UNIVERSITY OF SOUTHAMPTON

FACULTY OF SCIENCE

# EFFECTS OF COARSE WOODY DEBRIS ACCUMULATION, CHANNEL STRUCTURE AND LAND USE ON FISH POPULATIONS IN FORESTED LOWLAND STREAMS

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## ABSTRACT

FACULTY OF SCIENCE

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# EFFECTS OF COARSE WOODY DEBRIS ACCUMULATION, CHANNEL STRUCTURE AND LAND USE ON FISH POPULATIONS IN LOWLAND FORESTED STREAMS

by T.E.L. Langford

A detailed study of forested streams in the New Forest in southern England has shown that the accumulation of coarse woody debris (CWD) in channels affects the within-reach structure of the stream channel and the abundance, diversity and community structure of the fish fauna. Samples in *a priori* selected habitat units, namely 43 riffles, 80 pools and 39 CWD accumulations showed significantly higher densities of fish (as numbers $m^{-2}$) in the riffles than in the other habitat units and significantly higher biomass (as $gm^{-2}$) in the CWD accumulations. Effects on each of the six species present differed. There was a significant correlation between maximum size of salmonids and abundance of CWD in a habitat unit.

Habitat diversity and fish diversity were lowest in riffles and dense CWD accumulations and highest in habitats with moderate amounts of CWD present. Communities of habitat units could not be separated clearly based on the *a priori* selection but both the physical structure and fish communities showed a gradient of change from an erosional to a depositional condition with riffles and the deepest CWD habitats as the opposing extremities of the gradient. On the reach scale habitat diversity was related to the abundance of CWD but overall fish diversity and abundance were not. Salmonid density was negatively correlated with CWD abundance on the reach scale. The fish community of the forested streams was highly structured and characteristic of a deterministic community relatively undisturbed by human influences.

In contrast, the fish communities of adjacent and confluent streams with partially deforested catchments showed higher species richness (12spp), higher diversity and higher abundance as a result of immigration of species from downstream. The difference between the communities was commensurate with a change from an allochthonous energy system to an autochthonous system and indicated high level of disturbance. Higher abundance was a result of higher overall productivity in the latter system. The two types of fish community appear to persist separately in adjacent and confluent streams despite the mobility of the fish.

A simple, partly hypothetical model is proposed to predict the effects of varying abundance of CWD on the fish communities of the forested streams. Future stream habitat management is discussed and potential applied and fundamental topics for research outlined.

**THIS WORK WAS CONCEIVED, INITIATED AND MAINLY CARRIED OUT BY THE AUTHOR WHILE A REGISTERED POSTGRADUATE CANDIDATE OF THE UNIVERSITY OF SOUTHAMPTON. A SMALL AMOUNT OF DATA HAS BEEN PUBLISHED IN THIRD YEAR UNDERGRADUATE REPORTS. THE AUTHOR PROPOSED AND SUPERVISED ALL OF THESE PROJECTS AND PARTICIPATED IN ALL FIELD WORK. WHERE DATA FROM OTHER SOURCES HAVE BEEN USED THESE ARE CLEARLY MARKED AND IDENTIFIED.**

# CONTENTS

## CHAPTER 1. INTRODUCTION

**CHAPTER 3. METHODS, DATA ANALYSIS AND SITE DESCRIPTIONS**

## CHAPTER 4. DESCRIPTION OF THE HIGHLAND WATER AND ASSOCIATED STREAMS: HABITAT DIVERSITY, WATER QUALITY AND INVERTEBRATE FAUNA

**CHAPTER 5. ABUNDANCE AND SIZE DISTRIBUTION OF INDIVIDUAL FISH SPECIES IN RELATION TO CHANNEL STRUCTURE AND WOOD DEBRIS ACCUMULATIONS**

**APPENDICES**

**Appendix I**. List of all sampled reaches showing the location, total reach length and total catch of each species in the summed habitat unit samples.

**Appendix II.** Appendix II comprises two OS maps (1:25000) of parts of the New Forest showing the locations of the sites (reaches) used for this research. Each site is labelled with its ID number (see Appendix I). The stream and overall sampling areas are summarised diagrammatically in Fig. 2.1.

**Appendix III.** Appendix III is a 1.4Mb computer disc containing the raw fish catch data for each successive sub-sample from each habitat unit at each site on the Highland Water streams during 1996-98. The format is MS Excel spreadsheet.

## FIGURES AND TABLES

### CHAPTER 2

## CHAPTER 3

## CHAPTER 4

**CHAPTER 5**

## CHAPTER 6

**CHAPTER 7**

## CHAPTER 8

## PREFACE

The streams of the New Forest in southern England are small, quiet and generally unobtrusive except after heavy precipitation when they rise rapidly and carry heavy loads of sediment. They drain a range of catchments, mostly with neutral or acidic soils. There is very little urban run-off and the main catchments comprise mostly forest, grazing land or pasture. There is no formal fishery except in a very few reaches although the streams hold significant numbers of large migratory sea-trout during autumn and winter. Because of their small size, relative unimportance as a fishery and the absence of a very local freshwater research group, there are few peer reviewed publications dealing with the ecology of the streams compared with those dealing with the nearby chalk streams of Hampshire and Dorset.

At the end of 1991, I began a long-term, on-going study of the detailed distribution of the invertebrates of the New Forest streams mainly to investigate the effects of different land-uses. During this work, I realised that despite the extensive geomorphological studies of the streams and the continuing discussions about the effects of coarse woody debris dams (CWD) on invertebrates and fish there was no relevant scientific information either from the New Forest or any other region of Britain. Originally, the intention was to study the invertebrates in relation to CWD but in the event this work was started elsewhere so I turned to the fish populations. This work is the result of the first three years of the programme, which is continuing. I have had a great deal of support for the work from all the Agencies involved and from my colleagues, friends and family all of whom are acknowledged.

In this thesis I have used the terms high-disturbance and low disturbance when referring to different streams. In fundamental ecological terms, disturbance generally refers to natural disturbance, for example from spates or extensive sediment movement to which species and communities become adapted over long time-scales. In this account, however, disturbance refers solely to human influences, for example channelisation, stream clearance and maintenance or human activities in the catchment or riparian zones. Natural disturbance is considered to have been similar in all the streams in their original state and therefore a common factor. Human disturbance on the other hand has differed in its extent in the streams studied.

As I continue to work in the New Forest, I realise that though the stream systems are unique they provide a diversity of opportunity for original fundamental and applied research because of their basic, pre-disturbance similarities, their geological isolation from surrounding waters, their hugely varying levels of human disturbance, lack of pollution, long history of land-use change and accessibility. They are also likely to be subjected to further land-use changes, channel changes and encroaching development over the next few decades and are thus highly threatened habitats.

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As ever, my wife Mrs.Jean Langford has been my major support, given much of her time in helping with all the tasks associated with this work, from vital assistance in the field to the helping with typescript, checking and proof reading of references, text, figures and tables and correcting syntax, in addition to all the other duties associated with consultancy and administrative work. Without her help, encouragement and support I would never have taken on, let alone completed this work. If I have failed to mention any other contributors or helpers I offer my sincere apologies but I hope they will accept my gratitude, even anonymously.

*Terry Langford*

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# CHAPTER 1

# INTRODUCTION

## 1. 1. OVERVIEW, AIMS AND OBJECTIVES

There is a widely accepted premise that the distribution, abundance and diversity of fish in rivers and streams are functions of the structural diversity of the channel provided that other conditions are not limiting (e.g. Heggenes, 1988a; Cowx & Welcomme, 1998; Maddock, 1999) and there have been many attempts to model these relationships (e.g. Binns & Eisermann, 1979; Bovee, 1982; Milner *et al.*, 1985; Korman *et al.*, 1994; Scruton *et al*, 1998; Milhous, 1999).

Timber debris, mainly large trunks or branches, known as "coarse woody debris" (CWD) is regarded as a major component of the structural diversity of stream channels (e.g. Keller & Swanson, 1979; Bisson *et al.*, 1987; Hawkins *et al.*, 1993). Indeed, the abundance and diversity of invertebrates and fish in many streams are regarded as closely related to the amounts of CWD present (Harmon *et al.*, 1986; Maser & Sedell, 1994; Bryant & Sedell, 1995; Gurnell *et al.*, 1995). Whilst this has been demonstrated for some streams (e.g. Wallace & Benke, 1984) and for some species (Angermeier & Karr, 1984; Dudley & Anderson, 1987) the relatively scarce peer-reviewed data (see Gurnell *et al.*, 1995) suggest that the relationship is by no means universally applicable to streams or species (e.g. Hortle & Lake, 1983; Langford & Hawkins, 1997). To date, there are no comprehensive data on the relationships between fish and CWD in British streams (Langford & Hawkins, 1997). Further, there are few data on the effects of channel structure or physical alterations on either fish abundance, diversity and community structure (Milner *et al.*, 1985; Cowx & Welcome, 1998). The overall aim of this research programme was to provide an ecological basis for the management and restoration of streams by quantifying the relationships between fish populations, communities and physical habitat with particular reference to CWD accumulation and the effects of land-uses in the catchments. The results of the studies were also intended to relate to stream management on the wider scale both where low-disturbance and high-disturbance communities are found. The streams selected are located in the New Forest in southern England.

The New Forest in southern England is an area with a relatively stable history of land use (Tubbs, 1968, 1986). A full description and maps are given in Chapter 2. It is drained by a network of small streams many of which contain substantial amounts of CWD. These streams have been the subject of geomorphological and biological studies for over 20 years many related to the presence and dynamics of CWD (e.g. Brookes, 1983; Gurnell & Gregory, 1984; Gregory *et al.*, 1985; Langford, 1996; Gurnell & Sweet, 1998). Thus the New Forest streams provided an excellent opportunity to investigate the roles of channel structure and CWD in the ecology of the fish communities. Further, because of the long-term relative stability of some land uses, there was an opportunity to study the effects of well established disturbances such as forest clearance and stream channel maintenance on the fish communities of streams in close proximity to the low-disturbance forested streams.

Because of the historical absence of tree clearance and low disturbance characteristics of some streams (Sear & Arnell, 1997) it was predicted that the stream fish populations were likely to be climactic and show well established within-reach and within-stream spatial patterns of habitat use, related to optimal habitat partitioning by the various species (e.g.Moyle & Vondracek, 1985; Prenda *et al.*, 1997; Martin-Smith, 1998). As CWD accumulations are considered as an integral part of the undisturbed stream system, their role in the abundance patterns and community structure of the fish on various spatial scales should also become evident. It was also predicted that the disturbed streams would show higher diversity and abundance of fish as a result of higher primary productivity together with significant changes in community composition.

Among the streams of the New Forest the Highland Water system is considered as one of the least disturbed (Sear & Arnell, 1997). Further, most of the background data on CWD distribution and dynamics have been obtained from studies of this stream system over more than 20 years (e.g. Gregory, 1992;

Gregory *et al*., 1993). As one of the streams nearest to its original state in the New Forest it shows "*features of pre-disturbance rivers systems that are rare in lowland Britain*" (Sear & Arnell, 1997). Thus, from September 1996 to February 1998 detailed studies of the distribution, abundance, community structure and diversity of fish were made in the Highland Water and its associated streams. Comparative studies included streams in the proximity of the Highland Water, mainly Dockens Water and the Ober Water where deforestation and land use changes have occurred over long timescales. This thesis reports the analyses and conclusions of this programme and provides an ecological background to guide the future management of woody debris, channel structure and fish conservation in the stream systems in the New Forest and other regions.

This introductory chapter (Chapter 1) places the studies in the wider context of fish distribution and abundance and their relationship with physical habitat structure. Distribution and abundance of species are clearly closely interrelated phenomena (Andrewartha & Birch, 1954; Andrewartha, 1970) and to a great extent the composition of the community and diversity in any habitat are also a function of these (Krebs, 1978). However, management of many habitats, particularly streams, has most often been based on enhancing or protecting the abundance of one or two species rather than protecting a community (Maitland, 1974; Maitland & Campbell, 1992; Cowx & Welcomme, 1998). More recently, however, the protection of habitats has focused attention on the composition of communities in such habitats, particularly where historical disturbance has been minimal (European Communities, 1992; Maitland & Campbell, 1992). In this thesis distribution and abundance of individual species are therefore considered separately from diversity and community structure with the aim of using the data for the future management both for particular species and communities.

The factors which influence the distribution of fish in streams are outlined briefly in this chapter on the basis of different scales, e.g. geographical, regional, catchment, stream reach and within-reach scales (Bayley & Li, 1992; Maddock, 1999). Although the fundamental basis of fish distribution and abundance in any region may depend on large scale characteristics, the long-term survival of any species and the management of a stream system for any species or species assemblage will depend to a large extent on small-scale variations in the habitat both temporally and spatially (Stalnaker, 1979; Cowx & Welcomme, 1998). The relationship between physical structure and fish habitat is therefore discussed with particular reference to the role of wood debris. The historical role of wood debris in stream ecology is also discussed, and published literature on CWD reviewed. Aspects of the effects of human disturbance including deforestation on streams are outlined. At the end of this chapter the aims and objectives of the subsequent chapters are described.

## 1. 2. FACTORS AFFECTING FISH DISTRIBUTION AND ABUNDANCE IN STREAMS

Freshwater fish distribution on various scales, *viz.*, geographic, catchment, stream, reach, mesohabitat and microhabitat, is influenced by many different factors (e.g. Norman & Greenwood, 1963; Hynes, 1970; Hawkes, 1975; Davies & Walker, 1986; Bayley & Li, 1992; Maitland & Campbell, 1992; Maddock, 1999). From the literature it is clear that the structural diversity of any reach of a stream channel is low in the ranking of factors which initially determine the species of fish which live in that reach both temporally and spatially. Geography, topography, physical obstruction to free movement, catchment conditions and water chemistry are the major factors determining species distribution and hence community composition on the regional, catchment and stream (river) scales. Within the reach, however, the distribution and abundance of specific structural features or substrates can influence the detailed distribution of any life-history stage of any species (e.g. Heggenes, 1988a). For example the amounts and placement of large timber debris in running waters could affect free-movement by causing physical blockage (e.g. Hawkins *et al.*, 1993), by altering the hydrological regime (Gurnell *et al.*, 1985; Bryant & Sedell, 1995) and to a small extent by changing water chemistry (Bisson *et al.*, 1987; Ridge *et al.*, 1995). It will not, however, exert much influence on the total species composition of the fish community on the catchment scale though the amounts of timber present in a stream can depend on the extent of woodland in the catchment (Triska, 1984; Gregory *et al.*, 1993; Bryant & Sedell, 1995; Gurnell *et al.*, 1995).

Geographic region and size of catchment are vital factors determining species composition on the catchment scale. For example river systems such as the Orange-Vaal, in southern Africa contains some 16 species, while the Mackenzie in Canada, and the Murray-Darling in Australia contain some 50-53 species and the Niger and the Zaire 180 and 600 species respectively. As an example of extreme size and fish diversity the Amazon contains more than 1300 species (see Davies & Walker, 1986). In Britain, the larger river systems such as the Thames or Severn contain between 35 and 40 species (Maitland & Campbell, 1992).

Within any one river system, the number of species and diversity typically increases with distance downstream from the source (e.g. Shelford, 1911; Thompson & Hunt, 1930; Larimore *et al.*, 1952; Huet, 1959; Sheldon, 1968; Hynes, 1970; Hawkes, 1975). Whilst this may be a function of the size or depth of the channel it may also be a function of physical diversity and the increasing presence of instream vegetation. In the uppermost reaches of streams of comparable size, the diversity and species richness of the fish community is also a function of the chemical tolerance of the fish rather than size or physical diversity of habitat. For example in naturally acidic British streams salmonids may be the only fish present and even these may disappear if the pH falls (e.g. Howells, 1990). In circumneutral or more alkaline streams species of Cottidae, Cobitidae, Cyprinidae and Petromyzonidae may also be present along with other species (e.g. Mann, 1971; Turnpenny *et al.*, 1987; Howells, 1990). However, stream size may also be an important limiting factor to species richness (Hawkes, 1975; Moyle & Vondracek, 1985).

Models which predict fish presence or absence on the catchment scale have been developed for both tropical and temperate regions. For example (Welcomme, 1989) showed a significant relationship between floodplain area and species richness in African rivers. More recently, Lamouroux *et al.* (1998, 1999) developed statistical habitat models based on the frequency distributions of relatively simple physical measurements and a low flow hydraulics model which predicted community composition and abundance of fish in the Rhone basin in France. They found that the relative abundance of species present depended mostly on zoo-geographic factors. They concluded that "*geographic regions act as filters*" on the relative abundance of species in a river catchment but that hydraulic characteristics affect communities within this scale. They also found that hydraulic models could explain up to 95% of the variation of community structure indices.

The use of physical and hydraulically based habitat models for assessment of usable fish and invertebrate habitat is widespread and well established (e.g. Bovee, 1982) though there are many criticisms of such models e.g. (Korman *et al.*, 1994; Maddock, 1999). At the reach scale or smaller, relating the diversity and abundance of any fish community directly to the physical structure of the habitat may be over simplistic. Indeed, Elliott (1994) has questioned the basic assumptions on which physical habitat models for predicting fish abundance are founded, particularly the assumption which many models adopt that there is a linear or log-linear relationship between abundance and structural diversity of the channel. The lack of uniformity in measurement, the differential ranking of variables, the mixture of subjective and objective observations and the lack of consideration for interspecific or intraspecific factors are considered to be other reasons for the failure of such models (Elliott, 1994; Korman *et al.*, 1994; Scruton *et al.*, 1998). Sheldon (1968) also noted that because of the cross-correlation of many environmental variables in streams "*almost any variable chosen will have some predictive value*" and the choice of factors to be measured is difficult.

Even within any single stream channel, the relationship between the physical habitat and the diversity of the fish community may vary over time. For example, Gorman & Karr (1978) found that there was a significant relationship between habitat diversity and species diversity in a stream in June but that this was not so in September when dense filamentous algal growths restricted the use of the habitats by some species of fish. At this time fish abundance was negatively related to habitat diversity. Gorman & Karr (1978) suggested that the method of habitat assessment predicted higher fish diversities than occurred. In fact the published graphs show that the algal mat decreased the measured physical diversity of the channel bed and the species diversity declined accordingly. Other temporally variable physical factors such as depth, current velocities and sedimentation can also have direct effects on habitat diversity (see Jungwirth, 1984; Cowx & Welcomme, 1998) and the responses to habitat features such as cover or substrate can vary with both species and life-history stages of fish (Cowx & Welcomme, 1998, Heggenes,1988a).

At the microhabitat level the presence of a single stone or structure, weed beds, bankside vegetation or the presence of a single competitor or predator may be the primary influence on the presence of an individual fish (e.g. Heggenes, 1988a; Copp, 1992; Ault & White, 1994; Ibbotson *et al.*, 1994; Copp & Bennetts, 1996). With all the provisos and variability both within and between stream habitats it would, therefore, seem optimistic to expect any single model of channel structure or hydraulics to achieve more than a very broad scale of prediction of fish diversity or abundance even for any one stream system.

## 1. 3. DEFINITIONS OF FISH HABITAT

The physical habitat of any fish species is, therefore, only one component of the environment in which the fish lives. Elton (1949) defined habitat in terms of the observer as " *an area that seems to possess a certain uniformity with respect to physiography, vegetation or some other quality that the ecologist decides is important or easily recognised. He (sic) decides on the limit of the habitat arbitrarily and in advance as a first step toward the study of the community*". Later definitions have been less anthropomorphic and based more on the requirements of the organism. Cowx & Welcomme (1998) for example define habitat as " *where a fish (species) lives without specifying resource availability or use*" and Southwood (1988) also relates the habitat directly to the organism and concludes that "*the features of the habitat must be expressed on temporal and spatial scales that are organism-related*". Maddock (1999) defines physical habitat as " *the living space of instream biota, a spatially and temporally dynamic entity determined by the interaction of the structural features of the channel and the hydrological regime*".

Clearly the definition of habitat has to include both spatial and temporal components, and the scale varies with season and the life history stages of any species. For example, within any stream, shallow gravel reaches are vital for spawning and early growth and development of salmonid fry and parr (e.g.

Maitland & Campbell, 1992) while deeper waters are used for shelter by older juveniles and for upstream migrants (e.g. Egglishaw & Shackley, 1982; Kennedy & Strange 1982; Maitland & Campbell, 1992; Cowx & Welcomme, 1998). The entire habitat for migratory species such as *Salmo* spp. and *Anguilla* spp. however, includes that used by all life history stages. This comprises a very wide range of spatial dimensions encompassing various lotic and possibly lentic environments from headwater streams through to large river channels (possibly via interconnected lakes) and subsequently to estuarine and marine environments. These can be affected by a vast array of both natural and human induced factors, including impoundment, abstraction, diversions and pollution.

In contrast the habitat for a more sedentary lotic species such as the bullhead (*Cottus gobio* Linnaeus 1758) will only include relatively short lengths of a stream channel possibly tens of metres or less (e.g. Cowx & Welcomme, 1998). For other species, such as barbel (*Barbus barbus* (Linnaeus 1758)) the habitat is larger with two major components. Spawning occurs in clean gravels in shallows and feeding habitats are the deeper, slower flowing reaches and backwaters where a temporary zoo-plankton can develop in summer (Maitland & Campbell, 1992). Even in small streams species may show temporal variations in habitat use. For example Roussel & Bardonnet (1997) showed diel patterns of movement by small fishes between pools and riffles in summer in a small stream in France. Also, Garner *et al.* (1998) showed varying habitat use by minnows (*Phoxinus phoxinus* (Linnaeus 1758)) in a chalk stream which depended upon a trade-off between temperature and prey-density.

Given the different scales and definitions of fish habitat therefore it is clear that alterations or disturbances in a relatively short length of stream channel will influence the success of one or two stages or the whole of the life history for that particular part of the population. The effect on the population at different scales will depend on the extent of the disturbance.

Thus for any one stream system, given the requisite chemical conditions and limitations set by the larger scale factors, the variations and diversity of physical habitat at reach and sub-reach levels will at least partly determine both the diversity of the community and the survival of relevant species, at least at the local population level, through certain stages in their life history. A physically diverse habitat may be necessary not only for the maintenance of community diversity but also for the survival of any one species. Therefore the management of species diversity or populations of any one species in any one reach of river will depend on very localised variability in hydraulic conditions and the consequent sedimentation in addition to the structural diversity of the channel. In streams and small rivers this can be determined by channel configuration and structure which in turn is determined by flow and physical structures within the channel such as boulders, rocks, tree-root matrices and large pieces of timber debris (CWD) (e.g. Hawkins *et al.*, 1993).

Because of the definitions of habitat and the variety of ecological levels of study in stream ecosystems the choice of a scale on which to attempt to analyse or describe the relationship between any species and its environment or part of its environment can be difficult. Levin (1992) considered that because ecological systems show variability over a wide range of spatial temporal and organisational scales and because of the innate patchiness over a broad range of scales, "*there is no single correct scale at which to describe populations or ecosystems*". Given the various spatial and temporal scales of habitat definitions, the choice of scale at which to study fish in relation to their habitat or any part of their habitat must initially be based on the testing of a hypothesis which in turn may be based on some requirement for information which can be readily applied in stream management. River management and restoration work are typically carried out at the reach or sub-reach (channel-unit) scale (Cowx & Welcomme 1998; Gurnell & Sweet, 1998) and a knowledge of the distribution, abundance and diversity at this scale is generally applicable for practical purposes. To date, timber debris in rivers (CWD) has been mainly studied on the catchment scale (e.g. Triska, 1984) or stream reach scale (e.g. Gurnell & Gregory, 1984; Gregory *et al.*, 1985; Gregory, 1992; O'Connor 1992; Shields & Smith, 1992) and the biological relationships mainly on the reach or sub-reach scale (e.g. Angermeier & Karr, 1984; Benke *et al.*, 1984, 1985; Dudley & Anderson, 1987; Smock *et al.*, 1989; Davies & Nelson, 1994; Braaten & Berry, 1997; Hildebrand *et al.*, 1997; Langford & Hawkins, 1997). Thus far there are few data on fish and CWD where the CWD accumulations themselves have been isolated as habitat units (Langford & Hawkins, 1997).

In Britain, studies of timber debris in streams (e.g. Gregory *et al.*, 1985) have been mainly concerned with its significance to channel form, physical diversity, hydrological processes and the distribution and quantity of woody debris in relation to management practices and effects of storms (Gurnell & Gregory, 1984; Gregory *et al.*, 1985; Gregory, 1992; Gregory & Davis, 1992; Gregory *et al.*, 1993; Gurnell *et al.*, 1995). Most of the data originate from studies in the New Forest. Published data on fish in relation to instream timber debris in British rivers and streams are scarce and so far originate only from the studies described in this account (see Langford & Hawkins, 1997).

## 1. 4. WOOD DEBRIS AND STREAM MORPHOLOGY

The role of timber debris in streams has been reviewed by a number of authors in relation to river channel morphology and processes in recent years, (e.g. Sedell & Froggatt, 1984; Triska, 1984; Harmon *et al.*, 1986; Bisson *et al.*, 1987, 1988; Gregory, 1992; Dolloff, 1994; Maser & Sedell, 1994; Bryant & Sedell, 1995; Gurnell *et al.*, 1995). The overall perception is that in-stream timber debris, particularly coarse woody debris (CWD) is a vital natural component of the natural river processes, particularly influencing flood regimes, floodplain interactions and sediment movement. It is also considered to be a significant factor in the maintenance of structural complexity of channels and the enhancement of habitat diversity and hence biological diversity (Bryant & Sedell, 1995). These timber accumulations may either act directly by increasing the availability of physical refugia in the habitat, or indirectly through influences on the physical, chemical and biological processes in the stream system (e.g. Inoue & Nakano, 1998). The biological role of CWD is discussed in later sections of this chapter.

## 1. 5. HISTORICAL STUDIES OF WOOD IN RIVERS

There are very few accounts of the history of woody debris (CWD) in the rivers of Europe prior to the beginnings of the main clearance for drainage and navigation in Europe (see Gurnell *et al.*, 1995). In contrast the history of the clearance of woody debris from rivers in the United States, mainly for navigation purposes or for transport of cut timber (Bisson *et al.*, 1987) is well documented (e.g. Lobeck, 1939; Swanson *et al.*, 1976). Before human influences began to have their marked effects on the catchments of rivers, most channels contained woody debris ranging from twigs and pieces of bark to whole trees complete with roots and attached soil (Sedell & Luchessa, 1982; Sedell & Froggatt, 1984; Michenor, 1985; Keller & MacDonald, 1995; Gurnell *et al.*, 1995). Some of the woody debris accumulations such as the Red River debris dam in the USA were massive and created large natural impoundments (Triska, 1984). Prior to the clearance of the Red River channel in the early 20th century, the accumulation of trees, woody debris and soils stretched for some 300 km in the channel, effectively preventing navigation in over two-thirds of the river system (Lobeck, 1939). In the Mississippi some of the items of debris, known as "snags", removed during clearance were up to 20m long, 1m in diameter and were embedded up to 6m in the river bed (Keller & MacDonald, 1995). The removal of the large accumulations of woody debris led to significant changes in river channels and floodplain processes which were subsequently exacerbated by channelization and embankment.

In European rivers the process of clearance is not well documented though it was no doubt similar to that in the USA but took place over a far longer period. Bryant & Sedell (1995) and Harmon *et al.* (1986) cite examples of the uses of European rivers and rivers in the Middle East which accounted for the clearance of timber debris from the channels. In Europe, navigation of the major and minor river channels increased with the spread of the more advanced civilisations and rivers such as the Rhine, the Danube, the Volga and their tributaries must have experienced the clearance of timber debris much as did the rivers of North America. Today, ancient riparian forest is rare in Europe and almost entirely absent from the lowlands. The presence of the remains of lignicolous aquatic invertebrates in ancient timber unearthed from the River Rhine (Amoros & Van Urk, 1989) is some indication of the long-term occurrence of wood debris prior to clearance. Lignicolous Diptera still form an important component of stream faunas in other parts of the world (Armitage *et al.*, 1995)

In the UK, the clearance of forests in the major river catchments occurred mainly between 5000 BC and 1000AD (Wiltshire & Moore, 1983). A bibliography of the effects of deforestation in streams is given by Blackie *et al.* 1980. Even though rivers and small streams had been used for navigation and fishing for many centuries it is unlikely that much clearance of timber debris occurred until more commercial navigation began and water mills increased the need for clear channels and the channelization of the more anastomosed streams. Most of the rivers in the UK are today relatively free of wood debris accumulations, particularly where drainage or navigational operations are the priority uses of the river. Riparian vegetation is also scarce along many rivers such as those in the fenlands and drained marshes where access for land-drainage machinery is a priority. The few stream systems with significant amounts of timber debris are mostly in hill forests or the few remaining lowland forests such as the New Forest (Langford, 1996) and the Forest of Dean. In the past decade, wooden structures, simulating CWD matrices or timber pieces have been used as components of river channel modifications ostensibly to enhance both the invertebrate and fish populations (e.g. de Jalon, 1995; Cowx & Welcomme, 1998).

## 1. 6. ORIGINS AND DISTRIBUTION OF WOOD IN RIVERS

Many factors control the introduction and distribution of CWD in running waters, of which some are the result of human activities, mainly forestry or agricultural land clearance (Maser & Sedell, 1994; Bryant & Sedell, 1995; Gurnell *et al.*, 1995). Before the advent of these human activities, storms, floods, fires and natural growth and decay processes provided dead or dying trees and branches either directly to a stream channel or to the forest floor. Delivery from the forest floor to the stream channel is a function of various forces including avalanches, movement down hill slopes, direct blowdown, flooding and human activity (Gurnell *et al.*, 1995). Triska (1984) notes that in the well managed forests of Europe amounts of wood on the forest floor are small, 0-5 tha$^{-1}$ compared with up to 200 tha$^{-1}$ in the more pristine forests of the north west of the USA. Much of the timber debris in streams at the present time originates from clear-felling where smaller logs and branches are discarded on the forest floor and are swept by floods or placed by human activity in the stream channel. Data for the streams studied for this research are given in Chapter 4.

## 1. 7. WOOD DEBRIS AND THE BIOTA IN RIVERS

### 1. 7. 1. Organic sediments and micro-organisms

There are several comprehensive reviews dealing with wood in rivers (e.g. Bisson *et al.*, 1987; Maser & Sedell, 1994; Bryant & Sedell, 1995; Gurnell *et al.*, 1995) most of which refer to aquatic ecology or aquatic habitats either in the title or in the text. The number of publications in the open literature which include biological data of direct relevance to the role of wood in streams and rivers is relatively small (Gurnell *et al.*, 1995). CWD has however been recorded or measured as a habitat variable in many studies particularly of salmonids in streams (e.g. Gorman & Karr, 1978; Binns & Eiserman, 1979; Heggenes, 1988a; Martin-Smith 1998).

Apart from its roles in the river and floodplain processes CWD provides a physical resource at various levels and scales in the ecosystem including surface structures for colonisation by bacteria and fungi (see Langford, 1983,1990) algae (e.g. Shamsudin & Sleigh, 1994) and invertebrates (Cudney & Wallace, 1980; Smock *et al*, 1985; Chergui & Pattee, 1991; Langford, 1996), physical refugia for invertebrates and fish (e.g. Angermeier & Karr, 1984; Fausch & Northcote, 1992; Harvey *et al.*, 1999), direct food resources for micro-organisms and invertebrates (e.g. Triska & Cromack, 1980; Dudley & Anderson, 1987; Shearer & Webster, 1991; Armitage *et al.*, 1995) and an indirect food resource for fish, preying on colonising or sheltering invertebrates or other fish (e.g. Benke *et al.*, 1985; Smock *et al.*, 1989). Woody debris dams can also be a major influence in temporary inundation of the ecotone habitat by promoting overbank flow (Gurnell *et al.*, 1995). These wet margins of streams harbour both plants and animals that are dependent upon regular and frequent partial inundation and the deposition of dead plant material and sediments.

Reach and sub-reach scale studies show considerable discontinuity in the distribution of finer sediments and organisms caused by CWD. For example, Anderson & Sedell (1979) noted that 25% of the area of the bed in some small streams was wood and another 35% was organic debris stacked up behind the wood. In larger streams the wood or wood-created habitat fell to some 12%. Bilby & Likens (1980) also showed that debris dams contained 75% of the standing stock of organic matter in the first order streams of the Hubbard Brook system in the USA. This fell to 58% and 20% in second and third order streams respectively.

CWD was also reported to be a major contributor to the fine particulate organic matter (FPOM) in some Oregon streams (Ward & Aumen, 1986). Conservative estimates indicated a contribution of $90gm^{-2}$ but the authors suggested that the real figure could be several times that contributed by leaves and pine needles. Aumen *et al.* (1990), using *in-situ* manipulations, subsequently showed that CWD did not affect dissolved nutrient retention directly in streams. CWD and cobbles both adsorbed nitrates and phosphates more readily than finer substrates under experimental conditions but the low densities of CWD *in situ* did not affect the total adsorption significantly.

### 1. 7. 2. Macro-invertebrates

Some 40 taxa of invertebrates have been reported as associated with wood (Anderson & Sedell, 1979) in Oregon streams and the texture, water saturation, species and state of decay were all considered to be factors determining the rate of colonisation and the taxa present on wood in rivers. Wood debris is a direct food resource for some of these invertebrates and aquatic xylophagous invertebrates recorded include the midge, *Brilla* sp. the elmid beetle *Lara avara*, the caddisfly *Heteroplectron californicum*, and the cranefly *Lipsothrix* sp. all of which colonise or use wood in various stages of decay. Amoros & Van Urk (1989) note that the lignicolous chironomid *Symposiocladius lignicola* has disappeared from the Rhine as a result of "snag" clearance and deforestation. In an Australian stream system studied by O'Connor (1992) only two macro-invertebrate species, the chironomids *Stenochironomus* sp. and *Dicrotendipes* sp. were found to be feeding on decaying wood. Most wood-feeders occur among the Diptera. Dudley & Anderson (1987) for example, noted that the Chironomidae contains the most xylophilous species, but the Tipulidae are the most conspicuous of the wood boring insects in waterlogged wood in streams. They also noted that the low standing crop and species richness of insects on wood debris compared with leaves shows there are barriers to exploitation of wood. The advantages of living within logs are protection from freezing and drying, shelter from predation or competition and stability of habitat. Larval densities increase as decay proceeds.

