AN INVESTIGATION INTO THE EFFECTS OF CATCHMENT PROCESSES ON THE WATER QUALITY OF SOUTHERN CHALK RIVERS

by

ALISON GAIL ARBUTHNOTT B.Sc. (Hons), M.Sc.

Centre for Environmental Sciences,
University of Southampton

This thesis is submitted in fulfillment
of the degree of Doctor of Philosophy

August 2001

UNIVERSITY OF SOUTHAMPTON

ABSTRACT

FACULTY OF SCIENCE

CENTRE FOR ENVIRONMENTAL SCIENCES

Doctor of Philosophy

AN INVESTIGATION INTO THE EFFECTS OF CATCHMENT PROCESSES ON THE WATER QUALITY OF SOUTHERN CHALK RIVERS

By Alison Gail Arbuthnott

The physical, chemical and biological characteristics of five southern English chalk streams from neighbouring catchments were investigated seasonally over two years. Stream characteristics were studied at both a catchment scale and at a local, reach scale.

At the catchment scale, the five chalk rivers had very similar physico-chemical properties. Differences between years and between seasons were much greater than those between rivers or within-river longitudinal differences. Elevated inorganic nutrient concentrations, relative to 'pristine' systems, indicated some degree of catchment water quality deterioration in all five rivers.

The effect of physical habitat degradation (channel overwidening and sedimentation) on biological river 'health' was investigated at a within-river, reach scale. Overwidening was found to have little impact on benthic invertebrates over the two years. In contrast, the increased proportion of fine sediment within the coarse gravel substratum had a marked effect on invertebrate communities. Communities showed a high degree of stability between years, and between individual catchments, reflecting the overall biological stability of these systems.

Manipulation experiments at a smaller, within-reach scale demonstrated the importance of substratum characteristics to macroinvertebrate communities. Invertebrate colonisation over time of newly exposed substrata was quantitatively and qualitatively different for coarse and fine particle sizes, interpreted as a function of shifting resource utilisation and microhabitat preferences. The relatively long time taken to fully colonise new substrata demonstrated the weak ability of chalk stream invertebrate communities to respond rapidly to changes in habitat – a reflection of the stable physical nature of natural chalk streams and the vulnerability of their communities to anthropogenic disturbance.

The implications of these findings for the management of chalk rivers at hierarchical spatial and temporal scales is discussed, within a framework of water quality enhancement.

LIST OF CONTENTS

LIST OF FIGURES

Page number

LIST OF TABLES

LIST OF PLATES

ACKNOWLEDGEMENTS

I would like to thank Dr. Peter Shaw for his very great help, time and patience and the odd pint or two of beer. Thanks are also due to Professor Steve Hawkins for his continued support and encouragement. I am extremely grateful to the Potter family trust for their financial support, and I must also thank Mr. and Mrs. Bray for their help and assistance during fieldwork.

Many thanks are also due to all my colleagues in the Centre of Environmental Sciences, University of Southampton, including Jacqui, for her endless patience and wit during the tough times (of which there were many), Dawn and Cathy for putting up with my random requests, Mal who was going through the same trials and tribulations as me, Terry for all the cups of tea, Simon Kemp for putting up with me both at work and at home (not easy I must admit), and Simon Bray for his all his help in the field and belief that I would finish one day.

A big thank you all my friends for making my time in Southampton so great: Nat, Sally, Airasa, Gary, Isabel, Jane, Danni, Maggie, Anneke, Dean, Lee Ann and Kevin. Sorry to all of those I have forgotten.

Many thanks must also go to my parents, as without both their financial support and belief in me, this thesis would never have been completed. Also, thank you mum for reading everything that I wrote, it must have been love. I would also like to thank Simon Harrison for his constant encouragement. Thank you for living every moment with me, again it must have been love. The dark days have come to an end and your second PhD is finished.

Lastly, I would like to remember Rob. Although he isn't here to call me Doctor, I like to think he isn't far away.

CHAPTER ONE: INTRODUCTION

CHAPTER ONE: INTRODUCTION

1.1 RIVER SYSTEMS AND MANAGEMENT

1.1.1 River ecosystems

Throughout history, humans have had a close association with rivers, and it is widely accepted that river ecosystems have been utilised to a greater extent than any other natural environmental system (Boon 1992). Intensive agriculture, industrial development, fisheries and tourism have altered the size, shape, course and content of many of our major rivers (Angermeir and Bailey 1992).

Exploitation of any resource inevitably leads to changes in the surrounding environments. However, in an aquatic context, the effects of man's interference often do not occur at the point of alteration, thus making it difficult to relate effects to possible causes. Societal demands made upon many water resources are often in direct conflict with the supply capabilities of these systems. In this sphere of conflicting interests, the need for intervention through resource management has become evident (Van Stokkom *et al.* 1993). It can therefore be concluded that rivers are highly complex and variable natural systems that pose a major challenge to both scientific understanding and management (Cannan 1998).

The intensity and distribution of catchment activities ultimately dictate the direction of water resource management through their impact on water quality. Understanding the environmental controls on water quality is recognised as an important consideration in the management of rivers and their catchments (Mullholland and Hill 1997). As awareness of the importance of aquatic environments increases and recognition of the anthropogenic perturbations being imposed upon these systems becomes widespread, much research is being directed to understanding the impacts of human activity on riverine ecosystems.

1.1.2 River management

Land and water constitute two of the basic resources in the life-support system on which humans depend (Lundqvist *et al.* 1985, Mellquist 1992). With increasing standards of living, growing populations and concentrations of anthropogenic activities, the pressures on global and local water resources are becoming intensified. Concerns about the environmental impacts associated with the development of these activities have grown rapidly in recent decades (Pugh 1997).

The most noticeable response in this respect has been in the increasing attention paid to the management of natural environmental resources. Management seems to be perceived as a means by which resource utilisation in the contemporary setting is assured and which encompasses the necessary conservation of the productivity of resources on a long-term basis (Lundqvist *et al.* 1985). In particular water management has become an issue of high priority (Loucks 2000).

The management of a river or river system in its simplest form could be the movement of a few stones in a river bed in order to improve the passage of water, fish or timber, and the subsequent maintenance of these changes over a period of time (Mellquist 1992). At the other end of the spectrum is a complete alteration to the hydrological and ecological conditions of particular stretches/entire length of a river system. Ideally the objectives of river management should include:

- Balancing users' interests
- Optimising the use of resources (e.g. abstraction, recreation, industrial)
- Recognition of environmental interests and those of the general public when exploiting water resources
- Cleaning up after previous activities (Ten Brink and Woudstra 1991, Loucks 2000)

It has been recognised that problems will not be solved in the channel alone because these issues, for the most part, originate in the rivers' catchment areas. This recognition has permeated through to scientific and management communities and recently there has been an increase in the study of river systems at the catchment scale (Cannan 1998).

1.1.3 Holistic catchment research

A river can be thought of as a transport system for waters, sediments, solutes and biota (Newson 1992). This broad definition encompasses many complex and interactive processes, operating at a variety of scales. In response to their inherent variability and complexity, numerous scientific disciplines have been engaged in the investigation of these systems, including, for example, geomorphologists, hydrologists, engineers and ecologists. However, in the early stages of research on lotic systems, little communication and discussion of ideas took place between these disciplines. This division had the effect that each discipline developed its own approach to the study of river environments in isolation (Cannan 1998).

Consequently, in the past, there have been few attempts to treat streams and rivers holistically (Minshall *et al.* 1983). Indeed, a truly holistic view of streams did not emerge until the late 1950s with the recognition that stream biota were influenced by the surrounding landscape (Ross 1963, Hynes 1975). There has been a subsequent shift in perspective from that of viewing streams or their component stretches as individual entities to that of a synthetic (integrated) view that seeks generalisations about streams as whole ecosystems (Minshall *et al.* 1985).

The precursor to the current holistic view of river ecosystems was the River Continuum Concept (RCC) developed by Vannote *et al.* (1980). The RCC proposed a synthetic view of stream ecosystems, with the antecedents emphasising the unifying aspects of flowing water (especially geomorphology and hydraulics) in structuring stream communities (Margalef 1960, Hynes 1963, Wallace *et al.* 1977, Elwood *et al.* 1983). Vannote *et al.* (1980) regarded the entire fluvial system as a continuously integrated series of physical gradients and associated biotic adjustments. This approach provided useful generalisations concerning temporal and spatial variations in the supply of energy, and biological community structures within rivers (Minshall *et al.* 1985).

It is now widely recognised by practitioners in all disciplines that it is far more beneficial to scientific understanding to consider together all the interacting hydrological, geomorphological and biological processes occurring within rivers (Cannan 1998). Petts (1994) divided the research on the functioning of river systems into five principles and summarised rivers as:

- three dimensional systems;
- driven by hydrology and fluvial geomorphology;
- structured by food webs;
- characterised by spiralling processes; and
- dependent upon change – changing flows, moving sediments and shifting channels.

Townsend (1996), in his catchment hierarchy approach, argued for an integration of existing river concepts with the patch dynamics concept, viewing streams as mosaics of patches (Pringle *et al.* 1988, Downes *et al.* 1993). An hierarchical framework of both river and patch dynamics concepts on the scale of the river catchment enables the prediction of spatial and temporal patterns of ecological variables in the river basin, which can be fed into the river management framework.

Utilising the holistic view of rivers provides a pathway towards integrated river basin management (IRBM). IRBM requires an holistic view not only of the river systems themselves, but also of the sources and processes causing water quality problems throughout the surrounding areas (Pegram and Bath 1995). It therefore follows that not only do the rivers have to be regarded as a whole entity, but so do the catchments surrounding the rivers, giving rise to the study of rivers on a catchment scale (Petts 1994). The underlying principle of the catchment theory is that strong links exist between rivers and the land over which they flow (Cannan 1998). The catchment can therefore provide us with a natural and clearly defined geographical unit on which to base the study, and management, of rivers (Imhof *et al.* 1996, Cannan 1998). The increasing catchment pressures on rivers have led the authorities responsible for the management of these systems to adopt ‘sustainable management policies’ (Cannan 1998). For example, the Environment Agency (in the UK) is responsible for the sustainable management of many ecosystem functions, one of which being riverine environments (Petts *et al.* 1995, Cannan

1998). The link between river ecosystems and their surrounding drainage areas is clear, it follows that their management should be based on a catchment scale. Therefore an understanding of catchment processes and functions is fundamental to the efficient and effective management of lotic ecosystems (Cannan 1998).

1.2 WATER QUALITY

1.2.1 Factors affecting water quality

Society benefits immeasurably from rivers (Karr 1999). However, for centuries, UK freshwater resources have been taken for granted. Exploitation of rivers was fundamental to the Industrial Revolution for steam power, for canal transport, for industry, and for disposing of wastes (Boon and Howell 1997). Over the past 200 years the sustainability of the use of this resource has been under threat.

Chemically 'pure' water (water devoid of any dissolved substances) is virtually never found within the natural environment (UNESCO 1978). Hence, the term 'natural waters' (i.e. waters containing chemicals in sufficiently low concentrations to cause no impairment to the surrounding environments; Pagenkopf 1978) is more commonly associated with the monitoring of riverine environments. It has long been accepted that the water flowing in any stream channel has undergone many natural interactions before becoming a component of the running water. These interactions arise from a combination of hydrological, chemical, biological and physical factors, all of which exert an influence on the quality of water. Together with this, anthropogenic activities often influence the water quality of a riverine environment. The influence of anthropogenic disturbances upon aquatic systems are summarised in Figure 1.1. At one end of the continuum of human influence on the environmental condition of an environment, (Figure 1.1 (a)), severe disturbance can result in reductions in water quality, ultimately leading to extremely degraded systems, whilst at the other end of the gradient conditions can be pristine. A parallel gradient passes through healthy, or sustainable, condition or activities (Figure 1.1 (b)). Below a threshold defined by specific criteria, the conditions or activities are no longer healthy or sustainable in terms of supporting healthy systems (Karr 1999).

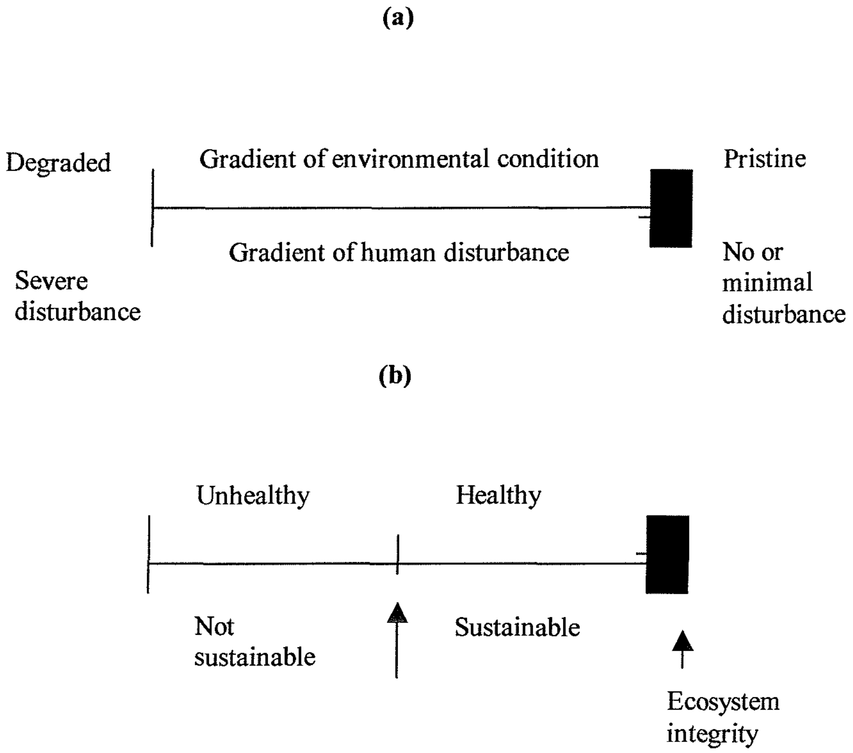


Figure 1.1. Continuum of human influence on environmental condition (adapted from Karr 1999).

The fundamentals of water quality monitoring have been recognised for some time. For example, Fish (1973) stated that the purpose of the assessment of water quality is to establish the present position and how this changes under the influence of controllable and uncontrollable events in the short and long term.

The term 'water quality' is liberally used throughout the literature as if a single concept of water quality were universally recognised (Robards *et al.* 1994). However, even though 'quality' is a widely used term, it is a difficult concept, at least in absolute terms, to explain (Hellawell 1997). It has been defined as '*that which makes a thing what it is, property, attribute, grade of goodness and excellence*' (Maitland 1997). Given that there are scores of elements associated with freshwater and many thousands of chemical components, let alone a myriad of biological species, to derive an absolute description of water quality is extremely difficult (Pugh 1997). Therefore there would appear to be no satisfactory, absolute measure of water quality other than the extent to which a sample deviates from pure dihydrogen oxide. In the field of quality management, quality has a more specific application, which may be defined as the '*degree to which a product is fitted to the purpose for which it is made*' (Pugh 1997). Quality used in this context is a relative rather than an absolute criterion and this pragmatic approach can be applied to the definition of water quality.

Water that is suitable for human consumption, for example, may be toxic to fish (Hellawell 1988). At the same time, water in which fish thrive may be completely unacceptable as a potable water supply. Therefore the definition of river water quality depends upon the use of the water. Definitions exist for potable water supplies, industrial processes, effluent disposal, fisheries, recreation, wildlife and nature conservation (Hellawell 1997, Pugh 1997). In general, the term "water quality" is used in monitoring surveys to give an indication of the condition/state of the water at the time of sampling (UNESCO 1978). As such, water quality guidelines have been used over the past years as an important component of the management of water resources (Johnes *et al.* 1996, Hart *et al.* 1999).

1.2.2 Methods of assessment

Physico-chemical methods

There is almost an infinite range of methods available for the assessment of river water quality. Historically, the monitoring assessment and regulation of aquatic ecosystems has been largely based on chemical measures of water quality (Metcalfe-Smith 1994). Chemical analyses can be undertaken on water samples in order to determine the quantities of specified substances (Hellawell 1997). The range of substances for which analytical methods are available is huge and increasing all the time, as is the accuracy with which these substances can be detected. Coupled with this, increases in technology have led to automation processes which enable a high number of samples to be analysed, thus increasing the frequency at which samples can be collected. If reliable chemical quality criteria for a particular use can be established and if the sampling regime is robustly designed, then chemical water quality assessment can be extremely efficient (Hellawell 1997). For example, many studies concerned with freshwater quality relate to chemistry and some limnologists would argue that good chemical data are adequate to categorise the quality of any water both for human needs and ecological purposes (Maitland 1997).

However, the current view in aquatic research is that chemical monitoring alone provides insufficient information and evidence to enable meaningful conclusions to be drawn and cannot provide a reliable assessment of the health of an aquatic system (Roux *et al.* 1993, Metcalfe-Smith 1994, Wright *et al.* 1994). Consequently, with the increasing importance of ecological conservation in catchment management, there has been an increase in the support for a biological approach to water quality assessment (Karr 1993), when previously biological methods only played a supporting role in the formal regulatory system (Pugh 1997).

Biological methods

Biological methods have been viewed as superior to chemical approaches, which, due to their snapshot nature, only reveal the environmental state at the moment of sampling (Hellawell 1997). Both the quality and quantity of flowing waters are highly variable and help shape the structure of communities found within and surrounding water bodies (UNESCO 1978). Biological communities are subject to the totality of chemical and physical influences, hence exhibit an integrated response to a fluctuating environment over time (Roux *et al.* 1993). Biological methods are regarded as being particularly relevant for assessing environmental water quality, particularly for those aspects of water which have ecological consequences which cannot be assessed readily in the laboratory from a limited number of samples (Hellawell 1997). It follows that any perturbation/disturbance imposed upon a system will ultimately impact upon the supply or quality of flowing waters, leading to a shift in the community structure. A monitoring of this shift will therefore enable an assessment to be made of an imposed stress or disturbance event over time.

Biological monitoring can be directed at a variety of levels, investigating individuals, populations and communities. The scales required by the investigation being undertaken determine the level chosen (Cannan 1998). It is possible to use any group of organisms to examine the biological condition of a river, and in Britain early attempts at detecting pollution used both faunal and floral data (Armitage *et al.* 1983, Cannan 1998). In an ideal situation it would be preferable to use many aspects of the biotic community of a river as indicators (Cannan 1998). However, this is often not practical. The biota occupying any particular water body are determined by a number of factors which have combined to make the specific environment what it is in the locality, i.e. to give it its own particular water quality (Pugh 1997). Whilst there are many components of the aquatic biota of a river, attention has been focused on macroinvertebrates as they are present in all but the most polluted rivers and are relatively easy to sample (Hellawell 1978, 1986, Logan and Brooker 1983, Norris 1995). Macroinvertebrates are those organisms inhabiting the bottom substrates of freshwater habitats for at least part of their life cycle (Cannan 1998). This group of organisms is generally regarded as being an important component in aquatic ecosystems, and provides one approach to studying the functioning of streams and rivers (Hildrew and Giller 1994, Cannan 1998).

Water quality monitoring using macroinvertebrates

Benthic macroinvertebrates are used widely in the biological assessments of freshwater quality principally because of their biology: they are ubiquitous, with a large number of species occupying most aquatic habitats and environments which results in a wide range of responses to environmental stresses (Cannan 1998). Their relatively long life cycles and sedentary nature mean that they provide an analysis of conditions over time relative to the site from which they were sampled (Cannan 1998). The taxonomy of many groups is well known, as are the responses they exhibit to stresses imposed (Hogg *et al.* 1995, Cannan 1998). A wide range of expertise has been gained on the techniques of collection and analysis, for which simple and inexpensive equipment can be used (Rosenberg and Resh 1993, Cannan 1998). Also an extensive knowledge of these organisms has been accumulated through a long history of research (Cannan 1998).

Research on these communities has been wide and varied. Much attention has been focused on patterns of distribution and abundance of individuals, and the underlying factors controlling these patterns (Armitage *et al.* 1983, Logan and

Brooker 1983). Biological studies open up the possibility of investigating temporal and spatial variation in the composition of macroinvertebrate communities and hence the water quality of riverine systems on a temporal and spatial scale (Jenkins *et al.* 1984). Through their time-integrative nature, these approaches enable the understanding of the responses of riverine communities to both natural and anthropogenic perturbations. There are many reasons for research being concentrated on macroinvertebrate sampling approaches. Numerous advantages of using these individuals have been proposed, and are summarised below:

- macroinvertebrates contribute significantly to riverine species diversity and act as the main pathway between primary production and many other species occupying higher trophic levels (Harper *et al.* 1992)
- individuals are sedentary in nature and have relatively long life spans (Mason 1996)
- the group is taxonomically heterogeneous, with different families and species having different responses to stresses imposed upon them (Metcalfe-Smith 1994). This opens the potential to study combined influences on macroinvertebrate communities
- their taxonomy is well established
- macroinvertebrates are present in the majority of aquatic habitats, especially within riverine systems
- individuals from each family are relatively abundant, enabling accurate measurements of abundance to be made.
- a wide range of experience has been gained on the techniques of collection and analysis, for which simple and inexpensive equipment can be used (Rosenberg and Resh 1993)
- once collected, the individuals are usually large enough to be observed by the human eye (Cummins 1992), aiding the identification procedure

All of these facets of these organisms make them extremely valuable for identifying shifts in community structure.

However, as with any seemingly ideal monitoring tool, potential disadvantages have been recognised and need to be noted prior to the commencement of any biological study. Metcalfe-Smith (1994) stated that macroinvertebrates respond to apparently minor natural changes in substrate particle size, organic content and texture. Their life histories can be complex and monitoring results can vary seasonally. Spatial heterogeneity within communities can be high, requiring considerable sampling replication to achieve statistically reliable results. Despite these disadvantages, because of the value of the qualitative and quantitative results they generate, macroinvertebrates have been widely used in the assessment of water quality of running waters for many years (Hynes 1960, 1970, Armitage *et al.* 1983, Wright *et al.* 1984, Battegazzore *et al.* 1996, Wood and Armitage 1997, Charvet *et al.* 2000).

1.2.3 Holistic approach to water quality monitoring

Many studies that have examined water quality in relation to one or more aspects have failed to appreciate the importance of other elements of the ecosystem (Pinay *et al.* 1990). Therefore, in recent years it has become increasingly recognised that individual water quality methods have both advantages and disadvantages (examples of those associated with chemical and biological methods are outlined in Table 1.1). Consequently, attention has been focused on developing monitoring schemes that adopt a wider perspective than simply describing and assessing water quality through a series of measurements (Maitland 1997). Although chemistry and biology are different scientific disciplines, the distinctions between chemical and biological methods of assessing water quality are somewhat artificial. For example, measuring dissolved oxygen is often regarded as a chemical water quality assessment but the result obtained often depends on many biological factors (Hellawell 1997). It has been suggested that no single method is sufficient to measure water quality: a combination of two or more, preferably of different kinds, is essential for an adequate understanding (Hellawell 1997).

Therefore for a catchment approach to water quality monitoring a combination of assessment methods is often regarded as essential (Boon and Howell 1997, Howell and Mackay 1997). When water quality methods are used in conjunction (Figure 1.2), the approaches can be complementary in nature. As such, studies that utilise both chemical and biological (in particular macroinvertebrate) methods of assessment are seen to provide a unique insight into the understanding of prevailing ecosystem controls (Wright *et al.* 1994).

Table 1.1. Comparison of the differences between chemical and biological water quality methods

Chemical methods	Biological methods
Procedure closely defined Standard analytical methods have been developed for most determinands	**Procedure often adapted** Some biological procedures have been standardised but many field methods are modified by operators to meet local circumstances
Variables controlled Laboratory conditions enable control to be effected	**Variables uncontrolled** The complexity of biological systems and natural variability together with the vagaries of field conditions limits control
Determination precise Chemical analyses are often able to determine minute quantities with a known precision	**Determination imprecise** Biological methods may be imprecise; for example, it may be difficult to identify species with certainty
Results repeatable Replicate samples or sub-samples may be used to refine determination	**Results unrepeatable** Intrinsic biological variability and the destructive nature of some sampling methods limits repeatability
Statistically robust Reduced variability and the applicability of standard statistical procedures assists in the assessment of results	**Statistically tenuous** Considerable variability and complexity and limited replication impair statistical confidence
Relevance extrapolated Results can only be applied to environmental assessments where considerable additional data on the likely or potential consequences for biota are known	**Relevance immediate** Direct involvement of biological systems facilitates the application of results to environmental management

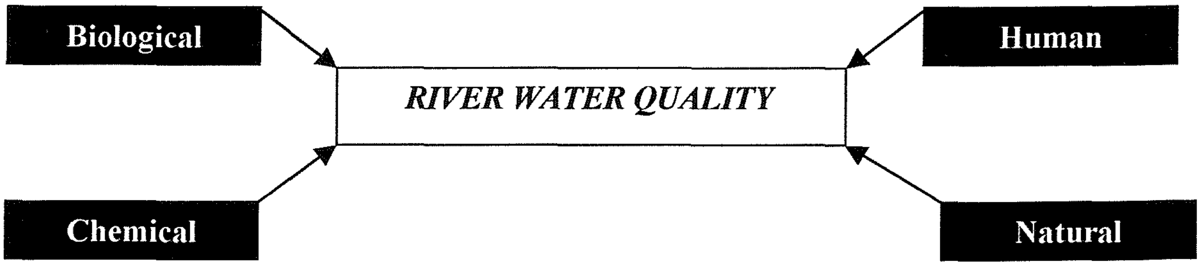


Figure 1.2. Factors affecting the water quality of rivers (adapted from Howell and Mackay 1997).

1.3 WATER QUALITY IN CHALK STREAMS

Potential inputs detrimental to water quality may be derived from many sources within a catchment area but are often grouped and classified as either point or diffuse (non-point) sources (Klapper 1991). Point source pollutants enter water bodies at discrete, identifiable locations and are therefore relatively easy to estimate and control compared with pollution from diffuse sources. By definition non-point source pollution results from materials entering a stream from sources which usually cannot be clearly defined (Delong and Brusven 1992).

Diffuse pollution of surface and groundwaters is still a major problem in many countries (Sarmiento *et al.* 1988, Kronvang *et al.* 1993). The area over which non-point pollutants can exert an influence is large, potentially affecting ecosystem processes in reaches located at considerable distance from the area in which pollutants enter a river. Although numerous countries have implemented measures to reduce point source pollution, non-point source pollution remains at the same magnitude, or is increasing, due to intensive agricultural production (Edwards *et al.* 1990, Kristensen and Hansen 1994). For example, nutrient enrichment and increased sediment supplies are often correlated with agricultural land use (Clarke *et al.* 1985). The potential impacts of these pollutants have received much research and are the focus of the current study.

1.3.1 Sediments

It is widely recognised that fine sediments can be easily transported from surrounding catchments into water courses (Olssoon and Persson 1988, Gurnell *et al.* 1992). Fine sediments have been recognised for some time as a common and significant pollutant to aquatic bodies defined in both granulometric and chemical terms (Robinson 1973). In recent years there has been increasing evidence of sediment problems in chalk rivers (Walling and Amos 1995). Although many studies have investigated the effects of deposited sediments on salmonid spawning grounds in chalk stream systems, the impacts of sedimentation can be observed throughout whole riverine ecosystems. Therefore, there is a need to identify the physico-chemical effects of sedimentation and the impact that this process has on all in-stream communities in order to gauge water quality impacts (Wood and Armitage 1997).

To date, numerous studies have been carried out on the causes and deleterious effects of fine sediment deposition on the ecology of running waters at a variety of scales, especially with regard to declining fish populations (Carling 1984, Berkman and Rabebi 1987, Sear 1993, Andrews 1995, Byres 1996, Magee *et al.* 1996, Wood and Armitage 1997). All streams carry loads of suspended solids under natural conditions. However, superimposed upon the natural sediment load a river transports is the anthropogenic sediment component produced as a by-product of catchment activities (Ryan 1991). The effects of sediment addition are often difficult to quantify, especially where diffuse land uses are concerned (Wood and Armitage 1997, Bryson 1998). Quantification difficulties have been attributed to the lack of information relating to natural baseline conditions, the relatively unknown cumulative effects of sediments on stream biota and the extreme spatial and temporal variability in the extent and degree of sedimentation on stream ecosystems (Carling 1984, Ryan 1991).

Sources of fine sediments

The relative contribution to the solids budget from near and far field diffuse sources is a subject of great debate (Mainstone 1999). In the chalk streams of southern England reduced river flows, caused by over-abstraction from surrounding aquifers, have resulted in reduced channel scour and sediment transport capacity (Walling and Amos 1995). Land uses have changed and resulted in increased sediment mobilisation and delivery to streams. Increased livestock numbers and stocking densities and the tradition of using the streams for livestock watering have caused increased poaching and trampling of channel banks with associated increased sediment inputs to channels (Rabeni and Smale 1995, Walling and Amos 1995). The poaching of the banks has increased channel widths, which reduces flow depths and velocities and encourages increased sediment deposition. Geologic characteristics of individual drainage basins exert a strong influence on degree of sedimentation and sensitivity to land disturbance (Everest 1987).

The shift in agricultural land use is of specific relevance to the chalk streams of southern England. Within the River Test catchment, sediment yields have risen with the increase in the amount of land dedicated to agriculture since the Second World War (Byres 1996). Changes in the agricultural cropping patterns and soil conservation practices have influenced sediment addition to riverine systems. The switch from spring to winter cereals has been instrumental in increasing erosion rates (Gurnell *et al.* 1992), leaving large areas of land bare during the winter months when the soil erosion hazard is recognised as being at its maximum (Boardman 1990). These changes from agriculture based on crop rotations to increasing dependence on continuous growth of cereals have caused increases in soil erosion in recent years. Changes in the type and weight of machinery used in cultivation have also affected soil erodability leading to higher levels of soil loss (Gurnell *et al.* 1992). It is thought therefore that changes in agricultural cropping patterns are likely to be one of the most influential factors which cause changes in on-site erosion within catchments in southern England.

The activities of the Ministry of Defence (MoD) are also important since they have large landholdings in a number of chalk river catchments (such as the Hampshire Avon and River Frome in Dorset). Whilst most of their land is given over to rough grassland used for training purposes, soil disturbance and the creation of highly compacted tracks by the heavy vehicles create the potential for enhanced delivery of particulate loads should be considered (Mainstone 1999).

Point source inputs can also be substantial, particularly from these fish farms and commercial watercress beds without physical treatment of effluents. Sewage treatment discharges usually contain low levels of solids due to secondary treatment. It has been estimated that a large cress farm without any form of sediment trapping can deliver 100 tonnes of particulates per year (Casey and Smith 1994), whilst a sizeable fish farm without effluent treatment can generate 54 tonnes (derived from figures given by Solbe 1982), equivalent to sewage treatment works with secondary treatment serving 34,000 and 63,000 population equivalents respectively (Mainstone 1999). Many fish and cress farms now remove solids using settlement lagoons, but their proper maintenance is vital to high removal efficiency.

Effects of fine sediments

Enhanced sediment loads can have many varied and profound effects on the whole riverine ecosystem. From an ecological point of view, it is unclear whether localised gravel bed siltation has had any significantly adverse effects on aquatic habitats (Petts *et al.* 1989). In the short-term, slow rates of change have been observed to create a number of transitional habitats that increase the overall habitat diversity within rivers. However, when sediment addition to a riverine system is extended over a prolonged period, the morphology and ecology of specific channels may be completely altered (Wood and Armitage 1997), warranting further examination of the processes in operation.

Variations in flow and concomitant increases in both suspended and deposited sediment are a natural occurrence in all streams (Ryan 1991). It follows therefore that stream organisms should be adapted to withstand occasional increases in suspended and benthic sediment levels. However, in extreme situations, fine sediments can smother the riverbed, change channel morphology, kill aquatic flora, clog the interstices between substrate clasts, increase macroinvertebrate drift, and reduce the available habitat for benthic organisms (Petts *et al.* 1989, Ryan 1991, Wood and Armitage 1997). Direct stream bed sediment deposition, although an inevitable result of sediment transport, is not the only important effect. An increase in turbidity due to suspended sediments can affect the photosynthetic activity and production of aquatic organisms, if light is their limiting factor (Ryan 1991).

1.3.2 Nutrients

In recent years much emphasis has been placed on the enrichment of natural waters by man's various activities, with considerable speculation as to nutrient sources and their variations. Eutrophication of most freshwaters is dependent upon supplies of nitrogen and phosphorus (Vollenweider 1968, Spink *et al.* 1998). These concentrations depend on a number of catchment characteristics, including land use, fertiliser application rate, soil type and the hydrological flow pathways linking the land to the stream (Roberts and Marsh 1987, Johnes *et al.* 1994a, 1994b, Heathwaite and Johnes 1996).

The definition of what constitutes a nutrient species is somewhat dependent on the perspective from which the river is being studied (Webb and Walling 1992). In this study selection included those elements that are vital to the health of plants and animals, and are strongly involved in cycling processes between the inorganic and organic compartments of rivers and their drainage basins. The generally low levels of the nutrients in natural water make them ideal indicators of surface water quality. Therefore, the determination of nutrient levels contributes much to the meaningful assessment of the water quality of river systems.

Sources of nutrients

Modern agricultural practices, particularly the use of fertilisers (Kronvang *et al.* 1995), have been strongly linked with elevated nutrient concentrations in aquatic environments (Neill 1989, Stribe and Fleischer 1991). The application of fertilisers has not only resulted in higher crop yields and nutrient removal via increased yields, but also has led to an improved nutrient level in the soil which subsequently leads to increased loss of nutrients (Isermann 1990). The realisation of this link is not recent, with studies in the late 1960s and early 1970s relating increased nutrient levels in chalk streams to increases in the application of fertilisers (Casey 1975, 1977). As early as 1965 there was considerable speculation and concern directed at the rise in nitrate concentrations in chalk water, the sources of these nutrients and their variations. Studies on the River Frome (Dorset) indicated that from 1965 to 1972 there was an overall increase in nitrate of 41%, which was proposed to be linked to the fact that the use of nitrogen fertilisers had doubled in Dorset since 1965 (Casey 1975).

As with the causes of sedimentation (§ 1.3.1) the timing and length of crop coverage influences the extent of nutrient export. For example, exports of nutrients are higher from arable land compared to grassland as the soil is left bare for a set period each year. The specific crop type is also important in the loss of nutrients. For example, the breakdown of oilseed rape litter in the soil has been observed to lead to greater export than the decomposition of cereals (Addiscott *et al.* 1991). Many studies have demonstrated the importance of terrestrial processes to stream water nutrient concentrations. Weathering of parent material is known to be an important source of phosphorus to stream water (Dillon and Kirchner 1975). Perturbation studies of small catchments have shown the importance of vegetation in retaining nutrient inputs and minimising losses to the stream (Likens *et al.* 1977), with the amount of riparian vegetation therefore influencing nutrient concentrations in rivers. Agriculture has intensified greatly in many areas of chalk downland over the past sixty years, resulting in the use of very thin and nutrient poor soils for arable production and intensive grazing (Environment Agency 1998a). Nutrient application rates to sustain arable crops and higher grass yields are high, placing chalk aquifers (particularly shallow ones) in an extremely vulnerable position (Mainstone 1999).

Sewage treatment works tend to dominate the point source load of nutrients (even with treatment), although fish farms and watercress beds can also both be important contributors. A sizeable fish farm with no effluent treatment can generate the same load of phosphorus as a sewage treatment works serving 1000 people (assuming secondary treatment). Whilst cress beds are amongst the smaller point sources, the usage of phosphorus per unit area is very high and plant uptake can be highly inefficient (Mainstone 1999). This is of particular concern since cress farms are often located in the headwaters of chalk rivers, particularly in Hampshire where 80% of the UK's national production is concentrated. The losses from 1 hectare of cress beds are equivalent to nearly 100 hectares under intensive arable production located on high-risk impermeable soils (Parr *et al.* 1998), much of which enters chalk headwaters during the growing season.

Discharges from Ministry of Defence (MoD) facilities are a concern in that the MoD has large landholdings in a number of chalk stream catchments (for example the Hampshire Avon catchment), and, until recently, their activities lay outside the normal framework of environmental regulation. Consequently little is known about the nutrient loads discharged in their effluents (Mainstone 1999).

Effects of nutrients

Nutrients are essential for the survival and growth of any organism but problems can arise when these substances are present in systems in excess amounts. The presence of inorganic nutrients in river water has only recently become a serious water quality consideration. In a relatively natural unpolluted state, concentrations of inorganic nutrients often occur in excess of the requirements of the organisms present. However, in certain conditions, some compounds can be present at concentrations that may be utilised to the point of exhaustion, limiting the amount and rate of primary production and therefore controlling the whole basis of aquatic ecosystems. Any alteration in the amount or concentration of these compounds can therefore carry profound implications for organisms higher up the trophic levels.

The growth of algae to nuisance proportions can be an eventual expression of eutrophication, leading to blankets of benthic algae and blooms of phytoplankton. Increased algal growth rates have been proposed as the principal impact of increased nutrient additions to freshwater bodies (Marsden *et al.* 1997).

Nutrient enrichment, leading to eutrophication, is now recognised as being widespread and a common environmental problem (Wilson *et al.* 1996). Consequently, there is a recognised need to reduce the current nutrient loads delivered to many water bodies in the UK in order to restore higher ecological and chemical quality to these systems (Johnes and Heathwaite 1997). Considerable effort has been directed at determining the factors affecting nutrient levels in freshwaters (Casey and Farr 1982), and it was considered that examining nutrient concentrations in a chalk stream context would provide an important insight into the trophic status of these freshwater environments.

1.4 AIMS AND RATIONALE OF THE THESIS

1.4.1 Catchment research

Increasing human population and the associated agricultural and industrial activity have resulted in chalk stream systems becoming increasingly impacted, in terms of greater use of the water resources that are available in the catchments, and increased pollution entering the system from point and non-point sources (Pegram and Bath 1995). In recent decades, greater public awareness of environmental issues has prompted demands for conservation measures and pollution reduction practices (Stonehouse *et al.* 1997).

With respect to rivers, a multidisciplinary approach is frequently being adopted which addresses scientific, engineering, social and administrative aspects of the systems (Pegram and Bath 1995). A central component of this system requires the qualitative and quantitative description of the impacts of pollution within the catchments, which will provide management with information about the impact of activities on water quality. A catchment-wide assessment of water quality is therefore a necessity. However, the implementation of such an approach to river water quality assessment is difficult. Not only does the scale at which monitoring is undertaken require definition, but more fundamentally the aspects of the systems to be assessed also requires consideration (Cannan 1998). Ideally, experimental scales should be defined with respect to the ecological/management criteria of interest to the investigation (Poizat and Pont 1996, Cannan 1998).

The inherent problems associated with collection of data at a catchment level are common to studies of any component within the system (Cannan 1998). However, the research carried out for this thesis is an attempt to develop sampling methodologies adaptable at a catchment level with a high degree of replication.

1.4.2 Aims of thesis

All natural and anthropogenic activities affect the natural riverine environment in some way and to some extent, either beneficially or detrimentally. The extent and nature of these impacts depends on the situation, manner and timing which the activities occur. Activities occurring within river catchments are numerous, with many of them impacting upon the water quality of the associated rivers. Alteration to the water quality of chalk streams is not a new phenomenon. The overall objective of my investigation was to evaluate the potential impacts of catchment activities on the water quality of chalk streams in southern England, focusing on the extent to which catchment activities impact upon the water chemistry and ecology of chalk streams.

Several sampling programmes were designed to facilitate this research. These included a biological assessment of a matrix of sites encompassing various catchment activities, coupled with physical and chemical surveys of five chalk river systems. Also, the patterns of chemical water quality within and between river systems were investigated, along with activities occurring within the catchments. The data derived from these programmes should indicate the role that catchment activities play in determining the water quality and ecological health of chalk streams in southern England, leading to the assessment of the relative stability of these systems.

1.4.3 Chapter outlines

Chapter Two: *Chalk river ecosystems*: The aim of this research programme was to assess and quantify the impacts of catchment activities on the water quality of chalk streams in southern England. Given that chalk rivers are unique systems, this chapter provides an introduction into the characteristics of chalk rivers, a justification of selection of study rivers, a background to the study rivers and an introduction into the rationale behind the sampling programme employed.

Chapter Three: *Physico-chemical water quality of chalk rivers*: This chapter describes the physico-chemical composition of different chalk river systems. Initially an introduction into the chemical composition of unimpacted chalk rivers is given, coupled with the impacts and consequences of excess nutrient loading. The second section provides an outline of the field survey undertaken from spring 1998 to spring 2000. The methods adopted for the processing of the collected samples in the laboratory are also detailed. The third section presents the results of the field survey, examining longitudinal, seasonal and interannual patterns in the composition of flowing chalk river waters. Long term variability of nutrients in a chalk river over an eighteen year period is also examined. The data provide a quantitative description of the chemical environment of chalk rivers through the presentation of various primary chemical parameters, including nutrient levels, pH, conductivity, dissolved oxygen, temperature and SPM values. The final section will discuss the current findings and place them into a wider context.

Chapter Four: *Macroinvertebrate community composition of chalk rivers*: This chapter describes the relative impacts of various catchment processes on the biological quality of chalk rivers. The causes and consequences of two catchment processes previously identified as being, on the whole, detrimental to the ecological quality of five chalk rivers are introduced. The processes highlighted are channel widening and physical sedimentation of channel beds. The second section of this chapter provides details of the field survey undertaken from spring 1998 to spring 2000. The methods adopted for the processing of collected samples and subsequent analyses of data are also described. The third section presents the results of the field survey, examining patterns both within individual rivers systems and between the five catchments studied. The data will be shown to provide an indication of the potential impacts, both beneficial and detrimental, on the ecological quality of chalk river catchments using macroinvertebrate communities. The final section discusses the ecological impacts of catchment processes on southern chalk rivers, placing current results into a wider context.

Chapter Five: *Recolonisation dynamics of chalk rivers*: This chapter describes the temporal response of macroinvertebrate communities to a simulated 'worst case' disturbance scenario. The disturbance theory in riverine systems is introduced and the factors potentially influencing the recolonisation of macroinvertebrates are outlined (e.g. substrate size and food availability). The causes and consequences of anthropogenic disturbance to streambeds are also highlighted (e.g. cattle trampling/poaching of streambeds, gravel washing and artificial substrate placement) which create new habitats. The second section of this chapter provides details of the manipulation experiment undertaken in the autumn of 1999: to determine substrate recolonisation patterns in a chalk stream. The methods adopted for the processing of collected samples are also outlined. The next section presents the results of the experiment, examining the temporal response of invertebrate communities along with the influence of substrate size on the colonisation of taxa. The data provide an indication into dynamics of recolonisation of invertebrates in a chalk stream environment. The final section discusses the ecological impacts of disturbance processes in southern chalk rivers, placing the temporal recolonisation dynamics and the substrate preference of taxa into a wider context.

Chapter Six: *General discussion*: The main findings of the thesis are summarised and implications for the future management of chalk rivers are discussed. Local versus catchment wide disturbances are reviewed leading to the formulation of a hierarchical management strategy for chalk rivers. Limitations of the study and considerations for future research are presented.

CHAPTER TWO: CHALK RIVER ECOSYSTEMS

CHAPTER TWO: CHALK RIVER ECOSYSTEMS

2.1 CHARACTERISTICS OF CHALK STREAMS

All rivers reflect the nature of their catchments and this is particularly true of chalk streams (Berrie 1992, Mainstone 1999). Many characteristics of chalk rivers can be attributed to the physical and chemical properties of the chalk over which they flow, including for example, their water chemistry, hydrology, stream flow, flora and fauna.

2.1.1 Geology

A typical chalk stream is the result of a specific combination of geology, plant and animal species, and management (Westlake *et al.* 1972). In the present study, all the rivers examined flow over catchments dominated by chalk. Chalk streams have many attributes that are related to the physical and chemical properties of the chalk geologies over which they flow and which set them apart from other lowland rivers (Berrie 1992), giving rise to a distinctive hydrochemistry and flow regime. Chalk is a very pure form of limestone, consisting almost entirely of calcite of organic origin, mainly from marine algae and fragments of calcified animal structures laid down during the Cretaceous period (Giles *et al.* 1991, Berrie 1992). Chalk is highly porous with pore space typically forming about 40 % of its volume, giving it great capacity to absorb water and form substantial aquifers. It is this high porosity that results in the rivers having all the typical characteristics: few tributaries (occurring at right angles to the main channel), intermittent headwaters (winterbournes), and short stream lengths in relation to the catchment area (Berrie 1992). The drainage densities of chalk rivers (km of stream/sq. km of land area) are low, around 0.12 compared with values of up to 0.47 for the clay-vale rivers of Sussex and Kent (Mainstone 1999). The drainage pattern of chalk rivers contrasts sharply with that of rivers on impermeable clay catchments, where rainfall flows overland, creating a host of tributaries with a branching pattern.

Features which typify the channels of chalk rivers include, for example, low longitudinal frequency of riffles and pools, infrequent gravel shoals and exposed riverine substrates, shallow cross sections and sinuous channel form (Sear *et al.* 1999). It is uncertain to what extent these features are the result of natural or human influences. Channel morphology is a function of hydrological regime, sediment delivery and the drift geology through which the river flows. Chalk rivers generally have low hydraulic energy, too low to mobilise gravel, and are, therefore, less able to shape channel morphology than other river types (Mainstone 1999).

2.1.2 Hydrology

The hydrology of any river is arguably the most important factor controlling the form and function of its ecosystem. The principal hydrological pathway within chalk streams is through slow percolation into the underlying aquifer and subsequent discharge via springs. Streams that flow over catchments dominated by chalk therefore originate almost entirely from ground water, often receiving little surface run-off (Berrie 1992, Mainstone 1999). Around 90% of the annual stream discharge may be derived from groundwater sources (Casey 1981, Mainstone 1999), and violent floods are rare (Casey and Smith 1994).

The slow release of water from aquifers greatly attenuates the sporadic nature of rainfall. With chalk rivers, ratios of maximum to minimum daily mean flows are generally less than 10:1, often in the order of 3:1 on pure chalk geologies (Ladle and Westlake 1995), contrasting with clay catchments where ratios are greater than 100:1. Importantly, whilst peak flows in aquifer fed rivers are relatively small compared to flashier river types, they are sustained for longer periods by the high base flow component. Spring fed channels have been found to equal or exceed bank full flows for approximately 30% of the time compared to 5% in run off dominated rivers (Whiting and Stamm 1995). This regime produces sustained waterlogging in riparian soils, a characteristic of chalk river floodplains in their pristine state (Mainstone 1999).

In general, a chalk aquifer will rise in winter months, producing a stronger discharge from perennial springs. Flows will increase throughout the winter and continue to rise well into spring months, after which they will decline steadily until the next winter (Berrie 1992). This whole process is governed by a consistent recharge occurring during winter months (Westlake *et al.* 1972). Where chalk rivers flow over mixed geology, run-off from impermeable soils or flows from aquifers with faster response times will distort this hydrograph. Such distortions may be generated by a variety of human activities, including urban development, land drainage and water abstraction (Mainstone 1999). Flows within chalk rivers are generally fast flowing and shallow, with velocities sufficient to prevent silt being deposited on beds predominately composed of gravel. Owing to the physical filtration of most of the streamflow by percolation through chalk, and the relatively low hydraulic energy of chalk rivers, inputs of solid material are naturally low in chalk rivers. The characteristic state is therefore one of clear waters with very low levels of suspended solids and low bed loads of fine sediment (Mainstone 1999).

2.1.3 Water chemistry and instream biota

The hydrology of chalk streams often produces a relatively stable and predictable environment (Bickerton *et al.* 1993, Casey and Smith 1994, Prenda *et al.* 1997). Due to the high proportion of discharge derived from groundwater sources, chalk stream flows, temperatures and most chemical components are strongly buffered against change (Ibbotson *et al.* 1994, Mainstone 1999). For example, water enters chalk rivers in southern England from springs at 11°C, ensuring that temperatures remain relatively warm in the winter but relatively cool in the summer. The characteristic annual temperature range of a sizeable southern chalk stream is 5 – 17°C (Mainstone 1999). Chalk streams have characteristically high alkalinity (180-260 mg l^{-1}), conductivity and pH (7.5-8.5) (Bickerton *et al.* 1993, Mainstone 1999). The water generally contains an excess of all major nutrients and low amounts of suspended solids (Casey and Newton 1972). Another common characteristic of chalk rivers is that they support a rich and diverse community of macroinvertebrates and macrophytes due to good light penetration (Giles *et al.* 1991, Berrie 1992).

All of the factors described above represent very favourable conditions for plant growth and production, and chalk streams typically have high densities of algae, diatoms and macrophytes (Berrie 1992). Chalk rivers have a characteristic plant community, often dominated in mid-channel by river water crowfoot *Ranunculus penicillatus* var *pseudofluitans* and starworts *Callitriche obtusangula* and *C. platycarpa*, and along the edges by watercress *Rorippa nasturtium-aquaticum* and lesser water-parsnip *Berula erecta*. They have low banks which support a range of water-loving plants. These conditions support important game fisheries, notably for brown trout *Salmo trutta*. Chalk rivers also support species listed on Annex II of the EC Habitats Directive (92/43/EEC), including Brook lamprey *Lampetra planeri*, salmon *Salmo salar*, crayfish *Austropotamobius pallipes* and otter *Lutra lutra* (Mainstone 1999).

2.1.4 Recognition of the importance of chalk rivers

Chalk rivers, like most lowland rivers in England, are generally highly physically modified systems (Mainstone 1999). Their current ecological significance and their social value (visual amenity and specific water and catchment uses) are intricately linked to certain aspects of historical modification. Examples include, fishing, water meadows, flood defence, land drainage and agricultural improvement which have all had detrimental impacts on chalk rivers and their associated habitats. A number of chalk rivers have been designated as Sites of Special Scientific Interest (SSSIs), including the Rivers Test, Itchen, Frome and Hampshire Avon, all of which were chosen as study rivers in this research. Under the EU Habitats Directive (92/43/EEC), Special Areas for Conservation (SACs) are being designated to protect listed species and habitats. Based on the SSSI network, the candidate list of cSACs includes two chalk river systems:

- the *Itchen* (designated for *Ranunculus* beds and the southern damselfly *Coenagrion mercuriale*), and,
- the *Hampshire Avon* (designated for *Ranunculus* beds, Atlantic salmon *Salmo salar*, bullhead *Cottus gobio*, sea and brook lampreys *Petromyzon marinus* and *Lampetra planeri*, and Desmoulin's whorl snail *Vertigo moulinsiana*).

2.2 EXPLOITATION AND MODIFICATION OF CHALK RIVERS

2.2.1 Anthropogenic exploitation and modification

The human pressure on chalk rivers has for the last centuries been intense, these systems being utilised and managed for a wide range of activities. As time has progressed, the nature of usage of chalk rivers has also shifted. However, the impacts upon rivers are due to both recent activities and those carried out many years ago. Examples of pressures being exerted on chalk rivers in southern England are presented below.

Abstraction: there is great demand for abstraction of the high quality water for a variety of purposes. Excessive abstraction mainly for public water supply from the chalk aquifer has contributed to low flows on a number of chalk rivers. This has led not only to drying out of upper sections and riparian zones, but also to accumulation of silt and changes in the aquatic vegetation structure (National Rivers Authority 1991, 1993, 1995). Artificial measures to counter these effects, such as sealing of the bed with concrete and narrowing of the channel, can themselves have negative ecological consequences (Mainstone 1999).

Physical modification: Like most lowland rivers, the beds of many chalk rivers have been dredged and lowered and flow has been confined to specific channels for flood defence, drainage, navigation, and other purposes. As low energy systems, chalk rivers are generally less able to revert to their original channel structure. Some have side channels, created during times of much higher flows after the last ice age. These have sometimes been modified to create lakes for ornamental or fishery purposes. The management of water meadows from a mill head was also a common practice in recent centuries. The full extent of these modifications on the animal and plant communities of chalk rivers is not known (Mainstone 1999).

Pollution: Chalk rivers serve the typical function of receiving discharges of various types. In common with most lowland rivers, chalk rivers have been significantly affected by sewage discharges and, in times of low flow, de-oxygenation may occur. High levels of nitrates (leaching from ploughed land into groundwater) and phosphate (from sewage effluent) can be found in many chalk rivers. Because of this enrichment, excessive growths of blanket-weed have been observed on what were previously free of instream vegetation. Changes in plant communities have occurred, including loss of water crowfoot (*Ranunculus penicillatus*) beds from some river stretches. Effluent from fish farms, water-cress beds and light industry can have similar detrimental effects (Ladle and Westlake 1995, National Rivers Authority 1995, Mainstone 1999).

Catchment land use: The land adjacent to chalk rivers is often dominated by arable cultivation for wheat and barley. Prior to the 1940s, the majority of chalk downland areas were used for extensive grazing regimes or under woodland. Land use change adjacent to rivers has led to conversion of permanent grassland to intensive leys and silage as well as arable crops. These can have significant consequences on river nutrient loads (Mainstone 1999). The ploughing of the catchments has led to high levels of nitrate leaching into the groundwater aquifer and to the run-off of soil particles, causing siltation and concretion of river gravels, which are vital for the spawning of salmon and trout (Berrie 1992, Wood *et al.* 1999). Alongside the rivers, the land is often used for cattle pasture. Some of these areas were formerly water meadows, with the early grass production used for hay and grazing (National Rivers Authority 1995). Over-grazing can exacerbate siltation of river gravels. Light trampling by cattle also creates muddy margins of importance for a range of uncommon invertebrates (Harrison 2000). Direct destruction as a result of development pressure may also occur. Flood defence operations can add to the ecological stresses on these systems (Mainstone 1999).

Fisheries management: On many chalk rivers this is intensive, with regular weed cuts in the channel; fencing off and mowing of strips along the bank; infilling and stabilisation of banks; removal of unwanted fish species (e. g. pike, grayling); and stocking with farm-reared trout. Some fisheries management practices are evidently beneficial to conservation, such as cleaning gravels, while others are neutral providing they do not either impact on characteristic plant and animal communities or are carried out in previously unmanaged areas. However, on the whole, it is considered that fisheries management modifies habitats and fish communities to suit the needs and preferences of game anglers (Mainstone 1999). Hence it is obvious that chalk rivers have in the past, and still continue, to experience a number of activities which impact upon the natural systems. It is widely agreed that these influences have greatly changed these river systems in their physical appearance over the past centuries. It is believed that there are no British examples of large chalk rivers with natural channel geomorphology (Ladle and Westlake 1995, Mainstone 1999). Table 2.1 outlines the major anthropogenic activities thought to impact upon the sediment and nutrient status of chalk river catchments, indicating their potential ecological implications.

Table 2.1. **An inventory of key human activities undertaken in chalk river systems and their links to ecological impact (modified from Mainstone 1999).**
Note: **Some activities can have beneficial effects (in italics), depending upon how they are performed.**

Sphere of activity	Specific activity	Further detail	Potential direct effect
Discharge of polluting substances (point sources)	Domestic sewage		Organic pollution, nutrient enrichment, increased solids load
	Fish farms		Organic pollution, nutrient enrichment, increased solids load Contamination with fish farm chemical
	Cress farms		Increase solids load Pesticide contamination
	Livestock farms	Yard washings, silage liquor, slurry, etc.	Organic pollution, nutrient enrichment, increased solids load
	Road run-off		Hydrocarbon and heavy metals contamination, solids load
	Other (e.g. industrial)		Organic/toxic/nutrient/solids pollution
Flood defence/land drainage	Channel modifications	Deepening	Loss of hydrological continuity with banks, reduced floodplain water table
		Straightening	Reduced physical diversity, particularly depth, current, velocity and substrate type. Loss of riffle pool/sequence
		Longitudinal reprofiling	Reduced variation in channel gradient and hence current velocities, depth and substrate type
		Widening	Reduced water depth, current velocity and floodplain water table. Loss of hydrological continuity with banks
	Bank reinforcement	Hard (concrete, piling etc)	Loss of habitat structure and riparian flora/fauna
		Soft (geotextiles, willow hurdles etc)	*Encouragement of natural regrowth* but altered habitat structure and flora/fauna

Table 2.1 cont.

Sphere of activity	Specific activity	Further detail	Potential direct effect
Flood defence/land drainage cont.		Disposal of weed cut (caught downstream)	Smothering of riparian and floodplain vegetation (which can also cause bank instability), pollution via run-off, soil enrichment, encouragement of ruderal vegetation
		Dredging	Reinstatement of overlarge channel, loss of benthic infauna, high solids remobilisation within channel
		Disposal of dredgings	Smothering of riparian and floodplain vegetation, steepening of bank edges, soil enrichment, encouragement of ruderal vegetation, pollution via run-off.
	Removal of bankside vegetation	Trees, scrub	Potential loss of habitat for animals. Increased light to river and encouragement of plant production
Agriculture	Intensive livestock grazing	In riparian areas with free access to river	Bank destabilisation, soil erosion, nutrient enrichment and disturbance of bankside vegetation, enrichment and siltation of channel. Loss of wetland/meadow plant species and associated fauna in favour of ruderal species
		In wider catchment (chalk downland and floodplain)	Soil erosion and loss of particulates, nutrients and agrochemicals to river network. Nutrient/agrochemical contamination of groundwater and subsequent baseflow

Table 2.1 cont.

Sphere of activity	Specific activity	Further detail	Potential direct effect
Agriculture cont.	Ploughing/arable cropping	In riparian areas	Bank destabilisation, soil/nutrient loss to river, damage to marginal and riparian flora/fauna from pesticide/herbicide spray drift, nutrient enrichment of bankside vegetation and encouragement of ruderal species. Pesticide/herbicide run-off to river
		In wider catchment (chalk downland and floodplain)	Soil erosion and loss of particulates, nutrients and to river. Contamination of groundwater and subsequent baseflow
Fisheries management	Cleaning of spawning gravels		*Improvement in gravel habitat for a range of fish species and some flora.* Possible destruction of silty habitats. Smothering of gravels downstream
Recreation, amenity and access	Public use of riparian areas and main channel for walking, canoeing etc.		Disturbance to sensitive mammals (e.g. otters) and nesting birds. Trampling of sensitive habitats. *Possible benefit to water voles from mink disturbance.*

2.3 STUDY RIVERS

The term chalk river is used in this research to describe watercourses dominated by groundwater discharge from chalk geology, including those that flow over a range of non-chalk surface geologies at various points along their length, particularly in the lower reaches (Mainstone 1999).

The majority of chalk rivers in the UK are located in England, in and downstream of areas of outcropping chalk in the south, East Anglia, and further north in Lincolnshire and Yorkshire. There are approximately 35 rivers and major tributaries that are classified as chalk rivers, ranging from 20 to 90 kilometres in length, from the Frome in Dorset to the Hull in Humberside.

2.3.1 Choice of river

After an initial survey of the area involved, five river systems were selected for study: the Rivers Test, Itchen and Meon in Hampshire, the River Frome in Dorset and the River Avon, the catchment of which covers parts of the counties of Wiltshire, Hampshire and Dorset.

These rivers were selected for numerous reasons:

- primarily, their catchments were small enough to maintain a sampling effort in sufficient temporal and spatial detail to meet the objective of chalk stream system comparison without encountering severe sampling problems
- the various systems represent chalk streams at a variety of scales (ranging from large: the River Avon, to small: the River Meon), all of which exhibit a high degree of habitat variability
- accessibility to sites along the rivers was of paramount importance, and,
- prior knowledge of certain systems (previous studies carried out by author) was also pivotal in the choice of the five river systems.

2.3.2 River Descriptions

The River Itchen

The Itchen (Figure 2.1) rises on the Upper Chalk of the Hampshire Downs, initially flowing west through Winchester before turning southwards, and entering the tidal estuary at Woodmill, Southampton (National Rivers Authority 1993). The population in the River Itchen catchment is approximately 250,000, with the main settlements being Southampton, Eastleigh and Winchester, which occupy approximately half of the total length of the main river corridor (National Rivers Authority 1992a). A broad riverside corridor of open land has been retained in many of areas, which is rarely intensively farmed. In upstream areas the catchment is predominately rural with a combination of arable and livestock farming. Further downstream the catchment is urban and includes areas of heavy industry. For much of its length the River Itchen is divided between two or more separate channels running parallel to each other, with many structures having been constructed to regulate river flows and levels (National Rivers Authority 1993). The multiplicity of channels has been linked with the harnessing of water power for milling, the use of the water meadow systems to promote early pasture development, and the use of the river for navigation. Traditional industries relying on water power included wool processing, paper making, tanning, flour and grist milling as well as the generation of electricity (National Rivers Authority 1992a). The high conservation value of the River Itchen is reflected in the statutory designation of sections as Sites of Special scientific Interest (SSSIs).

The catchment forms part of the northern flank of the Hampshire Basin, dipping from north to south. The total catchment area of the River Itchen is 400 km^2, of which 360 km^2 is classified as chalk geology. The oldest strata exposed in the River Itchen catchment is the Cretaceous Chalk, a porous, fine grained limestone that outcrops over the whole of the north of the catchment. There are three main sub-divisions of the Cretaceous Chalk: the Upper, Middle and Lower Chalk. The catchment is dominated by Upper Chalk, however there is an inlier of Lower Chalk to the east of Winchester. Further south, the river flows over younger sands, silts and clays that are less permeable than the chalk (National Rivers Authority 1993).

The average annual precipitation varies from 750 mm on the south coast to 1000 mm on the northern downland (National Rivers Authority 1993). The range in flows is slightly less than the neighbouring River Test, with the maximum flow typically only 3 to 4 times the minimum flow. This contrasts sharply with other rivers where surface run-off in winter can boost flows several hundred times, with summer flows not supported by groundwater recharge.

There are two surface water abstractions for public supply on the River Itchen. Along with these, there are four major licences for groundwater abstraction for public water supply in the Itchen catchment, each controlling a number of supply boreholes. There is a thriving watercress industry based on the river in its upper reaches, supplied by artesian springs and boreholes. Fish farming is also an important industry with four major fish farms located adjacent to the river or its tributaries (National Rivers Authority 1992a).

The Environment Agency sets objectives for river quality to protect natural stocks of fish and the uses to which the river is put. The highest objective in the classification is Class 1A (water of high quality suitable for potable supply abstraction, high class game and coarse fisheries and of high amenity value). Increases in population have resulted in three large sewage treatment works discharging directly into the lower and tidal reaches of the river. Altogether 55 km of the River Itchen and its tributaries above Eastleigh have an objective of Class 1A. Between Eastleigh and the tidal limit there is a 4 km length of the river which has an objective of Class 1B (high water quality, suitable for game and high class fisheries). The downstream tributaries are also Class 1B objective holders.

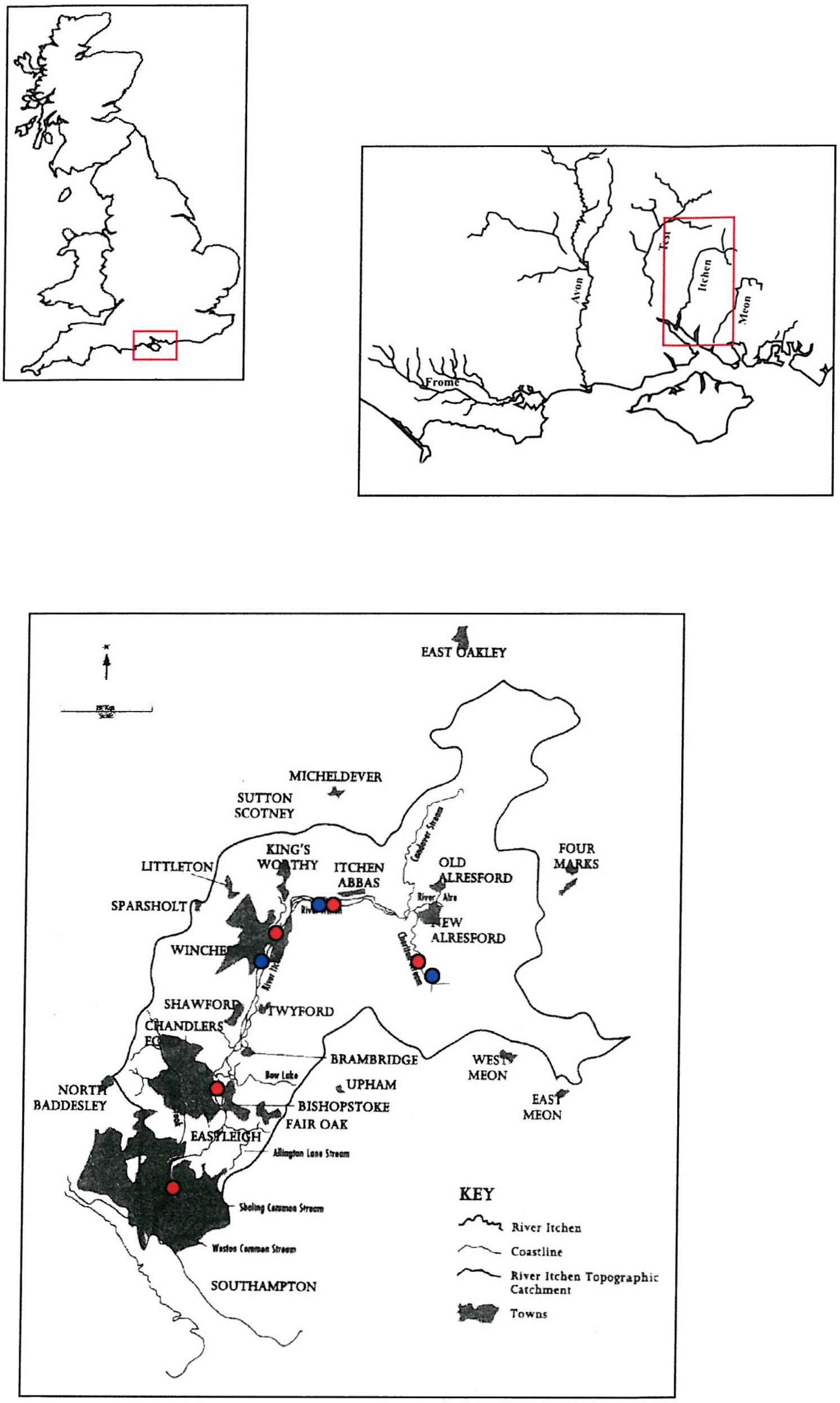


Figure 2.1. Location of the River Itchen, UK.
Blue spots represent biological monitoring sites
Red spots represent chemical monitoring sites

The River Test

The River Test (Figure 2.2) is considered by many people to be the finest chalk stream in the world (National Rivers Authority 1991). Initially the river flows south across the Upper Chalk outcrop on the southern flank of the North Downs ridge. From these upper reaches the river widens considerably and meanders through a wide floodplain, often flanked by water meadows (Environment Agency 1998a). Within the valley the main channel of the Test frequently splits to form a network of anastomised channels. Further south the river flows over less permeable soils before finally entering the estuary of Southampton Water. In its natural form, the precursor of the modern River Test would have flowed along a number of shallow silty channels with occasional gravel bars, fringed by marshlands and wetlands (Environment Agency 1998a). The braided channel system is again a legacy of the historic uses of the river for milling, for land irrigation of water meadows and for navigation (Environment Agency 1998a). The river once provided power for wool processing, paper making, tanning, flour and grist milling and the generation of electricity. There are still extensive remnants of the water meadow system.

The total catchment area of the River Test is 1260 km^2, with the underlying geology forming part of the northern flank of the Hampshire Basin. The oldest rock is the Cretaceous Chalk that outcrops over the whole of the north of the catchment. Upper Chalk outcrops over much of the catchment with some exposures of Lower Chalk, particularly in the north west region of the catchment (Environment Agency 1998a). Downstream of this, the Test flows off the chalk and on to a varied clay, silt and sand catchment comprising London Clay and the overlying Bagshot Beds (National Rivers Authority 1991). These are younger rocks of Tertiary origin, which are much more impermeable than the Chalk.

The mean annual rainfall across the catchment is fairly constant, varying from over 900 mm in the upper parts of the Hampshire Downs to 750 mm in the Test Valley (National Rivers Authority 1991). A vast majority of this rainfall falls on the chalk catchment and permeates into the aquifers and the range of flows in the River Test are low, with the maximum flow in any one year typically only 4 – 5 times the minimum (Environment Agency 1998a).

The development of south Hampshire has generated a continuing demand for water for domestic and industrial use. The chalk streams provide much of the water used in Hampshire, particularly from the lower reaches and boreholes penetrating the chalk aquifers. An important consideration is the extent to which the water is returned to the river and the point of its return. Water taken out of the catchment or 'consumptive use' is a loss to the river system. Some uses such as spray irrigation are entirely consumptive as little, if any, of the water returns to the system. Conversely almost all the watercress growing, gravel washing and fish farming users return all the water to the same location, so the overall effects on flow are small (Environment Agency 1998a). On average, 80% of water abstracted for domestic supply is returned to the river after proper treatment. On the River Test consumptive use represents only 2% of the total average flow. There is one major abstraction for public water supply on the River Test that is located close to the tidal limit so that water is taken just before it is lost to the estuary.

Approximately 135 km of the 139 km of the River Test meet the Environment Agency Objective Class 1A (water of high quality suitable for potable supply abstraction, high class game and coarse fisheries and of high amenity value) (Environment Agency 1998a). The catchment is predominately rural in nature, with a population located in small towns and villages in the North, or concentrated in the conurbations alongside Southampton Water. There is little manufacturing industry on the River Test, the only significant discharge in this category being from a paper mill at Overton (near the river source). The river has an objective of Class 1B (high quality water suitable for game and high class fisheries) for the 4 km below this discharge. There are two substantial discharges of treated sewage effluent made directly to the river (Environment Agency 1998a).

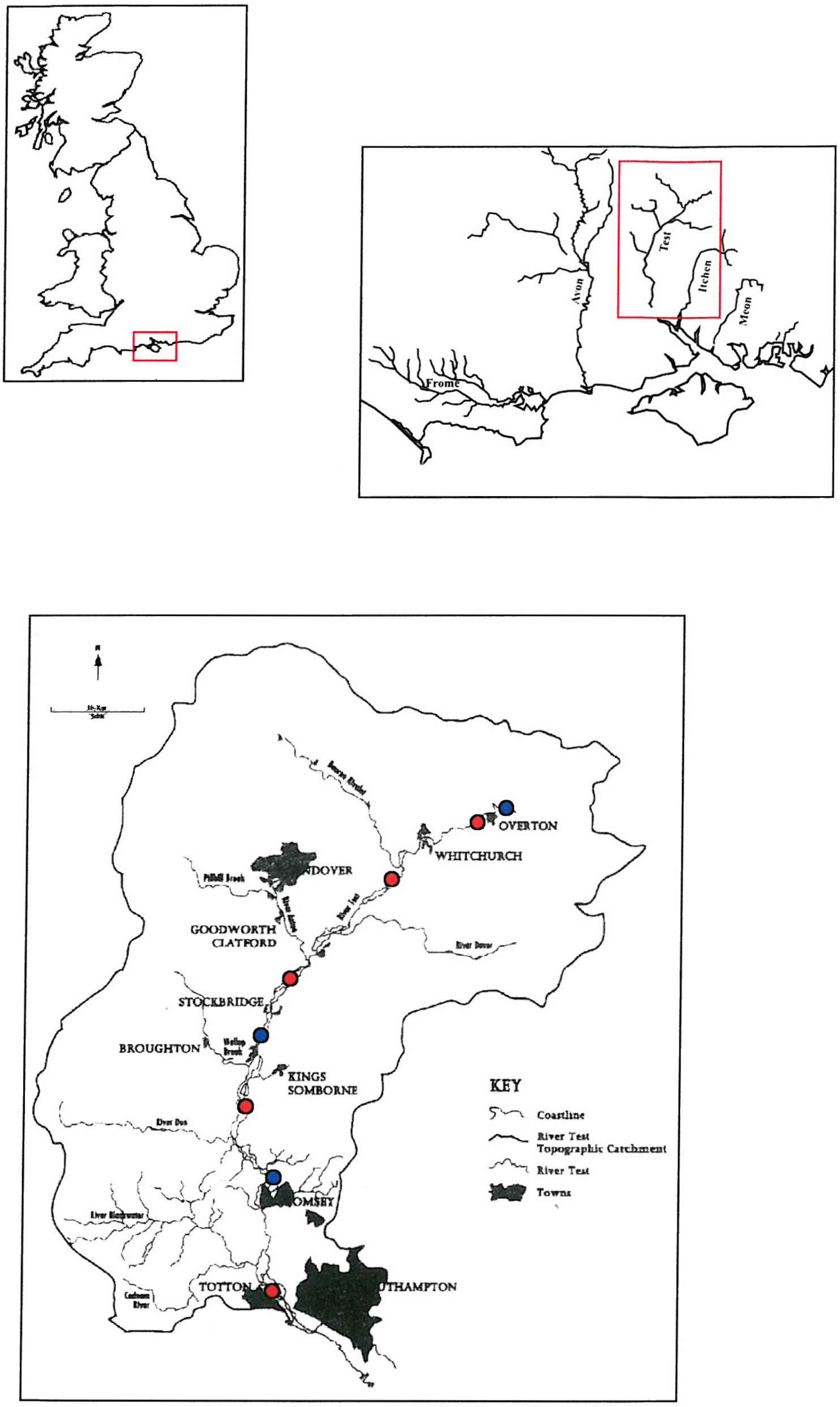


Figure 2.2. Location of the River Test, UK.
Blue spots represent biological monitoring sites
Red spots represent chemical monitoring sites

The River Meon

The River Meon (Figure 2.3) is the smallest river investigated in this study. Initially the river flows north-west, before turning south towards the sea, passing through several villages, and finally entering Southampton Water. The headwaters cross permeable chalk in the upper catchment, after which the river flows over less permeable sands, silts and clays of tertiary age (Environment Agency 2000a). Historically the steep gradients of the middle and upper reaches have enabled the river's water power to be harnessed for iron working, wool processing, paper making, tanning, flour and grist milling as well as for the generation of electricity (Environment Agency 1998b). Structures were also constructed to improve navigation, and a water meadow system designed to increase surface irrigation throughout the catchment. Over most of Hampshire the Upper Chalk outcrops at the surface but it is in the Lower Chalk that the river has its source, where the groundwater table intercepts the surface and springs occur (Environment Agency 1998b). Generally the river flows over the more permeable Upper Chalk, however there are outcrops of the Lower Chalk over which the river flows. As with the majority of Hampshire chalk rivers the flow is split between a number of channels over many sections of the river's length, this feature being attributed to the old milling and water meadow systems.

There are four major public water supply abstraction points throughout the catchment. These are all groundwater abstractions and there are no direct water supply abstractions from the river. The water is mostly used within the immediate catchment but there is some export towards the Portsmouth area. There are also a significant number of private abstractions in the Meon Valley, many of which are used for spray irrigation and agriculture (Environment Agency 1998b). The high quality of the water has resulted in a thriving water cress industry on the Meon river, along with numerous fish farms. Within the Meon Valley there are three small sewage treatment works that discharge into the river (Environment Agency 2000a). No industrial discharges are made into the river or its tributaries, apart from one very small cooling water discharge. The river meets the Environment Agency objective of Class 1B for its entire length.

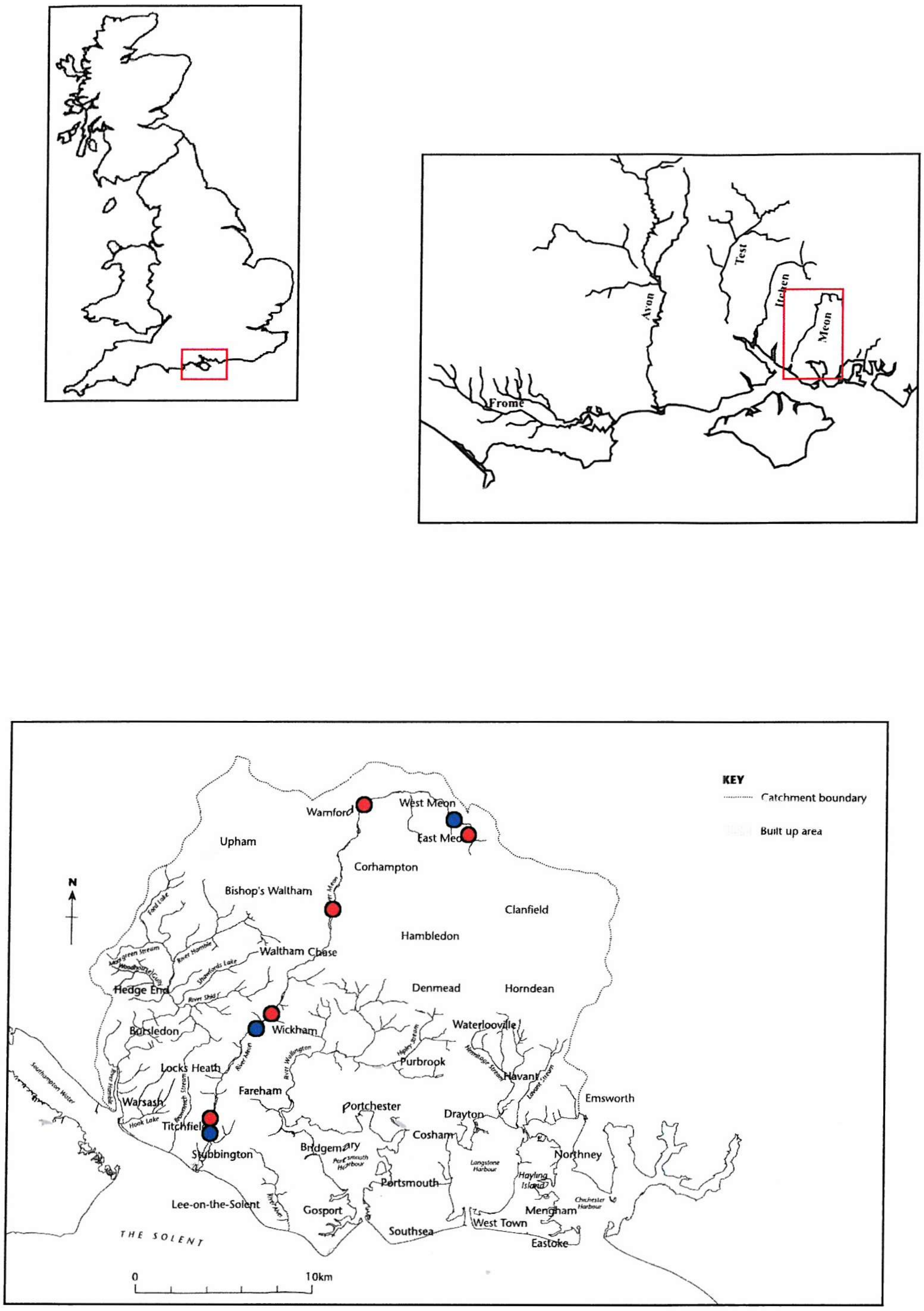


Figure 2.3. Location of the River Meon, UK.
Blue spots represent biological monitoring sites
Red spots represent chemical monitoring sites

The River Frome

The River Frome is the westernmost major chalk stream in England (Figure 2.4). It is 52 km long, with the source located in the North Dorset Downs. Initially the river heads west towards Dorchester, below which several small tributaries enter from the south. The Frome meanders in an easterly direction through a shallow chalk basin to Poole Harbour, before discharging into the English Channel.

The upper part of the catchment lies within the Dorset Area of Outstanding Natural Beauty and is characterised by steep-sided valleys (National Rivers Authority 1995), with the river dependent on springs and groundwater levels for its flow. Many of the tributaries in this upper section are winterbournes (cease to flow during summer months) or perched (flow underground for part of their lengths). Consistent with the other rivers sampled, the middle reaches of the River Frome have an anastomised network. The catchment through which the river flows is predominately rural in nature, with areas of high amenity and ecological value well distributed throughout (National Rivers Authority 1995). In the upper catchment, sheep grazing historically dominated land use, but, with changes in agriculture practices, arable crops now predominate coupled with dairying and stock rearing. Since prehistoric clearances the lower area of the catchment has been typically lowland heath in nature (National Rivers Authority 1995). The catchment has a population of 63,000, with two major settlements, Dorchester and Wareham. Industrial development throughout the catchment is light, with most activity being related to agriculture.

The River Frome drains a catchment area of approximately 660 km^2, flowing over a geologically complex area. The north and east of the catchment are predominantly Middle and Lower Chalk, along with Greensand and Gault Clay (exposed through erosion and geological faults). Further west the river flows through a section of Upper Chalk, before flowing through more the recent geological strata of tertiary deposits, including Reading Beds, London Clays, and the sands and gravel of Bagshot Beds. However, chalk is viewed as the most important geological formation in the catchment, outcropping over 46% of the area (Paolilo 1969) and providing approximately 80% of the discharge through natural springs and boreholes (Mann *et al.* 1989, Cannan 1998).

The highest annual rainfall of up to 1100 mm per year occurs in the north and south-west of the catchment and is thought to be attributed to the higher altitude of these areas. Rainfall decreases to the south and east, with records of 850 to 950 mm in these regions (National Rivers Authority 1995). River flow decreases throughout the summer months, and some springs and tributaries in the upper regions of the catchment have been observed to dry up progressively (National Rivers Authority 1995). Throughout the rest of the year, flows generally remain stable.

There are a number of abstraction points for water throughout the catchment. The largest licensed groundwater uses are public water supply and cress growing, although general farming has the largest number of licenses. The largest licensed surface water use is fish farming, followed by spray irrigation, industrial and other (commercial, public services and recreation) (National Rivers Authority 1995).

There are 11 sewage treatment works of varying sizes which discharge effluent into the River Frome. Two of these works receive a significant trade effluent component. There is also a treated surface water discharge from the oil/gas transfer station operated by British Petroleum at Furzebrook. There are a number of watercress beds and fish farms located along the river's length. All of these processes result in the discharge of effluents into the river channel. The Environment Agency have classified the river quality objectives for the River Frome as Class 1A for the majority of its length (Armitage 1997). Two stretches of the river have been designated as Class 1B, where the river flows through Dorchester, and further east, through the town of Wareham. However, it has been noted by the Environment Agency that parts of the Frome may be experiencing eutrophication through the input of nutrients from point source discharges (sewage treatment works and fish farms) and diffuse inputs (agriculture).

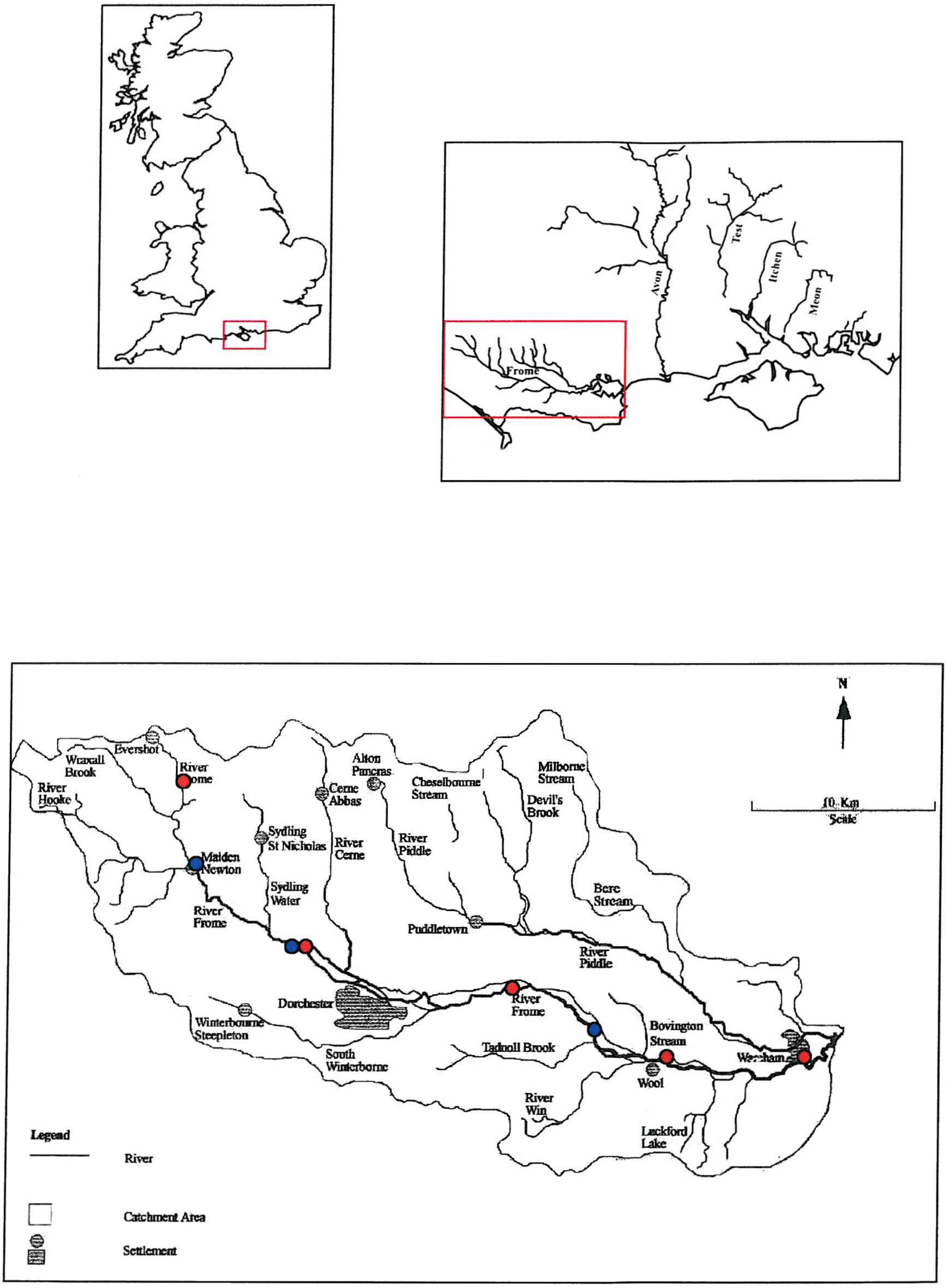


Figure 2.4. Location of the River Frome, UK.
Blue spots represent biological monitoring sites
Red spots represent chemical monitoring sites

The River Avon

The River Avon (Figure 2.5) rises in the Vale of Pewsey and flows directly south before entering Christchurch Harbour. Together with two tributaries the River Avon drains Salisbury Plain (National Rivers Authority 1992b). Below Fordingbridge the catchment area is concentrated on the left bank with a number of significant tributaries joining which drain the tertiary deposits of the New Forest. The overall catchment land use is predominantly rural, with light industry mainly situated in the towns. Although intensive agriculture is the main land use, the catchment retains a relatively high proportion of semi-natural habitats compared to other river catchments in southern England (National Rivers Authority 1992b). Tourism is economically important in the catchment and the water environment is an important feature, with extensive water meadow systems established along the river coupled with numerous fish farms and recreational fisheries (Environment Agency 1996). The catchment has a population of approximately 200,000, distributed throughout the towns of Salisbury, Christchurch, Ringwood and Fordingbridge. In addition there are several major military establishments on Salisbury Plain.

