in the palaeoecological samples
Valvata piscinalis	*Theodoxus fluviatilis*	*Lymnaea palustris*
Bithynia tentaculata	*Valvata cristata*	*Lymnaea peregra*
Lymnaea stagnalis	*Potamopyrgus antipodarum*	*Planorbarius corneus*
Zonitoides nitidus	*Anisus leucostoma*	*Planorbis carinatus*
Sphaeriidae	*Acroloxus lacustris*	*Anisus vortex*
Nebrioporus depressus	*Rhyacaphila dorsalis*	*Bathyomphalus contortus*
Elmis aenea	*Phryganea grandis*	*Brychius elevatus*
Oulimnius tuberculatus	*Phryganea bipunctata*	*Hydropsyche angustipennis*
Polycentropus flavomaculatus	*Limephilus lunatus*	*Potamophylax cingulatus*
Hydropsyche pellucidula	*Athripsodes albifrons*	*Molanna angustata*
Anabolia nervosa	*Mystacides longicornis*	*Goera pilosa*
Halesus radiatus	*Silo nigricornis*	
Athripsodes aterrimus	*Lepidostoma hirtum*	
Athripsodes bilineatus	*Sericostoma personatum*	
Chironomidae		

The species found exclusively within the palaeoecological samples suggest that historically, the river contained large areas of slow flowing, fine sediment dominated habitats, interspersed with some faster flowing coarse-grained habitats and abundant growths of instream macrophytes. The majority of taxa were classified in LIFE flow group IV indicating slow flowing waters (Extence *et al*., 1999). The past riverine environment contained more species associated with aquatic or emergent vegetation, with a preference for muddy or silty substrates and slow moving waters (e.g. the Gastropods; *Lymnaea palustris, Planorbarius corneus, Planorbis carinatus* and *Anisus vortex* - however *Anisus vortex* is very common within clean, well oxygenated waters) (Kerney, 1999). *Goera pilosa* is typically associated with high velocities/ fast flowing waters and is classified as LIFE flow group 1 representing flow velocities of >100 m s^{-1}. It also displays a preference for gravel substrates as it was found to prefer cobbles in deep water (20-30cm) with a smooth surface in a Slovenian stream (Wallace et al., 2003; Urbanič et al., 2005). Another species indicator of heterogeneous gravel substrates, found only within the palaeoecological samples, is *Potamophylax cingulatus* whereas in comparison *Molanna angustata* displays a preference for sand (Eutaxa, 2013). The combination of taxa suggests a diversity of habitats.

The species unique to the contemporary samples also suggest heterogeneous habitats and substrates but a larger range of flow velocity compared to the palaeochannel. *Theodoxus fluviatilis* and *Valvata cristata* indicate well-oxygenated waters, with *Acroloxus lacustris* showing preference towards clean waters (Kerney, 1999). The cased caddisfly larvae *Phryganea bipunctata* is associated with marginal vegetation as they construct cases from hollow stalks or from leaf litter fragments (Greenwood et al., 2006) and *Limephilus lunatus* indicate slower flowing marginal environments (Urbanič et al., 2005). *Sericostoma personatum* was abundant within the contemporary community and is associated with sands and silts, where the diatom community is well developed and *Potamopyrgus antipodarum* thrives in silt substrates within slow flowing steams of high nutrient content (Nilsson & Holmen, 1995; Wallace et al., 2003) but is absent from palaeoecological samples.

Through examining the substantial number of species common to both the contemporary and palaeoecological samples, this research suggests that the River Wensum has not experienced significant changes to its ecology since 1946 and diversity/heterogeneity of flow and habitats remain similar. Over time there is evidence for slower flow velocities as indicated by the presence of *Polycentropus flavomaculatus, a* species commonly found in slower flowing water or marginal areas of the lower reaches of the river system, and considered more tolerant of higher temperatures and lower oxygen levels than other Polycentropidae (Edington & Hildrew, 1995). Other species such as

Valvata piscinalis, *Bithynia tentaculata* and *Anabolia nervosa* are also common in slow-moving lowland rivers and show a particular preference for substrates comprised of mud and silt where dense growths of aquatic plants occur. *Bithynia tentaculata* shows an additional preference towards well-oxygenated waters and abundances indicate that it has a higher abundance within the palaeoecological samples. The family Leptoceridae is represented by *Athripsodes aterrimus*, *Athripsodes bilineatus* and *Athripsodes albifrons*. *Athripsodes aterrimus* and *Athripsodes bilineatus* are found in both contemporary and palaeoecological samples. *Athripsodes aterrimus* is often associated with mud and sandy substrates and prefers low flow velocities. Whereas *Athripsodes bilineatus* and *Athripsodes albifrons* both dwell in faster flowing water on stony or sandy substrata (Wallace et al., 2003). *Halesus radiatus*, a sub family of Limnephilidae, is an additional species that is commonly found in streams and rivers on stony substrates (Wallace et al., 2003).

6.8 Summary

This chapter has presented the case study of the River Wensum SSSI study reach. Detailed site and macroinvertebrate community descriptions and historical channel changes have been described through the use of maps and photographs. Key findings of this chapter include:

- By performing DCCA and DCA in Canoco, results indicate that there are clear environmental, seasonal and temporal gradients that charaterise the contemporary and palaeoecological samples.
- The combination of taxa found within the palaeo community suggests a diversity of rheophilic and limnophlic habitats (mesohabitats) were present within the historic river, but also indicates the river was characterised by taxa associated with slower flow velocities, due to its meandering morphology.
- Results gained from the ecological indices suggest that despite the slight differences, the current and historic ecologies and environment are mostly similar.
- The River Wensum's recovery rates after disturbance (the reattachment of the palaeochannel) show to be fast, with the macroinvertebrate community fully colonising the channel 6 months after its reattachment.

CHAPTER SEVEN: SYNTHESIS AND DISCUSSION

7.1 Introduction

The influence that humans have had, and continue to exert on river channels has been extensive throughout history to the extent that rivers are now widely perceived to be one of the most modified and degraded ecosystems globally (England et al., 2008). Human activity is by far the most significant factor influencing UK riverine evolution, primarily due to society's dependency on riverine water resources and ecosystem services (Mooney et al., 2009). Therefore it has been argued that society needs to assume responsibility for riverine health, in order to ensure ecological integrity in addition to managing the resource sustainably (Convertino et al., 2013). River restoration is a global issue and is increasingly becoming an integral part of national and international programs aimed at improving the health and integrity of riverine ecosystems (Mainstone et al., 2011). It provides an opportunity for society to test its ability to recreate and/or restore complex ecosystems from degraded states. However this represents one of the major challenges currently facing freshwater scientists (Bernhardt & Palmer, 2011).

The research presented in this thesis has explored the use of multiple lines of evidence using contemporary, palaeoenvironmental and palaeoecological data to define reference conditions that may form the basis for river restoration to meet needs of the WFD and biodiversity objectives for other conservation designations. This chapter synthesizes the research and analysis from the preceding chapters within the context of current published literature and in relation to the original hypotheses of this thesis:

4. Using historic information (historic maps and documents) it is possible to identify instream morphological and habitat features that may have been significantly degraded by previous management operations.
5. Elements of the instream faunal community have been significantly compromised or may have become locally extinct as a result of historic channel management operations and that examination of palaeoecological communities; aquatic beetle (Coleoptera), caddisfly larvae (Trichoptera) and snails (Gastropods), will enable identification of this and other changes to in-stream communities.

6. Through detailed examination of the contemporary riverine communities and those within former channels (palaeochannels and floodplain deposits) it is possible to determine historic environments that may form the basis for future river restoration schemes and inform the definition of 'reference conditions' for WFD purposes.

The wider implications of the results presented are considered in relation to the restoration and conservation of riverine systems and with reference to current policy legislation (including the EUs WFD, the Groundwater Directive and the Habitats and Species Directive). The implications of up-scaling site scale results to provide a landscape perspective are considered and the chapter provides evidence highlighting the essential need for post restoration monitoring, not only for current understanding but also to ensure future success of river restoration programmes. Finally, the potential implications that future climate variability and change could have on rivers and their possible restoration programmes are considered.

7.2 Overview of Case Study Results and Key Findings

The primary aim of this research project was to determine reference conditions and the historical ecological change for three SSSI rivers through the use of palaeoecological techniques, to help underpin the development and implementation of strategic restoration plans. By providing an overview of the potential benefits of palaeoecological techniques, it provides the basis on which to determine a rivers ecological condition at varying points in the past (1956, River Eye; 1820-1850, River Hull and 1946, River Wensum) and a way of defining reference condition, that may have otherwise not been possible. It is important to recognise that river SSSIs are not always pristine examples of rivers, they are sometimes the best remaining examples of their type. They may be subject to many of the impacts that are prevalent in the wider river network and also demonstrates why they are useful for exploring best practice approaches for future river restoration programmes. Each of the three rivers sampled within this thesis have been subject to a variety of modifications and some of the causes and consequences are highlighted in Figure 7.1. Given that restoration activities have already taken place on the River Wensum and are likely to occur on the other two rivers in the future, the information provided by examination of the palaeochannel sediments and the sub-fossil remains they contain, will clearly serve as a valuable reference condition for post restoration monitoring of the in-stream communities.

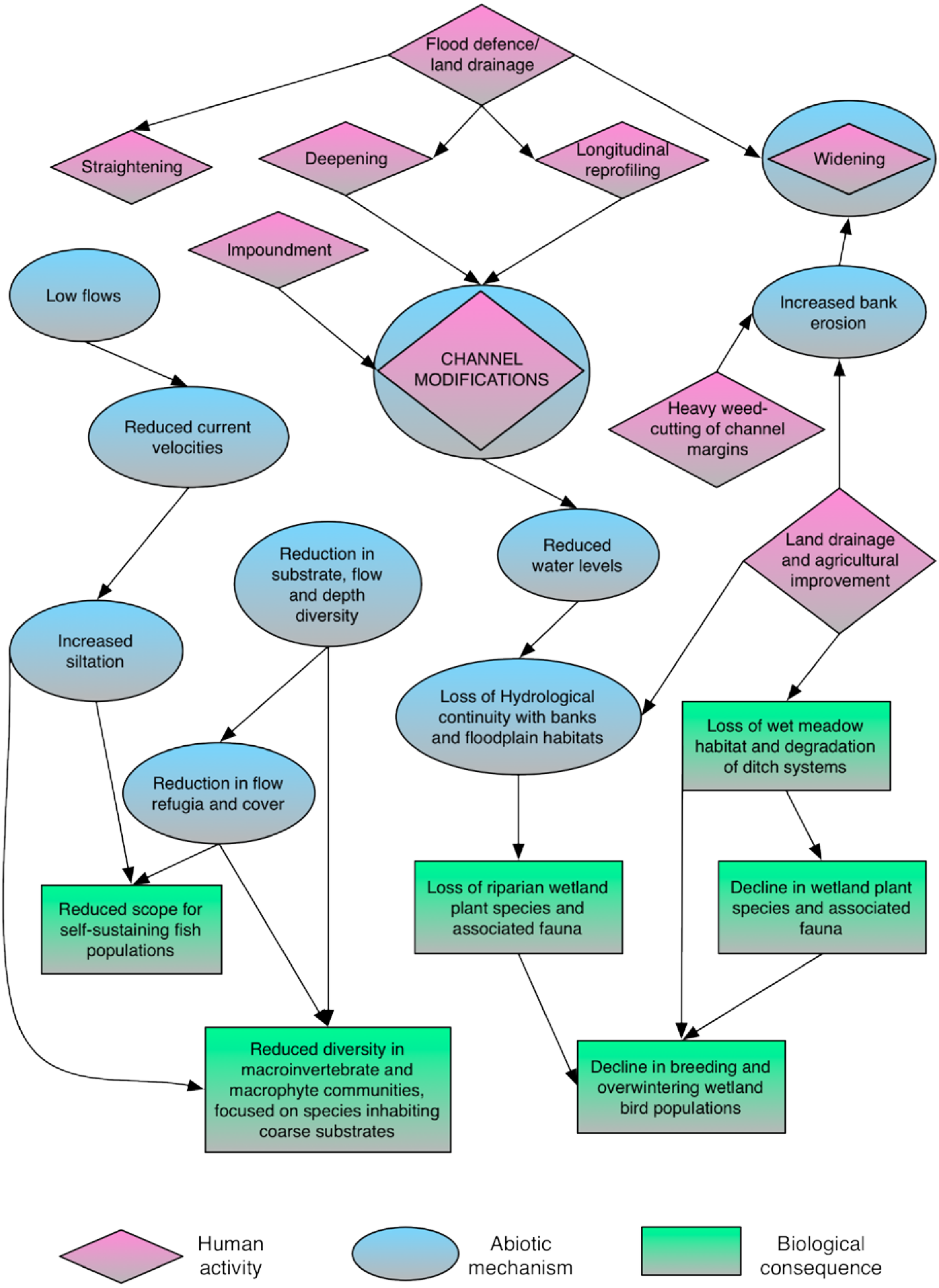

Figure 7.1 Causes and consequence of channel modification in river systems (Mainstone, 1999).

7.2.1 The River Eye SSSI

The heart of the problem currently affecting the River Eye is its surrounding agricultural land use. Changing land use is not shown on historical maps although personal communication with the local landowner (pers. comm. Julia Hawley, 2012) indicates that the majority of the land in the catchment was converted from pasture to the growing of crops with extensive ploughing since 1945. This also coincides with increased application of chemical fertilizers and treatments. These changes are mirrored in many lowland rivers across Western Europe over the past century which have seen dramatic landscape changes associated with land improvement and the intensification of agriculture (Harrison et al., 2004). Pretty *et al.* (2003) believe that the most intense period of modification occurred between 1930 and 1980; when rivers were heavily channelized for the purposes of flood control, land drainage and navigation. These dates coincide with the approximate time of when the River Eye was channelized and dredged. Dredging caused the channel to become wider and deeper than would be natural and was used as a way of alleviating flooding. However, even today the area is prone to flooding (Plate 7.1; which shows the extent of the August 2007 floods).

It is widely acknowledged that intensive farming practices can reduce habitat diversity and quality, and this is especially true for aquatic communities as they are highly susceptible to changes in land use (Barbour & Paul, 2010; Collins et al., 2011; Groves et al., 2012). Due to the farmland in the upstream surrounding catchment the River Eye converting from pasture into arable land, the river is known to have experienced an increased input of nutrients (largely nitrogen and phosphorus) via field runoff, from the increased use of fertilisers and pesticides. These nutrients increase the trophic status of freshwater ecosystems and lead to eutrophication. The bare soils within the River Eye catchment, exposed through ploughing and rotivating have also been more susceptible to erosion than in the past. The eroded fine sediments from agriculture have widespread impacts within rivers through increasing sedimentation rates and modifying instream morphology and communities (Wood and Armitage, 1997).

i.

ii.

Plate 7.1 Photographs (courtesy of Julia Hawley), providing an indication of the extent and severity of the August 2007 flooding; i) shows the closed gates of the flood control structure taken from sampling location 3, ii) view from the earth dam of the flooded pastoral field.

The macroinvertebrates recorded within both the contemporary and the palaeoecological samples of the River Eye support the conclusion that the river's ecology has not changed significantly over the past 50 years. The data supports the hypothesis that the river was already subjected to the effects of anthropogenic activities (intensification of agriculture and channelization) in the mid 1950s. Considering how little the channel morphology appears to have changed over this period, the problems affecting the River Eye may be more associated with long-term nutrient and sediment issues. Habitat availability may also be a limiting factor along the river however this provides an opportunity in terms of restoring the reach upstream of the earth dam/flood control structure, as some instream habitat recreation may be possible. In particular, the low number of riffles downstream of the dam, due to channelization and impoundment caused by a weir near Melton Mowbray, needs addressing. This provides is an opportunity to create a more varied channel morphology, improving flow and physical habitat diversity. Although this will need careful consideration due to the other conservation features within the River Eye (e.g. the presence of the White legged damselfly).

This research has shown that palaeobiology can provides an overview of species-specific requirements and the conditions associated with them. However, in order to understand their associations within the River Eye and its habitats, and make practical recommendations for future restoration, species level of identification is required (Monk et al., 2012). It has been identified that additional measures need to be undertaken to reduce the impact that fine sediment is having within the channel. The erection of fencing to create a buffer strip along the river corridor will help to reduce poaching and bank failure. Gates should also be included within the fences so that limited grazing takes place. The introduction of woody debris into the river system would also help to reduce and regulate sediment transfer by trapping mobile sediment. Further benefits associated with this management measure is its ability to create valuable invertebrate and fish habitats and due to the presence of a healthy macroinvertebrate community in the remaining riffle sites on the river, the potential for successful restoration and rehabilitation of the River Eye is high.

7.2.2 The River Hull SSSI

To evaluate the impact of anthropogenic activities on the headwaters of the River Hull, the contemporary observed community has been compared to other historic datasets including; Whitehead (1935) and Pearson and Jones (1984). Results from the palaeoecological cores suggest that the natural character of the river would be a mosaic of slow- and fast-flowing biotopes, supporting a wider range of species than exists today. The morphology and abiotic factors of the upper reaches of the River Hull in 1972 were reported to be similar to that of other chalk streams such as those in Dorset, for instance the River Frome (Westfield et al., 1972). Pearson and Jones (1984) reported that a range of factors including the presence and abundance of macrophytes, flow velocity patterns, substratum composition, the level of organic pollution through its effects on turbidity and dissolved oxygen levels, and temperature influenced the community composition in 1972. Whitehead (1935) suggested that low substratum stability within the River Hull was primarily responsible for the low number of taxa recorded when compared to other chalk streams, this is also supported by other available evidence (Pearson and Jones, 1984; and the contemporary results). The relative instability of gravel and sandy/silty substratum compared to coarse gravels, cobbles and boulders in headwater streams is largely associated with the low gradients of chalk streams. The stability of substrates in the River Hull could be affected by the channelization process, which has increased hydraulic energy and therefore substrate scour by: 1) shortening the river channel length and thus increasing stream gradient and 2) deepening the channel.

Findings from Whitehead (1935) and Pearson and Jones (1984) suggest that from a few measured physical parameters, the composition of the River Hull macroinvertebrate community could be readily predicted. Such predictive ability is valuable to river managers with respect to the effects of possible changes in the river morphology. When comparing results of Whitehead (1935) and Pearson and Jones (1984) the changes to the invertebrate community in the headwaters appear to be relatively small, however there was more diversity in the past. Examination of the communities within the cores collected suggests that flow velocity patterns may have changed. This is due to the headwaters of the river being an extensive marshland until anthropogenic interventions in the 18th and 19th centuries. The land drainage scheme straightened a previously meandering channel, embanked and channelized it creating a uniformed and dredged channel that was over-deepened (2m below floodplain) for most of its course (Environment Agency, 2003), thus significantly increasing local flow velocities (refer to the historical map - Figure 5.6).