Marked discontinuities of invertebrate distribution at the sub-reach level are associated with CWD. For example, Cudney & Wallace (1980) sampled Trichoptera from snag habitats in a large USA river, mainly roots and branches of *Salix* sp. immersed in the stream. These snags formed the only substrate suitable for the net-spinning caddis larvae in this lowland river with an unstable sand substrate. In fact in such rivers the amount of habitat is probably the limiting factor in controlling numbers. They found that logged streams contained more invertebrates than unlogged streams irrespective of the amounts of wood present as did other workers (Murphy *et al.*, 1981; Carlson *et al.*, 1990). There were no significant differences in overall diversity but densities of Trichoptera, Coleoptera and some Diptera were greater at logged sites than unlogged sites. There was no difference evident for Ephemeroptera, Plecoptera and some other Diptera. The effects of light on the stream's productivity, e.g. algal cover and other vegetation are probably primary causes of variation in invertebrate densities.

Davies & Nelson (1994) concluded indirectly that higher CWD volumes in forested streams were associated with lower invertebrate abundance. They found that the volume of CWD snags decreased with increasing buffer strip width in Tasmanian forests and unlogged forests contained fewest snags. There was an 80% decrease in abundance of invertebrates with the increase in logging and buffer widths. However this was also related to an increase in snags. Densities of all invertebrates and Ephemeroptera were positively correlated with buffer width, but snag volume was negatively related to buffer width. Therefore the densities of invertebrates were negatively related to snag volume. Similarly, as snag volume increased with buffer strip width, brown trout densities also decreased. Clearfelling and buffer-strip width also had effects on leaf input, water quality, algal densities, channel stability, dimensions and stream morphology as well as invertebrate densities. Carlson *et al.* (1990) found that CWD abundance in channels was not significantly different between logged and unlogged stream systems. In contrast macro-invertebrate densities were significantly higher in logged streams. Algal growth and vascular plants encouraged by light were supposedly the main reason. Plecoptera were less abundant in the logged streams. Ephemeroptera, Trichoptera, Coleoptera and Diptera were more abundant.

### 1. 7. 3. Fish

The relationship between fish abundance and CWD abundance varies considerably with stream type and species. Most studies of salmonid habitats and CWD have been at the reach and sub-reach scale and most have been carried out in the north west of the USA. Beechie & Sibley (1997) aimed to identify relationships between the abundance of wood debris and areas of spawning gravel for salmonids. In fact they found no correlation at all between the two in their streams though they noted that other authors had found the reverse. The percentage of gravel in the stream increased with stream width depending upon the gradient. In contrast Sedell *et al.* (1984) noted that CWD created "*high quality salmonid spawning*" by stabilising the channel substrates, notably gravel bars and marginal rearing habitats on bends. Side channels formed and protected by CWD had eight times more juvenile coho salmon than side channels without CWD. The CWD at the upstream end of the side channel protected the habitat from scouring and spates though boulders were as effective as CWD in this function. Most juvenile salmonids were reared in these habitats. The side channels accounted for 6% of the habitat but held 75% of the juvenile fish in one stream, 25% and 55% in another (Sedell *et al.*, 1982, 1984). Bryant & Sedell

(1995) concluded that refuge is probably more important than food where CWD is concerned in natural streams.

Bryant (1985) (*op.cit.* Bryant & Sedell, 1995) also showed similar results in Alaskan streams. The densities of 1+ salmonids increased from 0.09$m^{-2}$ (quartiles 0- 0.1) where there was no wood up to 0.65$m^{-2}$ (0.07 –1.41) where stream channels contained more than 10 pieces of CWD in the reach. The densities in backwaters (1.00 $m^{-2}$, (0.29-1.83)) and side channels (0.32$m^{-2}$) were generally higher than in the main stream. The numbers of samples were small and significance values were not shown. Further, the effects of CWD on the 0+ group were not investigated. The side channels clearly formed significant habitats for 1+ and older salmonids during winter at times of high discharge. Woody habitats in other streams also showed the same pattern in that the main stream contained few salmonids while the side channels contained many (Bryant & Sedell, 1995).

CWD provides seasonally important refugia for Salmonidae at the mesohabitat and microhabitat scale and is an important shelter from high discharges at low temperatures (Bisson *et al.*, 1987). For example, Murphy *et al.* (1986) showed that the density of juvenile coho rose by a factor of 10 from less than 0.02$m^{-2}$ to almost 0.25$m^{-2}$, as CWD abundance rose from 0-4 to over 100+$m^3$ per reach. Mean parr density rose from 0.1$m^{-2}$ to 0.37$m^{-2}$ as CWD loadings in streams draining clearcut, old-growth and partially buffer-strip blowdown forests rose from mean loadings of 0.05 to 0.12 $m^3m^{-2}$. Bisson *et al.* (1987) also noted that the storage capacity increases caused by CWD allow increases in invertebrates feeding on leaf detritus. Murphy *et al.* (1986) in their studies of the effects of logging on reaches of forested streams showed that CWD volumes were greater in the buffered streams than either old growth or clear cut. Fish densities also varied with species, reach and stream type. For example, coho salmon fry (*Oncorhynchus kisutch*) were densest in buffered and clearcut streams in summer and winter where CWD was both at its densest and least dense. Parr (1+) were densest where CWD was most abundant in the buffered streams. Dolly Varden parr (*Salvelinus malma* (Walbaum)) were most abundant in the densest CWD (buffered) but trout (*Salmo trutta*) parr were generally less abundant in the higher wood densities.

Cover either as CWD or other categories (roots, undercuts etc.) was considered as more important for fish in the winter than in the summer. In summer food abundance was considered to override the need for shelter from predators.

Hortle & Lake (1983) found that the numbers, biomass and species richness of fish were significantly correlated with the number of CWD pieces (snags) present and areas of slack water in the Bunyip River in Australia. Channelized sites contained fewer fish, fewer species and a lower biomass than the non-channelized reaches. The absence of habitat diversity was believed to be the reason for the poor fish fauna and stock. Eels (*Anguilla australis*) and trout (*Salmo trutta*) were the species which showed the strongest correlations with CWD.

On the reach scale and above, debris dams cause biological changes in streams where they form obstacles and steps or waterfalls (Bilby, 1981). In such situations they may impede migrating invertebrates or fish apart from acting as retention structures for sediments and organic materials (see Section 1.7.1). Blockage of migration can occur at very low flows and with very dense wood jams (Bisson *et al*., 1987), but as soon as flows increase the accumulations of CWD become passable. The migrations of smaller fish may be hindered but usually dams are in a matrix form with sufficient spacing between individual pieces to allow free passage. Although these dams are localised phenomena serious blockage of migratory species could affect distribution on the catchment scale.

Rotting wood may also have directly adverse chemical effects in streams for example as an inhibitor of algal growths (Ridge *et al*., 1995). Brown rotted wood (BRW), the residue of an attack by brown-rot fungi was found to inhibit algal growths in cultures by up to 90%. Bisson *et al*. (1987) also noted that organic acid leachates from the western cedar (*Thuja plicata*) have been associated with low pH and other chemical changes which led to poor fish populations in streams. Insects were more tolerant than coho salmon and eggs and fry of this species were most sensitive. Other species e.g. Sitka spruce (*Picea sitchensis*) and western hemlock (*Tsuga heterophylla)* also produce toxic leachates which have been tested on fish and invertebrates and showed adverse effects on these organisms.

## 1. 8. WOODY DEBRIS, HABITAT DIVERSITY AND CONSERVATION

The concept that community diversity is related to habitat diversity originated from the work on bird communities and vegetation (MacArthur & MacArthur, 1961; MacArthur, 1964) and was later applied to aquatic communities including fish, (e.g. Gorman & Karr, 1978). However, although the principle may be generally acceptable, there are provisos for streams, apart from the large scale limiting factors (see Sections 1.2, 1.3). For example the definition of habitat diversity is difficult for running waters and has been measured in such a variety of methods that there is no universally applicable measurement unit. In some assessments only physical habitat (e.g. Gorman & Karr, 1978; Heggenes, 1988a) has been measured, in others water chemistry has been included (e.g Binns & Eiserman, 1979; Milner *et al.*, 1985). Small scale discontinuities in the channel may also have wider implications. For example Cowx & Welcomme (1998) note that a relatively small obstruction in a stream, with height of only 40cm restricted the species richness of the fish community upstream to 1 compared with 8 downstream by destroying free movement. Turnpenny *et al.* (1988) also indicated that a 5m waterfall prevented trout migrating upstream and acidity also limited the distribution of eels and trout in a Scottish stream system. Clearly in such cases physical habitat diversity at reach or sub-reach level is of little consequence to the fish populations above the obstacles.

Maitland & Campbell (1992) concluded that the major single cause of the extinction of populations of fish is the destruction of habitat. In rivers much of this has come from pollution, land use changes and the physical modification of the channels. Introductions of fish by humans have also changed the composition and diversity of many communities. Although habitat diversity is regarded as vital to maintain species diversity on the broader scale, habitat diversity *per se* may not be the vital factor in the maintenance of particular species assemblages at the stream level and below. In some cases low disturbance and the maintenance of a particular level of habitat diversity may be the key to the maintenance of pristine communities or populations while increasing habitat diversity could allow other species to colonise and destabilise the original pristine assemblage. In forested streams it has been long known that clearfelling, forestation and the reduction of riparian vegetation have been responsible for such changes (e.g. Burton & Odum, 1945;

Hynes, 1970). Gotelli & Graves (1996) compared results from various stream studies and noted that stream fish communities could show unexpected trends in diversity depending upon the methods of analysis. They noted that for stream fish assemblages the lowest diversity could be shown to be in high-order stable streams instead of the reverse pattern suggested by earlier studies (Shelford, 1911; Huet, 1959; Hawkes, 1975). The general inference was that "*species diversity may be highest in non-equilibrial assemblages that are frequently disturbed*". Whatever the underlying principle on the larger scale, in small temperate streams the typical fish fauna is of limited species richness and diversity on the reach and within-reach scales (e.g. Mann, 1971; Hawkes, 1975; Langford & Hawkins, 1997).

## 1. 9. WOODY DEBRIS AND COVER FOR FISH

The measurement of "cover" or refugia for fish is one of the inconsistencies of habitat analysis, though there have been some attempts to standardise the methods. CWD is one of the major categories of cover in most studies but here again, methods of assessing the "abundance" of cover vary widely (Heggenes, 1988a). The definition of cover most widely used is " *structures which obscure areas of the stream bed from overhead vision*". This usually assumes that protection from predation is the main function of cover. However, for territorial species such as salmonids, objects in the stream either in two or three dimensions can reduce visual contact and thus reduce the probability of aggressive territorial behaviour (Bisson *et al.*, 1982). Also cover in two or three dimensions can provide refugia from which predators can forage.

Cover usually implies some physical structure such as a bank overhang or undercut, tree-roots, instream vegetation, trailing marginal vegetation, rock or item of wood debris (dead snags) or other debris (Binns & Eiserman, 1979). It can also include turbulent water (Binns & Eiserman, 1979; Heggenes, 1988a; Heggenes & Saltveit, 1990), though this category is by no means universally used (e.g. Heggenes, 1988a; Williams *et al.*, 1996; Inoue & Nakano, 1998). The measurement of cover has varied from visual estimates of areas of bed obscured (e.g. Heggenes & Saltveit, 1990; Fausch & Northcote, 1992; Williams *et al.*, 1996; Harvey *et al.*, 1999) to measurements of undercuts, overhangs, tree-root matrices and CWD by width and length (Murphy *et al.*, 1986; Nielsen, 1986; Inoue *et al.,* 1997) to detailed mapping (Lewis, 1969; Hunt, 1976; Moore & Gregory, 1988) and planimetry (Elser, 1968). It is commonly presented as percentage cover of the bed area or bank length or as an absolute value of area (see Milner *et al.*, 1985; Heggenes, 1988a).

By definition "cover" also implies some physical space or refuge in which a fish of a given size may reside either permanently or temporarily. The relevant size of any area of cover thus is related to the size of the fish. As most habitat studies have related to salmonids and usually larger salmonids (Binns & Eiserman, 1979; Milner *et al.*, 1985; Heggenes 1988a) the definitions of cover have often been restricted to those which will suit individuals or groups of salmonids over 1 year old. More recently studies in the UK have concentrated on other species and given varying definitions of both instream and riparian cover (e.g. Copp, 1990; Ibbotson *et al.*, 1994; Copp & Bennetts, 1996; Prenda *et al.*, 1997; Watkins *et al.*, 1997; Garner *et al.*, 1998).

For CWD accumulations to provide "physical cover" for any given species or individual the component pieces of the accumulation must be arranged in such a manner as to provide water-filled spaces between, i.e. in some open matrix formation. Thus the actual volume of wood in any accumulation may not be as important in determining the available refugia as the space to wood ratio. It is likely therefore that any studies simply using total volumes or biomass of wood present in relation to fish biomass or density may be based on an incorrect premise. Some form of classification of wood matrices based on the space/wood ratio may therefore be necessary. Methods of quantifying CWD in streams are reviewed in Chapter 3.

## 1.10. AIMS AND STRUCTURE OF THE THESIS

As already stated (Section 1.1) the initial aim of the research was to quantify the spatial and temporal relationships between CWD and the fish populations of the

Highland Water and associated streams by detailed studies at the sub-reach (channel-unit) and reach level. In addition the effects of changes in land use and low CWD loadings at the stream and catchment level would be determined by comparisons with other New Forest streams.

It was clear from the literature that despite the work on CWD in north America, Australia and Japan, relatively few studies have quantified the effects of the presence of CWD in streams from the perspective of the total fish community on the reach scale or smaller and on the different life-stages of species in assemblages. Further, there is little information on the differential effects of CWD on all species in an assemblage which might be used to manage streams both for species of commercial and conservation importance. As shown above, most of the published data concern salmonids even where other species were present in a stream. There are no data from studies in UK streams despite the geomorphological work described above and the controversy about the management of CWD in streams (Lappin, 1991; Langford, 1996).

The thesis proceeds from the general review in this chapter (**Chapter 1**) to focus on the unique study area, the New Forest in **Chapter 2**. **Chapter 3** reviews methods of habitat analysis and determining CWD abundance. It also describes the methods and defines the terms used throughout the thesis. The streams used for the main study are described in **Chapter 4** and their physical, chemical and biological characteristics compared. This chapter also includes a review of earlier studies on CWD in these streams and an analysis of the effects of CWD on habitat diversity at the within-reach and reach scales. Comparisons between streams in forested and de-forested catchments are made. The abundance and size-distribution of individual fish species in relation to habitat structure and CWD is described in **Chapter 5.** In **Chapter 6** the abundance data are used to explore the composition of fish communities at the sub-reach, reach and stream scales in relation to channel structure, CWD and catchment disturbance. Salmonids are the focus of a special-case population level study in **Chapter 7** because of their use of the different components of the stream habitat at different stages in their life history. The final chapter, **Chapter 8**, then synthesises the data from the whole study in both an ecological and managerial context and outlines potential management strategies and future research for the New Forest. A provisional model is proposed to relate the abundance of CWD dams to the abundance of fish in a hypothetical 100m reach of a wooded stream. The relevance of the studies to stream management on the regional, national and international scale is noted. Potential programmes for fundamental ecological research and applied studies are briefly outlined.

Figures and tables for each chapter are interleaved as near as possible to their first mention in the text. Appendices I, II and III are located at the end of the thesis, following the references. Appendix I contains tabulated data on locations of sites, dates of sampling and fish catches. Appendix II (Parts 1 & 2) comprises two OS maps showing the locations of all the sites sampled during 1996-2000. Appendix III is a 1.4 Mb computer disc containing the raw data from each fish sample, population estimates, standard errors and biomass estimates in MS Excel worksheet format.

# CHAPTER 2

# ECOLOGICAL ASPECTS OF NEW FOREST STREAMS: AN OVERVIEW

## 2. 1. INTRODUCTION

The New Forest, in the south of England (Fig. 2.1), is a unique area of Britain. It contains a collection of habitats now rare in lowland Europe. It is also under severe pressure from development and recreational activities which threaten the continued existence of the habitats despite apparent protection from new European legislation. The formation of a new National Park centred on the Forest will also increase recreational pressure. The acidic and circumneutral soils of the New Forest are drained by a number of small self-contained stream and river systems, all of which have their sources within the Forest boundary and drain to the sea or to chalk rivers on the Forest borders.

Because of its unique social and ecological history, the Forest has been the subject of many biological studies mainly concerned with the terrestrial flora and fauna and the populations of birds and large mammals (see Tubbs, 1968, 1986). To date, relatively few studies of the plant and animal communities of the streams have been published. In 1992, a series of long-term studies of the invertebrate communities was begun and in 1996 studies of the effects of timber debris on fish in the streams also began with the aim of providing scientific data to aid the future management and conservation of the stream system. As part of the background to the work, this chapter describes the streams of this unique area of Britain and reviews the published and some unpublished chemical, physical and biological information that is currently available. This chapter formed the basis of a published paper (Langford, 1996) and is only slightly modified from this publication. As a review, the chapter includes names of species of plants and animals recorded in various publications. No authorities are given where they were not included in the original publication. In

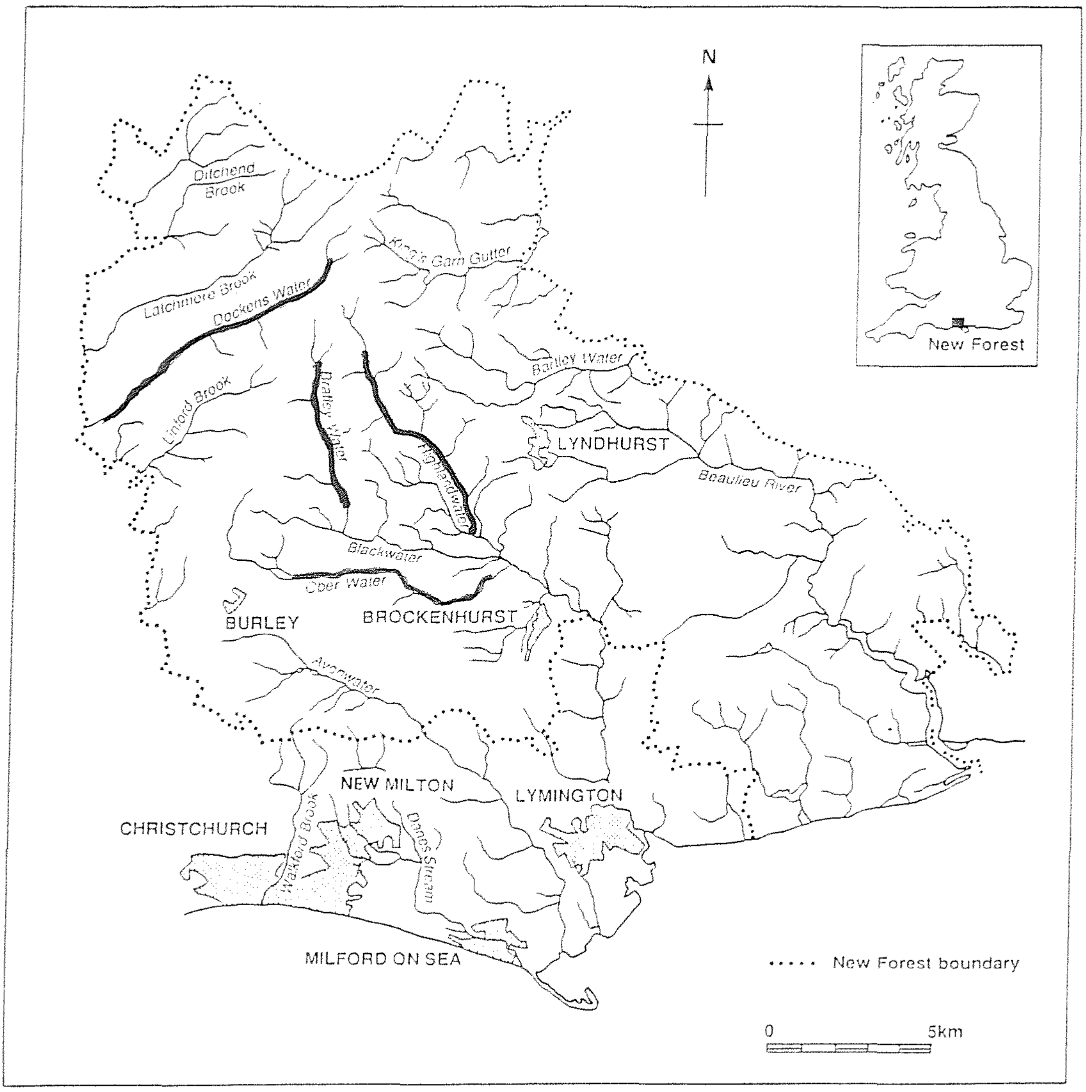


**Fig. 2.1. Major river systems of the New Forest, Hampshire, in Southern England. Main study sites are on Highland, Bratley, Ober and Dockens Waters**

***( Reaches sampled are indicated by darker lines)***

subsequent chapters authorities are shown where they refer to species directly relevant to the current research.

## 2. 2. DESCRIPTION AND HISTORY OF THE AREA

The New Forest was established as a hunting forest sometime between 1066 and 1086 by William I. Since that time, the ecological and sociological evolution of the Forest has resulted in the creation of a unique collection of habitats, making it one of the most important conservation areas in Northern Europe. It was given a Grade 1 listing in the Nature Conservancy Council's Nature Conservation Review (Ratcliffe, 1977), and has recently been declared a site of international importance within the terms of the RAMSAR Convention (Anon, 1976). Much of the Forest is also a Site of Special Scientific Interest (SSSI) and is to be protected by the European "Habitats" Directive, ratified in 1995.

The New Forest includes *"the largest area of unenclosed wild or 'unsown' vegetation in lowland Britain"* together with large tracts of lowland heath, valley mire and ancient pasture woodland, all three of which were once common in Northern Europe but are now *"fragmented and rare"* (Tubbs, 1986). It has a rich and diverse flora and fauna. For example, 46 rare species of plants occur within its boundary of which eight are listed in the British Red Data Book (Perring & Farrell, 1983). Some 55% of the 2251 species of British Lepidoptera and almost 48% of the 1539 species of British Coleoptera, have been recorded in the Forest, and 27 of the 38 species of British Odonata breed in Forest waters (Cowley, 1950-51; Welstead & Welstead, 1984; Tubbs, 1986).

The boundary of the New Forest encloses some 37,900 hectares, though the proposed New Forest Heritage Area would include a total of 54,400 ha. which may eventually increase to 57,300 ha (Land Use Consultants, 1991; Anon 1992b; New Forest Committee 1994). Because of the unique nature of its habitats and ecosystem, the Forest has been the subject of historical, sociological, ecological and other scientific studies over many years (Flower, 1977) and much of the information has been summarized and reviewed in two excellent books (Tubbs, 1968; 1986).

## 2. 3. HISTORICAL INFORMATION ON STREAMS

The main river systems of the New Forest are shown on Fig. 2.1. In addition to this there is a finer network of trickles and small drainage channels in the woodlands and across the open "lawns" many of which are flooded only temporarily during heavy rainfall or high river discharge. Even so, these can form part of the total habitat of some opportunistic species of fish such as the minnow (*Phoxinus phoxinus Linnaeus* 1758) as will be seen in Chapter 5.

Historically, general descriptions of Forest streams occur in the writings of naturalists and in various travellers books (e.g. Cornish, 1895; Begbie, 1934; Everard, 1957). For example, the channel and catchment of Latchmoor Brook, on the western slope of the Forest, were described as "*deep grottos, fox holes (so large that they look more like dens for wolves) and bogs which heave*" (de Bairacli-Levy, 1958). The stream and spring waters are also reputed to have medicinal advantages and Shore (1890) describes the sources of some Forest streams and the curative properties of the iron-rich chalybeate springs in their catchments. Such a spring, at Fritham in the northern part of the Forest, was at one time known as Iron Well or Lepers' Well and was noted for the treatment of leprosy in the Middle ages. Brierly (1886) analysed the water from this well and noted that there "was a copious deposit of a reddish brown colour at the bottom of the bottles" but the water was "of a pale green colour and free from odour". He also found no dissolved iron in the water but high concentrations in the deposits. Similar deposits are still found on the stream bed at the outflow of the spring as a result of the rapid oxidation of dissolved iron in the water. In the 19th century the waters of the same spring were still used for medicinal purposes, but at that time to treat mange in dogs. Near Lymington, in the southern part of the Forest, the similar ferruginous springs which feed Passford Brook were noted for the treatment of ophthalmic disorders.

The Forest streams do not have a history of consistent organic pollution, apart from intermittent problems caused by farm effluents and small sewage works (e.g. New Milton Advertiser & Lymington Times, 1995). The worst and most persistent inorganic effluent problems originated with the Schultze Gunpowder factory at Fritham on the upper reaches of Latchmoor Brook. This factory was reputed to cause serious contamination and fish mortalities in the latter half of the 19th century (Pasmore, 1993). The problems stemmed from the frequent leakages of nitric and sulphuric acids used in the manufacture of explosives. Begbie (1934) described the effects of pollution in the Latchmoor Brook and found it containing *"ale coloured water with little trout where once the proud salmon used to spawn". " The powder mills ...... so tainted the water that cattle refused to drink it and the fish, holding their noses, fled, in the case of the salmon never to return".* The gunpowder works also used a great deal of water which was obtained from a reservoir formed by damming the brook and augmented by water obtained by tapping nearby springs. The effluent problems ceased with the closure of the factory over 50 years ago. The general water quality in most streams is thus consistently high (Sear & Arnell, 1997; Environment Agency, Public Register 1988-1998).

At the end of the 19th century the streams were not highly regarded as fisheries, and De Crespigny & Hutchinson (1899) concluded that *"practically speaking there is no fishing in the Forest" and " the fishing in the Forest... may be said to be nil".* At the same time they noted the large trout (probably sea-trout *Salmo trutta* (Linnaeus 1758)) spawning in the Avon Water (Fig.2.1) but did not comment on their availability to anglers. These authors did, however, note that bankside trees and timber debris were a hindrance to fly-fishers, and the role of timber in streams is still debated by anglers and conservationists (Lappin, 1991; Gregory & Davis, 1992). Humby (1961) also describes the clearance of woody debris from the Lymington River as part of the management of the river, and as a means of increasing the area of spawning gravels and to easing the passage of upstream-migrating sea-trout.

Several contentious issues related to the management of streams and drainage in the New Forest have a long history, particularly with regard to the ecological role and effects of large timber debris, and the effects of land-drainage works, and these issues have not been fully resolved (Gregory *et al.*, 1985; Tubbs, 1986; Lappin, 1991; Gregory & Davis, 1992). Although there are data from other regions of the world (e.g. Gregory, Gurnell & Petts, 1994), no specific studies of the biological role of timber debris and the effects of its removal had been carried out until this present work began, though some work on the micro-algae noted the importance of wood debris as a substrate (Moore, 1997a).

Published information on the plant and animal communities of the streams comes mainly from comparative studies of single streams or from localised studies of particular groups of organisms, and these are described later in this chapter. There are also records of stream-living insects in the Proceedings of the Hampshire Field Club, reaching back to the 19th century, though most of these refer to adults caught at various distances from the actual stream channels (Hampshire Field Club, 1890 et seq.).

## 2. 4. DESCRIPTION OF THE WATERCOURSES

### 2. 4. 1. Topography, geology and climate

Tubbs (1986) describes the New Forest, when seen from the air, as " *a series of eroded flat terraces, highest in the north, lowest in the south. The middle terraces are scoured into wide hollows drained by two south-flowing stream systems which empty into the Solent.* " The rocks beneath the New Forest are mostly sedimentary clays, which were once capped by gravel or brickearth which has subsequently been eroded to reveal the underlying strata. There is a wide variation in the texture and chemistry of the soils, which is reflected in the variation in vegetation across the Forest (Tubbs, 1986). The characteristic rocks and soils are listed in Table 2.1 together with chemical data from streams which drain soils on them. In general, the soils become richer and more productive toward the fringes of the forest, and these fringes merge into the more improved pasture and agricultural lands of the chalk-stream valleys to the east and west and the Solent to the south. Some areas of these fringes are urbanised to varying extents and the lower reaches of the streams receive road and urban surface drainage.

**Table 2.1. Chemical constituents of New Forest streams in relation to their underlying geology; values are means from a minimum of 5 samples.** *(From Langford, 1996 after Le Rossignol, 1977)*. **Lead (Pb) and Copper (Cu) concentrations were below the levels of detection.**

| Rock type | Conductivity at 25°C (µS/cm) | Total dissolved solids (mg/l) | Ca (mg/l) | Mg (mg/l) | Na (mg/l) | K (mg/l) | Li (µg/l) | Fe (µg/l) | Mn (µg/l) | Zn. (µg/l) |
|---|---|---|---|---|---|---|---|---|---|---|
| Headon Beds | 396 | 428 | 41.5 | 12.0 | 27.5 | 8.44 | 25 | 2300 | 6000 | 47 |
| Headon Beds | 486 | 540 | 62.5 | 16.0 | 42.0 | 4.7 | 33 | 460 | 1525 | 33 |
| Barton Clay | 156 | 193 | 11.0 | 6.3 | 12.7 | 3.2 | 36 | 1840 | 119 | 34 |
| Plateau Gravel | 120 | 124 | 5.0 | 4.5 | 12.7 | 1.05 | 25 | 1250 | 113 | 30 |
| Barton Sand | 156 | 170 | 6.0 | 5.2 | 16.9 | 3.20 | 14 | 630 | 80 | 25 |
| Barton Sand | 147 | 212 | 13.5 | 6.0 | 12.0 | 4.35 | 24 | 4800 | 381 | 72 |
| Barton Clay | 373 | 349 | 26.0 | 25.8 | 17.9 | 6.10 | 86 | 460 | 440 | 440 |
| Barton Clay | 129 | 123 | 6.5 | 4.0 | 11.2 | 2.43 | 16 | 2010 | 116 | 121 |

The climate of the New Forest is moderately warm and wet for Britain, with temperatures and rainfall towards the middle of the range for the country as a whole. The mean annual temperature over the region varies between 9°C inland to 10.5°C in the coastal belt, compared with 7°C in the Shetlands and 11°C in the extreme south west (Anon, 1990). The relative humidity averages about 85% and exceeds 95% for 25% of the year. Average annual sunshine is about 1500 hours, compared with 1100 hours in Scotland and 1928 hours in the south west of England. Average rainfall for the region is 806mm (1957-1981) with a range of 509mm in the driest year to 993mm in the wettest year. All of the New Forest lies within the 800mm isohyet. Over the period from 1957 to 1981 snow lay in the region for an average of 5 days each year, though there were differences of 1-2 days between the higher parts of the Forest and the warmer areas nearer the coast.

### 2. 4. 2. Geomorphology and hydrology

The main watershed in the New Forest runs approximately in the direction of north to south, and the highest point is at 128m (OD) on the north-western escarpment. Six streams flow westward from this watershed to the River Avon (Fig. 2.1). They are mostly short, run approximately parallel to each other, and have few tributaries. In contrast the river systems which run eastward to the Test and south to the Solent have many small tributaries and tend to be longer. The more developed and urbanized lands of the southern and eastern fringes of the Forest are drained by a small number of short streams which mostly run directly to the Solent (part of the English Channel, see Fig. 2.1). The main categories of land bordering stream-reaches are: open heath, deciduous woodland, conifer plantation, forest lawn, and improved agricultural land.

None of the rivers are more than 30 km long. The highest source of any stream is at around 90m (OD), though most rise at 50m (OD) or less. The maximum overall gradient is ca. 1.0% in the short streams of the western catchment while the longer streams and those of the southern fringes have overall gradients of less than 0.6%. Many reaches of all the streams have been artificially straightened, channelized and regraded since the 1840s, mainly for improving drainage of forest lawns or woodlands or for protecting forest roads. Small ditches, most of which have been dug in the last 100 years, drain the Forest lawns and plantations.

Stream channels are mostly less than 7 m. wide. Freshwater reaches that are more than 7m. wide and 1m deep at dry-weather flows are relatively short, and are mainly limited to the lowest reaches of the Lymington and Beaulieu rivers. Some scoured pools in the middle reaches may exceed 1m depth.

Typically, maximum current velocities during dry-weather flows are less than 0.5 metres per second. For example, in a 1992 survey by the author, 62% of the 87 reaches where current velocities were measured in mid-stream showed values of between 0.11 and $0.5ms^{-1}$ and 18% showed values over 0.5 $ms^{-1}$. The highest current velocity recorded so far by the author during dry-weather flows was 0.76 $ms^{-1}$ in a channelized reach of a small tributary of the Lymington River.

A typical stream reach comprises riffles, usually less than 30 m. long and 5 to 15 cm deep, interspersed with deeper pools and undercut banks where the water may exceed 1m in depth and where leaves and twig debris accumulate. The sequence of riffles and pools is very similar to that found in other streams, with riffles occurring at intervals of about 5 to 7 channel widths (Gregory, Gurnell, Hill & Tooth, 1994). Tree roots and overhanging grasses trail in the water at the margins and both marginal and in-stream aquatic vegetation occurs where the tree canopy is sparse or absent. Accumulations of timber debris (CWD) occur at irregular intervals in some streams whilst other nearby streams are relatively clear of CWD as a result of forest clearance and stream maintenance for drainage purposes.

