

**THE EFFECTS OF HABITAT IMPROVEMENT TECHNIQUES
ON INVERTEBRATE COMMUNITIES IN TWO MANAGED
LOWLAND RIVERS**

by

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who bestows knowledge*

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ABSTRACT

The effects of habitat improvement techniques on invertebrate communities in two managed lowland rivers.

by Mohammad Ebrahimnezhad

Most rivers and streams in England and Wales have been channelised for reasons of flood control, drainage of wetlands and channel improvement for navigation. Channelisation is one of the most dramatic aspects of man's impact on the riverine system. It involves the direct modification of the river channel and has impacts on the environment and ecosystem of the river.

In order to decrease or ameliorate the impacts of channelisation and restore the riverine habitat, different restoration and rehabilitation techniques are used. Improvement structures in a river may produce many effects, but their overall function is to increase the diversity of the river habitat. Although the initial objectives of the installation of such structures are improvements to fish stocks and fisheries, they also improve biodiversity in invertebrates and plants and hence increase the conservation value.

The effectiveness of two river habitat improvement techniques; artificial riffles and current deflectors, installed in two rivers, Harper's Brook and the River Smite were studied by comparing the macroinvertebrate communities of a control site on each river with those of channelised and improved sites by means of benthic kick-sampling.

Eighty four samples from seven sites of Harper's Brook and 57 samples from five sites of the River Smite were collected in four seasons. One hundred and eight taxa with the total abundance of 210,045 from Harper's Brook and 101 taxa with the total abundance of 149,397 from the River Smite were identified.

The results of ANOVA showed that the sites in Harper's Brook were significantly different in mean diversity (H') and in general the mean diversity of natural and two artificial riffles were greater whilst those of run sites were smaller. The natural riffle of the River Smite had also the greater mean diversity than the other sites.

The results of DECORANA indicated that there was a clear separation between the riffle sites and run sites in all seasons in Harper's brook, and the separation of the natural riffle from the other sites in the River Smite. The results of TWINSpan agreed with DECORANA, indicating that the two artificial riffles were associated with the natural riffle and the third with the run sites.

The overall results showed that the riffle reinstatement was successful in improving the hydrology and biology of some sites close to or similar to that of a natural site. Current deflectors have also been successful in changing the hydrology of the river and creating pool and shoal sequences. Although the biology of the created shoal was not comparable to the natural riffle, it had a better biology compared with an unimproved site.

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CHAPTER ONE

Macroinvertebrate Biodiversity and Habitat Improvement

CHAPTER ONE – MACROINVERTEBRATE BIODIVERSITY AND HABITAT IMPROVEMENT

1.1 OBJECTIVES AND INTRODUCTION

The aim of this project was to study aspects of the ecological effects of different habitat improvement devices installed in channelised rivers. The initial objectives of such installations are improvements to fish stocks and fisheries, but they may also improve biodiversity in invertebrates and plants and hence have conservation value. This project investigated the effects of such structures upon the abundance and diversity of macroinvertebrates.

The effectiveness of two river habitat improvement techniques; artificial riffles and current deflectors, installed into two rivers, Harper's Brook and the River Smite were studied by comparing the macroinvertebrate communities of a control site on each river with those of channelised and improved sites.

Most rivers and streams of England and Wales have been channelised for reasons of flood control, drainage of wetlands and channel improvement for navigation. Channelisation is one of the most dramatic aspects of man's impact on the riverine system. It involves the direct modification of the river channel and has impacts on the environment and ecosystem of the river.

In a natural state, a river is in equilibrium having a diversity of habitats and micro habitats for invertebrates and other aquatic organisms. Once channelised, a river usually becomes straight, deep and homogenous. The uniformly inclined banks reduce stands of marginal vegetation which provide habitat for many aquatic and terrestrial organisms. Channelisation inevitably affects the entire food chain. The primary producers (algae, aquatic macrophytes, etc.), primary consumers (invertebrates) and secondary consumers (fishes), which are the top of the food chain of the river become diminished or destroyed.

In order to decrease or ameliorate the impacts of channelisation and restore the riverine habitat, different restoration and rehabilitation techniques are used and the consequences are now being studied.

In order to gain a background understanding of the biodiversity of macro-invertebrates in lotic waters, and their responses to habitat modification, the literature has been reviewed following two major topics. These are: 1) distributional ecology of macroinvertebrates, mostly related to substrate and flow, and 2) river channelisation and restoration devices.

1.2 DISTRIBUTIONAL ECOLOGY OF MACROINVERTEBRATES

1.2.1 INTRODUCTION

There are spatial differences in the distribution of stream benthos, not only for long stretches of stream but also within small sections. Factors affecting the distribution of stream benthos fall into two broad groups; biotic and abiotic. Both can be significant in determining where species can occur. The focus of this study is upon the abiotic factors. These factors (physical and chemical) are of fundamental importance for the occurrence of the biota. Existing abiotic conditions at any given location determine which species can be present as they must be adapted to the specific condition to survive. Abiotic factors vary from place to place. For each important abiotic factor a species possesses a range of tolerances within which it can survive and a narrower range within which it can reproduce (Townsend, 1980).

Many attempts have been made to evaluate the importance of each environmental factor which determines the occurrence and distribution of stream-dwelling macroinvertebrates. Many factors which affect distribution of stream macroinvertebrates exert their influence over a wide area and usually can be ignored or controlled when microhabitat is dealt with. For example, temperature, water chemistry and dissolved

oxygen may be considered to work together locally in a stream and attention therefore may be directed to the heterogeneous conditions within a small area of stream bottom. Here certain factors, most notably current velocity, substrate conditions and detritus distribution, are quite varied (Rabeni & Minshall, 1977).

The three principal environmental factors; substrate (Percival, 1929; Cummins, 1966; Cummins & Lauff, 1969), current velocity (Jaag & Ambuhl, 1964; Edington, 1968; Chutter, 1969; Degani, 1993) and availability of preferred food items (Scott, 1958; Egglshaw, 1964, 1969; Wallace & Merritt, 1980), considered the major determinants of macroinvertebrate distribution in lotic water, are discussed here.

1.2.2 THE EFFECT OF SUBSTRATE ON MACROINVERTEBRATE DISTRIBUTION

Substrate particles range from boulders and cobbles of a mountain streambed to the silts and sands that are more typical of lowland rivers, and many kinds of organic substrates, from minute organic fragments up to fallen trees, along with filamentous algae, mosses and macrophytes. In general, the substrate includes everything on the bottom or sides of streams or projecting out into streams, including a variety of human artifacts and debris, on which organisms inhabit (Minshall, 1984).

1.2.2.1 Mineral and organic substrates

The bed material of all streams consists primarily of inorganic particles ranging in size from silt to boulder and bed rock. The most easily quantified measurement of these particles is size. Thus direct measurement of particle diameter is obtained by sieving or from settling velocities. Cummins (1962) introduced to river ecology the scale of particle size classification (Table 1). Phi (Φ) is the negative \log_2 of the smallest diameter in each size group, in millimetres. Larger particles are easily sieved, but silt requires elutriation and settling. There is a general tendency for particle size to decrease as one proceeds

downstream. This is because, as the river slope declines progressively downstream the flow velocity decreases and smaller particle sizes settle out. In many regions one finds larger stones and boulders in rivers of mountainous areas, and sandy-silty bottom in lowland rivers. Mineral substrates have characteristics beyond their average size. The surface area of individual particles and the degree of texture are rarely quantified in field measurements. Texture in particular requires some arbitrary ranking scale making it somewhat subjective (Allan, 1995).

Table 1.1 Mineral substrate particle size classification (After Cummins, 1962; Allan, 1995).

Size Category	Particle Diameter (range in mm)	Phi (Ø) Value ($-\log_2$ smallest diameter)
Boulder	>256	≤ -8
Cobble		
large	128–256	-7
small	64–128	-6
Pebble		
Large	32–64	-5
Small	16 – 32	-4
Gravel		
Coarse	8–16	-3
Medium	4–8	-3
Fine	2–4	-1
Sand		
Very coarse	1–2	0
Coarse	0.5–1	1
Medium	0.25–0.5	2
Fine	0.125–0.25	3
very fine	0.063–0.125	4
Silt	<0.063	≥ 5

Very small organic particles (> 1 mm) usually serve as food rather than as substrate. Larger organic material, from plant stems to submerged logs, generally function as substrate rather than food. Autumn-shed leaves on the streambed are a substrate to macroinvertebrates that graze algae from their surfaces, and food to macroinvertebrates that eat the leaves themselves. More commonly, however, large organic substrates serve as attachment sites from which to capture food items transported in the water, as sites where fine detrital material accumulates and as surfaces for algal growth.

1.2.2.2 Substrate and macroinvertebrate abundance and diversity

It is generally believed that benthic invertebrates increase in numbers as particle size increases from sand to boulders (Bell, 1969). The increase in particle size is sometimes equated to an increase in the complexity of the substrate (Hynes, 1970). There is a positive relationship between substrate diversity and species richness. The more types of substrate, the more species are expected. Colonisation of mixed substrates always resulted in higher mean species richness than did colonisation of a single substrate type (Allan, 1975). Substrate particle size is a determining factor in the distribution of the benthos. This is reasonable, since boulders and cobbles provide more space and more diverse habitat than a sand and gravel substrate. Substrate stability and the presence of organic detritus are further factors which increase the diversity and abundance of macroinvertebrates (Percival, 1929; Egglisshaw, 1964).

Effects of large and stable substrates on fauna

Larvae of the water penny (Coleoptera: Psephenidae) in North America occur mainly on the undersides of rocks, and often under boulders in torrential flows. Attached forms require a substrate such as larger boulders which are not easily overturned by the current. The longer the faunal life-span, the more critical this is. Due to the slow growth of mosses, bryozoans and sponges are found mainly on larger boulders or in locations where scouring is infrequent (Allan, 1995).

Among molluscs, *Ancylus* (Gastropoda: Ancyliidae) occurs only on solid surfaces no matter how fast or slow the rate of flow. In North America the genus *Goniobasis* (Gastropoda: Pleuroceridae), on the other hand, requires a solid surface only if the current is swift, in slow flowing water it occurs also on silt and sand. The flat, limpet-like *Ancylus* can move around only on rock and stone, but the long narrow, *Goniobasis*, being heavy, only needs a solid surface when it is necessary to secure itself against the current (Hynes, 1970).

It is not clear why certain species seem to prefer particular substrates. In Britain the two very similar stoneflies, *Dinoceras cephalotes* and *Perla bipunctata* (Plecoptera:

Perlidae) divide certain river beds between them, the former occurring primarily in stable stony areas and the latter on unstable stones and gravels. The reasons for this are unknown, but it may be connected to colour, as the fairly uniform, darkly coloured *D. cephalotes* is less conspicuous on the generally darker-coloured stable substrata, while the strongly patterned *P. bipunctata* is more camouflaged against the varied colours of the shifting stones (Hynes, 1941). Some species of net-spinning caddis larvae and *Simulium* larvae also prefer a stony and stable substrata. These animals need a firm base for a sedentary way of life, and thus are not found where the substrate is unstable or fine-grained (Townsend, 1980).

Effects of silt and sand on fauna

Hynes (1960) described the two principal ways in which the fauna of streams and rivers may be affected by inert solids. First, suspended solids reduce light penetration and may therefore render all higher plant and algal growth impossible. There is then no food for herbivores and the detritus feeders have to rely on allochthonous detritus. Secondly, where inert solids settle out of the water they not only blanket algae, mosses and rooted plants, cutting off their light and oxygen supply, but they also alter biotopes in other ways. For instance the interstices between stones in the current may become clogged, obliterating the habitat of many animals.

Sand is generally considered to be a poor substrate, especially for macroinvertebrates, due to its instability and because tight packing of sand grains reduces the trapping of detritus and can limit the availability of oxygen. Nevertheless, a variety of taxa, termed psammophilous, are specialists of this habitat. The meiofauna, defined as invertebrates passing through a 0.5 mm sieve, can be very abundant, dwelling interstitially to considerable depths. Palmer (1990) reported meiofauna densities (rotifers, oligochaetes, early instar chironomids, nematodes and copepods) that averaged over 2000 per 10 cm² and at times reached nearly 6000 per 10 cm². The psammophilous fauna includes some macroinvertebrates as well, and they can exhibit distinctive adaptation, often associated with respiration. The dragonfly nymph *Lestinogomphus africanus*, found burrowing deep

in sandy-bottom pools in India, has elongated respiratory siphons that reach above the sand surface (Hora, 1928). On stony substrata the presence of silt reduces and changes the fauna. Sprules (1947) was able to observe this effect on insects when beavers built a dam across a stream in Ontario, on which he was quantitatively trapping insects. The dam raised the water level about 40 cm and caused the deposition of silt. This reduced the total number of insects emerging, especially of Ephemeroptera, Plecoptera and Trichoptera, and increased the proportion of Chironomidae.

Cummins & Lauff (1969) have found that silt, in small amounts, benefits at least some taxa. Working in North America they found that, when silt was added to larger mineral substrates in laboratory preference tests, silt enhanced the preference for coarse substrates in the mayfly *Caenis latipennis* and the stonefly *Perlesta placida* (Plecoptera). In large amounts, silt is generally harmful to macroinvertebrates. It causes scour during high flow, fills interstices thus reducing habitat space and exchange of gases and water, and reduces the algal and microbial food supply. Species such as *Ephemera simulans* (burrowing mayfly nymph), however, require fine particles into which its modified front legs can dig effectively (Townsend, 1980). Silt may be colonised by larger burrowing oligochaetes such as *Eiseniella tetraedra* (Lumbricidae) and species of Naididae and Tubificidae.

Effects of vegetation on fauna

The presence of vegetation also greatly affects the fauna. Percival (1929) found that there are more animals in moss, rooted plants, and filamentous algae than there are on stones, and that certain species or groups are confined to, or more abundant in, one or more of these habitats. Minckley (1963), in a study of a spring stream in Kentucky, showed that the presence of vegetation affects the fauna of nearby bare areas. His two upper stations were densely covered with beds of moss, *Fissidens* sp., which housed enormous populations of the isopod *Asellus bivittatus* and the amphipod, *Gammarus minus*. Further downstream, where the *Fissidens* was rare, the isopods and amphipods in the riffles were much scarcer, and their place was taken by Ephemeroptera, and Diptera,

mostly Chironomidae. Minckley's study also showed that there were considerable differences between different types of plant. Genera such as *Nasturtium*, *Myriophyllum*, and *Myosotis* contained relatively, and absolutely, fewer *A. bivittatus* than did *Fissidens* and, presumably because of the silt they collect round their roots, many more oligochaetes. Other similar differences can be seen; for instance molluscs, mostly genus *Goniobasis*, were much commoner in the loose growth of the higher plants than they were in the moss. It is also clear that all types of plants were more heavily colonised than were the non-vegetated areas of substrate.

Townsend (*op. cit.*) pointed out that, in regions of silty substrates, rooted macrophytes such as *Potamogeton pectinatus* and *Elodea canadensis* are often common. The distribution of many invertebrates is related to the presence of macrophytes. In general, more individuals and more species of animals occur on plants than on nearby mineral substrates. He stated that some species or groups are confined to macrophytes. For example, larvae of the chironomid *Eucricotopus brevipalpis* feed only on *Potamogeton natans*. Most invertebrates associated with macrophytes do not, however, feed on the plant tissue. Some graze on epiphytic algae which grow on the surface of the macrophyte, others use the plants as a stable site from which to filter food from passing water and others are predators. Townsend (*op. cit.*) also mentioned that artificial weeds, made of plastic or string, generally become colonised by the same invertebrate community as their natural forms. This emphasises the role of macrophytes as a substratum rather than a food supply. The plants are also important to many fish in providing a surface to which they attach their eggs.

Conclusions

Substrate is one of the most important abiotic factors influencing the microdistribution, diversity and abundance of macroinvertebrates in running waters. The substrate includes everything on the bottom and sides of the stream; from boulder, gravel and sand (inorganic substrates) to vegetations such as macrophytes, algae and dead plant materials (organic substrates). Determination of the role of substrate is further complicated

by its tendency to interact with other environmental factors. For example, slower currents, finer substrate particle size and (possibly) lower oxygen are often correlated. In addition, the size and amount of organic matter, which affect algal and microbial growth, vary with substrate.

The idea that a mixed substrate provides more kinds of living places, and therefore can support a greater variety of invertebrates than a simple one, is evident in the writings of many freshwater ecologists (e.g. Sprules, 1947; Hynes, 1970). In general, diversity and abundance of benthic invertebrates increase with median particle size, and some evidence suggests that diversity declines with stones at or above the size of cobbles (Minshall, 1984). Stability of the substrate is an important factor in the distributional patterns of stream invertebrates. Reduced species richness and abundance are commonly associated with areas of shifting substrate.

1.2.3 THE EFFECTS OF CURRENT VELOCITY ON MACROINVERTEBRATE DISTRIBUTION

1.2.3.1 Introduction

Water velocity and the associated physical forces collectively represent perhaps the most important environmental factor affecting the organisms of running water. The speed of the current influences the size of particle of the substrate (the higher the current speed the larger substrate particle size can be carried). Current affects food resources *via* the delivery and removal of nutrients and food items. Finally, organisms within the water column and at the substrate surface experience a physical force exerted directly by the current velocity. Because most organisms of running waters live near the river bed and not in the middle of the water column the complexities of flow around obstructions and near the stream bed are of particular importance (Allan, 1995).

1.2.3.2 Physical characteristics of flow velocity

In rivers, the intensity of physical factors vary from source to mouth. The following

equation describes the longitudinal changes in flow velocity (Townsend, 1980):

$$V = (8 g s d / f)^{1/2}$$

Where, V= mean flow velocity

g= gravitational constant

s= river slope

d= mean flow depth

f= resistance to flow (from river bed and banks)

The river slope decreases downstream while depth and volume of discharge increase, due to the additional input from adjoining tributaries. Resistance to water flow also decreases in response to increasing depth and the reduction of substrate particle size, and this also tends to increase mean flow velocity.

Flow velocity varies across the width of the channel. It is usually highest near the middle and reduced at banks and among macrophytes. Flow velocity also changes down the length of short stretches. In small streams, for instance, shallow fast-flowing riffles alternate with deeper pools. It also varies with time, in response to fluctuation in discharge during wet and dry periods of weather.

1.2.3.3 Macroinvertebrate adaptations to current velocity

Aquatic invertebrates exhibit a number of anatomical features that apparently enhance their ability to move about, or secure them against water flow. The functional benefits of such features are not always clear (Hynes, 1970). Direct attachment devices including silk and other sticky secretions, hooks and suckers help animals hold their position against the current. Blepharocerid larvae (Diptera) which occur in northern hemisphere mountains are found on smooth rocks in fast-flowing water, where their row of six ventral suckers allows them to move against very high current velocities. Simuliidae larvae as another example are able to occupy high-velocity habitats by spinning a mat of silk onto a stone surface to which they attach with specialised prolegs. Circlets of outwardly directed hooks on both anterior and posterior prolegs aid the larvae in attachment and movement. Chironomids and other dipteran larvae, and caddis larvae also use silk to attach their pupal

case to stones while they moult (Allan, 1995).

There is a relatively still boundary just above the substrate of the stream bed and some species are able to live there or in narrow crevices under stones through the extreme flattening of their bodies. In this way they avoid the current. Townsend (*op. cit.*) stated that mayfly nymphs of the genus *Baetis* have a dorsal surface which closely approaches the ideal streamlined shape (a fusiform body, widest at about 36% along its length and tapering to a point at the rear), and therefore offers little resistance to flow. The long tail cerci swing from side to side and rather like a rudder, serve to keep the body facing into the current. Townsend (*op. cit.*) also suggested that caddis larvae that are typical of fast-flowing water, build their cases out of large sand grains and gravel; this has the effect of weighing them down.

1.2.3.4 Macroinvertebrates and current velocity relationships

Current velocity is an important factor which controls the occurrence and abundance of species and therefore the whole structure of the animal community (Jenkins *et al.*, 1984; Scott, 1958). Many macroinvertebrates rely on the current either for feeding or because their respiratory requirements demand it. Current velocity is important to the benthic invertebrate because it governs the rate of oxygen renewal to the boundary layer between water mass and the stream substrate (Ambuhl, 1959). Logically the faster the current, the faster and more efficient the renewal rate. The speed of the current also determines the composition of the substrate, which in turn affects the fauna. Larger particles such as rocks, settle out in fast water and finer particles such as sand, accumulate on the river bed in slower flowing reaches.

Current velocity can also directly affect the constitution of the fauna. Edington (1968) found that some species of caddisfly larvae like *Wormaldia* spp., *Philopotamus montanus* (Trichoptera: Philopotamidae), *Diplectrona felix* and *Hydropsyche instabilis* (Trichoptera: Hydropsychidae) occur predominantly in rapids. Other species like *Polycentropus flavomaculatus* and *Plectrocnemia conspersa* (Trichoptera: Polycentropodidae) occur predominantly in pools. The separation of the larvae into

'rapids' species and 'pools' species seems to be related to water velocity. For example *P. conspersa* typical of stream pools and sheltered sites in rapids, usually occurs in the velocity range 0 – 20 cm s⁻¹, while *H. instabilis* is found in the more exposed rapids within the range 15 – 100 cm s⁻¹.

There is some evidence in the literature for respiratory differences between the two groups of species. Philipson (1954) found that *Polycentropus flavomaculatus*, a low velocity species, could utilise oxygen at low concentrations even in still water. On the other hand *H. instabilis*, a high velocity species, could not survive at low oxygen concentration unless the water was flowing. He suggested that *H. instabilis* is specialised to use oxygen at a rapid rate. High velocity species in general may need to expend extra energy to maintain in position. As there is more oxygen available to respiratory surfaces at high water velocities, this probably satisfies the animal's needs.

Harrod (1964) found that a high flow rate was an absolute requirement for larval Simuliidae. All species of this family are filter feeders and have modified mouthparts that form a fan-like structure. In still or slow flowing water, however, Simuliidae can only keep the fan open at will for a short period. Harrod found that a current of 19 cm s⁻¹ or more was needed for the force of the water to keep the structure open without expenditure of energy by the animal. The results of the laboratory experiments of Phillipson (1956) on the effect of velocity on larvae of *Simulium ornatum* showed that they aggregate in the velocity range 50 – 120 cm s⁻¹, the greatest number occurring between velocities of 80 – 90 cm s⁻¹. The main advantage accruing from this preference for certain velocities is probably an increased food supply rather than increased oxygen.

Minshall & Minshall (1977) working in North America showed that, there were three different relationships of invertebrate populations to current velocity. *Baetis intermedius*, *Epeorus* sp., *Cinygmula mimus* (Ephemeroptera: Heptageniidae) and *Optioservus quadrimaculatus* (Coleoptera: Elmidae) increased in numbers as velocity increased; those of the genus *Capnia* (Plecoptera: Capniidae), *Paraleptophlebia heteronea* (Ephemeroptera: Leptophlebiidae) and *Ephemerella inermis* (Ephemeroptera: Ephemerellidae) decreased. The third type of response, seen in the genus *Alloperla*

(Plecoptera: Chloroperlidae) showed an optimum in mid-range, with the numbers declining on either side. Similar responses to current have been documented by Ambuhl (1959). The data indicated that the smaller number of *Baetis intermedius* in the pools than in the riffles was due to the reduced current in the former habitat.

These are just a few examples from numerous studies made by different investigators, about the effects of current velocity on stream macroinvertebrates. Clearly the way in which current velocity affects animals is very complicated and a detailed review of the related literature is beyond this general introduction.

1.2.3.5 Conclusions

Current speed is a factor of major importance in running water, and it controls the occurrence and abundance of species and hence the whole structure of the animal community. Its mode of action is, however, complex; it is variable in time and over very short distances, and it is almost impossible to quantify except in general terms (Hynes, 1970).

The influence of current on the biota remains poorly understood. With regard to adaptive morphology and behaviour the principal features of many organisms are inherited within a taxonomic lineage and therefore subject to multiple selective forces. Current has indirect as well as direct effects on organisms. Water velocity affects substrate size composition, the delivery of gases and food items and other environmental factors, making causation difficult to ascertain even with imaginative experimentation. Lastly but importantly, we are just beginning to understand the actual forces that organisms experience, and this, of course, limits our interpretation. A better understanding is needed of the complexities of flow, of velocities near the streambed and around obstructions, and of the fluid forces that organisms actually experience (Allan, 1995).

1.2.4 THE EFFECTS OF FOOD AVAILABILITY ON MACROINVERTEBRATE DISTRIBUTION

The availability of food is an obvious factor controlling the occurrence and abundance of species. In general, species occur, or are more common, only where their food is readily available.

1.2.4.1 Food resources of stream invertebrates

There are two sources of energy to lotic food webs: autotrophic and heterotrophic.

Autotrophic sources

Autotrophs are organisms that acquire their energy from sunlight and their materials from non-living sources. Periphyton, macrophytes and phytoplankton constitute three very different groups of autotrophs occurring in streams and rivers.

The periphyton, comprising an abundance of Bacillariophyceae (diatoms), Chlorophyta (green algae), Cyanobacteria (blue-green bacteria) and a few other groups, occur on virtually every surface in running waters including stones (epilithon), soft sediments (epipelon) and macrophytes (epiphyton). Epipellic taxa form films or mats on silt and mud bottoms, and typically are mobile and easily swept away by increased current. Epiphytic taxa occur on macrophytes, particularly angiosperms, where such loading can be detrimental to the host plant. Unlike epipellic species, epiphytic and epilithic taxa are usually firmly attached by mucilaginous secretions or via a basal cell and stalk securing them against currents except when the flow increases substantially (Allan, 1995).

Flowering plants, mosses and liverworts, a few species of encrusting lichens, the Charales and other large Chlorophyta constitute the macrophytes of flowing waters (Hynes, 1970). They are found mainly where neither the depth nor current is great. Rivers of intermediate size, canals and river margins usually support the greatest biomass of these groups. Macrophytes can be classified according to their growth form. Four major growth forms are recognized by Westlake (1975):

(1) Emergent plants – These occur on river banks and shoals. They are rooted in soil that is close to or below water level much of the year, and their leaves and reproductive organs are aerial; e.g. *Glyceria maxima*, *Phragmites communis*.

(2) Floating-leaved plants – These are rooted in submerged soils with many of their leaves floating on the water surface and their reproductive organs floating or completely aerial; e.g. *Nuphar luteum*, *Potamogeton natans*. These are usually seen along the margins of slower-flowing rivers.

(3) Free-floating species – These are normally not attached to the substrate and can form large mats, often entangled with other species and debris, in slow rivers, e.g. *Ceratophyllum demersum*, *Lemna minor*.

(4) Submerged taxa – These are attached to the substrate, their leaves are entirely submerged and they typically occur in midstream unless the water is too deep, e.g. *Elodea canadensis*, *Ranunculus calcareus*, *Vallisneria spiralis*.

Small autotrophs suspended in the water column and transported by the current, including algae, protists and cyanobacteria, comprise the phytoplankton. In small, fast-flowing streams, sloughing of attached autotrophs is probably the main food source for primary consumers and any cells in the water column are simply this material in transit (Swanson & Bachman, 1976). In sluggish, lowland streams, inside channels and within macrophyte beds and in rivers of considerable length, however, the residence time of a water mass can be sufficient for true plankton to colonise and reproduce. Under these conditions large populations of phytoplankton and zooplankton can develop (Hynes, 1970).

Heterotrophic sources

Heterotrophs obtain energy and materials by consuming living or dead organic matter. All animals are heterotrophs, as are fungi and those bacteria that gain nourishment through the processing of dead organic matter. Particulate and dissolved non-living organic matter are important energy inputs to most food webs, and this is especially true in running water ecosystems. While primary production by the autotrophs of running waters

can be substantial, much of the energetic support of lotic food webs derives from non-living sources of organic matter. These energy pathways are referred to as heterotrophic and the immediate consumers of this material are decomposers and detritivores, in contrast to autotrophic pathways linked to higher trophic levels by herbivores (Allan, 1995).

Heterotrophic production requires a source of non-living organic matter, and the presence of microorganisms (bacteria, fungi) to break down the organic matter and release its stored energy. Plant litter and other coarse debris that falls or blows into stream channels, collectively called coarse particulate organic matter (CPOM, > 1 mm), fine particulate organic matter (FPOM, 1 mm to 0.5 μm) that originate from many sources including the breakdown of larger particles and dissolved organic matter (DOM, < 0.5 μm) constitute the three main categories of non-living organic matter (Townsend, 1980). Some of this material originates within the stream (autochthonous) (such as dying macrophytes, animal faeces, extracellular release of dissolved compounds) and some is transported into the stream from outside (allochthonous) (such as leaf fall, soil particulates and compounds dissolved in soil water). Collectively these sources can substantially exceed the energy transformed within a stream by photosynthesis (Allan, *op. cit.*).

Allochthonous organic matter enters a particular river stretch either from upstream in the water column or from the surrounding catchment area in groundwater, overland water flow or wind. The input consists of two principal components: organic matter dissolved in the water and coarse particles such as tree leaves, fruits and twigs. Of the autochthonous production, only an insignificant proportion of macrophytes are eaten alive. Most enters the CPOM compartment and is processed in the same manner as its allochthonous counterpart. Aquatic microphytes may be consumed alive by herbivorous grazers, such as limpets (*Ancylus* spp.) and certain stonefly nymphs (e.g. *Brachyptera* spp.), or they may be filtered or gathered with other FPOM by the collectors (Townsend, *op. cit.*).

A fraction of CPOM is quickly lost to the DOM compartment by leaching. The remainder is converted by three processes to FPOM. First, mechanical disintegration by battering and abrasion causes a break up of the particles. Secondly, processing by microorganisms causes gradual break down of particles. Thirdly, the invertebrate

shredders fragment the large particles and produce large quantities of faeces which comprise a component of the FPOM compartment. FPOM constitutes the food supply for invertebrate collectors (Townsend, *op. cit.*). DOM also derives from many sources, mostly outside the stream channel, and often is the largest single pool of organic carbon in lotic ecosystems. Much DOM is, however, a poor source of energy to the bacteria that are its primary entry point to the biota (Allan, *op. cit.*).

1.2.4.2 Macroinvertebrate trophic relationships

Food is the ultimate determinant of macroinvertebrate distribution and abundance in non-perturbed running waters (Cummins, 1975). Since, however, the majority of macroinvertebrates are non-selective feeders (Hynes, 1970; Cummins, 1973), taking in a wide range of food substances of acceptable particle dimensions, the specific details of a population's distribution pattern within a section of stream having a suitable food supply may often be controlled by such factors as sediment particle size, current, competition for space and predation.

Macroinvertebrates are classified into four functional feeding groups based on feeding mechanisms rather than food eaten (Cummins, 1975; Cummins & Klug, 1979; Townsend, 1980):

- (1) Grazer and scraper – herbivores feeding on attached algae.
- (2) Shredders – large particle feeding detritivores.
- (3) Collectors—both suspension (filter) and deposit (surface) fine particle feeding detritivores.
- (4) Predators – carnivores.

Grazers and scrapers (consumers of autotrophs)

The living primary producers are consumed by both grazers and piercers. The former ingest periphyton and the latter macrophytes. Piercers refer primarily to the micro-caddisflies (Hydroptilidae), which pierce individual cells of algal filaments and imbibe cell fluids (Cummins & Klug, *op. cit.*). Of significance in the grazing pathway are the

periphyton mat and scraping mouthparts. Scraping of surfaces is an important feeding role, complete with specialised structures including the rasping radula of gastropods (e.g. *Ferrissa* spp.) and the modified mandibles of caddis larvae such as *Neophylax* spp. (Limnephilidae) and *Glossosoma* spp. (Glossosomatidae), the beetle larvae of the genus *Psephenus* of North America and some ephemeropterans. In order to remain attached to exposed surfaces in fast and turbulent water, scrapers have evolved modifications such as dorso-ventral flattening. The North American mayfly, *Ephemerella doddsi* also has a ventral sucker formed by its gills (Cummins & Klug, *op. cit.*).

Shredders (consumers of CPOM)

The trophic pathway involving CPOM which has been most studied, is the consumption of autumn-shed leaves in woodland streams by invertebrates (Cummins, 1973; Cummins & Klug, *op. cit.*). Invertebrates that feed on decaying leaves include crustaceans (especially amphipods, isopods, crayfish and freshwater shrimp), gastropods and several groups of insect larvae (Cummins *et al.*, 1989). The last group includes tipulid larvae and several families of trichopterans (Limnephilidae, Lepidostomatidae, Sericostomatidae, Oeconesidae) and plecopterans (Peltoperlidae, Pteronarcidae, Nemouridae).

The shredding role is important not only in providing a larger surface area for microbial activity, but also in contributing to FPOM and thus making food available to the collectors. Short & Maslin (1977) demonstrated this in an experiment using leaves of an alder (*Alnus rubra*) labelled with the radioactive isotope of phosphorus (^{32}P) in an artificial stream. In the presence of a shredder (the stonefly, *Pteronarcys californica*), two collector species (larvae of the net-spinning caddis, *Hydropsyche californica* and the black fly, *Simulium arcticum*) were found to accumulate a significantly larger amount of ^{32}P in their bodies than when the shredder was absent.

Invertebrate detritus feeders unquestionably prefer leaves that have been conditioned by microbial colonisation in comparison to uncolonised leaves. The benefits to the consumers include greater efficiency of converting ingested leaf biomass into consumer

biomass and a higher individual growth rate (Lawson *et al.*, 1984).

Woody debris has been largely ignored as an energy source, because few invertebrates feed on it directly and wood appears to be a poor food. Nevertheless, wood can contribute 15–50% of the total litter fall in small, deciduous forest streams, and even more in coniferous regions (Anderson & Sedell, 1979). Although its importance diminishes downstream and it is utilised only very slowly (a residence time of years to decades, in comparison with weeks to months for leaves), wood provides food and habitat for many species. Anderson *et al.* (1978) found some 40 taxa associated with this resource in wood rich Oregon streams. Prominent aquatic xylophages included one that colonise phloem of newly fallen branches (chironomid larva, genus *Brilla*) and a tipulid larvae (*Lipsothrix* sp.) that ate partially decomposed woody material. A further two species were found that gouged the microbially conditioned surface of water-logged wood (the elmids, *Lara* sp. and the caddis, *Heteroplectron* sp.). In comparison with leaves, invertebrate standing crop biomass on wood was about two orders of magnitude lower per kilogram of substrate.

Lotic consumers are relatively unspecialised for xylophagy. The elmids beetle *Lara avara* which lives in montane streams of western North America possesses robust mandibles capable of slicing away thin strips of wood, but apparently lacks digestive enzymes or gut symbionts to aid digestion. Microscopic inspection of material progressing through the gut indicated no change to the wood (Steedman & Anderson, 1985). Presumably the larva is nourished by microbiota and their exudates occurring on the wood surface. Not surprisingly, *L. avara* grows very slowly and requires 4–6 years to attain maturity.

Collectors (consumers of FPOM)

There are two sub-categories of collectors. The collector-gatherers, including many larval mayfly species, oligochaetes and orthoclad chironomid larvae, inhabit areas where FPOM is abundant on the stream bed and simply ingest whole particles. The collector-filterers, such as hydropsychid and *Simulium* larvae, sieve FPOM from the flowing water

(Townsend, 1980).

Collectors exhibit a wide range of morpho-behavioural adaptations for acquiring fine particle detritus. Cummins & Klug (1979) state that animals feeding on FPOM in suspension are called filtering collectors to distinguish them from gathering collectors which feed on deposited, sediment-related detritus.

Caddisfly larvae in the superfamily Hydropsychoidea (which includes the Philopotamidae, Psychomyiidae, Polycentropodidae and Hydropsychidae) spin a variety of intricate silken capture nets (Wallace & Merritt, 1980).

Philopotamid larvae construct elegant, sac-like nets which have the smallest mesh openings known for any trichopteran larva (as small as $0.4 \times 0.4 \mu\text{m}$ for *Wormaldia* and $0.5 \times 5.5 \mu\text{m}$ for final instar *Dolophilodes* and some *Chimarra*) (Wallace & Malas, 1976). A philopotamid net may be composed of millions of individual meshes and each larvae possesses the ability to secrete about 70 silk strands simultaneously. Larval food, primarily fine detritus and diatoms, is swept from the net's surface with a flexible, brush-like labrum (Williams & Hynes, 1973). European *Plectrocnemia* spp. occur in regions of low current velocities ($< 10 \text{ cm s}^{-1}$) and construct large nets lacking organised meshes. The nets are apparently used more as snares for trapping benthic prey than as sieves for drifting organisms (Hildrew & Townsend, 1976).

Net-spinning hydropsychid larvae are a predominant group of lotic insects throughout the world and comprise about 80% of all trichopterans in larger North American streams (Wallace & Merritt, 1980). Most hydropsychid larvae construct fixed silken nets, perpendicular to the current, that are used to capture food. Hydropsychid larvae respond readily to changes in velocities and the proportion of larvae that spin nets decreases with a reduction in current velocity or temperature (Philipson, 1969). Apparently, specialised head setae are current-sensing devices. The general pattern suggests that increased filtration rates compensates for lower seston-capture efficiencies (larger meshes at faster flows); whereas smaller meshes, although more efficient at seston-capture, filter less water per time interval due to velocity-resistance limitations imposed by their small meshes.

Simuliid larvae are highly specialised suspension feeders. They have been studied extensively because the adults include important disease vectors and nuisance pests. Larvae attach to the substrate in rapid, often shallow water. The paired cephalic fans of suspension-feeding larvae each consist of primary, secondary and medial fans. It is thought that particles are snared by sticky material on the primary fans, which are the main suspension-gathering organs, while secondary and median fans act to slow and deflect the passage of particles. Food items are removed by a combing action of the mandibular and labral bristles. These adaptations to a filtering existence are lacking in some Simuliidae species that scrape substrates instead. Fans are opened for feeding and closed at other times (Allan, 1995).

Other dipteran families with representatives adapted to a suspension-feeding existence in running waters include the Culicidae, Dixidae and Chironomidae (Wallace & Merritt, 1980). Some Chironominae construct tubes or burrows with catchnets and create a current by body undulations; others such as the genus *Rheotanytarsus* feed passively on suspended particles by means of a sticky secretion supported by rib-like structures on the anterior end of the case.

Mechanisms of deposit feeding on FPOM are either less diverse in comparison with the suspension feeding mode, or less is known about the subject. Deposit feeders are also called collector-gatherers in the functional group classification (Cummins 1973). Deposit feeding is well-represented in most running water environments in terms of both species and total abundances. Among the macroinvertebrates in swifter streams, representatives of the Ephemeroptera (e.g. *Caenis* spp., *Ephemerella ignita*), Trichoptera, Chironomidae (e.g. *Paratendipes albimanus*, *Stictochironomus annulicrus*, *Brillia flavifrons*) Crustacea and Gastropoda are prominent deposit feeders. In slower currents and finer sediments oligochaetes, nematodes and other members of the meiofauna might also be found. It would not be possible for these animals to all feed in the same way and consume the same food. As well as possessing different food-gathering adaptation, deposit feeders live either on the surface or in the sediments. They also differ in a variety of other ways such as mobility and body size, their ability to produce mucus and in their digestive capabilities

(Allan, 1995).

Predators

This group includes all macroinvertebrates that are adapted specifically for the capture of live prey. Representatives of other functional groups (shredders, collectors, scrapers) may ingest live animals whilst feeding. Most predators engulf their prey entire or in pieces, but hemipterans and rhagionid dipteran larvae have piercing mouthparts (Cummins, 1973). Other distinctions can be made between hunting by ambush versus searching (Allan, 1995), and whether prey are obtained from suspension, as in large hydropsychids, or strictly from the substratum, as in flatworms.

Among carnivores, as in most habitats, there is a fair amount of selection. For example, the one large species of *Perla* sp. (Plecoptera: Perlidae) in Britain certainly selects simuliid and chironomid larvae, and possibly also baetid nymphs, in preference to other animals. Some groups, most notably amphipods, mites, and elmids beetles, appear to be largely avoided by invertebrate predators, even where they are abundant (Hynes, 1970).

1.2.4.3 Conclusions

Food resources are divided on the basis of particle size and whether active (prey), stationary (periphyton, vascular plants, deposition detritus), or in suspension (plankton and fine particle detritus in standing waters, particulate drift in stream and rivers). The information at hand supports the consensus that most aquatic macroinvertebrates are best termed polyphagous or generalists and that availability, most frequently defined by food particle size and texture, is the key to trophic relationships among aquatic invertebrates (Cummins, 1973).

A variety of feeding strategies are used by stream macroinvertebrates to compensate for changing dietary sufficiency of ingested foods. Shredders, and possibly some collectors, feed preferentially on particulate organic detritus colonised by microorganisms, utilising the associated microorganisms and partially hydrolysed (microbially digested)

substrate. Collectors, scrapers and facultative shredders increase the consumption of low quality food to compensate for its decreased nutritional benefit (Cummins & Klug, 1979).

As mentioned earlier, since the majority of macroinvertebrates are non-selective feeders, distributions of a taxon within a section of stream having a suitable food supply may often be controlled by such factors as sediment particle size, current, competition for space and predation.

1.2.5 THE RIFFLE-POOL HABITAT AND MACROINVERTEBRATE DISTRIBUTION

1.2.5.1 Introduction

Since artificial riffles are studied in this research as one of the habitat improvement devices, this section focuses upon riffle-pool habitats.

Riffle and pool can be identified as distinct habitats in most streams based primarily on flow, depth and slope of the water surface, although the method of their formation, their specific physical characteristics (e.g. substrate composition) and the relative percentage of the stream area that can be classified as pool or riffle habitat varies considerably among stream channel forms. Distinct, regularly-spaced alluvial gravel riffles and pools are characteristic of the middle reaches of most rivers, and are the dominant physical feature of the streams of mid-continental North America and Europe (Brussock *et al.*, 1985). Riffles are characterised by having a greater than mean velocity, a less than mean depth and substrate composed of gravel-cobble. Pools are characterised by having a less than mean velocity, a greater than mean depth and substrate composed of silt-sand.

1.2.5.2 Factors involving differences between riffle and pool

Although pools and riffles differ most obviously in flow rate and depth, other less obvious factors may also influence their suitability as habitats for various macroinvertebrate species. Availability of suitable substrate is very important (Minshall & Minshall, 1977). A coarse substrate is suitable for some taxa, while a fine substrate is

necessary for others. The frequency, severity and intensity of disturbances vary between riffles and pools and may also have significant impacts on their community structures (Resh *et al.*, 1988). Invertebrates may be more subject to fish predation in pools than riffles, especially if the pools have fewer interstitial refugia because of a lack of coarse substrate.

1.2.5.3 Distribution of taxa in riffles and pools

Several studies of mountain streams that compared erosional (equivalent to riffles) and depositional (equivalent to pools) zones have reported greater diversity, abundance and biomass in riffles (Minshall & Minshall, 1977; Rabeni & Minshall, 1977; Ormerod, 1988) but with less distinct differences between riffle and pool faunal assemblages or densities. A few studies on the mountain stream channel form reported equal or greater numbers or biomass of macroinvertebrates in depositional areas (Egglishaw & Mackay, 1967; Armitage *et al.*, 1974; Hynes *et al.*, 1976; Armitage, 1976). Similarly, lowland sand bed channel streams apparently have significantly higher numbers, biomass and diversity of invertebrates in pools (McCulloch, 1986).

A wide variety of environmental factors affect in-stream distributions (Minshall & Minshall, 1977; Brown & Brown, 1984). In a comparison between the fauna of riffles and pools in a number of studies from North America and the United Kingdom, Logan & Brooker (1983) found that riffles most often had a greater number of organisms and total biomass. However, the majority of these studies dealt with upland streams with predominantly stony substrates (McCulloch, 1986).

McCulloch (*op. cit.*) in his study of riffle–pool communities of two east Texas streams found that Chironomidae comprised the greatest percentage of both sample communities, with all genera reaching highest numbers in pools. Approximately 40% of the numbers in pools were chironomids, which made up less than 20% in riffles. Species of *Ablabesmyia*, *Polypedilum*, and *Rheotanytarsus* all had significantly higher numbers in pools while *Chironomus*, *Cryptochironomus*, *Tribelos*, *Paralauterborniella*, and *Paratendipes* were absent from riffles. Hunt (1930) found that Chironomidae accounted

for 91% of the pool fauna in Otter Creek, Oklahoma, with Harrel (1969) reporting values ranging from 19 to 31% (in the same river). Armitage *et al.* (1974) and Minshall & Minshall (1977) observed greater numbers of chironomids in pools, as did Hynes *et al.* (1976). The other major component of the Diptera are the Simuliidae. Most studies indicate that species of *Simulium* are restricted to, or are more abundant in riffles (Hunt, 1930; Harrel, 1969; Armitage *et al.*, 1974; Hynes *et al.*, 1976), with distribution influenced by current velocity (Phillipson, 1956, 1957). The genus *Simulium*, however, showed no significant difference in numbers between riffles and pools in two east Texas streams studied by McCulloch (*op. cit.*). This was due to the sandy unstable riffle substrate.

McCulloch (*op. cit.*) found that mayflies did not exhibit a clear preference for pools in both streams, with significantly higher densities only in pools of Alazan Creek. This result contrasts with Logan & Brooker (1983), who found that Ephemeroptera was the only major group with significantly higher densities in riffles than in pools. Ephemeroptera as a group are adapted to a wide range of current velocities and stream habitats (Crisp & Crisp, 1974), and it is this variety among taxa that probably led to these contrasting results. Major differences between ephemeropteran assemblages would make comparisons difficult, but a number of genera were collected during the McCulloch study which displayed similar distributions in other studies. Species of *Leptophlebia*, *Ephemerella*, *Stenacron*, *Caenis* and *Hexagenia* all had greater numbers in pools. Armitage *et al.* (1974) found that Leptophlebiidae were always more common in pools, and showed that species of *Leptophlebia* inhabit sheltered areas of reduced flow. Harrel (1969) also found that species of *Caenis* occur in higher numbers in pools. *Caenis* as well as *Ephemerella*, are well adapted to a life in a region of high silt deposition, with gills placed dorsally on their abdomen, and in the case of *Caenis* and some species of *Ephemerella*, covered by large operculate gills (Eastham, 1932). Eastham (*op. cit.*, 1937, 1939) also reported that burrowing mayflies possess gill and body adaptations which enabled them to keep the body free of fine particles. As a result, species of Ephemeridae in general, are found in higher numbers in pools (Harrel, 1969).

The ephemeropteran genus that displayed the greatest preference for riffles was *Baetis*, with significantly higher numerical densities in riffles at both sampling locations (McCulloch, 1986). Species of *Baetis* have been reported by numerous authors as restricted to or more abundant in riffles (Egglishaw & Mackay, 1967; Harrel, 1969; Armitage *et al.*, 1974; Hynes *et al.*, 1976), and are felt to be more highly adapted to a habitat exposed to rapid currents than other ephemeropterans.

Edington (1968) pointed out that some species of net-spinning caddis larvae occur predominantly in riffles. This group includes *Wormaldia* spp., *Philopotamus montanus*, *Diplectrona felix* and *Hydropsyche instabilis*. Another group is made up of species which occur predominantly in pools. In this category are *Polycentropus flavomaculatus* and *Plectroemia conspersa*. Observations made by Edington in other regions where *Hydropsyche fulvipes* is more common showed that it also occurs on riffles and *Plectrocnemia geniculata* probably occurs in pools. The results of McCulloch's (1986) study have shown that of the caddisflies collected, *Cheumatopsyche* had significantly higher numbers in riffles at both sampling sites, while species of *Hydatophylax* and *Psncnopsyche* were absent from riffles.

Coleoptera and Odonata were significantly higher in pools at both Alazan Creek and Bernaldo Bayou (McCulloch, *op. cit.*). Species of *Hydroporus* (Dytiscidae) and *Ancyronyx* (Dryobidae) were collected only in pools, while *Stenelmis* (Elmidae) had significantly higher pool densities at Alazan Creek. Other coleopteran genera displaying higher densities in pools were *Helichus*, *Heterelmis*, and *Macronychus* (Elmidae). *Gomphus* was the only dragonfly displaying a significant preference towards pools at both locations. The genera *Progomphus* and *Boyeria* attained their greatest numbers in pools, while *Hagenius* was restricted to pools. Damselflies were represented by the genera *Agrion* and *Argia*, both of which had higher numerical densities in pools.

1.2.5.4 Conclusions

It could be concluded that riffles and pools are amongst the most distinct features of most streams and rivers. The physical characteristics of riffles and pools are different.

Riffles are known to be in the erosional part of the river with greater mean flow, less depth and larger substrate particle size (e.g. cobble, gravel) whilst pools are the depositional opposites. The faunal composition of riffle and pool is normally different, because some taxa are restricted to the erosional part and some to the depositional part of the river. However, some taxa do not show much preference for riffle or pool and therefore it is normally the physical characteristics of the habitat which determines the occurrence of any taxon.

1.3 RIVER CHANNELISATION AND RESTORATION

1.3.1 INTRODUCTION

The impact of man on river and river channels has been widespread throughout the period of habitation of the planet. According to Brookes (1988) water supply and land drainage schemes were implemented as early as 3200 BC and the Hwang He (Yellow River) in China has been regulated for at least 4000 years. Prior to the 11th century in Europe, embankments were systematically built for flood control and land reclamation, and primitive weirs were constructed to regulate river flows for water power, whilst in Britain river regulation was widely used by Domesday times. Concern for the morphological, hydrological and biological impacts of man on rivers has been reported by many researchers (e.g. Hynes, 1960, 1970; Whitton, 1975).

Channelisation is the term used to embrace all processes of river channel engineering for the purposes of flood control, drainage improvement, maintenance of navigation, reduction of bank erosion or relocation for highway construction (Brooker, 1985). In Scandinavia watercourses have been cleared and straightened in order to float logs out from forest (Hynes, 1960). The principal purpose of much of the earliest channelisation was to increase arable land and this has been applied extensively. Flood control is essential to protect buildings from damage in both urban and rural areas. Navigation is limited to larger watercourses, but it has often been necessary to modify the channels to provide sufficient depth of water to enable movement of boats (Brookes, *op. cit.*).

Channelisation involves the reshaping of waterways and can include shortening, straightening, widening, realigning, deepening, removing obstructions to flow and increasing the gradient (Woods, *et al.*, 1981). Such modifications greatly affect natural stream morphology and hydrology, while the effects on river fauna and flora are usually detrimental (Swales & O'Hara 1980). The majority of studies on the effects of stream channelling have been carried out in North America. These studies have largely dealt with game fish populations. Studies on the effects on rivers in Europe have been sparse, but results have substantiated those obtained in North America (Crisp & Gledhill, 1970; Swales & O'Hara, *op. cit.*; Brooker, *op. cit.*).

1.3.2 ENVIRONMENTAL EFFECTS OF RIVER CHANNELISATION

According to McCarthy (1985) stream channelisation activities often result in the elimination of meanders and the straightening of stream beds. The immediate effect is to produce a channel devoid of typical pool-riffle sequences and without vegetation and in-stream cover which may be of considerable importance to many organisms (Jenkins *et al.*, 1984). In a natural stream system, channel width and depth are adjusted to flow regime, probably to bankfull discharge and its recurrence interval. Any destruction of this balance may lead to the erosion of bed and bank material, with increased suspended material in the water column and subsequent sedimentation (Brooker 1985). In particular, the removal of bankside vegetation and decreased soil stability are likely to increase sediment loads to rivers. Changes in water depth and the removal of bankside vegetation, either for machinery access or to reduce frictional effects, may lead to temperature changes in the stream (Brooker, *op. cit.*). Since most streams receive their primary source of energy in the form of allochthonous organic matter (Cummins, 1974, 1979), often as tree leaves, losses of bankside vegetation may also greatly reduce energy flow in the aquatic system. Additionally, the loss of trees, scrub and vegetation and the general disturbance during channelisation is likely to have a substantial effect on birds and mammals and dominant vegetation.

Gebhards (1973) stated that changes in the physical rather than the chemical form of the habitat are mainly responsible for changes in the fish populations. pH values, dissolved minerals and gases may undergo little change. Changes of sediment load, temperature and physical habitat normally caused by channelisation may also be important (Duvel *et al.*, 1976). The majority of studies (e.g. Moyle, 1976; Swales, 1982a) indicate that reductions of habitat diversity are responsible for changes in fish populations.

Increased turbidity caused by high levels of suspended solids affects light penetration and reduces visibility. Decrease in light availability reduces photosynthesis and the production of attached algae and macrophytes, thereby negatively affecting the food supply of consumer species. Turbidity and high levels of suspended solids greatly interfere with fish feeding, movement, migration, spawning and species diversity, and also reduce respiration efficiency, causing physiological damage to fish and, at a very high concentration, even death (McCarthy, 1985). Silt deposits may have more devastating effects on fish. Food, in the form of bottom-dwelling organisms, is killed and potential spawning sites eliminated when parts of the stream bed are blanketed by silt (Apman & Otis, 1965).

The increase in the rate of flow caused by channelisation has a major effect upon stream life. Spates affect the distribution and abundance of invertebrates by washing animals out of their original habitat and depositing them lower down the channel. Streams which experience spates more often have less abundant and varied fauna than others (Hynes, 1970). Low flows caused by channelisation can also have severe impacts on invertebrates and fish communities, affecting spawning, survival, distribution and migration of cold water and warm water species through low concentrations of dissolved oxygen and high concentrations of carbon dioxide (McCarthy, *op. cit.*).

Duvel *et al.* (*op. cit.*) found that the major reason for sparse adult trout populations in channelised reaches appeared to be the lack of cover normally provided by undercut banks, overhanging shoreline vegetation, deep pools and other obstructions such as logs and boulders. As all trout, in particular brown trout (*Salmo trutta*), exhibit some degree of negative phototropism, they tend to stay under features which provide a measure of

concealment. Channelisation often destroys these entities.

The removal of overhanging vegetation as a consequence of channelisation has been cited as a significant factor affecting the distribution of fish in several studies (e.g. Swales, 1982b). One reason for this is the effect of increased light and water temperature caused by vegetation removal. Temperature plays a significant role in the occurrence and distribution of fish species, particularly cold-water fish such as trout. Hansen & Muncy (1971) found that the mean daily water temperature was 0.3 °C higher in a channelised section than in an unchannelized section of the Little Sioux River in Iowa. Duvel *et al.* (*op. cit.*) showed how channelisation of the Fishing Creek in Pennsylvania raised the water temperature by 1.7 °C above that of a well-shaded natural site upstream, in which the water temperature was 18.9 °C. Natural shade is one of most significance on clear sunny days during mid-summer and it is at this time that such a temperature increase is likely to occur. Duvel *et al.* (*op. cit.*) pointed out that water flowing through a number of channelised areas can become progressively warmer as it approaches the mouth of the stream. The temperatures in the lower reaches of the stream may then exceed the temperature that trout can tolerate, resulting in the loss of productive sport fishing waters. Such a situation may account for the loss of legal sized trout observed on the streams studied by the above authors.

1.3.3 BIOLOGICAL EFFECTS OF STREAM CHANNELISATION

Stream channelisation is an extreme physical disturbance that disturbs not only the physical aspects of the channel but changes the entire riverine ecosystem. Channelisation can alter the original dimensions and shape of a channel, the slope and the channel pattern, changing the heterogeneous system into a homogeneous one. The removal of in-stream cover, riffle–pool pattern, riparian and bank vegetation and substrate modification have adverse effects on aquatic flora, invertebrate and fish communities (Cummins & Lauff, 1969; Hynes, 1970).

1.3.3.1 Impacts on macroinvertebrates

The majority of the impacts of channelisation on macroinvertebrates come from channel excavation. Animals such as mussels may be physically removed during the process of excavation (Clark, 1944), or macroinvertebrates may fail to establish thereafter because of a changed substrate (Hansen & Muncy, 1971).