Despite the River Hull Headwaters being designated as a SSSI for its chalk stream ecology and physical characteristics, it has been modified extensively along its length over time for a variety of reasons including land drainage, water supply, flood defence, fish farming and navigation. These anthropogenic changes have all had localized effects upon the ecology of the river and habitats characteristics. The river is now over deepened, straightened, subject to siltation and major structures along its channel impound the river leading to localized siltation. The nature of the modifications are spatially very patchy and significantly influence the changes in the habitat and biota on the river. In order to address these issues, a range of potential management solutions have been identified by Natural England (2009) (Table 7.1). All sampling sites in this study were located in a free-flowing channelized section where flow velocities were relatively uniform and fine sediment accumulation limited. However if sampling had taken place upstream of impounding structures then markedly different results may have been recorded, as they could have included limnophilic and fine sediment burrowing taxa. Nevertheless the short taxon list obtained from the River Hull, reflects the homogenous nature of the study reach. The results of the analysis of the River Hull SSSI demonstrate that anthropogenic channelisation has resulted in the loss of channel length and increased stream gradient and flow velocity. Habitat simplification has also occurred as a result of channel straightening, with the loss of backwaters resulting in homogeneous rheophilic faunal community, potentially at the expense of limnophilic species. In addition, instream refugia (flow and habitat diversity) have been lost, which some rheophilic species may require at some stage in their life cycle (Collins et al., 2011). The dredging and over-deepening of the channel has also resulted in loss of connectivity with the floodplain resulting in the loss of riparian communities and the wetland character of the historic headwaters.

The River Hull Headwaters SSSI is primarily notified for its riverine habitat, therefore restoration measures need to be targeted at restoring ‘natural’ geomorphological processes in ways that support lost habitat heterogeneity upon which characteristically diverse biological assemblages depend. By ensuring the river contains and supports a diversity of biotopes (shallow fast-flowing water, deeper slow-flowing water, different substrate types, exposed tree root systems, marginal and submerged vegetation, shaded and un-shaded areas) characteristic of the river under conditions of low anthropogenic impact generated by natural processes, then this mosaic of biotopes will cater for all components of the characteristic community. Species characteristic of chalk streams include *Ranunculus* spp, trout and grayling, and the life cycle of all these species is supported by a river channel with high levels of biotope heterogeneity.

Table 7.1 Modified summary of the issues identified as affecting the River Hull Headwaters and their potential solutions (Natural England, 2009).

Key Issues	Potential solutions
Fine sediment deposition	• **Measures to prevent sedimentation** Increase morphological and flow diversity by removing or modifying in-channel structures. • **Measures to address sources** Introduce riparian buffer strips through the erection of fencing to reduce bankside poaching and change land use management practices.
Channelisation and disconnection of the river from the floodplain	• **Measures to address over-deepening** Changing the bank profile of the channel will provide a degree of flow diversity if performed on alternate banks. • **Increasing morphological diversity** The reintroduction of historical meanders will increase morphological and flow diversity and the use of woody debris will help increase habitat diversity.
Lack of bankside shelter and over shading	• **Increase riparian vegetation** Planting suitable shrub and tree species in areas where shelter is sparse and the introduction of woody debris will increase bankside shelter. • **Measures to reduce over-shading** Targeted coppicing, pollarding or tree thinning in the upper reaches of the headwaters will increase light into the channel.
In-channel structures (weirs and sluices)	• **Measure to modify key structures** To address this issue, some structures may need to be removed or modified to ease fish passage. The creation of fish passes or additional spawning habitats could result in improved breeding success.

7.2.3 The River Wensum SSSI

The majority of the River Wensum SSSI is currently deemed to be in an unfavourable condition when WFD criteria are applied. This is, in part, associated with high levels of siltation and historic channel modification that has resulted in a loss of natural geomorphological functioning. The high fine sediment input from the catchment is being addressed through Defra's 'Catchment Sensitive Farming' project and changes in the agricultural support system to encourage environmentally sensitive land use practices (Natural England, 2013). However, fine-grained substrates do have their place in natural chalk river systems and provide an important habitat for a range of species that were recorded in the sub-fossil record and contemporary community. Silt naturally accumulates in rivers behind logs and woody debris, however by reducing the silt ingress from catchment sources (e.g. by the introduction riparian buffer strips), this will help to ensure that fine sediment accumulations forms part of the balanced biotope mosaic within the existing channel.

Changes in river management, in addition to physical changes to the channel are required in order to return the river to a more naturally functioning system. Due to a combination of its naturally low hydraulic energy, low contemporary coarse sediment supply and high baseflow from the underlying chalk aquifer, the River Wensum would have a naturally meandering channel form with a relatively high width to depth ratio (fairly shallow), a high occurrence of gravel substrate (i.e. low silt content) and glide habitats (Hiscock et al., 2001; Smith et al., 2003). In natural chalk river systems heterogeneity develops through vegetative processes, such as riparian trees root systems, large woody debris and marginal vegetation encroachment. If their natural meandering planform is un-impacted by anthropogenic activities then this leads to a highly heterogeneous channel, as the work on the River Hull in particular has shown (Chapter 5).

The restoration strategy for the River Wensum focuses on the re-creation of a natural long profile, water depth/level and cross-sectional form with gravel glides and occasional riffle/pool type sequences so as to reinstate the characteristic form and function of the chalk stream (JBA, 2007). The average stream gradient would have been naturally lower through the study section in historical times as river length was greater, but the drop in altitude through the reach would have remained constant. This means that average flow velocities would be lower when compared to the contemporary, deeper channel velocities. However, this potential historic lower flow velocity is likely to have been counteracted by higher spatial variation in flow conditions, with a greater diversity of biotopes including slow-flow and higher flow zones associated with a meandering river geomorphology. In the past, less bank edge definition may have given rise to more extensive marginal transition zones, and therefore more opportunities for wetland species of muddy vegetated margins. Through the channelization process the river channel length would have been dramatically reduced. Therefore the changes reported in the results (Chapter 6) may have been brought about by a loss of slow-flowing silty habitats and limnophilic biotopes at the channel margins, due to habitat simplification caused by channelization. However, the results also demonstrated that through physical restoration habitat heterogeneity could be increased.

During the restoration process the palaeochannel (former meander) was excavated to the depth of the former riverbed which was shallower than most of the current channel. Due to the historic riverbed being used, the river flow eroded the silt and peat covering the natural river-worked gravels thus allowing the channel to benefit from the natural stability of water-worked gravels. By decreasing the depth of the channel, flow velocity was locally increased at reach scale; this was especially noticeable on the riffle at the apex of the meander. The results presented in Chapter 6 indicate that reinstatement of the palaeo channel encouraged limnophilic biota, whilst maintaining the rheophilic biota through this reach of river.

Monitoring post project completion helps to ensure that the objectives of the restorative work have been met and that ecological conditions are improving. By undertaking sampling that encompassed the existing channel, the palaeochannel and the reinstated channel following restoration and reconnection of the meander bend, the past and present conditions could be characterised and recolonisation rates could be monitored. Recolonisation of the reconnected meander did not appear to take place until February 2011, four months after its reattachment. However, six months later (June 2011) the community was largely indistinguishable from that of the original channel. This demonstrates that the river community recovered quickly, probably due to the pre-existing communities upstream and downstream of the reconnected meander in this instance.

In order to secure a long-term future of wildlife along the River Wensum SSSI, it will be necessary to protect the existing wildlife resources by ensuring WFD and SAC objectives are met and maintained. In addition, it is increasingly apparent that the area of wildlife and riverine habitat needs to be greatly increased and re-connected if it is to survive in a human dominated landscape and be allowed to adapt to climate change.

7.3 Defining Reference Conditions

The need to identify ecological 'reference conditions' within freshwater ecosystems is fundamental to river restoration and is explicitly embedded in environmental legislation such as the European Union Water Framework Directive (European Commission, 2000) and the United States Environment Protection Agency guidelines (USEPA, 2006). However, defining reference conditions is far from simple, particularly where little or no historical data is available for a site. Reference conditions in freshwater habitats are typically based on physical and biological monitoring data from 'unimpacted' or 'pristine' sites, from which modeling simulations may be created of reference biological communities (Sandin & Verdonschot, 2006; Hawkins et al., 2010; Nestler et al., 2010). These data typically provide short to medium-term records (typically <20-years) of in-stream community and habitat composition (e.g., Monk *et al.*, 2007). However, the definition of 'reference conditions' should not solely be inferred from contemporary and palaeoecological data (Figure 7.2), there are multiple lines of evidence and data that can be used to underpin the identification of historical characteristics (e.g. multiproxies such as coleoptera, tricoptera and gastropoda).

Despite rivers having been influenced by anthropogenic activities for centuries, this thesis has focused on records of in-stream ecological dynamics within the period following the most significant human impact on riverine systems, such as the industrial revolution (1750-1850) or after the end of World War II (1946). This however, only provides a narrow set of conditions within the long history of each river. 'Reference sites' are typically located in headwater regions (upstream of the most severe human impacts), whilst lowland and somewhat larger sites, which have experienced the most severe deleterious impacts, are poorly represented (Pardo et al., 2012). For some lowland river types, even headwater reference sites may be unavailable due to large-scale land use intensification across whole catchments.

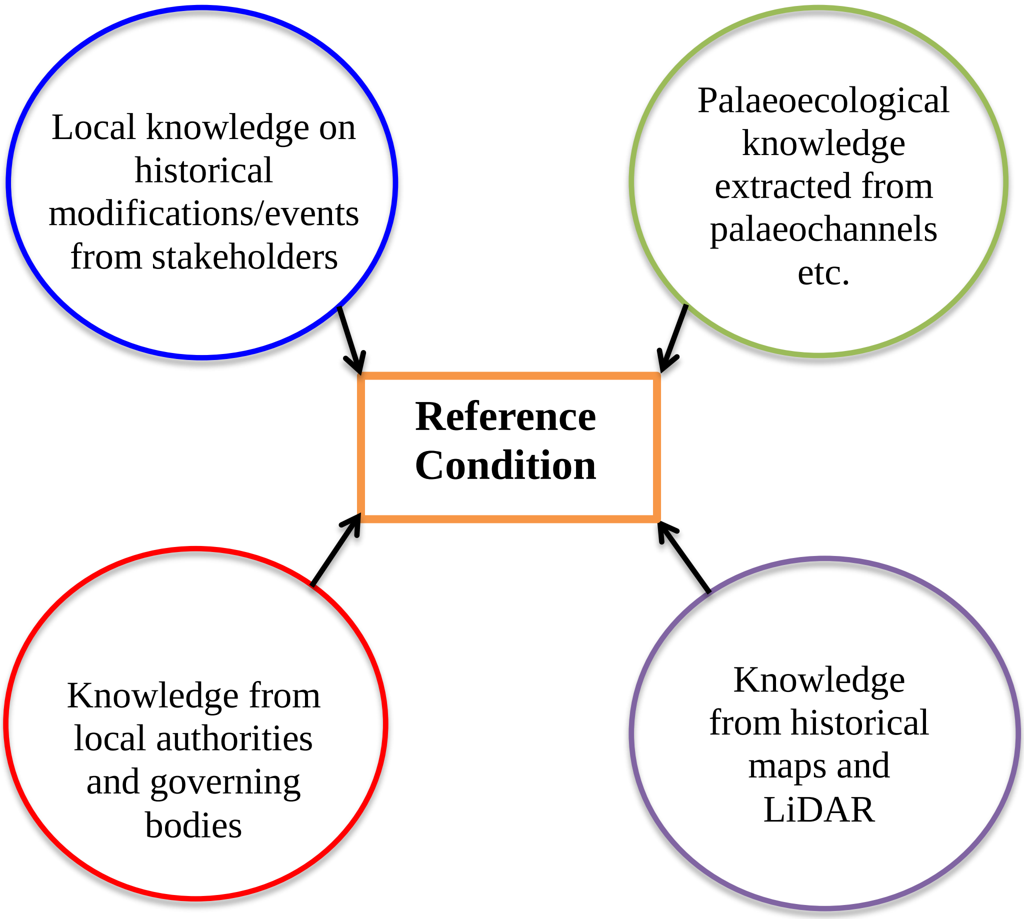

Figure 7.2 Sources of information that could be used to increase accuracy when creating reference conditions.

There are a range of methods available to river scientists to help define reference conditions for biological communities in riverine systems in the UK such as RIVPACS (Wright, 2000) and DARES (Kelly et al., 2008). RIVPACS (River InVertebrate Prediction and Classification System) assesses a rivers quality by predicting the macroinvertebrate fauna expected to be found in the absence of major environmental stress and comparing this with the observed fauna at each site (Wright, 2000). DARES (Diatoms for Assessing River Ecological Status) utilises modern diatom assemblages to determine reference conditions based on comparisons across a number of sites. However, whilst diatoms can be used to determine the level of impact, the ecological and biodiversity significance of changes in TDI (Trophic Diatom Index) are not clear, and while they can be used for assessing the ecological status of rivers, care needs to be taken (Kelly *et al.*, 2008). In addition, neither RIVPACS nor DARES can accurately define a reference condition for a riverine site in the absence of high quality/low impact reference sites of a 'similar' type and output still needs to be tested against a realistic reference point. The concept of 'Shifting Baseline Syndrome' also needs to be taken into consideration (see section 2.6) as false perceptions on the extent of a river systems degradation may result in an underestimation of the ecosystems functions (Pitcher, 2001). As a result, this thesis highlights the need for alternative approaches, which draw on alternative palaeoecological and palaeoenvironmental techniques such as those employed in this thesis. For many lowland rivers, with a long history of management and anthropogenic channel modifications, such as straightening (Figure 7.3), this may be the only viable method available for the identification of historic in-stream biological conditions that may help in the definition of 'reference conditions'.

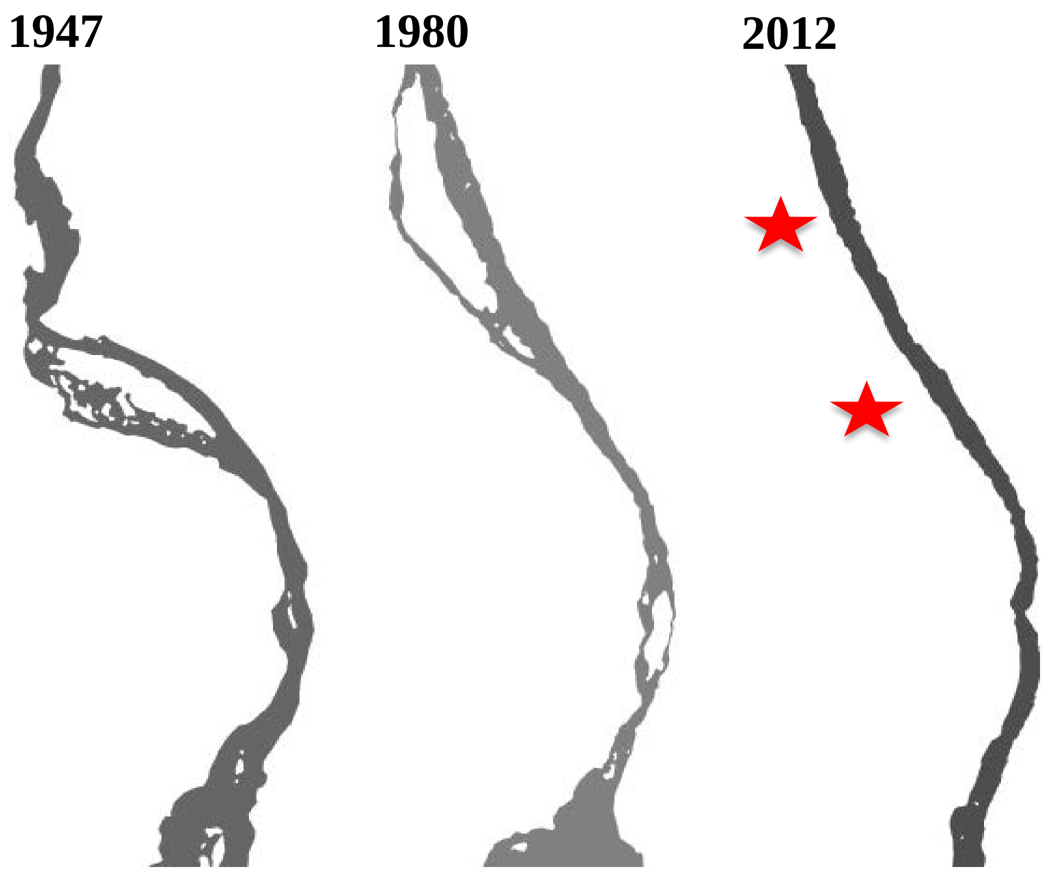

Figure 7.3 Idealized representation of channel change (straightening) typically identified from examination of historic maps or aerial photographs between historic surveys of the schematic river. Stars indicate potential palaeoecological sampling locations, where palaeochannel sediments may be extracted.

Methods such as comparing maps as in Figure 7.2, or evaluating records of engineering works, can provide very important insights into physical reference conditions without resorting to palaeoecology. These methods are embedded in geomorphological evaluation as a precursor to river restoration (at least in the SSSI network in England, managd by Natural England). From these methods, biological communities can be inferred based on an understanding of the likely pattern of biotopes, but palaeoecology provides the only means of testing those inferences.

Palaeoecological techniques are well established and predominantly applied in the study of lakes across the globe and their application to riverine ecosystems has been limited (notable exceptions being: Ogden et al., 2001; Brown, 2002; Smith & Howard, 2004; Greenwood et al., 2006; Davis et al., 2007; Kelly et al., 2008; Howard et al., 2010; Reavie & Edlund, 2010). It has been demonstrated that it is possible to identify lentic and lotic sediment deposits prior to the first evidence of human activity and use these to define 'pristine' or 'reference conditions' in some instances (Bennion et al., 2010). However, this and other research, such as that on 'Shifting Baseline Syndrome' has also clearly recognized that pristine systems are not static and that natural and anthropogenically forced variability occurs over shorter (inter-annual to decadal) and longer (centennial to millennial) time-scales. It is also increasingly acknowledged that a 'pristine' reference condition is not always the most useful, appropriate or achievable in the study of lakes or rivers and that a more pragmatic approach may be required (Hawkins et al., 2010). Bennion *et al.*, (2010) propose the use of a 'reference' concept that incorporates: (1) the extent of degradation recorded contemporaneously compared to a historic state / condition; and (2) the potential for recovery (natural or directly related to restoration) in the absence of human impacts. However following disturbances, multiple recovery pathways may occur resulting in different end points (Figure 7.4 – points A and B) rather than return to the pre-disturbance community. Subsequently the pre-disturbance community may only result by implementing restoration procedures (Milner, 1994). However, depending upon the recovery time-scale, the pre-disturbed community structure may have changed through development and successional processes, in response to habitat changes. Hence, if a pre-disturbed state is required, restoration will need to mimic the presumed successional stage had the system been permitted to continue with normal community processes (Cairns, 1990), which is a difficult scenario for river managers to achieve. As a result it may be more appropriate to consider multiple potential reference conditions rather than a single static target for river restoration.