The river has a geographical catchment area of approximately 1701 km^2 (Environment Agency 1996) and covers parts of the counties of Wiltshire, Hampshire and Dorset. Chalk is the predominant rock type throughout the Avon catchment. In the uppermost sections of the catchment the chalk is underlain by the Greensand, which is exposed in the Vale of Pewsey. The Greensand is also an aquifer but inputs a more acidic groundwater into the river. As the river progresses south, it crosses a succession of strata including Upper Chalk, Barton, Bracklesham and Bagshot beds which are acidic clays, sands, silts and gravels, which support different land uses compared to those undertaken on the chalk deposits, including greater levels of dairy farming. The river is largely spring fed from the chalk aquifers that provide a relatively stable discharge throughout the year, consistent with all chalk rivers. The average annual precipitation throughout the catchment is 769 mm.

Public water supply and spray irrigation form the largest consumptive use of surface water within the Avon catchment. Other uses of river water include fish farms, gravel washing and watercress farms. The main occurrence of groundwater abstractions is in the upper tributaries of the River Avon, with the main uses being public water supply, agricultural spray irrigation, watercress farms, and fish farming. Minor uses such as the Ministry of Defence, gravel extraction and industrial uses make up the remainder.

Sewage effluents from 32 sewage treatment works are discharged to the watercourses in the catchment. The two largest are located in Salisbury and Christchurch. There is only one significant industrial discharge from a Tannery which process effluent is discharged directly to the river. River quality objectives for the River Avon are Class 1A for the majority of the river length, with relatively short stretches of river classified as Class 1B. These areas are located downstream of sewage treatment works.

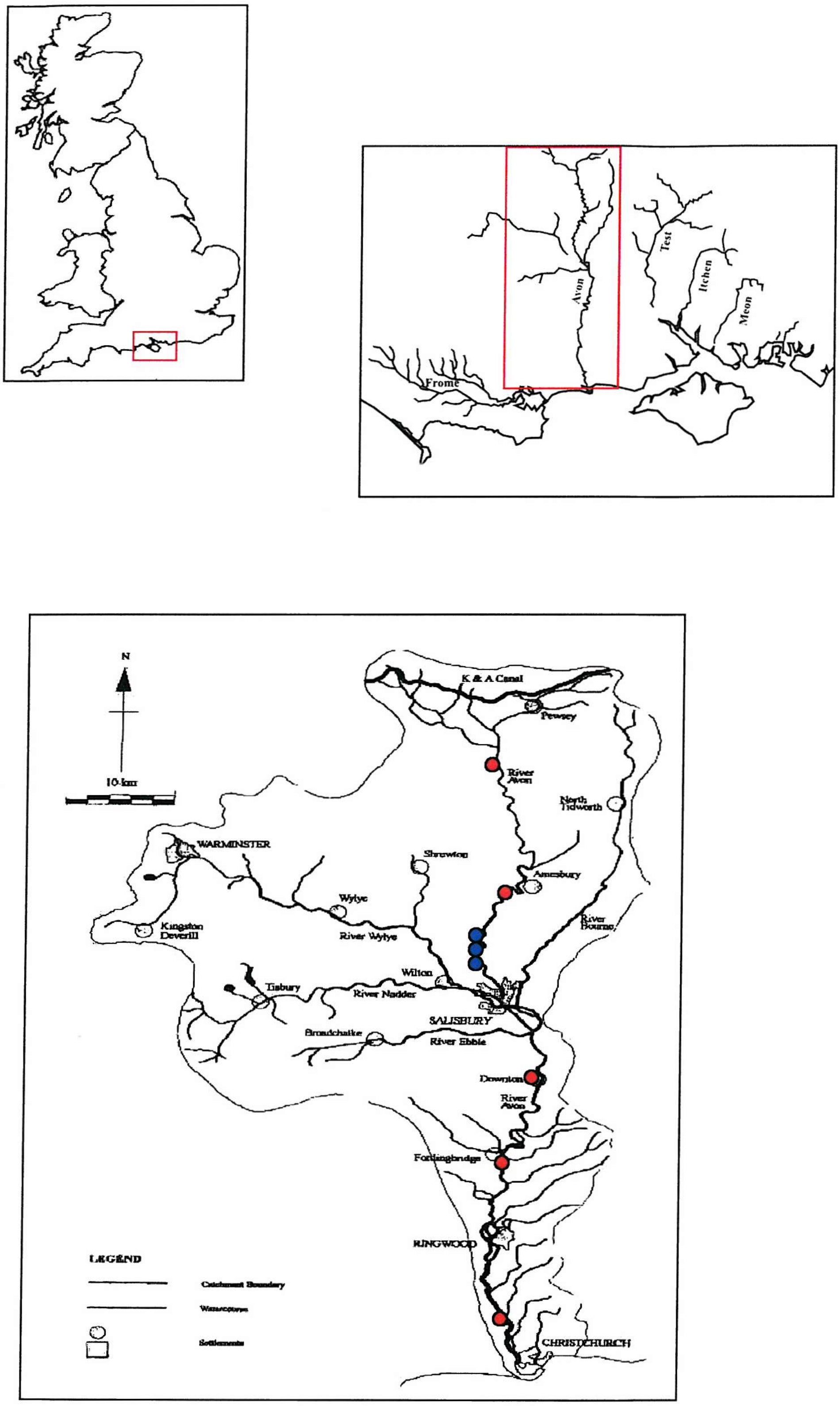


Figure 2.5. Location of the River Avon, UK.
Blue spots represent biological monitoring sites
Red spots represent chemical monitoring sites

2.4 SAMPLING STRATEGY

The objectives of water quality monitoring have been widely discussed together with the constraints, and many different reasons have been suggested for establishing water quality monitoring programmes (Alabaster 1977, Whitfield 1988). For example, Ellis and Lacey (1980) recognised time as an important dimension in the consideration of objectives. Ward and Loftis (1986) also highlighted spatial coverage as being an important criterion for water quality monitoring systems. There are no hard and fast rules for water quality monitoring since every monitoring programme requires special determinands, has different aims, and needs to take account of characteristics which are unique to the particular river/catchment studied (Parr 1994). However, examination of literature concerning the design of any monitoring programme has highlighted the following four factors as major considerations:

- The types of data to be collected at each site
- The number and locations of these sites
- The frequency of measurement of each data series
- The period of time during which the data will be collected (Sanders *et al.* 1983, Parr 1994).

2.4.1 Sampling considerations

Most rivers encompass a wide range of environmental conditions, and support many different types of habitat and organisms (Kerans *et al.* 1992, Cannan 1998). These variations obviously present many difficulties to those charged with responsibilities for monitoring river ecosystems (Cannan 1998). All water quality monitoring techniques and sampling methodologies introduce some bias (Norris *et al.* 1992). For chemical analysis, criticisms are largely concerned with the absence of any ecological impact relevance and unrepresentative nature of discrete samples; for example, in a medium-large river a monthly 1 litre discrete sample may be used to represent the quality of over 0.5×10^9 m^3 of water passing a given point. Continuous monitoring provides a greater resolution, but is available for very few parameters (Mar *et al.* 1986). In order to take representative samples, a wide range of equipment and techniques to cope with many different situations may be required, or alternatively large areas of the river channel will remain unsampled (Cannan 1998).

Ecological studies have frequently shown the patchy and random distribution of organisms (Merritt *et al.* 1995, Cannan 1998) making sampling an important consideration. Patchiness (where the variance of the population is significantly greater than the mean; Resh 1979, Resh and McElravy 1993, Cannan 1998) is a commonly encountered issue in rivers. For sampling programmes assessing potential ecological shifts it is vital to recognise this variability and build a degree of replication into the monitoring (Ward *et al.* 1986, Boulton and Lake 1992, Cannan 1998).

2.4.2 Catchment sampling programme

In arriving at the sampling strategy adopted for this investigation, the above factors were considered. The overall aim of my research was to establish the water quality of southern chalk rivers in the UK, and in order to carry this out a sampling strategy was devised to facilitate the sampling of biological and physico-chemical elements of chalk streams at a catchment scale, enabling catchment-wide comparisons to be made.

In order to assess the water quality of chalk rivers, with particular reference to nutrients and sediment, two simultaneous sampling approaches were adopted, enabling a reasonably high level of spatial coverage and resolution to be achieved within each of the chosen river catchments (catchment monitoring). However, changes in the water quality of a river can be expected to occur not only longitudinally within a river system, but also temporally. Rivers are extremely dynamic systems, undergoing marked changes, both seasonally throughout the year and on an annual basis from one year to the next (Cannan 1998). Therefore, a sampling programme that encompassed temporal monitoring as well as spatial coverage was developed enabling an assessment of the temporal stability of chalk streams (stability monitoring).

Catchment monitoring

For the chemical sampling programme, five sites on each of the Rivers Itchen, Test, Meon, Frome and Avon, were selected as sampling locations. These sites were selected to incorporate specific catchment activities, e.g. water cress discharges and sewage treatment works discharges. The locations of these sampling sites on each river are shown (as red circles) in Figure 2.1 to 2.5, and photographs of selected sites are shown in Appendix A.

For the biological sampling programme, three sampling reaches were established along each of the five sample rivers. Each river had a sedimented, over-widened and control reach, in effect making the rivers themselves the replicates. The sites were not targeted at particular locations, for example, they were distributed along the lengths of the rivers and covered a broad range of hydrological and physical characteristics of chalk rivers. The locations of the 15 biological sampling sites are shown (as blue circles) in Figures 2.1 to 2.5, and photographs of selected sites are shown in Appendix B.

The precise location of the sampling sites was decided primarily through a pilot survey of the catchments involved, but also consideration was given to practical issues. For example, accessibility of the site from the road and access to the channel was of paramount importance (specifically the River Avon). As far as possible, relatively undisturbed locations were chosen for the control sites, although in such a highly managed system it must be recognised that no part of a chalk river will be completely free from human influence (Cannan 1998).

Stability monitoring

Along with being spatially variable in terms of their water quality, rivers are also temporally variable. They undergo marked changes, both seasonally throughout the year and on an annual basis from one year to the next (Cannan 1998). For example, animal communities, which are always moving through their own life cycles, can also be expected to be influenced by these changes in habitat, flow and temperature (Cannan 1998). Therefore a temporal aspect was felt to be an extremely important element of this sampling strategy, as indicated by Cannan (1998).

There is evidence to suggest that in the UK broad changes in invertebrates can be detected by sampling two or three times a year, and even at a seasonal scale (Morgan and Egglishaw 1965, Furse *et al.* 1984, Cannan 1998). Ideally, a high sampling resolution would be used to follow temporal change, however, accepting the time constraints of this study, sampling four times a year was felt to provide an adequate temporal coverage (Cannan 1998). It was also decided that chemical samples would be taken on a seasonal basis as water composition has been shown to vary across this time scale (Walling and Webb 1992). Sampling was carried out on eight occasions spanning two years for both the biological and chemical surveys: summer 1998, autumn 1998, winter 1998, spring 1999, summer 1999, autumn 1999, winter 1999 and spring 2000, as suggested by Cannan (1998).

CHAPTER THREE: PHYSICO-CHEMICAL WATER QUALITY OF CHALK RIVERS

CHAPTER THREE: PHYSICO-CHEMICAL WATER QUALITY OF CHALK RIVERS

3.1 INTRODUCTION

3.1.1 Background

There are many different properties and parameters that describe the chemical characteristics of rivers. These range from general descriptors, such as measures of conductivity and acidity, to composition in terms of major cation and anion content, and the concentration of organic and inorganic pollutants (Hem 1970, Webb and Walling 1992). The present chapter focuses on establishing the chemical characteristics of chalk river systems of varying scales. Key physico-chemical water quality descriptors considered included temperature, conductivity, pH, dissolved oxygen and suspended particulate matter (SPM), along with measurements of the concentrations of inorganic nutrients (e.g. nitrate, nitrite and phosphate).

The transfer of chemical elements through the hydrological cycle involves a complex interaction of hydrological, chemical and biological systems and processes (Neal and Hornung 1990). It follows therefore that the solute content and composition of rivers may vary greatly, both spatially and temporally, depending on the particular sources, processes and pathways that are dominant in a given drainage basin (Webb and Walling 1992). Coupled with this, the extent of variation and the controlling factors responsible for the composition of water will vary depending on the scale of the area under consideration. The scale at which a study is undertaken will also determine the patterns observed (Webb and Walling 1992). Detailed investigations in water composition can be carried out at selected sites, although the examination of the longitudinal patterns of water composition within an entire catchment has both scientific value for furthering our understanding of the river catchment, and practical implications for river management. The increased use of the catchment unit for management purposes has focused attention on evaluating sites in a wider context and raised interest in the spatial pattern of water quality along the length of individual rivers (Sweeting 1994, Cannan 1998).

3.2 CHEMICAL CHARACTERISTICS OF CHALK RIVERS AND CONSEQUENCES OF NUTRIENT LOADING

3.2.1 Characteristic water properties of chalk rivers

Aspects of the water chemistry of chalk streams of southern England have been the subject of much research (Casey 1969, 1975, 1977, 1981, Casey and Ladle 1976, Casey and Newton 1972, 1973, Casey and Smith 1994, Casey *et al.* 1981, 1989, 1990, House and Casey 1989, Ladle and Casey 1971, 1979, 1991, Marker and Casey 1983, Westlake 1968, Westlake *et al.* 1972). Numerous studies have shown that chalk streams often produce a very stable and predictable environment in their unimpacted state (Bickerton *et al.* 1993, Casey and Smith 1994, Prenda *et al.* 1997, Mainstone 1999). Due to the high proportion of chalk stream discharge being derived from groundwater sources, their flows, temperatures and most chemical components are strongly buffered against change (Berrie 1992, Ibbotson *et al.* 1994, Mainstone 1999). Long residence times in the chalk aquifer stabilise water temperatures against the extremes of the seasons (Smith 1980). The characteristic annual temperature range of a sizeable southern UK chalk stream is 5 – 17°C (Mainstone 1999).

Chalk streams have characteristically high alkalinity (180-260 mg l^{-1}) and high conductivity, due mainly to calcium carbonate, and pH values typically lie within the range 7.4-8.5 (Bickerton *et al.* 1993, Mainstone 1999). Owing to the physical filtration of stream water through chalk and the relatively low hydraulic energy of chalk rivers (generating minimal natural bankside erosion), inputs of solid material are naturally low in chalk rivers. The characteristic state is therefore one of clear waters with very low levels of suspended solids and low bed loads of fine sediment. In the unimpacted state, chalk streams have a suspended solids concentration of less than 5 mg l^{-1} (Casey and Newton 1972), and generally contain high levels of all major nutrients (Berrie 1992). Indicative values for key water quality parameters that might be expected in chalk rivers in near pristine conditions are given in Table 3.1, drawing heavily on observations by Robach *et al.* (1996) in French rivers, but also based on subjective judgement of historical UK conditions (Mainstone 1999).

It is recognised that these values should be used as a broad guide only, and were not intended for use in evaluations of specific rivers or river reaches without more detailed assessment of local circumstances. Values for upper, middle and lower reaches take account of nutrient spiralling and the tendency to encounter non-chalk geologies further down the catchment, and also admit some low-level anthropogenic impact from extensive agriculture and low population levels (Mainstone 1999).

Table 3.1. Indicative values (annual means) of key water quality parameters in chalk rivers under near pristine conditions (adapted from Mainstone 1999).

Parameter	Upper reaches	Middle reaches	Lower reaches
Suspended solids (mg l^{-1})	<2	4	6
Total Phosphorous (mg l^{-1})	<0.01	0.02	0.03
Nitrate-N (mg l^{-1})	0.2	0.5	1.0
pH	7.8-8.0	7.8	7.4

3.2.2 Consequences of nutrient loading

Although chalk rivers are recognised as having a naturally high nutrient content, increased anthropogenic activities have been highlighted as adding to these loads. Nutrient concentrations in excess of the natural background levels can cause major consequences for aquatic systems. The term 'eutrophication' describes the nutrient enrichment of aquatic systems and the resultant impact upon primary production (Marsden *et al.* 1997). Many studies have been published relating changes in primary production to nutrient enrichment in standing waters and describing the trophic status of lake waters (Dillion and Rigler 1974, OECD 1982, Moss *et al.* 1996, Marsden *et al.* 1997). In contrast, however, few studies have been carried out defining the impact of nutrient loading on running waters (Descy 1992). Not until relatively recently has the presence of inorganic nutrients in river water become a serious water quality consideration (Environment Agency 2000b). Estimation of the actual loading of surface waters from nutrients is extremely difficult since data referring to the problem are scarce (Modest *et al.* 1992). Subsequently, establishing guidelines for the limitation of nutrient concentrations in running waters has been a much slower process than that for lakes (Welch *et al.* 1989). However, the enrichment of riverine waters by nutrients can have deleterious consequences.

Eutrophication brings about an increase in biomass and productivity of vegetation (Moss 1998), and further increases in nutrient levels potentially produce more severe changes, including reduced species diversity and disruption of community structure (Marsden *et al.* 1997). In particular the growth of algae to nuisance proportions can be an eventual expression of this process, leading to blankets of benthic algae and blooms of phytoplankton. Although eutrophication has been seen to be both beneficial, (with small additions of nutrients resulting in an increase in biomass and productivity of vegetation (Moss 1998)), and harmful to various systems under different conditions, it is assumed that eutrophication as defined should be avoided. Consequently, eutrophication is now recognised as one of the main threats to the ecological quality of freshwaters (Wilson *et al.* 1996). Due to the naturally high levels of nutrients supplied to chalk streams, these systems are potentially highly susceptible to eutrophication, with sections already designated as nitrate-sensitive (e.g. the lower reaches of the River Test).

A wide range of factors determines primary productivity in rivers, the relative importance of which changes seasonally, with the prime limiting factors being light, temperature and nutrients (Reynolds 1984). In addition, factors such as flow, the stability of the riverbed, and biological interactions will determine the extent of algal biomass development (Dodds and Gudder 1992). In most circumstances phosphorus has been found to be the most important nutrient determining primary production in fresh water (Vollenweider 1968). The range of phosphorus concentrations found in fresh water has been divided into a series of trophic categories according to the development of algal biomass. Typically this classification may be described as:

- ultraoligotrophic (extremely unproductive; very low levels of nutrients)
- oligotrophic (very unproductive; waters low in nutrients and algae)
- mesotrophic (moderately productive)
- eutrophic (a very biologically productive water body due to the relatively high rates of nutrient inputs)
- hypertrophic (excess of nutrients impacting upon biological communities)

Any classification scheme based upon the continuum of nutrient/biological interactions is by necessity an arbitrary process and the limitations of such a scheme must be accepted prior to use. The value of such a classification is that it provides a structure that facilitates the description and assessment of aquatic resources (Marsden *et al.* 1997). The trophic status of a catchment depends on the relative scale of nutrient input and river flow. Clearly the relative importance of point and diffuse sources will vary between catchments (Marsden *et al.* 1997). The principal sources and routes of entry of nutrients into chalk river systems have been examined by Mainstone (1999) and are outlined in Figure 3.1. Hence the relative importance of diffuse and point source loads depends on the intensity of human activities in the catchment (Mainstone 1999). For example, the magnitude of non-point source nitrogen and phosphorus loading has been shown to vary considerably depending on the proportion of agricultural land in the catchment (Taylor *et al.* 1986, Wright *et al.* 1991). Runoff rates, timing and extent of crop cover and the hydrological flow pathways linking the land to the stream (Heathwaite and Johnes 1996, Johnes *et al.* 1996), soil type and geology (Rast and Lee 1983) of the catchments have all been investigated with regard to potential nutrient sources.

Modern agricultural practices, particularly the use of fertilisers (Kronvang *et al.* 1995), have been strongly linked with elevated nutrient concentrations in aquatic environments (Neill 1989, Stribe and Fleischer 1991). The application of fertilisers has not only resulted in higher yields and nutrient removal via increased yields, but has also led to an improved nutrient level in the soil which leads to increased loss of nutrients (Isermann 1990). Point sources of nutrients are also important, especially with regard to phosphorous additions to riverine systems. Sewage treatment works tend to dominate the point source load of phosphorus even in rural areas, although fish and watercress farms can be important contributors. A typical fish farm with no effluent treatment can generate the same load of phosphorus as a sewage treatment works serving 1000 people (assuming secondary treatment). Whilst cress beds are amongst the smaller point sources, the usage of phosphorus per unit area is very high and plant uptake can be highly inefficient.

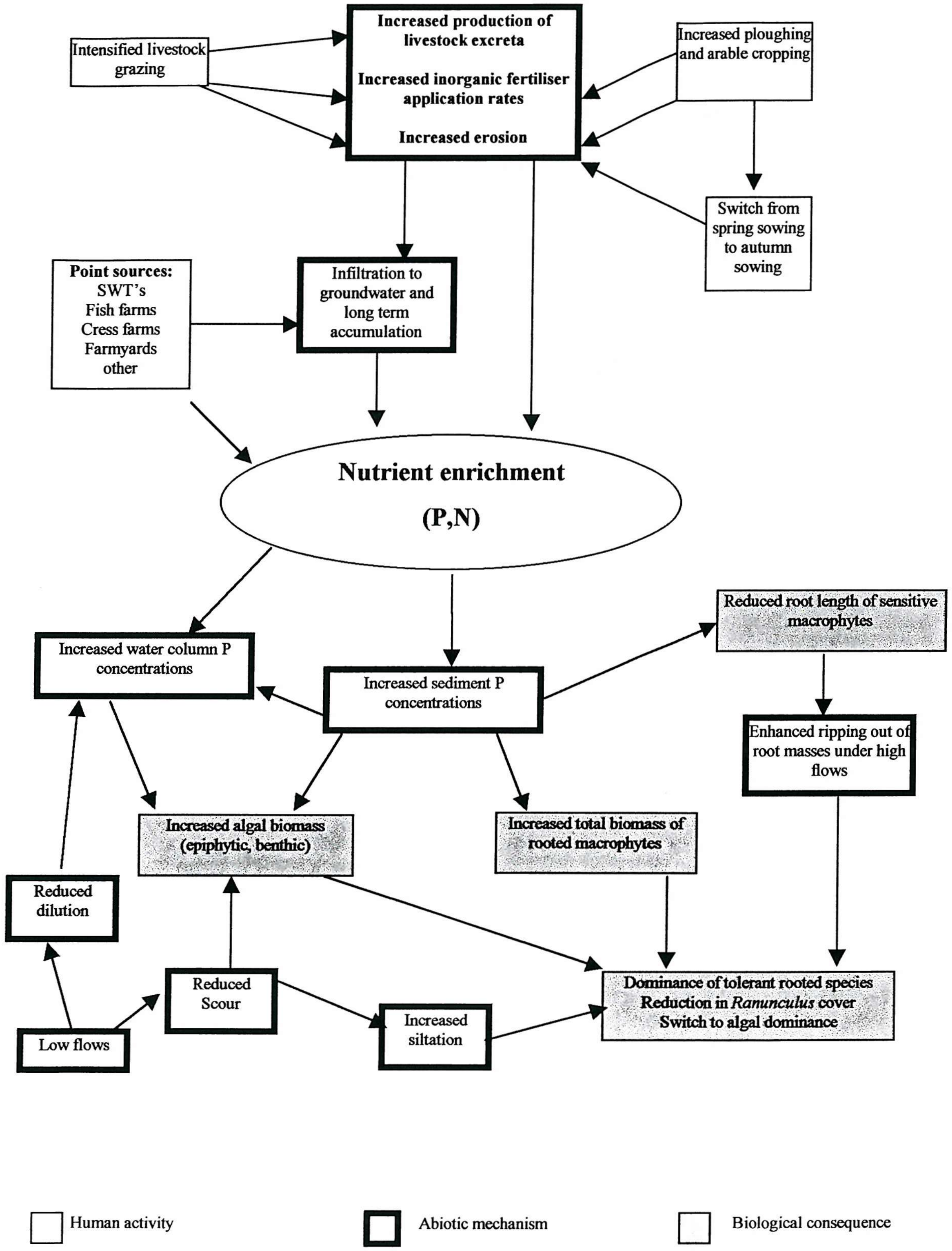


Figure 3.1. Causes and consequences of nutrient enrichment in chalk river systems (adapted from Mainstone 1999).

This is of particular concern since cress farms often constitute the headwaters of chalk rivers, particularly in Hampshire where 80% of the national production of watercress is concentrated (Casey and Smith 1994, Mainstone 1999). Hence, identification of the relative importance of factors in modifying stream water quality is extremely important and requires an understanding of the spatial and temporal variation in water quality parameters and, in particular, nutrient concentrations (Heathwaite and Johnes 1996).

3.2.3 Study aims

Chemical water quality is an important issue in the management of freshwater bodies (Johnes *et al.* 1996). My study aims to investigate longitudinal and seasonal patterns in the chemical water quality and composition of chalk river systems, using several key chemical parameters, at a catchment level. A fundamental distinction is drawn in this study between chemical constituents that are present in dissolved/suspended form and those that are transported in sediment-associated form (Webb and Walling 1992).

It is generally considered that catchment activities impact upon the water quality of southern chalk rivers. My specific aims were to investigate:

- the longitudinal patterns of (a) general water quality parameters and (b) nutrients within individual rivers at local and catchment scale
- the seasonal patterns of (a) general water quality parameters and (b) nutrients within the individual rivers
- the inter-annual variation in the water quality parameters measured within the rivers

Long term analyses were investigated using data acquired from the Environment Agency and published literature. An 18 year data set was established including 11 years of nitrate and phosphate concentration data from previous studies carried out to establish the origin and variation in nutrient concentrations in a Dorset river (the River Frome) (Casey 1977, Casey and Clarke 1979, 1986).

3.3 METHODS AND MATERIALS

3.3.1 Field Methodology

Site selection

Chemical water quality measurements were taken in samples collected from five chalk river catchments located throughout several counties in southern England. An initial survey of the chalk streams throughout Hampshire, Dorset and Wiltshire was undertaken during early spring 1998. Numerous river systems were considered, from which five were selected for study, details of which were outlined previously (Chapter Two). Major consideration was given to choosing representative chalk rivers, primarily selecting sampling areas that are subject to the wide ranging land and river uses/practices normally associated with chalk stream catchments. In terms of land use, the majority of chalk streams flow through agricultural catchments and urban areas which have been intensively used by man for thousands of years (Mainstone 1999). The original braided channels have been replaced by modified channels flowing in a pastoral floodplain, while the chalk downs have been cleared of forest and converted, originally to grassland, but recently to arable fields (Ladle and Westlake 1995). More recently, land uses have been dominated by intensive arable, sheep and dairy production in many parts of the catchment (National Rivers Authority 1992b).

Five sites were located along the length of each of these river systems, encompassing different catchment activities. For example, sites were chosen located upstream and downstream of sewage treatment works, water cress farms and anthropogenically managed fishing lakes. Sites were also located around the few remaining original water meadow systems (especially on the River Avon). Control sites were also selected close to the headwaters of each river course and details of all sites are outlined in Table 3.2. Chemical water quality measurements were taken on a quarterly basis for two years, from summer 1998 to the spring of 2000.

Table 3.2. Location of sites chosen for chemical survey of southern chalk streams and distance downstream from source (shown in Italics).

RIVER				
ITCHEN	TEST	MEON	FROME	AVON
Titchbourne (SU 5795 2975) *2.0 km*	**Southington** (SU 3065 4970) *4.4 km*	**Drayton** (SU 6720 2265) *2.0 km*	**Wraxall** (SU 5765 0085) *6.5 km*	**Endford** (SU 1415 5145) *19.4 km*
Itchen Stoke (SU 5575 3215) *9.0 km*	**Whitchurch** (SU 4610 4725) *11.5 km*	**Warnford** (SU 6210 2220) *8.2 km*	**Bradford Peverell** (SY 6695 9245) *20.5 km*	**Upper Woodford** (SU 1245 3710) *43.5 km*
Easton (SU 5110 3245) *12.0 km*	**Chilbolton** (SU 3895 4010) *23.7 km*	**Droxford** (SU 6095 1810) *13.8 km*	**Woodsford** (SU 7690 9095) *31.5 km*	**Downton** (SU 1775 2160) *63.5 km*
Highbridge (SU 4680 2144) *27.8 km*	**Kimbridge** (SU 3300 2545) *42.1 km*	**Mislingford** (SU 5855 1410) *20.0 km*	**Wool** (SU 8440 8715) *44.0 km*	**Fordingbridge** (SU 1495 1425) *76.6 km*
Woodmill (SU 4420 1525) *36.2 km*	**Nursling Mill** (SU 3515 1550) *54.6 km*	**Titchfield** (SU 5410 0550) *30.0 km*	**Wareham** (SU 9230 8715) *55.0 km*	**Sopley** (SU 1495 9775) *84.1 km*

Following site selection a rigid sampling, processing and analytical strategy was devised, the outline of which is shown in Figure 3.2. The following sections provide details of the field methodologies and laboratory procedures carried out with the aim of determining the chemical composition of different chalk river systems.

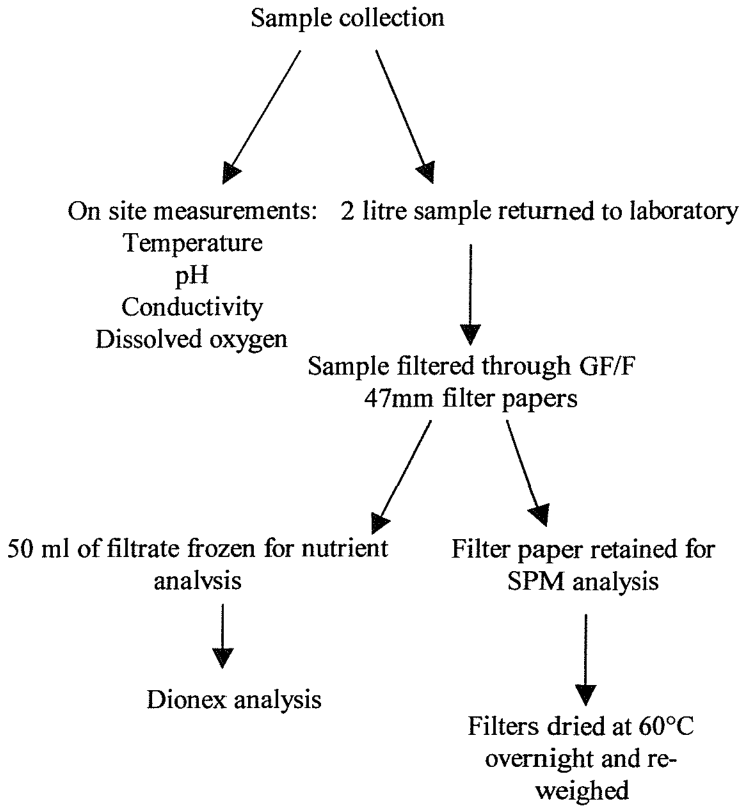


Figure 3.2. Strategy for the chemical sampling and analytical process.

Field techniques

All water samples were collected using a grab technique from the centre of the river channel. Using this sample, a number of parameters were recorded using a variety of hand held instruments, including: temperature (LF 330/SET WTW Instruments, accuracy ± 0.5% of measured value), conductivity (LF 330/SET WTW Instruments, accuracy ± 0.5% of measured value), pH (pHep 3 microprocessor pH meter, accuracy ± 0.1 pH, standardised with pH 4.0, 7.0 and 10.0 standard buffer solutions), and dissolved oxygen: per cent saturation and mg l^{-1} (Oxi 330/SET WTW Instruments, accuracy ± 0.5% of measured value). Following this a 2 litre volume of river water was transferred into a washed, plastic bottle container. A biological preservative was not added to these samples, as they were processed (filtered and stored) within 24 hours of collection (Heathwaite and Johnes 1996).

Following initial visits to the sampling sites, field observations were recorded. For example, any changes in the channel morphology or bankside vegetation were mapped and noted. These observations were made from the bank, without detailed measurements being taken as chalk streams are relatively stable systems (Berrie 1992) and an assumption was made that channel morphology would not change significantly during the period of study (two years), in accordance with Cannan (1998).

3.3.2 Laboratory procedures

Inorganic nutrients

Routine chemical analysis of flowing waters has tended to focus on inorganic determinands: nitrate (NO_3^-) and total phosphorus ($PO^3_4{}^-$). This approach is based on the assumption that inorganic nutrient forms are the key fractions available to organisms in freshwaters (Heathwaite and Johnes 1996). Accepting this, it was decided that nitrate, nitrite and phosphate would be sampled for during this research and used as indicators of the trophic status of southern chalk rivers.

After collection, the 2 litre water samples were transported back to the laboratory and filtered through GF/F 47mm Millipore filters (filter pore size <0.7 μm). These were retained for subsequent particulate matter analysis. For nutrient analysis 50 ml of the filtrate was transferred into washed, labelled glass bottles and stored at -20°C.

Inorganic nutrients (nitrate, nitrite and phosphate) were analysed using ion chromatography and a Dionex system with an ED40 electrochemical detector, GP40 gradient pump and an AS40 autosampler, (a full methodology of the procedure is shown in Appendix C).

Suspended Particulate Matter (SPM)

In order to assess the suspended particulate matter, once returned to the laboratory, the water samples were shaken and a sub-sample immediately poured into a measuring cylinder ready for filtration. This sub-sample was filtered though GF/F

47mm Millipore filters (pore size <0.7 µm). Tests have shown that filters of this pore size retain 90% of the solids from chalk streams (Farr and Clarke 1984).

Between 0.25 and 0.8 litre of each water sample collected was filtered according to the suspended particulate matter concentration of the specific sample and the water volume available (Farr and Clarke 1984). After filtration the papers were placed in freezer proof containers, labelled and stored at - 20°C. At a convenient time after sample collection the filters were removed from the freezer and defrosted. The filter papers were dried at 60°C overnight with each paper triplicate weighed to ensure analytical accuracy.

3.3.3 Data analysis

The results generated from this physico-chemical survey of chalk rivers have been analysed through univariate methods. The longitudinal and seasonal variation in parameters has been investigated using Analysis of Variance (ANOVA). ANOVA is a well established and powerful parametric technique for comparing means of a single variable (Sokal and Rohlf 1981). The tests can be used to investigate for effects that may be quite small compared to the overall variability of the system (Dijk 1987). Its major uses include assessing the statistical significance of differences in a measure among various water bodies, among different locations in the same water body, or among samples taken from one location at different times; and partitioning total variability in values of a measure into components attributable to variation among sites or times and components attributable to each of one or more levels of replication (Norris and Georges 1993, Underwood 1994). Further detail for the specific ANOVA used will be outlined in the individual results sections.

3.3.4 Construction of interannual data sets

Following the analyses of data generated through this study, an investigation was carried out into the interannual variation in the nutrient status of selected chalk rivers. Initially an eight year data set was constructed through the acquisition of five years nitrate and phosphate data from the Environment Agency. Data were obtained from the Environment Agency that corresponded with the sites and months sampled during my research, and were averaged to provide annual means. In the case of nitrate concentrations, my results are added to Environment Agency data spanning a five year period, generating an eight year data set. The analysis of the eight year phosphate concentrations has been carried out on Environment Agency data alone, a factor which will be discussed in detail at a later stage.

An even longer timescale for nitrate and phosphate comparisons was available through the extensive research carried out during the 1960s, 1970s and 1980s on the River Frome. Much research was concentrated on establishing the origin and variation of nitrate and phosphate in Dorset chalk rivers of varying scale (Casey 1977, Casey and Clarke 1986). Results derived from this literature were added to the eight year data sets to establish any trends in the key nutrients over an eighteen year period.

3.4 VARIATIONS IN THE PHYSICO-CHEMICAL COMPOSITION OF CHALK RIVERS

3.4.1 Spatial and temporal variations

The physico-chemical data comprised a total of 200 measurements for each of the eight water quality parameters (temperature, pH, conductivity, dissolved oxygen, SPM, nitrate, nitrite and phosphate) collected in Summer 1998, Autumn 1998, Winter 1998, Spring 1999, Summer 1999, Autumn 1999, Winter 1999 and Spring 2000. A summary of the raw data collected is shown in Appendix D.

The aim of this study was to investigate the longitudinal and seasonal variation in certain physico-chemical parameters of chalk streams, from which conclusions will be drawn about their overall water quality status. In order to achieve this, the significance of any between site and season differences were assessed statistically using ANOVA. A three way randomised block ANOVA (Zar 1996) was chosen to analyse the data. All parameters (with the exception of pH) were log transformed to reduce the influence of extreme values. Results have been presented for each water quality parameter on each river, establishing the degree of spatial (between sites and rivers) and temporal (between season and sampling years) variation. Figures 3.3 to 3.10 show the variation of physico-chemical parameters for each of the five rivers sampled.

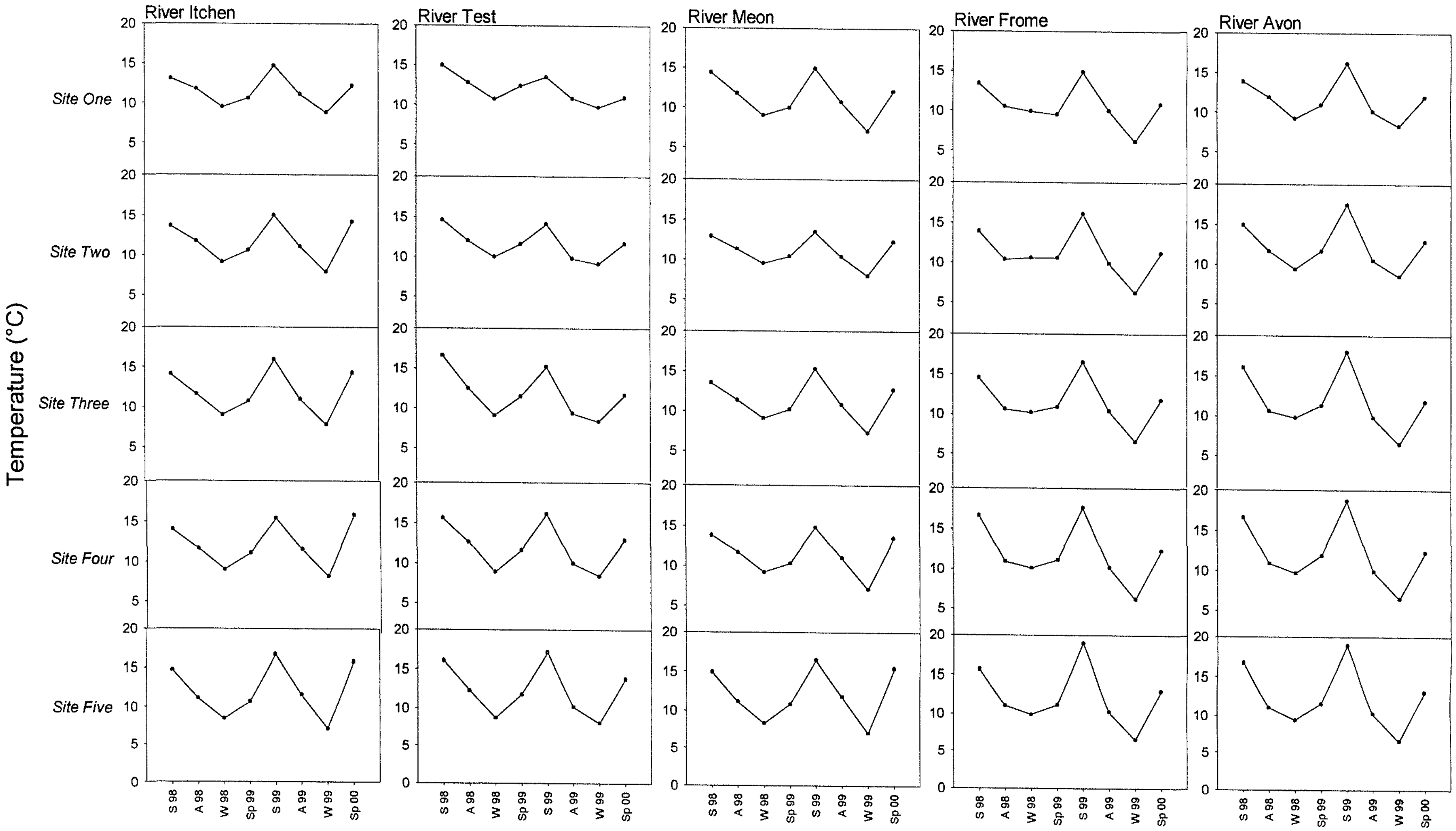


Figure 3.3. Seasonal temperature at each site across the five rivers.

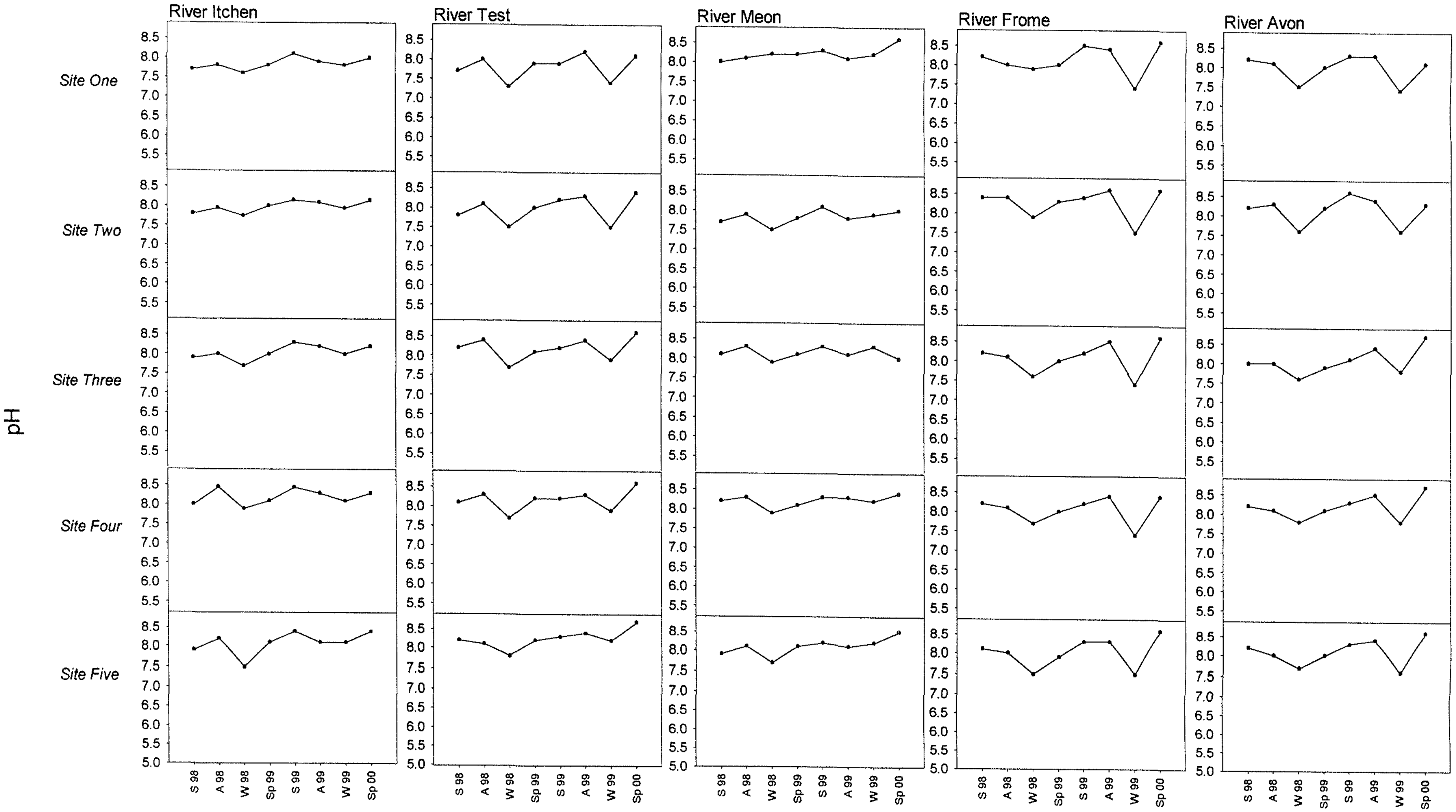


Figure 3.4. Seasonal pH at each site across the five rivers.

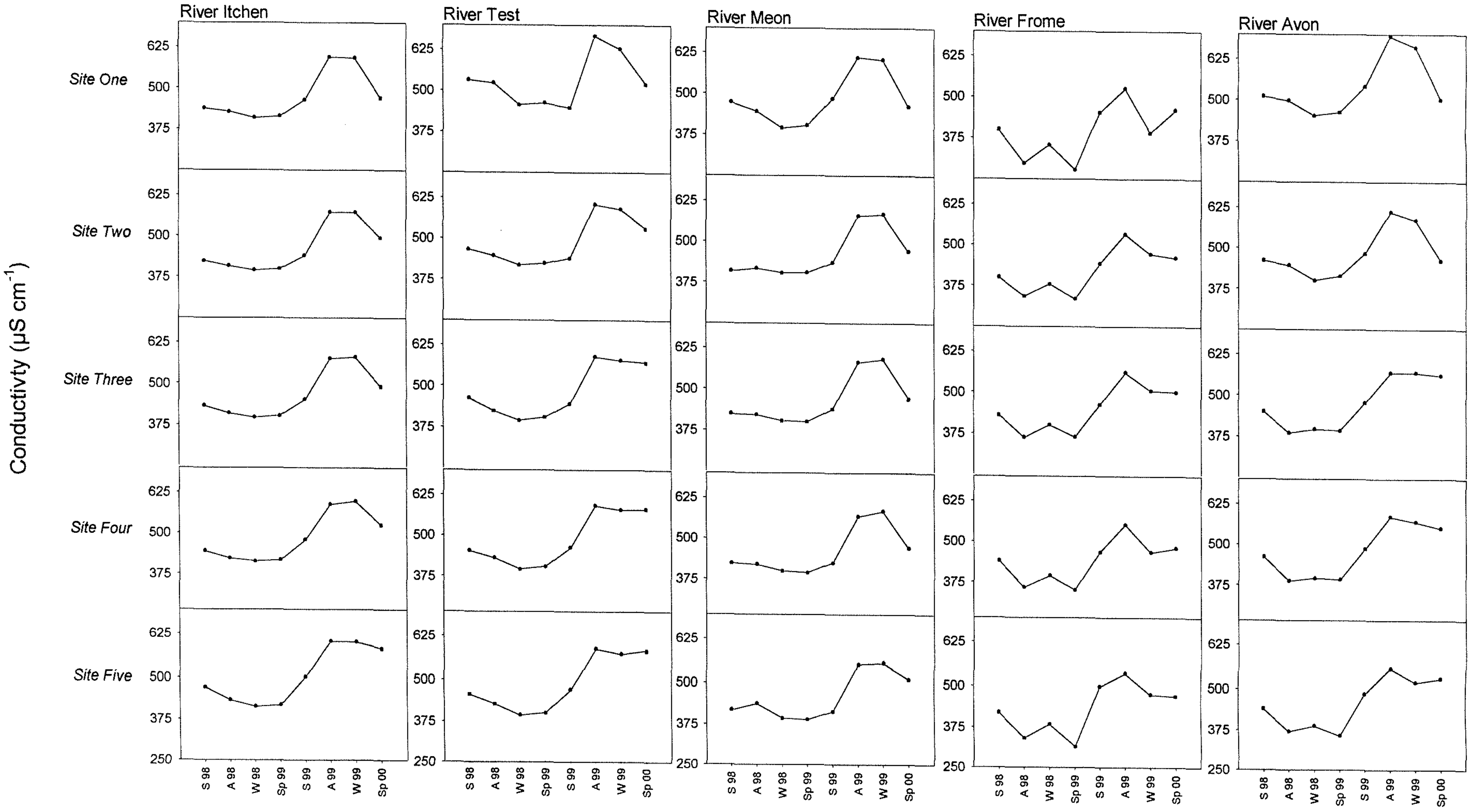


Figure 3.5. Seasonal conductivity at each site across the five rivers.

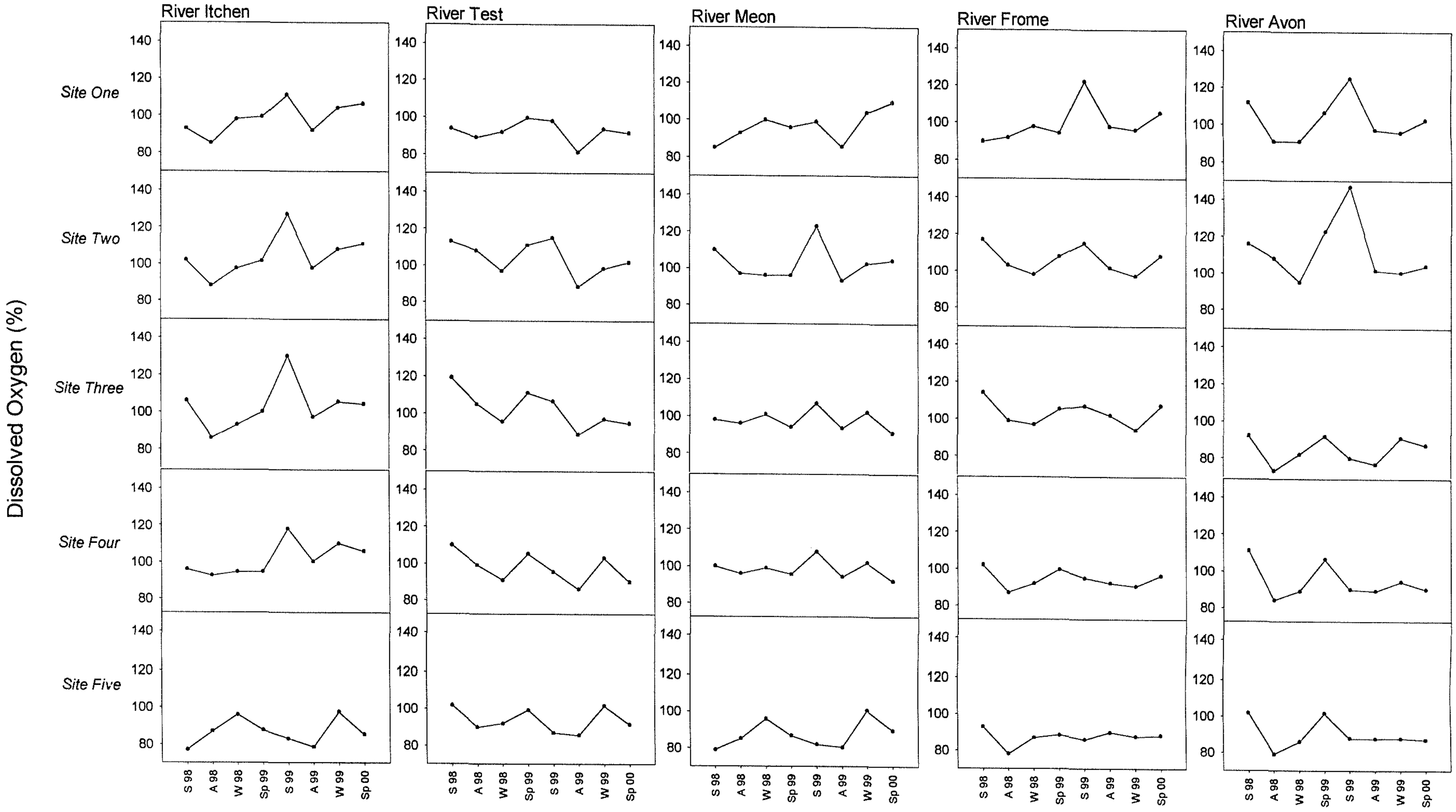


Figure 3.6. Seasonal dissolved oxygen at each site across the five rivers.

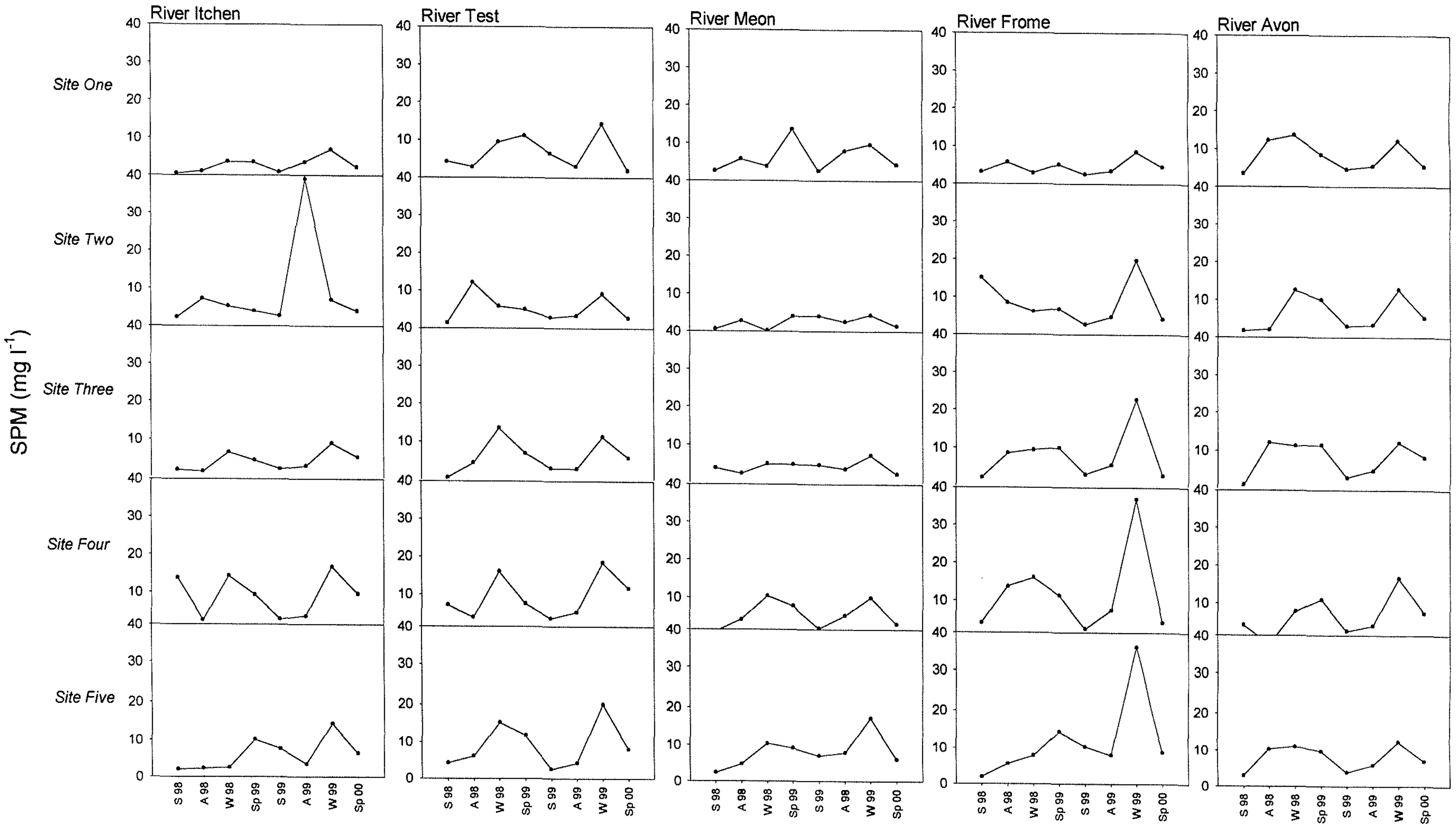


Figure 3.7. Seasonal SPM at each site across the five rivers.

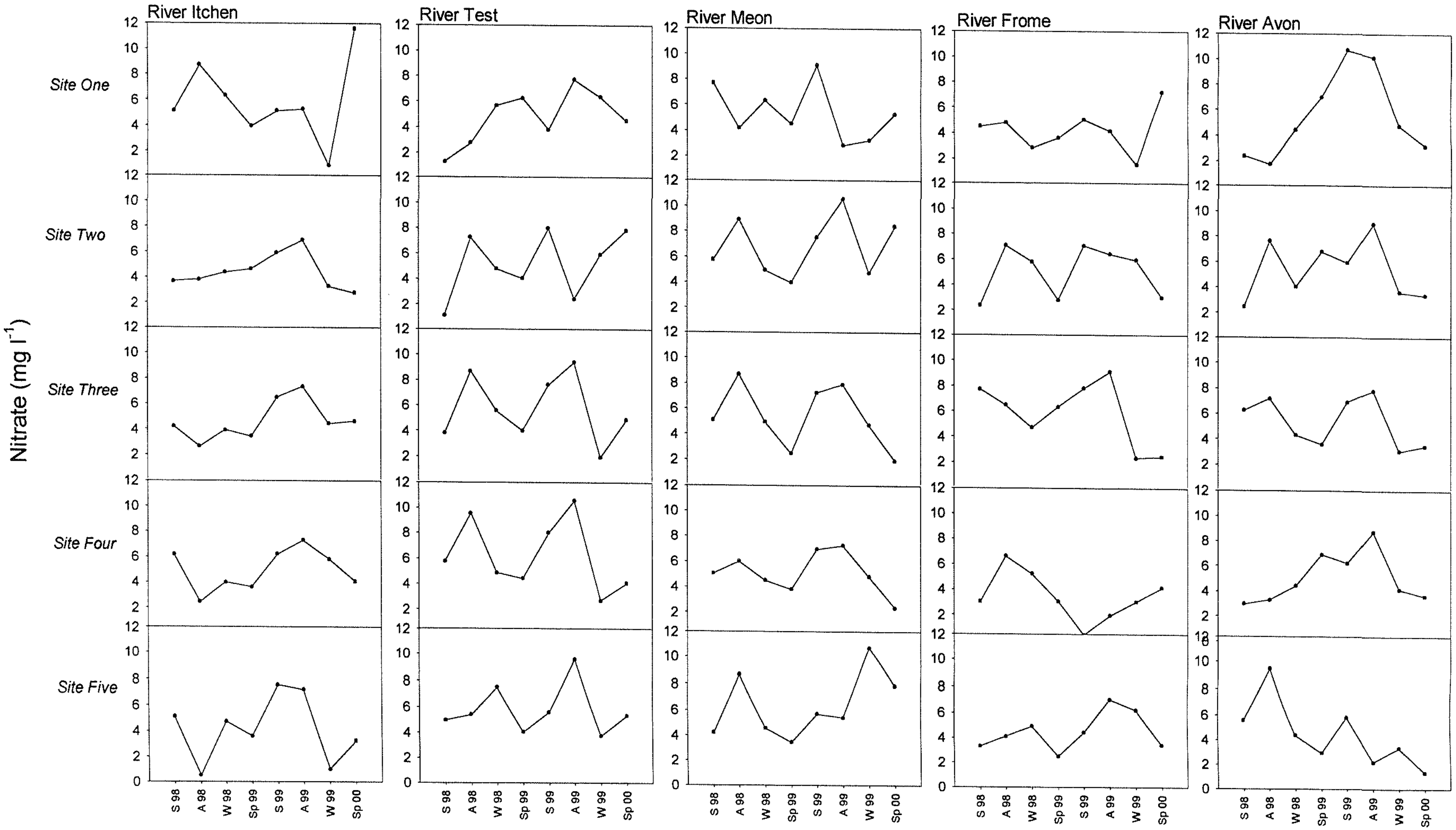


Figure 3.8. Seasonal nitrate at each site across the five rivers.

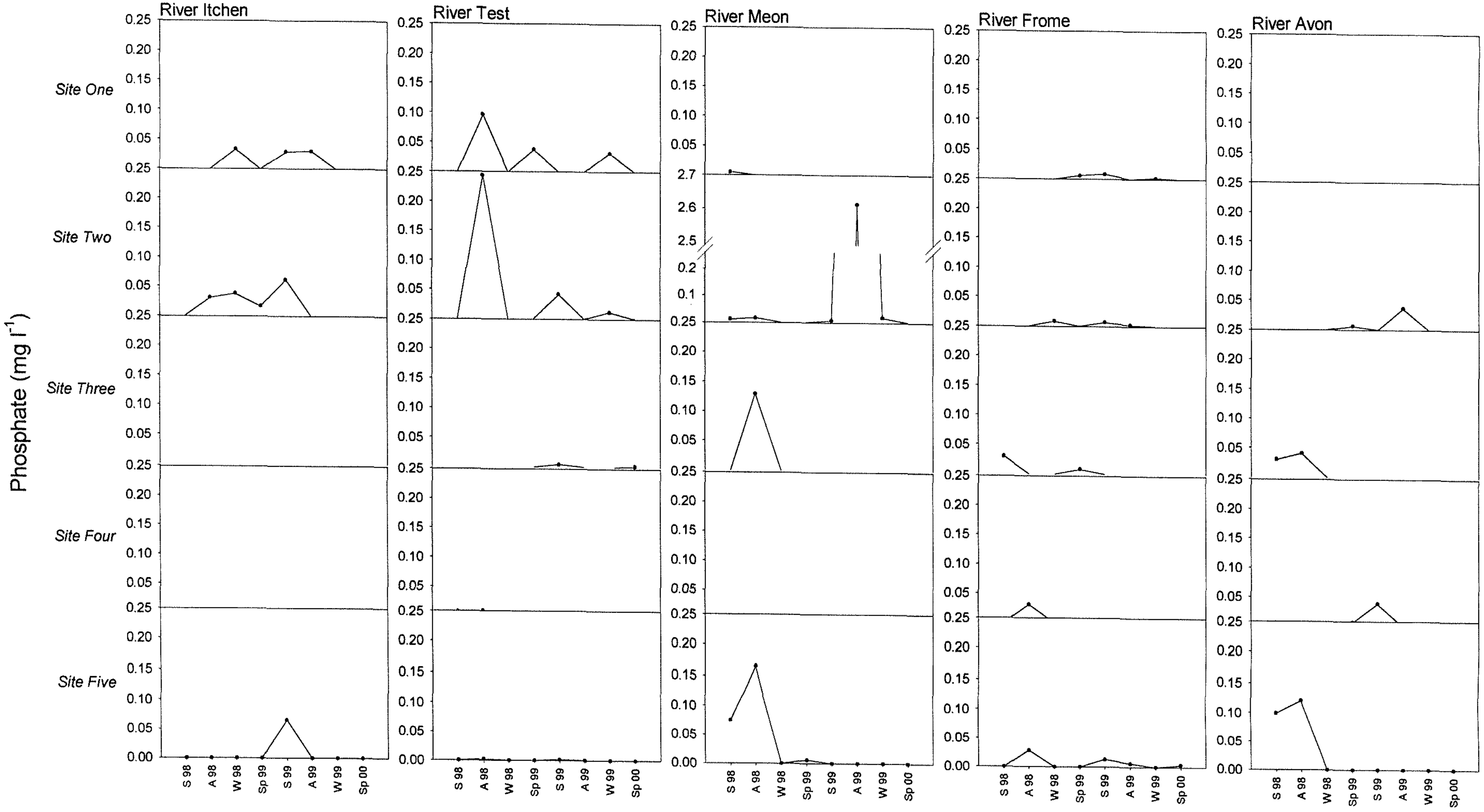


Figure 3.9. Seasonal phosphate at each site across the five rivers.

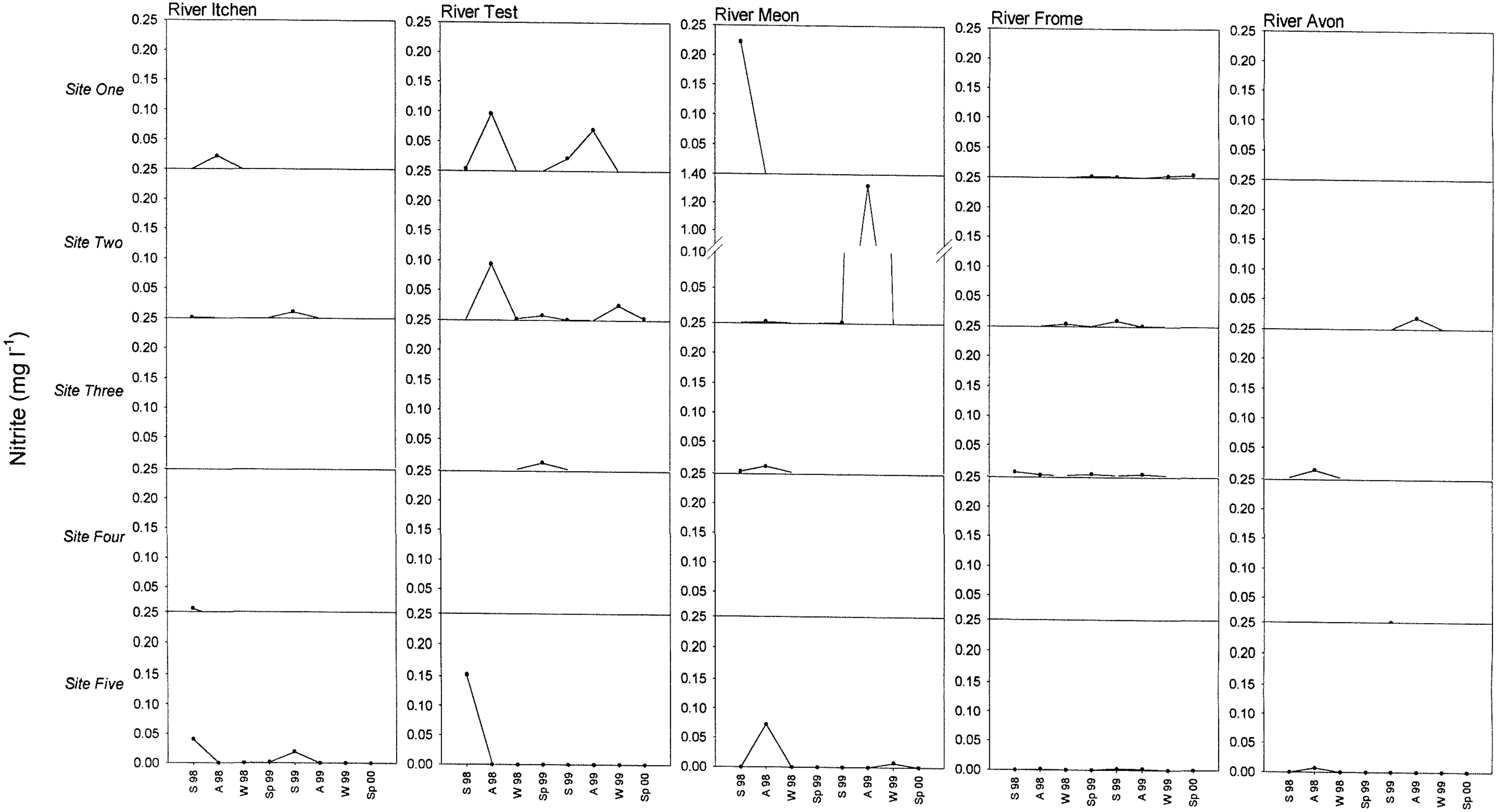


Figure 3.10. Seasonal nitrite at each site across the five rivers.

Table 3.3. Three way randomised block ANOVAs with site, season and year as main factors and river as block. Significant results (p-value <0.05) are in bold.

Measure	Site	Season	Year	River
Log Temperature	0.277	**<0.001**	0.146	0.349
pH	**0.004**	**<0.001**	**<0.001**	0.418
Log Conductivity	0.173	**<0.001**	**<0.001**	**<0.001**
Log Dissolved Oxygen	**<0.001**	**<0.001**	0.726	0.331
Log SPM	**0.008**	**<0.001**	0.332	**0.006**
Log Nitrate	0.780	**<0.001**	0.149	0.072
Log Nitrite	0.393	0.176	0.727	0.239
Log Phosphate	0.236	0.080	0.745	0.353

Measure	Site*Season	Site*Year	Season*Year	Season*Site*year
Log Temperature	0.103	0.820	**<0.001**	0.985
pH	0.993	0.574	**0.013**	0.942
Log Conductivity	0.370	0.383	**<0.001**	0.672
Log Dissolved Oxygen	**0.006**	0.468	0.774	0.196
Log SPM	0.150	0.972	**0.001**	0.370
Log Nitrate	0.192	0.821	**0.003**	0.807
Log Nitrite	0.347	0.326	0.483	0.645
Log Phosphate	0.307	0.402	0.920	0.639

Temperature

All five rivers displayed a very similar longitudinal and temporal pattern in temperature regime. For example, on all rivers it can be seen that the amplitude or range of temperatures increased with distance downstream (Figure 3.3). This is especially the case on the Rivers Itchen, Test and Avon. A very predictable seasonal pattern was seen at each site sampled, with winter minima and summer maxima. When statistically analysed some interesting patterns were highlighted (Table 3.3). There was no significant difference between the five rivers, as suggested by Figure 3.3, and there was no significant difference between the sampling years. Slightly surprising was the lack of statistical difference between sites (p-value 0.277), given the increase in temperature range with increasing distance downstream on all five rivers. The lack of statistical difference could be attributed to the extremely significant seasonal pattern masking this secondary variation within the rivers. Indeed, when the range of temperatures and maximum temperatures were assessed by one-way ANOVAs, with site as main factor, the results were significant (p-values <0.001 and <0.001 for temperature range and maximum temperature respectively), indicating a clear downstream pattern (Figure 3.11).

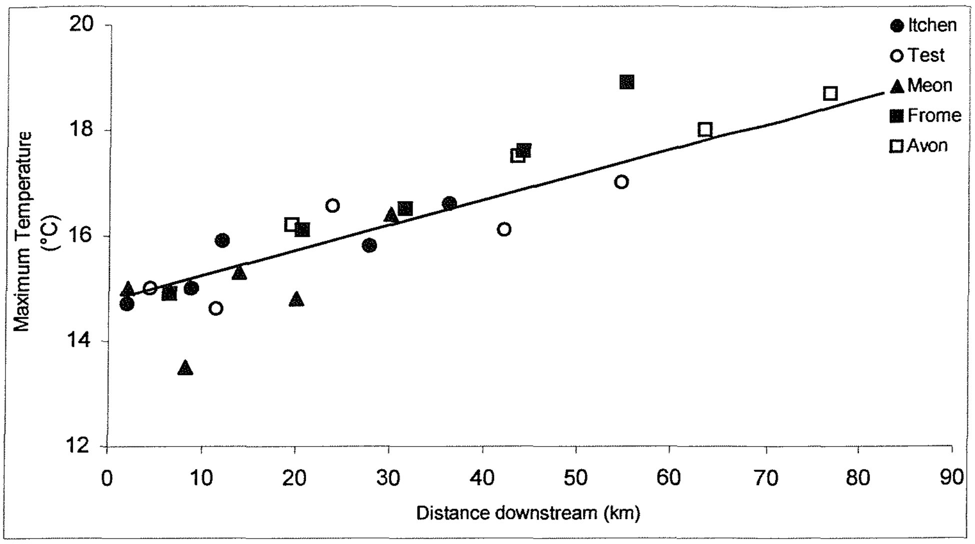


Figure 3.11. Relationship between distance downstream and maximum temperature on the five sample rivers.

The only significant interaction was that of season and year (p-value <0.001). This indicates that the seasonal pattern of temperatures were not the same between the two sampling years.

pH

Again there were no significant differences between the rivers in terms of their pH measurements (p-value 0.418), a pattern clearly shown in Figure 3.4. Another pattern shown in Figure 3.4 is that pH values tend to increase with distance from source, a pattern supported by the significant difference between sites (p-value 0.004, Table 3.3). Again, there was a highly significant difference between seasons (p-value <0.001) and sampling years (p-values <0.001). The difference between sampling years can perhaps be attributed to higher levels of variation on the Rivers Frome and Avon. Again, the only significant interaction was that of season and year suggesting again that the seasonal pH varied differently between the two years, the pattern previously highlighted on the Rivers Frome and Avon.

Conductivity

Figure 3.5 clearly shows that conductivity values varied between the two sampling years (p-value <0.001, Table 3.3). Very little between site differences were observed in Figure 3.5, a finding supported by the p-value of 0.173 (Table 3.3). For the first time there was a significant difference between the rivers (Table 3.3), with a post hoc test revealing that the River Frome had consistently significantly lower conductivity values compared to the other four rivers. Coupled with this seasonal variation was highly significant (p-value <0.001). Again, the only significant interaction was that of season and year (p-value <0.001). This shift in seasonal conductivity values between the two years can be attributed to the relative increase in conductivity values in Autumn 1999 and Winter 1999 compared to the previous sampling year.

Dissolved Oxygen

A downstream pattern can be seen in Figure 3.6, with a general decrease in dissolved oxygen with increasing distance from source on all five rivers. The pattern was especially clear on the River Itchen and Frome. This pattern is supported by the p-value of <0.001 for between site differences (Table 3.3). Coupled with this, there were significant differences between seasons (p-value <0.001) (Table 3.3), generally thought to be attributable to summer increases. There were no differences between sampling years (p-value 0.726) and rivers (p-value 0.331) (Table 3.3). However, there was a significant interaction between site and season (p-value 0.006) suggesting that the sites did not behave in the same way with respect to their dissolved oxygen measurements over the seasons.

SPM

Figure 3.7 again clearly shows a difference between the sites on the five rivers, with SPM concentrations significantly increasing with distance downstream (p-value 0.008) (Table 3.3). The seasonal variations in SPM concentrations were also significant (p-value <0.001), with generally higher SPM concentrations observed during winter months (e.g. River Frome and Avon). There were no significant differences between the sampling years (p-value 0.332) (Table 3.3). However, there were significant differences between rivers (p-value 0.006) and a post hoc test revealed that the River Frome had significantly higher concentrations compared to the other four rivers. The only significant interaction was that between season and year (p-value 0.001) (Table 3.3), suggesting that the seasons varied differently between the two sampling years with respect to their SPM concentrations. This could perhaps be due to the higher levels of seasonal variation in the second year of sampling (e.g. River Frome).

Nitrate

Nitrate concentrations varied greatly, both temporally and spatially. There were no significant differences between sites (p-value 0.780), suggesting that upstream and downstream sites had similar concentrations of nitrates. However, there was a significant seasonal difference (p-value <0.001) (Table 3.3). Although rather difficult to highlight from Figure 3.8 Autumn values were generally elevated, especially during the second sampling year (e.g. River Avon, Test and Meon). No significant differences were found between sampling years (p-value 0.149) and rivers (p-value 0.072) (Table 3.3). Again, the only significant interaction was that between season and year (p-value 0.003), indicating that seasonal nitrate concentrations behaved differently between the two sampling years.

Phosphate and Nitrite

Phosphate was generally found in extremely low concentrations throughout the two years of sampling, with the exception of site two on the River Meon. There were no significant patterns between sites, seasons, years and rivers or interactions (Table 3.3), however slightly elevated concentrations were recorded during summer and autumn (Figure 3.8). Nitrite concentrations showed an extremely similar pattern to that of phosphate concentrations previously described. The only exception to this pattern was site two on the River Meon, where concentrations reached 1.39 mg l^{-1}, far in excess of anything found at any of the other sites. There were no significant differences (Table 3.3), however, Figure 3.9 does again show slightly elevated concentrations during summer and autumn.

3.4.2 Interannual variations in the physico-chemical composition of chalk rivers

When chalk rivers are discussed in general terms much weight is given to their temporal stability, especially in terms of their chemical composition (Berrie 1992). The term temporal stability often refers to the lack of interannual variation of key chemical parameters. Therefore, consideration of water quality/composition also needs to be examined beyond the two years of study feasible within the current research. Nitrate and phosphate were chosen as general descriptors of the nutrient status of chalk rivers as they are often used to determine the trophic status of rivers (Marsden *et al.* 1997).

Nitrate concentrations from my study were added to the previous five years of Environment Agency data. In order to assess the long term patterns of nitrate and phosphate concentrations in the rivers an unreplicated randomised one way block ANOVA (Zar 1996) was carried out on the annual means of these nutrients.

Statistical results

It is recognised at this stage that different methods were used in the determination of the nutrients. However, these results will (1) act as a quality control measure with regards to under or over estimation of values, and (2) provide an idea of the interannual variation in nitrate concentrations. There were significant differences between the annual nitrate mean over the eight sampling years (p-value 0.003). A post hoc Tukey test revealed that these year differences were between the Environment Agency sampling year 1995 and my sampling years of 1998 and 2000. None of the other sampling years differed from one another. Although this result does not fully rule out the potential of operator bias in the determination of nitrate, the lack of significant difference between all of my sampling years and the Environment Agency data does suggest that my determination methods were not consistently under or over estimating nitrate values. The result also indicates that there was little over all interannual variation despite the differences in nitrate determination methods, operator and number of samples taken. This finding is further shown in Figures 3.12, 3.14, 3.16 and 3.18.

Similar statistical analyses were carried out on the eight years of Environment Agency phosphate data in order to gauge the degree of interannual variation. There was no issue of operator bias as all the data were obtained from the Environment Agency. There were no significant differences between the eight sampling years (p-value 0.478), indicating very little interannual variation in the phosphate concentrations of the sampled rivers. This pattern is further displayed in Figures 3.13, 3.15, 3.17 and 3.19.

River Itchen

Figures 3.12 and 3.13 show the annual mean concentrations of nitrate and phosphate concentrations in the River Itchen. Figure 3.12 shows that nitrate concentrations exhibited a degree of interannual variation. For example, concentrations increased from 1993 to 1995 and declined for the next three years. Figure 3.13 displays phosphate concentrations from 1998 to 2000. No data were available for the sites sampled from 1993 to 1997. There was no evidence of any major shift in phosphate levels over the three years.

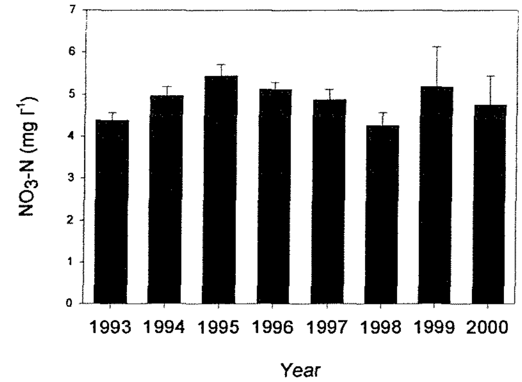


Figure 3.12. Annual mean nitrate concentrations from 1993 - 2000, River Itchen, Hampshire.
Vertical bars show 1 Standard Error.
Data source: 1993 to 1997 nitrate concentrations, Southern Environment Agency.

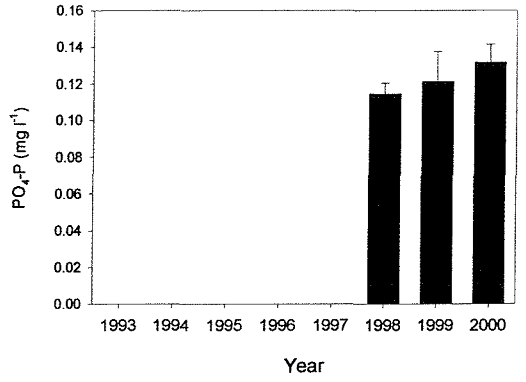


Figure 3.13. Annual mean phosphate concentrations from 1998 - 2000, River Itchen, Hampshire.
Vertical bars show 1 Standard Error.
Data source: 1998 to 2000 phosphate concentrations, Southern Environment Agency.

River Test

Nitrate values in the River Test showed a very similar interannual pattern to those recorded in the River Itchen, increasing from 1993 to 1995 and decreasing during the following three years. However, the differences observed between years were small, varying by less than 1 mg l^{-1} each year, indicating that there was little overall interannual difference in nitrate concentrations.

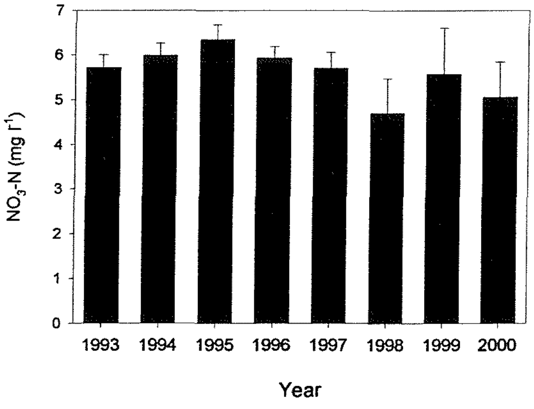


Figure 3.14. Annual mean nitrate concentrations from 1993 - 2000, River Test, Hampshire.
Vertical bars show 1 Standard Error.
Data source: 1993 to 1997 nitrate concentrations, Southern Environment Agency.

Figure 3.15 displays phosphate concentrations from 1993 to 2000 and a degree of interannual variation in concentrations is evident. For example, concentrations decreased from 0.26 to 0.16 mg/l between 1995 and 1996. Overall, a general decrease was seen in phosphate concentrations during the period considered.

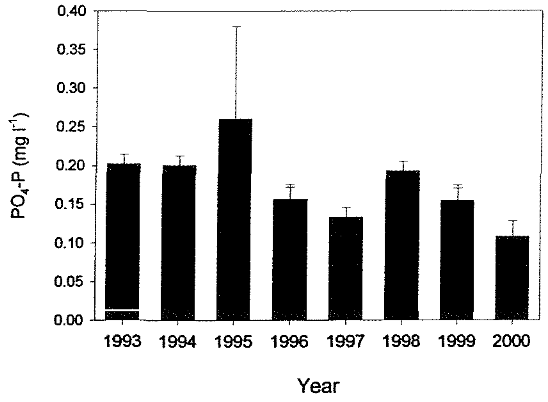


Figure 3.15. Annual mean phosphate concentrations from 1993 - 2000, River Test, Hampshire.
Vertical bars show 1 Standard Error.
Data source: 1993 to 2000 phosphate concentrations, Southern Environment Agency.

River Meon

A distinct trend of decreasing nitrate concentrations over the eight years (from 6.52 mg l^{-1} in 1993 to 5.15 mg l^{-1} in 2000) is evident in Figure 3.16. Figure 3.17 displays the phosphate concentrations over a six year period. No data were available for 1993 and 1994. During the six year period little interannual variation in phosphate concentrations was evident increasing from 0.8 mg l^{-1} to 0.13 mg l^{-1}.

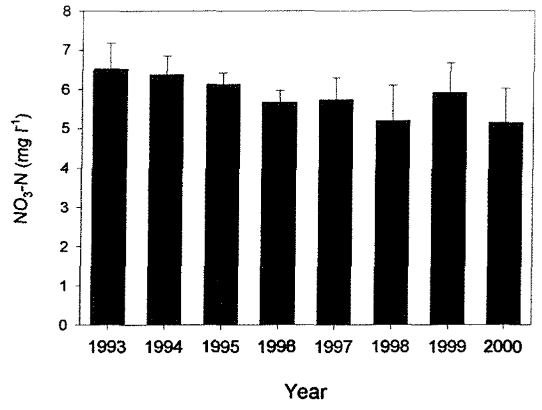


Figure 3.16. Annual means of nitrate concentrations form 1993 - 2000, River Meon, Hampshire.
Vertical bars show 1 Standard Error.
Data source: 1993 to 1997 nitrate concentrations, Southern Environment Agency.

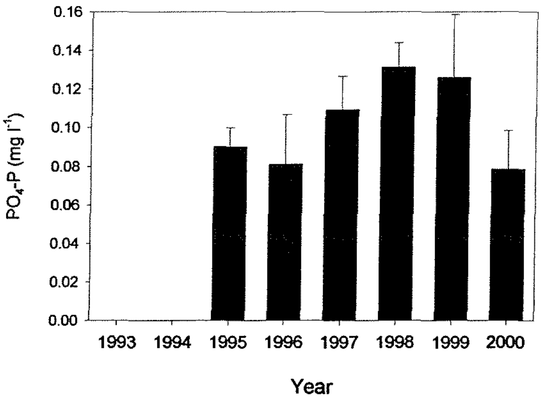


Figure 3.17. Annual means of phosphate concentrations from 1995 - 2000, River Meon, Hampshire.
Vertical bars show 1 Standard Error.
Data source: 1993 to 2000 phosphate concentrations, Southern Environment Agency.

River Frome

Figure 3.18 displays the nitrate concentrations in the River Frome. Concentrations increased from 1993 to 1995, after which values decreased. However, these changes in concentration were extremely low, varying less than 1 mg l^{-1} over the eight year period.

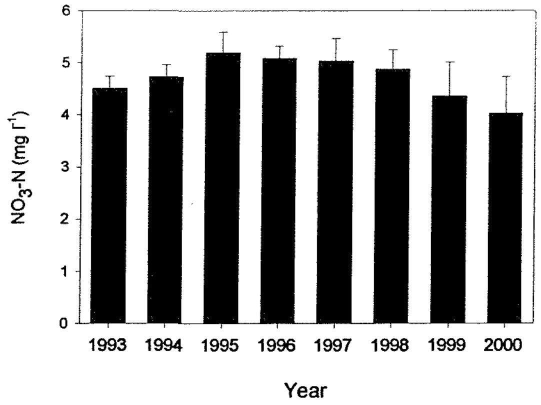


Figure 3.18. Annual means of nitrate concentrations from 1993 - 2000, River Frome, Dorset.
Vertical bars show 1 Standard Error.
Data source: 1993 to 1997 nitrate concentrations, South West Environment Agency.