Silt deposition as a consequence of channel excavation can kill many benthic invertebrates (Apman & Otis, 1965). Silt screens out light and tends to hold extraneous substances, such as industrial wastes, on the stream bed. The increased number of drift organisms in a channelised section of the Little Sioux River, Iowa, was attributed to the lack of suitable attachment areas (Hansen & Muncy, *op. cit.*). Etnier (1972) found that channelisation of the Middle Creek in Sevier County, Tennessee, resulted in decreased riffle habitat and gravel and boulder substrate. This was reflected in significant decreases in macroinvertebrate density, species richness and composition, particularly a reduction in the abundance of Ephemeroptera, Trichoptera and Plecoptera in sections which had been channelised for less than a year.

Standing crop, productivity, species diversity and numbers of macroinvertebrates were lower in channelised sections of the Luxapalila River, Mississippi even 52 years later (Arner *et al.*, 1976). This was attributed to differences in the substrate, pebbles being common in natural reaches and fine sand being typical of channelised sections.

Channel modifications undertaken in 1972 following Hurricane Agnes on six cold water streams in Pennsylvania (Fishing, Money, Beaver, Clover, and Mill Creeks and Freeman Run) appeared to have had no long-term effect on the number of taxa, number of organisms, density, standing crop and diversity of benthic communities (Duvel *et al.*, 1976). The lack of difference between natural and channelised sites was explained by the availability of suitable substrate and habitat for benthic fauna in both natural and channelised stream reaches. The substrate of the channelised sections of the streams included in the study was composed primarily of cobble. The substrate of almost all the natural sections was predominantly cobble with varying amounts of gravel, silt, bedrock, and/or boulders. Thus, while the benthic habitat varied from channelised to natural sites,

there was sufficient diversity of habitat in both instances to provide abundant living space for benthic populations to increase.

Dredging on a regularly maintained mill-stream in southern England (River Frome) had only a limited impact on the population densities of benthos, with rapid recovery in approximately two years (Crisp & Gledhill, 1970). The study area was a muddy-bottom reach containing Chironomidae, Oligochaeta and Mollusca which together formed 80% of the benthos in samples. A study by the Anglian Water Authority (UK) in 1982 (Brookes, 1988) showed a marked decline in invertebrate diversity following dredging in the Great Ouse however, which was attributed to silt covering the gravel bed. Only where the gravel bed subsequently became re-exposed did the invertebrate community recover, 1.5 years after dredging. Other studies have cited more rapid recovery rates. Harper *et al.* (1994) found that the taxonomic richness and density of macroinvertebrates in an artificial riffle reached the levels of adjacent natural riffles within two months. I suspect that the differences between these time scales are due to different types of substrate. Rapid recovery of a benthic community was also achieved on a chalk stream in Yorkshire (River Hull) following excavation of the gravel bed to a depth of 30 – 40 cm, and this was attributed to the behaviour of the animals which produced a rapid redistribution of the fauna over the available habitats. It was suggested that much of the substrate and its fauna escaped the bucket and many animals probably fell out during the lifting operation (Pearson & Jones, 1975b). The timing of dredging may influence the degree of impact, a spring to summer dredge having the least effect, since breeding of most species occurs shortly after redistribution. The results of a study by Smith, *et al.* (1990) on the river Welland indicated that the effects of canalization on benthic macroinvertebrates were to reduced family richness by about 50% and biomass by about 80%.

In the River Moy, Ireland, samples of macroinvertebrates showed a 90% reduction in density following dredging activities and the rate of recovery was slow (McCarthy, 1981). In the study of a flood alleviation scheme on the River Usk in Wales a variety of habitats including pools, riffles, tree roots and marginal vegetation was sampled at 10 sites before engineering works began (Brooker, 1985). A total of 80 taxa was recorded,

compared to only 50 taxa collected at the same sites a year after channelisation.

The annual cutting of weeds has been shown to have a dramatic impact on the invertebrate fauna. The immediate effects of weed cutting on the River Hull in northern England on the macroinvertebrate fauna were the removal of large numbers of animals in the weed, and increased drifting of some plant-dwelling animals (Brookes, 1988). Weed cutting on the Gjern stream in Jutland (Denmark) increased the total drift density by 173 times, to a maximum of 24,722 invertebrates per 100 m³ (Kern-Hansen, 1978). The dominant plants were *Ranunculus peltatus* and *Sparganium simplex*. In particular, species of invertebrates such as Hydroptilidae, Empididae and adults of Dytiscidae and *Elmis* spp. were found only in drift samples from reaches affected by cutting. Even several days after cutting the drift density of many species (e.g. Oligochaeta, *Gammarus pulex*, *Baetis*, *Caenis*, *Heptagenia*, and *Sialis* and larvae of Haliplidae and Dytiscidae) was still significantly higher than before cutting. Macrophyte cutting causes loss of habitat and unstable condition in the substrate and changes in drift density were attributed to this.

It has also been demonstrated that the macroinvertebrate fauna can rapidly recover after weed cutting, and that the community composition changes very little. The timing of such disturbances can, however, be very important for individual species. A June weed cut on the River Hull in Yorkshire would have affected the hatch of insects because some species such as Chironomidae and *Caenis horaria* attach their eggs to plants. By contrast a cut in July has less effect as numbers in the water are already declining (Pearson & Jones, 1978).

1.3.3.2 Impacts on fish and fisheries

The principal habitat requirements of fish are temperature, both directly and indirectly through the influence on oxygen consumption, rate of flow and discharge fluctuations, and the availability of suitable shelter and spawning area. Each of these factors has been shown to have been altered by channelisation (Swales, 1982a).

Channelisation greatly reduces the standing crop and diversity of fish populations of streams in several regions of the United States of America. A study of 23 channelised and

36 natural streams in North Carolina revealed that channelisation reduced the number of game fishes (over 15 cm in length) by 90% and reduced the weight by 80% (Bayless & Smith, 1967). In Rush Creek, in Northeast California, Moyle (1976) found that trout biomass was over seven times greater in unchannelized sections than it was in channelised sections. The loss of pools and cover such as overhanging bushes and large boulders was probably the main cause of a reduction in fish carrying capacity in the channelised sections of Rush Creek. Only small riffle-dwelling fish (speckled dace, pit sculpin) that were able to use the scant cover provided by small rocks and turbulent water maintained large populations in the channelised sections. The rate of recovery of fish populations from the effects of channelisation is extremely slow, some streams showing no significant recovery after 30 – 40 years (Bayless & Smith, *op. cit.*).

There are few published reports of the effects of channelisation on fish in the UK. Swales (1980) investigated the long and short term effects of river channel works on the ecology of fish populations in small lowland rivers. He found that channel works carried out since the turn of the century in the River Perry, a lowland tributary of the River Severn in Shropshire, have had adverse effects on fish habitat. The affected areas had low habitat diversity, little in-stream cover and few natural river characteristics. Consequently the fish community of the affected areas was reduced in abundance, diversity and growth, compared to nearby natural areas. In the River Soar, a lowland river in Leicestershire, reductions in total density (70%) and biomass (76%) (principally dace, chub and roach) were found four months after land drainage works (Swales, 1982a). Changes were attributed to the modification of habitat features, particularly the elimination of in-stream cover. In one other study in England on the effects of a flood alleviation scheme on the River Roding, Weeks (1982) suggested that engineering works had no detrimental effect on the coarse fishery, but few data were available.

In a long-term study of the impact of a land drainage scheme on the River Camowen in Northern Ireland, Kennedy *et al.* (1983) found that salmonid densities were reduced by the effects of dredging but subsequently recovered down stream from upland areas which were unaffected by the works. There were also changes in population structure: two sites

were deepened and finally contained larger numbers of older fish, whilst sites which became shallower generally supported higher densities of fry at the end of the survey period. In the River Boyne, McCarthy (1981) found that the ratio of the number of salmonids to other fish species before a land drainage scheme was 14:1, changing to 1: 5 after drainage works. This was primarily attributed to the production of silt and subsequent sedimentation.

1.3.4 IN-STREAM COMPONENT OF FISH HABITAT

The four fundamental components of salmonid habitat are acceptable water quality, food producing areas, spawning-egg incubation areas and cover. The extent to which each of these components is present in a given stream is dependent upon the stream's physical, chemical and hydraulic characteristics (Wesche, 1985). The emphasis is placed on physical and hydraulic aspects of the fish habitat which are the most important features subjected to changes during channel works but able to improve after rehabilitation.

1.3.4.1 Food producing areas

Riffles and pools are distinct habitats of most streams and rivers. Of the two, riffles are the primary fish food-producing areas. Substrate, velocity and depth combine in riffle sections to provide optimal conditions for the majority of invertebrate species.

Trout feed on macroinvertebrates which are themselves dependent on a suitable habitat. Morphological variability of channels is essential to trout, to provide areas of shelter from high velocities whilst the trout waits for prey. Higher velocity areas may also be important for carrying food from upstream (drifting invertebrates). Overhanging vegetation can be important as a source of food, cover for fish and shade from excessive temperatures (Gibson & Power, 1975).

Coarse fish consume either plants, invertebrates, fish, detritus, or combinations of these. Roach and carp eat substantial amounts of aquatic and terrestrial plant material. Carp also eat seeds which have fallen into the water. Many cyprinids whose main diet consists of invertebrates, also eat vegetation. The majority of coarse fish eat macroinvertebrates

with the exception of pike, which from a young age becomes purely fish-eating. Invertebrate feeders tend to eat larger prey as they grow (Brookes, 1988).

1.3.4.2 Spawning and egg incubation areas

Salmonids (e.g. Atlantic salmon (*Salmo salar*), brown trout (*Salmo trutta*), sea trout (*Salmo trutta trutta*)) require definite nest-sites in gravel substrates. After fertilisation, the female displaces stones by vigorous movements of the tail in order to bury newly laid eggs (Jones & King, 1950). These locations are known as "redds". Salmonids usually select places at the downstream end of pools where there is a downward movement of water into the gravel, or at the tail-end of riffles where there is upward flow of water. At such locations the buried eggs are constantly washed and supplied with oxygen. In Great Britain most salmonids and some cyprinid species such as dace (*Leuciscus leuciscus* L.) and chub (*Leuciscus cephalus* L.), are gravel spawners, depositing their eggs only where the substrate is suitable and where water depth, velocity and temperature are favourable. Gravel spawning cyprinids usually deposit their eggs on the gravel surface (Swales & O'Hara, 1980).

Spawning habitat has been defined by numerous investigators (e.g. Reiser & Wesche, 1977; Smith, 1973) who have measured the hydraulic and physical parameters existing in the stream sections utilised by actively spawning salmonid fish. Generally, acceptable spawning areas exhibit water velocities between 0.15–0.9 m s⁻¹, water depth of 0.15 m or greater, and substrate size between 0.6–7.6 cm. To a large degree fish size will determine if an area is acceptable for spawning, as larger fish dislodge larger substrate and endure swifter currents than smaller fish (Wesche, 1985).

Certain chemical, hydraulic and physical parameters must be in place for an incubating egg to develop successfully. The most important chemical factor is dissolved oxygen (at least 5.0 mg l⁻¹). The development of salmonid eggs is directly related to dissolved oxygen. The eggs require more and more dissolved oxygen as they develop and have a maximum requirement just prior to hatching (Hayes *et al.* 1951).

As unpolluted running water is generally sufficiently oxygenated it is not a limiting factor for fish. Oxygen levels are related to water velocity, as well as to plant growth and decay. However, for fish species the rate of the metabolism rises with temperature. Since oxygen concentration falls with increasing temperature of the water, and because the oxygen consumption of fish is higher in such conditions, excessive water temperature is harmful to the fish by reducing its activity. Brook trout are intolerant of temperatures above 25 °C. The tolerance of coarse fish species to warm water varies: perch have an upper limit of tolerance of about 30 °C, European cyprinids perish at 29 – 31 °C and pike die at about 29 °C. For trout to breed the temperature must fall below 14 °C at some time of the year (Brookes, 1988). Temperature and oxygen are affected by shade provided by bankside vegetation.

A good incubation environment is affected by hydraulic parameters such as the percolation rate of water through the spawning gravels, a pool–riffle sequence and ground water seepage. The percolation rate influences the length of the incubation period and the relative size of new fry (Shumway *et al.*, 1964), because it brings the necessary oxygen to the incubating eggs and removes the metabolic waste materials. This of course, is dependent on the concentration of dissolved oxygen.

Numerous investigators (e.g. Greeley 1932; Webster & Eiriksdotter 1976) have shown that brown trout (*Salmo trutta*) and particularly brook trout (*Salvelinus fontinalis*) select spawning sites in areas with ground water seepage. There is a direct relationship between the amount of ground water, size of trout population and number of redds. Ground water provides a constant flow over the eggs ensuring sufficient dissolved oxygen for development. Also as ground water temperatures are often warmer than surface waters in the winter, the eggs are protected from freezing conditions and time to hatching is reduced.

1.3.4.3 Cover

Cover can be defined as those stream areas providing the fish protection from the effects of high current velocities and predation. Cover for fish in streams can be provided

by overhanging vegetation, undercut banks, submerged vegetation, submerged objects (stumps, logs, roots, rocks), floating debris and water turbulence. The extent to which each of these forms is used is dependent upon species preference and, of course, upon its availability in the stream.

Swales and O'Hara (1980) revealed that shelter, both in and adjacent to streams, serves many important functions in that it provides;

- (1) concealment for both predator and prey fish species.
- (2) increased habitat diversity
- (3) cover in the form of aquatic weeds and detritus which is used by some fish species as a spawning substrate.
- (4) shading from direct illumination, lowering the water temperature.
- (5) cover may be important in determining fish behaviour, particularly with regard to territoriality.
- (6) stream invertebrates utilise vegetation cover, both as a food source and a habitat.

Generally, fish establish a territory around the selected cover type. This tends to spread the fish population throughout the stream system leading to a more efficient utilisation of the food supply. It is within this micro-habitat that the fish spend the majority of their time, feeding and resting. Hooper (1973) stated that the abundance of suitable cover determines the number of territories and thus the fish population. The greater the variability then the greater the diversity of type and size of fish. Structurally diverse natural streams also have a buffering capacity: meanders moderate flood effects, pools serve as refuges for fish during dry periods and the shading effect of trees protects against excessive temperatures (Gorman & Karr, 1978). By contrast channelised streams may have little or no buffering capacity.

The relationship between salmonid populations and protective cover has been investigated by several authors (e.g. Shetter *et al.*, 1946). These studies have demonstrated that the addition of artificial cover can cause an increase in number and size of the trout in a given section of the stream. Boussu (1954) found that the addition of bush

cover (consisting of plank frames with interwoven willow branches) to four sections of a stream in Gallatin County, Montana, caused a 258% increase in the weight of fish, whilst removal of natural bush from two sections led to a 40.5% decrease in the fish population. The natural vegetation consisted of a heavy cover of sedge (*Carex* sp.) and willow (*Salix* sp.). At a further two sections an undercut bank was removed, causing a 33% decrease in both the numbers and size of fish. Elser (1968) also found 78% more trout in an unaltered stream section than in an altered section which had 80 % less cover.

1.3.5 RIVER RESTORATION AND IN-STREAM HABITAT IMPROVEMENT DEVICES

1.3.5.1 Introduction

River restoration means many different things to different people and interest groups. For some it is restoring or improving water quality whilst for others it may be reversing the impacts of engineering or other activities on the fishery, landscape, recreation or ecological interests. The terms rehabilitation, enhancement and improvement refer to measures taken to improve various aspects of a river, but not to change it back to its pre-damaged state. Holmes (1993) believes that the term 'river restoration' should be reserved for the more complete restoration of the system, not single-function improvements. Wesche (1985) considered that the enhancement of in-stream aquatic habitat for fish life involves a wide variety of activities such as stream flow regulation, watershed improvement and regulation of land use activities, overall channel design and alignment, stream bank stabilisation and improvement, obstruction removal, construction of spawning facilities and installation of in-channel structures. Obviously the employment of any one or combination of these general approaches depends upon the particular problem at hand and the philosophy of the management agency involved.

The principal objective of most in-stream habitat improvement techniques is to modify and diversify the habitat in such a way that it becomes more favourable for fish survival. These stream improvement structures are designed to recreate, to some extent,

environmental features present before habitat degradation. Improvement structures have also been used to increase fish abundance in areas of river which, although not directly affected by man's activities, do not provide optimal environmental conditions (Swales & O'Hara, 1980).

Improvement structures in a river may produce many effects, but their overall function is to increase the diversity of the river habitat. This may be achieved in a variety of ways, by providing shelter, altering flow, channel morphology or substrate composition. The amount of available cover may be increased by means of artificial shelters placed in the river, or indirectly, using structures which increase water depth by impoundment, or by increasing current velocity, causing the scouring out of bed material. The substrate size composition of the river bed can be altered, increasing the area suitable for fish spawning and rearing, or improving existing spawning areas. Habitat improvement devices may also serve to improve water quality, for example by increasing the dissolved oxygen content of the water, or lowering the concentration of suspended solids (Swales & O'Hara, *op. cit.*).

From this variety of activities, my project is concerned with the installation of in-stream structures, particularly current deflectors and artificial riffles. Hence, this review deals with some of the more commonly used rehabilitation devices and studies of the effects of these improvement techniques. The most commonly used in-stream treatments are current deflectors, weirs, artificial riffles and substrate placement.

1.3.5.2 Current deflectors

Current deflectors have historically been one of the most commonly used in-channel treatments to improve fish habitat. They work by scouring the stream bed and creating pool-riffle habitat. In general they are easy to construct, inexpensive and easily modified to suit on-site conditions. Deflectors are built from a variety of materials (logs, rocks, boulders, gabions, and concrete), are applicable to a wide range of stream sizes, can be adapted for use with other treatments and, when properly designed and implemented, are successful in providing habitat diversity. Deflectors have been built with a variety of

purposes in mind, including: deepening and narrowing channels, scouring pools, increasing water velocities, and protecting stream banks from erosion (Wesche, 1985).

In order to recreate pools and riffles natural fluvial processes are used, thereby increasing current velocity and removing deposited sediment. The river channel is constricted by these devices and if they are angled in a downstream direction, the current is diverted away from the deflector and its velocity increased due to the reduction in channel width. In this way the current tends to scour out the river bed in an area immediately downstream of the tip of the deflector. The immediate effect of this is to carry away deposited sediment, but eventually the river bed itself is eroded which produces a pool area with a riffle area downstream (Swales & O'Hara, *op. cit.*).

Wesche (*op. cit.*) stated that, prior to deflector construction the main aspects to consider are the shape of the structure, its height, the angle of the deflector, the length it will extend into the channel and the materials to be used. Regarding shape, several forms have been used over the years, the most common being the peninsular wing (jetty) (Figure 1. 1) and the triangular wing. White & Brynildson (1967) recommend the use of the latter because it reduces the tendency for erosion of the bank and bed behind the structure during high flow (the deflector included in my study is the triangular boulder form (Figure 1. 2) which is installed in the River Smite). Structure height is generally dictated by the elevation of water surface at low flow. To avoid excessive damage to the structure itself and the opposite bank during high flow, the structure should not extend more than 0.15 – 0.3 m above the low flow elevation (White & Brynildson *op. cit.*).

Typically, deflectors are angled downstream at approximately 45° to the current direction, while the back brace is set at approximately 90° to the deflector (Swales 1982a). The distance that the deflector extends into the channel will vary depending upon the specific results desired. For example Swales (1982a) achieved the desired result on a small lowland river in England by extending his deflectors one-third to one-half the distance across the channel. For general planning purposes deflectors across 50% of river width would probably be appropriate. On-site knowledge of relative bank stability, substrate size and composition, and design flow and associated hydraulic characteristics are necessary to

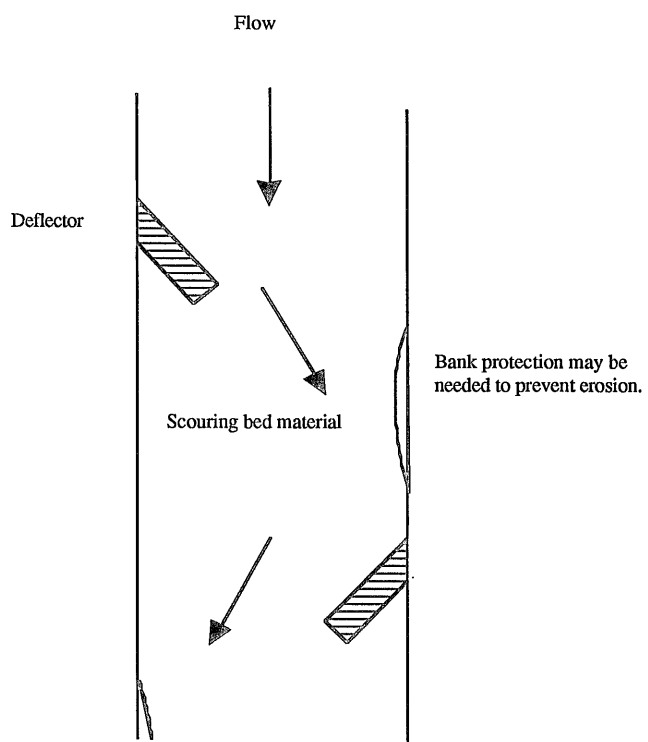


Figure 1.1 Showing wing deflectors and the effect of current flow on channel morphology.

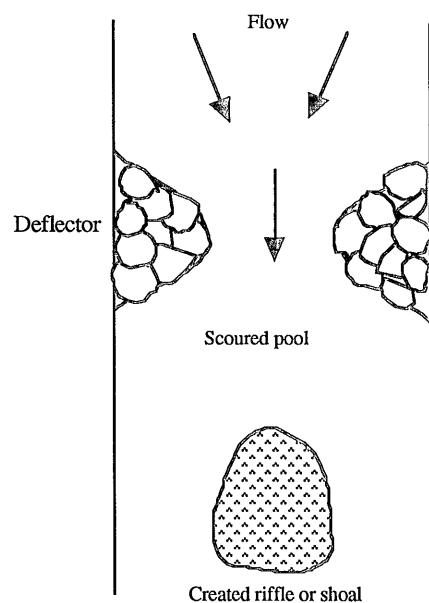


Figure 1.2 Showing triangular boulder deflectors and the effect of current flow on creating riffle-pool.

determine exact lengths (Wesche, 1985). As current deflectors partially recreate the habitat characteristics associated with pools and riffles, wherever possible, they should be installed five to seven stream widths apart, imitating the natural river pattern (Swales & O'Hara, 1980).

1.3.5.3 Low dams or weirs

River impoundment using low dams is a commonly used improvement method in fisheries management projects. The main effects on the river habitat involve changes in channel morphology, hydrology and water quality. Above the dam, river depth is increased by the impoundment, providing more shelter in depth for fish populations.

Water quality is also improved due to the deposition of suspended solids caused by a reduction in water velocity. Dam overspill aerates the water, further improving water quality. Below the dam water depth is reduced, and its velocity increased. This is accompanied by a change from laminar to turbulent water flow, immediately below the dam. The increase in velocity and turbulent water scours out the stream bed creating a pool area followed by a riffle area produced by the change in flow pattern and the deposition of eroded material. (Swales & O'Hara, *op. cit.*).

These structures extend across the entire channel, although some have a notch to concentrate flows locally. Weirs can be constructed from logs, rocks, gabions, sheet piles or concrete, and must be keyed into the bed and banks to prevent side-scour. According to Gore (1985), low dams (approximately 0.3 m high) are successful on smaller (1–9 m wide), high-gradient (0.5–20% slope) headwater streams. Successive structures should be placed no closer than 5–7 channel widths apart, and the stream bed substrate should be stable.

1.3.5.4 Artificial riffles

Riffle and pool habitats are the major features of the middle reaches of natural streams and rivers. The importance of these habitats in the river has been discussed earlier in this chapter. The abundance and the diversity of macroinvertebrate communities and fish populations are directly related to the presence of these habitats. Riffles and pools have been eliminated as a major consequence of channelisation. Recreating riffle–pool habitat by introducing adequate substrate to the river bed or through other types of habitat improvement techniques will ameliorate the adverse effects of channelisation. The study of artificial riffles of Harpers Brook is a part of my project.

The diversity of flow conditions, substrate types and channel morphological characteristics provided by pool and riffle habitats means they are of great value to fisheries (Swales & O'Hara, 1980). Turbulence in riffle areas serves to oxygenate the water and lower water temperature, and water flowing through the gravel in a riffle provides conditions suitable for the development of fish eggs. Riffles are also the major

fish food producing areas. As substrate particle size tends to be larger in riffles than pools, the riffle habitat is physically more diverse than the pool habitat. In general it is known that the more diverse the substratum, the more diverse is the invertebrate fauna associated with it. This is so in pools and riffles, with the invertebrate populations of riffles being more diverse and greater in numbers of species and total biomass than the fauna of the pools (Hynes, 1970). Pools provide a favourable habitat for fish. The increased depth provides shelter and minimum energy is required by a fish to hold its position as the current velocity is low.

In general a pool–riffle sequence in streams is important in providing cover, resting, and food producing areas. The interchange area between a pool and riffle (run) provides an excellent spawning environment, with velocities great enough to carry away silt and debris that may clog the redd substrate (Wesche, 1985). The elimination of this pattern by river channel works has often been shown to result in the deterioration of a fishery (Etnier, 1972). These adverse effects may be ameliorated by the use of artificial riffles. Riffle reinstatement is considered to be an excellent method for the improvement of habitat diversity in rivers devoid of this due to past engineering works (Harper *et al.*, 1994). They report that, the construction of riffles is a central technique of in-channel restoration. Replacing riffles, even in a straight channel, will provide substrate and current speed heterogeneity immediately and depth heterogeneity due to subsequent erosion and deposition caused by the redirection of energy during subsequent bankfull discharges. Despite the central place of riffle construction within river restoration, there are relatively few examples of the consequences.

As a preliminary step in riffle–pool construction, it is necessary to assess the flow characteristics and channel morphology to determine if pools and riffles are appropriate in-stream habitat features (Brookes, 1988). Pools and riffles are not usually installed on ephemeral streams, in channels with a steep gradient where there is a high sediment transport, or where the banks are unstable. Generally an average of 5–7 channel width has been found to be sufficient to emulate natural conditions (Keller, 1978). In Britain the range for the natural pool–riffle sequence is 3 –10 channel widths. Meanders should be

incorporated with riffles located in straight reaches and pools at bends. Proper spacing will facilitate self-maintenance (Brookes, 1988).

In those cases where the riffles are to be dynamic and self-maintaining, they should be constructed from natural stream gravels with a size distribution typical of the existing bed material. Otherwise they can be constructed from gabions, cobbles, or boulders which will withstand high discharges (Edwards *et al.*, 1984).

1.3.5.5 Other in-stream treatments

According to Wesche (1985), the placement of individual boulders or boulder clusters is one of the simplest and most commonly applied in-stream treatments that can improve fish habitat on streams of any size. Such an activity may not be appropriate for wider ecological reasons on lowland rivers where boulders have not naturally occurred in the past. Generally, boulder placements are made with one or more of the following management objectives in mind: to provide additional rearing habitat; to provide fish cover, to improve pool-riffle ratio, to restore meander and pools in channelised reaches (by deflecting the current and eroding the opposite bank), to protect eroded banks by deflecting flow. Rock-filled gabion structures have been used in larger streams by Cooper & Wesche (1976) to create additional holding water and cover for trout.

The replacement of natural bed sediments following completion of a channelisation scheme is another type of in-stream treatment which may speed recovery. This is most successful where well-sorted gravel replaces unsorted sediments or bedrock. Placement of artificial materials such as crushed rock may also improve the habitat for fish and for macroinvertebrates (Spillett & Armstrong, 1984).

Devices which provide direct cover may either be fixed to the bed or banks of a channel or allowed to float with varying discharge rates (Brookes, *op. cit.*).

1.3.5.6 Previous studies of the effects of improvement devices

Research into the use of improvement devices began in North America in the 1930's, when several habitat improvement programs were initiated. Current deflectors and

low dams were the most widely used in-stream devices in these early studies, and were very successful in improving fish abundance (Swales & O'Hara, 1980).

One of the largest studies of current deflectors has been undertaken on Lawrence Creek in Wisconsin. Hunt (1976) showed that the mean annual biomass of trout, mean annual number of legal-sized trout and annual production, increased markedly in the first 3 years, reaching a maximum development after 5 years. Saunders & Smith (1962) reported that one year after the installation of deflectors and dams on Hayes Brook, the number of age 1 and older brook trout had doubled in the modified reach. Shetter *et al.* (1946) found that five years after 24 current deflectors were installed on a section of 600 m of Hunt Creek (Michigan), the number of good quality pools had increased from 9 to 29, mean pool depth had increased by 15 cm and additional spawning gravel had been exposed.

Many river improvement schemes have successfully used dams to improve fisheries. Shields *et al.* (1995) gives the hydraulic results of restoration utilising weirs in the 1 km long reach of Goodwin Creek (USA) and the consequences for fish fauna. Restoration increased the pool area to 72% of the water area and bed types became more heterogeneous. The fish species composition altered and the median lengths of five species increased. Gard (1961) used dams in an attempt to create a habitat suitable for brook trout in the headwaters of a California stream, and from a four year study concluded that brook trout, which were introduced, survived, grew rapidly and reproduced in the stream following deepening and widening in places by damming. No trout had previously survived before habitat modification of this stream.

Macroinvertebrate abundance, diversity, standing stock and drift were significantly higher in a channelised reach of the Olentangy River in Ohio which had been mitigated with pools and riffles, and approached values found in natural streams (Edwards *et al.*, 1984). The diversity and abundance of game fish were also higher in the mitigated reach. Certain non-game species were, however, relatively more abundant in the mitigated area when compared to the natural area. An earlier study had shown that, 24 years after channelisation, the number of species of fish compared with a natural reach was 22% less in a conventional channelised reach but only 5% less in a reach where artificial riffles had

been constructed (Edwards *et al.*, 1975). They also pointed out that there was an obvious shift from torrential fauna such as Hydropsychidae, Psephenidae and Heptageniidae in natural areas to slower-water forms such as Ephemerellidae, Oligochaeta (primarily Tubificidae) and Chironomidae in the channelised areas. Benthic diversity was also higher in natural areas than in the mitigated areas, and higher in the mitigated than in the channelised areas.

Recently, research has been carried out in Great Britain to evaluate the effects of improvement devices installed in lowland rivers which have been affected by extensive river channel works.

Taylor (1995) in the study of artificial reefs in the River Bure found greater invertebrate abundance and species richness on reefs than on the riverbed substratum (silt dominated) throughout his study.

A study by Harper *et al.* (1994) on two rivers in eastern England mitigated by artificial riffles revealed that :

- (a) the artificial riffles colonised rapidly with fauna similar to that of natural riffles;
- (b) there was no significance difference between natural and artificial riffles of the river Ivel, each holding around 20 taxa after twelve days of riffle-instatement;
- c) 50 days after riffle construction the artificial riffles stabilised at around 30 taxa, compared with a maximum recorded for natural riffles of 33;
- d) between $1-2 \times 10^3$ individual macroinvertebrates m^{-2} occurred at the natural riffle site through the study, and similar densities were achieved at the artificial riffle within the first month. They believed this implies that the artificial riffle may provide similar food quality for fish fairly rapidly after its creation.

For the River Usk in Wales, Spillett & Armstrong (1984) introduced crushed limestone and flint gravels over a bare clay bed. Surveys at 4, 10 and 20 weeks after reinstatement indicated a significant increase in invertebrate populations and improvement in biological quality. The limestone substrate supported a much higher invertebrate density than the clay bed, with higher abundance and diversity. In the Afon Gwyrfai (UK) recolonization of a reinstated gravel bed was a gradual process, taking about a year to

complete (Brooker, 1982). The stability of retained gravels is important: if the gravel is too fine and therefore unstable, species diversity and abundance will be less.

Jungwirth *et al.* (1995) found in a three year study that the addition of instream bed structures (groynes and bedfalls) to straightened Austrian streams provided riverbed variability that resulted in increased fish fauna diversity, density and biomass. Number of fish species increased from 10 to 19 and density and biomass tripled during the period of study.

White (1975) reported how fixed cover on a stream in Wisconsin increased the number of trout by over 100% in 3 years. Compared to four reaches of the Big Roche-A-Cri Creek left unmanaged, the youngest trout numbers were up to 11% higher, whilst older or larger trout were 200% more numerous in the spring and autumn in the managed channel. Hooper (1973) stated that abundance of suitable cover determines the number of territories, and thus the fish population. Elser (1968) found 78% more trout in an unaltered stream section than in an altered section having 80 % less cover.

While the literature detailing the results of boulder placements is not abundant, Wesche (1985) cited the following applications which were successful. Lere (1982) found that after eight years a majority of the boulders were still functioning properly by creating in-stream habitats and trout numbers were greatest in a river reach mitigated with random boulders. Knox (1982) found that random boulders placed in the Eagle River (Colorado) were successful in creating pool habitats in a channelised reach. In British Columbia, Haugen (1978) noted a twenty-fold increase in coho salmon numbers one year after rock clusters were installed on the Keough River. Also, Kanaly (1971) found that the trout population in a channelised section of Rock Creek mitigated with large boulders quickly recovered to levels comparable with unaltered reaches.

CHAPTER TWO

Materials and Methods

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MATERIALS AND METHODS

2.1 THE AREA OF STUDY

2.1.1 INTRODUCTION

The study was carried out in two lowland rivers in the east midlands of England; Harper's Brook in Northamptonshire and the River Smite in Nottinghamshire (Figure 2.1). Both rivers were channelised and have subsequently been rehabilitated by the National Rivers Authority (NRA). The techniques applied were current deflectors in the River Smite and artificial riffles in Harper's Brook. The improvement techniques have been used in certain small sections of each river, but the other parts of both rivers are either semi-natural (untouched and recovering after being channelised) or remain degraded (dredged by regular weed cutting operations).

2.1.2 HARPER'S BROOK

Harper's Brook, a tributary of the River Nene, is a small lowland alluvial river. In the study reach it forms the boundary between the Titchmarsh Nature Reserve to the south and arable farmland to the north (Figure 2.2). It is typical of many small rivers of the region in that it has been channelised extensively. The current channel is completely artificial where the stream has been re-routed across the Nene floodplain around gravel workings which have been subsequently flooded and are now part of the nature reserve. The channel was thus formed in the reworked sediments of the Nene floodplain. The result of this is a stream with a straight channel, uniform cross-section, very low gradient and sluggish flow velocities. The effects of this low gradient are enhanced by the effect of the Nene which causes backing up of the water in the stream (Smith & Youdan, 1994).

The diverted channel initially resembled an inland drain as it was straight and virtually devoid of tree cover. This led to prolific weed growth, especially in the lower reach, causing increased siltation and reduction in the already low channel velocity.

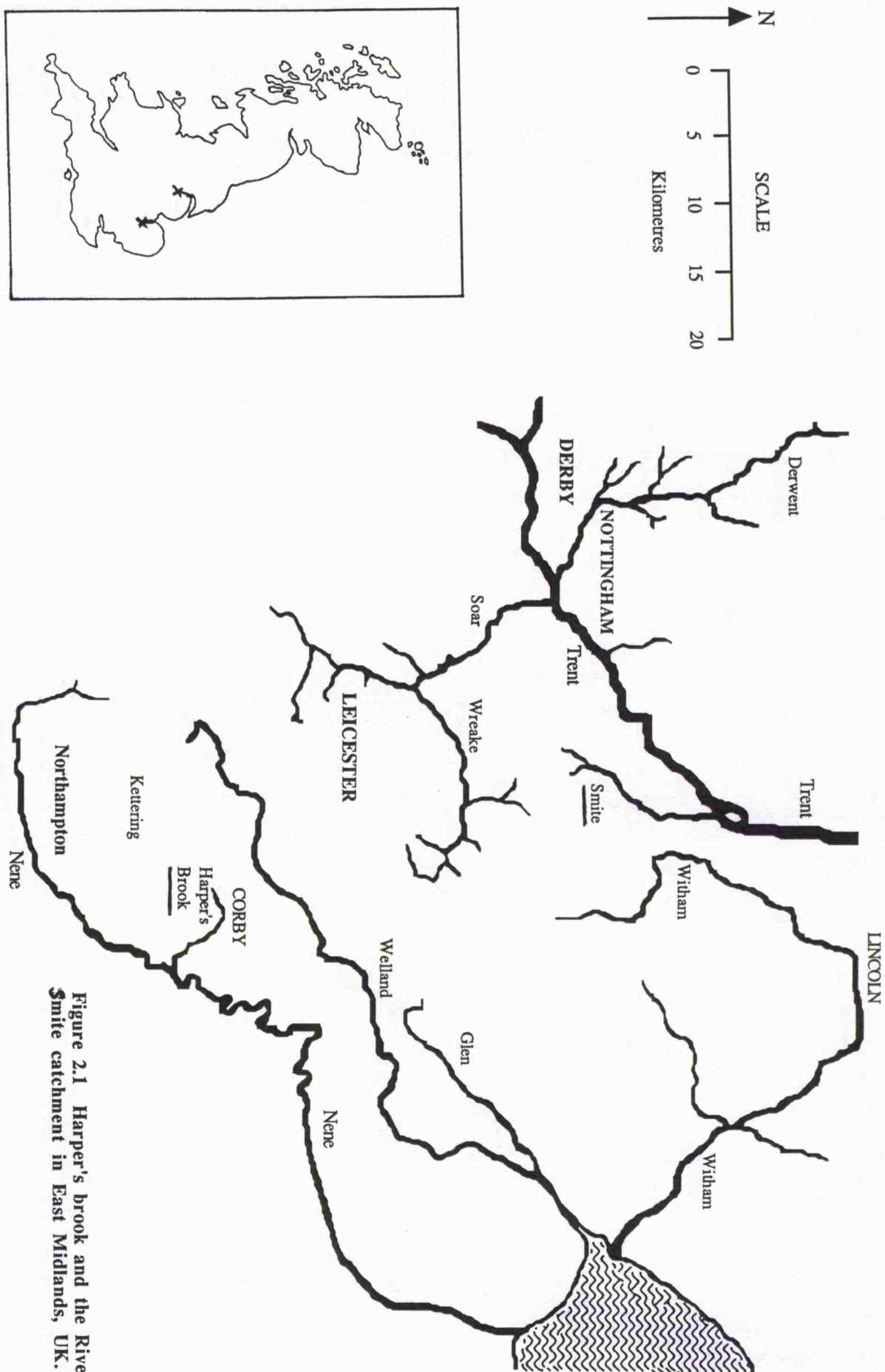


Figure 2.1 Harper's brook and the River Nene catchment in East Midlands, UK.

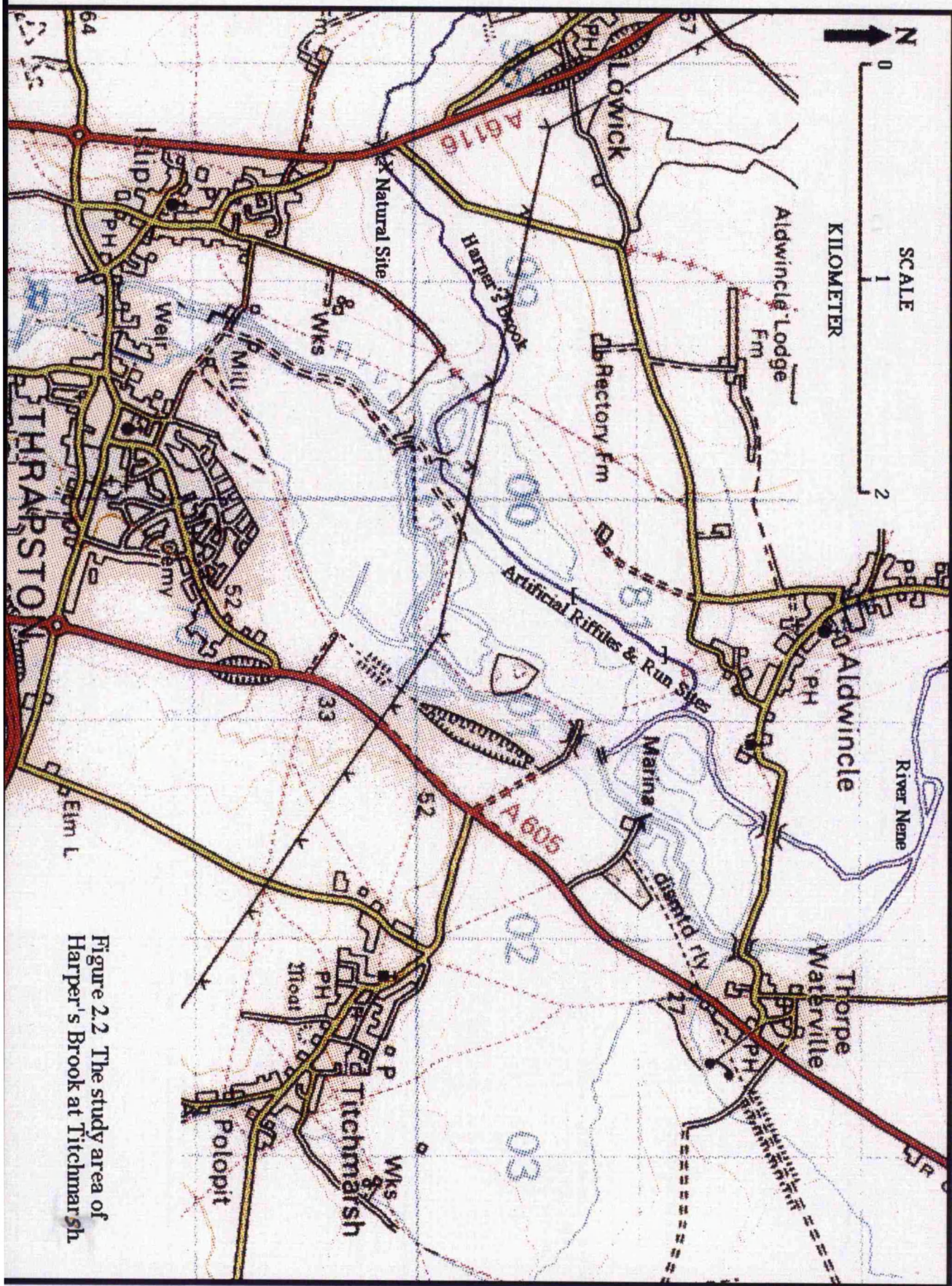


Figure 2.2 The study area of Harper's Brook at Titchmarsh.

Gravels comprise the main substrate of the brook and these are covered by sand and silt. Most of the larger substrate such as cobbles and boulders had been removed previously and there was little transport of coarse substrate from upstream. In common with many streams in the region, the brook sides are uniformly steep to below the summer water level. The bankside trees and marginal vegetation have been largely cleared, and there was no vegetation planted in the sections which were diverted.

A scheme drawn up between the National Rivers Authority, the Northamptonshire Wildlife Trust and English Nature to rehabilitate the physical structure of the river was implemented in 1991. Several basic objectives for future management of the Harper's Brook were established. These were to;

- (a) reduce the frequency of weed-cutting operations,
- (b) increase the heterogeneity of substratum / flow / depth,
- (c) provide a more varied channel margin,
- (d) provide a more diversely-vegetated riparian zone.

The rehabilitation program for Harper's Brook was carried out in two stages. First the introduction of artificial riffles was implemented in summer 1992. The artificial riffles constructed from local materials were introduced at sites where coarser sediments had accumulated naturally. The existing substrate was removed from the stream bed and the artificial riffles were constructed to about 7–8 m length and 0.5 m height with replacement of the finer material (Smith & Youdan, 1994). The artificial riffles in my study are made of cobble which differ from those of 'finer material' cited by Smith & Youdan (*op.cit.*).

Secondly, the physical aspects of the river banks have been changed in parts. This has been carried out by the modification of the bank profile through changing the bank slope conditions. Embayments were created and shallow berms cut at or above the normal water level. Furthermore, trees such as alders and willows, with a mix of oak, ash, hawthorn, elder, blackthorn and dog rose have been planted on the bankside (Smith & Youdan (*op.cit.*)).

At present the stream comprises the pool–riffle–run sequence. Artificial riffles on the study reach are sited between 25 m and 155 m apart, separated by pools and runs. The

pools rapidly increase and decrease in depth reaching up to c. 1 m. Runs are situated between two successive riffles and occupy a longer area. The study sites are located along c. 500 m of the lower half of the riffle re-instatement reach. The control site is located c. 1 km upstream of the rehabilitated reach which was the first suitable accessible semi-natural site (Figure 2.2 & 2.3).

2.1.3 RIVER SMITE

The River Smite is a lowland river in Nottinghamshire. It originates from highlands of the Wolds, flows northward, joins to the River Devon in its continuation and finally joins the River Trent (Figure 2.1 and 2.4). The study site was at Whatton, where the river had been over-widened in the past by the Internal Drainage Board, creating an average width of 5 m and average depth of 0.4 m. The downstream end of the section was wider and deeper than the middle and upstream reaches.

The objective of the current deflectors was to improve the diversity of habitat over approximately 400 m of the River Smite at Whatton. The aim of the improvements was to spread the resident fish population by creating diverse habitats over the entire stretch and later to allow the owner to introduce brown trout. In October 1991, it was decided to construct a set of stone deflectors in the river every 6–8 river widths along the water course. Six of these would be pairs of triangular shaped deflectors and one would be a double wing deflector (Easton, pers. com.). No further information was available from the Nottingham office of the NRA regarding the construction of the deflectors and the geomorphology of the river; it appears that little was recorded at the time of construction.

The areas within the study sites were; (a) the last deflector, near the railway bridge, which was chosen because it appeared that a riffle or shoal had been created downstream of it, (b) a uniform run in mid-reach which appeared to be unaffected by the deflectors, (c) a riffle upstream of the improved reach which represented a 'control' section presumed to have recovered from the original widening and deepening independently of the current deflectors (Figure 2.5).

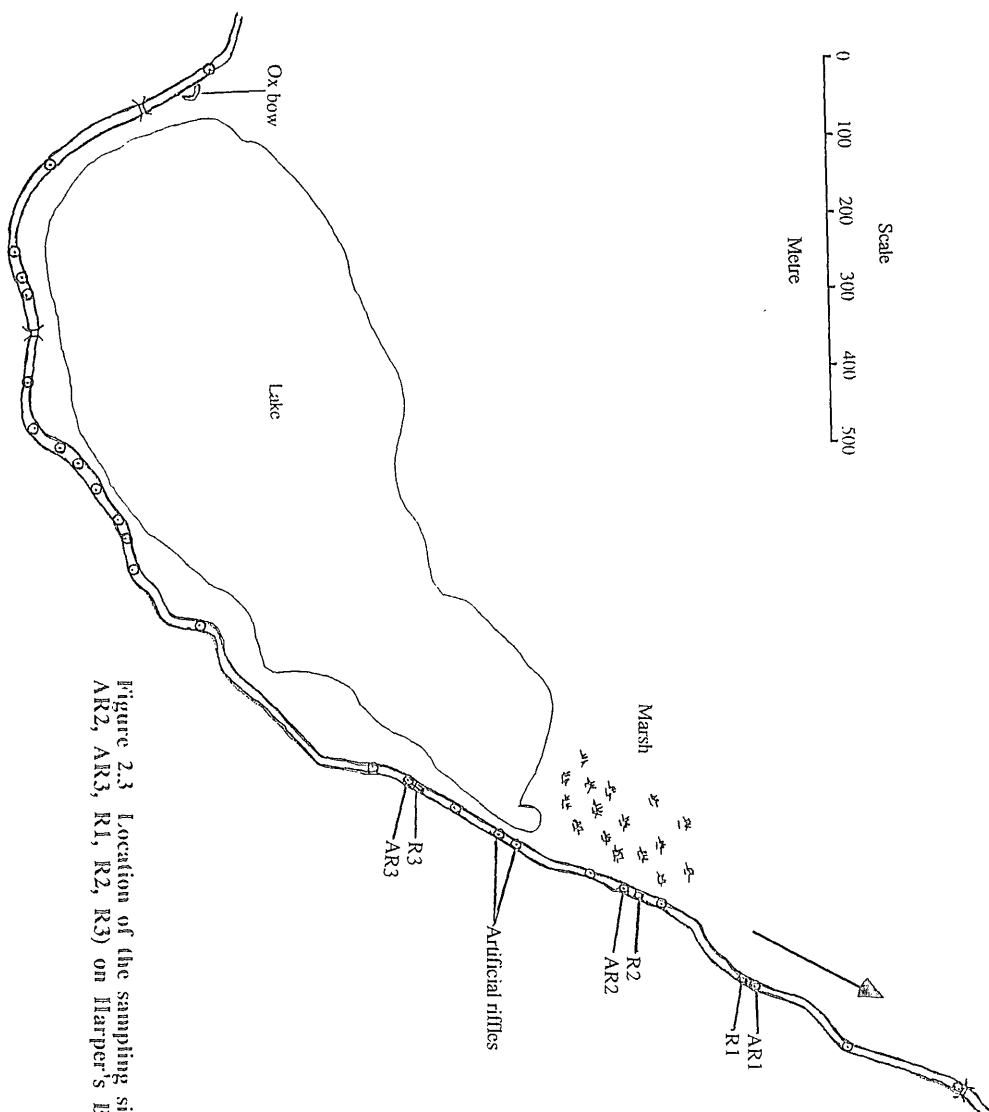
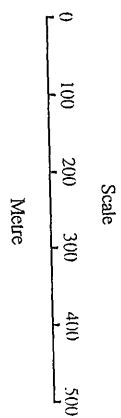
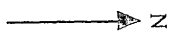


Figure 2.3 Location of the sampling sites (AR1, AR2, AR3, R1, R2, R3) on Harper's Brook.

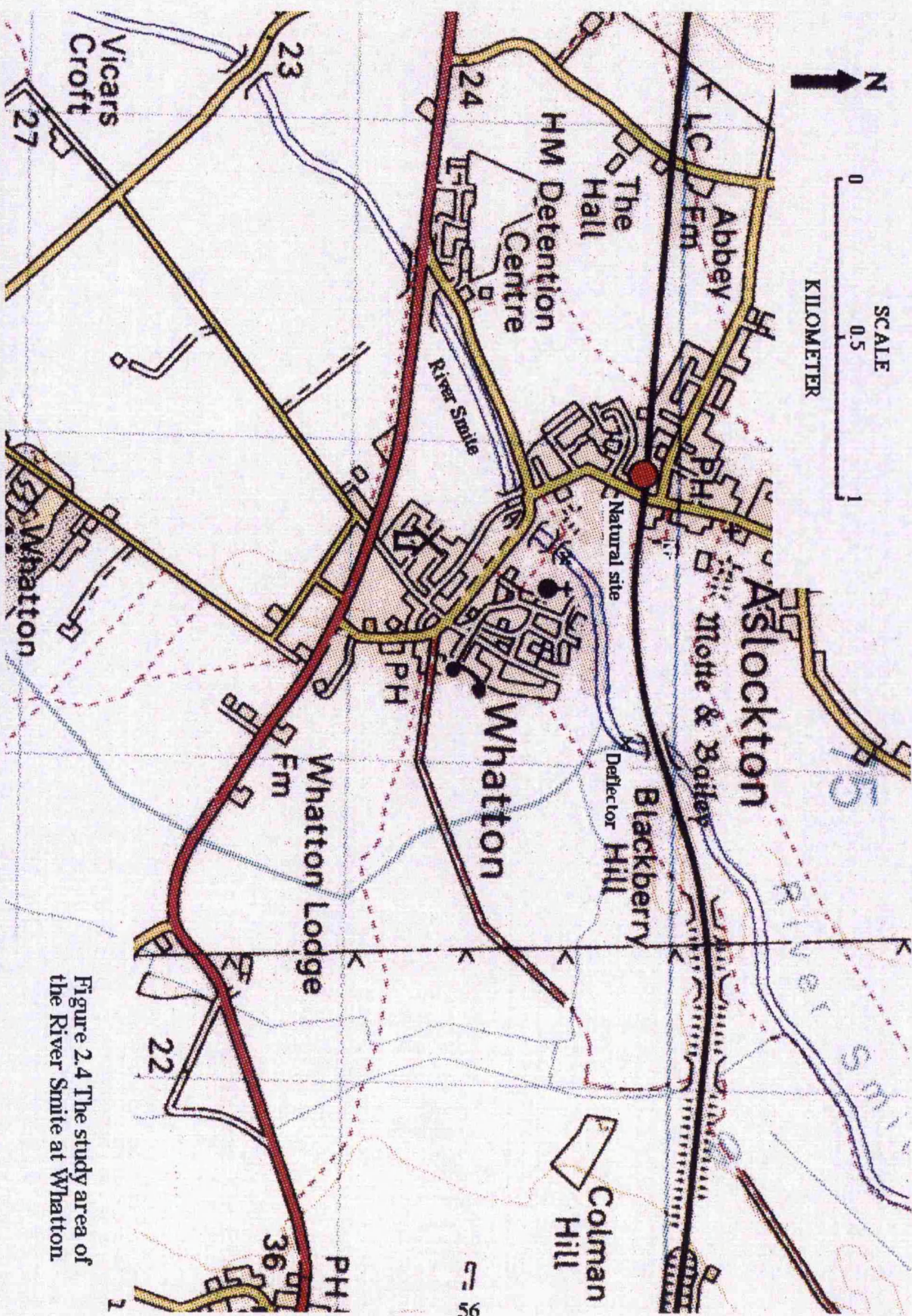


Figure 2.4 The study area of the River Smite at Whetton

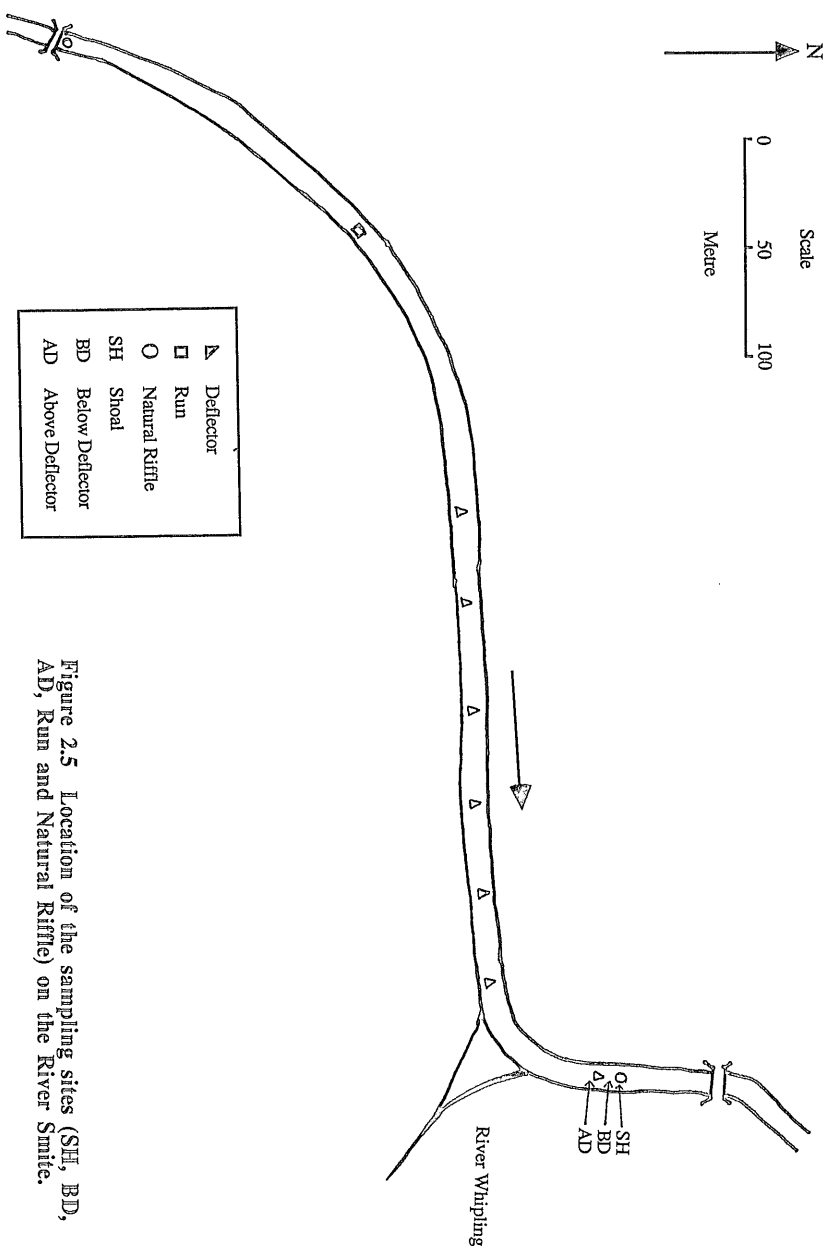


Figure 2.5 Location of the sampling sites (SH, BD, AD, Run and Natural Riffle) on the River Smite.