The streams are generally of low volume, though they rise and flood very rapidly after heavy rainfall and can be described as *"flashy"* (Gurnell *et al.*, 1985; Gurnell & Gregory, 1987). Fig. 2.2 shows the effects of rainfall on the discharge of a stream on two separate occasions. Peak flows occurred very soon after the rain began and the stream level subsequently fell rapidly over a few hours. On a seasonal basis, the streams with mires and bogs in their catchments tend to have more consistent summer discharges than those without these features (Tubbs, 1986). In summer,

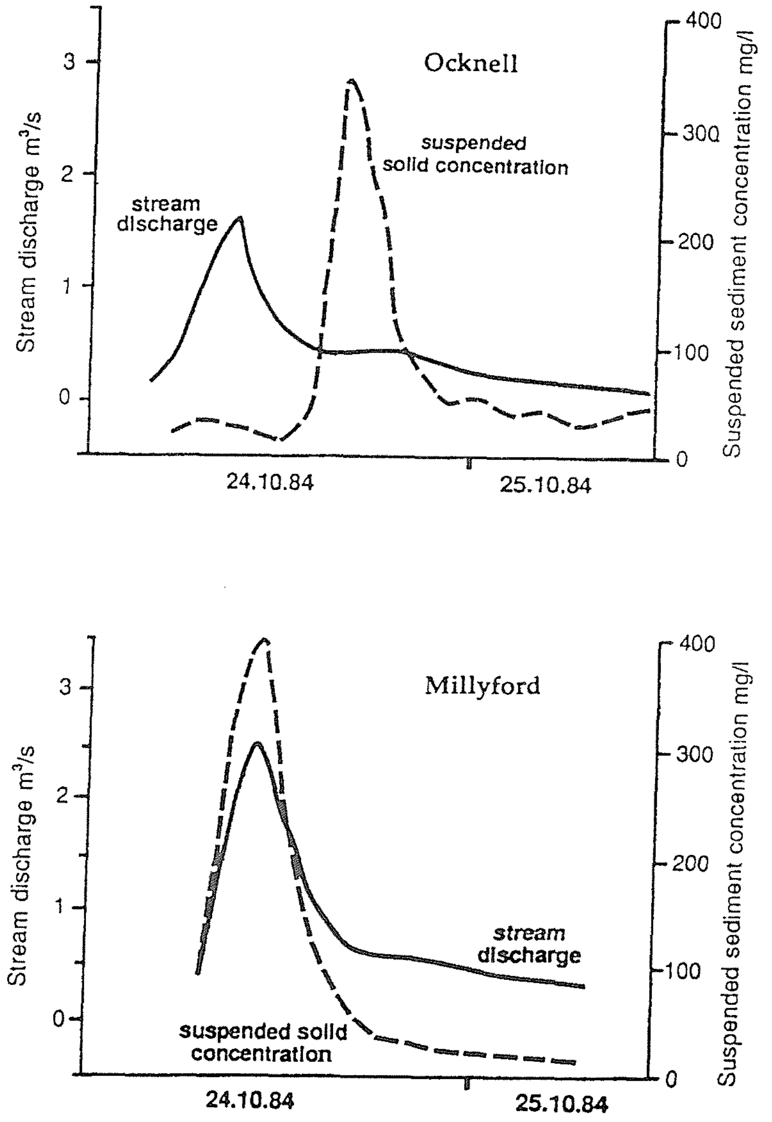


**Fig.2.2. Effects of heavy rainfall on stream discharge (m3s-1) and concentrations of suspended solids (mgl-1) at two sites (Ocknell and Millyford) on the Highland Water.(redrawn from Langford, 1996, After Futter, 1985)**

some of the streams, particularly those flowing westward, are reduced to pools separated by dry gravel reaches, and in several recent years the upper reaches of several streams were completely dry for up to three months, after about six weeks without rain. Previous records show similar patterns (Shore, 1890).

The effects of human influence on the hydrology of the streams has not been studied in great detail, though Gregory (1992) noted an increase in the peak discharge of the Highland Water near the head of the catchment, which he associated with increased road drainage following road improvements in 1980.

The substrata of Forest streams consist mainly of sand or clay, covered with small gravel or pebbles. Substrata with gravel and pebbles ranging from 10 to 65mm in diameter are moderately frequent but cobbles over 12cm in diameter occur rarely and are only found consistently in the reaches of streams on the gravels. The phi-scale range is typically from 3 to -6. The substrata are mobile for the most part though in some reaches sediments are stabilized by weedbeds or compaction. There is no exposed bare bedrock and no large rocks or boulders in any of the streams. Where the channels have been deepened or straightened the substrata are mainly loose, coarse sands or gravels. The largest structures in the stream channels are large pieces of wood debris (CWD) either singly or as accumulations (debris dams) of various sizes (see Chapter 4).

The streams remove and carry considerable quantities of sediments during spates. Tubbs (1986) quotes data (Tuckfield, 1964, 1973, 1976, 1980) showing an annual removal 0.64 to 0.75 $m^3$ $yr^{-1}$ of gravel and smaller sediments in very tiny streams. Human influences, either from drainage work or through the wear from the movement of people or cattle, increase the rates of erosion and even very small drains can erode 0.1 to 0.3 $m^3$ $m^{-1}$ $yr^{-1}$. In extreme cases this can reach 0.5 $m^3$ $m^{-1}$ $yr^{-1}$. For example, in one gully created by such wear, 1084 $m^3$ of gravel was eroded away over 10 years at an annual rate of 98.5 $m^3yr^{-1}$, and in a second gully, 280 $m^3$ of material was eroded away in 2 years.

Channelization has also led to changes in both sedimentation and the growth of aquatic plants in unshaded streams. Brookes (1983) showed that channelization of the Ober Water caused increased sedimentation and the growth of *Elodea* sp. over a 175m stretch downstream of the channelized reach. *Elodea* is also common in similarly altered reaches in other stream systems in the Forest.

### 2. 4. 3. Timber debris dams

Timber debris accumulates in many reaches of New Forest rivers, forming as debris dams of various sizes and configurations (Gurnell & Gregory, 1984; Gregory & Davis, 1992). These dams, usually composed of tree trunks and large branches augmented by smaller branches, can have significant effects on the channel processes, including sedimentation, the travel times of flood peaks and channel migration (Gurnell & Gregory, 1984; Gregory *et al.*, 1985; Gregory, Gurnell, Hill & Tooth, 1994). Impoundment of water by such dams is also reputed to have adverse effects on the drainage of Forest lawns and plantations and hence on the grazing of livestock and survival of trees. Furthermore, anglers believe that the migration of sea-trout to their spawning reaches is impeded by the dams (Anon, 1992a) though there are no scientific data to support or refute these suggestions.

Gregory *et al.* (1985) showed that timber debris dams, with a density averaging 1 per 27 m of channel in the Highland Water, extended the travel times of flood peaks and affected the channel processes in the vicinity of the dams. Over a distance of 4028 metres, the presence of some 93 dams causing varying degrees of impedence, delayed the smaller flood peaks by some 100 minutes and the larger flood peaks by about 10 minutes when compared with unimpeded travel.

A detailed survey of the Lymington River basin (Gregory *et al.*, 1993) covering 110.4 km$^2$, found a total of 754 debris dams which generally decreased in frequency of occurrence with distance downstream from the sources of feeder streams, though the peak occurrences were in reaches between 3 and 10 km downstream (Fig. 2.3). Gross and net loading of timber in the streams followed a similar pattern. The number and total loading of timber in dams also varied with land use in the catchments, with deciduous forest being the greatest contributor. The net loadings measured, which included only timber actually in the stream and impeding flow, ranged from 0.03 kg.m$^2$ to 2.49 kg.m$^2$. The authors estimated that the net timber

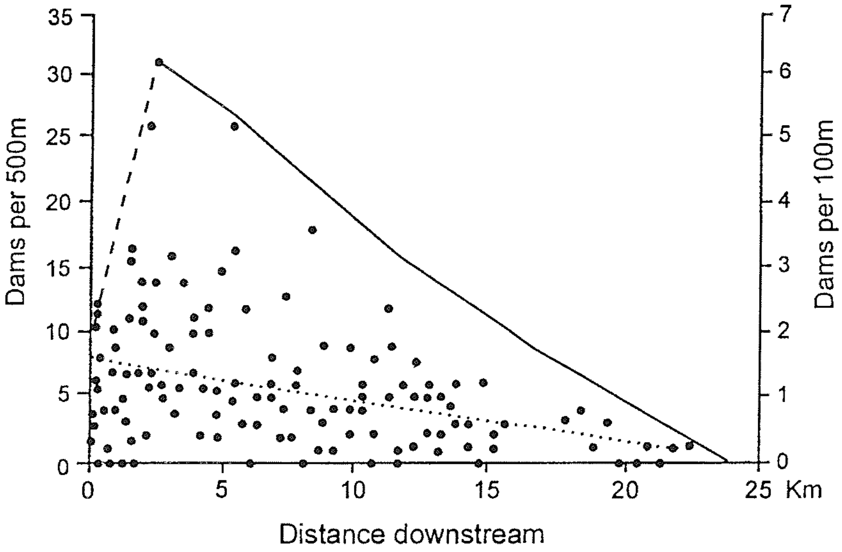


**Fig. 2.3. Numbers of debris dams occurring in 500-metre reaches of the Lymington River and Highland Water (*Redrawn from Langford, 1996 after Gregory et al.*, 1993).**

loadings were only 7% of those which could occur if the streams were not managed and the timber was not cleared.

The differences between CWD loadings in streams were marked and could be considered to affect the stream habitat and consequently the various stream ecosytems. For example in the Highland Water, Bratley Water and Bagshot Gutter, all with mostly wooded catchments, the average numbers of dams per 500m were 4.99, 5.44 and 11.84 respectively. Net timber loadings were 0.59, 0.43 and 2.49kgm$^{-2}$. In contrast, in the Ober Water, with a large proportion of open lawn in the catchment, there were 2.49 dams per 500m with a net loading of 0.11kgm$^{-2}$. These differences have considerable implications for the fish communities (see Chapters 5, 6, 7).

### 2. 4. 4. Sedimentation and suspended solids in streams

Suspended sediment concentrations during dry-weather flows are typically between 5 and 25 mgl$^{-1}$, frequently rising to over 200 mgl$^{-1}$ during spates and reaching over 1000mgl$^{-1}$ at times (Sear & Arnell 1997). Fig. 2.4 shows the relationships between suspended sediment concentrations and stream discharge for two sites on the Highland Water in the 24 hours following heavy rainfall in October 1984 (Futter, 1985) (Fig. 2.2 shows the temporal course of the same event). The maximal concentrations (335 mgl$^{-1}$ and 394 mgl$^{-1}$) were considered as unusual because the rainfall occurred after a long dry spell. A major rainstorm in the following month produced peaks of 214 and 228 mgl$^{-1}$ of suspended sediments, which were regarded as more characteristic of Forest streams under conditions of high discharge. The suspended solids load peaked at 13.65 kg. ha$^{-1}$ hr$^{-1}$ during this episode.

The removal of timber debris increases the current velocities (Fig. 2.5) and the movement of sediment along a stream channel, the amount of sediment stored and the duration of storage decreases (Beschta, 1979; Gregory, 1992). At the same time localised bank erosion increases and the sequence of pools and riffles becomes less clearly defined.

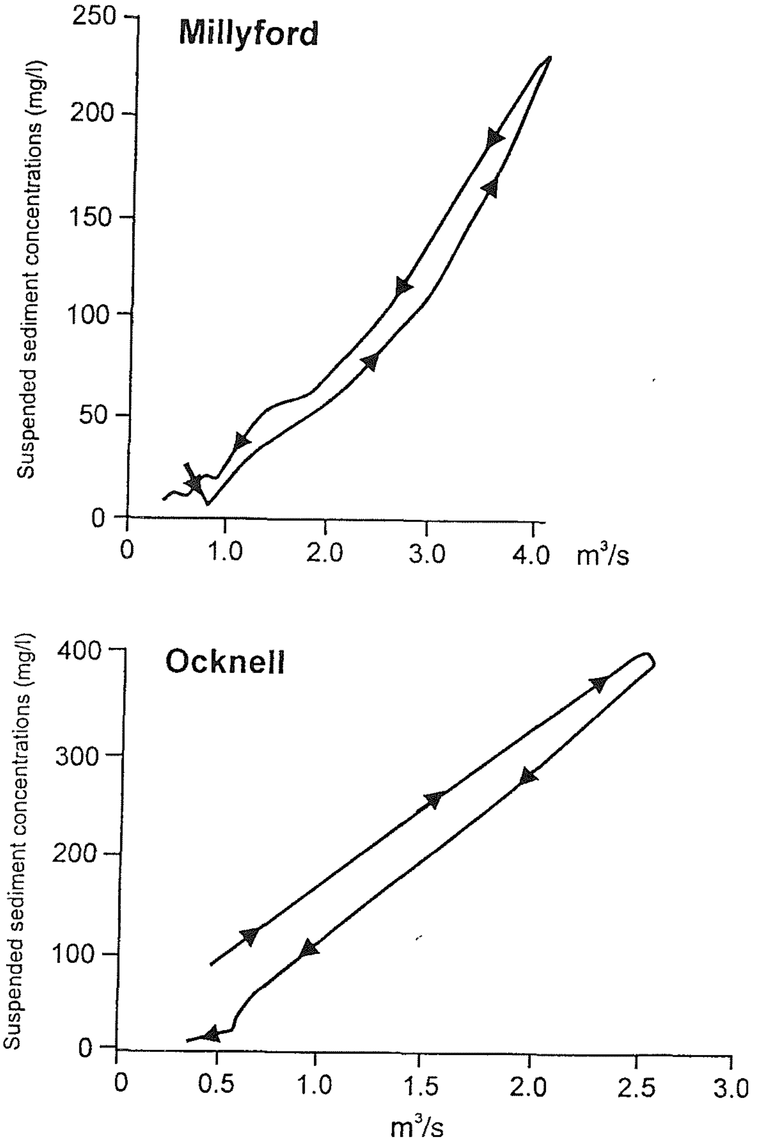


**Fig. 2.4. Concentrations of suspended solids (mgl$^{-1}$) in relation to stream discharge (m$^3$ s$^{-1}$) at two sites (Millyford and Ocknell) on the Highland Water after heavy rainfall.** ***(Redrawn from Langford, 1996 after Futter, 1985*****).**

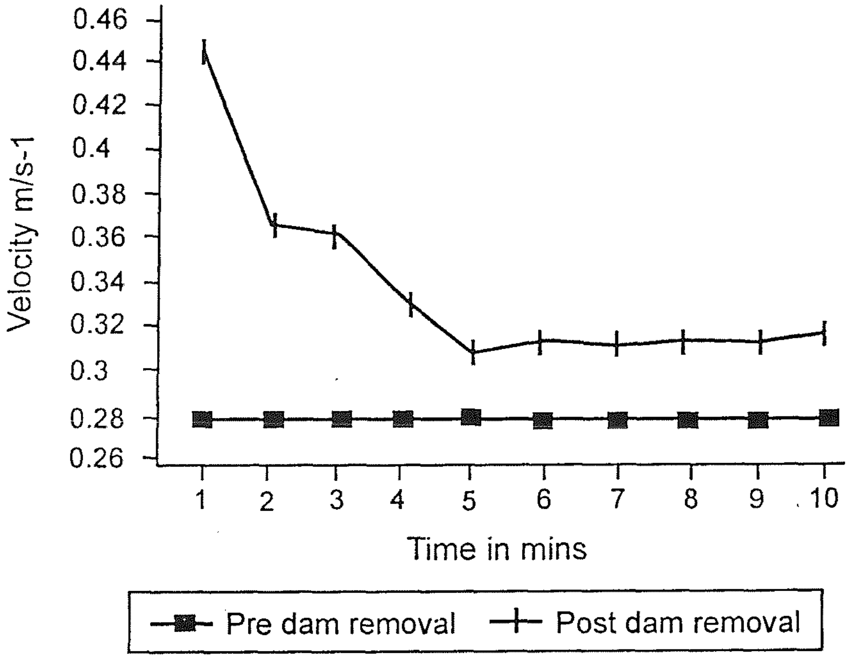


**Fig. 2.5. The immediate effects of removing a debris dam on the current velocity (m s$^{-1}$) measured some 10 metres downstream.**
***(Redrawn from Langford, 1996 after Evans, 1995).***

Pollen and spores from plants form a significant part of the suspended solids in New Forest streams, ranging in one study from 230 grains.ml$^{-1}$ at flood discharges to less than 1 grain ml$^{-1}$ during baseflows (Brown, 1985). The variations in the concentrations and species composition of pollen and spores can be used to trace the sources of suspended sediments in catchments and assess the hydrological contributions from different geological features and types of vegetation cover. In pollen hydrographs recorded for September 1981, the contribution from tree species at peak discharge rates was clearly much greater than that from other vegetation. At lower flows the proportions were more or less equal.

### 2. 4. 5. Chemical composition of streams

A general description of the water chemistry in Forest streams is given here using published and public record data. A more detailed account and comparisons of the chemistry of various streams are included in Chapter 4.

The stream waters of the New Forest are typically base-poor, with low nutrient concentrations. The acidity of the waters of the uppermost reaches of some streams was first reported in the last century (Brierly, 1890). He noted *"the very great corrosive nature of the water in the New Forest upon metals"* and after tests with various metals concluded that the acidity was *"due to the presence of a free acid, crenic acid "* in the water. He also noted the presence of humic acids in a number of stream water samples. Subsequent studies have shown that the chemistry of the stream water in the various catchments differs, as might be expected, with the underlying rocks and soils and the varying land-uses across the Forest (Tables 2.1 & 2.2). Le Rossignol (1977) recorded specific conductance values (electrolytic conductivity) ranging typically from 74 to 447 μScm$^{-1}$ (μS per cm at 25°C) in unpolluted Forest waters in the Lymington River catchment, though values of up to 1170 μS cm$^{-1}$ were recorded where a small discharge from houses and farmland entered a stream. Conductivities measured at 66 sites in various catchments by the author in June 1992 showed that 79% of the values ranged between 101-200 μS cm$^{-1}$. Some 6% were below 101 and 15% above 200. The higher conductivities all occurred in reaches passing through the more improved and developed land on the

**Table 2.2. Ranges of selected determinands in water from streams flowing through different types of marginal land use in the New Forest.**

| Determinand | Wooded land | Open heath/forest lawn | Urban/developed land |
|---|---|---|---|
| Conductivity (µS/cm) | 122 - 185 | 89 - 293 | 258 - 788 |
| Alkalinity (mequiv./l) | 0.10 - 0.38 | 0.14 - 0.86 | 0.48 - 2.09 |
| Phosphate (µg/l) | 0.32 - 6.85 | 0.74 - 138.0 | 23.0 - 144.0 |
| Nitrate (µg/l) | 6.80 | - | 9.97 |
| Silicate (µg/l) | 0.91 - 1.02 | 0.99 - 1.02 | 0.98 - 1.04 |
| Aluminium (µg/l) | 20.8 - 112.0 | 9.8 - 45.0 | 2.2 - 23.4 |

*From Langford 1996*

fringes of the Forest, and values exceeding 400 $\mu$S $cm^{-1}$ were found in at least one of these reaches.

Fig. 2.6 shows the frequency occurrences of the conductivities of water draining different rock types within the New Forest. There was some overlap but water draining the Barton Sands is typically in the lower ranges while that from the Headon Beds is generally richer in ionic content. The variation in both types is because of the range of surface land uses, though much of the improved land is on the areas of the Headon Beds. In general, water draining deciduous woodland had higher conductivity than that from heathland (Le Rossignol, 1977). There was also a significant relationship ($r^2 = 0.98$) between the conductivity (SC, $\mu$S $cm^{-1}$ at 25°C) and concentrations of total dissolved solids (TDS, $mgl^{-1}$), expressed by the equation,

$$SC = 0.94\ (TDS - 5.16)$$

The annual variations in some chemical characteristics of two New Forest streams were shown by Marker & Gunn (1977). They found that pH values varied between 6 and 7, and alkalinities were ca. 0.2 $meq.l^{-1}$. Phosphate concentrations peaked at about 150 $\mu gl^{-1}$, with typical values of about 50 $\mu gl^{-1}$ in dry weather. Temperatures exceeded 21°C in both streams.

Typical pH values for the Forest streams are 5.9 to 7.3, though surveys have shown values as low as pH 4.2 in small streams (Winsland, 1985) and as high as pH 8.1 in the lower reaches of the larger rivers (National Rivers Authority, 1993).

New Forest streams are generally unpolluted by major point sources. Sewage works at Brockenhurst and Lyndhurst discharge into streams but neither consistently cause major adverse effects on river chemistry, though intermittent events occur which can cause fish mortalities. Higher levels of some nutrients have also been recorded downstream of such works. For example, in the Lymington River the concentrations of phosphate downstream of Brockenhurst sewage disposal works

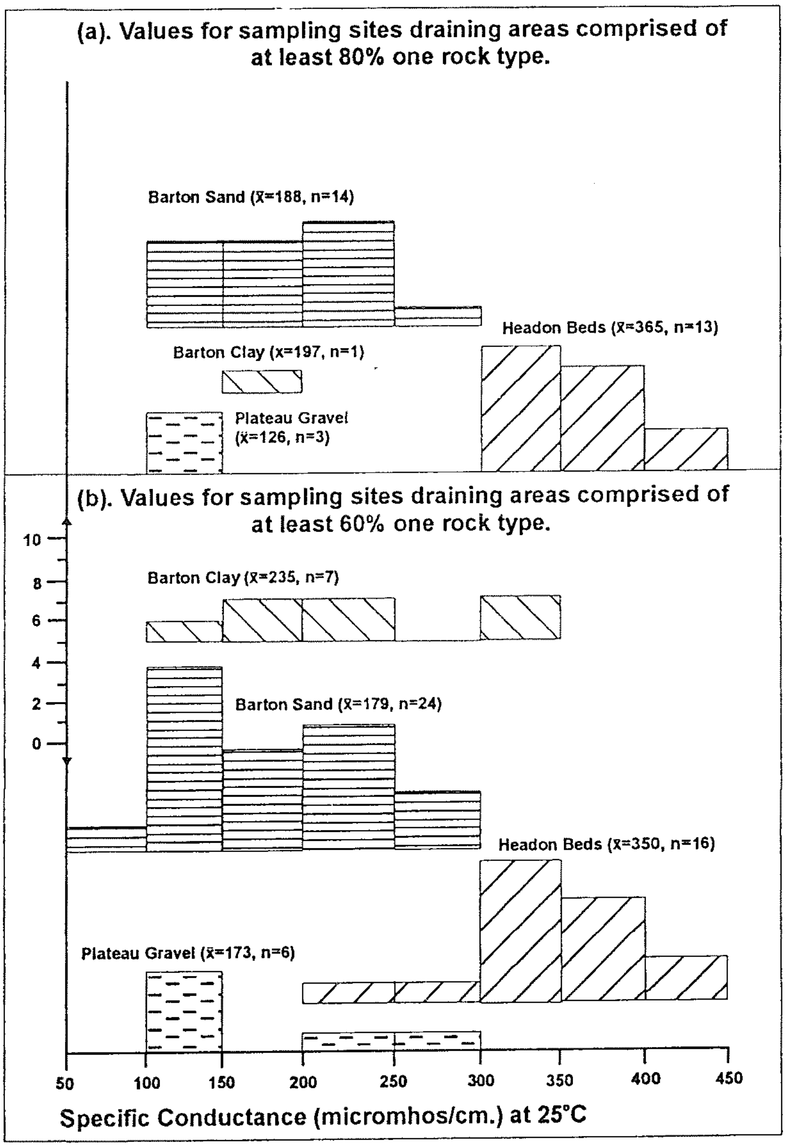


**Fig. 2.6. Frequency distribution of specific conductance values ($\mu$ s cm$^{-1}$ at 25°C) for New Forest streams draining different rock types.**
***(redrawn from Langford, 1996 after Le Rossignol 1977)***

exceeded 1.0mg $l^{-1}$ on four sampling occasions in ten years with a maximum of 2.2 mg $l^{-1}$ (National Rivers Authority, 1993).

## 2. 5. FLORA AND FAUNA

### 2. 5. 1. Micro-organisms and algae in streams

The primary productivity and standing crops of algae in New Forest streams are predictably low when compared with other streams carrying higher concentrations of minerals and nutrients, such as chalk streams. For example the biomass of benthic algae in two New Forest streams was much lower than in the Bere Stream, a hard water Dorset stream (Marker, 1976). Levels of chlorophyll-*a* were much higher during the summer and varied much more in the hard water stream than in the Forest stream. There was also a marked April peak in the hard water stream which was not apparent in the Forest streams (Fig. 2.7). The seasonal variation and abundance of chlorophyll-*a* in the suspended matter showed similar comparative patterns between the hard water and soft water streams, as was found for the benthic algae (Marker & Gunn, 1977). In this study, however, the seasonal variations in the two Forest streams differed from each other. In Dockens Water, draining the western edge of the Forest, there was no spring peak, while in the Ober Water, draining toward the south, there was a peak in April, though much lower than the April peaks in the hard water streams.

In a more recent study, chlorophyll-*a* concentrations of epilithic algae varied from 2.2 to 44 $mgm^{-2}$ of stream bed in the Ober Water representing an annual mean biomass of 1.0 $gm^{-2}$ of algae (Shamsudin & Sleigh, 1994). In contrast, the values for the River Itchen, a nearby chalk stream were 115 to 415 $mgm^{-2}$, representing a mean annual biomass of about $8gm^{-2}$. These authors estimated that the annual production of epilithic algae in the River Itchen was about eight times that of the Ober Water, being respectively 600 g and 75 g organic dry weight $m^{-2}$.

Diatoms dominated the epilithic flora in both streams, comprising 70 to 95% of the total number of algal cells. Numbers peaked in April and May. Chlorophyte cells

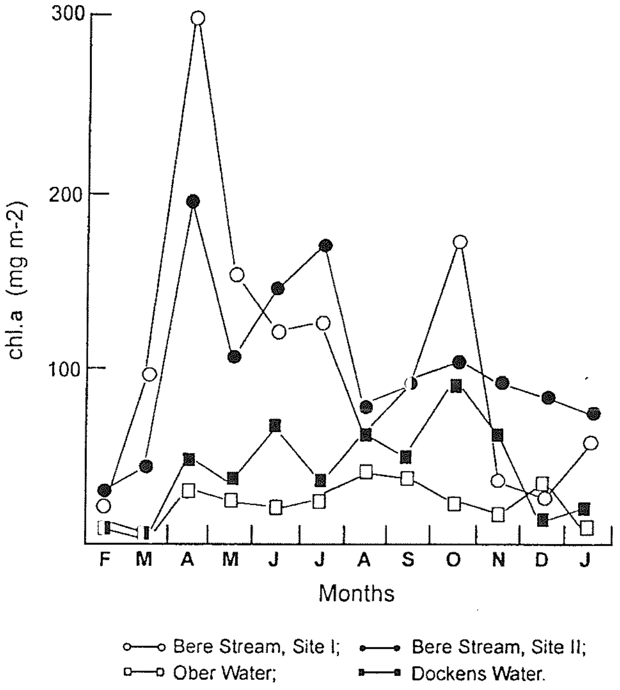


**Fig. 2.7. Seasonal variation in chlorophyll-*a* content (mg m-$^2$, corrected for phaeopigments) of the epilithic flora of a Dorset hardwater stream . (Bere Stream at two sites, ○-○ and ●-●) and two softwater New Forest streams (□-□, Ober Water; ■-■, Dockens Water).**

***(Redrawn from Langford, 1996 after Marker, 1976).***

peaked in summer and the Cyanophytes peaked in autumn. Densities ranged from 8 and 320 cells mm $^{-2}$ in the Ober Water and 500 and 7,000 cells mm$^{-2}$ in the Itchen.

McCollin (1993) recorded 30 species of epilithic diatoms (Bacillariophyceae) and epiphytic diatoms at 15 sites in New Forest streams. There was no clear relationship between the occurrences of particular species and marginal land uses, except that *Fragilaria* cf. *pinnata* was more common in streams flowing through wooded areas than in streams flowing through open areas. *F. pinnata* was the most common and abundant of all the species recorded at the 15 sites, particularly at those sites with low phosphate concentrations and low light levels. It was uncommon or absent at sites enriched by agricultural drainage or 'urban' development. At the highest phosphate levels, *Cocconeis placentula* and *Achnanthes minutissima* were the dominant species.

Densities of epipelic, epilithic and planktonic algae were not correlated with temperature, phosphate, nitrate or silicate concentrations over a yearly cycle in the Highland Water (Moore, 1977a). Water velocity appeared to be a major influence on the growth rate of the predominant epilithic species, *Achnanthes saxonica,* though shade affected both standing crop and density. Where the current velocity averaged 40 to 50 cm$^{-1}$, *A. saxonica* accounted for 95% of the standing crop throughout the year. At 30 to 40 cm$^{-1}$ it dominated only in the winter and in summer was replaced by aggregations of other species. Decaying timber was the dominant substratum for epiphytic species in the Highland Water. Here again *A. saxonica* was dominant, though the community found on wood, also included species of algae that are normally epilithic or epipsammic. Clearly there was not a separate, distinct community on the woody debris. Only *Surirella ovata* var *minuta* appeared to be much more common on dead wood than in all other niches. In common with most small streams, the 'planktonic' cells were derived from the sediment communities and densities of the two communities generally varied in parallel.

There are no published studies of filamentous algae in New Forest streams, though Marker (1976) noted the growths of the red alga *Batrachospermum* sp. from August to November in Dockens Water, and this has since been found by the author in most Forest streams.

Studies of ciliate protozoa in the Ober Water, showed that the densities of organisms were also significantly lower than in a nearby chalk stream (Baldock & Sleigh, 1988). As might be predicted, the Ober Water showed lower microbial activity, estimated by alkaline phosphatase assay. Peak abundance of ciliates in both the chalk stream and the Ober Water coincided with diatom blooms. Flagellates were more abundant than ciliates in the Ober Water, in contrast to the chalk stream, and photosynthetic flagellates, dominated by the genus *Synura,* reached densities of 148,000 cells $cm^{-3}$ in this soft-water stream. In a later study (Harmsworth, *et al.*, 1992) the densities of peritrich ciliates on stones in a small New Forest stream were, perhaps less predictably, higher for most of the year than in a nearby chalk stream and the population size was found to be negatively correlated with discharge. *Carchesium* spp, *Vorticella* spp. and *Platycola* spp. were the dominant genera in both types of stream.

### 2. 5. 2. Macrophytes

The macrophyte communities of *"sandy New Forest streams"* have been classified in a separate category from those of streams elsewhere because they have a unique assemblage of species (Haslam & Wolseley, 1981; Holmes, 1983). Common species of macrophytes in New Forest Streams, identified in surveys since 1992, are listed in Table 2.3, where the species considered to be typical are indicated. *Elodea canadensis* (possibly *E.nuttalli* at some sites) occurs mainly in streams which have been deepened or channelized as described earlier (Brookes, 1983), and in some of these reaches, most notably in the upper Ober Water, *Elodea* lines the margins for distances of over 100 metres and can practically block the channel in some places. *Ranunculus aquatilis* occurs mainly in deeper and slower reaches and *Callitriche stagnalis* mainly occurs in reaches with relatively stable substrata, generally consisting of small, accreted gravel or stones in a firm sand matrix.

*Potamogeton polygonifolius* is found in streams and in standing water but the species is most common in the upper reaches of streams with lower conductivities

**Table 2. 3. Common macrophytes in New Forest streams.**

| **Species** | **Common name** |
|---|---|
| ***Emergent/marginal plants*** | |
| *Agrostis stolonifera* L. | Fiorin, Creeping bent |
| *Alisma plantago-aquatica* L. | Water plantain |
| *Apium nodiflorum* (L.) Lag. | Fools watercress |
| *Caltha palustris* L. | Marsh marigold, Kingcup |
| *Glyceria fluitans* (L.) R. Br. | Flote-grass, Floating sweet grass |
| *Hypericum elodes* L. | Marsh St. John's wort |
| *Juncus acutifloris* Ehr. Ex Hoffm | Sharp-flowered rush |
| *Juncus bulbosus* L. | Bulbous rush |
| *Ludwigia palustris* (L.) Elliott | Hampshire purselane |
| *Mentha aquatica* L. | Water mint |
| *Menyanthes trifoliata* L. | Bogbean |
| *Myosotis scorpioides* L.* | Water forget-me-not |
| *Oenanthe crocata* L. | Hemlock water dropwort |
| *Ranunculus flammula* L. | Lesser spearwort |
| *Rorippa nasturtium-aquaticum* (L.) Hayek | Summer watercress |
| *Scrophularis auriculata* L. | Water betony |
| *Sparganium erectum* L.* | Bur reed |
| *Veronica beccabunga* L. | Brooklime |
| ***Submerged/instream plants*** | |
| *Callitriche hamulata* Kutz ex. Koch | Water starwort |
| *Cullitriche platycarpa* Koch | Starwort |
| *Callitriche stagnalis* Scop. | Starwort |
| *Elodea canadensis* Michx.† | Canadian pondweed |
| *Potamogeton polygonifolius* Pourret | Bog pondweed |
| *Ranunculus peltatus* Schrank* | Water crowfoot |
| *Ranunculus omiophyllus* Lam.* | Water crowfoot |

†Some fronds appear to be very like *E. nuttalli* (P. Angold, personal communication).
*Species regarded as typical of New Forest sandy streams (Haslam & Wolseley 1981).

*From Langford 1996*

and pH values. It occurs in fast-flowing water where the leaves tend to be partly or wholly submerged as they stream in the current, and also in slow reaches or backwaters, where the leaves float on the surface. In the many reaches shaded by woodland or with unstable gravel substrata, stands of macrophytes are rare or non-existent, though *P. polygonifolius* is unusual in that it occurs in very small streams with dappled to dense shade. Most of the common macrophytes in the streams maintain stands throughout the year.