In contrast to previous rivers, phosphate values in the River Frome (Figure 3.19) were relatively constant from 1993 to 2000. Any increase or decrease in concentration between years was extremely small.

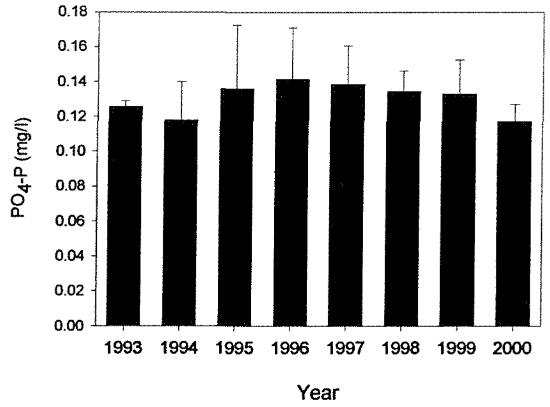


Figure 3.19. Annual mean phosphate concentrations from 1993 - 2000, River Frome, Dorset.
Vertical bars show 1 Standard Error.
Data source: 1993 to 1997 phosphate concentrations, South West Environment Agency.

3.4.3 Long term patterns

Nitrate concentrations 1965 – 2000, River Frome

Figure 3.20 shows the long term trends in the nitrate concentrations recorded in the River Frome from 1965 to 2000. Casey and Clarke (1979) carried out statistical analyses on 11 years of weekly data for nitrate concentrations from the River Frome from 1965 to 1975 (Figure 3.20), establishing concentrations increased by 0.11 mg l^{-1} per year. Nitrate concentrations continued to rise through the early 1990s, reaching a peak in 1995 after which values declined. Throughout the period covered by the Environment Agency there was an overall increase of 0.53 mg l^{-1}. Results from my study indicate that annual nitrate concentrations continued to decline, and throughout 1998 – 2000 there was an overall decline of 0.85 mg l^{-1}.

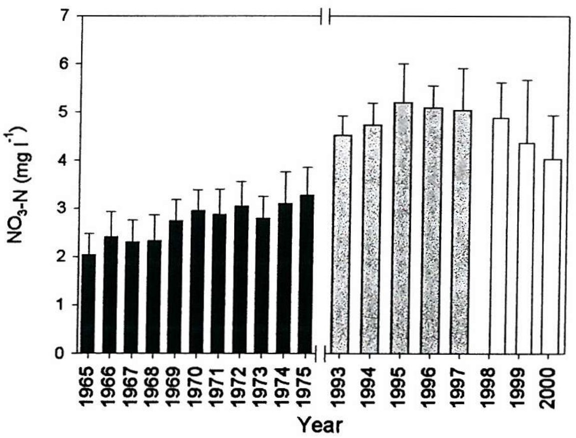


Figure 3.20. Long term trends in nitrate concentrations, River Frome Dorset. Vertical bars show 1 standard error.

■ Data derived from Casey and Clarke (1979)
▨ Data derived from the South West Environment Agency
□ Data from my research

Phosphate concentrations 1965 – 2000, River Frome

Figure 3.21 displays the long term variation in phosphate concentrations from 1965 to 2000 in the River Frome. Casey and Clarke (1986) established that levels increased from 1968 and 1971 (from 0.1 to 0.14 mg l^{-1}), after which there was no coherent variation. Considerable variation was observed in the concentrations within each year, generally showing a spring depression in levels, possibly related to biological activity (Casey and Clarke 1986). Figure 3.21 also shows that Environment Agency phosphate continued to fluctuate, exhibiting a high degree of interannual variation.

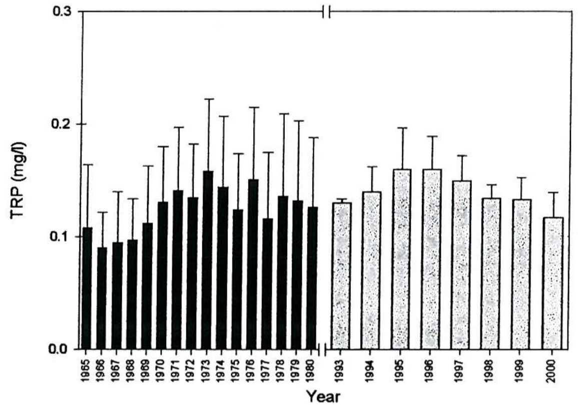


Figure 3.21. Long term trends in phosphate concentrations, River Frome Dorset. Vertical bars show 1 standard error.

■ Data derived from Casey and Clarke (1986)
▨ Data obtained form South West Environment Agency

3.5 DISCUSSION

Rain falling on chalk land percolates through the soil accumulating in large aquifers. The speed of water movement through the chalk is not known precisely but tritium dating indicates that downward movement is slow (0.8 m yr^{-1}), taking many years to return to surface flows as spring water (Casey 1977). When the water returns to the river channel, given the length of time percolating through the chalk bedrock, it is thought to be extremely pure in terms of its water chemistry and quality (Berrie 1992). However, over the past centuries there has been extensive anthropogenic use of chalk rivers and their surrounding catchments, many of which have been linked with an overall degradation in ecosystem quality.

3.5.1 Potential method limitations

Given the availability of the Environment Agency data for similar sites and time periods sampled during my research it was decided a comparison should be made prior to examination and discussion of nutrient water quality trends. Some discrepancies became obvious for phosphate and nitrite concentrations. Simple visual examination of these nutrient data indicated that concentrations were extremely low, which in itself raised concern as chalk rivers are generally referred to as nutrient rich (Berrie 1992, Mainstone 1999). When the data were compared in more detail to Environment Agency measurements it appeared that concentrations for both phosphate and nitrite were a factor of 10 lower and, hence not surprisingly, highly significant differences existed between the two data sets.

A variety of methods have been used for the analysis of inorganic nutrients, including traditional spectrophotometric techniques such as colorimetry; methods such as gravimetric analysis, turbidimetry and titrimetry; and electrochemical techniques such as ion selective electrodes and amperometric titrations (Greenberg *et al.* 1992). Phosphate is relatively immobile in all but very sandy soils and therefore the natural levels of phosphate in waters tend to be very low (Marr and Cresser 1983). Similarly, nitrite concentrations are often found in extremely low concentrations in natural waters, especially if dissolved oxygen is plentiful converting nitrite to nitrate. Given the low concentrations of these nutrients normally encountered in natural waters, very sensitive methods are necessary for their determination. Traditionally, methods such as colorimetry have been used, which is known to give excellent sensitivity to extremely low levels of phosphate. Coupled with this, the most widely used methods for nitrite determination are based upon diazotization reactions, which can detect extremely low concentrations. However, many of these methods are not specific, suffer from interferences, can be labour intensive, and are often difficult to automate. Given the spatial and temporal intensity of the chemical sampling programme undertaken, one of the primary considerations was the efficient processing of collected samples. A suite of the individual test procedures described above can be replaced by one chromatographic separation, and ion chromatography has been quickly accepted by regulatory agencies world-wide for the detection of inorganic nutrients in environmental waters (Dionex 1999). Using commercially available automated apparatus many samples may be analysed in a working day for a wide range of anions (Marr and Cresser 1983). Hence, ion chromatography was deemed to offer the best analytical option for the analysis of inorganic nutrients.

However, given the deviation between results generated from Dionex analysis and Environment Agency data it would appear that the above method does not provide sufficient detection levels to measure the low concentrations of phosphate and nitrite found in chalk rivers. In contrast to phosphate and nitrite, current nitrate concentrations showed no consistently significant difference to equivalent Environment Agency data. The relatively high concentration of nitrate in chalk waters (compared to nitrite and phosphate), and the subsequent lack of difference between data sets, supports the conclusion that the detection levels of Dionex methods were not sufficiently low for nitrite and phosphate measurement. However, this does not mean the phosphate and nitrite results presented are meaningless, rather, that comparisons for these nutrients should only be made within the current database, with further interpretation limited to Environment Agency data.

3.5.2 Spatial variation

Generally all five sampled rivers were extremely similar in terms of their physico-chemical parameters, with only conductivity and SPM differing significantly between systems (attributed to conditions in the River Frome). A key point to this general comparability was the similarity in the general downstream patterns observed in many of the parameters. For example, all five rivers displayed a strong increase in maximum temperatures with increasing distance from source. The general increase recorded in temperature with distance from source is a general characteristic of flowing waters. For example, the water issuing from groundwater springs enters the main channel at a relatively low temperature (Crisp *et al.* 1982). As the river flows downstream, the water body is exposed to solar heat to a far greater degree and the increasing volume of the river combines to increase water temperature (Walling and Webb 1992). Therefore, it is felt that the overall pattern of increasing temperatures is a strong characteristic of southern chalk rivers, displayed by all five sampled rivers, indicating this is a natural pattern rather than one induced by catchment activities (Berrie 1992).

No detectable downstream patterns were observed in the conductivity measurements. Groundwater normally carries salts into the river water, thus increasing the conductivity of the river water (Otley 2000). It follows therefore that chalk rivers would generally have high conductivities that remain relatively stable longitudinally due to the relatively large amount of groundwater recharge they receive along their length (Berrie 1992). Therefore the lack of spatial variation is perhaps not surprising. Conductivity is affected primarily by catchment geology, but flow and anthropogenic discharges can also significantly alter natural conductivity levels (Otley 2000). The lack of spatial variation therefore also suggests that catchment activities have a relatively small influence on the conductivity of southern UK chalk streams.

A stream system both naturally produces and consumes oxygen (e.g. photosynthesis, organism respiration, organic matter decomposition, and various chemical reactions). However, anthropogenic discharges can also influence dissolved oxygen, e.g. wastewater from sewage treatment discharges often contains organic materials that are decomposed by microorganisms using oxygen in the process. All five rivers sampled had a high level of dissolved oxygen indicating that none of these systems suffer deoxygenation problems, an important aspect for ecosystem quality (Walling and Webb 1992). However, dissolved oxygen levels again displayed longitudinal variation with levels decreasing with increasing distance from source. This potentially suggests that local catchment activities in the lower regions of catchments were exerting an influence on dissolved oxygen. For example, on the River Test there was a substantial fish farm effluent discharging above site 5 (Environment Agency 1998a). This discharge is believed to have a high biological oxygen demand that could result in the slightly depressed dissolved oxygen levels at this site. Therefore, local catchment impacts cannot be ruled out as an important factor in determining dissolved oxygen levels.

Chalk rivers have historically been associated with low levels of suspended sediment and low bed loads. This has been attributed to the filtration that water undergoes with movement through the chalk aquifers. Coupled with this, chalk rivers characteristically have a relatively low hydraulic energy that minimises natural bank erosion and hence lowers suspended sediment loads (Mainstone 1999). Again, there was a longitudinal pattern within each river with an increase in SPM concentrations with distance from source. This result is not unexpected as the catchment area available to supply fine particulate matter increases with increasing distance downstream (Walling and Webb 1992). However, several catchment activities can also supply large quantities of SPM. For example, discharges from a water cress farms have been highlighted as a potential source of SPM. Casey and Smith (1994) carried out an investigation into the impacts of a watercress farm on the water chemistry of chalk rivers and observed that when watercress beds are cleaned out, suspended solids in streams below the farms can be as high as 17,192 mg l^{-1}. Depending on the management practices of bed cleaning, high values of suspended matter can be found in the receiving stream over varying time periods (Casey and Smith 1994). A potential example of this is the extremely high value at site two on the River Itchen during Autumn 1999. Shifts in geology, and hence land uses, have also been associated with increased SPM concentrations (Farr and Clarke 1984, Environment Agency 1998a), which may be an important factor in the five river catchments studied.

Unlike lake systems, river water quality is not easily characterised by the concentrations of nutrients (Wong *et al.* 1979, Sas 1989). Hence evaluating the patterns observed in nutrient concentrations are often difficult. Nitrate concentrations showed no significant longitudinal variation. This lack of marked longitudinal variation indicated that there was relatively little input of point sources of this nutrient (Mainstone 1999). However, this finding does suggest that sites near the source of the rivers had relatively high nitrate concentrations. This is an indication that the groundwater feeding these chalk rivers is rich in nitrate, a pattern that has previously been reported for chalk rivers (Casey and Clarke 1979). These results therefore suggest that the relative influence of catchment activities, in terms of point sources, are small compared to the influence of diffuse land use sources and groundwater recharge along the length of the rivers (Edwards *et al.* 1985, Murdoch and Stoddard 1992, Black *et al.* 1993, Heathwaite and Johnes 1996, Pinoke *et al.* 1996, Pinoke *et al.* 1999).

In contrast to nitrate concentrations, both the nitrite and phosphate results presented for each of the five rivers show a slightly higher degree of longitudinal variation, however, there were no significant differences between sites. For example, there was a trend of low nitrite concentrations at sites in the lower reaches of the rivers. This pattern is perhaps not unexpected as previously highlighted these systems are extremely well oxygenated. The presence of consistently high dissolved oxygen could result in the nitrite component of the nitrogen budget of these rivers being extremely small (Casey and Clarke 1979, Walling and Webb 1992, Mainstone 1999).

Elevated levels of these nutrients at certain sites (e.g. phosphate at sites one and two on the Rivers Itchen and Test and site two on the River Meon) could potentially be attributed to point source discharges of these nutrients. In contrast to nitrate, which is thought to enter river courses in a diffuse manner, phosphate and nitrite have been associated with local, point sources (Casey and Clarke 1986, Heathwaite and Johnes 1996). An example of this is at site two on the River Meon. This site was located downstream of a watercress farm and a set of fish farm discharges. Borehole water usually has sufficient nitrogen for watercress, but does not supply enough phosphorus or potassium for optimum growth. Traditionally phosphorus has been added to the beds in the form of basic slag, rock phosphate, phosphoric acid or as a mixed nutrient solution (Casey and Smith 1994). Traditional methods of cress bed fertilisation involve the monthly application of 125 kg of slag, a material containing approximately 6.5 % phosphorus, which is applied to the water surface and results in around 10 – 15 % uptake efficiency, the rest being lost downstream. The losses from 1 hectare of cress beds under this system are equivalent to nearly 100 hectares under intensive arable production located on high-risk impermeable soils (Parr *et al.* 1998), much of which enters chalk headwaters during the growing season. More modern methods of fertilisation are now being used by the larger growers, resulting in lower loss rates. Even so, Casey and Smith (1994) recorded a trebling of mean soluble reactive phosphate concentrations in a chalk headwater due to a modern cress farm, from $25\mu g\ l^{-1}$ upstream to $72\mu g\ l^{-1}$ below the farm. Therefore the additions of these compounds could potentially explain the elevated concentrations of phosphate recorded at this site below a watercress farm.

3.5.3 Seasonal variation

A clear annual cycle of solute behaviour has been observed in many UK rivers (Neal *et al.* 1990). Factors such as the autumnal flushing of soluble material accumulated over summer months and seasonal variations in the chemistry of incoming precipitation (Feller and Kimmins 1979, Williams *et al.* 1983) can affect solute concentrations. The annual cycle of biological activity may also strongly influence water chemistry through the impact of autumnal vegetation dieback and leaf fall (Slack and Feltz 1968), the seasonal uptake by plants and animals (Casey and Ladle 1976, Casey *et al.* 1981) and the effects of microbial populations on the reactions that build up and decompose organic matter in catchment soils (Blackie and Newson 1986, Stevens *et al.* 1989).

A high degree of seasonal variability was observed in my data set, with temperature, pH, conductivity, dissolved oxygen, SPM and nitrate all varying significantly between seasons. The most obvious seasonal variation observed in all five rivers sampled was the summer maximum and winter minimum temperatures. In southern England spring water from chalk aquifers emerges at about 11°C throughout the year (Crisp *et al.* 1982). This baseflow produces a marked effect on the amplitude of the annual temperature regime in chalk streams (Smith 1981, Mackey and Berrie 1991), giving rise to the term 'winter warm and summer cool' which is often associated with chalk rivers. Mackey and Berrie (1991) observed that stream temperatures are seldom less than 5°C or more than 17°C. All five of the rivers sampled exhibited stable and similar temperature patterns, all falling within the range of 8 to 15°C, which is even less than the range predicted by Berrie (1992) and further indicates the temporal stability of these systems.

Another seasonal pattern was that of lowered dissolved oxygen levels in autumn compared to summer. Depressed dissolved oxygen concentrations during late autumn and early winter could be linked with the several factors: these months represent the peak time for decay of instream organic matter and a decline in photosynthesis from lowered levels of macrophyte production (Mainstone 1999), and therefore this pattern is perhaps not unexpected. Seasonal variation in suspended particulate matter is another well-established pattern (Webb and Walling 1992), often linked to the discharge regime of the rivers. Increases in discharge during the winter months are associated with increased runoff from the surrounding catchment, which often brings elevated SPM to the river channel due to erosion. An inverse pattern is observed during summer months, where the lack of over land run-off reduces the amount of SPM supplied to the river (Mainstone 1999), a seasonal pattern seen at sites on all five rivers.

A high degree of seasonality has been seen in nitrate concentrations (Casey and Clarke 1979). In chalk rivers, the general seasonal pattern is one of maximum concentrations during winter and a minimum in summer (Berrie 1992). In other river systems, similar patterns have been observed. For example, Owens *et al.* (1991) found nitrate concentrations and export in both baseflow and storm flow from a hill-land watershed to be lowest in summer and highest in winter. Murdoch and Stoddard (1992) observed the highest nitrate concentrations to occur in late autumn/winter through to early spring and the lowest to occur in summer. Pionke *et al.* (1999) found that highest nitrate concentrations were associated with winter months, and the lowest in summer periods in a small hill-land watershed in the US over a twelve year period.

Although it is difficult to pinpoint a distinct seasonal pattern in nitrate concentrations (Figure 3.8), statistically there was a significant pattern. Generally autumn was associated with high nitrate concentrations and winter with lower levels. This would appear to present a deviation from the pattern of winter maxima previously observed in chalk rivers (Casey 1977, Casey and Clarke 1979). The pattern could be intimately linked to their flow regimes, with increasing discharges in winter months resulting in a dilution of river water nitrate levels (Mullholland and Hill 1997). Casey (1977) found an inverse relationship with discharge, with heavy rainfall associated with reduced nitrate concentrations in some small chalk rivers. Biggs and Close (1989) also reported a similar pattern of nitrate concentrations and discharge. They observed a significant negative correlation between nitrate and flow, with the relatively high nitrate concentrations derived from groundwater diluted by increasing flows in winter, and concluded this to be a characteristic of groundwater dominated rivers. It is a well established that nitrate concentrations in the chalk rivers are predominately derived from groundwater sources (Casey and Clarke 1979), and this could be a major factor in determining the seasonal patterns observed in this study.

Although it is accepted that phosphate concentrations measured during this research may be underestimated, the general seasonal patterns can be compared to those previously described in literature (Casey and Clarke 1986, Berrie 1992, Mainstone 1999). Generally, phosphate showed increased levels during autumn, coupled with elevated summer levels (e.g. the River Test and Meon). In all of the rivers sampled, spring phosphate concentrations were extremely low. The typical seasonality in phosphorus concentrations of many chalk rivers show a peak through the autumnal growing season, in spite of this being the most intense period of biological uptake

(Mainstone *et al.* 1998). Casey and Clarke (1986) observed a substantial reduction in phosphate concentrations during the spring period, with a maximum recorded during summer/autumn (Berrie 1992). Pinoke *et al.* (1999) observed that summer and early autumn months were associated with phosphorus concentrations consistently in excess of other times of year.

These seasonal patterns observed are considered to reflect a combination of the discharge regime of chalk rivers and the sources of phosphate to the rivers. For example, Cahill *et al.* (1974), Harms *et al.* (1978) and Casey and Farr (1982) all suggested that dissolved phosphate concentrations in streams generally decrease during high flow events (normally associated with winter months). Such decreases have been attributed to dilution by inputs of low nutrient content. Throughout the growing season in chalk rivers, bioavailable phosphorus concentrations in the water generally appear to be driven by continuous point sources (for example sewage treatment works, fish and watercress farms) and the amount of dilution afforded by available river flow (Mainstone *et al.* 1996). Data presented by Mainstone (1999) from the chalk river Wylye illustrate the typical seasonality observed in phosphorus, peaking in summer/autumn at the time of minimum dilution despite being the time of highest biological uptake. This lower dilution effect is particularly noticeable in spring, the time of maximum *Ranunculus* growth. Hence it would appear that the current seasonal phosphate patterns are extremely similar to those previously reported in chalk river literature and might be being controlled/influenced by common factors (Casey and Clarke 1986). This could be due to the low dilution afforded to continuous point sources during low flow conditions in the growing season (summer and autumn) and the tendency for much of the diffuse load to be carried out of the catchment under high winter flows (Mainstone 1999).

3.5.4 Interannual variations in the trophic status of chalk rivers

The aim of any water quality monitoring programme is to establish the long term trends exhibited by parameters, upon which management options can be based (Spink *et al.* 1998). However, the absence of historical data often renders this aim unachievable. Long term nutrient data were obtained from several sources at a variety of temporal scales with the aim of establishing interannual variation in the nutrient levels in chalk rivers. Brown (1988) reviewed long term studies of nitrate concentrations in Norwegian rivers, showing a significant systematic increase over time. Hauhs (1989) observed increases in nitrate concentrations in streams between 1977 and 1986 in the Hatrz Mountains in Germany. Hence, increases in nutrients in river waters are not a new phenomenon (Roberts and Marsh 1987), and given the important role these nutrients play in the potential eutrophication of freshwater systems the level of current interest is justified.

The analysis of the trophic status of several southern chalk rivers over an eight year period reveals some interesting patterns. 1995 nitrate concentrations were significantly higher than 1998 and 2000 values. Although the potential of operator bias, through the use of differing methods, cannot be ruled out, this does suggest that nitrate concentrations were generally decreasing. However, the lack of consistently lower values in the latter years does suggest that this trend is still subject to between year fluctuations. In contrast, the lack of significant interannual variation in phosphate suggests there was no long term trends over the eight year period, with a high level of interannual fluctuation.

However, the analysis of nitrate and phosphate concentrations from 1995 – 2000 does reveal some important temporal variations. Nitrate concentrations in the river Frome increased from 2.05mg l^{-1} NO_3-N in 1965 to 3.06 mg l^{-1} NO_3-N in 1972. Most of this increase in nitrate was attributed to the increasing use of nitrogen fertilisers during this time (Casey and Clarke 1979). The amount of nitrogen fertilisers delivered to Dorset rose from 11022 tonnes in 1964 - 1965 to 32430 tonnes in 1971 –1972 (Casey and Clarke 1979). The percentage of nitrate derived from sewage effluent appeared to be fairly constant around 7 % (Casey 1977, Casey and Clarke 1979), suggesting that the majority of nitrate contamination of the river Frome was derived from diffuse sources, in particular from agricultural lands.

Nitrate concentrations have continued to increase, reaching a maximum during the late 1990s after which there has been a slight decline. This decrease could be a result of increased management and legislative regulations imposed upon the land

and fertiliser usage of chalk river catchments (Environment Agency 2000b). However, it should be noted at this point that borehole investigations have revealed that nitrate concentrations in groundwaters are relatively high in chalk catchments (Casey and Clarke 1979). Therefore any management and legislative induced decrease in surface nitrate contributions to river waters will be buffered against through the supply of nitrate rich groundwater to these systems.

Research carried out on the river Frome on phosphate concentrations indicated that levels increased from 108 $\mu g\ l^{-1}$ in 1965 to 134.7 $\mu g\ l^{-1}$ in 1972 (Casey and Clarke 1986). Over 70% of the phosphate concentrations in the river were calculated as coming from sewage effluent, with discharge increasing by 29% during this period. However, considerable variation in phosphate concentration was found within each sampled year, and no coherent trend was observed from 1973 to 1980. When the Environment Agency data were considered alongside these historical results, no major trends were observed. This observation is perhaps slightly unexpected as the majority of phosphate additions to the river Frome are from point sources (primarily sewage treatment works, fish and watercress farms). Given the increased level of sewage treatment it might have been expected that a general decrease in phosphate concentrations would be observed. However, this does not seem to be occurring, with interannual variation occurring to the same extent during the 1990s as was observed during 1965 - 1980.

3.5.5 Comparison to unimpacted chalk rivers

When the results of this chemical survey were compared against the indicative values of key water quality parameters in chalk rivers under near pristine conditions (derived from Mainstone 1999), some clear deviations were seen. For example, Mainstone (1999) generally considered that chalk rivers decrease in pH with increasing distance from source (Table 3.1). However, my results displayed the opposite trend, with increasing pH further downstream. This finding can perhaps be attributed to the precise geology/land uses over which the river flows (Environment Agency 1998a). However, even the lowest pH values on the rivers were far in excess of the 7.4 suggested by Mainstone (1999), perhaps reflecting the true chalk geology of the area.

Research also indicates that in an unimpacted state, chalk streams can have a suspended solids concentration of between <2 mg l^{-1} (upper reaches) and 6 mg l^{-1} (lower reaches) (Casey and Newton 1972, Mainstone 1999). All five rivers sampled had suspended particulate concentrations in excess of 6 mg l^{-1} indicating that there was a degree of catchment-wide enhancement of fine sediments. All the catchments chosen for study are predominantly rural (dominant land uses are pasture and arable farming) with no major effluent associated with high suspended particulate matter concentrations (Farr and Clarke 1984). This suggests that suspended sediment in chalk rivers is primarily derived from diffuse sources, for example runoff from agricultural land.

However, the patterns in nutrient concentrations observed are perhaps the most obvious deviation from Mainstone's (1999) 'pristine' guidelines. Elevated nitrate concentrations are common in rivers draining catchments with a high proportion of land designated as agricultural (Black *et al.* 1993). For example, Hill (1978) found a highly significant positive relationship between nitrate mobilisation and active agricultural land use in 21 watersheds in Southern Ontario. Edwards *et al.* (1990) demonstrated a direct link between river water nitrate concentration and upstream land use and fertiliser application for two major rivers in northeast Scotland.

All five rivers sampled were extremely similar in terms of their nitrate concentrations, with values exceeding 4 mg l^{-1}. Values proposed for unimpacted chalk rivers by Mainstone (1999) range from 0.2 and 1.0 mg l^{-1} in upper and lower reaches respectively. Clearly current results are in excess of these values. The impacts of increased nitrate concentrations have wide ranging implications. Elevated nitrate levels are of some clinical concern because of methaemoglobinaemia and gastric and stomach cancers (Comly 1945, Robards *et al.* 1994). Environmental effects are concentrated on the growth of macrophytes and microalgae. The increasing concentration of available phosphorus allows plants to assimilate more nitrogen before the phosphorus is depleted. Thus, if sufficient phosphorus is available, high concentrations of nitrates can lead to increased production of epilithic algae and macrophytes (Bartenhagen *et al.* 2001). Thus the increased concentrations of nitrate in chalk rivers are of potential concern. However, there are few ecologically related guidelines for the upper amount of nitrate that should be found in flowing freshwater systems. Robards *et al.* (1994) state that heavily polluted rivers contain nitrate concentrations in excess of 10 mg l^{-1}, derived mainly from diffuse agricultural land. There are alternative nitrate guidelines of 10 mg l^{-1} NO_3-N for waters used for recreation and leisure activities, and less than or equal to 40 mg l^{-1} NO_3-N before impacts can be detected on freshwater aquatic life (Robards *et al.* 1994). Given the concentrations encountered throughout my research it can be concluded that although nitrate levels are in excess of unimpacted chalk rivers, concentrations are not reaching levels that can impact upon freshwater aquatic life. However, assuming that phosphorus concentrations are in excess of plant requirements, the levels of nitrate measured have the potential to contribute to increased macrophyte/algal production.

Phosphorus is considered to be one of the key elements in the problem of nutrient enrichment of streams, primarily due to the fact that rivers are often said to be phosphorus limited (Newbold 1992, Robards *et al.* 1994). That is, if all phosphorous is used, plant growth will cease, no matter the amount of nitrogen available. Excessive loading of phosphorus in its various physico-chemical forms is known to be a causal factor in the eutrophication of both lotic and lentic waters (Robards *et al.* 1994). As awareness of water quality issues in river systems has increased, much attention has been focused on establishing a trophic classification system for rivers, based primarily on phosphorus concentrations (Environment Agency 2000b).

Comparisons of recorded phosphorus concentrations in rivers have been used as a means of defining five trophic categories on the basis of macrophyte/total phosphate (TP) data and a literature review, suggesting boundaries at 20, 60, 100, 200, and 1000 μg l^{-1} PO_4-P (Mainstone *et al.* 1993). Marsden *et al.* (1997) proposed a guideline limit of 30 μg l^{-1} TP, and similar standards (30 – 40 μg l^{-1}) intended to prevent nuisance algal growth in rivers in Scotland were put forward by the Clyde

River Purification Board (1989). Higher limits of 100 and 300 µg l^{-1} TP were proposed by the US EPA (1986), and for The Netherlands (Ministry of Transport and Public Works 1989) respectively. The discrepancies between these limits illustrate the difficulties of applying the concept of EQS to nutrients that naturally vary spatially and temporally. In the European Union, two EC Directives have defined phosphorus limits (Freshwater Fisheries Directive 78/659/EEC sets the limit at 200 µg l^{-1}, and the Drinking Water Directive 75/440/EEC sets the guideline at 400 µg l^{-1}). In addition, the UK guidelines to the EC Urban Waste Water Treatment Directive define eutrophic rivers by reference to factors such as soluble reactive phosphate (SRP) concentrations (100 µg l^{-1}), abundance and diversity of algae/macrophytes/invertebrates and diurnal oxygen variation (Marsden *et al.* 1997). Therefore a degree of consensus is evident in the classification of the trophic status of rivers according to SRP/TP concentrations, and the widely accepted threshold of 100 µg l^{-1} can be used to define mesotrophic/eutrophic conditions. However, the definition of boundaries above 100 µg l^{-1} is more arbitrary and little agreement is evident (Marsden *et al.* 1997). Assuming that the concentrations of phosphorus recorded during this work may be underestimated, due to concentrations being below the detection level of the analytical method, it was decided that interpretation of phosphorus levels should be based on Environment Agency data (Figure 3.22).

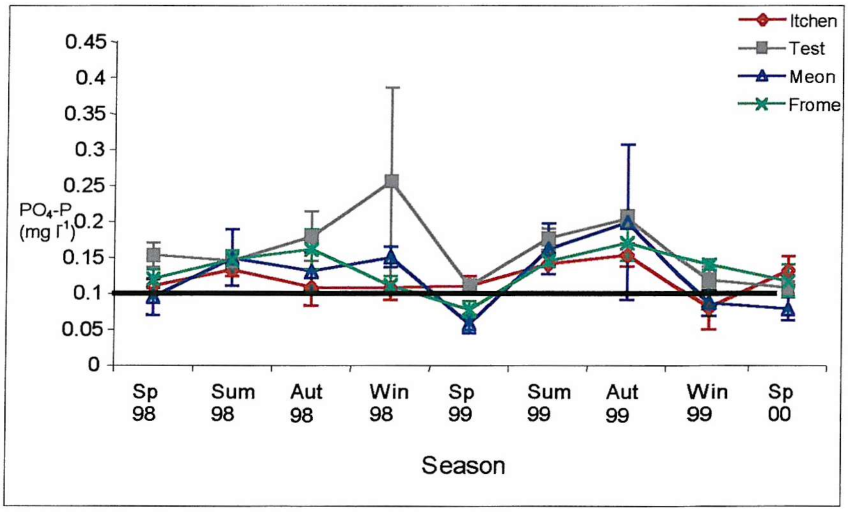


Figure 3.22. Environment Agency phosphate data for rivers Itchen, Test, Meon, and Frome.
Data presented are for equivalent sampling sites and times to those sampled during this research. 0.1 mg l^{-1} PO_4-P equivalent to 100 µg l^{-1}.

During these two years, phosphorus concentrations in the chalk rivers sampled showed a degree of eutrophication, with concentrations in excess of the 100 μg l^{-1} classification suggested by Marsden *et al.* (1997). Coupled with this, Mainstone (1999) suggested that phosphate levels in rivers nearing pristine conditions should range from 10 μg l^{-1} in the upper reaches to 30 μg l^{-1} in the lower reaches. Phosphate concentrations recorded by the Environment Agency throughout this research are well in excess of these concentrations, which again suggests that the chalk rivers are either naturally phosphate-rich (a characteristic classification of chalk rivers) or that a degree of anthropogenic contribution of phosphate is occurring within these systems. Either way, all the chalk rivers studied are at the lower limit of the classification of eutrophication, suggesting that a classification such as *boundary eutrophic* should be applied. However, it is difficult to determine whether this status is due to natural processes or anthropogenically derived phosphate contributions. What is suggested is that any further increases in phosphate concentrations could exceed the accumulation threshold of the rivers and result in a shift to truly eutrophic systems.

3.6 CONCLUSIONS

The longitudinal results of stream water quality in the five chalk streams reveal some interesting patterns. One of the more obvious patterns is that exhibited by SPM concentrations increasing with distance from source. This pattern is generally attributed to the increasing catchment area available to supply SPM to the chalk rivers, suggesting that diffuse additions of sediment are potentially impacting upon chalk river water quality. However, in contrast, it is extremely difficult to attribute downstream patterns in pH and temperature to potential anthropogenic activities, suggesting that the natural properties of chalk catchments could be responsible for some of the patterns previously described. With regard to nutrients, it can be concluded that nitrate is generally elevated along the length of the rivers, again indicating a catchment wide, diffuse source of this nutrient or a consistent recharge from groundwater. In contrast, supplies of both phosphate and nitrite appear to be more locally driven. Therefore, although catchment activities have been potentially identified as impacting detrimentally upon water quality, natural variations in catchment properties must also be acknowledged in determining certain water quality parameters.

Seasonality of nutrient concentrations in streamflow is subject to many controls. Climate establishes the seasonality of flow, which subsequently controls nutrient export. Nutrient availability and supply can affect the seasonality of nutrient concentration in this export, and the characteristic concentration difference is very much tied to flow origin (Pionke *et al.* 1999). A degree of seasonality is also observed in other water quality parameters, for example temperature and SPM concentrations. Much attention has been given to the temporal stability in water chemistry offered by chalk rivers (Berrie 1992, Mainstone 1999). However, results from this investigation reveal that season often significantly influences the water quality patterns of the sampled rivers and must be considered when discussing the overall water quality of chalk river systems.

Another conclusion that can be drawn from these results is that on a catchment scale all five rivers sampled are extremely similar in terms of their water quality and composition. Very few water quality studies have been carried out on a catchment scale, let alone on replicate catchments over the same period. Given the difference in anthropogenic activities undertaken within the five catchments the overall similarity is remarkable. Also, the relatively high level of anthropogenic activity occurring within individual systems indicates that chalk rivers have a high capacity to buffer anthropogenically-induced changes in water quality.

All five rivers currently experience elevated water quality parameters compared to the pristine values given by Mainstone (1999), for example, SPM, nitrate and phosphate concentrations. In terms of ecological implications, the elevated nitrate concentrations do not seem to be at a level at which ecological impacts may occur. This could be attributed to the naturally high levels of nitrate being supplied to these systems through groundwater discharges. However, in terms of phosphate concentrations there is more potential cause for concern. All the rivers sampled would appear to support phosphate concentrations of 100 $\mu g\ l^{-1}$, and can consequently be classified as *boundary eutrophic*. Currently, any consequences of elevated phosphate levels appear to be extremely localised if they do occur at all. However, if concentrations were to increase there may be severe implications for the ecology and water quality of chalk rivers on a much broader scale.

Long term analysis has indicated that over 1993 - 2000 there has been little significant change in nitrate or phosphate concentrations, suggesting that chalk rivers are relatively stable in terms of their long term interannual trophic status. However, between year variation is often high. Both nitrate and phosphate showed increasing trends throughout the 1960s and 1970s. Previous research has suggested that these increases were due to increased usage of nitrate fertilisers and inputs of phosphate rich sewage. Nitrate concentrations were shown to level off during the 1990s, suggesting that the usage of fertilisers has either decreased or at least been controlled through the designation of nitrate sensitive areas. Phosphate concentrations appeared to level off during the mid 1970s, however a high level of interannual variation was evident, making determination of the benefits of increased sewage treatment difficult to achieve.

CHAPTER FOUR: MACROINVERTEBRATE COMMUNITY COMPOSITION OF CHALK RIVERS

CHAPTER FOUR: MACROINVERTEBRATE COMMUNITY COMPOSITION OF CHALK RIVERS

4.1 INTRODUCTION

4.1.1 Background

Chalk river systems have been restructured by human activity for centuries for the purposes of agriculture and water power, and, until recent decades, this has largely served to add value to their ecological character (Mainstone 1999). However, activities in the past 100 years have proved detrimental to the physical form of lowland watercourses in general, caused largely by flood defence and land drainage operations, followed by agricultural and urban development (Boon 1992). In many instances, chalk rivers have fared better than other river types during this period due to the fact that the angling interests of riparian landowners have been respected. The trend towards enhanced drainage and agricultural improvement of riparian meadows has been less acute on prime angling reaches of chalk rivers (particularly the classic chalk stream), since the revenue generated from angling has reduced the need for high agricultural returns from riparian land (Mainstone 1999).

The intrinsic nature of chalk rivers however makes their characteristic communities vulnerable to certain catchment impacts, especially alterations to channel conditions. Key human activities occurring in chalk river catchments have been listed in Table 2.1, along with their principal direct effects upon the riverine environment and associated floodplain areas. A principal area highlighted by literature covering chalk rivers is the vulnerability of the channel bed and banks to physical alteration by, for example, channel widening and sedimentation. Associated direct and indirect effects of channel widening and sedimentation are illustrated in Table 4.1. Although the majority of impacts upon channel quality are negative, elements of sedimentation can be beneficial to localised areas of chalk rivers. Increased banks of sediment in the main channel can increase the diversity of rarer riverine species. However, on the whole, channel widening and siltation of chalk river channels have been highlighted as two areas of ecosystem concern (Mainstone 1999, Environment Agency 2000a).

4.2 CAUSES AND CONSEQUENCES OF CATCHMENT PROCESSES AND THE MONITORING ROLE OF MACROINVERTEBRATE COMMUNITIES

4.2.1 Causes and consequences of catchment processes

Catchment activities have altered numerous properties of many rivers. For example, reduced river flows, caused by over abstraction from surrounding aquifers, have resulted in reduced sediment transport capacity and channel scour (Boon 1992, Mantle and Mantle 1992, Armitage 1995). Land uses have shifted resulting in increased sediment mobilisation and delivery to streams. Increased livestock numbers and stocking densities, coupled with the tradition of using chalk rivers for livestock watering, have resulted in increased 'poaching' (trampling of banks), which have not only caused increased sediment yields but also increased channel widths (Walling and Amos 1995). Channelisation, land drainage and changes in agricultural cropping patterns have all resulted in channel modifications and increased the quantity of sediment transported by chalk rivers (Olssoon and Persson 1988, Gurnell *et al.* 1992). These prolonged hydrological and morphological modifications have all altered the water and habitat quality of chalk river systems in southern England.

As previously discussed (Chapter Two) channel modifications (widening and sedimentation) can impact upon chalk rivers in a variety of ways (Table 4.1).

Table 4.1. Summary of direct and indirect effects caused by two catchment processes on chalk rivers (- negative impacts, ± positive and negative impacts).

Channel modifications	WQ	Flow	Subst q	Chan	Rip	Floodpl
a. Widening	-	-	-	-	-	-
b. Sedimentation			-	-	±	±

WQ	All aspects of water column quality, including source waters and run off
Flow	Flow regime, including flow rate, current velocity and diversity of velocities
Substr q	Substrate quality, including levels of siltation, quality of pore waters and diversity of substrate types
Chan	Channel form, including cross-section, variation in cross section and water depth
Rip	Riparian areas
Floodpl	Floodplain habitats, including flood meadows, ditch systems and other habitats dependent upon water levels

Channel modification

Dramatic resectioning has occurred on some chalk rivers involving straightening, deepening, widening and reprofiling (Environment Agency 1998a, 2000a). Post-war engineering works are often likely to have been driven by the desire for land drainage, permitting intensive livestock grazing and arable crop production (Mainstone 1999). Where such works have occurred, the result has been considerable loss of habitat structure and diversity within the channel (e.g. loss of macrophyte beds), detrimental modifications to bank profile (e.g. loss of riparian vegetation), and the loss of hydrological continuity between the river channel and its banks (Brookes 1994). Consequent impoverishment of channel and riparian communities follows. Channel deepening results in the loss of gravel substrates that are characteristic of chalk rivers (particularly the upper and middle reaches) and so vital to chalk river communities (Mainstone 1999). These substrates can take centuries to acquire and are effectively irreplaceable by natural processes, due to the lack of downstream delivery of coarse material and the lack of movement of modern chalk rivers across alluvial sediments (Berrie 1992).

Modern channel widening in chalk rivers is more often a result of indirect processes rather than intentional engineering. The natural channel form of chalk rivers appears to be wider and shallower than other river types (average width to depth ratio of 33), possibly due to bank erosion through groundwater seepage from riparian areas (Keller *et al.* 1990). However, widespread bank erosion has been caused in recent decades by high densities of livestock with unrestricted access to the river, trampling and destabilising river banks (Quinn, Williamson, Smith and Vickers 1992). This process allows the river to further erode the exposed and unconsolidated soil with large volumes of suspended solids entering the system (Lewis 2000). Riparian vegetation can be severely damaged by overgrazing, causing an associated loss of bank stability (Guthery and Bingham 1996). This artificially induced widening is exacerbated by heavy weed cutting programmes that have denuded river banks of protective marginal vegetation and left them exposed to the full erosive force of the river (Madsen 1997). In addition to altering the physical dimensions of riverine habitats, channel widening has served to dissipate hydraulic energy and reduce the river's ability to maintain solids in suspension, thereby contributing to siltation problems, whilst temperature fluctuations become more extreme (Mainstone 1999, Lewis 2000). A summary of the causes and consequences of channel modifications in chalk river systems are presented in Figure 4.1.

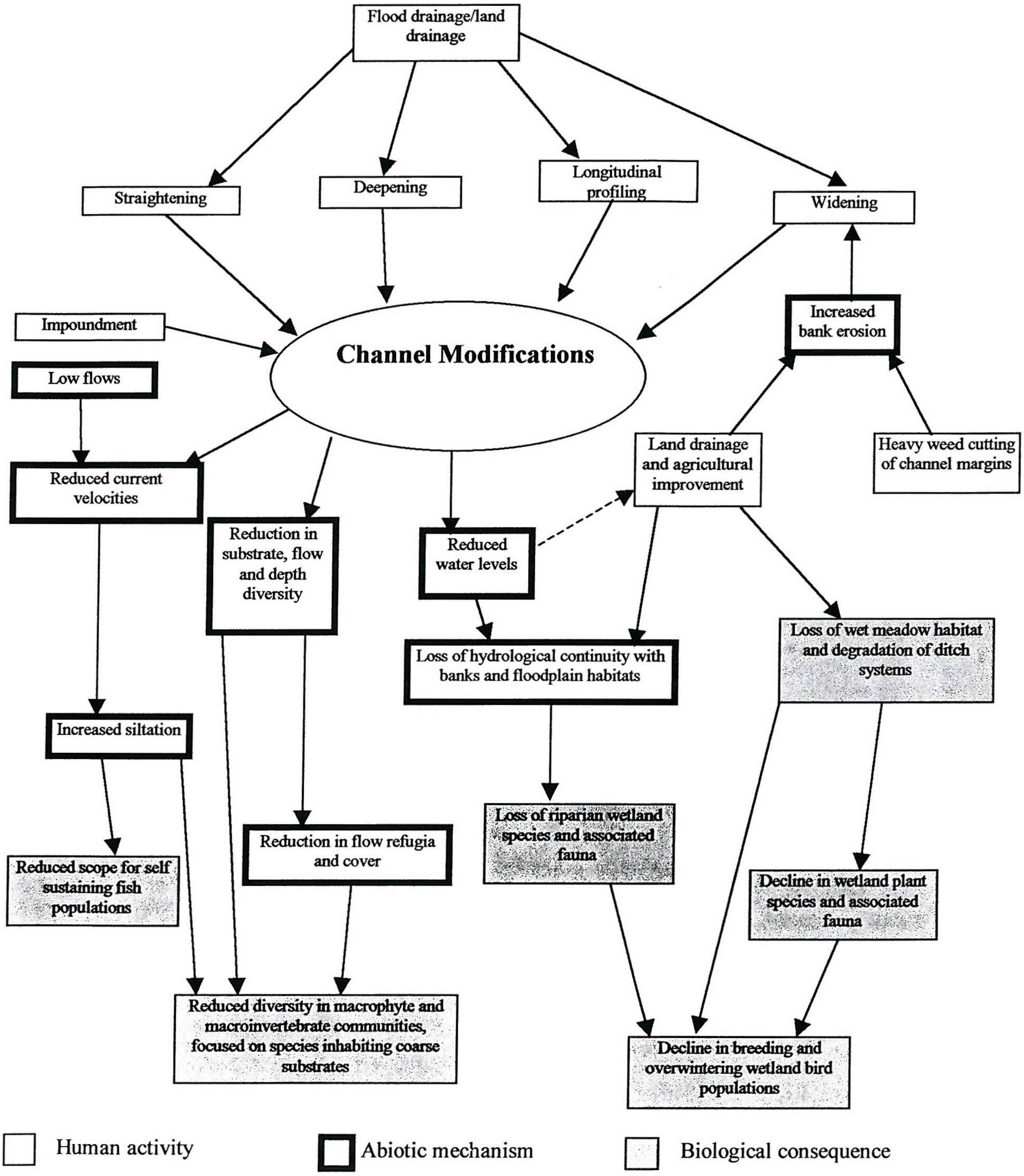


Figure 4.1. Causes and consequences of channel modifications in chalk river systems. (Modified from Mainstone 1999).

At a floodplain scale, the lowering and widening of the channel bed has reduced water table levels throughout the year and decreased the frequency of winter flooding, exacerbated by enhanced land drainage often made possible by channel resectioning. The result has been the large scale conversion of lightly grazed wet meadows to improved grassland and arable production, which can be more prone to sediment run off via overland flow (Berrie 1992, Mainstone 1999, Environment Agency 2000a).

As is evident from Figure 4.1 channel widening is only one of many modifications to be carried out in chalk rivers, with deepening, straightening and impoundment all important processes influencing chalk river profiles. Channel widening is, and has been for some time, intimately linked with the surrounding catchment land uses/practices (Brookes 1994). However, although relatively easy to predict, the impacts of channel widening have not been investigated in detail, especially the potential effects on instream macroinvertebrate communities (Brookes 1986). It is in response to the lack of scientific research on the biological impacts of channel widening that the current work has been initiated.

Sediments

The suspension and deposition of fine sediment (<2 mm) in lotic systems has received considerable attention in recent years (Carling and Reader 1982, Carling 1984, Carling and McCahon 1987, Ryan 1991, ASCE Task Committee 1992, Schälchli 1995, Wood and Armitage 1997, Acornley and Sear 1999). It is widely recognised that anthropogenic activities such as agriculture (Lenat *et al.* 1981, Walling 1990, Richards *et al.* 1993), forestry (Scrivener and Brownlee 1989), groundwater abstraction (Bickerton *et al.* 1993, Castella *et al.* 1995) and periodic in-channel management activities (Brookes 1986, Hearne and Armitage 1993) can result in significant changes to instream deposition, storage and erosion of fine sediments (Figure 4.2). In addition to these anthropogenic activities, natural events such as low flow periods can significantly reduce river energy, enhancing fine sediment deposition in areas of channel not usually subject to sedimentation (Wood *et al.* 1999).

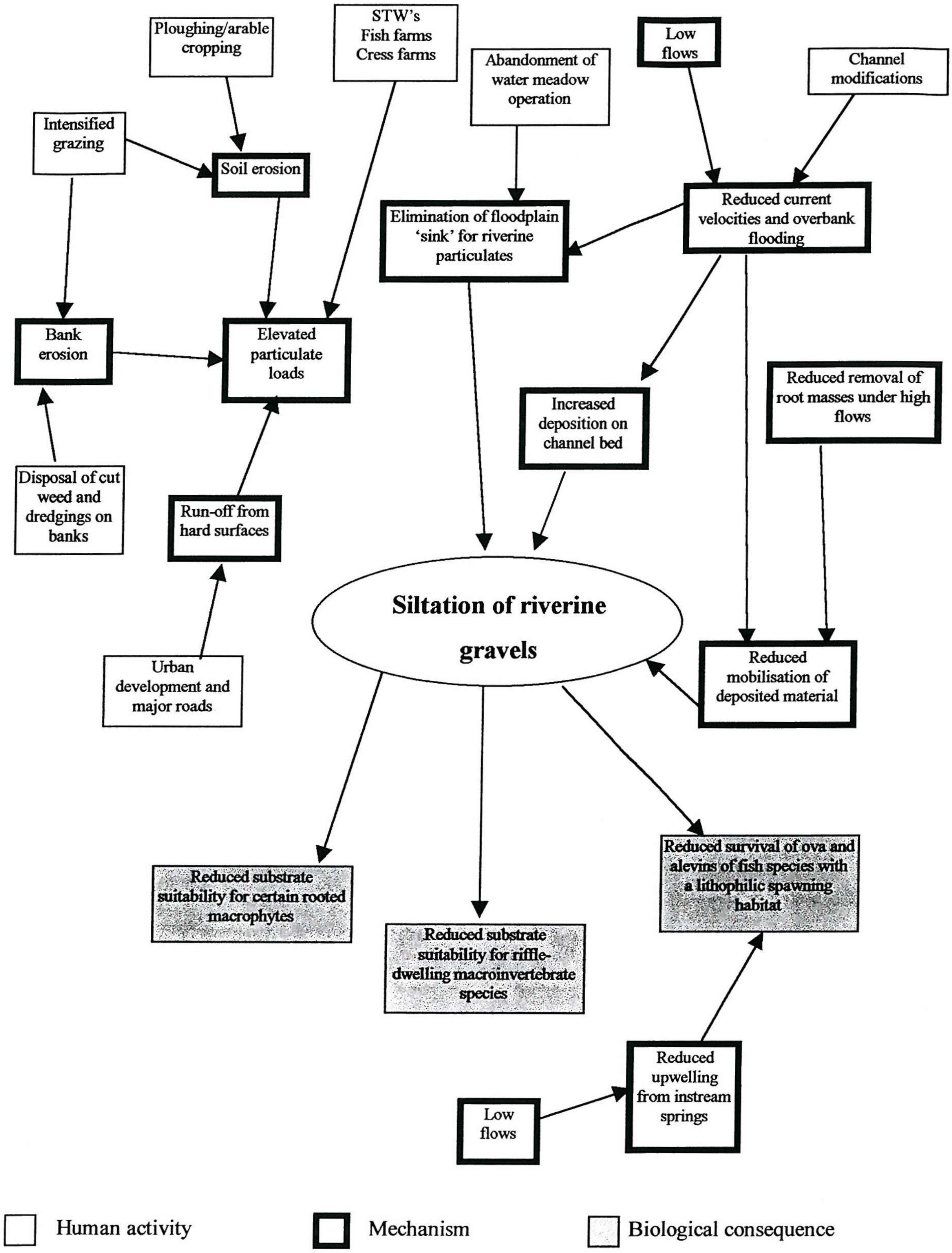


Figure 4.2. Causes and consequences of siltation in chalk river systems. (Modified from Mainstone 1999).

Clean gravel substrates are a key habitat requirement for a range of riverine species typical of chalk rivers, and their maintenance is therefore vital to the proper functioning of characteristic chalk river communities (Berrie 1992, Wright 1992). However, silt is also a crucial component of the habitat matrix of typical chalk rivers and it is important that a proper perspective is maintained when discussing the problems of siltation and in planning mitigation measures (Mainstone 1999). As previously mentioned, chalk river systems have a limited natural flushing capacity and are therefore highly susceptible to siltation of gravels. It has been estimated that natural freshets in chalk rivers only scour the top few centimetres of channel bed, and extensive silt deposits have been found in gravels down to a depth of at least 50 cm (Davey *et al.* 1987). Whilst increased particulate inputs to the system and low flows appear to be the main causes, siltation can be made worse by the reduction in river/floodplain interactions, the development of oversized channels (Wood and Armitage 1997), the presence of in-river structures (Mainstone 1999), all of which reduce current velocities and therefore scouring forces. Importantly, whilst the deposition of fine silts results in deep infiltration of gravels, the deposition of coarser sands and fragments of tufa (a product of the precipitation of calcium out of solution) can result in the development of a consolidated layer just below the gravel surface that is resistant to scour during high flows. This makes the self-cleaning of silted gravels through natural processes very difficult.

Given the natural lack of energetic flushing flows, chalk rivers must rely heavily on receiving low inputs of solids and the maintenance of these inputs in suspension over bare gravel. The physical filtration of the majority of stream flow through the chalk aquifer means that suspended solids levels are naturally low throughout the year, but SPM loads can be greatly enhanced by a range of human activities. Even under high flows, the deposition of suspended particulate matter (SPM) can be high in bare gravel, where water forces its way into the permeable gravel layer and leaves its silt load behind in the gravel interstices (Mainstone 1999). It is evident therefore that increasing levels of fine sediments in chalk rivers could potentially be extremely detrimental to the ecology of these systems (Hamilton 1961, Oschwald 1972, Lenat *et al.* 1981, Wood *et al.* 1999).

4.2.2 The role of invertebrates as biological indicators of substrate quality

The nature of the streambed is a major influence on invertebrate communities (Armitage 1995, Bourassa and Morin 1995, Beisel *et al.* 1998). This has been recognised since the work of Behning (1924) on the Volga and subsequently by many workers round the world (e.g. Hynes 1970, Collier 1995, Rempel *et al.* 2000, Doisy and Rabeni 2001). Spatial and temporal changes in streambed habitats provide a shifting mosaic of abiotic and biotic conditions that play a major role in organising stream communities (Pringle *et al.* 1988, Resh *et al.* 1988, Townsend 1989, Palmer and Poff 1997, Townsend *et al.* 1997, Usseglio-Polatera *et al.* 2000).

Hence, streambed communities are affected by numerous factors, related to both water quality and stream morphology (Hynes 1960, Macan 1974, Meyer *et al.* 1988, Statzner *et al.* 1988, Metcalfe-Smith 1994). The identification of the predominant environmental factors that influence community composition thus involves measurement and analysis of numerous, often intercorrelated, factors (Hynes 1975, Richards and Host 1993). Macroinvertebrate communities play an important functional role in stream ecosystems and are therefore effective as monitors of environmental conditions (Hynes 1970, Cairns *et al.* 1971, Rosenberg *et al.* 1986). Macroinvertebrate community composition has been shown to respond relatively predictably to physical and environmental variables within specified geographical areas (Furse *et al.* 1984, Wright *et al.* 1984). They can thus provide a means to identify important causal factors and to monitor the improvement or degradation of the stream environment over time on a specified spatial scale (Crisp and Gledhill 1970, Richards and Minshall 1992, Rosenberg and Resh 1993).

The large volume of sediments present in chalk catchments is thought to reflect antecedent conditions (Wood and Armitage 1999). It has been suggested that a major contribution of fine sediment was developed during the severe drought of 1988 to 1992 and during the subsequent period of groundwater recovery (Wood *et al.* 1999). The increased flows since 1995 have remobilised and redistributed some of these sediments within the systems. However, on a local scale, large volumes of sediment, especially at the margins (e.g. Great Bridge on the River Test), have remained in the river channels of chalk streams. Therefore, the spatial variability in the deposition and accumulation of fine sediments for individual river reaches reflects local redistribution patterns and site-specific characteristics (Wood *et al.* 1999).

The impacts of excess sediment of the fauna of streams are numerous. Sediment may impact stream communities through a variety of direct and indirect processes (Oschwald 1972). Reduced light penetration, smothering, habitat reduction and the introduction of absorbed pollutants (pesticides, metals, and nutrients) are all reported effects (Lenat *et al.* 1981). Hynes (1960) described the two principal ways in which the fauna of streams and rivers may be affected by inert solids: firstly, when the solids are suspended in the water, they may render all plant and algal growth impossible through reducing the penetration of light. There is then no food for herbivores and the detritus feeders have to rely on detritus of an allochthonous origin. Secondly, where inert solids settle out of the water, they not only smother algal growths and kill rooted plants and mosses but they also alter biotopes in other ways. For instance the interstices between stones may become clogged, obliterating the habitat of many animals (Chutter 1968, Ritchie 1972, Hansmann and Phinney 1973, Rosenberg and Snow 1975).

However, other studies of the effects of sediment on macroinvertebrate communities have resulted in some apparently contradictory conclusions. Most studies have revealed that the density/abundance of benthic macroinvertebrates is severely reduced by sediment addition (Tebo 1955, Gammon 1970, Reed 1977). This results in a shift in the composition of the benthic community (Rosenberg and Snow 1975, Dance 1978). However, other investigations have found community structure largely unchanged by sediment addition (Herbert *et al.* 1961, Pearson and Jones 1975). The results of my research should help to resolve these apparent conflicts and illustrate the variable effects of sediment addition.

One objective of this research was to quantify variation in the composition and abundance of stream macroinvertebrate fauna within five catchments of varying size in response to various catchment processes. Analytical methods considered the effects on community structure (e.g. through diversity indices), indicator organisms (e.g. through biotic indices), and the taxonomic composition of communities (e.g. multivariate analysis) (Charvet *et al.* 2000).

4.2.3 Study aims

This study aims to determine the impact of catchment processes on the ecological quality of chalk rivers on a catchment level, by examining macroinvertebrate communities. Increasingly, water quality investigations are being made on a catchment level, however, replication at this level is often lacking. In order to address this, the current investigation has incorporated five chalk catchments aimed at establishing ecological trends on a replicated catchment level. Macroinvertebrate communities have been selected as biological indicators of the ecological health of these systems (Armitage *et al.* 1983, Mason 1996), and control sites will act as reference communities against which ecological impacts can be compared.

It is generally considered that physical catchment processes impact upon the ecological quality of southern chalk rivers. My specific aims were to investigate:

- the impact of channel widening and sedimentation upon the ecological quality of southern chalk rivers. Ecological quality was assessed using diversity and biotic measurements, along with the examination of community structures.
- the impacts on a replicable catchment level using macroinvertebrate communities as a means of detecting the impacts of catchment processes

A decrease in macroinvertebrate community diversity is predicted, with an associated shift in community structure towards communities dominated by species adapted to substrates containing a high level of fine sediment and lower velocities. It is also predicted that lower biotic scores, indicating degradation in ecological quality, will be encountered (Armitage *et al.* 1983).

4.3 FIELD AND LABORATORY METHODS

4.3.1 Field methodologies

The following section provides details of the various methodologies undertaken to determine the ecological status of chalk rivers. These include considerations of site selection, the sampling protocols adopted, and the laboratory procedures carried out.

Site selection

Rivers typically show geomorphic, hydraulic and biological characteristics along the length of their course (Cannan 1998). Such changes in character are a combination of both natural and anthropogenic influences. Chalk rivers provide excellent examples of the myriad of interacting habitats that are often found in lotic systems (Cannan 1998). However, catchment processes over the past century have resulted in channel modifications potentially to the detriment of benthic macroinvertebrate communities (Ladle and Westlake 1995).

After a preliminary examination of numerous chalk river systems throughout Hampshire and Dorset, five catchments were selected within which biological and chemical water quality investigations were carried out. As mentioned previously, the five systems selected were the Rivers Itchen, Test and Meon (Hampshire), Frome (Dorset) and Avon (Wiltshire). In order to assess the potential ecological implications of channel widening and sedimentation on a catchment level, several sites encompassing these processes were selected on each river providing a matrix of sites with which to work, locational details of which are presented in Table 4.2.

Ideally, it would have been preferable to replicate site type within each river system. However, certain site types were not found in abundance and it was felt if site replicates were taken within individual rivers, pseudoreplication might become a major factor. It was therefore felt that the individual rivers could be used as the replicating factors rather than individual sites.

This subsequently raises the issue of site placement. Clearly sites should have been located in a random order in each river so as to minimize potential confounding upstream and downstream factors. This was followed as far as possible, with treatment stretches selected at random except where the paucity of treatment reaches along the river made this impossible. For subsequent analysis treatment sites were assumed to be randomaly located along the streams and no correction was made for upstream/downstream placement.

Extensive examination of each river system in turn facilitated rigorous site selection, with each location meeting specific criteria outlined prior to going out in the field. For example, a control site was chosen for inclusion into the matrix to provide a baseline against which catchment influenced sites could be compared. Characteristics of a pristine chalk stream include diverse macrophyte beds and mineral substrates (Wood *et al.* 1999). After an extensive review of literature and research pertaining to chalk rivers (including work carried out by the author), five principal mesohabitats were identified as representing classic chalk river stretches: main channel gravel/cobbles, instream vegetation, for example *Ranunculus* beds, sand, silt, and riffle gravel. Each of these habitats had to be present during initial site selection otherwise further locations were investigated regarding their suitability as unimpacted locations with classic chalk stream features.

Table 4.2. Matrix of sites selected for the study of impacts of sediment deposition and channel widening on macroinvertebrate communities.

RIVER	SITE TYPE		
	CONTROL	OVERWIDENED	SEDIMENTED
ITCHEN	**Titchbourne** (SU 5795 2975)	**Easton** (SU 5110 3245)	**Itchen Stoke Main Channel** (SU 5575 3215)
TEST	**Overton** (SU 5170 5045)	**Chilbolton Carrier** (SU 3860 4045)	**Great Bridge** (SU 3545 2270)
MEON	**Drayton** (SU 6720 2265)	**Mislingford** (SU 5855 1410)	**Titchfield** (SU 5410 0550)
FROME	**Bradford Peverell** (SY 6695 9245)	**Moreton** (SY 8560 8945)	**Maiden Newton** (SY 5985 9735)
AVON	**Lower Woodford Carrier** (SU 1255 3485)	**Lower Woodford Main Channel Downstream** (SU 1240 3470)	**Lower Woodford Man Channel Upstream** (SU 1275 3535)

Overwidened site selection was based upon width to depth ratios. Mainstone (1999) states that chalk rivers generally have relatively shallow cross sections, with average width to depth ratios of 33. In order for sites to fit into this classification, during initial site consideration, widths and depths for each site were recorded. If the ratio of width to depth exceeded 33, the sites were included in the matrix as overwidened. Table 4.3 outlines the width to depth ratios of selected sites.

Table 4.3. Width:Depth ratios of overwidened sites selected for study.

River	Site	Width:Depth
Itchen	Itchen Stoke Main Channel	41
Test	Chilbolton Carrier	50
Meon	Mislingford	80
Frome	Moreton	114
Avon	Lower Woodford	69
	Average Width:Depth ratio	71

As Table 4.3 indicates, all overwidened sites chosen for study were in excess of the width to depth ratio of 33 proposed by Mainstone (1999), especially Mislingford on the River Meon, Moreton on the River Frome and Lower Woodford on the River Avon.

Although it is recognised that fine sediments are an integral component of chalk river habitats (Berrie 1992), increasing supplies of these materials to the channel have resulted in localised sedimentation of reaches. The importance of instream channel and habitat heterogeneity to the overall aquatic biodiversity of riverine systems has been widely recognised (Power 1992, Harper *et al.* 1997, Guegan *et al.* 1998). However, the increased sediment availability and reduced flushing capacity of many chalk rivers may have resulted in long term changes in substratum characteristics (Wood *et al.* 1999). Visual examinations were carried out to determine the dominant substratum category, in accordance with Wood and Armitage (1999). A site was classified as sedimented if the majority of the reach (in excess of 60 %) experienced sediment deposition in areas of the channel not usually subject to sedimentation (for example in high flow areas such as the main channel). Details of the classification scheme used to identify sites are summarised in Table 4.4.

Table 4.4. Description of site classification scheme.

CLASSIFICATION	DESCRIPTION
CONTROL	Classic chalk stream reaches characterised by high habitat heterogeneity (Berrie 1992)
SEDIMENTED	Reach dominated by fine sediment beds in areas not usually subject to sediment deposition (Wood and Armitage 1999)
OVERWIDENED	Width:Depth ratios in excess of 33 (Mainstone 1999)

These site classifications were chosen as they represent the most common disturbances within chalk rivers on a catchment wide basis. It is recognised that this is not an exhaustive list and other disturbances are present within chalk rivers, for example, bank instability problems. However, after a review of the literature, it became apparent that sedimentation is perhaps the most pressing current problem facing chalk rivers (Wood *et al.* 1999). Apart from the physical constraint on the number of samples it is possible to process, which in itself will restrict the number of catchment activities that can be sampled, the aim of this work was to assess, on a catchment wide basis the issue of channel modification. For sufficient replication of catchments it was necessary to limit the number of activities investigated.

Sample collection

Environmental variables were recorded at sites on each visit, allowing any changes in the channel morphology or bankside vegetation to be recorded over the two sampling years. An assumption was made that channel morphology would not change significantly during the period of study (two years), in accordance with Cannan (1998), as chalk streams are thought to be extremely stable systems (Berrie 1992).

On every sampling occasion, a survey of the 50 m river reach was undertaken to establish the position of sampling sites for macroinvertebrate collection, as suggested by Cannan (1998). It should be noted at this point that the majority of sample collection and processing techniques were carried out in accordance with Cannan (1998), as I had worked in close conjunction with Cannan (1998) previously. At each site, three replicate kick samples were taken using a standard Freshwater Biological Association pond net (dimensions: 0.9 mm diameter mesh, 230 mm by 255 mm frame, 275 mm deep net and 1.5 m handle; Armitage and

Pardo 1995). Samples were taken from areas typical of the survey reach as a whole (Cannan 1998, Wright *et al.* 1984), sampling all available habitats in the sample area in proportion to their occurrence (Wright *et al.* 1984). This sampling time period was considered to provide an adequate sample of individuals in the time available whilst minimising the destruction of habitats (Wright *et al.* 1992). A variety of times have been used in the collection of invertebrate samples, ranging from 15 seconds to 3 minutes (Cannan 1998) and there is little consensus as to what time period should be used. For example, Morgan and Egglishaw (1965) demonstrated that the first two kicks of a sample collected the majority of taxa, and that it was rare that new animals occur towards the end of the sample. In theory shorter sampling times would allow greater numbers of samples to be taken. However, the overall aim of my investigation was to investigate the community composition of each site type a longer sampling time period was decided upon (Cannan 1998). In accordance with Cannan (1998), no precise definition of area to be sampled was made, and the length of the river covered in three minutes at each point depended on the habitat diversity, channel width and depth. Samples were collected starting downstream and moving upstream, so as to reduce the possibility contaminating samples with organisms from areas previously sampled.

Whilst it is unlikely that the three replicate samples taken are sufficient to fulfil statistical requirement, as determined by sample size statistics, this number was a pragmatic compromise (Cannan 1998). The aim of this investigation was to cover a large spatial area and for this to be possible it was felt that smaller numbers of samples should be collected. Three to five replicates are common in studies on macroinvertebrates in rivers, particularly in monitoring studies where budget considerations are important (Cannan 1998). Three replicates may provide a good compromise between ideal requirements and the practical limits of the project, and are sufficient to allow acceptable estimation of the presence and abundances of common taxa (Boulton and Lake 1992, Cannan 1998). For example, Barmuta (1989) found that three replicate samples retained 74% of taxa collected by nine replicates. Resh and McElravy (1993) also showed that increasing the number of replicates from two to three resulted a substantial increase in the ability to detect community changes, whilst between five and ten provided only a small increase (Cannan 1998).

Coupled with invertebrate samples, environmental variables were also recorded at each sampling reach, e.g. width, depth, substrate composition and plant cover (Table 4.5).

Table 4.5. Environmental variables recorded for each macroinvertebrate survey reach.

Variable	Measurement
Width	30 m tape measure
Depth	Metre rule
Substrate	Visual evaluation of % substrate
Macrophyte	Visual evaluation of instream macrophytes

Each sample was taken from a different stretch of the survey reach to avoid the problems of pseudoreplication, a phenomenon discussed in detail by Hurlbert (1984). Replication is an inherent requirement of any biological survey (Cannan 1998). Replication allows the magnitude of any differences between habitats and sites to be compared against the magnitude of natural differences (Norris and Gorges 1993). In benthic macroinvertebrate sampling there is a very important distinction to be made between sampling unrelated patches for replicates, rather than merely sub-sampling the same patch, resulting in samples not wholly independent from one another (Cannan 1998). Buikema and Vohell (1993) outlined the problems of pseudoreplication in the subsequent analysis of data.

Sample storage

Once collected the sample was washed, removing as much fine sediment as possible. Large stones, pieces of detritus and weed were removed from the net, if present, to reduce the volume of the sample, in accordance with the suggestions of Wright *et al.* (1984). Attached organisms on these substrates were removed prior to the removal of these objects and the sample was placed in a labelled plastic bag along with a small amount of river water and sealed (Cannan 1998). After each sample was collected, the pond-net was washed thoroughly to reduce the chance of contamination. In the laboratory, the samples were preserved by adding 40% formaldehyde. Where samples contained a high proportion of plant material, for example sites with a high proportion of in-stream vegetation mesohabitats, slightly more preservative was added (Cannan 1998).

Formaldehyde was chosen as the preservative in preference to alcohol due to its superior preserving qualities; alcohol leads to a loss of pigment and increased brittleness that may hamper the identification of specimens (Cannan 1998). Samples preserved in alcohol are also likely to decompose more rapidly. Due to the time taken to process the relatively large number of samples collected formaldehyde was thought to be the most appropriate preservative, in accordance with Cannan (1998).

4.3.2 Laboratory Techniques

Macroinvertebrate sample processing

The animals were picked from the samples for subsequent identification. Large numbers of organisms are also often found in macroinvertebrate samples collected. In many circumstances it is not practical to count and identify every individual (Cannan 1998) and sub-sample has become common in order to reduce sample processing time (Hynes 1970). Two sub-sampling methods are commonly used: fixed fraction and fixed count. Fixed fraction methods have been traditionally used for benthic studies (Cannan 1998). Numerous techniques have been developed and are reviewed by Hickley (1975), involving apparatus that randomly mixes and splits the invertebrates into a known fraction of the total sample (Cannan 1998). There are many problems associated with sub-sampling apparatus, including high construction costs, complexity of construction and lack of reliability and restrictions on the types of samples with which it can be used (Wrona *et al.* 1982, Cannan 1998). Other methods have been developed, for example the use of divided trays (Furse *et al.* 1984, Cannan 1998). In fixed count methods a certain, predetermined number of organisms are randomally picked from the sample and then identified, e.g. 100 organisms are picked in the rapid biological assessment methods developed by the US Environmental Protection Agency (Cannan 1998). The debate over these different methods continues and to date there is no consensus as to which method is preferable (Cannan 1998).

However, whatever method is use to pick samples, the process is relatively time consuming and it is this stage of data collection which limits the number of samples that can be dealt with in a given time (Cannan 1998). The sample processing methods used in this work was based on the methodology developed on the RIVPACS project (Furse *et al.* 1984, Cannan 1998), combining the removal of individuals and sub-sampling large samples simultaneously. This method can be used to obtain two types of data from the samples: semi-quantitative data (i.e. a list of families present in the sample with a relative abundance total) and qualitative species data (i.e. list of species present in sample) (Cannan 1998).

Sample preparation

Each sample was thoroughly washed through a 500 μm sieve to wash off excess formaldehyde and remove fine particles. Larger objects, e.g. pebbles and woody debris, were also removed if this could be achieved without the substantial loss of animals (Wright *et al.* 1984). Following this, the sample was placed into a large container and covered with sufficient water to avoid dehydration (Cannan 1998).

Sample sorting

Aliquots were transferred to a white-bottomed tray and distributed evenly. A grid was drawn on the tray dividing it into 16 equal sections, facilitating the sub-sampling of aliquots (Cannan 1998). Once the aliquot was evenly distributed over the tray the material was sorted by eye (with the aid of a magnifying glass) (Wright *et al.* 1984). If the sample was small, then all individuals were removed, however, most of the time samples proved too large for this method to be carried out and were therefore sub-sampled, according to Cannan (1998) methodologies. All animals were removed from a fixed area of the tray, usually one quarter (i.e. 4 out of 16 squares) or one eighth (i.e. 2 squares out of 16). The remainder of the tray was checked for less common individuals that had not previously been found in the sub-sampled area (Cannan 1998). The individuals removed from the remainder of the tray were placed in a separate bottle. The individuals that had been picked from the fixed fraction of the tray were then multiplied by the appropriate factor (i.e. 4 or 8) and the individuals removed from the remainder of the tray were not picked from a fixed fraction and so did not require a multiplication factor to work out their relative abundances (Cannan 1998). Any individuals from a family found in the sub-sampled fraction but of a different size/appearance to those already collected were also picked out of the remainder of the tray. This was to maximise the species list (Cannan 1998). In accordance with the methodology used by Cannan (1998) any individuals which were very abundant in the sample, and that could be readily identified, were counted using a counter once approximately 30-50 individuals had already been picked out. Large numbers of individuals were required to be removed as Wright *et al.* (1984) state that since many macroinvertebrates cannot be identified without a microscope, large numbers of specimens should be removed to ensure that most, if not all, of the species present in the sample were available for identification (Cannan 1998).

Sample identification

The individuals were stored in labelled sample bottles containing 70% alcohol (Cannan 1998). Using microscopes and standard identification keys, all the individuals were identified and counted. Relative abundances were calculated through applying the correct multiplication factor (Cannan 1998). Wherever possible taxa were identified to species level, but Oligochaeta and Chironomidae were left at a higher taxonomic level, and others such as Baetidae were only identified to genus level owing to the large number of early instars in this family (Wood and Armitage 1999, Wood *et al.* 1999). There is no consensus in the literature as to the level of identification required for benthic samples (Lenat *et al.* 1980, Cannan 1998). The aim of this investigation was to examine subtle differences in the macroinvertebrate communities at different site classifications, and it was therefore felt that identification should to be to species level wherever possible (Cannan 1998). If identification had only been to family level it is possible that the more subtle differences in the invertebrate communities of the sampling sites would have been less likely to be detected (Plafkin *et al.* 1989, Resh and McElravy 1993, Cannan 1998). The individuals were kept for checking.

4.3.3 Characterising study sites

Geomorphological and physico-chemical variables

Variables relating to the channel morphology of the sampling sites were recorded at site visits and arrived at from maps (Table 4.6). Width and depth measurements were recorded using 30 m tape measures and a metre rule, in accordance with Cannan (1998). The values were averaged to give a mean width and depth, enabling the width:depth ratio to be calculated (as previously described in § 4.3.1). Altitude, distance downstream and catchment area were calculated from 1:50,000 Ordnance Survey Maps and Environment Agency literature (e.g. catchment management plans). Chemical variables were also measured after invertebrate samples had been collected. These measurements were taken in order to establish relationships with specific faunal occurrences (Cannan 1998).

Table 4.6. Physical and chemical variables used to describe the character of study reaches.

Variable	Method
Channel Width (m)	Field measurement
Channel Depth (cm)	Field measurement
Width:Depth ratio	Field measurement
% Gravel	Visual examination
% Fine sediment	Visual examination
% Macrophyte	Visual examination
Temperature (°C)	Field measurement
pH	Field measurement
Conductivity (µS/cm)	Field measurement
SPM (mg/l)	Field and lab measurements
Dissolved Oxygen (%)	Field measurement
Altitude (m)	Map
Distance downstream (km)	Map
Catchment area (km^2)	Map
Geology	EA literature

4.3.4 Data analysis: univariate methods

The remainder of this section describes the methods used in the analysis of the data collected from this field study with the aim of providing a background to the univariate and multivariate techniques available for investigating community data. Numerous options are available for the analysis of biological macroinvertebrate data. Studies of spatial and temporal changes in benthic communities have required quantification of the state of the communities, often in the form of a single summary measure. This measure may be simply biomass, total abundance of benthic macroinvertebrates or species richness, or it may be an index of community structure combining elements of both abundance and species number (e.g. species diversity). The measure may also be based on data selected from one or more indicator species (e.g. a biotic index) (Norris and Georges 1993).

Diversity indices

Species diversity has been defined as a function of the number of species present (species richness or abundance) and the evenness with which the individuals are distributed among these species (species evenness or equability) (Margelef 1958, Norris and Georges 1993). Diversity indices are of particular use in aquatic ecosystems for the study of pollution or stress and its effects on the macroinvertebrate community (Washington 1984), due to the relative ease with which species abundance and evenness can be measured. Ecological stresses produce changes in the pattern of species distributions, for example inert silt can reduce species numbers and abundances (Godfrey 1978). Diversity indices attempt to combine the data on abundance of individual species within a community into a single number (Godfrey 1978, Washington 1984). From this value conclusions can then be drawn regarding the state of the community in terms of its diversity.

One of the most commonly used diversity indices is the Shannon-Weiner function (Godfrey 1978). This index (H') is based on the probability of predicting correctly the species of the next individual to be collected. The index combines both the number of species in a sample and the evenness of distribution of individuals amongst species, with H' reaching a maximum value when all species are distributed evenly. Biologically, this is assumed to be the most desirable situation (Norris and Georges 1993). The index is expressed as:

$$H' = -\Sigma\,(p_i)\,(\log_e p_i)$$

H'=index of species diversity; $p_i = n_i/N$; n_i=number of individuals in the ***ith*** species; **N**=total number of individuals

As the proportion of individuals per species increases and proportion of individuals per species becomes more constant (i.e. even) the diversity index will increase. The index remains stable in any species distribution but can be relatively insensitive to rare species. In general, the higher the index value, the greater the diversity of river invertebrate communities (Washington 1984). This index relies upon two fundamental assumptions: (1) that individuals are randomly sampled from an indefinitely large population, and (2) that all species are represented in the sample, both of which must be considered prior to calculation of the index (Houston 1994).

Biotic indices

Score systems using macroinvertebrates have been developed to allow the interpretation of large quantities of data resulting from the biological monitoring (Hynes 1970, Armitage *et al.* 1983). Ecosystem change in response to pollution/stress has led to the search for means of quantifying such changes (Washington 1984). Biotic indices have been developed empirically as a means for assessing pollution impact (Norris and Georges 1993). By definition, a biotic index is a score that takes into account the sensitivity or tolerance of individual species or groups to stresses (Mason 1996). Calculation of biotic indices requires a total count of individuals or total counts of taxa, counts of specific groups, detailed lists of the responses of different taxa to pollution, or division of invertebrates into groups with different feeding strategies.

The most widely calculated biotic indices are the Biological Monitoring Working Party (BMWP) and Average Score Per Taxon (ASPT) (Mason 1996). The BMWP index is a score system used to assess the biological condition of river courses, based on the varying degrees of tolerance exhibited by species and/or families to perturbations or stresses (Ward *et al.* 1995). More sensitive organisms are characterised by having a high score, with low scores being attributed to tolerant species. The sum of these scores of the individual families or species recorded at a site constitutes the total BMWP score (Armitage *et al.* 1983). The ASPT index of a sample can be calculated from the BMWP index by dividing the total score by the number of scoring taxa (Metcalfe 1989). This index is often calculated as (1) it limits all values to within a scale of 1 – 10 (Metcalfe 1989), and, (2) it is relatively independent of sample size, sampling technique and season (Armitage *et al.* 1983, Pinder *et al.* 1987). Although the scores of the indices are specific to individual river systems, to place these indices into the wider context of river quality; Alvarez (1995) regarded a BMWP and ASPT score in excess of 100 and 4 respectively as indicating good water quality.

Statistical analyses

As previously outlined in § 3.3.3 Analysis of Variance techniques are extremely useful in determining the significance of variation within a data set, and have been used throughout the analysis of my results. Further detail for the specific ANOVA used will be given in the subsequent results sections.

Univariate methods involve reducing the full taxon information for a sample into a single coefficient (Cannan 1998). Standard statistical methods can then be applied to these extractions from the matrix, and analysis is relatively straightforward. However there are a number of factors that make these analyses inapplicable to biological data. For example, the responses of indices to environmental conditions are relatively unknown (Clarke and Warwick 1994). Coupled with this, these methods are not based on species identity and in theory two samples with completely different set of species could bring the same result (Godfrey 1978, Cannan 1998). Moreover, as only the relative abundance of species is considered in the calculation of indices, a wealth of information available on the environmental adaptations and responses of aquatic macroinvertebrate communities is ignored. Hence, multivariate techniques are widely used in the analysis of macroinvertebrate community data (Gauch 1982).

4.3.5 Data analysis: multivariate techniques

Community data are inherently multivariate (Digby and Kempton 1987). Multivariate analysis is the branch of statistics that deals with the examination of numerous variables simultaneously (Gauch 1982, Townsend *et al.* 1983). It serves two basic roles in community ecology: helping to discover structure in ecological data and providing a relatively objective summarisation of the data, which facilitates understanding and comprehension (Gauch 1982, Jongman *et al.* 1995, ter Braak and Smilauer 1998). The overall purpose of multivariate analysis is to treat data as a whole, summarise them and reveal structure and patterns (Gauch 1982). Sheldon and Haick (1981) summarised the role of multivariate analyses as descriptive and exploratory tools that summarise complex observations or generate hypotheses for rigorous testing. Multivariate techniques have been highlighted as useful tools for the stream ecologist who has to deal with many species interacting in environments that are truly multi-dimensional (Sheldon and Haick 1981).

Ordination

Three basic multivariate strategies exist: direct gradient analysis, ordination and classification (Gauch 1982). As ordination was chosen as the primary analytical technique throughout this study, the remainder of this section provides an outline of the underlying theories behind the methods used. Ordination primarily endeavours to represent sample and species relationships as faithfully as possible in a low-dimensional space (Clymo 1980). The end result is a graph, usually two-dimensional, in which entities are arranged in a continuous way, based on taxon composition, in such a way that samples/species with comparable compositions are arranged close to one another (Wiegleb 1981, Townsend *et al.* 1983, Verdonscholt and Higler 1989).

Coupled with this, an environmental interpretation of the sample/species arrangement is often considered (Gauch 1982). This use of environmental variables can aid in the formulation of hypotheses as ideas can be based upon the relationships revealed between the species composition of the samples and the associated environmental variables (Jongman *et al.* 1995). The relative success of this approach depends on the extent to which the arbitrary axes, derived mathematically, correspond to ecologically meaningful environmental gradients (Townsend *et al.* 1983). Numerous reviews of the different ordination techniques have been undertaken with the only conclusion arrived at is that there is not a definitive answer as to which one should be used. It would appear that in general, the test used depends on the nature of the data set that is to be analysed (Gauch 1982, Jongman *et al.* 1995).

Two different ordination techniques are generally used: indirect ordination and gradient analysis. Indirect ordination examines the variation of the sample independently of the environmental data. Patterns of species composition can therefore be examined on their own, and then subsequently be related to environmental conditions, leading to indirect environmental interpretation (ter Braak 1995). Examples of this approach include multidimensional scaling (MDS) (Clarke and Warwick 1994, Clarke 1999), component analysis, factor analysis and correspondence analysis (CA) (Gauch 1982). Direct gradient analyses include environmental variables directly into the analysis carried out (Kovach 1999). The gradients extracted in this case are constrained to represent the best distribution of the samples in the light of environmental variables. The results are then interpreted in comparison with indirect techniques to assess the influence of the environmental variables on the patterns in the data (Kent and Cocker 1992).

There are numerous multivariate techniques available and choice is not intuitive. The main multivariate approaches that have been used in this thesis are Principal Components Analysis (PCA) and Redundancy Analysis (RDA). PCA is classified as an indirect gradient technique, and the related RDA as a direct gradient analysis technique. These techniques are readily accessible via the CANOCO package (version 4) (ter Braak and Prentice 1988), an important consideration when analysing large data sets with the frequency necessary in this project (Cannan 1998).

Principal Components Analysis (PCA)

Indirect gradient analysis can be undertaken either with linear or unimodal methods (ter Braak and Smilauer 1998). One example of linear ordination is that of PCA. These techniques summarise gradients in the faunal data alone, and therefore require only a sample by taxa matrix as input (Cannan 1998) and can be considered to be an extension of fitting straight lines and planes by least squares regression.

PCA is the ordination technique that constructs the theoretical variable (the first PCA Axis) that minimises the total residual sum of squares after fitting straight lines to the species data. If a single variable does not explain the species data sufficiently well, two variables will be used and for each species a least squares regression is carried out on the two explanatory variables, giving rise to two PCA axes. The first two axes of PCA are now the theoretical variables minimising the total residual sum of squares, and therefore explaining the majority of variance in species data (Gabriel 1971, ter Braak and Verdonchot 1995). Analogously, the first three PCA axes minimise the total residual sum of squares by fitting the data to hyperplanes, and so on.

PCA is therefore a multi-species extension of multiple (least squares) regression. The difference is that in multiple regression the explanatory variables are supplied environmental variables whereas in PCA the explanatory variables are theoretical variables estimated from species data alone. In summary therefore, the first PCA axis is the variable that explains the species data best, and second and later axes also explain the species data best but subject to the constraint of being uncorrelated with previous PCA axes. In practice, higher numbered PCA axes are ignored as they explain only a small proportion of variance in the species data (ter Braak 1995).

Redundancy Analysis (RDA)

RDA is a linear example of direct gradient analysis, detecting the pattern of variation in the faunal data that can be best explained by the environmental variables (Cannan 1998). As with PCA, attempts are made to explain the data of all species by fitting a separate straight line to the data of each species. As a measure of how badly a particular environmental variable explains the species data, the total residual sum of squares is considered, as in PCA. The best environmental variable is the one that gives the smallest total residual sum of squares. From this an ordination technique can be derived that selects a linear combination of environmental variables that gives the smallest total residual sum of squares, hence maximising the difference between species according to their environmental characteristics (ter Braak 1995).