2.2 GEOMORPHOLOGY OF THE IMPROVEMENT DEVICES

2.2.1 ARTIFICIAL RIFFLES (HARPER'S BROOK)

In all the NRA placed 26 artificial riffles at this 2 km-long site. The artificial riffles are superficially similar but there are some geomorphological and hydrological differences between them. In order to show the extent of differences in depth and velocity of the riffles, measured during low-flow in October – November (raw data collected by a group of MSc. students for practical work) the depth and velocity frequency distributions were analysed.

2.2.1.1 Depth and velocity of the riffles

The depth frequency distribution is shown in figure 2.6. Fourteen percent of the artificial riffles are < 100 mm, 23% between 100 – 124 mm, 36% between 125 – 149 mm, 13% between 150 – 200 mm (including two classes) and 14% greater than 200 mm. These categories can be described as very shallow, shallow, moderate, deep and very deep riffles.

The results of velocity frequency distribution are shown in figure 2.7. The histogram shows five major velocity classes; 14% of the riffles with velocity of > 50 cms^{-1} can be identified as very slow riffles, 5% with the velocity of 50 – 59 cms^{-1} called slow, 36% with velocity of 60 – 69 cms^{-1} called moderate-flowing riffles, 27% with velocity of 70 – 79 cms^{-1} called fast-flowing riffles and 18% with velocity of > 80 cms^{-1} called very fast-flowing riffles.

2.2.1.2 Pools created by riffles

The effectiveness of the riffles in creating pools, is shown by the frequency distributions of the deepest point below riffles. The results indicated six classes: pools which were < 60 cm deep formed 4% of the total number(22), those with 60 – 79 cm depth formed 14%, those with 80– 99 cm depth comprised 23%, those with 100 – 119 cm

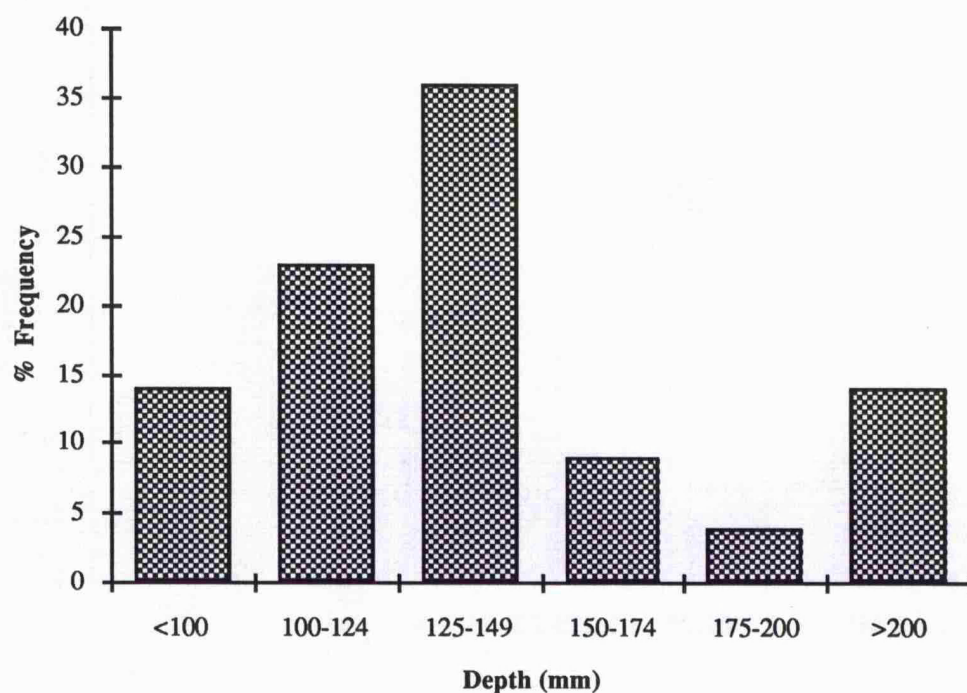


Figure 2.6 Percentage frequency distribution of depth of the 22 artificial riffles from Harper's Brook.

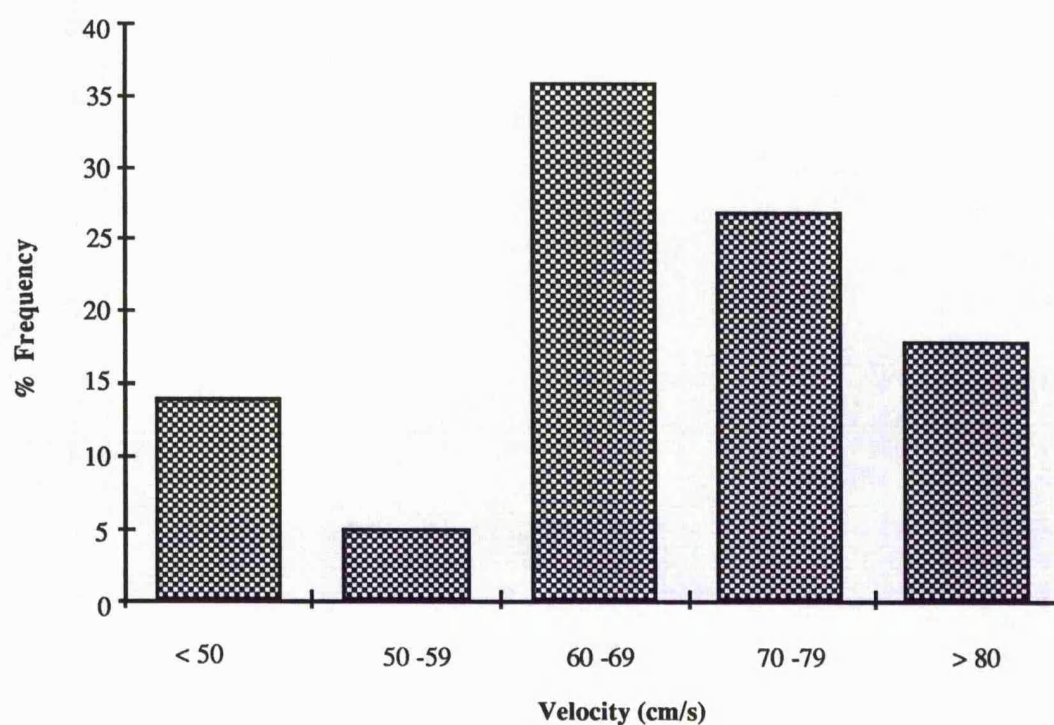


Figure 2.7 Percentage frequency distribution of velocity of the 22 riffles from Harper's Brook.

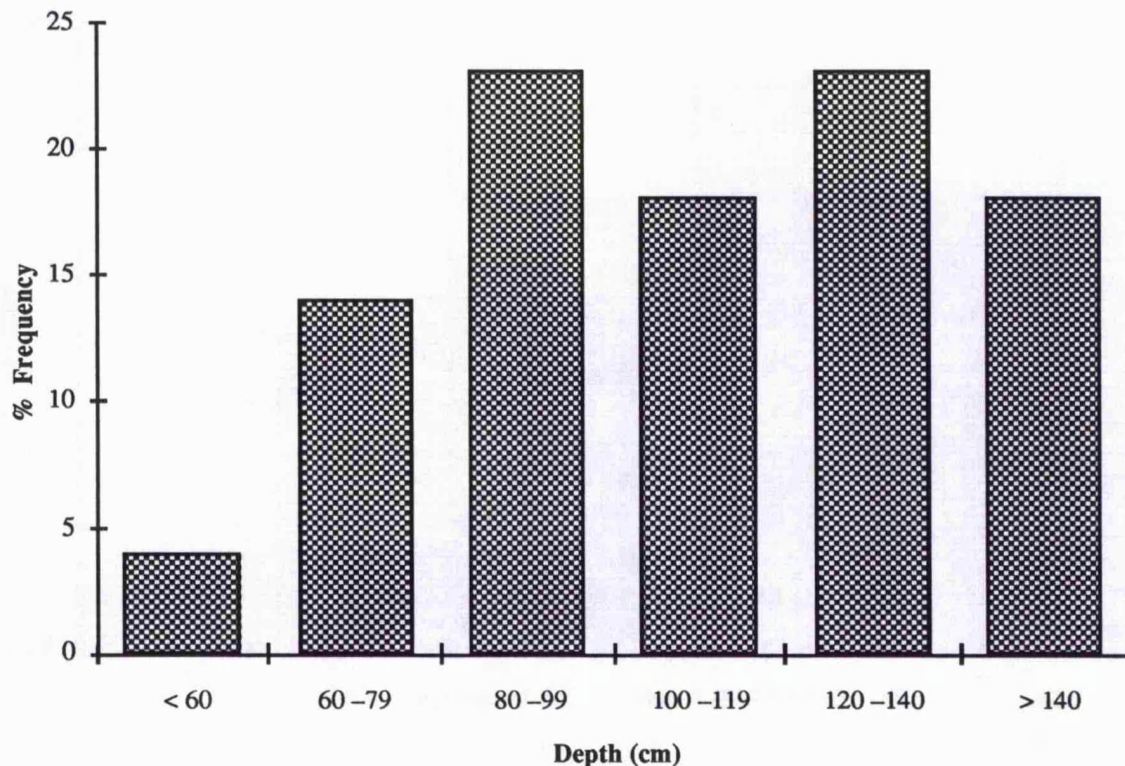


Figure 2.8 Percentage frequency distribution of maximum depths below each of 22 riffles from Harper's Brook.

depth formed 18%, those with 120 – 140 cm depth comprised 23% and those with > 140 cm depth formed 18% (Fig. 2.8).

In order to show the relationship between the depth of pools and riffles, these were plotted against each other on a graph after removing the three deepest riffles from the data. The results indicate that there is a negative relationship between the depth of pools and riffles which means the shallower riffles have tended to create deeper pools (Fig. 2.9). The three riffles which were not included the analysis are all within the most downstream seven riffles (riffles no. 2, 3 and 7). This is probably because the channel in the downstream reaches of the 2 km stretch within which the riffles were reinstated is deeper overall, because it is backed up by the river's confluence with the river Nene immediately below the section. In conclusion 15 out of 22 (68%) of the artificial riffles were successful in that they are beginning to create pools.

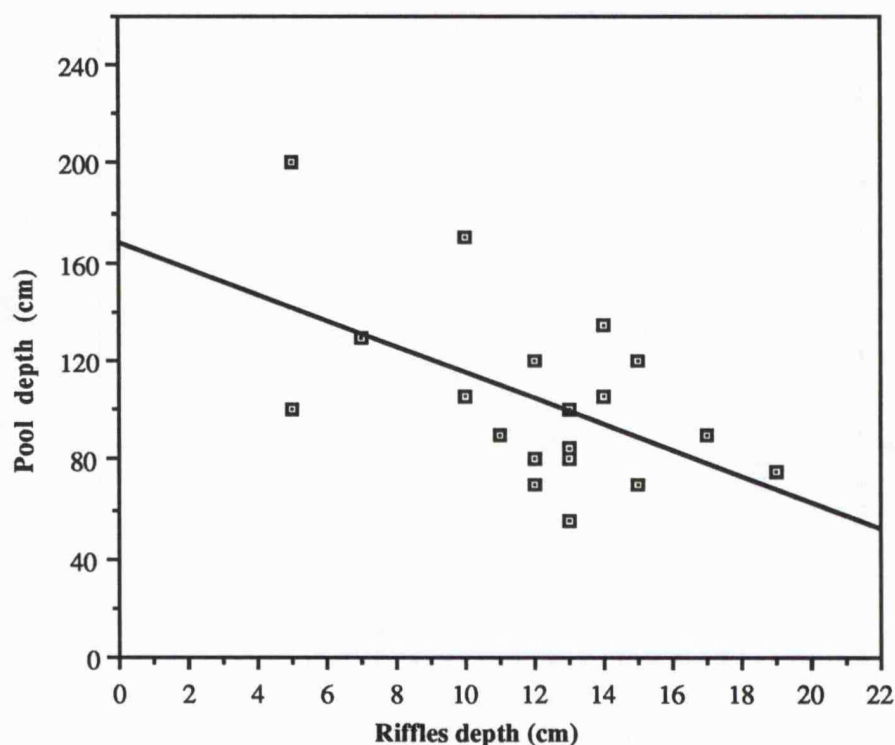


Figure 2.9 Relationship between depth of the artificial riffles and depth of the pools from Harper's Brook. The regression line is shown.

2.2.1.3 Choice of sites for biological sampling

The three artificial riffles (AR1, AR2 and AR3) selected for this study were chosen from different depth classes which represent the range of riffles present. AR3, belongs to the second greatest percentage class (23%) a shallow site; AR2, belongs to the greatest percentage class (36%) of moderate depth; and AR1, belongs to the third highest percentage class (14%) and is very deep (Fig. 2.6).

Again the three selected artificial riffles in this study lie in the three different velocity classes. AR1 is in the sluggish class which has the second smallest percentage frequency (14%), AR2 is in the moderate-flowing class with the greatest percentage frequency (36%) and AR3 is in the fast-flowing class with the second greatest percentage frequency (27%) (Fig. 2.7).

It can be concluded that AR1 is a very deep, sluggish riffle which is typical of 14% of the riffles. AR2 is a moderate riffle both in depth and velocity and is typical of 36% of the riffles. AR3 is a shallow and fast-flowing, and typical of 23% (depth) to 27% (velocity) of the riffles.

The depth of three runs selected for this study lie in the first three classes of the frequency distribution of the pools (Fig 2.8). Run 2 (38 cm) is the shallowest one and lies in the first class, Run 3 (72 cm) is a moderate run and lies in the second class, and Run 1 (87 cm) is a deep one and lies in the third class.

The data from pre riffle reinstatement time is not available, but since about 50% of the pools have the depth more than the deepest run it seems that some of the artificial riffles were successful in creating pools below them.

2.2.2 CURRENT DEFLECTORS (RIVER SMITE)

In order to show the effectiveness of the seven current deflectors put in the River Smite by the NRA in changing the geomorphology and hydrology of the channel, the depth and velocity in the current deflector plus the pool and the shoal below each deflector (when present) were measured at the midpoint of each habitat. The depth of the river bed was also measured in a longitudinal transect at every metre over the three most downstream deflectors.

The results (Fig. 2.10) showed that each deflector is followed by a deeper pool and shallower shoal downstream except for deflector 7 below which a shoal was not found. Since the pools which are created by the deflectors are relatively deeper than the other natural depressions it seems that the deflectors have changed the geomorphology of the river bed. The substrate which is scoured from the pool, by the greater turbulence produced by the deflector, has been transported and deposited downstream and created a shoal. Deflector 1, which was chosen for this study has a greater water depth than the mean depth of the seven deflectors (32 cm cf. 20 cm), similarly pool 1 has the greater water depth than the mean of seven pools (73 cm cf. 68 cm) and shoal 1 has the greater depth than the mean of the six shoals (21 cm cf. 17 cm). Deflector 1 created a relatively longer pool and shoal than deflector 2 and 3. This is possibly because the River Whipling joins the River Smite at the midpoint between deflector 1 and 2 and produces a greater discharge. Deflector 1 however was chosen over the others because of a greater pool-shoal development combined with easier physical access.

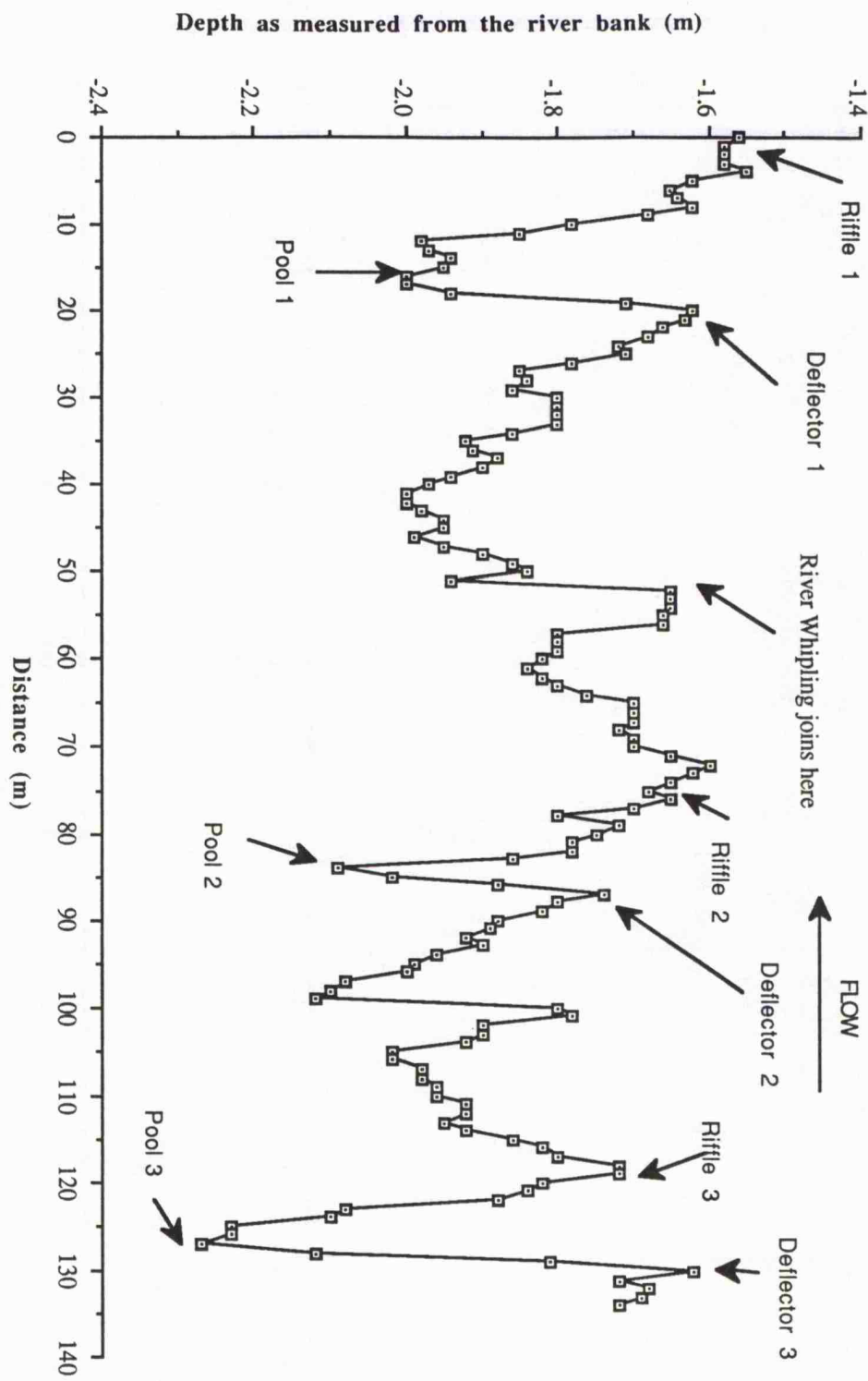


Figure 2.10 The profile of the river bed and the position of the three most downstream deflectors from The River Smite.

2.3 SAMPLING METHODOLOGY

2.3.1 SAMPLING SITES

Samples were collected from three different sites within each river: the control site, an improved site and a still-degraded site. The control site was a riffle habitat in the semi-natural section of the river, which was selected from a location as similar as possible to a natural riffle. The improved site was either an artificial riffle in the case of Harper's Brook or a shoal created downstream of the current deflector in the case of the River Smite. The still-degraded site was a small uniform section (5–6 m long) of the channelised reach, normally with a sand/silt bed and with no trees on its bank.

Although the artificial riffles in Harper's Brook are superficially similar, there are some differences in depth and velocity in different parts of each riffle and in individual riffles which led to the site selection described above. Therefore, three different riffles were selected as improved sites. The riffles were: deep sluggish and silted, designated Artificial Riffle 1; moderate in both depth and flow, designated Artificial Riffle 2; and shallow with fast flow, designated Artificial Riffle 3 (Plates 1 and 2). Similarly, three different runs were sampled: a deep sluggish run; a run of moderate depth and flow; and a shallow run. These were chosen as degraded sites (Plate 3). The control site is called 'natural' riffle (Plate 4). The locations of the different sites on Harper's Brook are shown in Figure 2.3 and the sample abbreviations are listed in Table 2.1.

The sites selected on the River Smite were; a 'natural' riffle as the control site; a run as a still-degraded site; and sites above the deflector, below the deflector and the shoal as the improved sites (Plate 5). The locations of the different sites on River Smite are shown in Figure 2.5, and sample abbreviations are listed in table 2.2. In each site three replicates; left bank, right bank and middle, were taken.



Plate 1 Harper's Brook, artificial riffle 2, looking downstream.



Plate 2 Harper's Brook, artificial riffle 3, looking upstream.



Plate 3 Long run and artificial riffle from the mid-reach of the rehabilitated section, Harper's Brook.



Plate 4 Natural riffle from Harper's Brook.



Plate 5 Current deflector and a downstream shoal from the River Smite.

Table 2.1 Sampling sites and abbreviations for Harper's Brook.

Site	Habitats	Samples	Abbreviations
Control	Natural Riffle	Left	NRL
		Middle	NRM
		Right	NRR
Improved	Artificial Riffle 1	Left	AR1L
		Middle	AR1M
		Right	AR1R
	Artificial Riffle 2	Left	AR2L
		Middle	AR2M
		Right	AR2R
	Artificial Riffle 3	Left	AR3L
		Middle	AR3M
		Right	AR3R
Degraded	Run 1	Left	R1L
		Middle	R1M
		Right	R1R
	Run 2	Left	R2L
		Middle	R2M
		Right	R2R
	Run 3	Left	R3L
		Middle	R3M
		Right	R3R

Table 2.2 Sampling sites and abbreviations for River Smite.

Sites	Habitats	Samples	Abbreviations
Control	Natural Riffle	Left	NRL
		Middle	NRM
		Right	NRR
Improved	Above Deflector	Left	ADL
		Middle	ADM
		Right	ADR
	Below Deflector	Left	BDL
		Middle	BDM
		Right	BDR
	Shoal	Left	ShL
		Middle	ShM
		Right	ShR
Degraded	Run	Left	RL
		Middle	RM
		Right	RR

2.3.2 SAMPLING DEVICES

Many authors (e.g. Macan, 1958; Cummins, 1962; Hynes, 1970) have described and used different sampling devices, primarily in quantitative studies. The literature was reviewed before final selection of methods was made.

2.3.2.1 Kick sampling

Kick sampling is a popular method used by many authors (Hynes, 1961; Morgan & Egglshaw, 1965; Egglshaw, 1969; Armitage *et al.*, 1974; Furse *et al.*, 1981; Ormerod, 1988; Wright, *et al.*, 1993), primarily in qualitative studies of lotic macroinvertebrates. It is a very effective, easy to use and time-saving method of collecting samples of macroinvertebrates from a wide variety of substrates in stream beds which vary from silt to boulders.

According to Hynes (1970), if some numerical assessment of the differences between areas is merely required, then simpler methods can be used, and they are often more satisfactory. He recommended collecting samples with a pond-net for a fixed time or kicking up the substrate in a standardised way in front of the mouth of a net. Hynes (1961) used a triangular pond-net, which was held vertically against the stream bed, to collect the animals whilst the area of the stream bed immediately upstream was vigorously stirred by the collector.

Morgan & Egglshaw (1965) suggested that it was not necessary to obtain absolute values of the quantity of the bottom fauna in a stream, as long as the differences between catches were directly proportional to the differences between streams. These authors used the kick sampling technique and provided justifications for the use of pond-nets in their surveys. They collected a series of six-kick samples at Allt Leathan, Perthshire, and the net was emptied after every second kick so that each sample was the sum of three pairs of kicks at one site. Of the total catch at each site, 51–87% was taken in the first two kicks, 9–36% in the second two kicks and 4–15% in the third two kicks. Therefore, they decided that four kicks at each site would be adequate. More kicks would give very few more

animals but would increase the amount of debris from which they would have to be separated.

The main sampling method used throughout the Armitage *et al.* (1974) survey was kick sampling. The operator kicked and stirred up the stream bed for 60 seconds upstream of a net 25 cm in diameter with a mesh of 10 threads per cm. These authors provided justification for the use of pond-nets in their surveys, and found that 10.5 kicks gave a catch equivalent to the populations of 1 m².

Furse *et al.* (1981) believed that in the River Communities project comparable standardisation of netting techniques is not feasible, because the nature of sites is different and efforts at standardisation should be confined to the equipment used, the duration of netting and the sampling aims. These authors used a standard Freshwater Biological Association (FBA) pond-net throughout their survey. This net with 900 µm mesh, 230 × 255 mm frame and 275 mm bag depth, was fitted to a 1.5 m handle. Each sample was of 3 minutes duration; this was also the period most commonly used by Water Authority biologists.

2.3.2.2 Surber sampler

The most widely used sampling device for investigations of stream benthos is the Surber-type "square-foot" sampler. The animals in a one square foot area, enclosed by a frame, are washed into a net on the downstream side of the frame (Surber, 1937). The substrate is stirred up by a rod or the operator's hand. Modifications of the general type were made by Leonard (1939) for work in slow flowing water and Hess (1941) for investigations in fast flowing waters. In both cases the addition of a screen enclosure prevented animals from moving into or out of the area. Although modified and improved since its initial introduction there are some limitations to using the Surber sampler in sampling invertebrate communities from river bed. These are:

- (a) The sampler is relatively heavy and is not easy to use.
- (b) It samples a small area and a large sample size is needed to cover a site.
- (c) It can not be used in sampling deeper sites (e.g. pools).

(d) Sampling time is longer than kick sampling.

2.3.2.3 Grabs and Corers

Sampling devices suitable for collection of substrate and animals in the stream investigations are grabs which were originally developed for marine and lake studies. Only two grabs have been used extensively by limnologists; the Ekman and the Petersen grabs. Both devices are best suited for fairly soft sediments which are relatively free of aquatic vegetation, although the latter can be used in gravel (Cummins, 1962). The Petersen grab is also especially useful for sampling in weed beds (Needham & Usinger, 1956). Both grabs bite the bottom from above.

Corers are also devices suitable for collection of substrate and animals simultaneously. The device is pushed by hand into the substrate and so bites out a definite area (Minckley, 1963). An interesting addition to this technique is the use of very low-temperature fluids to freeze the sample *in situ* before the core is withdrawn, thus retaining it in the sampler without disturbance (Efford, 1960). As in the Surber sampler there are limitations on using grabs and corers. The most important one is these samplers are not suitable for sampling riffle habitats. Using grabs is a troublesome operation and more than one operator is needed for sampling. Another disadvantage is the loss of part of the sample while transferring the sample from the river bed to the container. Pugsley & Hynes (1983) believed that, although the freeze core method is reliable for quantitatively sampling benthos in a variety of habitat, it is difficult to use and expensive to build and operate.

2.3.2.4 Boxes and cylinders

Boxes and cylinders enclose an area of stream bed and the animals contained are removed in a number of ways. The boxes are cylindrical or square and open at both ends, the ends are pushed into the substrate. In some of the circular ones the bottom is jaggedly toothed and the top is fitted with lateral handles so that the sampler may be rotated while being pushed and thus cut its way downward. This modification is particularly useful on rough stony bottoms (Hynes, 1970).

Kamler & Riedel (1960) suggested the use of a metal plate which could be pushed through the substrate under the box and used to lift both the box and its contents out. By washing the stones which were enclosed in the box and stirring up the remaining substrate, Needham (1934) removed animals with a net until no further animals could be found.

Obviously boxes can not be used in sampling deep sites. Furthermore sampling involves spending much time recovering animals from the sampler.

2.3.2.5 Artificial substrates

Trays or baskets filled with natural substrate or materials such as hardboard layers or bushes (Scott, 1958), or even artificial materials, are alternative methods of quantitative sampling. These devices are left on the river bed and afterwards removed to obtain data on bottom fauna. In addition to the extra effort required for setting and recovering, artificial substrates are subject to loss (e.g. flooding) during the colonisation period (Brown *et al.*, 1987). Furthermore they cannot copy the exact natural substrate size and texture of the diverse habitat of the stream bed.

2.3.2.6 Vacuum benthos sampler

Brown *et al.* (1987) introduced a vacuum benthos sampler for quantitative study of benthic fauna. The sampler is suitable for sampling diverse habitats. It is independent of natural flow (as are the Surber and kick sampling) and is effective in standing water, low or high current velocities. Brown *et al.* (*op. cit.*) believe that the device has overcome the several problems involved in suction sampling. Namely overestimating the abundance of some species, because they draw animals from substrate interstices deeper than or outside the sampler, or underestimating the abundance of strong swimmers (e.g. *Baetis*) and organisms with heavy cases or shells. Suction samplers also damage delicate organisms and most require two or more persons for transportation and operation. The authors compared the vacuum benthos sampler with other samplers such as the Hess sampler and stated that it is a very effective quantitative sampler for sampling all invertebrate taxa when

properly used. The vacuum benthos sampler has a relatively sophisticated structure and needs special care once it is in operation (see Brown *et al.*, *op.cit.* for detailed information regarding the structure and operation of the equipment). Although it is a heavy piece of equipment, one operator can carry and operate it.

2.3.3 REASONS FOR CHOOSING KICK SAMPLING

In comparison with the other methods reviewed above, there are several advantages of kick sampling. These are:

- (1) it is a simple and easily used method,
- (2) several samples can be collected over a relatively short period of time,
- (3) different habitats; riffles, pools and runs with a range of substrate from silt to cobble can be sampled by the same method,
- (4) the semi-quantitative nature of the objective of the study, which was the comparison of the invertebrate communities of different sites, could be covered by this method.

Therefore, in view of the ease of use, speed and objective of the study, it was decided to use the kick sampling technique for the collection of macroinvertebrate samples during this project. A pond-net, (1 mm mesh, 230 × 280 mm frame, 330 mm bag depth), fitted to a 1.2 m handle, was used on all sampling occasions. In using it the net was held vertically on the stream bed facing upstream, and the substrate was disturbed just upstream of it with a booted foot for 60 seconds in an upstream direction. Samples were transferred to buckets, containing some water, with watertight lids, and preserved with 4% Formalin. After each sampling the net was washed, inside out, by sweeping it upwards through the water several times.

2.3.4 SAMPLING STRATEGY, FREQUENCY AND SIZE

A sampling procedure was needed that would yield a good approximation of the whole community from a small number of samples. There are three types of sampling

strategies: simple random sampling, stratified random sampling and systematic sampling (Brown & Brussock, 1991).

For the execution of the simple random sampling a grid is superimposed on the study area and intercepts numbered. Then sample sites are determined using a table of random numbers.

Stratified sampling involves the selection of habitat types, such as riffle or pool, followed by the application of random sampling to each of these discrete areas.

A third and very common procedure, is systematic or transect sampling. Because the substrate characteristics of aquatic environments are determined, within limits, by currents, the transect is most likely to cut across the range of habitats present. Since the deposition of fine particles along the shores of a stream is dependent on channel gradient and specific location within the channel, several complete transects are required to adequately sample a given area (Cummins, 1962).

Although the artificial riffles in Harper's Brook and the other selected sites are superficially similar, there are some differences (depth and velocity) in different parts of each riffle and in individual riffles. In order to partially overcome the variation in sample composition, which is due to the differences in the nature of sampling points, the systematic sampling strategy was chosen and samples were collected along three transects (left, middle and right) of each selected site (5–6 m long).

The following constraints (based on a preliminary investigation) had to be considered in regard to the sample size and number of replicates:

- (1) Time required for collecting samples (one day for each river on each occasion).
- (2) Time required for sorting and counting animals (one day per replicate sample).
- (3) Time required for identifying animals to lowest possible taxonomic levels (two days per replicate sample).
- (4) The three year time scale of the research programme.

Thus it was decided to have three replicate samples, collected along three transects at each site in each season. The sampling was repeated in spring, summer, autumn and winter to take account of seasonality.

Three replicate samples were the minimum that could be collected for statistical analysis of each habitat. Three representatives of the improved habitat were the minimum that could be taken to include potential variation within them. Less variation was apparent in the degraded or the 'control' sites and so only the three replicates were sampled here, except in Harper's Brook where there appeared to be greater physical differences in the "still-degraded" run sections.

In total 33 samples from the two rivers were collected on the first sample occasion in the summer of 1993. Processing these samples was going to take approximately 100 days and so for reasons of practicality it was decided to sample in a similar fashion once each season, i.e. four samples in a year. Therefore, approximately 400 days were spent processing all samples. sample dates and frequencies are shown in Tables 2.3 & 2.4.

Table 2.3 Sampling dates for Harper's Brook and the River Smite.

Rivers	Summer	Autumn	Winter	Spring
Harper's Brook	Aug. 20, 1993	Nov. 23, 1993	Mar. 7, 1994	May, 16, 1994
River Smite	Sep. 8, 1993	Nov. 24, 1993	Feb. 24, 1994	May, 17, 1994

Table 2.4 Sampling frequencies for Harper's Brook and the River Smite.

Rivers	Number of sites	Number of replicates	Sampling frequency	Total number of samples
Harper's Brook	7	3	4	84
River Smite	5*	3	4	57

* Only four sites were sampled during the summer.

2.3.5 MEASURING ENVIRONMENTAL FEATURES

The physical and environmental characteristics of each site at the position of the replicate sample; depth, flow velocity and width were measured on each sampling occasion. Substrate type, vegetation type and cover were estimated by eye.

A measuring tape was fixed along the left margin of the site (6 m). Using a one meter metal ruler, the water depths were measured every half meter along left, middle and right transects, corresponding to the three sampling points of the site. Therefore the depth of each sampling point was taken as the mean of the transect depth measurements.

The current velocity was measured at the middle point of each transect using a portable current meter (Type C2 "10.150"). The instrument was positioned at 60% depth from the surface and the number of revolutions was recorded over 30 seconds. The current velocity was then calculated using the following equations, being particular to the instrument used.

$$\text{For } n < 0.59 \quad v = 0.2347 n + 0.015$$

$$\text{For } n \geq 0.59 \quad v = 0.2535 n + 0.004$$

Where, v = velocity of water (m s^{-1}) and n = number of revolutions per second.

Three width measurements were taken at the upper, middle and lower segments of each site on all sampling occasions, and the mean width was calculated.

Substrate type was recorded and the percentage of each was estimated. The following particle size categories were used:

Boulder	> 20 cm smallest diameter.
Cobble	< 20 – 5 cm "
Gravel	< 5 – 0.5 cm "
Sand	< 0.5 cm, fine sediment
Silt	very fine sediment

2.4 LABORATORY WORK

2.4.1 SORTING SAMPLES

Samples usually contain animals, aquatic vegetation, detritus and some mineral substrate. A small portion of each sample was emptied into a small hand net (*ca.* 0.08 mm mesh size) and washed with tap water to remove the silt and fine debris. The contents of the hand net was then emptied into a plastic tray. Small stones, coarse gravels and aquatic vegetation were removed after being searched for clinging animals. Then the organic part of the tray content was separated from minerals by several decantations and transferred to a tray. The mineral part was searched for animals before being discarded. The remaining part of the sample was treated similarly. The recovered material of each sample was transferred to a plastic white sorting tray and the animals were sorted and counted as described in the following section.

2.4.2 COUNTING MACROINVERTEBRATES AND SUBSAMPLING

Animals were picked up by hand, fine forceps, or wide-mouth plastic pipette and transferred to 100 ml plastic containers with 70% Industrial Methylated Spirit (I.M.S.) for preservation. At this stage animals were sorted to the higher taxonomic levels of class, order or family and each taxon was kept in separate containers for further identification. The larger animals such as leeches, gastropods, bivalves and odonates were first sorted, counted and transferred to the containers. Any sample having populations less than 100 of any taxon (estimated visually) was fully sorted and the number of each taxon recorded. Data sheets containing the following information; site, habitat, sampling date and taxa, were prepared in advance and used for recording the number of each taxon at sorting times. Samples having larger populations were subsampled. The sorting tray was divided into two, four and eight equal parts by a permanent marker pen. The sample was transferred into the tray with some water and an effort was made to distribute it around the tray as evenly as possible using forceps. A half, quarter, or an eighth (depending on the

number of individuals in the sample) was carefully removed from the tray using a spoon, forceps and a pipette. Care was taken not to include any of the rest of the sample in this subsection. The subsample was then placed into another tray and sorted, all invertebrates were removed and counted. The other subsample parts were sorted in every case in order to recover any taxon which was not found in the subsample, although none was ever found.

2.4.3 TESTING THE VALIDITY OF SUBSAMPLING RANDOMNESS

In order to test the validity of subsampling randomness, the Chi-square test was applied. A sample from each habitat (e.g. artificial riffle, natural riffle) was divided into eighths and all invertebrates found in five of these subdivisions were removed and counted. The numbers found were tested to see if invertebrate distribution in the tray was random.

The Chi-square was calculated from the following equation:

$$\chi^2 = \frac{s^2(n-1)}{\bar{x}}$$

or Chi-square = (Variance x Degree of freedom) / sample mean (Elliott, 1971).

According to Elliott (*op.cit.*) if the value of χ^2 lies between the upper limit (suggesting contagious distribution) and lower limit (suggesting regular distribution) of the table 4 in Fisher & Yates (1963) (p. 47), it will show agreement with a Poisson distribution which is an accepted method for testing random distribution. The results which are shown in tables 2.5–2.7 indicate that the distribution of invertebrate in the tray was random.

Table 2.5 Number of animals in 5 of 8 tray subdivisions from a Natural Riffle sample of Harper's Brook. χ^2 upper limit = 9.49, χ^2 lower limit = 0.71, Degrees of freedom = 4, P \leq 0.05.

Species name	Number in subsample	Mean	Variance	Chi-square
<i>Caenis luctuosa</i>	56 45 46 39 35	44.2	63.7	5.8
<i>Hydroptila</i> sp.	49 56 44 32 47	45.6	77.3	6.8
<i>Elmis aenea</i>	40 22 38 30 23	30.6	68.8	9.0
Chironomidae	118 120 129 125 108	120.0	63.5	2.1

Table 2.6 Number of animals in 5 of 8 tray subdivisions from an Artificial Riffle sample of Harper's Brook. χ^2 upper limit = 9.49, χ^2 lower limit = 0.71, Degrees of freedom = 4, P \leq 0.05.

Species name	Number in subsample	Mean	Variance	Chi-square
<i>Gammarus pulex</i>	28 32 52 42 40	38.8	87.2	9.0
<i>Caenis luctuosa</i>	72 45 51 68 60	59.2	127.7	8.6
<i>Hydropsyche siltalai</i>	15 15 15 19 16	16	3	0.8
<i>Elmis aenea</i>	37 29 28 38 41	34.6	33.3	3.8

Table 2.7 Number of animals in 5 of 8 tray subdivisions from a Natural Riffle sample of the River Smite. χ^2 upper limit = 9.49, χ^2 lower limit = 0.71, Degrees of freedom = 4, P \leq 0.05.

Species name	Number in subsample	Mean	Variance	Chi-square
<i>Caenis luctuosa</i>	19 21 15 28 22	21	22.5	4.3
<i>Baetis vernus</i>	3 4 6 5 8	5.2	3.7	2.8
<i>Hydropsyche angustipennis</i>	6 6 8 10 5	7	4	2.3
<i>Athripsodes</i> spp.	9 12 13 12 9	11	3.5	1.3
<i>Elmis aenea</i>	17 30 19 32 19	23.4	49.3	8.4

2.4.4 IDENTIFICATION

Most animals were identified, in I.M.S., using a Kyowa Stereo microscope with a zoom objectives from x0.7 to x4.5. The microscope was supplied with a Schott cold light source. Keys of the Freshwater Biological Association and others (Table 2.6) were used. Some groups such as oligochaetes and chironomid larvae needed more detailed examination under a Nikon phase contrast microscope at x40 or, using oil immersion, at x100, and had to be mounted on microscope slides first.

Pinder (1983) suggested the following procedure for mounting and preparation of the chironomid larvae: "The head capsule should be removed and macerated in a hot 10% solution of caustic potash (KOH) for 5–10 minutes. The specimen is then passed through glacial acetic acid (5 minutes), alcohol (e.g. ethanol or iso-propanol) (15 minutes) and alcohol layered over either cedarwood or clove oil (15 minutes) before mounting in Canada Balsam. If Euparal is used as a mounting medium the alcohol/cedarwood oil stage can be omitted" (p. 9). Taylor (1995) used a similar method, by placing the head capsules into lactic acid for approximately two weeks for clearing, and used Euparal as the mounting medium. Obviously both methods are laborious and very time consuming.

Di-methyl hydantoin formaldehyde (DMHF) resin has customarily been used as a mounting medium by the members of the Ecology Unit, University of Leicester. I used DMHF for both clearing and mounting chironomid larvae and the result was satisfactory. Head capsules were dissected from the body just above the anterior parapods. This was accomplished by holding the body of the animal, ventral side uppermost, with a pair of fine forceps and teasing the head off with a tungsten needle. Needles were mounted on short pieces of hollow perspex tubing which had been melted around the wire, and then sharpened as described by Harding & Smith (1960). Head capsules were mounted on microscope slides (76 x 26 x 1.0 mm) ventral side up in DMHF and covered with 13 mm diameter coverslips, and then left on a hot plate (approximately 40 °C) for one week. The bodies of some of the chironomid larvae (orthocladines) were treated similarly and mounted on the same slides on which their head capsules were mounted under separate coverslips.

The oligochaetes were mounted on microscope slides in DMHF, covered by a 22 x 50 mm coverslip and left on a hot plate (approximately 40 °C) for one week.

The following taxa were identified to species; Oligochaeta, Hirudinea, Isopoda, Amphipoda, Ephemeroptera, Trichoptera, Odonata, Megaloptera, Coleoptera, Hemiptera, Simuliidae, Gastropoda and Bivalvia. Chironomidae and Tipulidae were identified to genus, whilst other taxa were identified to either genus or family and in a few cases to higher taxonomic level.

Table 2.8 Keys and references used for identification of macroinvertebrates.

TAXA	AUTHOR	PUBLISHER
Oligochaetes	Brinkhurst, 1971	Freshwater Biological Association
Leeches	Elliott & Mann, 1979	Freshwater Biological Association
Malacostraca	Gledhill <i>et al.</i> , 1993	Freshwater Biological Association
Ephemeroptera	Elliott <i>et al.</i> , 1988	Freshwater Biological Association
Caseless caddis larvae	Edington & Hildrew, 1981	Freshwater Biological Association
Cased caddis larvae	Wallace <i>et al.</i> , 1990	Freshwater Biological Association
Megaloptera	Elliott, 1977	Freshwater Biological Association
Elminthidae	Holland, 1972	Freshwater Biological Association
Hemiptera	Savage, 1989	Freshwater Biological Association
Orthocladinae	Cranston, 1982	Freshwater Biological Association
Simuliidae	Davies, 1968	Freshwater Biological Association
Gastropods	Macan, 1960	Freshwater Biological Association
Coleoptera	Friday, 1988	Field Studies Council
Freshwater Invertebrates	Macan, 1974	Longman
Chironomidae	Cranston & Reiss, 1983	Entomologica Scandinavica
Chironominae	Pinder & Reiss, 1983	Entomologica Scandinavica
Orthocladinae	Cranston <i>et al.</i> , 1983	Entomologica Scandinavica
Prodiamesinae	Saether, 1983	Entomologica Scandinavica
Diamesinae	Oliver, 1983	Entomologica Scandinavica
Tanypodinae	Fittkau & Roback, 1983	Entomologica Scandinavica
Larvae of other Diptera	Smith, 1989	Royal Entomological Society
Odonata	Hammond, 1985	Harley Books

2.5. DATA ANALYSIS

The data which were originally recorded as species or taxa abundance were collected from a number of sites at four seasons from each river. The recorded data were arranged in species sample tables for each season and river separately using Microsoft Excel (version 4.0) on an Apple Macintosh computer. From the wide variety of analytical techniques two main approaches were chosen; first univariate, and second multivariate analyses.

2.5.1 UNIVARIATE METHODS

In order to test whether differences between sites were significant these methods were used in three steps.

2.5.1.1 Summarising the data

The full set of species counts for each sample was summarised into a single coefficient, the Shannon-Wiener diversity index. Species diversity is one of the most obvious and characteristic feature of a community. It has two distinct components:

a) *Species richness*, which is the total number of species in a community.

b) *Evenness*, that is how equally abundant the species are. Diversity describes the relative abundances of species using a measure which combines the two components, species richness and species abundance. The Shannon-Wiener index of diversity (H') takes into account not only the number of species but also the number of individuals per species (Krebs, 1989).

The Shannon-Wiener diversity index is probably the most widely used diversity index in community ecology and therefore appropriate in terms of comparability with similar studies. As the present study is comparing samples (communities) collected from different sites, the Shannon-Wiener diversity index is a good indicator for discriminating sites. According to Krebs (*op.cit.*) this index should be used only on random samples drawn from a large community in which the total number of species is known, which is not the case for most community samples, and he recommended using the Brillouin index

which does not use the same assumptions and seems to be more appropriate. But a few lines later he stated "there is much argument in the literature about whether the Brillouin index or Shannon index is a better measure of species diversity. In practice, this argument is irrelevant to field ecology because H (Brillouin index) and H' (Shannon-Wiener index) are nearly identical for most ecological samples" (p. 362). Thus, the Shannon diversity index was selected for use in this study. The following equation is used for the calculation:

$$H' = -\sum (p_i)(\ln p_i)$$

where, H' is the index of species diversity

p_i is the proportion of individuals found in the i th species.

\ln is the natural log (any log base can be used).

If base 2 logs are used the units of H' are bits per individual; if base e (i.e. natural) logs, nits per individual; and if base 10 logs, decits per individual (Krebs, *op. cit.*). The value of the Shannon diversity index is usually found to fall between 1.5 and 3.5 nits and only rarely surpasses 4.5 nits (Magurran, 1988).

According to Taylor (1978) if the Shannon-Wiener diversity index is calculated for a number of samples the indices themselves will be normally distributed. This makes it possible to use parametric statistics, including the powerful Analysis of Variance (ANOVA) methods, to compare sets of samples for which the diversity has been calculated. This is a useful method of comparing the diversity of different habitats, especially when a number of replicates have been taken. This has been used as clear evidence for applying ANOVA to the present data.

2.5.1.2 Comparison of the sites

In order to demonstrate any difference between the sites, one-way analysis of variance (ANOVA) was used. ANOVA is one of the most powerful statistical techniques and can be used to test the differences between numerous samples. The total variation in a set of data is divided into components associated with possible sources of variability, which include the variation between sample means and the variation between sample units within the samples (residual or error variation). Then the relative importance of the

different sources is assessed by an F-test (Elliott, 1977). ANOVA is a test of the hypothesis that there is no difference between sample means. Thereafter a further test must be applied to determine which means are significantly different from the other mean values. The 95% confidence intervals test, Tukey test (Föler & Cohen, 1994), and the least significant range test (Parker, 1983) are commonly recommended tests. In this study the 95% confidence intervals test was chosen for its better results and graphical representation.

2.5.1.3 Linking to environmental variables

In order to examine any relationship between diversity and independent environmental factors (depth and velocity) the techniques of regression and correlation analysis have been used.

2.5.2 MULTIVARIATE METHODS

2.5.2.1 Introduction

Multivariate analysis is the branch of mathematics that deals with the examination of numerous variables simultaneously. Community data are multivariate because each sample site is described by the abundances of a number of species, and because numerous environmental factors affect communities. The purpose of multivariate analysis is to treat multivariate data as a whole, summarising the data and revealing their structure (Gauch, 1985). Thus multivariate analysis provides relatively objective summaries of the data which gives a better understanding of the data structure and provides a means for effective communication of results.

2.5.2.2 Ordination and classification

Multivariate methods include two well-known techniques; ordination and classification. They have the common goal of organising data for the purposes of description, discussion and understanding of communities. Ordination and classification techniques organise community data on species abundances exclusively, apart from

environmental data, leaving environmental interpretation to a subsequent, independent step (Gauch, *op. cit.*).

The result of ordination is the arrangement of species and samples in a low-dimensional space such that similar entities are close and dissimilar entities far apart. The result of classification is the assignment of species and samples to classes; classes may or may not be arranged in a hierarchy. These two approaches are complementary.

DECORANA and TWINSpan, two well-known ordination and classification techniques, are used for the analysis of the present data. These techniques achieved considerable prominence and became widely used with the widespread availability of the related computer program in the 1980s. In the following account both techniques are briefly described.

Detrended Correspondence Analysis (DCA); DECORANA

This account is based on Hill (1994). DECORANA is a computer program designed primarily for ecologists who have collected data on the occurrence of a set of species in a set of samples. Its main purpose is to ordinate the data by the method of detrended correspondence analysis.

DCA is an improvement upon the reciprocal averaging (RA) or correspondence analysis (CA) ordination technique. RA has two main faults: the second axis is often an 'arch' or 'horseshoe' distortion of the first axis, and the distances in the ordination space do not have a consistent meaning in terms of compositional change (in particular distances at the ends of the first RA axis are compressed relative to the middle). DCA corrects these two faults. Tests with simulated and field data show DCA to be superior to RA and non metric multidimensional scaling in giving clear interpretable results (Hill & Gauch, 1980). DCA has several advantages.

- (a) Its performance is the best of the ordination techniques, and both species and samples ordinations are produced simultaneously.
- (b) The axes are scaled in standard deviation units with a definite meaning.

(c) As implemented in a FORTRAN program called DECORANA, computing time rises only linearly with the amount of data analysed, and only positive entries in the data matrix are stored in memory, so very large data sets present no difficulty.

According to Kent (1994), at the present time DCA and Canonical Correspondence Analysis (CCA) are widely accepted as the best available methods. CCA is a substantial improvement if a good set of environmental data can be supplied with the species data. A great deal of effort in environmental measurement is necessary, however, in order to obtain such data. In summary, where a good set of environmental data are available in addition to species data, then CCA is more suitable. Where such data are not available, DCA probably still remains the most appropriate choice. DCA is selected for ordination of the sites of the present study.

Although DECORANA has been primarily used for the analysis of plant community data, there are numerous accounts in the literature about the application of the method for analysis of macroinvertebrate community data (e.g. Furse *et al.*, 1984; Wright *et al.*, 1984; Bunn *et al.*, 1986; Ormerod & Edwards, 1987; Rutt *et al.*, 1989, 1990; Rundle *et al.*, 1990, 1993).

Two-way Indicator Species Analysis; TWINSpan

TWINSpan is now the most widely used technique for polythetic divisive classification. TWINSpan is a computer program in FORTRAN designed primarily for ecologists and phytosociologists who have collected data on the occurrence of a set of species in a set of samples. TWINSpan is a development of a method previously published under the name "indicator species analysis" (Hill, 1994).

In TWINSpan the data are first ordinated by reciprocal averaging (RA). Then those species that characterised the RA axis extremes are emphasised in order to polarise the samples, and the samples are divided into two clusters by breaking the ordination axis near its middle. The sample division is refined by a reclassification using species with maximum value for indicating the poles of the ordination axis. The division process is then

repeated on the two sample subsets to give four clusters, and so on, until each cluster has no more than a chosen minimum number of members. A corresponding species classification is produced, and the sample and species hierarchical classification are used together to produce an arranged data matrix. The resultant sample hierarchy (and species hierarchy) may also be displayed as a dendrogram, using the sequences of divisions as integral levels or computing the levels as the average distances between samples in ordination space (Gauch & Witteraker, 1981).

The computer program for TWINSpan, unlike any other hierarchical classification program, deliberately arranges the two clusters at each node in a way that results in the most similar samples being placed together in the dendrogram's sample sequence. This makes the information in the dendrograms more lucid (Gauch, 1985).

CHAPTER THREE

Harper's Brook

CHAPTER THREE

HARPER'S BROOK

3.1 PHYSICAL AND ENVIRONMENTAL CHARACTERISTICS

Depth, velocity, substrate and vegetation are the most important factors which make one site different from another, and subsequently affect the community composition of the site. Tables 3.1– 3.4 show the physical and environmental data for Harper's Brook which were collected in four seasons in 1993–1994.

Table 3.1 Environmental data from Harper's Brook in summer 1993.

Site	Length (m)	Mean width (m)	Mean depth (m)	Mean velocity (m/s)	Substrate	Vegetation
NR	7	3.4	0.15	0.56	60% cobble, 40% gravel & sand	50% <i>Cladophora</i> , 10% <i>Ranunculus</i>
AR1	6	4.3	0.34	0.16	10% boulder, 50% cobble, 40% gravel & sand	70% <i>Cladophora</i>
AR2	8	4.6	0.18	0.5	70% cobble, 30% gravel & sand	80% <i>Cladophora</i>
AR3	6	3.4	0.15	0.41	90% cobble, 10% gravel & sand	90% <i>Cladophora</i> 5% <i>Sparganium</i>
R1	6	5.5	0.72	0.14	sand & silt	5% <i>Sparganium</i>
R2	6	4.5	0.3	0.15	gravel & sand	50% <i>Cladophora</i>
R3	6	3.5	0.41	0.15	gravel & sand	80% <i>Cladophora</i>

The riffle sites (NR, AR1–3) of Harper's Brook in summer (August)1993 were dominated by cobble substrate (> 50%) and the run sites (R1–3) had gravel and sandy substrate (R1 had sand and silty substrate) (Table 3.1). Riffles and run sites were also different in depth and current velocity. The natural riffle (NR) had the highest velocity (0.56 m s⁻¹) and Run 1(R1) had the lowest velocity (0.14 m s⁻¹). The deepest site was R1

(0.72 m) and the shallowest were NR and AR3 (0.15 m). Generally as the depth increases the current velocity decreases, but this is not always true, because other factors such as the substrate and discharge are also involved. The amount of vegetation is also related to the depth, as the depth increases the vegetation cover due to the reduced light penetration, decreases. R1 which was the deepest site had the least vegetation (5%), and the greatest amount of vegetation occurred on AR3 (95%) which had the least depth.

The physical data in autumn (November) 1993 (Table 3.2) show that the amount of vegetation had declined at all sites, two levels ranging from none in run sites and in AR1 to the maximum of 50% in AR3. In comparison with the summer data the mean depth of all sites increased, because of the higher discharge in autumn. Current velocity showed an increase at riffle sites and decrease at run sites compared with summer. Riffles had the higher velocity and lower depth and the runs *vice versa*. The dominant substrate in riffle sites was cobble with the exception of AR1 which had 70% gravel and sand (compared with 40% in summer). The substrate of R3 changed from gravel and sand to sand and silt, and thus it is probable that the finer substrate were deposited during the low flows between summer and autumn.

Table 3.2 Environmental data from Harper's Brook in autumn 1993.

Site	Length (m)	Mean width (m)	Mean depth (m)	Mean velocity (m/s)	Substrate	Vegetation
NR	7	4	0.22	0.68	10% boulder, 50% cobble, 40% gravel & sand	20% <i>Cladophora</i>
AR1	6	4.9	0.5	0.23	10% boulder, 20% cobble, 70% gravel & sand	10% <i>Cladophora</i>
AR2	8	4.9	0.23	0.66	50% cobble, 50% gravel & sand	20% <i>Cladophora</i>
AR3	6	3.7	0.23	0.67	60% cobble, 40% gravel & sand	50% <i>Cladophora</i>
R1	6	5.2	0.79	0.08	sand & silt	none
R2	6	4.8	0.35	0.4	gravel & sand	none
R3	6	3.9	0.72	0.07	sand & silt	none

The physical data collected in winter (February) 1993/1994 (Table 3.3) show that the riffle and run sites generally had slightly greater depth and velocity, due to the higher discharge, than in the autumn. AR1 was again different from the other riffles in being deeper and slower, as in summer and autumn. R1 was again the deepest (0.87 m) and slowest (0.14 m s^{-1}) site. The vegetation cover was low; all run sites and AR1 had no vegetation, AR3 had 70% vegetation and the others had 20% to 30%. Cobble was the dominant substrate of the riffle sites, and the substrate of run sites was composed of gravel and sand except for R1 which had sand and silt.