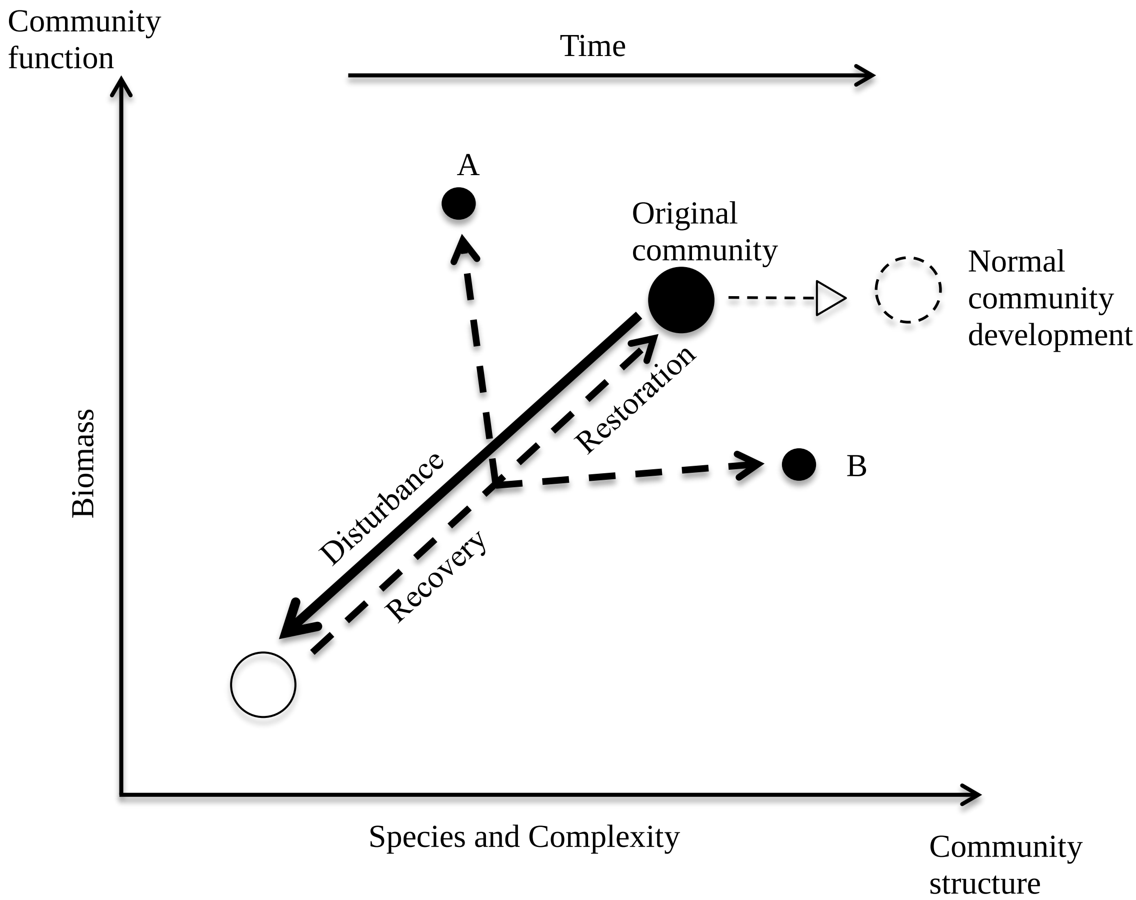

Figure 7.4 Conceptual model of the multiple recovery pathways of a disturbed community, indicating the possibility of the creation of multiple reference conditions (adapted from Bradshaw, 1988).

It has been demonstrated that reference conditions can be determined in a number of ways (Hawkins et al., 2010). However in order to ensure that river restoration is successful, the reference conditions established need to be sustainable to achieve the objectives set. Due to rivers being naturally variable ecosystems, understanding the range of natural variability and being able to identify natural thresholds and when they are exceeded is important for the management of ecosystem health (Saunders & Taffs, 2009). Therefore, this is essential when defining reference conditions. Palaeoecological data integrates information from surrounding habitats and up-stream biotopes through the use of multiproxies (not solely the past condition at the point of sampling), thus providing an insight into the mosaic of biotopes within the riverine environment present at the time of flow, and directly after palaeochannel cutoff. Thus, reference conditions that are based on available records and draw upon palaeoecological techniques, may serve as a management strategy for riverine floral and faunal communities. It is important to realise that a river's capacity for self repair is preserved and minimal external support (apart from monitoring) is received following restoration wherever possible (Verdonschot, 2000). However in order to guarantee this self-repairing capability, there needs to be habitat available and riverine processes capable of maintaining the physical conditions. Physical restoration of rivers should be driven by natural processes since this provides the most sustainable and naturally dynamic pattern of biotopes; it should also be the least interventionist and most cost-effective. However, this desirable approach rarely happens as large mechanical diggers usually 'engineer' the river and depending on the scale of the river, this may not be the most environmentally sensitive approach.

The research in this thesis has highlighted that the methods used to define reference conditions for each of the three case studies are restrictive. Strategic site selection was critical for the success of this research as the methods used are currently limited to rivers where there is evidence of palaeochannels or instream islands. The collection of cores from palaeoechannels can increase our understanding of river channel evolution and successional patterns (Greenwood et al., 2003) but it only provides a relatively short temporal snap shot of these past environments. Figure 7.5 shows how palaeochannel deposits can be used in the identification of the timing of channel cut-off or isolation following anthropogenic diversion through the change in the macroinvertebrate community structure. In these idealized results, the 1945 transition from lotic to lentic environments can be easily observed by the change in fauna, clearly pinpointing when anthropogenic intervention (channel cut off) occurred.

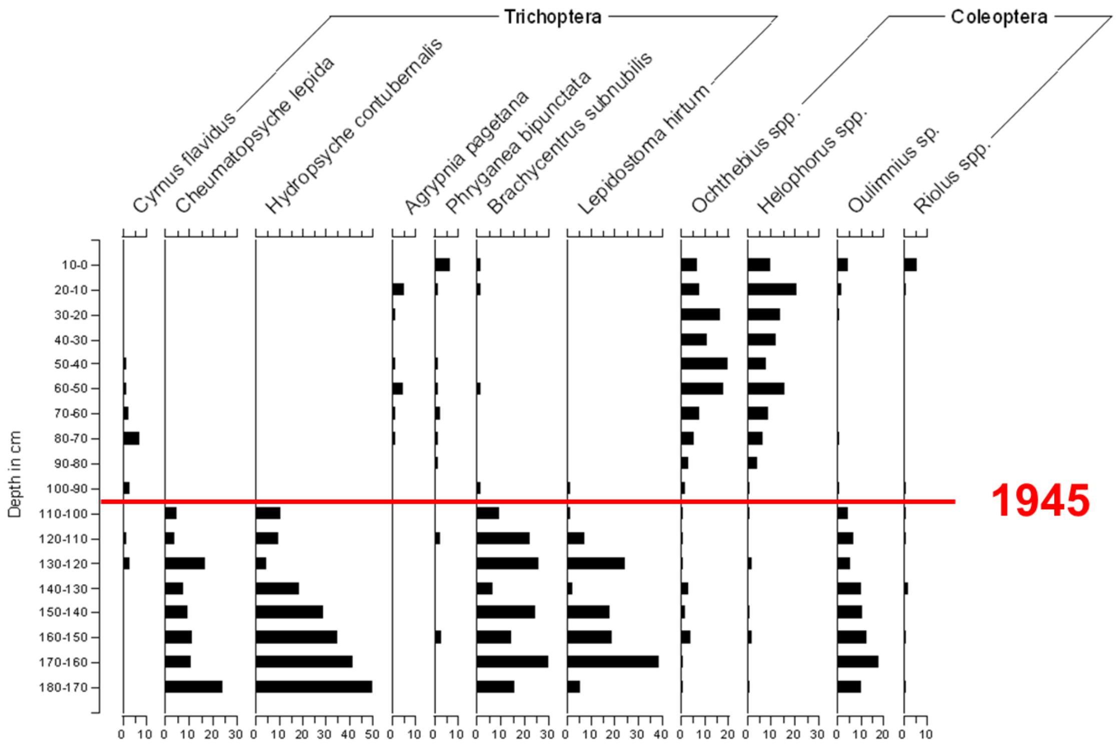

Figure 7.5 Vertical distribution of sub-fossil Trichoptera and Coleoptera from an idealized palaeochannel deposit indicating the transition in community structure from a predominantly lotic to lentic community (between 110-100cm in depth) resulting from channel cut-off (after Howard, 2007).

7.4 The Issue of Scale and Bringing Landscape Perspectives to Restoration Schemes

Despite the considerable interest and activity centred on reach-scale river channel restoration activities in the UK, there has been a growing recognition that this needs to be set within a coherent integrated catchment scale approach to managing rivers and their floodplains. Rivers cannot be separated from the land they drain, due to the connectivity of the rivers (longitudinal, transverse and vertical dimensions) across the floodplain. Rivers are hierarchical systems across a range of spatial and temporal scales from microhabitats to the entire stream network (Benda et al., 2004) (see Figure 7.6). River systems and their instream communities vary over time due to flow regime variability (e.g. floods and droughts), and variations in the organisms associated with them (e.g. size and mobility of taxa - Monk et al., 2007). Therefore river management and restoration strategies require appropriate linkages between spatio-temporal scales with their relevant ecological processes (Thoms & Parsons, 2003).

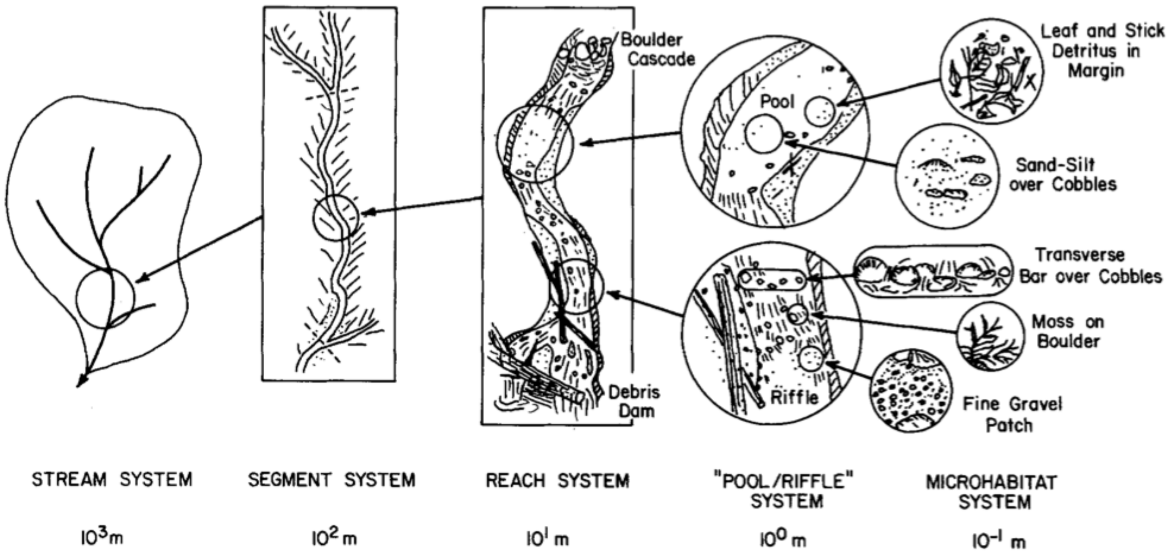

Figure 7.6 Stream hierarchal classification and associated habitat systems from Frissell et al. (1986).

Contemporary instream habitat structure is primarily determined by local conditions, such as flow velocity hydraulics, whereas hydrology and channel characteristics are influenced by regional conditions, including landscape features some distance upstream and lateral to stream sites (Thorp et al., 2006). It is widely recognized that landscape and catchment characteristics have a significant influence on in-stream communities and that the climate, topography, geology and vegetation within the catchment drive the geomorphic processes (Sheldon et al., 2012). Catchment features influence the water chemistry, hydrology and ecosystem processes that account for a significant proportion of the ecological variation within individual rivers (Sponseller et al., 2001). Previous research exploring relationships between in-stream ecological and physical variables have demonstrated that the spatial scale significantly influences the importance of environmental variables at each level. For example Gray & Harding (2011) demonstrated that communities recorded from all habitat types were influenced by catchment scale hydrology with the main channel communities being equally influenced by catchment and habitat scale characteristics and headwater spring communities being primarily regulated by habitat level characteristics. Gray & Harding (2011) concluded that even though there was considerable cross-scale correlation regarding the influence of environmental variables it was highly suggestive of complex hierarchical controls. Similar scale related results have been reported in other locations across the globe (Frissell et al., 1986; Poff et al., 1997; Weigel et al., 2003).

With such strong linkages between catchments and streams, the catchment boundary should define the spatial extent of the riverine ecosystem instead of the focus being concentrated solely on the river channel itself. However, the scales at which river management is applied are often different from the scales at which the ecological information that informs the management schemes are collected. For instance management schemes could be based on information from gauging stations that are not positioned on the affected reach. Therefore it is necessary to consider the effects of up scaling ecological information gathered from river reaches for restoration and management purposes. The problems associated with scaling are the result of the mismatch between the scales of observation (e.g. ecological, hydrological and morphological) to the characteristics of the wider environment (Frissell et al., 1986; Cooper et al.,1998). The spatial and temporal variations experienced and recorded in rivers may not be the same and this results in high levels of heterogeneity (Frissell et al., 1986; Sponseller et al., 2001; Thoms & Parsons, 2003; Parsons et al., 2004). There is therefore a need to ensure that the data collected, is at the appropriate spatial and temporal resolution (Biggs et al., 2005), such as those used within this thesis. This will allow results of restoration projects to be placed in appropriate catchment/river/reach/site scale contexts to provide a more holistic overview of river restoration activities on which their potential success can then be gauged. With regards to the positioning of palaeoecological sampling within this hierarchy, the historical conditions defined need to be related to the spatial scale of evolution in order to inform the definition of reference conditions and a restoration vision.

7.5 Monitoring Restoration Schemes

Post project appraisal and monitoring is a very important aspect of any restoration procedure, as it seeks to determine if measures undertaken have been effective and successful. In this sense the process of river restoration should be no different. Monitoring a river restoration project provides a number of important benefits such as learning from experience and for regulatory purposes (England et al., 2008). By setting measureable objectives at the start of the scheme it makes it possible for an appropriate post project monitoring strategy to be developed. Lessons learnt from both failures and successes of projects are vital for adaptive management with regard to river restoration and are valuable in optimising future schemes (Woolsey et al., 2007).

Historically, monitoring of physical river restoration has focused on changes in physical habitat, vegetation and instream restoration techniques, such as log structures and boulders with inadequate attention paid to techniques used in restoring basic watershed and ecosystem processes (Roni, 2005). Biotopes can be constructed or allowed to develop through natural processes, and in both cases monitoring tends to focus on changes in physical habitat structure (e.g. biotope types and distribution), as the direct result of the restoration. To ensure that restoration is undertaken incorporating natural processes wherever possible, monitoring regimes should be put in place and tied to the natural regime of the river. Despite the large financial investment in restoration of aquatic ecosystems in recent decades, only a small fraction of the funds are allocated to the research and evaluation of the project effectiveness after the engineering work has been completed. Poorly designed restoration and monitoring programmes may ultimately be costly in terms of negative impacts to the ecosystem and in terms of financial resources allocated to ineffective monitoring (Downs & Kondolf, 2002).

The benefits of restoration monitoring are slowly being recognised. For instance as part of the EU WFD regulatory bodies have defined monitoring requirements within their protocols. The directives requirements come in three forms; 1) surveillance, 2) operational and 3) investigative:

1. Surveillance monitoring is linked to characterisation and risk assessment.
2. Operational monitoring is used when classifying water bodies that show a risk of failing to reach good ecological status
3. Investigative monitoring aims to determine why a river is not meeting the set ‘good ecological status’. It is also used to identify problems which may have previously been undetected within river reaches (UK Technical Advisory Group, 2005).

The monitoring framework outlined in the WFD addresses broad characterisation of ecological status and activities associated with problem detection. However, it doesn’t deal with the issue of pre- and post-project appraisal and the surveillance monitoring put in place to monitor ecological status is not geared towards detecting physical impacts. These are often spatially very patchy and result in changes in the spatial extent of different biotopes to which WFD monitoring tools are not sensitive. The need for post project appraisal has been proposed by a number of scholars and is now widely accepted by academics and practitioners (Boon, 2000; Bond & Lake, 2003; Buchananet et al., 2012). Effective monitoring requires an understanding of the temporal and spatial scales at which the restoration measures are aimed, the nature of both the restoration actions and the response within the ecosystem, and historic and current conditions (Downes et al., 2002). However despite this, detailed monitoring of river restoration schemes has rarely been undertaken in the past, with less than 10% of restoration projects reported in literature being subjected to post project evaluation (Convertino et al., 2013). Among the few that have been objectively evaluated, monitoring usually occurred over a short time scale and thus did not consider lag effects on morphology or habitats due to high magnitude flow events (Bond and Lake, 2003). Very few restoration projects to date have significantly contributed towards our post restoration knowledge about whether or not restoration scheme outcomes have achieved their aims and objectives. Learning from past experiences will help to improve future project designs, increasing success and help ensure restoration measures are sustainable.

Attitudes towards river restoration has changed over the recent years due to the realization of how serious the state of decay of some rivers is (Lake, 2001; Naiman et al., 2002; Buijs, 2009). However, without being able to demonstrate river restoration success, there is a great risk that the current level of public support will decline, particularly when large investments of money are required (Woolsey et al., 2007). Nevertheless, monitoring is made more difficult due to the fact that there are still considerable debates over which are the most suitable methods to use when assessing the ecological health of rivers (Bunn et al., 2010). Traditionally monitoring has relied on water quality methods coupled with qualitative sampling of biotic communities, particularly benthic macroinvertebrates and macrophytes. The key challenge for restoration monitoring is to develop cost effective ecosystem health monitoring programs that can be used prior to, during and post restoration which takes into account a wide range of ecological, physical, chemical and geological parameters (Jungwirth et al., 2002; Convertino et al., 2013).

A well designed monitoring and evaluation programme is a critical component of river restoration, with broad catchment scale monitoring determining the success of overall restoration schemes to project/site level monitoring assessing whether site-specific actions have been successful. Monitoring helps to reduce uncertainties surrounding the effects of management decisions on ecosystem dynamics and recovery (Roni et al., 2005; Harris & Heathwaite, 2011). Roni et al. (2005) indicated that before initiating a study to evaluate restoration actions, the overall goals of the project and the objectives of the monitoring program must be clearly laid out to help gauge its success (Figure 7.6). Without these well-accepted criteria there is little incentive for practitioners to assess and report restoration outcomes, especially given budgetary constraints on time and resources that usually exist (Plummer, 2005; Song & Frostell, 2012). This impediment needs to be addressed in order for restoration ecology to grow as a respected science. Monitoring of river restoration projects requires an understanding of the controls, such as climate and geology, and the processes such as flow regime dynamics and sediment load, that shape and influence riverine ecosystems. Therefore to provide a complete overview, both physical and biological parameters need to be considered when conducting a monitoring program.

Physical monitoring concentrates on the pattern and shape of river channels through the study of morphology (Pess et al., 2005). Channel morphology should consider channel slope, width, depth, sinuosity, in channel features (e.g. pools, riffles and channel bars) and the degree of connection the channel has with the surrounding floodplain. Through monitoring erosion rates the transport and deposition of sediments can be assessed. This will help to determine present and future channel stability and help inform both pre and post restoration programs (Bunn et al., 2010). Biological monitoring involves using species assemblages to provide information about ecosystem structure and functioning, together with floodplain connectivity. This form of monitoring has been conducted less frequently and has resulted in inconsistent results, depending upon the duration, technique, species, region and life stage being examined (Roni et al., 2005).