In other words RDA chooses the smallest total residual sum of squares for the environmental variables, giving the first RDA axis. The second and further RDA axes also select linear combinations of environmental variables that maximise the dispersion of the species scores, but subject to the constraint of being uncorrelated with previous RDA axes. As many axes can be extracted as there are environmental variables.

Environmental variables to be included in the ordination are selected by forward selection, which is an option in CANOCO (version 4) (ter Braak 1988). Ter Braak and Smilauer (1998) describe forward selection as being analogous to the selection process employed in stepwise multiple regression. The variables are selected for the RDA such that each will account for the greatest possible proportion of the remaining variation. Only variables that explain a significant ($p<0.05$) additional proportion of the total variance are included in the analysis, with significance assessed on the basis of the Monte Carlo Permutation test available in CANOCO (version 4) (Cannan 1998).

Partial PCA and RDA

Partial PCA and RDA techniques are an extension of the PCA and RDA procedures. These methods can be carried out when the effect of particular environmental variables needs to be singled out from the background variation imposed by other variables (ter Braak 1995, Cannan 1998). For example, in an environmental impact study, the effects of impact variables are to be separated from those of other sources of variation, represented by covariables. One way to achieve this is to eliminate or partial out the effects of the covariables and to relate the residual variation to the impact variables. The usual environmental variables are replaced by the residuals obtained by regressing each of the impact variables on the covariables (ter Braak 1995). These techniques can be used to get a more complete picture of how different variables explain the faunal data (Cannan 1998).

4.5 CATCHMENT PROCESSES RESULTS

4.5.1 Impacts of channel widening and sedimentation: univariate results

Table 4.7 provides catchment information for each of the sites investigated. Locations of the sampling sites are shown in Figures 2.2 to 2.6 (blue dots) in § 2.3.

Table 4.7. Environmental context of the 15 study reaches.

River	Site classification	Variables			
		Distance Downstream (km)	Catchment Area (km^2)	Altitude (m)	Geology
Itchen	Control	2	60	65	Chalk
	Overwidened	9	120	55	Chalk
	Sedimented	12	144	40	Chalk
Test	Control	1.9	82	95	Chalk
	Overwidened	20	380	30	Chalk
	Sedimented	43.2	986	10	Bagshot beds
Meon	Control	2	12	105	Chalk
	Overwidened	20	80	30	Bagshot beds
	Sedimented	30	128	5	Bagshot beds
Frome	Control	20.5	186	60	Upper chalk
	Overwidened	38	317	30	Bagshot beds
	Sedimented	11	90	90	Greendsand/Gault
Avon	Control	50.3	347	65	Chalk
	Overwidened	50.5	349	65	Chalk
	Sedimented	49	345	65	Chalk

A total of 342 macroinvertebrate samples (3 replicates per site type, 3 site types per river and 5 rivers) were collected in Summer 1998, Autumn 1998, Winter 1998, Spring 1999, Summer 1999, Autumn 1999, Winter 1999 and Spring 2000. The site samples were composite samples collected from all habitats in relation to their occurrence at that point of the river (Cannan 1998). A summary of the raw data collected is shown in Appendix E.

The potential impacts of channel modifications (through channel widening and sedimentation) were assessed by examining abundance data, taxon richness, BMWP and ASPT scores, and Shannon Diversity (H'). The significance of any differences highlighted was assessed statistically using a three way randomised block ANOVA (Zar 1996). The ANOVA tests were carried out on non-transformed data unless stated. Prior to the ANOVA being carried out, data were tested for conformation to normality and equal variance. If these tests were failed, data were log transformed. Results have been presented for each site type on each river establishing the degree of influence that catchment processes have on individual chalk systems. Following these analyses, data have been investigated using multivariate methods.

Principal Components Analysis (PCA) has been used to summarise the patterns in the overall composition of the samples. Redundancy Analysis (RDA) was then used to examine the structure in the faunal data and relate it to the environmental characteristics of the channel and catchment at that point (Cannan 1998). Table 4.6 and 4.7 outline the environmental variables used in this analysis. Seasonal and inter-annual variation was also examined at the sample sites over the two year sampling period using similar analysis techniques.

A total of 101 macroinvertebrate taxa were recorded in the five rivers (Table 4.8).

Table 4.8. Macroinvertebrate taxa recorded in the biological survey. Taxa were classed into the higher taxonomic headings of orders and sub-classes.

Taxa	**Itchen**	**Test**	**Meon**	**Frome**	**Avon**
TRICLADIDA					
Polycelis tenuis	X	X	X	X	X
Dugesia polychroa	X			X	
Dugesia tigrina				X	X
Dendrocoelum lacteum	X	X	X	X	X
Bdellocephala punctata	X				
GASTROPODA					
Theodoxus fluviatilis				X	X
Viviparus viviparus	X				
Valvata cristata			X		X
Valvata macrostoma	X		X		X
Valvata piscinalis		X	X	X	X
Potamopyrgus jenkinsi		X	X	X	X
Bithynia leachi	X	X		X	X
Lymnaea trunctula	X				X
Lymnaea glabra					X
Lymnaea peregra	X	X	X	X	X
Physa fontinalis	X	X	X	X	X
Planorbis albus		X	X		
Planorbis carinatus		X	X	X	X
Planorbis contortus	X	X			
Planorbis planorbis		X			X
Planorbis vortex	X	X	X	X	X
Ancylus fluviatilis	X	X	X	X	X
BIVALVIA					
Sphaeriidae	X	X	X	X	X
OLIGOCHAETA	X	X	X	X	X
HIRUDINEA					
Piscicola geometra	X	X		X	X
Theromyzon tessulatum					X
Glossiphonia complanata	X	X	X	X	X
Helobdella stagnalis	X	X	X	X	X
Erpobdella octoculata	X	X	X	X	X
HYDRACARINA	X	X	X	X	X
ISOPODA					
Asellus aquaticus	X	X	X	X	X
AMPHIPODA					
Gammaridae	X	X	X	X	X
EPHEMEROPTERA					
Baetis sp.	X	X	X	X	X
Rithrogena semicolorata	X	X	X		
Heptagenia sulphurea	X	X	X	X	X
Ecdyonurus torrentis				X	
Ecdyonurus venosus	X	X	X	X	X
Leptophlebiidae				X	
Ephemerella ignita	X	X	X	X	X
Ephemerella notata					X
Ephemera danica		X	X	X	X
Caenis sp.	X	X	X	X	X
PLECOPTERA					
Nemoura sp.		X			
Leuctra geniculata	X	X	X	X	X
Leuctra hippopus	X	X	X	X	X
Isoperla grammatica	X	X	X	X	
Chloroperla sp.				X	X
ODONATA					

Table 4.8 cont.

Taxa	Itchen	Test	Meon	Frome	Avon
Calopteryx splendens					X
HEMIPTERA					
Mesovelia furcata	X	X	X		
Gerris sp.	X				
Aphelocheirus aestivalis		X			
Cymatia bonsdorffii	X				
Sigara sp.			X		X
COLEOPTERA					
Haliplus sp.	X	X	X	X	
Hyphydrus ovatus		X		X	
Platambus sp.	X	X	X	X	
Colymbetes sp.	X	X		X	X
Gyrinus natator	X	X	X	X	
Orectochilus villosus		X			
Elmis aenea	X	X	X	X	X
Esolus parallelepipedus	X		X	X	X
Limnius volckmari	X	X	X	X	X
Oulimnius sp.					X
Donacia sp.		X	X		
Curculionidae					X
NEUROPTERA					
Sialis lutaria	X	X	X	X	X
TRICHOPTERA					
Rhyacophila dorsalis	X	X	X	X	X
Agapetus fuscipes	X	X	X	X	X
Hydroptila sp.	X	X		X	
Polycentropus flavomaculatus	X	X	X	X	X
Tinodes unicolor		X	X	X	X
Psychomyia pusilla				X	
Hydropsyche pellucidula	X	X	X	X	X
Hydropsyche siltalai	X	X	X	X	X
Drusus annulatus	X	X			
Limnephilus lunatus	X	X	X		
Limnephilus sp.	X	X	X	X	X
Anabolia nervosa				X	
Potamophylax latipennis	X	X	X	X	X
Halesus radiatus	X	X	X	X	X
Melampophylax mucoreus	X	X			
Odontocerum albicorne	X	X	X	X	X
Athripsodes albifrons		X			X
Athripsodes sp.	X	X	X	X	X
Mystacides azurea	X	X			X
Goera pilosa				X	X
Silo pallipes	X	X	X	X	X
Crunoecia irrorata	X	X	X	X	
Lepidostoma hirtum	X	X	X	X	X
Brachycentrus subnubilis				X	X
Philopotamidae	X				X
Sericostoma personatum	X	X	X	X	X
DIPTERA					
Dicranota sp.	X	X	X	X	X
Tipula sp.			X	X	X
Culicoides sp.	X	X	X	X	X
Atherix sp.				X	
Chironomidae	X	X	X	X	X
Dixa sp.	X		X		
Empididae	X		X	X	X
Muscidae	X			X	
Simuliidae	X	X	X	X	X

Table 4.9 presents a summary of the faunal characteristics of the three site types on each of the five rivers investigated, with respect to abundance, taxon richness, BMWP, ASPT and Shannon diversity (H'). Figures 4.3 to 4.7 display the differences in site type in terms of their summary macroinvertebrate community parameters for each of the five rivers sampled.

Table 4.9. Summary of the Abundance, Taxon Richness, BMWP, ASPT, and H' for the site classifications on each river.

River	Site classification	Faunal parameters				
		Abundance	Taxon richness	BMWP	ASPT	H'
Itchen	Control	576	16	78	5.3	1.58
	Overwidened	662	18	87	5.4	1.81
	Sedimented	165	8	31	4.0	1.16
Test	Control	715	15	71	5.3	1.38
	Overwidened	309	16	83	5.8	1.71
	Sedimented	47	7	32	4.2	1.51
Meon	Control	463	13	48	4.1	1.63
	Overwidened	725	16	82	5.8	1.39
	Sedimented	160	9	37	4.2	1.29
Frome	Control	1012	21	110	5.7	1.93
	Overwidened	324	17	87	5.8	1.96
	Sedimented	245	11	51	4.9	1.39
Avon	Control	905	19	92	5.2	1.84
	Overwidened	436	18	86	5.3	1.80
	Sedimented	230	12	61	5.2	1.60

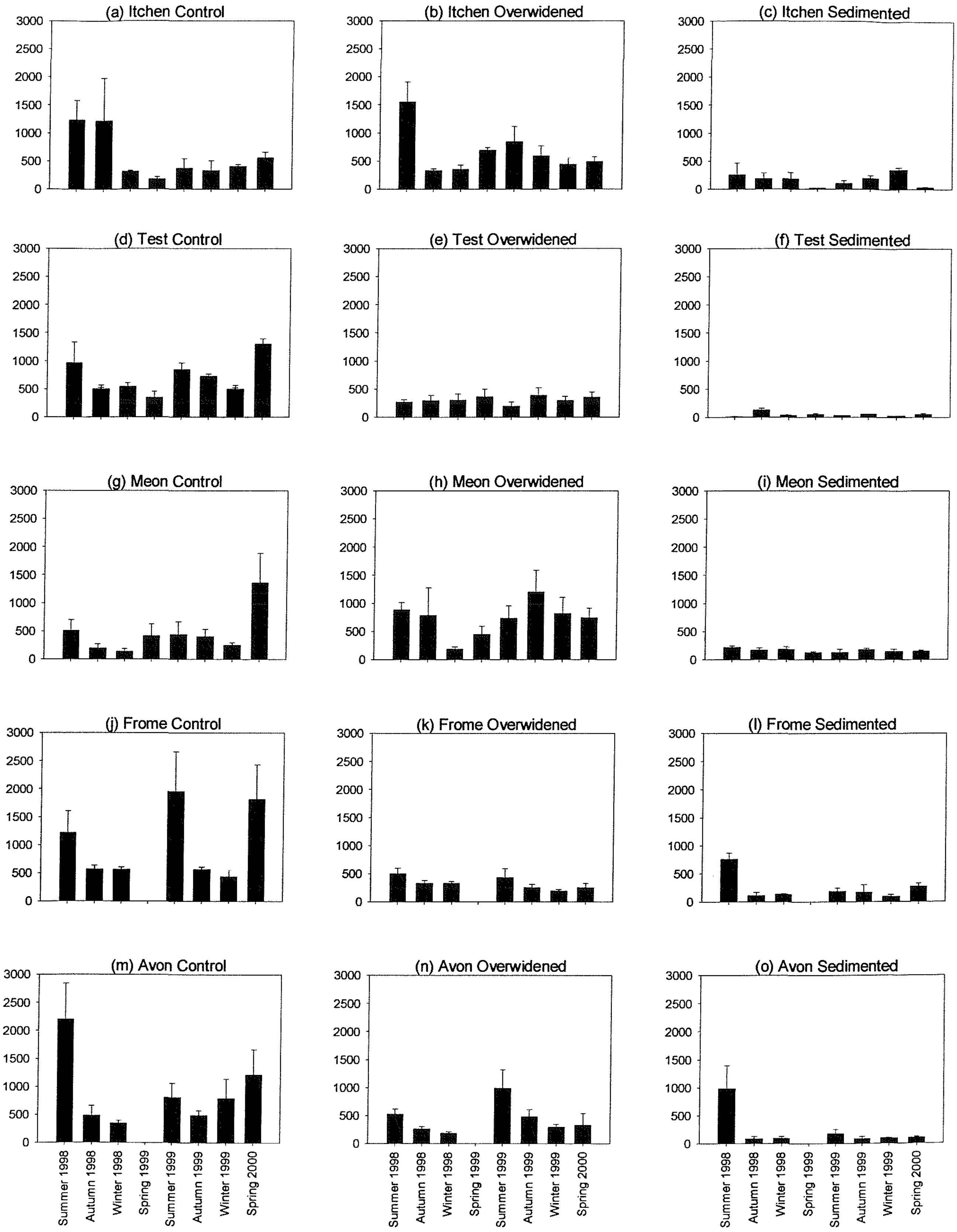


Figure 4.3. Mean abundance for the eight sampling seasons at control, overwidened and sedimented sites on the Rivers Itchen, Test, Meon, Frome and Avon. Vertical bars show ± 1 Standard Error.

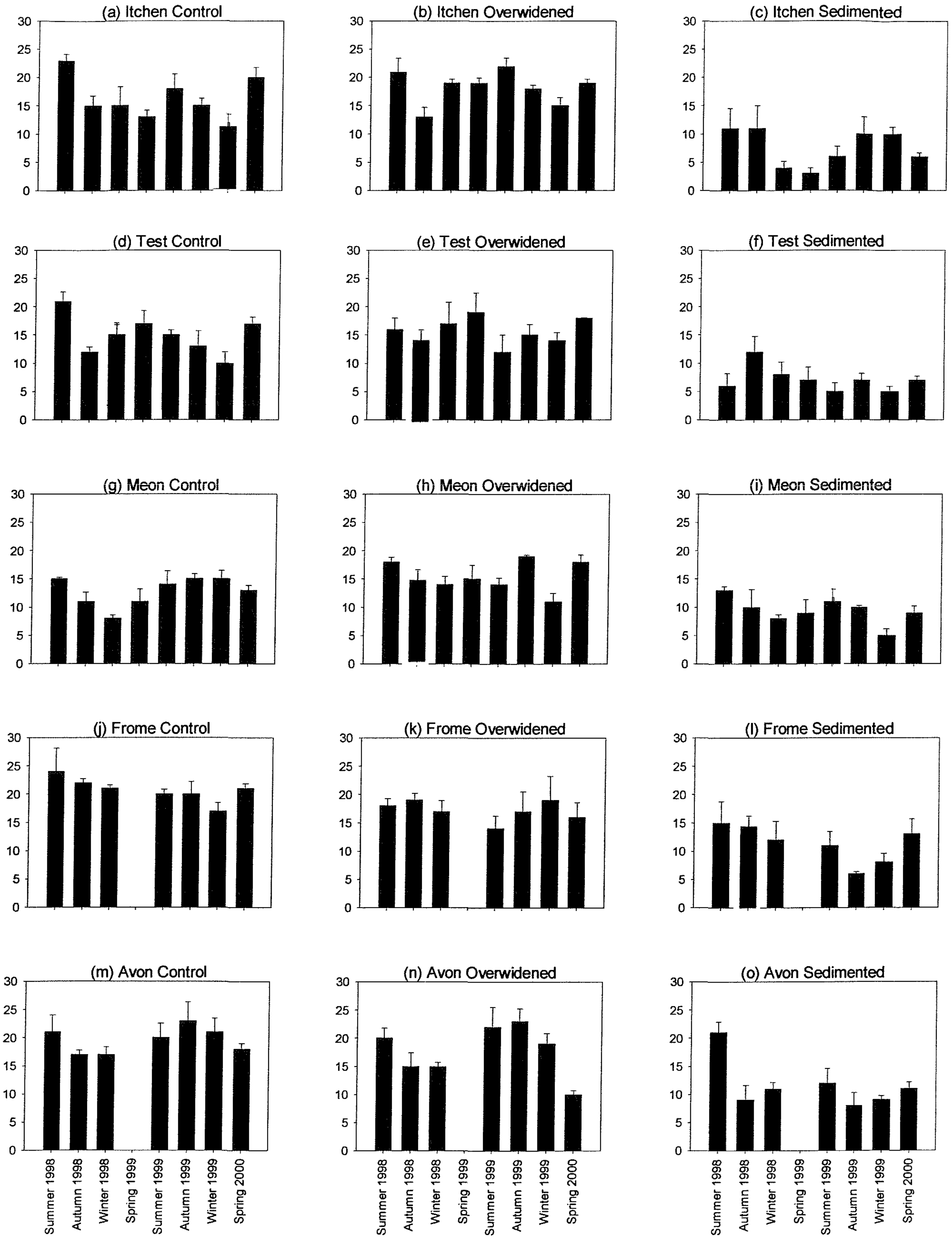


Figure 4.4. Mean taxon richness for the eight sampling seasons at control, overwidened and sedimented sites on the Rivers Itchen, Test, Meon, Frome and Avon.
Vertical bars show ± 1 Standard Error.

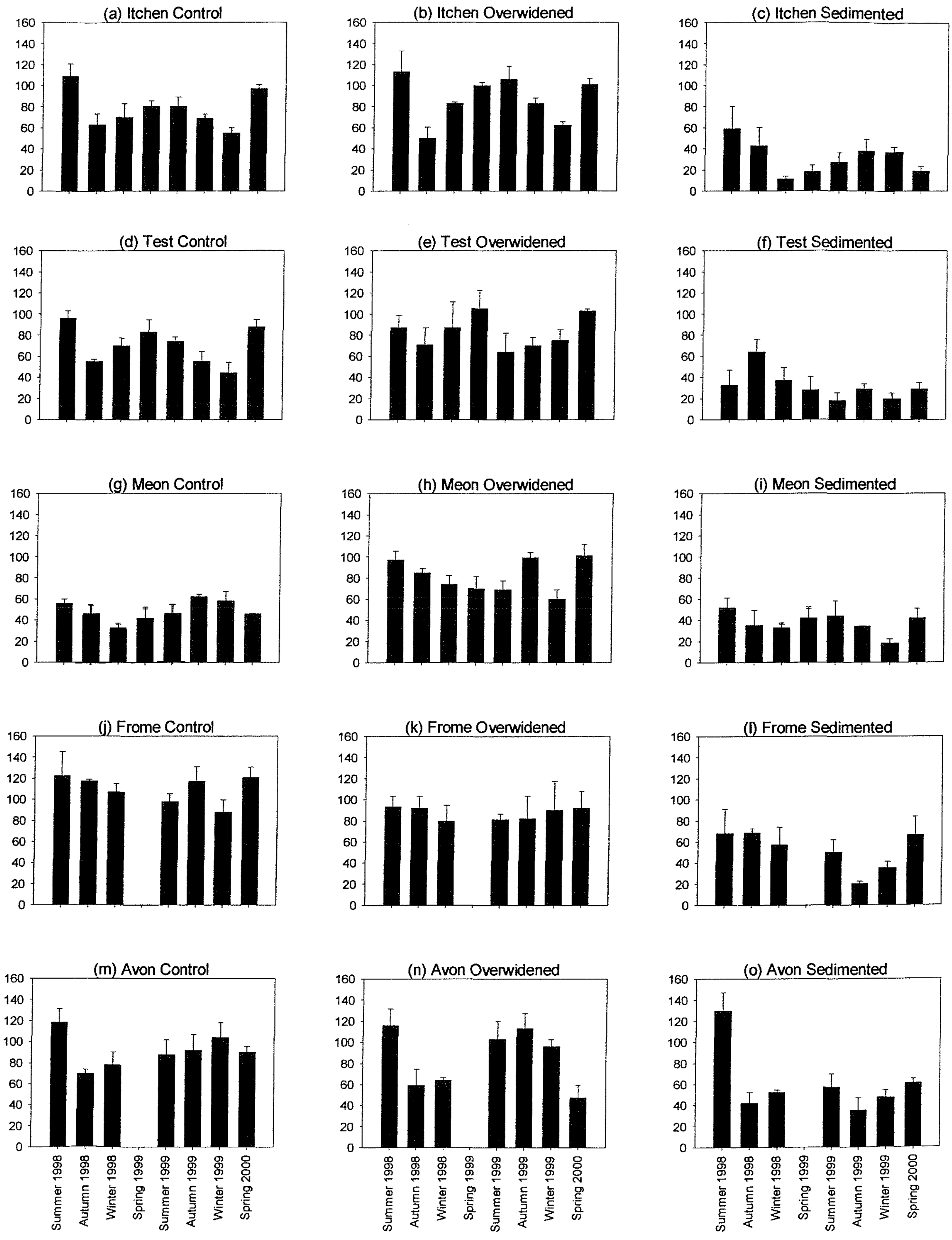


Figure 4.5. Mean BMWP for the eight sampling seasons at control, overwidened and sedimented sites on the Rivers Itchen, Test, Meon, Frome and Avon. Vertical bars show ± 1 Standard Error.

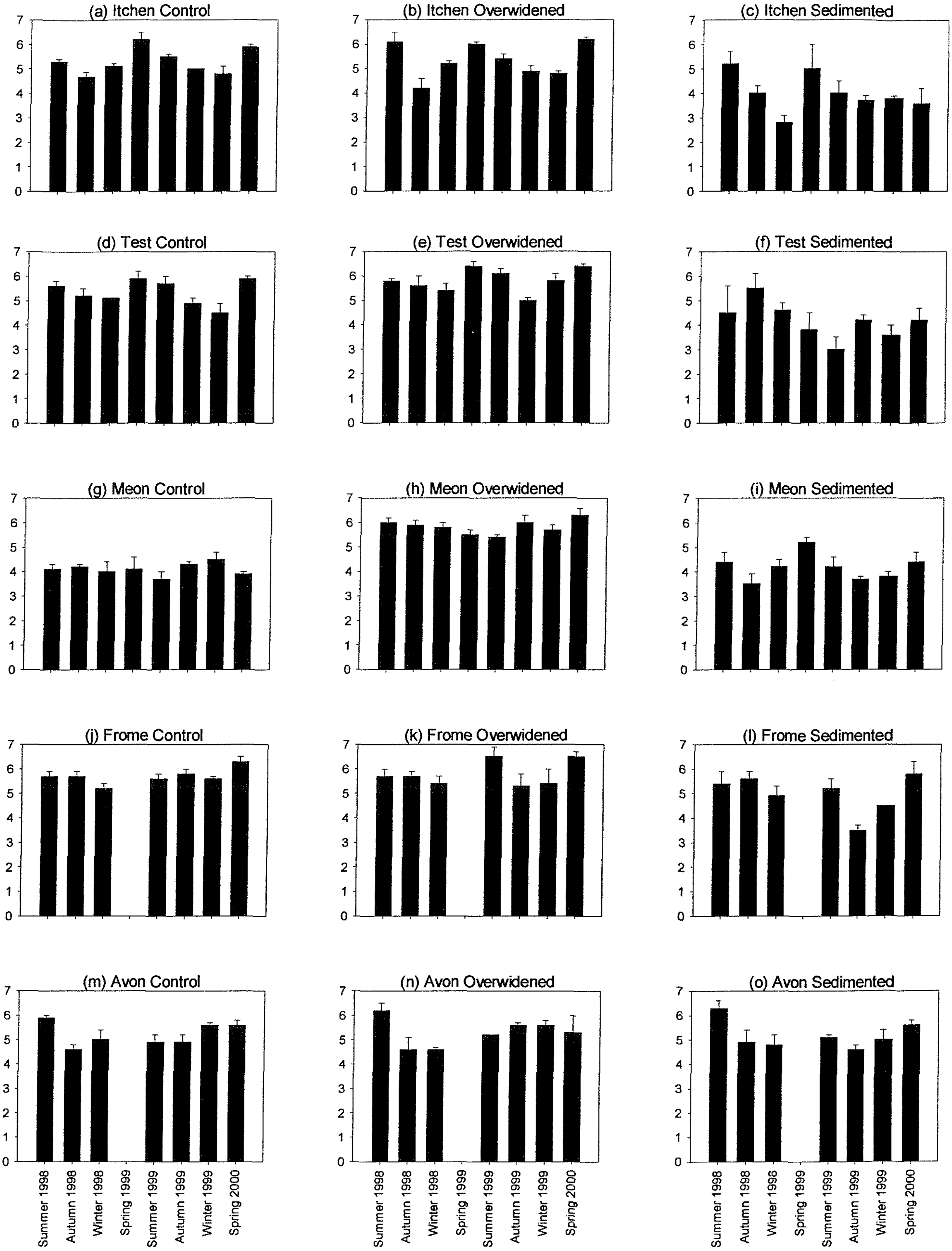


Figure 4.6. Mean ASPT for the eight sampling seasons at control, overwidened and sedimented sites on the Rivers Itchen, Test, Meon, Frome and Avon. Vertical bars show ± 1 Standard Error.

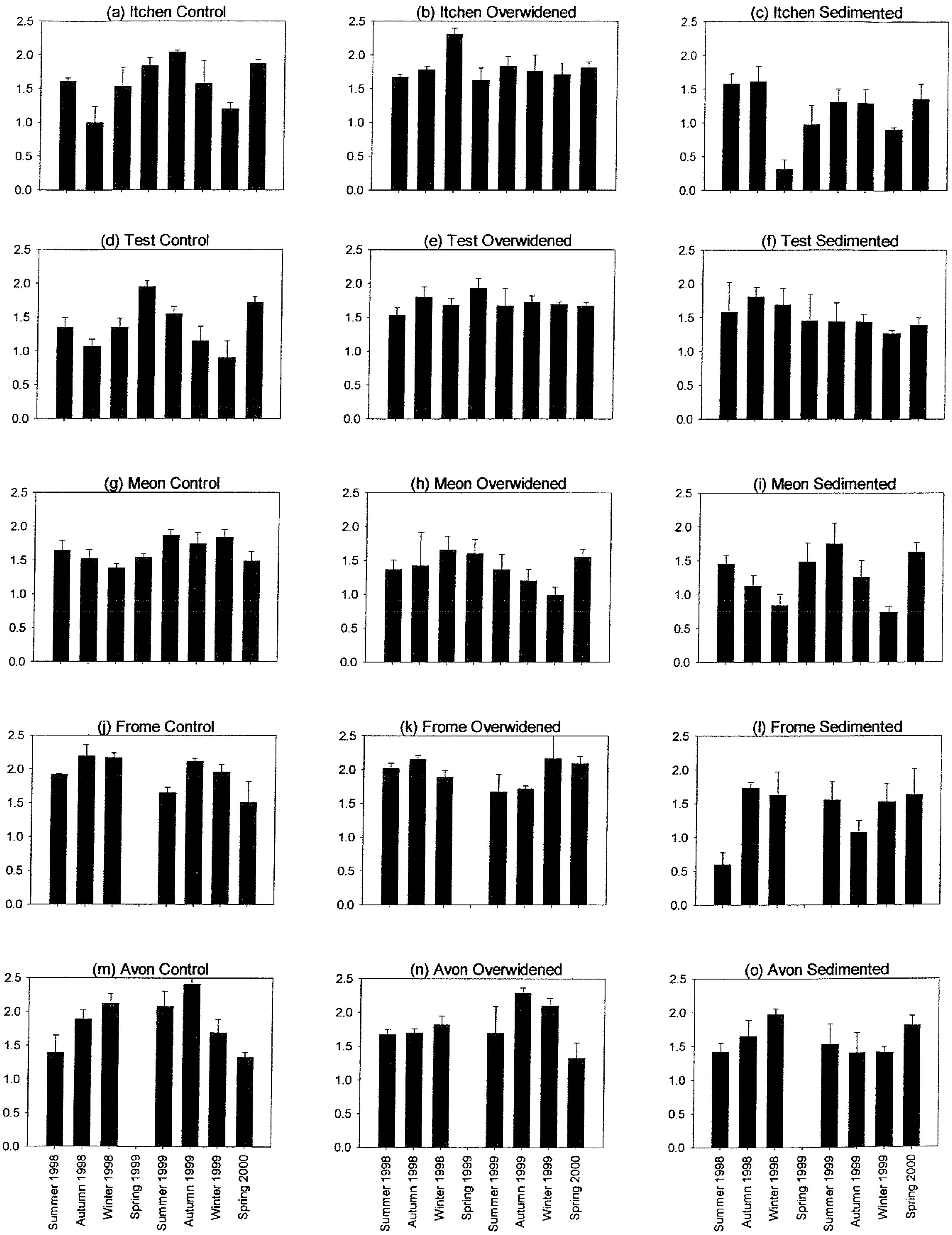


Figure 4.7. Mean Shannon Diversity (H') for the eight sampling seasons at control, overwidened and sedimented sites on the Rivers Itchen, Test, Meon, Frome and Avon.
Vertical bars show ± 1 Standard Error.

All five rivers showed lowered abundances at the sedimented sites compared to control and overwidened stretches (Figure 4.3 (c), (f), (i), (l) and (o)). With regard to abundances at control and overwidened sites, two basic patterns were seen. The pattern observed in the Rivers Test, Frome and Avon was that control sites (Figure 4.3 (d), (j) and (m)) supported higher total abundances than overwidened sites (Figure 4.3 (e), (k) and (n)). A slightly different pattern was seen in the abundances at sites on the Rivers Itchen and Meon where overwidened sites (Figure 4.3 (b) and (h)) had slightly greater abundances than the control sites (Figure 4.3 (a) and (g)).

Similar to the abundance results, depressed taxon richness was observed at sedimented sites on all five rivers (Figure 4.4 (c), (f), (i), (k) and (o)) compared to the control sites (Figure 4.4 (a), (d), (g), (j), and (m)). These figures also indicated that there was very little difference between the taxon richness of overwidened sites (Figure 4.4 (b), (e), (h), (k) and (n)) when compared to control sites on all of the rivers. Table 4.9 shows that mean taxon richness was slightly greater at the overwidened sites on the Rivers Itchen, Test and Meon compared to the control sites on these rivers.

BMWP scores displayed a similar pattern to both abundance and taxon richness at the control, overwidened and sedimented sites. Sedimented sites again showed a consistently lower BMWP score (Figure 4.5 (c), (f), (i), (l) and (o)) compared to both control (Figure 4.5 (a), (d), (g), (j) and (m)) and overwidened sites (Figure 4.5 (b), (e), (h), (k) and (n)). Table 4.9 again indicated the similarity of BMWP scores at control and overwidened sites. Overwidened sites on the Rivers Itchen, Test and Meon had higher BMWP scores compared to the control sites.

Although not quite as clear as the previous parameters, ASPT scores were again consistently lower at sedimented sites (Figure 4.6 (c), (f), (i), (l) and (o)) compared to the control (Figure 4.6 (a), (d), (g), (j) and (m)) and overwidened sites (Figure 4.6 (b), (e), (h), (k), and (n)) on all five rivers. Table 4.9 also shows that mean ASPT values were lower at these sites. Generally, control and overwidened sites had extremely similar mean ASPT values, especially on the Rivers Itchen, Frome and Avon.

Similar to Figure 4.6 the pattern of lowered Shannon diversity was slightly less clear when compared to abundance and taxon richness. However, Table 4.9 showed that mean Shannon diversity was lower at sedimented sites on the Rivers Itchen, Meon, Frome and Avon (Figure 4.7 (c), (i), (l) and (o)). Shannon diversity at control (Figure 4.7 (a), (d), (g), (j) and (m)) and overwidened sites (Figure 4.7 (b), (e), (h), (k) and (n)) were again extremely similar to one another.

4.5.2 Catchment level investigations

Figures 4.3 to 4.7 indicate that there are certain trends in the ecological quality of the sites monitored on all five rivers. Generally, control and overwidened sites had greater abundances, taxon richness, BMWP, ASPT and diversity values compared to sedimented sites. One of the principal aims of this research was to investigate the influence of channel modifications at a catchment level and it was therefore decided to consider the individual rivers as replicates rather than investigate each system individually. Table 4.10 provides a summary of the ecological quality parameters calculated for the control, overwidened and sedimented sites. The values are means covering the seasonal sampling periods over the two sampling years.

Table 4.10. Summary of the Abundance, Taxon Richness, BMWP, ASPT, H' for the site classifications (average of five rivers), ± 1 Standard Error.

Site classification	**Faunal parameters**				
	Abundance	**Taxon richness**	**BMWP**	**ASPT**	**H'**
Control	722 ± 81.9	17 ± 0.7	78 ± 4.1	5.11 ± 0.1	1.66 ± 0.1
Overwidened	497 ± 49.5	17 ± 0.5	85 ± 2.9	5.61 ± 0.1	1.73 ± 0.1
Sedimented	166 ± 30.2	9 ± 0.6	42 ± 3.4	4.47 ± 0.1	1.38 ± 0.1

Three way unreplicated randomised block ANOVAs were performed on the means for each faunal parameter, with treatment (site type), season and year as main factors and river as a blocking factor. This allowed for comparison between site type, taking the variation encountered between rivers into account. Table 4.11 provides a summary of the findings.

Table 4.11. Three way randomised block ANOVAs with treatment, season and year as main factors and river as block.

Measure	Treatment effect	Season	Year	River
Log Abundance	**<0.001**	**0.004**	0.402	**0.028**
Taxon richness	**<0.001**	**0.003**	0.249	**<0.001**
BMWP	**<0.001**	**0.002**	0.139	**<0.001**
ASPT	**<0.001**	**<0.001**	0.169	**<0.001**
Shannon diversity (H')	**0.001**	0.642	0.607	**0.001**

It is evident that for each faunal parameter difference between treatment (site type) was extremely significant, supporting the observations of difference between the control and overwidened sites and the sedimented sites. A degree of seasonal difference is also evident for log abundance, taxon richness, BMWP and ASPT. Significant between rivers differences were also observed for each parameter.

It is therefore evident that, on a catchment level, there is a significant difference in the ecological character/quality of control, overwidened and sedimented sites, even when variation between rivers is taken into account. The recognition of patterns in community composition is an important aspect of stream ecology (Townsend 1989) and an understanding of the natural and anthropogenic shifts in populations is essential for the interpretation of river ecology. Therefore the remainder of the results section deals with the analysis of the faunal composition of samples collected at the different site types, combining the data at a catchment level and considering the individual rivers as replicates.

4.5.3 Faunal composition: Abundance of the 15 most common taxa

The relative abundance of the 15 most common taxa from control, overwidened and sedimented sites is shown in Figure 4.8. Control and overwidened sites had relatively similar abundances of many of the common taxa, for example *Baetis* sp., *Caenis* sp., *L. volckmari*, *E. aenea*, and Oligochaeta were all found in similar quantities. However, some subtle shifts in community composition were seen when overwidened sites were compared to control sites. Certain taxa showed reductions in their abundance, for example, Simuliidae, and less obviously, Gammaridae, *P. jenkinsi*, *Agapetus* sp., and *E. danica*. Sphaeriidae and *H. siltalai* were not among the 15 most common taxa at overwidened sites.

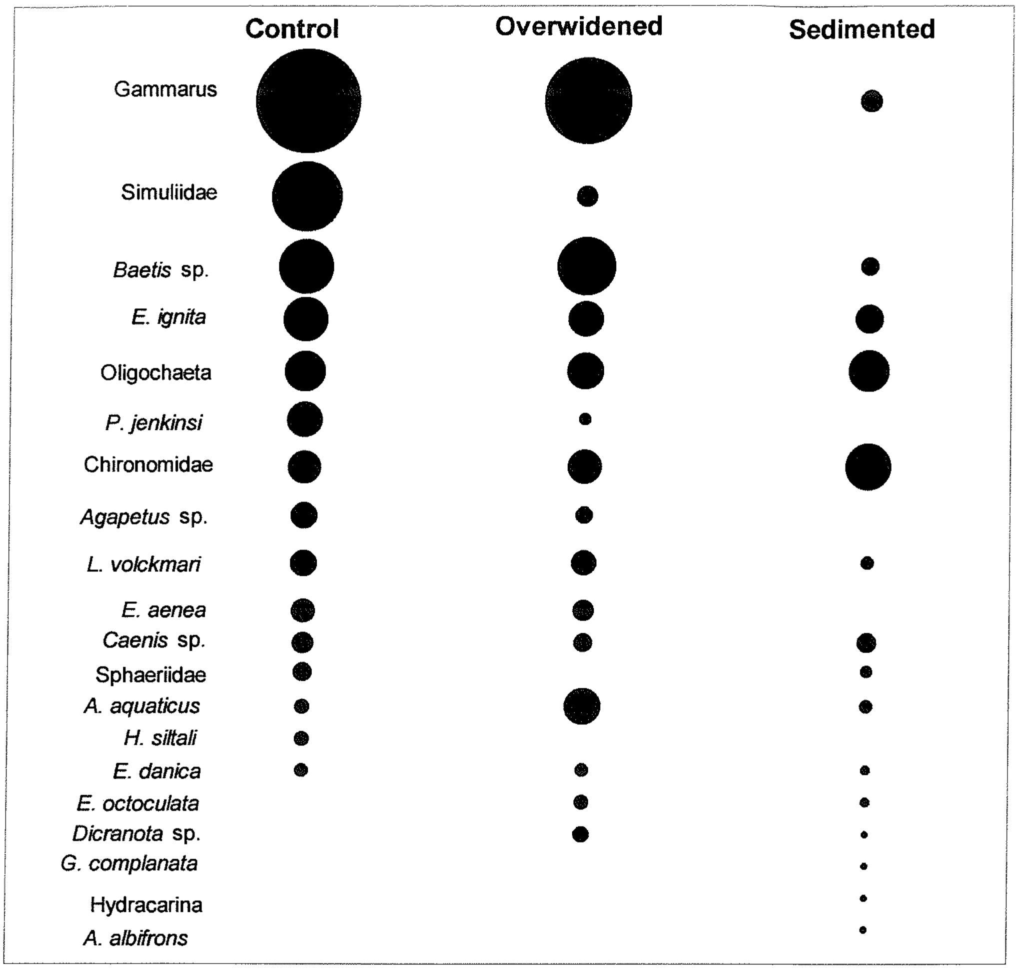


Figure 4.8. Mean abundance of the 15 most common taxa for control, overwidened and sedimented site types averaged across the five rivers. Size of circle represents the mean relative abundance of each taxon.

However, two different taxa were found more commonly at these sites, namely *E. octoculata*, and Dicranota. In addition, *A. aquaticus* and Chironomidae both increased in abundance. However, when sedimented sites were considered, Chironomidae and Oligochaeta dominated the community, with Gammaridae and *Baetis* sp. much reduced in abundance. A number of taxa were not among the 15 most common taxa at sedimented sites, including Simuliidae, *P. jenkinsi*, *Agapetus* sp., *E. aenea* and *H. siltalai*. Additional taxa found more commonly in sedimented sites included *Glossiphonia complanata*, Hydracarina, and *A. albifrons*. Therefore, distinct differences in the macroinvertebrate community were observed between control, overwidened and sedimented. Further community analyses have been carried out utilising multivariate techniques.

4.5.4 Community analysis

Initial exploratory community analysis of species data was undertaken using Detrended Correspondence Analysis (DCA), an indirect unimodal ordination method. This option determines how strongly unimodal data are by looking at the lengths of the resulting ordination axes, which further determines which multivariate tests you use. In all cases the lengths of the ordination axes in DCA were less than 3.0 SD, which justified the use of linear ordination methods (PCA and RDA) (ter Braak and Prentice 1988).

Principal Components Analysis results

Summaries of the results of the PCA performed on log transformed abundance data for the three site classifications (control, overwidened and sedimented) are given in Table 4.12, and Figure 4.9. The eigenvalue of the first Axis was 0.255. In a PCA, where eigenvalues for individual axes all lie between 0 and 1, values of greater than 0.5 are taken to indicate good separation of the taxa (ter Braak 1995, Cannan 1998).

Table 4.12. Ordination diagnostics from a PCA of abundance data from the five rivers sampled.

	Axis 1	Axis 2	Axis 3	Axis 4
Eigenvalues	0.255	0.113	0.079	0.072
Cumulative % variance of data	25.5	36.8	44.8	52.0

52.0 % of the variance found within the species data was explained by the four axes, with Axes 1 and 2 explaining the greatest amount of species variance (36.8%). The sample scores on Axis 1 and Axis 2 were plotted (Figure 4.9).

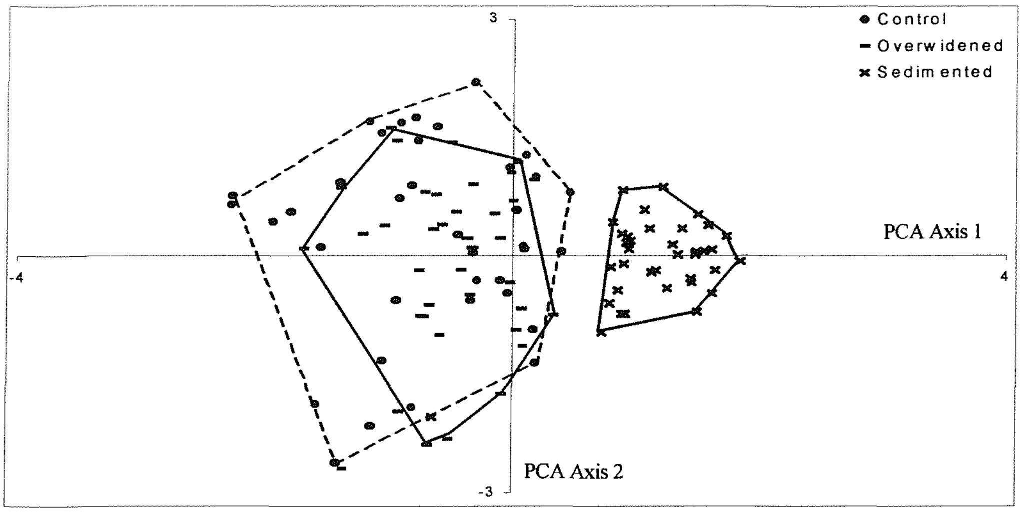


Figure 4.9. Plot of sample scores along the first two axes of a Principal Components Analysis.
Samples are log transformed abundances for site classifications (control, overwidened and sedimented) for each of the five rivers.

Figure 4.9 displays a clear separation of sedimented sites from the other two site types on each of the five rivers. Also evident from the figure is that control and overwidened sites were extremely similar to one another (as shown by the overlap). However, more variation was evident at control sites which was suppressed elsewhere. By applying ordination analysis many response variables have been reduced to a few ordination axes which aim to encompass as much of the variation as possible (Cannan 1998). These axes can be interpreted with environmental variables to establish which factors may be related to the distribution of species and samples in the data (Cannan 1998). Correlation coefficients can be used as adequate summaries of scatter plots of environmental variables against ordination axes (ter Braak 1995, Cannan 1998). In accordance with methodologies outlined by Jongman *et al.* (1995) rank correlation coefficients between the ordination scores for Axes 1 to 4 and a selection of environmental variables were calculated (Table 4.13).

Table 4.13 shows that Axis 1, which separates sedimented sites from control and overwidened reaches, showed the highest correlation with % fine sediment. % gravel, channel depth, width:depth ratio and % macrophyte were also significantly associated with Axis 1. Distance downstream, % fine sediment and temperature were all significantly associated with Axis 2. pH, catchment area, dissolved oxygen and % gravel were significantly related to Axis 3. However, no variables proved to be significantly associated with Axis 4 sample scores.

Table 4.13. Rank correlation coefficients between PCA sample scores for Axes 1 to 4 and environmental variables. Values in bold are significant (p-value: <0.001).

Variable	Axis 1	Axis 2	Axis 3	Axis 4
% Fine sediment	**-0.732**	**0.347**	0.117	0.063
% Gravel	**0.648**	-0.041	**-0.312**	-0.252
Channel Depth (m)	**-0.591**	0.225	0.179	0.125
Width:Depth	**0.510**	-0.111	-0.028	0.026
% Macrophyte	**0.336**	-0.195	-0.109	0.259
Dissolved Oxygen (%)	0.275	0.041	**0.322**	0.058
SPM (mg/l)	-0.251	-0.236	-0.113	0.053
Altitude (m)	0.230	-0.134	-0.186	-0.113
pH	0.170	-0.102	**0.407**	-0.158
Temperature (°C)	0.159	**0.332**	0.298	-0.234
Channel Width (cm)	0.091	0.111	-0.061	-0.053
Catchment area (km^2)	-0.040	-0.154	**0.371**	0.244
Conductivity (µS/cm)	-0.015	-0.024	-0.115	-0.017
Distance downstream (km)	0.005	**-0.446**	0.259	0.166

Redundancy Analysis results

In order to quantify the contribution of environmental variables to explaining the variation in the species composition, these can be incorporated directly into the ordination analysis using RDA techniques (Cannan 1998). The environmental variables shown in Table 4.13 were included in the RDA that identified the variables significantly explaining the biological variation (ter Braak 1995, Cannan 1998). It was decided at this point to partial out the influence of river, season and sampling year. This decision was made because the rivers were chosen as replicate sites as opposed to five individual studies. Year and season were also partialed out as full temporal analysis will be carried out subsequently (§ 4.6).

The RDA results are summarised in Table 4.14, which shows that six variables were significant, explaining 32 % of the faunal variation. 88.7 % of the total environment-species relation was explained by the first four axes. RDA restricts the axes to linear combinations of the environmental variables and when compared to the PCA, the RDA axes explain less variation, indicating that the environmental variables do not predict exactly the variation in the faunal composition extracted by the PCA (Cannan 1998). However, the high biological-environment correlations mean that a large part of the remaining variation is predicted, and the Monte Carlo tests results suggest that this result is significant (Cannan 1998). Figure 4.10 displays the partial RDA plot for sample scores for Axis 1 and 2.

Table 4.14. Ordination diagnostics from a RDA of abundance data from the five rivers sampled (river and season partialed out). (Significance: *** p=0.05).

	Axis 1	Axis 2	Axis 3	Axis 4	
Eigenvalues	0.190	0.042	0.027	0.025	
Species-environment correlations	0.920	0.875	0.769	0.793	
Cumulative percentage variance:					
of species data	26.6	32.5	36.2	39.7	
of species-environment relation	59.3	72.6	81.0	88.7	
Sum of canonical eigenvalues					0.715
Sum of unconstrained eigenvalues					0.320
Monte-Carlo significance test results:	*Variable*				
	Channel Depth***				
	Channel Width***				
	Width:Depth ratio				
	% gravel ***				
	% fines ***				
	% macrophyte ***				
	Temperature				
	pH				
	Conductivity				
	SPM				
	Dissolved oxygen (%)				
	Altitude				
	Distance downstream				
	Catchment area				
	Orientation ***				

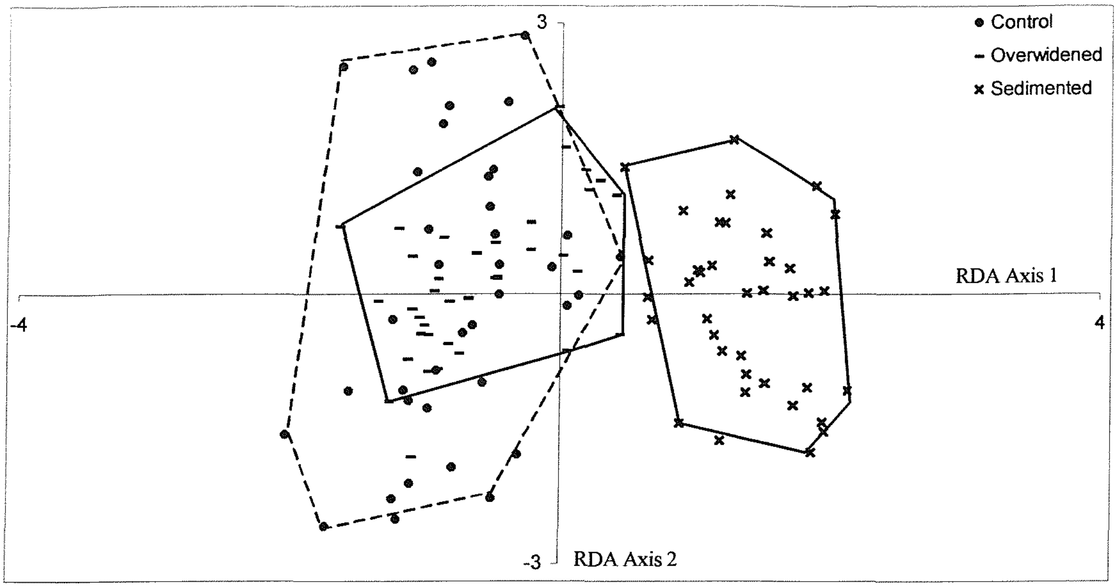


Figure 4.10. Plot of sample scores along the first two axes of a Redundancy Analysis.
Samples are log transformed abundances for site classifications (control, overwidened and sedimented) for each of the five rivers.

Figure 4.10 (constrained ordination) is extremely similar to Figure 4.9 (unconstrained ordination). Again the sites classified as sedimented were separated from control and overwidened sites. A difference between the two ordination plots was that sedimented sites exhibited a slightly more dispersed pattern and overwidened sites were slightly more closely clustered. A higher level of heterogeneity was again seen in the control site scores. However, the overall pattern in site classification was the same in the two ordination plots. This therefore indicates that the community composition of sedimented sites differs from control and overwidened site types.

To investigate further this pattern a species biplot with the significant environmental variables is displayed in Figure 4.11.

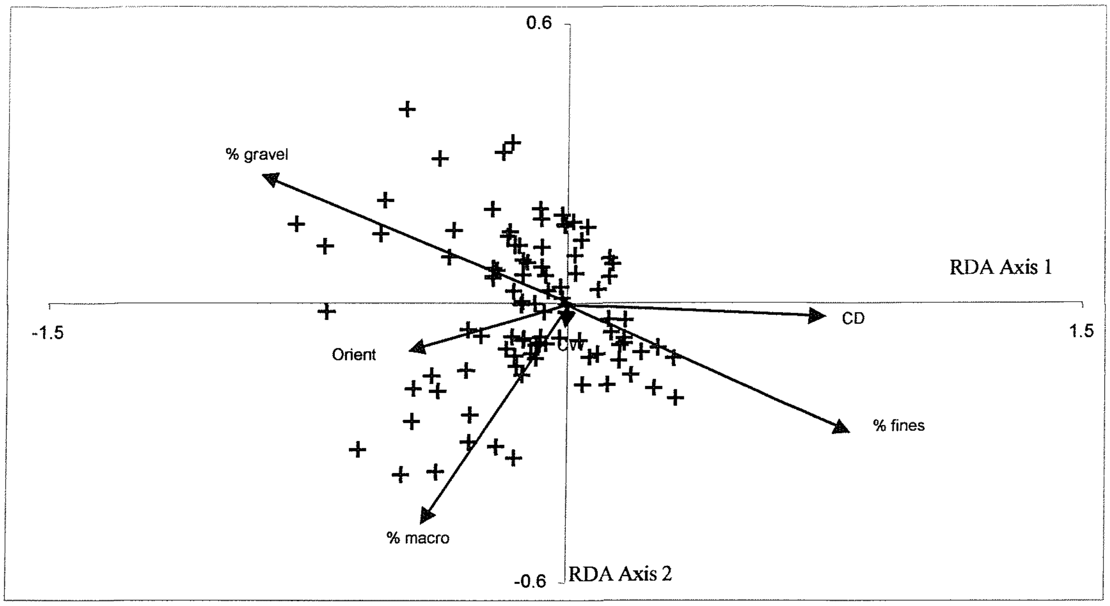


Figure 4.11. Biplot of species scores and significant environmental variables along the first two axes of a Redundancy Analysis.
Species are log transformed abundances for each of the site types sampled on the five rivers. River, season and year are partialed out. Significant environmental variables: CD = channel depth, % fines = proportion of fine sediment, % macro = proportion of macrophytes, Orient = geological orientation, % gravel = proportion of gravel, CW = channel width.

The biplot of species and environmental scores displays, in linear methods, approximate values of correlations between species and environmental variables. In the species-environment biplot, arrows often represent environmental variables (Cannan 1998). The biplot scores of environmental variables give the co-ordinates of the head of the arrows (Figure 4.11). Environmental variables with long arrows are the most important in the analysis: the longer the arrow the larger the effect of the variable on the species (ter Braak 1995). Each arrow determines an axis in the diagram and the species points must be projected on to the axis in question (ter Braak 1995). Figure 4.11 shows the environmental variables in relation to the species scores, and according to the length of the arrows, % fines, % gravels and % macrophytes were the most important environmental variables in this analysis.

If species are represented by arrows as well as the environmental variables, the cosine of the angle between the arrows of a species and an environmental variable is an approximation of the correlation coefficient between the species and the environmental variables (ter Braak 1995). A qualitative observation can be made from such correlations by noting that arrows pointing in the same direction indicate a high positive correlation, and that arrows pointing in opposite directions indicate a high negative correlation.

Given that we are primarily interested in the influence of catchment processes on the substrate quality of chalk rivers, Figure 4.11 was redrawn with instream environmental variables, including % gravels, % fines, % macro, channel width and channel depth (Figure 4.12). The physical nature of these environmental variables suggests that the invertebrate communities were influenced primarily by physical factors. Taxa near the intersection of the axes were found across a wide range of samples. Taxa located away from the centre were associated with a narrower range of samples whose environmental characteristics can be related to the environmental variables plotted. Many of the species (those unlabelled in Figure 4.12 and Figure 4.13) that were found in this investigation showed no clear relationship with any environmental variable. For example, the cased caddis *G. pilosa*, *L. lunatus* and *Hydroptila* sp., the mayfly family Leptophlebiidae, and the stonefly *L. geniculata* showed no evident relationship with any of the environmental factors measured. The unusually stable physical conditions, excellent water quality and abundant food (algae and detritus) in chalk streams may enable taxa to live in a much broader range of habitats than in more disturbed or polluted systems (Harrison 2000). It is therefore often difficult to tie specific taxon distributions to any one particular environmental factor.

The species correlated with % gravel represented a variety of taxonomic groups including the crustacean, Gammaridae, the cased and caseless caddisflies, *P. latipennis* and *R. dorsalis*, the Coleoptera, *E. aenea*, the gastropod, *A. fluviatilis*, the Plecoptera, *I. grammatica*, the Ephemeroptera, *E. ignita* and the Diptera, *Dicranota* sp.. All of these species are classic gravel organisms, requiring fast flows and coarse substrates for habitat and functional purposes (Fritter and Mannuel 1995).

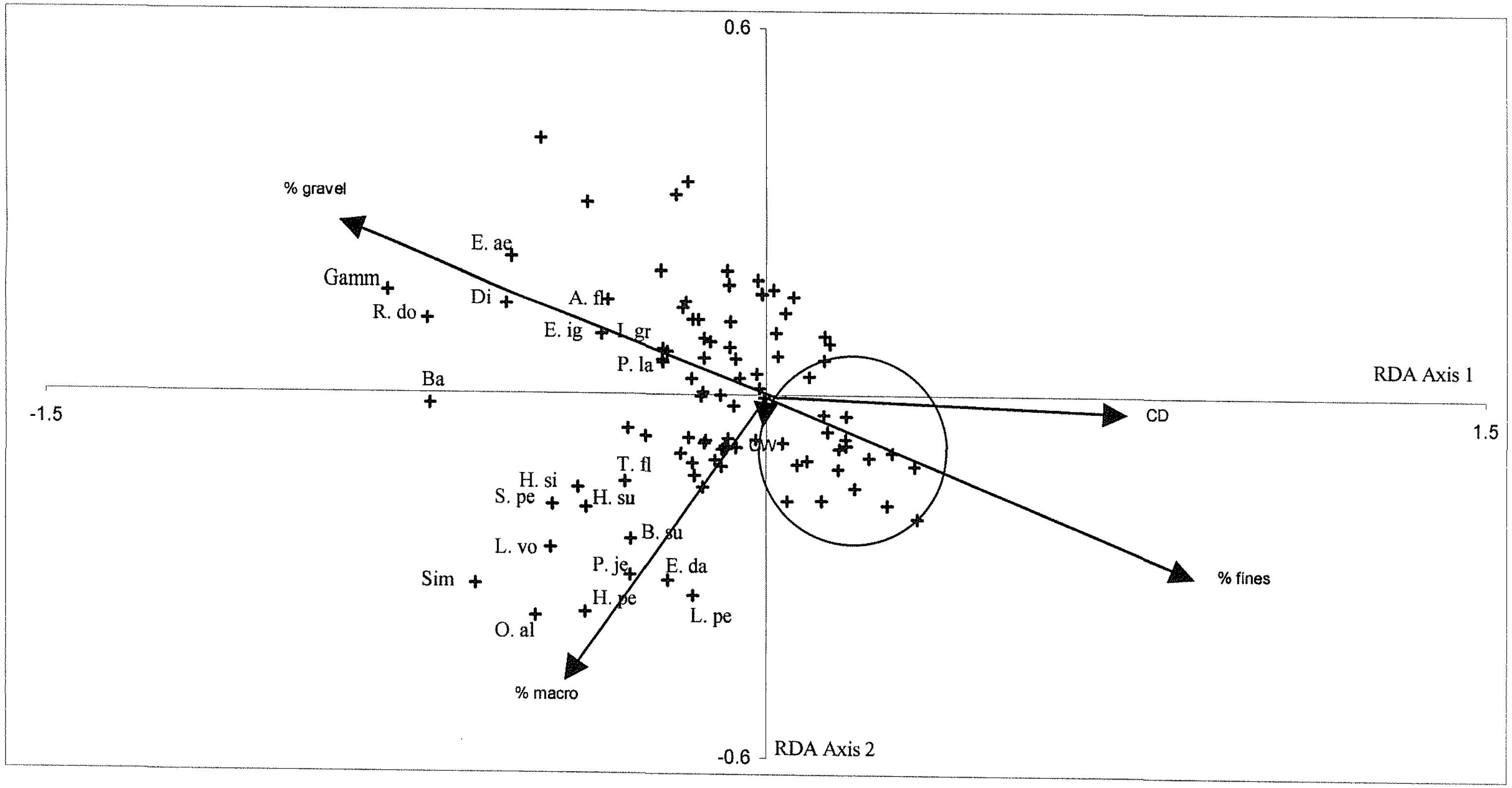


Figure 4.12. Biplot of species scores and significant instream environmental variables (RDA). Species are log transformed abundances. River, season and year are partialed out. Environmental variables: CD = channel depth, % fines = proportion fine sediment, % macro = proportion macrophytes, % gravel = proportion gravel, CW = channel width. Species associated with % gravel: Gamm: Gammaridae; R.do: *R. dorsalis*; Ba: *Baetis* sp.; Di: *Dicranota*; E.ae: *E. aenea*; A.fl: *A. fluviatilis*; E.ig: *E. ignita*; P.la: *P. latipennis*; I.gr: *I. grammatica*. Species associated with % macrophytes: Sim: Simuliidae; O.al: *O. albicorne*; H.pe: *H. pellucidula*; L.pe: *L. peregra*; E.da: *E. dancia*; P.je: *P. jenkinsi*; L.vo: *L. volckmari*; S.pe: *S. personatum*; H.su: *H. sulphurea*; H.si: *H. siltalai*; T.fl: *T. fluviatilis*; B.su: *B. subnubilis*.

The circle in Figure 4.12 has been expanded in Figure 4.13 to identify those species associated with fine sediments.

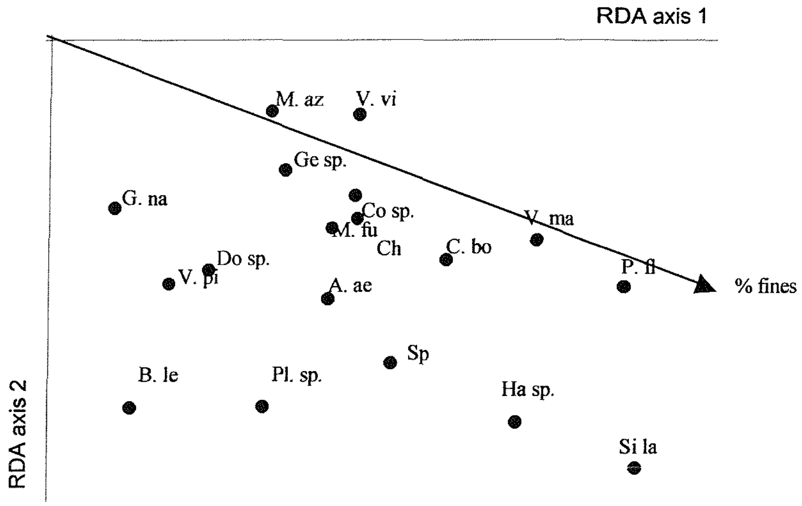


Figure 4.13. Sub plot of Figure 4.12 displaying those species associated with the % fine environmental variable.
Species associated with fine sediment include: Si sp.: *Sialis lutaria.*; Ha sp.: *Haliplus* sp.; P. fl: *Polycentropus flavomaculatus*; B. le: *Bithynia leachi*; Pl. sp.: *Platambus* sp.; Sp: Sphaeriidae; V. ma: *Valvata macrostoma*; C. bo: *Cymatia bonsdorffii*; A. ae: *Aphelocheirus aestivalis*; V. pi: *Valvata piscinalis*; Do: *Donacia* sp.; M. fu: *Mesovelia furcata*; Ch: Chironomidae; Co sp.: *Colymbetes* sp.; V. vi: *Viviparus viviparus*; Ge sp.: *Gerris* sp.; G. na: *Gyrinus natator*; M. az: *Mystacides azurea.*

The taxa highlighted above are all characteristic of low energy systems, functionally relying on fine sediment, and can be divided into six taxonomic groups. Those species related to low energy systems included the gastropods *B. leachi*, *V. macrostoma*, *V. piscinalis* and *V. viviparus*, all typically found in still or slow flowing waters. The Coleoptera species *Haliplus* sp., *Platambus* sp., *Colymbetes* sp., *G. natator* and *Doncia* sp. occupy slow flowing, silty habitats often associated with emergent vegetation (Fritter and Manuel 1995). Another taxonomic group represented in this subsection are the Hemipteran species, for example *C. bonsdorffii*, *Gerris* sp., and *M. furcata* all of which can be associated with the emergent vegetation linked with slow flowing, sedimented areas. Two caddis species were found to be associated with fine sediment, *P.*

flavomaculatus and *M. azurea*, both of which are found in still or slow flowing regions of rivers. For example, *M. azurea* has been found amongst tree roots and other vegetation, and *P. flavomaculatus* prefers organic rich areas for feeding. The presence of Sphaeriidae, Chironomidae and *S. lutaria* largely reflects the burrowing habit of these organisms.

There would also appear to be an association between species and the % macrophyte environmental variable. Species correlated with this % macrophyte included the filter feeder Simuliidae and cased caddis *B. subnubilis*, which attach themselves to the instream macrophyte, and the gastropod *P. jenkinsi*. Other species associated with macrophyte beds in chalk rivers included *L. peregra, H. siltalai, T. fluviatilis and E. danica*, which have all previously been associated with these habitats (Harrold 1964, Wright 1992). However, interestingly, other species, more generally associated with gravel habitats were correlated with macrophyte presence (Figure 4.12), for example, the caddisflies *O. albicorne*, *H. pellucidula*, *S. personatum*, the Elminthidae beetle *L. volckmari*, and the mayfly *H. sulphurea.* The presence of these species may be explained by the fact that sites were sampled proportionally according to the presence/absence of mesohabitats. Therefore, accepting that instream macrophyte beds were not sampled individually and that macrophyte beds are often found in gravel areas with fast flows, it was not unexpected that gravel species were found associated with macrophyte species.

4.5.5 Flow relationship

Throughout descriptions of the species associated with certain instream habitats it is evident that species were not only associated with the physical substrate characteristics of chalk rivers, but are also intimately linked with the flow regime of the rivers (Extence *et al.* 1999). This link is not unexpected, as reduced/slow flows will promote increased deposition of sediments (Cordone and Kelly 1961, Everest 1987, Wood *et al.* 1999), giving rise to higher fine sediment levels throughout certain river reaches. In order to investigate the relationship with flow, the species analysed in the RDA previously were assigned a flow category derived from Extence *et al.* (1999). These classified species were subsequently plotted in Figure 4.14.

A clear shift can be seen in the species-flow relationship in Figure 4.14. Species associated with rapid/moderate flows were located on the left-hand side of the figure and the species primarily associated with slow flowing/standing waters were displayed on the right-hand side.

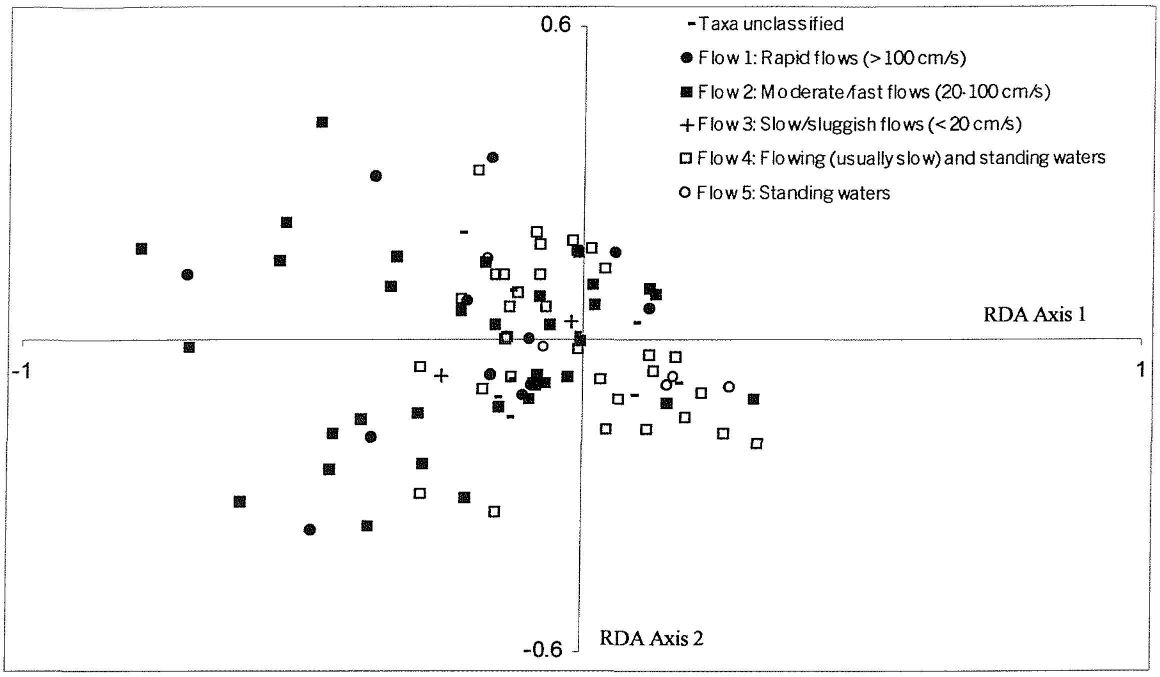


Figure 4.14. RDA species plot with each species assigned a flow category. Flow data derived from Extence *et al.* (1999).

The majority of species previously identified as being associated with fine sediment were classified as flow category 4 defined by Extence *et al.* (1999) as taxa primarily associated with flowing (usually slow) and standing waters. It therefore becomes evident that there is a clear link between the flow regime and physical substrate properties. There were a few exceptions to this general pattern, with two species associated with moderate/fast flows located on the right-hand side of Figure 4.14. These species were *P. flavomaculatus* and *A. aestivalis*, both of which were assigned flow category 2. It can therefore be assumed that these species were located in this section of Figure 4.14 due to reasons other than the physical substrate and flow; for example, this finding may be a functional response due to the high levels of organic matter deposited in sedimented sites.

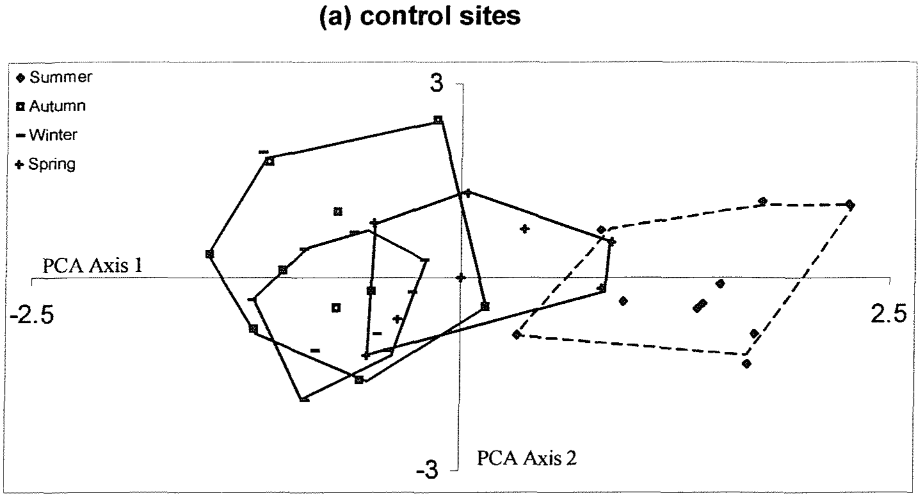
4.6 TEMPORAL ANALYSIS

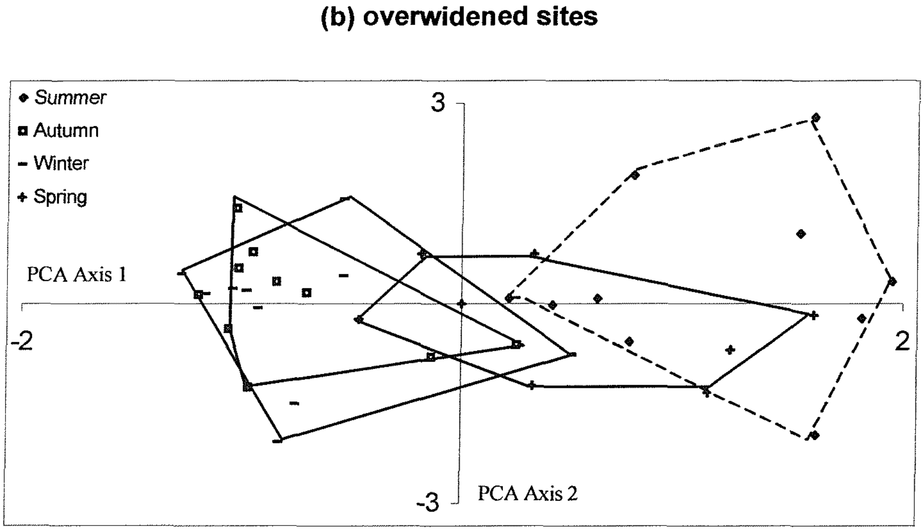
The temporal scale over which the current investigation was undertaken provides an excellent opportunity to examine both inter-and intra-annual stability of macroinvertebrate communities, as samples were collected at each of the 15 sampling sites over eight seasons from summer 1998 to spring 2000. Temporal analyses were carried out on abundance data alone as it was felt that this parameter would detect any shifts in macroinvertebrate communities over time particularly well. Control, overwidened and sedimented sites were dealt with individually so to highlight any temporal trends, both intra- and inter-annual, at the three site types.

4.6.1 Intra-annual stability

Figure 4.3 displays the seasonal abundance of macroinvertebrates at the three site types for each of the five rivers. The three way randomised block ANOVA on log abundance showed that abundances displayed a significant seasonal difference, taking yearly variation into account (p-value 0.004, Table 4.11). A seasonal increase in macroinvertebrate abundance during summer months was detected at each of the site types, e.g. Figure 4.3 (b), (d), (j), (k), (m), (n) and (o).

Seasonal patterns were also investigated using PCA ordination for each of the site types (Figure 4.15 (a) to (c)). The influence of river and year were partialed out during the analysis resulting in partial PCA plots reflecting the influence of season alone on the community structure of macroinvertebrate communities at the three site classifications. Figure 4.15 (a) displays the sample scores from control sites showing a high level of overlap between winter, autumn and spring sampling periods. Also evident was a degree of separation during the summer months. A similar pattern was exhibited at the overwidened sites (Figure 4.15 (b)), with summer months again differing from the remaining three seasons. However, there is a greater degree of variability in community composition at these sites throughout all four seasons.





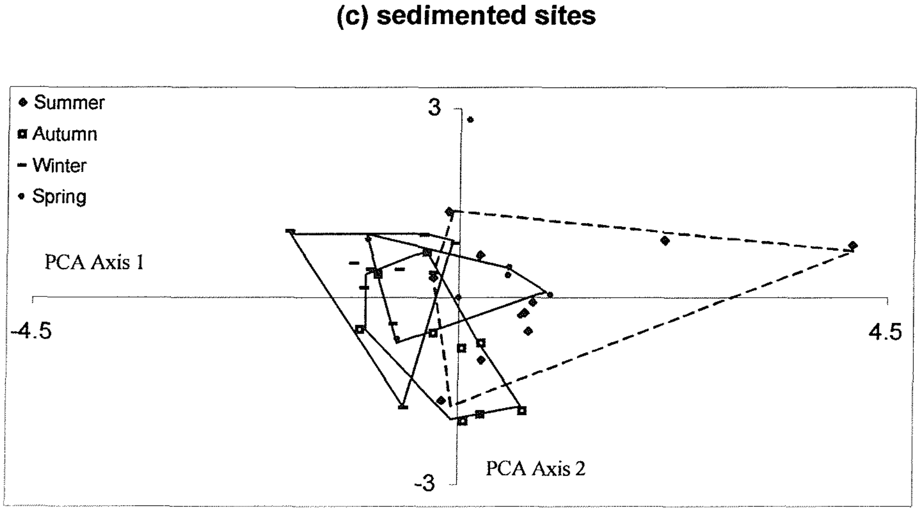


Figure 4.15. Partial PCA plots of samples scores on abundance data from (a) control, (b) overwidened and (c) sedimented sites.
PCA sample scores for axes 1 and 2 are plotted.

A slightly different seasonal pattern was observed at the sedimented sites (Figure 4.15 (c)). Overall, there was little separation of the community during the four seasons, with winter, autumn and spring months extremely similar to one another. This was also the case for the majority of summer sample scores with the exception of two outliers, which represented the communities at the sedimented sites on the Rivers Itchen and Avon.

The abundance data for each site classification were converted to a presence/absence basis. Further analyses were carried out on the control and overwidened sites with the aim of establishing whether the separation of the summer communities in Figure 4.15 (a) and (b):

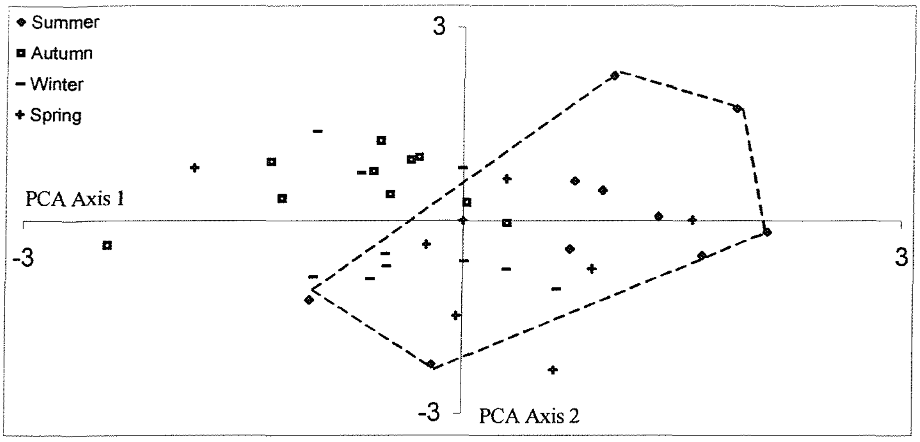
(a) reflected an actual shift in the community composition, or,

(b) was solely related to a change in the abundance of species consistently present.

Analyses were not carried out on the sedimented sites as Figure 4.15 (c) displayed very little seasonal variation.

Figure 4.16 (a) and (b) displays the partial PCA results on presence/absence data for control and overwidened sites. Figure 4.16 (a) and (b) shows that there was a greater degree of seasonal overlap in the communities at both control and overwidened sites compared to Figure 4.15 (a) and (b). This result suggests that the summer separation of invertebrate communities observed in Figure 4.15 (a) and (b) was primarily due to the abundances of individual taxa peaking during the summer months, rather than there being a community composition shift.

(a) control sites



(b) overwidened sites

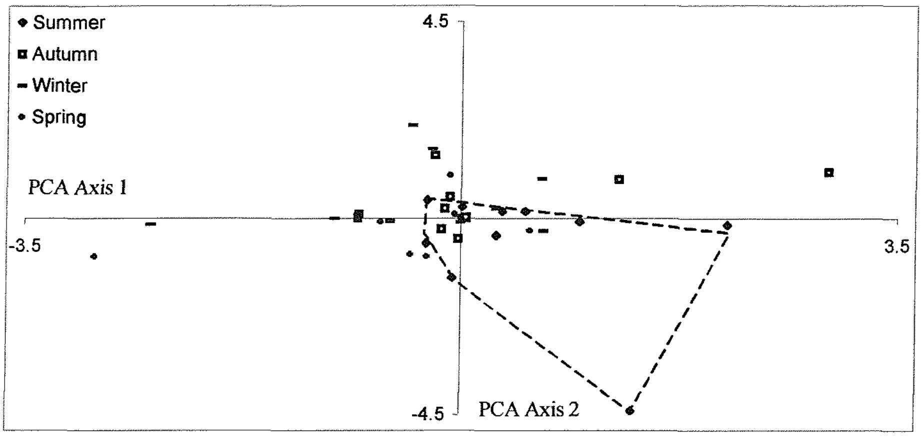


Figure 4.16. Partial PCA plots of sample scores on abundance data from (a) control and (b) overwidened sites.
PCA sample scores for axes 1 and 2 are plotted.

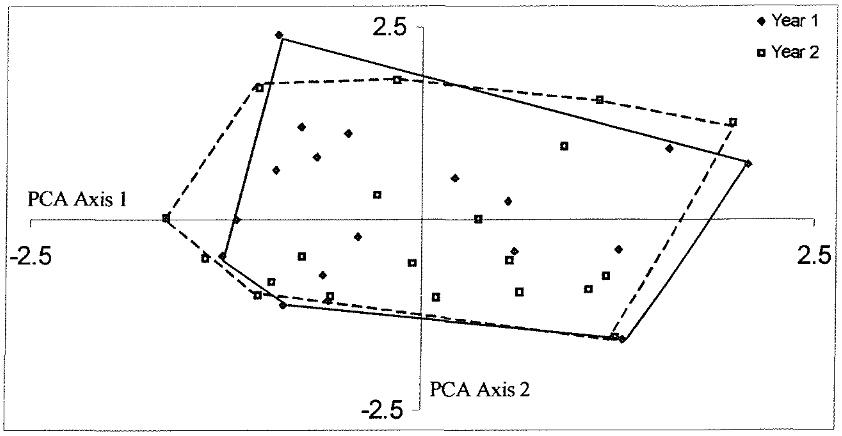
4.6.2 Inter-annual variation

The normally predictable flow regime that characterises a chalk stream is important in promoting the growth of macrophytes and providing a relatively stable environment for the associated macroinvertebrate fauna (Wright 1992). However, there are few long term studies on river systems. Such studies have the value of providing knowledge of the temporal stability of macroinvertebrate communities (Armitage *et al.* 1995, Wright and Symes 1999). The current investigation has generated a two year database of macroinvertebrate abundances for each of the site classifications on each of the five river systems. Given the knowledge of relative stability of many physical, chemical and biological aspects of chalk river systems, very little inter-annual variation in macroinvertebrate communities might be expected. Again, analyses have focused on abundance data alone as it is felt that these data provide the most insight into the temporal dynamics of macroinvertebrate communities.

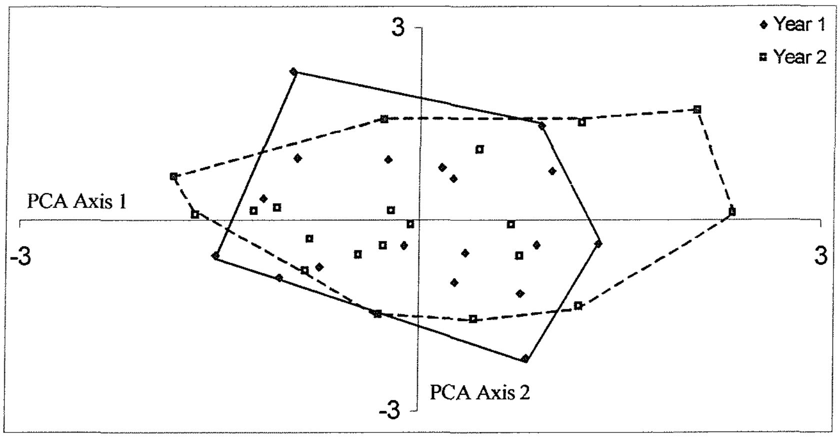
Figure 4.3 (a) to (o) shows the mean abundance data for the three site types on each river over the two years that sampling took place. It is clear from this figure that there was little between sampling year variation. The three way randomised block ANOVA supports this finding with a p-value of 0.402 (Table 4.11) indicating that there was no statistical difference between sampling years.

Similar multivariate analyses to those undertaken during seasonal pattern examination were performed on abundance data for the different site types to examine the between year stability in macroinvertebrate communities. In this section the influence of river and season were partialed out allowing the full examination of community composition on an annual basis. Figure 4.17 (a), (b) and (c) displays the PCA sample scores for Axis 1 and 2. These plots indicate that there was very little inter-annual variation in the community composition of samples from control, overwidened and sedimented sites. Figure 4.17 (a) and (b) suggests that control and overwidened sites were extremely similar in terms of the community composition and abundance over the two sampling years. Samples collected during the second sampling year at overwidened sites (Figure 4.17 (b)) showed slightly more variation in terms of community composition, but generally there was a large amount of overlap in sample scores indicating that inter-annual variation was extremely low. Figure 4.17 (c) indicates that samples collected at sedimented sites showed slightly more variation in the first sampling year. However, the interannual variation in community composition was extremely low.

(a) control sites



(b) overwidened sites



(c) sedimented sites

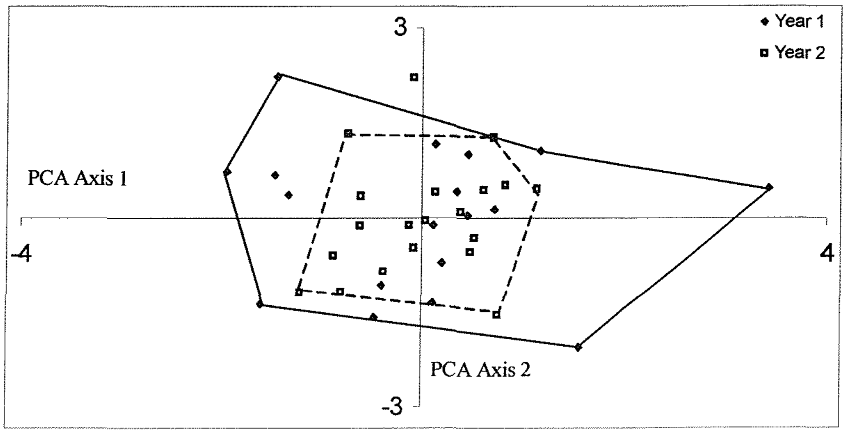


Figure 4.17. Inter-annual stability of the macroinvertebrate community composition over the two sampling years for (a) control, (b) overwidened and (c) sedimented sites.
All figures are partial PCA, with river and season as covariables. Sample scores plotted.

4.7 DISCUSSION

Overwidening has been recognised for some time as a consequence of channel modification (Mainstone 1999), but the relative impact this process has had on benthic macroinvertebrate communities has received little attention to date. Sedimentation of chalk rivers has also recently been identified as a major concern (Mainstone 1999, Wood *et al.* 1999, Environment Agency 2000a). Clean mid-stream gravels are a key habitat requirement for a range of riverine species and are extremely characteristic of chalk stream systems. Hence, any process that compromises the quality of these habitats may affect the overall functioning of chalk river systems.

4.7.1 Confounding issues

The overall aim of this study was to determine the relative significance of channel modification on the community structure of macroinvertebrates in chalk streams in southern England. The longitudinal changes in the physical conditions of rivers, which in turn lead to longitudinal patterns in biotic variables, have been highlighted. For example, general patterns include a decrease in the channel slope, an increase in discharge, and a change in the variability and nature of flow (Giller and Malmqvist 1998). Downstream shifts in fauna have not often been examined in detail due to the intensive nature of the sample collection and processing techniques (Minshall 1988, Downes *et al.* 1993). However, of the work carried out in the UK, information indicates that increases in taxonomic richness may be expected with increasing distance from source (Vannote *et al.* 1980, Wright *et al.* 1984, Ivol *et al.* 1997, Cannan 1998). These shifts have been associated with changes in the physical parameters of the rivers. For example, Statzner and Higler (1986) suggested that high taxon richness occurred at points of transitions in hydraulic stress, where a relatively large number of animals live near the limits of their tolerance.