### 2. 5. 3. Macro-invertebrates

The earliest detailed records of macro-invertebrates with stream-living stages in their life histories are from the observations of natural historians beginning around the end of the 19th century (e.g. Hampshire Field Club, 1890; Lucas, 1932; Haines, 1933 et seq. Jones, 1930). Lucas (1932) noticed the relative scarcity of Ephemeroptera, and commented that "" *Judging by the water supply in the county, the Ephemeroptera (Mayflies) should be common, but in the New Forest, they are not very noticeable."* By 1940, over 20 species of Trichoptera and 10 species of Plecoptera had been recorded as adults, but only 4 species of Ephemeroptera were noted. Some species of Trichoptera were also very abundant at times and Haines (1940) noted "*the usual autumnal abundance of Stenophylax stellatus and Halesus radiatus*" around Dockens Water.

Also among the early records were Coleoptera and some Diptera, notably the Tabanidae of which the most frequently recorded stream species was *Pedicia rivosa*. T.T. Macan and A.H. Moon both collected species of insects from New Forest streams in the late 1930's and their records are noted in the Hampshire Field Club Proceedings (Haines, 1940). Haines also quotes data on the density and diversity of the insect fauna of the streams, noting that 68 insects in 10 species were found, on average, "*per square foot*" of the bed of the Latchmoor Brook. The comparable numbers for the nearby River Avon, a chalk stream, were 384 insects per square foot in 15 species.

Research on the fauna of the streams, as distinct from merely recording the presence of species, appears to have begun in the late 1940's, based in the

University of Southampton. In one of the earliest studies, Hall (1951) compared the species of Chironomidae in a chalk stream and three more acidic streams in the New Forest. Of the 23 species recorded, 18 were in the chalk stream, 13 in the acid streams and 7 species were common to both. Later, Hall and his colleagues transferred attention to other invertebrate groups including the rarer species of the standing waters (Hall, 1953, 1954, 1959a, b, c, 1961, 1977; Khalaf, 1973; Khalaf & Hall, 1975).

Studies on the food of fish in two streams have included records of invertebrates from stomachs (Mann & Orr, 1969; Mann, 1971). More recently, lists of species have been compiled for several sites as part of a national study of river habitats by the NERC'S Institute of Freshwater Ecology, in order to develop the RIVPACS programme (e.g. Furse *et al.*, 1986; Wright *et al.*, 1994). The provisional list from New Forest streams includes some 262 species. Table 2.4 shows the number of species in each major group recorded so far from some 70 sites in relation to the number of species in that major group known from mainland Britain. The relative numbers of representatives in these groups (expressed as percentages of total numbers in each group) reflects both the actual occurrence of species for well known groups such as Plecoptera and Ephemeroptera, and the relative lack of expertise and collections for the lesser known groups such as Hydracarina.

Life-history studies of invertebrates in New Forest streams are rare, though in a series of observations on animals in the superficial gravel beds of one stream, the Ober Water, the biology and life histories of several species were described. Gledhill (1969) found that the phreaticolous water-mite *Neoacarus hibernicus* Halbert, breeds in early spring in the interstitial gravel water. The maximum densities of the species were just over 1 individual per litre of water sampled in July and August. Very few were found in the winter months, probably because of floods washed out specimens from the gravel.

In the same gravel bed, the subterranean amphipod *Niphargus aquiliex aquilex* had two breeding seasons in a year, the earlier one occurred in May, originating from overwintering adults, and the later one in October, originating from a fast-growing and maturing summer generation (Gledhill & Ladle, 1969). The breeding males of

**Table 2.4. The numbers of species in major macroinvertebrate groups recorded from New Forest streams in recent surveys, compared with numbers found in the UK. Data from the author and from lists provided by the IFE, from Maitland (1977) and NCC database.**

| Major groups | No. of species UK | No. of species New Forest | Percentage in New Forest |
|---|---|---|---|
| PLATYHELMINTHES | | | |
| Turbellaria/Tricladida | 11 | 3 | 27 |
| MOLLUSCA | | | |
| Gastropoda | 52 | 14 | 26 |
| Bivalvia | 27 | 7 | 26 |
| ANNELIDA | | | |
| Oligochaeta | 118 | 24 | 28 |
| Hirudinea | 14 | 5 | 36 |
| ARTHROPODA | | | |
| Hydracarina | 322 | 11 | 3 |
| Malacostraca | 33 | 5 | 15 |
| Ephemeroptera | 47 | 19 | 40 |
| Plecoptera | 34 | 14 | 41 |
| Odonata | 45 | 7 | 16 |
| Hemiptera | 62 | 6 | 10 |
| Coleoptera | 300 | 29 | 10 |
| Megaloptera | 3 | 2 | 67 |
| Neuroptera | 4 | 1 | 25 |
| Trichoptera | 193 | 42 | 22 |
| Diptera | 1138 | 73 | 7 |
| VERTEBRATA | | | |
| Agnatha | 3 | 1 | 33 |
| Pisces | 55 | 20 | 38 |

*From Langford, 1996*

this second generation were significantly larger than the overwintering males. The densities of *N. aquilex* ranged from 2 per litre of water sampled in March to about 8 per litre in October. In later surveys of the streams, *N. aquilex* has been recorded from kick-samples taken in the free-flowing reaches both by the author and by others, though occurrences are rare and numbers usually very small.

In a study of the feeding biology of benthic herbivores, Moore (1977b) collected individuals of mayfly larvae (*Ephemera danica* and *Ecdyonurus* sp.) and the amphipod *Gammarus pulex* from the Highland Water. The amounts of food varied very little throughout the day for all three species and there was no obvious diel pattern of feeding. Although Moore did not list the species of algae taken by these animals, from his studies on the succession and abundance of algae (Moore, 1977a) the assumption can be made that each of the invertebrates ingested the most common and abundant algae found in their specific niche. Thus *Ephemera danica* a collector-filterer, living in sand and coarse silt might contain the commonest epipelic species such as *Achnanthes minutissima, A. saxonica* and *Cymbella naviculiformis,* whilst *Ecdyonurus* sp., living on the stones, grazed on *Achnanthes saxonica, A. minutissima* and *Gomphonema acuminatum.*

Early observations of the invertebrates inhabiting the coarse woody debris in streams (Langford, *unpublished data*) indicate that the community can differ markedly from that of nearby riffles. In preliminary winter collections one wood-pile community was dominated by mayflies (Leptophlebiidae), stoneflies *(Nemoura* spp.) and the cased larvae of the caddis fly, *Halesus radiatus.* In comparison the fauna of the riffle some 30 metres upstream was dominated by mayflies (Heptageniidae) and blackfly larvae (*Simulium* spp). In the nearby stream margin the most abundant macro-invertebrate was the cased caddis, *Anabolia nervosa.* Seven taxa were common to all three habitats from a total of thirty-four recorded taxa. Some stoneflies (Leuctridae) were more or less equally abundant in all three niches. Other preliminary studies also showed that the removal of a debris dam created a pulse of invertebrate drift downstream. An increase in the abundance of drifting invertebrates was measured in the first few minutes immediately after removal of the debris, using a linear series of six small nets, set at ten metre intervals downstream of the dam site. Fig. 2.8 shows the numbers of invertebrates caught in the nets after removal of the dam, in relation to normal background drift. The effects of the sequential removal of drifting organisms by the nets has not been quantified (Evans, 1995), but the effects of the debris dams on drift and community structure are the subject of further studies.

The Odonata are perhaps the most consistently observed aquatic insects in the New Forest (Welstead & Welstead, 1984) and the status of the scarcer species has been reviewed recently (Winsland, 1994). One of the most obvious common species associated with the streams is the damsel-fly *Calopteryx virgo,* which is more or less ubiquitous. For example, during stream surveys during June 1992, adults were observed at 45 of 50 sites sampled by the author. The larvae occur in the stream margins throughout the Forest. The species most studied is the relatively rare damselfly, *Coenagrion mercuriale* (Corbet, 1957). Most of the recent work on this damselfly has been a detailed study of the adult populations in one small stream on Beaulieu heath (Winsland, 1985; Jenkins, 1986a, b, 1987, 1991, 1994, 1995). *C. mercuriale* appears to favour streams with a pH range between 5.5 and 7.0, and the adults are most abundant where the vegetation bordering the streams is dense.

### 2. 5. 4. Fish

Of the 55 British species of fish found in fresh waters (Maitland, 1977; Maitland & Campbell, 1992), 20 regularly occur in the streams of the New Forest. Nomenclature follows that given by Maitland & Campbell (1992). Both common names and shortened binomials are used in the following text and subsequent chapters. The most common species recorded in an electrofishing survey of 19 sites in five streams (National Rivers Authority, 1993) were brown trout (*Salmo trutta,* Linnaeus 1758), minnow (*Phoxinus phoxinus* (Linnaeus 1758)), bullhead (miller's thumb or sculpin) (*Cottus gobio* Linnaeus 1758), stone loach (*Noemacheilus barbatulus* Linnaeus 1758), brook lamprey (*Lampetra planeri* (Bloch 1754)) and eel (*Anguilla anguilla* (Linnaeus 1758)), all six being recorded at every site. Roach (*Rutilus rutilus* (Linnaeus 1758)), rudd (*Scardinius erythrophthalmus* (Linnaeus 1758)), dace (*Leuciscus leuciscus* (Linnaeus 1758)), chub (*Leuciscus cephalus* (Linnaeus 1758)), pike (*Esox lucius* Linnaeus 1758) and

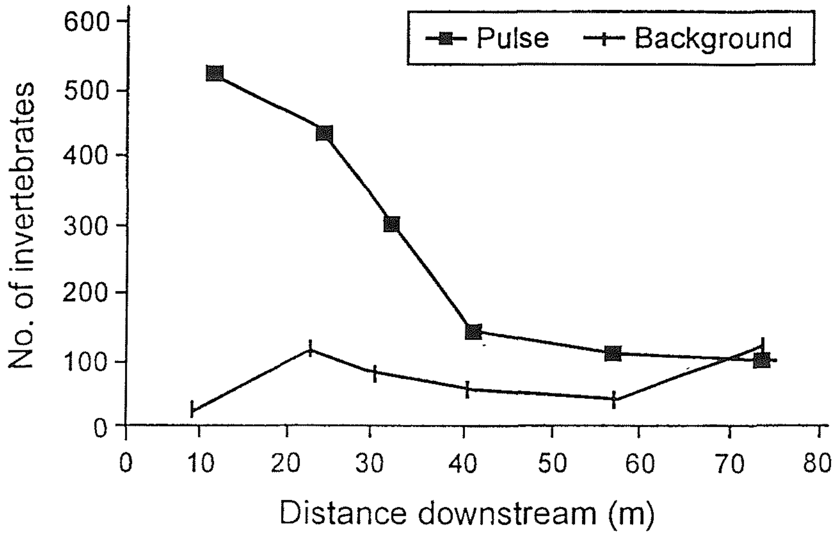


**Fig. 2.8. Numbers of invertebrates caught by drift samplers in 10 minutes following the removal of a debris dam from a New Forest stream. (Background numbers were from the sites before disturbance). (0m represents the position of the dam site).**
***(Redrawn from Langford, 1996 after Evans, 1995).***

perch (*Perca fluviatilis* Linnaeus 1758), were recorded mostly from the lower reaches of the rivers. Flounder (*Platichthys flesus* (Linnaeus 1758)) were recorded from the lower reaches of the Beaulieu River. There is also likely to be some exchange of fish between the neighbouring chalk streams and their more acidic feeders draining from the Forest, but there are no data on the extent of their migrations. A detailed analysis of fish distribution and abundance is given in Chapters 5, 6 and 7.

Although surveys suggest that the bullhead (*C. gobio*) is ubiquitous in New Forest streams, its distribution appears to be discontinuous both between and within streams. For example, Mann & Orr (1969) and Mann (1971) recorded no bullheads at a sampling site in the upper reaches of Dockens Water (Fig. 2.1), one of the larger streams among the western catchments of the Forest. Subsequent work reported later in this account (Chapter 5, 6) also found no bullheads at the same site almost 30 years later, though the species is present further downstream together with stone loach (*N. barbatulus*). This latter species, however, appears to replace *C. gobio* completely in the upper reaches.

The most important items in the diet of both trout and minnows in Dockens Water were the freshwater shrimp *Gammarus pulex* and larvae of chironomid midges (Mann & Orr, 1969). Stoneflies (Plecoptera) were also prominent in the diet of trout, with most being eaten in the winter months. Both trout and minnows apparently exploited the same food resources, though minnows appeared to take less stoneflies and more chironomid midges than the trout. The "flashy" nature of stream-flows resulted in terrestrial invertebrates such as lumbricid worms and spring-tails (Collembola) being washed in and eaten by both trout and minnows. When compared with those of a chalk stream, trout in Dockens Water fed more on *G. pulex*, terrestrial organisms and stoneflies, in the last because stoneflies did not occur in the chalk stream studied. In contrast, mayfly nymphs and molluscs were relatively more important in the diet of trout from the chalk stream.

The growth of trout in Dockens Water was slower throughout life than in nearby chalk streams (Mann, 1971) and by the end of their third year trout averaged about 20 cm in length, some 20% less than in the chalk streams. Growth continued throughout the year at all sites sampled, with maximum rates in May, June and July. Spawning occurred in November in Dockens Water some two months earlier than in two Dorset chalk streams although 1st of March was reckoned to be the "birth" date of alevins in both hard-water and soft-water streams.

Minnows spawned during April and May in both the hard-water and soft-water streams (Mann, 1971). First-year growth was approximately the same for both types of stream, but in the second year minnows grew faster in the hard-water, spring-fed stream. The oldest fish were aged at just over 3yrs. Stone loaches grew faster in the hard-water streams but, as with the minnows, the maximum age in both streams was just over three years.

## 2. 6. RARE AND ENDANGERED SPECIES

Despite the fact that the New Forest includes several terrestrial and aquatic habitats now rare in Britain and to some extent in lowland Europe, very few species living in the streams are listed as endangered or vulnerable in the various Red Data Books (e.g. Perring & Farrell, 1983; Wells *et al.*, 1983; Shirt, 1987) or other publications (Falk, 1991; Maitland & Lyle, 1991; Wallace, 1991; Kirby, 1993). Most of the rare or threatened species of Diptera, including those believed to be extinct or on the point of extinction from the Forest such as *Tipula seibekei* and *Tipula yerburyi,* have larvae and pupae which live in damp moss, wet heath or on decaying wood rather than in the streams (Falk, 1991). Of the insects listed as rare, the southern damselfly *Coenagrion mercuriale,* is perhaps the most notable and this has been the subject of several studies which are referred to above and in the bibliography. One hemipteran, the lesser water measurer *Hydrometra gracilenta,* was recorded from Forest streams and pools before the 1950's but has not been recorded since the end of that decade (Kirby, 1993). Of the fish, the brook lamprey (*L. planeri)* and the bullhead (*C. gobio*) have been afforded some protection as Annex II species under the Bern Convention (see Boon *et al.*, 1992), but are not generally regarded as endangered in England.

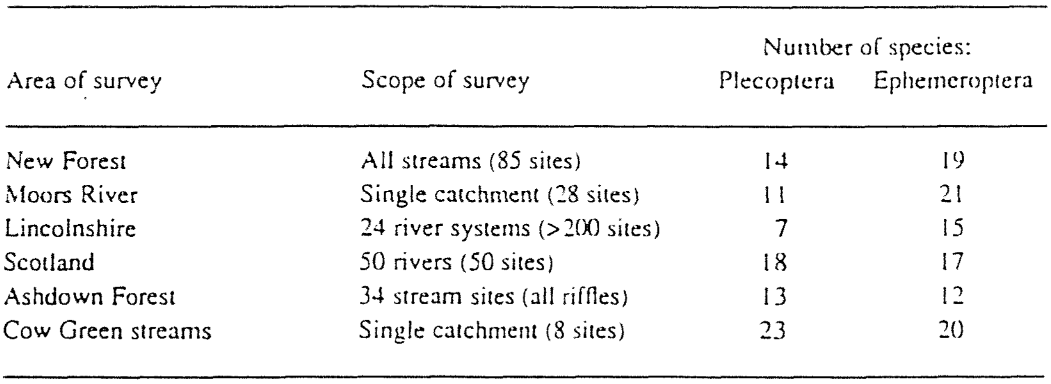
The aquatic species which do occur most prominently in the Red Data Books are mostly those from standing waters or temporary ponds in the Forest and include such rarities as the tadpole shrimp *Triops cancriformis,* the fairy shrimp *Chirocephalus diaphanus,* and the medicinal leech *Hirudo medicinalis* (Elliott & Tullett, 1984; Wilkin, 1987; Bratton, 1991; Boon *et al.*, 1992).

## 2. 7. COMPARISON OF INVERTEBRATES IN NEW FOREST STREAMS WITH STREAMS IN OTHER REGIONS OF BRITAIN

Because there is an extensive knowledge of these groups, the faunal lists of Ephemeroptera and Plecoptera occurring in New Forest streams are unlikely to be extended significantly and it is interesting to compare the numbers of species recorded from Forest streams briefly with those of other surveys (Table 2.5). The scope and intensity of the surveys varies very much and no attempt has been made to quantify the comparisons or to compare the relative abundance of species. Although such a comparison is not conclusive, the overall predeliction of mayflies and stoneflies for areas of high relief and stony streams is shown by the species-richness of streams in Scotland and around Cow Green Reservoir in the north Pennines, of which the latter ranks highest. The New Forest drainage system is in the middle range for species-richness of Plecoptera but matches the paucity of the Lincolnshire streams for species-richness of Ephemeroptera (Langford & Bray 1969). This is probably a result of the low relief and relative chemical uniformity of the streams in both regions, although those in Lincolnshire are much more alkaline than those of the New Forest. Perhaps the most comparable species-richness is in the streams of the Ashdown Forest, also situated in southern England, though these streams are generally more acidic than the New Forest streams and fish are absent from some of them (Townsend *et al.*, 1983).

The Moors and Uddens River system (Wright *et al.*, 1988), which lies about 20 km. to the west of the New Forest, is relatively very rich in species of both groups (Table 2.5) considering the small, single catchment area. The species-richness here apparently reflects the high chemical and physical diversity within this particular river system. The species-richness of the Ephemeroptera in the Scottish streams

**Table 2.5. A comparison of species-richness for Plecoptera and Ephemeroptera from stream systems in different regions of Britain. Data were obtained from the following sources. New Forest: Langford (1996), plus IFE RIVPACS data-set; Moors River: Wright *et al.* (1988); Lincolnshire: Langford & Bray (1969); Scotland: Morgan & Egglishaw (1965); Ashdown Forest: Townsend *et al.* (1983); Cow Green streams: Armitage *et al.* (1974).**



| Area of survey | Scope of survey | Number of species: Plecoptera | Ephemeroptera |
|---|---|---|---|
| New Forest | All streams (85 sites) | 14 | 19 |
| Moors River | Single catchment (28 sites) | 11 | 21 |
| Lincolnshire | 24 river systems (>200 sites) | 7 | 15 |
| Scotland | 50 rivers (50 sites) | 18 | 17 |
| Ashdown Forest | 34 stream sites (all riffles) | 13 | 12 |
| Cow Green streams | Single catchment (8 sites) | 23 | 20 |

*From Langford 1996*

studied by Morgan & Egglishaw (1965) is relatively low, despite the wide variety of geology, stream types and the large geographical area covered by the survey. However, rather more species of Plecoptera were recorded here than in the other regions, except at Cow Green.

From various studies it is clear that Ephemeroptera such as *Ephemera danica, Ephemerella ignita* and *Habrophlebia fusca* and Plecoptera such as *Isoperla grammatica* and *Brachyptera risi,* are tolerant to a wide range of natural water chemistry and were common to most of the regions listed in Table 2.5. The most notable "lowland stream" absentee from the New Forest and the Ashdown Forest is the stonefly, *Taeniopteryx nebulosa,* which is clearly intolerant of waters of low electrolytic conductivities and pH, though it is recorded from the general southern region, presumably from more calcareous streams nearby outside the New Forest. More detailed comparisons will be made in future publications.

## 2. 8. FUTURE MANAGEMENT AND RESEARCH

The regulation and administration of the New Forest rivers, at the time of writing, is shared by two regions of the Environment Agency, Southern and Wessex. Within the Crown Lands, the rivers are managed by Forest Enterprise (part of the Forestry Commission). The Commission is also regarded as a Land Drainage Authority and is responsible for the proper drainage of the forest. The fact that many of the rivers fall within the New Forest SSSI ensures that any work on the channels requires consultation with English Nature, though English Nature does not have the power to veto or prosecute the Forestry Commission should consultation not take place. The inclusion of the Lymington River as a new SSSI will increase protection. Further protection would come from the proposal to declare the New Forest as a Special Area for Conservation (Sanderson, 1995).

Despite the extra protection to be afforded by the EC Habitats Directive and other legislation (e.g. Jones, 1991) the streams of the New Forest, like the Forest itself, are likely to come under considerably increased pressure from recreational pursuits and peripheral population growth before the end of the 20th century. A recent review noted that *"The New Forest lies between two of the fastest growing conurbations in the country"* (New Forest Review Group, 1988) and the use of the Forest by the larger peripheral populations, in addition to the tourist and weekend recreational influxes, has considerable management and conservation implications.

In many places the margins of stream channels are already eroded badly by children wielding buckets and spades and by walkers and grazing animals. In several areas there is no vegetation remaining at the edge of the streams. At the same time, the use of mountain bicycles is causing erosion within and alongside streams and there is evidence from tracks and direct observation that, in some areas, the stream channels are being used as "cycle-ways".

Despite the increasing conservation activities associated with forestry, (Forestry Commission, 1970, 1988) the felling of trees, the need for drainage and the presence of conifer plantations can have short-term effects on water quality and on the flora and fauna. To date no specific research has been done on the effects of these activities on the streams within in the New Forest.

There are also a number of management and ecological problems related to the streams which require further study, and research is in hand on some of these. The most obvious, as has been noted, are the effects of timber dams and their removal on the macrophyte and invertebrate communities, together with their effects on the migrations and spawning of fish, particularly Salmonidae. In addition, the mobility of the substrates in New Forest streams must have marked effects on the resident invertebrates and spawning fish, and the extent of such effects and the consequences of any proposed or even past drainage operations require quantification.

The view of groups representing grazing, forestry and angling interests is that debris dams should be removed for the most part to allow better drainage and access for migratory fish. The conflicting view from conservationists is that the dams and timber aggregations cause natural changes in stream processes and form very specialized natural habitats in forest streams (Gregory & Davis, 1992; Gregory, Gurnell & Petts, 1994) and, with a few exceptions, they should be left undisturbed.

There are, however, no scientific data on the direct or indirect effects of timber debris or its management on the animal or plant communities in New Forest streams and this requires urgent attention.

Very few of the proposals for the preservation and protection of the New Forest refer specifically to the streams, though the importance of the drainage system is well recognised among conservation groups. Data from the studies reported here are aimed at helping and encouraging policies and strategies aimed at the specialised protection of the stream habitats within the New Forest. The most recent initiative by The Environment Agency, English Nature and the Forestry Commission is aimed at the restoration of channel sinuosity and physical diversity in stream channels most affected by land-drainage engineering and channelization.

# CHAPTER 3

# METHODS, DATA ANALYSIS AND SITE DESCRIPTIONS

## 3. 1. INTRODUCTION

This chapter reviews literature on the structural diversity of stream habitats and the quantification of wood in streams before focussing on the methods used in this research. The methods include site selection, habitat definition, physical measurements, electrofishing, fish measurements and data analysis. The aim of the chapter is to place the methodolology used in the New Forest streams in the context of the variety of methods described in previous publications so that critical comparisons can be made. The primary aim of the methodology was to define within-reach habitat units on the basis of physical variables including CWD accumulations, and to relate fish distribution and abundance to these habitat units or to measurable physical variables. From the measurements the aim was to define as far as possible the role of CWD in habitat structuring and the distribution abundance and community structure of the fish fauna. The same methodology was used on the stream scale to compare the physical variables and fish populations between streams. All measurements were based on the scale of the channel-unit or a sub-division of this unit (Bisson *et al.*, 1987; Hawkins *et al.*, 1993, Maddock, 1999). There are many examples of studies of the relationships of fish to physical features at the microhabitat scale and a review of these is given by Heggenes (1988a). Many of these studies have used direct observation usually by snorkelling or from the bank. Unless there are data in the publications directly relevant to this work, these studies are not generally reviewed further in this thesis.

It can be assumed that to date there is no single universally applicable model which can predict the abundance of fish from physical variables in stream channels. Models for species of salmonids, of which there have been many proposed (e.g. Binns & Eisermann, 1979; Bovee, 1982; Milner *et al.*, 1985; Heggenes 1988a; Korman *et al.*, 1994) are not generally consistent in their use of variables and none is widely applicable with any precision to a range of rivers and streams. As noted in Chapter 1 Elliott (1994) suggested that the reasons for the lack of generic models include this lack of consistency in the use of variables, little agreement in the relative importance of variables and a tendency to ignore interactions between the various physical and biological variables. He also stressed that the assumption that there is a generic linear or log-linear relationship between fish abundance and the physical nature of streams may also be false, though many of the publications quoted here have used such relationships (see Milner *et al.*, 1985). Further, the problems with many models is that they have used data from a variety of streams sometimes in different catchments so that the numbers of potential variables is unmanageably large and they ignore other factors, for example larger scale influences (see Chapter 1) (e.g. Binns & Eiserman, 1979). The species composition of the streams may also differ, which introduces new interactions which are not accounted for by the models. Another basic assumption is that the distribution and abundance of species may be deterministic rather than stochastic (e.g. Moyle & Vondracek, 1985; Martin-Smith, 1998) and that external physical variables are a stronger influence than simple probability or a result of density pressures (e.g. Angermeier & Smogor, 1995). For highly mobile organisms such as fish, distribution can be deterministic or stochastic or a combination of both depending upon the species, habitat and life-history stage. Studies on other organisms have shown that distribution is typically patchy (e.g. Hildrew & Giller, 1994; Giller *et al.*, 1994) and this can also apply to fish, at least in some life-history stages (e.g. Copp, 1992; Copp & Bennetts, 1996; Prenda *et al.*, 1997; Watkins *et al.*, 1997). Thus within-reach physical variations in a stream channel, either in currents, structures or substrates can influence fish distribution and abundance on that scale.

In one of the earlier comprehensive studies Binns & Eiserman (1979) noted that " *in reality any investigation of the limiting factors acting on a trout stream is limited more by...... the ability to measure than by theoretical considerations as to the true limiting factors*". It may be therefore that the search for any universally applicable model is fruitless or that the relevant factors or combination of factors have not yet been identified. It is however clear that the structure of fish communities in continuous streams can be related to some combination of physical variables most likely in the form of a gradient or continuum (e.g. Paller, 1994; Williams *et al.*, 1996; Prenda *et al.*, 1997).

## 3. 2. MEASUREMENTS OF HABITAT DIVERSITY

### 3. 2. 1. Definitions of habitat diversity

The perception of physically diverse aquatic habitats is universally accepted and instinctively understood. The difference to the human observer between a sinuous river channel with overhanging vegetation, undercut banks, riffles and pools, abundant instream and marginal vegetation and large obstructions creating shelter, backwaters and refugia and a channelized, straightened, embanked river with little riparian, marginal or instream vegetation is obvious. Whether this anthropomorphic perception equates with the optimal requirements of the various fish species is unknown (see Chapter 1, Section 2.3). Indeed despite the comprehensive studies, the precise nature of the relationship between morphological diversity of river channels and biological diversity is still largely unresolved and this applies particularly to fish (e.g. Gorman & Karr, 1978; Bayley & Li, 1992; Cowx & Welcomme, 1998; Martin-Smith, 1998; Lamouroux *et al.*, 1998). It is notable that the heavily dredged, straightened trapezoidal channels of East Anglian rivers and many navigable canals contain large and diverse populations of fish often with good growth rates as do the semi-natural channels of rivers such as the Severn or Wye, (e.g. Langford, 1966; Townsend & Peirson, 1988; Murphy *et al.*, 1995).

The variety of measurements used to describe habitat for brown trout (*S. trutta*) were summarised by Heggenes (1988a). He showed that for seventeen studies reviewed nine variables were consistently measured of which four were the most consistent and probably most important. These were water depth, water velocity (or stream gradient), substrate and cover. Over the 17 studies, however, the number of variables ranged from 2-7 and there was no consistency apart from the four specified. Binns & Eiserman (1979) in one of the earliest studies of physical habitat and fish measured or estimated 22 variables, (13 physical, 5 chemical and 4 biological) in a total of 36 Wyoming streams to develop a Habitat Quality Index

(HQI) for salmonids. The alternative Habitat Suitability Index (HSI) proposed by Raleigh *et al.* (1986) included measurements of 18 variables, all of which were predicted to be important factors determining the distribution and abundance of salmonids. Milner *et al.* (1985) listed 6 intra-catchment parameters, 15 localised stream channel parameters, 6 hydrological features, and 8 chemical parameters used in various habitat evaluation schemes (see Table 3.1). These authors also list 21 categories for classifying stream transects, reaches or habitat scale channel units (Table 3.2). Using a questionnaire and point scores Milner *et al.* (1985) defined a habitat scoring system (HABSCORE) which can be used at least within specific regions or stream catchments for classifying trout habitats. Like other systems of habitat assessment, the methodology uses both quantitative and qualitative data and like other systems its uses are limited to specific types of streams in a specific region.

Heggenes (1988a) noted two main difficulties in ranking the importance of habitat variables for salmonids, namely that most studies have focussed on measuring features of habitats actually occupied by trout without quantifying total available habitat and that different variables may be limiting in different rivers. Indeed, Gorman & Karr (1978) suggested that physical variables should only be measured at certain times of year when fish were likely to be present. Also, within any reach where there are riffle-pool sequences different size groups of the same species may choose different habitats (e.g. Egglishaw & Shackley, 1982; Kennedy & Strange, 1982; Martin-Smith, 1998). Such size-segregation indicates that the different size-groups could be regarded as different species as far as habitat preferences are concerned (Schlosser, 1987). Copp (1992) and Garner (1996) used the term "ecospecies" to describe young cyprinids with overlapping microhabitats which were different from those of the adult fish. This term could also apply to species such as *S. trutta* where the 0+ year class and older fish are clearly separable on habitat preferences (e.g. Egglishaw & Shackley, 1982; Heggenes, 1988a) (see Chapters 5, 6, & 7).

The most common method of representing structural diversity in a stream channel has been to use a combination of physical measurements and visual estimations (e.g. Binns & Eiserman, 1979; Milner *et al.*, 1985; Heggenes, 1988a;

**Table 3. 1. Examples of some habitat attributes used in assessments and evaluation of physical habitat structure for fish abundance. (from Milner et al, 1985)**

**Appendix 1.** Examples of habitat attributes used in evaluation schemes

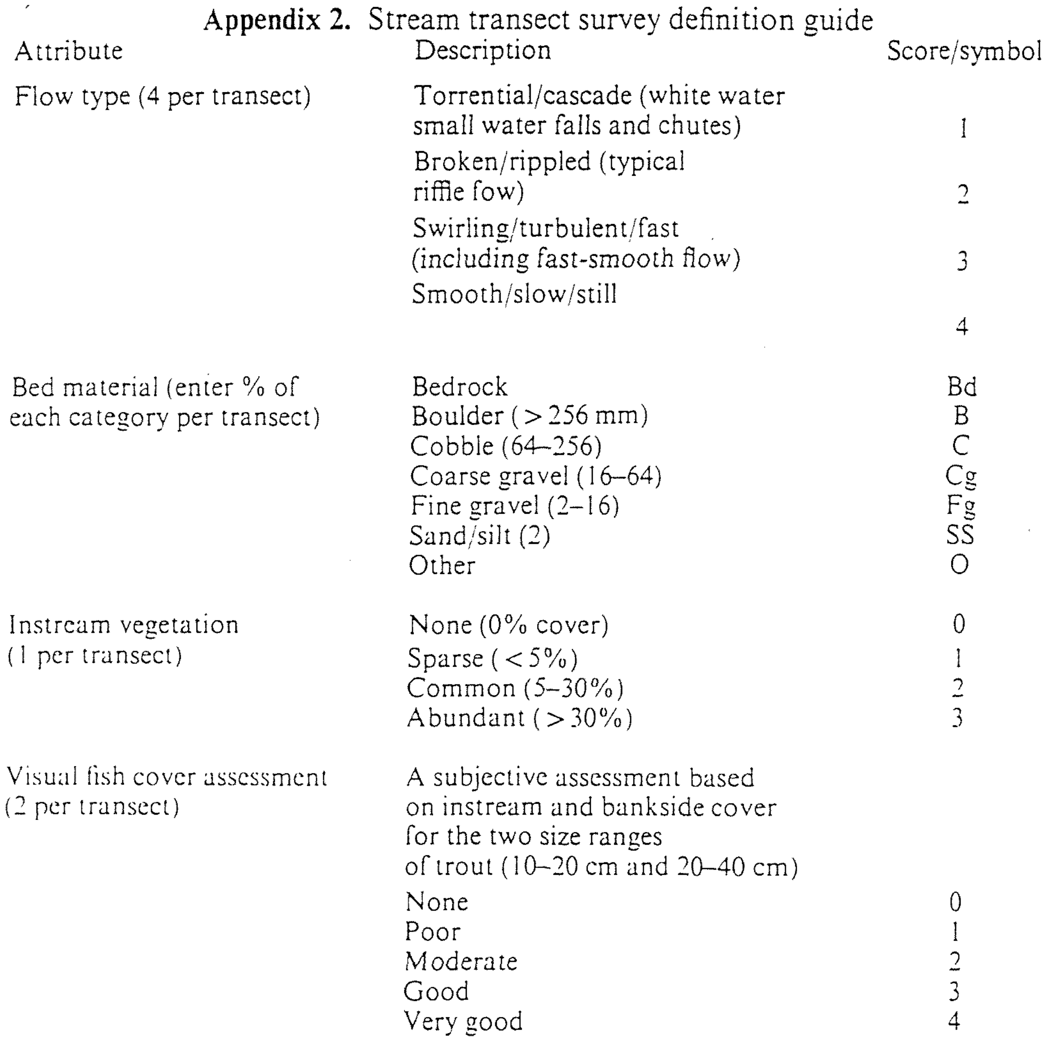
(A) Catchment attributes

| Geomorphological features | Hydrological features | Water chemistry features |
|---|---|---|
| Altitude | Average daily flow | pH |
| Geology | Average seasonal flow | Hardness |
| Catchment area | Pattern | Alkalinity |
| Total channel lengths | Extreme flow variations | Nitrogen ($NO_2$) |
| Drainage density | Stability of flow regime | Phosphorus |
| Mean basin length | Precipitation | Dissolved solids |
| Mean basin slope | | Conductivity |
| Forest ratio | | Temperature |

(B) Site attributes
Width
Depth
Substrate composition
Instream cover—debris, rocks, macrophytes
Bankside cover—undercut banks, overhanging vegetation, tree roots
Sinuosity
Bank erosion
Water surface area
Volume
Flow type
Riffle: pool ratio
Velocity
Gradient
Fish food abundance
Fish food diversity

**Table 3. 2. Guide and categories for stream transect surveys used for assessment of fish habitat. (from Milner *et al,* 1985)**

HABITAT EVALUATION



**Appendix 2.** Stream transect survey definition guide

| Attribute | Description | Score/symbol |
|---|---|---|
| Flow type (4 per transect) | Torrential/cascade (white water small water falls and chutes) | 1 |
| | Broken/rippled (typical riffle fow) | 2 |
| | Swirling/turbulent/fast (including fast-smooth flow) | 3 |
| | Smooth/slow/still | 4 |
| Bed material (enter % of each category per transect) | Bedrock | Bd |
| | Boulder (> 256 mm) | B |
| | Cobble (64–256) | C |
| | Coarse gravel (16–64) | Cg |
| | Fine gravel (2–16) | Fg |
| | Sand/silt (2) | SS |
| | Other | O |
| Instream vegetation (1 per transect) | None (0% cover) | 0 |
| | Sparse (< 5%) | 1 |
| | Common (5–30%) | 2 |
| | Abundant (> 30%) | 3 |
| Visual fish cover assessment (2 per transect) | A subjective assessment based on instream and bankside cover for the two size ranges of trout (10–20 cm and 20–40 cm) | |
| | None | 0 |
| | Poor | 1 |
| | Moderate | 2 |
| | Good | 3 |
| | Very good | 4 |

Simonson *et al.*, 1994). These can then either be used as empirical measurements or as categorical data using predetermined criteria (Watkins *et al.*, 1997). Some of these categorical data, e.g. substrate types, (Bain *et al.*, 1985) are defined by standardised empirical limits which are generally applied, but others, for example "cover" or "current velocities" may be differently defined as categorical variables in different streams and studies (e.g. Shields & Smith, 1992). Even when standardised limits are set, for example to the size of substrate particles (Bain *et al.*, 1985), the methods of estimating actual areas of substrate present in a channel reach vary between the external visual estimate and an estimate of area based on point or line-transect measurements (see Wallace & Benke, 1984; Kent & Coker, 1992).