Table 3.3 Environmental data from Harper's Brook in winter 1993/94.

Site	Length (m)	Mean width (m)	Mean depth (m)	Mean velocity (m/s)	Substrate	Vegetation
NR	7	3.9	0.24	0.58	20% boulder, 40% cobble, 40% gravel	20% <i>Cladophora</i> 10% <i>Ranunculus</i>
AR1	6	5.2	0.53	0.29	50% boulder, 50% cobble & gravel, covered by sand & silt	none
AR2	8	5.1	0.27	0.67	90% cobble & gravel, 10% sand	20% <i>Cladophora</i>
AR3	6	3.9	0.23	0.78	50% cobble, 50% gravel & sand	70% <i>Cladophora</i>
R1	6	5.3	0.87	0.14	sand & silt	none
R2	6	4.8	0.38	0.39	80% gravel, 20% sand	none
R3	6	4.1	0.77	0.17	10% gravel, 90% sand	none

In spring (May) 1994 the depth and velocity of all sites were slightly lower than winter (Table 3.4), apparently due to less discharge. The vegetation cover did not change much in most of the sites, except for AR3 which decreased to 15%. The occurrence of spates could have been the reason for the low vegetation cover in spring.

Table 3.4 Environmental data from Harper's Brook in spring 1994.

Site	Length (m)	Mean width (m)	Mean depth (m)	Mean velocity (m/s)	Substrate	Vegetation
NR	7	3.6	0.18	0.51	60% cobble, 40% gravel & sand	10% <i>Ranunculus</i> , 5% <i>Cladophora</i>
AR1	6	5.4	0.48	0.1	50% boulder, 50% cobble & gravel	none
AR2	8	4.9	0.22	0.47	80% cobble, 20% gravel & sand	15% <i>Cladophora</i>
AR3	6	4	0.22	0.49	80% cobble, 20% gravel & sand	15% <i>Cladophora</i>
R1	6	5.2	0.82	0.01	sand & silt	none
R2	6	4.5	0.31	0.22	gravel & sand	none
R3	6	4.3	0.69	0.07	sand & silt	none

It can be concluded that the physical characteristics of the sites differed from each other within and between seasons. AR2 and AR3 had almost identical physical characteristics and were similar to NR while AR1 was different from the other riffle sites and was more like the run sites in depth and velocity, despite being "riffle" in the substrate type. Cobble was the dominant substrate of riffle sites and vegetation was present in all riffle sites in all seasons except for AR1 in winter and spring. Run sites were deeper and slower than riffles and there was no vegetation present in autumn, winter and spring in any of the run sites. Gravel and sand or sand and silt formed the substrate of runs. There was a tendency for decreasing vegetation from summer to spring. Due to the higher discharge, there was also a tendency for the water depth to increase from summer to spring.

3.2 MACROINVERTEBRATE COMPOSITION AND ABUNDANCE

The full lists of species composition and the abundance of 21 samples from Harper's Brook in four seasons are in Appendix one.

3.2.1 TAXONOMIC LEVEL OF THE DATA

In order to specify which taxonomic level is the best for representing the data and comparing the sites the summer data were compared at three different levels of taxonomy; species, family and higher level (e.g. order). Tables 3.5 – 3.7 show relative abundances of taxa at different taxonomic levels, comparing control, improved and degraded groups of sites. Only the taxa forming $\geq 1.0\%$ of the total abundance at one or more habitat types were considered.

Since the representatives of some families belonging to the same order are generally collected from different habitats, it was decided not to represent the data at the highest taxonomic level. For example, in the Ephemeroptera, Caenidae are more abundant in degraded sites (runs), whereas Baetidae are dominant on the improved and control sites (riffles). Other examples are Elmidae and Dytiscidae, both Coleoptera. Elmidae is generally associated with improved and control sites, whereas Dytiscidae is dominant in degraded sites.

Some species such as *Stylaria lacustris*, *Gammarus pulex*, *Caenis luctuosa* and *Asellus aquaticus* were the only representatives of their families, and different species of the same family were generally collected from the same site. For example *Hydropsyche angustipennis* and *H. pellucidula* of the family Hydropsychidae were both characteristic of control and improved sites and were rarely found in degraded sites. The same was true for *Elmis aenea* and *Oulimnius tuberculatus* which belong to the family Elmidae. Thus representing the data at the species level of taxonomy was not adding much more information to the family level, at this stage. Consequently the family level was chosen for representing the data and comparing the biology of the sites in the subsequent sections.

Table 3.5 Macroinvertebrate species abundance $\geq 1\%$ of the three habitats from Harper's Brook in summer 1993.

SPECIES	CONTROL	IMPROVED	DEGRADED
<i>Stylaria lacustris</i>	1.9	0.8	1.3
Tubificidae A *	0.2	2.4	6.3
Tubificidae B *	0.2	1.4	1.1
<i>Gammarus pulex</i>	0.9	1.2	0.3
<i>Asellus aquaticus</i>	0.0	0.5	1.9
<i>Caenis luctuosa</i>	3.4	1.7	6.5
<i>Baetis scambus</i>	3.7	1.7	0.1
<i>Baetis rhodani</i>	1.2	0.3	0.1
<i>Baetis vernus</i>	0.7	1.0	0.1
<i>Hydropsyche angustipennis</i>	2.4	9.0	0.1
<i>Hydropsyche pellucidula</i>	1.5	0.3	0.0
<i>Elmis aenea</i>	2.1	1.9	0.2
<i>Oulimnius tuberculatus</i>	2.1	1.0	0.3
<i>Tanytarsus</i> spp.	7.9	7.4	16.7
<i>Paratanytarsus</i> spp.	5.0	14.6	17.0
<i>Cladotanytarsus</i> spp.	0.1	1.2	0.2
<i>Microtendipes</i> spp.	0.0	7.3	12.2
<i>Stictochironomus</i> spp.	0.0	3.1	8.4
<i>Thienemannimyia</i> spp.	5.0	5.1	1.2
<i>Macropelopia</i> spp.	5.7	0.4	1.0
<i>Procladius</i> spp.	0.0	0.2	1.8
<i>Orthocladius</i> spp.	1.8	2.8	1.4
<i>Eukiefferiella</i> spp.	7.4	4.1	0.1
<i>Tvetenia</i> spp.	17.4	4.7	0.1
<i>Cricotopus</i> spp.	8.5	5.9	1.4
<i>Simulium</i> spp.	6.3	7.9	0.2
<i>Culicoides</i> spp.	2.4	0.1	0.3
Hydracarina	7.3	2.4	1.3
Copepoda	0.1	0.3	1.7
Cladocera	0.0	0.2	6.6
Total Abundance $\geq 1\%$	95.3	91.0	90.0

* Tubificidae A were those with hair chaetae, and B were those without hair chaetae.

Table 3.6 Macroinvertebrate family abundance $\geq 1\%$ of the three habitats from Harper's Brook in summer 1993.

FAMILY	CONTROL	IMPROVED	DEGRADED
Naididae	1.9	0.8	1.3
Tubificidae	0.7	5.9	9.3
Gammaridae	0.9	1.2	0.3
Asellidae	0.0	0.6	2.3
Caenidae	3.4	1.7	6.8
Baetidae	5.7	3.1	0.4
Hydropsychidae	3.8	9.3	0.1
Elmidae	4.2	2.9	0.5
Dytiscidae	0.0	0.3	1.2
Chironomidae	59.4	59.2	64.7
Simuliidae	6.3	7.9	0.2
Ceratopogonidae	2.4	0.1	0.3
Hydracarina	7.3	2.4	1.3
Micro-crustacea	0.1	0.5	8.3
Total Abundance $\geq 1\%$	96.3	95.9	97.0

Table 3.7 Percentage of the highest taxa abundance $\geq 1\%$ of the three habitats from Harper's Brook in summer 1993.

HIGHEST LEVEL	CONTROL	IMPROVED	DEGRADED
Oligochaeta	2.6	6.8	10.6
Hirudinea	0.6	1.3	0.2
Malacostraca	0.9	1.8	2.6
Ephemeroptera	9.6	5.1	7.3
Trichoptera	4.5	9.9	1.1
Odonata	0.0	0.1	0.2
Megaloptera	0.0	0.1	0.4
Coleoptera	4.3	3.4	1.9
Hemiptera	0.0	0.1	0.3
Chironomidae	59.4	59.2	64.7
Simuliidae	6.3	7.9	0.2
Other Diptera	3.6	0.9	0.4
Mollusca	0.4	0.4	0.6
Hydracarina	7.3	2.4	1.3
Other Taxa	0.4	0.5	8.3
Total Abundance	100.0	100.0	100.0

3.2.2 SUMMER

3.2.2.1 Description

A total number of 121,954 macroinvertebrates belonging to 93 species or higher taxonomic groups were collected from Harper's Brook in summer 1993. Table 3.8 shows the relative abundance of the taxa at family level on the seven sampling sites. Only the families forming $\geq 1.0\%$ of total abundance are considered here, those with abundance of $< 1\%$ are shown in Appendix one. Data from the other seasons are treated in the same way.

There were 18 families with $\geq 1\%$ which formed at least 97.3% of the total abundance, whereas 15 families with $< 1\%$ formed at most 2.7% of the total abundance. The most abundant family at all sites was Chironomidae, ranging from 75.4% on AR1 to 44.7% on R1.

Table 3.8 Macroinvertebrate family abundance $\geq 1\%$ of the sites from Harper's Brook in summer 1993.

FAMILY	NR	AR1	AR2	AR3	R1	R2	R3
Naididae	1.9	0.8	0.6	1.1	0.6	1.3	1.9
Tubificidae	0.7	8.6	5.2	3.8	21.0	3.9	3.1
Glossiphoniidae	0.0	2.1	0.1	0.1	0.1	0.1	0.1
Gammaridae	0.9	0.3	1.6	1.7	0.1	0.2	0.5
Asellidae	0.0	0.5	0.4	0.8	4.1	0.8	1.9
Caenidae	3.4	2.1	1.9	1.0	8.0	7.6	5.0
Baetidae	5.7	0.7	6.1	2.6	0.1	0.7	0.5
Hydropsychidae	3.8	0.9	11.5	15.4	0.0	0.4	0.0
Hydroptilidae	0.6	0.1	0.4	1.2	0.0	0.2	0.1
Sialidae	0.0	0.4	0.0	0.0	1.1	0.1	0.1
Elmidae	4.2	0.7	3.5	4.6	0.2	0.7	0.5
Dytiscidae	0.0	0.9	0.0	0.0	0.9	0.6	2.1
Chironomidae	59.4	75.4	49.9	52.4	44.7	74.4	74.9
Simuliidae	6.3	0.3	13.1	10.3	0.1	0.4	0.0
Ceratopogonidae	2.4	0.1	0.0	0.2	0.2	0.3	0.3
Empididae	0.9	0.0	1.0	0.2	0.0	0.0	0.0
Hydracarina	7.3	2.9	1.9	2.5	0.3	2.2	1.4
Microcrustacea	0.1	1.0	0.4	0.0	16.0	3.8	5.1
Total Abundance $\geq 1\%$	97.9	97.8	97.6	98.0	97.3	97.8	97.6

Chironomidae (59.4%) was the dominant* taxon of the control site. Hydracarina (7.3%), Simuliidae (6.3%) and Baetidae (5.7%) formed the abundant taxa of the site. Elmidae, Hydropsychidae, Caenidae, Ceratopogonidae and Naididae formed the frequent taxa (4.2% – 1.9%).

The improved sites on average were characterised by Chironomidae (59.2%) as being the dominant taxon with Hydropsychidae (9.3%), Simuliidae (7.9%) and Tubificidae (5.9%) the abundant families. Although all other taxa were present, some like Baetidae, Elmidae, Hydracarina, Caenidae and Gammaridae were frequent (3.1% – 1.2%), and the others were occasional (0.8% – 0.1%) (Table 3.6). AR1 was markedly different from the other artificial riffles in having the highest Chironomidae percentage (75.4%) with Tubificidae (8.6%) as the second most abundant taxa. Glossiphoniidae (2.1%) was frequent at the site. Simuliidae and Baetidae were occasional in AR1, whereas they were more abundant (> 10% and > 2.5%) in the other artificial riffle sites (Table 3.8).

The degraded sites on average were characterised by Chironomidae (64.7%) as being the dominant family, and Tubificidae (9.3%), microcrustacea (Copepoda and Cladocera) (8.3%) and Caenidae (6.8%) were abundant. Although all other taxa were present, some like Asellidae, Naididae, Hydracarina and Dytiscidae were frequent (2.3% – 1.2%), and the other families were occasional (Table 3.6). R1 was markedly different from the other degraded sites in having the lowest Chironomidae percentage (44.7%) and highest Tubificidae percentage (21.0%). Microcrustacea were also abundant (16.0%) compared with 5.1% and 3.8% at the other degraded sites. Sialidae (1.1%) was also more abundant in R1 than in the other sites (0.1%) (Table 3.8).

* Terms used in the text regarding the abundance percentages

Dominant	> 17%
Abundant	5 – 17%
Frequent	1 – 5%
Occasional	< 1%
Rare	< 0.1

3.2.2.2 Comparison between habitats

The most important differences between the control site and the improved sites were; the greater abundance of Hydropsychidae and Tubificidae in improved sites (9.3%, 5.9%) compared with the control site (3.8%, 0.7%), and the greater abundance of Baetidae and Elmidae in the control site (5.7%, 4.2%) than the improved sites (3.1%, 2.9%). Hydracarina were also more abundant in the control site (7.3%) than improved sites (2.4%). The differences between improved and degraded sites were more distinct. Apart from Chironomidae which was more abundant in degraded sites (64.7%) than improved sites (59.2%), Hydropsychidae, Simuliidae, Tubificidae, Baetidae, Elmidae and Hydracarina formed the abundant taxa of improved sites, whereas Tubificidae Microcrustacea, Caenidae and Asellidae formed the abundant taxa of degraded sites. The former group (except Tubificidae) are typical taxa of faster-flowing riffle habitats, and the latter are more typical of slower-flowing run or pool habitats. The three habitat types also differed in Chironomidae subfamily abundance (Table 3.9). Chironominae was more abundant in degraded sites (55.4%) than in improved (34.4%) and control (13.3%) sites and, conversely, Orthocladinae was more abundant in the control site (35.2%) than in improved (17.6%) and degraded sites (3.0%). The same was true for Tanypodinae which was more abundant in the control site (10.7%) than in the improved (5.8%) and degraded sites (4.9%).

Table 3.9 Percentage of the subfamily abundance of Chironomidae at three habitat types from Harper's Brook in summer 1993.

SUBFAMILY	CONTROL	IMPROVED	DEGRADED
<i>Chironominae</i>	13.3	34.9	55.4
<i>Tanypodinae</i>	10.7	5.8	4.9
<i>Orthocladinae</i>	35.2	17.6	3

3.2.3 AUTUMN

3.2.3.1 Description

A total number of 34,836 macroinvertebrates belonging to 70 species or higher taxonomic group were collected from Harper's Brook in autumn 1993. Table 3.10 shows the relative abundance of the taxa at family level at all seven sampling sites. Ten families with $\geq 1.0\%$ abundance formed at least 96.4% of the total abundance, and 23 families with $< 1\%$ abundance formed at most 3.6% of the total abundance. The most abundant taxon at all sites (except at R1) was Caenidae, ranging from 67.8% on AR1 to 36.0% on R2. The relative abundance of taxa found at the control, improved and degraded sites is shown in table 3.11.

Table 3.10 Macroinvertebrate family abundance $\geq 1\%$ of the sites from Harper's Brook in autumn 1993.

FAMILY	NR	AR1	AR2	AR3	R1	R2	R3
Tubificidae	3.4	13.7	4.3	5.2	64.1	28.0	24.8
Lumbriculidae	2.1	0.0	1.7	0.8	0.0	0.2	0.0
Gammaridae	2.0	1.2	3.1	5.6	0.3	3.7	2.5
Asellidae	0.6	2.3	1.6	5.5	0.6	1.8	3.4
Caenidae	45.3	67.8	52.7	37.1	20.2	36.0	39.4
Hydropsychidae	8.1	0.8	8.5	13.3	0.0	0.5	0.7
Elmidae	16.4	1.1	13.2	12.7	0.0	2.6	1.8
Chironomidae	3.3	8.9	2.4	9.2	12.9	20.9	17.1
Simuliidae	16.6	1.7	8.6	7.2	0.2	3.6	8.6
Hydracarina	0.0	1.2	0.3	0.3	0.0	0.0	0.0
Total Abundance $\geq 1\%$	98.0	98.7	96.4	96.8	98.3	97.2	98.4

Caenidae (45.3%) was the dominant family of the control site whilst Simuliidae (16.6%), Elmidae (16.4%) and Hydropsychidae (8.1%) formed the abundant taxa. Tubificidae, Chironomidae, Lumbriculidae and Gammaridae were frequent (3.4% – 2.0%), Hydracarina was absent and the other taxa were occasional.

The improved sites on average were characterised by Caenidae (52.5%) as the dominant taxon, and Tubificidae (7.7%), Hydropsychidae (7.6%), Chironomidae (6.8%)

and Simuliidae (5.8%) as abundant taxa. Gammaridae and Asellidae formed the frequent taxa (3.3%, 3.1%), and the other taxa were < 1% abundant. AR1 was markedly different from the other improved sites in having the highest percentage of Caenidae (67.8%), and Tubificidae (13.7%) which formed the second most abundant taxon at the site. Hydracarina was also more abundant in AR1 (1.2%) than in the other improved sites (0.3%). There was a great difference between the abundance of Hydropsychidae and Elmidae in AR1 (0.8%, 1.1%) and the other improved sites (AR2 8.5% – 13.2%, AR3 13.3% – 12.7%) (Table 3.10).

Table 3.11 Macroinvertebrate family abundance \geq 1% of the three habitats from Harper's Brook in autumn 1993.

FAMILY	CONTROL	IMPROVED	DEGRADED
Tubificidae	3.4	7.7	39.0
Lumbriculidae	2.1	0.8	0.1
Gammaridae	2.0	3.3	2.2
Asellidae	0.6	3.1	1.9
Caenidae	45.3	52.5	31.8
Hydropsychidae	8.1	7.6	0.4
Elmidae	16.4	9.0	1.5
Chironomidae	3.3	6.8	17.0
Simuliidae	16.6	5.8	4.1
Hydracarina	0.0	0.6	0.0
Total Abundance > 1%	98.0	97.3	98.0

The degraded sites on average were characterised by Tubificidae (39.0%), Caenidae (31.8%) and Chironomidae (17.0%) as the dominant taxa. Simuliidae, Gammaridae, Asellidae and Elmidae were the frequent taxa of these sites. Hydracarina was absent, and the other taxa were occasional (Table 3.10). R1 was markedly different from the other degraded sites in having the greatest abundance of Tubificidae (64.1%). Hydropsychidae and Elmidae were absent from R1, whereas the former was occasional and the latter frequent on the other degraded sites. Simuliidae was abundant in the other degraded sites, whereas it was occasional on R1 (Table 3.10).

3.2.3.2 Comparison between habitats

The most important differences between the control and improved sites were; the greater abundance of Elmidae (7.4%) and Simuliidae (10.8%) in the control site, and the greater abundance of Tubificidae (4.3%), Caenidae (7.2%) and Chironomidae (3.5%) in the improved sites. The most distinctive difference between the improved and degraded sites was the greatest percentage of Tubificidae (39.0%) in degraded sites, whereas it was 7.7% abundant in the improved sites. The abundance of Chironomidae was also 10.2% greater in degraded sites than in the improved sites. Hydropsychidae and Elmidae were much more abundant in improved sites (7.5%, 9.0%) than in degraded sites (0.4%, 1.5%).

3.2.4 WINTER

3.2.4.1 Description

A total number of 15,669 macroinvertebrates belonging to 59 species or higher taxa were collected from Harper's Brook in winter 1993/1994. Table 3.12 shows the relative abundance of the taxa at family level on seven sampling sites. There were 16 families with $\geq 1\%$ which formed at least 96.8% of the total abundance, whereas a further 13 families with an abundance of $< 1\%$ formed at most 3.2% of the total abundance.

Chironomidae (27.2%) and Caenidae (21.1%) were the dominant taxa of the control site. Elmidae (11.9%), Simuliidae (9.9%), Baetidae (8.6%), Hydracarina (6.1%) and Hydropsychidae (5.6%) were the abundant taxa. Ceratopogonidae (4.1%), Gammaridae (2.2%) and Tubificidae (1.4%) were frequent while Tipulidae and Hydroptilidae were absent from the site and other families were occasional (Table 3.12).

The improved sites on average were characterised by Caenidae (32.8%) and Chironomidae (21.5%) as the dominant taxa. Simuliidae (8.4%), Baetidae (8.4%), Tubificidae (7.6%) and Elmidae (5.5%) were the abundant taxa. Hydropsychidae, Gammaridae, Lumbriculidae and Hydracarina were frequent and the other taxa were occasional (Table 3.13). Although AR1 contained the greatest abundance of Caenidae (59%) when compared with the other improved sites, AR2 was also different to the others

in having the greatest abundance of Baetidae (22.0) and the least abundance of Caenidae (12.5%).

Table 3.12 Macroinvertebrate family abundance $\geq 1\%$ of the sites from Harper's Brook in winter 1993/1994.

FAMILY	NR	AR1	AR2	AR3	R1	R2	R3
Tubificidae	1.4	6.3	13.3	3.1	56.3	58.6	24.0
Lumbriculidae	0.4	0.0	4.6	1.1	0.0	0.0	0.0
Erpobdellidae	0.3	0.1	2.3	0.3	0.2	0.2	0.0
Gammaridae	2.2	2.2	2.7	3.0	1.2	1.4	2.9
Asellidae	0.1	1.2	0.2	0.3	1.6	0.6	0.3
Caenidae	21.1	59.5	12.5	26.5	21.9	22.7	36.3
Baetidae	8.6	0.1	22.0	3.0	0.0	0.5	0.0
Hydropsychidae	5.6	0.7	7.1	5.3	0.2	0.4	0.0
Hydroptilidae	0.0	0.1	1.3	0.0	0.0	0.0	0.0
Elmidae	11.9	1.9	7.0	7.6	0.2	0.3	0.9
Chironomidae	27.2	20.0	10.6	33.9	15.6	11.3	25.1
Simuliidae	9.9	0.4	12.5	12.3	0.2	0.6	0.3
Tipulidae	0.0	0.1	0.2	0.0	0.2	0.7	1.5
Ceratopogonidae	4.1	0.2	1.1	0.9	0.0	1.0	0.6
Empididae	0.3	1.1	0.7	0.8	0.3	0.7	1.2
Hydracarina	6.1	4.6	0.0	1.2	0.8	0.3	3.8
Total Abundance $\geq 1\%$	99.1	98.4	98.3	99.4	98.7	99.4	96.8

The degraded sites on average were characterised by Tubificidae (46.3%), Caenidae (27.0%) and Chironomidae (17.4%) as the dominant taxa while Gammaridae and Hydracarina were frequent, and all other taxa were present at $< 1\%$ abundance. There was no distinct difference between R1 and the other degraded sites in winter.

3.2.4.2 Comparison between habitats

There were no marked differences between the control site and the improved sites in winter, except for the greater abundance of Tubificidae (6.2%) and Caenidae (11.7%) on the latter sites. Elmidae (6.4%) and Hydracarina (4.2%) were also present in greater abundance in the control site. The most obvious difference between the degraded and improved sites was the very high abundance of Tubificidae (46.3%) in the degraded sites. Baetidae, Hydropsychidae, Elmidae and Simuliidae were much more abundant in the

improved than in the degraded sites. Other taxa, except Asellidae and Tipulidae were also more abundant in the improved sites (Table 3.13).

Table 3.13 Macroinvertebrate family abundance $\geq 1\%$ of the three habitats from Harper's Brook in winter 1993/1994.

FAMILY	CONTROL	IMPROVED	DEGRADED
Tubificidae	1.4	7.6	46.3
Lumbriculidae	0.4	1.9	0.0
Erpobdellidae	0.3	0.9	0.2
Gammaridae	2.2	2.6	1.8
Asellidae	0.1	0.6	0.8
Caenidae	21.1	32.8	27.0
Baetidae	8.6	8.4	0.2
Hydropsychidae	5.6	4.4	0.2
Hydroptilidae	0.0	0.5	0.0
Elmidae	11.9	5.5	0.5
Chironomidae	27.2	21.5	17.4
Simuliidae	9.9	8.4	0.3
Tipulidae	0.0	0.1	0.8
Ceratopogonidae	4.1	0.7	0.5
Empididae	0.3	0.8	0.7
Hydracarina	6.1	1.9	1.6
Total Abundance $\geq 1\%$	99.1	98.7	98.3

3.2.5 SPRING

3.2.5.1 Description

A total number of 38,409 macroinvertebrates belonging to 70 species or higher taxa were collected in spring 1994. The relative abundance of the taxa from seven sampling sites at family level are shown in table 3.14. Seventeen families with $\geq 1\%$ abundance formed at least 98.4% of the total abundance, and 13 families with $< 1\%$ abundance formed at most 1.6% of the total abundance. The relative abundance of taxa found at control, improved and degraded sites is shown in table 3.15.

Baetidae (40.1%) and Chironomidae (21.1%) were the dominant taxa of the control site. The abundant taxa of the site were; Elmidae (8.7%), Caenidae (8.2%), Hydroptilidae (5.7%) and Hydracarina (5.0%). Dytiscidae and Corixidae were absent, but other families were frequent or occasional (Table 3.14).

The improved sites on average were characterised by Caenidae (27.6%) as dominant and Baetidae (16.2%), Chironomidae (13.1%), Tubificidae (9.4%), Elmidae (8.4%) and Gammaridae (5.1%) as abundant. Other taxa were frequent or occasional. AR1 was markedly different from the other improved sites in having the greatest abundance of Caenidae (52.9%). Tubificidae was also present in much greater abundance (approx. 9%) in AR1 than in the others. Corixidae was abundant (3.5%) in AR1, whereas it was absent from the other improved sites. Lumbriculidae was absent in AR1, but was frequent or occasional in the others (Table 3.14).

Table 3.14 Macroinvertebrate family abundance \geq 1% of the sites from Harper's Brook in spring 1994.

FAMILY	NR	AR1	AR2	AR3	R1	R2	R3
Naididae	1.2	1.1	3.3	2.7	2.7	9.8	8.6
Tubificidae	1.3	15.9	4.9	7.4	37.4	24.9	4.1
Lumbriculidae	0.2	0.0	2.8	0.3	0.0	0.4	0.1
Gammaridae	1.0	2.2	7.2	5.8	0.5	0.3	1.0
Caenidae	8.2	52.9	12.7	17.2	43.3	46.4	67.6
Baetidae	40.1	0.4	30.8	17.3	0.8	0.4	0.0
Ephemereidae	1.2	0.9	3.9	1.4	0.0	0.4	0.1
Hydropsychidae	1.1	0.1	1.6	3.7	0.0	0.2	0.0
Hydroptilidae	5.7	1.4	1.5	1.8	0.5	2.1	1.8
Elmidae	8.7	2.6	6.6	16.1	0.1	0.7	0.7
Dytiscidae	0.0	0.6	0.1	0.1	5.5	0.1	0.6
Corixidae	0.0	3.5	0.0	0.0	0.0	0.4	0.6
Chironomidae	21.2	8.9	16.3	14.2	5.6	6.6	4.3
Simuliidae	2.6	0.1	1.6	1.7	0.0	0.0	0.0
Ceratopogonidae	2.0	1.1	3.1	7.1	0.1	0.7	0.8
Sphaeriidae	0.0	1.2	0.0	0.3	1.4	0.0	0.0
Hydracarina	5.0	5.7	2.7	1.8	0.6	6.1	8.6
Total Abundance \geq 1%	99.4	98.7	99.1	99.1	98.4	99.5	98.8

The degraded sites on average were characterised by Caenidae (52.4%) and Tubificidae (22.1%) as the dominant taxa of the sites. Naididae (7.0%), Chironomidae (5.5%) and Hydracarina (5.1%) were the abundant taxa while Simuliidae was absent and the other taxa were frequent or occasional (Table 3.15). R1 was different from the other degraded sites in not having Lumbriculidae, Ephemereidae and Corixidae, whereas these taxa were frequent or occasional on the other degraded sites. Sphaeriidae was frequent in

R1 (1.4%), whereas it was absent from the other degraded sites. Hydracarina was abundant in the other degraded sites, but was occasional in R1 (Table 3.14).

3.2.5.2 Comparison between habitats

The control and improved sites were markedly different in having the greatest abundance of Baetidae (40.1%) in the control site. Chironomidae, Simuliidae, Hydracarina and Hydroptilidae were also more abundant in the control site, whereas the other taxa were present in greater abundance in the improved sites (Table 3.15). The most obvious difference between degraded and improved sites was the greatest abundance of Caenidae (52.4%) in degraded sites. The abundance of Tubificidae, Naididae and Dytiscidae was also much greater in degraded sites than in improved sites. Gammaridae, Baetidae, Ephemerellidae, Hydropsychidae and Elmidae were much more abundant in the improved than in the degraded sites in spring (Table 3.15).

Table 3.15 Macroinvertebrate family abundance $\geq 1\%$ of the three habitats from Harper's Brook in spring 1993.

FAMILY	CONTROL	IMPROVED	DEGRADED
Naididae	1.2	2.4	7.0
Tubificidae	1.3	9.4	22.1
Lumbriculidae	0.2	1.0	0.1
Gammaridae	1.0	5.1	0.6
Caenidae	8.2	27.6	52.4
Baetidae	40.1	16.2	0.4
Ephemerellidae	1.2	2.1	0.2
Hydropsychidae	1.1	1.8	0.1
Hydroptilidae	5.7	1.6	1.5
Elmidae	8.7	8.4	0.5
Dytiscidae	0.0	0.3	2.1
Corixidae	0.0	1.2	0.4
Chironomidae	21.2	13.1	5.5
Simuliidae	2.6	1.1	0.0
Ceratopogonidae	2.0	3.7	0.5
Sphaeriidae	0.0	0.5	0.5
Hydracarina	5.0	3.4	5.1
Total Abundance $\geq 1\%$	99.4	99.0	98.9

3.2.6 COMPARISON BETWEEN SEASONS

The most obvious difference between summer and autumn macroinvertebrate composition and abundance was the greater abundance of taxa in summer than autumn (approx. fourfold). Although the number of families was equal (33), 23 fewer species were recorded in autumn than summer. The dominant taxon in summer was Chironomidae, whereas the dominant taxon in autumn was Caenidae. Naididae, Glossiphoniidae, Baetidae, Hydroptilidae, Sialidae, Dytiscidae, Ceratopogonidae, Empididae and microcrustacea were absent or rare in autumn, whereas they were present ($\geq 1\%$) in summer. Lumbriculidae was the only family which was present in autumn, but was absent or rare in summer. Tubificidae was also more abundant in autumn than summer, especially at degraded sites (39.0% in autumn, 9.3% in summer).

The macroinvertebrate abundance (approx. 50%), and the species (11) and family richness (4) were markedly decreased in winter in comparison with autumn. Baetidae was present in greater abundance (22.0%–3.0%) in three sites (NR, AR2 and AR3) in winter, whereas it was absent or occasional in autumn. Erpobdellidae, Hydroptilidae, Tipulidae, Ceratopogonidae and Empididae were also present in $> 1\%$ abundance in some of the sites in winter, whereas they were absent or rare in autumn.

The major differences between spring and winter macroinvertebrate composition and abundance were as follows: the total abundance of taxa increased more than twofold in spring compared with winter, although the number of families increased by only one, the species richness rose by 11 species in spring. Naididae, Ephemerellidae, Dytiscidae, Corixidae and Sphaeriidae were present in spring, whereas they were absent or rare in winter. The abundance of Baetidae also increased markedly in spring (Table 3.14).

The results also showed that the degraded habitats were more consistent in macroinvertebrate composition through the seasons than the other habitats. Tubificidae, Caenidae and Chironomidae formed the most abundant taxa of the degraded habitats in three of the seasons, although in summer microcrustacea replaced Chironomidae in the third place and Caenidae dropped to fourth place. Simuliidae, Hydropsychidae, Elmidae and Baetidae, all of which are characteristic of riffles, were among the abundant taxa of the

control and improved habitats in all seasons. Caenidae was the dominant taxon of the improved habitat in most seasons except for summer when Chironomidae was dominant. The dominant taxon of the control site varied between seasons.

3.2.7 BIODIVERSITY DIFFERENCES BETWEEN HABITATS AND SEASONS

In general there was a tendency towards decreasing species and family richness in the control and improved sites from summer to winter (Table 3.16). This was possibly caused by the decrease in marginal and in-stream vegetation together with some taxa overwintering in a quiescent state. The degraded sites did not follow this pattern; their richness decreased dramatically from summer to autumn and increased slightly in winter and spring. A possible explanation for this could be that weed cutting occurs in late summer and is mostly on the deeper straighter sections.

Table 3.16 Species and family richness of the three habitats from Harper's Brook in different seasons.

HABITAT	SUMMER		AUTUMN		WINTER		SPRING	
	Species	Family	Species	Family	Species	Family	Species	Family
CONTROL	57	30	41	24	37	22	39	23
IMPROVED	59	31	42	24	41	22	42	24
DEGRADED	60	30	32	17	35	18	35	19

3.3 SPECIES DIVERSITY

In order to determine whether any significant differences in species diversity existed between the sites from Harper's Brook, the Shannon-Wiener diversity index (H') was calculated for each sample in each season and the null hypothesis (that there was no difference between mean diversity of the sites) was tested, using the one-way ANOVA and F test of Microsoft "Excel" (version 4.0). Then the 95% confidence intervals were calculated to show which means differed significantly from each other. The ANOVA output for each season is shown in Appendix One.

3.3.1 DIFFERENCES IN SPECIES DIVERSITY OF EACH SEASON

Table 3.17 shows the calculated values of H' of each replicate and the mean diversities for each site in summer. ANOVA shows that the mean diversities of the sites are significantly different ($F = 3.29$, d.f. = 6,14, $P < 0.05$).

Table 3.17 Shannon-Wiener diversity index of 21 samples and the mean diversity of the sites from Harper's Brook in summer 1993.

Sites	H' of Replicates		Mean H'	
NR	2.76	2.91	3.03	2.90
AR1	2.48	2.64	2.50	2.56
AR2	2.86	2.79	2.91	2.85
AR3	2.91	2.82	2.57	2.77
R1	2.30	2.66	2.40	2.45
R2	3.09	2.42	2.61	2.70
R3	2.44	2.45	2.50	2.46

Table 3.18 shows the H' values of the replicates and the mean diversities of the sites in autumn. The mean diversities of the sites are also significantly different ($F = 4.50$, d.f. 6,14, $P < 0.01$).

Table 3.18 Shannon-Wiener diversity index of 21 samples and the mean diversity of the sites from Harper's Brook in autumn 1993.

Site	H' of Replicates			Mean H'
NR	1.84	2.08	1.58	1.84
AR1	1.40	1.60	1.26	1.42
AR2	2.27	2.04	1.30	1.87
AR3	2.39	2.09	2.58	2.35
R1	1.76	1.56	1.60	1.64
R2	2.24	2.15	2.07	2.15
R3	2.00	2.52	2.06	2.19

The calculated diversity values of the replicates and the mean diversities of each site in winter are shown in Table 3. 19. The mean diversities of the sites are significantly different ($F = 9.80$, d.f. 6,14 $P < 0.001$).

Table 3.19 Shannon-Wiener diversity index of 21 samples and the mean diversity of the sites from Harper's Brook in winter 1993/94.

Site	H' of Replicates			Mean H'
NR	2.54	2.40	2.37	2.43
AR1	1.74	1.86	1.54	1.71
AR2	2.80	2.70	2.37	2.62
AR3	2.51	2.42	2.37	2.43
R1	2.13	2.16	1.71	2.00
R2	2.32	2.18	2.39	2.30
R3	1.96	2.36	2.20	2.17

Table 3.20 shows the calculated diversity values of the replicates and the mean diversity values of each site in spring. The mean diversities of the sites are significantly different ($F = 5.79$, d.f. 6,14, $P < 0.003$).

Table 3.20 Shannon-Wiener diversity index of 21 samples and the mean diversity of the sites from Harper's Brook in spring 1994.

Site	H' of Replicates			Mean H'
NR	1.86	2.07	2.40	2.11
AR1	1.87	1.77	2.31	1.98
AR2	2.31	2.47	2.76	2.51
AR3	2.75	2.50	2.70	2.65
R1	1.34	2.35	2.18	1.95
R2	2.18	1.96	1.63	1.92
R3	1.52	1.46	1.01	1.33

3.3.2 DIFFERENCES IN SPECIES DIVERSITY BETWEEN HABITATS

Although the sites of Harper's Brook are significantly different in mean diversity, there is significant variation within the group. One way of finding out if all possible combinations of pairs are different from each other or just one is different, is by displaying the 95% confidence intervals (Fowler, 1994). The sites whose intervals do not overlap are presumed to be reasonably different. The 95% confidence intervals of the mean diversity of the sites were calculated using Excel (version 4.0) ($\alpha = 0.025$, for small sample size, $n = 3$) for each season separately. Figures 3.1– 4 show the 95% confidence intervals of the mean diversity of the sites in the four seasons.

The summer results indicate that AR1 was significantly different from NR and AR2, and R3 was different from NR, AR2 and AR3. NR had the greatest mean diversity followed by AR2 and AR3. R3 had the least mean diversity, similar to R1 and close to AR1 (Fig. 3.1). The error bars of R2 are the largest and those of R3 the smallest, i.e. the variation between replicates of R2 is the greatest and between those of the R3 is the smallest.

Figure 3.2 shows the results of 95% confidence intervals in autumn. The results indicate that AR1 and R1 were significantly different from AR3, R2 and R3. AR1 and R1 also had the least mean diversity making them similar. However, AR3 had the greatest

mean diversity with medium size error bars. The largest error bars i.e. the greatest variation between samples, belonged to AR2 and the smallest belonged to R2 and R3 respectively.

In winter AR1 was significantly different from NR, AR2, AR3 and R2, and had the least mean diversity which was relatively close to R1 and R3 (Fig. 3.3). AR2 had the greatest mean diversity followed by NR and AR3 which were different from AR1 and R1 representing two relatively distinct groups. The largest error bars were found for AR2 and R1, and AR3 and NR had the smallest one.

Figure 3.4 shows the results of 95% confidence intervals in spring. The results indicate that AR3 had the greatest mean diversity and was significantly different from R2, R3 and AR1. AR2, which had the second greatest mean diversity, was also different from R3 in having least mean diversity. The variability between AR3 samples was the least (smallest error bars) and that of R1 samples was the greatest (largest error bars).

3.3.3 SUMMARY OF DIVERSITY DIFFERENCES

The overall results of ANOVA showed that the mean diversity of macroinvertebrate communities of the seven sites from Harper's Brook was significantly different in all four seasons. The results of 95% confidence intervals agreed with the results of ANOVA, although not all sites were significantly different in any season. In general the mean diversities of the sites NR, AR2 and AR3 were highest, whilst those of sites AR1 and the Run sites were lowest in summer, winter and spring. AR3 had the highest mean diversity in autumn, followed by R3 and R2. AR2 was in the middle range followed by NR and R1 and AR1 had the smallest mean diversity. In winter and spring the differences between riffles and runs were more accentuated than during summer and autumn. This is possibly due to the more consistent physical environment provided by the coarse substrate of riffles than by the sand/silt and macrophytes of runs. Macrophyte dieback in runs in winter reduces physical heterogeneity.

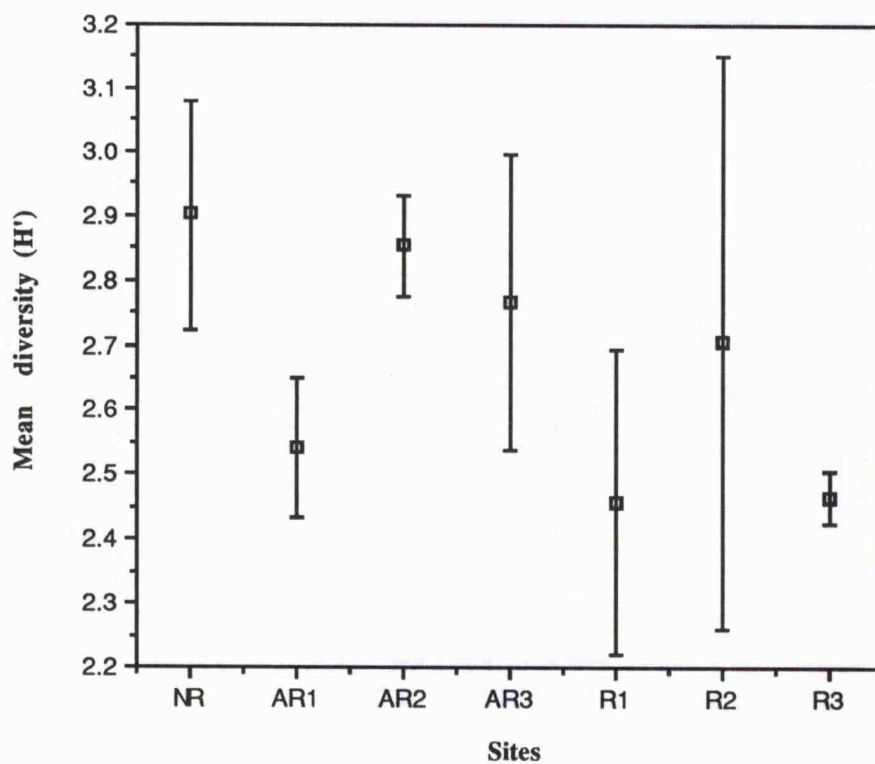


Figure 3.1 Mean diversity (H') and 95% confidence intervals of the sites from Harper's Brook in summer 1993.

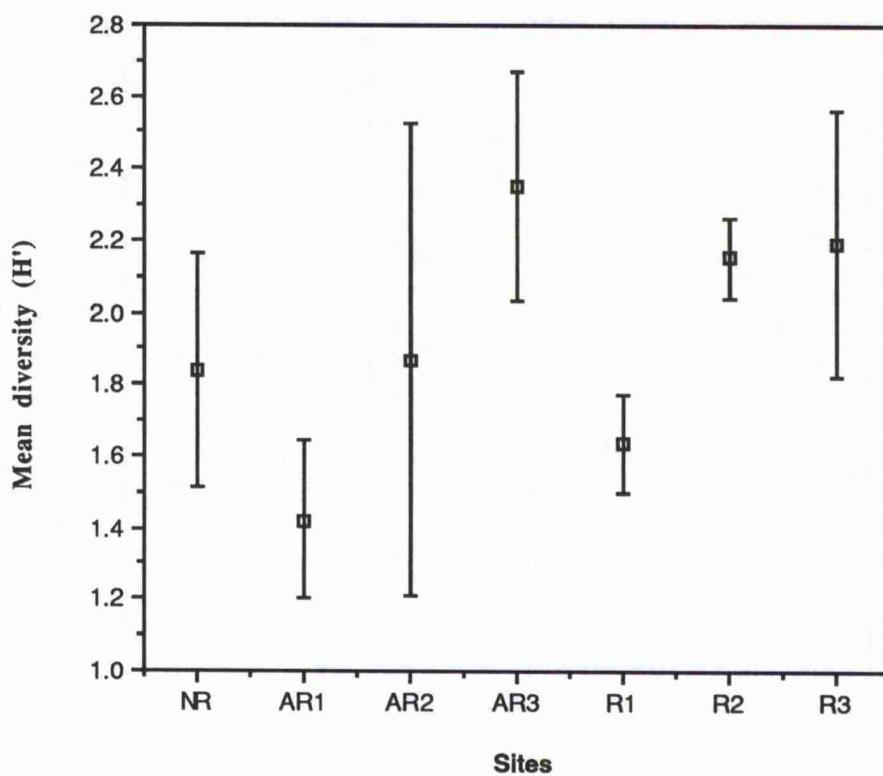


Figure 3.2 Mean diversity (H') and 95% confidence intervals of the sites from Harper's Brook in autumn 1993.

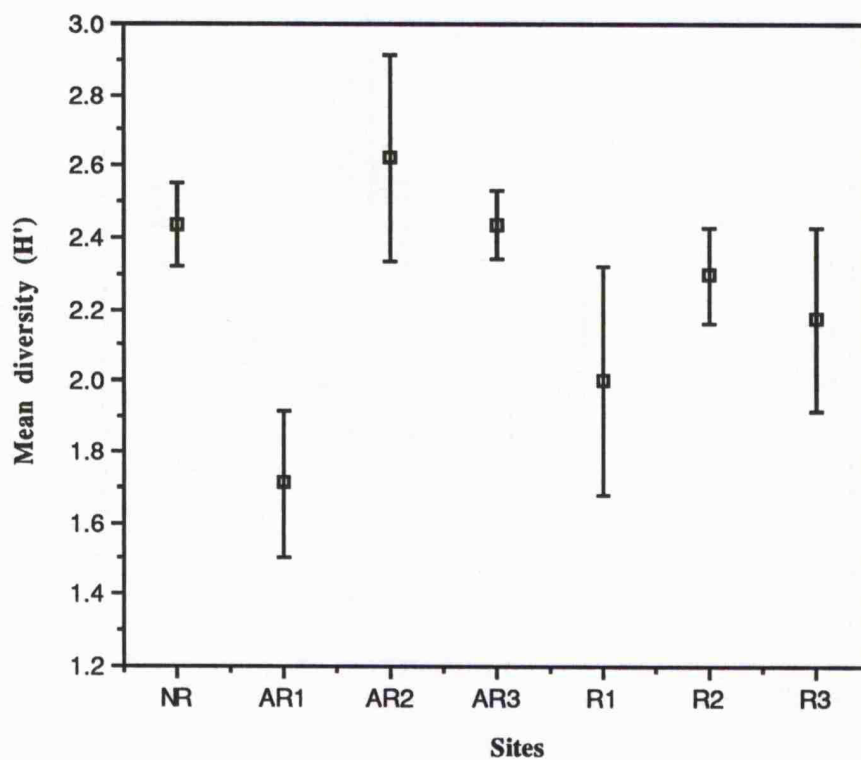


Figure 3.3 Mean diversity (H') and 95% confidence intervals of the sites from Harper's Brook in winter 1993/1994.

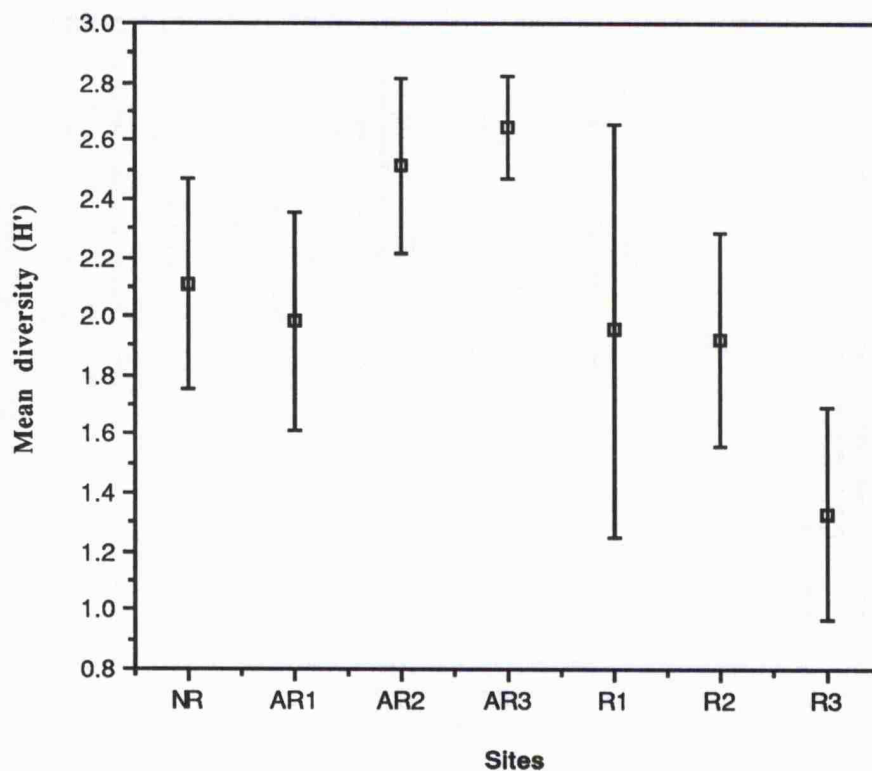


Figure 3.4 Mean diversity (H') and 95% confidence intervals of the sites from Harper's Brook in spring 1994.

3.3.4 DIVERSITY AND ENVIRONMENTAL VARIABLES

In order to discern any relationship between the diversity of the invertebrate communities and the depth and velocity of the related samples and sites, diversity values as dependent variables were plotted against depth and velocity values as independent variables on two separate graphs for each season using Cricket graph (version 1.3.2). Two graphs were used for each pairing of variables in each season, one using the 21 replicate samples and the other using the seven sites with mean diversity values. A logarithmic regression gave stronger correlation between depth and diversity in all seasons, whilst a linear relationship was strongest between velocity and diversity.

3.3.4.1 SUMMER

Figures 3.5 & 3.6 show the correlation between depth and diversity of the sites and the replicate samples. Both graphs show a negative logarithmic relationship, which means sites with greater diversity values are shallower. Figure 3.5 shows a strong correlation between the depth and diversity ($r = -0.89$) in summer, i.e. 89% of the variation in diversity is accounted for by the variation in the depth. Figure 3.6 shows a similar trend which is less powerful ($r = -0.62$) because of the variability between the samples. R values of the sites and the samples both are significant at $P < 0.01$. The result also show NR, AR2 and AR3 are placed closely together around the top of the fitted line. AR1 and R3 also are close to each other (Figure 3.5).

Figures 3.7 & 3.8 show the correlation between velocity and diversity of the sites and the samples respectively. Both graphs show a positive linear correlation between velocity and diversity, i. e. sites with greater diversity values are faster flowing sites and those with lesser diversity values are the more sluggish sites. The correlation between the velocity and diversity of the sites is strong ($r = 0.89$) and those of the samples is clearly weaker ($r = 0.6$), but both are significant at $P < 0.01$. Again NR, AR2 and AR3 are similar to each other occupying the upper end of the fitted line. AR1, R1 and R3 also are close and are positioned lower on the line (Figure 3.7).

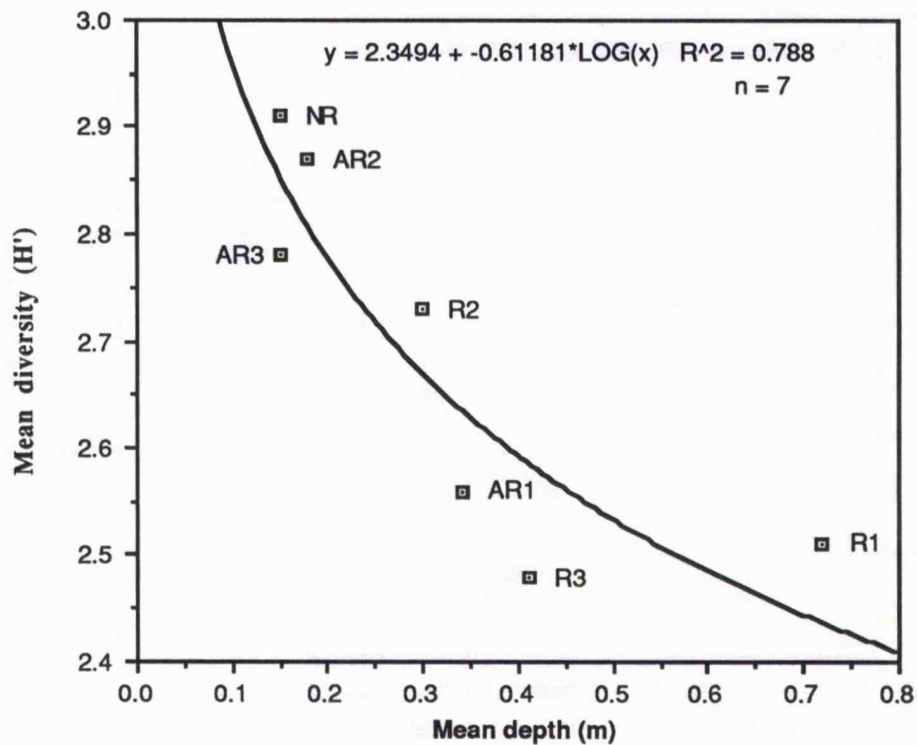


Figure 3.5 Correlation between depth and diversity of the sites from Harper's Brook in summer 1993.

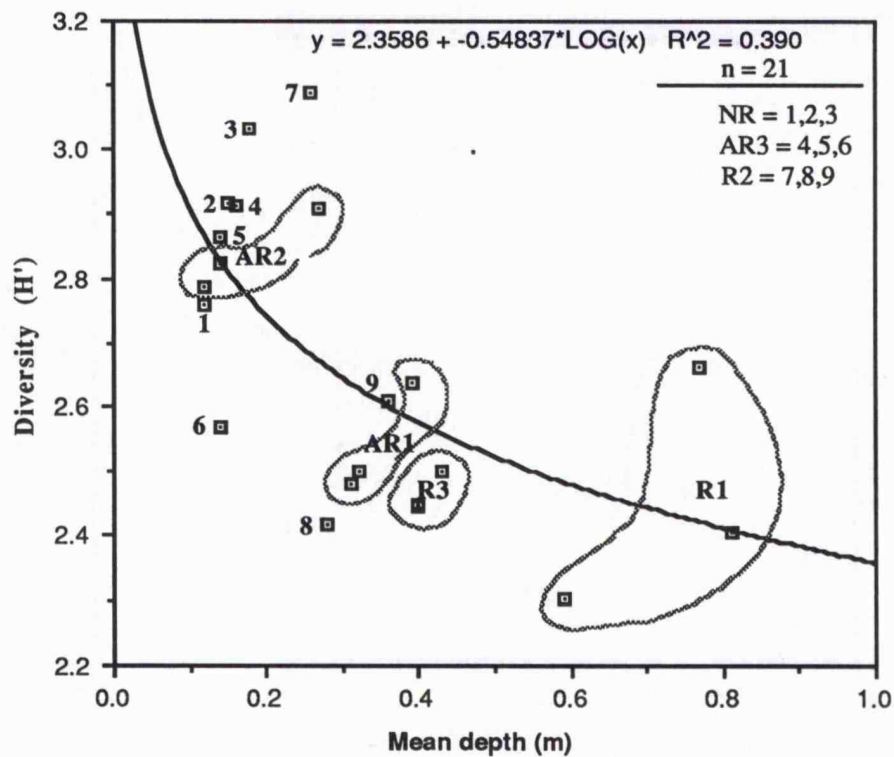


Figure 3.6 Correlation between depth and diversity of the 21 samples from Harper's Brook in summer 1993.