There was no consistent order of sampling sites within rivers as they were effectively randomaly selected. Therefore if longitudinal factors were playing a significant role in determining faunal communities, we may expect a downstream increase in taxon richness and hence associated faunal parameters (e.g. community diversity (H')). However none of the downstream sites were the most taxon rich (Figure 4.4 (c), (d), (f), (k) and (n)) indicating that downstream effects were not that important in this research. Also, although distance downstream did correlate significantly with Axis 2 scores from the PCA (correlation coefficient -0.446), this correlation was not as strong as that exhibited by treatment effect (% sediment and % gravel, -0.732 and –0.648 respectively) on Axis 1 (which explained the greatest amount of variation in the PCA). This indicates that site treatment was far more influential than distance downstream in determining the faunal characteristics of the invertebrate communities.

4.7.2 Impacts of sedimentation

The increasing input of sediment to river systems has been recognised since the early 1920s (Wu 1931, Ellis 1936). The direct and indirect impacts of fine sediment on the fauna of streams are numerous. Effects include reduced light penetration, smothering, habitat reduction and the introduction of adsorbed pollutants (pesticides, metals, nutrients) (Ellis 1936, Lenat *et al.* 1981). Most of the literature concerned with the effects of sand and silt on the aquatic fauna has dealt with cases in which there has been complete or partial smothering of biotopes (Chutter 1968). Cordone and Kelly (1961), in a review of the literature on the effects of silt and sand, dealt entirely with cases in which stony bottom biotopes were altered by the deposition of fine material. In many instances the abundance of fauna of the stream bed was considerably reduced where there was sedimentation (e.g. Herbert *et al.* 1961).

Previous research has also suggested that in addition to lower abundance of benthic macroinvertebrates, taxonomic richness can be severely reduced by sediment addition (Tebo 1955, Gammon 1970, Hynes 1960, 1970, Reed 1977). Results from previous research indicate that there is a marked association between certain taxa within different substrates (Wood and Armitage 1999). For example, Wood and Armitage (1999) and Wood *et al.* (1999) found a lower taxonomic richness associated with sedimented stretches. Coupled with this, studies that have examined macroinvertebrates at sites naturally composed of fine sediments (Soluk

1985, Strommer and Soluk 1989) or subject to severe sedimentation owing to human activity such as mining (Nuttall and Bielby 1973, Quinn *et al.* 1992) reported low species diversity. This suggests that rivers subject to increased sedimentation may be characterised by a low abundance, taxon richness and diversity of macroinvertebrates (Wood and Armitage 1997).

Overall, the majority of the results from this study agree with the detrimental influence of fine sediment on macroinvertebrate communities previously observed, with lower abundances, taxon richness, BMWP and ASPT scores associated with sedimented sites (Figures 4.3 to 4.6 (a) – (o), Table 4.9). For example, in the current study, the number of taxa recorded at sedimented sites ranged from 7 to 12 (Table 4.9). Wood and Armitage (1999) observed that 21 taxa were associated with a sedimented reach and concluded that this figure was well below that recorded in other English lowland streams. Coupled with this, the reduction in BMWP and ASPT scores at sedimented sites indicates that the process of sediment addition to chalk rivers is having a detrimental ecological impact.

The findings of this research generally agree with the assumption that sediment addition reduces the diversity of macroinvertebrate communities. There was one exception to this pattern, the River Test, where the sedimented site had a higher mean diversity value than the control site samples (Figure 4.7 (f)). Silt is recognised as a crucial component of the habitat matrix so typical of chalk river systems. Many chalk river species are dependent upon silt for all or part of their life cycle (Mainstone 1999). Therefore, certain species may in fact exhibit a preference for the conditions associated with increased levels of sedimentation. For example, many studies have found the lotic invertebrate community in marginal areas has a higher diversity compared to mineral habitats (Scheffer *et al.* 1984, Painter 1999, Harrison 2000). Marginal vegetation was noted intruding into the main channel at the River Test sedimented site and therefore it was impossible to avoid sampling these areas.

Marginal areas are structurally complex, and, importantly, possess physical attributes common to other mid-channel habitats. For example, where the part of the marginal vegetation is entrained by high flows, the habitat will resemble that of the trailing stems and leaves of mid-channel *Ranunculus*. Margins therefore contain not only specialist marginal taxa but also many taxa more common to other habitats. Gravel and *Ranunculus* possess fewer attributes of other habitats, containing fewer taxa that normally live elsewhere and are more dominated by a few characteristic taxa as a result (Harrison 2000). This is certainly the case with these results, with gravel samples dominated by Gammaridae, *Baetis* sp. and the Elminthidae beetles. As the site on the Test was the only sedimented site with extensive marginal areas it is perhaps unsurprising that an increased diversity was recorded. Coupled with this, Shannon diversity not only takes into account the number of species recorded but also the proportion of individuals within that species. Therefore, perhaps the higher diversity in the River Test is a reflection of a more equally distributed community compared to those associated with control and overwidened sites, which tend to be dominated by several taxa.

However, much debate exists over the influence of sediment on invertebrate community structure (Chutter 1968, Rosenberg and Resh 1996, Wood and Armitage 1997). Most studies indicate that sediment addition may cause a change in the composition of the benthic community, usually involving a shift towards certain chironomid species (Rosenberg and Snow 1975, Dance 1978). In contrast other investigations have found community structure largely unchanged by sediment addition (Herbert *et al.* 1961, Pearson and Jones 1975). However, community analyses carried out on the current data show that there were markedly different communities associated with sedimented site types compared to control and overwidened (Figure 4.8, 4.9 and 4.10). For example, several taxa were highly correlated with the % of fine sediment (Figure 4.13). The correlation of *S. lutaria* and Sphaeriidae are largely thought to reflect the burrowing nature of these organisms (Ellis 1978, Elliott 1996). The poaching action of cattle at the water's edge along grazed reaches has been noted to create shallow, silty, low gradient margins with small pools of standing water (e.g. conditions noted at the sedimented sites on the River Frome, Itchen and Avon). These conditions have been shown to favour gastropod and hemipteran taxa possibly through increased abundance of fine particulate organic matter and periphyton (Harrison 2001). Painter (1999) also observed a community shift towards Mollusca and Coleoptera in fenland drainage ditches due to the low gradient and high accumulation of organic detritus. A similar shift in community structure can be seen in the present data with the community associated with fine sediment dominated by the mollusc family Valvatidae and the beetles *Haliplus* sp., *Platambus* sp., and *Colymbetes* sp.

Therefore, the generally depressed ecological character of sedimented sites is perhaps not unexpected. However, the fact that this result can be detected on a catchment scale is extremely interesting, suggesting that chalk stream communities respond in a relatively predictable way to this form of ecological stress.

4.7.3 Impacts of channel widening

Since overwidening involves manipulation of one or more of the dependent hydraulic variables of slope, depth, width and roughness, feedback effects may be expected on the resident biological communities (Brookes 1994). The effects of channel modification on aquatic habitat and associated organisms may be both direct and indirect (Petts 1987, Brookes 1988). Overwidening of channels may dramatically change the channel morphology to the detriment of the biology of watercourses (Fruget 1992). For example, alteration of the width and depth of variables in a channel may create shallow and unnatural hydraulic conditions that create an unsuitable habitat for fish (Brookes 1994).

When overwidened sites were considered their results proved to be most interesting. Generally very little difference was highlighted between overwidened and control sites on any of the five rivers (Figure 4.3 to 4.7 (a) to (n), Figure 4.9, 4.10), especially for taxon richness, BMWP, ASPT and Shannon diversity (Table 4.9). The only differences were in terms of abundance, where overwidened sites on the Rivers Test, Frome and Avon (Figure 4.3 (e), (k) and (n)) had lower abundances compared to control sites. However, although experiencing reduced abundances, the overall community quality was not changing when compared to control values (e.g. ASPT and H' Table 4.9).

The general similarity between control and overwidened sites therefore suggests that the process of overwidening has been less severe than predicted. In fact, the increased mean abundances, taxon richness, BMWP, ASPT and H' values at several overwidened sites may also indicate that the physical conditions brought about as a result of overwidening actually prove favourable to certain benthic macroinvertebrates. For example, Figure 4.8, which displays the mean abundances of the 15 most common taxa at the three site types, shows that two species (*E. octoculata* and *Dicranota* sp.) were found in greater abundances at overwidened sites compared to control reaches. In addition, *A. aquaticus* and *Baetis* sp. were found in higher numbers at these sites than at control reaches. The associated reduction in Simuliidae is perhaps not unexpected as there was often a lack of instream vegetation associated with the overwidened sites, which is the primary habitat for this taxon. However, the overall similarity suggests that overwidened sites were extremely similar to control sites in terms of their community composition, (as indicated by the lack of separation in Figure 4.9 and 4.10). Therefore it can be concluded that although there is a slight shift in community composition at overwidened sites in terms of abundance (Figure 4.8), it was not sufficient to separate these sites from control reaches using PCA.

The general similarity between overwidened and control sites may not be as surprising as one might first think. The natural channel form of chalk rivers appears to be wider and shallower than other river types (Mainstone 1999, Sear *et al.* 1999). Therefore the actual physical widening of chalk rivers may not impact upon benthic communities to such a degree as might be expected in other deeper, non-chalk lowland rivers. Indirect effects of channel widening could also account for the increased diversity and taxon richness and general biotic quality of some overwidened sites compared to control reaches. For example, shading by bankside vegetation has often been found to limit primary and secondary production in streams (Hawkins *et al.* 1982, Dudgeon and Chan 1992, Reed *et al.* 1994, Friberg *et al.* 1997). During this study overwidened sites were generally associated with little marginal tree growth and hence subject to less shading than control sites (e.g. the River Test control site experienced almost entire canopy closure). A slight increase in *Baetis* sp. at the overwidened sites could be attributed to less shading as these highly mobile invertebrates have been found to track even small, local variations in algal abundance (Kohler 1984). Another factor could be that overwidened sites were generally shallower with less marginal areas than control reaches making them less favourable to predatory fish (Harrison 2000), therefore potentially reducing the influence of this limiting factor for invertebrates.

The associated impacts of channel widening outlined in Figure 4.1 (reduced current velocities, reduction in substrate, flow and depth diversity and reduced water levels) are obviously having minimal impact on benthic macroinvertebrate communities in these rivers. However, the issue of reduced water levels may not have been addressed directly in this study and hence not reflected in the results as all samples were collected from the benthic substrate and not from marginal areas. Reduced water levels may well impact upon marginal areas, which traditionally support increased macroinvertebrate diversity and taxon richness (Harrison 2000). These areas were not sampled during this study, as the focus was placed on examining the benthic substrate response of invertebrates, which may contribute to the relative similarity between control and overwidened sites.

4.7.4 Temporal patterns

The understanding of the natural temporal changes in populations is essential for the interpretation of river ecology and particularly for any programme of management and conservation (Underwood 1994). The two year sampling period over which this investigation was carried out provides an ideal opportunity to assess the relative contribution of temporal variation within the data set, on both a seasonal and inter-annual basis.

Seasonal and between year differences in the environmental regimes of the rivers are normal, and the macroinvertebrate assemblages observed within any given system must be adapted to withstand the variability they encounter (Wright and Symes 1999). For example, Brewin *et al.* (2000) observed strong seasonality in the invertebrate communities of monsoon disturbed streams in Nepal, with a 20 % reduction in taxon richness observed between winter and pre-Monsoon periods. However, chalk streams represent the stable end of a continuum from stable to flashy discharge regimes. They have an annual discharge, temperature and chemical regime that favours the growth of macrophytes and offers a relatively stable range of habitats and food resources for the macroinvertebrate community throughout the year (Wright 1992, Wright and Symes 1999). Therefore it might be expected that these riverine systems exhibit relatively little temporal variation in terms of their community structure compared to other river systems.

Seasonal patterns

Despite the general stability of chalk river systems, a degree of seasonality in invertebrate communities has previously been identified. Wright and Symes (1999) showed that numerous taxa displayed seasonal changes in their abundance and biomass between summer and winter months, and Wright (1992) observed significant seasonal differences in the biomass data of macroinvertebrates associated with five biotopes. Cannan (1998) also showed a degree of seasonality in the invertebrate data analysed whilst investigating the significance of habitat in structuring macroinvertebrate communities. More specifically, Wright *et al.* (1983) (River Lambourn, Berkshire), Armitage *et al.* (1995) (the Mill stream, Dorset) and Cannan (1998) (the River Frome, Dorset) all noted that faunal assemblages appeared to be most abundant in the summer months.

Analysis of the present data revealed that seasonal change was apparent in the data set despite the relative temporal stability of chalk streams in terms of flow and water chemistry (Casey and Newton 1973, Berrie 1992). Generally peak abundances of invertebrates were observed during summer months (Figure 4.3), with the PCA also revealing a degree of summer separation in the invertebrate community composition at control and overwidened sites (Figure 4.15). It therefore becomes evident that seasonal differences in the community composition detected during this research show similar patterns to previous work on chalk rivers.

The PCA carried out on presence/absence (rather than abundances) showed a reduced summer separation, suggesting that the seasonal separation of summer communities was a result of changes in abundances rather than actual shifts in the taxon composition of the communities. This indicates that species turnover in chalk streams is extremely constant despite the potential impact of catchment processes. This finding is again in agreement with previous research carried out on chalk rivers (Wright *et al.* 1983, Armitage *et al.* 1995).

However, it is often difficult to link the community composition solely to seasonal influences. Cannan (1998) found that patterns within the seasonal mesohabitat data were complex due to the interaction of habitat, site and seasonal factors, all of which contributed significantly to the variation in the faunal assemblages. This finding is common in studies where numerous factors are considered. For example, Boulton and Lake (1992) had to restrict temporal analysis to individual habitats in order for patterns to emerge. Although the multivariate assessment of catchment processes on invertebrate communities were temporally restricted (with season and year partialed out), initial examinations of temporally unrestricted data exhibited similar patterns of catchment process impact. This therefore suggests that the patterns of catchment processes observed generally dominated the seasonal variation, and that the seasonal variation in communities in chalk rivers is small compared to other systems, which is in agreement with previous studies by Armitage *et al.* (1995), Humphries *et al.* (1996) and Cannan (1998).

Inter-annual patterns

The importance of temporal macroinvertebrate data has been known for some time (Lancaster *et al.* 1996). However, to date there have been few long term studies on the ecological quality of river systems (McElravy *et al.* 1989, Giller *et al.* 1991). Long term ecological studies are potentially valuable as they can provide information on both community stability and the response of invertebrates to stress (Hildrew and Giller 1994). Extremely low levels of inter-annual variation were exhibited in the abundance and community structure of invertebrate communities associated with the different site types (Table 4.11, Figure 4.17), which further suggests that the community structure of macroinvertebrate communities exhibits strong temporal stability.

The normally predictable flow regime that characterises a chalk stream is important in promoting growth of macrophytes and providing a relatively stable environment for the macroinvertebrate fauna (Wright 1992). The most notable long term variation in macroinvertebrate communities has been linked to changes in the discharge regime (Wright 1992). Catastrophic events such as major spates, which can lead to habitat disturbance (McElravy *et al.* 1989), are not generally encountered in chalk streams. Therefore it is at the other end of the spectrum that changes in discharge may impact upon chalk rivers.

During the early 1990s there were several low flow years and subsequently much work was carried out on the potential ecological impacts of these changes in discharge regime (Mainstone 1999, Wood *et al.* 1999). General effects on the biota were associated with reduced current velocities, the loss of water depth and the drying out of marginal and riffle areas, reduced dilution of effluents, increased water residence time in the river, and reduced reoxygenation from turbulent flows (Mainstone 1999). Wright (1992) also concluded that the between year differences in the invertebrate communities on the River Lambourn, Berkshire, were related to conditions of low discharge.

However, few low flow conditions were experienced in the period of this investigation. For example, there was no significant difference in the discharge at equivalent sampling sites on the River Frome and Avon throughout the two years of sampling (p-value 0.919 and 0.927 for the Frome and Avon respectively). This therefore suggests that the primary source of inter-annual variation (change in discharge regime) on invertebrate communities was minimal throughout this study. It can therefore be concluded that during periods of normal or non-low flow, and despite the influence of catchment processes, macroinvertebrate communities exhibit little between year variation in their abundance and community structure at a catchment scale.

4.7.5 Influence of instream factors

Microhabitat diversity

A clear finding shown by the multivariate analyses was that although control sites were extremely similar to overwidened reaches in terms of community composition, control sample scores tended to show more heterogeneity which was suppressed elsewhere (Figure 4.9 and 4.10). One potential explanation for this is that habitat variability at control sites was greater than that experienced at overwidened and sedimented sites. This is perhaps not unexpected as the definition of control sites required the presence five principle mesohabitats: main channel gravel/cobbles, instream vegetation, sand, silt and riffle gravel, the extent of which will vary naturally, a classic feature of chalk rivers (Armitage 1997). In contrast, overwidened sites were dominated almost entirely by gravel beds, with no or little instream vegetation. It follows therefore that the macroinvertebrate communities associated with control sites would show greater levels of variation. Previous studies focusing on the macroinvertebrates associated with individual mesohabitats have shown these habitats to exhibit temporal and spatial variation, the amount of heterogeneity specific to the environment (Armitage *et al.* 1995, Cannan 1998).

For example, although it was not an objective of this research to investigate the specific importance of instream macrophyte beds, it is recognised that these habitats support specific communities (Wright *et al.* 1983, Minshall 1984, Rutt *et al.* 1989, Palmer *et al.* 1991, Harper *et al.* 1992, Wright 1992, Cogerino *et al.* 1995). It is also recognised that these habitats vary seasonally (Armitage *et al.* 1994) therefore potentially introducing an element of habitat variability not experienced at the impacted sites. Macrophyte presence was often limited at these sites due to the shallow flows and increased levels of fine sediment (Armitage *et al.* 1994). However, given that the influence of season was partialed out in the multivariate analyses it is proposed that the greater control heterogeneity can be attributed to the higher number of habitats available for macroinvertebrate community occupation, each experiencing their own limiting factors.

Flow relationship

Relating species traits to habitat characteristics can provide important insights to the structure and functioning of stream communities (Usseglio-Polatera *et al.* 2000). Variations in instream hydraulic and substratum characteristics have been shown to have a strong influence on the faunal composition (Sheldon and Haick 1981, Meffe and Minkly 1987, Giller *et al.* 1991, Wood *et al.* 1999). Rempel *et al.* (2000) established that the distribution of invertebrates was correlated significantly with hydraulic variables and suggested that hydraulic conditions represent a major physical gradient along which the benthic community is organised. However, they noted that substratum texture and size was also an important factor in determining community structure. Therefore, separating the effect of substrate and stream hydraulics is often difficult since flow is a primary determinant of substrate composition (Clausen and Biggs 1997, Townsend *et al.* 1997). The analysis of the species scores related to flow ratings (Figure 4.14) suggests that the differentiation between mesohabitat biodiversity is at least in part related to flow variability. For example, the majority of the species associated with high % fine sediment also show a relationship with slow and slack flows.

However, this was not the case for all the taxa, e.g. *P. flavomaculatus*. On closer examination it was discovered that the micro-distribution of polycentropodid larvae is primarily related to functional aspects rather than water velocity. Edington (1965, 1968) found that *P. flavomaculatus* was most common in stream pools with low velocities as it is tolerant of decreased oxygen concentrations (Philipson 1954, Edington and Hildrew 1981, Fjellheim 1996). *P. flavomaculatus* often aggregate in parts of the stream bed where benthic prey density is highest (Hildrew and Townsend 1976), and in these favourable areas larvae may fight over net spinning sites. Polycentropodidae are exclusively predatory and the exact composition of the diet is variable from place to place according to the prey available (Edington and Hildrew 1981), with Chironomidae and Oligochaeta favourite prey items in the slow flowing areas (Higler 1978). Therefore the presence of this species in slow flowing, sedimented areas could be a functional response rather than an association with flow conditions.

4.7.6 Management implications and future work

Assessing and managing catchment disturbances, as well as understanding natural processes, is often difficult (Wood *et al.* 1999). This is due to many factors, especially the absence of baseline data on channel morphology. Spatial and temporal variability in the fauna within stream on both the meso- and macro-scale also make assessment difficult (Wright *et al.* 1983, Wright 1992).

As previously highlighted many factors have significantly altered the channel morphology of many lowland streams (Walling 1990, Mantle and Mantle 1992, Casey and Smith 1994). Given that the impacts of overwidening have been shown to be minimal the remainder of this section deals with the management implications regarding fine sediment. The most desirable way to mitigate the effects of this catchment disturbance would be to prevent sediment influx into the watercourse (Wood *et al.* 1999). This would require addressing the causal factors of the problems within the catchment rather than the impact within the stream (Wood and Armitage 1997, Mainstone 1999). However, this is often difficult. For example, in the case of watercress production large volumes of compost are added to the growing beds and these sediments may subsequently be released into the stream as a result of erosion or cleaning operations between crops (Casey and Smith 1994).

Several instream measures can be used to mitigate against sedimentation. An example of which is gravel cleaning (both mechanical disturbance and washing techniques have been undertaken; Scott and Beaumont 1994, Mundie and Crabtree 1997). The selective management of instream vegetation could be used to maintain water level and habitat diversity (Wood *et al.* 1999). Macrophyte growth helps maintain areas of fast flow velocities and gravel habitats between individual clumps, as well as slow flow velocity and fine sediment mesohabitats in backwater areas, providing refugia for many riverine taxa (Armitage *et al.* 1994). The development of clearly defined best management practices and the use of buffer strips (Wood *et al.* 1999) or the removal of arable productions from sites adjacent to water courses (which in turn reduce potential overwidening) may all be desirable (Mainstone 1999), although economically unfeasible (Rabeni and Smale 1995).

Future work should aim to address the fate of sediments within stream systems as a whole (Wood *et al.* 1999). Local gravel cleaning operations tend to shift the problem downstream whilst causing major disturbance to resident macroinvertebrate communities. It is particularly important that long-term studies follow changes in siltation from drought to ‘normal’ years, both as a result of instream hydraulics and fluctuations in sediment supply from catchment, and river management activities (Mainstone 1999). The long-term response of instream biota to these changes in the physical environment will identify the most sensitive biotic components and provide data on the ecological sustainability of water resources under stress (Wood *et al.* 1999).

4.8 CONCLUSIONS

Patterns of distribution and abundance of invertebrates in streams are a result of the function of the specific requirements of different taxa and the particular environmental complex that exists at any section of stream (Hawkins *et al.* 1982). Substrate and food are two of the most important aspects of the environment and have accordingly received a great deal of study (Hynes 1970). In this study I have attempted to discover if the impacts of overwidening and sedimentation on the ecological quality of benthic invertebrate communities are detectable at a catchment wide scale.

The influence of channel modification, in terms of overwidening, was found to be minimal. In fact, some overwidened sites actually supported increased abundances and other associated faunal parameters. On a catchment level, univariate and multivariate analyses revealed little difference in the community structure between control and overwidened sites. All of these findings support the observation that overwidening is having minimal detrimental impact on the ecological quality of chalk streams, and in fact at times was proving beneficial to certain taxa.

In contrast, sedimentation was generally associated with decreased abundance, taxon richness, BMWP, ASPT and diversity. On a catchment level, a clear difference was seen in the community structure of sedimented sites when compared to control and overwidened reaches. When analysed it was found that this community shift was detected at a species level, attributed to a variety of factors including substrate characteristics, hydrological influences and functional responses of individual taxa. No chemical factor was found to be significantly associated with explaining the variation in the community data set.

An element of seasonality was detected in the data set, with summer communities separate from the other seasons. This finding was in agreement with previous studies carried out on chalk river invertebrate communities. When further analysed, the separation of summer communities was attributed to increased abundances of many taxa during these months as opposed to actual shifts in the community structure.

Little between year variation was detected in the catchment data set suggesting the macroinvertebrate communities were extremely temporally stable, and that the primary source of between year variation in environmental conditions, e.g. changes in discharge regimes, was not impacting upon invertebrate communities despite the influence of catchment processes.

In conclusion, the ecological quality of benthic invertebrate communities was found to be significantly influenced by the catchment process of sedimentation rather than overwidening. However, the specific response of the invertebrate communities depended upon the relative scale of the disturbance.

CHAPTER FIVE: RECOLONISATION DYNAMICS OF MACROINVERTEBRATE COMMUNITIES IN A CHALK STREAM

CHAPTER FIVE: RECOLONISATION DYNAMICS OF MACROINVERTEBRATE COMMUNITIES IN A CHALK STREAM

5.1 INTRODUCTION

5.1.1 Background

Chalk streams are characterised by gravel streambeds (Mainstone 1999), which are known to provide the basic habitat for many macroinvertebrate communities (Mackay 1992). However, quantifying the abundance of macroinvertebrates within these habitats is difficult due to their patchy distribution. Many factors influence the distribution of macroinvertebrate communities in streambeds. In particular, disturbance has been recognised as a major structuring force in many natural communities (Matthaei *et al.* 1997); natural and anthropogenic disturbances of streambeds are common. Natural variations in flow have been observed as primary factors determining community structure (Lake 2000, Doisy and Rabeni 2001). For example, droughts can markedly reduce flow volumes resulting, in extreme cases, in complete exposure of streambeds, impacting upon the benthic macroinvertebrate communities (Rincon and Cressa 2000). On the other hand, spates and floods can remove macroinvertebrate individuals from the streambed, resulting in decreases in the abundance and taxon richness of communities (Angradi 1997, Lytle 2000).

However, as discussed in § 4.6, chalk streams are rarely exposed to such dramatic temporal hydrological change, and therefore streambed disturbance is often more associated with anthropogenic catchment activities. Examples of catchment activities identified as disturbing chalk rivers are poaching and trampling of banks and beds by cattle using the rivers as watering holes, and the process of gravel washing. To date, little research has been carried out to establish the instream disturbance impacts of these activities, and hence very little is known of the responses of macroinvertebrate communities in chalk streams to anthropogenic disturbance.

5.2 DISTURBANCE THEORY AND RECOLONISATION PROCESSES

5.2.1 Recolonisation and disturbance theory

The need to find ways of predicting responses to disturbance is particularly urgent in the case of streams and rivers, primarily because benthic habitats are often disrupted indirectly by practices and pollutants in surrounding catchments (Mackay 1992). A disturbance, either real or simulated, has enabled stream ecologists to explore theories on island biography and the relative importance of stochastic and deterministic processes in shaping community structure (MacArthur and Wilson 1967). Consequently, there is a large body of literature focusing on the rates at which bare patches of stream substrate, both large and small, are colonised by invertebrates. The ways in which animals colonise new habitats (or recolonise disturbed ones) have been studied for some time (e.g. Shelford 1913). Studies specific of this process in freshwaters have progressed from the purely qualitative (e.g. Grensted 1939) to the quantitative (e.g. Williams and Hynes 1973, Wallace *et al.* 1986, Malmqvist and Otto 1987, Boothroyd and Dickie 1989, O'Connor 1991). Studies of community-recovery following disturbance have indicated the colonisation mechanisms responsible for restoring a community to its predisturbance state. For example, Gore and Milner (1990) and Wallace (1990) reviewed many of these mechanisms, and Shaw and Minshall (1980) and Sheldon (1984) discussed some numerical responses of colonising aquatic insects.

Spatial and temporal scales of colonisation range from biogeographic to extremely local and immediate phenomena. The scale of most experimental work in streams has been small (<1m^2 of streambed) and has described the establishment of macroinvertebrate assemblages following a variety of treatments that ensured an initial absence, or lowered density, of organisms. A recurring theme in such studies has been the MacArthur-Wilson model of island colonisation, the islands being the bare stones or patches of introduced substrates (Gore and Milner 1990). Debate exists as to whether such an approach is a proper use of the model given that:

(1) the original theory was developed for species breeding on islands whereas residency in stream patches is transitory,

(2) stream islands are not surrounded by a foreign and inhospitable sea, and

(3) the species pool equivalent to the model's mainland, which is the source of colonisers, is usually the immediately adjacent streambed (Mackay 1992).

In spite of the imperfections of streambeds in terms of the MacArthur-Wilson model, small scale studies of colonisation have provided valuable information on the dynamics of macroinvertebrate communities and their particular responses to bare patches of substrate.

Disturbance concepts have received more attention with regard to terrestrial (e.g. Connell 1978, White 1979) and marine (e.g. Drayton 1971, Paine and Levin 1981) ecosystems than to freshwater systems (e.g. Fisher *et al.* 1982, Ward and Stanford 1983). A natural disturbance common to all streams and rivers is flooding, which influences the ecosystem through a downstream displacement of biomass as well as through changes in channel morphometry. As with natural disturbances in terrestrial systems, the frequency, predictability, and magnitude of such events vary among aquatic systems and are important in the evolution of adaptive strategies of the biota. All disturbances in a watershed also affect the streams, since a stream is intimately dependent on its catchment (Hynes 1975). However, in spite of disturbances to their environment, benthic invertebrates are often diverse and abundant. Given this, we must consider how macroinvertebrates respond to disturbances to their environments, and identify those factors affecting subsequent colonisation of disturbed habitats.

5.2.2 Factors affecting colonisation

Recolonisation mode

Examinations of recolonisation have often been based on lotic benthic macroinvertebrate communities because of their micro-scale, their relative ease of manipulation, and their wide distribution throughout riverine systems. Bare substrates placed in a stream have been shown to house invertebrates within 24 hours, even when the substrates had been thoroughly dry or acid-scoured (and were without eplithic growths), rather than merely brushed free of fauna (e.g. Wise and Molles 1979, Beckett and Miller 1982, Lake and Doeg 1985, Boulton *et al.* 1988, Doeg *et al.* 1989, Quinn *et al.* 1998). This rapid colonisation indicates that some macroinvertebrates are extremely mobile. Some of the early colonisers may merely inspect the substrate during the course of their normal widespread foraging, and then depart just as rapidly (Giller and Cambell 1989). In contrast, organisms more sedentary in nature may be regarded as true colonisers in that they remain in the habitat until some other factor removes them or stimulates their departure some time later (Mackay 1992).

Therefore it is evident that the colonisation mode of invertebrate taxa is an extremely important aspect when considering recolonisation dynamics. When substrates have been introduced and positioned to allow access from only certain sources (drift, adjacent natural substrates, air), drift seems to be one of the commonest movement techniques (Townsend and Hildrew 1976, Williams and Hynes 1976, Bird and Hynes 1981, Benson and Pearson 1987). Examples of the taxa most often listed among the earliest arrivals on bare substrates include Baetidae, Gammaridae, Simuliidae, and Chironomidae, all of which are known to be prominent drifters (e.g. Waters 1964, Ulfstrand *et al.* 1974, Hynes J.D. 1975, Gray and Fisher 1981, Lake and Doeg 1985, Malmqvist and Otto 1987, Dudley 1988). Minshall and Peterson (1985) observed that drift was the primary means of redistribution of benthic organisms in streams and that it is a particularly important relocating mechanism for invertebrates that do not move far under their own power such as the relatively sedentary filter-feeding caddisflies, black flies, and chironomids.

However, recent research indicates that many extremely mobile invertebrates swim and crawl in all directions as well as drifting (Hart and Resh 1980, Wiley and Kohler 1981, Kohler 1984, Richards and Minshall 1988, Kohler and McPeek 1989). Doeg *et al.* (1989) established that drift contributed to less than 50% of colonisers after 10 days, and noted that the design of some substrate-holding trays hindered accessibility by crawling invertebrates in the surrounding streambed, resulting in overestimates of drift in these cases.

The vigorous swimming of some mayfly nymphs is well known, for example Baetidae. *Gammarus* are also strong swimmers (Hynes 1970). Swimming down to the substrate allows some drifters to leave the water column deliberately, as seen in *Baetis* and *Gammarus* species. (Ciborowski and Corkum 1980). For example, mayfly larvae can use dorsoventral undulations to swim downwards reaching the streambed far sooner than stonefly and caddisfly larvae (Otto and Sjostrom 1985). Therefore the rapidity with which many mayflies and other lotic invertebrates reach local bare substrates may be due partly to drift and partly to their swimming, either from place to place on the substrate or when leaving the drift (Mackay 1992).

Nearly all benthic macroinvertebrates can crawl, and some do so rapidly (Mackay 1992). Giller and Cambell (1989) concluded that crawling contributed as many mayflies to their colonisation trays as drift did. The crawling mobility of grazers has attracted particular interest. Kohler (1984) showed that the direction and extent of movement by *Baetis* were related to periphyton levels. When the quantity of algal food was low, *Baetis* larvae moved more quickly and turned over more often as they searched for a food rich patch. Highly mobile grazers are able to cope with patchy distribution of periphyton (Hart and Resh 1980). Richards and Minshall (1988) estimated that 40 – 50% of *Baetis* arriving on a single rock reached it by crawling. After studying sources of colonisation in a Sri Lankan river, Benzie (1984) concluded that all colonisation resulted from random foraging activities over short distances. Therefore on a small scale, crawlers could colonise bare patches as effectively as drifters and swimmers (Mackay 1992).

Influence of substrate on colonisation

When small patches of substrate are physically disturbed (naturally, or when substrates are treated to mimic the disturbance effects) many factors have been recognised as having an influence on subsequent colonisers. Examples include changes in substrate size and in the food resources associated either with the substrate themselves or with their arrangement on the streambed (Mackay 1992). Although stones in experimental substrate trays do not necessarily offer the same range in stone size as natural substrate patches, they do mimic physically disturbed patches in that they are loose, with interstitial spaces which are initially free of silt and detritus (Mackay 1992). The size of substrate particles, and hence the size of interstitial spaces, can affect colonisation. Percival and Whitehead (1929), de

March (1976) and Minshall and Minshall (1977) all found characteristic animal associations in different substratum types. For example, large pebbles and cobbles (>40 mm in diameter) tend to be more stable and, once the surface is suitably conditioned, are more likely than finer substrates to attract clinging sedentary organisms (Khalaf and Tachet 1980, Malmqvist and Otto 1987). The interstitial spaces between such larger substrate particles, however, do not provide as much shelter for small invertebrates as those in gravel, which, if silt free, attract greater densities of organisms than the surrounding substrates (Minshall and Minshall 1977, Williams and Mundie 1978, Shaw and Minshall 1980). Finer experimental substrates tend to collect a different fauna, including more burrowers such as Oligochaeta, bivalves and chironomidae (Williams and Hynes 1976, Doeg *et al.* 1989).

Another effect of substrate size is on the accumulation of detritus particles. Gravel and small pebbles tend to collect more fine particulate organic matter (FPOM, <1 mm) than do coarser substrates, and are colonised by the greatest number of macroinvertebrates (Rabeni and Minshall 1977, Wise and Molles 1979, Parker 1989). This characteristic could be associated with a functional preference by certain taxa resulting in small substrates potentially characterised by greater abundances of shredders/detritivores. Gatherers also take advantage of the FPOM that rapidly accumulates within smaller substrates and are reported to be early (but not always the earliest) colonisers in experiments (Fisher and Gray 1983, Boulton *et al.* 1988, Parker 1989).

A general conclusion from the literature is that substrate particle size has numerous effects on the abundance and taxonomic composition of the colonising fauna due to both physical and functional responses of the invertebrates (Mackay 1992). Therefore substrate type is an extremely important aspect of recolonisation experiments. Several generalisations have arisen with regard to substrate type and the associated macroinvertebrate communities. These include certain benthic species being restricted to particular types of substrate, different types of substrates supporting different assemblages of invertebrates, and that larger substrate particles support greater numbers of stream invertebrates, both in terms of abundance and number of taxa. Given the current atmosphere of river restoration, and especially the reconstruction of riffle sequences through the addition of new substrate, the choice of substrate for such initiatives could therefore be extremely important.

5.2.3 Disturbances to chalk streambeds

Within chalk river catchments there are numerous activities which can cause disturbance to benthic macroinvertebrates. The main disturbances include: cattle poaching of river banks and streambeds; the practice of gravel cleaning; and the placement of artificial substrates to create riffle sequences aiming to restore the instream ecological diversity of chalk rivers. In the past ecologists have come to realise that disturbance may play an influential role in community dynamics (Sousa 1984, Strong *et al.* 1984). Although it is unlikely that such catchment disturbances will result in a complete removal of invertebrates, the biological consequences of these disturbances are unknown and could potentially play an extremely important role in determining the stability of macroinvertebrate communities in chalk rivers.

Bank poaching by cattle

Over the past centuries there have been significant changes in the land use practices of chalk river catchments. An example of this is the increase in number of cattle grazing the fields that are adjacent to the rivers (Plate 5.1).

Plate 5.1 Cattle grazing.

Where these cattle have unlimited access to the river for drinking (Plate 5.2), severe damage can be done to both the bank and the bed of the stream (Plate 5.3). The result is a potential local disturbance to benthic macroinvertebrate communities.

Plate 5.2 **Instream disturbance.**

Plate 5.3 Bank and bed damage.

Gravel cleaning

For centuries, gravel cleaning has been practised to improve chalk streams for salmonid spawning. The siltation of spawning gravels in upland rivers appears to be an increasing hindrance to salmonids' spawning success (Shackle *et al.* 1999). Salmonid fish have specific requirements for spawning, primarily clean gravel beds with low fine sediment content. However, in many British rivers, especially chalk rivers, the poor spawning success of brown trout, *Salmo trutta* L., has been partly attributed to increases in the fine sediment content of gravel streambeds (Edwards *et al.* 1972). When present in sufficient quantity, silt can reduce gravel porosity and permeability (Carling and Glaister 1987) and so also reduce intragravel water velocity, oxygen supply to fish eggs and metabolic waste removal; the net result is reduced egg survival to hatching (Iwamoto *et al.* 1978).

Since the second world war, chalk rivers have received higher fine sediment loads. Numerous factors have been identified as contributing to this problem. Carling

(1984) suggested that deforestation and deep ploughing for drainage increases sediment release to rivers. Other suggestions as to why chalk rivers have received increased sediment loads include: the decline in use of water meadows which trap silt; channel widening, which reduces water velocity, so increasing deposition; cattle trampling unfenced banks; changing agricultural practices where rapid drainage carries away topsoil and an increase in the area of land ploughed; fish farms, which discharge water containing faecal matter and unused fish foods.

The process of gravel cleaning removes fine sediment that has accumulated in the gravel matrix, making the streambed unfavourable to salmonid fish. Traditional chalk stream management involved regular gravel cleaning, e.g. by harrowing (Plate 5.4) and the decline of this labour intensive process may have exacerbated siltation problems (Shackle *et al.* 1999). Similarly, regulated rivers may not receive the flushing flows that would remove fines and maintain stream quality (Gilvear 1987).

Plate 5.4 Traditional chalk stream gravel cleaning.

A significant reduction in silt has been shown to occur after intensive gravel cleaning (Scott and Beaumont 1993). In the past, low technology practices were used, however more recent gravel cleaning treatments include tractor rotavating, high-pressure jet washing, and pump washing. Although viewed as a necessary procedure in most chalk rivers, primarily aimed at increasing production at one trophic level, namely salmonids, it can cause extreme local disturbance to other communities, e.g. macroinvertebrates, which are recognised as an important trophic level within chalk rivers (Mundie and Crabtree 1997).

Another form of potential disturbance to stream communities in chalk rivers is the introduction of artificial riffle sequences. This practice generally involves the placement of several tonnes of artificial substrate directly into the stream channel. The principle behind this restoration technique is that the riffle sequences will increase the instream habitat and flow heterogeneity, which will be reflected in an increase in biological diversity of both macroinvertebrates and fish populations. An example of such a scheme on a chalk river was carried out by the Environment Agency which involved the placement of over 2,000 tonnes of gravel into the river Allen (Dorset) to create a series of fast running riffle areas, slower glides and pools. This programme was carried out to benefit the various life stages of salmon and trout as well as endangered creatures such as the native white-clawed crayfish and the water vole (Environment Agency 2000c). However, relatively little work has been carried out on the ecological effects of the introduction of artificial substrate into chalk rivers, especially on the macroinvertebrate communities.

Therefore, disturbance in streams is a common phenomenon and can lead to variable effects on community structure within areas upon subsequent recolonisation of disturbed habitat (Johnson and Vaughn 1995). Communities are potentially maintained in a non-equilibrium state and prevented from achieving ecological saturation and tight species packing, which theoretically should characterise a stable, resource limited and niche-controlled structure (Resh *et al.* 1988, Townsend 1989). Some studies have shown rapid community recovery following disturbance (Townsend *et al.* 1987, Boulton *et al.* 1991), whereas others have found the resulting community to be different from the original (Fisher *et al.* 1982, Townsend 1989, Hildrew and Giller 1994). Studies of ecosystem disturbance and subsequent recovery provide an insight to the mechanisms underpinning community structure (Hurlbert 1975, Gray 1981, Wallace *et al.* 1989). Therefore disturbance studies have become an important component of stream ecosystem research (Resh *et al.* 1988) and it has been concluded that the patterns of recovery (e.g. recolonisation of macroinvertebrate communities) in lotic systems are dependent upon the disturbance (Whiles and Wallace 1992).

5.2.4 Study aims

This study aimed to quantify the temporal responses of macroinvertebrate communities to a worst case scenario disturbance, i.e. the complete removal of macroinvertebrate communities through gravel cleaning or the placement of new, artificial substrate into the river system, a well used practice to increase instream habitat diversity (Environment Agency 2000c). Most studies of invertebrate colonisation in running waters have been conducted on a small scale or have been restricted to specific taxa (Malmqvist *et al.* 1991). Also, little work has been carried out on establishing the recolonisation dynamics of macroinvertebrate communities in chalk rivers. It is in response to this lack of general knowledge of chalk stream colonisation that the current manipulation experiment was initiated.

It is generally considered that there will be a clear long term temporal recolonisation response of macroinvertebrate communities. Coupled with this, the influence of substrate size will also greatly influence the recolonisation dynamics of invertebrate taxa. More specifically, I hypothesise that:

- a specific temporal recolonisation pattern will be detected in the invertebrate community, with early populations significantly different to end communities in terms of abundance, taxon richness and diversity
- temporally, the abundance, number of taxa and community structure will progress towards an equilibrium, following the MacArthur-Wilson island biogeographic model (Mackay 1992)
- substrate size will be a significant factor determining the specific temporal dynamics of recolonisation
- individual taxa will display significant associations with the different substrate sizes resulting in the two different substrate types being associated with different invertebrate communities
- successional changes in the macroinvertebrate community will be associated with feeding groups

5.3 METHODS

It is therefore evident that for a meaningful representation of the recolonisation dynamics of invertebrates, much consideration must be given to specifics of the experimental design and the colonisation modes described above. The length of the experiment may well dictate the results obtained, for example, if the experiment covers a few days the dominant colonising invertebrates will be drifters. Therefore, it has been suggested that the longer the colonisation experiment is undertaken, the more representative the result will be, indicating true colonisation by individuals not solely related to drift movement (Peckarsky 1980). Therefore, in order to assess the long term dynamics of macroinvertebrate colonisation in chalk rivers, a relatively long exposure length was decided upon. Coupled with this, the actual design of recolonisation trays is an extremely important aspect, for example, trays with raised edges and those which are not flush with the streambed may well hinder the colonisation of crawling taxa. Therefore, if a representative result of these invertebrates is to be acquired a suitable substrate holder was of primary consideration in the experimental design stage.

5.3.1 Field methods

Experiments were carried out on a reach of the river Meon, a chalk stream in east Hampshire (Figure 2.4, Plate 5.5). A description of the river Meon and the catchment through which it flows is given in Chapter Two (§ 2.3.1). The experimental reach was classified as a classic chalk stream stretch, being dominated almost entirely by gravel beds. The study reach experienced a variety of flow conditions, but was dominated by an extensive riffle with fast flows. The study was conducted in this riffle area in order to reduce the influence of flow conditions. The riffle area on which the experiment was carried out was approximately 15 m wide, 20 m long and 0.1 m deep.

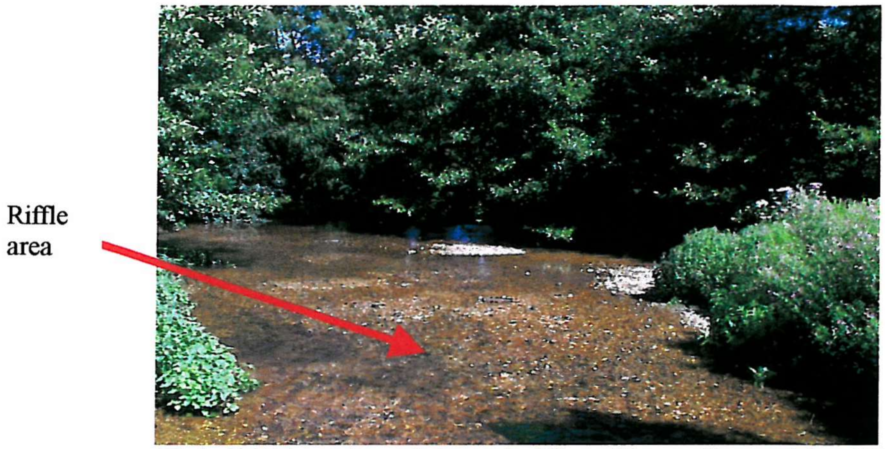


Plate 5.5 Experimental reach, River Meon, Hampshire, UK.

Experiments were designed to quantify the effects of substrate size on recolonisation by invertebrates. 15 replicates were filled with clean small substrate and 15 with clean large substrate (plates 5.6 (a) and (b)) and then implanted into the streambed. These recolonisation 'implants' consisted of plastic baskets 33 x 24 x 14 cm high, with 1 cm^2 holes to allow the passage of water through the substrate. A mesh was attached inside each basket, which allowed the free passage of water, and enabled easy removal of the substrate and retention of material accumulated. Plate 5.6 shows examples of the recolonisation baskets used throughout the experiment and Table 5.1 summarises the physical characteristics of the substrate in these baskets.

(a) **(b)**

Plate 5.6 Examples of the recolonisation baskets used in the experiment. (a) small substrate, (b) large substrate.

Table 5.1. Physical characteristics of the substrate in the two treatment baskets.

Property	Small substrate	Large substrate
Range of diameters of substrate particles (mm)	12 – 26	40 - 75
Average number of substrate particles per basket	2646	60
Average surface area of substrate particles (cm^2)	53040	12353
Average interstitial volume (%)	44	54

In the Autumn 1999 recolonisation baskets were prepared and implanted within the experimental reach after removal of gravel (Plate 5.7).

Plate 5.7. Placement of recolonisation baskets.

The baskets were embedded so that the top of the implant was level with the surface of the surrounding bed. The natural substratum consisted primarily of cobbles and gravels, with slight sand (Plate 5.8).

Plate 5.8. Recolonisation basket in place, flush with the streambed.

Each basket was positioned so that one implant did not impinge upon any other (Plate 5.9). They were positioned throughout the midsections of the reach, starting upstream and working downstream.

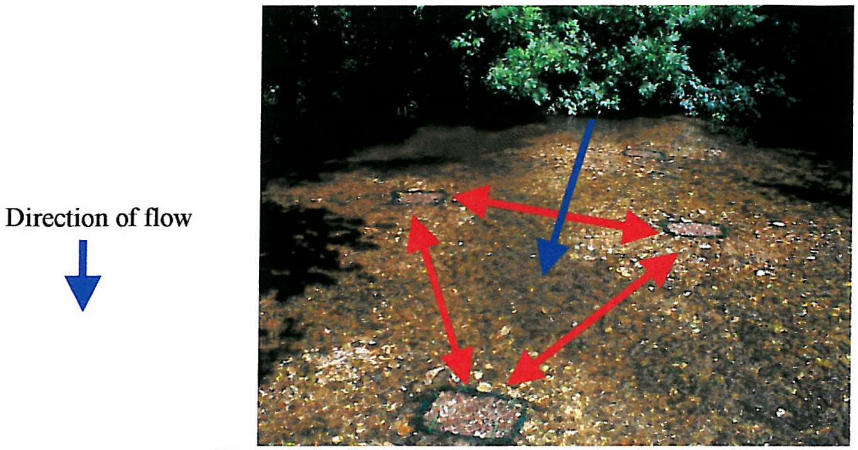


Plate 5.9. Positioning of recolonisation baskets.

Three replicates for each substrate size were left in the streambed for 16, 33, 55, 69 and 85 days. At these intervals three replicates were removed from the streambed and, treating each basket individually, the contents emptied immediately into a bucket. Each emptied basket was washed on site to ensure the complete removal of animals. The mesh was also placed in the bucket and returned to the laboratory where it was washed free of organisms.

5.3.2 Laboratory procedures

Once the baskets were returned to the laboratory the contents were preserved by adding formaldehyde, and the samples processed. The contents of each basket were washed through a 500 μm sieve that retained the substrate. Insects were washed free from the substrate prior to it being discarded. The macroinvertebrate organisms were picked from the resultant material, identified to species level wherever possible and their abundances counted. The same methods were applied to the processing of these samples as were used in Chapter Four, (§ 4.3.2).

At the initial starting point of the experiment and at the same time as the baskets were removed from the river, benthic background samples were taken, enabling a comparison with background macroinvertebrate communities to be undertaken.

These samples were returned to the laboratory, preserved, sorted and identified to species level using the same methodologies outlined above.

5.3.3 Data analysis

The data were analysed using several techniques, including both univariate and multivariate techniques. The experiment covered a period of 85 days, with the artificial substrates resident in the streambed for a variety of exposure times. Replicates for the two substrate treatments were exposed for 16, 33, 51, 69, and 85 days. The experimental design was a 2 X 5 factorial with three replicates per class. Treatments were substrate (small and large) and exposure time (5 periods). Results of the experiment have been analysed by two way ANOVAs. Data were transformed if they did not pass the normality and equal variance tests undertaken prior to the ANOVA. Univariate analyses have been based on abundance, taxon richness and Shannon diversity (H'). Multivariate analyses were also carried out on the data, a full description of which is given in Chapter Four, § 4.4.2.

5.4 RESULTS: ABUNDANCE, TAXON RICHNESS, DIVERSITY

A total of 49 macroinvertebrate taxa were recorded in the samples processed (Table 5.2). A summary of the raw data collected is shown in Appendix F.

Table 5.2. Macroinvertebrate taxa recorded in the small, large and background samples throughout the experiment.

Taxa	Small substrate	Large substrate	Background
TRICLADIDA			
Dugesia polychroa	X	X	X
Dendrocoelum lacteum	X	X	X
GASTROPODA			
Potamopyrgus jenkinsi	X	X	X
Bithynia leachi		X	
Lymnaea peregra	X		
Lymnaea stagnalis	X	X	
Physa fontinalis	X	X	X
Planorbis planorbis		X	
Planorbis vortex	X	X	X
BIVALVIA			
Sphaeriidae	X	X	X
OLIGOCHAETA	X	X	X
HIRUDINEA			
Glossiphonia complanata	X	X	X
Erpobdella octoculata	X	X	X
HYDRACARINA	X		
ISOPODA			
Asellus aquaticus	X	X	
AMPHIPODA			
Gammaridae	X	X	X
EPHEMEROPTERA			
Baetis sp.	X	X	X
Rithrogena semicolorata		X	
Heptagenia sulphurea	X	X	X
Ephemerella ignita	X		X
Ephemera danica	X	X	X
Paraleptophebia wernei	X	X	
Caenis sp.	X	X	X
PLECOPTERA			
Leuctra hippopus	X		X
ODONATA			
Calopteryx splendens			X
COLEOPTERA			
Platambus sp.	X		X
Gyrinus natator	X	X	X
Elmis aenea	X	X	X
Limnius volckmari	X	X	X
TRICHOPTERA			
Rhyacophila dorsalis	X	X	X
Agapetus fuscipes	X		X
Polycentropus flavomaculatus	X	X	
Tinodes unicolor	X		X
Hydropsyche pellucidula	X	X	X
Hydropsyche siltalai	X	X	X
Potamophylax latipennis		X	X
Odontocerum albicorne	X	X	X
Athripsodes albifrons		X	
Goera pilosa	X	X	X
Sericostoma personatum	X	X	X
DIPTERA			
Dicranota sp.	X	X	X
Tipula sp.		X	
Culicoides sp.	X	X	
Chironomidae	X	X	X
Muscidae	X		
Simuliidae	X	X	X

5.4.1 Abundance

Figure 5.1 shows that initially, small substrate treatments supported greater abundances than the large substrates for equivalent exposure time, a pattern that was maintained for the remainder of the experiment.

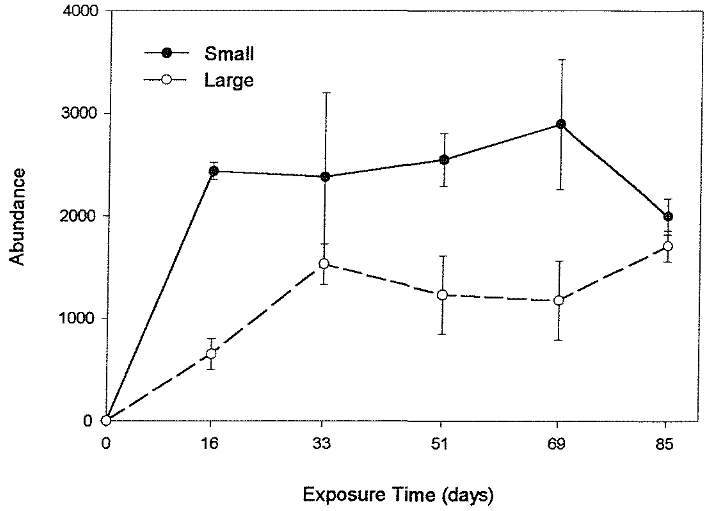


Figure 5.1. Mean total abundance of macroinvertebrates on small and large substrate treatments for each exposure time.
Vertical bars show 1 Standard Error.

Abundances on small substrate treatments remained relatively constant after 16 days, apart from a slight drop from 2893 to 1997 between 69 and 85 days. The larger substrates supported lower abundances throughout the length of the experiment, reaching comparable levels after an exposure of 85 days. After initial increases, abundances remained relatively constant throughout the experiment after 33 days. A two-way ANOVA revealed that both stone size (p-value <0.001) and exposure time (p-value <0.001) were significant, as was the interaction between substrate size and time (p-value 0.034) (Table 5.3). A post hoc Tukey test showed that time zero abundances significantly differed to the five exposure times, after which there were no significant differences. The significant interaction shows that the effect of time on colonising abundances was not the same on the large and small substrates. As Figure 5.1 shows, abundances increased in large substrates more slowly than in small substrates, reaching an equilibrium after 33 days compared to 16 days on the small substrates.

Table 5.3. Summary of two way ANOVAs with substrate, size and time as main factors. Significant results (p-value <0.05) are in bold.

Measure	Substrate size	Exposure time	Substrate*Time
Log Abundance	**<0.001**	**<0.001**	**0.034**
Taxon Richness	**<0.001**	0.251	**0.010**
Shannon diversity (H')	**<0.001**	**0.05**	0.062

5.4.2 Taxon Richness

The taxon richness of both substrate treatments over the 85 exposure days showed a pattern similar to that of abundances, with an initial increase between 0 and 16 exposure days, after which there was a leveling off of taxon richness (Figure 5.2). After a 16 day exposure large substrates supported higher taxon richness than small substrates (20 compared to 13 taxa). This difference was maintained at the 33 day exposure length (24 compared to 19 taxa), after which both substrate treatments supported similar a taxon richness (Figure 5.2).

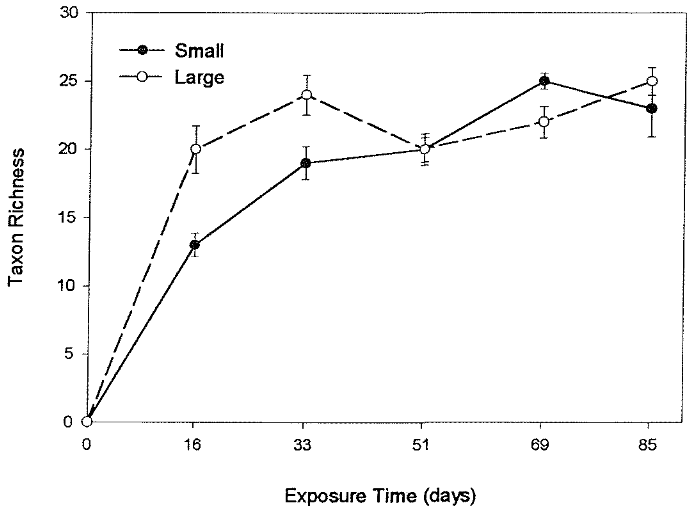


Figure 5.2. Mean taxon richness on small and large substrate treatments for each exposure length.
Vertical bars show 1 Standard Error.

A two way ANOVA established that there was no significant difference between the taxon richness of small and large substrates (p-value 0.251) (Table 5.2). However, both time (p-value <0.001) and the interaction between substrate size and time (p-value 0.010) were significant (Table 5.2). A Tukey post hoc test again revealed that taxon richness at time zero was significantly different to the other five exposure times. Coupled with this the taxon richness at 16 days was significantly different to that at 69 and 85 days, and the 51 day taxon richness was significantly different to 69 and 85 days. The significant interaction (p-value 0.010) shows that the effect of time on taxon richness was not consistent for small and large substrates. This was primarily thought to be due to the early time periods where large substrates had a greater taxon richness compared to small substrates.

5.4.3 Shannon diversity

Mean Shannon diversity (H') initially increased between 0 and the 33 days in both the large and small substrate treatments (Figure 5.3). After this point, there was a degree of levelling off of diversity in the large substrates, however, in the small substrates diversity continued to increase up until an exposure of 85 days.

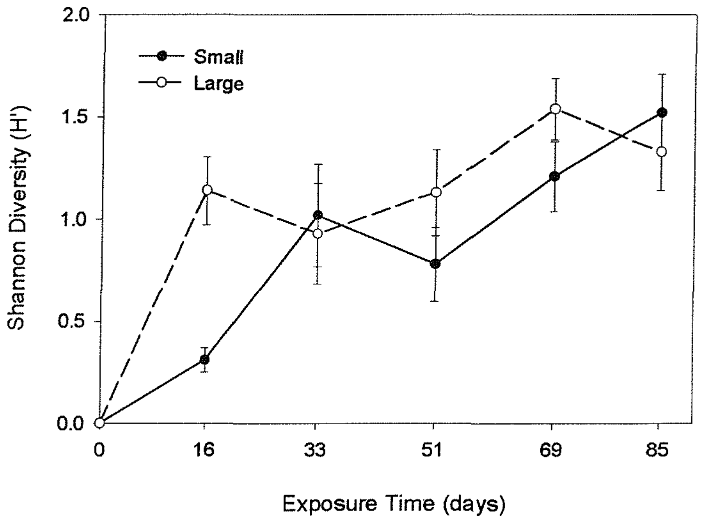


Figure 5.3. Mean Shannon diversity (H') on small and large substrate treatments for each exposure length.
Vertical bars show 1 Standard Error.

The two way ANOVA showed that both exposure time (p-value <0.001) and substrate size (p-value 0.05) were significant (Table 5.2). However, the interaction between time and size was not significant (p-value 0.062). The post hoc Tukey test revealed that Shannon diversity at time zero was significantly different to the other five times. Coupled with this, the diversity at 16 days was significantly different to 69 and 85 days, showing that diversity was still increasing towards the end of the experiment.

5.4.4 Temporal colonising responses and substrate selectivity of individual taxa

The analyses carried out indicated that there were distinct duration and substrate related patterns in the data, both in the raw variables (abundance and taxon richness) and derived (diversity) parameters. It was therefore decided that further analysis would be carried out to determine the temporal patterns and substrate selectivity of individual colonising taxa. Both the temporal and substrate response of the 15 most common taxa are shown in Figure 5.4.

Temporal responses

Three main temporal patterns were observed when individual taxa were examined. Firstly, some taxa showed high colonisation numbers after short exposure times that were maintained throughout the experiment (e.g. the mayfly *Baetis* spp., the crustacean Gammaridae, the true fly Chironomidae, the flatworms *D. lacteum* and *D. polychroa*, and the beetle *L. volckmari*; Figure 5.4 (a), (b), (c), (d), (g)). All of these taxa showed significant temporal patterns (Table 5.4) and Tukey tests revealed that their abundances at early exposure times (time zero and 16 days) were significantly different to the later exposure times (e.g. 33, 51, 69, and 85 days). After this there were no significant differences over the exposure times. This indicates that once initially colonised numbers of these taxa remained relatively constant over the experiment, suggesting they can be classed as 'quick' colonisers.

The second temporal pattern observed was that of taxa increasing in abundance over the entire experiment (e.g. the true fly *Dicranota* sp., Oligochaeta, and the cased caddisflies *G. pilosa*, *O. albicorne* and *S. personatum*; Figure 5.4 (d), (e), (f), (g), (h), (i), (j) and (l)). Again these taxa showed significant temporal patterns (Table 5.4), although slightly different to those previously described. For example, Tukey tests for these taxa revealed that their abundances at early and mid exposure times (e.g. time zero, 16, 33 and 51 days) were significantly different to later stages (e.g. 69 and 85 days). This indicates that abundances were still increasing towards to end of the experiment, suggesting that these taxa can be classed as 'slow' colonisers. A third, less frequent pattern, that was also observed was an initial increase in abundance followed by a subsequent decrease in numbers over time (e.g. the true fly Simuliidae, the stonefly *L. hippopus*, and the beetle larvae *G. natator*; Figure 5.4 (m), (n) and (o)). Again, these taxa showed significant temporal patterns (Table 5.4). Post hoc Tukey tests for these taxa revealed that their abundances were significantly different between time zero and 33 days (relating to the initial colonisation) followed by a significant decrease at 69 and 85 days. This pattern has previously been associated with 'resource' colonisers that adapt to changing conditions.

Table 5.4. Summary of two way ANOVAs for the 15 most common taxa with substrate size and time as main factors. Significant results (p-value <0.05) are in bold.

Taxa	Exposure time p-value	Substrate size p-value	Interaction p-value
Baetis spp.	**<0.001**	0.356	0.976
Gammaridae	**<0.001**	**0.001**	**0.033**
Chironomidae	**<0.001**	0.600	0.908
Dendrocoelum lacteum	**<0.001**	**<0.001**	**0.029**
Dicranota sp.	**0.019**	0.808	0.898
Dugesia polychroa	**<0.001**	**<0.001**	**0.006**
Limnius volckmari	**<0.001**	0.098	0.622
Oligochaeta	**<0.001**	**0.011**	**0.020**
Goera pilosa	**0.021**	0.085	0.925
Odontocerum albicorne	**<0.001**	**0.004**	0.467
Agapetus fuscipes	0.581	0.459	0.966
Sericostoma personatum	**<0.001**	**<0.001**	**0.012**
Simuliidae	**0.005**	0.559	0.793
Leuctra hippopus	**<0.001**	**0.050**	**0.007**
Gyrinus natator	**0.004**	0.112	0.240

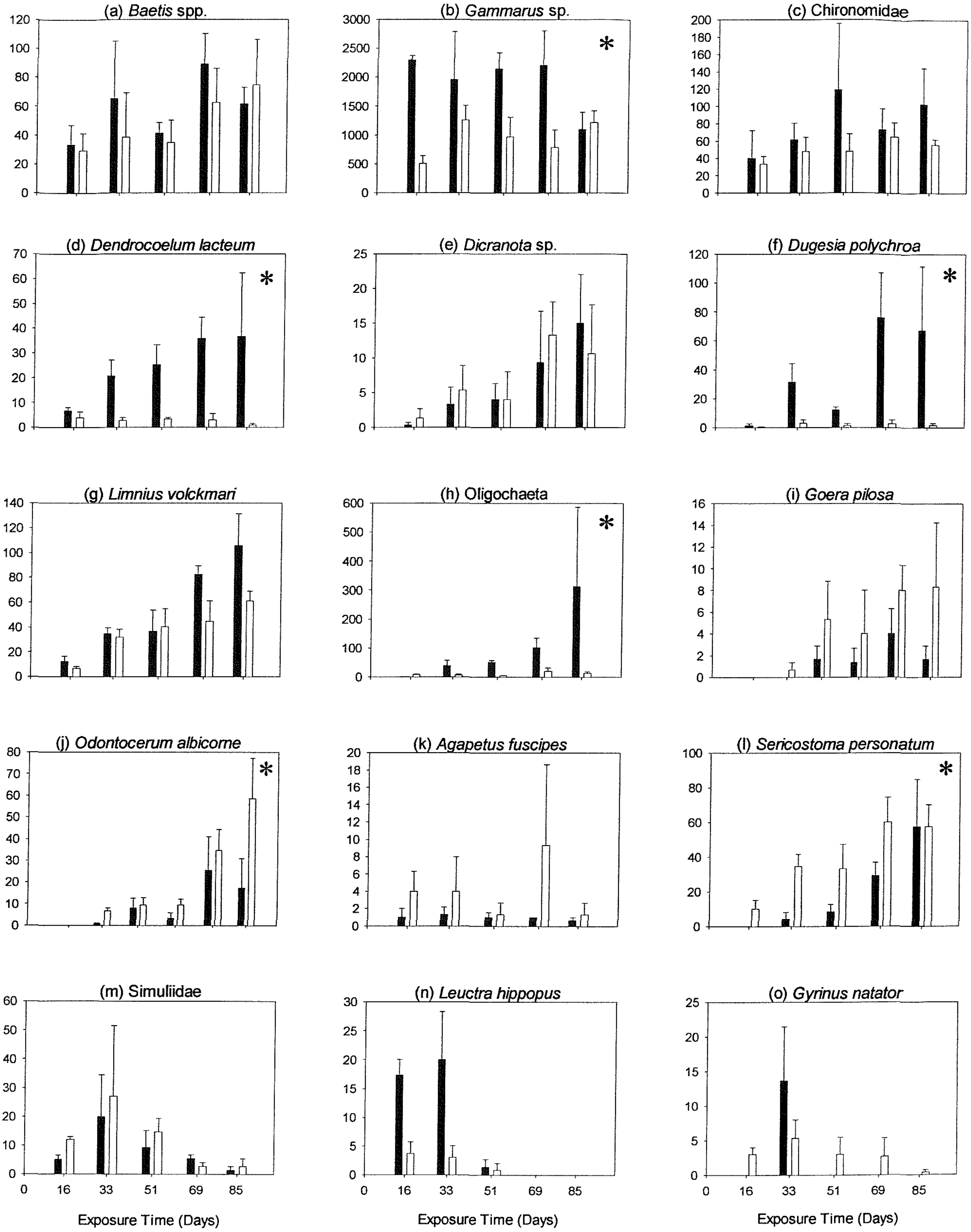


Figure 5.4. Mean abundances of the 15 most abundant taxa over the 85 days.
*** indicate taxa significantly different between small and large substrates.**
Black bars represent mean abundances of taxa in small substrate.
White bars represent mean abundances of taxa in large substrate
Vertical bars show 1 Standard Error.

Substrate selectivity

Coupled with these general temporal patterns, a degree of substrate selectivity was shown by individual taxa in Figure 5.4. For example, *D. lacteum* and *D. polychroa*, Gammaridae, *L. hippopus* and Oligochaeta were all found in significantly higher numbers in the small substrates (Figure 5.4 d), (f) and (h) and Table 5.4). In contrast, the cased caddisflies *G. pilosa*, *A. fuscipes*, *O. albicorne* and *S. personatum* were found in higher numbers in the large substrates (Figure 5.4 (i), (j), (k), (l)), significantly so for *O. albicorne* and *S. personatum* (Table 5.4).

Table 5.4 also shows that for certain taxa the interaction between exposure time and substrate size was also significant (e.g. Gammaridae, *D. lacteum*, *D. polychroa*, Oligochaeta, *S. personatum* and *L. hippopus*). This shows that the effect of time on the abundances of these taxa was not consistent on the two substrate sizes, suggesting that individual taxa exhibited different colonisation patterns over time on the small and large substrates. An example clearly shown by the flatworms *D. lacteum* and *D. polychroa*, which increased steadily throughout the experiment in small substrates, but remained relatively constant in the large substrates.

5.4.5 Community development

Principle Components Analysis results

Initial exploratory community analysis of species data was undertaken using Detrended Correspondence Analysis (DCA), an indirect unimodal ordination method. This option is used to establish how strongly unimodal data are by looking at the lengths of the resulting ordination axes, which further determines which multivariate tests are appropriate. In all cases, the lengths of the ordination axes in DCA were less than 3.0 SD, which justified the use of linear ordination methods (Principle Components Analysis and Redundancy Analysis) (ter Braak and Prentice 1988).

A summary of the results of the PCA performed on log transformed abundance data on the two substrate treatments over the five exposure times is given in Table 5.5, and Figure 5.5. The eigenvalue of the first Axis was 0.244. In a PCA, where eigenvalues for individual axes all lie between 0 and 1, values of greater than 0.5 are taken to indicate good separation of the taxa (ter Braak 1995, Cannan 1998).

Table 5.5. Ordination diagnostics from a PCA of abundance data from the two substrate treatments over the five exposure lengths.

	Axis 1	**Axis 2**	**Axis 3**	**Axis 4**
Eigenvalues	0.244	0.163	0.095	0.072
Cumulative % variance of data	24.4	40.7	50.3	57.5

The first four PCA axes explained 57.5 % of the variance found within the species data. Axes one and two explained the greatest amount of variance within the data and a plot was carried out on the sample scores on Axis 1 and Axis 2 (Figure 5.5). In Figure 5.5 the symbols of each sample indicate the duration that the replicates were left in the streambed.

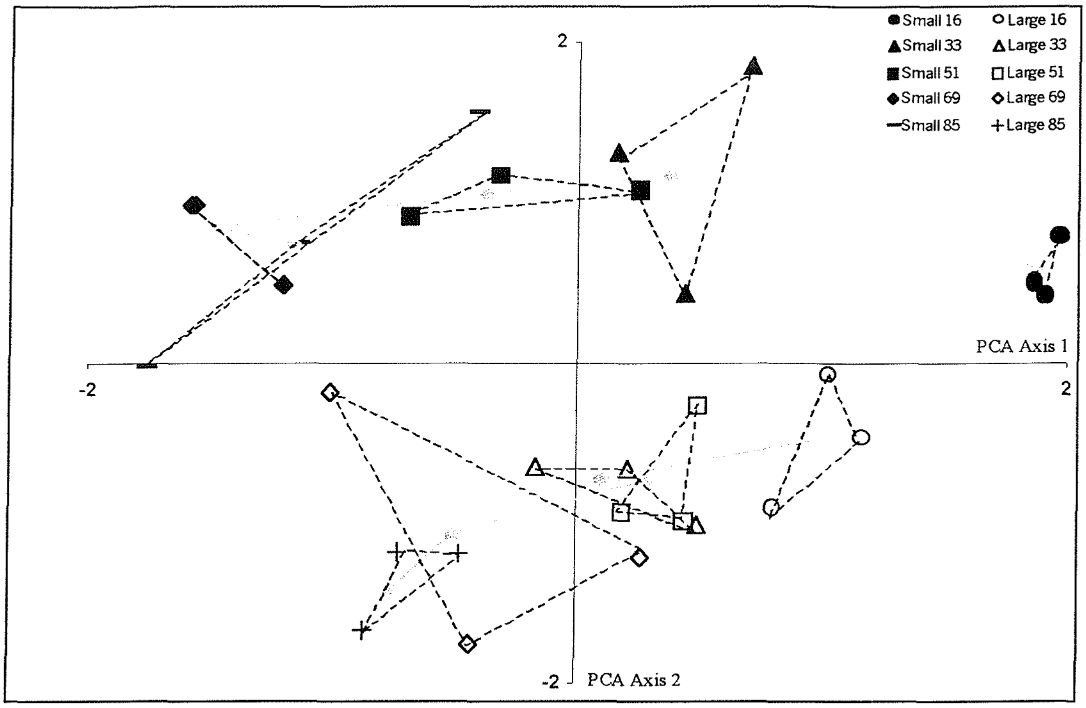


Figure 5.5. Sample scores along the first two axes of a Principal Components Analysis.
Samples are log transformed abundances for the two substrate treatments over the five exposure times. Grey line connects the centroids for each exposure time period.

Figure 5.5 displays a clear separation of the communities associated with the two different substrate types, with the small sample scores lying above Axis1 and the large sample scores located below Axis 1. However, within both of the substrates a clear temporal pattern can be seen with communities exposed for short time periods located on the right hand side of Axis 2 and the communities exposed for longer found on the left hand side of Axis 2, therefore indicating that length of exposure to recolonisation was extremely important in determining the structure of colonising communities. It is also evident that there was a greater spread, or temporal variability, in the structure of communities on small substrates compared to the large substrate community development, with sample scores extending along the entire length of Axis 1 (Figure 5.5).

By applying ordination analysis many response variables have been reduced to a few ordination axes which aim to encompass as much of the variation as possible (Cannan 1998). These axes can be interpreted with environmental variables to establish which factors may be related to the distribution of species and samples in the data (Cannan 1998). Correlation coefficients can be used as adequate summaries of scatter plots of environmental variables against ordination axes (ter Braak 1995, Cannan 1998). Rank correlation coefficients between the ordination scores for Axes 1 to 4 and the exposure time and substrate diameter are given in Table 5.6.

Table 5.6. Rank correlation coefficients between PCA sample scores for Axes 1 to 4 and environmental variables. Significance of coefficients is indicated: ***=P<0.001.

Variable	Axis 1	Axis 2	Axis 3	Axis 4
Exposure length (days)	**-0.842*****	-0.141	-0.002	0.088
Substrate diameter (cm^2)	-0.094	**0.879*****	-0.183	0.047

Table 5.6 details that on Axis 1, which represents a temporal factor in Figure 5.5, the highest correlation was with exposure length (days), reinforcing the temporal aspect of Axis 1. Substrate diameter is significantly associated with Axis 2, which indicates that the separation between small and large communities in Figure 5.5 is related to the physical properties (i.e. substrate size) of the recolonisation baskets. Exposure time and substrate diameter show no significant correlation with Axes 3 and 4.

Redundancy Analysis results

In order to quantify the contribution that environmental variables have in explaining the variation in the species composition, these can be incorporated directly into the ordination analysis using RDA techniques, (Cannan 1998). Three possible explanatory variables were included in the analysis, the summary of the RDA outcome is given in Table 5.7. Exposure time and substrate diameter were significant and were therefore included in the full analysis. The two significant variables explained 35 % of the faunal variation, with Axis 1 and 2 explaining 100 % of the total environment-species relation.

RDA restricts the axes to linear combinations of the environmental variables and when compared to the PCA, the RDA axes explain less variation, indicating that the environmental variables do not predict exactly the variation in the faunal composition extracted by the PCA (ter Braak 1995, Cannan 1998). However, the high biological-environment correlations mean that a large part of the remaining variation is predicted, and the Monte Carlo tests results suggest that this result is significant (Cannan 1998). Figure 5.6 displays the partial RDA plot for sample scores.

Table 5.7. Ordination diagnostics from a RDA of abundance data from the two substrate treatments over the five exposure lengths rivers sampled. (Significance set at *** p=0.005).

	Axis 1	Axis 2	Axis 3	Axis 4	
Eigenvalues	0.184	0.138	0.102	0.093	
Species-environment correlations	0.883	0.936			
Cumulative percentage variance:					
of species data	18.4	32.1	42.3	51.6	
of species-environment relation	57.1	100			
Sum of canonical eigenvalues					0.321
Sum of unconstrained eigenvalues					1.000
Monte-Carlo significance test results:	*Variable*				
	Exposure length (days)***				
	Substrate diameter***				

Figure 5.6 (constrained ordination) is extremely similar to Figure 5.5 (unconstrained ordination). Again the two substrate treatments separate out, and the temporal pattern is exactly the same which is why each point has not been designated an exposure length.

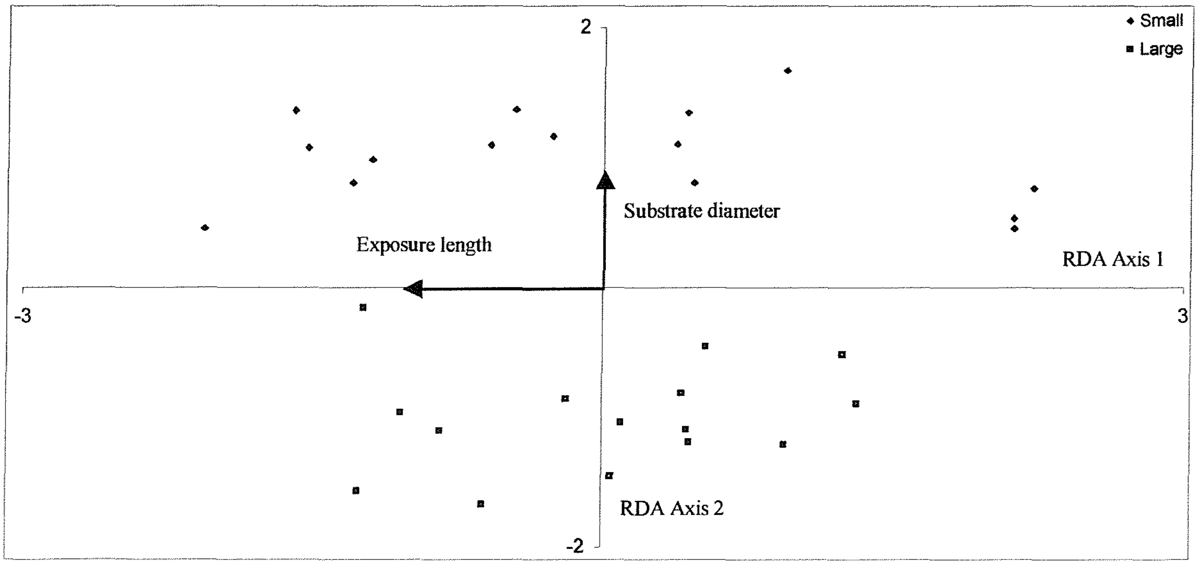


Figure 5.6. Sample scores along the first two axes of a Redundancy Analysis. Samples are log transformed abundances for the two substrate treatments over the five experimental time periods.

Coupled with the sample scores, the scores for the significant environmental variables are also plotted in Figure 5.6 (represented by the arrows). The plot confirms that Axis 1 represents a temporal separation of community development, and Axis 2 represents the influence of physical difference in the recolonisation baskets.

To investigate further this pattern, a species biplot with the significant environmental variables was constructed (Figure 5.7). The full arrows represent the significant environmental variables of substrate diameter and exposure period. The majority of taxa in Figure 5.7 were located on the left hand side of the diagram. Examples included the caseless caddisflies *H. siltalai* and *H. pellucidula*, the mayflies *H. sulphurea* and *Caenis* sp., the beetle *L. volckmari*, the leech *G. complanata* and the snail *P. vortex*. All of these taxa were located close to the arrow of exposure time indicating that they increased in abundances as the length of the experiment increased. It follows therefore that taxa close to the mirror image of the exposure arrow represent individuals that were present initially in high numbers before suffering reduced abundances. Examples included the fly larvae Simuliidae, the beetle *G. natator*, the stonefly *L. hippopus* and the mayfly *E. ignita*. Several taxa were also associated with the second significant environmental factor, substrate diameter. Examples included the flatworm *D. lacteum*, the crustacean Gammaridae, and the snail *L. peregra*, all of which were strongly associated with substrates of small diameters. In contrast, the cased caddisflies *P. latipennis* and *A. fuscipes* and the snails *B. leachi* and *P. planorbis* were all correlated with substrates with large diameter.

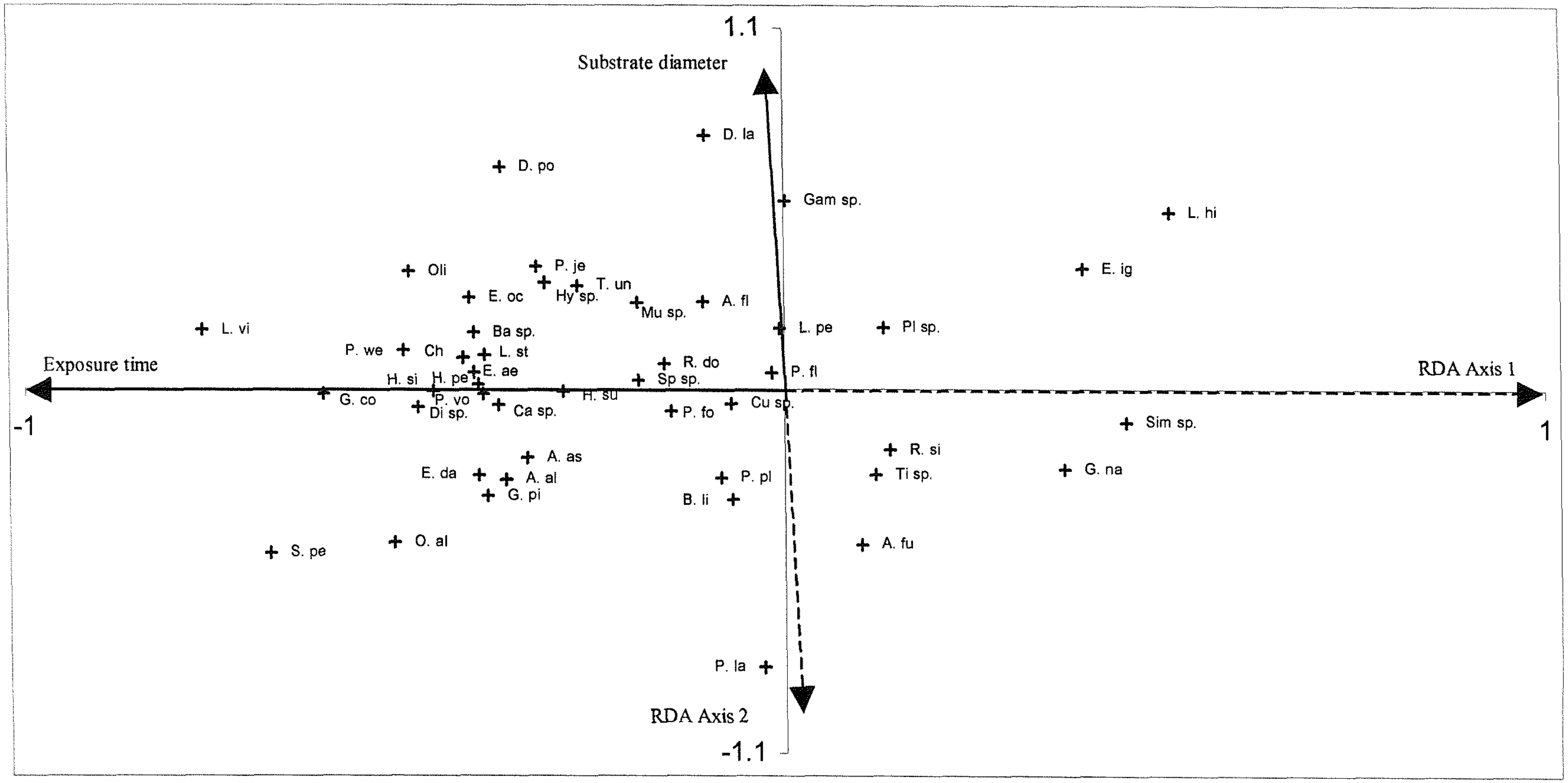


Figure 5.7. Biplot of species scores and significant instream environmental variables (RDA).
Species are log transformed abundances. Environmental variables: Exposure time and substrate diameter. Species associated with exposure time: H. si: *H. siltalai*, H. pe: *H. pellucidula*, H. su: *H. sulphurea*, Ca sp.: *Caenis* sp., L. vo.: *L. volckmari*, G. co: *G. complanata*, P. vo: *P. vortex*, Sim: Simuliidae, G. na: *G. natator*, L. hi: *L. hippopus*, E. ig: *E. ignita*. Species associated with substrate diameter: D. la: *D. lacteum*, Gam sp.: Gammaridae, L. pe: *L. peregra*, P. la: *P. latipennis*, A. fu: *A.* fuscipes, B. le: B. *leachi*, P. pl: *P. planorbis*

5.4.6 Trophic patterns

The colonisation patterns of many invertebrate taxa have been linked with the trophic grouping that the individuals belong to and the food availability within colonisation areas (Mackay 1992). It has been hypothesised that initial colonisers are often grazers, which are present in high numbers after relatively short periods of time due to their mobility and the periphyton availability. Later colonisers are often detritivores and omnivores, which exploit the accumulation of detritus in the substrate, and predators, which are attracted to the relatively high abundance of prey. Functional analyses were undertaken on the invertebrate taxa colonising the two substrate treatments, with individuals assigned the trophic classifications of grazer, grazer/detritivores, predator, filterer, or omnivore. These classifications were felt to encompass the main trophic pathways utilised by chalk river invertebrates (Mihuc 1997). The abundances of taxa classified to the trophic groupings were calculated and plotted for small (Figure 5.8) and large (Figure 5.9) substrates, and a fully factorial three way ANOVA was carried out with time, substrate and trophic group as main factors (Table 5.8).