Many of the studies have used combinations of continuous variables such as velocity or depth with categorical variables (see Milner *et al.*, 1985; Heggenes, 1988a). However, current velocity can vary widely over short time scales in any one system and any categories should have some empirical limits. Stream gradient is usually more stable and more reliable as an indicator of current type (e.g. Binns & Eiserman, 1979; Milner *et al.*, 1985; Heggenes, 1988a) though the use of gradient is limited for studies on very small spatial scales.

Structural diversity has been represented in terms of classic diversity indices (e.g. Gorman & Karr, 1978) with one of the commonest being the Shannon-Wiener index H' (Shannon & Weaver, 1963) usually complemented by the Equitability index J' The use of the various methods has been well reviewed by Magurran, (1988) and Kent & Coker, (1992). However, these indices are liable to be "richness-biased" in that they respond to the number of different categories used. They have been used in least squares regression to investigate relationships with fish abundance or diversity (e.g. Gorman & Karr, 1978; Binns & Eiserman, 1979). Alternatively, the physical parameters have been tested separately against fish abundance, diversity or size data using single variate or multi-variate least squares regression analysis (Egglishaw & Shackley, 1982; Kennedy & Strange, 1982). Where the data are not normally distributed and transformation is not satisfactory, Spearman Rank correlation analysis is used (e.g. Milner *et al.*, 1985). Where physical parameters such as depth and width or depth and specific substrates show significant co-linearity, some form of ordination method has been used to combine parameters into orthogonal or near orthogonal axes (e.g. Gilliam *et al.*, 1993; Paller, 1994; Martin-Smith, 1998). From such analyses various levels of correlation with both individual variables and combinations of variables have been used to explain relationships. These models may be among the most acceptable in terms of community and habitat studies though they are not always usable as precise methods for predictions of abundance or diversity.

For the studies reported here, the initial site and sampling-unit selection was based on visual selection of geomorphological channel units using the basic *fast-water/slow water* categories (e.g. Bisson *et al.*, 1982; Hawkins *et al.*, 1993; Martin-Smith, 1998). Within these categories CWD accumulations were selected out and isolated as separate *habitat units* as described later in this chapter. The physical characteristics of these pre-selected habitat units were subsequently compared and contrasted using measured variables. The measured physical variables were then used to try to explain the variations in fish abundance and community diversity within streams and between streams. Categorical variables were not used because of the difficulties of standardisation but, as will be seen in the next section it is difficult to make precise measurements for all aspects of the physical habitat. Because of the significance of the detailed physical structure of the stream channel and CWD to the analysis of the small-scale variations in fish abundance and diversity, the physical characteristics of the streams are described and analysed separately in Chapter 4. Chemical and invertebrate data are also summarised.

### 3. 2. 2. Definitions and measurement of "cover"

The extent of instream "cover" or refugia where fish can avoid predation or competitive aggression is regarded as a vital component of any physical model related to fish abundance (e.g. Heggenes 1988a; Ibbotson *et al.*, 1994; Copp & Bennetts, 1996; Prenda *et al.*, 1997). External cover in the form of overhanging trees or other vegetation have also been used to explain fish distribution on the microhabitat and mesohabitat scales (e.g. Binns & Eiserman, 1979; Hegennes, 1988a; Ibbotson *et al.*, 1994; Copp & Bennetts, 1996; Watkins *et al.*, 1997).

By far the majority of studies of the use and effects of cover have related to Salmonidae (Heggenes, 1988a; Fausch & Northcote, 1992), or larger non-salmonid species of commercial angling interest (Korman *et al.*, 1994; Harvey *et al.*, 1999). Thus in many studies the perception of desirable habitat features for Salmonidae or larger non-salmonids has generally related to perceptions of cover for fish over 1 year old or over a defined size threshold, accepting that there is clear habitat partitioning between smaller and larger fish in most streams (e.g. Binns & Eiserman, 1979; Egglishaw & Shackley, 1982, 1985; Milner, 1983; Milner *et al.*, 1985; Heggenes 1988a). Instream cover is generally defined as areas of water where fish over a given age or size can shelter from competitors, predators or from the more extreme flow conditions (Milner *et al.*, 1985; Heggenes, 1988a). Similar definitions have been given for various non-salmonid fishes (e.g. Gorman & Karr, 1978; Hortle & Lake, 1983). Estimates of "instream cover" or preferred habitat for other, "non-commercial" species, particularly the smaller species, are generally lacking from most of the habitat models. Some estimate of cover for larger fish is desirable in any study so that the effects of this physical characteristic can be assessed on both smaller and larger species. For example it could be predicted that the distribution of potential prey species has a negative relationship to cover features which may shelter predators.

The extent of fish "cover" is generally difficult to assess quantitatively and most studies have used a combination of objective measurements and visual estimates of cover features (e.g. Binns & Eiserman, 1979; Milner *et al.*, 1985; Heggenes, 1988a). More quantitative methods have included line transects using a weighted rope (e.g. Heggenes *et al.*, 1990) or point-transects (e.g. Binns & Eiserman, 1979; Heggenes & Saltveit, 1990). The weighted-line transect method involves assessing the length of the line on any transect which is obscured from above by in-stream structures such as bank overhangs, undercuts, logs, boulders or tree-root matrices. The point-transect method involves using a rod, usually the same rod used for depth measurements, one end of which is dropped on to the substrate at set points along a transect. The dominant substrate type under the rod is recorded at each point and the total number of points at which each is recorded used to estimate the approximate percentages of each substrate type in any reach.

Both line and point methods can also be used to give approximate areas of each substrate if the dimensions of the reach are known and both can provide reasonable estimates of instream cover provided the density of observations is satisfactory (Wallace & Benke, 1984; Heggenes, 1988a). For these studies point-transect measurements were used as described later in this chapter.

### 3. 2. 3. Quantification of wood debris in rivers

The methods used for the quantification of wood debris (CWD) in stream channels have not been standardised. Thus there is some inconsistency in the various studies. A selection of the methods is therefore outlined below to illustrate the range of approaches. The reasons for quantifying CWD (or LWD, large woody debris) have included comparisons of loadings between streams with different riparian vegetation or different degrees of logging, effects on stream channel morphology and change and effects on invertebrate and fish distribution and abundance. Thus the methods have reflected the requirements for precision and accuracy and have varied from visual estimates based on percentage of stream-bed cover to detailed measurements of individual wood pieces and calculated volumes and biomass.

The most detailed measurements have originated from studies where the CWD itself was the subject of the study. For example, Ralph *et al.* (1994) used the basic definition of LWD or CWD as any log over 0.1m in diameter and 3m in length for their study on the hydraulic effects of CWD on streams in the north - western states of the USA. They counted the number of logs in set lengths of stream and used wetted area as the basis for the analysis. They also used two categories of log based on diameter (0.1-0.5, >0.5) and estimated the number and size of logs in log jams over 10 pieces because of the physical difficulty of access to the individual logs. They used the morphological classification of Bisson *et al.* (1982) (macrohabitat units) as the basis for the analysis of the effects of logging on CWD loadings. Streams were compared on the basis of the degree of harvesting of trees in the various catchments categorised as *unharvested*, *moderate* and *intensive* and used the relative proportions of stream area formed into pools, riffles and cascades as an indication of stream habitat condition. Intensive harvesting reduced the pool areas but gradient also affected the number of pools present as might be expected. Riffles and cascades were not affected.

In low-gradient streams (<4%) which anadromous salmonids were known to use and where channel widths were less than 20m. Beechie & Sibley (1997) used 0.1m diameter and 2m long as their size definition of CWD logs but they classified the pieces as 0.1-0.2m x 2m, 0.2-0.5m x 3m and >0.5m x 5m as small, medium and large respectively. Visual estimates were mainly used, with some 10% of logs being measured to calibrate the estimates. CWD volumes were estimated as from lengths and diameters and volume per unit area of bankfull channel was used for comparisons. Areas of gravel and pool areas were also calculated from visual estimates. Volumes ranged from 0.001 to 0.42 $m^3$ $m^{-2}$ of channel area. There were no significant relationships between channel widths and the number of CWD pieces per metre length but there was a significant relationship when number per metre squared was used. Areas of gravel (regarded as potential spawning areas for salmonids) were not related to the abundance of LWD either as numbers of pieces or volumes of CWD in any reach despite the fact that woody debris is generally perceived as increasing local areas of spawning gravels in some streams (see House & Boehne, 1985).

Two categories of CWD were used in studies of the effects of buffer strips and logging activities on invertebrate and fish abundance in Tasmanian streams (Davies & Nelson, 1994). CWD volumes were divided into "*total volume of snags in* $m^3m^{-2}$ *of wetted area*" and "*submerged CWD as* $m^3m^{-2}$". Volumes of snags (CWD) were significantly correlated with stream length and volumes of submerged wood were correlated with open length, pool area and negatively correlated with riffle areas. Non-volumetric methods have used inventories to assess the amount of wood present in logged and unlogged streams in the American north west (Carlson *et al.*, 1990). Criteria include the 10cm x 1m size classification and position in the stream channel which was classified in relation to the CWD formation, i.e. clump or single. Estimation and visual assessment were checked by periodic measurements. In more detailed studies Robison & Beschta (1990) measured the large end diameter, mean diameter and length of individual pieces of CWD in a reach of an Alaskan stream and from this, calculated volumes. They also noted the horizontal orientation as an angle (0-180°) in relation to the direction of flow, classified each piece in relation to its degree of observable decay, and noted whether each piece was in contact with another or not.

Wallace & Benke (1984) working in the Ogeechee River in Georgia, USA, used the line intersect methods originally developed to measure logging residues (de Vries, 1974) for estimating the abundance of CWD in streams. Using stem diameters they calculated volumes and surface areas from the estimators for the surface area of a cylinder. They measured stem diameters at below the water surface, 0-1m above it and 1-2m above it. The specific gravity of different sizes and species of wood were used to calculate wood mass in the stream channel. O'Connor (1992) working in an Australian stream, used similar methods to estimate surface areas of cover, mean-log diameter and wood volume. All the log diameters were measured in each transect length and the data used to estimate the volume of wood per unit area. Smock *et al.* (1985) estimated the surface areas of snags using their mean diameter and length for studies of invertebrate distribution and abundance.

Categorical variables based on size data were used by Shields & Smith (1992) following visual survey methods from within the channel. Surveys consisted of counting all the pieces and formations in an area of the plane of the water surface larger than $1m^2$. Each was assigned to one of 9 categories of size. The size categories were assessed as a proportion of the stream width (B) as 0.25B, 0.5B etc. It was assumed that the depth of the formation was equal to the reach mean depth, therefore area multiplied by depth was the estimated volume. Volume of wood per unit volume of water and unit area of stream bed were calculated. Point-transect methods were used to determine substrates and current velocities. Cover classification was based on features such as undercut banks, log dams and marginal vegetation and points within 0.5m of each was regarded as "cover". Shannon-Wiener diversity indices (H') were calculated from the point-transect data. Hildebrand *et al.* (1997) used 4 size- categories of CWD in their stream to map existing structures before placing new structures in the various mesohabitats. They increased the loadings from between 0.45 and 1 piece of CWD per habitat unit (riffle or pool) to 5.2 – 20 pieces per pool in addition to the natural wood. Most assessments of CWD abundance in New Forest streams has been based on the densities of debris dams (accumulations) per 100m length of channel (Gregory *et al.*, 1985; Gregory & Davis 1992; Gregory *et al.*, 1994; Gurnell & Sweet, 1998) (see Chapter 4). Comparisons from sites in different regions of the world tabulated by Gurnell *et al.* (1985) show variations from 0-40 dams per 100m in 1st to 3rd order streams. Because of the lack of consistency in the methodology it is thus difficult to compare data on CWD abundance in relation to fish habitat from stream to stream or region to region with confidence though broad comparisons can be made. The relationships of CWD matrix structure to habitat diversity and biological diversity are not generally addressed in the literature.

Given the variety of methods used by previous authors it was difficult to find a common methodology though most studies have included measurements of some common variables (see Section 3.2.1). Of these common variables, substrate type, channel width, water depth, instream cover, and cover by CWD were the primary categories used in the New Forest. Most were related to the stream channel itself. Comparisons at the stream scale were made using the physical diversities of reaches, where the measurements of physical variables for the individual habitat units in a specified reach were summed to provide a reach diversity index. Comparisons of catchments were based on the extent of tree cover. primarily in the riparian zone of the respective streams.

## 3. 3. METHODS

### 3. 3. 1. Site selection

The Highland Water, Bratley Water and Bagshot Gutter were selected for the main programme on within-reach variation in fish distribution, abundance and diversity (see Chapter 2, Fig. 2.1), mainly because of previous studies of CWD and because land use in much of the catchment has been relatively consistent for many years (e.g. Tubbs, 1968, 1986; Gregory *et al.*, 1993). These streams are referred to collectively as the "Highland Water streams" in the text. All three run through mostly wooded catchments (see Chapter 4), and were known to have high densities of wood debris dams and high net loadings of timber debris relative to adjacent streams (Gregory *et al.*, 1993). Although results of detailed surveys have not been published since 1993, qualitative observations by the author and the physical assessments of structural diversity of the channel indicate that they can be ranked as "high-load". Of the 23 sites selected 20 were on the Highland Water, 2 were on the Bagshot Gutter and 1 was on the Bratley Water. The full list of sites and dimensions is given in Appendix I and locations shown on maps in Appendix II. For comparisons with other streams these three were classified as "wooded" (see Chapter 4).

To investigate the effects of different land-uses on the fish communities and to evaluate further the "between-stream" effects of varying amounts of CWD and riparian tree cover, a small series of samples was taken on two other streams, the Ober Water and Dockens Water, both of which were known to have lower proportions of riparian woodland (Gregory *et al.*, 1993), lower numbers of dams and lower net loadings of woody debris (see Chapters 2 & 4). These can be regarded as "low-load" streams (see Chapter 4). Both have lengths of non-wooded channel and for comparison were classified as "open" streams. No streams were found which combined high CWD loadings with a low amount of riparian tree cover (see Chapter 4). CWD loadings were found in earlier studies to be closely related to riparian land use (Gregory *et al.*, 1993), particularly the extent of woodland.

### 3. 3. 2. Sampling site selection and definitions

Potential sampling sites (reaches) in the Highland Water, Bratley Water and Bagshot Gutter were selected, *a priori*, from visual inspections of the stream to include a variety of structural and geomorphological features. The choice of each sampling reach was based on the presence of an accumulation of CWD. Reaches were selected to cover a range of forms and sizes of CWD formations based on the classification outlined by Gregory *et al.* (1993) and described more fully below. The aim was to select accumulations of varying sizes so that the amounts or proportions of CWD in a stream reach might be used as a more or less "continuous" variable in the subsequent analyses. This was considered as more representative of the actual stream system than using relatively simple but less realistic two-way comparisons, i.e. presence and absence of CWD. Most reaches selected were therefore based around a three dimensional CWD accumulation which extended either across all or part of the channel width (Fig. 3. 1) (Gregory, 1992; Gurnell & Sweet, 1998). All the CWD accumulations consisted of both submerged and exposed wood at dry weather flows.

Within the reaches, habitat sampling units were based on the channel-unit categories of "pools" and " riffles" (Bisson *et al.*, 1982; Hawkins *et al.*, 1993; Martin-Smith, 1998). These definitions use major physical discontinuities in the stream bed and characteristics of the flow and depth as assessed visually from the bank of the stream, initially to distinguish and define physically dissimilar sub-reaches of the channel or channel units (Hawkins *et al.*, 1993). Riffles and pools are relatively low level categories in the hierarchical system proposed by Hawkins *et al.* (1993), and can be regarded as sub divisions of their "fast-water" and " slow-water" categories respectively. The sub-reach scale was selected because of the need to quantify within-reach discontinuities in abundance, diversity and community composition of the fish in relation to detailed physical structures (Bohlin *et al.*, 1989; Angermeier & Smogor, 1995). The scale of sampling in relation to the provision of both ecological and management data is discussed in Chapter 8.

The sampling reach therefore comprised a group of sub-reach scale, *a priori* " habitat units" which were the units used for sampling fish. These habitat units were isolated from each other by stop-nets (blocking nets) prior to fish sampling (Fig. 3.1). The difference between this method and those in other similar studies (e.g. Inoue & Nakano, 1998) was that here the CWD accumulations were designated as separate habitat units and subsequently isolated and sampled as such. Because of the low gradients in New Forest streams (see Chapter 2), the number of different categories of channel unit is limited.

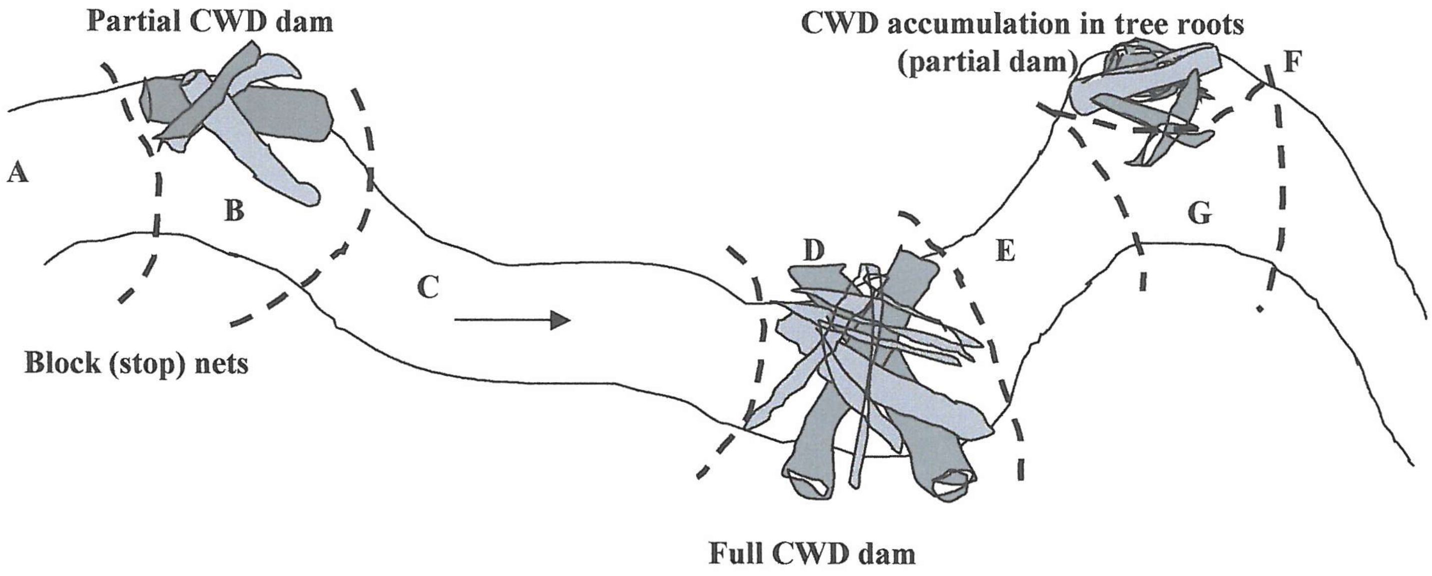


Figure 3. 1 Diagram of a New Forest stream reach with typical CWD formations, showing various stop net positions. The percentage area of each isolated habitat unit covered by CWD varied with the position of the net and the shape,size and position of the CWD matrix in relation to the channel morphology. A – G represent isolated habitat units of varying size, depending upon the visual identification of channel and flow changes.

The three categories of habitat unit on which fish sampling was based were as follows:-

> *Riffles:-* shallow, fast flowing reaches, including runs, with rippled or broken water surface and substrata almost entirely of gravel which varied in size from approximately 1cm to 12cm in diameter but mostly less than 6cm. Because of very localised irregularities in the channel structure, riffles could also include small areas of back eddies or slack water when isolated by stop nets across the channel. In the channelized reaches gravels were deposited on a base substrate of harder clay.
>
> *Pools:-* deeper, slow flowing reaches, including slow glides, often almost static in summer, with a smooth water surface and a variety of substrata usually including silt and leaves or twigs. The category also includes reaches which may be described more accurately as slow "glides" where there was some discernible current but no riffle area and the water was generally deeper than in riffles. (For comparison with CWD reaches, pools with no obvious accumulations of coarse wood debris were selected where possible). Most of these were close to or contiguous with the CWD or riffles (Fig.3.1) to reduce chemical and other habitat variability as far as possible. Any timber debris present was rarely in a definable matrix and was typically submerged scattered single pieces.
>
> *CWD accumulations (habitats):-* (CWD) typically deep, slow moving, unbroken water physically similar to and often contiguous with the pools (Fig.3.1) but containing clearly definable matrix structures of CWD either as a full-width or partial dam (see below). In most cases the dam was not the major impoundment structure, though it may have been originally. Sections in which dams were sited were mainly pools impounded by gravel bars immediately downstream of the dam (Bisson *et al.*, 1988; Gregory, 1992; Gurnell & Sweet, 1998). Effectively most CWD habitat units were components of a pool but separable as habitat units for sampling because of the close aggregation of the CWD.

In the following text and figures the habitat units are described separately as riffles, pools and CWD or generically as habitat units. In the "open" streams, Ober Water and Dockens Water, sites were chosen in both open and partly wooded reaches using the same visual selection methods as for the Highland Water streams.

Coarse woody debris (CWD) was defined for this study as all pieces of timber exceeding 10cm in diameter (Beechie & Sibley, 1997) and over 0.5m in length. Gregory (1992) classified CWD accumulations in New Forest streams in low to average discharge conditions mainly as "*partial dams, complete dams* and *active dams*" depending on the proportion of the channel width they occupied and their effect on flow and water level. A fourth category, which spanned the channel but only affected flows at near bank-full discharge was also identified. For this study on fish only two main categories, namely *partial* dams and *full-width* dams (Fig. 3.1) were used for initial site selection. *Full-width* dams include the *complete* and *active* categories of Gregory's definitions. Usually CWD dams and other accumulations were based on pieces of timber longer than 1m but shorter pieces and branches with diameters less than 10cm often formed significant parts of the structures.

A *partial dam* was defined as a three dimensional accumulation of timber debris, with a core structure of trunks and branches over 10cm in diameter, **none** of which occupied the full width of the channel. These were usually found in the channel margins either separately or partly among tree roots. Smaller diameter branches and twigs were often enmeshed in the larger branches. There was usually no general impedance of the main flow in the channel though the main flow of water could be diverted laterally. Diagrammatic examples are shown in Fig. 3.1. At some sites, where a full-width dam had been partly removed or washed away by floods the remaining CWD formed a partial dam.

A *full-width dam* was defined as a three dimensional accumulation of timber debris, with a core structure of trunks or branches over 10 cm in diameter, **some** of which spanned the full width of the channel (see Fig. 3.1). Typically, smaller branches and twigs were enmeshed in the lattice of larger pieces of wood and in autumn the spaces between these could become blocked by leaves causing differences in hydraulic head of up to 1m at higher flows between upstream and downstream of the dam. In other seasons such dams usually allowed water to pass through but in the more dense dams, there was clearly some impedance of the flow even at low stream discharge. Most of the full-width dams were associated with plunge or scour pools and gravel plunge bars caused by heavy scour at high discharges removing substrate from beneath the dam and displacing it downstream (see Chapter 4, Fig. 4.17) (Bisson *et al.*, 1988; Gregory, 1992).

Both partial and full-width dams were typically in the form of three-dimensional latticed matrices of timber debris extending throughout the depth of the water column, often with trunks and branches emerging to more than 0.5m above the water surface at low flows. Other timber debris was also scattered in other parts of the stream away from the major accumulations. These pieces of submerged timber debris occurred either singly or in very small numbers mostly fully submerged and lying on the stream bed. These were not identified as three dimensional matrices though they were subsequently recorded during the physical assessment of the habitats.

Physical variables measured within each sampling unit were subsequently used to explain the within-stream and within-reach variations in physical diversity of channel morphology and substrate types in relation to the presence and amounts of CWD. The physical data were also subsequently used in the analysis of fish abundance and distribution.

For comparisons between streams, chemical and invertebrate data were used. (see Chapter 4). The chemical data were obtained from the Environment Agency (Environment Agency, Public Register, 1988-1998). Invertebrate data were extracted from long-term studies of the distribution and composition of the invertebrate fauna of New Forest streams which began in 1992 (see Langford, 1996). Standard hand-net sampling was used (Southwood and Henderson, 2000) in mid-stream and marginal habitats. Identification was to family level for the description and comparison of the streams and only biological scores (BMWP, ASPT) (see Spellerberg, 1991) and numbers of families are used in Chapter 4

### 3. 3. 3. Physical measurements and substrate assessments

All the sites in the Highland Water, Bratley Water and Bagshot Gutter had relatively consistent tree cover and no polluting discharges. Thus chemical variation and riparian vegetation were considered as more or less constant and were not included in the within-reach habitat assessments, though chemical data from the Highland Water were used for overall comparisons with other streams (see Chapter 4). All physical measurements were taken within the areas of the defined and isolated habitat units in every reach selected. Wetted widths and water depths clearly vary with stream discharge in all streams. Thus sampling was carried out as near as possible at similar discharges (see Fig. 3.2) to reduce variation as much as possible. Even so, the dimensional differences between riffles, pools and CWD habitats may include a temporal as well as a spatial component. The structural features and substrate types included in the assessment of the physical diversity of each sampling unit are listed in Table 3.3.

In the New Forest streams "cover" (see Section 3.2.2) was assessed as an area in which a fish over 7cm total length (TL) would be obscured from above. Although this was to some extent an arbitrary definition, 7cm equates approximately to the total length of a 1yr old salmonid in these streams (see Chapter 7, also Mann, 1971). Thus open, shallow gravel riffles, with no overhanging banks and no vegetation or instream objects such as weed beds or timber debris were assessed as having no "cover". The major "cover" features included overhanging banks, bankside vegetation close to or trailing in the water, instream weed beds, undercut banks, submerged tree-root matrices and timber debris accumulations (Table 3. 3).

The occurrence of each category of cover (Table 3. 3) was indicated by noting the presence of the respective cover feature either contacted by or touching the measuring rod used for the point-contact measurements (see Section 3.2.2). The percentage occurrence of cover was estimated from the proportion of cover contacts in relation to the total point contact measurements. In some studies turbulent 'broken' water in riffles or runs has been included as a cover feature for

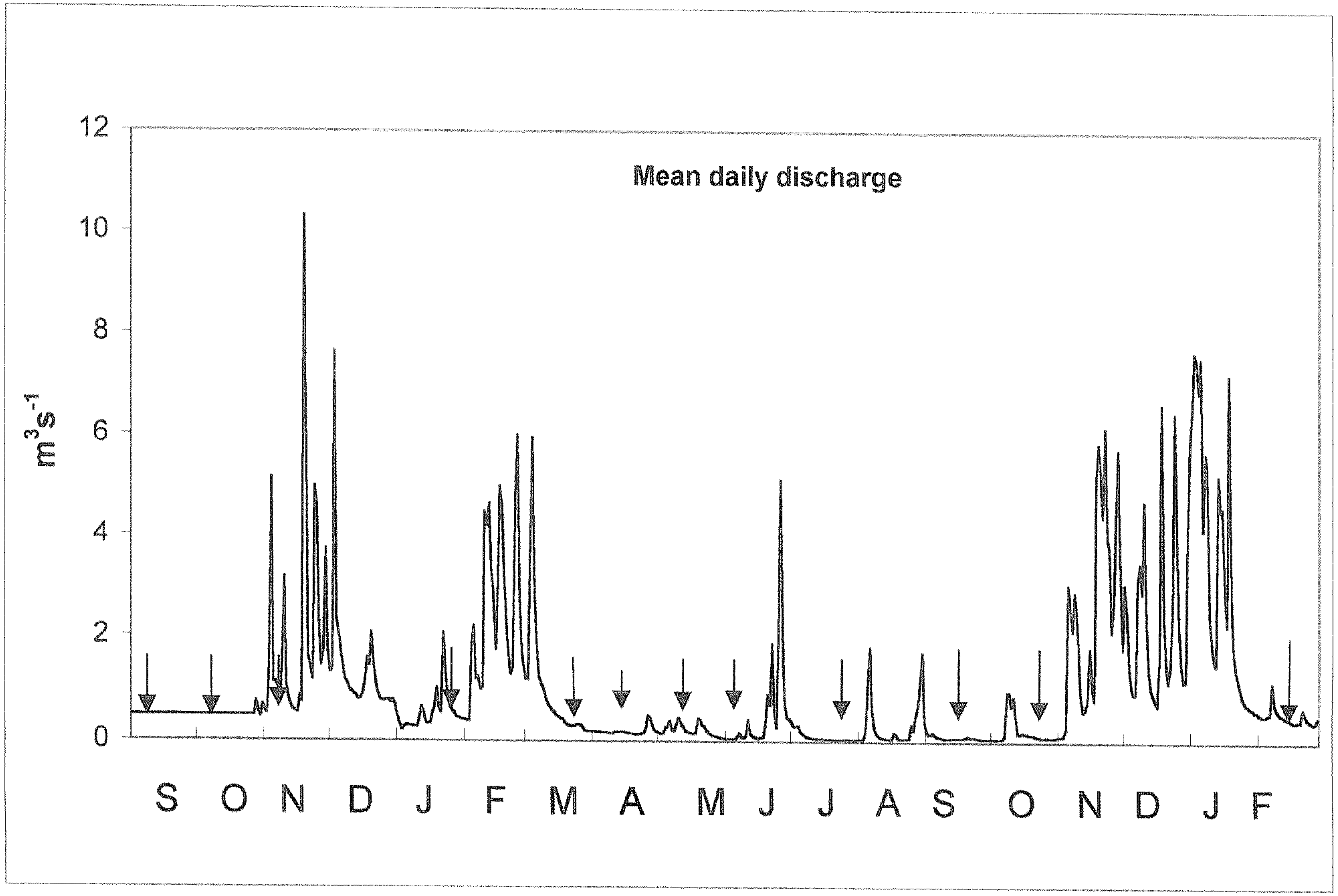


**Figure 3. 2 Mean daily discharges in the Lymington River at Brockenhurst gauging station 1996-1998.Fish sampling dates are indicated by vertical arrows**

**Table 3. 3 Physical features and substrate categories used in point-transect habitat assessments in each habitat unit sampled in New Forest streams**

| Feature/substrate | Description/comments |
|---|---|
| Length of reach | *In metres assessed as pooled length of both banks/2* |
| Mean width | *From all width measurements (min n= 5)* |
| Median depth | *From all point-transect data in each habitat unit* |
| Maximum depth | *From point-transect data* |
| Gravel/cobbles | *Includes diameters from 0.5- 12cm* |
| Silt /Mud | *Fines, including organic and inorganic particles* |
| Clay | *Mainly solid clay substrata exposed in channels* |
| Leaves | *Leaf packs and accumulations* |
| Wood | *Coarse wood debris, submerged /water column* |
| Twigs | *Finer wood, stem debris* |
| Weed | *Instream weed beds, (all species)* |
| Undercuts | *Undercuts and very low overhanging banks* |
| Roots | *Submerged tree root matrices* |
| Trailing vegetation | *From bankside grasses and shrubs* |

the larger fish (see Heggenes, 1988a), but at normal dry-weather flows the water depth in New Forest riffles is so low that this was not considered. Although the *occurrence* of cover was noted no estimates of the width of undercuts or areas of roots as cover were made for this account.