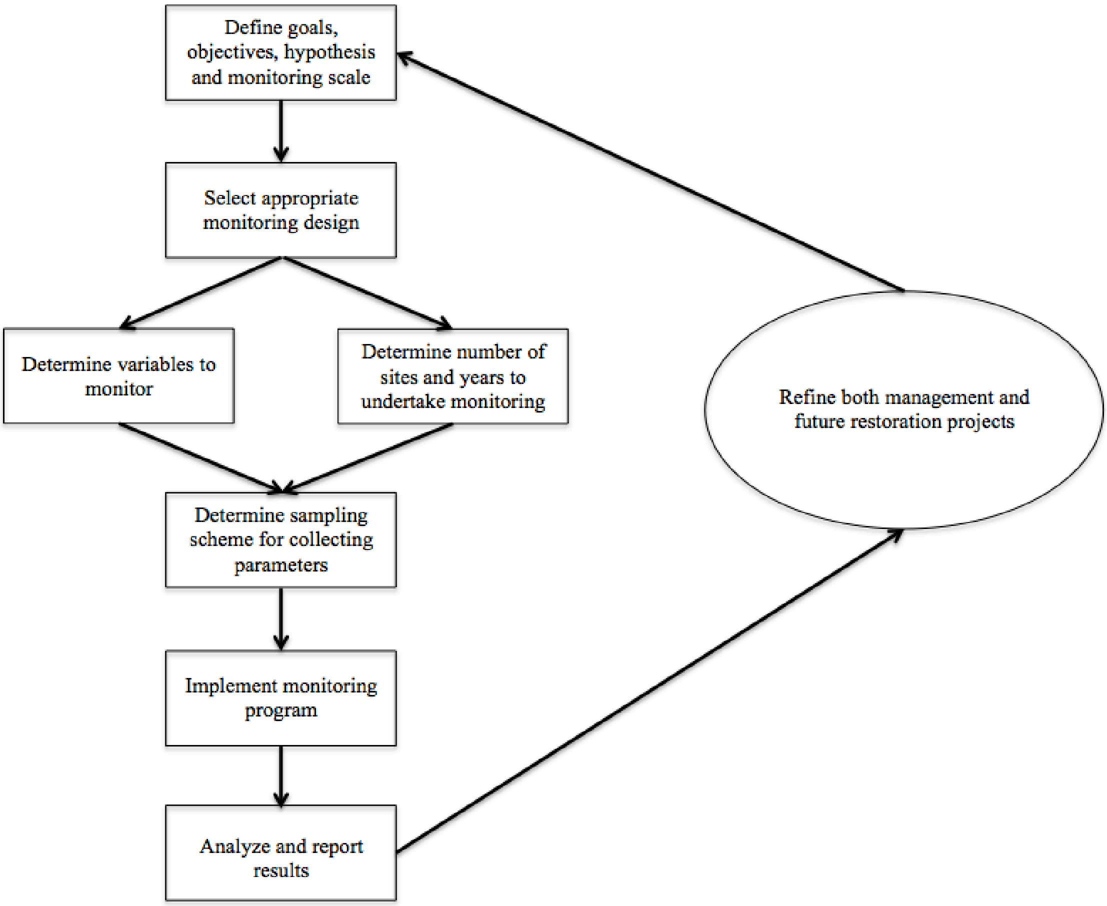

Figure 7.6 Diagram adapted from Roni et al. (2005) depicting the basic steps for creating a monitoring and evaluation program from river restoration.

Whilst the evidence base relating to river restoration success or failure is growing, post-project appraisals that examine river dynamics linked to ecological functioning remain rare. It has been recognized that there is a lack of appropriate frameworks to guide and define what level of appraisal is appropriate for different restoration techniques (Dollar et al., 2007; Vaughan et al., 2007). Woolsey et al. (2007) have taken steps to provide guidance on selecting indicators focusing on river restoration success. They highlight the practicalities of the monitoring process and categorise key indicators required such as collection methods and the associated financial and time constraints. Steps need to be taken towards the development of post project appraisal guidelines that are grounded on realistic objectives and goals. If planning is undertaken based on sound ecological principles then the chances of making the wrong restoration decisions are greatly reduced. This will only prove successful if it is underpinned by a 'top down' approach where government agencies are in a position to deliver appropriate funding and policies that help inform project managers about the most appropriate actions to take (Clifford et al., 2006). However, both ‘top down’ and ‘bottom up’ approaches operate and getting them to work together for the benefit of river restoration is a large problem.

7.6 A Critique of the Benefits of Palaeoecological Techniques to River Restoration

The methods used in this thesis are relatively inexpensive, quick and an efficient way of identifying reference conditions by drawing on a range of proxies and temporal scales. However in order for this to be beneficial, the identification of reference conditions needs to be integrated into the ecological/geomorphological appraisal at the start of the SSSI river restoration planning process; before river restoration plans are finalised. For instance with regard to the River Hull, restoring the river to the conditions identified from the headwater palaeochannel, the riverine environment would be markedly different to the headwaters typically described for a chalk-stream and what it is designated for as a SSSI. River restoration comes down to a site-specific restoration vision that is an outcome of the restoration planning process, built upon a picture of historic changes and their effects based on a site-specific analysis of reference conditions. River managers and society have to decide between what is best for the river and what is most desirable, not just in terms of government policies but also in the eyes of the public and the people that the river influences e.g. farmers and anglers. The SSSI river restoration planning process however, has a technical evaluation, methods for the development of a vision, a stakeholder process, and the constraints assessment (Mainstone, 2006). This guidance is designed to go through these issues and help bring the restoration vision to life for stakeholders involved in participation process. For instance, if the River Hull were to be restored to the ‘natural’ historic condition, the headwaters would need to revert back to a marshland environment. This would change the dynamics of the floodplains by decreasing the value of agricultural land and impacting upon the current fish and invertebrate populations. These restoration activities could be a source of resistance and conflict from local communities and stakeholders as river corridors are significant to peoples local living space and everyday environment (Junker et al., 2007). Junker *et al*. (2007) found that people strongly relate to rivers, attaching importance to them not only as recreational and natural spaces, but also as landscapes associated with local identity. The cultural value that local populations place on rivers is high, therefore adopting a purely scientific approach to river management and restoration strategies could lead to alienation of, and resentment by, those directly affected by the changes implemented (Maynard, 2013). However, river restoration has been increasingly understood not only as a process of improving and modifying riverine environments, but one which captures the interactions between science and societal goals (McDonald et al., 2004; Petts, 2007; Sear, 1994). Public engagement and participation in environmental decision-making is now widely supported by policy makers and has been institutionalised at the international level in the drive for ‘sustainable development’ (Cook et al., 2012). Although the SSSI river restoration planning process is based on conservation goals as a statutory requirement, it does have elements that try and reconcile those goals with stakeholder needs and ambitions in ways that can enhance the river for everyone.

The WFD encourages Member States to engage all interested parties as part of the implementation of the Directive, thus involving stakeholders and the public in planning and co-decision on factors that could impact upon their livelihoods and surrounding environment (Henriksen et al., 2009). This is also true of the Habitats Directive and the Countryside and Rights of Way (CRoW) Act (2000), which help to underpin SSSI management and are predicated on finding mutually agreeable (to Natural England and the owner-occupier) management regimes typically embedded in management agreements. Guidance provided on public participation by the WFD (EC, 2003) can be categorised into four types;

1. Information provision (including management timetables, issues and the participants, which is considered the foundation of further participation activities).
2. Consultation (this encourages written and oral responses).
3. Engagement (actively involving people in “developing and implementing plans” that could form the final restoration plans decided upon).
4. Co-decision making (helping to make the final decision about which restoration plan to implement and taking responsibility for the decision).

Despite these requirements, there is ambiguity around the exact way participation should be approached. For example, consultation and information provision are obligatory, while higher degrees of participation such as ‘co-decision making’ are optional (Henriksen et al., 2009). Even though public participation is not legally required, the benefits it brings may be essential to reaching the environmental goals set by the WFD (Henriksen et al., 2009). However, the objectives that are driving restoration of the case study rivers used within this thesis are the SSSI and SAC conservation objectives, which go further than ecological status objectives and can therefore lead to decisions being made in different ways.

Despite the growing calls for public participation, it has been suggested that truly deliberative public engagement is still the exception rather than the rule (Petts, 2007). Even though it is perceived that the involvement of the public and stakeholders helps to decrease conflicts in river restoration planning and implementation, river managers tend to perceive the risks rather than the potential benefits. Therefore, river managers may opt for the ‘easy’ approach by just drawing on expert understanding of restoration objectives and the constraints associated to achieving those objectives (Pahl-Wostl et al., 2011). More involvement in decision making is frequently perceived as too complicated and expensive, and there is a concern that the public will make ill informed and therefore bad decisions (Mostert, 2003). For instance it has been found that local stakeholders can bring heavy conservatism to change (even though the current state of the river was brought about by massive change of the type they would have resisted if those stakeholders had been born in an earlier age). They can also bring perspectives that, if acted upon, would result in a continuation of existing damage and even exacerbation of damage to freshwater habitats for which there are statutory obligations to protect (Maynard, 2013). Furthermore, stakeholder perspectives are often based on self-interest without due regard for societal need. Flood management is a good example where farmers upstream of urban areas want/need their land defended via dredging and flood banks alongside oversized and straightened channels, even though the social imperative is to alleviate flooding downstream. Furthermore, project managers often believe that they already know local needs and interests and therefore can best represent them (Junker et al., 2007). However ,if involvement is restricted to small circles of influential stakeholder groups then stakeholder interests (e.g. landowners, farmers and anglers), that may be opposed to restoration aims, could be over represented. Therefore involving the local public could weaken any potential resistance to restoration activities, thus leading to more realistic restoration solutions that are based on a more representative range of interests (Luyet et al., 2012).

The role of public participation and its associated advantages within river management and restoration is increasingly recognized as important (Lane et al., 2011).

Comprehensive public participation makes it possible to highlight public concerns and values and use local knowledge to better inform more creative decision-making (Le Lay et al., 2013). Involving the wider public has enhanced social objectives, such as trust building, identification of people with their local environment and conflict reduction (Luyet et al., 2012). Stakeholders are an essential part of river restoration planning but their input should be injected at optimal points in the planning process. A starting vision for restoration needs to be provided based on statutory objectives with a clear and accessible rationale, highlighting wider societal benefits. Stakeholder perspectives on this vision have to be heard and acted upon to the extent that they are reasonable, and efforts should be made to identify restoration measures that meet restoration objectives whilst enhancing societal benefits from the river. It is important to engage a holistic, integrative environment for river restoration planning, drawing on all pools of knowledge from scientific to local cultural values. The wants and needs of local governing bodies and the public may differ and discussions about restoring rivers to a 'natural' state may have little meaning between these relationships and the ones between humans and the natural environment. But by fully understanding all the drivers behind restoration projects it will be possible to draw upon the full potential of river restoration projects and what can be realistically achieved sustainably, will become clearer.

7.7 Future Protection and Adaptation of Rivers

Rivers by their very nature are dynamic systems, constantly adjusting to changes in sediment and flow (Lytle & Poff, 2004). However, the changes resulting from climate change may occur much quicker than naturally occurring historical trends (Wilby et al., 2010). Predictions for the UK suggest considerable reductions in summer flows and significant increases in winter flows (Mainstone et al., 2012) and combined with shifts in temperature regimes this could have major implications for riverine biodiversity (Graham & Harrod, 2009). Therefore during implementation of river restoration schemes climate change future proofing needs to be taken into consideration. Proactive management is essential in aquatic environments to protect existing resources, but it should also be recognized that its effectiveness is dependent on existing monitoring programmes and modeling capabilities (Palmer et al., 2009). Unfortunately, even the WFD does not explicitly mention the risks posed by climate change in its environmental objectives

(Wilby et al., 2006). This is notwithstanding the fact that the time scale for achieving a number of the WFD objectives, extends into the 2020s, when it is predicted that average temperatures and precipitation rates may have changed (Wilby, 2004). However, the SSSI and SAC conservation objectives for rivers are designed to maximize resilience to climate change and allow for climate change adaptation. Restoring natural processes in rivers is the key adaptation challenge for riverine ecosystems, and is also central to future flood risk management strategies. Climate change has the potential to impact WFD decision-making processes through additional pressures on rivers associated with changing flow rates and thermal regimes (Irvine, 2004; Webb et al., 2008). Therefore, one of the largest policy challenges is how best to implement decision making in the face of long term climate variability and change (Wilby et al., 2006). In order to address this, improvements need to be made in the understanding of the natural variability of ecosystems and their process responses.

There is great uncertainty surrounding local climate change projections and the associated impacts it might have on freshwater ecosystems. Therefore there is a strong case for devising adaptation management strategies that produce benefits to riverine environments regardless of the climate outlook (Wilby et al., 2010). Two examples of popular adaptation techniques that are classified as 'soft', reversible and no regret solutions that are also valuable in terms of river restoration are increasing shading of vulnerable river reaches and the concept of hands off flows (Nel et al., 2009; Bowler et al., 2012). Increasing the shading of river reaches through the planting of bank side trees will help aid in the reduction of water temperature (Hallegate, 2009). Due to river temperature being the master water quality variable that affects physical, chemical and biological processes, higher water temperatures could potentially affect macroinvertebrate species distributions and abundances through changes in metabolic rates, feeding and migration patterns (Malcolm et al., 2008; Kaushal et al., 2010). Planting trees and restoring riparian vegetation can increase shade and help to counter rising stream temperatures by creating thermal refugia (Wilby et al., 2010), however this may also increase woody debris which in some cases could be considered to have detrimental effects due to the increase in channel roughness and the risk of exacerbating local flooding (Thomas & Nisbet, 2007; Wilby & Wood, 2012). According to Hallegate (2009) this adaptation measure is easily reversible if it shows to be unnecessary, ineffective or harmful, therefore shading should be monitored and assessed on the long term effects it has on managed riverine ecosystems with respect to catchment wide changes and hydrological regimes.

'Set hands off flows' is an adaptation that protects riverine ecosystems by halting potentially harmful water abstraction during flow low episodes because when flow falls below a critical threshold ecosystems can become damaged (Fung et al., 2009). Variations in river flow may have indirect consequences through reduced dilution of nutrients, organic contaminants and disease agents found within the water (Wilby, 2004). However Poole & Berman (2001) believe that rising water temperature due to catchment wide climatic or non-climatic pressures may pose greater threats to the ecology of rivers than an altered flow regime. Either way river managers are being confronted by the choice of trying to build resistance or resilience to climate change (Hansen et al., 2003).

Both SSSI/SAC conservation objectives and WFD ecological status objectives are based on constraining the level of anthropogenic impact on natural habitats, operating under natural processes. However there is no legal requirement of river managers to resist climate change. Conservation objectives have expended a great deal of effort in trying to ensure that objectives focus on relieving catchment management pressures in a changing climate. However research by Dunbar *et al.* (2010) shows that less modified channels with higher quality freshwaters offered a greater diversity of habitats and refugia for fauna during extreme high and low flows. This suggests that more 'natural' channels may have higher resilience to climate change. Due to the large uncertainties surrounding the impacts of climate change on rivers and the associated adaptation techniques, the protection and enhancement of riverine ecosystems should involve restoration of natural processes and low regret techniques that have multiple benefits, example being the planting of bank side trees that provide shade, alternative habitat, act to diffuse pollution and decrease bank erosion (Harrison et al., 2008). Monitoring of implemented adaptations should become part of routine programs to determine if hypothesized benefits materialize or to determine if further restoration/remediation measures are required.

7.8 Summary

The relevance of these research findings to the field of restoration science has been brought together in this chapter and the advances made have been outlined. The integration of contemporary and palaeo-riverine ecologies has provided a method to explore a river's current and past status. The results from each of the three case studies show how the macroinverterberate communities have changed and demonstrates how an understanding of riverine palaeoecology across spatial and temporal scales could provide an important role in the future identification of reference conditions for river restoration and management strategies.

By accurately characterising how biotopes have changed, and the potential reasons for this, it will provide relevant information to help towards ensuring river restoration success. These insights will help restoration managers understand the eventual effects of restoration schemes, such as the meander reconnection on the River Wensum. The benefits of identifying historic reference conditions include the creation of realistic aims and objectives for river restoration. However these methods do require a considerable amount of time, effort and expertise and unfortunately will not work everywhere; only rivers with palaeoecological sources such as palaeochannels. Restoration schemes also need to take into account catchment scale to identify and prioritize the most influential stressors affecting river ecology. By basing restoration schemes on catchment scale analysis, appropriate and sustainable measures (e.g. buffer strips) for high priority reaches can be identified. Post-restoration monitoring is essential for future restoration schemes and the successful communication of restoration results will benefit other river managers. To enhance the probability of achieving desired ecological and morphological outcomes, a long-term vision and linkage to a catchment context is required. This coupled with the encouragement of public participation in the creation of restoration schemes will not only decrease any friction present but also provide insight into local knowledge on past riverine environments. The integration of these disciplines will prove essential in increasing our understanding of reference conditions. It is another tool that can be drawn upon to provides a series of opportunities that will assist river managers and organizations in the development of river restoration programmes in the fulfillment of the WFD's, the SSSI's and SAC's conservation objectives.

CHAPTER EIGHT: SUMMARY AND FUTURE RESEARCH PLANS

8.1 Introduction

This thesis has sought to characterise 'reference conditions' for river restoration purposes through the comparison of contemporary and palaeoecological riverine macroinvertebrate communities by employing a multi-proxy investigation of three lowland SSSI rivers in England. This chapter summarises the main findings of the research based on the original aims and objectives underlying it and concludes with suggestions for future research within the field of restoration ecology and palaeoecology for applied river management, conservation and monitoring purposes.

The results of this research indicates that the methodology employed enables the identification of 'reference' communities with regards to its composition, providing evidence of taxa that have historically been extirpated and also identifies more recent colonisers. This information will potentially help agencies (e.g. Natural England and the Environment Agency) engaged in managing contemporary river systems for planning future conservation and restoration activities by directly supporting ongoing research and management efforts centered on the physical and ecological restoration of river SSSI sites.

The research in this thesis has demonstrated that, historic conditions can be identified to aid in the process of defining 'reference conditions'. However, it is concluded that the sites (selected river reaches) used in this study were specifically selected for investigation, based on the presence of either a palaeochannel or instream deposits. The availability of sites may be more limited in other lowland rivers; although the use of airborne remotely sensed data captured by a combination of LiDAR and aerial photography can be used to help identify the presence of former channel deposits (Howard, 2005). These techniques may help identify former channel courses, aid restoration planning and reconstruction of river floodplains (Howard et al., 2008). However realistically, research of this nature presented in this thesis, may only be feasible at sites with higher-level conservation designations (SSSI, SAC and DTC) in order to ensure they meet WFD or other specific designations.

8.2 Fulfillment of Aims and Objectives

The principle aim of this thesis was to explore whether characterising the historic ecological communities and in-stream conditions within three lowland rivers through the combined analysis of contemporary river ecology and palaeoecological techniques could enable the identification of reference conditions for river restoration planning purposes. The thesis aims identified in Chapter 1.2 have been fulfilled through the completion of the outlined objectives;

6. Examine the national and international drivers for contemporary river management, conservation and restoration (Chapter 2).
7. Examine the use of sub-fossil macroinvertebrates in riverine palaeoecological analysis in relation to the identification of historic reference conditions, through detailed examination of the literature (Chapter 2). The potential use of multiproxy methods were explored (Chapter 2) and put into practice (Chapters 3, 4, 5 and 6).