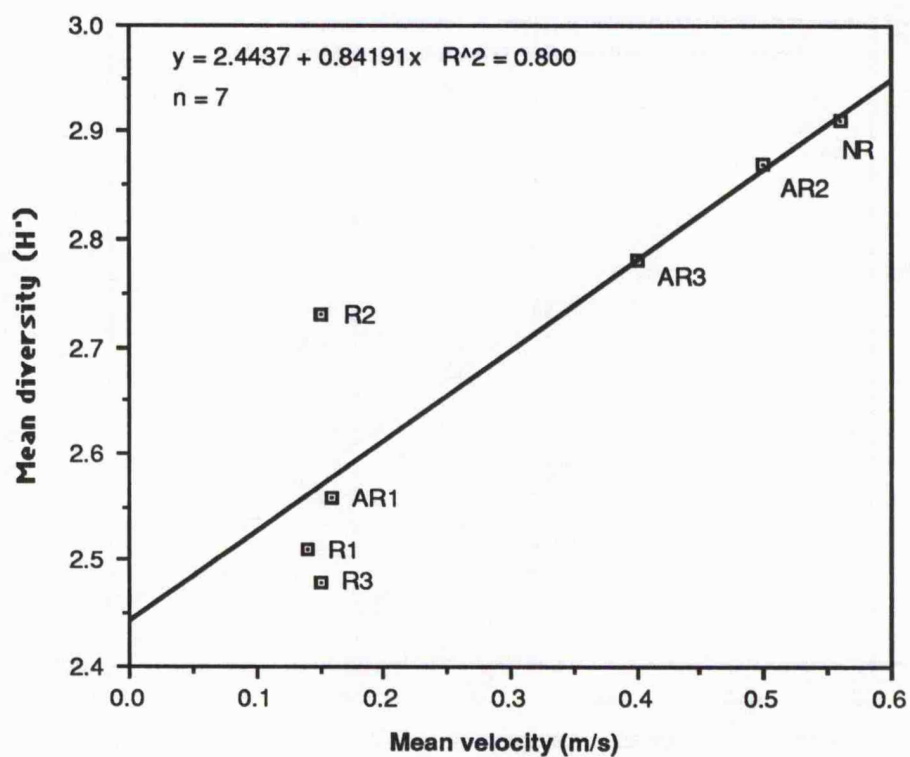


Figure 3.7 Correlation between velocity and diversity of the sites from Harper's Brook in summer 1993.

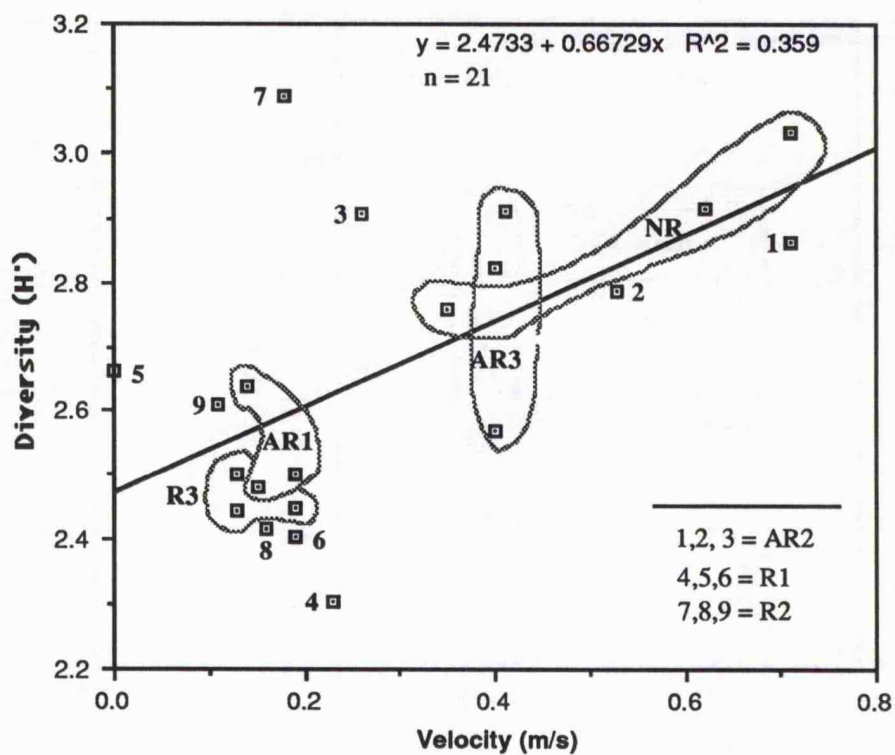


Figure 3.8 Correlation between velocity and diversity of the 21 samples from Harper's Brook in summer 1993.

3.3.4.2 AUTUMN

Figures 3.9 & 3.10 show the correlation between depth and diversity of the sites and samples. Both graphs show a weak negative relationship between depth and diversity ($r = -0.29$ for sites and -0.27 for samples), and none are significant. The variability between sites was very high, only AR2 and NR were placed close to each other (Figure 3.9).

Figures 3.11 & 3.12 show the correlation between velocity and diversity of the sites and samples. Both graphs show a weak positive linear relationship between velocity and diversity. $r = 0.31$ for the sites and $r = 0.35$ for the samples and none are significant. Here again most of the sites are very scattered around the fitted line except for AR2 and NR which show a great similarity, lying very close to each other (Fig. 3.11).

3.3.4.3 WINTER

Figures 3.13 & 3.14 show the correlation between depth and diversity of the site and the samples. Both graphs show negative relationship between depth and diversity which is strong for sites and modest for samples. Approximately 73% of the variation in diversity is accounted for by the variation in depth. R values are -0.73 and -0.69 for the sites and samples respectively, significant for samples at $P < 0.01$, but sites at $P < 0.1$. NR and AR3 are very close to each other and show some association with AR2 and R2. R3 and R1 are also close to each other to some extent (Figure 3.13).

The correlation between the velocity and diversity of the sites and the samples in winter is shown in figures 3.15 & 3.16. Both graphs show positive relationship between velocity and diversity which is strong for sites and modest for samples. R values are 0.75 and 0.69 for the sites and samples respectively, with r value of the samples significant at $P < 0.01$, and of the sites at $P < 0.1$. NR, AR2 and AR3 lie close to each other on the upper end of the fitted line and the same is true for R3 and R1 which are located lower down the line. R2 is in the middle position and AR1 is far apart from the others (Figure 3.15). The variability between the samples of AR3, NR, AR2 and AR1 is relatively low so the samples are located close to each other (Fig. 3.16).

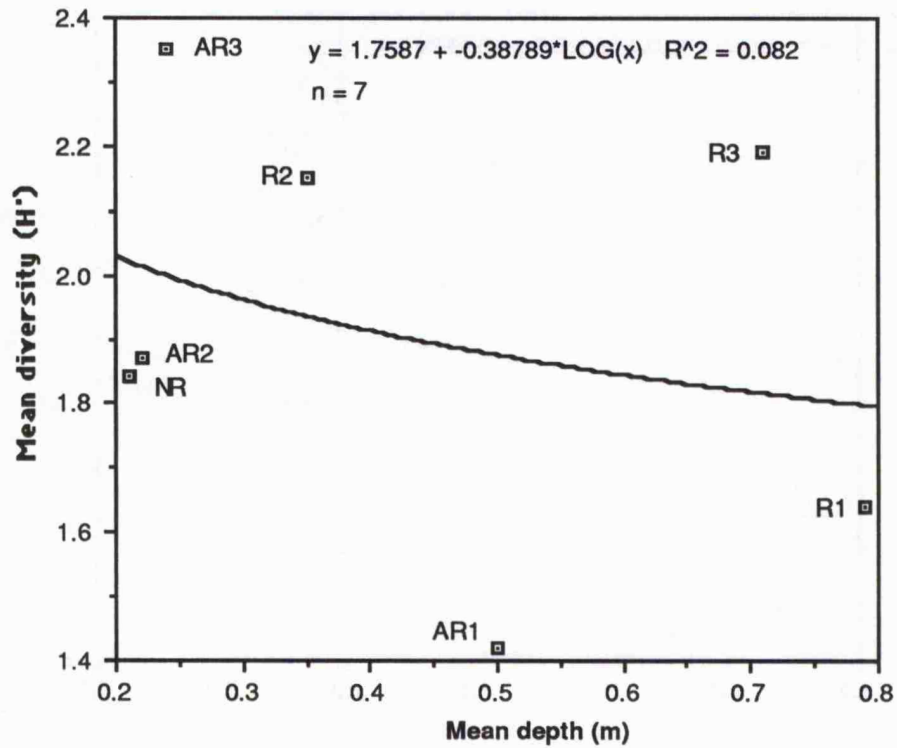


Figure 3.9 Correlation between depth and diversity of the sites from Harper's Brook in autumn 1993.

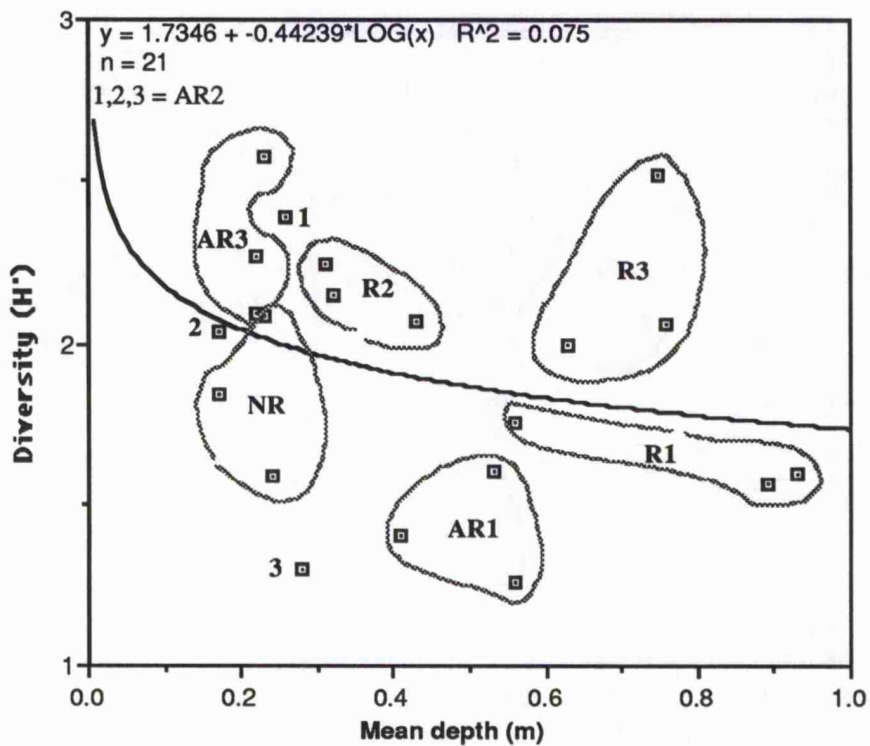


Figure 3.10 Correlation between depth and diversity of the 21 samples from Harper's Brook in autumn 1993.

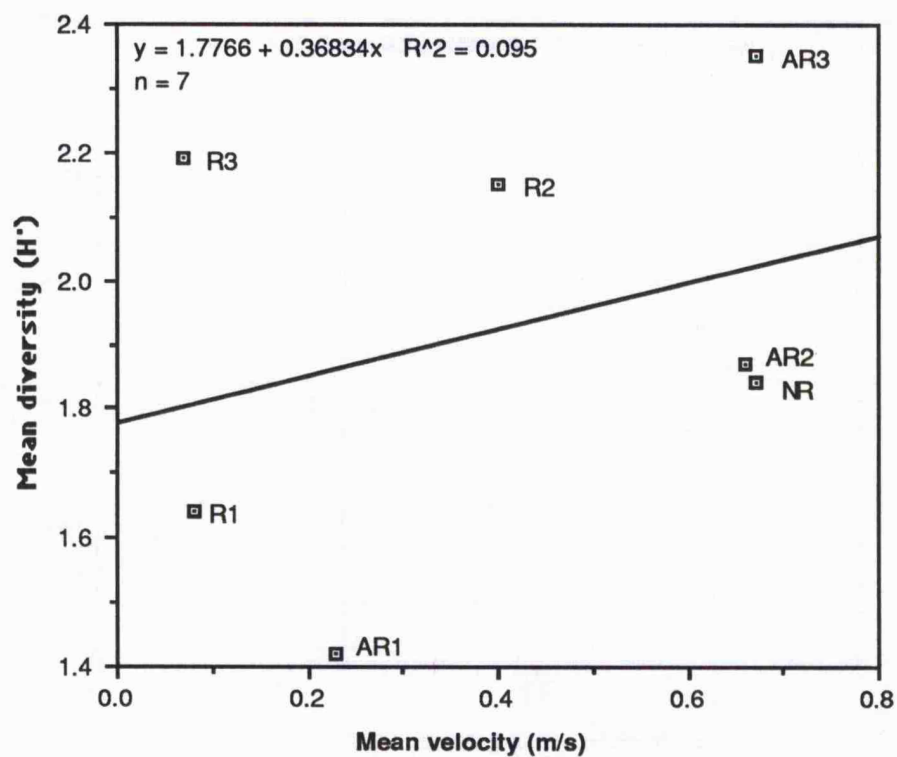


Figure 3.11 Correlation between velocity and diversity of the sites from Harper's Brook in autumn 1993.

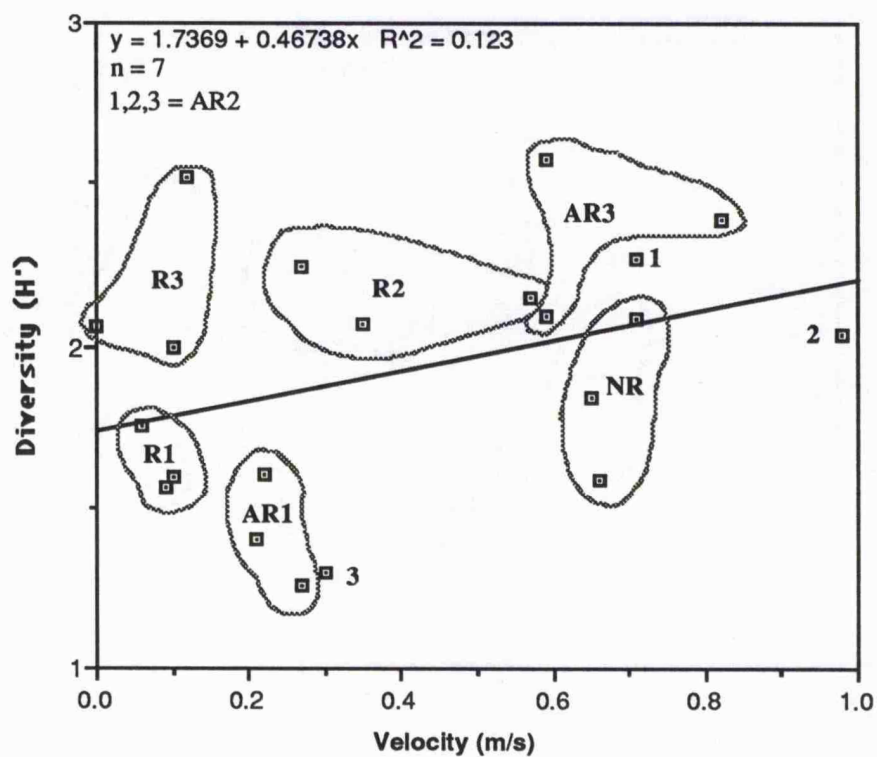


Figure 3.12 Correlation between velocity and diversity of the 21 samples from Harper's Brook in autumn 1993.

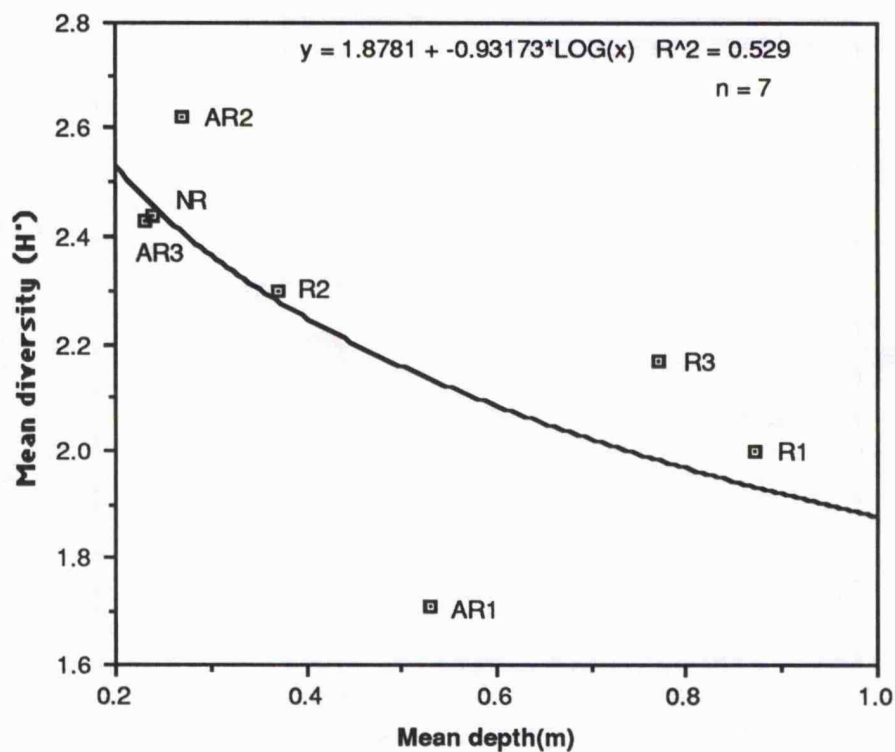


Figure 3.13 Correlation between depth and diversity of the sites from Harper's brook in winter 1993/94.

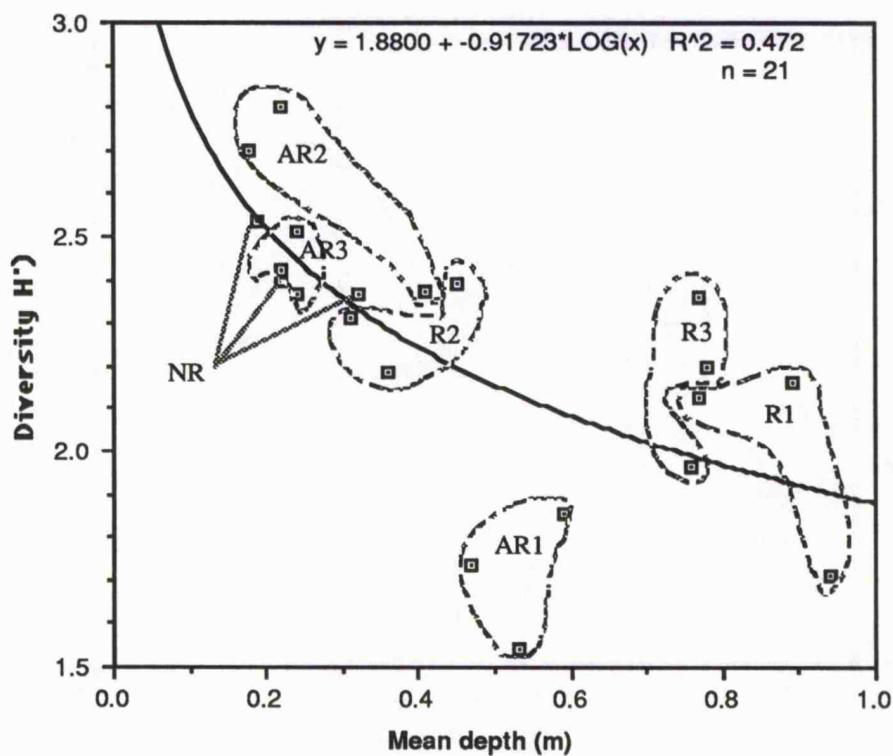


Figure 3.14 Correlation between depth and diversity of the 21 samples from Harper's Brook in winter 1993/94.

3.3.4.4 SPRING

Figure 3.17 & 3.18 show the relationship between depth and diversity of the site and samples. Both graphs show a negative relationship between the two variables. R values are -0.72 and -0.5 for sites and samples respectively, with more than 70% of the variation in diversity accounted for by the variation in depth in spring. The great variability between the samples of some of the sites is responsible for the smaller value of r of the samples (Fig. 3.18). R value of the samples is significant at $P < 0.05$, and for sites at $P < 0.1$. AR3 and AR2 are the closest sites to each other on the upper end of the fitted line and show some association with NR. R2 and AR1 are in the middle and R1 and R3 are located at the lower down of the fitted line (Fig. 3.17).

The relationship between velocity and diversity of the sites and samples in spring is shown in Figures 3.19 & 3.20. Both graphs show a positive linear relationship between velocity and diversity. R values are 0.76 and 0.49 for sites and samples respectively, with more than 70% of variation in diversity accounted for by the variation in velocity. R values of both sites and samples are significant at $P < 0.05$. Here again the great variability between the samples is responsible for the smaller value of r of the samples (Figure 3. 20). AR2 and AR3 are the most similar sites which are placed close to each other at the upper end of the fitted line. NR is closest to them, with other sites on the lower end of the line almost located in a similar distance from each other (Figure 3.19).

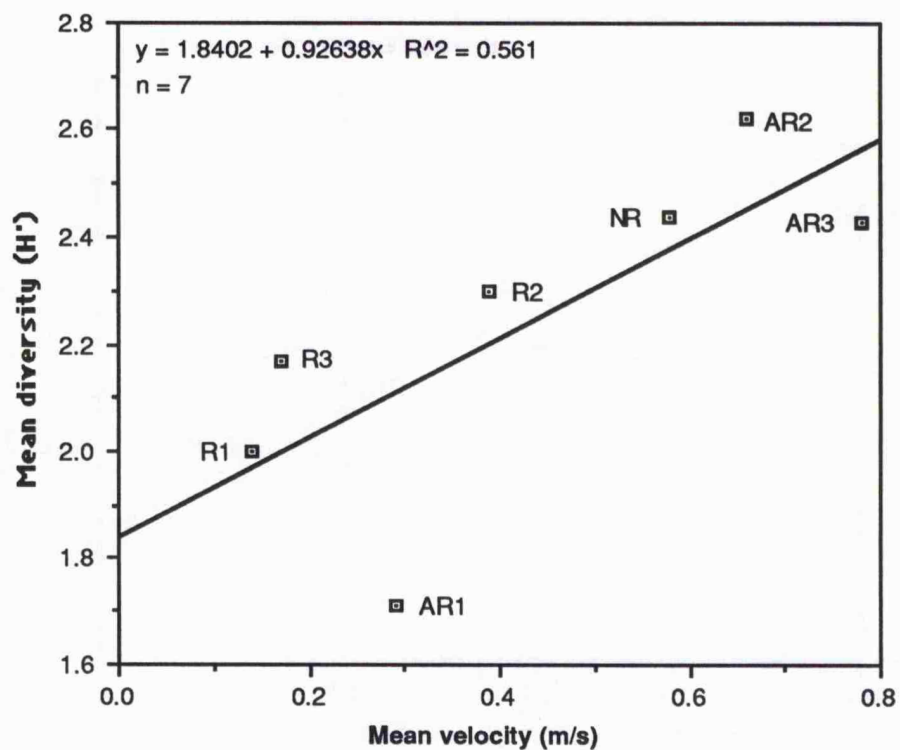


Figure 3.15 Correlation between velocity and diversity of the sites from Harper's Brook in winter 1993/94.

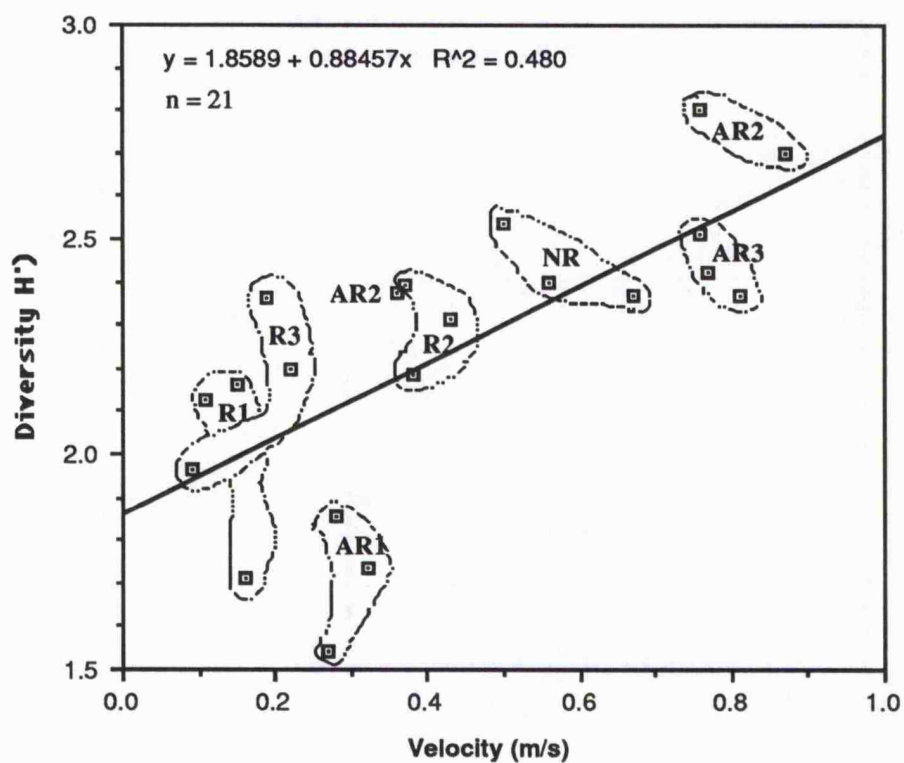


Figure 3.16 Correlation between velocity and diversity of the 21 samples from Harper's Brook in winter 1993/94.

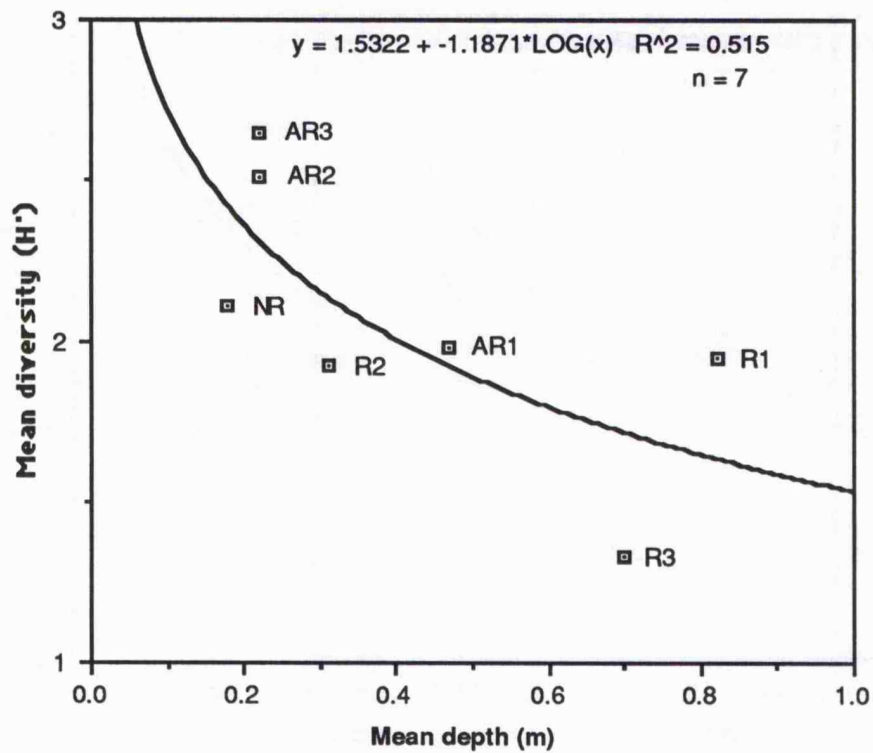


Figure 3.17 Correlation between depth and diversity of the sites from Harper's Brook in spring 1994.

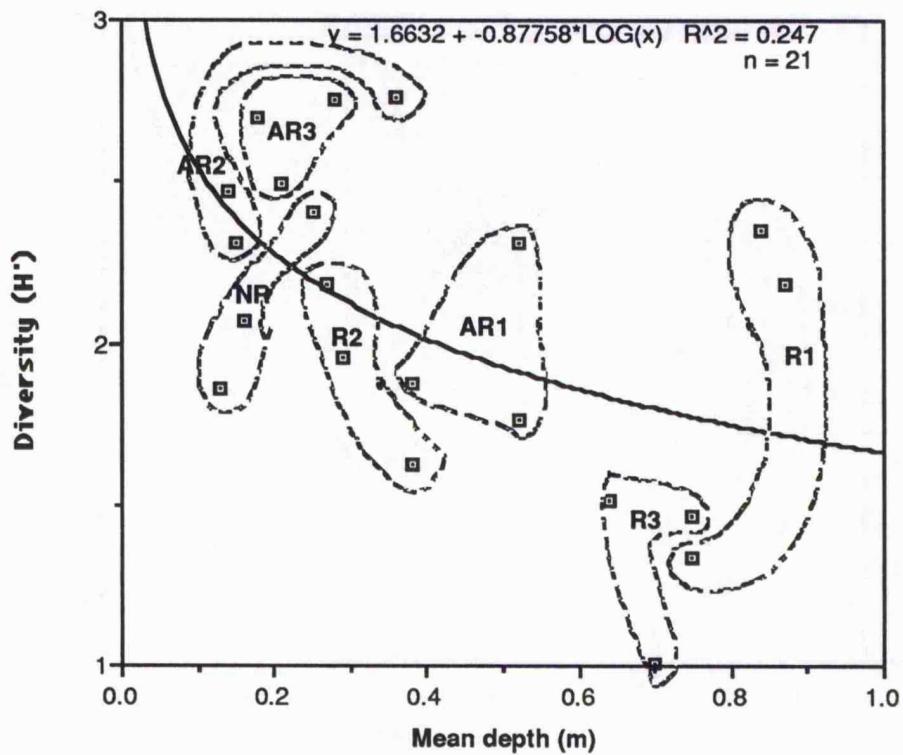


Figure 3.18 Correlation between depth and diversity of the 21 samples from Harper's Brook in spring 1994.

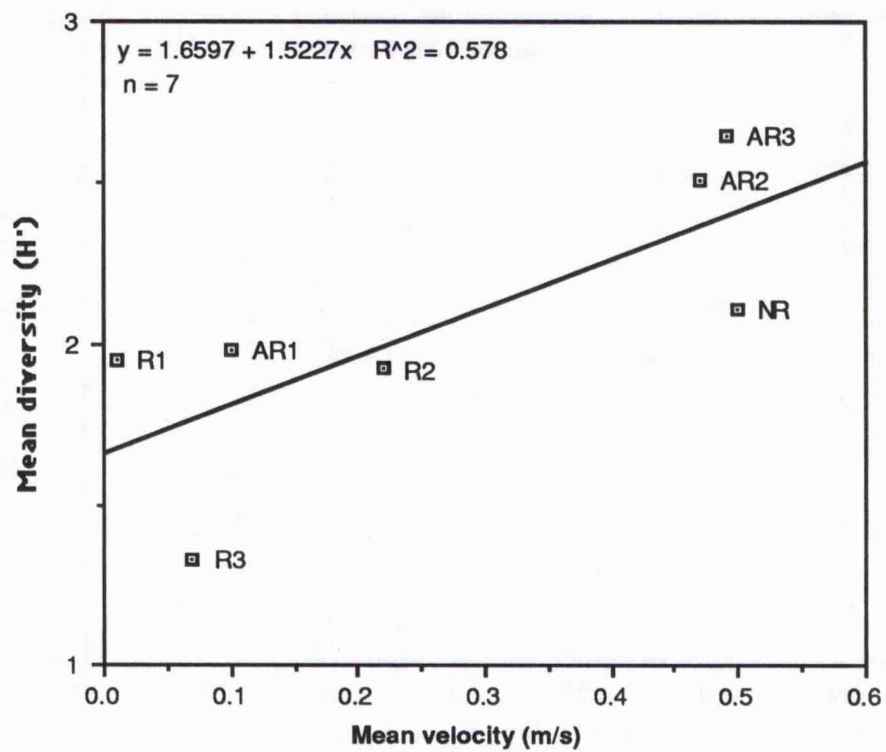


Figure 3.19 Correlation between velocity and diversity of the sites from Harper's Brook in spring 1994.

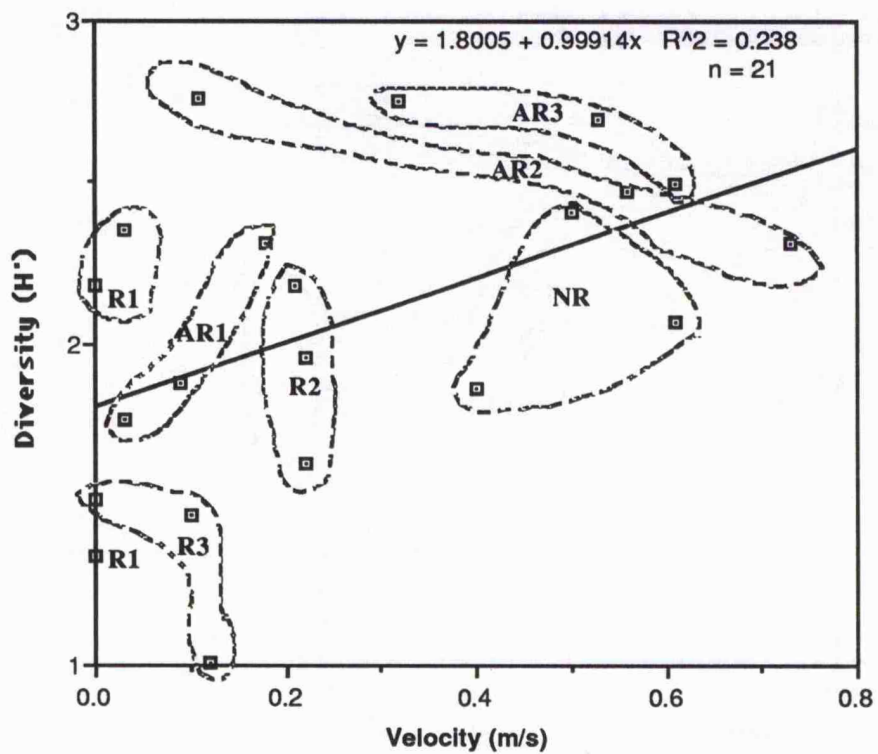


Figure 3.20 Correlation between velocity and diversity of the 21 samples from Harper's Brook in spring 1994.

3.3.4.5 CONCLUSIONS

From the above results it can be concluded that:

- There was a negative correlation between depth and diversity, and a positive correlation between velocity and diversity of the sites and the samples in all the seasons.
- The correlations were strong (70% – 89%) in summer, winter and spring, and weak (30%) in autumn.
- The Natural Riffle (NR) was more closely placed with either Artificial Riffle 2 (AR2) or Artificial Riffle 3 (AR3) which united them as a group in all the seasons.
- Thus two of the artificial riffles behaved hydrologically and biologically like a natural riffle.
- There was also similar relationship between Artificial Riffle 1 (AR1), Run 1 (R1) and Run 2 (R2) in summer, winter and spring, but not as clear as the former group.
- One of the artificial riffles, whose construction had given it greater depth (AR1) seemed to behave hydrologically as well as biologically like a Run, despite its coarse substrate.
- Since diversity decreases with greater depth and increases with greater velocity, deeper sites (Runs) have poorer macroinvertebrate communities and faster flowing sites (Riffles) have richer macroinvertebrate communities.

The environmental conditions (temperature, light, availability of food, shelter, etc.) are normally optimal in summer and late spring, and macroinvertebrate populations occur on all types of habitats in high abundance. Hence, the differences between different sites (habitats) are not as clear as in autumn and winter. In autumn and winter the differences between sites will become clearer as the environmental conditions decline, the diversity and abundance decrease dramatically, and the animals are eliminated from the poor habitats. The environmental conditions are also responsible for the differences between seasons which means the better the conditions the greater the diversity of the macroinvertebrate communities.

3.3.5 THE MULTIVARIATE ANALYSES OF COMMUNITIES

In order to determine the existence of any similar patterns among the samples or sites DECORANA and TWINSpan (DECOTWIN, Hill 1994) were used for ordination and classification of each data set from each season separately. A log (x+1) transformation of abundance values of species or higher taxonomic levels was applied before using both multivariate techniques.

3.3.5.1 DECORANA

In ordination by DECORANA, sites or replicate samples are arranged objectively into an order, those with similar taxonomic composition occurring most closely together. In each occasion the option of downweighting was used in order to minimise the influence of rare species (Hill, 1979). The axis 2 sample scores were plotted against axis 1 for each season. Only the first two axes of the ordination were used as the eigenvalue for the remaining axes were too low. In general, the higher the eigenvalue the more important the ordination axis. The results of DECORANA analysis are shown in Table 3.21. The output of DECORANA for each season is shown in Appendix Two.

Table 3.21 Eigenvalues for the first 3 axes of DECORANA from Harper's Brook in four seasons.

Season	Axis 1	Axis 2	Axis 3
Summer	0.264	0.055	0.028
Autumn	0.256	0.071	0.041
Winter	0.247	0.073	0.029
Spring	0.268	0.058	0.039

Figures 3.21–24 show the results of the analysis and the arrangement of the samples or sites between axis 1 and axis 2 in the four seasons. The major separation occurred along axis 1 which separated samples of NR, AR2 and AR3 from those of AR1, R1, R2 and R3. The riffle sites (except AR1) tended to have lower axis 1 sample scores than the run sites. This was true for all seasons. In summer the distinction between the

two groups was very clear and the replicate samples of most of the sites tended to be close to each other. There was also a clear differentiation between sites along axis 2 especially among riffle sites in summer (Fig. 3.21). In autumn, the three groups separated along axis 1 were riffle sites (except AR1), AR1, R2 & R3 and R1. AR1 was also separated from R2 & R3 along axis 2 (Fig. 3.22). In winter, in addition to the separation of riffle and run sites along axis 1, AR1 and run sites were separated, while riffle sites stayed close together (Fig. 3.23). The spring result showed clear separation of riffle sites (except AR1) from run sites and also the separation of R1 from the other run sites along axis 1. AR1 also separated from R2 and R3 along axis 2 (Fig. 3.24).

From the above results it can be concluded that there is a clear separation between the riffle sites (with the exception of AR1) and the run sites (including AR1) in all seasons.

In order to show the extent to which depth and velocity were linked to the DECORANA results the axis 1 sample scores were plotted against each variable for each season separately. Figures 3.25 & 3.26 show the results in summer. There was strong positive correlation between axis 1 and mean depth of the sites ($r = 0.87$, $P < 0.001$). Riffle sites (except AR1) separated from the run sites, with the former on the lower end of the fitted line and the latter on the upper. R1 also was separated from the run sites on the upper end of the line (Figure 3.25). There was strong, but negative correlation between axis 1 and velocity ($r = -0.83$, $P < 0.001$). Run sites and AR1 were close to each other lying on the upper end of the fitted line while riffle sites were scattered along the lower end (Figure 3.26).

Figures 3.27 & 3.28 show the results for autumn. There was strong positive correlation between axis 1 sample scores and mean depth of the sites ($r = 0.87$, $P < 0.001$). The riffle sites (except AR1) lie close to each other at the lower end of the fitted line and run sites were scattered along the rest of the line (Figure 3.27). The correlation between axis 1 and velocity of the sites was also strong but negative ($r = -0.81$, $P < 0.001$). Although the riffle sites stay fairly close together on the lower end of the line the runs are scattered along the rest of the line (Figure 3.28).

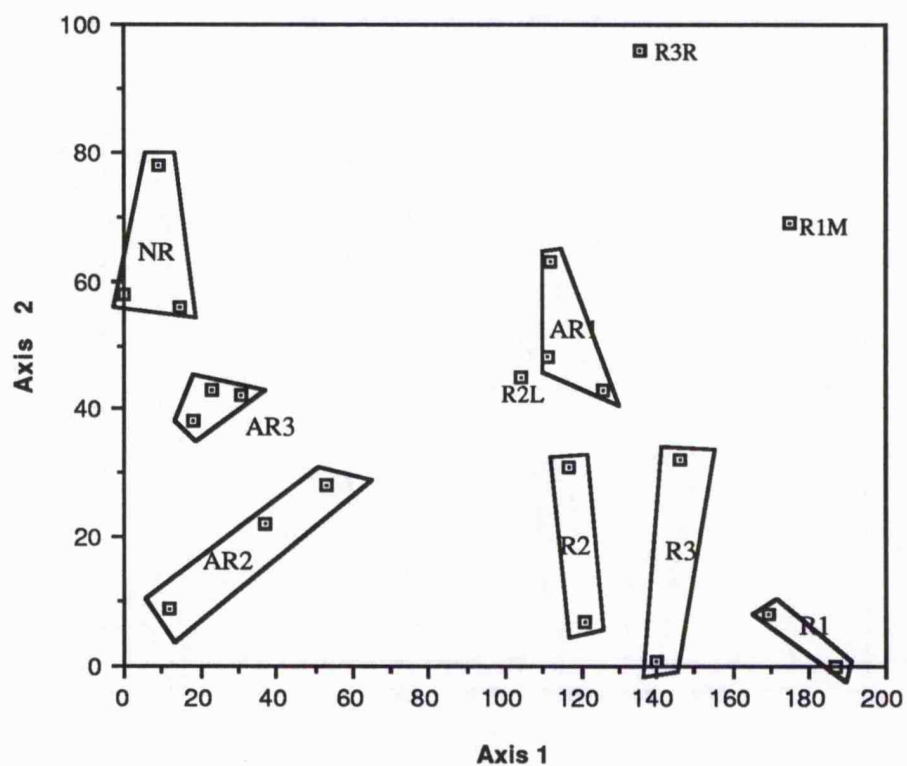


Figure 3.21 DECORANA of samples and sites from Harper's Brook in summer 1993.

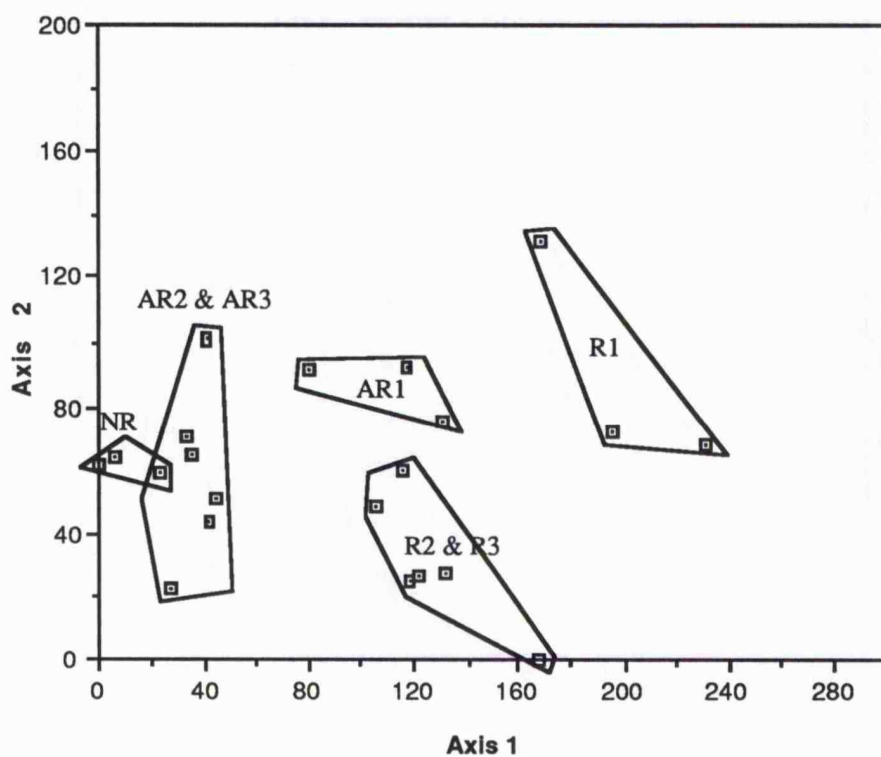


Figure 3.22 DECORANA of samples and sites from Harper's Brook in autumn 1993.

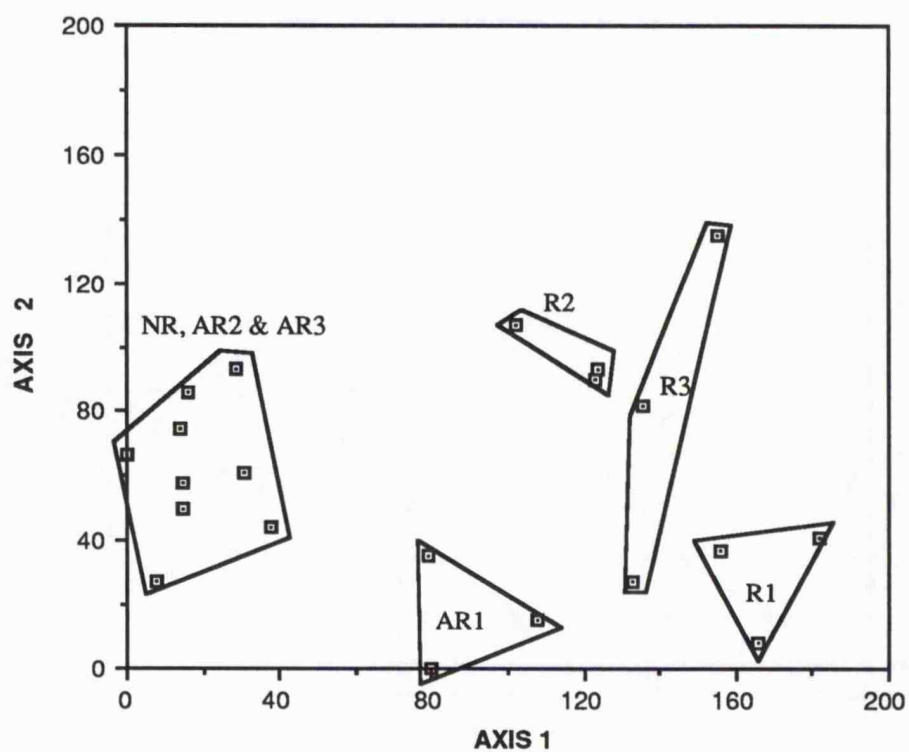


Figure 3.23 DECORANA of sites and samples from Harper's Brook in winter 1993/94.

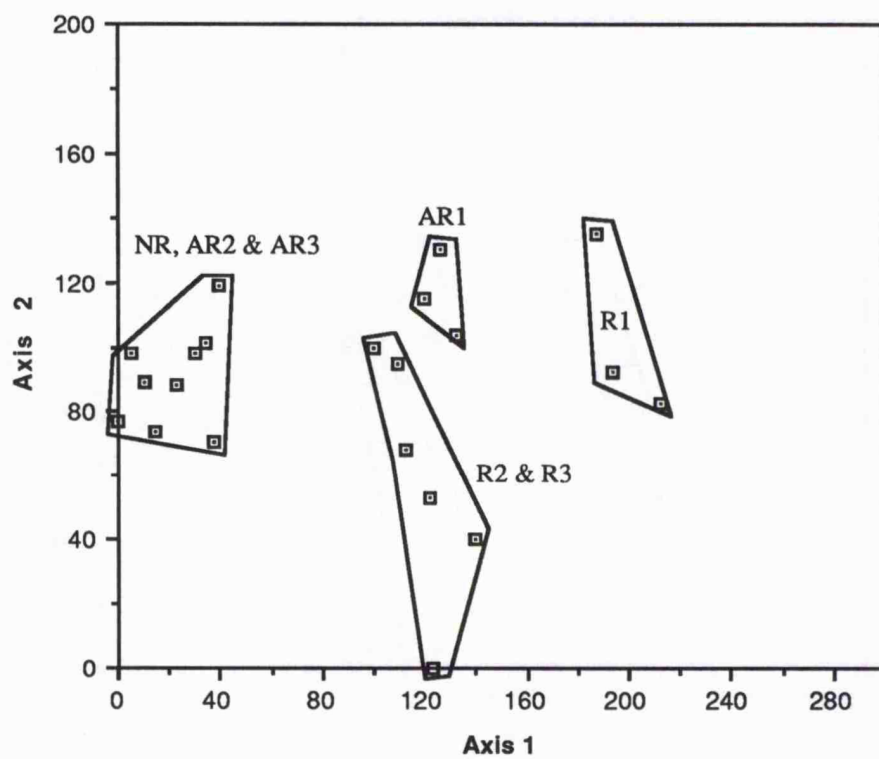


Figure 3.24 DECORANA of sites and samples from Harper's Brook in spring 1994.

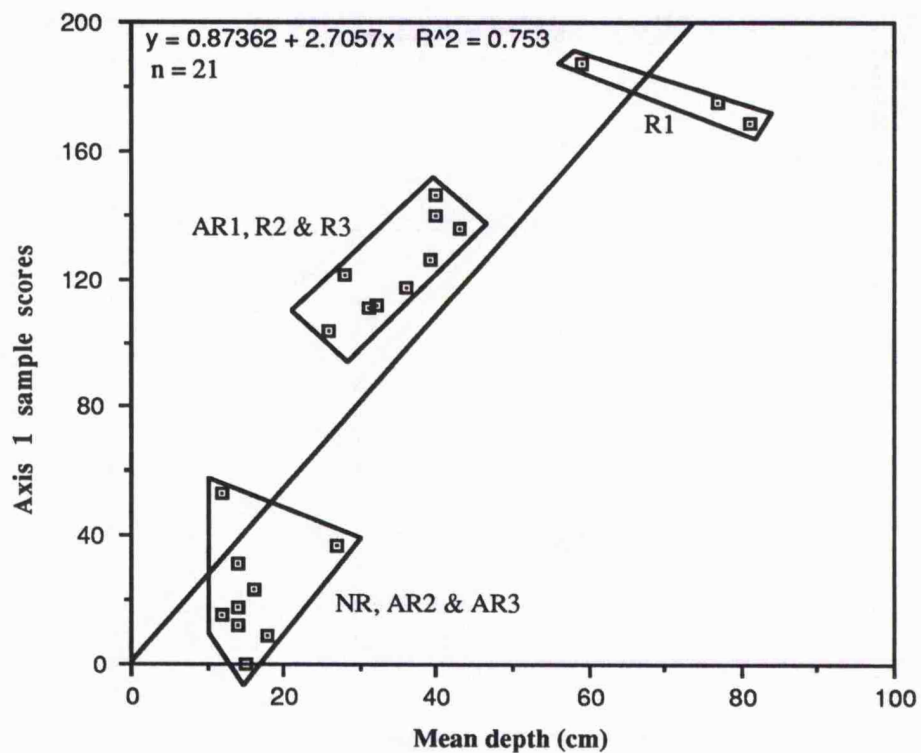


Figure 3.25 Correlation between axis 1 of DECORANA and the mean depth of the sites from Harper's Brook in summer 1993.

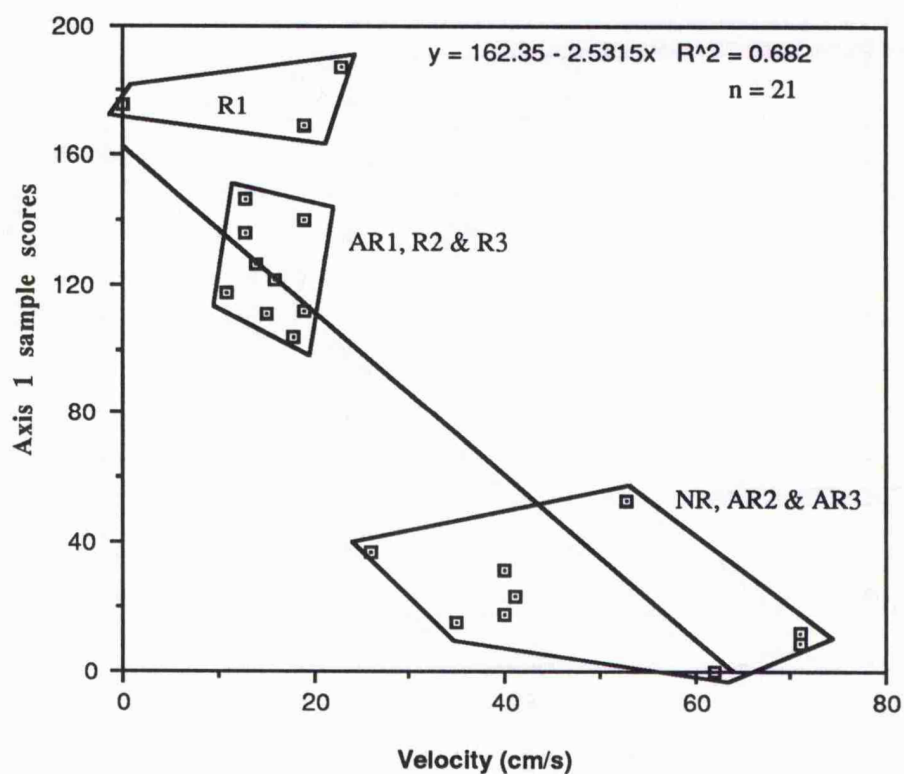


Figure 3.26 Correlation between axis 1 of DECORANA and the velocity of the sites from Harper's Brook in summer 1993.

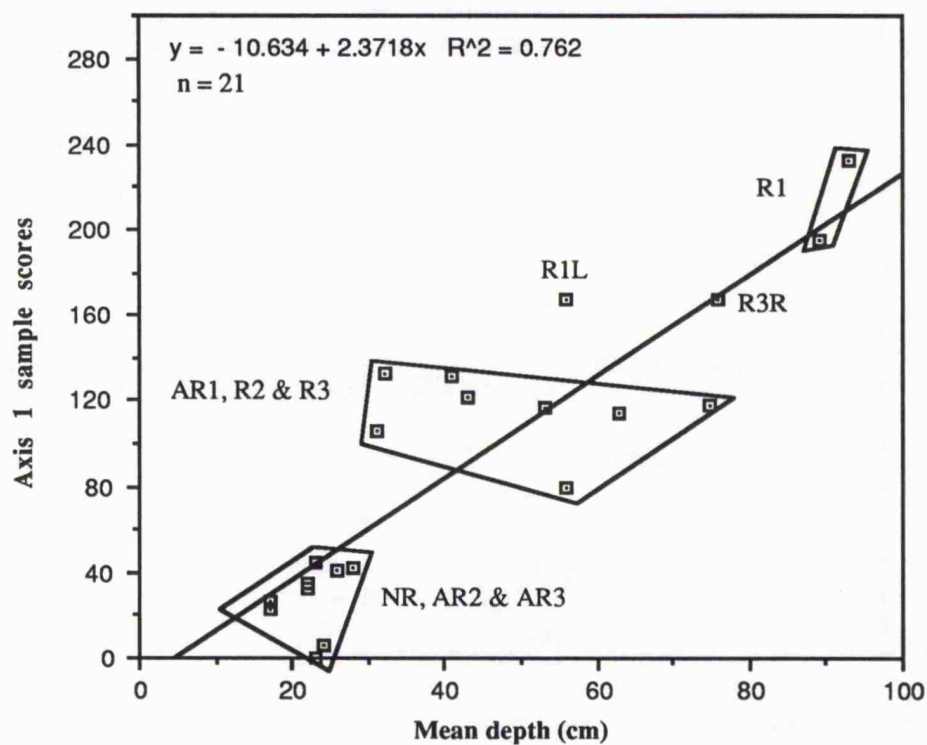


Figure 3.27 Correlation between axis 1 of DECORANA and the mean depth of the sites from Harper's Brook in autumn 1993.