The point-transect method recorded contact with CWD as with other substrata and thus the percentage area of CWD cover was assessed as a percentage of the wetted area in each habitat unit. Depths of each habitat unit were measured across transects. Depths within and along the upstream and downstream extremities of the CWD matrix were also recorded. It was also noted whether the CWD contacted was part of a three-dimensional matrix or simply a submerged individual item on the stream bed. The shape of the channel and the configuration of each CWD accumulation in New Forest streams varied considerably. Thus the enclosure of each accumulation also varied in the closeness that the net could be sited to the woody debris (Fig. 3.1). Consequently the areas of open water and other substrata enclosed in the *a priori* CWD habitats varied with the site and sampling unit. Further, where sites were sampled more than once over the months, changes had occurred in the shape and size of the CWD accumulation between visits. In consequence the point-transect measurements showed a wide variation in CWD cover even in the same visually selected CWD reaches. Similarly, some pools and riffles selected visually for the absence of CWD accumulations actually contained small numbers of CWD pieces either spaced over the section or in a small group. Thus point-transect and cover estimates included varying amounts of wood in all three categories of habitat sampling unit providing a wide range of CWD densities. From the dimensions of each habitat unit and from point-contact assessments the approximate overall volumes of each CWD matrix were estimated as an overall volume.

$$M_v = A_w \times D_m$$

$M_{v\_}$= *volume of CWD matrix submerged, inclusive of interstices*, $A_w$= *Area of CWD as assessed from point transects,* i.e. *number of point contacts with CWD/Total number of points in the habitat sampling unit x 100*, $D_m$= *Median* *depth, i.e. median of depth measurements from water surface to the stream bed within and alongside the CWD accumulation.*

Other dimensional measurements included the length of both banks between the stop nets and the wetted width measured at approximately 1.8-2m intervals in longer reaches, though in shorter sections the intervals were reduced so that at least five widths were measured including depths very close to each bank. For each transect, water depth was measured at 50cm intervals across the stream. Again in very narrow reaches these intervals were reduced to give at least 4 measurements. Thus for each section at least 20 measurements of depth were obtained.

The proportional occurrence of substrate and cover features was recorded as percentage occurrence which was calculated as *"the number of recorded occurrences of a substrate/feature, divided by the total number of observations for the habitat unit sampled, multiplied by 100"* The classification of the substrate types is a simplified version of the modified Wentworth scale used by other authors (e. g. Bain *et al.*, 1985; Heggenes, 1988a; Heggenes & Saltveit, 1990). At the stream margins the presence of an undercut bank, overhanging vegetation or root mass was noted and recorded as an "occurrence". Full point-transect measurements were made from November 1996 to February 1998 and in subsequent measurement in other streams. In September and October 1996 only percentage of total natural cover and the percentage of CWD cover were estimated visually from measurements in each sampling reach. To calibrate the visual estimates, both point-transect and visual estimates were made after November 1996 (see Chapter 4).

### 3. 3. 4. Fish sampling

The main sampling period was from September 1996 to February 1998. Subsequent samples were taken until February 2000 for additional data and comparisons with other sites. The timing of any sampling was dictated mainly by stream discharge patterns and in some months no sufficiently long periods of low flows occurred (Fig. 3.2). Sampling was carried as nearly as possible under conditions of similar stream discharge. Prior to electric fishing, stop nets were placed carefully to isolate the selected habitat units as indicated in Fig. 3.1. Activity in the stream was restricted to a minimum during this isolation, although in such small streams some disturbance was inevitable. Following the isolation of the habitat units, time was allowed for any disturbed sediments to settle and the water to clear where necessary. Fish were subsequently sampled with a 240 volt smoothed DC electric fishing machine using a single operator moving upstream (e.g. Cowx, 1983; Bohlin *et al.*, 1989). Each section was fished at least three times (runs) even when the second fishing produced no fish. All individuals of each fish species were removed to holding tanks at the end of each run.. The catch from each sub-sample was kept separately. All fish except minnows (*P. phoxinus*) were measured, weighed and returned. Samples of minnows were retained for separate investigations. All electric fishing occurred during daylight hours. Sampling within the CWD accumulations was carried out by inserting the electrode between the timber pieces as far as possible from different angles. By careful insertion most areas of the CWD matrix could be reached and the direct current caused the fish to swim toward the electrode for capture. At various times individual fish were initially lost within the CWD but subsequently drifted into view while still immobilised.

Given that there was a finite number of debris dam sites and that sampling too frequently could cause permanent disruption to any reach, the sampling regime was based on a stratified random pattern so that reasonable spatial and temporal replication could be attained. Thus of the 23 sites originally selected in the Highland Water, Bratley Water and Bagshot Gutter nine were fished once only, nine twice, four three times and one site four times during the 12 monthly sampling visits (see Appendix I). The minimum time between successive samplings was one month. A total of 162 habitat units were sampled in the Highland Water streams comprising 43 riffles, 80 pools and 39 CWD sections with at least two and up to five replicate habitats units included in most months except in September 1996, June 1997 and July 1997 when only one riffle was sampled each month. Full point-transect data were obtained for 125 sampling units. A further 15 habitat units were sampled in the Ober Water and Dockens Water for comparative purposes. Chemical data were obtained from Environment

Agency surveys and additional fish data for comparisons with other streams from the relevant Environment Agency Fisheries Sections (Southern Area and Wessex Area).

### 3. 3. 5. Data analysis

The dimensions and physical characteristics of *a priori* riffles, pools and CWD habitats were initially compared using the measurements and the point-transect data. Mean and median dimensions were used for initial comparisons but frequency distributions of the means or medians were also compared. Frequency distributions of the proportional occurrences of substrates and structural features were also compared as part of the analysis to illustrate the extent of variation in riffles, pools and CWD habitats.

From the substrate data a Shannon-Wiener diversity index (H') and evenness index (J') was calculated for each habitat unit (Shannon & Weaver, 1963), using the computer program "Species Diversity and Richness" (Pisces Conservation Ltd., 1998). Simpsons diversity index (D) was also calculated for comparison with H' and J' (Gorman & Karr, 1978; Magurran, 1988).

Most of the physical data were not normally distributed and therefore comparisons of the median dimensions and occurrences of substrates were carried out using mainly non-parametric methods on untransformed or arcsine transformed proportional data. Despite the relatively low power of non-parametric statistical methods the more simple and conservative interpretation of the results was regarded as advantageous in the analysis of the relationships between habitat variables and habitat diversity and ultimately the relationships between the fish and their habitat. *A priori* comparisons of the physical dimensions, characteristics and diversity of the three categories of sampling unit were made using Kruskal-Wallis analysis of variance on ranks and *a posteriori* comparisons using Dunn's multiple pairwise test from the computer program Sigma-Stat v2.03. Spearman rank correlation was used to investigate colinearity between variables and Detrended Correlation Analysis (DCA) was used to test the relationship between physical variables and habitat diversity. DCA is a method of ordination of complex data, reducing numbers of highly correlated variables to a small number of axes which can then be used to compare data two dimensionally and display relationships in terms of distances along two axes. The method resolves some of the mathematical problems of Principal Component Analysis and is described and compared with other ordination methods by Kent & Coker (1992).

Population densities of each species of fish were estimated for each *a priori* sampling unit were possible using a modification of the constant probability of capture method (Zippin, 1956; Carle & Strub, 1978). The maximum weighted likelihood modification of the Zippin method described by Carle & Strub (1978) was used even where data where satisfactory for the normal Zippin method (Cowx, 1983). The equations for all methods are given by Cowx (1983) and included in the computer program REMOVAL (Pisces Conservation Ltd., 1998). Where the data did not fit the Carle & Strub model a regression method was used (see REMOVAL, Pisces Conservation Ltd., 1998). Catches, population estimates and 95% confidence limits are shown in Appendix III.

The basic conditions for the successful use of the constant probability methods were considered to be met in most samples, namely that the probability of capture remained constant, the population was stable with no immigration or emigration between samplings and the capture of any individual did not interfere with the capture of any other. The "chance of equal capture for all individuals" is known not to apply to electric fishing for certain fish species where larger individuals can be more susceptible to capture than smaller individuals (e.g. Cowx, 1983; Bohlin *et al.*, 1989). Thus, where possible, separate population estimates were carried out using the catches of different size groups. This applied particularly to Salmonidae in some seasons (see Chapter 7). In many cases, catches were so small that it was not practical to separate the size classes of species. In such cases catches equated with population estimates.

Catch efficiencies were estimated for all the species where possible (see Chapter 5), using catch in the first fishing run divided by the population estimate and are given in the relevant results sections in later chapters (see Chapter 5). Where population estimates were not feasible, efficiency was estimated as catch in the first run divided by total catch. Where three catch data were not acceptable in the models, two catch estimates (Seber & Le Cren, 1967) were used if the second and third catches showed a suitable level of reduction. The catch from the first run was then added to produce the population estimate. In the event of failure of all population estimate models, the total catch was used as the minimum estimated population (Mann, 1971).

Each fish was weighed or its length measured or both. Where only one measurement was available the other was estimated from length-weight regressions. Biomass estimates were obtained by multiplying the average catch weight by the population estimate for each species or, in the event of a failure to estimate the population, the total catch number. For salmonidae, with a large individual size range, the biomass was estimated by using mean individual weights of designated length-groups.

Densities and biomass are expressed here initially in relation to wetted area of stream bed for the within-stream and between-stream comparisons. To assist the interpretation of the areal data, abundance in relation to standing volume is also used. Standing volumes of each habitat sampling unit $m^3$ (V) were estimated as:-

$$V = L_c \text{ x } W_m \text{ x } D_d$$

Where $L_c$ = *channel length expressed as total length of both banks/2 in m*
$W_m$ = *mean width in m,* $D_d$ = *the median depth in m*

Fish distribution, abundance and community data were initially analysed in relation to riffles, pools and CWD habitats as selected by visual observation. To explain the data DCA axes were constructed from the population data and subsequently related to the DCA axes from the analysis of physical variables. The data for Salmonidae are partially analysed in Chapters 5 and 7 but a full analysis of the data for this group is dealt with separately in Chapter 7.

As with the physical data the variance in the fish abundance data was large. Tests for normal distribution of fish density and biomass data failed. Further, transformation using $\log_{10}$ or $\log_{10}+1$ also failed to normalise the data and homogenize variances and thus *a priori* comparisons of the fish data from habitat units were made using the Kruskal-Wallis analysis of variance on ranks (see Elliott, 1977; Sokal & Rohlf, 1981). *A posteriori* tests used Dunn's pairwise multiple comparison test. Correlation coefficients were calculated using Spearman rank correlation. Percentage and proportional data were subjected to arcsine transformation where necessary to allow for problems with closed data sets (see Kent & Coker, 1992). Detrended correspondence analysis (DCA) was used to determine the closeness of relationships between variables and species. The data for fish abundance as numbers $m^{-2}$ from all the habitat units were inserted into the DCA together with the data from physical measurements to determine their relative positions on the primary and secondary axes (see Townsend & Peirson, 1988). All data were processed using the program CAP (Pisces Conservation Ltd., 1998), Sigma-Stat Version 2.03, Microsoft Excel 97, and XL Stat. To analyse the interactions between fish diversity and abundance in relation to season and habitat unit, two-way analyses of variance (General Linear Model) were performed on the ranked data (Sigma-Stat ver 2.03). To improve replication and statistical viability seasonal differences were further analysed using fish data pooled in three monthly (seasonal) groupings and the Kruskal-Wallis analysis of variance on ranks. The seasonal grouping were as follows:- *Winter:-* December, January, February; *Spring*:- March, April, May; *Summer:-* June, July, August; *Autumn:-* September, October, November.

Preliminary models of the potential effects of changes in the management of CWD in the Highland Water streams on the fish populations are proposed in Chapter 8 based on the analyses of structural changes and the species abundance and community data. Median and quartile densities of individual species were used in conjunction with physical changes predicted from physical measurements and with data from the long-term studies of CWD and channel structure (Gurnell & Sweet, 1998). A general discussion of the methodology and limitations is included in Chapter 8.

# CHAPTER 4

# HABITAT DIVERSITY, CWD, WATER QUALITY AND INVERTEBRATES IN THE HIGHLAND WATER AND ASSOCIATED STREAMS

## 4. 1. INTRODUCTION

The organisation of fish assemblages along environmental gradients (Matthews *et al.*, 1992; Paller, 1994; Williams *et al.*, 1996; Prenda *et al.*, 1997), including definable zones (e.g. Huet, 1959; Elser, 1968) or in association with specific micro, meso or macrohabitat features (e.g. Townsend & Peirson, 1988; Angermeier & Schlosser, 1989; Watkins *et al.*, 1997; Martin-Smith, 1998) depends upon the requirements of the species within the assemblage and the degree of discontinuity within the habitat at varying scales (Capone & Kushlan, 1991). Within–stream zonation or habitat association will also depend upon the degree of discontinuity within the stream and the mobility of species between the recognisable habitat units. For example species living in streams which become separate pools in dry periods (e.g. Capone & Kushlan, 1991) can vary between close association with those specific habitats (pools) in dry times and living in an environmental gradient at times of increased flow when the pools become contiguous with riffles and runs. An important part of any analysis is to determine whether the physical environment itself can be considered as a series of separate readily classifiable units or whether there is a continuum (Vannote *et al.*, 1980) based on a clearly definable gradient of physical variables.

Coarse woody debris (CWD) is considered to be a major contributor to habitat diversity in streams both directly and indirectly (e.g. Bisson *et al.*, 1982; Hawkins *et al.*, 1993; Gurnell *et al.*, 1995; Cowx & Welcomme, 1998). Whether CWD creates a specialised habitat which provides a specific and exclusive niche for any fish species or groups of species is not well known as much of the work on CWD has only involved salmonid habitat. Further in all the studies quoted here CWD accumulations were not isolated from other habitat units. There are indications from some studies of particular species being more abundant in or near CWD accumulations than in other locations in a channel (Hortle & Lake, 1983; Thévenet & Statzner, 1999) but data on fish within CWD accumulations do not appear in the literature with the exception of early data from this work (Langford and Hawkins, 1997).

Although CWD contributes to the structural diversity of the stream channel on the reach scale and above (Shields & Smith, 1992; Hawkins *et al.*, 1993) the physical diversity of CWD accumulations themselves as habitat units has not been assessed on the within-reach scale. In the Highland Water and its associated streams the role of CWD as a channel modifier and the dynamics of CWD have been studied extensively (e.g. Gregory, 1992; Gregory *et al.*, 1993) but again no attention has been paid to the CWD accumulations as separate habitat units for fish or invertebrates (Langford, 1996).

The removal of riparian woodland canopies has long been known to cause geomorphological, temperature and biological changes in the streams which drain the relevant catchments (e.g. Burton & Odum, 1945; Bilby & Bisson, 1992; Reeves *et al.*, 1993; Davies & Nelson, 1994; Copp & Bennetts, 1996). In the longer term the changes in light penetration result in a change from an allochthonous energy to an autochthonous energy driven ecosystem with consequent changes in the biota. In the New Forest woodlands have been cleared to create grazing lawns over many centuries (Tubbs, 1968, 1986) and the streams draining these areas are maintained by weed cutting and CWD clearance. They form long-established contrasting habitats to the wooded streams of the Highland Water catchment and were predicted to have different structural diversity and hence fish populations.

The aim of this chapter is, therefore, to describe and compare the physical habitats of the Highland Water and its associated streams and to review the data on the distribution and abundance of CWD as a background to the analysis of the data on fish populations. The aim is also to determine whether CWD can be identified as a cause of within-reach structural habitat discontinuity and as a separate habitat unit from riffles and pools based on physical variables and dimensions. This analysis will provide the basis for the quantitative analyses of fish abundance, diversity and community structure. The physical habitat diversity of the Highland Water streams is also compared on the reach scale and on the stream scale with that of the channels in the less-wooded (open) catchments.

The partial removal of a large, full-width CWD dam prior to winter spates provided the opportunity for a preliminary analysis of the within-reach effects of the partial removal of a such a dam on the physical habitat. These could be compared with effects within a reach where a dam was retained during the same period. These preliminary analyses also form the basis of the subsequent analysis of the abundance of fish species, community diversity and composition in relation to CWD removal and retention.

## 4. 2. DESCRIPTION OF THE HIGHLAND WATER AND ASSOCIATED STREAMS

### 4. 2. 1. The streams and catchments

The general descriptions of the five streams sampled for this work are given in Table 4.1. The Highland Water is a 1st – 4th order stream in the study area (Horton, 1945; Gurnell & Sweet, 1998) based on the 1:25000 scale map of the region (see Appendix II). Part of the stream drains the New Forest research catchment of the University of Southampton (Sear & Arnell, 1997). It rises as two main tributaries on the higher part of the heathland in the northern part of the New Forest (see Chapter 2, Fig. 2.1, Appendix II). The stream and its catchment are described in various publications and have been the subject of a series of studies for over two decades (e.g. Moore, 1977a, b; Gregory *et al.*, 1985; Langford, 1996; Sear & Arnell, 1997; Gurnell & Sweet, 1998). All of the study sites were upstream of the A35 (see OS map, Outdoor Leisure 22, Ref. 274068). The catchment is underlain by a mixture of clays, sands and gravels which strongly influence the morphology and hydrology (Tubbs, 1968, 1986; Sear &

**Table 4.1 Approximate dimensions,gradients and catchment uses of streams used for fish studies in the New Forest**

| Stream | Approx.length (km) | Approx.gradient | Height of Source (m) | Land uses |
|---|---|---|---|---|
| **Highland Water** | 12.5 | 0.007 | 100 | Heath,wood |
| **Bratley Water** | 12.5 | 0.007 | 95 | Heath,wood |
| **Bagshot Gutter** | 3 | 0.013 | 75 | Wood |
| **Ober Water** | 13 | 0.006 | 90 | Heath,lawn,pasture,gardens |
| **Dockens Water** | 11 | 0.009 | 95 | Heath,lawn,pasture,wood |

***lengths and gradients are approximate and estimated from 1:25,000 OS map.***

Arnell, 1997). The vegetation in the catchment is partly semi-natural and partly plantation. The upper reaches are bordered by dry and wet heathland and the reach within which most of the sampling sites lie is heavily wooded. Some road-drainage water enters the upper reaches from the main A31 dual carriageway.

A good part of the length of the Highland Water channel is highly sinuous and meandering with large numbers of wood debris accumulations though land drainage works at various times have included straightening and deepening parts of the channel (Tubbs, 1986). In some reaches remains of the original channel meanders are clearly identifiable though the bed in these is up to 0.5m above the present bed level. Despite the management of the Highland Water and the removal of debris dams, the pool-riffle sequence over a 6km length was found to be very similar to that of most rivers with an inter-riffle distance of 5-7 bankfull channel widths (Gregory, Gurnell, Hill & Tooth, 1994). Both the inter-riffle distance and the bankfull width increased downstream from averages of about 15m and 2m respectively in the upper reaches to 30m and 4.5m at the downstream end of the reach studied. The length of the reach studied by Gregory and subsequent authors (e.g. Gurnell & Sweet, 1998) equates almost exactly with the length of the Highland Water studied for the work described in this thesis.

Discharge in the Highland Water was not measured specifically for this work but as a main tributary of the Lymington River it followed the general pattern of New Forest streams (Langford, 1996) in that it is subject to rapid rise and fall in response to precipitation (flashy) and is highly variable seasonally (see Chapter 3, Fig. 3.2). In dry periods, particularly in summer, flows in the reaches studied became so low that there were only small trickles of water over the riffles between the pools. At such times current velocities in the pools and CWD habitats were undetectable with standard current meters. Examples of the rapid rise and fall of stream discharge after rain are given in Chapter 2 (Fig. 2.2).

Suspended solids loads in excess of 1000mgl$^{-1}$ have been recorded (Sear & Arnell, 1997) and the gravel substrata are highly mobile as in other New Forest streams (see Chapter 2). Annual run-off from the catchment is approximately 320mm, which corresponds to a mean daily discharge of 0.12m$^3$s$^{-1}$. Sear &

Arnell (1997) consider that the Highland Water represents "*features of pre-disturbance river systems that are rare in lowland Britain*" and the stream is "*a natural analogue for the study of past river processes*".

The Bratley Water (also known as the Black Water and Fletchers Water in different reaches) (see Chapter 2, Fig. 2.1, Appendix II) is a first to second order stream which originates a short distance to the west of the main source of the Highland Water and flows more or less parallel to it to join the Lymington River at Boldreford Bridge (OS:- SU 291041). The stream is slightly smaller than the Highland Water with very similar overall flow patterns. The riparian vegetation of the Bratley Water was similar to that of the Highland Water during the course of this study, namely heathland in the upper reaches and woodland lower down but in 1997-98 marginal conifers were cleared from some reaches.

The Bagshot Gutter is a short first order stream which joins the Highland Water upstream of Millyford Bridge (OS:- SU 284084) (Fig.2.1). It is a small, short stream, (ca 2.5km) with very low average flows and the catchment is fully wooded from its source to its confluence with the Highland Water.

The Ober Water and Dockens Water, both first to second-order streams, were selected for comparisons with the Highland Water streams (see Chapter 2, Fig. 2.1, see Appendix II). The Ober Water is also a major tributary of the Lymington River, which it joins almost at the same point as the Highland Water near Boldreford Bridge. The stream has a high proportion of non-wooded riparian land in the catchment, including grazing lawns (see Chapter 2; Tubbs, 1986) and some low density, semi-urban areas. In earlier studies, the stream was found to carry a relatively low density of CWD in contrast to the Highland Water and its associated streams (Gregory *et al.*, 1993). Dockens Water was the subject of earlier work on fish (Mann & Orr, 1969; Mann, 1971). Its source is some 3km north of the Highland Water on the same ridge but it flows approximately westwards to the River Avon. It has grazing lawn, improved pasture, woodland and heath in the catchment (Table 4.2) with an estimated occurrence of CWD similar to that of the Ober Water.

Table 4. 2. **Approximate percentages of different riparian land uses along the lengths of streams used in fish studies in the New Forest. 1996-2000**

| River/Stream | Heath/mire | Woodland | Lawn/Open | Agricultural | Semi-urban |
|---|---|---|---|---|---|
| Highland Water | 14 | 82 | 2 | 2 | 0 |
| Bratley Water | 16 | 82 | 1 | 1 | 0 |
| Bagshot Gutter | 0 | 100 | 0 | 0 | 0 |
| Ober Water | 16 | 20 | 40 | 15 | 9 |
| Dockens Water | 34 | 33 | 6 | 27 | 0 |

*NB % are as lengths of stream bordered by the land uses*

### 4. 2. 2. Water quality

Water quality in the three Highland Water streams is generally good (Sear & Arnell, 1997; Environment Agency, Public Register, 1988-1998) despite occasional run-off from the A31 road in the uppermost reaches. Fig. 4.1 shows the variations in the main chemical determinands taken from monthly samples at Boldreford Bridge on the Highland Water just upstream of its confluence with the Lymington River (Environment Agency, Public Register, 1988-1998) over the period from September 1996 to April 1998. pH ranged from 6.4 -7.5, conductivity (@ 25°C) from 150-250 $\mu Scm^{-1}$, temperature from 4-17°C, dissolved oxygen from 78-110% saturation, BOD from 1-2.1$mgl^{-1}$, ammoniacal nitrogen from 0.03–0.13$mgl^{-1}$ and nitrate from 0.2-0.85$mgl^{-1}$. There was an atypical rise in ammoniacal nitrogen and nitrate in the winter of 1996 which coincided approximately with the peak flows for that period (see Chapter 3, Fig. 3.2). There is no specific explanation for this other than association with the high discharge though the concentrations took some 5 months to reach pre-peak levels. For most determinands the Highland Water and Bratley Water were within the typical ranges for New Forest streams without significant urban run-off.

Fig. 4.2 shows the median and quartile values for five chemical determinands for selected New Forest streams. Data from streams other than those used for fish sampling are included to show the ranges within the area and place the study streams in context. The streams fall into two groups, viz. those of the "forest proper" (Highland Water (Hi), Ober Water (Ob), Bratley Water (Bl), Huckles Brook (Hu) and Dockens Water (Do), with little or no urban or agricultural run-off or effluents, and the streams of more urbanised or agricultural catchments, viz. Danes Stream (Da) and Lymore Brook (Le). The Highland Water is more or less typical of the forest streams. Both Dockens Water and the Ober Water have varying amounts of improved pasture or open lawn in their catchments (Table 4.2) but the basic chemistry is similar to the more wooded Highland Water and Bratley Water.

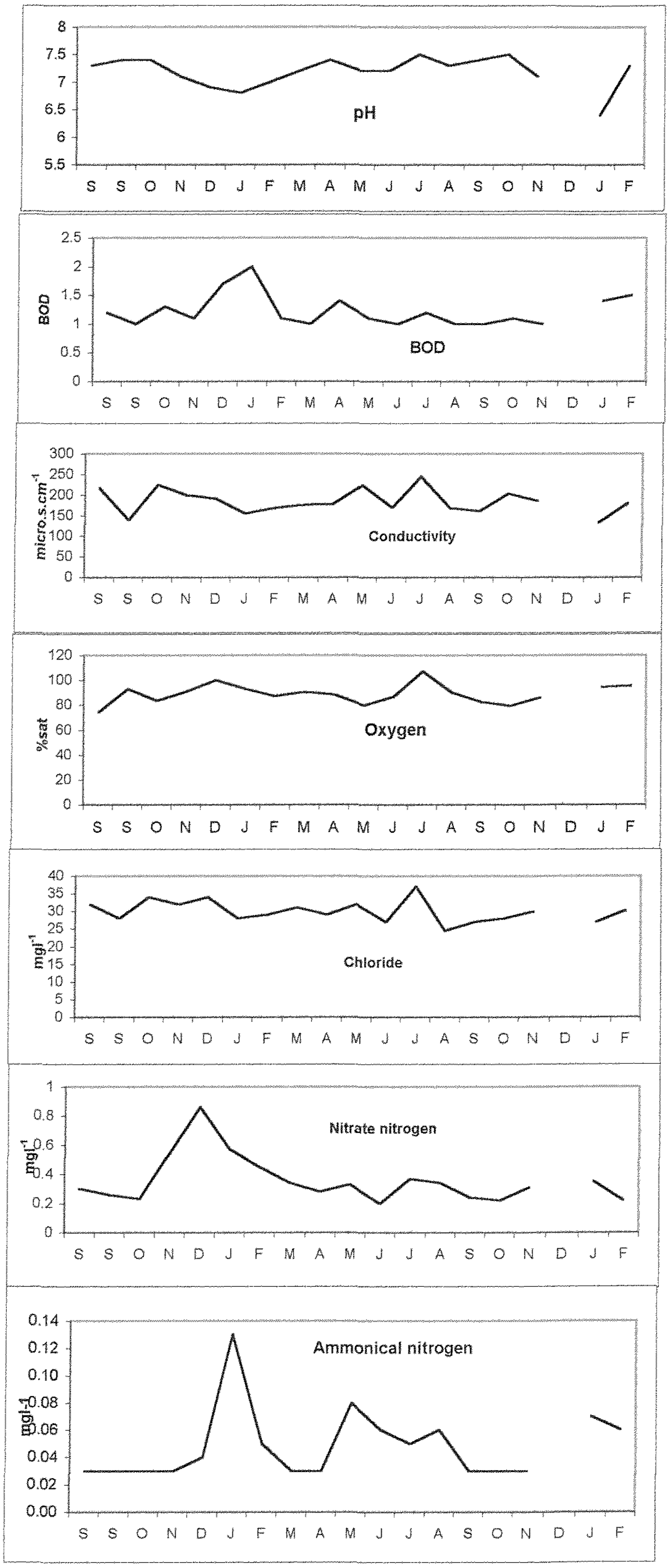


**Fig. 4.1** **Chemical data from monthly samples taken from the Highland Water 100m upstream of the confluence with the Lymington River. (Environment Agency, Public Register, 1988-98)**

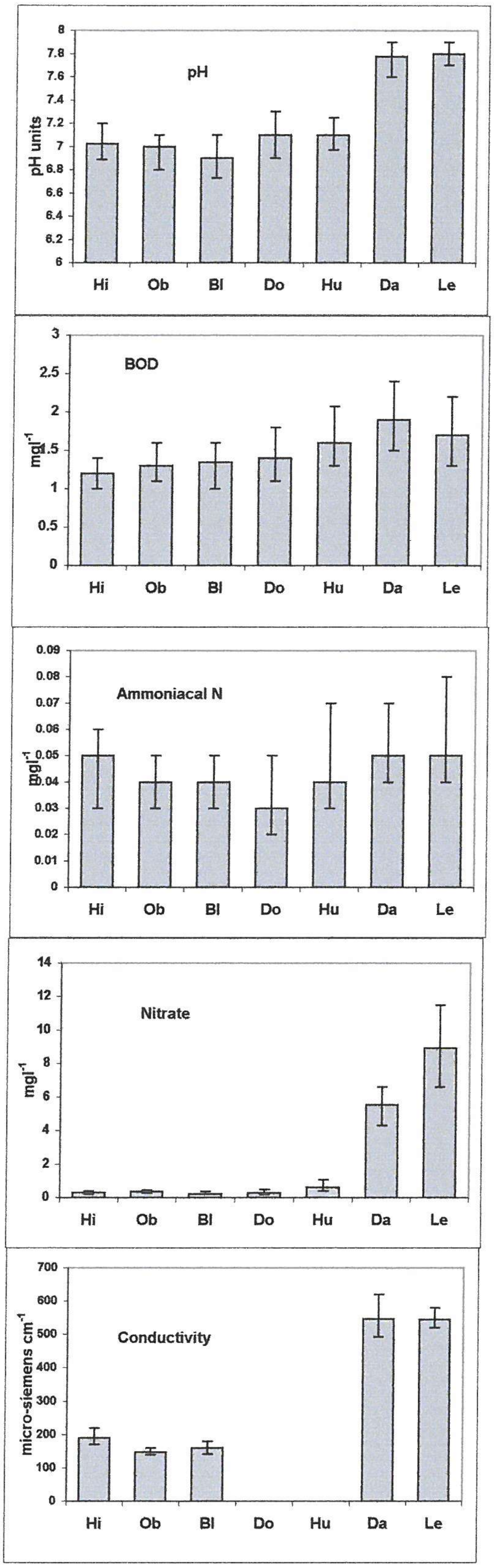


**Fig.4.2. Median and quartile values for selected chemical determinands from New Forest streams.** *(Hi=Highland Water, Ob = Ober Water, Bl = Bratley Water (also known as Black Water, Do = Dockens Water, Hu = Huckles (Latchmoor) Brook, Da = Danes Stream, Le = Lymore Brook.* ***Danes Stream and Lynore Brook are examples of streams darining semi-urban and agricultural land respectively.***

Table 4.3 shows the results of Kruskal-Wallis one way ANOVA on ranks for selected chemical determinands for four streams, viz. The Highland Water, Bratley Water (or Black Water), the Ober Water and Dockens Water (N = 35-122). Overall the chemistry of the four streams was very similar with no stream showing consistently different water quality overall. Where differences were statistically significant the large numbers of samples was an important factor. Dockens Water showed a significantly higher ($p<0.05$) median pH than the Bratley Water and a significantly higher median BOD ($p<0.05$) than the Highland Water. The Highland Water showed significantly higher median ammoniacal nitrogen concentrations ($p<0.05$) than Dockens Water and significantly higher conductivity values ($p <0.05$) than the Bratley Water or Ober Water. Conductivity was not measured for Dockens Water (Environment Agency, Public Register, 1988-1998), but data from other surveys (McCollin, 1993) gave a conductivity measurement of 122 $\mu Scm^{-1}$ for Dockens Water, which is at the lower end of the scale for New Forest streams. The differences between streams with urban, agricultural and forested catchments were also clear. Conductivity in forested streams ranged from 89 – 293$\mu Scm^{-1}$, urban and agricultural from 258 – 788$\mu Scm^{-1}$.

In 1969-1972 (Marker, 1976), nitrate concentrations were similar to those of 1993-1999 (Fig. 4.2), with ranges of 0.1–0.8$mgl^{-1}$ for Dockens Water and 0.1–1.4$mgl^{-1}$ for the Ober Water. The data from various sources suggest that the chemistry of the streams studied has been relatively consistent for several decades at least.