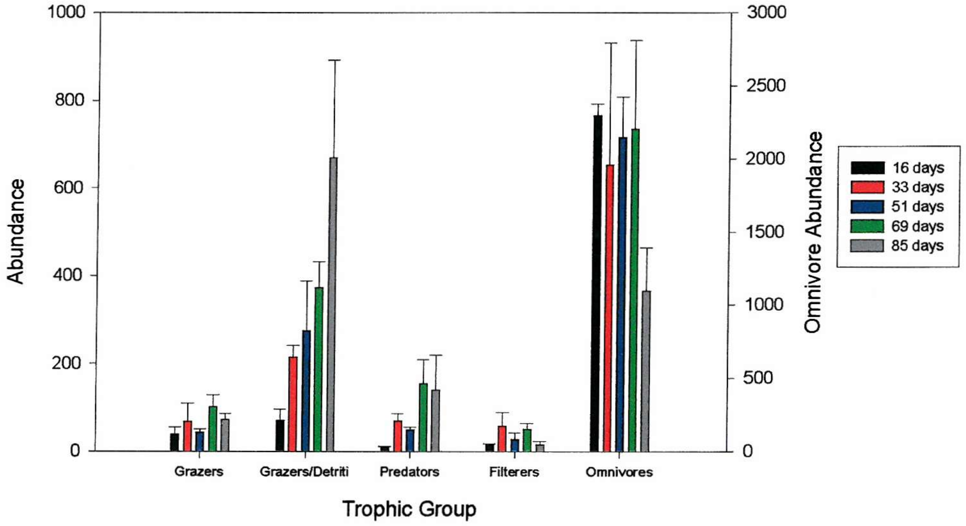


Figure 5.8. Shift in trophic groups colonising small substrate treatments. Grazers, grazer/detritivore, predator and filterer abundance are on the left x axis and omnivore abundance is on the right x axis.

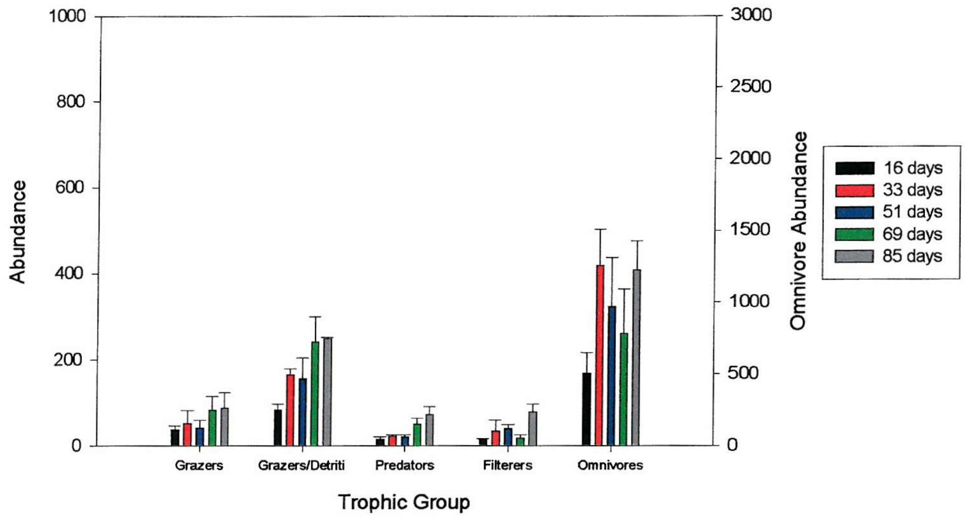


Figure 5.9. Shift in trophic groupings colonising large substrate treatments. Grazers, grazer/detritivore, predator and filterer abundance are on the left x axis and omnivore abundance is on the right x axis.

Table 5.8. Summary of three way ANOVAs with time, substrate and trophic group as main factors.

Measure	p-value
Time	**0.007**
Substrate	**<0.001**
Trophic Group	**<0.001**
Size*Time	0.092
Size*Trophic Group	**0.050**
Time*Trophic Group	**<0.001**
Size*Time*Trophic Group	0.331

Some clear temporal trophic patterns can be seen in the small substrates (Figure 5.8). For example, there was a slight decrease in the abundance of filterers. In contrast, the numbers of predators and grazer/detritivores increased as exposure time lengthened. There was little overall trophic pattern for grazers and omnivores. Trophic succession was also seen in the large substrate treatments (Figure 5.9), with the majority of trophic groups increasing over time. There was a significant difference between size, time and trophic grouping (Table 5.8). There were significant interactions between time and trophic group (indicating that the trophic groups showed different colonisation patterns) and size and trophic (suggesting that the trophic groups colonised differently on large and small substrates, a pattern seen in Figures 5.8 and 5.9).

However, a point that should be addressed at this point was the observation of the Bullhead (*Cottus Gobio*) solely present within the large substrate treatments over the entire experimental exposure length. The predatory nature of this fish species and the influence it may be exerting upon the community/trophic composition of the macroinvertebrates colonising the large substrate treatments will be dealt with within the discussion section of this chapter.

5.4.7 Community development in comparison to background communities

Multivariate analyses were carried out on the taxa colonising the small and large substrate treatments and also on benthic samples taken from the streambed at the time of basket placement and every removal phase. It was decided that multivariate analyses would be carried out on presence/absence data for the taxa present rather than abundances; this removed any bias in terms of extremely common taxa (e.g. Gammaridae). Initial exploratory multivariate analyses (DCA) indicated that linear analytical methods were most applicable to the data set. Thus a PCA was carried out and a summary of the results of the PCA performed is given in Figure 5.10 and Table 5.8.

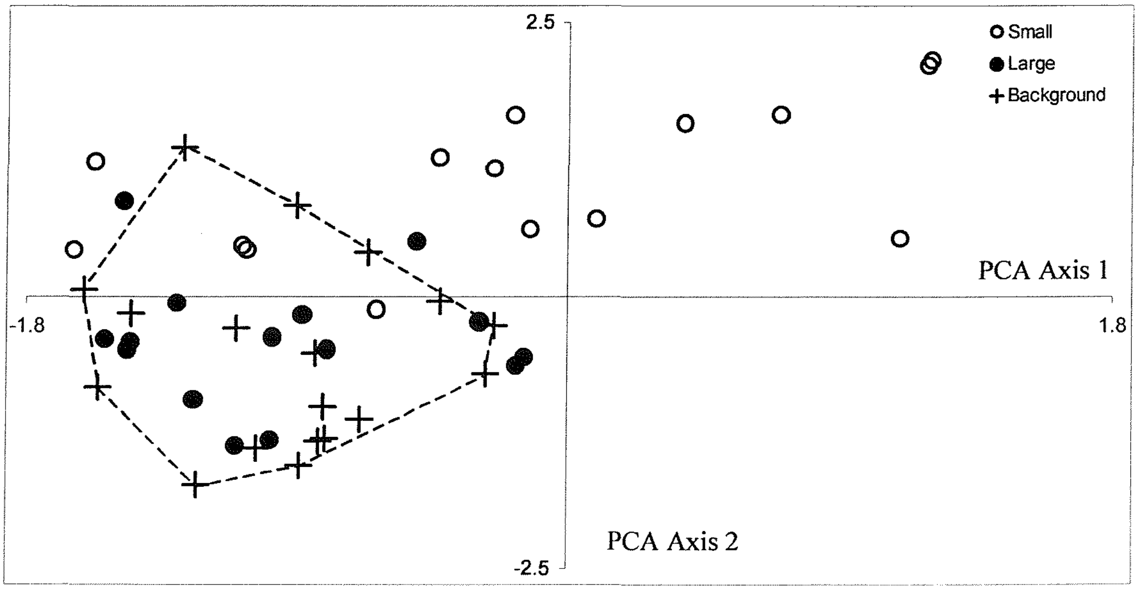


Figure 5.10. PCA plot of sample scores for small, large substrates and benthic background samples over the experimental time period.

Table 5.8. Ordination diagnostics from a PCA of presence absence data from the two substrate treatments and background samples over the exposure length of the experiment.

	Axis 1	**Axis 2**	**Axis 3**	**Axis 4**
Eigenvalues	0.162	0.091	0.082	0.075
Cumulative % variance of data	16.2	25.2	33.4	40.9

The eigenvalue of the first Axis was 0.162. The PCA explained a total of 40.9 % of the total variance found within the species data, with axes 1 and 2 explaining the majority of variance within the data. A plot was carried out on the sample scores of Axis 1 and Axis 2 (Figure 5.10).

The dashed line in Figure 5.10 encompasses the spread of background samples taken over the duration of the experiment. The temporal development of communities previously highlighted (Figure 5.5) is still evident in this figure, with early communities of small substrate treatments located on the right hand side of Figure 5.10, and the samples exposed for greatest lengths located close to the background samples. Also evident from Figure 5.10 is the similarity of communities associated with large substrates with background communities over the entire experimental time period. Although a temporal development was still evident in the communities on large stones, initial colonising communities were extremely similar to background communities. This therefore suggests that there was much less temporal development/variation in terms of community structure in the large substrate treatments compared to the small substrate treatments.

5.5 DISCUSSION

Disturbance events in aquatic environments vary among systems but are important factors in the evolution strategies of the resident biota (Gurtz and Wallace 1984). Mayr (1964) defined colonisation as the 'invasion of species into disturbed and newly created habitats'. Recolonisation studies on running water benthos have shown that animals may quickly re-appear in disturbed areas (Muller 1954, Waters 1964). However, former diversity and background abundances may not be re-established for some time (Cairns *et al.* 1971). The processes of recolonisation and subsequent changes in dominance of taxa (succession) are common in streams subject to natural and anthropogenic disturbances (Sousa 1984, Peckarsky 1986, Brooks and Boulton 1991, Malmqvist *et al.* 1991, Whiles and Wallace 1992, Tikkanen *et al.* 1994). However, it has been recognised that stream ecologists know little about the process of site specific temporal succession by macrobenthos (Fisher 1983, Peckarsky 1986).

5.5.1 Succession

Population changes

Analyses carried out on the colonising patterns on small and large substrates over time indicated that there were distinct temporal patterns at a taxon level. The classic MacArthur-Wilson method of examining temporal colonisation events focuses on the examination of temporal change at a community level. An alternative is to follow changes in numbers of individual taxa (Reice 1980, Gurtz and Wallace 1984, Peckarsky 1986, Mackay 1992). However, this method shares a problem with the 'classical' approach in that it is difficult to differentiate between colonists truly exploiting the resources of the patch and individuals that are merely exploring the potentials and moving on. For example, it has been suggested that the often recorded early rise in numbers of *Baetis* spp. probably includes much coming and going while larvae test periphyton resources before eventually staying longer to feed (Ulfstrand *et al.* 1974, Allan 1975, Shaw and Minshall 1980, Fisher *et al.* 1982, McAuliffe 1983, Minshall, Andrews and Manuel-Faler 1983, Ciborowski and Clifford 1984, Lake and Doeg 1985, Dudley 1988). Other early colonisers are thought to include Simuliidae larvae (Ulfstrand *et al.* 1974, Lake and Doeg 1985). However, these taxa are often thought to reflect true colonisation of bare sites for filter feeding. A later decline in Simuliidae numbers has been explained as part of the colonisation sequence as the larvae move to avoid the increasing densities of algae and other filter feeders (Lake and Doeg 1985, Hemphill 1988, Downes and Lake 1991). Therefore, the replacement of pioneer species by superior competitors that are slower colonisers has been widely reported (Hemphill and Cooper 1983, Malmqvist *et al.* 1991).

Three general colonisation patterns were highlighted within the current study (e.g. Figure 5.4). Some taxa showed high colonisation numbers after a relatively short temporal period, e.g. *Baetis* spp., Gammaridae, *D. lacteum*, *D. polychroa* and *L. volckmari*, suggesting that these taxa were quick colonisers. In contrast, certain taxa were classified as slower colonisers, e.g. *Dicranota* sp., Oligochaeta, *O. albicorne*, *S. personatum* and *G. pilosa*, whose numbers were still increasing after approximately 85 days. A third colonisation pattern observed was that certain taxa which were found in higher numbers after relatively short exposure times decreased significantly as time progressed (e.g. Simuliidae, *L. hippopus* and *G. natator*). This therefore indicates that, although these results do not fully resolve the uncertainties associated with defining true colonists versus exploration, several temporal recolonisation patterns can be expected to occur within locally disturbed chalk streams.

Community assemblage

The sequence of events during colonisation of bare stream patches has been described in several ways. Classical approaches follow the MacArthur-Wilson island biogeographic model in which community structure progresses towards an equilibrium (MacArthur and Wilson 1967). It has been argued that stream communities are unlikely to exist in equilibrium (Lake and Barmuta 1986), although Minshall and Petersen (1985) suggested that equilibrial conditions could occur under stable flow regimes. Equilibrium is reached when the total number of species (or taxa due to the limitations of identification) becomes more or less constant (Mackay 1992). Alternative end points in this scheme are the stabilisation of either total abundances or species diversity per patch.

When equilibrium is equated with a plateau in the number of taxa, the recolonisation period has been as short as 4 to 6 days in some experiments (Townsend and Hildrew 1976, Lake and Doeg 1985), but more usually is in the range of 10 to 25 days (Wise and Molles 1979, Lake and Doeg 1985, Minshall and Petersen 1985, Peckarsky 1986). When invertebrate abundances are considered in these experiments equilibrium has been reached after 10 to 30 (Mackay 1992). However, much longer time periods have also been associated with recolonisation of invertebrates. For example, Williams and Hynes (1977) followed the establishment of the benthic macroinvertebrate community after a disturbance event and concluded that the abundances of recolonising taxa reached an equilibrium after approximately 100 days.

Analyses showed a clear temporal community development within the substrate treatments (Figures 5.1 and 5.2). Both abundances and taxon richness levelled off after 16 to 33 days. Therefore, when compared to previous studies, taxon richness appears to take slightly longer to reach equilibrium than the 10 to 25 days suggested by Lake and Doeg (1985) and Minshall and Petersen (1985). In contrast, the time taken for abundances to reach equilibrium are more closely comparable to the end point of the 10 to 30 days suggested by Mackay (1992). The relatively longer time period required for the taxon richness to reach equilibrium is perhaps not unexpected. The previous chapter highlighted the extremely diverse nature of undisturbed chalk stream macroinvertebrate communities (Berrie 1992, Wright 1992, Harrison 2000). It may therefore be expected that these systems take a relatively long time to reach stability in terms of the number of taxa colonising.

A slightly different temporal pattern was seen when the diversity of the recolonisation baskets were examined, with an equilibrium difficult to identify (Figure 5.3). It could be argued that the diversity of small substrates was beginning to level off towards the end of the experimental time period. However, the diversity within large substrates was still increasing in the baskets exposed for 85 days. This could again be attributed to the extremely diverse nature of the macroinvertebrate communities found within chalk streams (Mainstone 1999).

Many factors can determine the diversity of colonising invertebrates over time (McCabe and Gotelli 2000). Biotic interactions such as predator/prey interaction (Peckarsky 1985) and competition for available resources, e.g. space and food (Hart 1986, Hemphill 1988, Matczak and Mackay 1990) combine to determine the specific community composition over time. Hence, the extremely dynamic nature of aquatic systems, especially chalk rivers (Berrie 1992), could be impacting upon diversity values (Malmqvist *et al.* 1991). For example, the accumulation of detritus over time will present an additional food source for certain invertebrate taxa, potentially attracting additional organisms (Gurtz and Wallace 1984) and increases in trapped fine sediment can offer new habitats for certain taxa (Peckarsky 1986). Therefore the ongoing interaction of abiotic and biotic factors could potentially be playing an important role in temporal community diversity patterns (Malmqvist *et al.* 1991).

Therefore, generally the temporal community colonisation patterns of abundances and taxon richness reached equilibrium, a similar pattern observed by previous researchers (Allan 1975, Wise and Molles 1979, Shaw and Minshall 1980, Fisher *et al.* 1982, Minshall, Andrews and Manuel-Faler 1983, Ciborowski and Clifford 1984, Peckarsky 1986, Mackay 1992), although the exact pattern differed between the substrate treatments. In contrast, no clear equilibrium was identified in the diversity of the small and, particularly, large substrate treatments. Therefore, it can be concluded that substrate size significantly influences temporal succession at both a population and community level.

5.5.2 Substrate preferences

Historically, substratum has been considered to be one of the most important factors influencing the distribution of in-stream macroinvertebrates (Bell 1969, Wise and Molles 1979). This view has developed from early observations that certain benthic species are restricted to particular types of substrate and that different types of substrate support assemblages of invertebrates that differ in regard to total numbers and/or numbers of taxa (Hynes 1970). Substrate selection by stream invertebrates has been identified as an important mechanism determining invertebrate abundances, species composition, and species richness (Egglishaw

1969, Hart 1978, Reice 1980, Minshall 1984, Magoulick 1998). Stones in experimental substrate trays do not offer the same range in stone size as natural substrate patches, but do mimic physically disturbed patches in that they are loose, with interstitial spaces at first free of silt and detritus. It is generally accepted that lotic organisms display substratum preferences in the laboratory (Scott 1958, Cummins and Lauff 1969, Higler 1975, Walton 1978) and in the field (Percival and Whitehead 1929, Linduska 1942, Ulfstrand 1968, Reice 1974, de March 1976, Reice 1977).

In an experimental manner similar to the pioneering substratum manipulation carried out by Wene and Wickliff (1940) patches of two substratum sizes were established during this investigation. These patches were set up in a single riffle to minimise the current velocity differences among patches, allowing any differences in population distributions and community structure to be ascribed to substratum differences alone, the null hypothesis being that the distribution of colonists among the two substrates would be the same. Alternatively a hypothesis was also formulated that organisms would select substrata on the basis of their characteristics and produce distribution patterns accordingly. This research clearly showed that small and large substrate treatments supported different communities, which were temporally colonised in different manners (Figure 5.4). Therefore these results suggest that substrate size is an extremely important factor in determining the temporal colonising dynamics of invertebrates.

Previous recolonisation patterns have suggested that greater numbers of individuals colonise larger substrates (Tarzwell 1936, Moon 1939, Pennak and Van Gerpen 1947, Ward 1975 and Wise and Molles 1979). These results clearly show that this was not the case, and only after 85 exposure days did the substrate treatments have similar abundances (Figure 5.1). However, the current results are consistent with those of Minshall and Minshall (1977), Williams and Mundie (1978) and Shaw and Minshall (1980). Minshall and Minshall (1977) attributed the higher numbers on smaller substrates to the greater surface area available within these treatments. Small substrate treatments had a surface area 4.3 times greater than that associated with the large substrate treatments (53, 040 cm^2 compared to 12, 353 cm^2 respectively, Table 5.1). This difference in surface area could be a potential factor in explaining the apparent discrepancy with earlier research.

When individual taxa were assessed the results clearly demonstrate that certain taxa exhibited substrate preference. For example, the flatworms *D. lacteum* and *D. polychroa* were found in significantly higher numbers on the small substrate treatments and the cased caddisflies *S. personatum* and *O. albicorne* had significantly higher numbers in the large substrates (Figures 5.4). This finding is inconsistent with Sheldon (1977) who compared numbers of stream insects colonising trays of large and small stones and found no significant effect of substrate size for any individual taxon. Minshall and Minshall (1977) also failed to show significant differences between substrate sizes when taxa were analysed individually. However, the failure to demonstrate significant effects probably resulted from the higher variance in numbers for individual taxa than for numbers of stream insects (Hellawell 1977). In contrast, both Gurtz and Wallace (1984) and Malmqvist *et al.* (1991) clearly demonstrated the importance of substrate size in the colonisation preference of individual taxa and attributed this finding to the physical characteristics of the substrate available to colonisation.

Minshall (1984) listed several substrate parameters that appeared to be important in determining substrate preference in taxa. These included type of substrate, particle size, heterogeneity, texture, pore space, and silt/detritus accumulation. Cummins and Lauff (1969), Reice (1980) and Erman and Erman (1984) studied a selection of these factors and found them to influence taxon colonisation. A number of these factors could be playing an important role in the results presented. For example, the greater surface area of the smaller substrate treatments could be an important aspect for flatworm species that exhibit an extremely close relationship with the surface of the substrate they colonise (Fritter and Mannuel 1995). Another example of the substrate properties that could be exerting an influence on colonisation of individual taxa includes interstitial pore size. The interstitial volume within large substrate treatments was 10 % greater than that in the small substrate treatments. When the individual taxa were examined, the taxa with significantly higher numbers in large substrates were cased caddisflies. Large substrates tend to be more stable than smaller substrates (Taylor and Littler 1982, Zaret 1982, Gurtz and Wallace 1984), and, once the surface is suitably conditioned, are more likely attract clinging, more sedentary organisms, such as cased caddisflies (Khalaf and Tachet 1980, Malmqvist and Otto 1987). The greater interstitial space available could also be beneficial in terms of case development for these species (Mackay 1992).

The presence of *Cottus gobio*, the Bullhead, was observed solely within the large substrate treatments. Although not explicitly studied, the presence of this fish species could potentially be attributed to the usability of the interstitial spaces available within the large substrate treatments. Kohler and McPeek (1989) investigated the prey preference of *Cottus* sp., using the mayfly *Baetis* sp. and caddisfly *Agapetus* sp. Results indicated that *Cottus* sp. favoured mobile prey rather than more sessile organisms as they rely on visual detection of prey when feeding. Cased caddisflies were found to be less vulnerable to predation than *Baetis* sp. and Gammaridae as the stone case deterred predation and the movement of the cased caddis was less (Kohler and McPeek 1989). Therefore, a combination of physical and biological characteristics of the substrate treatments could potentially explain the significantly higher numbers of certain taxa associated with the small/large substrate treatments. In fact, it is a suggestion of this research that the predatory presence of *C. gobio* could be a factor in the overall lower abundances of taxa within the large substrate treatments (Figure 5.1). Therefore the results of this study lend support to the theory that specific physical characteristics and related biological aspects of the substrate can significantly influence the colonisation patterns of macroinvertebrate communities within a chalk river.

5.5.3 Trophic succession

An additional method utilised in the analysis of recolonisation data is the classification of individual taxa to trophic feeding groups. As previously identified Baetidae (mostly *Baetis* spp.) are abundant and early colonisers in most parts of the world (Ulfstrand *et al.* 1974, Hynes J.D. 1975, Gore 1979, Gray and Fisher 1981, Benke and Jacobi 1986, Parker 1989, Lamberti *et al.* 1991). *Baetis* spp. are often classified as grazers, removing thin organic films which develop quickly on the bare substrate. The early arrival of these taxa is consistent with their ability to exploit the earliest food materials on and among bare substrates. Fisher *et al.*

(1982) found that grazers composed 87 % of numbers in the invertebrate community of a desert stream disturbed by flash floods. This initial increase in grazers, especially mayfly genera, is a nearly universal response in streams following disturbances (Hawkins *et al.* 1982). These generalist feeders typically have short generation times and high fecundity (*r*-strategists) and hence, can rapidly increase in numbers following environmental changes that favour their food resources, temperature requirements, etc. Elminthidae and Chironomidae are also consistently recorded among the first colonisers (Gurtz and Wallace 1984, McElravy *et al.* 1989).

If colonists cannot graze, an alternative early feeding strategy is filtering, making use of suspended fine detritus (Mackay 1992). Simuliidae and Hydropsychidae larvae are also often reported as early colonists. However, because the Hydropsychids can only establish on rough surfaced rocks they may not be the earliest filterers. Nevertheless, they have some of the characteristics of opportunists in that the larvae in this family tend to have unspecialised nets and a broad diet (Fuller and Mackay 1980), and they can feed on drifting animals as well as detritus (Lamberti and Moore 1984, Diamond 1986).

Later in the recovery of streams, herbivorous taxa contribute more to colonisation. For example, detritivore dominated communities have been reported as being rather slow to recover after disturbance (Gray and Fisher 1981, Wallace 1990). The relatively few numbers of detritivores among the early colonisers of introduced substrates has been attributed to the slow rate of coarse particulate organic matter accumulation (Peckarsky 1980). Ulfstrand *et al.* (1974) reported that when coarse particulate organic matter did collect on the substrate trays in a suitable stream, many shredders were among the colonisers.

This general pattern of trophic succession has been previously reported within recolonisation experiments, for example, Malmqvist *et al.* (1991) correlated the gradual colonisation by grazers, e.g. *Baetis* spp., with the gradual establishment of benthic algae, while predators, e.g. *O. albicorne* and *R. dorsalis* and collectors/gathers, e.g. Chironomidae, arrived later with the accumulation of prey organisms and fine organic material respectively. The relative proportion of trophic groups shifted over time within both of the substrate treatments. For example, the mean abundance of both predators and grazer/detritivores increased with time in the small substrate treatments. In contrast the abundance of filterers and grazers showed little temporal pattern (Figure 5.8). A similar pattern was observed within the large substrates (Figure 5.9), with grazers and filterers showing little temporal pattern. One potential explanation for the relative stability in grazer numbers in the large substrates, in contrast to the general decline previously reported in literature, was the predatory presence of *C. gobio* which have been established to select mobile prey such as grazers and could therefore be regulating the numbers of these taxa. However, this does not explain the little temporal succession of grazers in small substrates, which suggests that their functional resources did not become limiting during the experiment.

As the numbers of grazers/detritivores, filterers and grazers grow, invertebrates begin to interact with each other. For example, intra- and interspecific competition for space has been shown to determine the relative abundances of colonising filterers (Hart 1986, Hemphill 1988, Matczak and Mackay 1990) and grazers (McAuliffe 1984, Hart 1985). Besides causing emigration from the patch under consideration, competition may be occurring upstream, resulting in drift, and hence affecting the rate at which the patch receives immigrants (Mackay 1992). It has also been suggested that early colonisers may enjoy an initial period of enemy-free space because dominant predators tend to be later immigrants (e.g. Pianka 1970, Ulfstrand *et al.* 1974, Allan 1975, Meier *et al.* 1979, Shaw and Minshall 1980, Peckarsky 1986, Boothroyd and Dickie 1989). Therefore, there are many internal biotic factors that could be contributing to the trophic colonisation pattern observed. Also, recolonisation by invertebrate taxa would appear to be a combination of both mobility of individual taxa and the trophic pathways they exploit.

5.5.4 Management implications

Much debate exists over the similarity of recolonisation community composition compared to background/control communities. In theory, given adequate time, the composition of recolonisation communities should approach that of the background, given that the undisturbed streambed provides the recolonising pool of invertebrates (Williams and Hynes 1976, Malmqvist *et al.* 1991, Mackay 1992, Tikkanen *et al.* 1994). This aspect of invertebrate recolonisation has potential management implications for chalk rivers that experience a variety of instream disturbances. Therefore an extremely important aspect to this experiment was to establish how long it takes for recolonisation and existing communities to show similar compositions, abundances and diversities.

The multivariate analysis provided an interesting insight to similarities between colonising and existing communities. For example, it is evident that communities associated with large substrates were extremely similar to background communities in terms of composition over the entire exposure period (Figure 5.10). Communities of the small substrates did reach similar composition levels to background samples, but only after a longer period. These findings have potential links with disturbance due to the introduction of artificial substrates into chalk rivers in order to restore instream habitat and hydraulic heterogeneity (§ 5.2.3). Currently little regard is given to the choice of substrate used in restoration schemes, however the results presented within this chapter suggest that the choice of substrate can be pivotal in determining the equilibrium macroinvertebrate communities observed. For example, in terms of similarity to background communities large substrates would appear to be preferable, with a comparable community composition observed after relatively short exposure times compared to small substrates. However, if consideration is given to the relative abundances that substrates supported (an important ecological aspect, especially for fish populations), then small substrata would be preferred to larger ones. Therefore the specific physical and biological characteristics associated with substrates of different sizes need to be considered prior to their introduction to riverine systems.

5.6 CONCLUSIONS

A clear temporal pattern was seen in the abundance and taxon richness of recolonising macroinvertebrate communities on both small and large substrate treatments. Both the abundances and taxon richness reached an asymptote after approximately 16 to 33 days, a temporal equilibrium that compares favourably with many previous studies. In contrast however, the diversity of colonising communities on both substrate treatments appeared to take longer than the experiment time period to reach equilibrium. This is perhaps not unexpected due to the extremely diverse nature of chalk stream macroinvertebrate communities and the ongoing shift of internal conditions within the recolonisation baskets. It also highlights the potentially weak ability of chalk stream invertebrates to respond rapidly to physical disturbance, a reflection of the physically stable nature of chalk streams.

With regard to individual taxa, three general colonisation patterns were observed. The first pattern, common to the majority of taxa, was one of increasing abundances over exposure time ('slow' colonisers). The second colonisation pattern was that of initially high numbers that were maintained over the whole experiment ('quick' colonisers). The third colonisation pattern was that of initially high numbers followed by a dropping off of abundances, potentially due to a shift in resources available for specific taxa ('resource' colonisers).

The influence of substrate size was also identified as a highly important factor in determining colonising communities. Throughout the experiment, large substrate treatments supported lower numbers compared to small substrates. This has potentially been linked with the predatory presence of *C. gobio* suppressing colonising numbers. When individual taxa were investigated, the influence of substrate size was again important, with certain properties such as surface area and interstitial pore size highlighted as potentially explaining the association of certain taxa. However, the significant interaction exhibited between exposure time and substrate size means that neither of these factors can be considered in isolation when examining the recolonisation dynamics of chalk stream invertebrates.

Trophic succession of communities again compared closely to previous recolonisation studies, with patterns attributed to a combination of the mobility of individual taxa and exploitation of resources available (Grimm and Fisher 1989, Boulton *et al.* 1992). The biological interaction of competition and predation within invertebrate and higher trophic groups was also highlighted.

Substrate size was observed to influence the temporal colonisation towards similarity with background communities in terms of composition and abundance in the small substrates. Large substrate communities were extremely similar to background samples over the entire experimental period, however, smaller substrates consistently supported greater numbers. This finding has potential implications for the substrate used in river restoration techniques, where substrates are added to recreate habitat and flow heterogeneity (Environment Agency 2000c).

Recolonisation studies to date have established recurring ecological patterns (Mackay 1992). The current results presented within this chapter agree with many previous colonisation studies, especially the trophic succession of communities and the physical influence of substrate characteristics in determining community composition. Depending on hydrologic regime and season, predictions can be made as to the likely responses of macroinvertebrate communities to disturbance, according to their ecological traits and dynamics. This relative predictability of invertebrate recolonisation has not been previously considered in chalk rivers and therefore provides an ideal opportunity to assess the ability of a stream to cope with disturbance. This information could be invaluable to managers debating the necessity of protecting/restoring a stream. For although our ecological knowledge at the species level is still incomplete, some recognition of typical faunal patterns can provide the constructive advice needed in environmental decision-making (Mackay 1992).

CHAPTER SIX: GENERAL DISCUSSION

CHAPTER SIX: GENERAL DISCUSSION

6.1 INTRODUCTION

6.1.1 Chalk rivers revisited

Great Britain has many geological formations, and a range of temperate climates and approaches to land management. These geographical differences give rise to a wide variety of rivers and streams (Lewin 1981), ranging from the fast, rocky, clean and oligotrophic salmon rivers of the north and west, to the slow, muddy, eutrophic and often polluted coarse fish rivers in the south and east (Ladle and Westlake 1995). Chalk rivers are generally classified as lowland rivers. However, lowland rivers are often highly stressed systems due to changes in land cover and use; this results in increased flashiness and a largely unrepresentative river channel structure (Harper and Everard 1998). Therefore, it is evident that the physical and biological characteristics of chalk rivers set them apart from the more recognisable 'lowland' systems.

Chalk rivers are found only where chalk outcrops on the surface of the earth (Berrie 1992), hence they are a special type of calcareous stream with quite distinctive features. Many of their characteristics can be attributed to the physical and chemical properties of the land over which they flow and from which their waters emerge. For example, the hydrological regime of a chalk river is a direct result of the groundwater nature of these systems. Passage of water through the aquifers is relatively slow and hence smooths out irregularities in rainfall. Consequently, chalk rivers tend to have a regular annual flow with a relatively small difference between winter and summer, and no spate conditions. The temperature range in chalk rivers is relatively small compared to other lowland rivers which receive greater quantities of surface runoff. The water is calcareous, with naturally high quantities of inorganic nutrients present. The undisturbed bed of chalk streams is a pavement of gravel cemented by calcium carbonate with overlying silt and detritus at the edges and under weed beds. In contrast, more typical UK lowland rivers flow over clay, cobbles, or an anaerobic organic ooze (Ladle and Westlake 1995).

British chalk streams and lowland rivers flow through agricultural catchments and urban areas that have been intensively used by man since at least Roman times (Ladle and Westlake 1995). The relative stability of conditions within chalk streams has been widely noted (Giles *et al.* 1991, Berrie 1992, Mainstone 1999). However, it has also been recorded widely that the previous 200 years has seen much change throughout chalk river catchments, primarily driven by anthropogenic development and exploitation of the ecosystems. For example, early use of the rivers for water power and navigation has been widely reported (Environment Agency 2000a). More recent catchment activities influencing chalk rivers include the abstraction of potable water and the discharge of various wastes (Mainstone 1999). This exploitation raises the issue of the degree to which current catchment processes influence the ecological quality of chalk river ecosystems, the main theme of this thesis.

6.2 ECOSYSTEM BIODIVERSITY

It is evident that rivers cannot be separated in theory or practice from the lands they drain (Hynes 1975). Hence, the catchment basin defines the spatial dimensions of river ecosystems (Stanford *et al.* 1996). The catchment landscape is composed of interactive, biophysical resources (e.g. water, minerals, nutrients, habitats, food webs) that are used by the assemblage of animals and plants (biodiversity) that live within the ecosystem (Stanford *et al.* 1996). Biodiversity encompasses such phenomena as genetic, morphological, and life history variation within species and the richness, distributions, biomass and productivity of populations, species, guilds and other taxonomic and trophic categories across the landscape. It also takes in the myriad of biophysical processes (functional attributes) that control these phenomena (Hall *et al.* 1992, Doppelt *et al.* 1993, Noss and Cooperrider 1994, Stanford *et al.* 1996). However, the biodiversity of a system is determined by the availability of the resources that are needed by animals and plants in order to reproduce successfully (Andrewartha and Birch 1954) and thereby sustain ecosystem integrity (Frissell and Bayles 1996). Therefore it follows that a shift in the resources available (e.g. an increase in nutrients and fine sediment) could carry potential implications for the biodiversity of chalk ecosystems.

It is recognised that river ecosystems have a certain natural capacity to maintain their ecological status despite natural and anthropogenic disturbances occurring within their catchments (Warren *et al.* 1979, Frissell *et al.* 1996, Stanford *et al.* 1996), and that biodiversity is dynamic in time and space in relation to availability of resources (Benke *et al.* 1988). Biotic dynamics derive from natural variation in the environmental setting; equilibrium conditions rarely exist for long time periods because specific productivity increases opportunities for other species to use resources, inferring that levels of ecosystem biodiversity and bioproduction are generally related to the intensity, frequency and duration of disturbance events (Huston 1979, Resh *et al.* 1988, Pimm 1991, Huston 1994, Reice 1994, Stanford *et al.* 1996).

Anthropogenic activities have almost universally played an important role in shaping and disturbing stream ecosystems (Resh and Grodhaus 1983, Petersen *et al.* 1987, Resh *et al.* 1988). Clearly the impact of humans on streams can vary in spatial scale from affecting extremely localised microhabitats (e.g. displacement of an individual rock) to affecting large regions (e.g. acidic precipitation). Thus, the ecological capacity to accommodate these disturbances varies from place to place and high ecosystem biodiversity is most likely to occur in catchments with a long legacy of high spatial and temporal environmental heterogeneity (Connell 1978, Ward and Stanford 1983, Salo *et al.* 1986, Poff and Ward 1990).

Humans tend to dominate ecosystems, thereby superimposing pervasive, continual perturbation on the natural disturbance regimes. The result can be suppression, and in some cases permanent loss, of environmental heterogeneity and biodiversity, fundamentally reducing the ecological functioning of riverine environments (Stanford *et al.* 1996). It is evident from the review of physical, chemical and biological conditions experienced within chalk rivers that certain components of these systems are potentially susceptible to the imposition of anthropogenic disturbances. For example, the presence of mid-stream gravels used by spawning fish populations (and the macroinvertebrate communities upon which they feed) is a classic feature of chalk rivers. This instream feature is potentially extremely susceptible to degradation through the physical smothering by fine sediment supplied from the surrounding catchment, and the flushing of wastewaters from watercress and fish farms. Implications of this smothering include a degradation of spawning grounds for salmonid fish and a general lowering of water quality, e.g. reduction in macroinvertebrate community diversity.

Eutrophication is also a primary concern and cause of community changes in chalk rivers (Ladle and Bass 1981, Giles *et al.* 1991, Ladle and Casey 1991). Abundant instream macrophyte beds are another common feature of chalk rivers (Armitage 1997). However, an increase in available nutrient resources can result in a shift in macrophyte community composition. For example, at eutrophic sites *Potamogeton pectinatus* can be particularly abundant, representing a shift from the more common *Ranunculus* sp. (Ladle and Westlake 1995). The growth of *Cladophora* (filamentous green algae) has also been noted as a potential community shift associated with increased inorganic nutrients. The growth of *Cladophora* has been seen to stress the growth of *Ranunculus* sp., whereas the competing plant *Potamogeton pectinatus* is less affected (Spink *et al.* 1993).

My research was initiated with the principle aim of investigating the influence of catchment disturbance on the water quality of five chalk stream catchments that vary in spatial scale. The underlying philosophy behind this work was to identify a series of point and diffuse disturbances thought to encompass many of the current anthropogenic activities undertaken throughout chalk river catchments, and to investigate their influence at a replicated catchment scale. It is recognised that it is virtually impossible to locate an undisturbed/unmanaged chalk river in the UK (Ladle and Westlake 1995). However, with the placement of control sites the current research is able to provide an in-depth assessment of the current water quality of chalk rivers and provide a basis for the assessment of the response of these systems to local and catchment disturbances. From these results a conclusion can be drawn regarding the relative stability of southern UK chalk streams.

6.3 CATCHMENT DISTURBANCES

Humans vastly reduce the capacity of river ecosystems to sustain natural biodiversity by severing or compromising the dynamic interactive pathways of the river continuum (Stanford *et al.* 1996). Environmental heterogeneity is often associated with high biodiversity, indicating that individual components of river systems must be adaptable to local and catchment scale changes in conditions. However, anthropogenically mediated environmental change often exceeds the ability of the system to adapt (Stanford *et al.* 1996). The perturbations that potentially influence ecological processes linking ecosystem components in river basins have been classified by Stanford *et al.* (1996) into three groupings: (a) water pollution of all types; (b) food web manipulation by harvest, stocking and exotic invasion (an especially important issue in chalk rivers with reference to their fish populations); (c) alteration of conditions by instream manipulations (e.g. over widening and channel sedimentation).

Catchment land uses can therefore create direct and diffuse inputs of pollutants (Hynes 1960, Warren 1971), accelerate erosion and sediment loading (Waters 1995), and alter the flux rate of materials in rivers (e.g. eutrophication), all of which ultimately impact upon the ecological functioning of river communities. For example, the harvest/stocking of fishes, and the purposeful introduction of non-native species (e.g. rainbow trout) can induce strong interactions that alter food webs by causing biomass and bioproduction shifts, species replacements and other trophic effects (Mooney and Drake 1986) that may cascade through all trophic levels and even involve terrestrial species that feed on aquatic biota (Spencer *et al.* 1991).

Compared to nutrient concentrations considered representative of pristine chalk streams (Mainstone 1999), nutrient loads in the rivers Itchen, Test, Meon, Frome and Avon are high. Therefore, this does indicate the potential for a degree of ecosystem degradation at a catchment level. All five rivers investigated displayed extremely similar levels of nutrients at a catchment level. It has been widely documented that chalk rivers experience naturally high nutrient levels (Casey and Smith 1994). Although elevated against pristine conditions (Mainstone 1999), the concentrations (especially those of phosphate) recorded throughout this research indicate that currently the rivers studied are classified as *boundary eutrophic.* Despite the elevated nutrient levels observed in the five study rivers, there is little evidence of marked impacts on their biodiversity or ecology (Chapter Three). Therefore it would appear that chalk rivers are not currently adversely affected by the nutrient inputs they receive. Given that chalk rivers generally have more than enough nutrients to support primary production (Westlake 1975) it is perhaps not unexpected that additional concentrations of these compounds do not bring about the typical eutrophic changes observed in other lowland rivers (Ladle and Westlake 1995).

Results also clearly show elevated levels of suspended particulate matter (SPM) entering the systems, compared to the levels suggested by Mainstone (1999) in pristine chalk rivers. The ecological problems associated with turbid water have been widely reviewed, with implications not only for fish populations (e.g. reduced vision, clogging of gills, reduction in spawning areas; Stuart 1953, Burton 1985, Wood and Armitage 1997), but also for macroinvertebrates - an extremely important trophic level within chalk rivers (Ryan 1991, Wood and Armitage 1997). The link between increased suspended solids and increased benthic fine sediments, an obvious potential problem within chalk rivers given salmonid spawning requirements, has also been reported (Mainstone 1999). However, the increases in suspended solids in the water column, mainly from diffuse sources, observed (Chapter Three) would appear not to be being translated into an overall degradation of benthic substrate conditions within chalk rivers at a catchment level. Indeed, although the deleterious effects of fine sediment addition to the benthic substrate matrix have been discussed (Chapter Four), it is evident from longitudinal surveys that this issue is currently a local disturbance derived from point source grazing and trampling of banks resulting in bankside erosion.

Other local disturbances were highlighted in Chapter Five (e.g. cattle grazing, gravel cleaning, and artificial riffle placement). Results of a simulated disturbance indicated again that catchment activities could potentially influence the ecological functioning of chalk rivers at a local scale. For example, the use of substrate differing in size to the local streambed (e.g. the placement of artificial riffles in the restoration of chalk rivers) could potentially alter macroinvertebrate communities in terms of their diversity and overall functional structure, a recognised important trophic level in chalk rivers (Berrie 1992).

It should be remembered that all five rivers investigated were found to respond in an extremely similar manner to the localised disturbance of increases in fine sediment. Although it is recognised that the activities examined within the individual catchments varied in the intensity and frequency of disturbance, when the sites were combined to represent the whole catchment, all of the river systems responded in a very similar manner. This suggests that chalk rivers appear to be relatively robust at a catchment level, with local disturbances potentially exerting a greater influence on the ecological function of these riverine environments.

6.4 CHALK STREAM STABILITY

As previously mentioned in § 6.1.1 much weight has been given to the relative stability of chalk streams, in terms of their biological and chemical status (Berrie 1992). Aspects of this have been highlight through my research. For example, a high degree of interannual variability in macroinvertebrate communities was evident (Chapter Four). However, the relatively long time taken for communities to colonise new substrata (Chapter Five) does demonstrate the relatively weak ability of chalk stream invertebrate communities to respond rapidly to physical disturbances. This further suggests that chalk streams are naturally undisturbed systems, at least in physical terms. This general lack of physical disturbance can be attributed to chalk streams being dominated by groundwater (Berrie 1992). The groundwater recharge results in stable hydrological conditions throughout the year, minimising physical disturbances to the stream bed and banks. In contrast, certain physico-chemical properties of chalk streams displayed marked variation, especially between seasons (e.g. temperature, SPM and nitrate). However, these local and catchment scale variations appear not to be adversely affecting the water quality of chalk streams at the current time. This finding suggests that, although physico-chemical properties of chalk streams are relatively temporally and spatially variable, their overall impacts are strongly buffered against, resulting in a high quality river system. This result therefore indicates that chalk rivers are able to accommodate physico-chemical variation (at current levels) and that physical disturbances present a major threat to the stability of these systems. Therefore physical catchment disturbances need to be minimised.

6.5 CHALK RIVER MANAGEMENT

The findings of this study have important implications for the management of chalk rivers. The management of land and water within entire catchments in order to ensure river 'health' is emerging as a popular, albeit largely undefined, response to the widespread recognition that river ecosystems are increasingly threatened (Benke 1990, Allan and Flecker 1993, Allan *et al.* 1997). It is widely recognised that river management needs to move towards an environmentally sustainable future; this means that solutions to water quality and water resource problems must seek technically self-sustaining options which are both economically efficient and enhance biodiversity (Department of the Environment 1996). Coupled with this, there is now a general convergence of views in fluvial science that physical and biological elements of catchment systems are intrinsically linked (Cannan 1998).

Hence river management of lowland systems has shifted towards a catchment approach, focusing on the fact that a holistic approach is required rather than isolated manipulation, repair, replacement or mitigation of individual sites (Wissmar and Beschta 1998). It is well known that the entire catchment can influence the rivers within its boundaries via large scale disturbances/controls on chemistry and sediment delivery (Allan and Johnson 1997). Although this is a fundamental consideration of river management, the responses of chalk rivers highlighted through this research appear to set them apart from the general lowland management 'norm'. Results indicate that local disturbances (e.g. localised benthic sedimentation) potentially result in greater disturbance than catchment scale processes.

This finding implies that the management of chalk systems requires a degree of local scale consideration prior to implementing a catchment wide approach. For example, it is suggested that catchment management of chalk rivers should not ignore the individual components that comprise chalk river catchments, and must recognise that the different factors are not static in response to disturbance, but shift on a spatial and temporal scale, as clearly shown by the current results. In terms of a management structure, Figure 6.1 displays the hierarchical levels at which management could be focused within chalk river catchments.

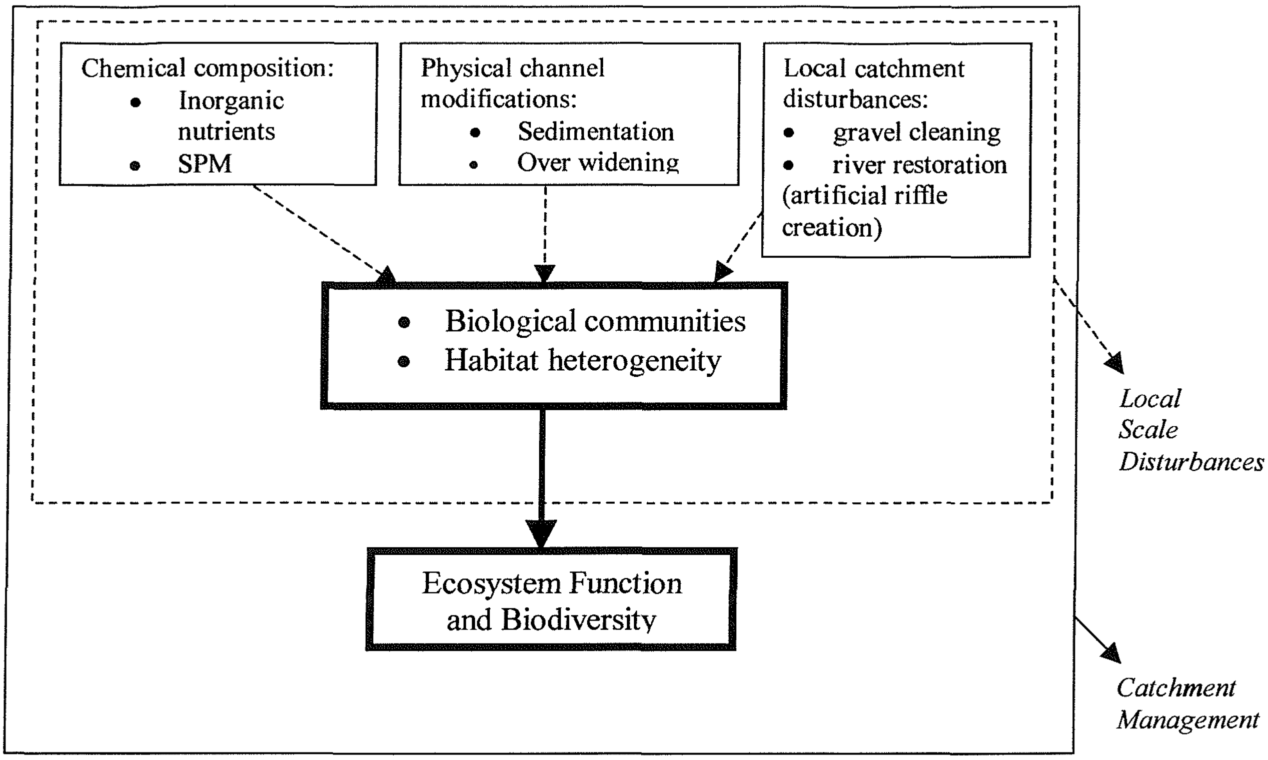


Figure 6.1. Schematic diagram of the management hierarchy in chalk rivers.

This spatial conceptualisation of aquatic ecosystems suggests that a hierarchical organisation of units: habitat-reach-segment-catchment (Frissell *et al.* 1986, Hawkins *et al.* 1993, Allan and Johnson 1997) could be the way forward in overall catchment management of chalk rivers. This hierarchical organisation will also overcome the potential issue of 'missing' local scale disturbances if management is focused only at a catchment level.

6.6 LIMITATIONS AND SUGGESTIONS FOR FURTHER RESEARCH

6.6.1 Fish Management

On the biological side, this work examines the response of macroinvertebrate communities to catchment activities. Other trophic levels within chalk streams are extremely important. Any consideration of the future management options available within chalk river catchments would be seriously flawed if it did not pay due attention to the fish populations of these river systems (Ladle and Westlake 1995). Fishery owners have played an important role in the protection of chalk river catchments for centuries (Mainstone 1999) and the importance of fishery managers in maintaining the ecological quality of chalk rivers in the future is widely recognised (Mainstone 1999). By aim and design this study examined chalk rivers at a replicated catchment scale. It was felt that to sample fish populations of these systems at this scale was not feasible due to time, labour and financial constraints.

It is generally accepted that the aim of any fisheries management within chalk rivers should be one of moving away from the intensive regimes of some fisheries, focusing on natural recruitment within a balanced biological community that is characteristic of the river reach in question (Mainstone 1999). This encourages the localised management view previously discussed and outlined in Figure 6.1. For example, many angling clubs are (a) reducing stocking levels of non-native fish and (b) stopping the removal of non-salmonid fish e.g. grayling and pike. One of the key problems is where salmonid fisheries have been artificially created in river sections (typically downstream reaches). These areas are naturally unsuited to strong salmonid recruitment, in that any habitat enhancement measures are likely to meet with limited success and indeed potentially cause more disturbance to a naturally robust system (Mainstone 1999). However, it is recognised that adoption of less intensive management regimes on such reaches may result in the loss of the salmonid fishery and greatly reduce revenues, which is an economic issue that requires consideration (Mainstone 1999).

Therefore, the basis for the management of salmonid fisheries in chalk rivers should involve angling within the context of a more natural fish fauna (through the reduction of stocking) and an increased habitat diversity. These management techniques may facilitate the sustainability of the biological communities of chalk streams (Mainstone 1999).

6.6.2 Further research considerations

The aim of this study was to examine chalk rivers across a large spatial area, encompassing rivers of differing scale. Primarily due to time constraints, a limited number of environmental parameters were investigated, relating specifically to physical elements of the channel. It is recognised that additional environmental variables could have been examined and this is a suggestion for further research. As previously mentioned the variation in fish populations has not been examined, especially in relation to habitat complexity and the interactions with non-native species. This has been attributed to time constraints and is an area for additional research.

The aim of the data analysis was to investigate patterns in water quality within and between catchments using several analytical methods. However, it was not possible to directly link the elevated nutrients and suspended sediments observed with individual source inputs (Chapter Three). The need to support correlative data by improved understanding of cause and effect links is clear (Cannan 1998). Therefore, it is suggested that further work is required to identify direct critical pathways of these pollutants.

Reach scale disturbances (overwidening and sedimentation) and catchment disturbances (nutrient enrichment) have been examined. However, there are many other disturbances occurring within catchments that have not been investigated. For example, further research could be carried out into the specific chemical and physical effects of watercress and fish farms, sewage treatment works and the biological consequences of habitat restoration.

It was decided to base the macroinvertebrate analysis in Chapters Four and Five on conventional community parameters, e.g. abundance, taxon richness, diversity and community composition. Other methods of examining macroinvertebrate communities are available, especially the use of species traits (Usseglio-Polatera *et al.* 2000). An example of which was presented in Chapter Four (assigning species with flow categories). However, other traits such as feeding method, microhabitat preference and locomotion method could have been used. Biomass/production estimates (Grubaugh *et al.* 1996, 1997) could also be used as an alternative to abundance (Cannan 1998). A combination of these additional analytical techniques could provide further information on the influence of sediment addition on macroinvertebrate communities.

6.7 CONCLUSIONS

With management taking an increasingly holistic view of rivers, the dynamic and variable nature of fluvial systems has had to be recognised in applied studies. In turn, this has required the recognition of spatial and temporal scales over which rivers operate and the disturbances they experience (Cannan 1998). My work clearly illustrates different scales of disturbance occur within the chalk stream environment and this therefore has important implications for managers. With the increasing use of ecological information in management, ecological data are vital (Allan and Johnson 1997, Johnston *et al.* 1997). However, this raises the question of the appropriate scale on which to collect these data for management purposes. Sampling local as well as catchment wide scale disturbances has revealed various patterns, and therefore sampling programmes providing information for management/assessment need to be carefully designed (Cannan 1998).

An appropriate scale for information for any management purpose has to be identified with consideration of the purpose for which the information is to be used (Cannan 1998). For example, when nutrient concentrations are considered on a catchment scale, although elevated, current levels are not thought to represent a significant ecological threat. However, when viewed on a local scale, significant elevations were recorded, suggesting that local scale variation is potentially a greater threat to the ecological integrity of chalk rivers. Therefore, it is felt that information derived from the different components integrated into a hierarchical framework can provide an evaluation of the disturbances chalk river systems face in an holistic fashion.

The importance of the landscape and vegetation of a valley to its river was clearly stated by Hynes (1975) and is an integral part of the river continuum concept (Vannote *et al.* 1980). A clear implication of this perspective is that local conditions are under some degree of regional influence, perhaps strongly so (Hildrew and Giller 1994, Wissmar 1997). However, there is only a limited understanding of the relative importance of local versus regional/catchment disturbances (Allan *et al.* 1997). A deeper understanding of these issues is necessary to resolve the spatial scale at which disturbances should be evaluated for their subsequent management.

The main limitation of many studies lies in the fact that they examine patterns in one river only. In such cases, the patterns observed will be constrained by the regional bias and the scale of the river. It has been suggested that further research is required on a larger geographical scale (Cannan 1998). This aspect has been addressed in the current research through the sampling of five different chalk river catchments, encompassing systems varying in longitudinal scale. This therefore suggests that the general robustness of chalk rivers at a catchment scale is an attribute that can be applied, albeit carefully, to all southern UK examples of chalk river systems, irrespective of catchment scale.

Therefore, this research clearly indicates that chalk rivers are extremely robust and resilient systems, currently assimilating the pressures/disturbances they face. For example, increases in nutrients and sediments at a catchment scale appear to be buffered. This could be due to naturally high velocities resulting in an extremely low residence time for pockets of water, meaning they have minimal influence on the ecological habitats and communities before being transported from the system. The continuous supply (not only from the source of the river but the entire length of the system through groundwater seepages) of extremely high quality water from aquifers also buffers against catchment wide inputs. This potentially provides a considerable scope for addition of pollutants before a degradation in water quality is transferred to a loss of ecosystem biodiversity.

In conclusion, a catchment approach to the study of chalk river systems indicates that water quality is strongly influenced by catchment impacts and disturbances operating at different spatial scales. My research suggests that although the management of catchment processes is vital, local disturbances may be of greater importance; hence managers must adopt a hierarchal approach to catchment management, thereby encompassing a local scale element. Therefore, an understanding of the disturbances operating at spatial scales ranging from the local to the regional is fundamental to sound management of freshwater ecosystems (Allan and Johnson 1997). Improving this understanding and making the linkages to management are important challenges for all aquatic ecologists.

REFERENCES

REFERENCES

Acornley R. M., and Sear D. A., (1999) Sediment transport and siltation of brown trout (Salmo trutta L.) spawning gravels in chalk streams, *Hydrological Processes*, **13**, 447 - 458.

Addiscott T.M., Whitmore A.P. and Powlson D.S., (1991) Nitrogen cycling in upland pastures of the U.K, *Philosophical Transactions of the Royal Society of London B*, **296**, 551 - 556.

Alabaster J., (1977) Biological monitoring of inland fisheries, *Proceedings of the Symposium of the European inland Fisheries Advisory Commission, Helsinki.*

Allan J.D. and Flecker A.S., (1993) Biodiversity conservation in running waters, *Bioscience*, **43**, 32 – 43.

Allan J.D. and Johnson L.B., (1997) Catchment scale analysis of aquatic ecosystem, *Freshwater Biology*, **37**, 107 – 111.

Allan J.D., (1975) The distribution ecology and diversity of benthic insects in Cement Creek, Colorado, *Ecology*, **56**, 1040 – 1053.

Allan J.D., Erickson D.L. and Fay J., (1997) The influence of catchment land use on stream integrity across multiple spatial scales, *Freshwater Biology*, **37**, 149 – 161.

Alvarez M., (1995) *An Investigation of the Spatial Variation of Freshwater Macroinvertebrates of the river Frome, Dorset*, Project report, MSc. Biology of Water Resource Management, Department of Biology, Napier University.

Andrewartha H.G. and Birch L.C., (1954) *The Distribution and Abundance of Animals*, University of Chicago Press, Chicago.

Andrews C., (1995) *The Availability of Suitable Salmonid Spawning Gravels in the river Test With Respect to Gravel Size Distribution and the Occurrence of Tufa Concretions*, BSc. Dissertation, Centre for Environmental Sciences, University of Southampton.

Angermeir P.L. and Bailey A., (1992) Use of a Geographical information System in the conservation of rivers in Virginia, USA, in: Boon, P.J., Callow, P. and Petts, G.E., (Eds) *River Conservation and Management*, John Wiley and Sons, Chichester.

Angradi T.R., (1997) Hydrologic context and macroinvertebrate community response to floods in an Appalachian headwater stream, *American Midland Naturalist*, **138**, 371 - 386.

Armitage P.D., (1995) Faunal community changes in response to flow manipulation, in: Harper D.M. and Ferguson A.J.D. (Eds), *The Ecological Basis for River Management*, John Wiley and Sons, Chichester.

Armitage P.D. and Pardo I., (1995) Impact assessment of regulation at the reach level using macroinvertebrate information from mesohabitats, *Regulated Rivers: Research and Management*, **10**, 147 – 158.

Armitage P.D., (1997) Personal Communication.

Armitage P.D., Blackburn J.H., Winder J.M. and Wright J.F., (1994) Impact of vegetation management on macroinvertebrates in chalk streams, *Aquatic Conservation: Marine and Freshwater Ecosystems*, **4**, 95 – 104.

Armitage P.D., Moss D., Wright J.F. and Furse M.T., (1983) The performance of a new biological water quality score system based on macroinvertebrates over a wide range of unpolluted running-water sites, *Water Research*, **17**, 333 – 347.

Armitage P.D., Pardo I. and Brown A., (1995) Temporary constancy in "mesohabitats" – application to management, *Archives für Hydrobiologia*, **133**, 367 – 387.

ASCE Task Committee on Sediment Transport and Aquatic Habitats, Sedimentation Committee (1992) Sediment and aquatic habitats in rivers, *Journal of Hydrological Engineering ASCE*, **118**, 669 – 687.

Barmuta L.A., (1989) Habitat patchiness and macrobenthic community structure in an upland stream in temperate Victoria, Australia, *Freshwater Biology*, **21**, 223 - 236.

Bartenhagen K.A., Turner M.H. and Osmond D.L., (2001) A Decision Support System for Non Point Source Pollution Control (WWW) Available from: http://h2osparc.wq.ncsu.edu/ (Accessed 19/02/2001).

Battegazzore M., Guzzini A., Pagnotta R. and Marchetti R., (1996) The importance of investigatory and analytical techniques in biological water quality investigation, in: Harper D.M. and Ferguson A.J.D., (Eds) *The Ecological Basis for River Management*, John Wiley and Sons, Chichester.

Beckett D.C. and Miller M.C., (1982) Macroinvertebrate colonisation of multiplate samplers in the Ohio River: the effect of Dams, *Canadian Journal of Fisheries and Aquatic Sciences*, **39**, 1622 – 1627.

Behning A., (1924) Einige ergebnisse qualitativer und quantitiver untersuchungen der bodenfauna sder Wolga, *Verhandlungen der Internationalen Vereinigung für Theoretische und Angewandte Limnologie*, **2**, 71 – 94.

Beisel J.N., Usseglio-Polatera P., Thomas S. and Moreteau J.C., (1998) Stream community structure in relation to spatial variation: the influence of mesohabitat characteristics, *Hydrobiologia*, **389**, 73 - 88.

Bell H.L., (1969) Effect of substrate types on aquatic insect distribution, *Journal of the Minnesota Academy of Sciences*, **35**, 79 – 81.

Benke A.C. and Jacobi D.J., (1986) Growth rates of mayflies in a subtropical river and their implications for secondary production, *Journal of the North American Benthological Society*, **5**, 107 - 114.

Benke A.C., (1990) A perspective on America's vanishing streams, *Journal of the North American Benthological Society*, **9**, 77 – 88.

Benke A.C., Hall C.A.S., Hawkins C.P., Lowe-McConnell R.H., Stanford J.A., Suberkropp K. and Ward J.V., (1988) Bioenergetic considerations in the analyses of stream ecosystems, *Journal of the North American Benthological Society*, **7**, 480 – 502.

Benson L.J. and Pearson R.G., (1987) Drift and upstream movement in Yuccabine Creek, an Australian tropical stream, *Hydrobiologia*, **153**, 225 – 239.

Benzie J.A.H., (1984) The colonisation mechanisms of stream benthos in a tropical river (Menik Ganga: Sir Lanka), *Hydrobiologia*, **111**, 171 – 179.

Berkman H.E. and Rabeni C.F., (1987) Effect of siltation on stream fish communities, *Environmental Biology of Fisheries*, **18**, 285 - 294.

Berrie A.D., (1992) The chalk stream environment, *Hydrobiologia*, **248**, 3 - 9.

Bickerton M., Petts G., Armitage P. and Castella E., (1993) Assessing the ecological effects of groundwater abstraction on chalk streams: three examples from eastern England, *Regulated Rivers: Research and Management*, **8**, 121 - 134.

Biggs J.F. and Close M.E., (1989) Periphyton biomass dynamics in gravel bed rivers: the relative effects of flows and nutrients, *Freshwater Biology*, **22**, 209 – 231.

Bird G.A. and Hynes H.B.N., (1981) Movement of immature aquatic insects in a lotic habitat, *Hydrobiologia*, **77**, 103 – 112.

Black K.E., Lowe J.A., Billett M.F. and Cresser M.S., (1993) Observations of the changes in nitrate concentrations along streams in seven upland moorland catchments in northeast Scotland, *Water Research*, **27**, 1195 – 1199.

Blackie J.R. and Newson M.D., (1986) The effects of forestry on the quantity and quality of run off in upland Britain, in: Slobe J.F de L.G (Ed) *Effects of Land Use on Freshwaters*, Ellis Horwood Series in water and Wastewater Technology, Water Research Centre Publication, Chichester.

Boardman J., (1990) Soil erosion on the South Downs: A review, in: Boardman J., Foster I.D.L. and Dearing J.A., (Eds) *Soil Erosion on Agricultural Land*, John Wiley and Sons, Chichester.

Boon P.J. and Howell D.L., (1997) *Freshwater Quality: Defining the Indefinable?* Scottish Natural Heritage, Edinburgh.

Boon P.J., (1992) Essential elements in the case for river conservation, in: Boon P.J., Calow P. and Petts G.E. (Eds) *River Conservation and Management*, John Wiley and Sons, Chichester.

Boothroyd I.K.G. and Dickie B.N., (1989) Macroinvertebrate colonisation of perspex artificial substrates for use in biomonitoring studies, *New Zealand Journal of Marine and Freshwater Research*, **23**, 467 – 478.

Boulton A.J. and Lake P.S., (1992) The ecology of 2 intermittent streams in Victoria, Australia, 2: comparisons of faunal compositions between habitats, rivers and years, *Freshwater Biology*, **27**, 99 – 121.

Boulton A.J., Petersen C.G., Grimm N.G. and Fisher S.G., (1992) Stability of an aquatic macroinvertebrate community in a multiyear hydrologic disturbance regime, *Ecology*, **18**, 2192 – 2207.

Boulton A.J., Spangaro G.M. and Lake P.S., (1988) Macroinvertebrate distribution and recolonisation on stones subjected to varying degrees of disturbance – an experimental approach, *Archiv für Hydrobiologie*, **113**, 551 – 576.

Boulton A.J., Stibbe S.E., Grimm N.B. and Fisher S.G., (1991) Invertebrate recolonisation of small patches of defaunated hyporheic sediments in a Sonoran Desert stream, *Freshwater Biology*, **26**, 267 – 277.

Bourassa N. and Morin A., (1995) Relationships between size structure of invertebrate assemblages and trophy and substrate composition in streams, *Journal of the North American Benthological Society*, **14**, 393 – 403.

Brewin P.A., Buckton S.T. and Ormerod S.J., (2000) The seasonal dynamics and persistence of stream macroinvertebrates in Nepal: do monsoon floods represent disturbance? *Freshwater Biology*, **44**, 581 - 594.

Brookes A., (1986) Response of aquatic vegetation to sedimentation downstream from river channelisation works in England and Wales, *Biological Conservation*, **38**, 352 – 367.

Brookes A., (1988) *Channelised Rivers: Perspectives for Environmental Management*, John Wiley and Sons, Chichester.

Brookes A., (1994) River channel change, in: Calow P. and Petts G.E. (Eds) *The Rivers Handbook * Volume 2*, Blackwell Science, London.

Brooks S.S. and Boulton A.J., (1991) Recolonisation dynamics of benthic macroinvertebrates after artificial and natural disturbances in an Australian temporary stream, *Australian Journal of Marine and Freshwater Research*, **42**, 295 – 308.

Brown D.J.A., (1988) Effect of atmospheric N deposition on surface water chemistry and the implications for fisheries, *Environmental Pollution*, **54**, 275 – 284.

Bryson P., (1998) Personal Communication.

Buikema A.L. and Vohell J.R., (1993) Toxicity studies using freshwater benthic macroinvertebrates, in: Rosenberg D.M. and Resh V.H. (Eds) *Freshwater Biomonitoring and Benthic Macroinvertebrates*, Chapman and Hall, New York and London.

Burton M.N., (1985) The effects of suspendoids on fish, *Hydrobiologia*, **125**, 221 - 241.

Byers T., (1996) *Fine Sediments in Salmonid Spawning Gravels in the Rivers Test and Wallop Brook: Their Sources and Processes of Deposition*, BSc. Dissertation, Centre for Environmental Sciences, University of Southampton.

Cahill T.H., Imperato P. and Verhoff F.H., (1974) Evaluation of phosphorous in a watershed, *Journal of Environmental Engineering*, **10445**, 439 –458.

Cairns J., Crossman J.S., Dickson K.L. and Herricks E.E., (1971) The recovery of damaged streams, *Association of the South-Eastern Biological Bulletin*, **18**, 79 – 106.

Cannan C., (1998) *Scales of Macroinvertebrate-Habitat Relationships in Fluvial Systems: A Case Study of the River Frome*, PhD Thesis (unpublished), Department of Geography, University of Southampton.

Carling P.A. and Glaister M.S., (1987) Rapid deposition of sand and gravel mixtures downstream of a negative step: the role of matrix infilling and particle overpassing in the process of bar front accretion, *Journal of the Geological Society*, **29**, 223 - 239.

Carling P.A. and McCahon C.P., (1987) Natural siltation of brown trout (*Salmo trutta* L.) spawning gravels during low-flow conditions, in: Craig J.F. and Kemper J.B. (Eds.) *Regulated Streams: Advances in Ecology*, Plenum Press, New York.

Carling P.A. and Reader N.A., (1982) Structure, composition and bulk properties of upland stream gravels, *Earth Surface Processes and Landforms*, **7**, 349 – 365.

Carling P.A., (1984) Deposition of fine and coarse sand in an open-work gravel bed, *Canadian Journal of Fisheries and Aquatic Sciences*, **41**, 263 - 270.

Casey H. and Clarke R.T., (1979) Statistical analysis of nitrate concentrations from the River Frome (Dorset) for the period 1965 – 1976, *Freshwater Biology*, **9**, 91 - 97.

Casey H. and Clarke R.T., (1986) The seasonal variation of dissolved reactive phosphate concentrations in the river Frome, Dorset, England, *Monitoring to Detect Changes in Water Quality Series*, IAHS Publication **157**, 257 – 265.

Casey H. and Farr I.S., (1982) The influence of within stream disturbance on dissolved nutrient levels during spates, *Hydrobiologia*, **92**, 447 – 462.

Casey H. and Ladle M., (1976) Chemistry and biology of the south Winterbourne, Dorset, *Freshwater Biology*, **6**, 1 – 12.

Casey H. and Newton P.V.R., (1972) The chemical composition and flow of the South Winterbourne in Dorset, *Freshwater Biology*, **2**, 229 - 234.

Casey H. and Newton P.V.R., (1973) The chemical composition and flow of the river Frome and its main tributaries, *Freshwater Biology*, **3**, 317 – 333.

Casey H. and Smith S.M., (1994) The effects of watercress growing on chalk headwater streams in Dorset and Hampshire, *Environmental Pollution*, **85**, 217 - 228.

Casey H., (1969) The chemical composition of some southern English chalk streams and its relation to discharge, *River Authorities Association Yearbook*, **1969**, 100 – 113.

Casey H., (1975) Variation in chemical composition of the River Frome, England, from 1965 – 1972, *Freshwater Biology*, **5**, 507 - 514.

Casey H., (1977) Origin and variation of nitrate nitrogen in the chalk springs, streams and rivers in Dorset and its utilisation by higher plants, *Progression in Water Technology*, **8**, 225 - 235.

Casey H., (1981) Discharge and chemical changes in a chalk stream headwater affected by the outflow of a commercial watercress-bed, *Environmental Pollution*, **2**, 373 - 385.

Casey H., Clarke R.T. and Marker A.F.H., (1981) The seasonal variation in silicon concentration in chalk streams in relation to diatom growth, *Freshwater Biology*, **11**, 335 – 344.

Casey H., Clarke R.T., Clinch J.R., Smith S. and Worsfold P.J., (1989) Statistical analysis of the results obtained by an automated spectrophotometric field monitor measuring nitrate concentrations in river water, *Analytica Chimica Acta*, **227**, 379 – 385.

Casey H., Smith S.M. and Clarke R.T., (1990) Trends and seasonality in the nitrate concentrations and loads of the Bere stream (Dorset) for the period 1966 – 1986, *Chemical Ecology*, **4**, 85 – 94.

Castella E., Bickerton M., Armitage P.D. and Petts G., (1995) The effects of water abstractions on invertebrate communities in UK streams, *Hydrobiologia*, **308**, 167 – 182.

Charvet S., Statzner B., Usseglio-Polatera P. and Dumont B., (2000) Traits of benthic macroinvertebrates in semi-natural French streams: an initial application to biomonitoring in Europe, *Freshwater Biology*, **43**, 277 – 296.

Chutter F.M., (1968) The effects of silt and sand on the invertebrate fauna of streams and rivers, *Hydrobiologia*, **34**, 54 – 76.

Ciborowski J.J.H. and Clifford H.F., (1984) Short-term colonisation patterns of lotic macroinvertebrates, *Canadian Journal of Fisheries and Aquatic Sciences*, **41**, 1626 – 1633.

Ciborowski J.J.H. and Corkum L.D., (1980) Importance of behaviour to the re-establishment of drifting Ephemeroptera, in: Flanagan F. and Marshall K.E. (Eds) *Advances in Ephemeroptera Biology*, Plenum Press, New York.

Clarke E.H., Haverkamp J.A. and Chapman W., (1985) *Eroding Soils*, Washington.

Clarke K.R. and Warwick R.M., (1994) Similarity-based testing for community pattern - the 2-way layout with no replication, *Marine Biology*, **118**, 167 - 176.

Clarke K.R., (1999) Nonmetric multivariate analysis in community-level ecotoxicology, *Environmental Toxicology and Chemistry*, **18**, 118 – 127.

Clausen B. and Biggs B.J.F., (1997) Relationships between benthic biota and hydrological indices in New Zealand streams, *Freshwater Biology*, **38**, 327 – 342.

Clyde River Purification Board (1989) *Eutrophication in Flowing Waters*, Internal Technical Report No. 92, Clyde River Purification Board, East Kilbride.

Clymo R.S., (1980) Preliminary survey of the peat bog Hummell Knowe Moss using various numerical methods, *Vegetatio*, **42**, 129 – 148.

Cogerino L., Cellot B. and Bournaud M., (1995) Microhabitat diversity and associated macroinvertebrates in aquatic banks of a large European river, *Hydrobiologia*, **304**, 103 – 115.

Collier K.J., (1995) Environmental factors affecting the taxonomic composition of aquatic macroinvertebrate communities in lowland waterways of Northland, New Zealand, *New Zealand Journal of Marine and Freshwater Research*, **29**, 453 – 465.

Comly H.H., (1945) Cyanosis in infants caused by nitrates, *Journal of the American Medical Association*, **129**, 112 – 116.

Connell J.H., (1978) Diversity in tropical rain forests and coral reefs, *Science*, **199**, 1302 – 1310.

Cordone A.J. and Kelly D.W., (1961) The influences of inorganic sediment on the aquatic life of streams, *Californian Fish and Game*, **47**, 189 – 228.

Crisp D.T. and Gledhill T., (1970) A quantitative description of the recovery of the bottom fauna in a muddy reach of a mill stream in southern England after draining and dredging, *Archiv für Hydrobiologia*, **67**, 502 – 541.

Crisp D.T., Matthews A.M. and Westlake D.F., (1982) The temperatures of nine flowing freshwaters in southern England, *Hydrobiologia*, **89**, 193 – 204.

Cummins K.W. and Lauff G.H., (1969) The influence of substrate particle size on the microdistribution of stream macrobenthos, *Hydrobiologia*, **34**, 145 – 181.

Cummins K.W., (1992) Invertebrates, in: Calow P. and Petts G.E., (Eds) *The Rivers Handbook * Volume 1*, Blackwell Science, London.

Dance K.W., (1978) *Macroinvertebrate Diversity in Two Streams as Affected by Agricultural Land Use*, Paper presented at the annual meeting of the North American Benthological Society, Winnipeg, Manitoba.

Davey G.W., Doeg T.J and Blyth J.D., (1987) Changes in benthic sediment in the Thomson River, Victoria, during construction of the Thomson Dam, *Regulated Rivers: Research and Management*, **1**, 1 - 12.

de March B.G.E., (1976) Spatial and temporal patterns in macrobenthic stream diversity, *Journal of the Fisheries Research Board of Canada*, **33**, 1261 – 1270.

Delong M.D. and Brusven M.A., (1992) Patterns of periphyton chlorophyll *a* in an agricultural nonpoint source impacted stream, *Water Resources Bulletin*, **28**, 731 - 741.

Department of the Environment (1996) *Statutory Guidance Under Section 4 of the Environment Act 1995 with respect to Objectives of the Environment Agency and its Contribution Towards Achieving Sustainable Development*, Department of the Environment, London.

Descy J.P., (1992) Eutrophication of the River Meuse, in: Sutcliffe D.W. and Jones J.G. (Eds) *Eutrophication Research and Application to Water Supply*, Freshwater Biological Association, Windermere.

Diamond J.M., (1986) Effects of larval retreats of the caddisfly *Cheumatopsyche* on macroinvertebrate colonisation in piedmont, USA streams, *Oikos*, **47**, 13 – 18.

Digby P.G.N. and Kempton R.A., (1987) *Multivariate Analysis of Ecological Communities*, Chapman and Hall, London and New York.

Dijk P.A.H. Van, (1987) *Statistical Analysis of a Routine Sampling Scheme for Monitoring Water Quality*, WRc, Medmenham.

Dillion P.J. and Rigler F.H., (1974) The phosphorus-chlorophyll relationship in lakes, *Limnology and Oceanography*, **19**, 767 - 773.

Dillion. P.J. and Kirchner. W. B., (1975) The effects of geology and land use on the export of phosphorus from watersheds, *Water Research* **9**, 135 - 148.

Dionex, (1999) Determination of inorganic anions by ion chromatography, *Application Note 133*, Dionex Corporation.

Dodds W.K. and Gudder D.A., (1992) The ecology of *Cladophora*, *Journal of Phycology*, **28**, 415 – 427.

Doeg T.J., Marchant R., Douglas M. and Lake P.S., (1989) Experimental colonisation of sand, gravel and stones by macroinvertebrates in the Acheron River, southeastern Australia, *Freshwater Biology*, **22**, 57 – 64.

Doisy K.E. and Rabeni C.F., (2001) Flow conditions, benthic food resources, and invertebrate community composition in a low-gradient stream in Missouri, *Journal of the North American Benthological Society*, **20**, 17 - 32.

Doppelt B., Scurlock M., Frissell C.A. and Karr J., (1993) *Entering the Watershed*, Island Press, Washington.

Downes B.J. and Lake P.S., (1991) Different colonisation patterns of two closely related stream insects (*Austrosimulium* spp.) following disturbance, *Freshwater Biology*, **26**, 295 – 306.

Downes B.J., Lake P.S. and Schreiber E.S.G., (1993) Spatial variation in the distribution of stream invertebrates - implications of patchiness for models of community organisation, *Freshwater Biology*, **30**, 119 - 132.

Drayton P.K., (1971) Competition, disturbance, and community organisation: the provision and subsequent utilisation of space in a rocky intertidal community, *Ecological Monographs*, **41**, 351 – 389.

Dudgeon D. and Chan I.K.K., (1992) An experimental study on the influence of periphytic algae on invertebrate abundance in a Hong Kong stream, *Freshwater Biology*, **27**, 53 – 63.

Dudley T.L., (1988) The roles of plant complexity and epiphyton in colonisation of macrophytes by stream insects, *Verhandlungen der Internationalen Vereinigung für Theoretische und Angewandte Limnologie*, **23**, 1153 – 1158.

Edington J.M. and Hildrew A.G., (1981) *Caseless Caddis Larvae of the British Isles*, Freshwater Biological Association, Scientific Publication No. 43.

Edington J.M., (1965) The effect of water flow on populations of net spinning Trichoptera, *Mitteilingen der Internationalen Vereinigung für Theoretische und Angewandte Limnologie*, **13**, 40 – 48.

Edington J.M., (1968) Habitat preferences in net spinning caddis larvae with special reference to the influence of water velocity, *Journal of Animal Ecology*, **37**, 675 – 692.

Edwards A.C., Creasey J. and Cresser M.S., (1985) Factors influencing nitrogen inputs and outputs in two Scottish upland catchments, *Soil Use and Management*, **1**, 83 – 87.

Edwards A.C., Pugh K., Wright G., Sinclair A.H. and Reaves G.A., (1990) Nitrate status of two major rivers in N.E. Scotland with respect to land use and fertiliser additions, *Chemical Ecology*, **4**, 97 – 107.

Edwards R.W., Benso-Evans K., Learner M.A., Williams P.F. and Williams R., (1972) A biological survey of the river Taff, *Water Pollution Control*, **71**, 144 – 166.

Egglishaw H.J., (1969) The distributional relationship between the bottom fauna and plant debris in streams, *Journal of Animal Ecology*, **33**, 463 – 476.

Elliott J.M., (1996) *British Freshwater Megaloptera and Neuroptera: A Key with Ecological notes*, Freshwater Biological Association, Scientific Publication No. 54, Ambleside.

Ellis A.E., (1978) *British Freshwater Bivalve Mollusca*, Synopses of British Fauna, The Linnaean Society, Academic Press, London.

Ellis J.C. and Lacey R.F., (1980) Sampling: defining the task and planning the scheme, in: Stiff M.J. (Ed) *River Pollution Control*, Ellis Horwood, Chichester.

Ellis M.M. (1936) Erosion silt as a factor in aquatic environments, *Ecology*, **17**, 29 – 42.

Elwood J.W., Newbold J.D., O'Neill R.V. and Van Winkle W., (1983) Resource spiraling: an operational paradigm for analysing lotic ecosystems, in: Fontaine T.D. and Bartell S.M. (Eds) *The Dynamics of Lotic Ecosystems*, Ann Arbor Science Publishers inc., Ann Arbor, MI.

Environment Agency (1998b) *The River Meon - Fact Sheet*, Environment Agency, Southern Region.

Environment Agency (2000a) *Local Environment Agency Plan – East Hampshire*, Environment Agency, Southern Region.

Environment Agency (2000b) *Aquatic Eutrophication in England and Wales*, Environment Agency, Bristol.

Environment Agency (2000c) Special Green Boost for Dorset River Allen (WWW) Available from: http://www.environment-agency.gov.uk//modules/MOD44.2753.html (Accessed 10/04/2001).

Environment Agency, (1996) *Hampshire Avon – Catchment Management Plan – Second Review*, Environment Agency, South West Region.

Environment Agency, (1998a) *The River Test – Fact Sheet*, Environment Agency, Southern Region.

Erman D.C. and Erman N.A., (1984) The response of stream macroinvertebrates to substrate size and heterogeneity, *Hydrobiologia*, **108**, 75 – 82.

Everest F.H., (1987) Fine sediment and salmonid production - a paradox, in: Salo E. and Cundy T., (Eds) *Streamside Management: Forestry and Fishery Interactions*, University of Washington, Seattle.

Extence C.A., Balbi D.M. and Chadd R.P., (1999) River flow indexing using British benthic macroinvertebrates: a framework for setting hydroecological objectives, *Regulated Rivers: Research and Management*, **15**, 543 – 574.

Farr I.S. and Clarke R.T., (1984) Reliability of suspended load estimates in chalk streams, *Archiv für Hydrobiologie*, **102**, 1 – 19.

Feller M.C and Kimmins J.P., (1979) Chemical characteristics of small streams near Haney in southwestern British Columbia, *Water Resources Research*, **15**, 247 – 258.

Fish H., (1973) *Principles of Water Quality Management*, Thunderbird Enterprises Limited.

Fisher S.G. and Gray L.J., (1983) Secondary production and organic matter processing by collector macroinvertebrates in a desert stream, *Ecology*, **64**, 1217 – 1224.

Fisher S.G., (1983) Succession in streams, in: Barens J.R. and Minsall G.W. (Eds.) *Stream Ecology, Application and Testing of General Ecological Theory*, Plenum, New York.

Fisher S.G., Gray L.J., Grimm N.B. and Busch D.E., (1982) Temporal succession in a desert stream ecosystem following flash flooding, *Ecological Monographs*, **52**, 93 – 110.

Fjellheim A., (1996) Distribution of benthic invertebrates in relation to stream flow characteristics in a Norwegian river, *Regulated Rivers: Research and Management*, **12**, 263 – 271.

Friberg N., Winterbourn M.J., Shearer K.A. and Larsen S.E., (1997) Benthic communities of forested streams in the South Island, New Zealand: effects of forest type and location, *Archiv für Hydrobiologie*, **138**, 289 – 306.

Frissell C.A. and Bayles D., (1996) Ecosystem management and the conservation of aquatic biodiversity and ecological integrity, *Water Resources Bulletin*, **32**, 229 – 240.

Frissell C.A., Liss W.J. and Bayles D., (1996) An integrated, biophysical strategy for ecological restoration of large watersheds, in: Spangenborg N.E. and Potts E.D. (Eds) *Proceedings of the Symposium on Changing Roles in Water Resources Management and Policy*, American Water Resources Association, Bethesda.

Frissell C.A., Liss W.J., Warren C.E. and Hurley M.D., (1986) A hierarchical framework for stream habitat classification: viewing streams in a wetland context, *Environmental Management*, **10**, 199 – 214.

Fritter R. and Mannuel R., (1995) *Lakes, Rivers, Streams and Ponds of Britain and North-West Europe*, Harper Collins Publishers.

Fruget J.F., (1992) Ecology of the Lower Rhone after 200 years of human influence - a review, *Regulated Rivers: Research and Management*, 7, 233 - 246.

Fuller R.L. and Mackay R.J., (1980) Feeding ecology of three species of *Hydropsyche* (Trichoptera: Hydropsychidae) in southern Ontario, *Canadian Journal of Zoology*, **58**, 2239 – 2251.

Furse M.T., Moss D., Wright J.F. and Armitage P.D., (1984) The influence of seasonal and taxonomic factors on the ordination and classification of running-water sites in Great Britain and on the prediction of their macroinvertebrate communities, *Freshwater Biology*, **14**, 257 – 280.

Gabriel K.R., (1971) The biplot graphic display of matrices with application to principal components analysis, *Biometrika*, **58**, 453 – 467.

Gammon J.R., (1970) *The Effect of Inorganic Sediment on Stream Biota*, Water Pollution Control Research Service 18050DWC12/70, US EPA.

Gauch H.G., (1982) *Multivariate Analysis in Community Ecology*, Cambridge University Press, Cambridge.

Giles N., Philips V. and Barnard S., (1991) *The Current Crisis: Ecological Effects of Low Flows on Chalk Streams*, Report complied for the Wiltshire Trust for Nature Conservation.

Giller P.S. and Cambell R.N.B., (1989) Colonisation patterns of mayfly nymphs (Ephemeroptera) on implanted substrate trays of different size, *Hydrobiologia*, **178**, 59 – 71.

Giller P.S. and Malmqvist B., (1998) *The Biology of Streams and Rivers*, Oxford University Press.

Giller P.S., Sangpradub N. and Twomey H., (1991) Catastrophic flooding and macroinvertebrate community structure, *Verhandlungen der Internationalen Vereinigung für Theoretische und Angewandte Limnologie*, **24**, 1724 – 1729.

Gilvear D.J., (1987) Suspended solid transport within regulated rivers experiencing periodic reservoir releases, in: Craig J.F. and Kemper B.J., (Eds) *Regulated Streams: Advances in Ecology*, Plenum, New York.

Godfrey P.J., (1978) Diversity as a measure of benthic macroinvertebrate community response to water pollution, *Hydrobiologia*, **57**, 111 – 122.

Gore J.A. and Milner A.M., (1990) Island biogeography theory: can it be used to predict lotic recovery rates? *Environmental Management*, **14**, 737 – 753.

Gore J.A., (1979) Patterns of initial benthic recolonisation of a reclaimed coal strip-mined river channel, *Canadian Journal of Zoology*, **57**, 2429 – 2439.