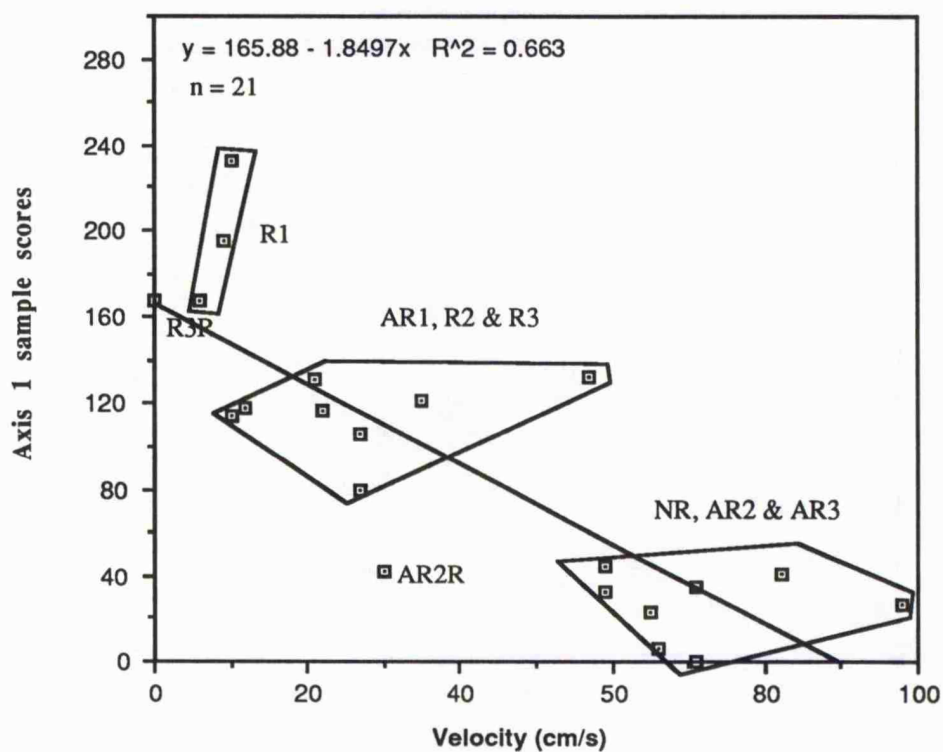


Figure 3.28 Correlation between axis 1 of DECORANA and the velocity of the sites from Harper's Brook in autumn 1993.

Figures 3.29 & 3.30 show the results for winter. The results indicated that there was again strong positive correlation between axis 1 scores and the mean depth of the sites ($r = 0.87$, $P < 0.001$). NR, AR2 and AR3 are positioned on the lower end of the fitted line, AR1 and R2 on the middle and R1 and R3 at the upper end of the line (Figure 3.29). The correlation between axis 1 and velocity was also strong and negative in winter ($r = -0.83$, $P < 0.001$). The location of the sites along the fitted line is similar to that of the depth, but the riffle sites are more scattered (Figure 3.30).

Figures 3.31 & 3.32 show the results in spring. The results showed that there was strong positive correlation between the axis 1 scores and the mean depth of the sites ($r = 0.89$, $P < 0.001$). Riffle sites (except AR1) are positioned close together on the lower end of the fitted line, but the other sites are scattered along the rest of the line; AR1, R2 and R3 in the middle but R1 at the upper end (Figure 3.31). There is strong negative correlation between axis 1 scores and the velocity of the sites in spring ($r = -0.86$, $P < 0.001$). Although the separation of riffle and run sites is clear along axis 1 most of the sites are scattered along the fitted line (Figure 3.32).

3.3.5.2 CONCLUSIONS

It can be concluded that :

- There is a strong correlation ($r > 0.8$) between axis 1 sample scores of DECORANA and the depth and velocity of the sites in all seasons. In three seasons the r values from the results of DECORANA were greater than the r values from the results of the univariate analysis (except in summer when the r values were slightly greater).
- The correlation between the axis 1 scores and depth is positive (i.e. the shallower the site, the smaller axis 1 scores and *vice versa*) and the correlation between axis 1 scores and velocity is negative (i.e. the faster flowing the site the smaller the axis 1 score and *vice versa*).
- The riffle sites (except AR1) and run sites are separated along the fitted line and in general, the former are positioned on the lower end of the line and the latter on the upper end, in all occasions.

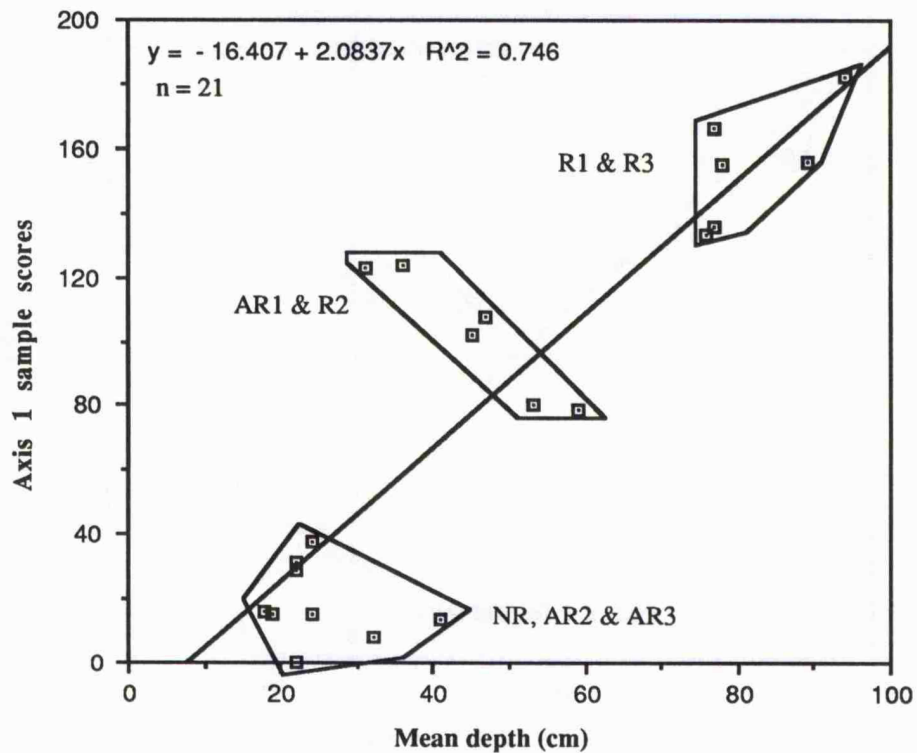


Figure 3.29 Correlation between axis 1 of DECORANA and the mean depth of the sites from Harper's Brook in winter 1993/94.

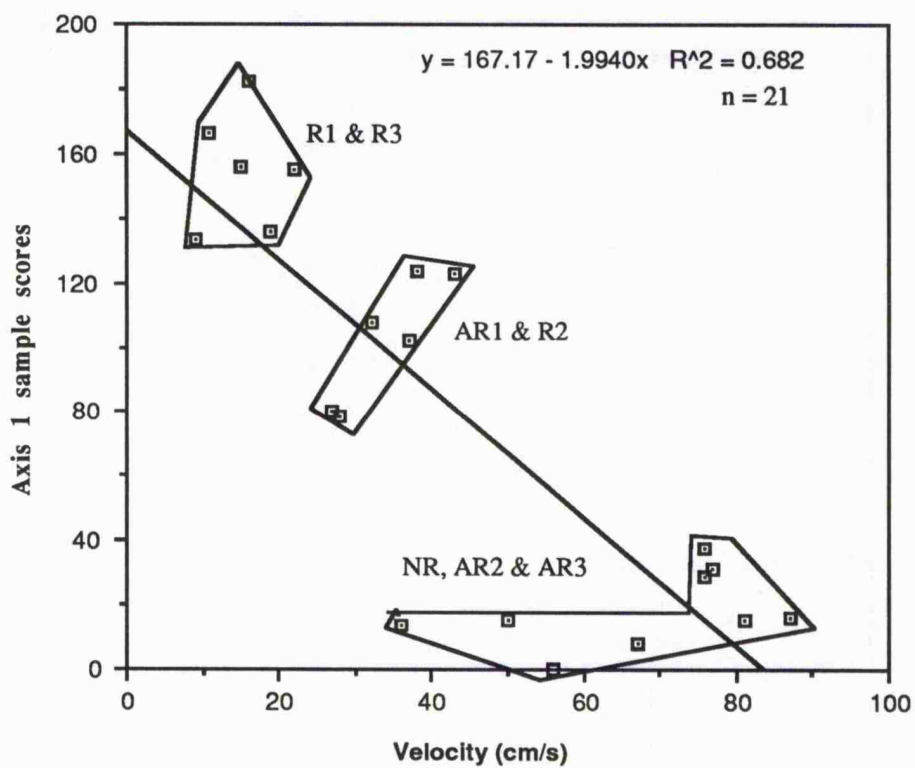


Figure 3.30 Correlation between axis 1 of DECORANA and the velocity of the sites from Harper's Brook in winter 1993/94.

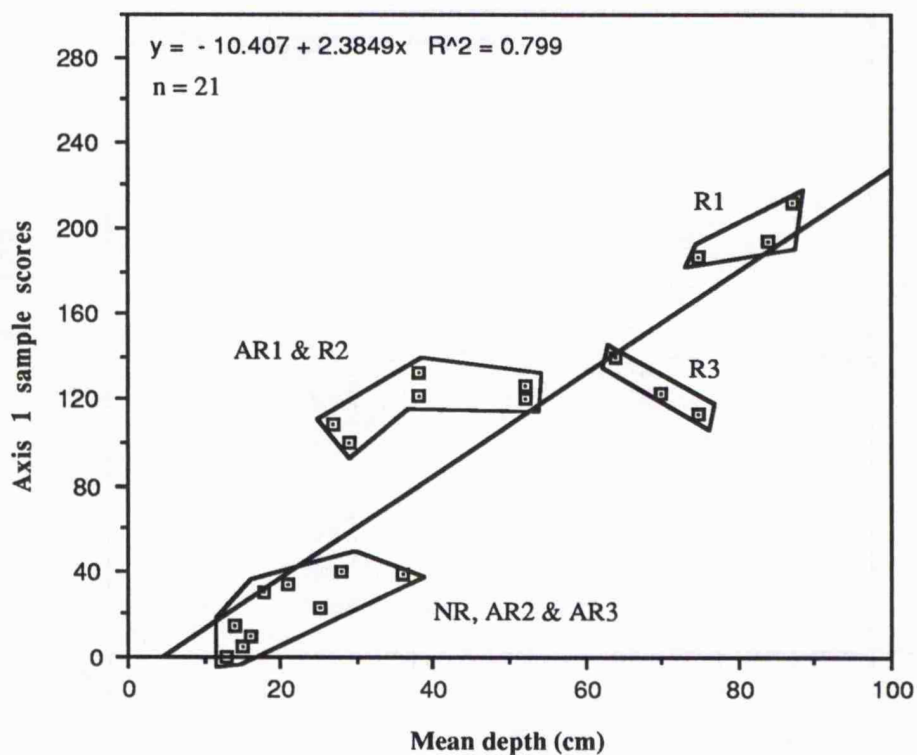


Figure 3.31 Correlation between axis 1 of DECORANA and the mean depth of the sites from Harper's Brook in spring 1994.

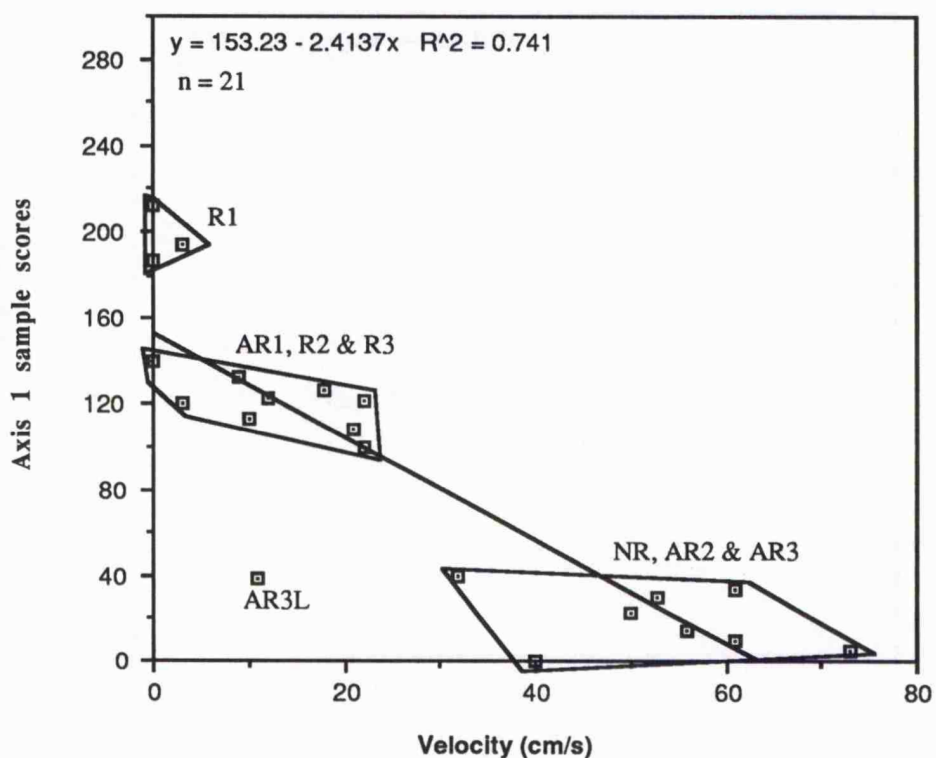


Figure 3.32 Correlation between axis 1 of DECORANA and the velocity of the sites from Harper's Brook in spring 1994.

3.3.5.3 TWINSpan

Samples were classified using two-way indicator species analysis, TWINSpan (Hill, 1994). The analyses were carried out for each season separately and all seasons together. The abundance values were transformed to $\log(x+1)$ values before analysis. The pseudospecies cut levels 1, 2 and 3 were chosen, as this produced greater eigenvalues for each dichotomy at each level of classification and the results corresponded with the DECORANA results. The weights 1, 2 and 1 were selected for levels of pseudospecies, giving double the weight to the second pseudospecies cut level, as the species with the log abundance of 2 were the common taxa and this produced clear results in all cases except in winter. Since the common species of the winter samples were in the first level of abundance, the weights 2, 1 and 1 were selected for this season. All samples and species were included in the analysis because the omission of the rare species did not affect the results. The output of TWINSpan for each season and all seasons together is shown in Appendix Two.

Figures 3.33– 3.36 show the results of TWINSpan in each season. In summer the 21 samples divided into group 0 which contained riffle sites (except AR1) on the negative side of the dichotomy and group 1 which contained AR1 and run sites on the positive side of the dichotomy at the first level of classification (Fig. 3.33). *Eukiefferiella*, *Tvetenia* (Chironomidae), *Hydropsyche angustipennis* (Hydropsychidae) and *Simulium* spp. (Simuliidae) were the indicator taxa of riffle sites (group 0). The Artificial Riffle 2 (group 00) separated from NR and AR3 group (01) at the second level of classification. *Culicoides* (Ceratopogonidae), *Oulimnius tuberculatus* (Elmidae) and Hydracarina were the indicator taxa of this dichotomy, associated with group 01. Group 1 divided into groups 10 and 11 at the second level of classification. Group 10 contained all samples from R2 and one sample from R3, and group 11, contained all AR1, R1 samples and two samples from R3. The indicator taxa of this division were *Hydroptila* spp. (Hydroptilidae), *Paratanytarsus* (Chironomidae) and *Psammoryctes barbatus* (Tubificidae) which are belonged to group 10. NR and AR3 separated at the third level of classification. *Baetis rhodani* and *Macropelopia* (Chironomidae) were the indicator taxa of

the Natural Riffle (NR) and *Euiyodrilus moldaviensis* (Tubificidae), Tubificidae juvenile, Lumbriculidae and *Asellus aquaticus* were the indicator taxa of AR3. Site AR1 (group 110) also separated from R1 and two remaining samples of R3 (group 111) at the third level of classification, and *Hydropsyche angustipennis* was the indicator species of AR1. Two samples of R2 (group 100) separated from R2L and R3R (group 101) at the third level of classification. Tubificidae juvenile was the indicator taxa of R2 and *Sigara* larvae (Corixidae), *Potthastia* (Chironomidae) and *Athripsodes aterrimus* (Leptoceridae) were the indicator taxa of the other (group 101). The classification was stopped at the third level as this revealed the separation of most of the sites which corresponded to the DECORANA result.

Figure 3.34 shows the result of TWINSpan for autumn 1993. The 21 samples divided into groups 0 and 1 at the first level of classification. Group 0 contained all samples from NR, AR2, AR3 and sample AR1R. *Hydropsyche angustipennis*, *H. siltalai* and *Elmis aenea* (Elmidae) were the indicator species of this group. Group 1 contained AR1L, AR1M and all samples from R1, R2 and R3. *Euiyodrilus hammoniensis* (Tubificidae) was the indicator species of this group. At the second level of classification, group 0 divided into group 00 which contained NR, AR2 and two samples of AR3 on the negative side of the dichotomy and group 01 which contained AR1R and AR3L on the positive side. *Athripsodes aterrimus* and *Haliphus* sp. (Halipidae) were the indicator species of this dichotomy belonged to AR1R and AR3L (group 01). Group 1 also divided into groups 10 and 11 at the second level of classification. AR1L, AR1M, R3L and all samples from R2 comprised group 10 on the negative side of the dichotomy with *Gammarus pulex* being the indicator species. R3M, R3R and all samples from R1 comprised group 11 on the positive side of the dichotomy. It was decided to end the classification at the second level, as this revealed the best results which corresponded to the DECORANA result for autumn.

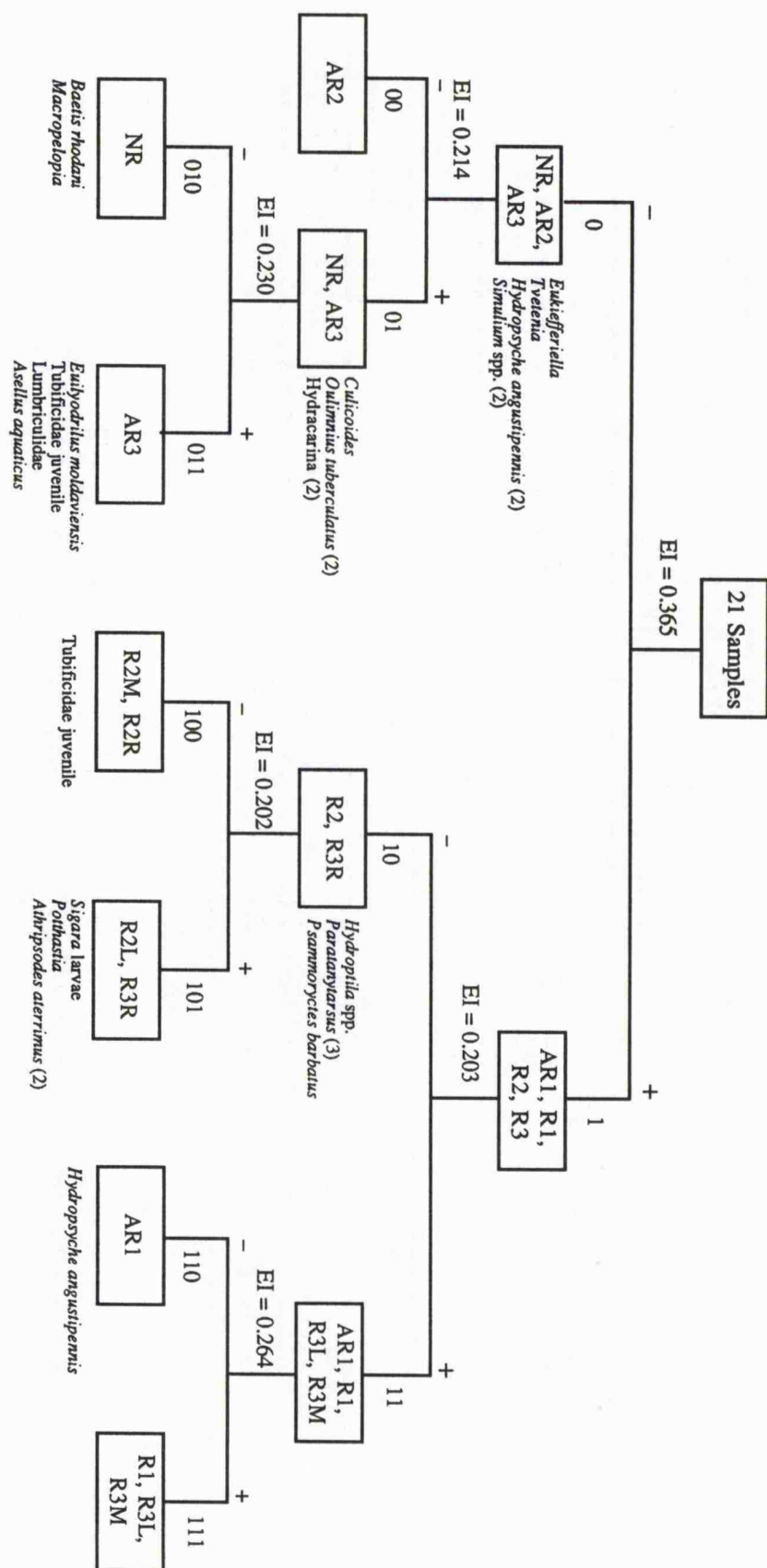


Figure 3.33 Classification of the 21 samples from Harper's Brook in summer 1993 by TWINSpan. EI = eigenvalues. Indicator species are shown (where present) for each group; numbers for pseudospecies levels (other than 1) are indicated.

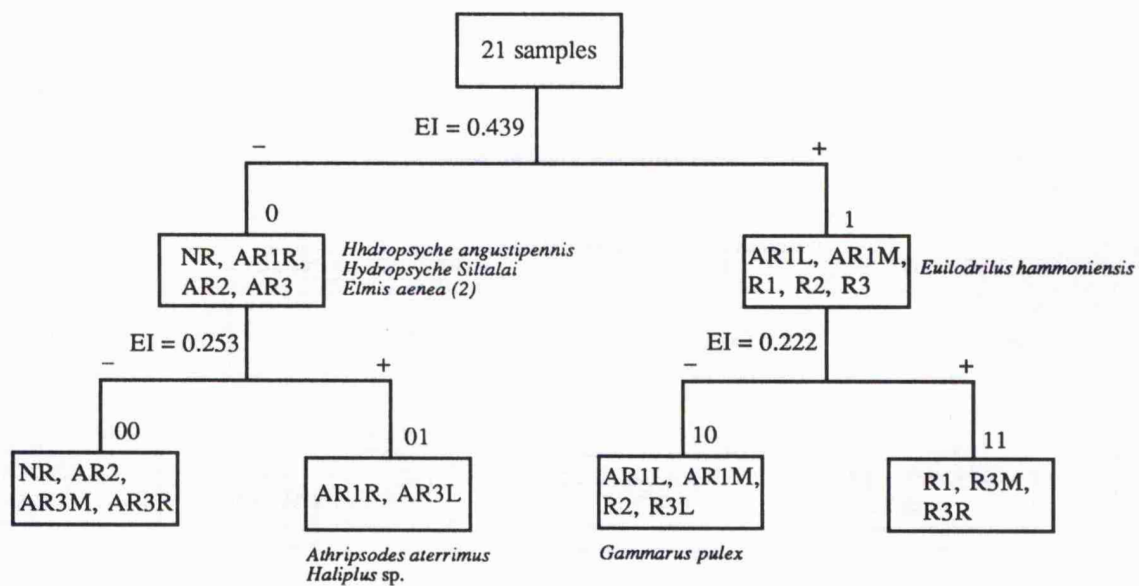


Figure 3.34 Classification of the 21 samples from Harper's Brook in autumn 1993 by TWINSpan. EI = Eigenvalues. Indicator species are shown (where present) for each group; numbers for pseudospecies levels (other than 1) are indicated.

Figure 3.35 shows the results of TWINSpan in winter. The 21 samples divided into group 0 which contained all riffle sites on the negative side of the dichotomy and group 1 which contained all run sites on the positive side. *Elmis aenea* was the indicator species of the division characteristic of riffle sites. AR1 separated from the other riffle sites (NR, AR2 and AR3) at the second level of classification and *Baetis rhodani* was the indicator species of the dichotomy characteristic of other riffle sites (group 00). Group 1 also split into group 10 which contained two samples from R1 and two samples from R3 and group 11 which contained R1M, R3R and all R2 samples at the second level of the classification. *Psammoryctes barbatus*, *Limnodrilus hoffmeisteri* (Tubificidae) and Tubificidae spp.B were the indicator species of the dichotomy, belonged to group 11. The classification was terminated at this level.

Figure 3.36 shows the results of TWINSpan in spring. Riffle sites (NR, AR2 & AR3) separated from run sites (R1, R2 & R3) and AR1 (group 1) at the first level of the classification. *Hydropsyche siltalai*, *Simulium* spp. and *Baetis scambus* were the indicator species of the dichotomy and characterised the riffle sites. The control site (NR) separated from the improved sites (AR2 & AR3) at the second level of classification while *Gammarus pulex* and *Euliyodrilus moldaviensis* were the indicator species of the improved sites (group 01). R1 also separated from AR1, R2 & R3 (group 10) at the second level of classification and Hydracarina was the indicator taxon of the dichotomy which belonged to group 10. The two improved sites (group 01) split at the third level of classification with *Orthocladius* (Chironomidae) and *Culicoides* being the indicator taxa of the dichotomy characteristic of AR3. Group 10 divided into group 100 which contained AR1, R3 and one sample from R2 and group 101 which contained two samples of R2, at the third level of classification. Lumbriculidae was the indicator taxon of the dichotomy belonged to group 101. AR1 split from R3 and R2L (group 1001) at the fourth level where the classification was terminated. *Oulimnius tuberculatus* was the indicator species of AR1 and *Thienemannimyia* (Chironomidae) was the indicator taxon of group 1001.

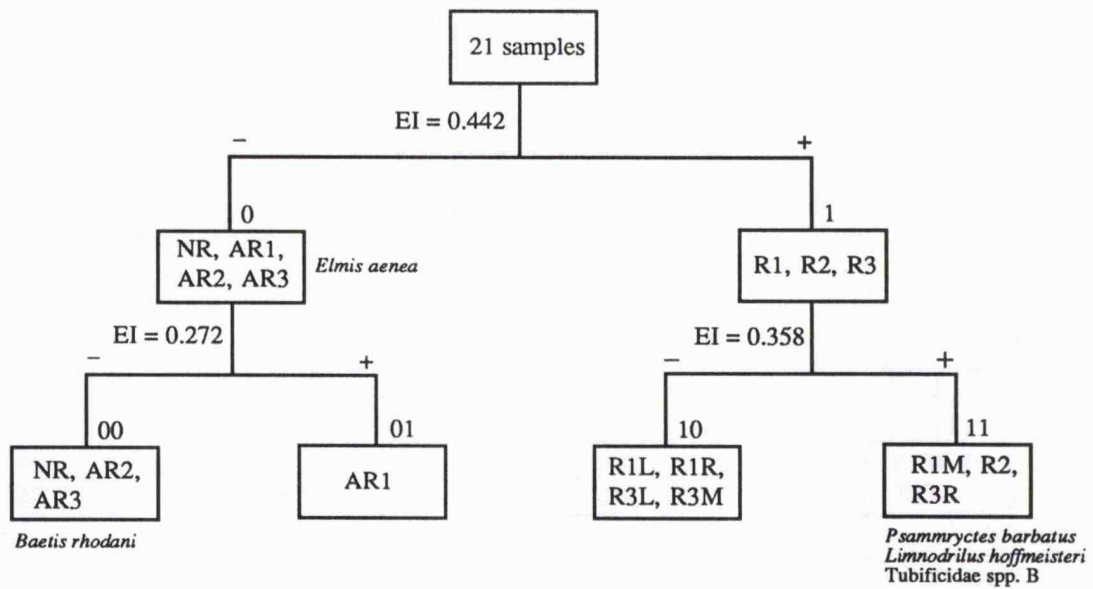


Figure 3.35 Classification of the 21 samples from Harper's Brook in winter 1993/1994 by TWINSpan. EI = Eigenvalues. Indicator species are shown for each group (where present), numbers for pseudospecies levels (other than 1) are indicated.

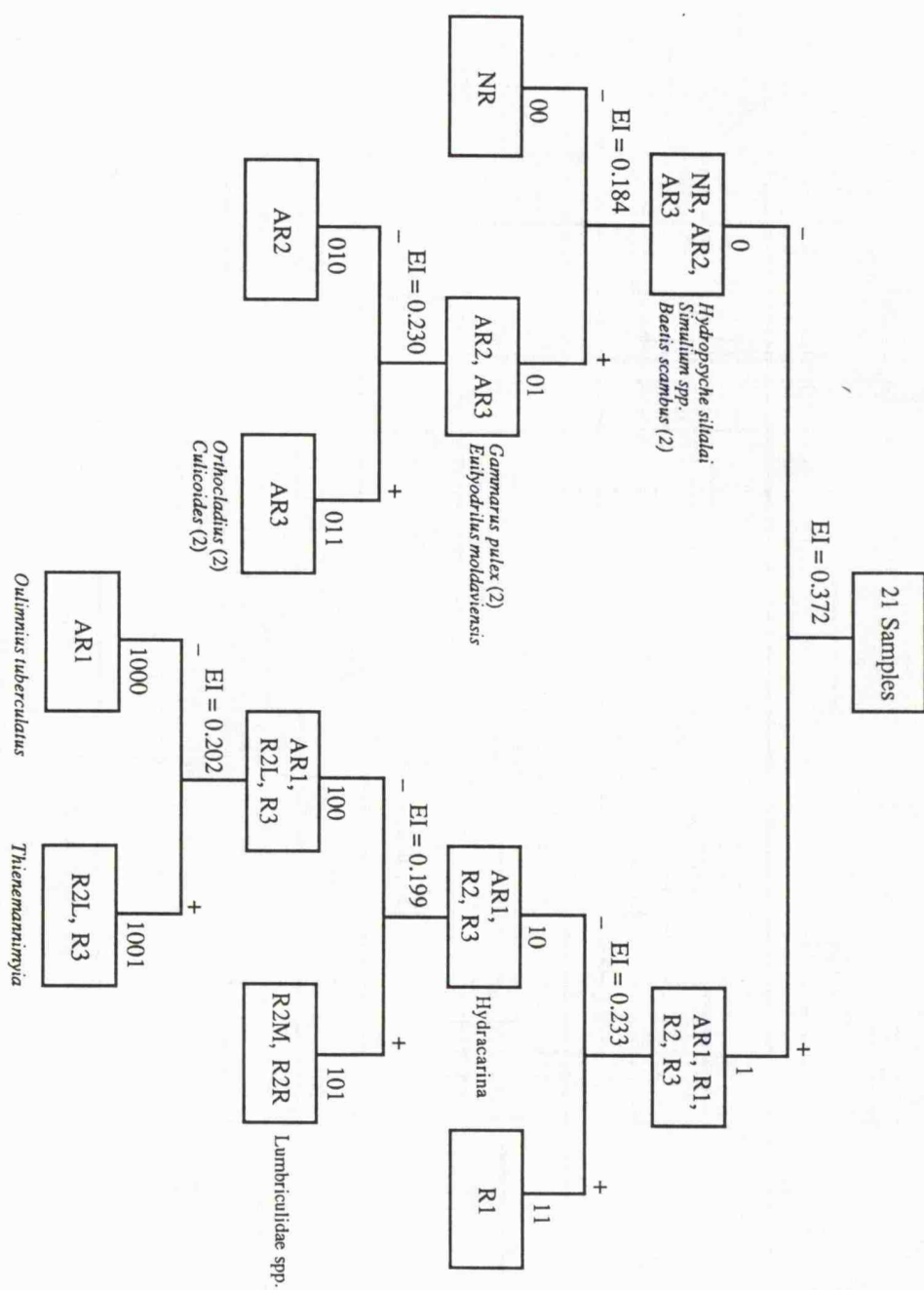


Figure 3.36 Classification of the 21 samples from Harper's Brook in spring 1994 by TWINSPLAN.
 EI = Eigenvalues. Indicator species are shown for each group, numbers for pseudospecies levels (other than 1) are indicated.

3.3.5.4 CLASSIFICATION OF ALL SEASONS AND SITES COMBINED

The abundance values of the three samples of each site were pooled and $\log(x+1)$ of the pooled values was calculated for each site in each season before analysis and the resultant 28 sites by 108 species/taxa data matrix was then subjected to TWINSpan. The pseudospecies cut levels 1, 2, 3 and weights 1, 2, 1 for levels of pseudospecies were selected and all species and samples were involved in the analysis.

The results (Figure 3.37) show that all run and AR1 sites of all seasons (except AR1 (winter) and R2 and R3 (spring) which are included in group 0) separated from NR, AR2 & AR3 of all seasons at the first level of classification. The second level of classification gave a separation between summer and the other seasons. Riffle sites (except AR1) of summer (group 00) separated also from the rest of the sites of group 0 which belonged to the other seasons (group 01). Run and AR1 sites of summer (group 10) were also separated from the rest of the sites of group 1 (group 11) at the second level of classification. The third level of classification involved the separation between spring and autumn and winter. Sites NR, AR2, AR3, R2 and R3 of spring (group 000) separated from sites NR, AR2, AR3 of autumn and sites NR, AR1, AR2 and AR3 of winter (group 001). Sites AR1 and R1 of spring (group 111) separated also from sites AR1, R1, R2 and R3 of autumn and sites R1, R2 and R3 of winter (group 110) at the third level of classification. The major event of the fourth level of classification was the separation of winter sites from autumn sites (only site R1 of autumn was included the winter sites). Spring run sites (R2 & R3) separated also from spring riffle sites (NR, AR2 & AR3), winter riffle sites (NR, AR1, AR2 & AR3) separated from autumn sites (NR, AR2 & AR3) and autumn sites (AR1, R2 & R3) separated from winter sites (R1, R2 & R3) and autumn site R1 at the fourth level of classification.

3.3.5.5 CONCLUSIONS

From the above results it can be concluded that:

- The separation of the sites in all seasons (excluding spring) corresponded to the results of individual seasons, which in general indicated the separation of riffle sites (except AR1) from those of run sites. Two artificial riffle sites AR2 and AR3 were associated with the Natural Riffle and the third AR1 was associated with the run sites (same as the DECORANA results).
- All seasons separated from each other; summer at the second level, spring at the third level and winter and autumn at the fourth level of classification. This indicated that the community composition of the sites in winter and autumn was relatively similar and those of summer and winter and autumn differed the most whilst those of spring lie in between.
- The variability between the seasons was greater than the variability between the sites, as the seasons were completely separated from each other (Figure 3.37) whereas the sites separated from each other only in spring (Figure 3.36). However, most of the riffle and run sites were also separated at the first level (TWINSpan for all seasons).

3.5 THE BIOLOGICAL INDICATORS OF THE SITES

In order to show the major indicator species of each site, the mean $\log(x+1)$ abundance values of three replicate samples of each site was calculated for summer and spring data, as the abundance and species richness of the samples were very high compared to autumn and winter. Those taxa with markedly greater abundance in one site and none or with very low abundance in the other were chosen as indicators and discussed here. Numbers in the brackets indicate the abundance values.

In summer *Baetis scambus* (2.4), *B. rhodani* (1.8), *B. vernus* (1.8) *Hydropsyche angustipennis* (2.3), *H. pellucidula* (2.1), *Rheotanytarsus* (1.0), *Macropelopia* (2.2), *Eukiefferiella* (2.6), *Tvetenia* (3.1) and *Simulium* spp. (2.7) formed the indicator taxa of natural riffles. The same taxa (except *Macropelopia*) comprised the indicator taxa of artificial riffles AR2 and AR3 but in different abundance (Table 3.22). The indicator taxa

of AR1 were *Microtendipes* (2.8), *Stictochironomus* (2.0), Microcrustacea (1.6) and *H. angustipennis* (1.5) which were relatively similar to those of run sites except for *H. angustipennis* which was absent from two run sites. *Caenis horaria* (0.3 – 1.6), *Microtendipes* (0.3 – 2.9), *Stictochironomus* (0.4 – 2.7) *Procladius* (1.0 – 1.8) and Microcrustacea (Copepoda and Cladocera) (2.5) were the major indicator taxa of run sites. R2 was different from the other run sites and behaved like a riffle in which some species such as *Baetis scambus* (1.0), *B. rhodani* (1.2), *B. vernus* (1.2), *H. angustipennis* (1.5) and *Simulium* spp. (1.5) were abundant whereas they were absent or rare in the other run sites .

In spring *B. scambus* (2.6 – 3.1), *H. pellucidula* (0.3 – 0.6), *H. siltalai* (1.5 – 1.8), *Tvetenia* (1.8 – 2.3) and *Simulium* spp. (1.3 – 1.5) were the indicator taxa of riffle sites (except AR1) and were absent or rare in run sites including AR1. *Sialis lutaria* (0.1 – 0.4), *Cryptochironomus* (0.2 – 0.6) and *Paratendipes* (0.4 – 0.8) formed the indicator species of runs including AR1 and were absent from riffle sites.

Although in general some taxa are associated with riffles and some with runs it is the collaboration of flow, depth, substrate and vegetation that determine the occurrence and abundance of the taxa. Genera such as *Baetis*, *Hydropsyche* and *Simulium* are generally found in fast-flowing habitats and are the indicator taxa of the riffles. The above results indicated that *B. rhodani* was exclusively found in riffle sites but not in the AR1 and runs (except R2), whereas *B. scambus* and *B. vernus* were found in all riffles, while *B. scambus* were rare in AR1 in spring. Thus *B. rhodani* and *B. scambus* could be good indicators of good riffles. *H. pellucidula* and *H. siltalai* were also exclusively found in riffles in spring but not in AR1. *H. pellucidula* was rare in AR1 in summer, thus these two species could also be good indicators of good riffles when they occurred in relatively high abundance. *Simulium* spp. was found abundant in all riffles (it was rare in AR1) in spring but not in any of run sites. In summer it occurred in all sites but in different abundance. *Eukiefferiella* and *Tvetenia* were also exclusively found in riffles but not in AR1 in both summer and spring and are good indicators of good riffles.

Table 3.22 Major indicator species of the sites from Harper's Brook in summer 1993 and spring 1994 (values are mean log(x+1) abundance of three replicate samples of each site).

SUMMER	NR	AR1	AR2	AR3	R1	R2	R3
<i>Caenis horaria</i>	0.0	0.0	0.0	0.0	0.8	1.6	0.3
<i>Baetis scambus</i>	2.4	0.9	1.7	2.2	0.2	1.0	0.0
<i>Baetis rhodani</i>	1.8	0.0	1.0	0.2	0.0	1.2	0.0
<i>Baetis vernalis</i>	1.8	1.0	1.6	2.0	0.0	1.2	0.6
<i>Hydropsyche angustipennis</i>	2.3	1.5	2.2	3.1	0.0	1.5	0.0
<i>Hydropsyche pellucidula</i>	2.1	0.2	0.8	1.7	0.0	0.0	0.0
<i>Rheotanytarsus</i>	1.0	0.6	0.4	1.6	0.0	0.0	0.0
<i>Microtendipes</i>	0.0	2.8	1.6	0.6	2.0	2.9	3.0
<i>Stictochironomus</i>	0.0	2.0	2.1	0.5	0.4	2.6	2.7
<i>Macropelopia</i>	2.2	1.6	0.0	0.0	0.5	1.4	0.8
<i>Procladius</i>	0.0	1.0	0.0	0.0	1.8	1.4	1.0
<i>Eukiefferiella</i>	2.6	0.0	1.9	2.8	0.0	0.6	0.0
<i>Tvetenia</i>	3.1	0.0	2.1	2.8	0.0	0.7	0.0
<i>Simulium</i> spp.	2.7	1.0	2.5	2.5	0.2	1.5	0.3
Microcrustacea	0.5	1.6	1	1.1	2.5	2.5	2.5
SPRING							
<i>Baetis scambus</i>	3.1	0.2	2.8	2.6	0.0	0.6	0.0
<i>Hydropsyche pellucidula</i>	0.6	0.0	0.3	0.6	0.0	0.0	0.0
<i>Hydropsyche siltalai</i>	1.5	0.0	1.5	1.8	0.0	0.3	0.0
<i>Sialis lutaria</i>	0.0	0.2	0.0	0.0	0.4	0.1	0.0
<i>Cryptochironomus</i>	0.0	0.2	0.0	0.0	0.3	0.6	0.2
<i>Paratendipes</i>	0.0	0.8	0.0	0.0	0.4	0.8	0.6
<i>Orthocladius</i>	2.4	1.2	1.9	2.2	0.5	1.6	0.4
<i>Tvetenia</i>	2.3	0.0	2.2	1.8	0.0	0.5	0.3
<i>Simulium</i> spp.	1.3	0.2	1.5	1.5	0.0	0.0	0.0

CHAPTER FOUR

The River Smite

CHAPTER FOUR

THE RIVER SMITE

4.1 PHYSICAL AND ENVIRONMENTAL CHARACTERISTICS

Depth, velocity, substrate and vegetation data collected from the River Smite in four seasons in 1993–1994 are summarised in Tables 4.1 – 4.4.

In summer (September) 1993 gravel and sand comprised the substrate of three sites; Above Deflector (AD), Below Deflector (BD), and Shoal (SH) whilst Natural Riffle (NR) had 50% cobble and boulder, and 50% gravel & sand (Table 4.1). The deepest site was BD (0.39 m) and the shallowest was SH (0.14 m), NR was the next shallow site (0.2 m). NR had the greatest vegetation cover (100%) and BD had the least (20%). Depth and water turbulence, which was produced by the scouring velocity below the deflector possibly were the reasons for the low vegetation at the latter site. AD also had high vegetation (90%) while it had relatively moderate depth (0.27 m).

Table 4.1 Environmental data form the River Smite in summer 1993.

Site	Length (m)	Mean width (m)	Mean depth (m)	Substrate	vegetation
NR	6	4.5	0.2	40% cobble, 10% boulder, 50% gravel & sand	80% <i>Cladophora</i> , 20% <i>Potamogeton</i>
AD	5	5.5	0.27	gravel & sand	90% <i>Cladophora</i>
BD	6	5.5	0.39	gravel & sand	20% <i>Cladophora</i>
SH	6	6	0.14	gravel & sand	40% <i>Cladophora</i>

The physical data in autumn (November) 1993 (Table 4.2) showed that the amount of vegetation had declined in all sites, ranging from none in BD and Run to a maximum of 40% in NR. In comparison to the summer data, due to the higher discharge the mean

depth and mean width of all sites had increased. The deepest site was the newly sampled Run (0.6 m), followed by BD (0.48 m) and the shallowest was NR (0.22 m) followed by SH (0.33 m). NR had the greatest velocity (0.73 ms^{-1}) and Run had the least (0.24 ms^{-1}) which indicated that, as expected, there was a negative relationship between depth and velocity, that is the deeper the site the lower the velocity and *vice versa*. The substrate type of the sites did not change in autumn and the substrate of Run was composed of gravel – sand (50%) and sand – clay (50%).

Table 4.2 Environmental data from the River Smite in autumn 1993.

Site	Length (m)	Mean width (m)	Mean depth (m)	Mean velocity (m/s)	Substrate	vegetation
NR	6	5.3	0.22	0.73	40% cobble, 20% boulder, 40% gravel & sand.	30% <i>Cladophora</i> , 10% <i>Potamogeton</i>
RUN	5	5.8	0.6	0.24	50% gravel & sand, 50% clay & silt.	none
AD	5	8.5	0.47	0.27	gravel & sand	10% <i>Cladophora</i>
BD	6	8.5	0.48	0.33	gravel & sand	none
SH	6	8.5	0.33	0.3	gravel & sand	5% <i>Cladophora</i>

In winter (February) 1993/94 the vegetation was absent from all the sites except NR (40%). The substrate of all of the sites had also changed except for SH (gravel – sand). More gravel and sand (60%) accumulated in NR, the same happened to the Run (90%) and some cobbles were found in BD. These changes in substrate type could be due to the higher discharges and the transportation of the larger substrate from upstream during winter. The mean depth and velocity of sites were relatively similar to those of the autumn.

In spring (May) 1994 more changes in vegetation occurred at all sites compared to autumn and winter, ranging from 90% in NR to 10% in AD and BD. Gravel and sand formed the dominant substrate in most of the sites (except for NR), ranging from 80% in AD to 20% in NR. The dominant substrate of NR was cobble (60%). Some silt, ranging

between 50% – 20% was found in slower flowing sites (Run, AD and SH) which could be due to the settling velocities of the sites. The velocity of the sites had also changed. NR had the greatest velocity (0.54 ms^{-1}) and Run the least (0.09 ms^{-1}), while other sites had also lower velocities ($0.11\text{--}0.14 \text{ ms}^{-1}$) compared to the other seasons.

Table 4.3 Environmental data from the River Smite in winter 1993/94.

Site	Length (m)	Mean width (m)	Mean depth (m)	Mean velocity (m/s)	Substrate	vegetation
NR	6	5.2	0.21	0.77	30% cobble, 10% boulder, 60% gravel & sand	40% <i>Cladophora</i> ,
RUN	5	5.9	0.6	0.21	90% gravel & sand, 10% boulder & clay	none
AD	5	7.4	0.47	0.29	85% gravel & sand, 15% silt/slate	none
BD	6	8.3	0.52	0.34	50% sand/slate, 50% slate & cobble	none
SH	6	8.7	0.34	0.21	gravel & sand	none

Table 4.4 Environmental data from the River Smite in spring 1994.

Site	Length (m)	Mean width (m)	Mean depth (m)	Mean velocity (m/s)	Substrate	vegetation
NR	6	5.5	0.17	0.54	60% cobble, 10% boulder, 30% gravel & sand	70% <i>Cladophora</i> , 20% <i>Potamogeton</i>
RUN	5	5.6	0.54	0.09	50% gravel & sand, 50% silt/mud	30% <i>Cladophora</i> , 10% <i>Potamogeton</i>
AD	5	7.5	0.41	0.14	80% gravel & sand, 20% silt	5% <i>Cladophora</i> , 5% <i>Potamogeton</i>
BD	6	8.4	0.47	0.11	60% gravel & sand, 20% bedrock, 20% clay	10% <i>Cladophora</i>
SH	6	8.8	0.3	0.12	70% gravel & sand, 30% silt	20% <i>Cladophora</i> , 10% <i>Potamogeton</i>

From the above results it can be concluded that the environmental characteristics of the sites of the River Smite changed within and between seasons. In general, the dominant substrate of Natural Riffle was cobble, whilst gravel and sand comprised the other sites. NR was the shallowest site with highest velocity which followed by SH, and Run was the deepest with lowest velocity which followed by BD. There was a tendency for the water depth to increase from summer to spring due to the higher discharge. The amount of vegetation was maximum in summer, declining in autumn and winter, and increasing again in spring. Although the substrate of the "improved" sites (AD, BD and SH) were relatively similar, the velocity and depth of the sites differed, because of the presence of the deflector.

4.2 MACROINVERTEBRATE COMPOSITION AND ABUNDANCE

The full lists of species composition and abundance of the five sites from the River Smite in the four seasons are presented in Appendix one. The occurrence and percentage of the family abundance on each site in each season are discussed below.

4.2.1 SUMMER

4.2.1.1 Description

A total number of 82,098 macroinvertebrates belonging to 79 species or higher taxonomic groups was collected from the River Smite in summer 1993. Table 4.5 shows the relative abundance of the taxa at family level on the four sampling sites, the presentation of the taxa at family level has been discussed earlier (chapter three, section 3.2.1). Only the families forming $\geq 1\%$ of the total abundance are shown here, those with abundances of $< 1\%$ are shown in Appendix one. Data from the other seasons are treated in the same way.

Thirteen families formed at least 96% of the total abundance, whereas 24 families with $< 1\%$ formed at most 4% of the total abundance. The dominant family at all sites was Caenidae ranging from 60.9% in SH to 25.7% in NR in summer (Table 4.5).

Baetidae (12.6%), Elmidae (11.9%), Chironomidae (11.5%), Simuliidae (9%) and Leptoceridae (8.3%) were the abundant families of the Natural Riffle (NR). Tubificidae (4.8%), Sphaeriidae (3.9%), Hydracarina (3.5%), Hydropsychidae (3.4%) and Asellidae (2.4%) formed the frequent families of the site (Table 4.5). Sialidae and Dytiscidae were absent from NR, and others were either occasional (0.1% – 1%) or rare ($< 0.1\%$).

Chironomidae (31.3%) was the second dominant family of the Above Deflector site (AD). Asellidae (8.9%), Tubificidae (6%), and Hydracarina (5.8%) were the abundant families of the site. Microcrustacea (Copepoda and Cladocera) (4.9%), Elmidae (2.1%), Leptoceridae (1.9%) and Naididae (1.6%) formed the frequent families of the site (Table 4.5). Simuliidae, Ancylidae, Hydra and Empididae were absent, other families were either occasional or rare.

Chironomidae (20.3%) formed the second dominant family of the Below Deflector site (BD). Asellidae (13%) and Elmidae (6.5%) were the abundant families of the site. Tubificidae (4.4%), Leptoceridae (3.8%), Hydracarina (2.5%) and Microcrustacea (1.3%) formed the frequent families of the site (Table 4.5). Hydra was absent from the site and other families were either occasional or rare.

In the Shoal site (SH), Caenidae was the dominant family with 60.9% abundance and Chironomidae with 14% abundance and Asellidae (6%) formed the abundant families. Tubificidae (4.4%), Elmidae (4.4%), Hydracarina (3.2%), Leptoceridae (2.5%) and Microcrustacea (1.5%) were the frequent families of the site (Table 4.5). All other families were either occasional or rare.

Table 4.5 Macroinvertebrate family abundance $\geq 1\%$ of the sites from the River Smita in summer 1993.

Taxa	NR	AD	BD	SH
Naididae	0.23	1.62	0.69	0.31
Tubificidae	4.77	6.03	4.37	4.35
Asellidae	2.35	8.89	12.96	5.95
Caenidae	25.67	32.49	42.71	60.88
Baetidae	12.56	0.35	0.44	0.19
Hydropsychidae	3.44	0.34	0.32	0.09
Leptoceridae	8.33	1.92	3.84	2.52
Elmidae	11.93	2.08	6.54	4.38
Chironomidae	11.45	31.34	20.27	14.02
Simuliidae	8.97	0.00	0.01	0.01
Sphaeriidae	3.90	0.38	0.08	0.20
Hydracarina	3.48	5.82	2.54	3.15
Microcrustacea	0.07	4.92	1.27	1.46
Total % Abundance	97.16	96.19	96.04	97.52

4.2.1.2 Comparison between habitats

Natural Riffle differed from the other sites in having the greatest abundance of Baetidae (12.6% v. 0.4% – 0.2%), Hydropsychidae (3.4% v. 0.3% – 0.1%), Leptoceridae (8.3% v. 3.4% – 1.9%), Elmidae (11.9% v. 6.5% – 2.1%), Simuliidae (9% v. 0.01%) and Sphaeriidae (3.9% v. 0.4% – 0.1%). These taxa are commonly found in

fast-flowing and shallower habitats with greater abundance. The most distinctive differences of Above Deflector with the other sites were the greatest abundance of Naididae (1.6% v. 0.7% – 0.2%), Chironomidae (31.4% v. 20.3% – 11.5%), Hydracarina (5.8% v. 3.5% – 2.5%) and Microcrustacea (4.9% v. 1.5% – 0.1%). This site also had the greatest abundance of Tubificidae. Asellidae (13% v. 8.9% – 2.4%) was the only taxon found with the greatest abundance on BD. Shoal had the greatest abundance of Caenidae (60.9% v. 42.7% – 25.7%) and differed from the other sites. The three sites AD, BD and SH were similar in having Baetidae, Hydropsychidae and Simuliidae with very low abundance (< 1%), and Asellidae with greater abundance (13% – 6% v. 2.4% in NR).

4.2.2 AUTUMN

4.2.2.1 Description

A total number of 34,389 macroinvertebrates belonging to 63 species or higher taxonomic level was collected from the River Smite in autumn (November) 1993. Table 4.6 shows the relative family abundance of the taxa found on five sampling sites in autumn. There were 9 families with $\geq 1\%$ abundance which formed at least 96.6% of the total abundance, whereas 27 families with < 1% formed at most 3.4% of the total abundance. Caenidae was the dominant family of all sites ranging from 86.8% in Shoal to 40.5% in Natural Riffle (Table 4.6).

Natural Riffle was characterised by Caenidae (40.5%) and Elmidae (18.8%) as the dominant families, and Simuliidae (15.1%), Leptoceridae (10.3%) and Hydropsychidae (6.1%) being the abundant families of the site (Table 4.6). Asellidae (4.6%), Chironomidae (1.1%) and sphaeriidae (1.4%) formed the frequent families. Piscicolidae, Corixidae, Valvatidae, Planorbidae and Hydracarina were absent from NR and other families were either occasional or rare.

Caenidae (60.3%) was the dominant family of the Run site and Tubificidae (12.4%) and Simuliidae (10.8) formed the abundant families of the site. Asellidae (2.4%), Hydropsychidae (2.5%), Leptoceridae (4.4%), Elmidae (2.8%) and Chironomidae (1.7%)

comprised the frequent families of Run in autumn (Table 4.6). Naididae, Piscicolidae, Goeridae, Hydrobiidae, Valvatidae, Ancyliidae, Limnaeidae, Planorbidae and Muscidae were absent from the site whilst the other families were either occasional or rare.

The Above Deflector site (AD) was characterised by Caenidae (63.8%) as the dominant family and Tubificidae (16.1%), and Chironomidae (11.4%) as the abundant taxa of the site. Asellidae (1.3%), Hydropsychidae (1.5%), Leptoceridae (1.9%), Elmidae (1.5%) and Simuliidae (1.3%) were the frequent families (Table 4.6). Sphaeriidae, Lumbriculidae, Goeridae, Hydrobiidae, valvatidae and Muscidae were absent from the site while the other families were either occasional or rare.

Table 4.6 Macroinvertebrate family abundance $\geq 1\%$ of the sites from the River Smite in autumn 1993.

Taxa	NR	RUN	AD	BD	SH
Tubificidae	0.74	12.43	16.08	13.49	3.14
Asellidae	4.63	2.41	1.27	2.88	1.41
Caenidae	40.52	60.33	63.76	67.19	86.82
Hydropsychidae	6.10	2.45	1.47	0.99	0.55
Leptoceridae	10.27	4.39	1.89	1.15	2.16
Elmidae	18.76	2.78	1.49	3.37	2.86
Chironomidae	1.07	1.72	11.35	6.17	1.29
Simuliidae	15.12	10.82	1.29	0.58	0.27
Sphaeriidae	1.36	0.07	0.00	0.82	0.09
Total % Abundance	98.58	97.40	98.60	96.63	98.59

Caenidae (67.2%) was the dominant taxon and Tubificidae (13.5%) and Chironomidae (6.2%) were the abundant taxa of the Below Deflector site (BD). Asellidae (2.9%), Hydropsychidae (1%), Leptoceridae (1.2%) and Elmidae (3.4%) formed the frequent taxa of the site (Table 4.6). Lumbriculidae, Sialidae, Dytiscidae, Corixidae, Tipulidae, Ancyliidae, Limnaeidae, Planorbidae, Hydracarina, and Muscidae were absent from the site, while other taxa were either occasional or rare.

The Shoal site (SH) was dominated by Caenidae (86.8%). Tubificidae (3.1%), Asellidae (1.4%), Leptoceridae (2.2%), Elmidae (2.9%) and Chironomidae (1.3%) comprised the frequent taxa (Table 4.6). Lumbriculidae, Piscicolidae, Sialidae, Dytiscidae,

Halipidae, Corixidae, Limnaeidae, Hydracarina and Muscidae were absent; other families were either occasional or rare.

4.2.2.2 Comparison between habitats

In autumn NR differed from the other sites in having the greatest abundance of Hydropsychidae (6.1% v. 2.5% – 0.6%), Leptoceridae (10.3% v. 4.4% – 1.2%), Elmidae (18.8% v. 3.4% – 1.5%), Simuliidae (15.1% v. 2.8% – 0.3%), Sphaeriidae (1.4% v. 0.8% – 0.1%) and Asellidae (4.6% v. 2.8% – 1.3%) (in summer it had the lowest percentage of Asellidae). NR had also the least abundance of Tubificidae (0.7% v. 16.1% – 3.1%) in autumn. Run site differed from the others in having Hydropsychidae (2.5%), Leptoceridae (4.3%) and Simuliidae (10.8%) with the second highest abundance (NR was in the first place). AD had the greatest abundance of Tubificidae (16.1% v. 13.5% – 0.7%) and Chironomidae (11.4% v. 6.2% – 1.1%) compared with the other sites. BD had the second greatest abundance of Tubificidae (13.5%) and Chironomidae (6.2%) (AD was in first place). Shoal differed from the other sites in having Caenidae with the greatest abundance (86.8% cf. 67.2% – 40.5%). The three sites AD, BD and SH were similar in having the least abundance of Hydropsychidae, Leptoceridae and Simuliidae and Run was closer to NR in this respect.