### 4. 2. 3. Invertebrates

Invertebrate surveys of New Forest streams have been carried out as part of the overall studies since 1992 (Langford, 1996; see Chapter 3). Data extracted from various sites on four streams, the Highland Water, Bratley Water, Dockens Water and Ober Water show that there were no significant differences in diversity (as H') with distance downstream in the reaches sampled (Fig.4.3). Further, there were no significant differences between the overall diversity, numbers of

**Table 4.3 Comparisons of selected chemical determinands from New Forest streams used for fish studies. 1996-2000**

***Data from the Environment Agency*** ***(Medians and quartiles)***
***(results of Kruskal-Wallis and Dunns pairwise tests shown)***

*a) pH*

| Streams | N | Median | 25% | 75% |
|---|---|---|---|---|
| Bratley Water | 106 | 6.9 | 6.73 | 7.1 |
| Highland Water | 45 | 7.025 | 6.89 | 7.2 |
| Ober Water | 35 | 7 | 6.8 | 7.1 |
| Dockens Water | 122 | 7.1 | 6.9 | 7.3 |

*H = 12.672, df = 3, p<0.005* *Dockens>Bratley ,All others NS (p=0.05)*

*b) BOD*

| Streams | N | Median | 25% | 75% |
|---|---|---|---|---|
| Bratley Water | 115 | 1.35 | 1 | 1.6 |
| Highland Water | 109 | 1.2 | 1 | 1.4 |
| Ober Water | 102 | 1.3 | 1.1 | 1.6 |
| Dockens Water | 122 | 1.4 | 1.1 | 1.8 |

*H= 15.436, df=3, p=0.001)* *Dockens>Highland, All other NS (p =).05)*

*c) Ammoniacal Nitrogen*

| Streams | N | Median | 25% | 75% |
|---|---|---|---|---|
| Bratley Water | 74 | 0.05 | 0.03 | 0.05 |
| Highland Water | 74 | 0.05 | 0.03 | 0.06 |
| Ober Water | 75 | 0.04 | 0.03 | 0.05 |
| Dockens Water | 122 | 0.03 | 0.03 | 0.05 |

*H=13.173,df=3,p=0.004* *Highland > Dockens ,All others NS, p=0.05*

*d) Nitrate Nitrogen*

| Streams | N | Median | 25% | 75% |
|---|---|---|---|---|
| Bratley Water | 74 | 0.22 | 0.2 | 0.37 |
| Highland Water | 74 | 0.3 | 0.22 | 0.4 |
| Ober Water | 75 | 0.36 | 0.28 | 0.458 |
| Dockens Water | 122 | 0.29 | 0.19 | 0.49 |

*H = 14.730, df=3,p=0.002* *Ober > Dockens, Ober >Bratley,*
*All others NS, p= 0.05*

*e) Conductivity*

| Streams | N | Median | 25% | 75% |
|---|---|---|---|---|
| Bratley Water | 94 | 148 | 139 | 160.25 |
| Highland Water | 92 | 190 | 170 | 220 |
| Ober Water | 91 | 160 | 141 | 180 |
| Dockens Water | No data | | | |

*H = 71.631,df=3,p=<0.001* *Highland>Ober>Bratley (p<0.05)*

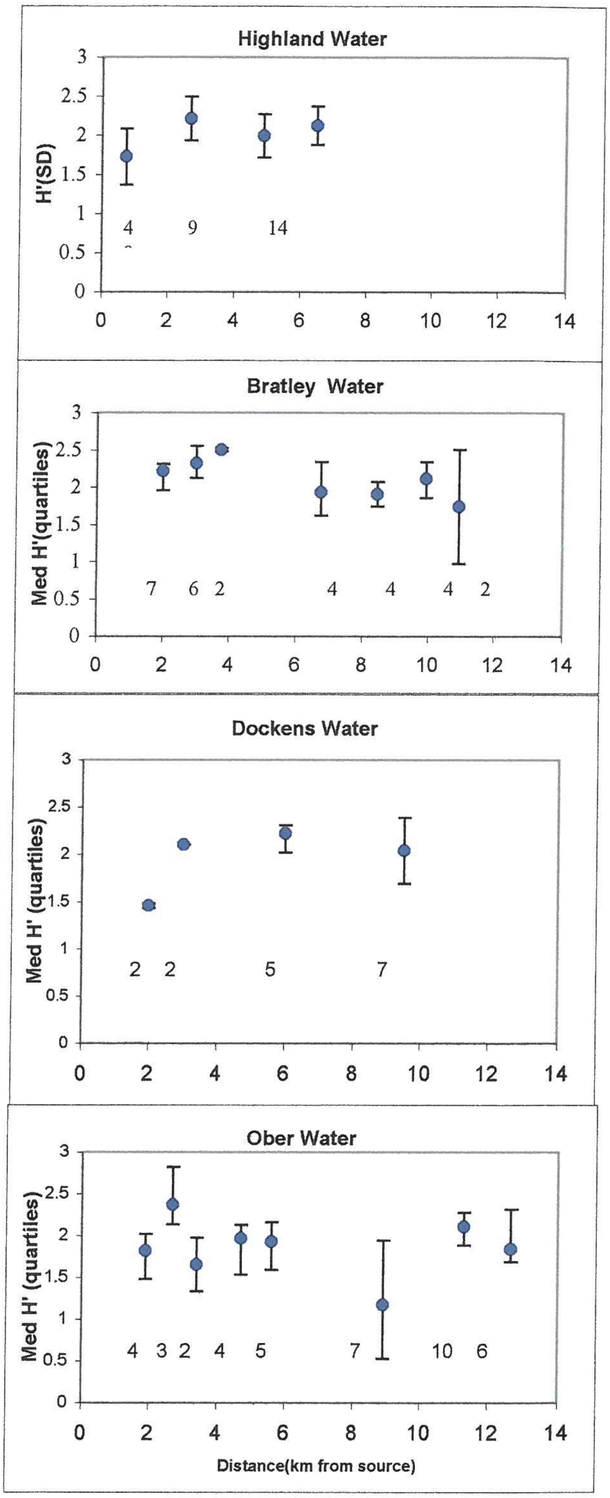


**Fig.4.3** **Invertebrate diversity expressed as Shannon-Wiener diversity indices (H') for locations along four New Forest streams**
*(Mean+/- 1SD, or median and quartiles as shown)*
*(numbers on plots are numbers of samples analysed for that site)*

families or numbers of animals collected in samples from the four streams (Table 4. 4). A comparison of the weighted biological scores used for water quality monitoring (Fig 4. 4) for the data from each stream showed that there were no significant differences in BMWP scores between the streams. Both BMWP and ASPT scores indicate high water quality for small circumneutral streams. The higher ASPT scores in the wooded streams indicate a higher proportion of "clean" water species present, mainly Plecoptera (Langford, *unpublished data*).

### 4. 2. 4. Previous studies of wood debris in the Highland Water streams

The Highland Water, Bratley Water and Bagshot Gutter were all in the higher ranges of CWD loading and debris dam concentrations relative to other streams in the New Forest in the 1980's and early 1990's (Gregory *et al.*, 1993) (see Chapter 2). For example the Highland Water contained an average of approximately 5 dams per 500m reach, the Bratley Water 5.4 dams and the Bagshot Gutter 11.8 compared with the 2.5 in the Ober Water. The net loads in kg $m^{-2}$ were 0.59, 0.43, 2.49 and 0.11 respectively. Quantitative studies were not carried out for this present study on fish populations but qualitative assessment and observations indicate that the streams are still similarly ranked as far as CWD loadings are concerned. Dockens Water was not studied in the work by Gregory and his co-authors but qualitative assessment for this work again indicates that the stream has a similar ranking to the Ober Water. In fact only one of the stream reaches sampled in both Dockens Water and the Ober Water contained an obvious CWD accumulation. This was caused by a living tree which had begun to slip into the stream and was leaning at such an angle that branches were immersed and had entangled drifting debris.

The distribution of CWD dams studied by Gregory and his co-authors varied significantly with distance downstream of the source of the streams (Gregory *et al.*, 1993). Peak frequencies occurred at about 2.5km from the headwaters and the decrease was rapid downstream until dams were relatively scarce in the lower reaches. Partial dams (see Chapter 3, Fig. 3.1) were most numerous at about 5km downstream of the source of the Highland Water. The reasons for the differences between the distributions of partial and full-width dams were the increasing

**Table 4.4 Comparisons of invertebrate data from four New Forest streams used for fish studies.**

*(Non-parametric test is Kruskal-Wallis test).*

a) Numbers of taxa per collection

| | N | Mean | Std Dev |
|---|---|---|---|
| **Highland** | 35 | 13.171 | 3.785 |
| **Bratley** | 28 | 13.321 | 3.916 |
| **Ober** | 43 | 14.186 | 5.086 |
| **Dockens** | 16 | 14.375 | 3.222 |

*F = 0.572, p = 0.634 (NS) (one-way ANOVA)*

b) Numbers of animals per collection

| | N | Median | 0.25 | 0.75 |
|---|---|---|---|---|
| **Highland** | 35 | 75 | 46 | 149 |
| **Bratley** | 28 | 53 | 36 | 118 |
| **Ober** | 43 | 95 | 35 | 273 |
| **Dockens** | 16 | 95 | 51 | 222 |

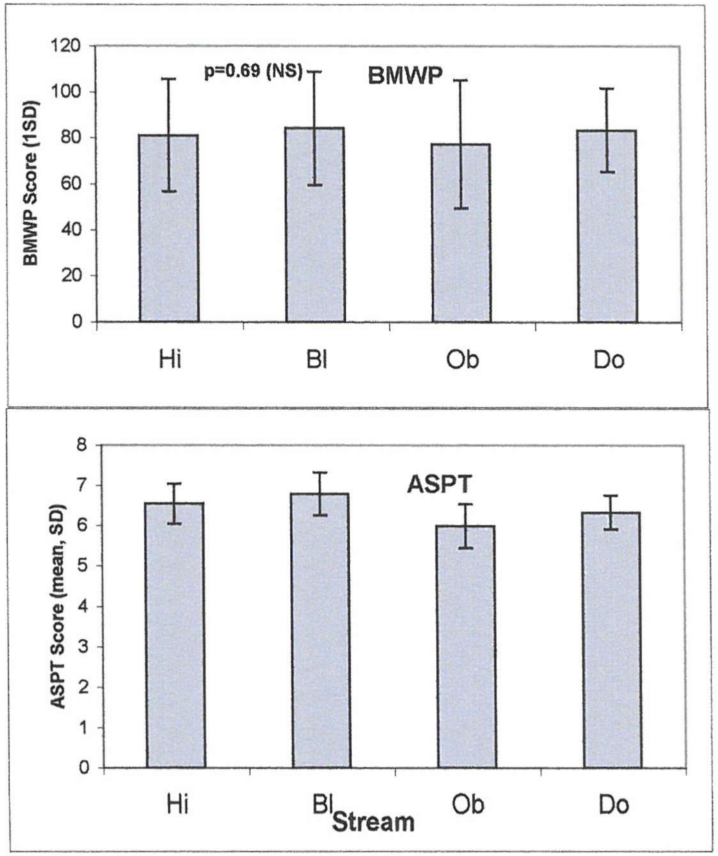
*(KW) H = 5.300 (df = 3) p =0.151 (NS)*

c) Diversity (as Shannon-Wiener H')

| | N | Median | 0.25 | 0.75 |
|---|---|---|---|---|
| **Highland** | 35 | 2.063 | 1.894 | 2.283 |
| **Bratley** | 28 | 2.209 | 1.905 | 2.41 |
| **Ober** | 43 | 1.949 | 1.648 | 2.191 |
| **Dockens** | 16 | 2.104 | 1.583 | 2.302 |

*(KW) H = 7.303 ,df = 3, p =0.063 (NS)*

N = number of samples



***(ASPT, (KW) p<0.001, Hi>Ob, Bl>Ob, Bl>Do)***

**Fig.4.4 Median BMWP and ASPT scores for all sites (summed) on four New Forest streams.** ***(Error bars are quartiles)***

channel width and the consequent increasing inability of wood pieces to span the width of the channel.

There was also a clear relationship between net CWD loadings and riparian land use. The highest loadings occurred in reaches which ran through areas dominated by deciduous forest. Lower loadings were associated with coniferous forest and the lowest loadings were associated with lawn, improved pasture or scrubland. The Ober Water fell into this category in the 1990-91 surveys and Dockens Water has a similar open catchment (Table 4. 2) but with more improved pasture. The Highland Water, Bratley Water and Bagshot Gutter on the other hand, run through substantial areas of deciduous woodland and coniferous plantation which is the primary reason for their higher CWD loadings (Gregory *et al.*, 1993).

## 4. 3. PHYSICAL CHARACTERISTICS AND HABITAT CLASSIFICATION

### 4. 3. 1. Dimensions of habitat sampling units

Riffles, pools and CWD habitats were selected for sampling *a priori* by visual inspection. Using selected geomorphologically defined habitat units as the basis for fish sampling rather than the more typical defined length or area (say 50 or 100m) obviously produced sampling units of different dimensions (Bohlin *et al.*, 1989). The larger habitat units in terms of channel length or wetted area were predicted to show higher physical diversity simply based on the increasing probability of the occurrence of an increasing number of features. The division of the slow-water category into pools and CWD habitat units also created an *a priori* category which does not fit strictly into previous classifications (Hawkins *et al.*, 1993) though these habitat units may be regarded as a further sub-division of "debris pools" (Hawkins *et al.*, 1993). Isolation of the CWD accumulation in many reaches effectively divided some pools into three parts, i.e. upstream and downstream of the CWD and the CWD habitat unit itself (see Chapter 3, Fig. 3.1).

Mean lengths, median widths and depth characteristics of the three habitat unit types are shown in Table 4.5. *A priori* comparisons using the Kruskal-Wallis one way ANOVA on ranks showed that there were significant differences in median dimensions between the selected habitat sampling units. Pairwise tests (Dunn's) showed that the median lengths of the isolated CWD sampling units (5m) were significantly less than those of both pools and riffles ($p>0.05$). Riffles and pools had median lengths about 10 and 11m respectively and were not significantly different ($p>0.05$). Median wetted widths of riffles (2.3m) were significantly smaller ($p<0.05$) than those of both pools (3.1m) and CWD habitats (3.1m) but pool and CWD widths were not significantly different ($p>0.05$). The median depth of CWD habitats (0.44m) was some 7 times that of riffles (0.06m) and twice that of pools (0.22m) and the differences were significant ($p<0.05$). The mean width/depth ratio of riffles was more than 2.5 times that of pools and 4 times that of CWD habitats (Table 4.5). The width/depth ratio for pools was about twice that of the CWD habitats. Differences between the pairs were all significant ($p<0.05$). On the basis of comparing medians or mean dimensions therefore, there appear to be clear and significant differences between the selected habitat units with CWD habitats being shorter and deeper than pools.

The differences are, however, less clear when the dimensions are compared using frequency distributions of individual habitat units. Fig. 4.5 shows the distributions for median wetted widths, depths and depth ranges. Median wetted widths were normally distributed and modal values were similar in all three sampling units at 2.5m. The modal value for median depths was 0.15m for riffles and 0.3m for pools with the frequency distributions showing negatively skewed distributions. The median depth distribution in CWD sampling units was, in contrast, distinctly bimodal with modes at 0.4 and 0.8m. The bimodality was probably caused by differences between units with partial dams and those with full dams, though the age and stability of the dams could also be a factor. For example, it is likely that more established dams have been affected by scour more than newer dams (see Section 4.4). Depth ranges were also generally lower in the riffles, with a mode at 0.1m. Here both pools and CWD habitats showed a bi-modal frequency range of depth variation with modal frequencies of around 0.4 and 0.5m respectively. Clearly, however, there was considerable overlap in

**Table 4. 5. Comparisons of median lengths, widths, depths and width/depth ratios for riffles, pools and CWD habitats in Highland Water streams.**

| Habitat | n | Median | Quartiles<br>25 % | <br>75% |
|---|---|---|---|---|
| **Lengths (m)** | | | | |
| Riffles | 43 | 10 | 9.5 | 12.2 |
| Pools | 80 | 11 | 7.6 | 5.9 |
| CWD habitats | 39 | 5 | 4.2 | 7.3 |

*Riffles/pools (NS), Riffles>CWD (p<0.05), Pools>CWD (p<0.05)*

| | | | | |
|---|---|---|---|---|
| **Widths (m)** | | | | |
| Riffles | 43 | 2.3 | 2.0 | 2.7 |
| Pools | 80 | 3.1 | 2.3 | 4.1 |
| CWD habitats | 39 | 3.1 | 2.5 | 3.9 |

*Riffles < Pools (p<0.05), Riffles < CWD (p<0.05) Pools/CWD (NS) (p>0.05)*

| | | | | |
|---|---|---|---|---|
| **Depths (m)** | | | | |
| Riffles | 43 | 0.06 | 0.04 | 0.09 |
| Pools | 80 | 0.22 | 0.15 | 0.32 |
| CWD habitats | 39 | 0.44 | 0.30 | 0.71 |

*Riffles<pools<CWD (p<0.05)*

| | | | | |
|---|---|---|---|---|
| **Width/depth ratios** | | | | |
| Riffles | 43 | 38.54 | 24.02 | 51.73 |
| Pools | 80 | 14.49 | 11.02 | 18.91 |
| CWD habitats | 39 | 6.90 | 5.67 | 9.53 |

**Riffles>Pools>CWD (p<0.05)**

***a priori*** **Kruskal Wallis, *a posteriori,* Dunn's multiple pairwise test.**

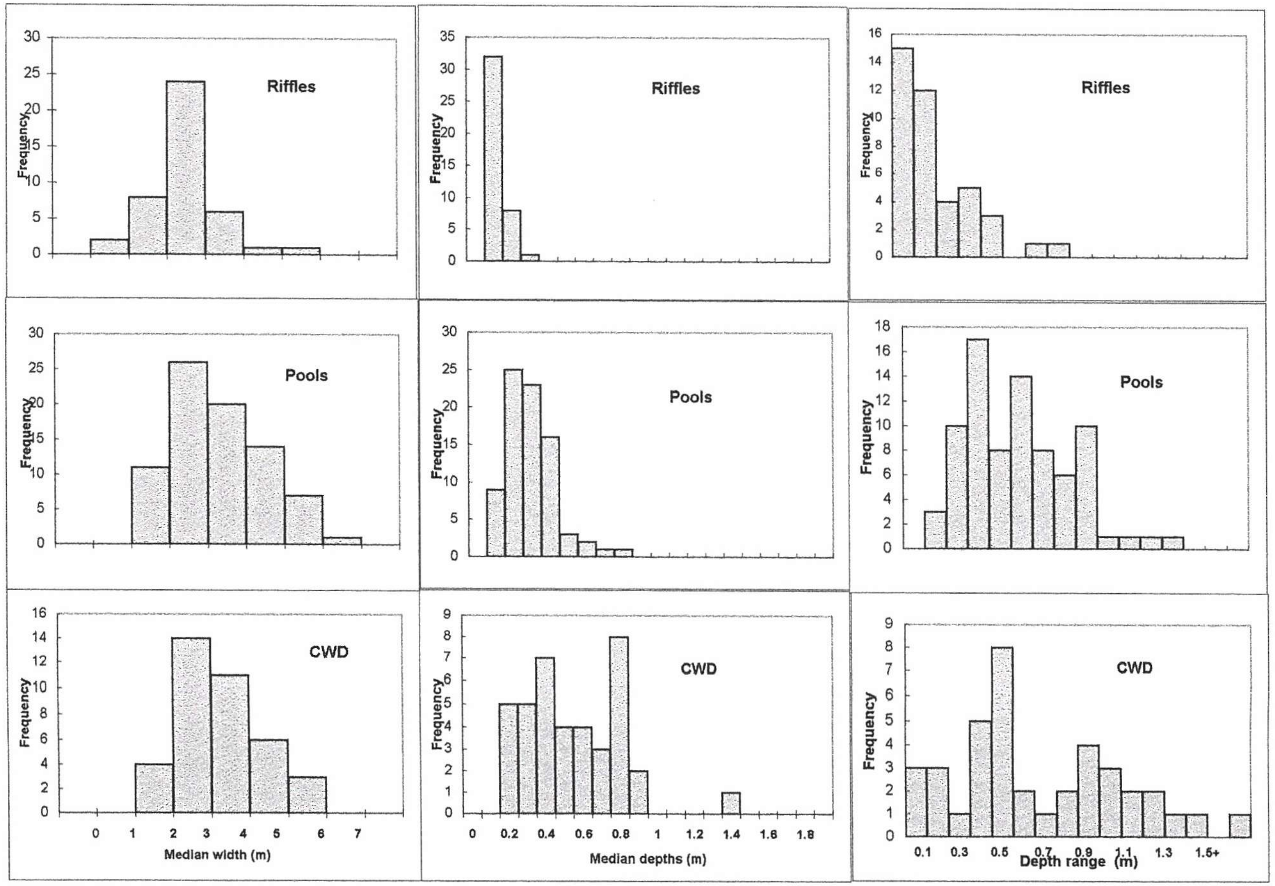


**Fig.4.5** **Frequency distributions for stream channel dimensions from habitat units in Highland Water streams**

*(a) Mean widths (m)* *(b) median depths (m)* *c. Depth ranges (m)*

*(these are selected for characterisation, further data are included in text and tables)* *(note different scales on y-axes)*

the ranges of all the channel dimensions and the indication was of a continuum of dimensions among the habitat units rather than clearly definable categories.

### 4. 3. 2. Substrates and channel structure

The overall substrate composition of riffles, pools and CWD habitat units is summarised in Table 4. 6 using the ranges of percentage occurrence of the different substrates and structures from point-transect measurements in every individual unit. Data from the habitats in open streams is included. The two most common categories, were gravels and CWD which showed inverse trends in frequency of occurrence in all three habitat types (Fig. 4. 6). This phenomenon was also partly a function of the methodology in that the point-transect measurements recorded only the dominant substrate at each point which was usually the first substrate contacted. Thus, where CWD was relatively dense it was the only substrate recorded though it may have been located on or just above another substrate, usually gravel. CWD and gravel were therefore mutually exclusive where CWD was abundant. Even so the method was sufficiently robust to show the physical differences between the habitat units when used together with dimensional data.

The basic substrate in most riffles, pools and CWD habitat units was gravel varying from small pea-sized (ca 0.5-1.5cm diameter) gravel to small cobbles (ca 10-12cm diameter) (Langford, 1996). CWD was also present in all three types of habitat unit but was generally more abundant in pools than riffles and, as expected from the visual selection of sampling units, most abundant in the CWD habitat units. The areal cover of CWD in the CWD habitat units varied from as little as 13% to over 90% as a result of the *a priori* selection which together with the ranges of percentage cover of CWD in riffles and pools gave a wide variation against which to test fish abundances. Some pools, originally selected for their apparent absence of a definable CWD accumulation, contained up to 43% CWD by areal coverage and one riffle site contained 34% CWD by areal cover. This was mostly in the margins downstream of a dismantled debris dam. Much of the CWD in pools and riffles was in the form of separate or small groups of wood pieces not aggregated as dams.

**Table 4. 6 . Ranges of percentage occurrences of identified substrates and physical habitat features as assessed from point-transect measurements in Highland Water and associated streams**

| | Highland Water, Habitat Unit | | | |
|---|---|---|---|---|
| **Substrate/Feature** | **Riffles** | **Pools** | **CWD** | **Open** |
| Gravel | 27-100 | 0-85 | 0-58 | 30-67 |
| Sand | 0-13 | 0-10 | 0-23 | 0-13 |
| Silt | 0-25 | 0-49 | 0-23 | 2-15 |
| Twigs | 0-7 | 0-26 | 0-13 | 0-3 |
| Leaves | 0-12 | 0-28 | 0-10 | 0-8 |
| Clay | 0-25 | 0-60 | 0-23 | 0-10 |
| Undercut banks | 0-33 | 0-24 | 0-27 | 0-15 |
| Instream root wads | 0-27 | 0-21 | 0-33 | 0-5 |
| Instream weed | 0-7 | 0-22 | 0 | 0-70 |
| Wood ((CWD) | 0-34 | 0-43 | 13-92 | 2-9 |

*(NB. "Open" refers to data from the Ober Water and Dockens Water habitat units with reduced woodland in the catchments and low CWD loadings)see text for descriptions and discussion.*

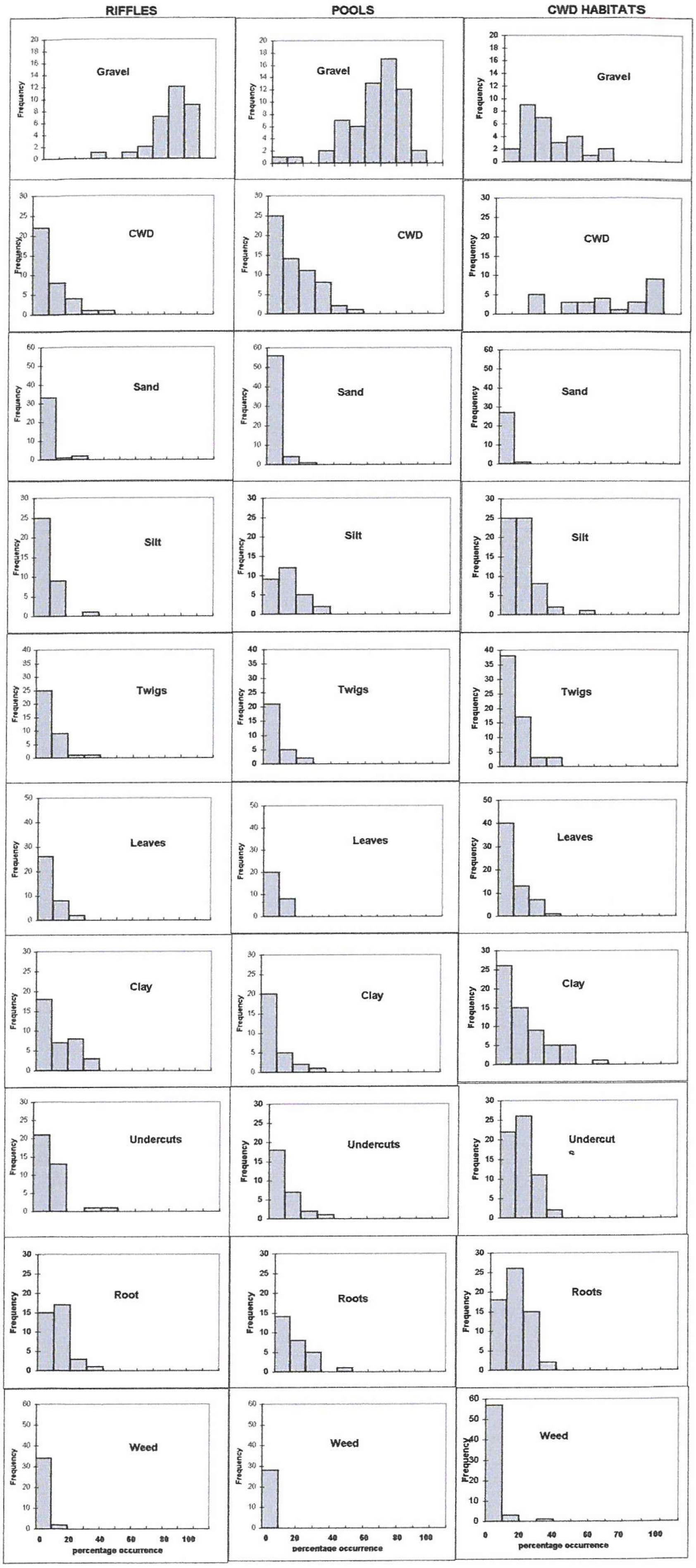


Fig.4.6 **Frequencies of occurrence of substrate types and structural features in point-transect surveys of habitat-units in Highland Water streams.** *(y-axes are scaled to the maximum for the variable)*
*x-axis denotes occurrence in each habitat unit, y-axis denotes proprtion of habitat units with that %*

Silt was most abundant in pools and CWD habitats usually in margins and back eddies. Cover features other than CWD, notably undercut banks and tree root wads were similarly distributed and of similar frequency in all three habitats. The size of the undercuts or roots is not indicated here but was largest in pools with little CWD. Instream weed was very scarce in the Highland Water streams as might be expected because of the dense riparian woodland. Most reaches contained none and the maximum for any one sampling unit was less than 30% cover. This was in direct contrast to the reaches sampled in some "open" streams where instream vegetation covered up to 60% of the substrata (see 4.3.7).

Fig. 4.6 shows clearly that there was considerable overlap in the frequency distributions of the various physical features and substrates which again indicates that there was a physical continuum rather than discrete habitat units. The bimodal distribution of CWD was a result of the initial site identification (see Chapter 3), where habitat units with varying amounts of CWD were selected.

Estimates of the overall volumes of submerged CWD accumulations (wood plus interstices) were made using point-transect data and median depth measurements (see Chapter 3, Section 3. 3). The estimated median and quartile CWD matrix volumes for riffles, pools and CWD habitat units are shown in Table 4. 7. There were significant differences between the matrix volumes as might be expected though there were clear overlaps between the amounts of CWD recorded in pools and CWD habitats. Spearman rank correlation showed that the many of the physical variables showed significant correlation (Table 4. 8). There was significant co-linearity between the stream "size" variables so one variable was selected to represent depth. From the various measures of depth, maximum depth was selected as the depth parameter for use in the analyses. The measure was relevant, convenient and efficient in that multiple measurements were not needed and calculations not necessary. It was significantly positively correlated with mean depth (Coeff. = 0.916, $p < 0.001$), standard deviation of depth (Coeff. = 0.913, $p < 0.001$), range of depth (Coeff. = 0.921, $p < 0.001$) and median depth (Coeff. = 0.868, $p < 0.001$). Maximum depth was also significantly negatively correlated with mean width/median depth ratio (Coeff. = -0.588, $p < 0.001$).

**Table 4.7 Estimated volumes ($m^3$) of CWD accumulations in each habitat type in Highland water streams used for fish studies 1996-98. (n = number of replicates)**

| Habitat | n | Median | quartiles |
|---|---|---|---|
| Riffles | 43 | <0.001 | 0 - 0.15 |
| Pools | 80 | 0.87 | 0.32 - 3.10 |
| CWD habitats | 39 | 4.38 | 1.78 - 9.94 |

*Riffles<Pools< CWD (p<0.05)*
*All statistical analysis using Kruskal-Wallis one way ANOVA and Dunns multiple pairwise tests*

Table 4.8 **Spearman rank correlation coefficients for selected physical variables in Highland Water streams. Data are from point-contact measurements, expressed as percentage occurrences.**

| | gravel | sand | silt | twigs | leaves | clay | under | roots | CWD | distance |
|---|---|---|---|---|---|---|---|---|---|---|
| depth | ***-0.616*** | -0.0317 | ***0.4*** | 0.0763 | 0.0277 | ***-0.313*** | 0.0505 | 0.0937 | ***0.623*** | ***0.552*** |
| gravel | | -0.0243 | ***-0.38*** | **-0.279** | -0.0941 | 0.0647 | -0.0832 | 0.0991 | ***-0.831*** | **-0.291** |
| sand | | | -0.104 | 0.0972 | 0.0259 | -0.0345 | 0.105 | **-0.246** | 0.0679 | ***0.304*** |
| silt | | | | 0.107 | 0.0822 | -0.0228 | -0.0347 | -0.113 | **0.225** | 0.222 |
| twigs | | | | | 0.0517 | 0.164 | 0.0732 | -0.199 | 0.176 | **0.217** |
| leaves | | | | | | -0.151 | -0.214 | -0.144 | 0.0233 | 0.102 |
| clay | | | | | | -0.0326 | **-0.239** | **-0.269** | **-0.291** | **-0.291** |
| under | | | | | | | 0.0449 | -0.0099 | -0.00705 | -0.007 |
| roots | | | | | | | | -0.0922 | -0.129 | -0.129 |
| CWD | | | | | | | | | ***0.446*** | ***0.446*** |

***Significance is shown as follows:- bold italics, $p<0.001$, bold only, $p<0.05$, ordinary type, NS ($p>0.05$)***

Notable significant correlations were between distance downstream of the source, depth, CWD and silt. Negative correlations were significant between distance and gravel, gravel and silt, depth and gravel and CWD and gravel. Abundance of CWD was also positively correlated with wetted width (Coeff. = 0.225, $P < 0.05$) and negatively correlated with length of the habitat unit (Coeff. = 0.214, $p < 0.05$).

### 4. 3. 3. Physical habitat diversity

Physical diversity as indicated by the Shannon-Wiener index (H') (e.g. Gorman & Karr, 1978) and the Equitability index (J') were calculated from the point-transect data for all the substrate and feature variables (Chapter 3, Table 3.3) for each individual habitat sampling unit. Values for each *a priori* selected habitat type were compared. Shannon-Wiener indices were normally distributed but Equitability indices were not. Differences between means and medians were therefore tested for significance by one-way ANOVA (H') and Kruskal-Wallis one way ANOVA on ranks (J') respectively (Table 4. 9). Physical habitat diversity indices were, on average, significantly higher in pools than in riffles but not greater in pools than in CWD. The difference between CWD and riffles was also significant ($p=0.003$). Riffles as habitat units were therefore on average more homogeneous than the deeper water habitats but there was no significant difference between habitat diversity in pools and CWD habitat units.

The factors which most influence the diversity were initially explored using correlation analysis. Table 4.10 shows Spearman-rank correlation coefficients for the major physical variables in relation to habitat diversity. Distance downstream of the source was added to the list because Gregory *et al.* (1993) showed that the concentration of debris dams in the channel could be related to this factor. There are significant positive correlations between diversity (H'), maximum depth, CWD, and silt. The highest significance is for the negative correlation between H' and the % gravel. Equitability (J') was correlated most strongly with % CWD and % gravel, the former positively, the latter negatively. More surprisingly, despite the strong correlation between distance downstream of the source, depth and % wood, there was no significant correlation ($p>0.05$) between distance

**Table 4.9 Comparisons of Shannon-Wiener diversity indices and equitability indices for habitat unit types in New Forest streams, based on the frequency of occurrence of identified physical variables (see Chapter 3, Table 3.3)**

| Habitat Unit | n | Mean | SD (+/-) |
|---|---|---|---|
| ***Shannon-Wiener*** | | | |
| **Riffles** | 36 | 0.73 | 0.33 |
| **Pools** | 61 | 1.17 | 0.29 |
| **CWD habitats** | 28 | 1.00 | 0.42 |

***F = 19.667, df = 2, p <0.001******
***Riffles < Pools (p<0.001)***, Riffles < CWD (p = 0.003)***
***Pools/CWD (NS) p= 0.073) (Tukey test)***

| ***Equitability*** | **n** | **Median** | **25%** | **75%** |
|---|---|---|---|---|
| **Riffles** | 36 | 0.30 | 0.24 | 0.40 |
| **Pools** | 61 | 0.22 | 0.20 | 0.38 |
| **CWD** | 28 | 0.47 | 0.26 | 0.61 |

***KW. H = 28.91, df = 2, (p<0.01)*****

***Dunn's test Pools < Riffles (p < 0.05), pools< CWD (p < 0.05), Riffles/CWD (NS) (p > 0.05)***

**Table 4.10 Spearman rank coefficients for the correlation of diversity indices and selected physical variables.**

*(Shannon-Wiener H'    Equitability, J')*

| Variable | H' | J' |
|---|---|---|
| Depth | **0.284** | 0.053 |
| Gravel | ***-0.459*** | ***-0.409*** |
| Silt | ***0.38*** | **0.276** |
| Wood | **0.322** | **0.326** |
| Distance DS | 0.143 | -0.26 |

***bold italics ,p<0.001, bold, p<0.05***

downstream and habitat diversity or equitability. Thus the physical diversity of the sampling units is clearly related to combinations of physical factors of which stream depth, amounts of gravel and CWD are major influences.