Gray L.J. and Fisher S.G., (1981) Postflood recolonisation pathways of macroinvertebrates in a lowland Sonoran Desert stream, *American Midlands Naturalist*, **106**, 249 – 257.

Gray L.J., (1981) Species composition and life history of aquatic insects in a lowland Sonoran desert stream, *American Midland Naturalist*, **106**, 249 – 257.

Greenberg A.E., Clesceri L.S., and Eaton A.D., (1992) *Standard Methods for the Examination of Water and Wastewater*, American Public Health Association, Washington DC.

Grensted L.W. (1939) Colonisation of new areas by water beetles, *Entomologist's Monthly Magazine*, **75**, 174 – 175.

Grimm N.B. and Fisher S.G., (1989) Stability of periphyton and macroinvertebrates to disturbance by flash floods in a desert stream, *Journal of the North American Benthological Society*, **14**, 315 – 323.

Grubaugh J.W., Wallace J.B. and Houston E.S., (1996) Longitudinal changes of macroinvertebrate communities along an Appalachian stream continuum, *Canadian Journal of Fisheries and Aquatic Sciences*, **53**, 896 – 909.

Grubaugh J.W., Wallace J.B. and Houston E.S., (1997) Production of benthic macroinvertebrates along an Appalachian stream continuum, *Freshwater Biology*, **37**, 581 – 596.

Guegan J.F., Lek S. and Oberdoff T., (1998) Energy availability and habitat heterogeneity predict global riverine fish diversity, *Nature*, **391**, 382 – 384.

Gurnell A.M., Ball J.H., Gatward T., Feaver J. and Ogle C., (1992) *River Test Sediment Investigation: The Development of a Sediment Erosion and Delivery Model Using GIS*, GeoData Institute, NRA (southern region).

Gurtz M.E. and Wallace J.B., (1984) Substrate mediated response of stream invertebrates to disturbance, *Ecology*, **65**, 1556 – 1569.

Guthery F.S. and Bingham R. L., (1996) A theoretical basis for study and management of trampling by cattle, *Journal of Range Management*, **49**, 264 - 269.

Hall C.A.S., Stanford J.A. and Hauer F.R., (1992) The distribution and abundance of organisms as a consequence of energy balances along multiple environmental gradients, *Oikos*, **65**, 377 – 390.

Hamilton J.D., (1961) The effect of sand-pit washings on a stream fauna, *Verhandlungen der Internationalen Vereinigung für Theoretische und Angewandte Limnologie*, **14**, 435 – 439.

Hansmann E.W. and Phinney H.K., (1973) Effects of logging on periphyton in coastal streams in Oregon, *Ecology*, **54**, 194 – 199.

Harms L.L., Vidal P.H. and McDermott T.E., (1978) Phosphorus interactions within stream sediments, *Journal of Environmental Engineering*, **13682**, 271 – 288.

Harper D. M. and Everard M., (1998) Why should habitat level approach underpin holistic river survey and management, *Aquatic Conservation: Marine and Freshwater Ecosystems*, **8**, 395 – 413.

Harper D.M., Mekotova J., Hulme S., White J. and Hall J., (1997) Habitat heterogeneity and aquatic invertebrate diversity in floodplain forests, *Global Ecology and Biogeography Letters*, **6**, 275 – 285.

Harper D.M., Smith C.D. and Barham P.J., (1992) Habitats as the Building Blocks for River Conservation Assessment, in: Boon P.J., Calow P. and Petts G.E., (Eds) *River Conservation and Management*, John Wiley and Sons, Chichester.

Harrison S.S.C., (2000) The importance of aquatic margins to invertebrates in English chalk streams, *Archiv für Hydrobiologie*, **149**, 213 – 240.

Harrison S.S.C., (2001) Personal Communication.

Harrold J.J. (1964) The distribution of invertebrates on submerged aquatic plants in a chalk stream, *Journal of Animal Ecology*, **33**, 335 – 348.

Hart B.T., Maher B. and Lawrence I., (1999) New generation water quality guidelines for ecosystem protection, *Freshwater Biology*, **41**, 347 – 359.

Hart D.D. and Resh V.H., (1980) Movement patterns and foraging ecology of a stream caddisfly larva, *Canadian Journal of Zoology*, **58**, 1174 – 1185.

Hart D.D., (1978) Diversity in stream insects: regulation by rock size and microspatial complexity, *Verhandlungen der Internationalen Vereinigung für Theoretische und Angewandte Limnologie*, **20**, 1376 – 1381.

Hart D.D., (1985) Causes and consequences of territoriality in a grazing stream insect, *Ecology*, **66**, 404 – 414.

Hart D.D., (1986) The adaptive significance of territoriality in filter feeding larval black flies (Diptera: Simuliidae), *Oikos*, **46**, 88 – 92.

Hauhs M., (1989) Large Bramke: an ecosystem study of a forested catchment, in: Adriano D.C. and Havas M. (Eds) *Acid Precipitation, Volume 1: Case Studies*, Springer, New York.

Hawkins C.J., Kerschner J.L., Bisson M.D., Decker L.M., Gregory S.V., McCoullough D.A., Overton C.K., Reeves G.H., Steedman R.J. and Young M.K., (1993) A hierarchical approach to classifying stream habitat features, *Fisheries*, **18**, 3 – 11.

Hawkins C.P., Murphy M.L. and Anderson N.H., (1982) Effects of canopy, substrate composition and gradient on the structure of macroinvertebrate communities in cascade range streams of Oregon, *Ecology*, **63**, 1840 – 1856.

Hearne J.W. and Armitage P.D., (1993) Implications of the annual macrophyte growth cycle on habitat in rivers, *Regulated Rivers: Research and Management*, **8**, 314 – 322.

Heathwaite A.L. and Johnes P.J., (1996) Contribution of nitrogen species and phosphorus fractions to stream water quality in agricultural catchments, *Hydrological Processes*, **10**, 971 - 983.

Hellawell J.M., (1977) Biological surveillance and water quality monitoring, in: Alabaster J.S., (Ed) *Biological Monitoring of In-land Fisheries*, Applied Science Publishers, London.

Hellawell J.M., (1978) *Biological Surveillance of Rivers, A Biological Monitoring Handbook*, WRc, Medmenham.

Hellawell J.M., (1986) Effects of suspended solids, in: *Biological Indicators of Freshwater Pollution and Environmental Management.* London and New York, Elsevier Applied Science Publishers.

Hellawell J.M., (1988) Toxic substances in rivers and streams, *Environmental Pollution*, **50**, 61 – 85.

Hellawell J.M., (1997) The contribution of biological and chemical techniques to the assessment of water quality, in: Boon P.J. and Howell D.L. (Eds) *Freshwater Quality: Defining the Indefinable?* Scottish Natural Heritage, Edinburgh.

Hem J.D., (1970) Study and interpretation of the chemical characteristics of natural water, *United States Geological Survey Water Supply Paper 1473.*

Hemphill N. and Cooper S.D., (1983) The effect of physical disturbance on the relative abundances of two-filter feeders, *Oecologia*, **77**, 73 – 80.

Hemphill N., (1988) Competition between two stream dwelling filter feeders, *Hydropsyche oslari* and *Simulium virgatum*, *Oecologia*, **82**, 56 – 61.

Herbert D.W., Alabaster J.S., Dart M.C. and Lloyd R., (1961) The effect of china-clay wastes on trout streams, *International Journal of Air and Water Pollution*, **5**, 56 – 74.

Hickley P., (1975) An apparatus for sub-dividing benthos samples, *Oikos*, **26**, 92 – 96.

Higler L.W.G., (1975) Reaction of some caddis larvae (Trichoptera) to different types of substrate in an experimental stream, *Freshwater Biology*, **5**, 151 – 157.

Higler L.W.G., (1978) Observations on caddis larvae in *stratiotes* vegetation, in: *Proceedings of the 2nd International Symposium on Trichoptera 1977,* 269 – 281, Junk, The Hague.

Hildrew A.G. and Giller P.S., (1994) Patchiness, species interactions and disturbance in the stream benthos, in: Giller P.S., Hildrew A.G. and Rafaelli D. (Eds) *Aquatic Ecology, Pattern, Process and Scale*, Proceedings of the British Ecological Society No. 34, Blackwell Science, London.

Hildrew A.G. and Townsend C.R., (1976) The distribution of two predators and their prey in an iron-rich stream, *Journal of Animal Ecology*, **45**, 41 – 57.

Hill A.R., (1978) Factors affecting the export of nitrate-nitrogen from drainage basins in southern Ontario, *Water Research*, **12**, 1045 – 1057.

Hogg I.D., de Lafontaine Y. and Eadie J.M., (1995) Phenotypic variation in benthic invertebrates: an indicator of freshwater quality? in: Boon P.J. and Howell D.L. (Eds) *Freshwater Quality: Defining the Indefinable?* Scottish Natural Heritage, Edinburgh.

House W.A. and Casey H., (1989) Transport of phosphorus in rivers, in: Tiessen H. (Ed) *Phosphorus Cycles in Terrestrial and Aquatic Ecosystems Regional Workshop 1: Europe*, Saskatchewan Institute of Pedology, Saskatoon, Canada, 251 – 282.

Houston M.A., (1994) *Biological Diversity*, Cambridge University Press, Cambridge.

Howell D.L. and Mackay D.W., (1997) Protecting freshwater quality through legislation: enforcement, inducement or agreement? in: Boon P.J. and Howell D.L. (Eds) *Freshwater Quality: Defining the Indefinable?* Scottish Natural Heritage, Edinburgh.

Humphries P., Davies P.E. and Mulcahy M.E., (1996) Macroinvertebrate assemblages of littoral habitats in the Macquarie and Mersey Rivers, Tasmania: implications for the management of regulated rivers, *Regulated Rivers: Research and Management*, **12**, 99 – 122.

Hurlbert S.H., (1975) Secondary effects of pesticides on aquatic ecosystems, *Residue Reviews*, **57**, 81 – 148.

Hurlbert S.H., (1984) Pseudoreplication and the design of ecological field experiments, *Ecological Monographs*, **54**, 187 – 211.

Huston M., (1979) A general hypothesis of diversity, *American Naturalist*, **113**, 81 – 101.

Huston M., (1994) *Biological Diversity: The Coexistence of Species on Changing Landscapes*, Cambridge University Press, Cambridge.

Hynes H.B.N, (1970) *The Ecology of Running Waters*, Liverpool University Press, Liverpool.

Hynes H.B.N., (1963) Imported organic matter and secondary productivity in streams, *Proceedings of the XVI International Congress of Zoology*, **4**, 324 - 329.

Hynes H.B.N., (1960) *The Biology of Polluted Waters*, Liverpool University Press, Liverpool.

Hynes H.B.N., (1975) The stream and its valley, *Verhandlungen der Internationalen Vereinigung für Theoretische und Angewandte Limnologie*, **19**, 1 – 15.

Hynes J.D., (1975) Annual cycles of macroinvertebrates of a river in southern Ghana, *Freshwater Biology*, **5**, 71 – 83.

Ibbotson A., Armitage P.D., Beaumont W., Ladle M. and Welton S., (1994) Spatial and temporal distribution of fish in a small lowland stream, *Fisheries Management and Ecology*, **1**, 143 - 156.

Imhof, J.G., Fitzgibbon, J. and Annable, W.K. (1996) A hierarchical evaluation system for characterising watershed ecosystems for fish habitat, *Canadian Journal of Fisheries and Aquatic Sciences*, **53**, 312-326.

Isermann K., (1990) Share of agriculture in nitrogen and phosphorus emissions into the surface waters of Western Europe against the background of their eutrophication, *Fertiliser Research*, **26**, 253 - 269.

Ivol J.M., Guinand B. and Richoux P., Tachet H., (1997) Longitudinal changes in Trichoptera and Coleoptera assemblages and environmental conditions in the Loire River (France), *Archive für Hydrobiologie*, **138**, 525 - 557.

Iwamoto R, N., Salo E.O., Madej M.A., McComas R.L. and Rulifson R.L., (1978) Sediment and water quality: a review of the literature including a suggested approach for water quality criteria with summary of workshop and conclusions and recommendations, *U.S. Environmental Protection Agency 910/9-78-048.*

Jenkins R.A., Wade K.R. and Pugh E., (1984) Macroinvertebrate-Habitat Relationships in the River Teifi Catchment and The Significance to Conservation, *Freshwater Biology*, **14**, 23 - 42.

Johnes P.J., Moss B. and Phillips G., (1996) The determination of total nitrogen and total phosphorus concentrations in freshwaters from land use, stock headage and population data: testing of a model for use in conservation and water quality management, *Freshwater Biology*, **36**, 451 - 473.

Johnes P.J. and Heathwaite A.L., (1997) Modelling the impact of land use change on water quality in agricultural catchments, *Hydrological Processes*, **11**, 269 - 286.

Johnes P.J., Moss B. and Phillips G.L., (1994a) *Lakes - Classification and Monitoring, A Strategy for the Classification of Lakes*, R and D Note 253, National Rivers Authority.

Johnes P.J., Moss B. and Phillips G.L., (1994b) *Lakes - Classification and Monitoring, A Strategy for the Classification of Lakes*, R and D Project Record 286/6/A, National Rivers Authority.

Johnson S.L. and Vaughan C.C., (1995) A hierarchical study of macroinvertebrate recolonisation of disturbed patches along a longitudinal gradient in a prairie river, *Freshwater Biology*, **34**, 531 – 540.

Johnston B.L., Richardson W.B. and Naimo T.J., (1997) Past, present, and future concepts in large river ecology, *Bioscience*, **45**, 134 – 141.

Jongman R.H.G., ter Braak C.J.F. and Van Tongeren O.F.R., (1995) (Eds) *Data Analysis in Community and Landscape Ecology*, Cambridge University Press, Cambridge.

Karr J.R., (1993) Defining and assessing ecological integrity: beyond water quality, *Environmental Toxicology and Chemistry*, **12**, 1521 – 1531.

Karr J.R., (1999) Defining and measuring river health, *Freshwater Biology*, **41**, 221 – 234.

Keller E.A., Kondolf G.M. and Hagerty D.J., (1990) Groundwater and fluvial processes; selected observations, in: Higgins C.G. and Coates D.R. (Eds) *Groundwater Geomorphology: The Role of Sub-Surface Water in Earth-Surface Processes and Landforms*, Geological Society of America Special Paper No. 252, 319 – 340.

Kent M. and Cocker P., (1992) *Vegetation Description and Analysis – A Practical Approach*, Belhaven Press, London.

Kerans B.L., Karr J. R. and Ahlstedt S. A., (1992) Aquatic invertebrate assemblages - spatial and temporal differences among sampling protocols, *Journal Of The North American Benthological Society*, **11**, 377 - 390.

Khalaf G. and Tachet H., (1980) Colonisation of artificial substrata by macroinvertebrates in a stream and variations according to stone size, *Freshwater Biology*, **10**, 475 – 482.

Klapper, H., (1991) *Control of Eutrophication in Inland Waters*, Ellis Horwood, New York.

Kohler S.L. and McPeek M.A., (1989) Predation risk and foraging behaviour of competing stream insects, *Ecology*, **66**, 1811 – 1825.

Kohler S.L., (1984) Search mechanism of a stream grazer in patchy environments: the role of food abundance, *Oecologia*, **62**, 209 – 218.

Kovach W.L., (1999) *MVSP – A Multivariate Statistical Package for Windows, Version 3.1*, Kovach Computing Services, Pentraeth, Wales.

Kristensen P. and Hansen O.H., (1994) *European Rivers and Lakes - Assessment of their Environmental State*, European Environment Agency, EEA.

Kronvang B., Aertebjerg G., Grant R., Kristensen P., Hovmand M. and Kirkegaard J., (1993) Nationwide monitoring of nutrients and their ecological effects: state of the Danish aquatic environment, *Ambio*, **22**, 176 - 187.

Kronvang B., Grant R., Larsen S.E., Svendsen L.M. and Kirstensen P., (1995) Non-point-source nutrient losses to the aquatic environment in Denmark: impact of agriculture, *Marine and Freshwater Research*, **46**, 167 - 177.

Ladle M. and Casey H., (1979) Chalk stream ecology, *Salmon Trout*, **217**, 52 – 57.

Ladle M. and Bass J.A.B., (1981) The ecology of a small chalk stream and its responses to drying during drought conditions, *Archiv für Hydrobiologie*, **78**, 17 – 24.

Ladle M. and Casey H., (1971) Growth and nutrient relationships of *Ranunculus penicullatus* var. *calcareous* in a small chalk stream, *Proceedings of the European Weed Research Council 3rd International Symposium on Aquatic weeds 1971*, 53 – 64.

Ladle M. and Casey H., (1991) The chalk stream revisited, *Salmon and Trout Magazine*, **242**, 20 – 23.

Ladle M. and Westlake D.F., (1995) River and stream ecosystems of Great Britain, in: Cushing C.E., Cummins K.W. and Minshall G.W. (Eds) *Ecosystems of the World 22*, Elsevier Applied Science Publishers.

Lake P.S. and Barmuta L.A., (1986) Stream benthic communities: persistent presumptions and current speculations, in: deDecker P. and Williams W.D., (Eds) *Limnology in Australia*, Monographs of Biology, volume 61, Dr W. Junk Publishers Dordrecht, The Netherlands.

Lake P.S. and Doeg T.J., (1985) Macroinvertebrate colonisation of stones in two upland southern Australian streams, *Hydrobiologia*, **126**, 199 – 212.

Lake P.S., (2000) Disturbance, patchiness, and diversity in streams, *Journal of the North American Benthological Society*, **19**, 573 - 592.

Lamberti G.A. and Moore J.W., (1984) Aquatic insects as primary consumers, in: Resh V.H. and Rosenberg D.M. (Eds) *The Ecology of Aquatic Insects*, Praeger Publishers, New York.

Lamberti G.A., Gregory S.V., Ashkenas L.R., Wildman R.C. and Moore J.W., (1991) Stream ecosystem recovery following a catastrophic debris flow, *Canadian Journal of Fisheries and Aquatic Sciences*, **48**, 196 – 208.

Lancaster J., Real M., Juggins S., Monteith D.T., Flower R.J. and Beaumont W.R.C., (1996) Monitoring temporal changes in the biology of acid waters, *Freshwater Biology*, **36**, 179 – 201.

Lenat D.R., Penrose D.L and Eagleson K.W., (1981) Variable effects of sediment addition on stream benthos, *Hydrobiologia*, **79**, 187 – 194.

Lenat D.R., Smock L.A., and Penrose D.L., (1980) Use of benthic macroinvertebrates as indicators of environmental quality, in: Worf D. (Ed) *Biological Monitoring for Environmental Effects*, D.C. Heath, Lexington.

Lewin J., (1981) (Ed) *British Rivers*, Allen and Unwin, London.

Lewis V., (2000) Nature Conservation and Game Fisheries Management (WWW) http://www.wildtrout.freeserve.co.uk/archive/vaughan1.htm (Accessed 27/02/01).

Likens G.E., Bormann F.H., Pierce R.S., Eaton J.S. and Johnson N.M., (1977) *Biogeochemistry of a Forested Ecosystem*, Springer-Verlag, New York.

Linduska J.P., (1942) Bottom type as a factor influencing the local distribution of mayfly nymphs, *Canadian Entomologist*, **74**, 26 – 30.

Logan P. and Brooker M.P., (1983) The macroinvertebrate faunas of riffles and pools, *Water Research*, **17**, 263 - 270.

Loucks D.P., (2000) Sustainable water resources management, *International Water Resources Association*, **25**, 3 – 10.

Lundqvist J., Lohm U. and Falkenmark M., (1985) *Strategies for River Basin Management*, D. Reidel Publishing Company.

Lytle D.A., (2000) Biotic and abiotic effects of flash flooding in a montane desert stream, *Archiv für Hydrobiologie*, **150**, 85 - 100.

Macan T.T., (1974) Running Water, *Mitteilingen der Internationalen Vereinigung für Theoretische und Angewandte Limnologie*, **20**, 301 – 321.

MacArthur R.H. and Wilson E.O., (1967) *The Theory of Island Biogeography*, Princeton University Press, Princeton, New Jersey.

Mackay R.J., (1992) Colonisation by lotic macroinvertebrates: a review of processes and patterns, *Canadian Journal of Fisheries and Aquatic Sciences*, **49**, 617 - 629.

Mackey A.P. and Berrie A.D., (1991) The prediction of water temperatures in chalk streams from air temperatures, *Hydrobiologia*, **210**, 183 – 189.

Madsen B.L., (1997) Danish experiences in river restoration II: the effort beyond restoration. in: Hansen H.O. and Madsen B.L. (Eds) *River Restoration 1996: Plenary Lectures*, National Environmental Research Institute, Denmark.

Magee J.P., McMahon T.E. and Thurow R.F., (1996) Spatial variation in spawning habitat of Cutthroat Trout in a sediment-rich stream basin, *Transactions of the American Fisheries Society*, **125**, 768 - 779.

Magoulick D.D., (1998) Effect of wood hardness, condition, texture and substrate type on community structure of stream invertebrates, *American Midland Naturalist*, **139**, 187 – 200.

Mainstone C.P., (1999) *Chalk Rivers: Nature Conservation and Management*, WRc, Medmenham.

Mainstone C.P., Davis R.D., House A. and Parr W., (1996) A review of the methods for assessing and controlling non-point sources of phosphorus, *NRA Project Record 562/5/W*, NRA, Bristol.

Mainstone C.P., Gulson J. and Parr W., (1993) Phosphates in freshwater – standards for nature conservation, *English Nature Research Reports No 73*, English Nature, Peterborough.

Mainstone C.P., Parr W. and Day M., (1998) Phosphorus in rivers: its ecological importance and the role of effluent stripping in the control of riverine eutrophication, *Draft Report to English Nature*, WRc, Medmenham.

Maitland P.S., (1997) Freshwater Quality: the use of the term in scientific literature, in: Boon P.J. and Howell D.L. (Eds) *Freshwater Quality: Defining the Indefinable?* Scottish Natural Heritage, Edinburgh.

Malmqvist B. and Otto C., (1987) The influence of substrate stability on the composition of stream benthos: and experimental study, *Oikos*, **48**, 33 – 38.

Malmqvist B., Rundle S., Bronmark C. and Erlandsson A., (1991) Invertebrate colonisation of a new, man-made stream in southern Sweden, *Freshwater Biology*, **26**, 307 – 324.

Mann R.H.K., Blackburn J.H. and Beaumont W.R.C., (1989) The ecology of brown trout (*Salmo trutta*) in English chalk streams, *Freshwater Biology*, **21**, 57 – 70.

Mantle A. and Mantle G., (1992) Impacts of low flows on chalk streams and water meadows, *British Wildlife*, **4**, 4 – 14.

Mar B.W., Horner R.R., Richey J.S., Palmer R.N. and Lettenmaier D.P., (1986) Data acquisition: cost-effective methods for obtaining data on water quality, *Environmental Science and Technology*, **20**, 545 – 551.

Margalef R., (1960) Ideas for a synthetic approach to the ecology of running waters, *Internationale Revue des Gesamten Hydrobiologie*, **45**, 133 - 153.

Margalef R., (1958) Information theory in ecology, *General Systems*, **3**, 36 –71.

Marker A.F.H. and Casey H. (1983) Experiments using an artificial stream to investigate the seasonal growth of chalk stream algae, *Annual Report for the Freshwater Biological Association*, **51**, 63 – 75.

Marr I.L. and Cresser M.S., (1983) *Environmental Chemical Analysis*, International Textbook Company, New York.

Marsden M.W., Smith M.R. and Sargent R.J., (1997) Trophic status of rivers in the Forth catchment, Scotland, *Aquatic Conservation: Marine and Freshwater Ecosystems*, **7**, 211 - 221.

Mason C.F., (1996) *Biology of Freshwater Pollution*, Longman, London and New York.

Matczak T.Z. and Mackay R.J., (1990) Territoriality in filter feeding caddisfly larvae: laboratory experiments, *Journal of the North American Benthological Society*, **9**, 26 – 34.

Matthaei, C.D., Uehlinger U., Meyer E.I. and Frutiger A., (1997) Recolonisation by benthic invertebrates after experimental disturbance in a Swiss prealpine river, *Freshwater Biology*, **35**, 233 – 248.

Mayr E., (1964) Summary, in: Baker H.G. (Ed) *The Genetics of Colonising Species*, Academic Press, New York.

McAuliffe J.R., (1983) Competition, colonisation patterns, and disturbance in stream communities, in: Barnes J.R. and Minshall G.W., (Eds) *Stream Ecology: Application and Testing of General Ecological Theory*, Plenum, New York.

McAuliffe J.R., (1984) Competition for space, disturbance, and the structure of a benthic stream community, *Ecology*, **65**, 894 – 908.

McCabe D.J. and Gotelli N.J. (2000) Effects of disturbance frequency, intensity, and area on assemblages of stream macroinvertebrates, *Oecologia*, **124**, 81 – 92.

McElravy E.P., Lamberti G.A. and Resh V.H. (1989) Year to year variation in the aquatic macroinvertebrate fauna of a northern Californian stream, *Journal of the North American Benthological Society*, **8**, 51 – 63.

Meffe G.K. and Minkly W.L., (1987) Persistence and stability of fish and invertebrate assemblages in a repeatedly disturbed Sonoran Desert stream, *American Midland Naturalist*, **117**, 177 – 191.

Meier P.G., Penrose D.L. and Polak L., (1979) The rate of colonisation by macroinvertebrates on artificial substrate samplers, *Freshwater Biology*, **9**, 381 – 392.

Mellquist P., (1992) River management – objectives and applications, in: Boon P.J., Calow P. and Petts G.E. (Eds) *River Conservation and Management*, John Wiley and Sons, Chichester.

Merritt R.W., Resh V.H. and Cummins K.W., (1995) Design of aquatic insect studies: collecting, sampling and rearing procedures, in: Merritt R.W. and Cummins K.W. (Eds) *An Introduction to the Aquatic Insects of North America*, Kendall/Hunt Publishing Company.

Metcalfe J.L., (1989) Biological water quality assessment of running waters based on macroinvertebrate communities: history and present status in Europe, *Environmental Pollution*, **60**, 101 - 139.

Metcalfe-Smith J.L., (1994) Biological water quality assessment of rivers: use of macroinvertebrate communities, in: Callow P. and Petts G.E. (Eds) *The Rivers Handbook * Volume 2*, Blackwell Science, London.

Meyer J.L., McDowell W.H., Bott T.L., Elwood J.W., Melack J.M., Peckarsky B.L., Peterson B.J. and Rublee P.A., (1988) Elemental dynamics in streams, *Journal of the North American Benthological Society*, **7**, 417 – 432.

Mihuc T.B., (1997) The functional role of lotic primary consumers: generalists versus specialist strategies, *Freshwater Biology*, **37**, 455 – 462.

Ministry of Transport and Public Works (1989) Water in the Netherlands: a time for action, *Summary of the National Policy Document on Water Management 1990 – 1994*, Ministry of Transport and Public Works, The Hague.

Minshall G.W. and Minshall J.N., (1977) Microdistribution of benthic invertebrates in a Rocky Mountain (U.S.A.) stream, *Hydrobiologia*, **55**, 231 – 249.

Minshall G.W. and Peterson R.C., (1985) Towards a theory of macroinvertebrate community structure in stream ecosystems, *Archiv für Hydrobiologie*, **104**, 49 – 76.

Minshall G.W., (1984) Aquatic insect-substratum relationships, in: Resh V.H. and Rosenberg D.M. (Eds) *The Ecology of Aquatic Insects*, Praeger Scientific, New York.

Minshall G.W., (1988) Stream ecosystem theory – a global perspective, *Journal of the North American Benthological Society*, **7**, 263 - 288.

Minshall G.W., Andrews D.A. and Manuel-Faler C.Y., (1983) Application of island biogeography theory to streams: macroinvertebrate recolonisation of the Teton River, Idaho, in: Barens J.R. and Minsall G.W. (Eds) *Stream Ecology, Application and Testing of General Ecological Theory*, Plenum, New York.

Minshall G.W., Cummins K.W., Petersen R.C., Cushing C.E., Burns D.A., Sedell J.R. and Vannote R.L., (1985) Developments in stream ecosystem theory, *Canadian Journal of Fisheries and Aquatic Sciences*, **42**, 1045 – 1055.

Minshall G.W., Petersen R.C., Cummins K.W., Bott T.L., Sedell J.R., Cushing C.E. and Vannote R.L., (1983) Interbiome comparison of stream ecosystem dynamics, *Ecological Monographs*, **53**, 1 - 25.

Modest, M., Small, H. and Gorniak, A., (1992) Ground water inflow of nutrients to a lake from differently utilised catchments, *Water Research*, **26,** 1237 - 1242.

Moon H.P., (1939) Aspects of the ecology of aquatic insects, *Transactions of the Society for British Entomology*, **6**, 39 – 49.

Mooney H.A. and Drake J.A. (1986) (Eds) *Ecology of Biological Invasions of North America and Hawaii*, Springer-Verlag, New York.

Morgan N.C. and Egglishaw H.J., (1965) A survey of the bottom fauna of streams in the Scottish Highlands, part I: composition of the fauna, *Hydrobiologia*, **25**, 181 – 211.

Moss B., (1998) *The Ecology of Fresh Waters*, Blackwell Science, London.

Moss B., Johnes P. and Phillips G., (1996) The monitoring of ecological qualify and the classification of standing waters in temperate regions: a review and proposal based upon a worked scheme for British waters, *Biology Research*, **71**, 301 - 339.

Muller K., (1954) *Investigations on the Organic Drift in North Swedish Streams*, Report for the institute of Freshwater Research, Drottingholm, **35**, 133 – 148.

Mullholland P.J. and Hill W.R., (1997) Seasonal patterns in streamwater nutrient and dissolved organic carbon concentrations: Separating catchment flow path and instream effects, *Water Resources Research*, **33**, 1297 – 1306.

Mundie J.H. and Crabtree D.G., (1997) Effects on sediments and biota of cleaning a salmonid spawning channel, *Fisheries Management and Ecology*, **4**, 111 – 126.

Murdoch P.S. and Stoddard J.L., (1992) The role of nitrate in the acidification of streams in the Catskill Mountains of New York, *Water Resources Research*, **28**, 2707 – 2720.

National River Authority, (1991) *River Test Catchment Management Plan,* National Rivers Authority, Southern Region.

National Rivers Authority, (1992b) *Hampshire Avon Catchment Management Plan,* National Rivers Authority, South West Region.

National Rivers Authority, (1992a) *River Itchen Catchment Management Plan: Consultation Report,* National Rivers Authority, Southern Region.

National Rivers Authority, (1993) *River Itchen Catchment Management Plan,* National Rivers Authority, Southern Region.

National Rivers Authority, (1995) *River Frome and Piddle Catchment Management Plan,* National Rivers Authority, South Western Region.

Neal C. and Hornung M., (1990) Special Issue: Transfer of elements through the hydrological cycle, *Journal of Hydrology*, **116**, August 1990.

Neal C., Smith C.J., Walls J., Billingham P., Hill S. and Neal M., (1990) Hydrogeochemical variations in Hafre forest stream waters, mid-Wales, in: Neal C. and Hornung M. (Eds) *Special Issue: Transfer of Elements Through the Hydrological Cycle*, *Journal of Hydrology*, **116**, 185 – 200.

Neill M., (1989) Nitrate concentrations in river waters in the south-east of Ireland and their relationships with agricultural practices, *Water Research*, **23**, 1339 - 1355.

Newbold J.D., (1992) Cycles and spirals of nutrients, in: Calow P. and Petts G.E. (Eds) *The Rivers Handbook * Volume 1*, Blackwell Science, London.

Newson M.D., (1992) River conservation and catchment management: a UK perspective, in: Boon P.J., Calow P. and Petts G.E. (Eds) *River Conservation and Management*, John Wiley and Sons, Chichester.

Norris R.H. and Georges A., (1993) Analysis and interpretation of Benthic Macroinvertebrate Surveys. in: Rosenberg D.M. and Resh V.H. (Eds) *Freshwater Biomonitoring and Benthic Macroinvertebrates*, Chapman and Hall, New York and London.

Norris R.H., (1995) Biological monitoring – the dilemma of data-analysis, *Journal of the North American Benthological Society,* **14**, 440 - 450.

Norris R.H., McElravy E.P. and Resh V.H., (1992) The sampling problem, in: Callow P. and Petts G.E. (Eds) *The Rivers Handbook * Volume 1*, Blackwell Science, London.

Noss R.F. and Cooperrider A.Y., (1994) *Saving Nature's Legacy: Protecting and Restoring Biodiversity*, Island Press, Washington.

Nuttall P.M. and Bielby G.H., (1973) The effects of china-clay waste on stream invertebrates, *Environmental Pollution*, **5**, 77 – 86.

O'Connor N.A., (1991) The effect of habitat complexity on the macroinvertebrates colonising wood substrates in a lowland stream, *Oecologia*, **85**, 504 – 512.

OECD (1982) *Eutrophication of Waters: Monitoring Assessment and Control*, Organisation for Economic Co-operation and Development, Paris.

Olssoon T. I. and Persson B., (1988) Effects of deposited sand on ova survival and alevin emergence in brown trout (*Salmo trutta* L), *Archiv für Hydrobiologie,* **113**, 621 - 627.

Oschwald W.R., (1972) Sediment-water interactions, *Journal of Environmental Quality*, **1**, 360 – 366.

Otley H., (2000) Healthy Rivers (WWW) Available from: www.huonvalley.tas.gov.au/healthyrivers/waterquality/parameters/ph.html (Accessed 8/2/2001).

Otto C. and Sjostrom P., (1985) Behaviour of drifting insect larvae, *Hydrobiologia*, **131**, 77 – 86.

Owens L.B., Edwards W.M. and Van Keuren R.W., (1991) Baseflow and stormflow transport of nutrients from mixed agricultural watersheds, *Journal of Environmental Quality*, **20**, 407 – 414.

Pagenkopf G.K., (1978) *Introduction to Natural Water Chemistry*, Marcel Dekker, inc., New York and Basel.

Paine R.T. and Levin S.A., (1981) Intertidal landscapes: disturbance and the dynamics of pattern, *Ecological Monographs*, **51**, 145 – 314.

Painter D., (1999) Macroinvertebrate distributions and the conservation value of aquatic Coleoptera, Mollusca and Odonata in the ditches of traditionally managed and grazing fen at Wicken Fen, UK, *Journal of Applied Ecology*, **36**, 33 – 48.

Palmer C.G., Okeefe J.H. and Palmer A.R., (1991) Are macroinvertebrate assemblages in the Buffalo river, Southern Africa, associated with particular biotopes? *Journal of the North American Benthological Society*, **10**, 349 – 357.

Palmer M.A. and Poff L.N., (1997) The influence of environmental heterogeneity on patterns and processes in streams, *Journal of the North American Benthological Society*, **16**, 169 – 173.

Paolilo S.A.G., (1969) Hydrogeology of the River Frome catchment (southern England), *Memoire Note dell Instituto di Geologia Aplicata, Napoli*, **10**, 57 - 84.

Parker M.S., (1989) Effect of substrate composition on detritus accumulation and macroinvertebrate distribution in a southern Nevada desert stream, *Southwestern Naturalist*, **34**, 181 – 187.

Parr W., (1994) Water quality monitoring, in: Calow P. and Petts G. E., (Eds) *The Rivers Handbook * Volume 2*, Blackwell Science, London.

Parr W., Mainstone C.P. and Morgan N., (1998) Nutrient budget for the upper reaches of the Hampshire Avon, *Report to English Nature and the Environment Agency*, WRc, Medmenham.

Pearson R.G. and Jones N.V., (1975) The effects of dredging operations on the benthic community of a chalk stream, *Biological Conservation*, **8**, 273 – 278.

Peckarsky B.L., (1980) Influence of detritus upon colonisation of stream invertebrates, *Canadian Journal of Fisheries and Aquatic Science*, **37**, 957 – 963.

Peckarsky B.L., (1985) Do predaceous stoneflies and siltation affect the structure of stream insect communities colonising enclosures? *Canadian Journal of Zoology*, **63**, 1519 – 1530.

Peckarsky B.L., (1986) Colonisation of natural substrates by stream benthos, *Canadian Journal of Fisheries and Aquatic Sciences*, **43**, 700 – 709.

Pegram G.C. and Bath A.J., (1995) Role of non-point sources in the development of a water quality management plan for the Mgeni river catchment, *Water Science and Technology*, **32**, 175 – 182.

Pennak R.W. and Van Gerpen E.D., (1947) Bottom fauna production and physical nature of the substrate in a northern Colorado trout stream, *Ecology*, **28**, 42 – 48.

Percival E. and Whitehead H., (1929) A quantitative study of some types of stream bed, *Journal of Ecology*, **17**, 282 – 314.

Petersen R.C., Madsen B.L., Wilzbach M.A., Magadza C.H.D., Paarlberg A., Kullberg A. and Cummins K.W., (1987) Stream management: emerging global similarities, *Ambio*, **16**, 166 – 179.

Petts G.E., (1987) Time-scales for ecological change in regulated rivers, in: Craig J.F. and Kempers J.B. (Eds) *Regulated streams: Advances in Ecology*, Plenum Press, New York.

Petts G.E., (1994) Rivers: dynamic components of catchments, in: Calow P. and Petts G.E. (Eds) *The Rivers Handbook * Volume 2*, Blackwell Science, London.

Petts G.E., Maddock I., Bickerton M. and Ferguson A., (1995) Linking hydrology and ecology: the scientific basis for river management, in: Harper D. and Ferguson A. (Eds) *The Ecological Basis for River Management*, John Wiley and Sons, Chichester.

Petts G.E., Thoms M.C., Brittan K. and Atkin B., (1989) A freeze-coring technique applied to pollution by fine sediment in gravel-bed rivers, *The Science of the Total Environment*, **84**, 259 - 272.

Philipson G.N., (1954) The effect of water flow and oxygen concentration on six species of caddis fly (Trichoptera) larvae, *Proceedings of the Zoological Society of London*, **124**, 547 – 564.

Pianka E.R., (1970) On *r*- and *k*-selection, *American Naturalist*, **104**, 592 – 597.

Pimm S.L., (1991) *The Balance of Nature? Ecological Issues in the Conservation of Species and Communities*, University of Chicago Press, Chicago.

Pinay G., Decamps H., Chauvet E. and Fustec E., (1990) Functions of ecotones in fluvial systems, in: Naiman R.J. and Decamps H. (Eds), *The Ecology and Management of Aquatic Terrestrial Ecotones*, UNESCO, Paris, 141 – 171.

Pinder L.C.V., Ladle M., Gledhill T., Bass J.A.B. and Matthews A.M., (1987) Biological surveillance of water quality – 1, a comparison of macroinvertebrate surveillance methods in relation to assessment of water quality, in a chalk stream, *Archiv für Hydrobiologie*, **109**, 207 - 226.

Pinoke H.B., Gburek W.J., Sharpley A.N. and Schnabel R.R., (1996) Flow and nutrient export patterns for an agricultural hill-land watershed, *Water Resources Research*, **32**, 1795 – 1804.

Pinoke H.B., Gburek W.J., Schnabel R.R., Sharpley A.N. and Elwinger G.F., (1999) Seasonal flow, nutrient concentrations and loading patterns in stream flow draining an agricultural hill-land watershed, *Journal of Hydrology*, **220**, 62 – 73.

Plafkin J.L., Barbour M.T., Porter K.D., Gross S.K. and Hughes R.M., (1989) *Rapid Bioassessment Protocols for Use in Streams and Rivers: Benthic Macroinvertebrates and Fish*, U.S. EPA, Office of water, EPA/444/4-89-001, Washigton, DC.

Poff N.L. and Ward J.V., (1990) Physical habitat template of lotic systems: recovery in the context of historical pattern of spatiotemporal heterogeneity, *Environmental Management*, **14**, 629 – 645.

Poizat G. and Pont D., (1996) Multi-scale approach to species-habitat relationships: juvenile fish in a large river section, *Freshwater Biology*, **36**, 611-622.

Power M.E., (1992) Habitat heterogeneity and functional significance of fish in river food webs, *Ecology*, **73**, 1675 – 1688.

Prenda J., Armitage P.D. and Grayston A., (1997) Habitat use by the fish assemblages of two chalk streams, *Journal of Fish Biology*, **51**, 64 - 79.

Pringle C.M., Naiman R.J., Bretschlko G., Karr J.R., Oswood M.W., Webster J.R., Welcomme R.L. and Winterbourn M.J., (1988) Patch dynamics in lotic systems: the stream as a mosaic, *Journal of the North American Benthological Society*, **7**, 503 – 524.

Pugh K.B., (1997) Organisational use of the term 'freshwater quality' in Britain, in: Boon P.J. and Howell D.L. (Eds) *Freshwater quality: defining the Indefinable?* Scottish Natural Heritage, Edinburgh.

Quinn G.P., Lake P.S. and Schreiber S.G., (1998) A comparative study of colonisation by benthos in a lake and its out flowing stream, *Freshwater Biology*, **39**, 623 – 635.

Quinn J.M., Davies-Colley R.J., Hickey C.W., Vickers M.L. and Ryan P.A., (1992) Effects of clay discharge on streams: 2. Benthic invertebrates, *Hydrobiologia*, **248**, 235 – 247.

Quinn J.M., Williamson R.B., Smith R.K. and Vickers M.L., (1992) Effects of riparian grazing and channelisation on streams in Southland, New-Zealand 2: Benthic invertebrates, *New Zealand Journal of Marine and Freshwater Research*, **26**, 259 - 273.

Rabeni C.F. and Minshall G.W., (1977) Factors affecting microdistribution of stream benthic insects, *Oikos*, **29**, 33 – 43.

Rabeni C.F. and Smale M.A., (1995) Effects of siltation on stream fishes and the potential mitigating role of the buffering riparian zone, *Hydrobiologia*, **303**, 211 – 219.

Rast W. and Lee G.F., (1983) Nutrient loading estimates for lakes, *Journal of Environmental Engineering*, **109**, 502 - 517.

Reed J.L., Campbell I.C. and Bailey P.C.E., (1994) The relationship between invertebrate assemblages and available food at forest and pasture sites in three south eastern Australian streams, *Freshwater Biology*, **32**, 641 – 650.

Reed J.R., (1977) Stream community responses to road construction sediments, *Virginia Water Resources Research Centre Bulletin*, **97**, 61 - 67.

Reice S.R., (1974) Environmental patchiness and the breakdown of leaf litter in a woodland stream, *Ecology*, **55**, 1271 – 1282.

Reice S.R., (1977) The role of animal associations, current velocity and sediments in lotic litter decomposition, *Oikos*, **29**, 357 – 365.

Reice S.R., (1980) The role of substratum in benthic macroinvertebrate microdistribution and litter decomposition in a woodland stream, *Ecology*, **61**, 580 – 590.

Reice S.R., (1994) Nonequilibrium determinants of biological community structure, *American Scientist*, **82**, 424 – 435.

Rempel L.L., Richardson J.S. and Healey M.C., (2000) Macroinvertebrate community structure along gradients of hydraulic and sedimentary conditions in a large gravel bed river, *Freshwater Biology*, **45**, 57 – 73.

Resh V.H. and Grodhaus G., (1983) Aquatic insects in urban environments, in: Frankie G.W. and Koehler C.S. (Eds) *Urban Entomology: Interdisciplinary Perspectives*, Praeger Publishers, New York.

Resh V.H. and McElravy E.P., (1993) Contemporary quantitative approaches to biomonitoring – using benthic macroinvertebrates, in: Rosenberg D.M. and Resh V.H. (Eds) *Freshwater Biomonitoring and Benthic Macroinvertebrates*, Chapman and Hall, New York and London.

Resh V.H., (1979) Sampling variability and life history features: considerations in the design of aquatic insect studies, *Journal of the Fisheries Research Board of Canada,* **36**, 290 – 311.

Resh V.H., Brown A.V., Covich A.P., Gurtz M.E., Li H.W., Minshall G.W., Reice S.R., Sheldon A.L., Wallace J.B. and Wissmar R.C., (1988) The role of disturbance theory in stream ecology, *Journal of the North American Benthological Society*, **7**, 433 – 455.

Reynolds C., (1984) *The Ecology of Freshwater Phytoplankton*, Cambridge University Press, Cambridge.

Richards C. and Host G., (1993) Identification of predominant environmental factors structuring stream macroinvertebrate communities within a large agricultural catchment, *Freshwater Biology*, **29**, 285 – 294.

Richards C. and Minshall G.W., (1988) The influence of periphyton abundance on *Baetis bicaudatus* distribution and colonisation in a small stream, *Journal of the North American Benthological Society*, **7**, 77 – 86.

Richards C. and Minshall G.W., (1992) Spatial and temporal trends in stream macroinvertebrate species assemblages: the influence of watershed disturbance, *Hydrobiologia*, **241**, 173 – 184.

Richards C., Host G. and Arthur J.W., (1993) Identification of predominant environmental factors structuring stream macroinvertebrate communities within a large agricultural catchment, *Freshwater Biology*, **29**, 285 – 294.

Rincon J. and Cressa C., (2000) Temporal variability of macroinvertebrate assemblages in a neotropical intermittent stream in Northwestern Venezuela, *Archiv für Hydrobiologie*, **148**, 421- 432.

Ritchie J.C., (1972) Sediment, fish and fish habitat, *Journal of Soil and Water Conservation,* **27**, 125 – 129.

Robach F., Merlin S., Rolland T. and Tremolieres M., (1996) Ecosphysical approach of water quality bioindication using aquatic plant materials, *Ecologie*, **27**, 203 – 214.

Robards K., McKelvie I.D., Benson R.L., Worsfold P.J., Blundell N.J. and Casey H., (1994) Determination of carbon, phosphorus, nitrogen and silicon species in waters, *Analytica Chimica Acta*, **287**, 147 - 190.

Roberts G. and Marsh T., (1987) The effects of agricultural practices on the nitrate concentrations in the surface water domestic supply sources of Western Europe, in: *Water for the Future: Hydrology in Perspective*, IAHS Publication **164**, 365 - 380.

Robinson A.R., (1973) Sediment, our greatest pollutant. in: Tank R.W. (Ed) *Focus on Environmental Geology*, Oxford Press, New York.

Rosenberg D.M. and Resh V.H., (1996) Use of aquatic insects in biomonitoring, in: Merritt R.W. and Cummins K.W. (Eds) *An Introduction to the Aquatic Insects of North America*, Kendall/Hunt Publishing Company, USA.

Rosenberg D.M. and Resh V.H., (Eds) (1993) *Freshwater Biomonitoring and Benthic Macroinvertebrates*, Chapman and Hall, New York.

Rosenberg D.M. and Snow N.B., (1975) Ecological studies of aquatic organisms in the Mackenzie and Procupine River drainages in relation to sedimentation, *Environment Canada Fisheries and Marine Services Technical Report No. 547.*

Rosenberg D.M., Danks H.V. and Lehmkuhl D.M., (1986) Importance of insects in environmental impact assessment, *Environmental Management*, **10**, 773 – 783.

Ross H.H., (1963) Stream communities and terrestrial biomes, *Archiv für Hydrobiologie*, **59**, 235 – 242.

Roux D.J., Van Vilet H.R. and Van Veelen M., (1993) Towards integrated water quality monitoring: assessment of ecosystem health, *Water SA*, **19**, 275 - 280.

Rutt G.P., Weatherely N.S. and Ormerod S.J., (1989) Microhabitat availability in Welsh moorland and forest streams as a determinant of macroinvertebrate distribution, *Freshwater Biology*, **22**, 247 – 261.

Ryan P.A. (1991) Environmental effects of sediment on New Zealand streams: a review, *New Zealand Journal of Marine and Freshwater Research*, **25**, 207 – 221.

Salo J., Kalliola R., Hakkinen I., Makinen P., Niemela A., Puhakka M. and Coley P.D., (1986) River dynamics and the diversity of Amazon lowland forest, *Nature*, **322**, 254 – 258.

Sanders T.G., Ward R.C., Loftis J.C., Steele T.D., Adrian D.D. and Yevjevich V., (1983) *Design of Networks for Monitoring Water Quality*, Water Resources Publications, Littleton, Colorado.

Sarmiento J.L., Herbert T. and Togweller J.R., (1988) Mediterranean nutrient balance and episodes of anoxia, *Global Biogeochemical Cycles*, **2**, 427 - 444.

Sas H., (1989) *Lake Restoration by Reduction of Nutrient Loading*, Academia Verlag, Richarz.

Schälchli U., (1995) Basic equations for siltation of riverbeds, *Journal of Hydrological Engineering ASCE*, **121**, 274 – 287.
Scheffer M., Achterberg A.A. and Belman B., (1984) Distribution of macroinvertebrates in a ditch in relation to the vegetation, *Freshwater Biology*, **14**, 367 – 370.

Scott A. and Beaumont A.R., (1993) *IFM Annual Study Course*, Cardiff.

Scott D., (1958) Ecological studies on the Trichoptera of the river Dean, Cheshire, *Archiv für Hydrobiologie*, **54**, 340 – 392.

Scrivener J.C. and Brownlee M.J., (1989) Effects of forest harvesting on spawning gravel and incubation survival of chum (*Oncorhynchus keta*) and coho salmon (*O. kisutch*) in Carnation Creed, British Columbia, *Canadian Journal of Fisheries and Aquatic Sciences*, **46**, 681 – 696.

Sear D.A., (1993) Fine sediment infiltration into gravel spawning beds within a regulated river experiencing floods: ecological implications for salmonids, *Regulated Rivers: Research and Management*, **8**, 373 - 390.

Sear D.A., Armitage P.D. and Dawson F.H., (1999) Groundwater dominated rivers, *Hydrological Processes*, **13**, 255 – 276.

Shackle V.J., Huges S. and Lewis V.T., (1999) The influence of three methods of gravel cleaning on brown trout, *Salmo trutta*, egg survival, *Hydrological Processes*, **13**, 477 – 486.

Shaw D.W. and Minshall G.W., (1980) Colonisation of an introduced substrate by stream macroinvertebrates, *Oikos*, **34**, 259 – 271.

Sheldon A.L. and Haick R.A., (1981) Habitat selection and association of stream insects: a multivariate analysis, *Freshwater Biology*, **11**, 395 – 403.

Sheldon A.L., (1977) Colonisation curves: application to stream insects on semi-natural substrates, *Oikos*, **28**, 256 – 261.

Sheldon A.L., (1984) Colonisation dynamics of aquatic insects, in: Resh V.H. and Rosenberg D.M. (Eds) *The Ecology of Aquatic Insects*, Praeger Scientific, New York.

Shelford V.E. (1913) *Animal Communities in Temperate America*, University of Chicago Press, Chicago.

Slack J.G. and Feltz H.R., (1968) Tree leaf control on low flow water quality in a small Virginia stream, *Environmental Science and Technology*, **2**, 126 – 131.

Smith C.J., (1980) *Ecology of the English Chalk*, Academic Press, London.

Smith C.J., (1981) The prediction of river water temperatures, *Bulletin Science Hydrology*, **26**, 19 – 32.

Sokal R.R. and Rohlf F.J., (1981) *Biometry*, W.H. Freeman, London.

Soluk D.A., (1985) Macroinvertebrate abundance and production of psammophilous chironomidae in shifting sand areas of a lowland river, *Canadian Journal of Fisheries and Aquatic Sciences*, **42**, 1296 – 1302.

Sousa W.P., (1984) The role of disturbance in natural communities, *Annual Review of Ecological Systematics*, **15**, 353 – 392.

Spencer C.N., McClelland B.R. and Stanford J.A., (1991) Shrimp stocking, salmon collapse, and eagle displacement: cascading interactions in the food web of a large aquatic ecosystem, *Bioscience*, **41**, 14 – 21.

Spink A., Sparks R.E., Van Oorschot and Verhoeven J.T.A., (1998) Nutrient dynamics of large river floodplains, *Regulated Rivers: Research and Management*, **14**, 203 – 216.

Spink A.J., Murphy K.J., Smith S.M. and Westlake D.F., (1993) Effects of eutrophication on *Ranunculus penicillatus* subsp. *Pseudofluitans* (Syme) and *Potamogeton pectinatus* L., *Proceedings of the International Symposium on the Biology and Control of Aquatic Plants*, Florida.

Stanford J.A., Ward J.V., Liss W.J., Frissell C.A., Williams R.N., Lichatowich J.A. and Coutant C.C., (1996) A general protocol for restoration of regulated rivers, *Regulated Rivers: Research and Management*, **12**, 391 – 413.

Statzner B. and Higler B., (1986) Stream hydraulics as a major determinant of benthic invertebrate zonation patterns, *Freshwater Biology*, **16**, 127 - 139.

Statzner B., Gore J.A. and Resh V.H., (1988) Hydraulic stream ecology: observed patterns and potential applications, *Journal of the North American Benthological Society*, **7**, 307 – 360.

Stevens P.A., Adamson J.K., Anderson M.A., and Hornung M. (1989) Effects of clearfelling on surface water quality and site nutrient status. in: Usher M.B. and Thomson D.B.A. (Eds) *Ecological Changes in the Uplands*, Special Publication, British Ecological Society, Blackwell Science, London.

Stonehouse D.P., Giraldez C. and van Vuuren W., (1997) Holistic policy approaches to natural resource management and environmental care, *Journal of Soil and Water Conservation*, **52**, 22 – 25.

Stribe L. and Fleischer S., (1991) Agricultural production methods - impacts on drainage water nitrogen, *Verhandlungen der Internationalen Vereinigung für Theoretische und Angewandte Limnologie*, **24**, 1749 - 1752.

Strommer J.L., and Soluk L.A., (1989) Vertical distribution and abundance of invertebrates within the sandy substrate of a low-gradient headwater stream, *Freshwater Biology*, **22**, 263 – 274.

Strong D.R., Simberloff D., Abele L.G. and Thistle A., (1984) (Eds.) *Ecological Communities: Conceptual Issues and the Evidence*, Princeton University Press, Princeton NJ.

Stuart T.A., (1953) Spawning migration, reproduction and young stages of loch trout (*Salmo trutta* L.), Freshwater, *Salmonid Fisheries Research*, **5**, 74 - 79.

Sweeting R.A., (1994) River pollution, in: Calow P. and Petts G.E. (Eds) *The Rivers Handbook * Volume 2*, Blackwell Science, London.

Tarzwell C.M., (1936) Experimental evidence on the value of trout stream improvement in Michigan, *Transactions of the American Fisheries Society*, **66**, 177 – 187.

Taylor P.R. and Littler M.M., (1982) The roles of compensatory mortality, physical disturbance, and substrate retention in the development and organisation of a sand influenced, rocky intertidal community, *Ecology*, **63**, 135 – 146.

Taylor R., Florezyk H. and Jakubowska L., (1986) Run-off of nutrients from river watersheds used for agricultural purposes, *Environment Protection Engineering*, **12**, 51 - 65.

Tebo L.B., (1955) Effects of siltation resulting from improper logging on the bottom fauna of a small trout stream in the southern Appalachians, *Progressive Fish Culturist*, **17**, 64 – 70.

Ten Brink B.J.E. and Woudstra J.H., (1991) Towards an effective and rational water management: the aquatic outlook project – integrating water management, monitoring and research, *European Water Pollution Control*, **1**, 20 – 27.

ter Braak C.J.F. and Prentice I.C., (1988) A theory of Gradient Analysis, in: Begon M., Fitter A.H., Ford E.D. and MacFadyen A., (Eds) *Advances in Ecological Research*, Academic Press, London.

ter Braak C.J.F. and Smllauer P., (1998) *CANOCO Reference Manual and User's Guide to Canoco for Windows: Software for Canonical Community Ordination (version 4)*, Microcomputer Power, Ithaca, New York.

ter Braak C.J.F. and Verdonchot P.F.M. (1995) Canonical correspondence analysis and related multivariate methods in aquatic ecology, *Aquatic Sciences*, **57**, 255 – 289.

ter Braak C.J.F., (1988) *CANOCO: a FORTRAN Program for Canonical Community Ordination by (Partial) (Detrended) (Canonical) Correlation Analysis, Principal Components Analysis and Redundancy Analysis (Version 2.1)*, Agricultural Mathematics Group, Wageningen, The Netherlands.

ter Braak C.J.F., (1995) Ordination, in: Jongman R.H., ter Braak C.J.F. and van Tongeren O.F.R. (Eds) *Data Analysis in Community and Landscape Ecology*, Cambridge University Press, Cambridge.

Tikkanen P., Lassonen P., Muotka T., Huhta A. and Kuusela K., (1994) Short term recovery of benthos following disturbances from stream habitat rehabilitation, *Hydrobiologia*, **273**, 121 – 130.

Townsend C.R. and Hildrew A.G., (1976) Field experiments on the drifting, colonisation and continuous redistributions of stream benthos, *Journal of Animal Ecology*, **45**, 759 – 772.

Townsend C.R., (1989) The patch dynamics concept of stream community ecology, *Journal of the North American Benthological society*, **8**, 36 – 50.

Townsend C.R., (1996) Concepts in river ecology: pattern and process in the catchment hierarchy, *Algological Studies*, **113**, 3 – 24.

Townsend C.R., Doledec S. and Scarsbrook M.R., (1997) Species traits in relation to temporal and spatial heterogeneity in streams: a test of habitat template theory, *Freshwater Biology*, **37**, 367 – 387.

Townsend C.R., Hildrew A.G. and Schofield K., (1987) Persistence of stream invertebrate communities in relation to environmental variability, *Journal of Animal Ecology*, **56**, 597 – 614.

Townsend C.R., Hildrew, A.G. and Francis J. (1983) Community structure in some southern English streams: the influence of physicochemical factors, *Freshwater Biology*, **13**, 521 – 544.

Ulfstrand S., Nilsson L.M. and Stergar A., (1974) Composition and diversity of benthic species collectives colonising implanted substrates in a South Swedish stream, *Entomologica Scandinavia*, **5**, 115 – 122.

Ulfstrand S., (1968) Benthic animal communities in Lapland streams, a field study with particular reference to Ephemeroptera, Plecoptera, Trichoptera and Diptera: Simuliidae, *Oikos Supplement*, **10**, 1 – 120.

Underwood A.J., (1994) Spatial and temporal problems with monitoring, in: Calow P. and Petts G.E., (Eds) *The Rivers Handbook * Volume 2*, Blackwell Science, London.

UNESCO WHO, (1978) *Water Quality Surveys*, UNESCO WHO.

US EPA, (1986) *Quality Criteria for Water*, United States Environmental Protection Agency EPA0/5-86-001, Office of Water Regulations and Standards, Washington DC, National Academy of Sciences, Washington D.C.

Usseglio-Polatera P., Bournaud M., Richoux P., Tachet H., (2000) Biomonitoring through biological traits of benthic macroinvertebrates: how to use species trait databases? *Hydrobiologia*, **422**, 153 - 162.

Van Stockkom H.T.C., Stokman G.N.M. and Hovenier J.M., (1993) Quantitative use of passive optical remote sensing over coastal and inland water bodies, *International Journal of Remote Sensing*, **14**, 541 - 569.

Vannote R.L., Minshall G.W., Cummins K.W., Sedell R. and Cushing C.E., (1980) The river continuum concept, *Canadian Journal of Fish and Aquatic Sciences*, **37**, 130 - 137.

Verdonscholt P.F.M. and Higler L.W.G., (1989) Macroinvertebrates in Dutch ditches: a typical characterisation and the status of the Demmerik ditches, *Hydrobiological Bulletin*, **23 2**, 135 – 142.

Vollenweider R.A., (1968) *Scientific Fundamentals of the Eutrophication of Lakes and Flowing Water, with Particular Reference to Nitrogen and Phosphorus as Factors in Eutrophication*, Technical Report DC5/SC1/68.27, Organisation for Economic Co-operation and Development, Paris.

Wallace J.B., (1990) Recovery of lotic macroinvertebrate communities from disturbance, *Environmental Management*, **15**, 605 – 620.

Wallace J.B., Lugthart G.J., Cuffney T.F. and Schurr G.A., (1989) The impact of repeated insecticidal treatments on drift and benthos of headwater streams, *Hydrobiologia*, **179**, 135 – 147.

Wallace J.B., Vogel D.S. and Cuffney T.F., (1986) Recovery of a headwater stream from an insecticide-induced community disturbance, *Journal of the North American Benthological Society*, **93**, 311 – 351.

Wallace J.B., Webster J.R. and Woodall W.R., (1977) The role of filter feeders in flowing waters, *Archiv für Hydrobiologie*, **79**, 506 - 532.

Walling D.E. and Amos C.M., (1995) *Source, Storage and Mobilisation of Fine Sediment in a Chalk Stream System.* NRA.

Walling D.E. and Webb B.W., (1992) Water quality 3: physical characteristics, in: Callow P. and Petts G.E. (Eds) *The Rivers Handbook * Volume 1*, Blackwell Science, London.

Walling D.E., (1990) Linking the field to the river: sediment delivery from agricultural land, in: Boardman J., Foster I.D.L. and Dearing J.A. (Eds), *Soil Erosion on Agricultural Land*, John Wiley and Sons, Chichester.

Walton O.E., (1978) Substrate attachment by drifting aquatic insect larvae, *Ecology*, **59**, 1023 – 1030.

Ward D., Holmes N. and José P., (1995) *The New Rivers Handbook*, The Royal Society for the Protection of Birds, Bedfordshire.

Ward J.V. and Stanford J.A., (1983) The intermediate disturbance hypothesis: an explanation for biotic diversity patterns in the lotic ecosystem, in: Fontaine III T.D. and Bartell S.M., (Eds) *Dynamics of Lotic Ecosystems*, Ann Arbor Science Publishers, Ann Arbor, Michigan, USA.

Ward J.V., (1975) Bottom fauna substrate relationships in a northern Colorado trout stream: 1945 and 1974, *Ecology*, **56**, 1429 – 1434.

Ward R.C. and Lofis J., (1986) Establishing statistical design criteria for water quality monitoring systems: review and synthesis, *Water Resources Bulletin*, **22**, 759 – 767.

Ward R.C., Loftis J. and McBride G.B., (1986) The data-rich but information poor syndrome in water quality monitoring, *Environmental Management*, **10**, 291 – 297.

Warren C.E., (1971) *Biology and Water Pollution Control*, W.B. Saunders, Philadelphia.

Warren C.E., Allen M.J. and Haefner J.W., (1979) Conceptual frameworks and the philosophical foundations of general living systems theory, *Behavioral Science*, **24**, 296 – 310.

Washington H.G., (1984) Diversity, biotic and similarity indices, *Water Research*, **18**, 653 – 694.

Waters T.F., (1964) Recolonisation of denuded stream bottom areas by drift, *Transactions of the American Fisheries Society*, **93**, 311 – 325.

Waters T.F., (1995) *Sediment in Streams: Sources, Biological Effects and Control*, American Fisheries Society, Bethesda.

Webb B.W. and Walling D.E., (1992) Water quality 2: chemical characteristics, in: Callow P. and Petts G.E. (Eds) *The Rivers Handbook * Volume 1*, Blackwell Science, London.

Welch E.B., Horner R.R. and Patmont C.R., (1989) Prediction of nuisance periphytic biomass: a management approach, *Water Research*, **23**, 401 - 405.

Wene G. and Wickliff E.L., (1940) Modification of a stream bottom and its effect on the insect fauna, *Canadian Entomologist*, **72**, 131 – 135.

Westlake D. F., Casey H., Dawson H., Ladle M., Mann R.H.K. and Marker A.F.H., (1972) The chalk stream ecosystem, in: *Productivity Problems of Freshwaters, Proceedings IBP-UNESCO Symposium, Kazimierz Dolny, Poland, 1970,* PWN, Polish Scientific Publishers, Poland, 615 – 635.

Westlake D.F., (1968) The weight of water weed in the river Frome, *River Authorities Association Yearbook*, **1968**, 59 – 68.

Westlake D.F., (1975) Macrophytes, in: Whitton B.A. (Ed) *River Ecology*, Cambridge University Press, Cambridge.

Whiles M.R. and Wallace J.B., (1992) First-year benthic recovery of a headwater stream following a 3-year insecticide-induced disturbance, *Freshwater Biology*, **28**, 81 – 91.

White P.S., (1979) Pattern, process, and natural disturbance in vegetation, *The Botanical Review*, **45**, 229 – 299.

Whitfield P.H., (1988) Goals and data collection design for water quality monitoring, *Water Resources Bulletin*, **24**, 775 – 780.

Whiting P.J. and Stamm J., (1995) The hydrology and form of spring dominated channels, *Geomorphology*, **12**, 233 – 240.

Wiegleb G., (1981) Application of multiple discriminant analysis on the analysis of the correlation between macrophyte vegetation and water quality in running waters of Central Europe, *Hydrobiologia*, **79**, 91 – 100.

Wiley M.J. and Kohler S.L., (1981) An assessment of biological interactions in an epilithic stream community using time-lapse cinematography, *Hydrobiologia*, **78**, 183 – 188.

Williams A.G., Ternan J.L. and Kent M., (1983) Stream solute sources and variations in a temperate granite drainage basin, in: *Dissolved Loads of Rivers and Surface Water Quantity/Quality Relationships* (Proceedings of the Hamburg Symposium, August 1983), International Association of Hydrological Sciences Publication No. 414, Wallingford, UK.

Williams D.D. and Hynes H.B.N., (1976) The recolonisation of stream benthos, *Oikos*, **27**, 265 – 272.

Williams D.D. and Hynes H.B.N., (1977) Benthic community development in a new stream, *Canadian Journal of Zoology*, **55**, 1071 – 1076.

Williams D.D. and Mundie J.H., (1978) Substrate size selection by stream invertebrates and the influence of sand, *Limnology and Oceanography*, **23**, 1030 – 1033.

Williams N.E. and Hynes H.B.N., (1973) Microdistribution and feeding of the net-spinning caddisflies (Trichoptera) of a Canadian stream, *Oikos*, **24**, 73 – 84.

Wilson H., Payne S., O'Sullivan P. and Gibson M., (1996) Policy and legislation relevant to the conservation of freshwater SSSIs subject to eutrophication, *Journal of the Chartered Institute of Water and Environmental Management*, **10**, 348 - 354.

Wise D.H. and Molles M.C., (1979) Colonisation of artificial substrates by stream insects: influence of substrate size and diversity, *Hydrobiologia*, **65**, 69 – 74.

Wissmar R.C. and Beschta R.L., (1998) Restoration and management of riparian ecosystems: a catchment perspective, *Freshwater Biology*, **40**, 571 – 585.

Wissmar R.C., (1997) Historical perspectives, in: Williams J.E., Wood C.A. and Dombeck M.P. (Eds) *Watershed Restoration: Principles and Practices*, American Fisheries Society, Bethesda.

Wong S.L., Clark B. and Kosciuw R.F., (1979) An examination of the effects of nutrients on the water quality of shallow rivers, *Hydrobiologia*, **63**, 231 – 239.

Wood P.J. and Armitage P.D., (1999) Sediment deposition in a small lowland stream – management implications, *Regulated Rivers: Research and Management*, **15**, 199 – 210.

Wood P.J. and Armitage P.D., (1997) Biological effects of fine sediment in the lotic environment, *Environmental Management*, **21**, 203 - 217.

Wood P.J., Armitage P.D., Cannan C.E. and Petts G.E., (1999) Instream mesohabitat biodiversity in three groundwater streams under base flow condition, *Aquatic Conservation: Marine and Freshwater Ecosystems*, **9**, 265 – 278.

Wright .F., Hiley P.D., Cameron A.C., Wigham M.E. and Berrie A.D., (1983) A quantitative study of macroinvertebrate fauna of five biotopes in the River Lambourn, Berkshire, England, *Archives für Hydrobiologie*, **96**, 271 – 292.

Wright G.G., Edwards A.C., Morrice J.G. and Pugh K., (1991) North East Scotland river catchment nitrate loading in relation to agricultural intensity, *Chemistry and Ecology*, **5**, 263 - 281.

Wright J.F. and Symes K.L., (1999) A nine-year study of the macroinvertebrate fauna of a chalk stream, *Hydrological Processes*, **13**, 371 – 385.

Wright J.F., (1992) Spatial and temporal occurrence of invertebrates in a chalk stream, Berkshire, England, *Hydrobiologia*, **248**, 11 – 30.

Wright J.F., Blackburn J.H., Westlake D.F., Furse M.T. and Armitage P.D., (1992) Anticipating the consequences of river management for the conservation of macroinvertebrates, in: Boon P.J., Calow P. Petts G.E., (Eds) *River Conservation and Management*, John Wiley and Sons, Chichester.

Wright J.F., Furse M.T. and Armitage P.D., (1994) Use of macroinvertebrate communities to detect environmental stress in running waters, in: Sutcliffe D.W. (Ed) *Water Quality Stress Indicators in Marine and Freshwater Systems: Linking Levels of Organisation*, Freshwater Biological Association.

Wright J.F., Moss D., Armitage P.D. and Furse M.T., (1984) A preliminary classification of running-water sites in Great Britain based on macroinvertebrate species and the prediction of community type using environmental data, *Freshwater Biology*, **14**, 221 – 256.

Wrona F.J., Culp J.M. and Davis R.W., (1982) Macroinvertebrate sampling: A simplified apparatus and approach, *Canadian Journal of Fisheries and Aquatic Sciences*, **39**, 1051 – 1054.

Wu Y.F., (1931) A contribution to the biology of *Simuliium, Paper of the Michigan Academic Sciences*, **13**, 543 – 599.

Zar J.H., (1996) *Biostatistical Analysis*, Prentice Hall International Inc.

Zaret T.M., (1982) The stability/diversity controversy: a test of hypotheses, *Ecology*, **63**, 721 – 731.

APPENDICES A - F

APPENDIX A
EXAMPLES OF CHEMICAL MONITORING SITES

Plate A. Titchbourne, River Itchen.

Plate B. Highbridge, River Itchen.

Plate C. Warnford, River Meon.

Plate D. Whitchurch, River Test.

Plate E. Wraxall, River Frome.

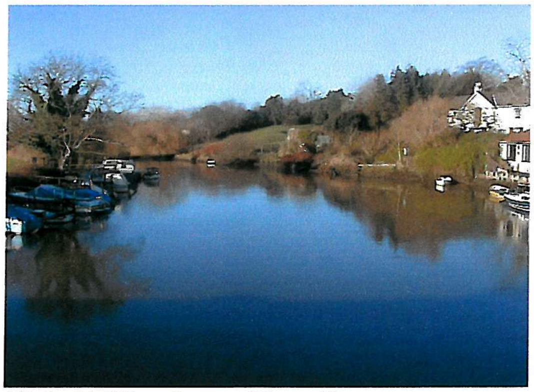


Plate F. Wareham, River Frome.

Plate G. Downton, River Avon.

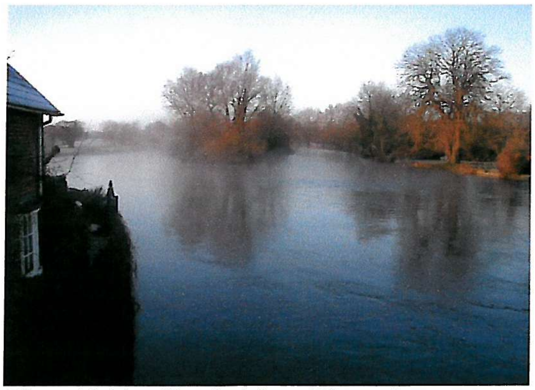


Plate H. Fordingbridge, River Avon.

APPENDIX B
EXAMPLES OF BIOLOGICAL MONITORING SITES

Plate I. Control site, River Test

Plate J. Control site, River Itchen

Plate K. Overwidened site, River Frome.

Plate L. Overwidened site, River Meon.

Plate M. Sedimented site, River Frome.

Plate N. Sedimented site, River Avon.

APPENDIX C

DETERMINATION OF ANIONS BY ION CHROMATOGRAPHY

Equipment

The Dionex DX 500 ion chromatography system was used for the determination of anion concentrations in stream water.

The DX 500 is a modular system, which in this case consisted of:

Gradient pump:	GP40
Electrochemical detector:	ED40 (in conductivity mode)
Auto sampler:	AS40 (5 ml vials)

Conditions

Anions

Guard column:	AG9-SC (4 mm)
Separating column:	ION PAC AS9-SC (4 mm)
Eluent:	1.8 mM Sodium Carbonate 1.7 mM Sodium Bicarbonate
Flow rate:	2 mls/minute
Injection volume:	25 µl
Detection:	suppressed conductivity, anion self regulating suppresser ASRS-1 (4 mm) 50 mA current

Isocratic

Sample Preparation

The sample (5 ml) was placed in the AS40 Automated Sampler vials, of which 25 µl was then injected into the separating column.

Sample Processing

Initially three replicate readings were taken for each anion being tested for in each sample. After adequate replicates were analysed it was decided to lower the number of reading to one as very little variation was experienced in the replicate readings.

APPENDIX D
CHEMICAL SURVEYS: SUMMER 1998 – SPRING 2000

ITCHEN Site 1

DATE	temperature	pH	conductivity	dissolved oxygen	SPM	nitrate	nitrite	phosphate
Summer 1998	13.1	7.7	438	93.0	0.6	5.15	0.000	0.000
Autumn 1998	11.8	7.8	427	85.0	1.2	8.73	0.022	0.000
Winter 1998	9.5	7.6	409	98.0	3.7	6.32	0.000	0.033
Spring 1999	10.6	7.8	414	99.3	3.5	3.95	0.000	0.000
Summer 1999	14.7	8.1	464	111.0	1.0	5.14	0.000	0.028
Autumn 1999	11.1	7.9	595	92.1	3.6	5.26	0.000	0.030
Winter 1999	8.8	7.8	592	104.3	6.9	0.86	0.000	0.000
Spring 2000	12.2	8.0	470	106.5	2.3	11.60	0.000	0.000

Site 2

DATE	temperature	pH	conductivity	dissolved oxygen	SPM	nitrate	nitrite	phosphate
Summer 1998	13.7	7.8	423	102.0	2.3	3.69	0.001	0.000
Autumn 1998	11.8	8.0	407.5	88.0	7.2	3.82	0.000	0.031
Winter 1998	9.2	7.8	395.5	97.5	5.3	4.38	0.000	0.038
Spring 1999	10.6	8.0	399.5	101.5	3.9	4.64	0.000	0.017
Summer 1999	15.0	8.2	439.5	127.0	2.7	5.89	0.010	0.061
Autumn 1999	11.1	8.1	572	97.7	39.1	6.92	0.000	0.000
Winter 1999	7.9	8.0	572.5	107.9	7.0	3.30	0.000	0.000
Spring 2000	14.2	8.2	495	111.0	4.0	2.79	0.000	0.000

Site 3

DATE	temperature	pH	conductivity	dissolved oxygen	SPM	nitrate	nitrite	phosphate
Summer 1998	14.1	7.9	432	106.0	1.9	4.21	0.000	0.000
Autumn 1998	11.6	8.0	410	86.0	1.5	2.63	0.000	0.000
Winter 1998	9.0	7.7	397	93.0	6.5	3.91	0.000	0.000
Spring 1999	10.7	8.0	403	100.0	4.4	3.44	0.000	0.000
Summer 1999	15.9	8.3	451	130.0	2.1	6.49	0.000	0.000
Autumn 1999	11.0	8.2	576	97.2	2.9	7.33	0.000	0.000
Winter 1999	7.8	8.0	580	105.4	9.0	4.45	0.000	0.000
Spring 2000	14.3	8.2	490	104.4	5.3	4.63	0.000	0.000

Site 4

DATE	temperature	pH	conductivity	dissolved oxygen	SPM	nitrate	nitrite	phosphate
Summer 1998	14.0	8.0	445	96.0	13.8	6.15	0.014	0.006
Autumn 1998	11.6	8.5	422.5	92.5	2.6	2.44	0.000	0.000
Winter 1998	9.0	7.9	413.5	94.5	14.3	3.98	0.000	0.000
Spring 1999	11.1	8.1	417.5	94.5	9.3	3.60	0.000	0.000
Summer 1999	15.4	8.5	480	118.0	2.9	6.21	0.000	0.000
Autumn 1999	11.6	8.3	588.5	100.4	3.6	7.30	0.000	0.000
Winter 1999	8.2	8.1	597.5	110.0	16.8	5.80	0.000	0.000
Spring 2000	15.8	8.3	525	105.9	9.6	4.07	0.000	0.000

Site 5

DATE	temperature	pH	conductivity	dissolved oxygen	SPM	nitrate	nitrite	phosphate
Summer 1998	14.7	7.9	470	77.0	2.0	5.12	0.040	0.000
Autumn 1998	11.1	8.2	433	87.0	2.3	0.52	0.000	0.000
Winter 1998	8.4	7.5	413	96.0	2.5	4.76	0.000	0.000
Spring 1999	10.7	8.1	418	87.6	10.1	3.61	0.001	0.000
Summer 1999	16.6	8.4	502	83.0	7.7	7.58	0.019	0.065
Autumn 1999	11.6	8.1	602	78.5	3.5	7.21	0.000	0.000
Winter 1999	7.1	8.1	600	97.6	14.5	1.01	0.000	0.000
Spring 2000	15.7	8.4	580	85.5	6.4	3.24	0.000	0.000

APPENDIX D Cont.

TEST	Site 1							
DATE	temperature	pH	conductivity	dissolved oxygen	SPM	nitrate	nitrite	phosphate
Summer 1998	15.0	7.7	532	94.0	4.3	1.31	0.0039	0.0000
Autumn 1998	12.8	8.0	522	89.0	2.9	2.78	0.0966	0.0961
Winter 1998	10.7	7.3	456	92.0	9.5	5.67	0.0000	0.0000
Spring 1999	12.4	7.9	462	99.5	11.3	6.25	0.0000	0.0378
Summer 1999	13.5	7.9	447	98.0	6.5	3.80	0.0219	0.0000
Autumn 1999	10.8	8.2	667	81.3	3.1	7.71	0.0697	0.0000
Winter 1999	9.7	7.4	627	93.6	14.3	6.36	0.0000	0.0316
Spring 2000	10.9	8.1	520	91.6	2.0	4.50	0.0000	0.0000

	Site 2							
DATE	temperature	pH	conductivity	dissolved oxygen	SPM	nitrate	nitrite	phosphate
Summer 1998	14.6	7.8	466	113.0	1.4	1.16	0.0000	0.0000
Autumn 1998	12.0	8.1	446	108.0	12.1	7.26	0.0937	0.2489
Winter 1998	10.0	7.5	418	97.0	5.8	4.80	0.0017	0.0000
Spring 1999	11.6	8.0	424	110.9	5.0	4.05	0.0074	0.0000
Summer 1999	14.1	8.2	439	115.0	2.9	7.99	0.0008	0.0412
Autumn 1999	9.8	8.3	604	88.3	3.3	2.43	0.0000	0.0000
Winter 1999	9.1	7.5	590	98.3	9.1	5.91	0.0246	0.0117
Spring 2000	11.7	8.4	530	101.8	2.7	7.83	0.0027	0.0000

	Site 3							
DATE	temperature	pH	conductivity	dissolved oxygen	SPM	nitrate	nitrite	phosphate
Summer 1998	16.6	8.2	463	119.0	0.4	3.82	0.0000	0.0000
Autumn 1998	12.5	8.4	424	105.0	4.3	8.68	0.0000	0.0000
Winter 1998	9.1	7.7	395	95.5	13.5	5.58	0.0000	0.0000
Spring 1999	11.5	8.1	405	111.1	6.9	4.00	0.0107	0.0000
Summer 1999	15.2	8.2	446	106.5	2.9	7.62	0.0000	0.0037
Autumn 1999	9.4	8.4	589	88.7	2.7	9.39	0.0000	0.0000
Winter 1999	8.4	7.9	578	97.0	11.3	1.90	0.0000	0.0000
Spring 2000	11.7	8.6	570	94.6	5.7	4.86	0.0000	0.0005

	Site 4							
DATE	temperature	pH	conductivity	dissolved oxygen	SPM	nitrate	nitrite	phosphate
Summer 1998	15.6	8.1	453	110.0	7.1	5.76	0.0000	0.0072
Autumn 1998	12.6	8.3	430	99.0	3.9	9.56	0.0000	0.0085
Winter 1998	8.9	7.7	396	91.0	16.0	4.88	0.0000	0.0000
Spring 1999	11.6	8.2	404	105.4	7.5	4.43	0.0045	0.0000
Summer 1999	16.1	8.2	462	95.5	3.6	8.03	0.0023	0.0026
Autumn 1999	10.0	8.3	591	86.0	5.2	10.56	0.0000	0.0000
Winter 1999	8.4	7.9	579	103.1	18.5	2.68	0.0000	0.0000
Spring 2000	12.9	8.6	580	90.1	11.6	4.06	0.0000	0.0000

	Site 5							
DATE	temperature	pH	conductivity	dissolved oxygen	SPM	nitrate	nitrite	phosphate
Summer 1998	16.0	8.2	456	102.0	4.3	4.97	0.1520	0.0000
Autumn 1998	12.3	8.1	427	90.0	6.1	5.42	0.0000	0.0019
Winter 1998	8.7	7.8	394	92.0	15.2	7.49	0.0000	0.0000
Spring 1999	11.8	8.2	401	99.4	11.8	4.06	0.0000	0.0000
Summer 1999	17.0	8.3	471	87.0	2.6	5.60	0.0000	0.0014
Autumn 1999	10.2	8.4	586	85.7	4.3	9.57	0.0000	0.0000
Winter 1999	8.0	8.2	572	101.9	20.1	3.75	0.0000	0.0000
Spring 2000	13.8	8.7	580	91.7	8.1	5.36	0.0000	0.0000

APPENDIX D Cont.

MEON Site 1

DATE	temperature	pH	conductivity	dissolved oxygen	SPM	nitrate	nitrite	phosphate
Summer 1998	14.4	8.0	473	85.0	2.7	7.66	0.2219	0.0051
Autumn 1998	11.8	8.1	445	93.0	5.8	4.17	0.0000	0.0000
Winter 1998	9.0	8.2	395	100.0	3.9	6.32	0.0000	0.0000
Spring 1999	10.0	8.2	403	96.0	13.7	4.49	0.0000	0.0000
Summer 1999	15.0	8.3	484	99.0	2.7	9.11	0.0000	0.0000
Autumn 1999	10.7	8.1	611	85.7	8.0	2.85	0.0000	0.0000
Winter 1999	7.1	8.2	603	103.8	9.7	3.22	0.0000	0.0000
Spring 2000	12.1	8.6	460	109.5	4.3	5.27	0.0000	0.0000

Site 2

DATE	temperature	pH	conductivity	dissolved oxygen	SPM	nitrate	nitrite	phosphate
Summer 1998	12.9	7.7	410	110.0	0.6	5.75	0.0000	0.0101
Autumn 1998	11.3	7.9	417	97.0	2.8	8.93	0.0021	0.0155
Winter 1998	9.5	7.5	404	96.0	0.2	4.95	0.0000	0.0000
Spring 1999	10.4	7.8	405	96.1	3.9	3.96	0.0000	0.0000
Summer 1999	13.5	8.1	435	123.0	4.1	7.52	0.0013	0.0067
Autumn 1999	10.4	7.8	579	93.2	2.6	10.55	1.3209	2.6133
Winter 1999	8.0	7.9	583	102.3	4.3	4.73	0.0000	0.0177
Spring 2000	12.3	8.0	470	104.2	1.5	8.41	0.0000	0.0000

Site 3

DATE	temperature	pH	conductivity	dissolved oxygen	SPM	nitrate	nitrite	phosphate
Summer 1998	13.5	8.1	425	98.0	3.7	5.09	0.0002	0.0000
Autumn 1998	11.4	8.3	421	96.0	2.3	8.69	0.0090	0.1290
Winter 1998	9.1	7.9	403	101.0	4.9	4.94	0.0000	0.0000
Spring 1999	10.2	8.1	400	94.1	4.7	2.45	0.0000	0.0000
Summer 1999	*15.3*	*8.3*	*438*	*107.0*	*4.5*	*7.22*	*0.0000*	*0.0000*
Autumn 1999	10.8	8.1	580	93.5	3.5	7.88	0.0000	0.0000
Winter 1999	7.3	8.3	590	102.2	7.1	4.71	0.0000	0.0000
Spring 2000	12.7	8.0	470	91.0	2.2	1.88	0.0000	0.0000

Site 4

DATE	temperature	pH	conductivity	dissolved oxygen	SPM	nitrate	nitrite	phosphate
Summer 1998	13.8	8.2	423	100.0	0.8	5.05	0.0000	0.0000
Autumn 1998	11.7	8.3	419	96.0	4.1	5.98	0.0000	0.0000
Winter 1998	9.2	7.9	399	99.0	10.3	4.50	0.0000	0.0000
Spring 1999	10.3	8.1	394	95.6	7.7	3.79	0.0000	0.0000
Summer 1999	14.8	8.3	425	108.0	1.7	6.96	0.0000	0.0000
Autumn 1999	11.0	8.3	568	94.3	5.1	7.26	0.0000	0.0000
Winter 1999	7.1	8.2	583	101.9	9.9	4.82	0.0000	0.0000
Spring 2000	13.5	8.4	470	92.0	2.9	2.34	0.0000	0.0000

Site 5

DATE	temperature	pH	conductivity	dissolved oxygen	SPM	nitrate	nitrite	phosphate
Summer 1998	14.9	7.9	418	79.0	2.3	4.23	0.0000	0.0744
Autumn 1998	11.2	8.1	437	85.0	4.7	8.66	0.0727	0.1652
Winter 1998	8.3	7.7	393	96.0	10.3	4.60	0.0000	0.0000
Spring 1999	10.9	8.1	389	86.7	9.1	3.45	0.0000	0.0059
Summer 1999	16.4	8.2	414	82.0	7.0	5.70	0.0000	0.0000
Autumn 1999	11.9	8.1	551	80.4	7.8	5.44	0.0000	0.0000
Winter 1999	7.0	8.2	555	100.7	17.3	10.71	0.0072	0.0000
Spring 2000	15.4	8.5	510	89.8	6.1	7.84	0.0000	0.0000

APPENDIX D Cont.