4.2.3 WINTER

4.2.3.1 Description

A total number of 15,557 macroinvertebrates belonging to 61 species or higher taxonomic level was collected from the River Smite in winter (February) 1993/94. Table 4.7 shows the relative abundance of the taxa at family level on the five sampling sites. There were 10 families with $\geq 1\%$ abundance which formed at least 97.5% of the total abundance, whereas 23 families $< 1\%$ comprised at most 2.5% of the total abundance. Caenidae was the dominant taxon of all sites except SH ranging from 39.6% in NR to 55.2% in AD. The dominant taxon of SH was Tubificidae (43.4%).

Natural Riffle was dominated by Caenidae (39.6%) and Simuliidae (22.6%). Elmidae (8.9%), Chironomidae (8%), Leptoceridae (7.9%) and Hydropsychidae (5.6%) were the abundant taxa. Naididae (1.6%), Tubificidae (1.9%), Lumbriculidae (1%) and Asellidae (1.5%) comprised the frequent taxa (Table 4.7). Sialidae, Corticidae, Limnaeidae, Physidae, Planorbidae and Hydracarina were absent the other families were either occasional or rare.

Caenidae (43.8%) and Tubificidae (34.7%) were the dominant taxa of Run, and Naididae (11.8) comprised the abundant taxon. The frequent taxa were Lumbriculidae (1.4%) Asellidae (1%) and Chironomidae (3.9%) (Table 4.7). Erpobdellidae, Hydrobiidae, Valatidae, Ancyliidae and Limnaeidae were absent from the site; other families were either occasional or rare.

The AD site was dominated by Caenidae (55.2%) and Tubificidae (21%), and Chironomidae (12.9%) was the abundant taxon of the site. Naididae (1.2%), Asellidae (2.1%), Hydropsychidae (1.2%), Leptoceridae (1.6%), Elmidae (1.4%) and Simuliidae (1%) formed the frequent taxa (Table 4.7). Piscicolidae, Hydrobiidae, Valvatidae, Ancyliidae, Planorbidae and Hydracarina were absent from the site but other taxa were either occasional or rare.

Caenidae (47.3%) and Chironomidae (18.8%) were the dominant taxa of BD, and Tubificidae (16.9%) and Simuliidae (7.5%) comprised the abundant taxa of the site. Naididae (2.4%), Asellidae (1.6%), Leptoceridae (2%) and Elmidae (1%) formed the frequent taxa (Table 4.7). Glssophoniidae, Piscicolidae, Goeridae, Sialidae, Haliplidae, Tipulidae, Hydrobiidae, Valvatidae, Ancyliidae, Physidae, Planorbidae and Hydracarina were absent but the other taxa were either occasional or rare.

The Shoal site was dominated by Tubificidae (43.4%), Caenidae (26.9%) and Naididae (17%); and Chironomidae (5.7%) was the abundant taxon. The frequent taxa were Asellidae (1%) and Simuliidae (2.2%) (Table 4.7). Goeridae, Sialidae, Haliplidae, Corixidae and Physidae were absent from the site while the other taxa were either occasional or rare.

Table 4.7 Macroinvertebrate family abundance \geq 1% of the sites from the River Smite in winter 1993/94.

Taxa	NR	RUN	AD	BD	SH
Naididae	1.60	11.75	1.17	2.44	16.96
Tubificidae	1.86	34.71	21.01	16.90	43.38
Lumbriculidae	0.99	1.38	0.02	0.20	0.59
Asellidae	1.49	0.96	2.09	1.63	0.98
Caenidae	39.60	43.78	55.22	47.25	26.94
Hydropsychidae	5.60	0.34	1.17	0.81	0.59
Leptoceridae	7.87	0.65	1.62	2.04	0.91
Elmidae	8.86	0.73	1.37	1.02	0.16
Chironomidae	7.96	3.98	12.86	18.74	5.68
Simuliidae	22.59	0.31	0.97	7.54	2.19
Total % Abundance	98.42	98.58	97.51	98.57	98.37

4.2.3.2 Comparison between habitats

The distinctive differences between the NR and the other sites in winter were the greatest abundance of Hydropsychidae (5.6% v. 1.2% – 0.3%), Leptoceridae (7.9% v. 2% – 0.7%) and Simuliidae (22.6% v. 7.5% – 0.3%), and the least abundance of Tubificidae (1.9%). AD differed from the other sites in having the least abundance of Naididae (1.2%) and the greatest abundance of Caenidae (55.2% v. 47.3% – 26.9%). BD differed from the other sites in having the greatest abundance of Chironomidae (18.8% v. 12.9% – 4%). This site also had the second greatest abundance of Simuliidae (7.5%) (NR was in the first place). SH differed from the other sites in having the greatest abundance of Naididae (17% v. 11.8% – 1.2%) and Tubificidae (43.4% v. 34.7 – 1.9%), and also the smallest abundance of Caenidae (27%) and Elmidae (0.2%).

4.2.4 SPRING

4.2.4.1 Description

A total number of 22,529 macroinvertebrate belonging to 65 species or higher taxonomic group was collected from the River Smite in spring 1994. Table 4.8 shows the relative abundance of taxa at family level on five sampling sites. There were 13 families

with $\geq 1\%$ abundance which formed at least 97.8% of the total abundance, whereas, 21 families with $< 1\%$ abundance comprised at most 2.2% of the total abundance.

Natural Riffle was characterised by Naididae (39.5%) and Chironomidae (18.5%) as the dominant taxa, and Caenidae (16.1%) and Elmidae (8.4%) as the abundant taxa. Tubificidae (1.9%), Lumbriculidae (1%), Baetidae (3.1%), Hydropsychidae (2.6%), Leptoceridae (4.1%), Ceratopogonidae (1.2%) and Hydracarina (1.8%) formed the frequent taxa (Table 4.8). In comparison to the other sites, Psychomyiidae was absent from NR but the other taxa were either occasional or rare.

The dominant taxon of Run was Caenidae (57.2%), and Tubificidae (15.5%) and Chironomidae (12.5%) were the abundant taxa. Naididae (2.1%), Lumbriculidae (2.1%), Leptoceridae (2.2%), Elmidae (2.1%), Dytiscidae (1.1%) and Simuliidae (1.6%) comprised the frequent taxa (Table 4.8). Polycentropodidae, Limnophilidae, Tipulidae, Hydrobiidae, Valvatidae and Planorbidae were absent from the site, whereas, the other taxa were either occasional or rare.

Caenidae (71.4%) was the dominant taxon of AD and Tubificidae (16.3%) was the abundant taxon. Naididae (2%), Leptoceridae (2.4%) and Chironomidae (4.4%) formed the frequent taxa (Table 4.8). Baetidae, Simuliidae, Polycentropodidae, Agriidae, Haliplidae, Tipulidae, Hydrobiidae and Planorbidae were absent, whereas the other taxa were either occasional or rare.

Caenidae (56.8%) was the dominant taxon of BD and Elmidae (13.3%), Leptoceridae (9.2%), Tubificidae (5%) and Hydracarina (5%) formed the abundant taxa. Naididae (2.5%), Hydropsychidae (1.1%), Chironomidae (2.1%) and Ceratopogonidae (1.6%) comprised the frequent taxa (Table 4.8). Lumbriculidae, Psychomyiidae, Limnophilidae, Agriidae, Haliplidae, Hydrobiidae, and Planorbidae were absent, the other taxa were either occasional or rare.

The Shoal site was characterised by Caenidae (63.7%) as the dominant taxon and Tubificidae (15.8%) and Elmidae (5.1%) as the abundant taxa. Naididae (2.4%), Leptoceridae (3.5%), Chironomidae (2.3%), Ceratopogonidae (2.3%) and Hydracarina (1.7%) formed the frequent taxa (Table 4.8). Polycentropodidae, Limnophilidae and

Agriidae were absent but the other taxa were either occasional or rare.

Table 4.8 Macroinvertebrate family abundance $\geq 1\%$ of the sites from the River Smite in spring 1994.

Taxa	NR	RUN	AD	BD	SH
Naididae	39.53	2.08	2.03	2.47	2.37
Tubificidae	1.93	15.49	16.28	5.01	15.82
Lumbriculidae	0.95	2.08	0.13	0.00	0.38
Caenidae	16.10	57.23	71.40	56.82	63.65
Baetidae	3.14	0.67	0.00	0.42	0.14
Hydropsychidae	2.56	0.07	0.03	1.09	0.31
Leptoceridae	4.07	2.22	2.37	9.23	3.54
Elmidae	8.38	2.08	0.63	13.27	5.07
Dytiscidae	0.12	1.11	0.08	0.90	0.24
Chironomidae	18.53	12.53	4.43	2.05	2.27
Simuliidae	0.28	1.63	0.00	0.00	0.86
Ceratopogonidae	1.15	0.89	0.73	1.57	2.30
Hydracarina	1.81	0.15	0.47	4.95	1.72
Total % Abundance	98.53	98.22	98.57	97.77	98.68

4.2.4.2 Comparison between habitats

In spring Natural Riffle differed from the other sites in having Naididae (39.5% v. 2.5% – 2%), Baetidae (3.1% v. 0.7% – 0.1%), Hydropsychidae (2.6% v. 1.1% - 0.03%) and Chironomidae (18.5% v. 12.5% – 2.1%) with the greatest abundance, and also Tubificidae (1.9%) and Caenidae (16.1%) with the least abundance. Run site differed from the other sites in having Lumbriculidae (2.1% v. 1% - 0.1%) with the greatest abundance, and Leptoceridae (2.2%) and Hydracarina (0.2%) with the least abundance. AD differed from the other sites in having the greatest abundance of Caenidae (71.4% v. 63.7% – 16.1%), and the least abundance of Hydropsychidae (0.03%) and Elmidae (0.6%). BD differed from the other sites in having Leptoceridae (9.2% v. 4.1% – 2.2%), Elmidae (13.3% v. 8.4% – 0.6%) and Hydracarina (5% v. 1.8% - 0.2%) with the greatest abundances. SH had the second greatest abundance of Caenidae (63.7%). The greater abundance of Tubificidae and Caenidae, and also the lesser abundance of Baetidae and Hydropsychidae made Run, AD, BD and SH the most similar to each other and different from NR.

4.2.5 SUMMARY OF THE ABUNDANCE AND COMPOSITION

In general there was a tendency toward decreasing family richness and the total macroinvertebrate abundance in all sites from summer to winter with both increasing from winter to spring (Table 4.9). The availability of more food, a more favourable environment and less disturbance by spate, etc. possibly provided conditions for the richer macroinvertebrate communities in summer and spring than in winter and autumn.

The results also showed that Natural Riffle had the greatest macroinvertebrate abundance in three seasons except summer and Below Deflector had the least abundance in three seasons except in spring. In summer and spring Shoal had the second greatest macroinvertebrate abundance and in autumn and winter in third place.

In general the macroinvertebrate composition (the most abundant taxa which comprises 79% to 92.5% of the total abundance) of Run and Above Deflector were more consistent through the seasons (except summer) than Natural Riffle, Below Deflector and Shoal. Caenidae was the dominant taxon of all sites in all seasons except NR in spring (Naididae was dominant and Caenidae in third place) and SH in winter (Tubificidae was dominant and Caenidae in second place). Baetidae was the second most abundant taxon of NR in summer whereas it was absent in autumn and winter, and frequent in spring. Simuliidae was amongst the most abundant taxa of NR in all seasons except in spring when it was occasional. Hydropsychidae was amongst the most abundant taxa of NR in autumn and winter whereas it was frequent in summer and spring. Tubificidae was the second most abundant taxon of Run in all seasons followed by Simuliidae in autumn, Naididae in winter and Chironomidae in spring. Tubificidae was also the second abundant taxon of AD in all seasons except in summer (Chironomidae was the second, Asellidae and Tubificidae in the third and fourth place) and followed by Chironomidae in autumn and winter, and none in spring. Taxa composition of BD and SH was different through the seasons except for the dominant taxon (Caenidae).

Table 4.9 Macroinvertebrate abundance and family richness of the sites from the River Smite in different seasons.

Site	Summer		Autumn		Winter		Spring	
	Abun.	Family	Abun.	Family	Abun.	Family	Abun.	Family
NR	10936	34	11381	25	5374	22	11504	28
Run	.*	.*	5470	24	2613	22	1349	23
AD	30606	31	8708	26	4013	23	3839	22
B D	8859	34	1216	21	491	17	1658	20
SH	28537	35	5598	21	3066	22	4179	24
Total	78938	38	32373	36	15557	33	22529	34

* Data were not collected.

4.3 SPECIES DIVERSITY

The species diversity was calculated and the statistical analysis was applied to the data of the River Smite in the same way as Harper's Brook (section 3.3). The ANOVA output for all seasons is shown in Appendix One.

4.3.1 DIFFERENCES IN SPECIES DIVERSITY IN EACH SEASON

Table 4.10 shows the calculated values of H' of each replicate and the mean diversities for each site in summer. NR and AD have the greatest and equal mean diversities, SH has the smallest and BD has the medium mean diversity compared with the others. However, ANOVA indicates that the mean diversities of the sites are not significantly different ($F = 1.58$, d.f. = 3, 8, $P = 0.27$).

Table 4.10 Shannon-Wiener diversity index of 12 samples and the mean diversity of the sites from The River Smite in summer 1993.

Sites	H' of Replicates			Mean H'
NR	2.45	2.67	2.58	2.57
AD	2.68	2.61	2.41	2.57
BD	2.00	2.22	2.40	2.21
SH	0.94	2.24	2.52	1.90

Table 4.11 shows the H' values of the replicates and the mean diversities of the sites in autumn. NR has the greatest and SH the smallest mean diversity values. There is a trend in which the mean diversities of the sites are in decreasing order from NR to SH. However, ANOVA shows the sites are not significantly different ($F = 2.37$, d.f. 4, 10, $P = 0.12$).

Table 4.11 Shannon-Wiener diversity index of 15 samples and the mean diversity of the sites from the River Smite in autumn 1993.

Sites	H' of Replicates			Mean H'
NR	1.78	1.92	1.81	1.84
RUN	1.14	1.60	2.06	1.60
AD	1.76	1.10	1.55	1.47
BD	1.16	0.29	1.95	1.13
SH	0.58	0.94	0.87	0.80

The calculated diversity values of the replicates and the mean diversities of the sites in winter are shown in Table 4.12. Shoal has the greatest and Run the smallest mean diversity, NR is in the second highest place and the mean diversities of AD and BD are equal . The mean diversities of the sites are relatively close to each other and ANOVA indicated that they are not significantly different ($F = 0.83$, d.f. 4, 10, $P = 0.53$).

Table 4.12 Shannon-Wiener diversity index of 15 samples and the mean diversity of the sites from the River Smite in winter 1993\1994.

Site	H' of Replicates			Mean H'
NR	2.17	1.87	1.87	1.97
RUN	0.93	2.04	1.95	1.64
AD	2.13	1.57	1.80	1.83
BD	1.73	1.76	1.94	1.81
SH	1.92	2.44	1.99	2.11

Table 4.13 shows the calculated diversity values of the replicates and the mean diversity values of each site in spring. In this season NR has the greatest and AD the smallest mean diversity, Run is in the second highest place, BD and SH are very close to each other. ANOVA showed that the mean diversity of the sites are significantly different ($F = 4.52$, d.f. 4, 10, $P < 0.05$).

Table 4.13 Shannon-Wiener diversity index of 15 samples and the mean diversity of the sites from the River Smite in spring 1994.

Site	H' of Replicates			Mean H'
NR	2.27	1.84	2.28	2.13
RUN	1.96	1.48	1.67	1.70
AD	1.02	1.15	1.59	1.25
BD	1.25	1.89	1.32	1.49
SH	1.44	1.68	1.56	1.56

4.3.2 DIFFERENCES IN SPECIES DIVERSITY BETWEEN HABITATS

In order to show which sites were different from the others, 95% confidence intervals of the mean diversity of the sites were calculated in the same way as Harper's Brook (section 3.3.2). The 95% confidence was applied only to spring data because the result of ANOVA revealed a significant difference between sites in that season. The result indicated that NR was significantly different from AD and SH. The variation between the three replicates of all sites was relatively similar (the size of error bars were similar) except for SH which has a smaller error bar size (Fig. 4.1).

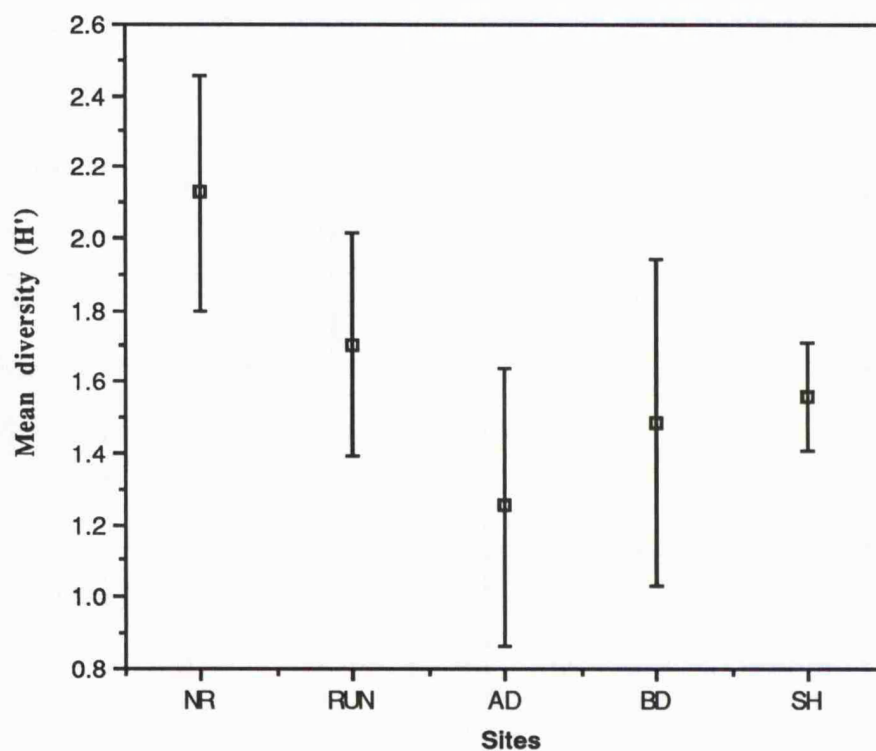


Figure 4.1 Mean diversity (H') and 95% confidence intervals of the sites from the River Smite in spring 1994.

4.3.3 DIVERSITY AND ENVIRONMENTAL VARIABLES

In order to investigate the relationship between the diversity of the invertebrate communities and the depth and velocity of the sample sites, the two measures were plotted against diversity as the dependent variable. Two graphs were used for each pairing of variables in each season, one using the 15 samples and the other using the five sites (except for summer when only 12 samples and four sites were involved).

The results (Table 4.14) indicated that there was no significant relationship between either the depth or the velocity and diversity of either samples or sites in any season, except for depth in winter ($r = 0.82$ (sites), $r = 0.53$ (samples), $p < 0.05$ for both) and velocity in spring ($r = 0.82$ (sites), $r = 0.55$ (samples), $p < 0.05$ for both).

Since the results of ANOVA also did not show significant differences between the sites, it is difficult to interpret the results clearly. The relatively similar environmental conditions of the sites is the most possible reason. In general the Natural Riffle has the coarser substrate (dominated by cobble and boulder), the highest velocity and least depth, and the greatest mean diversity. The substrate of the other sites is finer and relatively similar (gravel and sand or silt). Thus although depth and velocity of these sites (except Run) as the consequences of the deflector are relatively different, the overall mean diversities are either similar or the differences are not great enough to make them significantly different. In other word the deflector has not changed the substrate of the sites and the finer substrate does not produce the more diverse invertebrate communities than coarse substrate does.

Table 4.14 Statistical results of the relationship between depth, velocity and diversity of the sites and samples from the River Smite over four seasons in 1993–1994.

Season		N	Depth		Velocity	
			r	p < 0.05	r	p < 0.05
Summer	Sites	4	0.23	NS	.*	.*
	Samples	12	0.24	NS	.*	.*
Autumn	Sites	5	0.06	NS	0.54	NS
	Samples	15	0.22	NS	0.22	NS
Winter	Sites	5	0.82	S	0.27	NS
	Samples	15	0.53	S	0.06	NS
Spring	Sites	5	0.6	NS	0.82	S
	Samples	15	0.26	NS	0.55	S

N = number of samples or sites, r = correlation coefficient, S = significant, NS = not significant. * Data were not collected.

4.3.4 MULTIVARIATE ANALYSES OF COMMUNITIES

4.3.4.1 DECORANA

The data are treated in the same way as in section 3.3.5.1 and the results of DECORANA analysis are shown in table 4.15. The output of DECORANA for each season is shown in Appendix Two.

Table 5.15 Eigenvalues for the first four axes of DECORANA from the River Smite in four seasons in 1993–1994.

Season	Axis 1	Axis 2	Axis 3	Axis 4
Summer	0.200	0.048	0.022	0.001
Autumn	0.196	0.076	0.031	0.012
Winter	0.174	0.091	0.028	0.016
Spring	0.224	0.081	0.032	0.012

DECORANA analysis shows that the major separation occurs along axis 1 which separates samples of NR from those of AD, BD, SH and Run. Natural Riffle tended to have higher axis 1 sample scores over three seasons whereas it had lower values in spring. In summer the separation between NR and the other sites is clear and none of the other sites shows any clear distinction and their replicate samples are scattered along axis 2 (Fig. 4.2).

In autumn NR is clearly separated from the other sites. The Run site is relatively separated from the others although the separation is not very distinct. Two replicates of AD are also close together. The replicates of the other sites are scattered along axis 2 (Fig. 4.3).

The separation of NR from the other sites was very clear in winter but none of the other sites shows any distinction and their replicates are scattered along axis 2 (Fig. 4.4).

In spring NR is again a distinct site. The replicates of AD are relatively close and separated from the other sites. Two replicates of the Run are also close together. Most of the replicates of all sites are scattered along axis 1 (Fig. 4.5).

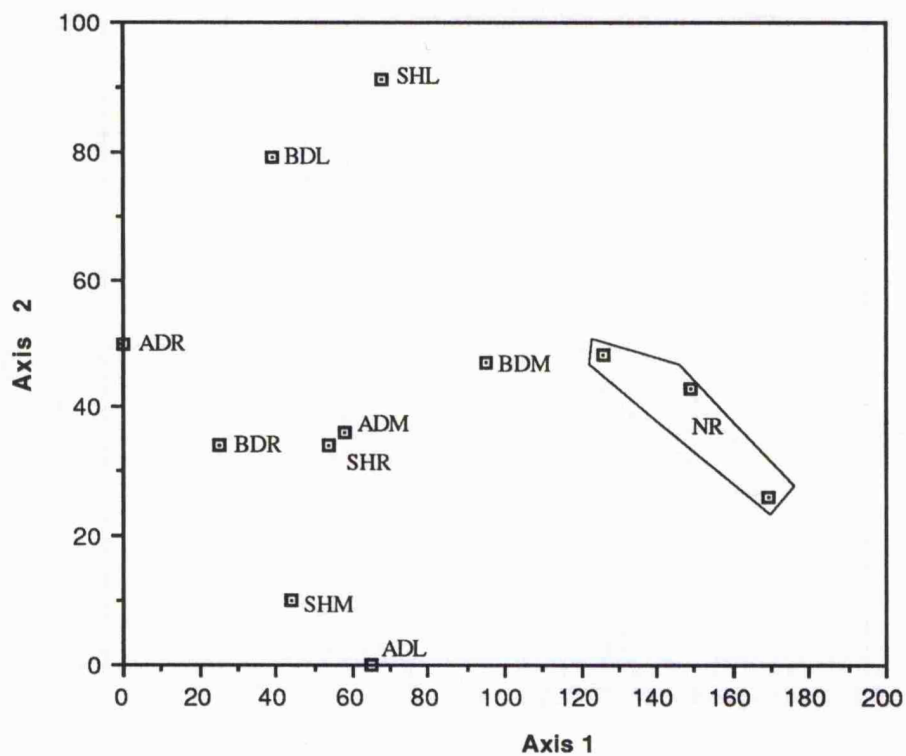


Figure 4.2 DECORANA of samples and sites from the River Smite in summer 1993.

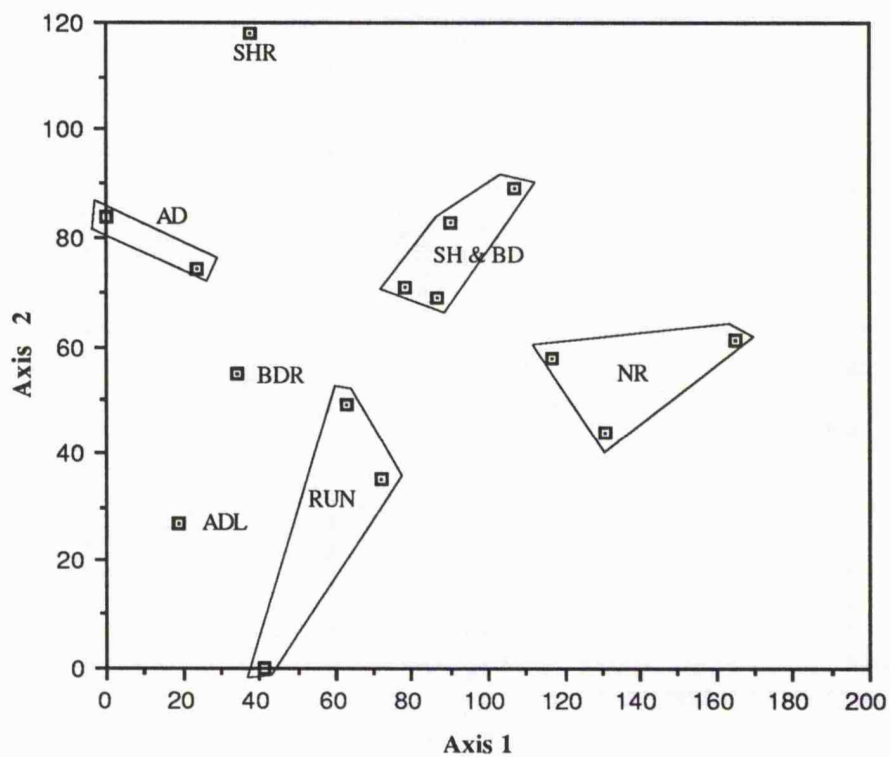


Figure 4.3 DECORANA of samples and sites from the River Smite in autumn 1993.

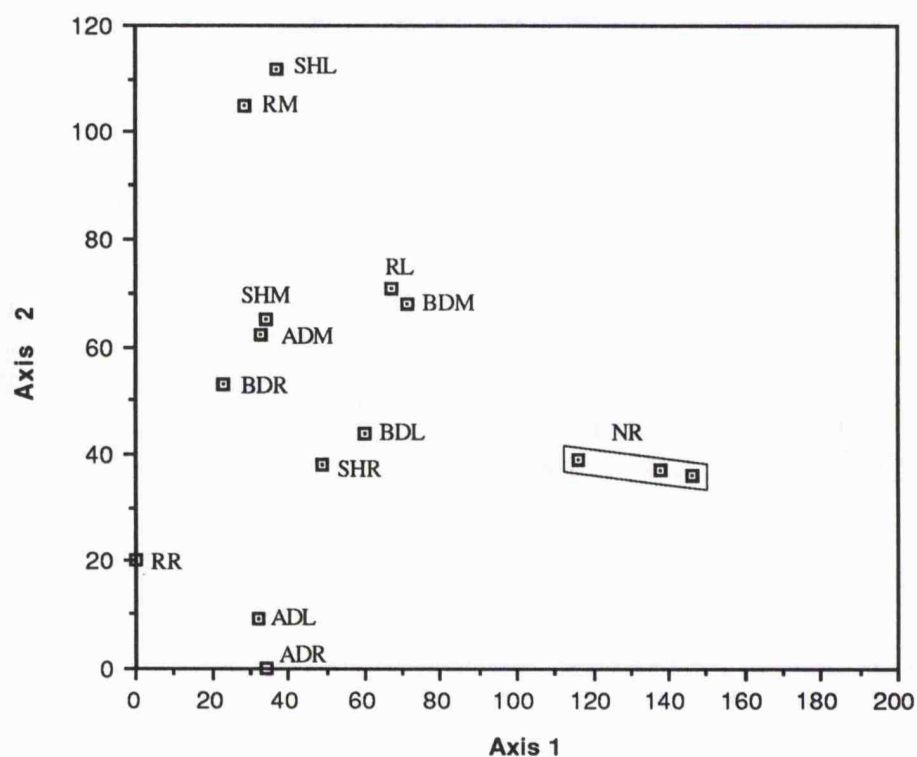


Figure 4.4 DECORANA of samples and sites from the River Smite in winter 1993/1994.

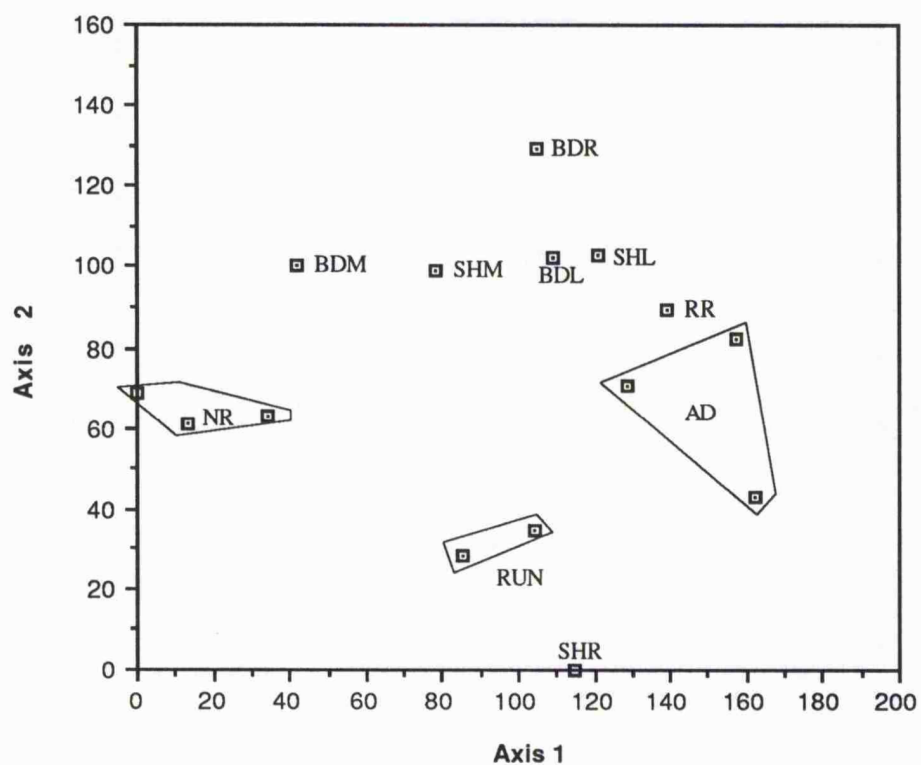


Figure 4.5 DECORANA of samples and sites from the River Smite in spring 1994.

In order to show the extent to which depth and velocity were linked to the DECORANA results the axis 1 sample scores were plotted against each variable for each season separately. In summer there is no significant correlation between axis 1 sample scores and the mean depth of the sites. Data for the velocity of the sites were not collected.

Figures 4.6 and 4.7 show the results for autumn. There is a negative correlation between axis 1 sample scores and the mean depth of the samples ($r = -0.51$, $P < 0.1$). NR was the shallowest site and is located on the upper end of the fitted line and followed by SH. Run and AD were the deepest sites and are located on the lower end of the line and BD is in the middle (Fig. 4.6). The correlation between axis 1 and velocity of the samples is strong and positive ($r = 0.82$, $P < 0.001$). A similar relationship between sites is apparent (Fig. 4.7).

Figures 4.8 and 4.9 show the results for winter. There is a significant negative correlation between axis 1 sample scores and the mean depth of the samples ($r = -0.54$, $P < 0.05$). NR with the smallest depth is positioned on the upper end of the fitted line and the other sites are positioned towards the middle and lower end of the line (Fig. 4.8). There is a strong positive correlation between axis 1 sample scores and the velocity of the samples ($r = 0.81$, $P < 0.001$). A similar relationship between sites is apparent (Fig. 4.9).

Figures 4.10 and 4.11 show the results for spring. The correlation between axis 1 sample scores and the mean depth of the samples is positive but not significant ($r = 0.42$). NR was the shallowest site and is located on the lower end of the fitted line, whereas most of the other sites (except RL and BDM) are relatively close together and located at the centre of the line (Fig. 4.10). There is strong negative correlation between axis 1 sample scores and the velocity of the samples ($r = -0.78$, $P < 0.001$). NR with the highest velocity is positioned on the lower end of the fitted line, whereas AD, BD, SH and Run with lower velocities (except SHM and BDM) are close together and positioned on the upper end of the line (Fig. 4.11).

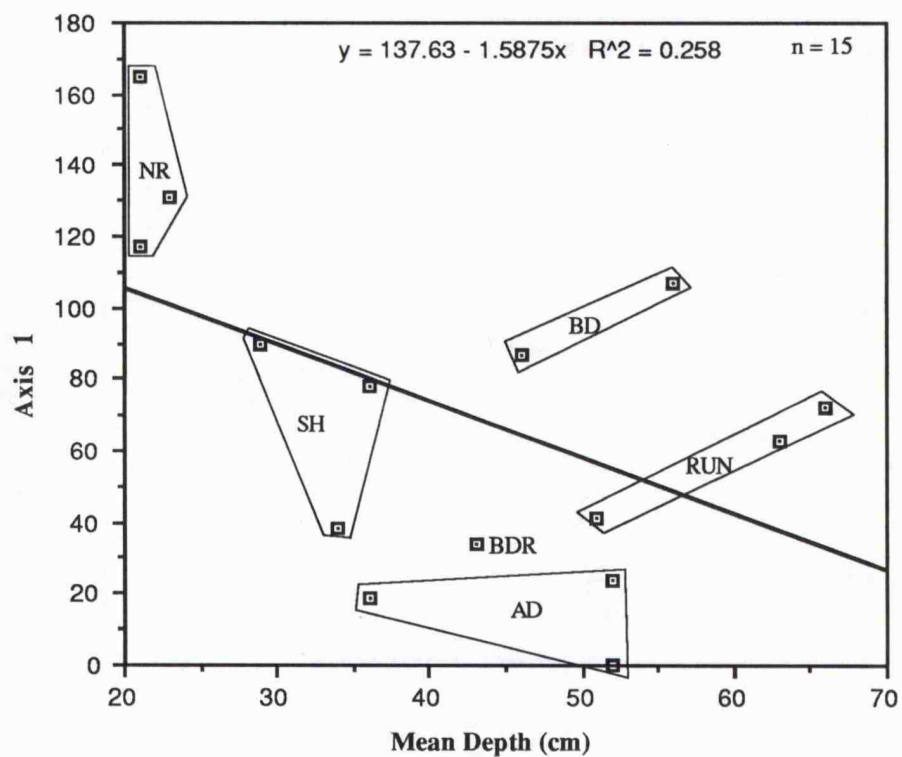


Figure 4.6 Correlation between axis 1 of DECORANA and the mean depth of the sites from the River Smite in autumn 1993.

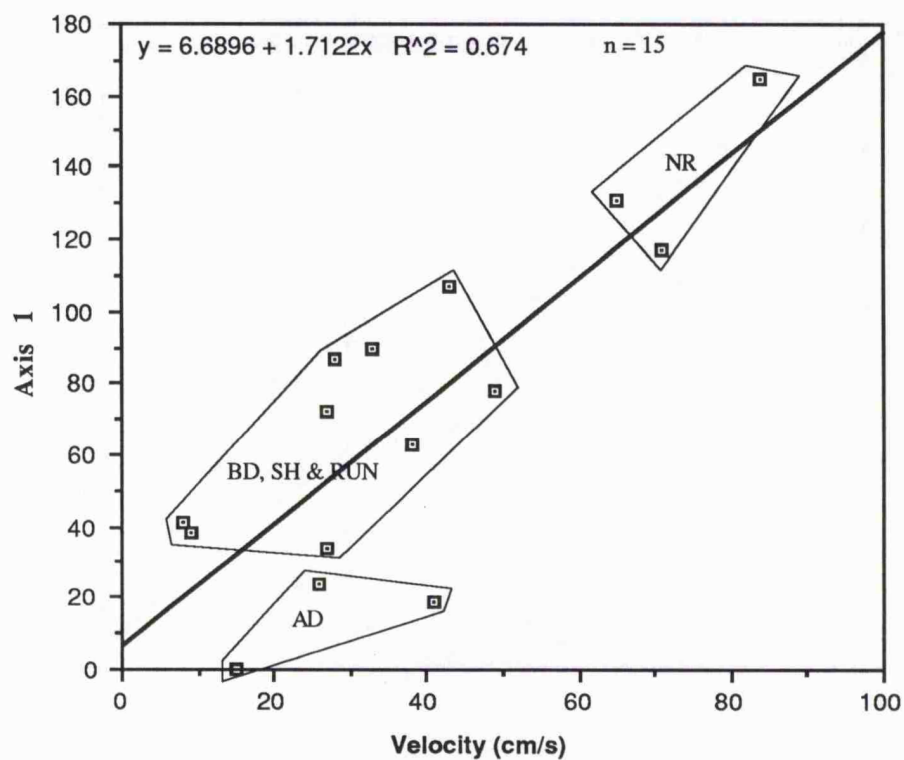


Figure 4.7 Correlation between axis 1 of DECORANA and the velocity of the sites from the River Smite in autumn 1993.

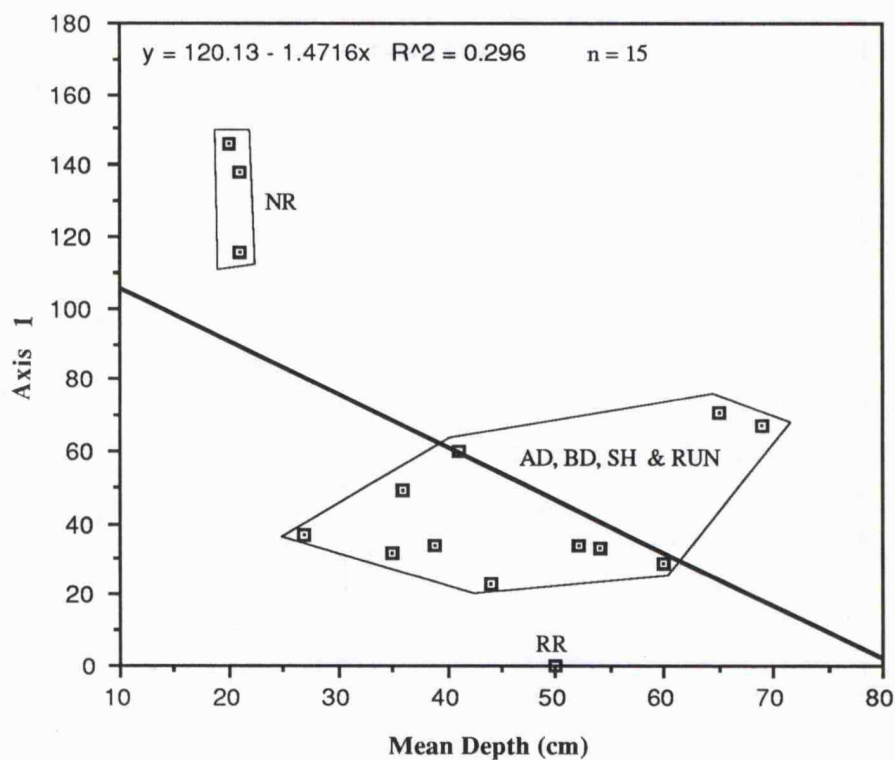


Figure 4.8 Correlation between axis 1 of DECORANA and the mean depth of the sites from the River Smite in winter 1993/1994.

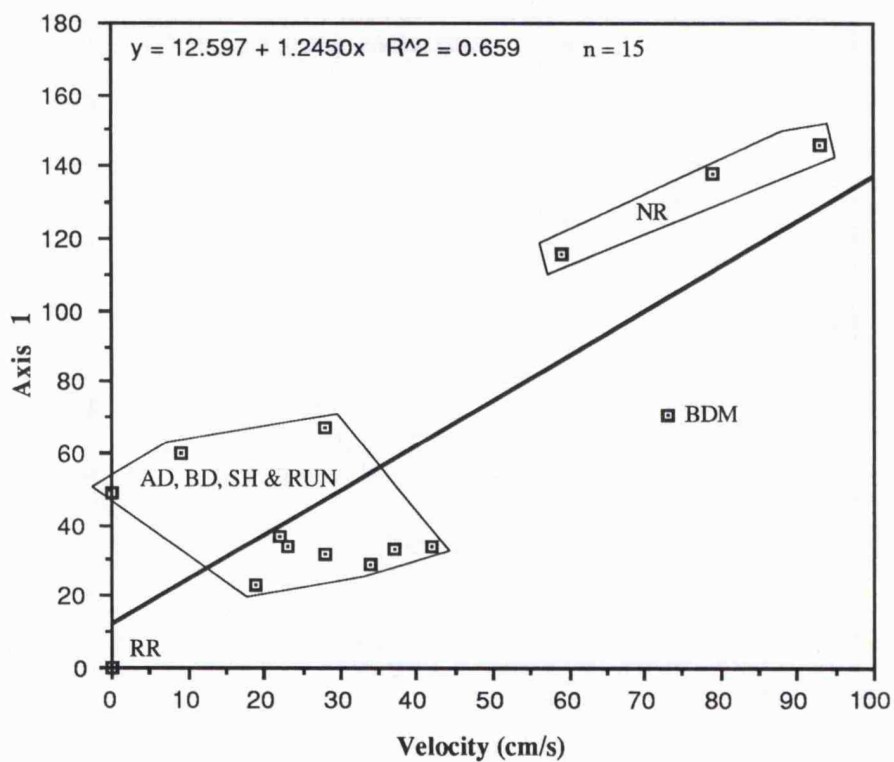


Figure 4.9 Correlation between axis 1 of DECORANA and velocity of the sites from the River Smite in winter 1993/1994.

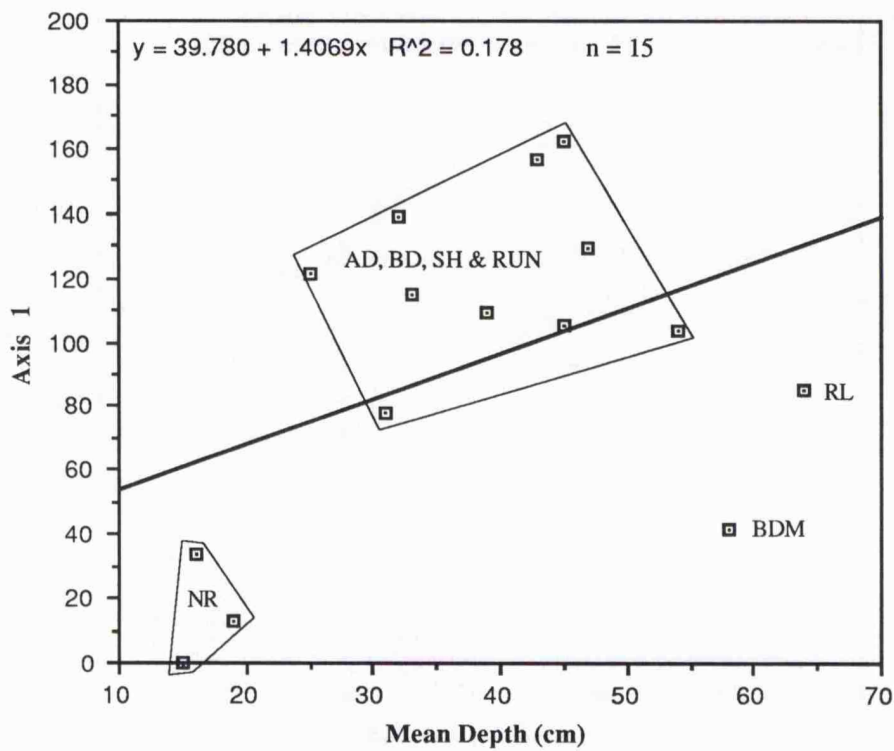


Figure 4.10 Correlation between axis 1 of DECORANA and the mean depth of the sites from the River Smite in spring 1994.

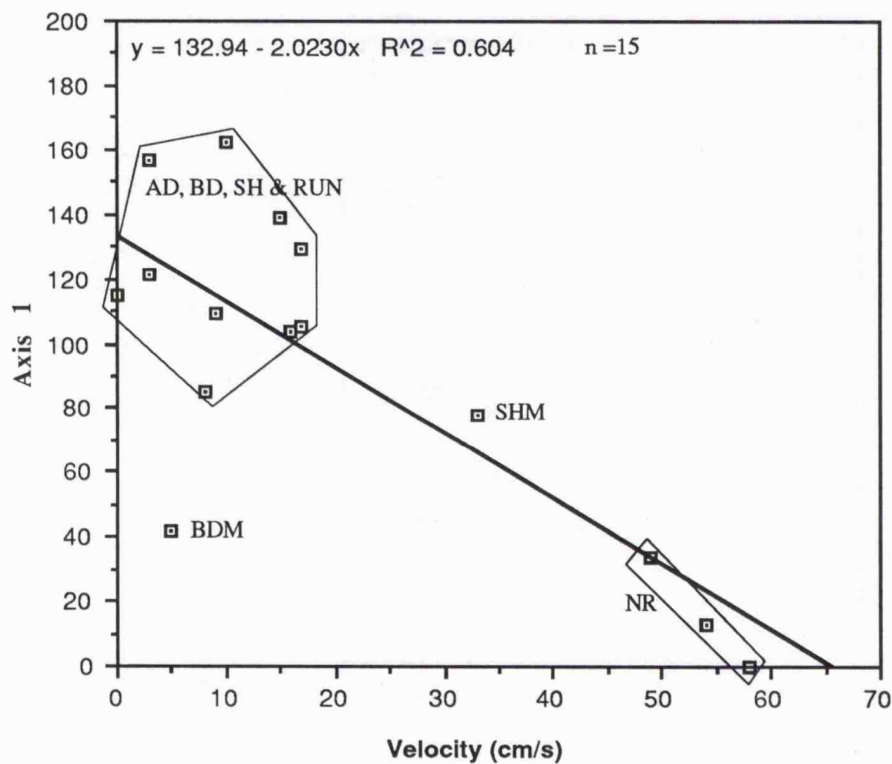


Figure 4.11 Correlation between axis 1 of DECORANA and the velocity of the sites from the River Smite in spring 1994.

4.3.4.2 CONCLUSIONS

It can be concluded that:

- Natural Riffle (with its replicates which were always close together) was clearly separated from AD, BD, SH and Run in the all seasons. In general the replicates of the other sites mixed and did not show them to be distinct sites. NR had the greatest axis 1 sample scores in three seasons and the smallest in spring and the other sites were the opposite.
- There was significant correlation between axis 1 sample scores of DECORANA and the depth and velocity of the sites (except for depth in summer, autumn and spring). As the results of the univariate analysis did not show significant correlation between diversity and depth and velocity (except for depth in winter and velocity in spring), whereas the multivariate analysis (DECORANA) showed significant correlations, it seems the latter is stronger than the univariate analysis and more useful in the interpretation of environmental data with high variability.
- The correlation between axis 1 sample scores and depth of the sites was negative in three seasons which means the deeper the site the smaller axis 1 scores and *vice versa*. This was reversed in spring.
- The correlation between axis 1 sample scores and the velocity of the sites was positive in three seasons which means the greater the flow, the greatest the axis 1 scores and *vice versa* ; again it was reversed in spring.
- The reason for the reversal of the axis one scores' relationship with depth and velocity in spring is possibly related to the fact that there was significant diversity differences in spring indicating considerable differences between the composition of the different sites. The low axis 1 scores of spring possibly reflect this.

4.3.4.3 TWINSPAN

Samples were classified using two-way indicator species analysis. The analyses were carried out for each season separately. The abundance values were transformed to $\log(x+1)$ values before analysis. The pseudospecies cut levels 1, 2 and 3 were chosen and the weights 1, 2 and 1 were selected for levels of pseudospecies, giving double the weight to the second pseudospecies cut level, as this produced greater eigenvalues for each dichotomy at each level of classification and the results corresponded with the DECORANA results. All samples and species were included in the analysis but omission of the rare species did not change the results. The output of TWINSPAN for each season is shown in Appendix Two.

Figures 4.12 – 4.15 show the results of TWINSPAN for each season. In summer the 12 samples divide into group 1 which contains all samples from Natural Riffle (NR) and samples BDM and SHL on the positive side of the dichotomy, and the remaining samples of BD, SH and all AD on the negative side of the dichotomy (Fig. 4.12). Copepoda is the indicator taxon of the dichotomy associated with the latter group. Natural Riffle (group11) separates from BDM and SHL at the second level of classification. *Baetis vernus* (Baetidae), *Simulium* spp. (Simuliidae), *Eukiefferiella*, *Tvetenia* (Chironomidae), *Sphaerium corneum* (Bivalvia) and Lumbriculidae are the indicator taxa of the dichotomy, associated with the Natural Riffle. The other group divides into a group containing ADR and BDL with *Glossiphonia complanata* (Glossiphoniidae) as the indicator species. The other group contains the remaining samples (ADL, ADM, BDR, SHM and SHR) with *Psammoryctes barbatus*, *Euliyodrilus bavaricus* (Tubificidae), *Hydropsyche angustipennis* (Hydropsychidae) and *Orthocladius* (Chironomidae) as the indicator taxa. The classification was stopped at the second level as none of the remaining samples were separated into a distinct site at the next level and this corresponded to the DECORANA result. This was true for the other seasons (except for autumn which classified to the third level).

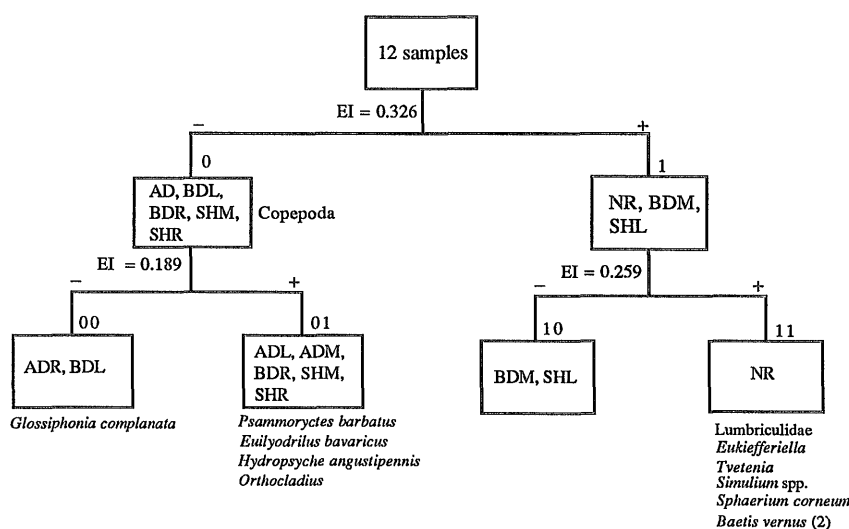


Figure 4.12 Classification of the 12 samples from the River Smite in summer 1993 by Twinspan. EI = Eigenvalues. Indicator species are shown for each group (where present), numbers for pseudospecies (other than 1) are indicated.

Figure 4.13 shows the results of TWINSpan for autumn. The 15 samples divided into two groups at the first level of classification. One group contains all samples from NR and sample SHL with *Oulimnius tuberculatus* as the indicator species. The other group contains all samples from Run, AD, BD, and two samples from SH. Group first separates at the second level of classification to produce groups containing one sample of NR (NRL) and SHL with Tubificidae B, *Helobdella stagnalis* (Glossiphoniidae) and *Erpobdella octoculata* (Erpobdellidae) as the indicator species. The second group contains the two remaining samples of NR with *Orthocladus*, *Eukiefferiella*, *Hydropsyche angustipennis* and *Simulium* spp. as the indicator taxa. At the second level of the classification, the remaining group divides to give a group of two samples of Run and sample ADL with *Athripsodes* spp. (Leptoceridae) as the indicator species. The other group contains the remaining samples. Two samples from AD (ADM and ADR) separate from the remaining samples at the third level of classification with *Paratanytarsus* and *Michrotendipes* (Chironomidae) as the indicator taxa of the dichotomy associated with the AD group. The classification terminated at this level.

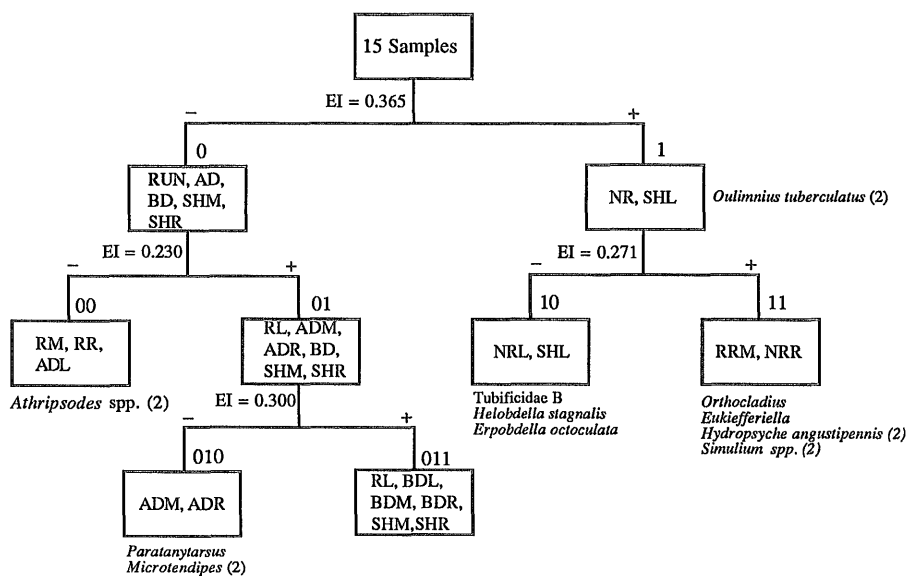


Figure 4.13 Classification of the 15 samples from the River Smite in autumn 1993 by TWINSpan. EI = Eigenvalues. Indicator species are shown for each group (where present), numbers for pseudospecies (other than 1) are indicated.

Figure 4.14 shows the results of TWINSpan for winter. The 15 samples divide into a group containing all samples from Run, AD, BD and SH with *Euilyodrilus hammoniensis* (Tubificidae) as the indicator species. All samples from NR form the other group with *Sphaerium corneum* as the indicator species, no further subdivision of this group is needed. The larger group separated into a group containing two samples of Run and sample SHL with Tubificidae A, Lumbriculidae, *Nais elinguis* (Naididae) and *Psammoryctes barbatus* as the indicator taxa. The other group contain the remaining samples of the sites with *Cricotopus* (Chironomidae) as the indicator taxon of the group.

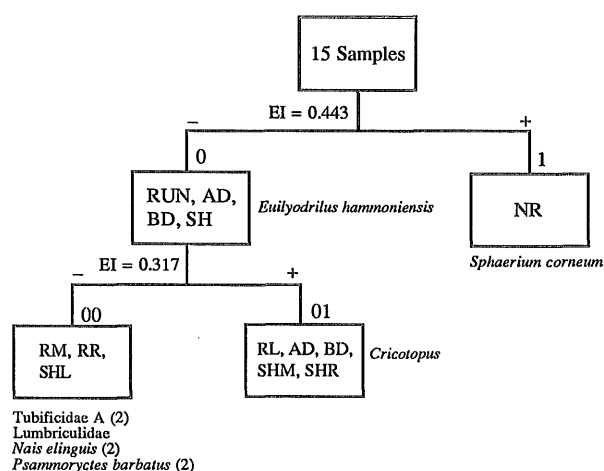


Figure 4.14 Classification of the 15 samples from the River Smite in winter 1993/1994 by TWINSpan. EI = Eigenvalues. Indicator species are shown for each group (where present), numbers for pseudospecies (other than 1) are indicated.