Detailed examination of recent publications provided an insight into the current government policies driving river restoration and the science used to inform them (Chapter 2). Dialogue with Natural England, the Environment Agency, landowners and end users allowed for a more varied insight into government policies and their influence. Various forms and methods of river restoration were researched and what constitutes 'ecologically successful' river restoration was discussed. The proxies subsequently used in the detailed research (Coleoptera, Trichoptera and Gastropoda) were examined and their benefits as indicators of past riverine environments were presented. The advantages of the multiproxy approach and how it provides clear reinforcement of lines of evidence from different environmental signals to provide a wider perspective was discussed.

8. Identify study sites from a list of potential rivers (provided by Natural England) and map the position of historic palaeochannels at each study site using GIS. Use this information to locate suitable areas to undertake contemporary and palaeo channel sampling (Chapters 4,5 and 6).

Historic and contemporary maps for each river were combined in order to highlight historical channel movements/changes. Information from local landowners was also utilized during the initial analysis of the floodplains. Suitable sites for the extraction of cores/sediment pits from within the palaeochannel sites were located through a combination of ground surveys and GIS analysis of historic maps.

9. Characterise the contemporary riverine environment of three lowland rivers via;

i) Sampling and examination of the biotic and abiotic elements of the rivers;

ii) Characterise the palaeoecology of each river, of when the palaeochannel was last flowing at the time of cut-off, using a multi-proxy approaches in order to aid the characterisation of 'reference conditions' (Chapters 4, 5 and 6).

A combination of sampling methods, using both kick and Surber samples, was necessary in order to provide a representative sample of the contemporary fauna within each river study reach. The palaeochannels from each river were either cored or a sediment pit was excavated in order to sample to sub-fossil macroinvertebrates. The identification of contemporary macroinvertebrates and where possible the sub-fossil taxa, to species-level was essential to help manage and conserve riverine systems where they may be specific species or habitats targets (Monk et al., 2012). Species-level data is also important in underpinning the design and management of river restoration schemes and conservation programmes, in which species-specific requirements would be overlooked if data were only resolved to family-level e.g. as part of routine biomonitoring activities (Vaughan & Ormerod, 2010)

The contemporary and palaeoecological taxa were assessed using a number of macroinvertebrate biotic indices (BMWP, ASPT, LIFE, PSI and finally CCI but only for the contemporary samples) to define each rivers typology and hydrology. Detrended Cononical Correspondence Analysis (DCCA) explored the relationships between the raw contemporary ecological data and the hydrological and geomorphological indices. Detrended Correspondence Analysis (DCA) was used to explore the structure of the contemporary macroinvertebrate community over three seasons to examine the similarities and differences in the contemporary and palaeoecological macroinvertebrate community structure.

10. Compare the results from the contemporary analysis to the palaeoecological analysis to understand the changes that have taken place on each river study reach since the palaeochannel/deposit was formed (Chapters 4, 5 and 6).

In order to allow a direct comparison, the sub-fossil community was compared to the contemporary Gastropoda, Coleoptera and Trichoptera community, in addition to the full community (including all taxa). These proxies proved to be robust, for instance due to chitinous exoskeletons, thus allowing for preservation within the palaeoechannel sediments. To determine the potential influence that species unique to one set of samples had on the comparison results, these taxa were removed from the analyses. This allowed each sampling period to be compared in an unbiased manner. Species abundance was standardised with the use of presence/absence data in order to remove any influence associated with very abundant or rare taxa.

11. Use the reference conditions identified for each river to inform and underpin future river restoration plans (Chapter 7).

Ecological interpretation of results from each river were undertaken to understand the changes that had occurred. The analysis of the macroinvertebrate communities highlighted biotopes that may have been lost or gained, and those habitats that supported unique taxa, communities or distinct habitat assemblages. Differences between contemporary and palaeoecological communities were explored in relation to known species requirements and habitat preferences.

8.3 Summary of results

The results of the research presented in this thesis have highlighted that muliproxy palaeoecological methods can successfully aid in identifying ‘reference conditions’. Insights into past riverine environments will help provide a vision for those responsible for river rehabilitation and restoration programs with a view to restoring ‘natural’ habitat quality and diversity to levels that were present before significant anthropogenic modifications took place; although this may represent a period in the recent past e.g. 1750 prior to the industrial revolution, and not necessarily a ‘natural (non anthropogenic influenced) condition. Results from all three rivers show that the combined ecological, hydrological and geomorphological aspects of rivers need to be considered wherever possible to provide a full understanding of the functioning of the system and to inform river restoration activities.

8.3.1 The River Eye SSSI

The historic and contemporary macorinvertebrate communities of the River Eye are broadly similar although the palaeoecological scores (BMWP and PSI) indicated the river had marginally higher sedimentation levels in the 1950s. The conversion of surrounding farmland from pasture to arable may largely account for these results. The river ecology does not appear to have changed significantly over the last 55 years. This supports the hypothesis that the river had already been subjected to significant anthropogenic influences by the mid 1950s and that the current problems facing the River Eye may be caused by long-term nutrient and sedimentation issues. The limited number of riffle biotopes is an additional limiting factor within the river and this can be attributed to man made structures present directly downstream of the SSSI that are having adverse effects on the river’s sediment regime. However due to the presence of a healthy macroinvertebrate community in the remaining riffles, the potential for successful restoration of the River Eye is high. The output of this research should provide a clearer focus for those undertaking future restoration activities to ensure that WFD and SSSI condition assessment criteria are met.

8.3.2 The River Hull SSSI

Currently the River Hull is over deepened, straightened, subject to periodic sedimentation and contains major structures along its course leading to localized ponding sedimentation. Faunal evidence collected from contemporary, historic (Whitehead 1935; Pearson and Jones, 1984) and palaeo samples show that the river had much more fine sediment present and slower flow velocities in the past than those experienced in the river today. This is especially evident when looking at the results from Pearson and Jones (1972). Results from the palaeoecological cores indicate the riverine environment was diverse with areas of slow flow and backwaters (containing fine sediment substratum) as well as areas of faster flow and gravel substrates. However channelization of the headwaters decreased the rivers length and resulted in habitat simplification and an increase in river gradient. Results from Whitehead (1935) and Pearson and Jones (1984) show a number of similarities to the contemporary riverine environment. Both surveys noted large volumes of macrophytes to be present within the river and a relatively mobile gravelly substratum due to fast flow velocities. This case study of the River Hull has highlighted the benefits of having access to multiple reference conditions as it allows a more in depth insight into the evolution of the river. The results obtained from using multiple reference conditions have identified the current issues affecting the River Hull Headwaters, therefore allowing potential solutions to be created that will help the river meet the WFD and SSSI condition assessment criteria.

8.3.3 The River Wensum SSSI

Ecological associations from the comparative study of modern and historic communities of the headwaters of the River Wensum demonstrated that the river has not experienced significant change to its riverine ecology since 1946. The historic channel probably had a greater diversity of biotopes including slow-flow and higher flow zones that are naturally associated with a meandering river geomorphology. However through the reinstatement and restoration of the palaeo meander a wider diversity of rheophilic and limnophilic habitats are returning to the contemporary site. Re-colonization occurred rapidly after the palaeochannel reconnection (river restoration), with the macroinvertebrate community of the reinstated meander being largely indistinguishable after only six months, thus demonstrating the rapid recovery capabilities of riverine ecosystems. This highlighted the importance of post project monitoring in certifying and ensuring that restoration objectives have been achieved.

8.4 Suggestions for Future Research

The following section considers the application of this research for future research needs for the identification of ‘reference conditions’ in the field of river restoration. A key future challenge, especially with regards to applied use and policy relevance, will be to develop protocols that maximise the relevance of paleoecological data for the development and underpinning of strategic physical river restoration plans. The application of these methods need to be realistic for use by river conservation managers and therefore the identification of reference conditions through the use of multiproxy palaeoecological techniques should be more widely available and understood. Judicious site selection for sediment coring is central to such protocols. Sites must be chosen to have sufficiently broad relevance to the experiences of river system catchments, to allow extrapolation of results to unsurveyed reaches of the river upstream and downstream. Combining the use of palaeoecological data and historic information on physical channel modifications will be key to informed evaluation of the likely spatial extent of losses of functional riverine habitats/biotopes and their associated biota.

This research has demonstrated the value of employing a multi-proxy approach to investigate palaeoecologies of rivers in order to underpin ‘reference condition’ definition to meet WFD and other habitat designation criteria. However, further analyses over greater geographical scales and a wider variety of river types would be beneficial for the identification of reference conditions for restoration programmes and for the successful implementation of the EU WFD/habitats directive/SSSI condition assessments. A catchment-based approach would provide opportunities to find more potential palaeochannel sampling sites and through incorporating multiple catchments, changes and anthropogenic impacts in both catchment and river profiles could be examined. This would achieve a better representation of past riverine environments and provide a larger dataset for analysis. This study could be extended to include a variety of rivers incorporating varying discharges and catchment geologies, providing a more holistic perspective. In order to achieve greater international significance, this research could be extended to incorporate a wider variety of rivers beyond the United Kingdom.

The palaeoecological methodology used within this study could also be adapted for the extraction of a greater variety of proxies in addition to Gastropoda, Coleoptera and Trichoptera. Additional faunal proxies such as Chironomidae, Ostacods, Cladocera, Simuliidae and mites could also be used as examples, as each of these species have been noted within the sub-fossil community during sediment analysis (Howard *et al*., 2009). Palaeoecological samples could also be processed for the presence of diatoms to help provide indications of changing conditions within the river and the wider floodplain environment. Terrestrial macroinvertebrate species e.g. Coleoptera, and the ecological information associated with them could also be incorporated into this research as this would provide a greater insight into the surrounding marginal riparian vegetation. This would help make connections between the contemporary in-channel fauna to the broader floodplain environment. Additional environmental information could be gained from the inclusion of plant macrophytes, seeds and pollen that are often abundant within palaeoechannel sediments. The incorporation of flora into the results allows corroboration of the terrestrial line of evidence with the aquatic instream vegetation enabling the definition of both terrestrial and aquatic reference conditions.

8.5 Conclusion

This thesis has investigated the changing ecologies of three SSSI rivers within the United Kingdom; the River Eye (Melton Mowbray), the River Hull (Driffield) and the River Wensum (Fakenham), through the integration of contemporary and palaeoecological analysis. Using historic information (historic maps and documents) it has been possible to identify instream morphological and habitat features that have been significantly degraded by historic management operations. Through detailed field based studies and examination of existing riverine communities this research has utilized a multiproxy approach, incorporating three biological proxies; Gastropoda, Coleoptera, Trichoptera, in order to reconstruct past riverine environments and to inform reference conditions for future river restoration purposes. This has identified elements of the instream faunal community that have been significantly compromised or made locally extinct as a result of historic channel management operations. The results further highlight the importance of understanding how riverine environments have changed and how these changes have impacted and influenced instream macroinvertebrate communities.

There are many instances where true restoration cannot be achieved and rehabilitation methods have to be adopted. However through the use of an ideal reference condition, to use as evaluation of what river managers should be aiming for, it will help enforce the basis of SSSI river restoration planning. The challenge for the future lies in protecting the ecological integrity and biodiversity of aquatic systems in the face of increasing pressures on our freshwater resources. Therefore restoring rivers back to a more ‘natural’ state using historic reference conditions is of great importance in encouraging the rebuilding of these fragmented habitats, thus securing their future.

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Appendix 1 Raw baseline seasonal survey data obtained from i) Kick sampling and ii) Surber sampling of the contemporary macroinvertebrate community of the River Eye.

i) Seasonal kick data	**Spring 2010**					**Winter 2011**					**Autumn 2011**				
	1	**2**	**3**	**4**	**5**	**1**	**2**	**3**	**4**	**5**	**1**	**2**	**3**	**4**	**5**
Theodoxus fluviatilis	3	34	3	12	21	5	49	14	37	30	10	**0**	4	1	2
Valvata piscinalis	1	5	0	17	9	0	0	0	1	3	1	0	0	1	1
Valvata piscinalis	0	0	0	0	0	0	0	0	24	21	0	0	0	4	6
Potamopyrgus antipodarum	21	87	29	29	27	18	27	9	12	19	9	3	5	2	3
Bithynia tentaculata	7	4	2	3	6	0	0	0	0	0	0	0	4	2	1
Lymnaea peregra	4	5	0	2	3	1	4	0	4	5	0	7	2	2	3
Lymnaea stagnalis	0	0	0	0	0	0	1	0	0	0	0	1	0	0	0
Gyrautus albus	2	0	0	0	1	0	0	0	0	0	3	0	0	0	0
Planorbis carinatus	2	0	0	0	0	1	5	1	4	4	0	0	0	0	0
Planorbis levcostoma	6	14	5	9	11	1	8	2	4	7	0	0	0	0	0
Planorbis planorbis	0	10	4	8	3	2	3	0	2	2	0	0	3	0	0
Anisus vortex	0	0	0	0	0	2	0	0	0	0	2	3	0	0	0
Bathyomphalus contortus	11	10	5	0	6	1	5	0	0	1	0	2	1	1	2
Acroloxus lacustris	0	8	3	0	1	4	15	0	5	3	0	1	3	4	2
Zonitoides nitidus	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0
Sphaeriidae	6	0	19	16	10	3	11	0	6	7	7	3	3	3	4
OLIGOCHAETA	7	8	17	7	5	6	0	0	6	6	5	2	4	3	4
Glossophanic complanata	4	4	0	10	9	0	4	0	11	14	0	0	0	0	0
Erpobdella octoculata	8	38	0	36	31	3	11	2	31	25	3	1	7	3	2
Asellus aquaticus	1	6	1	4	5	0	0	0	1	2	0	0	1	0	0
Platycnemis pennipes	0	0	0	0	0	0	0	0	0	0	0	3	0	2	2
Calopteryx splendens	0	0	0	0	0	0	0	0	0	0	0	4	0	1	2
Nemouride	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0

i) Continued

	Spring 2010					Winter 2011					Autumn 2011				
	1	**2**	**3**	**4**	**5**	**1**	**2**	**3**	**4**	**5**	**1**	**2**	**3**	**4**	**5**
Baetidae (cloeon)	0	0	0	0	0	0	6	2	6	5	0	19	19	15	19
Baetis rhodoni	0	0	0	0	0	0	18	7	22	24	33	0	0	0	0
Habrophlebia fusca	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0
Ephemera vulgate	0	0	0	0	0	0	0	0	0	0	0	0	1	1	2
Ephemera danica	5	3	12	2	4	0	0	0	0	0	0	0	1	1	3
Paraleptoplebia submarginata	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Caenis luctuosa	2	9	3	8	4	109	30	57	173	127	4	0	3	1	2
Calopteryx vigro	0	0	1	0	0	1	0	0	0	0	0	0	0	0	0
Haliplus ruficollis group	0	0	0	0	0	0	0	0	1	2	0	1	0	2	1
Haliplidae larvae	0	0	0	4	3	0	0	0	0	0	0	0	0	0	0
Potamonectes depressus elegans	0	0	0	0	0	2	0	0	8	3	0	9	0	4	7
Helophorus brevipalpis	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0
Elmis aenea (adult)	0	9	0	0	3	3	7	21	7	5	4	0	4	0	0
Elmis aenea (larvae)	1	41	0	0	1	0	8	0	0	0	5	1	1	0	0
Oulimnius tuberculatus (adult)	0	7	0	0	7	0	6	9	3	3	0	0	11	0	2
Oulimnius tuberculatus (larvae)	1	17	2	2	6	0	7	0	1	2	0	0	0	0	0
Rove beetles	0	0	0	0	0	0	0	0	0	0	3	1	0	0	0
Sialis lutaria	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Agapetus fuscipes	0	48	0	0	7	0	37	0	3	4	29	0	37	0	6
Polycentropus flavo	1	2	0	0	0	0	0	0	0	0	4	0	2	0	0
Tinodes waeneri	1	1	0	0	0	1	0	0	0	0	0	0	4	0	1
Hydropsyche pellucidula	0	29	1	3	16	1	33	3	42	46	7	0	0	0	0
Hydropsyche siltalai	0	1	0	0	0	0	1	0	0	0	37	0	8	0	3
Hydropsyche instabilis	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0
Hydropsyche angustipennis	0	0	0	0	0	0	0	0	5	2	0	0	0	0	0
Hydroptila sp.	0	0	0	0	0	39	22	0	7	8	42	0	0	3	2

i) Continued	**Spring 2010**					**Winter 2011**					**Autumn 2011**				
	1	**2**	**3**	**4**	**5**	**1**	**2**	**3**	**4**	**5**	**1**	**2**	**3**	**4**	**5**
Phryganea bipunctata	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Limephilus lunatus	0	8	1	4	2	0	18	21	9	15	0	0	0	2	5
Potamophylax angulatus	0	0	0	0	0	0	0	0	2	23	0	0	0	0	0
Halesus radiates	0	0	0	0	0	1	1	0	0	3	3	1	0	1	2
Athripsodes albifrons	0	0	0	0	0	0	0	0	29	21	0	0	0	0	0
Athripsodes bilineatus	4	13	3	3	11	4	5	19	10	14	25	0	55	11	15
Goera pilosa	2	4	1	59	37	4	10	14	98	87	0	0	24	2	6
Lepidostoma hirtum	0	1	0	0	0	0	9	0	0	1	312	0	44	4	4
Hydrometridae	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0
Sigara stagnalis	0	0	0	0	0	0	0	0	0	0	0	10	0	0	0
Chironomidae	14	27	33	17	23	14	31	0	2	10	17	10	3	9	6
Simuliidae	2	38	0	16	16	0	0	0	3	2	10	0	0	0	1
Tipulidae	0	15	0	0	1	0	0	0	0	0	0	0	0	0	0
Psychodidae	0	0	0	0	0	0	0	0	2	3	2	0	0	0	1
Dinocrota	0	12	0	3	10	0	0	0	1	1	0	1	0	0	1

ii) Spring Surber – Sites 1-3	1.1	1.2	1.3	1.4	1.5	2.1	2.2	2.3	2.4	2.5	3.1	3.2	3.3	3.4	3.5
Theodoxus fluviatilis	0	0	6	6	1	22	4	41	8	9	0	0	0	0	0
Valvata piscinalis	1	0	7	5	7	2	1	3	0	0	0	0	0	1	0
Potamopyrgus antipodarum	0	0	43	47	25	12	16	65	16	1	2	8	2	7	1
Bithynia tentaculata	0	0	2	1	0	2	0	0	0	1	0	0	0	0	0
Lymnaea peregra	3	0	1	0	0	0	3	1	3	2	0	0	0	0	0
Gyrautus albus	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Physa fontinalis	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Planorbis carinatus	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0
Planorbis levcostoma	0	0	7	5	4	1	2	7	2	1	2	0	1	0	0
Planorbis laevis	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Planorbis planorbis	1	0	4	3	0	3	0	4	2	0	1	0	0	0	0
Bathyomphalus contortus	0	1	12	9	7	5	4	12	3	0	3	2	1	0	0
Acroloxus lacustris	0	0	3	1	0	6	3	7	2	1	0	0	0	0	0
Zonitoides nitidus	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0
Sphaeriidae	0	0	6	30	5	0	0	0	0	0	0	0	1	9	11
OLIGOCHAETA	7	2	8	11	4	7	6	5	3	2	2	6	3	7	3
Glossophanic complanata	0	0	2	2	0	2	0	3	2	0	0	0	0	0	0
Erpobdella octoculata	0	0	3	2	0	7	5	5	2	1	0	0	0	0	0
Asellus aquaticus	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0
Ephemera danica	0	2	3	4	6	0	4	12	0	0	3	1	9	8	4
Paraleptoplebia submarginata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Caenis luctuosa	2	4	1	1	0	3	2	1	0	1	5	13	3	4	0
Haliplidae (larvae)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Elmis aenea (adult)	0	1	1	2	0	0	1	1	0	0	0	0	0	1	0
Elmis aenea (larvae)	0	1	1	1	0	6	16	12	4	3	0	0	0	0	0
Oulimnius tuberculatus (larv.)	0	1	1	2	0	2	7	6	0	2	3	0	0	1	0
Sialis lutaria	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Agapetus fuscipes	0	0	0	0	0	2	0	3	0	0	0	0	0	0	0
Polycentropus flavomaculatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tinodes waeneri	0	0	1	1	0	0	0	1	0	0	0	0	0	0	0
Hydropsyche pellucidula	0	0	0	1	0	0	1	3	1	0	0	0	0	1	1