FROME Site 1

DATE	temperature	pH	conductivity	dissolved oxygen	SPM	nitrate	nitrite	phosphate
Summer 1998	13.4	8.2	400	90.0	3.3	4.49	0.0000	0.0000
Autumn 1998	10.5	8.0	298	92.0	5.8	4.79	0.0000	0.0000
Winter 1998	9.9	7.9	354	98.0	3.0	2.84	0.0000	0.0000
Spring 1999	9.5	8.0	279	94.6	5.3	3.62	0.0024	0.0061
Summer 1999	14.9	8.5	452	122.0	2.7	5.03	0.0016	0.0088
Autumn 1999	10.0	8.4	525	98.0	3.5	4.14	0.0000	0.0000
Winter 1999	6.1	7.4	391	96.2	8.7	1.55	0.0035	0.0022
Spring 2000	10.8	8.6	460	105.2	4.7	7.20	0.0048	0.0000

Site 2

DATE	temperature	pH	conductivity	dissolved oxygen	SPM	nitrate	nitrite	phosphate
Summer 1998	13.9	8.4	400	117.0	15.1	2.37	0.0000	0.0000
Autumn 1998	10.4	8.4	342	103.0	8.5	7.09	0.0000	0.0000
Winter 1998	10.6	7.9	380	98.0	6.1	5.80	0.0044	0.0085
Spring 1999	10.6	8.3	335	108.0	6.7	2.80	0.0000	0.0000
Summer 1999	16.1	8.4	443	115.0	2.7	7.06	0.0095	0.0065
Autumn 1999	9.9	8.6	533	101.7	4.7	6.41	0.0007	0.0023
Winter 1999	6.2	7.5	472	97.1	19.7	5.94	0.0000	0.0000
Spring 2000	11.2	8.6	460	108.3	4.1	2.99	0.0000	0.0000

Site 3

DATE	temperature	pH	conductivity	dissolved oxygen	SPM	nitrate	nitrite	phosphate
Summer 1998	14.5	8.2	430	114.0	2.1	7.70	0.0047	0.0296
Autumn 1998	10.6	8.1	362	99.0	8.5	6.48	0.0002	0.0000
Winter 1998	10.2	7.6	400	97.0	9.5	4.69	0.0000	0.0000
Spring 1999	10.9	8.0	364	105.4	9.9	6.31	0.0015	0.0076
Summer 1999	*16.5*	*8.2*	*462*	*107.0*	2.9	*7.79*	*0.0000*	*0.0000*
Autumn 1999	10.4	8.5	560	102.1	5.5	9.09	0.0015	0.0000
Winter 1999	6.6	7.4	504	94.0	22.7	2.31	0.0000	0.0000
Spring 2000	11.8	8.6	500	107.1	2.7	2.40	0.0000	0.0000

Site 4

DATE	temperature	pH	conductivity	dissolved oxygen	SPM	nitrate	nitrite	phosphate
Summer 1998	16.6	8.2	440	102.0	4.1	3.04	0.0000	0.0000
Autumn 1998	10.9	8.1	359	87.0	13.7	6.62	0.0043	0.0310
Winter 1998	10.1	7.7	395	92.0	16.0	5.24	0.0000	0.0000
Spring 1999	11.1	8.0	351	99.8	11.3	3.05	0.0000	0.0000
Summer 1999	17.6	8.2	468	95.0	2.4	0.43	0.0000	0.0000
Autumn 1999	10.2	8.4	552	92.3	7.3	1.95	0.0056	0.0027
Winter 1999	6.2	7.4	467	90.6	36.9	3.02	0.0000	0.0000
Spring 2000	12.3	8.4	480	96.4	4.1	4.12	0.0000	0.0000

Site 5

DATE	temperature	pH	conductivity	dissolved oxygen	SPM	nitrate	nitrite	phosphate
Summer 1998	15.6	8.1	420	93.0	1.9	3.31	0.0000	0.0000
Autumn 1998	11.1	8.0	341	78.0	55.5	4.12	0.0009	0.0281
Winter 1998	9.9	7.5	384	87.0	7.8	4.94	0.0000	0.0000
Spring 1999	11.2	7.9	315	88.6	14.3	2.50	0.0000	0.0000
Summer 1999	18.9	8.3	498	86.0	10.3	4.45	0.0024	0.0130
Autumn 1999	10.3	8.3	536	90.1	7.9	7.06	0.0020	0.0060
Winter 1999	6.5	7.5	476	87.6	36.3	6.25	0.0000	0.0000
Spring 2000	13.0	8.6	470	88.2	8.9	3.42	0.0000	0.0036

APPENDIX D Cont.

AVON Site 1

DATE	temperature	pH	conductivity	dissolved oxygen	SPM	nitrate	nitrite	phosphate
Summer 1998	13.9	8.2	510	112.0	3.5	2.41	0.0000	0.0000
Autumn 1998	11.9	8.1	497	91.0	12.3	1.76	0.0000	0.0000
Winter 1998	9.2	7.5	451	91.0	13.7	4.44	0.0000	0.0000
Spring 1999	10.9	8.0	461	106.4	8.5	7.00	0.0000	0.0000
Summer 1999	16.2	8.3	540	125.0	4.7	10.71	0.0000	0.0000
Autumn 1999	10.1	8.3	694	97.2	5.5	10.14	0.0000	0.0000
Winter 1999	8.3	7.4	660	95.9	12.1	4.77	0.0000	0.0000
Spring 2000	11.9	8.1	500	102.4	5.3	3.21	0.0000	0.0000

Site 2

DATE	temperature	pH	conductivity	dissolved oxygen	SPM	nitrate	nitrite	phosphate
Summer 1998	14.9	8.2	460	116.0	1.7	2.46	0.0000	0.0000
Autumn 1998	11.7	8.3	446	108.0	2.1	7.63	0.0000	0.0000
Winter 1998	9.4	7.6	401	95.0	12.5	4.03	0.0000	0.0000
Spring 1999	11.7	8.2	414	122.7	10.0	6.78	0.0000	0.0062
Summer 1999	17.5	8.6	482	147.0	3.0	5.93	0.0000	0.0000
Autumn 1999	10.5	8.4	611	101.4	3.3	8.99	0.0187	0.0361
Winter 1999	8.5	7.6	585	100.3	12.7	3.62	0.0000	0.0000
Spring 2000	12.9	8.3	460	103.9	5.1	3.37	0.0000	0.0000

Site 3

DATE	temperature	pH	conductivity	dissolved oxygen	SPM	nitrate	nitrite	phosphate
Summer 1998	16.0	8.0	450	92.0	0.9	6.26	0.0000	0.0301
Autumn 1998	*10.6*	*8.0*	*384*	*73.0*	*12.1*	*7.17*	*0.0123*	*0.0403*
Winter 1998	9.8	7.6	396	82.0	11.3	4.30	0.0000	0.0000
Spring 1999	11.3	7.9	392	91.9	11.4	3.58	0.0000	0.0000
Summer 1999	18.0	8.1	477	80.0	2.8	6.91	0.0000	0.0000
Autumn 1999	9.8	8.4	568	76.6	4.7	7.80	0.0000	0.0000
Winter 1999	6.5	7.8	568	90.9	12.1	3.06	0.0000	0.0000
Spring 2000	11.8	8.7	560	87.0	8.2	3.45	0.0000	0.0000

Site 4

DATE	temperature	pH	conductivity	dissolved oxygen	SPM	nitrate	nitrite	phosphate
Summer 1998	16.6	8.2	460	111.0	4.1	2.97	0.0000	0.0000
Autumn 1998	10.9	8.1	387	84.0	-0.9	3.27	0.0000	0.0000
Winter 1998	9.7	7.8	396	89.0	7.8	4.41	0.0000	0.0000
Spring 1999	11.9	8.1	392	106.4	10.9	6.90	0.0000	0.0067
Summer 1999	18.7	8.3	486	90.0	2.5	6.21	0.0088	0.0384
Autumn 1999	9.9	8.5	585	89.1	4.0	8.70	0.0000	0.0000
Winter 1999	6.5	7.8	569	94.2	16.6	4.13	0.0000	0.0042
Spring 2000	12.3	8.7	550	90.1	7.2.	3.58	0.0000	0.0000

DATE	temperature	pH	conductivity	dissolved oxygen	SPM	nitrate	nitrite	phosphate
Summer 1998	16.6	8.2	440	102.0	2.9	5.60	0.0000	0.0990
Autumn 1998	11.1	8.0	370	79.0	10.3	9.46	0.0068	0.1207
Winter 1998	9.4	7.7	388	86.0	11.0	4.41	0.0000	0.0000
Spring 1999	11.6	8.0	357	101.8	9.7	2.93	0.0000	0.0000
Summer 1999	19.8	8.3	485	88.0	3.9	5.86	0.0000	0.0000
Autumn 1999	10.3	8.4	558	87.7	5.9	2.21	0.0000	0.0000
Winter 1999	6.5	7.6	519	87.8	12.4	3.37	0.0000	0.0000
Spring 2000	13.1	8.6	530	87.0	7.0	1.33	0.0000	0.0000

APPENDIX E
BIOLOGICAL MONITORING: SUMMER 1998 – SPRING 2000

	FCsum98	FCat98	FCwin98	FCsp99	FCsum99	FCat99	FCwin99	FCsp00	MCsum98	MCat98	MCwin98	MCsp99	MCsum99	MCat99	MCwin99	MCsp00
Agapetus fuscipes	0	0	1	0	11	0	3	0	0	0	0	0	6	4	9	0
Calopteryx splendens	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Anaboluia nervosa	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ancylus fluviatilis	0	0	0	0	0	2	3	0	0	0	3	3	17	27	11	12
Aphelocheirus aestivalis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Asellus aquaticus	1	13	3	0	0	2	0	0	36	5	0	0	2	1	1	0
Atherix sp.	0	0	0	0	0	3	1	0	0	0	0	0	0	0	0	0
Athripsodes albifrons	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Athripsodes sp.	3	1	0	0	2	0	0	3	0	0	0	0	0	0	0	0
Baetis sp.	100	17	20	0	369	28	19	110	82	5	3	41	16	25	7	21
Bdellocephala punctata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Bithynia leachi	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Brachycentrus subnubilis	35	0	1	0	2	5	20	1	0	0	0	0	0	0	0	0
Caenis sp.	1	41	3	0	3	0	0	29	0	0	0	0	0	0	0	0
Chironomidae	97	43	16	0	24	9	11	43	6	0	11	63	40	4	3	301
Chloroperla sp.	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
Colymbetes sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Crunoecia irrorata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Culicoides sp.	2	0	0	0	0	0	1	0	0	0	1	2	0	0	0	0
Curculionidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cymatia bonsdorffii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Dendrocoelum lacteum	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Dicranota sp.	2	7	5	0	5	3	0	7	1	2	1	5	3	0	2	29
Dixid sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Donacia sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Drusus annulatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Dugesia polychroa	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

APPENDIX E Cont.

	FCsum98	FCat98	FCwin98	FCsp99	FCsum99	FCat99	FCwin99	FCsp00	MCsum98	MCat98	MCwin98	MCsp99	MCsum99	MCat99	MCwin99	MCsp00
Dugesia tigrinia	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ecdyonurus torrentis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ecdyonurus venosus	0	0	7	0	0	0	0	0	0	0	0	0	0	0	0	0
Elmis aenea	10	0	0	0	10	0	6	0	3	4	0	5	11	27	5	1
Empididae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ephemera danica	0	47	31	0	5	5	10	9	0	3	0	0	0	1	1	0
Ephemerella ignita	142	1	0	0	47	3	0	48	0	0	0	3	0	0	0	0
Ephemerella notata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Erpobdella octoculata	1	4	4	0	0	0	0	2	2	5	0	0	2	3	1	2
Esolus parallelepipedus	0	0	0	0	74	0	0	0	0	0	0	0	1	3	0	0
Gammarus sp.	100	160	119	0	799	215	147	280	120	88	51	211	88	137	113	325
Gerris sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Glossiphonia complanata	3	7	7	0	8	4	5	0	4	10	3	0	5	3	0	0
Goera pilosa	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gyrinus natator	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Halesus radiatus	0	0	0	0	0	0	0	1	0	0	0	0	0	2	1	0
Haliplus sp.	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0
Helobdella stagnalis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0
Heptagenia sulphurea	1	9	0	0	0	7	4	3	0	0	0	0	0	0	0	0
Hydracarina	1	0	0	0	4	0	0	4	1	0	0	3	4	0	0	0
Hydropsyche pellucidula	4	20	4	0	23	20	4	24	0	0	0	0	0	1	0	3
Hydropsyche siltalai	0	0	0	0	0	0	0	10	0	8	1	3	1	10	9	13
Hydroptila sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hyphydrus ovatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Isoperla grammatica	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Lepidostoma hirtum	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Leptophlebiidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Leuctra geniculata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Leuctra hippopus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Limnephilus lunatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

APPENDIX E Cont.

	FCsum98	FCat98	FCwin98	FCsp99	FCsum99	FCat99	FCwin99	FCsp00	MCsum98	MCat98	MCwin98	MCsp99	MCsum99	MCat99	MCwin99	MCsp00
Limnephilus sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
Limnius volckmari	45	113	44	0	17	66	40	45	0	0	0	0	0	0	11	16
Lymnaea glabra	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lymnaea peregra	0	23	15	0	19	3	0	0	0	0	0	0	0	0	0	0
Lymnaea trunctula	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Melampophylax mucoreus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Mesovelia furcata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Muscidae	0	3	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Mystacides azurea	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nemoura sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Odontocerum albicorne	4	7	4	0	3	7	4	3	0	4	0	0	0	0	0	0
Oligochaeta	86	19	47	0	52	65	75	112	16	41	53	33	76	107	49	17
Orectchilus villosus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Oulimnius sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Philopotamidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Physa fontinalis	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Piscicola geometra	1	1	0	0	2	5	0	0	0	0	0	0	0	0	0	0
Planorbis albus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Planorbis carinatus	0	0	0	0	0	0	1	0	0	1	0	0	0	0	0	0
Planorbis contortus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Planorbis planorbis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Planorbis vortex	0	0	0	0	0	1	0	0	0	0	0	0	0	2	1	0
Platambus sp.	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0
Polycelus tenuis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Polycentropus flavomaculatus	0	1	0	0	0	0	0	0	0	0	0	0	1	0	0	0
Potamophylax latipennis	3	0	1	0	0	2	0	0	0	0	0	0	0	0	0	0
Potamopyrgus jenkinsi	60	5	20	0	180	36	1	3	15	7	2	40	149	31	21	573
Psychomyia pusilla	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Rhyacophila dorsalis	2	3	1	0	4	5	1	4	0	0	0	0	0	0	0	0
Rithrogena semicolorata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

APPENDIX E Cont.

	FCsum98	FCat98	FCwin98	FCsp99	FCsum99	FCat99	FCwin99	FCsp00	MCsum98	MCat98	MCwin98	MCsp99	MCsum99	MCat99	MCwin99	MCsp00
Sericostoma personatum	0	3	11	0	0	5	1	4	0	0	0	0	0	0	0	0
Sialis sp.	0	0	0	0	0	0	0	0	0	3	0	0	1	2	0	0
Sigara sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Silo pallipes	0	1	1	0	0	3	1	0	0	0	0	0	0	0	0	0
Simuliidae	513	7	161	0	284	63	71	1064	215	0	0	0	0	0	0	1
Sphaeriidae	1	0	2	0	0	0	1	1	0	0	1	3	13	9	4	47
Theodoxus fluviatilis	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0
Theromyzon tessulatum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tinodes unicolor	1	0	0	0	0	0	0	0	0	0	0	0	4	4	1	0
Tipula sp.	0	0	0	0	0	0	7	1	0	0	0	0	0	0	0	0
Valvata cristata	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Valvata macrostoma	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Valvata piscinalis	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0
Viviparus viviparus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

	ACsum98	ACat98	ACwin98	ACsp99	ACsum99	ACat99	ACwin99	ACsp00	TCsum98	TCat98	TCwin98	TCsp99	TCsum99	TCat99	TCwin99	TCsp00
Agapetus fuscipes	0	0	0	0	1	0	3	1	52	117	72	76	48	61	7	164
Calopteryx splendens	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Anaboluia nervosa	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ancylus fluviatilis	0	1	6	0	1	1	1	4	0	0	4	1	4	2	7	5
Aphelocheirus aestivalis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Asellus aquaticus	1	0	0	0	4	4	0	0	1	1	3	0	0	4	1	0
Atherix sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Athripsodes albifrons	4	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Athripsodes sp.	0	0	0	0	0	0	0	1	2	0	0	7	0	1	0	1
Baetis sp.	82	24	25	0	153	17	8	39	56	2	51	1	273	39	28	336
Bdellocephala punctata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Bithynia leachi	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Brachycentrus subnubilis	11	3	5	0	11	1	1	0	0	0	0	0	0	0	0	0
Caenis sp.	0	0	0	0	0	5	1	3	11	0	0	21	0	0	0	184

APPENDIX E Cont.

	ACsum98	ACat98	ACwin98	ACsp99	ACsum99	ACat99	ACwin99	ACsp00	TCsum98	TCat98	TCwin98	TCsp99	TCsum99	TCat99	TCwin99	TCsp00
Chironomidae	4	11	27	0	45	28	0	13	9	3	3	13	47	5	1	11
Chloroperla sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Colymbetes sp.	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Crunoecia irrorata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Culicoides sp.	0	0	0	0	0	0	0	0	0	0	1	3	0	0	0	3
Curculionidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cymatia bonsdorffii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Dendrocoelum lacteum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Dicranota sp.	2	5	0	0	8	3	4	0	4	3	3	3	4	3	1	1
Dixid sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Donacia sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Drusus annulatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2
Dugesia polychroa	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Dugesia tigrinia	0	1	0	0	1	3	2	0	0	0	0	0	0	0	0	0
Ecdyonurus torrentis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ecdyonurus venosus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Elmis aenea	66	47	45	0	40	0	0	26	7	13	7	9	8	7	3	16
Empididae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ephemera danica	8	0	8	0	0	6	3	4	0	0	0	0	0	0	0	0
Ephemerella ignita	286	0	0	0	128	27	1	4	487	0	4	4	77	0	0	203
Ephemerella notata	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Erpobdella octoculata	1	0	0	0	0	4	4	0	0	0	0	0	0	0	0	0
Esolus parallelepipedus	0	0	0	0	0	23	0	0	0	0	0	0	0	0	0	0
Gammarus sp.	1294	189	123	0	108	116	303	375	271	321	327	111	316	487	393	323
Gerris sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Glossiphonia complanata	0	0	7	0	2	1	2	0	0	2	4	3	0	4	1	0
Goera pilosa	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gyrinus natator	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Halesus radiatus	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
Haliplus sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

APPENDIX E Cont.

	ACsum98	ACat98	ACwin98	ACsp99	ACsum99	ACat99	ACwin99	ACsp00	TCsum98	TCat98	TCwin98	TCsp99	TCsum99	TCat99	TCwin99	TCsp00
Helobdella stagnalis	0	0	0	0	0	1	0	0	0	0	0	3	0	1	0	1
Heptagenia sulphurea	3	0	0	0	0	1	3	1	0	0	0	0	0	0	0	0
Hydracarina	0	0	0	0	0	0	0	1	0	0	0	0	3	0	0	0
Hydropsyche pellucidula	3	0	1	0	1	4	0	0	0	0	0	0	0	0	0	0
Hydropsyche siltalai	13	2	3	0	0	5	61	9	0	0	0	0	0	0	0	0
Hydroptila sp.	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
Hyphydrus ovatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Isoperla grammatica	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lepidostoma hirtum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Leptophlebiidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Leuctra geniculata	0	0	0	0	7	0	0	0	0	0	0	0	0	0	0	0
Leuctra hippopus	0	0	0	0	0	0	0	0	7	0	0	44	11	0	0	15
Limnephilus lunatus	0	0	0	0	0	0	0	0	4	0	3	0	1	0	0	2
Limnephilus sp.	0	0	0	0	0	0	1	0	5	0	0	0	0	0	0	1
Limnius volckmari	28	20	19	0	19	29	36	17	0	0	0	0	0	0	0	0
Lymnaea glabra	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lymnaea peregra	0	0	0	0	3	0	1	0	2	0	0	0	0	0	0	0
Lymnaea trunctula	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0
Melampophylax mucoreus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Mesovelia furcata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Muscidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Mystacides azurea	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nemoura sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
Odontocerum albicorne	1	0	1	0	0	5	1	3	0	0	0	0	0	0	0	0
Oligochaeta	14	19	32	0	16	45	25	17	11	19	43	33	33	85	44	8
Orectchilus villosus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Oulimnius sp.	0	0	0	0	0	0	15	0	0	0	0	0	0	0	0	0
Philopotamidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Physa fontinalis	1	25	2	0	6	25	5	0	3	0	0	0	0	0	2	0
Piscicola geometra	0	0	0	0	1	0	0	0	1	0	1	0	0	1	0	0

APPENDIX E Cont.

	ACsum98	ACat98	ACwin98	ACsp99	ACsum99	ACat99	ACwin99	ACsp00	TCsum98	TCat98	TCwin98	TCsp99	TCsum99	TCat99	TCwin99	TCsp00
Planorbis albus	0	0	0	0	0	0	0	0	0	0	0	0	0	4	0	0
Planorbis carinatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Planorbis contortus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3
Planorbis planorbis	0	0	0	0	0	2	0	0	0	0	0	0	0	0	1	0
Planorbis vortex	1	2	3	0	0	11	7	1	1	3	0	3	3	4	1	0
Platambus sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Polycelus tenuis	0	0	1	0	0	0	1	0	0	0	0	1	0	0	0	0
Polycentropus flavomaculatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Potamophylax latipennis	7	0	1	0	0	0	0	4	12	0	1	0	1	0	0	0
Potamopyrgus jenkinsi	4	1	3	0	0	0	0	1	0	0	0	0	0	0	0	0
Psychomyia pusilla	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Rhyacophila dorsalis	3	1	2	0	3	6	8	3	3	3	4	1	2	2	3	4
Rithrogena semicolorata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Sericostoma personatum	3	0	0	0	3	0	3	2	0	0	0	0	3	0	0	0
Sialis sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Sigara sp.	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0
Silo pallipes	0	0	0	0	0	0	2	0	2	6	8	5	0	9	8	2
Simuliidae	348	115	21	0	155	35	284	673	11	1	1	0	11	5	0	16
Sphaeriidae	3	7	1	0	89	71	8	8	0	0	0	2	1	1	0	0
Theodoxus fluviatilis	5	3	8	0	3	1	1	3	0	0	0	0	0	0	0	0
Theromyzon tessulatum	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0
Tinodes unicolor	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0
Tipula sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Valvata cristata	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Valvata macrostoma	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Valvata piscinalis	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Viviparus viviparus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

APPENDIX E Cont.

	ICsum98	ICat98	ICwin98	ICsp99	ICsum99	ICat99	ICwin99	ICsp00	FOWsum98	FOWat98	FOWwin98	FOWsp99	FOWsum99	FOWat99	FOWwin99	FOWsp00
Agapetus fuscipes	1	0	1	0	7	0	4	15	3	8	5	0	12	9	7	0
Calopteryx splendens	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Anaboluia nervosa	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ancylus fluviatilis	1	1	0	0	12	2	5	7	1	0	0	0	0	0	0	0
Aphelocheirus aestivalis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Asellus aquaticus	7	39	8	15	1	16	4	6	0	7	3	0	0	2	0	0
Atherix sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
Athripsodes albifrons	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Athripsodes sp.	1	0	0	7	2	0	0	3	11	0	0	0	12	0	0	13
Baetis sp.	517	4	27	21	43	4	69	107	52	63	25	0	23	5	27	5
Bdellocephala punctata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Bithynia leachi	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Brachycentrus subnubilis	0	0	0	0	0	0	0	0	1	1	4	0	4	1	3	0
Caenis sp.	21	0	0	4	0	0	0	66	5	0	0	0	0	0	0	9
Chironomidae	13	3	0	53	3	1	5	0	51	45	32	0	76	15	23	35
Chloroperla sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Colymbetes sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Crunoecia irrorata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3
Culicoides sp.	0	1	4	0	1	0	0	1	0	0	0	0	0	0	0	0
Curculionidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cymatia bonsdorffii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Dendrocoelum lacteum	0	0	0	0	0	3	0	0	0	0	0	0	0	1	0	0
Dicranota sp.	2	1	1	1	12	1	2	3	7	12	4	0	13	4	2	1
Dixid sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Donacia sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Drusus annulatus	1	0	4	0	1	0	1	3	0	0	0	0	0	0	0	0
Dugesia polychroa	0	8	0	0	0	1	1	0	0	0	0	0	0	0	0	0
Dugesia tigrinia	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ecdyonurus torrentis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ecdyonurus venosus	0	0	0	0	0	0	0	0	0	0	13	0	0	0	0	0

APPENDIX E Cont.

	ICsum98	ICat98	ICwin98	ICsp99	ICsum99	ICat99	ICwin99	ICsp00	FOWsum98	FOWat98	FOWwin98	FOWsp99	FOWsum99	FOWat99	FOWwin99	FOWsp00
Eimis aenea	5	1	3	3	9	0	12	55	4	5	4	0	1	7	4	8
Empididae	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0	0
Ephemera danica	0	0	0	0	0	0	0	0	0	9	4	0	2	3	2	2
Ephemerella ignita	288	4	2	11	20	0	1	19	97	1	0	0	15	1	1	20
Ephemerella notata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Erpobdella octoculata	4	7	1	0	2	2	4	5	1	8	3	0	1	1	2	2
Esolus parallelepipedus	1	2	0	0	11	3	0	0	0	0	0	0	8	0	0	0
Gammarus sp.	213	1020	185	24	128	224	267	224	131	79	143	0	199	131	68	80
Gerris sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Glossiphonia complanata	2	13	0	0	0	5	0	2	4	3	2	0	0	0	0	0
Goera pilosa	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0
Gyrinus natator	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Halesus radiatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Haliplus sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Helobdella stagnalis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Heptagenia suiphurea	0	0	0	0	0	0	0	0	0	1	0	0	0	1	5	3
Hydracarina	0	0	0	0	0	0	0	3	0	1	0	0	0	0	0	5
Hydropsyche pellucidula	0	0	0	0	0	0	0	0	0	3	3	0	7	3	2	2
Hydropsyche siltalai	4	0	1	0	0	0	0	0	0	1	0	0	0	0	0	0
Hydroptila sp.	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Hyphydrus ovatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Isoperla grammatica	1	0	1	0	0	0	0	3	1	0	1	0	0	0	0	0
Lepidostoma hirtum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2
Leptophlebiidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Leuctra geniculata	0	0	0	0	0	0	0	4	0	0	0	0	1	0	0	2
Leuctra hippopus	0	0	0	6	4	0	0	17	0	0	0	0	0	0	0	0
Limnephilus lunatus	1	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Limnephilus sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Limnius volckmari	1	0	0	0	1	0	0	3	8	10	7	0	8	23	10	18
Lymnaea glabra	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

APPENDIX E Cont.

	ICsum98	ICat98	ICwin98	ICsp99	ICsum99	ICat99	ICwin99	ICsp00	FOWsum98	FOWat98	FOWwin98	FOWsp99	FOWsum99	FOWat99	FOWwin99	FOWsp00
Lymnaea peregra	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lymnaea trunctula	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Melampophylax mucoraus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Mesovelia furcata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Muscidae	0	0	1	0	0	0	0	0	0	3	0	0	0	0	0	0
Mystacides azurea	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nemoura sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Odontocerum albicorne	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Oligochaeta	9	85	21	16	44	40	7	4	67	21	50	0	31	31	15	39
Orectchilus villosus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Oulimnius sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Philopotamidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Physa fontinalis	1	5	8	0	2	2	3	0	0	0	0	0	0	0	1	0
Piscicola geometra	1	0	1	0	0	0	1	0	0	0	0	0	0	1	0	0
Planorbis albus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Planorbis carinatus	1	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0
Planorbis contortus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Planorbis planorbis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Planorbis vortex	0	0	0	0	0	1	0	1	0	0	1	0	0	0	0	0
Platambus sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Polycelus tenuis	0	9	0	1	5	11	15	0	0	0	1	0	0	0	0	0
Polycentropus flavomaculatus	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0
Potamophylax latipennis	4	1	8	0	3	0	0	2	0	0	0	0	0	0	0	0
Potamopyrgus jenkinsi	0	0	0	0	0	0	0	0	1	33	4	0	0	0	1	0
Psychomyia pusilla	0	0	0	0	0	0	0	0	0	0	0	0	9	0	1	1
Rhyacophila dorsalis	3	8	4	7	6	2	2	2	6	2	1	0	0	2	2	0
Rithrogena semicoloreta	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Sericostoma personatum	4	3	1	1	0	4	0	1	0	0	0	0	0	1	3	0
Sialis sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Sigara sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

APPENDIX E Cont.

	ICsum98	ICat98	ICwin98	ICsp99	ICsum99	ICat99	ICwin99	ICsp00	FOWsum98	FOWat98	FOWwin98	FOWsp99	FOWsum99	FOWat99	FOWwin99	FOWsp00
Silo pallipes	0	1	3	1	3	4	1	0	0	2	0	0	0	0	0	0
Simuliidae	127	1	19	3	55	1	0	0	3	7	9	0	0	0	1	0
Sphaeriidae	0	0	0	0	0	0	1	0	0	1	0	0	4	0	1	0
Theodoxus fluviatilis	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0
Theromyzon tessulatum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tinodes unicolor	0	0	0	0	0	0	0	0	41	0	0	0	0	1	0	0
Tipula sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	1	2	0
Valvata cristata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Valvata macrostoma	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Valvata piscinalis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Viviparus viviparus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

	MOWsum98	MOWat98	MOWwin98	MOWsp99	MOWsum99	MOWat99	MOWwin99	MOWsp00	AOWsum98	AOWat98	AOWwin98	AOWsp99	AOWsum99
Agapetus fuscipes	0	3	1	4	1	12	0	0	0	0	0	0	3
Calopteryx splendens	0	0	0	0	0	0	0	0	0	0	0	0	0
Anaboluia nervosa	0	0	0	0	0	0	0	0	0	0	0	0	0
Ancylus fluviatilis	0	0	0	0	0	0	0	0	0	0	0	0	0
Aphelocheirus aestivalis	0	0	0	0	0	0	0	0	0	0	0	0	12
Asellus aquaticus	0	3	0	0	0	0	0	0	0	0	0	0	0
Atherix sp.	0	0	0	0	0	0	0	0	0	0	2	0	3
Athripsodes albifrons	0	0	0	0	0	0	0	0	0	0	0	0	0
Athripsodes sp.	0	0	1	0	0	2	0	3	10	0	0	0	0
Baetis sp.	55	8	8	67	40	47	136	140	7	3	8	0	92
Bdellocephala punctata	0	0	0	0	0	0	0	0	0	0	0	0	0
Bithynia leachi	0	0	0	0	0	0	0	0	0	0	0	0	0
Brachycentrus subnubilis	0	0	0	0	0	0	0	0	0	0	0	0	25
Caenis sp.	0	0	0	3	1	0	0	13	1	1	0	0	0
Chironomidae	17	5	1	192	25	1	1	126	1	2	2	0	44
Chloroperla sp.	0	0	0	0	0	0	0	0	1	0	0	0	0
Colymbetes sp.	0	0	0	0	0	0	0	0	0	0	0	0	0

APPENDIX E Cont.

	MOWsum98	MOWat98	MOWwin98	MOWsp99	MOWsum99	MOWat99	MOWwin99	MOWsp00	AOWsum98	AOWat98	AOWwin98	AOWsp99	AOWsum99
Crunoecia irrorata	0	0	0	0	0	0	0	1	0	0	0	0	0
Culicoides sp.	0	0	0	1	0	1	0	1	0	0	0	0	0
Curculionidae	0	0	0	0	0	0	0	0	0	0	0	0	0
Cymatia bonsdorffii	0	0	0	0	0	0	0	0	0	0	0	0	0
Dendrocoelum lacteum	0	0	0	0	0	2	0	0	0	0	0	0	7
Dicranota sp.	19	16	7	7	27	37	5	2	6	3	2	0	9
Dixid sp.	0	0	0	0	0	0	0	0	0	0	0	0	0
Donacia sp.	0	0	0	0	0	0	0	0	0	0	0	0	0
Drusus annulatus	0	0	0	0	0	0	0	0	0	0	0	0	0
Dugesia polychroa	0	0	0	0	0	0	0	0	0	0	0	0	0
Dugesia tigrinia	0	0	0	0	0	0	0	0	0	2	0	0	0
Ecdyonurus torrentis	0	0	0	0	0	0	0	0	0	0	0	0	0
Ecdyonurus venosus	0	0	4	0	0	0	0	0	0	0	0	0	0
Elmis aenea	15	9	7	5	9	39	15	31	66	71	10	0	23
Empididae	0	3	0	1	0	0	0	0	0	0	0	0	0
Ephemera danica	0	32	8	1	3	28	3	4	8	0	1	0	0
Ephemerella ignita	111	0	0	16	7	0	1	13	60	0	0	0	63
Ephemerella notata	0	0	0	0	0	0	0	0	0	0	0	0	0
Erpobdella octoculata	0	0	1	0	2	3	1	1	3	2	3	0	0
Esolus parallelepipedus	0	0	0	0	0	0	0	0	0	0	0	0	11
Gammarus sp.	565	624	105	99	492	892	568	372	283	104	77	0	544
Gerris sp.	0	0	0	0	0	0	0	0	0	0	0	0	0
Glossiphonia complanata	5	7	2	1	2	0	0	1	0	3	1	0	3
Goera pilosa	0	0	0	0	0	0	0	0	0	0	0	0	0
Gyrinus natator	0	0	0	0	0	1	0	0	0	0	0	0	0
Halesus radiatus	0	0	0	0	0	0	0	0	0	0	0	0	0
Haliplus sp.	0	0	0	0	0	0	0	0	0	0	0	0	1
Helobdella stagnalis	0	0	0	0	0	4	0	0	0	0	0	0	0
Heptagenia sulphurea	9	3	0	0	0	12	1	2	9	0	0	0	0
Hydracarina	0	0	0	0	0	0	0	0	0	0	0	0	0

APPENDIX E Cont.

	MOWsum98	MOWat98	MOWwin98	MOWsp99	MOWsum99	MOWat99	MOWwin99	MOWsp00	AOWsum98	AOWat98	AOWwin98	AOWsp99	AOWsum99
Hydropsyche pellucidula	0	1	0	0	0	1	0	0	0	0	0	0	1
Hydropsyche siltalai	2	2	2	2	4	7	4	3	0	0	2	0	0
Hydroptila sp.	0	0	0	0	0	0	0	0	0	0	0	0	0
Hyphydrus ovatus	0	0	0	0	0	0	0	0	0	0	0	0	0
Isoperla grammatica	0	0	1	0	0	0	0	0	0	0	0	0	0
Lepidostoma hirtum	0	1	0	0	0	0	0	0	0	0	0	0	0
Leptophlebiidae	0	0	0	0	0	0	0	0	0	0	0	0	0
Leuctra geniculata	0	0	0	0	0	0	0	0	0	0	0	0	2
Leuctra hippopus	1	0	0	0	0	0	0	1	0	0	0	0	12
Limnephilus lunatus	0	0	0	0	0	0	0	0	0	0	0	0	0
Limnephilus sp.	0	0	0	0	0	0	0	0	0	0	0	0	0
Limnius volckmari	18	8	5	5	48	21	5	4	28	30	24	0	22
Lymnaea glabra	0	0	0	0	0	0	0	0	0	0	0	0	0
Lymnaea peregra	0	0	0	0	0	0	0	0	0	0	0	0	0
Lymnaea trunctula	0	0	0	0	0	0	0	0	0	0	0	0	0
Melampophylax mucoreus	0	0	0	0	0	0	0	0	0	0	0	0	0
Mesovelia furcata	0	0	0	0	0	0	0	0	0	0	0	0	0
Muscidae	0	0	0	0	0	0	0	0	0	0	0	0	0
Mystacides azurea	0	0	0	0	0	0	0	0	0	0	0	0	0
Nemoura sp.	0	0	0	0	0	0	0	0	0	0	0	0	0
Odontocerum albicorne	2	5	3	1	0	5	7	0	0	1	1	0	0
Oligochaeta	7	37	11	35	60	68	56	12	14	17	21	0	29
Orectchilus villosus	0	0	0	0	0	0	0	0	0	0	0	0	0
Oulimnius sp.	0	0	0	0	0	0	0	0	0	0	0	0	0
Philopotamidae	0	0	0	0	0	0	0	0	0	0	0	0	0
Physa fontinalis	0	0	0	0	0	0	0	0	0	0	0	0	2
Piscicola geometra	0	0	0	0	0	0	0	0	0	0	0	0	0
Planorbis albus	0	0	0	0	0	0	0	0	0	0	0	0	0
Planorbis carinatus	0	0	0	0	0	0	0	0	0	0	0	0	0
Planorbis contortus	0	0	0	0	0	0	0	0	0	0	0	0	0

APPENDIX E Cont.

	MOWsum98	MOWat98	MOWwin98	MOWsp99	MOWsum99	MOWat99	MOWwin99	MOWsp00	AOWsum98	AOWat98	AOWwin98	AOWsp99	AOWsum99
Planorbis planorbis	0	0	0	0	0	0	0	0	0	0	0	0	0
Planorbis vortex	0	0	0	0	0	0	0	1	1	6	1	0	3
Platambus sp.	0	0	0	0	0	0	0	0	0	0	0	0	0
Polycelus tenuis	0	1	0	0	0	0	0	0	0	0	0	0	0
Polycentropus flavomaculatus	0	0	0	0	0	0	0	0	0	0	0	0	0
Potamophylax latipennis	0	0	5	0	0	0	4	0	0	0	0	0	0
Potamopyrgus jenkinsi	51	3	1	7	7	3	0	0	3	0	0	0	1
Psychomyia pusilla	0	0	0	0	0	0	0	0	0	0	0	0	0
Rhyacophila dorsalis	5	3	0	1	3	2	0	7	4	0	4	0	5
Rithrogena semicolorata	0	0	0	0	0	0	3	0	0	0	0	0	0
Sericostoma personatum	1	4	3	2	2	6	1	4	0	0	0	0	0
Sialis sp.	0	0	0	0	0	0	0	0	0	0	0	0	0
Sigara sp.	0	0	0	0	0	0	0	0	0	0	0	0	0
Silo pallipes	0	2	2	0	0	1	0	0	0	1	0	0	0
Simuliidae	3	2	0	1	0	0	0	0	9	4	3	0	60
Sphaeriidae	0	0	1	0	0	0	0	0	0	7	17	0	4
Theodoxus fluviatilis	0	0	0	0	0	0	0	0	2	0	1	0	2
Theromyzon tessulatum	0	0	0	0	0	0	0	0	0	0	0	0	0
Tinodes unicolor	1	0	0	0	0	1	0	0	4	0	0	0	0
Tipula sp.	0	0	0	0	0	0	8	0	0	0	0	0	0
Valvata cristata	0	0	0	0	0	0	0	0	0	0	0	0	0
Valvata macrostoma	0	0	0	0	0	0	0	0	0	0	0	0	0
Valvata piscinalis	0	0	0	0	0	0	0	0	0	0	0	0	0
Viviparus viviparus	0	0	0	0	0	0	0	0	0	0	0	0	0

	AOWat99	AOWwin99	AOWsp00	TOWsum98	TOWat98	TOWwin98	TOWsp99	TOWsum99	TOWat99	TOWwin99	TOWsp00	IOWsum98	IOWat98	IOWwin98
Agapetus fuscipes	3	2	3	0	22	13	28	15	0	23	11	0	0	33
Calopteryx splendens	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Anaboluia nervosa	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ancylus fluviatilis	10	8	3	0	0	0	1	0	0	0	0	0	1	0

APPENDIX E Cont.

	AOWat99	AOWwin99	AOWsp00	TOWsum98	TOWat98	TOWwin98	TOWsp99	TOWsum99	TOWat99	TOWwin99	TOWsp00	IOWsum98	IOWat98	IOWwin98
Aphelocheirus aestivalis	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Asellus aquaticus	2	2	0	4	1	4	0	1	7	4	0	455	88	68
Atherix sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Athripsodes albifrons	0	0	0	0	0	0	0	3	0	0	0	0	0	0
Athripsodes sp.	0	0	0	7	0	0	18	0	0	0	2	1	0	0
Baetis sp.	4	35	19	132	108	19	75	40	164	101	150	475	108	33
Bdellocephala punctata	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Bithynia leachi	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Brachycentrus subnubilis	0	9	0	0	0	0	0	0	0	0	0	0	0	0
Caenis sp.	2	1	3	5	0	1	0	0	0	0	5	64	0	1
Chironomidae	13	20	1	8	11	3	120	9	4	4	3	0	36	32
Chloroperla sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Colymbetes sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Crunoecia irrorata	0	0	0	0	0	0	0	0	0	0	3	2	0	0
Culicoides sp.	0	0	0	0	0	0	4	0	0	0	0	0	0	1
Curculionidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cymatia bonsdorffii	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Dendrocoelum lacteum	1	3	0	0	0	0	0	0	1	3	0	0	0	0
Dicranota sp.	3	5	2	2	6	5	3	1	4	2	3	0	0	0
Dixid sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Donacia sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Drusus annulatus	0	0	0	0	0	1	0	0	0	0	1	0	0	0
Dugesia polychroa	0	0	0	0	0	0	0	0	0	0	0	0	4	0
Dugesia tigrinia	15	5	0	0	0	0	0	0	0	0	0	0	0	0
Ecdyonurus torrentis	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ecdyonurus venosus	0	0	0	0	0	1	0	0	0	0	0	0	0	17
Elmis aenea	15	16	0	0	3	0	1	0	1	0	0	6	0	3
Empididae	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Ephemera danica	15	5	0	0	3	3	2	0	1	1	1	0	0	0
Ephemerella ignita	7	23	6	73	8	1	33	16	1	0	8	304	0	3

APPENDIX E Cont.

	AOWat99	AOWwin99	AOWsp00	TOWsum98	TOWat98	TOWwin98	TOWsp99	TOWsum99	TOWat99	TOWwin99	TOWsp00	IOWsum98	IOWat98	IOWwin98
Ephemerella notata	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Erpobdella octoculata	5	3	0	8	3	2	1	0	3	0	3	57	8	17
Esolus parallelepipedus	0	4	0	0	0	0	0	0	0	0	0	0	0	0
Gammarus sp.	147	256	232	17	72	167	35	81	99	107	120	116	23	7
Gerris sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Glossiphonia complanata	6	3	0	1	1	1	0	0	1	3	0	2	0	1
Goera pilosa	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gyrinus natator	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Halesus radiatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Haliplus sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Helobdella stagnalis	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Heptagenia sulphurea	1	0	0	1	0	0	0	0	0	4	3	3	4	0
Hydracarina	0	0	0	0	0	0	1	0	0	0	0	2	0	1
Hydropsyche pellucidula	2	1	1	0	0	0	0	0	2	1	0	0	0	0
Hydropsyche siltalai	13	5	4	0	0	6	0	0	3	2	2	0	0	0
Hydroptila sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hyphydrus ovatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Isoperla grammatica	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Lepidostoma hirtum	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Leptophlebiidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Leuctra geniculata	0	1	0	1	0	0	0	1	0	0	0	0	0	0
Leuctra hippopus	0	4	0	4	0	0	2	1	0	0	1	0	0	0
Limnephilus lunatus	0	0	0	0	1	1	0	1	0	0	0	0	0	0
Limnephilus sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Limnius volckmari	90	46	38	3	15	9	5	0	11	8	8	5	0	4
Lymnaea glabra	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lymnaea peregra	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lymnaea trunctula	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Melampophylax mucoreus	0	0	0	0	0	0	0	0	0	0	0	4	0	0
Mesovelia furcata	0	0	0	0	0	0	0	0	0	0	0	0	0	0

APPENDIX E Cont.

	AOWat99	AOWwin99	AOWsp00	TOWsum98	TOWat98	TOWwin98	TOWsp99	TOWsum99	TOWat99	TOWwin99	TOWsp00	IOWsum98	IOWat98	IOWwin98
Muscidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Mystacides azurea	0	0	0	0	0	0	0	1	0	0	0	0	0	0
Nemoura sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Odontocerum albicorne	5	2	0	0	0	8	2	2	2	1	0	2	0	0
Oligochaeta	33	28	16	2	9	12	11	16	24	16	3	17	24	79
Orectchilus villosus	0	0	0	0	0	0	0	0	0	1	0	0	0	0
Oulimnius sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Philopotamidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Physa fontinalis	0	1	0	0	0	2	0	0	4	0	0	0	17	4
Piscicola geometra	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Planorbis albus	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Planorbis carinatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Planorbis contortus	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Planorbis planorbis	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Planorbis vortex	2	2	0	0	0	0	0	0	0	0	0	1	5	8
Platambus sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Polycelus tenuis	1	1	0	0	0	0	1	2	6	0	0	0	0	20
Polycentropus flavomaculatus	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Potamophylax latipennis	0	0	0	0	0	0	0	0	0	0	0	5	0	3
Potamopyrgus jenkinsi	0	1	0	1	0	0	0	0	0	0	0	0	0	0
Psychomyia pusilla	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Rhyacophila dorsalis	0	3	0	1	5	2	2	0	1	3	7	1	2	3
Rithrogena semicolorata	0	0	0	0	0	0	1	0	0	0	4	0	0	0
Sericostoma personatum	6	2	0	1	1	7	3	0	3	3	3	14	1	7
Sialis sp.	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Sigara sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Silo pallipes	2	1	1	0	0	2	1	0	9	3	2	0	0	0
Simuliidae	39	34	0	0	28	29	13	2	38	8	12	18	3	3
Sphaeriidae	24	15	0	0	0	0	0	0	0	0	0	0	0	0

APPENDIX E Cont.

	AOWat99	AOWwin99	AOWsp00	TOWsum98	TOWat98	TOWwin98	TOWsp99	TOWsum99	TOWat99	TOWwin99	TOWsp00	IOWsum98	IOWat98	IOWwin98
Theodoxus fluviatilis	10	4	1	0	0	0	0	0	0	0	0	0	0	0
Theromyzon tessulatum	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tinodes unicolor	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tipula sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Valvata cristata	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Valvata macrostoma	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Valvata piscinalis	1	0	0	0	0	0	0	0	0	0	0	1	0	0
Viviparus viviparus	0	0	0	0	0	0	0	0	0	0	0	0	0	0

	IOWsp99	IOWsum99	IOWat99	IOWwin99	IOWsp00	FSsum98	FSat98	FSwin98	FSsp99	FSsum99	FSat99	FSwin99	FSsp00	MSsum98	MSat98	MSwin98
Agapetus fuscipes	3	16	0	5	5	0	0	0	0	0	0	1	0	0	0	0
Calopteryx splendens	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Anaboluia nervosa	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ancylus fluviatilis	0	1	1	1	0	0	0	0	0	0	0	0	0	0	0	0
Aphelocheirus aestivalis	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0
Asellus aquaticus	31	231	311	75	19	0	0	3	0	1	1	0	0	1	5	2
Atherix sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Athripsodes albifrons	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Athripsodes sp.	1	0	0	0	3	1	1	0	0	0	0	0	0	0	0	0
Baetis sp.	380	348	81	107	139	1	12	3	0	1	1	7	7	1	1	1
Bdellocephala punctata	0	5	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Bithynia leachi	0	0	1	3	0	0	0	0	0	0	0	0	0	0	0	0
Brachycentrus subnubilis	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
Caenis sp.	11	1	0	0	155	3	1	0	0	0	0	0	29	48	1	1
Chironomidae	28	19	5	10	4	664	57	33	0	87	50	7	19	61	3	16
Chloroperla sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Colymbetes sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Crunoecia irrorata	1	0	0	0	0	0	2	0	0	0	0	0	3	0	0	0
Culicoides sp.	1	1	1	0	0	1	1	3	0	0	0	0	0	0	0	0
Curculionidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

APPENDIX E Cont.

	IOWsp99	IOWsum99	IOWat99	IOWwin99	IOWsp00	FSsum98	FSat98	FSwin98	FSsp99	FSsum99	FSat99	FSwin99	FSsp00	MSsum98	MSat98	MSwin98
Cymatia bonsdorffii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Dendrocoelum lacteum	1	0	3	0	0	0	0	0	0	0	0	0	0	0	0	0
Dicranota sp.	0	3	0	1	1	0	1	2	0	0	0	1	0	0	1	0
Dixid sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Donacia sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Drusus annulatus	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0
Dugesia polychroa	0	0	0	3	0	0	0	0	0	0	0	0	0	0	0	0
Dugesia tigrinia	0	0	0	0	0	1	2	0	0	0	0	0	0	0	0	0
Ecdyonurus torrentis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ecdyonurus venosus	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0
Elmis aenea	0	1	7	1	5	0	0	0	0	0	0	0	0	1	1	1
Empididae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ephemera danica	0	0	0	0	0	1	0	1	0	0	1	4	0	0	0	0
Ephemerella ignita	95	43	1	0	99	14	0	0	0	7	0	0	11	1	0	0
Ephemerella notata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Erpobdella octoculata	4	6	7	14	5	2	0	2	0	0	1	0	0	1	2	1
Esolus parallelepipedus	0	0	0	0	0	0	0	0	0	11	1	2	3	0	0	0
Gammarus sp.	15	32	60	59	8	30	11	17	0	24	0	9	15	1	23	1
Gerris sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Glossiphonia complanata	1	2	0	0	1	1	0	1	0	3	0	0	0	6	14	9
Goera pilosa	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gyrinus natator	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Halesus radiatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Haliplus sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Helobdella stagnalis	0	1	1	2	0	0	0	0	0	0	0	1	0	0	0	0
Heptagenia sulphurea	2	4	1	4	0	0	0	0	0	0	0	0	2	0	0	0
Hydracarina	15	5	0	0	0	1	0	1	0	5	0	0	13	1	1	0
Hydropsyche pellucidula	0	0	2	1	0	0	0	0	0	0	0	0	0	0	0	0
Hydropsyche siltalai	1	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Hydroptila sp.	0	1	0	0	0	0	0	0	0	1	0	0	0	0	0	0

APPENDIX E Cont.

	IOWsp99	IOWsum99	IOWat99	IOWwin99	IOWsp00	FSsum98	FSat98	FSwin98	FSsp99	FSsum99	FSat99	FSwin99	FSsp00	MSsum98	MSat98	MSwin98
Hyphydrus ovatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Isoperla grammatica	1	0	0	0	3	0	0	0	0	0	0	0	1	0	0	0
Lepidostoma hirtum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Leptophlebiidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Leuctra geniculata	0	0	0	0	1	1	0	0	0	3	0	0	1	0	0	0
Leuctra hippopus	4	4	0	0	1	2	0	0	0	1	0	0	5	0	0	0
Limnephilus lunatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Limnephilus sp.	0	1	0	0	0	0	0	0	0	0	0	3	0	0	0	0
Limnius volckmari	0	1	4	0	5	0	0	0	0	1	0	0	0	0	0	0
Lymnaea glabra	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lymnaea peregra	0	0	0	0	0	0	0	0	0	1	0	0	0	1	0	0
Lymnaea trunctula	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Melampophylax mucoreus	0	2	0	5	0	0	0	0	0	0	0	0	0	0	0	0
Mesovelia furcata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Muscidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Mystacides azurea	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nemoura sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Odontocerum albicorne	2	1	3	2	5	0	0	0	0	0	0	0	0	0	0	0
Oligochaeta	55	64	40	143	16	33	8	54	0	31	111	48	152	87	108	147
Orectchilus villosus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Oulimnius sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Philopotamidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Physa fontinalis	0	1	5	0	0	0	0	0	0	0	0	0	0	0	0	0
Piscicola geometra	0	2	0	0	0	0	1	0	0	0	0	0	0	0	1	0
Planorbis albus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Planorbis carinatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Planorbis contortus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Planorbis planorbis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Planorbis vortex	0	2	9	0	3	0	0	0	0	0	0	0	0	0	0	0
Platambus sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0

APPENDIX E Cont.

	IOWsp99	IOWsum99	IOWat99	IOWwin99	IOWsp00	FSsum98	FSat98	FSwin98	FSsp99	FSsum99	FSat99	FSwin99	FSsp00	MSsum98	MSat98	MSwin98
Polycelus tenuis	4	8	21	3	0	0	0	0	0	0	0	0	0	0	0	0
Polycentropus flavomaculatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3
Potamophylax latipennis	0	1	1	0	1	0	1	2	0	0	0	0	0	0	0	0
Potamopyrgus jenkinsi	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Psychomyia pusilla	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Rhyacophila dorsalis	4	3	3	1	1	0	1	0	0	1	0	0	0	0	0	0
Rithrogena semicolorata	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Sericostome personatum	2	3	3	8	7	0	0	1	0	2	0	0	0	0	0	0
Sialis sp.	0	0	1	2	0	0	0	0	0	0	0	1	0	0	1	0
Sigara sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Silo pallipes	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	1
Simuliidae	32	23	10	1	3	0	1	1	0	0	0	0	0	1	1	0
Sphaeriidae	0	1	3	0	0	3	1	1	0	4	0	4	5	0	0	0
Theodoxus fluviatilis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Theromyzon tessulatum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tinodes unicolor	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tipula sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Valvata cristata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Valvata macrostoma	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Valvata piscinalis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
Viviparus viviparus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

	MSsp99	MSsum99	MSat99	MSwin99	MSsp00	ASsum98	ASat98	ASwin98	ASsp99	ASsum99	ASat99	ASwin99	ASsp00	TSsum98	TSat98	TSwin98	TSsp99
Agapetus fuscipes	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0
Calopteryx splendens	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Anaboluia nervosa	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ancylus fluviatilis	0	3	2	0	0	1	0	0	0	0	0	0	0	0	0	0	0
Apheldcheirus aestivalis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Asellus aquaticus	0	3	9	0	3	5	1	0	0	11	1	3	0	0	5	0	0
Atherix sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

APPENDIX E Cont.

	MSsp99	MSsum99	MSat99	MSwin99	MSsp00	ASsum98	ASat98	ASwin98	ASsp99	ASsum99	ASat99	ASwin99	ASsp00	TSsum98	TSat98	TSwin98	TSsp99
Athripsodes albifrons	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Athripsodes sp.	1	0	0	0	0	5	0	0	0	3	0	0	3	0	5	1	0
Baetis sp.	1	7	0	1	7	80	1	7	0	0	0	0	8	2	1	1	7
Bdellocephala punctata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Bithynia leachi	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Brachycentrus subnubilis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Caenis sp.	28	3	4	6	28	160	0	0	0	0	0	2	7	0	0	0	0
Chironomidae	37	20	19	19	37	5	33	3	0	20	1	3	3	3	19	7	12
Chloroperla sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Colymbetes sp.	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
Crunoecia irrorata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0
Culicoides sp.	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
Curculionidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cymatia bonsdorffii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Dendrocoelum lacteum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Dicranota sp.	0	3	0	0	3	3	1	6	0	0	1	0	1	0	0	0	0
Dixid sp.	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Donacia sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Drusus annulatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Dugesia polychroa	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Dugesia tigrinia	0	0	0	0	0	0	0	0	0	0	6	0	0	0	0	0	0
Ecdyonurus torrentis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ecdyonurus venosus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Elmis aenea	5	1	4	0	1	0	0	0	0	0	0	0	0	1	1	1	0
Empididae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ephemera danica	0	0	0	0	0	8	1	4	0	0	0	4	4	1	21	1	0
Ephemerella ignita	0	0	0	0	4	590	0	0	0	3	0	0	0	2	0	0	0
Ephemerella notata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Erpobdella octoculata	0	0	1	0	0	3	1	2	0	2	2	0	0	0	3	3	0
Esolus parallelepipedus	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0

APPENDIX E Cont.

	MSsp99	MSsum99	MSat99	MSwin99	MSsp00	ASsum98	ASat98	ASwin98	ASsp99	ASsum99	ASat99	ASwin99	ASsp00	TSsum98	TSat98	TSwin98	TSsp99
Gammarus sp.	3	5	5	0	4	70	11	4	0	4	3	0	5	1	44	1	3
Gerris sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Glossiphonia complanata	0	1	1	1	0	1	0	3	0	0	0	0	0	0	0	0	0
Goera pilosa	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gyrinus natator	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Halesus radiatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Haliplus sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3	0	0
Helobdella stagnalis	0	0	4	3	1	0	0	0	0	1	0	0	0	0	0	0	0
Heptagenia sulphurea	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0
Hydracarina	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1
Hydropsyche pellucidula	0	7	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hydropsyche siltalai	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hydroptila sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hyphydrus ovatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Isoperla grammatica	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lepidostoma hirtum	0	0	0	0	0	3	0	0	0	0	0	0	0	0	0	0	0
Leptophlebiidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Leuctra geniculata	0	0	0	0	0	2	0	0	0	1	0	0	0	0	0	0	0
Leuctra hippopus	1	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Limnephilus lunatus	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Limnephilus sp.	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0
Limnius volckmari	1	0	0	0	0	20	9	11	0	32	28	7	23	0	0	0	0
Lymnaea glabra	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lymnaea peregra	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0
Lymnaea trunctula	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Melampophylax mucoreus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Mesovelia furcata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Muscidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Mystacides azurea	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0	0
Nemoura sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

APPENDIX E Cont.

	MSsp99	MSsum99	MSat99	MSwin99	MSsp00	ASsum98	ASat98	ASwin98	ASsp99	ASsum99	ASat99	ASwin99	ASsp00	TSsum98	TSat98	TSwin98	TSsp99
Odontocerum albicorne	0	0	0	0	0	0	1	5	0	0	2	0	0	0	0	0	0
Oligochaeta	44	57	116	109	53	2	14	36	0	88	31	56	40	1	0	5	4
Orectchilus villosus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Oulimnius sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Philopotamidae	0	0	0	0	0	8	0	0	0	0	0	0	0	0	0	0	0
Physa fontinalis	0	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0	0
Piscicola geometra	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Planorbis albus	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Planorbis carinatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Planorbis contortus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Planorbis planorbis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Planorbis vortex	1	0	1	0	0	2	4	2	0	0	1	1	2	0	0	0	0
Platambus sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4	0	0
Polycelus tenuis	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
Polycentropus flavomaculatus	0	4	0	0	0	2	0	0	0	1	0	0	0	0	6	1	0
Potamophylax latipennis	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0	1	0
Potamopyrgus jenkinsi	1	3	0	0	1	0	0	0	0	0	0	0	0	0	2	0	1
Psychomyia pusilla	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Rhyacophila dorsalis	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0
Rithrogena semicolorata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Sericostoma personatum	0	0	0	0	0	1	0	0	0	0	1	2	3	0	1	1	1
Sialis sp.	0	3	0	0	0	0	0	0	0	2	0	0	0	1	2	2	0
Sigara sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Silo pallipes	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0
Simuliidae	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Sphaeriidae	1	1	3	0	3	0	5	5	0	4	5	9	2	0	0	4	17
Theodoxus fluviatilis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Theromyzon tessulatum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tinodes unicolor	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0
Tipula sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

APPENDIX E Cont.

	MSsp99	MSsum99	MSat99	MSwin99	MSsp00	ASsum98	ASat98	ASwin98	ASsp99	ASsum99	ASat99	ASwin99	ASsp00	TSsum98	TSat98	TSwin98	TSsp99
Valvata cristata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Valvata macrostoma	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Valvata piscinalis	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Viviparus viviparus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

	TSsum99	TSat99	TSwin99	TSsp00	ISsum98	ISat98	ISwin98	ISsp99	ISsum99	ISat99	ISwin99	ISsp00
Agapetus fuscipes	0	0	0	0	0	0	0	0	0	0	0	0
Calopteryx splendens	0	0	0	0	0	0	0	0	0	0	0	0
Anaboluia nervosa	0	0	0	0	0	0	0	0	0	0	0	0
Ancylus fluviatilis	0	0	0	0	0	0	0	0	0	0	0	0
Aphelocheirus aestivalis	0	0	0	0	0	0	0	0	0	0	0	0
Asellus aquaticus	0	0	0	0	24	21	0	0	5	15	15	1
Atherix sp.	0	0	0	0	0	0	0	0	0	0	0	0
Athripsodes albifrons	0	0	0	0	0	0	0	0	0	0	0	0
Athripsodes sp.	0	0	0	0	0	3	0	0	0	0	0	0
Baetis sp.	1	1	0	1	84	12	0	4	27	3	1	3
Bdellocephala punctata	0	0	0	0	0	0	0	0	0	0	0	0
Bithynia leachi	0	2	0	0	0	0	0	0	0	0	0	0
Brachycentrus subnubilis	0	0	0	0	0	0	0	0	0	0	0	0
Caenis sp.	0	0	0	0	0	0	0	1	0	0	0	0
Chironomidae	4	16	8	13	2	71	179	5	37	108	267	16
Chloroperla sp.	0	0	0	0	0	0	0	0	0	0	0	0
Colymbetes sp.	0	0	0	0	0	0	0	0	0	4	0	0
Crunoecia irrorata	0	0	0	0	0	0	0	0	0	0	0	0
Culicoides sp.	0	0	0	0	0	1	0	0	0	0	0	0
Curculionidae	0	0	0	0	0	0	0	0	0	0	0	0
Cymatia bonsdorffii	0	0	0	0	2	0	0	0	0	0	1	4
Dendrocoelum lacteum	0	0	0	0	0	0	0	0	0	0	0	0
Dicranota sp.	0	0	0	0	0	0	0	0	0	0	0	0
Dixid sp.	0	0	0	0	0	0	0	0	0	0	0	0

APPENDIX E Cont.

	TSsum99	TSat99	TSwin99	TSsp00	ISsum98	ISat98	ISwin98	ISsp99	ISsum99	ISat99	ISwin99	ISsp00
Donacia sp.	0	0	0	0	0	0	0	0	0	0	0	0
Drusus annulatus	0	0	0	0	0	0	0	0	0	0	0	0
Dugesia polychroa	0	0	0	0	0	0	0	0	0	0	0	0
Dugesia tigrinia	0	0	0	0	0	0	0	0	0	0	0	0
Ecdyonurus torrentis	0	0	0	0	0	0	0	0	0	0	0	0
Ecdyonurus venosus	0	0	0	0	0	0	0	0	0	0	0	0
Elmis aenea	0	0	0	0	0	0	0	0	0	0	0	0
Empididae	0	0	0	0	0	0	0	0	0	0	0	0
Ephemera danica	0	3	0	2	0	0	0	0	0	0	0	0
Ephemerella ignita	1	0	0	0	70	1	0	3	3	0	0	1
Ephemerella notata	0	0	0	0	0	0	0	0	0	0	0	0
Erpobdella octoculata	0	0	0	0	11	6	1	0	0	0	11	5
Esolus parallelepipedus	0	0	0	0	0	0	0	0	0	0	0	0
Gammarus sp.	2	8	1	2	49	36	0	0	13	39	4	0
Gerris sp.	0	0	0	0	0	0	0	0	0	0	0	0
Glossiphonia complanata	0	0	0	0	0	4	0	0	1	3	0	1
Goera pilosa	0	0	0	0	0	0	0	0	0	0	0	0
Gyrinus natator	0	0	0	0	0	1	0	0	0	1	0	0
Halesus radiatus	0	0	0	0	1	0	0	0	0	0	0	0
Haliplus sp.	0	1	0	0	0	0	0	0	1	5	1	0
Helobdella stagnalis	0	0	0	0	0	0	0	0	0	0	2	0
Heptagenia sulphurea	0	0	0	0	1	0	0	0	0	0	0	0
Hydracarina	0	0	0	0	0	0	0	0	0	0	0	1
Hydropsyche pellucidula	0	0	0	0	0	0	0	0	0	0	0	0
Hydropsyche siltalai	0	0	0	0	0	0	0	0	0	0	0	0
Hydroptila sp.	0	0	0	0	0	0	0	0	0	0	0	0
Hyphydrus ovatus	0	0	0	0	0	0	0	0	0	0	0	0
Isoperla grammatica	0	0	0	0	0	0	0	0	0	0	0	0
Lepidostoma hirtum	0	0	0	0	0	0	0	0	0	0	0	0
Leptophlebiidae	0	0	0	0	0	0	0	0	0	0	0	0

APPENDIX E Cont.

	TSsum99	TSat99	TSwin99	TSsp00	ISsum98	ISat98	ISwin98	ISsp99	ISsum99	ISat99	ISwin99	ISsp00
Leuctra geniculata	0	0	0	0	0	0	0	0	0	0	0	0
Leuctra hippopus	0	0	0	0	0	0	0	0	0	0	0	0
Limnephilus lunatus	0	0	0	0	0	0	0	0	0	0	0	0
Limnephilus sp.	0	0	0	0	0	0	0	0	0	0	1	0
Limnius volckmari	0	0	0	0	0	0	0	0	0	0	0	0
Lymnaea glabra	0	0	0	0	0	0	0	0	0	0	0	0
Lymnaea peregra	1	0	0	0	0	1	3	0	0	0	0	0
Lymnaea trunctula	0	0	0	0	0	0	0	0	0	0	0	0
Melampophylax mucoreus	0	0	0	0	0	0	0	0	0	0	0	0
Mesovelia furcata	0	0	0	0	0	0	0	0	0	0	0	0
Muscidae	0	0	0	0	0	0	0	0	0	0	0	0
Mystacides azurea	0	0	0	0	0	0	0	0	0	0	1	0
Nemoura sp.	0	0	0	0	0	0	0	0	0	0	0	0
Odontocerum albicorne	0	0	0	0	0	0	0	0	0	0	0	0
Oligochaeta	9	17	8	8	0	16	2	0	8	4	33	0
Orectchilus villosus	0	0	0	0	0	0	0	0	0	0	0	0
Oulimnius sp.	0	0	0	0	0	0	0	0	0	0	0	0
Philopotamidae	0	0	0	0	0	0	0	0	0	0	0	0
Physa fontinalis	0	0	0	0	0	7	0	0	0	3	0	0
Piscicola geometra	0	0	0	0	0	2	0	0	1	1	4	0
Planorbis albus	0	0	0	0	0	0	0	0	0	0	0	0
Planorbis carinatus	0	0	0	0	0	0	0	0	0	0	0	0
Planorbis contortus	0	0	0	0	0	0	0	0	0	0	0	0
Planorbis planorbis	0	1	0	0	0	0	0	0	0	0	0	0
Planorbis vortex	0	0	0	0	0	0	1	0	0	4	0	0
Platambus sp.	0	0	0	0	0	0	0	0	0	0	0	0
Polycelus tenuis	0	0	0	0	0	0	0	0	1	0	3	0
Polycentropus flavomaculatus	0	0	0	0	0	0	0	0	0	1	0	0
Potamophylax latipennis	0	0	0	0	2	1	0	0	0	0	0	0
Potamopyrgus jenkinsi	0	0	0	0	0	0	0	0	0	0	0	0

APPENDIX E Cont.

	TSsum99	TSat99	TSwin99	TSsp00	ISsum98	ISat98	ISwin98	ISsp99	ISsum99	ISat99	ISwin99	ISsp00
Psychomyia pusilla	0	0	0	0	0	0	0	0	0	0	0	0
Rhyacophila dorsalis	0	0	0	0	0	0	0	0	0	0	0	0
Rithrogena semicolorata	0	0	0	0	0	0	0	0	0	0	0	0
Sericostoma personatum	0	0	0	0	2	0	0	0	0	0	0	0
Sialis sp.	0	0	1	3	0	1	0	0	0	0	0	0
Sigara sp.	0	0	0	0	0	0	0	0	0	0	0	0
Silo pallipes	0	0	0	0	0	0	0	0	0	0	0	0
Simuliidae	0	0	0	0	0	0	0	0	0	0	0	0
Sphaeriidae	8	3	5	22	1	1	0	0	0	0	0	1
Theodoxus fluviatilis	0	0	0	0	0	0	0	0	0	0	0	0
Theromyzon tessulatum	0	0	0	0	0	0	0	0	0	0	0	0
Tinodes unicolor	0	0	0	0	0	0	0	0	0	0	0	0
Tipula sp.	0	0	0	0	0	0	0	0	0	0	0	0
Valvata cristata	0	0	0	0	0	0	0	0	0	0	0	0
Valvata macrostoma	0	0	0	0	0	0	1	0	0	2	0	3
Valvata piscinalis	1	0	0	0	0	0	0	0	0	0	0	0
Viviparus viviparus	0	0	0	0	0	0	0	0	0	0	0	0

APPENDIX F
RECOLONISATION DATA

	small 16	small 16	small 16	small 33	small 33	small 33	small 51	small 51	small 51	small 69	small 69	small 69
Agapetus sp.	0	0	0	0	0	0	0	0	0	0	0	0
Calopterxy splendens	0	0	0	0	0	0	0	0	0	0	0	0
Ancylus fluviatilis	0	0	0	0	0	0	0	4	0	4	0	0
Asellus aquaticus	1	0	0	0	0	4	0	0	0	4	0	8
Athripsodes albifrons	0	0	0	0	0	0	0	0	0	0	0	0
Baetis sp.	24	16	60	140	4	52	56	32	36	104	116	48
Bithynia leachi	0	0	0	0	0	0	0	0	0	0	0	0
Caenis sp.	0	0	0	0	0	0	0	0	0	8	0	0
Chironomidae	8	8	104	100	44	40	76	12	268	40	60	120
Culicoides sp.	0	0	0	0	0	0	4	0	0	0	0	4
Dendrocoelum lacteum	4	8	8	30	24	8	24	12	40	40	20	48
Dicranota sp.	0	0	1	0	2	8	0	4	8	0	4	24
Dugesia polychroa	0	4	0	26	56	12	16	12	8	128	20	80
Elmis aenea	4	0	0	16	28	56	16	12	30	0	8	8
Ephemera dancia	0	0	0	0	0	0	0	0	0	4	4	1
Ephemerella ignita	4	4	12	0	0	0	0	0	0	0	0	0
Erpobdella octoculata	0	0	0	4	8	4	4	1	3	8	1	1
Gammaridae	2448	2184	2260	1664	692	3528	1680	2636	2128	1428	1796	3396
Glossiphonia complanata	1	0	1	0	0	1	8	4	0	8	8	2
Goera pilosa	0	0	0	0	1	4	0	0	4	4	8	0
Gyrinus natator	0	0	0	28	12	1	0	0	0	0	0	0
Heptagenia sulphurea	0	0	0	8	0	0	0	8	16	16	28	16
Hydracarina	0	0	0	0	0	0	0	0	0	0	4	8
Hydropsyche pellucidula	0	4	4	15	1	32	1	4	12	0	28	24
Hydropsyche siltalai	4	4	12	45	0	20	8	4	24	4	40	24
Leuctra hippopus	20	20	12	4	32	24	0	4	0	0	0	0
Limnius volckmari	20	12	4	28	44	32	16	24	70	96	72	80
Lymnaea peregra	0	0	0	0	0	0	0	1	0	0	0	0
Lymnaea stagnalis	0	0	0	0	0	0	0	0	0	0	0	0
Muscidae	0	0	0	0	0	0	0	0	1	0	0	0
Odontocerum albicorne	1	1	0	0	8	16	2	8	0	56	4	16
Oligochaeta	2	0	0	64	48	1	60	36	52	80	56	168
Paraleptophlebia wernei	0	0	0	0	0	4	48	8	28	36	40	40
Physa fontinalis	0	0	0	0	0	0	0	0	0	0	0	0
Planorbis planorbis	0	0	0	0	0	0	0	0	0	0	0	0
Planorbis vortex	0	0	0	0	0	0	1	0	0	16	4	1
Platambus sp.	0	0	0	0	4	0	0	0	0	0	0	0
Polycentropus flavomaculatus	0	0	0	0	0	0	0	0	4	4	4	1
Potamophylax latipennis	0	0	0	0	0	0	0	0	0	0	0	0
Potamopyrgus jenkinsi	0	0	0	0	0	0	0	0	4	16	0	1
Rhyacophila dorsalis	3	0	1	3	4	8	1	1	0	1	1	8
Rithrogena semicolorata	0	0	0	0	0	0	0	0	0	0	0	0
Sericostoma personatum	0	0	0	12	0	0	8	1	16	44	20	24
Simuliidae	8	4	3	48	0	12	0	8	20	4	4	8
Sphaeriidae	0	0	0	0	0	0	0	0	0	16	0	0
Tinodes unicolor	0	0	0	4	0	0	0	0	0	0	0	0
Tipula sp.	0	0	0	0	0	0	0	0	0	0	0	0

	small 85	small 85	small 85	large 16	large 16	large 16	large 33	large 33	large 33	large 51	large 51	large 51
Agapetus sp.	0	0	0	0	4	8	0	12	0	0	0	4
Agrion virgo	0	0	0	0	0	0	0	0	0	0	0	0
Ancylus fluviatilis	0	0	0	0	0	0	0	0	0	0	0	0
Asellus aquaticus	0	0	64	1	1	4	4	1	1	1	0	0
Athripsodes albifrons	0	0	0	0	0	0	0	0	0	0	0	0
Baetis sp.	84	44	56	24	52	12	100	12	4	56	44	4
Bithynia leachi	0	0	0	0	0	0	0	0	1	0	1	0
Caenis sp.	4	4	8	0	0	8	0	0	4	0	0	0
Chironomidae	76	48	184	16	40	44	76	48	20	24	32	88
Culicoides sp.	0	0	0	4	0	0	0	0	0	1	0	0
Dendrocoelum lacteum	88	12	10	4	0	8	4	4	0	2	4	4
Dicranota sp.	20	1	24	0	0	4	4	0	12	0	0	12
Dugesia polychroa	156	28	18	1	0	0	1	0	8	4	0	0
Elmis aenea	8	20	39	4	12	0	4	8	8	0	12	15
Ephemera danica	4	0	20	0	0	4	2	8	2	4	0	4
Ephemerella ignita	0	0	1	0	0	0	0	0	0	0	0	0
Erpobdella octoculata	4	4	8	0	1	0	1	2	2	8	0	0
Gammaridae	1668	680	940	352	788	376	756	1448	1564	292	1340	1280
Glossiphonia complanata	4	1	12	0	0	1	1	4	3	2	0	4
Goera pilosa	4	0	1	0	0	2	4	0	12	0	12	0
Gyrinus natator	0	0	0	4	1	4	8	0	8	0	1	8
Heptagenia sulphurea	0	0	0	4	12	0	1	0	0	0	0	4
Hydracarina	0	0	4	0	0	0	0	0	0	0	0	0
Hydropsyche pellucidula	0	2	4	0	0	0	0	4	0	8	12	12
Hydropsyche siltalai	12	1	24	4	1	2	8	3	4	8	16	20
Leuctra hippopus	0	0	0	0	0	0	0	0	0	0	0	0
Limnius volckmari	76	84	157	4	8	8	24	28	44	12	48	61
Lymnaea peregra	0	0	0	0	0	0	0	0	0	0	0	0
Lymnaea stagnalis	0	1	1	0	0	0	0	0	0	0	0	0
Muscidae	0	1	0	0	0	0	0	0	0	0	0	0

APPENDIX F Cont.

	small 85	small 85	small 85	large 16	large 16	large 16	large 33	large 33	large 33	large 51	large 51	large 51
Odontocerum albicorne	44	0	8	8	4	8	8	16	4	12	12	4
Oligochaeta	8	860	72	12	8	4	4	2	12	4	4	4
Paraleptophlebia werner	40	0	4	0	0	0	12	4	16	4	12	0
Physa fontinalis	4	4	0	4	0	0	4	0	0	0	0	0
Planorbis planorbis	0	0	0	0	0	0	0	1	0	0	0	0
Planorbis vortex	0	4	4	2	0	4	0	0	0	0	4	0
Platambus sp.	0	0	0	0	0	0	0	0	0	0	0	0
Polycentropus flavomaculatus	0	0	0	0	0	1	4	1	0	4	0	0
Potamophylax latipennis	0	0	0	1	4	1	1	4	8	1	4	4
Potamopyrgus jenkinsi	0	8	4	0	1	0	0	0	0	0	0	0
Rhyacophila dorsalis	2	0	16	4	0	1	4	0	4	0	0	4
Rithrogena semicolorata	0	0	0	0	0	0	8	0	0	0	0	0
Sericostoma personatum	36	24	112	6	4	20	24	32	48	12	60	28
Simuliidae	4	0	0	12	12	12	76	4	1	8	12	24
Sphaeriidae	0	0	0	0	0	1	0	0	0	0	0	0
Tinodes unicolor	4	0	16	0	0	0	0	0	0	0	0	0
Tipula sp.	0	0	0	0	0	0	0	0	2	0	0	1

	large 69	large 69	large 69	large 85	large 85	large 85	background 1	background 1	background 1	background 2
Agapetus sp.	0	0	28	0	4	0	0	0	0	0
Calopteryx splendens	0	0	0	0	0	0	0	0	0	4
Ancylus fluviatilis	0	0	0	0	0	0	0	0	0	0
Asellus aquaticus	1	4	12	12	4	1	0	0	0	0
Athripsodes albifrons	0	0	0	0	4	4	0	0	0	0
Baetis sp.	28	52	108	136	32	56	364	128	96	60
Bithynia leachi	0	0	0	1	0	0	0	0	0	0
Caenis sp.	0	4	4	0	0	4	0	0	0	0
Chironomidae	40	60	96	48	52	68	16	4	0	12
Culicoides sp.	0	0	0	4	0	0	0	0	0	0
Dendrocoelum lacteum	1	8	0	0	2	1	0	0	0	3
Dicranota sp.	20	4	16	0	24	8	20	24	24	20
Dugesia polychroa	0	8	0	0	1	4	0	0	0	3
Elmis aenea	4	14	20	15	24	22	50	16	28	30
Ephemera dancia	1	0	4	4	8	0	0	0	3	0
Ephemerella ignita	0	0	0	0	0	0	24	20	4	0
Erpobdella octoculata	1	4	4	1	1	1	0	8	0	0
Gammaridae	284	720	1344	816	1432	1412	3572	904	1408	812
Glossiphonia complanata	4	8	0	4	4	8	1	4	2	0
Goera pilosa	12	4	8	20	1	4	4	0	4	4
Gyrinus natator	0	0	8	0	0	1	0	0	0	0
Heptagenia sulphurea	0	4	4	8	8	8	0	4	0	4
Hydracarina	0	0	0	0	0	0	0	0	0	0
Hydropsyche pellucidula	1	8	12	21	24	16	4	0	4	0
Hydropsyche siltalai	4	4	16	55	76	32	4	1	16	4
Leuctra hippopus	0	0	0	0	0	0	0	1	0	4
Limnius volckmari	12	58	64	45	72	66	150	52	80	94
Lymnaea peregra	0	0	0	0	0	0	0	0	0	0
Lymnaea stagnalis	0	0	1	0	0	0	0	0	0	0
Muscidae	0	0	0	0	0	0	0	0	0	0
Odontocerum albicorne	48	40	16	96	40	40	4	0	1	4
Oligochaeta	12	44	4	16	20	1	15	21	76	8
Paraleptophebia werner	0	16	8	12	4	4	0	0	0	0
Physa fontinalis	0	0	0	2	0	0	0	0	0	0
Planorbis planorbis	0	0	0	0	1	0	0	0	0	0
Planorbis vortex	8	0	0	2	4	0	0	0	0	1
Platambus sp.	0	0	0	0	0	0	0	1	0	0
Polycentropus flavomaculatus	0	0	0	0	0	0	0	0	0	0
Potamophylax latipennis	4	1	1	4	1	8	4	0	1	0
Potamopyrgus jenkinsi	0	0	0	0	0	0	1	8	4	0
Rhyacophila dorsalis	0	4	4	4	4	1	12	4	8	16
Rithrogena semicolorata	0	0	0	0	0	0	0	0	0	0
Sericostoma personatum	32	80	68	80	36	56	0	0	0	0
Simuliidae	4	0	4	0	8	0	20	0	0	1
Sphaeriidae	0	0	0	4	0	0	0	0	0	0
Tinodes unicolor	0	0	0	0	0	0	0	0	0	0
Tipula	0	0	0	0	0	0	0	0	0	0

	background 2	background 2	background 3	background 3	background 3	background 4	background 4	background 4
Agapetus sp.	0	4	0	1	0	1	4	4
Calopterxy splendens	4	0	0	0	0	0	0	0
Ancylus fluviatilis	0	0	0	0	0	0	0	0
Asellus aquaticus	0	0	0	0	0	0	0	0
Athripsodes albifrons	0	0	0	0	0	0	0	0
Baetis sp.	68	16	1	6	12	64	24	8
Bithynia leachi	0	0	0	0	0	0	0	0
Caenis sp.	0	0	0	0	0	4	0	0
Chironomidae	0	4	0	1	0	16	0	0
Culicoides sp.	0	0	0	0	0	0	0	0
Dendrocoelum lacteum	4	1	0	4	8	2	0	4
Dicranota sp.	16	8	4	8	4	40	16	1
Dugesia polychroa	0	4	1	4	12	20	0	0
Elmis aenea	22	16	28	20	16	54	20	8
Ephemera dancia	0	1	0	4	4	0	8	0
Ephemerella ignita	1	0	0	0	0	0	0	0

APPENDIX F Cont.

	background 2	background 2	background 3	background 3	background 3	background 4	background 4	background 4
Erpobdella octoculata	0	2	0	0	0	1	0	1
Gammaridae	652	348	332	460	360	1488	356	324
Glossiphonia complanata	0	1	0	0	0	0	2	1
Goera pilosa	0	4	4	3	4	4	4	16
Gyrinus natator	1	4	0	0	0	4	0	0
Heptagenia sulphurea	4	0	4	0	4	0	4	0
Hydracarina	0	0	0	0	0	0	0	0
Hydropsyche pellucidula	0	1	1	0	0	0	4	1
Hydropsyche siltalai	3	4	4	1	0	1	4	0
Leuctra hippopus	0	2	0	0	0	0	0	0
Limnius volckmari	66	44	80	56	52	156	64	28
Lymnaea peregra	0	0	0	0	0	0	0	0
Lymnaea stagnalis	0	0	0	0	0	0	0	0
Muscidae	0	0	0	0	0	0	0	0
Odontocerum albicome	4	1	0	4	16	8	0	4
Oligochaeta	64	16	12	20	68	60	16	8
Paraleptophlebia wernei	0	0	0	0	0	0	0	0
Physa fontinalis	0	0	0	0	0	0	0	0
Planorbis planorbis	0	0	0	0	0	0	0	0
Planorbis vortex	0	0	0	0	0	0	0	0
Platambus sp.	0	4	0	0	0	0	0	0
Polycentropus flavomaculatus	0	0	0	0	0	0	0	0
Potamophylax latipennis	0	0	0	0	0	0	0	0
Potamopyrgus jenkinsi	0	0	0	0	0	0	0	0
Rhyacophila dorsalis	2	1	12	4	0	4	0	0
Rithrogena semicolorata	0	0	0	0	0	0	0	0
Sericostoma personatum	0	4	0	3	16	8	4	2
Simuliidae	0	0	0	0	0	1	0	0
Sphaeriidae	0	0	0	0	0	0	0	4
Tinodes unicolor	0	0	0	0	0	0	0	0
Tipula sp.	0	0	0	0	0	0	0	0

	background 5	background 5	background 5	background 6	background 6	background 6
Agapetus sp.	1	0	0	32	0	60
Calopteryx splendens	0	0	0	0	0	0
Ancylus fluviatilis	1	0	0	0	0	0
Asellus aquaticus	0	0	0	0	0	0
Athripsodes albifrons	0	0	0	0	0	0
Baetis sp.	8	0	28	24	72	100
Bithynia leachi	0	0	0	0	0	0
Caenis sp.	0	0	0	0	4	0
Chironomidae	0	0	4	0	4	4
Culicoides sp.	0	0	0	0	0	0
Dendrocoelum lacteum	4	0	4	4	0	8
DIcranota sp.	2	2	8	24	24	4
Dugesia polychroa	2	1	0	0	5	8
Elmis aenea	12	4	28	16	28	8
Ephemera dancia	0	0	4	0	0	0
Ephemerella ignita	0	0	0	0	0	0
Erpobdella octoculata	2	0	0	0	1	1
Gammaridae	132	20	268	104	600	540
Glossiphonia complanata	1	1	0	0	0	0
Goera pilosa	4	4	1	4	4	0
Gyrinus natator	0	0	0	1	0	0
Heptagenia sulphurea	0	0	0	4	0	8
Hydracarina	0	0	0	0	0	0
Hydropsyche pellucidula	0	0	0	0	0	0
Hydropsyche siltalai	1	0	1	1	0	4
Leuctra hippopus	0	0	0	0	0	0
Limnius volckmari	32	12	88	24	84	52
Lymnaea peregra	0	0	0	0	0	0
Lymnaea stagnalis	0	0	0	0	0	0
Muscidae	0	0	0	0	0	0
Odontocerum albicome	0	1	1	4	1	20
Oligochaeta	3	1	36	4	52	92
Paraleptophlebia wernei	0	0	0	0	0	0
Physa fontinalis	0	0	0	0	0	1
Planorbis planorbis	0	0	0	0	0	0
Planorbis vortex	0	0	0	0	0	0
Platambus sp.	0	0	0	0	0	0
Polycentropus flavomaculatus	0	0	0	0	0	0
Potamophylax latipennis	0	0	0	0	0	0
Potamopyrgus jenkinsi	0	0	0	0	0	0
Rhyacophila dorsalis	1	0	0	0	0	0
Rithrogena semicolorata	0	0	0	0	0	0
Sericostoma personatum	0	0	3	4	24	16
Simuliidae	0	0	0	0	0	4
Sphaeriidae	0	0	4	0	0	0
Tinodes unicolor	0	0	0	0	0	4
Tipula sp.	0	0	0	0	0	0