Figure 4.15 shows the results of TWINSpan for spring. The first division of the 15 samples reveals a group containing all samples of NR, samples BDM and SHM with *Hydropsyche angustipennis* as the indicator species. The second group contains the remaining samples. Two samples of NR separate from NRR, BDM and SHM at the second level of classification with *Baetis vernus*, *Hydropsyche angustipennis*, *Elmis aenea* (Elmidae) and Lumbriculidae as the indicator taxa of the dichotomy associated with the first pair. The second group of the first dichotomy divides to produce groups containing two samples from Run, samples BDL and SHR on the one hand and all samples from AD, samples RR, BDR and SHL on the other. *Oulimnius tuberculatus*, *Nais elinguis*, *Potamonectes depressus* (Dytiscidae), *Cricotopus*, Tubificidae A and Lumbriculidae are the indicator taxa of the dichotomy associated with the first group.

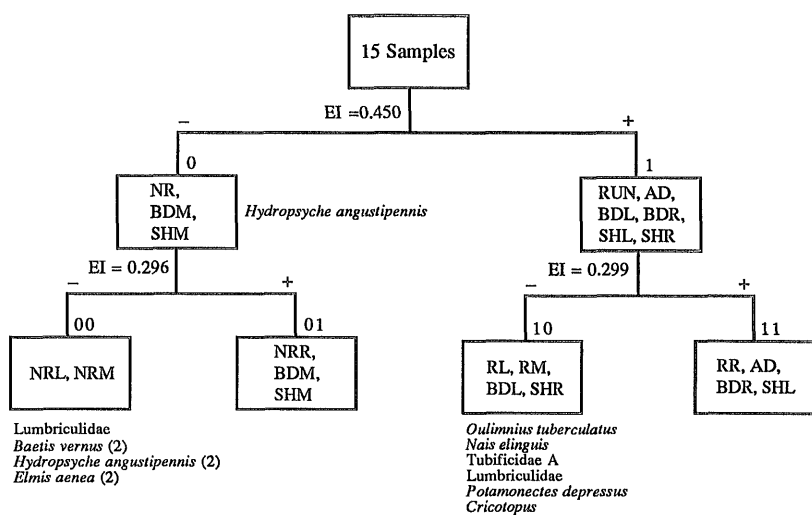


Figure 4.15 Classification of the 15 samples from the River Smite in spring 1994 by TWINSpan. EI = Eigenvalues. Indicator species are shown for each group (where present), numbers for pseudospecies (other than 1) are indicated.

4.3.4.4 CONCLUSIONS

From the above results it can be concluded that Natural Riffle is the only site that clearly differed from the others and this in general corresponds with the DECORANA results. NR has a coarse substrate (Cobbles dominate) and is shallower and faster flowing; the invertebrate community of the site is associated with the fast-flowing and shallow habitats. *Baetis vernus*, *Simulium* spp., *Hydropsyche angustipennis* (in higher abundance), *Eukiefferiella* and *Tvetenia* formed the indicator species of Natural Riffle all of which are normally found on cobble substrate, fast-flowing habitats. The substrates of Run, AD, BD and SH are in general finer (gravel and sand) and relatively similar to each other. Most of the indicator taxa of these sites such as Copepoda, *Psammoryctes barbatus*, *Euliyodrilus hamommoniensis*, *Nais elinguis*, *Paratanytarsus* and *Microtendipes* are normally found on finer substrate and slower-flowing habitats. Although there are some hydrological differences between these sites, the differences are insufficient to make them biologically different. In other word the current deflector can change the hydrology of the sites but it can not change the substrate which is one of the prime determinants for the invertebrate communities composition.

CHAPTER FIVE

**Biodiversity and comparison between
the two rivers**

CHAPTER FIVE

BIODIVERSITY AND COMPARISON BETWEEN THE TWO RIVERS

5.1 INTRODUCTION

The aim of this research was to investigate the effects of two habitat improvement techniques "artificial riffles" and "current deflectors" upon the macroinvertebrate taxonomic abundance and diversity in two lowland rivers. The abundance and diversity of the macroinvertebrate communities of the sites is compared and discussed for each river. The major taxonomic composition of each season and the differences between seasons is also discussed. The effectiveness of each technique is then evaluated and a comparison of the two techniques is made.

5.2 HARPER'S BROOK

5.2.1 BIODIVERSITY DIFFERENCES OF THE SITES

The results of both univariate and multivariate analyses showed that there was a clear distinction between artificial riffles and run sites based on the macroinvertebrate diversity and abundance of the communities. Two artificial riffles (AR2 and AR3) were similar to the natural riffle and one (AR1) was associated with run sites. The natural and the two artificial riffles had relatively greater diversity and richer communities than the run sites. The abundance and taxonomic richness of the sites are compared in table 5.1. AR3 had the greatest absolute abundance in summer, autumn and winter, and second greatest in spring. R1 had the smallest absolute abundance in summer, autumn and spring, and sixth smallest in winter. In spring the Natural Riffle had the greatest absolute abundance, with AR3, AR2, R2, AR1, R3 and R1 following respectively. Thus the abundance values in spring correlated best with the association and separation of the sites which was also found in the results of multivariate analysis. In general there was not a clear trend in number of taxa separating and associating the sites. In poorer sites like AR1 some taxa

occurred with an abundance of just one individual and although this increased the taxonomic richness, it did not necessarily increase the abundance and diversity.

Table 5.1 Abundance and number of taxa of the sites from Harper's Brook in four seasons.

Sites	Abundance				Number of taxa			
	Summer	Autumn	Winter	Spring	Summer	Autumn	Winter	Spring
NR	25086	6934	1928	10113	57	41	38	39
AR1	11296	9271	4362	3865	62	42	38	42
AR2	7401	4508	1371	6632	58	40	43	44
AR3	28302	8735	5164	7370	58	45	42	41
R1	5872	1214	1214	890	51	26	34	32
R2	25966	2456	1271	5340	68	38	39	39
R3	17903	1452	342	3787	59	33	34	34
Total	121826	34570	15652	37997	93	71	59	71

5.2.1.1 Species contribution to diversity

The taxonomic composition which is associated with the natural and two artificial riffles (AR2 and AR3) and those which are associated with the run sites are compared here. In general there were few taxa exclusive to either riffle (NR, AR2 and AR3) or run (R1, R2, R3 and AR1) groups, but the differences between the relative abundance of the taxa made them distinct.

Few similar studies have been undertaken in lowland rivers of the UK. Much of the literature which reports taxonomic composition of communities involves the study of highland rivers in Scotland and Wales with different taxa. In some cases the results did not include the diversity and abundance of all taxa in detail and the taxa were not identified fully (e.g. below Chironomidae). From the 93 taxa which were identified in summer samples those which differed the most between the sites are given below (Table 5.2). Summer samples were chosen because the taxa were the most diverse and abundant in this season: in general the results from summer also corresponded to those of the spring in which the taxa were also diverse and abundant (the list of taxa abundance in spring is in Appendix One).

Table 5.2 Absolute and % abundance of the taxa of the sites from Harper's Brook in summer 1993.

TAXA	NR		AR1		AR2		AR3		R1		R2		R3	
	Abso.	%	Abso.	%	Abso.	%	Abso.	Abso.	Abso.	%	Abso.	%	Abso.	%
Naididae	480	1.9	92	0.8	41	0.6	1.1	300	36	0.6	345	1.3	339	1.9
Tubificidae	138	0.6	942	8.3	384	5.2	3.8	1078	1231	21.0	1023	3.9	547	3.1
Lumbriculidae	0	0.0	0	0.0	10	0.1	0.2	50	0	0.0	4	0.0	0	0.0
Hirudinea	160	0.6	269	2.4	44	0.6	0.9	245	7	0.1	44	0.2	70	0.4
<i>Gammarus pulex</i>	222	0.9	31	0.3	116	1.6	1.7	488	6	0.1	50	0.2	96	0.5
<i>Asellus</i> spp.	1	0.0	60	0.5	28	0.4	0.8	238	240	4.1	221	0.9	348	1.9
<i>Caenis luctuosa</i>	860	3.4	233	2.1	142	1.9	1.0	282	443	7.5	1845	7.1	883	4.9
<i>Caenis horaria</i>	0	0.0	0	0.0	0	0.0	0.0	0	25	0.4	125	0.5	8	0.0
<i>Baetis scambus</i>	932	3.7	25	0.2	238	3.2	1.6	464	2	0.0	34	0.1	0	0.0
<i>Baetis rhodani</i>	309	1.2	0	0.0	75	1.0	0.0	4	0	0.0	50	0.2	0	0.0
<i>Baetis buceratus</i>	10	0.0	0	0.0	3	0.0	0.0	0	0	0.0	0	0.0	0	0.0
<i>Baetis vernus</i>	184	0.7	32	0.3	134	1.8	1.0	272	0	0.0	62	0.2	67	0.4
<i>Centroptilum luteolum</i>	0	0.0	16	0.1	3	0.0	0.0	0	0	0.0	6	0.0	27	0.2
<i>Procladius bifidus</i>	0	0.0	0	0.0	0	0.0	0.0	0	0	0.0	16	0.1	0	0.0
<i>Ephemerella ignita</i>	109	0.4	0	0.0	48	0.6	0.2	68	0	0.0	0	0.0	8	0.0
<i>Hydropsyche angustipennis</i>	594	2.4	100	0.9	827	11.2	14.9	4214	0	0.0	94	0.4	0	0.0
<i>Hydropsyche pellucidula</i>	370	1.5	2	0.0	25	0.3	0.6	156	0	0.0	0	0.0	0	0.0
<i>Hydroptila</i> spp.	156	0.6	16	0.1	26	0.4	1.2	338	0	0.0	48	0.2	10	0.1
<i>Athripsodes aterrimus</i>	12	0.0	7	0.1	6	0.1	0.0	12	36	0.6	208	0.8	197	1.1
<i>Holocentropus dubius</i>	0	0.0	0	0.0	2	0.0	0.0	13	0	0.0	6	0.0	1	0.0
<i>Agria splendens</i>	6	0.0	20	0.2	0	0.0	0.0	8	8	0.1	33	0.1	0	0.0
Coenagrionidae	0	0.0	0	0.0	0	0.0	0.0	0	2	0.0	4	0.0	29	0.2
<i>Sialis lutaria</i>	0	0.0	49	0.4	1	0.0	0.0	0	66	1.1	21	0.1	11	0.1
Elmidae	1064	4.2	83	0.7	260	3.5	4.6	1290	12	0.2	132	0.5	96	0.5
Dytiscidae	1	0.0	100	0.9	1	0.0	0.0	7	49	0.8	105	0.4	356	2.0
Halipidae	5	0.0	55	0.5	2	0.0	0.1	26	22	0.4	70	0.3	20	0.1
<i>Sigara dorsalis</i>	0	0.0	46	0.4	3	0.0	0.0	0	24	0.4	56	0.2	54	0.3
<i>Tanytarsus</i>	1978	7.9	1748	15.5	103	1.4	5.5	1546	1219	20.8	3932	15.2	2570	14.4
<i>Paratanytarsus</i>	1255	5.0	2838	25.1	912	12.3	6.5	1851	595	10.1	6917	26.7	2541	14.2
<i>Rheotanytarsus</i>	74	0.3	51	0.5	18	0.2	2.1	599	0	0.0	0	0.0	0	0.0
<i>Cladotanytarsus</i>	26	0.1	229	2.0	96	1.3	0.1	42	15	0.3	107	0.4	0	0.0
<i>Microtendipes</i>	0	0.0	2207	19.5	151	2.0	0.3	87	349	5.9	2203	8.5	3998	22.4
<i>Stictochironomus</i>	0	0.0	482	4.3	361	4.9	0.1	42	14	0.2	2303	8.9	2893	16.2
<i>Cryptochironomus</i>	0	0.0	76	0.7	0	0.0	0.3	87	0	0.0	0	0.0	0	0.0
<i>Polypedium</i>	0	0.0	0	0.0	0	0.0	0.0	0	69	1.2	172	0.7	21	0.1
<i>Micropsectra</i>	0	0.0	0	0.0	0	0.0	0.0	0	0	0.0	98	0.4	0	0.0
<i>Thienemannimyia</i>	1262	5.0	304	2.7	159	2.1	10.4	2933	0	0.0	715	2.8	173	1.0
<i>Ablabesmyia</i>	0	0.0	51	0.5	0	0.0	0.0	0	30	0.5	172	0.7	187	1.0
<i>Macropelopia</i>	1426	5.7	127	1.1	0	0.0	0.0	0	30	0.5	258	1.0	289	1.6
<i>Procladius</i>	0	0.0	76	0.7	0	0.0	0.0	0	233	4.0	297	1.1	76	0.4
<i>Orthocladius</i>	448	1.8	127	1.1	277	3.7	3.6	1028	49	0.8	637	2.5	187	1.0
<i>Eukiefferiella</i>	1868	7.4	0	0.0	286	3.9	8.6	2421	0	0.0	49	0.2	0	0.0
<i>Tvetenia</i>	4374	17.4	0	0.0	455	6.1	8.0	2269	0	0.0	98	0.4	0	0.0
<i>Cricotopus</i>	2140	8.5	101	0.9	762	10.3	6.6	1869	8	0.1	992	3.8	49	0.3
<i>Potthastia</i>	51	0.2	25	0.2	0	0.0	0.2	69	15	0.3	49	0.2	434	2.4
<i>Prodiamesa</i>	26	0.1	101	0.9	114	1.5	0.0	0	0	0.0	371	1.4	0	0.0
<i>Simulium</i> spp.	1592	6.3	33	0.3	973	13.1	10.3	2908	4	0.1	100	0.4	8	0.0
Other Diptera	909	3.6	45	0.4	114	1.5	0.7	205	16	0.3	128	0.5	60	0.3
Mollusca	108	0.4	57	0.5	33	0.4	0.2	68	61	1.0	130	0.5	37	0.2
Hydracarina	1832	7.3	326	2.9	139	1.9	2.5	700	16	0.3	582	2.2	256	1.4
Microcrustacea	32	0.1	112	1.0	29	0.4	0.0	12	938	16.0	998	3.9	921	5.2
Hydra	72	0.3	0	0.0	0	0.0	0.0	12	0	0.0	0	0.0	0	0.0

Gammarus pulex occurred on all sites but was more abundant on riffles (1.7% – 0.9%) than runs (0.5% – 0.1%). Friberg *et al.* (1994) in their study in the River Gelsa in Denmark, found that *G. pulex* was the most abundant species on restored gravelly substrate and highly abundant in control reaches (presumably riffle, but not stated in the original paper) with the mean density of 839 m⁻² and 534 m⁻² respectively. In the study by Harper *et al.* (1994) of the Rivers Wensum and Ivel, *G. pulex* was included in the taxa which were found in both natural and artificial riffles.

Baetis scambus had 3.7% to 1.6% abundance in NR, AR2 & AR3, compared with its absence from two runs, and abundance of only 0.1% in R2 and 0.2% on AR1. *B. rhodani* also occurred in NR and AR2 with 1.2% – 1% abundance and was absent from the other sites except R2 (0.2%). Species of *Baetis* have been frequently reported in the riffle sites and fast-flowing habitats in higher abundance by many authors. For example the total mean density of *Baetis rhodani* was 19,000 m⁻² in riffles compared with 9,147 m⁻² in pools in the study by Armitage (1976), in the River Tees. A comparative study of the macroinvertebrates of riffles and pools in the rivers Elan and Wye by Scullion *et al.* (1982) revealed that mean densities of two species of *Baetis* were clearly higher in riffles than pools. The mean density of *B. rhodani* was 2,572 m⁻² in riffles and 120 m⁻² in pools in the unregulated River Wye, and 34 m⁻² in riffles and 12 m⁻² in pools in the impounded River Elan where the hydraulic differences between these habitats were lessened. The mean density of *B. scambus* was 946 m⁻² in riffles and 718 m⁻² in pools in the River Wye.

Hydropsyche angustipennis comprised 14.9% of the total abundance in AR3, 11.2% in AR2 and 2.4% in NR compared with its absence from two run sites and scarcity on the others (0.4% on R2 and 0.9% in AR1). *H. pellucidula* was 1.5% on NR, 0.3% on AR2 and 0.6% on AR3 but absent from all runs including AR1. Larvae of different species of *Hydropsyche* which typically construct their nets in rapidly flowing waters, are often found in large numbers on the moss or algae covered surfaces of stones or in

crevices between and beneath stones in spring and summer (Edington & Hildrew, 1981). They are reported to be occurred in riffle habitats by many authors. For example in the study by Brooker & Morris (1980b) *H. siltalai* and *H. pellucidula* comprised the principal species recorded in the Rheidol catchment with maximum densities of 1,155 m⁻² and 355 m⁻² respectively. All samples in this study were collected across the riffle sites. *H. pellucidula* was also recorded at all sites of the study by Brooker & Morris (1980a) in the River Wye with a maximum density of 890 m⁻². Again all samples in this study were collected across riffle sites. *H. angustipennis* and *H. pellucidula* were also found in the artificial riffles in the rivers Wensum and Ivel, studied by Harper *et al.* (1994). *H. pellucidula* was also found with mean densities of 90 m⁻² and 965 m⁻² in restored and control reaches in the River Gelsa in Denmark (Friberg *et al.*, 1994).

Elmis aenea and *Oulimnius tuberculatus* were the most abundant Coleoptera which together comprised 4.6% – 3.5% of the taxa of riffle sites and 0.7% – 0.2% of the run sites. These species were also found in high densities elsewhere: *O. tuberculatus*, 4,245 m⁻² and *E. aenea*, 400 m⁻² on riffle sites in the lower reaches of the River Wye (Brooker & Morris, 1980a). *E. aenea* was found in the restored and control reaches in the River Gelsa (Friberg *et al.*, 1994) with mean densities of 487 m⁻² and 346 m⁻² respectively. Both species were also found in the artificial riffles of the rivers Wensum and Ivel (Harper *et al.*, 1994).

Thienemannimyia (Chironomidae; Tanypodinae) are characteristic of riffles whereas *Ablabesmyia* and *Procladius* of the same subfamily are occurred in runs. *Thienemannimyia* comprised 10.5% – 2.1% of the total abundance of riffle taxa, was absent from R1 and had 2.8% abundance in R2 and 1% in R3. *Ablabesmyia* was absent from riffles but occurred with 1% – 0.5% abundance in run sites. *Procladius* was also absent from riffles but had 4% – 0.4% abundance in runs. In the study of the macroinvertebrates of riffle fauna in the River Wye (Brooker & Morris, 1980a), *Thienemannimyia* had a density of 330 m⁻² and was included in the most abundant

Tanypodinae of the upland reach site (W3 a) whereas *Procladius*, with a density of 925 m⁻² occurred in the lowland reach site (W12).

Eukiefferiella, *Tvetenia* and *Cricotopus* (Chironomidae; Orthocladinae) were also characteristic of riffle sites, because they were always found in much greater abundance in these sites while they were either absent or with lower abundance in runs. *Eukiefferiella* had 8.6% – 3.9% abundance on riffles while was absent or rare on runs (R2 with 0.2% abundance). *Tvetenia* had 17.4% – 8% abundance in riffles while it was almost absent from runs (R2 with 0.4% abundance) and *Cricotopus* had 10.3% – 6.6% abundance in riffles and 3.8% – 0.1% in runs. In the study by Brooker & Morris (1980a) *Eukiefferiella* with a maximum density of 1,640 m⁻² and *Cricotopus* with a maximum density of 1,545 m⁻² were the principal taxa of the subfamily on riffle sites. The former was more abundant on sites in the upper reaches and the latter on sites in the lower reaches.

Simulium occurred with much greater abundance in riffles than runs. It comprised 13.1% – 6.3% of the taxa of riffles while was absent from R3 and had 0.4% – 0.1% abundance in the other runs. Species of *Simulium* have been reported to be occurred in fast-flowing and coarse substrate by many authors. For example, they formed > 18% of the fauna of the upper reach sites (W1, W4 and W6) of the River Wye (Brooker & Morris, *op. cit.*). They have been recorded in the taxa of artificial riffles of the rivers Wensum and Ivel (Harper *et al.*, *op. cit.*). In the River Tees (Armitage, 1976) Simuliidae were found in greater abundance in riffles (Total of 4,710 individuals) than pools (total of 222 individuals).

Tanytarsus and *Paratanytarsus* (Chironomidae; Chironominae) were the most abundant chironomids which occurred in this study with much greater abundance in runs than in riffles and were found to be characteristic of the slower-flowing and finer substrate habitats. Together they comprised 42% – 28% of the total abundance of the taxa in run sites and 13.7% – 12% in the riffle sites. *Microtendipes* (Chironominae) was also

found to be characteristic of run sites. It comprised 22.4% – 5.9% of the total abundance of the taxa of runs, was absent from NR, and had 2% to 0.3% abundance on AR2 and AR3. *Polypopedilum* (Chironominae) was only found in runs (1.2% – 0.1%) and was absent from all riffle sites. Thus all four genera described above are characteristic of slower-flowing and finer substrate habitats.

Microcrustacea including Copepoda and Cladocera were found in much greater abundance in run sites than riffles. They were 16% – 3.9% in runs (1% on AR1) while they were absent from AR3 and 0.4% – 0.1% in NR and AR2.

Hence there are relative similarities between the results of this study and those of other authors. Although some of the other studies did not involve habitat improvement techniques, their results confirm that the major taxa in artificial riffle sites found in this study are similar to those of natural riffles. This means in general that the artificial riffles of Harper's Brook are comparable to the natural riffles in their colonisation by similar invertebrate communities.

5.2.2 BIODIVERSITY DIFFERENCES IN SEASONS

The overall abundance of the taxa of the sites and taxonomic richness varied between seasons and decreased from summer to winter and then increased in spring. In general the macroinvertebrate abundance decreased about fourfold in autumn compared with summer, twofold in winter compared with autumn and increased twofold in spring. The number of taxa decreased from 93 to 71 in autumn, by a further 12 in winter and then increased by 12 in spring (Table 5.1).

As the environmental conditions change in different seasons the macroinvertebrate composition and the abundance varied. In autumn the major differences were the absence of some taxa notably Chironomidae such as; *Cladotanytarsus* (2% – 0%), *Ablabesmyia* (1% – 0%), *Macropelopia* (5.7% – 0%), *Procladius* (4% – 0%) and *Eukiefferiella* (8.6% – 0%). *Hydropsyche siltalai* was another species of the genus *Hydropsyche* which had a greater relative abundance in riffles (3.4% – 1.6%) than on runs (0.1%) in autumn while it

was absent in summer. This species is in the first instar in summer (Edington & Hildrew, 1981) which is possibly why it was not found in the summer taxa of Harper's Brook.

Some taxa occurred with greater abundance in winter than in autumn. These were: *Baetis rhodani* (22% – 0% v. 0.2% – 0%), *Tanytarsus* (1.9% – 0.1% v. 0.2% – 0%), *Paratanytarsus* (11.4% – 0.7% v. 2.3% – 0%), *Orthocladius* (14% – 0.4% v. 1.6% – 0.2%) and Hydracarina (6.1% – 0% v. 1.2% – 0%). As *Baetis rhodani* is basically bivoltine with overwintering and summer generations (Elliott *et al.*, 1988) it is appropriate for it to be recorded in greater abundance in winter than autumn.

Most of the taxa were found in greater abundance in spring than winter due to better environmental conditions and the emergence of larvae from their overwintering eggs. Naididae (8.6% – 1.1%), *Baetis scambus* (40.4% – 17.3% in riffles and 0.4% – 0% in runs) and *Hydroptila* spp. (5.8% – 0.5%) were found in greater abundance in spring than any other season. According to Learner *et al.* (1987) naidid worms are most abundant during summer months. This is to be expected because both growth rate and asexual reproduction are stimulated by higher temperature and a plentiful food supply. However, they found this was not true for *Nais elinguis* which was most abundant in winter and spring. *N. elinguis* formed more than 95% of the Naididae found in spring in this study and this corresponds to the Learner *et al.* (*op. cit.*) study. *Baetis scambus* is a bivoltine species with overwintering eggs (Elliott *et al.*, *op. cit.*) and the larvae are abundant during spring and summer. *Hydroptila* spp. were not recorded in autumn taxa, but were present in two sites (AR2, 1.3% and AR1, 0.1%) in winter and were rare in summer. The larvae of *Hydroptila* are very small and the final instar is markedly different from the first four instars which are caseless. "They found in all types of waterbody except small pools and temporary waters, the larvae feed on periphyton and fine organic particles and they are most likely to be encountered at the final instar when the larvae exhibits a distinctive seed-like cases which in most genera is strongly flattened laterally" (Wallace *et al.*, 1990). Thus it is that the final instar which is easily recognised in spring and this is a possible reason for the *Hydroptila* spp. being the most abundant in spring. *H. siltalai* was also present in riffles in spring.

5.2.3. THE EFFECTIVENESS OF ARTIFICIAL RIFFLES

From the variety of habitat improvement devices and river restoration techniques which are used to ameliorate the adverse effects of river channelisation, riffle reinstatement or artificial riffles have been reported to be one of the most effective techniques for the improvement of habitats in streams and rivers (Edwards 1984, Harper *et al.* 1994). Despite the importance of this technique, and the fact that there are about forty rivers in England which have been improved with artificial riffles (NRA unpublished information) very little has been written regarding the consequences. The study of the artificial riffles of Harper's Brook is possibly the first detailed investigation of this habitat improvement technique in the UK.

The results of this study have proven the effectiveness of artificial riffles in improving the habitats and increasing the abundance and diversity of macroinvertebrates similar to a natural riffle. The overall results of ANOVA, Regression, DECORANA and TWINSpan have shown that artificial riffles AR2 and AR3 were associated with natural riffle, and artificial riffle AR1 with run sites. This means that the riffle reinstatement, as an habitat improvement technique was successful in improving the hydrology and biology of some sites (AR2 and AR3) close or similar to a natural site (NR). In the case of the other artificial riffle (AR1) with a poor hydrological condition (deep and sluggish), however, the technique was not successful despite the deposition of coarse substrate. In a natural river it is the velocity that controls the substrate, because it is under the control of the discharge. As we change the substrate by constructing an artificial riffle, we create higher velocity and consequently improved conditions for biota, but depth is another important factor linked to velocity in controlling the biology of the river. Thus in rehabilitation we have "good" riffles which are shallower and "bad" riffles which are deeper.

5.3 THE RIVER SMITE

5.3.1 BIODIVERSITY DIFFERENCES OF THE SITES

Although the results of univariate analysis only showed a significant difference

between the sites in spring, there are some differences in mean diversity and abundance of the sites. The results of multivariate analysis also showed a clear distinction between the Natural Riffle and the other sites in all seasons. Natural Riffle had greater mean diversity than the other sites in all seasons except winter when the Shoal had a slightly greater value (2.11 v. 1.97). The abundance and taxonomic richness of the sites are compared in table 5.3. Natural Riffle had the greatest abundance in autumn, winter and spring, but was third in summer. BD had the smallest abundance in summer, autumn and winter, and was in the fourth place in spring when the Run had the smallest abundance. In summer AD and SH had the greater abundances which were about threefold that of the NR. The large number of *Caenis luctuosa* (9,944 on AD and 17,374 on SH) and *Asellus* spp.(2,721 on AD and 1,696 on SH) in comparison with NR (2,807 of the former and 257 of the latter), was possibly the reason for greater abundances in these sites. In general AD had the second greatest abundance in autumn and winter and SH was in third place but in spring the sequence was reversed. AD also had the greatest number of taxa in summer, autumn and winter, but was in third place (NR first and SH second) in spring. There was no other clear trend in the number of taxa between the sites.

Table 5.3 Abundance and number of taxa of the sites from the River Smite in four seasons.

Sites	Abundance				Number of taxa			
	Summer	Autumn	Winter	Spring	Summer	Autumn	Winter	Spring
NR	10936	11381	5374	11504	58	37	37	45
Run	-*	5470	2613	1349	-*	36	41	42
AD	30606	8708	4013	3839	65	44	44	36
BD	8859	1216	491	1658	60	34	26	33
SH	28537	5598	3066	4179	61	40	40	40
Total	78938	32373	15557	22529	79	63	61	65

* Data were not collected.

5.3.1.1 Species contribution to biodiversity

Since in general NR differed most from the other sites, the taxonomic composition differences between these two groups are discussed here. The data from summer are chosen for comparison because the abundance and richness of the taxa were the greatest in

Table 5.4 Absolute and % abundance of taxa of the sites from the River Smite in summer 1993.

Taxa	NR		AD		BD		SH	
	Abso.	%	Abso.	%	Abso.	%	Abso.	%
Naididae	25	0.2	497	1.6	61	0.7	88	0.3
Tubificidae	522	4.8	1847	6.0	387	4.4	1241	4.3
Lumbriculidae	74	0.7	49	0.2	5	0.1	7	0.0
Hirudinea	56	0.5	53	0.2	50	0.6	88	0.3
<i>Gammarus pulex</i>	18	0.2	48	0.2	19	0.2	9	0.0
<i>Crangonyx pseudogracilis</i>	0	0.0	6	0.0	4	0.0	11	0.0
<i>Asellus</i> spp.	257	2.4	2721	8.9	1148	13.0	1697	5.9
<i>Caenis luctuosa</i>	2807	25.7	9944	32.5	3784	42.7	17374	60.9
<i>Baetis scambus</i>	0	0.0	4	0.0	1	0.0	0	0.0
<i>Baetis vernus</i>	1374	12.6	66	0.2	26	0.3	32	0.1
<i>Cloeon dipterum</i>	0	0.0	18	0.1	0	0.0	0	0.0
<i>Centroptilum luteolum</i>	0	0.0	18	0.1	0	0.0	22	0.1
<i>Cetoptilum pennulatum</i>	0	0.0	0	0.0	12	0.1	0	0.0
<i>Hydropsyche angustipennis</i>	376	3.4	103	0.3	28	0.3	26	0.1
<i>Hydroptila</i> sp.	7	0.1	44	0.1	6	0.1	26	0.1
<i>Athripsodes</i> spp.	911	8.3	588	1.9	340	3.8	720	2.5
<i>Coenagrion</i> sp.	3	0.0	100	0.3	20	0.2	20	0.1
<i>Sialis lutaria</i>	0	0.0	94	0.3	26	0.3	19	0.1
Elmidae	1305	11.9	636	2.1	579	6.5	1250	4.4
Dytiscidae	0	0.0	93	0.3	56	0.6	30	0.1
<i>Haliphus</i> sp.	14	0.1	92	0.3	64	0.7	36	0.1
<i>Sigara dorsalis</i>	37	0.3	252	0.8	37	0.4	34	0.1
<i>Tanytarsus</i>	278	2.5	1533	5.0	532	6.0	1256	4.4
<i>Paratanytarsus</i>	71	0.6	2121	6.9	212	2.4	377	1.3
<i>Rheotanytarsus</i>	42	0.4	0	0.0	0	0.0	0	0.0
<i>Cladotanytarsus</i>	0	0.0	134	0.4	25	0.3	50	0.2
<i>Microtendipes</i>	297	2.7	1848	6.0	452	5.1	684	2.4
<i>Stictochironomus</i>	19	0.2	856	2.8	241	2.7	535	1.9
<i>Cryptochironomus</i>	0	0.0	0	0.0	0	0.0	14	0.0
<i>Thienemanuimya</i>	30	0.3	161	0.5	1	0.0	52	0.2
<i>Ablabesmyia</i>	0	0.0	58	0.2	12	0.1	107	0.4
<i>Macropelopia</i>	8	0.1	54	0.2	0	0.0	0	0.0
<i>Procladius</i>	10	0.1	543	1.8	40	0.5	143	0.5
<i>Orthocladius</i>	202	1.8	1364	4.5	186	2.1	565	2.0
<i>Eukiefferiella</i>	86	0.8	0	0.0	0	0.0	0	0.0
<i>Tvetenia</i>	120	1.1	29	0.1	0	0.0	0	0.0
<i>Cricotopus</i>	85	0.8	626	2.0	76	0.9	131	0.5
<i>Potthastia</i>	0	0.0	29	0.1	4	0.0	4	0.0
<i>Prodiamesa</i>	4	0.0	236	0.8	15	0.2	84	0.3
<i>Simulium</i> spp.	981	9.0	0	0.0	1	0.0	4	0.0
Other Diptera	60	0.5	48	0.2	25	0.3	71	0.2
Mollusca	454	4.2	404	1.3	44	0.5	388	1.4
Hydracarina	381	3.5	1780	5.8	225	2.5	900	3.2
Microcrustacea	7	0.1	1508	4.9	113	1.3	416	1.5
Other taxa	14	0.1	1	0.0	0	0.0	26	0.1

this season (Table 5.4), but the data from other seasons are also used in some instances (the list of taxa abundance in spring is in the Appendix one). Again although many rivers in the UK have been improved by current deflectors, there are few references in the literature regarding the consequences (e.g. Swales, 1982). Most of the studies have been carried out in North America and the results are only related to fisheries improvement (e.g. Hunt 1976, Sanderson & Smith 1962). In the following, the taxa which occurred on Natural Riffle in much greater relative abundance and did not occur on the other sites or occurred with low abundance are discussed. Other sites include AD, BD and SH, but the Run is not included because data were not collected from this site in summer. In the other three seasons when Run was sampled, there were no taxa inclusive to this site and the taxonomic abundance and richness of Run were low.

Baetis vernus was found in greater abundance in NR (12.6%) than other sites (0.3% – 0.1%). Species of *Baetis* are characteristic of riffles and found in greater abundance on fast-flowing habitats.

Hydropsyche angustipennis occurred in NR in greater abundance than the other sites in all four seasons. In summer it was found with 3.4% abundance in NR, but with 0.3% – 0.1% in the other sites. Species of *Hydropsyche* are also characteristic of fast-flowing waters and coarse substrate.

Athripsodes spp. (Leptoceridae) occurred in greater abundance in NR than in the other sites in summer (8.3% v. 3.8% – 1.9%), autumn (10.3% v. 4.4% – 1.2%) and winter (7.9% v. 2% – 0.7%) but in greater relative abundance in BD (9.2%) than in the other sites (4.1% - 2.2%) in spring, whereas the absolute abundance in spring was greater in NR (468) than BD (153). *Athripsodes* spp. was the third most abundant taxon in BD and the fifth in NR in spring. Species of *Athripsodes* are cased caddis with the case composed of sand grains. Two widespread and common species (*A. aterrimus* and *A. cinereus*) have been reported to occur in both slow and fast-flowing waters on stony and sandy substrate

in rivers and streams in the British Isles (Wallace *et al.*, 1990). The occurrence of *Athripsodes* spp. on both coarse and fine substrates in this study agrees with those findings of Wallace *et al.* (*op. cit.*).

Elmis aenea and *Oulimnius tuberculatus* (Elmidae) together, were also recorded in greater relative abundance in NR than in the other sites in summer (11.9% v. 6.5% – 2.1%), autumn (18.8% v. 3.4% – 1.5%) and winter (8.9% v. 1.4% – 0.2%) but with greater relative abundance in BD (13.3%) than in the other sites (8.4% – 0.6%) in spring, but again the absolute abundance was greater in NR (964) than BD (220). Elmidae was the second most abundant taxon in BD whilst it was the fourth in NR.

Tvetenia (Chironomidae) occurred with the abundance of 1.1% in NR, but was absent from the other sites. *Stictochironomus* was found in greater abundance in the other sites (2.8% – 0.2%) than NR (0.2%). *Eukiefferiella* was also occurred with 0.8% abundance in NR but was absent from the other sites. According to Smith (1989) many larvae of the subfamily Orthocladiinae, including *Eukiefferiella*, have adapted to life in swift flowing waters. Hayes (1991) found that species of *Eukiefferiella* were inhabiting moss growing on stones and vegetation in fast-flowing water. He also found larvae of *Tvetenia* in mosses, on stones in fast-flowing water and in *Ranunculus* sp. According to Cranston (1982) all species of *Eukiefferiella* and *Tvetenia* are primarily lotic, the maximum abundance for most species was in the upper reaches of streams among mosses and in the middle reaches on macrophytes. Larvae of *Stictochironomus* occur in profundal soft sediments or littoral sand of oligotrophic to mesotrophic lakes. They are also found in sandy sediments of stream and slowly flowing rivers (Cranston *op. cit.*). Pinder and Riss (1983) reported a similar distribution for the larvae of *Stictochironomus*. All these findings correspond to the results of this study which revealed that *Stictochironomus* was associated with fine substrate and slow-flowing sites and *Eukiefferiella* and *Tvetenia* were characteristic of coarse substrate and faster-flowing sites.

Simulium spp. were recorded in greater abundance in NR than in the other sites in three seasons; in summer 9% v. 0% , in autumn 15.1% v. 10.8% – 0.3% and in winter 22.6% v. 7.6% – 0.3% and in spring it occurred in greater relative abundance in Run (1.6%) than in the other sites (0.9% – 0.3%), but the absolute abundance was greater in NR (32) than in Run (22).

Microcrustacea were found in much greater abundance in the other sites (4.9% – 1.3%) than in NR (0.1%). The majority of freshwater species of Copepoda and Cladocera are found in ponds and ditches or in the muddy margins of larger water bodies (Scourfield & Harding, 1958 and Harding & Smith, 1960). This is the likely reason that these taxa were associated with the deeper and finer substrate sites of this study.

5.3.2 BIODIVERSITY DIFFERENCES IN SEASONS

The overall abundance and taxonomic richness varied between seasons; it decreased from summer to winter and then increased in spring. The abundance decreased about twofold in autumn, twofold in winter and increased less than twofold in spring. The number of taxa was 79 in summer and fell to 63 in autumn decreased 2 in winter and increased 4 in spring (Table 5.2).

The major difference between summer and autumn in the macroinvertebrate community structure was the greater relative and absolute abundance of two taxa in autumn than summer. *H. angustipennis* comprised 6.1% – 0.6% in autumn and 3.4% – 0.1% in summer. The reason for this is likely to be related to the animal's life cycle. According to Edington & Hildrew (1981) there is little information available on the life cycle of *H. angustipennis*. Adults of this species in southern England were found to have a single flight-period from May to September with peak activity in July, therefore they are likely to be found in earlier stages of their growth in the late summer and are more abundant in autumn. *Simulium* spp. were also found more abundantly in autumn (15.1% – 0.3%) than summer (9% – 0%).

The major difference between winter and autumn was the greater absolute and

relative abundance of the following taxa in winter: Naididae (11.7% – 1.2% abundance in winter v. 0.7% – 0.1% in autumn), *Orthocladius* (8.8% – 2% in winter v. 0.6% – 0.1% in autumn), *Tvetenia* (1.8% – 0% in winter v. 0.1% – 0% in autumn) and *Cricotopus* which had a 3.5% – 0.2% abundance in winter but was absent in autumn.

The major difference between spring and winter was related to the greater abundance of some taxa on NR. Naididae was found in much greater abundance in spring (39.5% – 2%) than winter (17% – 1.2%). *Nais elinguis* which is more abundant in winter and spring (Learner *et al.*, 1978) comprised more than 95% of the family in spring. *Baetis vernus* was found with 3.1% – 0% abundance in spring but it was absent in winter. Elmidae (13.3% – 0.6% in spring v. 8.9% – 0.2% in winter) and *Cricotopus* (14.1% – 0% in spring v. 3.5% – 0.2% in winter) were also found in greater abundance in spring than winter. *Simulium* spp. were found in less abundance in spring (1.6% – 0%) than winter (22.6% – 0.3%).

5.3.3 THE EFFECTIVENESS OF CURRENT DEFLECTORS

Current deflectors as habitat improvement device have been shown to be effective in enhancing fish population (Hunt, 1976). They are effective in changing the flow regime of the nearby areas. As they constrict the channel width the velocity increases and the current tends to scour out the river bed creating a pool below the deflector. The scoured sediments will be deposited downstream of the pool, creating a shoal or riffle. The shoal then is used by fish for spawning and the pool used for sheltering and resting. In the area above the deflector the current slows and the suspended sediments settle and this decreases the adverse effect of silt on fish spawning (Swales & O'Hara, 1980).

The results of this study of the current deflectors in the River Smite show that the devices have been successful in changing the hydrology of the river and creating pool and shoal sequences. Although the diversity and abundance of the biota of the created shoal was not comparable with the natural riffle, these were enhanced compared with Run which represents an unimproved site. AD also had a better biology than Run (greater total abundance in all seasons, table 5.2).

5.4 COMPARISON BETWEEN ARTIFICIAL RIFFLES AND CURRENT DEFLECTORS

The main objective of these habitat improvement techniques is the enhancement of fisheries and the overall biology of the river. Thus the final judgement of the effectiveness of the techniques would normally be based on the results of fisheries studies. Nonetheless, macroinvertebrates, which are a major component of fish food, can be used as a primary indicator of the improvement techniques.

Since both rivers are small, lowland, relatively unpolluted and the substrates of their natural riffles are similar, the comparison of the two techniques based on the abundance and taxonomic composition of the natural sites is appropriate. The total number of individuals of NR from Harper's Brook in four seasons was 34,061 which is close to that of the River Smite, 39,195. The diversity values of NR of the two rivers were also close, and identical in two seasons (autumn, 1.8 and spring, 2.1). The diversity of Harper's Brook was 2.9 – 1.8 while that of the River Smite varied from 2.6 to 1.8.

The taxonomic composition of NR of the two rivers were also relatively similar and characterised of faster-flowing and coarser substrate habitats. In general Baetidae, Hydropsychidae, Elmidae, Simuliidae and Chironomidae: *Eukiefferiella*, *Tvetenia* and *Cricotopus* formed the major characteristic taxa of NR of both rivers.

The results of both univariate and multivariate analyses showed that the sites in Harper's Brook were different in all seasons and two artificial riffles (AR2 and AR3) were similar to the natural riffle, while the results from the River Smite did not show the differences between the sites (except in spring) and none of them were similar to NR. The reason for this is because the physical characteristics (substrate, velocity and depth) of the two artificial riffles were similar to those of natural riffles and they produced macroinvertebrate communities comparable to those of the natural riffle. In the River Smite although the hydrology of the sites, due to the presence of the deflector, is changed, the diversity and abundance did not change to the level that it became comparable with a natural riffle, because the substrate of the sites were similar (gravel – sand/silt) and differed from that of the natural riffle dominated by cobbles. Therefore it can be concluded

that artificial riffles were more effective than a current deflector in creating a more diverse biota in the river.

5.5 SUMMARY OF THE CONCLUSIONS

- Artificial riffle or riffle reinstatement, as a habitat improvement technique was successful in improving the hydrology and biodiversity (macroinvertebrate abundance and diversity) of the improved sites close or similar to that of a natural riffle.
- Current deflector, another habitat improvement device, was successful in creating pool and shoal, by improving the hydrology, but not the biodiversity of the improved sites to a level similar to that of a natural riffle.
- Two of the artificial riffles in Harper's Brook behaved hydrologically and biologically like a natural riffle, and the third artificial riffle seemed to behave hydrologically and biologically like a run, despite its coarse substrate.
- Good riffles are shallower and fast-flowing and poor riffles are deeper and sluggish.
- The total abundance of macroinvertebrate communities of the sites decreased from summer to winter (fourfold – twofold) and increased in spring (twofold).
- In general the mean diversity of natural and two artificial riffles from Harper's Brook were higher, whilst those of run sites were lower in summer, winter and spring. The natural riffle of the River Smite had also a greater mean diversity compared with the other sites (Run, AD, BD and SH).
- There was a negative relationship between depth and diversity and a positive relationship between velocity and diversity of the sites from Harper's Brook in all seasons.
- Simuliidae, Hydropsychidae, Elmidae and Baetidae were among the abundant taxa of both natural and two artificial riffles (improved sites) in all seasons, whereas Tubificidae, Caenidae, Chironomidae and microcrustacea (Copepoda and Cladocera) formed the most abundant taxa of the run (degraded) sites of Harper's Brook. Simuliidae, Hydropsychidae and Baetidae also formed the most abundant taxa of the natural riffle of the River Smite. Some Chironomidae such as *Eukiefferiella*, *Tvetenia*

and *Cricotopus* were found to be characteristic of riffle sites, whereas *Tanytarsus*, *Paratanytarsus*, *Microtendipes* and *Stictochironomus* were characteristic of the slower-flowing and finer substrate habitats when they occur in greater abundance.

- The results of DECORANA indicated that there was a clear separation between the riffle sites (with the exception of AR1) and the run sites, including AR1, in all seasons in Harper's Brook. The results of the analysis also showed the separation of natural riffle from the other sites in the River Smite.
- The results of TWINSPAN agreed with the results of DECORANA, and indicated that the two artificial riffles were similar to the natural riffle and the third with the run sites in Harper's Brook. For the River Smite the results of TWINSPAN also agreed with the DECORANA results and indicated that the natural riffle was clearly different from the other sites.
- Since the r values from the results of DECORANA were greater than the r values from the results of the univariate analysis (in Harper's Brook), it seems the multivariate analysis is stronger and more useful in the interpretation of environmental data with high variability. In the River Smite the results of univariate analysis did not show significant correlation between diversity and depth and velocity but the multivariate analysis (DECORANA) did show significant correlations.

5.6 FISH SURVEY

5.6.1 HARPER'S BROOK

In order to show the existence of any differences in fish populations in different parts of Harper's Brook, three sites; natural (ca. 250 m), improved (ca. 221 m) and degraded (ca. 160 m), have been sampled in an electro-fishing survey by the NRA in March 1995. The natural riffle of this study is included in the natural site, the artificial riffles 2 and 3 are included in the improved and the run sites are included in the degraded site. The estimated density (no.m^{-2}) and biomass (gm^{-2}) of 11 fish sampled in this survey which were calculated by the NRA, are shown in table 5.5.

The total density at the natural site (0.591 no.m^{-2}) is greater than improved (0.094

no.m⁻²) and degraded (0.09 no.m⁻²) sites, although the total biomass of degraded site (30.03 gm⁻²) is greater than natural (18.48 gm⁻²) and improved (10.6 gm⁻²) sites. This indicates that smaller fish, like minnow, bullhead and stoneloach, occurred with higher density at the natural site whilst larger size fish like pike and eel with higher biomass occurred at the deeper degraded site. Although the total density of fish in improved and degraded sites is similar (0.094 no.m⁻² and 0.09 no.m⁻²), the total biomass of fish in the natural site is closer to the improved site than to that of the degraded site.

Pike and eel were not recorded from the natural site, but occurred in relatively much greater biomass in degraded (22.778 gm⁻²) and improved (8.127 gm⁻²) sites, and were the reasons for the higher biomass of the degraded site. This site is the lower reach of Harper's Brook which is affected by the confluence with the River Nene and this is the reason for the occurrence of pike and eel with relatively greater biomass in this site. Pike possibly migrates from the Nene to the brook for spawning.

When pike and eel are omitted from the data, the result shows that the natural site is the richest site in both density and biomass. The improved site has the second greatest density (0.087 no.m⁻²) and the degraded site the third (0.071 no.m⁻²), but the biomass of the two sites is reversed (7.252 gm⁻² v. 2.473 gm⁻²). The much greater biomass of some species especially chub (6.216 gm⁻² v. 0.91 gm⁻²) is the reason for the total biomass being greater in the degraded site than the improved site.

Table 5.5 Estimated density and biomass of fish from Harper's Brook in March 1995.

Species/sites	Natural		improved		Degraded	
	Density (no.m ⁻²)	Biomass (gm ⁻²)	Density (no.m ⁻²)	Biomass (gm ⁻²)	Density (no.m ⁻²)	Biomass (gm ⁻²)
Roach	0.042	1.729	0.007	0.240	0.013	0.301
Perch	0.005	0.085	0.019	0.541	0.026	0.628
Gudgeon	0.209	3.526	0.031	0.507	0.001	0.016
Bleak	0.004	0.054	0.000	0.000	0.003	0.025
Chub	0.087	5.730	0.005	0.910	0.008	6.216
Dace	0.219	7.272	0.008	0.225	0.015	0.059
Stoneloach	0.005	0.160	0.002	0.005	0.000	0.000
Bullhead	0.012	0.057	0.014	0.050	0.003	0.006
Minnow	0.008	0.018	0.002	0.002	0.004	0.005
Pike	0.000	0.000	0.005	7.945	0.013	22.140
Eels	0.000	0.000	0.002	0.182	0.006	0.638
Total	0.591	18.480	0.094	10.600	0.090	30.030

It can be concluded that, although the differences between total fish density and biomass in the natural and improved sites is relatively high, the improved site (artificial riffles) have started to become the habitats of the fish fauna. Thus the assessment of the artificial riffles in enhancing the fish population comparable to the natural site requires a longer period of time after surveying the Brook in some consecutive years. The degraded site which includes the run sites are generally deeper sites and inhabited by the larger fish. Some of them (e.g. Run 2) which are shallower with gravel and sand substrate can also be used by some fish for spawning and feeding. Thus, this site also plays positive role in the overall river ecosystem.

5.6.2 THE RIVER SMITE

Four sites in the River Smite were sampled for fish survey by the NRA in April/May 1995. Site 1 at Colston Basset is not degraded but consists of bed rock with no gravel or pools. Four fish species with the biomass of 6.3 gm⁻² was recorded from this site. Site 2 at Wiverton has gravel accumulated and is fenced off from cattle. Six species of fish with the abundance of 26 gm⁻² was recorded from this site. Site 3 at Whatton is rehabilitated by current deflectors. Ten species with the biomass of 32 gm⁻² recorded from this site. Site 4 at Cottam/Fox Cover which is located down stream of site 3 has more pools and overhanging trees. Twelve species of fish with the biomass of 14.2 gm⁻² was recorded from this site. The results indicate that the biomass of fish at site 3 was greater than the other sites, said to be very good diversity and biomass. Pools have scoured below each deflector and this has created holding areas for brown trout. Trout are stocked annually and each pool contained at least two of this fish. In previous years trout would not stay in this area so at this level these structures can be counted a success. These were the only information provided by the NRA.

5.7 SUGGESTIONS FOR FURTHER STUDY

The overall results of this study, based on different statistical methods, showed the effectiveness of artificial riffles in improving the macroinvertebrate diversity comparable to a natural riffle and hence realised the objective of the study. There are several aspects of the research that, with the experience gained during the course of this study, might have further improvement.

In order to get a better understanding of the consequences, the study should be continued for a longer period of time because this study was started just one year after the riffle reinstatement in Harper's Brook and two years after the construction of current deflectors in the River Smite and the ultimate hydrological and biological effects will not become fully apparent until a longer period has elapsed. I expect that the morphology of the rivers will change over longer period of time and create good pools and better habitat for fish. Longer term monitoring would ascertain what kind of invertebrate communities eventually develop and whether they were different from the non-rehabilitated parts of the river. The sampling of invertebrates in spring and summer months when the animals are most diverse and abundant have priority over autumn and winter. The number of replicates needs to cover the overall area of the sites and a minimum of four is recommended to cope with non parametric as well as parametric statistical analyses (in the case of using Kruskal–Wallis for the comparison of more than two sites or samples, the results of the test have to be tested by another test, the Mann–Whitney U-test which requires four replicates). A close preliminary examination of the sites based on their physical characteristics and subsequent classification of the sites for their final selection as sampling sites would cover all types of habitats and again four replicates would be desired. The study of some variables such as marginal vegetation and drifting animals would support the explanation of the changes in invertebrate communities. Marginal vegetation as well as the in-stream vegetation are used by some species for substrate, food, feeding sites, sheltering places and the attachment of invertebrate and fish eggs. Many invertebrates are subject to drift thus the occurrence of some of them on a particular site may be as a result of this activity and not being there because they are the actual

inhabitant of that site.

Along with the study of invertebrate communities, the fish survey of the river should also be carried out over some consecutive years in order to test the objective of the rehabilitation project which is the improvement in fisheries. The study should be carried out in the other rivers which have been improved with the same devices, to provide a point of comparison. Useful information would be gathered from studying rivers before they were rehabilitated and comparing the results with a study carried out after the rehabilitation on the same rivers.

5.8 CONCLUSIONS

The two different river habitat improvement devices, artificial riffles and current deflectors which have been studied in this project are among the most widely used techniques. Two artificial riffles are shown to be effective in enhancing the hydrology and biology to a level similar to the natural riffle, whereas the other artificial riffle which was deeper behaved as did the run sites in spite of its coarser substrate. Thus the construction of good riffles will increase the conservation value of the river. The installation of current deflectors also changed the hydrology of the river and created pools and shoals below them.

Clearly, the final judgement about the consequences of these devices requires further studies of both invertebrate and fish populations over a longer period of time. Obviously the use of any type of improvement devices depends on the management objectives and the river conditions. In the case of creating habitat heterogeneity and better biology, the current deflectors would be only effective when the natural substrate of the river bed is coarse enough to enhance the habitat after being scoured by the turbulent flow created by the deflector and deposited below as a shoal or riffle. But when the natural substrate is finer the riffle reinstatement, which includes coarser minerals, would be beneficial. However, changing the river bed and increasing the habitat heterogeneity is not the ultimate goal of any river restoration project. According to Sear (1994) river restoration is the complete structural and functional return to a pre-disturbance state, and Muhar *et al.*

(1995) proposed the following definition for river restoration, 'the totality of measures which change man-induced alterations to rivers in such a manner that the ecological functioning of the new state resembles a more natural river'. Besides the enhancement of in-stream habitat, river restoration has been used for reducing nutrient and sediment loads from intensively farmed agricultural land, for enhancing landscape quality and for the stabilisation of eroding stream system. From an ecological point of view the richness, diversity and age composition of the populations developing after restoration as a result of habitat improvement reveal the degree to which comprehensive concepts were applied.

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APPENDICES

The Appendices are on the enclosed disk which is formatted for Macintosh Excel (version 4.0) and Word (version 5.1). They are listed below:

APPENDIX ONE

ANOVA output from Harper's Brook.
ANOVA output from the River Smite.
Species abundance from Harper's Brook in autumn.
Family abundance < 1% from Harper's Brook in autumn.
Species abundance from the River Smite in autumn.
Family abundance < 1% from the River Smite in autumn.
Species abundance from Harper's Brook in spring.
Family abundance < 1% from Harper's Brook in spring.
Species abundance from the River Smite in spring.
Family abundance < 1% from the River Smite in spring.
Species abundance from Harper's Brook in summer.
Family abundance < 1% from Harper's Brook in summer.
Species abundance from the River Smite in summer.
Family abundance < 1% from the River Smite in summer.
Absolute and % abundance of the taxa of the sites from Harper's Brook in spring.
Absolute and % abundance of the taxa of the sites from the River Smite in spring.
Species abundance from Harper's Brook in winter.
Family abundance < 1% from Harper's Brook in winter.
Species abundance from the River Smite in winter.
Family abundance < 1% from the River Smite in winter.

APPENDIX TWO

DECORANA output for Harper's Brook in autumn (AUTHB.DEC).
TWINSpan output for Harper's Brook in autumn (AUTHB.TWI).
DECORANA output for the River Smite in autumn (AUTRS.DEC).
TWINSpan output for the River Smite in autumn (AUTRS.TWI).
TWINSpan output for Harper's Brook in four seasons (HBALL.TWI).
DECORANA output for Harper's Brook in spring (SPRHB.DEC).
TWINSpan output for Harper's Brook in spring (SPRHB.TWI).
DECORANA output for the River Smite in spring (SPRRS.DEC).
TWINSpan output for the River Smite in spring (SPRRS.TWI).
DECORANA output for Harper's Brook in summer (SUMHB.DEC).
TWINSpan output for Harper's Brook in summer (SUMHB.TWI).
DECORANA output for the River Smite in summer (SUMRS.DEC).
TWINSpan output for the River Smite in summer (SUMRS.TWI).
DECORANA output for Harper's Brook in winter (WINHB.DEC).
TWINSpan output for Harper's Brook in winter (WINHB.TWI).
DECORANA output for the River Smite in winter (WINRS.DEC).
TWINSpan output for the River Smite in winter (WINRS.TWI).