ii) Spring Sites 1-3 continued	1.1	1.2	1.3	1.4	1.5	2.1	2.2	2.3	2.4	2.5	3.1	3.2	3.3	3.4	3.5
Hydroptila sp.	0	0	0	3	0	0	0	0	0	0	0	0	0	0	0
Limephilus lunatus	0	0	0	0	0	0	2	1	0	1	0	2	0	0	1
Halesus radiatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Athripsodes bilineatus	2	2	0	0	2	7	3	2	4	3	0	0	3	0	0
Goera pilosa	0	0	0	1	0	1	0	0	0	0	0	1	1	0	0
Lepidostoma hirtum	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Chironomidae	16	6	39	31	12	11	3	7	3	6	7	8	11	9	20
Simuliidae	1	1	3	22	0	47	14	26	9	30	0	5	0	0	0
Tipulidae	0	0	0	1	0	5	2	0	0	1	0	0	0	0	1
Psychodidae	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Dinocrota	0	0	0	0	0	0	1	2	1	0	0	0	0	0	0

ii) Spring Surber – Sites 4-5	4.1	4.2	4.3	4.4	4.5	5.1	5.2	5.3	5.4	5.5
Theodoxus fluviatilis	0	2	0	7	0	4	5	11	0	0
Valvata piscinalis	3	2	5	15	2	6	30	17	3	1
Potamopyrgus antipodarum	4	6	14	12	1	17	18	19	16	5
Bithynia tentaculata	0	1	1	1	0	5	1	0	0	0
Lymnaea peregra	1	2	0	0	0	0	0	0	1	0
Gyrautus albus	0	0	0	0	0	0	0	0	0	0
Physa fontinalis	0	0	0	0	1	0	0	0	0	0
Planorbis carinatus	0	0	0	0	0	0	0	0	0	0
Planorbis levcostoma	0	2	3	3	0	3	2	5	4	1
Planorbis laevis	0	0	0	0	0	0	0	0	0	0
Planorbis planorbis	1	3	5	4	0	3	0	8	5	0
Bathyomphalus contortus	0	2	1	3	0	4	1	2	4	0
Acroloxus lacustris	0	2	0	0	0	0	0	0	0	0
Zonitoides nitidus	0	0	0	0	0	0	0	0	0	0
Sphaeriidae	2	0	11	12	0	0	7	3	3	1
OLIGOCHAETA	0	9	3	13	2	11	26	9	10	6

ii) Spring Sites 4-5 continued	4.1	4.2	4.3	4.4	4.5	5.1	5.2	5.3	5.4	5.5
Glossophanic complanata	0	7	5	7	1	6	4	3	0	6
Erpobdella octoculata	0	6	12	30	0	15	10	13	2	3
Asellus aquaticus	2	0	3	3	0	2	0	4	3	0
Ephemera danica	0	0	1	0	1	0	4	1	0	0
Paraleptoplebia submarginata	0	0	0	0	0	0	0	0	0	1
Caenis luctuosa	5	3	0	6	0	0	3	16	3	64
Haliplidae (larvae)	0	0	0	5	0	0	0	0	3	0
Elmis aenea (adult)	0	0	0	1	0	0	0	1	0	2
Elmis aenea (larvae)	0	0	0	0	1	0	0	0	0	2
Oulimnius tuberculatus (larv)	0	1	0	1	0	2	1	4	1	2
Sialis lutaria	1	0	0	0	0	0	0	0	0	0
Agapetus fuscipes	0	0	0	0	0	0	0	0	0	0
Polycentropus flavo	0	0	1	0	0	0	0	0	0	1
Tinodes waeneri	0	1	0	0	0	0	0	0	0	0
Hydropsyche pellucidula	0	0	0	4	0	0	0	0	2	0
Hydroptila sp.	0	0	0	0	0	0	0	0	0	0
Limephilus lunatus	2	0	0	4	0	2	2	2	0	0
Halesus radiatus	0	1	0	0	0	0	0	0	0	0
Athripsodes bilineatus	0	0	0	3	0	2	0	1	0	0
Goera pilosa	0	3	5	16	0	1	7	8	0	0
Lepidostoma hirtum	0	0	0	0	0	0	0	0	0	0
Chironomidae	142	7	3	9	40	0	0	0	0	0
Simuliidae	0	3	0	31	0	0	0	0	0	0
Tipulidae	0	1	6	15	0	0	0	0	0	0
Psychodidae	0	0	0	0	0	0	0	0	0	0
Dinocrota	0	0	0	3	0	0	0	0	0	0

ii) Winter Surber – Sites 1-3	**1.1**	**1.2**	**1.3**	**1.4**	**1.5**	**2.1**	**2.2**	**2.3**	**2.4**	**2.5**	**3.1**	**3.2**	**3.3**	**3.4**	**3.5**
Theodoxus fluviatilis	12	2	3	4	4	4	16	27	3	5	0	2	4	0	11
Valvata cristata	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Valvata piscinalis	0	1	2	0	0	0	0	0	2	0	0	0	3	0	1
Potamopyrgus antipodarum	21	3	0	0	6	9	11	12	0	0	3	5	4	0	0
Bithynia tentaculata	1	0	1	0	0	1	4	0	0	0	0	1	1	0	0
Lymnaea peregra	3	1	0	0	0	3	0	3	0	0	0	0	0	0	0
Planorbis carinatus	2	0	0	0	0	0	2	1	0	0	0	0	0	0	3
Planorbis leucostoma	3	0	1	1	2	0	3	2	0	0	0	1	0	1	1
Planorbis planorbis	1	0	0	0	1	0	0	1	0	0	0	0	0	0	0
Anisus vortex	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0
Bathyomphalus contortus	0	0	2	3	0	0	2	3	0	2	1	1	0	0	0
Acroloxus lacustris	2	0	0	3	1	0	7	2	0	0	0	0	0	0	1
Sphaeriidae	3	1	0	0	0	2	8	8	0	0	2	0	3	0	2
OLIGOCHAETA	4	0	3	2	3	3	0	0	0	5	0	2	0	0	3
Glossophanic complanata	0	0	1	0	0	0	0	0	3	0	1	0	0	0	0
Erpobdella octoculata	7	0	2	2	5	2	8	3	4	2	0	1	2	0	3
Asellus aquaticus	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0
Baetidae (cloeon)	0	0	0	0	0	2	3	0	0	7	3	1	0	0	2
Baetis rhodoni	0	0	0	0	0	0	7	2	0	1	8	0	2	4	11
Ephemeralla ignita	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0
Caenis luctuosa	147	56	57	4	15	17	15	6	12	48	41	19	17	32	42
Calopteryx vigro	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Haliplus ruficollis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nebrioporus depressus	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Elmis aenea (adult)	18	2	0	1	0	5	2	0	2	7	1	0	2	0	2
Elmis aenea (larvae)	0	0	0	2	3	2	3	6	0	2	4	0	2	3	8
Hydraenadae	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Oulimnius tuberculatus (adult)	0	0	0	0	0	4	2	0	0	2	0	0	1	0	1
Oulimnius tuberculatus (larv)	2	0	0	1	0	3	2	2	0	1	0	0	0	0	0
Rove beetles	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0
Sialis lutaria	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0

ii) Winter Sites 1-3 cont.	1.1	1.2	1.3	1.4	1.5	2.1	2.2	2.3	2.4	2.5	3.1	3.2	3.3	3.4	3.5
Agapetus fuscipes	0	0	0	0	0	0	0	108	0	0	0	0	0	0	0
Polycentropus flavo	2	0	3	0	0	0	0	0	0	0	0	0	0	0	0
Tinodes waeneri	8	0	2	4	0	0	0	0	0	0	0	0	0	0	0
Hydropsyche pellucidula	1	0	0	2	0	0	3	22	3	0	0	0	5	0	8
Hydroptila sp.	53	12	13	0	7	13	7	5	4	0	0	0	0	0	0
Phryganea bipunctata	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Limephilus lunatus	0	0	0	0	0	0	0	4	1	0	0	6	2	2	0
Halesus radiatus	0	2	1	0	2	0	3	1	0	0	0	0	0	0	0
Athripsodes albifrons	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Athripsodes bilineatus	1	1	2	3	1	5	8	12	2	3	5	6	7	4	1
Goera pilosa	1	1	1	2	4	0	0	0	34	0	0	1	3	1	0
Lepidostoma hirtum	0	0	0	0	0	0	9	5	3	1	1	0	1	1	0
Sericostoma personatum	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Chironomidae	8	4	0	3	2	7	69	4	18	4	0	0	0	0	3
Simuliidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	13

ii) Winter Surber – Sites 4-5	4.1	4.2	4.3	4.4	4.5	5.1	5.2	5.3	5.4	5.5
Theodoxus fluviatilis	0	3	0	2	10	0	4	0	5	15
Valvata cristata	0	0	0	0	0	0	3	0	0	0
Valvata piscinalis	0	14	5	8	7	0	3	6	0	9
Potamopyrgus antipodarum	0	9	0	4	8	0	9	2	2	5
Bithynia tentaculata	0	0	0	0	2	0	0	0	2	0
Lymnaea peregra	2	3	0	0	0	0	0	0	1	2
Planorbis carinatus	0	0	0	0	0	0	0	2	2	2
Planorbis leucostoma	0	1	0	0	2	0	0	1	0	3
Planorbis planorbis	0	0	0	0	0	2	0	0	0	0
Anisus vortex	0	0	0	0	1	0	0	0	1	0
Bathyomphalus contortus	1	0	2	0	0	0	0	0	0	0
Acroloxus lacustris	0	0	0	0	1	0	0	0	1	2

ii) Winter Sites 4-5 cont.	4.1	4.2	4.3	4.4	4.5	5.1	5.2	5.3	5.4	5.5
Sphaeriidae	2	2	0	3	0	0	3	1	4	2
OLIGOCHAETA	4	1	0	0	0	4	0	0	2	0
Glossophanic complanata	1	0	1	3	8	0	2	4	4	3
Erpobdella octoculata	1	5	4	16	5	9	4	7	14	13
Asellus aquaticus	0	0	1	0	0	0	0	0	0	0
Baetidae (cloeon)	0	0	0	0	7	0	0	1	0	1
Baetis rhodoni	0	8	0	0	19	2	2	7	7	11
Ephemeralla ignita	0	0	0	0	0	0	0	0	0	0
Caenis luctuosa	61	77	52	11	30	17	4	35	12	32
Haliplus ruficollis	0	0	0	0	0	0	0	0	1	0
Nebrioporus depressus	0	0	9	1	0	0	0	0	0	2
Elmis aenea (adult)	1	2	0	3	10	0	0	0	0	3
Elmis aenea (larvae)	0	0	1	0	6	0	0	0	0	0
Oulimnius tuberculatus	1	1	0	0	5	0	0	0	0	2
Sialis lutaria	1	0	0	0	0	0	0	0	0	0
Agapetus fuscipes	1	0	0	0	0	1	0	0	0	0
Polycentropus flavo	1	0	10	2	0	0	0	0	0	0
Tinodes waeneri	0	0	1	0	0	2	1	1	2	0
Hydropsyche pellucidula	0	2	0	0	12	3	1	5	6	14
Hydroptila sp.	0	0	0	0	3	0	0	0	0	0
Phryganea bipunctata	0	2	0	0	0	0	0	0	0	0
Limephilus lunatus	0	0	0	0	0	0	0	4	6	0
Halesus radiatus	1	0	1	1	0	0	0	1	0	0
Athripsodes albifrons	0	0	0	0	10	1	8	7	12	7
Athripsodes bilineatus	1	4	5	3	2	1	3	2	4	1
Goera pilosa	0	3	0	8	18	3	17	9	26	31
Lepidostoma hirtum	0	0	2	0	0	0	0	0	0	0
Chironomidae	0	0	1	2	6	0	0	1	0	0
Simuliidae	0	0	0	0	0	0	0	4	0	0

Appendix 2 The palaeoecological macroinvertebrate community of the River Eye extracted from the sediment pit on in-stream island.

	Pit 1				Pit 2			
	20-30 cm	30-40 cm	40-45 cm	45-50 cm	30-40 cm	40-45 cm	45-50 cm	50-55 cm
Theodoxus fluviatilis	0	3	1	2	0	2	0	0
Valvata piscinalis	0	0	0	0	1	0	1	0
Potamopyrgus antipodarum	2	8	10	4	7	4	8	0
Bithynia tentaculata	0	0	0	0	0	0	0	1
Planorbis planorbis	0	1	1	0	1	0	0	0
Anisus vortex	0	0	0	0	1	0	0	0
Acroloxus lacustris	0	1	1	0	0	0	0	0
Sphaeriidae	0	1	0	0	0	0	0	0
Sialis lutaria	3	3	5	3	17	9	7	0
Haliplidae	0	0	0	2	2	2	1	0
Helophorus	0	4	0	5	7	13	6	0
Cercyon	0	0	0	0	0	0	0	4
Hydraenidae	0	3	1	0	4	5	2	3
Elmis aenea	2	8	5	10	10	13	3	6
Oulimnius tuberculatus	0	2	1	2	4	3	0	0
Hydropsyche sp	0	0	0	0	0	0	0	1
Hydropsyche pellucidula	1	5	4	3	0	1	5	1
Hydropsyche angustipennis	5	5	5	2	4	4	4	0
Hydropsyche siltalai	3	5	0	0	0	0	0	0
Hydropsyche instabilis	0	5	2	2	0	3	3	0
Athripsodes aterrimus	0	5	4	3	5	5	2	3
Athripsodes bilineatus	0	3	3	1	0	1	2	0
Goera pilosa	0	1	3	0	0	2	1	0

Appendix 3 Raw baseline seasonal survey data obtained from i) Kick sampling and ii) Surber sampling of the contemporary macroinvertebrate community of the River Hull.

i) Seasonal Kick Data

	Spring 2011				Summer 2011				Autumn 2011			
	1	**2**	**3**	**4**	**1**	**2**	**3**	**4**	**1**	**2**	**3**	**4**
Potamopyrgus jenkinsi	0	3	0	2	0	0	3	0	0	2	0	0
OLIGOCHAETA	2	57	2	14	18	6	5	37	28	30	49	37
Piscicola geometra	0	0	0	0	0	0	2	1	0	0	0	0
Glossophania complenata	1	2	0	0	0	0	0	0	0	0	0	3
Erpobdella octoculata	6	3	22	5	16	3	4	0	17	19	0	14
Gammerus pulex	334	160	419	156	847	451	891	945	1712	999	801	573
Baetis rhodoni	15	6	9	2	0	0	0	0	0	0	0	0
Ephemeralla ignita	2	0	0	1	479	221	189	136	37	53	31	28
Elmis aenea (adult)	11	0	4	4	3	4	3	7	19	23	9	9
Elmis aenea (larvae)	0	0	1	0	4	8	3	7	11	19	11	9
Rhyacophila dorsalis	0	0	0	0	0	0	0	0	0	0	3	0
Agapetus fuscipes	96	249	41	265	12	28	63	78	283	416	217	498
Drusus annulatus	69	8	18	19	10	30	17	49	21	20	15	9
Potamophylax cingulatus	0	0	0	0	0	0	0	0	0	0	0	0
Silo nigricornis	22	1	6	13	9	5	10	8	17	8	7	5
Sericostoma personatum	0	1	4	2	3	6	19	4	12	21	10	12
Chironomidae	1	0	2	0	14	3	6	17	0	0	4	0
Simuliidae	11	38	54	2	0	0	0	4	0	0	0	0
Psychodidae	0	0	0	25	0	0	0	0	0	0	0	0
Dinocrota	1	6	4	0	0	8	19	38	30	33	23	14

ii) Spring Surber Sites 1-2

	Site 1					Site 2				
	1	**2**	**3**	**4**	**5**	**1**	**2**	**3**	**4**	**5**
Potamopyrgus jenkinsi	2	3	0	0	0	1	0	0	0	0
Zonitoides nitidus	0	0	0	0	0	0	0	0	0	0
OLIGOCHAETA	17	16	3	59	2	22	3	50	11	4
Piscicola geometra	0	0	0	1	0	0	0	1	1	0
Glossophania complenata	0	0	0	0	0	0	0	5	0	1
Erpobdella octoculata	0	3	1	22	4	5	6	2	2	3
Asellus meridianus	0	0	0	0	0	0	0	0	0	0
Gammerus pulex	45	47	163	178	46	1	117	48	33	62
Baetis rhodoni	0	1	4	0	1	0	0	1	0	1
Ephemeralla ignita	0	0	1	0	0	0	0	0	0	0
Elmis aenea (adult)	0	0	4	2	2	2	0	0	0	1
Elmis aenea (larvae)	1	1	0	0	0	1	0	0	0	0
Agapetus fuscipes	7	1	6	11	2	15	29	17	11	40
Drusus annulatus	8	4	0	36	21	2	10	3	24	64
Potamophylax cingulatus	0	0	0	0	0	0	0	0	0	0
Silo nigricornis	0	7	3	10	2	0	2	10	12	7
Sericostoma personatum	0	0	0	4	0	0	0	1	4	2
Chironomidae	1	2	0	0	0	0	0	0	0	0
Simuliidae	3	0	2	1	0	0	5	0	0	0
Psychodidae	0	0	0	0	0	0	0	0	0	0
Dinocrota	2	0	1	5	0	0	1	0	1	2

ii) Spring Surber Sites 3-4

	Site 3					Site 4				
	1	**2**	**3**	**4**	**5**	**1**	**2**	**3**	**4**	**5**
Potamopyrgus jenkinsi	1	0	4	3	5	0	1	8	1	1
Zonitoides nitidus	0	0	0	4	0	0	0	3	1	0
OLIGOCHAETA	3	5	9	21	19	1	34	8	7	69
Piscicola geometra	0	0	1	0	0	0	1	1	0	0
Glossophania complenata	0	0	0	0	0	0	3	0	0	0
Erpobdella octoculata	1	1	0	1	1	0	5	0	0	0
Asellus meridianus	0	0	0	0	0	1	0	0	0	1
Gammerus pulex	25	41	203	9	194	208	15	308	234	33
Baetis rhodoni	1	0	0	0	1	2	0	3	2	0
Ephemeralla ignita	0	0	0	0	0	1	0	0	1	0
Elmis aenea (adult)	1	0	4	0	22	3	0	24	5	0
Elmis aenea (larvae)	0	0	0	0	0	0	0	1	0	0
Agapetus fuscipes	29	34	4	11	156	37	22	183	45	80
Drusus annulatus	5	2	10	3	2	10	0	17	28	2
Potamophylax cingulatus	0	0	0	0	0	0	2	0	0	0
Silo nigricornis	1	0	2	0	3	2	0	6	1	1
Sericostoma personatum	0	0	1	0	3	1	1	3	3	0
Chironomidae	1	2	4	3	2	0	0	0	0	0
Simuliidae	10	0	5	31	12	0	0	0	4	0
Psychodidae	0	0	0	0	0	1	69	0	1	14
Dinocrota	0	0	5	2	7	0	0	0	0	0

ii) Summer Surber Sites 1-2

	Site 1					Site 2				
	1	**2**	**3**	**4**	**5**	**1**	**2**	**3**	**4**	**5**
Potamopyrgus jenkinsi	0	0	1	0	1	0	0	0	0	0
Zonitoides nitidus	0	0	0	0	0	0	0	0	0	0
OLIGOCHAETA	5	3	13	8	11	6	7	6	8	17
Piscicola geometra	0	0	0	0	0	0	0	0	0	0
Erpobdella octoculata	4	9	3	4	2	0	2	2	0	1
Gammerus pulex	776	348	410	130	623	170	114	72	76	103
Baetis rhodoni	7	0	19	3	21	0	0	0	0	0
Ephemeralla ignita	26	97	138	91	139	71	34	42	36	113
Elmis aenea (adult)	1	1	0	0	2	2	0	0	0	2
Elmis aenea (larvae)	0	1	2	1	0	2	0	0	4	1
Rhyacaphila dorsalis	0	0	0	0	0	0	0	0	0	0
Agapetus fuscipes	0	0	12	3	10	49	28	15	20	3
Drusus annulatus	0	8	8	1	4	3	3	4	17	11
Silo nigricornis	0	0	0	0	8	3	0	0	7	0
Sericostoma personatum	0	2	1	0	5	0	0	1	2	1
Chironomidae	0	0	3	0	10	0	2	2	0	0
Simuliidae	0	0	0	0	0	0	0	0	0	0
Dinocrota	0	0	0	0	0	3	2	5	1	3

ii) Summer Surber Sites 3-4

	Site 3					Site 4				
	1	**2**	**3**	**4**	**5**	**1**	**2**	**3**	**4**	**5**
Potamopyrgus jenkinsi	3	1	2	0	0	3	1	0	0	1
Zonitoides nitidus	0	0	0	0	0	0	1	0	0	0
OLIGOCHAETA	6	47	9	5	2	11	20	18	15	36
Piscicola geometra	0	1	0	0	0	0	0	0	0	0
Erpobdella octoculata	0	0	3	0	0	1	3	0	0	0
Gammerus pulex	220	314	210	254	72	457	314	666	143	203
Baetis rhodoni	17	12	11	4	6	0	27	13	4	9
Ephemeralla ignita	48	59	95	78	16	37	116	53	23	48
Elmis aenea (adult)	0	5	0	1	0	17	3	16	1	1
Elmis aenea (larvae)	1	1	2	1	0	2	4	9	0	1
Rhyacaphila dorsalis	0	1	0	0	0	0	0	0	0	0
Agapetus fuscipes	16	55	20	42	23	31	92	55	10	18
Drusus annulatus	1	13	4	11	5	14	17	24	13	3
Silo nigricornis	0	2	3	0	0	4	3	16	0	0
Sericostoma personatum	3	7	6	9	2	3	1	1	0	0
Chironomidae	0	3	2	0	0	2	6	0	0	2
Simuliidae	0	0	0	0	0	0	2	0	0	1
Dinocrota	3	7	6	9	2	6	8	5	6	2

ii) Autumn Surber Sites 1-2

	Site 1					Site 2				
	1	**2**	**3**	**4**	**5**	**1**	**2**	**3**	**4**	**5**
Potamopyrgus jenkinsi	0	0	0	2	0	1	0	0	0	0
OLIGOCHAETA	8	14	6	3	4	2	6	7	3	5
Glossophania complenata	0	0	0	0	0	0	0	0	0	0
Erpobdella octoculata	1	5	5	2	4	7	2	1	3	2
Gammerus pulex	254	887	118	417	471	348	301	171	149	200
Ephemeralla ignita	2	5	1	0	8	21	10	7	2	18
Elmis aenea (adult)	1	2	0	7	3	9	3	7	0	3
Elmis aenea (larvae)	0	6	0	0	1	12	5	0	0	4
Oulimnius tuberculatus	0	0	0	0	0	0	0	0	0	0
Rhyacaphila dorsalis	0	0	0	0	0	0	0	1	0	0
Agapetus fuscipes	19	13	103	29	75	147	139	124	61	111
Drusus annulatus	3	7	1	6	5	14	1	0	0	8
Silo nigricornis	0	5	2	1	4	1	0	0	1	2
Sericostoma personatum	3	3	0	2	3	1	3	0	2	3
Chironomidae	0	0	0	0	0	0	0	0	0	0
Dinocrota	11	4	3	10	8	4	7	14	0	8

ii) Autumn Surber Sites 3-4

	Site 3					Site 4				
	1	**2**	**3**	**4**	**5**	**1**	**2**	**3**	**4**	**5**
OLIGOCHAETA	27	12	6	5	17	10	14	6	4	19
Glossophania complenata	0	0	0	0	0	0	0	2	0	0
Erpobdella octoculata	0	0	0	0	0	1	0	1	0	2
Gammerus pulex	217	240	73	166	188	147	134	164	170	201
Ephemeralla ignita	11	2	4	6	0	4	7	2	0	1
Elmis aenea (adult)	2	0	3	1	0	1	0	5	4	4
Elmis aenea (larvae)	0	4	5	2	0	0	0	3	2	4
Oulimnius tuberculatus	0	0	1	0	0	0	0	1	0	0
Rhyacaphila dorsalis	0	0	1	0	0	0	0	0	0	0
Agapetus fuscipes	133	253	167	199	200	174	122	199	138	207
Drusus annulatus	5	0	4	1	0	1	0	2	5	0
Silo nigricornis	2	0	1	0	0	2	0	1	1	0
Sericostoma personatum	4	1	1	2	0	1	0	6	1	3
Chironomidae	1	1	0	0	0	0	0	0	0	0
Dinocrota	9	4	10	11	0	5	6	8	2	9

Appendix 4 The palaeoecological macroinvertebrate community of the River Hull extracted from the palaeo channel.

	Core 1					Core 2		Core 3			
	50-60 cm	60-70 cm	70-80 cm	80-90 cm	90-100 cm	50-60 cm	60-70 cm	50-60 cm	60-70 cm	70-80 cm	80-90 cm
Valvata piscinalis	0	0	0	0	0	0	0	0	0	0	0
Potamopyrgus jenkinsi	0	0	0	0	0	0	0	0	0	0	0
Physa fontinalis	0	0	0	0	0	0	0	0	0	0	0
Ancylus fluviatilis	0	0	0	0	0	0	0	0	0	0	0
Pisidium sp	0	0	0	0	0	0	0	0	0	0	0
Sialis lutaria	6	6	7	2	3	6	2	6	2	0	2
Nebrioporus elegans	0	2	2	1	3	0	0	0	2	2	0
Haliplus sp	0	0	0	0	0	0	0	0	0	0	0
Hydroporus sp	0	0	0	0	0	0	0	0	0	0	0
Helophorus	0	0	1	1	2	0	0	0	1	1	0
Cercyon	0	2	1	1	1	0	1	2	0	2	0
Elmis aenea (adult)	0	0	0	1	1	0	1	0	1	0	0
Elmis aenea (larvae)	0	0	0	0	0	0	0	0	0	0	0
Rhyacophila dorsalis	0	1	2	0	0	0	0	0	0	0	0
Agapetus fuscipes	0	0	0	0	0	0	0	0	0	0	0
Hydropsyche pellucidula	2	1	0	0	0	2	0	0	0	0	0
Hydropsyche angustipennis	0	0	0	4	0	0	0	2	0	0	0
Drusus annulatus	0	0	0	0	0	0	0	0	0	0	0
Limnophilidae	0	0	0	0	0	0	0	0	0	0	0
Limnephilus marmoratus	0	4	2	2	0	6	0	2	0	0	0
Limneph 3 (in det)	0	0	2	0	0	0	0	0	0	0	0
Limneph 4 (in det)	2	0	0	1	0	0	0	0	0	1	0
Anabolia nervosa	11	8	17	7	6	9	2	10	7	2	0

Palaeo count continued.

	Core 1					Core 2		Core 3			
	50-60 cm	60-70 cm	70-80 cm	80-90 cm	90-100 cm	50-60 cm	60-70 cm	50-60 cm	60-70 cm	70-80 cm	80-90 cm
Anabolia nervosa	11	8	17	7	6	9	2	10	7	2	0
Potamophylax cingulatus	4	3	5	2	1	4	4	0	2	0	0
Halesus radiatus	14	2	5	1	0	10	0	6	2	0	0
Melampophylax mucoreus	9	4	3	6	8	6	0	2	9	3	0
Athripsodes aterrimus	0	0	0	0	0	3	0	0	0	0	0
Athripsodes sp	0	0	0	0	0	3	0	0	0	0	0
Silo pallipes	0	2	0	0	2	0	0	0	2	2	0
Silo nigricornis	0	0	0	0	0	0	0	0	0	0	0
Sericostomatidae	0	0	0	0	0	0	0	0	0	0	0
Sericostoma personatum	12	5	5	3	0	5	3	2	2	0	0
Chironomidae	0	0	0	0	0	0	0	0	0	0	0

Appendix 5 Raw baseline seasonal survey data obtained from i) Kick sampling and ii) Surber sampling of the contemporary macroinvertebrate community of the River Wensum.

i) Seasonal Kick Data – Summer 2010

	Upstream top	**Upstream middle**	**Upstream bottom**
Theodoxus fluviatilis	0	1	0
Valvata cristata	0	0	1
Valvata piscinalis	0	0	7
Potamopyrgus antipodarum	287	36	39
Acroloxus lacustris	4	0	0
Zonitoides nitidus	0	0	1
OLIGOCHAETA	4	7	4
Glossophanic complanata	2	0	0
Erpobdella octoculata	8	1	7
Asellus aquaticus	0	0	1
Gammarus pulex	0	132	482
Baetis rhodoni	23	11	3
Ephemeralla ignita	137	23	37
Ephemera danica	49	44	21
Caenis luctuosa	0	0	23
Calopteryx vigro	0	2	5
Nebrioporus depressus	0	0	2
Dytiscidae larvae	0	0	1
Elmis aenea (adult)	47	0	0
*Elmis aenea (*larvae*)*	13	0	0
Oulimnius tuberculatus	0	1	0
Rhyacaphila dorsalis	0	1	0
Hydropsyche pellucidula	7	0	0
Limephilus lunatus	0	4	0
Anabolia nervosa	2	5	2
Halesus radiatus	0	0	1
Mystacides longicornis	0	0	2
Lepidostoma hirtum	0	5	5
Sericostoma personatum	16	14	0
Chironomidae	0	6	10
Simuliidae	1	0	0
Dinocrota	86	49	27

i) Seasonal Kick Data continued – Winter 2010

	Upstream top	Upstream middle	Upstream bottom	Top meander
Valvata cristata	0	1	0	1
Valvata piscinalis	0	1	0	0
Potamopyrgus antipodarum	671	86	81	0
Lymnaea stagnalis	0	1	0	0
Acroloxus lacustris	4	0	0	0
Zonitoides nitidus	0	1	0	0
Sphaeriidae	8	1	0	0
OLIGOCHAETA	24	15	28	3
Piscicola geometra	2	0	2	0
Glossophanic complanata	12	1	2	0
Erpobdella octoculata	17	3	2	0
Asellus aquaticus	5	5	5	0
Gammarus pulex	577	84	603	1
Baetis rhodoni	4	4	3	0
Ephemera danica	48	11	21	0
Caenis luctuosa	9	5	4	0
Nemouridae	3	0	2	0
Calopteryx vigro	1	0	1	0
Elmis aenea (adult)	4	2	0	0
Elmis aenea (larvae)	98	31	5	0
Oulimnius tuberculatus	29	12	2	0
Polycentropus flavomaculatus	0	0	2	0
Hydropsyche pellucidula	8	0	4	0
Athripsodes aterrimus	1	0	0	0
Silo nigricornis	0	1	0	0
Chironomidae	7	1	3	0
Simuliidae	0	2	0	0
Dinocrota	0	0	4	0

i) Seasonal Kick Data continued – Winter 2011

	Upstream top	Top meander	Side meander
Valvata cristata	4	0	1
Valvata piscinalis	2	0	0
Potamopyrgus antipodarum	532	3	6
Bithynia tentaculata	5	0	0
Acroloxus lacustris	4	0	0
Sphaeriidae	6	0	0
OLIGOCHAETA	35	4	4
Glossophanic complanata	16	0	0
Erpobdella octoculata	33	0	0
Asellus aquaticus	23	0	5
Gammarus pulex	1497	15	272
Baetis rhodoni	76	26	185
Ephemera danica	39	0	1
Caenis luctuosa	75	0	0
Nemouridae	1	0	0
Elmis aenea (adult)	57	0	0
Elmis aenea (larvae)	112	0	7
Nebrioporus depressus	3	0	0
Gyrinidae larvae	1	0	3
Rhyacaphila dorsalis	5	1	2
Polycentropus flavomaculatus	15	0	0
Hydropsyche pellucidula	28	0	5
Halesus radiatus	1	0	0
Athripsodes albifrons	2	0	0
Athripsodes bilineatus	7	0	0
Chironomidae	29	17	56
Simuliidae	36	2298	1116
Dinocrota	34	0	4

i) Seasonal Kick Data continued – Summer 2011

	Upstream top	Upstream bottom	Top meander	Side meander
Valvata cristata	3	2	2	0
Valvata piscinalis	2	3	0	0
Potamopyrgus antipodarum	797	981	388	750
Bithynia tentaculata	0	2	0	0
Lymnaea stagnalis	0	0	1	1
Acroloxus lacustris	1	0	0	0
Sphaeriidae	9	7	0	0
OLIGOCHAETA	31	14	17	4
Glossophanic complanata	6	5	0	0
Erpobdella octoculata	5	2	0	0
Asellus aquaticus	1	7	19	0
Gammarus pulex	222	396	303	871
Baetis rhodoni	13	11	60	26
Ephemera danica	6	13	8	7
Ephemeralla ignita	18	17	87	33
Caenis luctuosa	9	24	29	0
Nemouridae	0	0	11	0
Calopteryx vigro	1	4	0	0
Halipladae larvae	0	0	7	0
Elmis aenea (adult)	4	0	0	2
Elmis aenea (larvae)	13	7	7	9
Oulimnius tuberculatus (adult)	0	0	0	1
Oulimnius tuberculatus (larvae)	0	3	0	6
Rhyacaphila dorsalis	0	0	3	0
Polycentropus flavomaculatus	3	5	0	2
Hydropsyche pellucidula	2	0	3	4
Phryganea bipunctata	1	0	0	0
Anabolia nervosa	11	5	0	0
Athripsodes bilineatus	18	0	0	0
Mystacides longicornis	2	5	0	0
Silo nigricornis	1	0	0	0
Lepidostoma hirtum	4	3	0	1
Sericostoma personatum	5	3	0	11
Chironomidae	16	88	28	22
Dinocrota	4	4	127	144

i) Seasonal Kick Data continued – Winter 2011

	Upstream top	Upstream bottom	Top meander	Side meander
Potamopyrgus antipodarum	311	355	757	597
Anisus leucostoma	0	3	0	0
Zonitoides nitidus	2	0	0	0
Sphaeriidae	0	2	0	0
OLIGOCHAETA	31	27	23	0
Glossophanic complanata	0	9	0	0
Erpobdella octoculata	0	6	2	0
Asellus aquaticus	0	6	6	2
Gammarus pulex	419	236	207	573
Ephemera danica	7	9	9	3
Caenis luctuosa	0	0	0	2
Caenis robusta	0	2	0	0
Calopteryx vigro	0	3	0	0
Elmis aenea	3	0	21	13
Oulimnius tuberculatus	1	0	9	5
Polycentropus flavomaculatus	0	0	0	10
Hydropsyche pellucidula	0	2	12	5
Phryganea grandis	2	0	0	0
Chironomidae	22	47	27	12
Simuliidae	0	5	16	0

ii) Summer 2010 Surber counts

	Top of Contemporary Reach					Middle of Contemporary Reach					Bottom of Contemporary Reach				
	1	**2**	**3**	**4**	**5**	**1**	**2**	**3**	**4**	**5**	**1**	**2**	**3**	**4**	**5**
Theodoxus fluviatilis	0	0	0	0	0	1	0	0	0	1	0	0	0	0	0
Valvata cristata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Valvata piscinalis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Potamopyrgus antipodarum	21	13	41	28	9	60	0	30	1	3	0	16	1	2	21
Zonitoides nitidus	2	1	0	0	0	0	0	0	0	0	2	1	0	0	0
OLIGOCHAETA	0	0	3	2	0	2	1	2	4	5	1	5	0	2	3
Glossophanic complanata	1	2	0	0	0	0	0	0	0	0	0	0	0	0	0
Erpobdella octoculata	1	4	0	0	0	0	0	0	1	0	2	0	0	0	0
Asellus aquaticus	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
Gammarus pulex	0	0	0	0	0	21	9	5	98	14	34	2	35	27	16
Baetis rhodoni	0	0	2	0	0	2	0	3	0	0	0	1	0	1	0
Ephemeralla ignita	14	3	9	4	7	13	0	6	4	3	4	14	12	6	1
Ephemera danica	5	0	7	11	0	7	19	7	2	3	2	2	2	3	2
Caenis luctuosa	0	0	0	0	0	0	0	0	0	0	0	3	0	1	1
Elmis aenea (adult)	0	3	0	4	1	0	0	0	0	0	0	0	0	0	0
Elmis aenea (larvae)	0	1	0	1	0	2	0	0	0	0	0	0	1	0	1
Rhyacaphila dorsalis	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
Polycentropus flavomaculatus	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hydropsyche pellucidula	0	0	0	0	0	1	0	0	0	0	0	0	0	0	1
Anabolia nervosa	0	1	0	0	0	2	0	0	2	0	0	0	0	0	0
Athripsodes bilineatus	0	0	0	0	0	0	2	0	1	4	0	0	0	0	0
Mystacides longicornis	2	0	1	0	0	0	0	0	1	0	0	0	0	0	0
Goera pilosa	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0
Silo nigricornis	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0
Lepidostoma hirtum	1	0	1	0	0	0	0	0	3	0	1	0	0	0	0
Sericostoma personatum	1	0	0	0	1	1	0	3	2	0	4	1	3	0	5
Chironomidae	0	0	0	0	0	0	1	1	0	1	2	1	0	0	0
Dinocrota	2	2	1	5	7	20	12	5	2	1	7	7	6	2	10

ii) Winter 2010 Surber counts

	Top of Contemporary Reach					Middle of Contemporary Reach				
	1	**2**	**3**	**4**	**5**	**1**	**2**	**3**	**4**	**5**
Valvata cristata	0	0	0	0	0	0	1	0	0	0
Valvata piscinalis	0	0	3	0	0	0	1	1	0	0
Potamopyrgus antipodarum	81	304	41	97	54	0	2	153	10	28
Lymnaea stagnalis	0	0	1	0	0	0	0	0	0	0
Planorbis albus	0	0	0	0	0	0	0	0	0	0
Planorbis carinatus	0	0	0	0	0	0	0	0	0	0
Acroloxus lacustris	0	2	0	0	0	0	0	0	0	3
Sphaeriidae	0	3	0	0	0	1	0	1	0	3
OLIGOCHAETA	9	18	5	1	2	4	4	5	0	27
Glossophanic complanata	1	3	0	5	1	0	0	2	1	3
Erpobdella octoculata	1	4	3	3	2	0	0	1	2	1
Asellus aquaticus	0	0	3	1	3	0	0	3	1	1
Gammarus pulex	76	3	103	82	57	70	22	110	52	55
Baetis rhodoni	1	0	0	1	0	0	0	0	1	0
Ephemera danica	5	9	8	2	3	1	1	9	3	4
Caenis luctuosa	6	0	0	0	1	0	0	1	1	1
Nemouridae	0	0	0	0	0	0	0	0	0	0
Elmis aenea (adult)	0	0	0	0	3	0	0	0	0	0
Elmis aenea (larvae)	4	9	3	2	8	0	0	5	1	0
Oulimnius tuberculatus	3	2	0	0	0	0	0	0	0	0
Polycentropus flavomaculatus	1	2	0	0	0	0	0	0	0	1
Hydropsyche pellucidula	2	3	0	2	1	0	0	1	1	1
Athripsodes aterrimus	5	0	0	0	2	0	0	0	0	0
Silo nigricornis	0	0	0	1	3	0	0	0	0	0
Chironomidae	0	3	5	0	0	1	0	1	0	0
Simuliidae	0	0	0	0	0	0	0	0	0	0

ii) Winter 2010 Surber counts continued.

	Bottom of Contemporary Reach					Top of Reconnected Meander				
	1	**2**	**3**	**4**	**5**	**1**	**2**	**3**	**4**	**5**
Valvata cristata	0	0	0	0	0	0	0	0	0	0
Valvata piscinalis	0	0	3	1	1	0	0	0	0	0
Potamopyrgus antipodarum	41	64	12	14	20	0	0	0	0	0
Lymnaea stagnalis	0	2	0	0	0	0	0	0	0	0
Planorbis albus	0	0	1	0	0	0	0	0	0	0
Planorbis carinatus	0	0	1	0	0	0	0	0	0	0
Acroloxus lacustris	0	0	0	0	0	0	0	0	0	0
Sphaeriidae	0	0	0	0	0	0	0	0	0	0
OLIGOCHAETA	0	3	17	24	32	0	0	1	1	2
Glossophanic complanata	0	1	1	2	1	0	0	0	0	0
Erpobdella octoculata	0	6	1	2	0	0	0	0	0	0
Asellus aquaticus	0	7	4	7	2	0	0	0	1	0
Gammarus pulex	43	155	12	161	24	0	0	0	0	0
Baetis rhodoni	0	1	0	0	0	0	1	0	0	0
Ephemera danica	1	9	4	3	5	0	0	0	0	0
Caenis luctuosa	0	0	2	3	0	0	0	0	0	0
Nemouridae	0	1	0	0	0	0	0	0	0	0
Elmis aenea (adult)	0	0	0	0	0	0	0	0	0	0
Elmis aenea (larvae)	2	3	0	0	0	0	0	0	0	0
Oulimnius tuberculatus	0	1	0	0	0	0	0	0	0	0
Polycentropus flavomaculatus	0	0	0	2	0	0	0	0	0	0
Hydropsyche pellucidula	3	2	2	0	0	0	0	0	0	0
Athripsodes aterrimus	0	0	0	0	0	0	0	0	0	0
Silo nigricornis	0	0	0	0	0	0	0	0	0	0
Chironomidae	0	0	1	0	1	0	0	0	0	0
Simuliidae	0	0	0	0	0	0	0	0	0	2

ii) Winter 2011 Surber counts

	Top of Contemporary Reach					Top of Reconnected Meander					Side of Reconnected Meander				
	1	**2**	**3**	**4**	**5**	**1**	**2**	**3**	**4**	**5**	**1**	**2**	**3**	**4**	**5**
Valvata cristata	0	0	0	0	1	1	0	0	0	0	0	0	1	0	0
Valvata piscinalis	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Potamopyrgus antipodarum	37	59	131	21	53	0	0	1	0	0	1	0	3	1	1
Bithynia tentaculata	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0
Planorbis levcostoma	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0
Acroloxus lacustris	0	2	0	1	0	0	0	0	0	0	0	0	0	0	0
OLIGOCHAETA	4	11	16	6	3	0	1	0	1	0	0	0	0	1	0
Glossophanic complanata	2	0	4	0	2	0	0	0	0	0	0	0	0	0	0
Erpobdella octoculata	0	1	5	2	0	0	0	1	0	0	0	0	0	0	0
Asellus aquaticus	2	2	6	4	1	0	0	0	0	0	1	0	0	0	0
Gammarus pulex	52	43	146	31	101	3	0	1	3	1	44	19	63	30	4
Baetidae (cloeon)	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0
Baetis rhodoni	0	1	2	0	4	5	0	4	13	2	20	4	22	13	1
Ephemera danica	4	0	14	13	2	0	2	3	0	0	0	0	0	0	0
Caenis luctuosa	3	1	4	0	2	0	0	0	0	0	0	0	0	0	0
Nemouridae	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0
Calopteryx vigro	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Elmis aenea (adult)	2	0	1	7	0	0	0	0	0	0	0	0	0	0	0
Elmis aenea (larvae)	4	2	5	1	5	2	1	0	0	1	1	3	1	0	1
Gyrinidae larvae	0	0	0	0	0	0	0	1	1	0	1	1	0	0	0
Rhyacaphila dorsalis	0	0	0	0	1	0	0	0	0	0	0	0	0	1	1
Polycentropus flavomaculatus	0	8	0	9	2	0	0	0	0	0	0	0	0	0	0
Hydropsyche pellucidula	1	7	3	0	3	0	0	0	0	0	4	0	0	0	0
Athripsodes albifrons	1	0	6	0	0	0	0	0	0	0	0	0	0	0	0
Athripsodes bilineatus	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0
Goera pilosa	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Chironomidae	1	6	8	6	3	18	14	16	32	9	7	14	9	15	6
Simuliidae	0	2	6	0	2	244	49	145	260	145	183	298	346	22	1
Dinocrota	1	1	8	0	1	0	0	0	2	1	2	0	1	0	0

ii) Summer 2011 Surber counts

	Top of Contemporary Reach					Bottom of Contemporary Reach				
	1	**2**	**3**	**4**	**5**	**1**	**2**	**3**	**4**	**5**
Valvata cristata	0	4	0	1	2	0	0	0	0	0
Valvata piscinalis	0	2	0	0	2	0	0	0	0	2
Potamopyrgus antipodarum	301	297	39	518	362	22	288	178	53	77
Lymnaea peregra	0	0	0	1	0	0	0	0	0	0
Lymnaea stagnalis	0	0	0	0	0	0	0	1	0	0
Acroloxus lacustris	1	1	0	0	0	0	1	0	0	0
Zonitoides nitidus	1	1	0	2	0	0	0	1	0	0
Sphaeriidae	2	2	0	0	1	0	0	0	1	1
OLIGOCHAETA	17	63	42	29	25	6	4	24	23	20
Glossophanic complanata	4	7	0	3	0	0	1	6	0	3
Erpobdella octoculata	1	2	1	0	4	0	0	0	0	3
Asellus aquaticus	1	8	1	3	0	1	0	6	1	1
Gammarus pulex	10	184	9	91	177	7	17	105	16	45
Baetis rhodoni	5	8	0	3	4	2	1	6	2	2
Ephemera danica	3	1	3	1	2	3	2	2	9	8
Ephemeralla ignita	7	14	8	4	13	3	2	4	5	3
Caenis luctuosa	12	4	2	11	0	0	1	5	9	4
Nemouridae	0	2	0	0	0	0	1	0	0	0
Calopteryx vigro	0	0	0	3	0	0	0	5	0	0
Halipladae larvae	0	0	0	0	0	0	0	0	0	0
Elmis aenea (adult)	0	4	0	0	1	0	2	0	0	0
Elmis aenea (larvae)	7	12	3	4	5	3	2	0	0	0
Oulimnius tuberculatus (adult)	2	12	0	1	4	0	1	0	0	0
Oulimnius tuberculatus (larvae)	0	0	0	0	0	1	0	0	0	0
Rhyacaphila dorsalis	0	0	0	0	0	0	0	0	0	0
Polycentropus flavomaculatus	0	4	0	1	1	1	0	0	0	0
Hydropsyche pellucidula	0	1	0	0	0	0	0	0	0	0
Anabolia nervosa	2	9	4	1	4	0	0	1	0	4
Halesus radiatus	0	5	0	0	0	0	0	0	0	0

ii) Summer 2011 Surber counts continued.

	Top of Contemporary Reach					Bottom of Contemporary Reach				
	1	2	3	4	5	1	2	3	4	5
Athripsodes bilineatus	0	0	2	1	0	0	0	0	0	0
Mystacides longicornis	2	0	0	0	0	0	0	0	0	1
Silo nigricornis	0	0	0	0	0	0	5	0	8	0
Lepidostoma hirtum	0	4	0	5	1	0	0	1	0	1
Sericostoma personatum	2	10	0	6	4	0	0	0	21	4
Chironomidae	2	8	2	7	0	1	0	18	17	43
Simuliidae	0	0	0	0	0	0	0	3	0	0
Tipulidae	0	0	1	0	0	0	0	0	0	0
Dinocrota	0	2	0	1	0	1	0	0	0	0

ii) Summer 2011 Surber counts continued.

	Top of Reconnected Meander					Side of Reconnected Meander				
	1	2	3	4	5	1	2	3	4	5
Valvata cristata	0	0	0	0	0	0	2	0	0	0
Valvata piscinalis	0	0	0	0	0	0	0	0	0	0
Potamopyrgus antipodarum	34	10	74	47	26	389	67	112	48	81
Lymnaea peregra	1	0	0	0	0	1	0	0	0	0
Lymnaea stagnalis	0	0	0	0	0	0	0	0	0	0
Acroloxus lacustris	0	0	0	0	0	0	0	0	0	0
Zonitoides nitidus	0	0	0	0	0	0	0	0	0	0
Sphaeriidae	0	0	0	0	0	0	0	0	0	0
OLIGOCHAETA	11	28	8	1	2	2	1	0	2	0
Glossophanic complanata	0	0	0	0	0	0	0	0	0	0
Erpobdella octoculata	0	0	0	0	0	0	0	0	0	0
Asellus aquaticus	5	26	1	0	1	0	2	0	0	0
Gammarus pulex	173	102	31	10	17	172	166	151	208	6
Baetis rhodoni	56	9	5	6	7	8	14	17	31	1

ii) Summer 2011 Surber counts continued.

	Top of Reconnected Meander					Side of Reconnected Meander				
	1	**2**	**3**	**4**	**5**	**1**	**2**	**3**	**4**	**5**
Ephemera danica	0	2	1	0	0	4	3	1	0	2
Ephemeralla ignita	49	24	11	9	4	6	5	9	0	2
Caenis luctuosa	0	0	9	4	2	0	0	0	2	4
Nemouridae	5	13	5	0	0	0	0	0	0	0
Calopteryx vigro	0	0	0	0	0	0	0	0	0	0
Halipladae larvae	3	0	0	0	0	0	1	0	0	0
Elmis aenea (adult)	3	0	0	0	0	1	0	3	2	0
Elmis aenea (larvae)	3	0	4	0	2	0	0	4	3	0
Oulimnius tuberculatus (adult)	0	0	0	0	0	0	0	0	0	0
Oulimnius tuberculatus (larvae)	0	0	1	0	0	1	1	2	0	0
Rhyacaphila dorsalis	1	0	0	0	0	0	0	1	0	0
Polycentropus flavomaculatus	3	1	1	0	0	0	1	0	1	0
Hydropsyche pellucidula	2	0	0	0	0	0	1	0	1	0
Anabolia nervosa	0	0	0	0	0	0	0	0	0	0
Halesus radiatus	0	0	0	0	0	0	0	0	0	1
Athripsodes bilineatus	2	0	2	0	0	0	0	0	0	0
Mystacides longicornis	0	0	0	0	1	0	0	0	0	0
Silo nigricornis	0	0	0	0	0	0	0	0	0	0
Lepidostoma hirtum	0	0	0	0	0	0	0	0	0	0
Sericostoma personatum	0	0	0	0	0	0	0	0	1	1
Chironomidae	8	3	10	1	4	5	12	16	4	1
Simuliidae	2	0	0	0	0	0	0	0	0	0
Tipulidae	0	0	0	0	0	0	0	0	0	0
Dinocrota	12	29	61	7	10	29	12	9	15	1

ii) Winter 2011 Surber counts.

	Top of Contemporary Reach					Bottom of Contemporary Reach				
	1	**2**	**3**	**4**	**5**	**1**	**2**	**3**	**4**	**5**
Valvata piscinalis	0	0	0	0	0	0	0	0	0	0
Potamopyrgus antipodarum	270	263	141	146	207	210	201	132	129	187
Lymnaea peregra	0	0	0	1	0	0	0	0	0	0
Anisus leucostoma	0	0	0	0	0	0	0	0	0	2
Acroloxus lacustris	0	0	0	0	0	0	0	0	0	2
Zonitoides nitidus	0	1	0	0	1	2	0	0	0	0
Sphaeriidae	0	0	1	1	0	0	0	0	0	1
OLIGOCHAETA	11	7	20	31	14	17	27	28	25	3
Glossophanic complanata	3	2	0	4	5	0	0	0	0	0
Erpobdella octoculata	5	1	0	0	2	0	0	0	0	0
Asellus aquaticus	3	0	1	5	2	0	0	0	0	0
Gammarus pulex	115	121	67	106	119	211	86	106	163	170
Baetis rhodoni	1	1	0	0	0	0	0	0	0	0
Ephemera danica	4	5	2	3	1	3	0	4	0	0
Ephemeralla ignita	0	0	0	0	0	0	0	0	0	0
Caenis robusta	0	0	0	0	1	0	0	0	0	0
Halipladae larvae	0	0	0	0	0	0	0	0	0	0
Dytiscidae larvae	1	1	0	0	0	0	0	0	0	0
Gyrinidae larvae	0	0	0	0	0	0	0	0	0	0
Elmis aenea (adult)	0	0	0	0	0	0	0	0	0	0
Elmis aenea (larvae)	6	4	6	2	2	0	0	1	0	3
Oulimnius tuberculatus (larvae)	3	2	2	0	2	0	0	0	0	0
Polycentropus flavomaculatus	1	0	0	0	0	0	0	0	0	0
Hydropsyche pellucidula	0	0	0	0	0	1	0	0	0	0
Hydroptila sp	0	0	0	0	0	0	0	0	0	0
Silo nigricornis	0	0	0	0	0	0	0	0	0	0
Chironomidae	8	24	13	19	38	9	18	5	11	3
Simuliidae	4	1	0	7	9	0	0	0	0	0
Dinocrota	0	0	1	0	0	0	0	0	0	0

ii) Winter 2011 Surber counts continued.

	Top of Reconnected Meander					Side of Reconnected Meander				
	1	**2**	**3**	**4**	**5**	**1**	**2**	**3**	**4**	**5**
Valvata piscinalis	1	0	0	0	0	0	0	0	0	0
Potamopyrgus antipodarum	456	139	323	127	121	173	233	361	402	186
Lymnaea peregra	2	0	0	0	0	0	0	0	0	0
Anisus leucostoma	0	0	0	0	0	0	0	0	0	0
Acroloxus lacustris	0	0	0	0	0	0	0	0	3	0
Zonitoides nitidus	0	0	0	0	0	0	0	0	0	1
Sphaeriidae	0	0	0	0	0	0	0	0	0	0
OLIGOCHAETA	8	2	12	2	4	0	0	0	0	2
Glossophanic complanata	0	0	0	0	0	0	0	0	0	0
Erpobdella octoculata	1	0	0	0	0	0	0	0	0	0
Asellus aquaticus	4	0	3	0	0	0	0	0	0	0
Gammarus pulex	36	132	141	117	115	27	212	41	310	18
Baetis rhodoni	0	0	0	0	0	0	0	0	0	0
Ephemera danica	0	0	4	3	1	0	0	0	0	0
Ephemeralla ignita	2	0	0	0	0	0	0	1	0	0
Caenis robusta	0	0	0	0	0	0	0	0	0	0
Halipladae larvae	0	0	1	0	0	0	0	0	0	1
Dytiscidae larvae	0	0	0	0	0	0	0	0	0	0
Gyrinidae larvae	0	0	0	0	1	0	0	1	0	0
Elmis aenea (adult)	0	0	0	0	1	0	0	0	0	0
Elmis aenea (larvae)	9	4	10	7	6	53	4	18	3	2
Oulimnius tuberculatus (larvae)	0	0	4	5	2	0	2	7	0	0
Polycentropus flavomaculatus	2	1	0	0	0	0	0	0	0	0
Hydropsyche pellucidula	0	1	0	2	1	2	6	8	1	0
Hydroptila sp	3	0	0	1	0	0	0	0	0	0
Silo nigricornis	0	0	0	0	0	0	0	2	1	0
Chironomidae	0	4	0	0	7	0	0	0	13	4
Simuliidae	7	10	1	2	0	0	2	0	0	0
Dinocrota	0	0	4	0	2	0	1	2	0	0

Appendix 6 The palaeoecological macroinvertebrate community of the River Wensum extracted from the palaeo channel.

	50-60	60-70	70-80	100-110	Gravel Section
Valvata piscinalis	0	0	1	2	19
Bithynia tentaculata	0	0	0	1	14
Lymnaea palustris	1	0	0	0	6
Lymnaea peregra	5	43	18	0	56
Lymnaea stagnalis	0	3	1	0	0
Planorbarius corneus	0	1	0	0	2
Planorbis carinatus	1	8	1	1	21
Anisus vortex	0	0	0	0	24
Bathyomphalus contortus	0	0	0	0	5
Zonitoides nitidus	0	0	0	0	8
Sphaeriidae	21	22	8	3	42
Sialis lutaria	12	25	7	2	19
Brychius elevatus	0	0	0	0	2
Halipladae	3	5	0	0	10
Nebrioporus depressus	3	2	1	0	4
Helophorus	0	3	0	0	3
Elmis aenea	6	4	2	0	11
Oulimnius	2	0	0	0	3
Polycentropus flavomaculatus	0	0	2	0	0
Hydropsyche pellucidula	0	7	2	0	3
Hydropsyche angustipennis	17	21	6	8	31
Anabolia nervosa	3	5	2	0	2
Potamophylax cingulatus	0	5	4	2	7
Halesus radiatus	0	7	1	0	1
Molanna angustata	3	10	0	0	7
Athripsodes aterrimus	0	2	2	0	3
Athripsodes bilineatus	0	1	1	0	5
Goera pilosa	3	0	0	0	2
Sericostoma personatum	0	0	0	0	2