### 4. 3. 4. Habitat structure and organisation

To explore and explain the within-reach variations further and to provide a physical background model on which to base the analyses of fish populations the relationships between the physical variables and habitat diversity were analysed using ordination to try to define groups of sites with similar physical composition. From the comparisons of the mean or median values for dimensions and the occurrence of physical variables it was expected that the three selected channel unit types would separate out clearly into "habitat" types based on the composition of their physical components and on the clear differences in dimensions (e.g. Martin-Smith, 1998).

Fig. 4.7 shows the DCA plot of the primary physical variables and substrates in relation to the composition of the habitat in the habitat units. There are two identifiable groupings of physical variables on Axis I the first comprising clay and gravel and the second based on CWD a lesser extent, maximum depth, distance downstream of the source, twigs and silt. There is thus a physical gradient from shallow, gravel riffles to deeper water with increasing amounts of CWD and increasing distance downstream of the source. Organic and finer sediments (leaves, twigs and silt) are grouped together on Axis I but separated from roots and sand on Axis II. Non-wood cover variables (roots and undercuts) are grouped on the lower part of Axis II in the middle range of Axis I. Clay, although a fine sediment is in the shallow range because the category refers mainly to the hard clay underlying the substrates in the stream channel where gravel deposits form the superficial substrate. This occurs mainly where channelization and deepening has removed the original substrates. The indication is that the stream channel is structured on the basis of three main groups of physical variables with gravel and CWD as the main opposing variables.

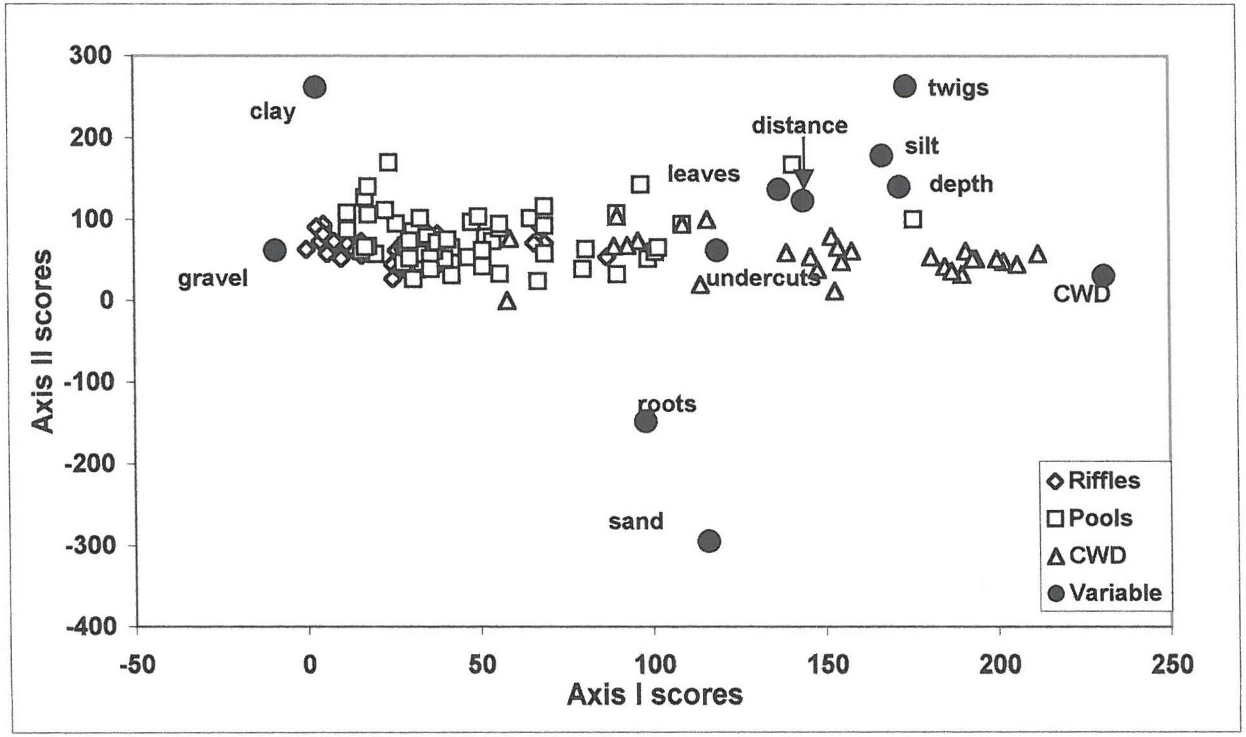


Fig.4.7 **DECORANA plot of physical variables in relation to habitat units based on the occurrence of variables in point-contact samples, maximum depth and distance from source.** (note locations of riffles, pools and CWD habitats in relation to the variables)

The *a priori* habitat units are not, however, clearly segregated into three types and are aligned along a gradient from the shallow, erosional habitats to the deeper, depositional habitats (Prenda *et al.*, 1997) with a high abundance of CWD. The general trend is predictable in that *a priori* riffles are grouped more toward the gravel and shallow water and CWD habitats toward the deeper water. Pools are distributed along the horizontal axis but are not clearly separated as a "physical habitat" group from riffles. Although any division into groups is somewhat subjective (e.g. Kent & Coker, 1992), there appears to be some separation of habitat units into two groups with the boundary at a score of about 70 on Axis I. Very few riffles are distributed beyond 100 on Axis I.

Table 4.11 shows the relative importance of the physical variables in the four axes from the DCA using the computed scores for the listed variables. In Axis I, wood, maximum depth, and downstream distance are the dominant positive factors with gravel and clay as the dominant negative. In Axis II, clay, twigs and silt dominate with sand and roots forming the opposing end of the gradient. Axes I & II account for the 79% of the variance in the physical variables.

Habitat diversity (H') was regressed against the habitat unit scores for DCA Axis I using least squares regression (Fig. 4. 8). It is clear that structural diversity on the within-reach scale peaks at some mid-point on the gradient from erosional to depositional habitats confirming that pools with lower concentrations of CWD were more diverse in their physical structure than either riffles or extreme CWD habitats. Habitat units with the highest amounts of CWD showed similar habitat diversities to those of riffles. Thus both riffles and CWD habitat units showed similar homogeneity on the basis of these measurements.

### 4. 3. 5. Effects of dam removal

There is clearly considerable variation in the physical habitat at the within-reach scale in the Highland Water streams which can be related to the presence of CWD. However, CWD dams vary in their permanence in these streams. For example, Gregory, Gurnell & Petts (1994) noted that some 36% of dams were moved after heavy floods and that after removal for management purposes there

Table 4.11 Component scores for physical variables from DCA of Highland Water habitat units, showing the relative weightings of individual variables

| Variable | Axis I | Axis II | Axis III | Axis IV |
|---|---|---|---|---|
| CWD | 231 | 31 | 36 | 102 |
| gravel | -9 | 62 | 119 | 36 |
| sand | 116 | -295 | 290 | 439 |
| silt | 167 | 178 | 253 | -51 |
| twigs | 174 | 263 | 305 | 120 |
| leaves | 137 | 137 | 277 | -23 |
| clay | 3 | 262 | -89 | 22 |
| undercut | 119 | 62 | -54 | 181 |
| roots | 98 | -148 | 53 | -152 |
| depth | 172 | 140 | 30 | -30 |
| distance | 144 | 123 | 88 | 75 |

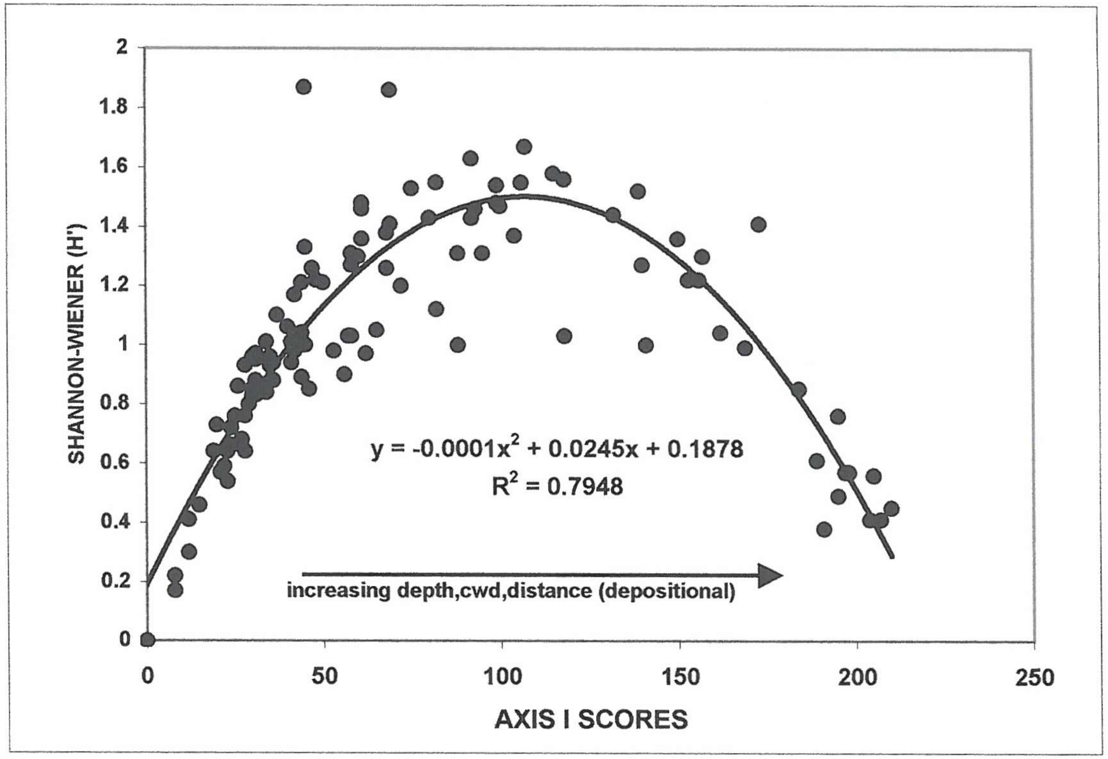


**Fig.4.8** Least squares regression of habitat diversity (H') and Axis I component scores from the DCA of physical variables. Highland Water streams. New Forest

was a rapid re-accumulation of CWD following a serious storm. From the evidence of the effects of the dams on physical habitat structure it can be predicted that the dynamics of CWD dams will affect both physical and biological components of the stream channel on the within-reach scale. There is thus a temporal factor to be considered.

During this study there was some natural removal and addition of CWD but not in the significant amounts as found by Gregory *et al.* (1993). Habitat units upstream and downstream of two major dams (Sites 16 and 22) (see Appendices I, II & III) were sampled on at least two occasions in the summer and autumn of 1997 as part of the routine research. Following high stream discharge in November and December 1997, the large dam at Site 16 on the Bratley Water (see Chapter 3, Fig.3.2) was breached and partially destroyed to relieve major flooding upstream. Figure 4. 9 shows in diagrammatic form the effects on the CWD matrix. Following the cutting and removal of the centre of the dam, there was a second period of high discharge which continued until late January 1998. In contrast, the dam at Site 22 on the Highland Water remained in place throughout the winter spates. This removal and retention provided the opportunity to make a preliminary comparison of the effects of the spates on both the physical structure of the habitat units and the fish. Both sites were therefore sampled in February 1998 which was the earliest opportunity after the spates. Four habitat units were sampled at both sites, and both comprised a downstream riffle, downstream pool, CWD habitat and upstream pool in the same sequence (Fig. 4.9). No allowance is made for differences in discharge in this preliminary analysis but as Fig.3.2 (see Chapter 3) shows samples were taken at similarly low discharges as far as possible. Stop nets were placed as near as possible to the same points on each sampling occasion.

The median depths of the downstream pool at Site 16 and 22 (Fig.4. 10) both increased following the spates but there may be some effect of higher winter flows. In the CWD habitat unit at Site 16 water depth decreased after the spates but at Site 22 the decrease was less marked. However at both sites there was a depth increase starting before the breach and before the spates. In the upstream

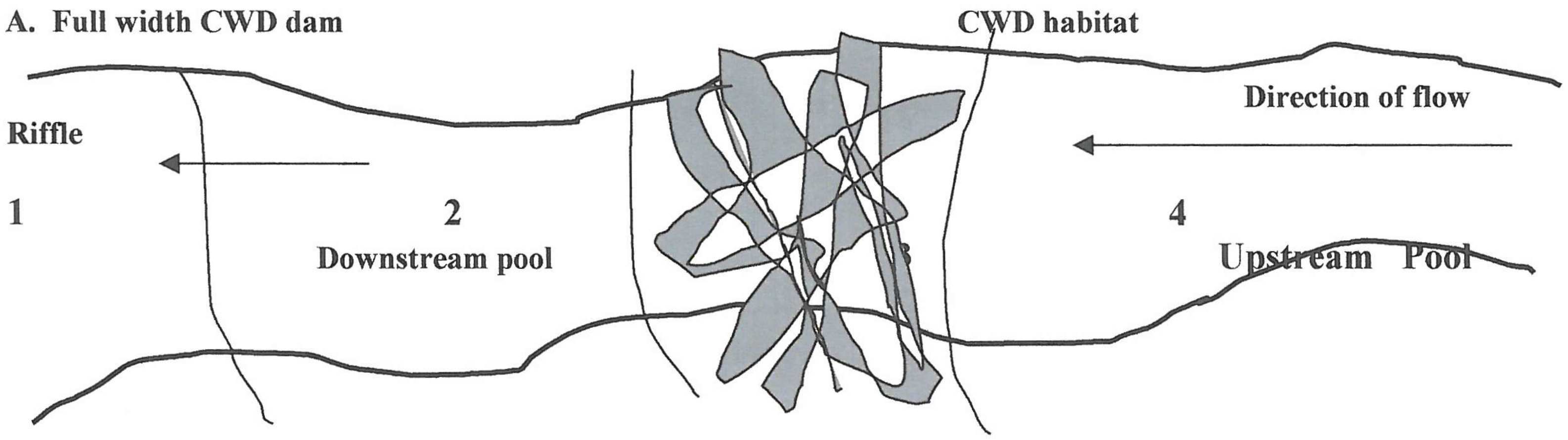
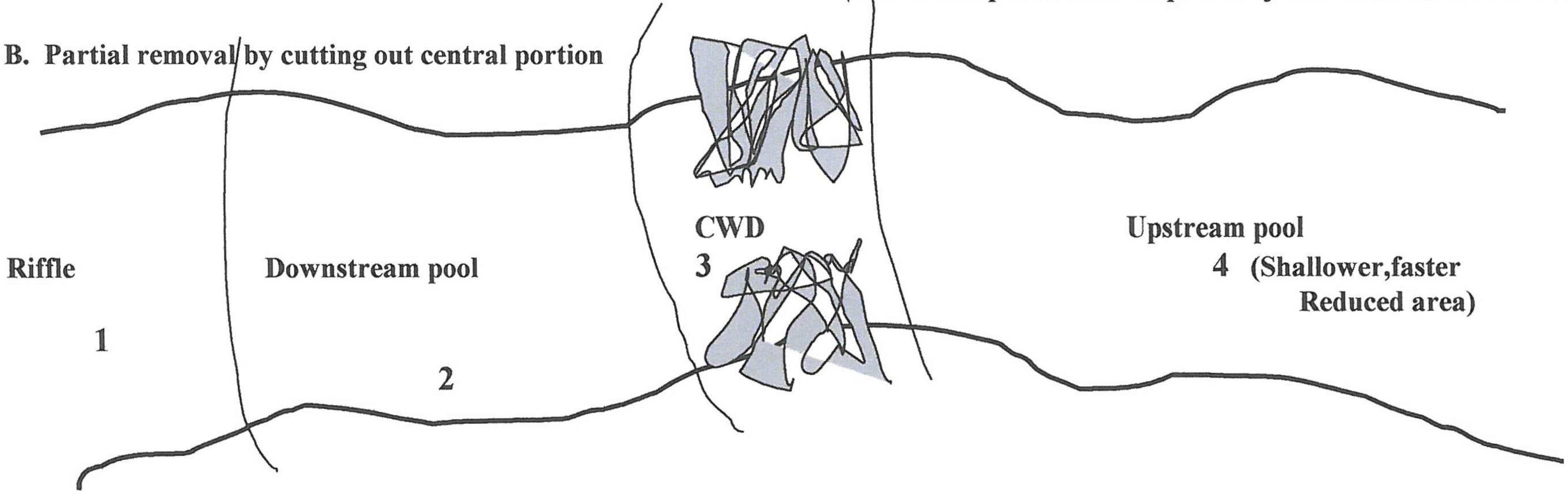
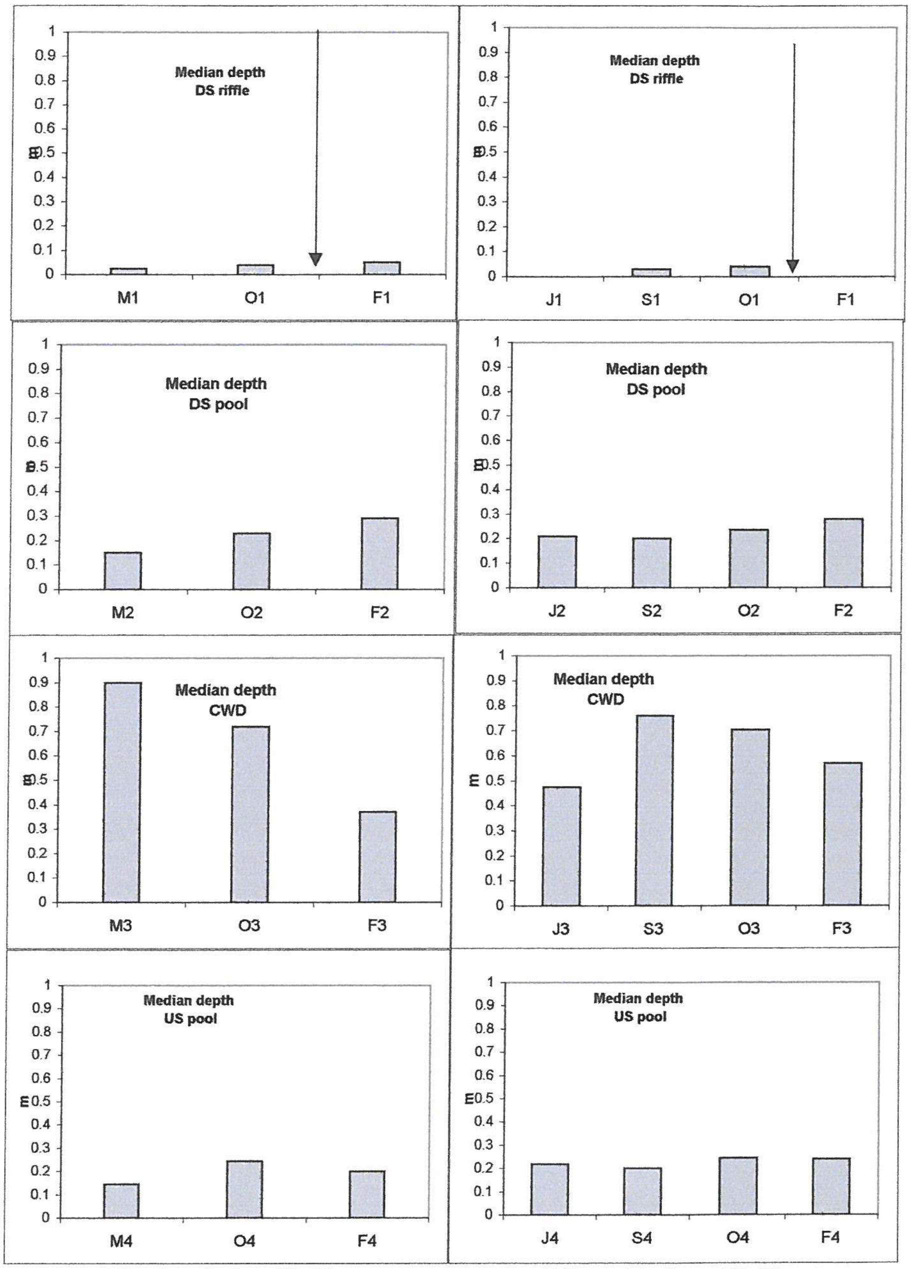


Figure 4. 9. Illustration of Site 16. Bratley Water before and after partial removal of a large CWD dam. 1997-98

*(Site 22 comprised same sequence of habitat units as Site 16,see text)*





**a) Dam breached Site 16** **b) Dam retained Site 22**

**Fig.4.10 Median depths in habitat units at two sites on Highland Water streams comparing effects of CWD dam removal and retention and winter spates.**

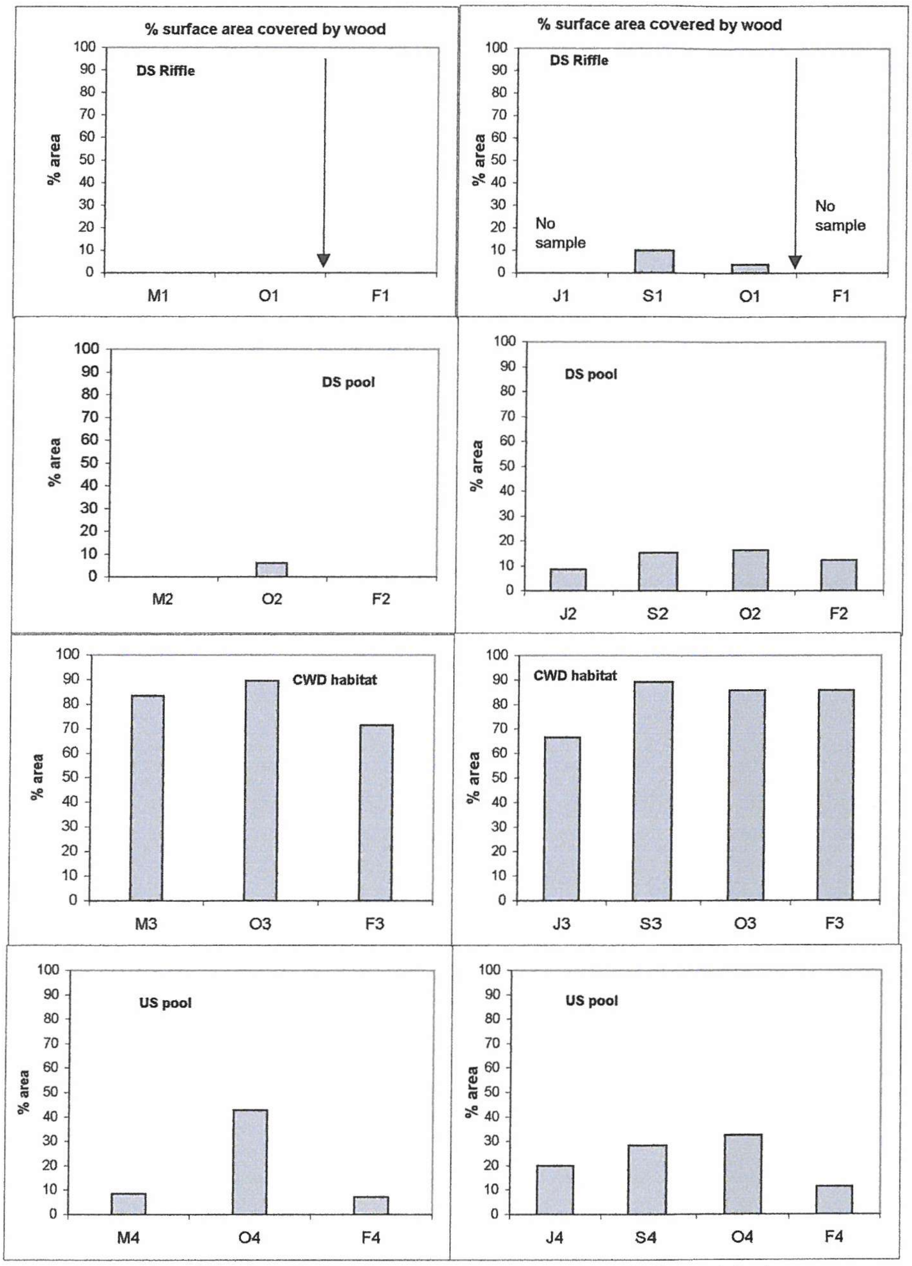
***Codes are as follows, letter denotes month, ie. M=May, S=September, J=July,O=October, F=February***
***Habitat units are as follows:- 1 =DS Riffle, 2= DS pool, 3= CDW habitat, 4= US pool.***
*Vertical arrows denote period when dam was removed at site 16*

pools median depth also decreased at Site 16 but no change was observed at Site 22. Median depth varied little in both upstream pools between October and February but slightly more at Site 16.

The breaching of the dam at Site 16 did not result in the removal of large amounts of CWD from the dam (Fig.4. 11) and there was no change at Site 22. At both sites, however, CWD was less in the downstream and upstream pools after the spates. The major changes in gravel area occurred in the upstream pool at Site 16 after dam removal (Fig.4.12) and in the CWD habitat at Site 22. The percentage area in the former rose from 4 -5% in May and October to almost 60% in February. The change in the upstream pool was much less at Site 22 namely from 40-50% in October to almost 60% after the spates. In the CWD habitat unit at Site 22 the area of silt fell from 17% to 4% between May and September but then remained unchanged through the winter spates.

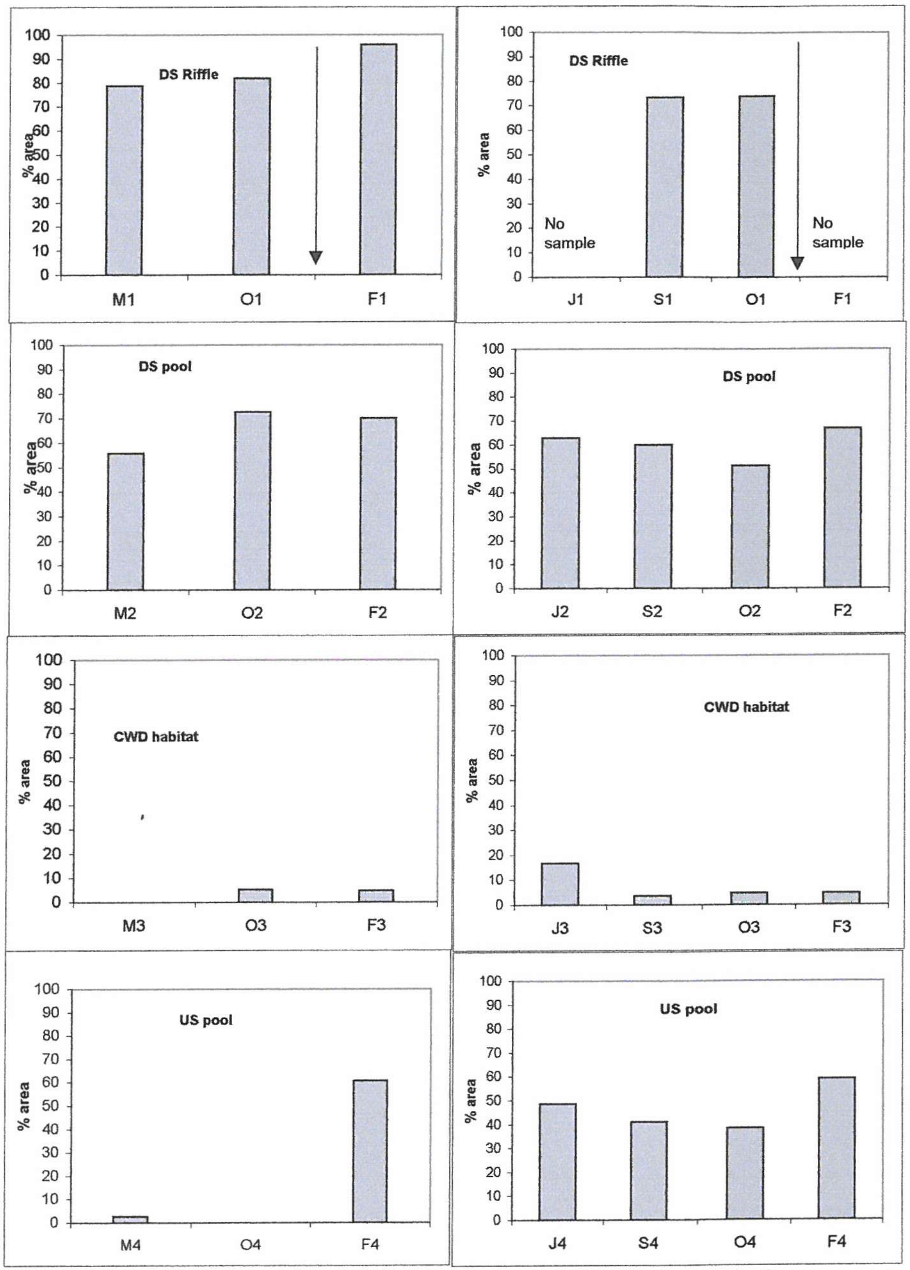
The changes in the area of silt showed the reverse trend at Site 16 (Fig.4.13). In all three deeper water habitat units silt decreased from May to October and further from October to February. No silt was recorded here in the February measurements. In contrast, the area of silt increased at Site 22 in the upstream pool after the winter spates though the amounts were generally small relative to the areas of the habitat units. In the CWD habitat there was virtually no change between October and February.

There were also changes in physical habitat diversity (H') (Fig. 4.14) of which the clearest was the increase within the CWD habitat unit after partial removal and spates at Site 16 and the decrease in the upstream pool coinciding with increased gravel and decreased depth. At Site 22 there were changes in habitat diversity through time but the most obvious was the increase in the upstream pool following the spates though it was still lower than in the September sampling. The overall effect of dam retention was mainly the retention of silt in the upstream pool, but the spates caused considerable silt removal downstream of the dams at both sites. At Site 16, the breaching of the dam mainly affected the depth and diversity of the upstream pool habitat.



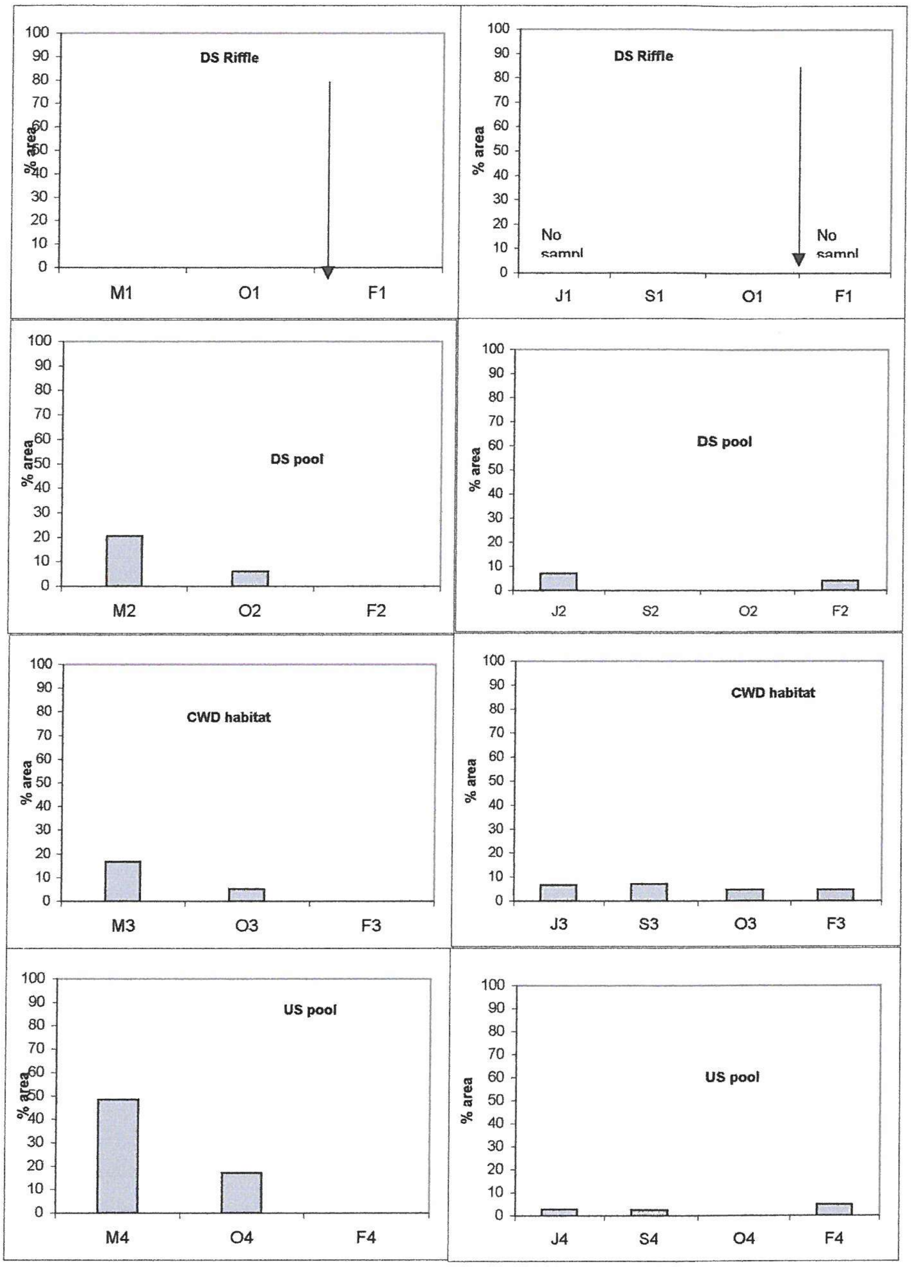
a) Dam breached b) Dam retained

Fig.4.11 Percentage cover of CWD in habitat units of two Highland Water streams in relation to dam removal and retention after a heavy winter spate.
(see Fig.4.10 for key to codes and explanation)



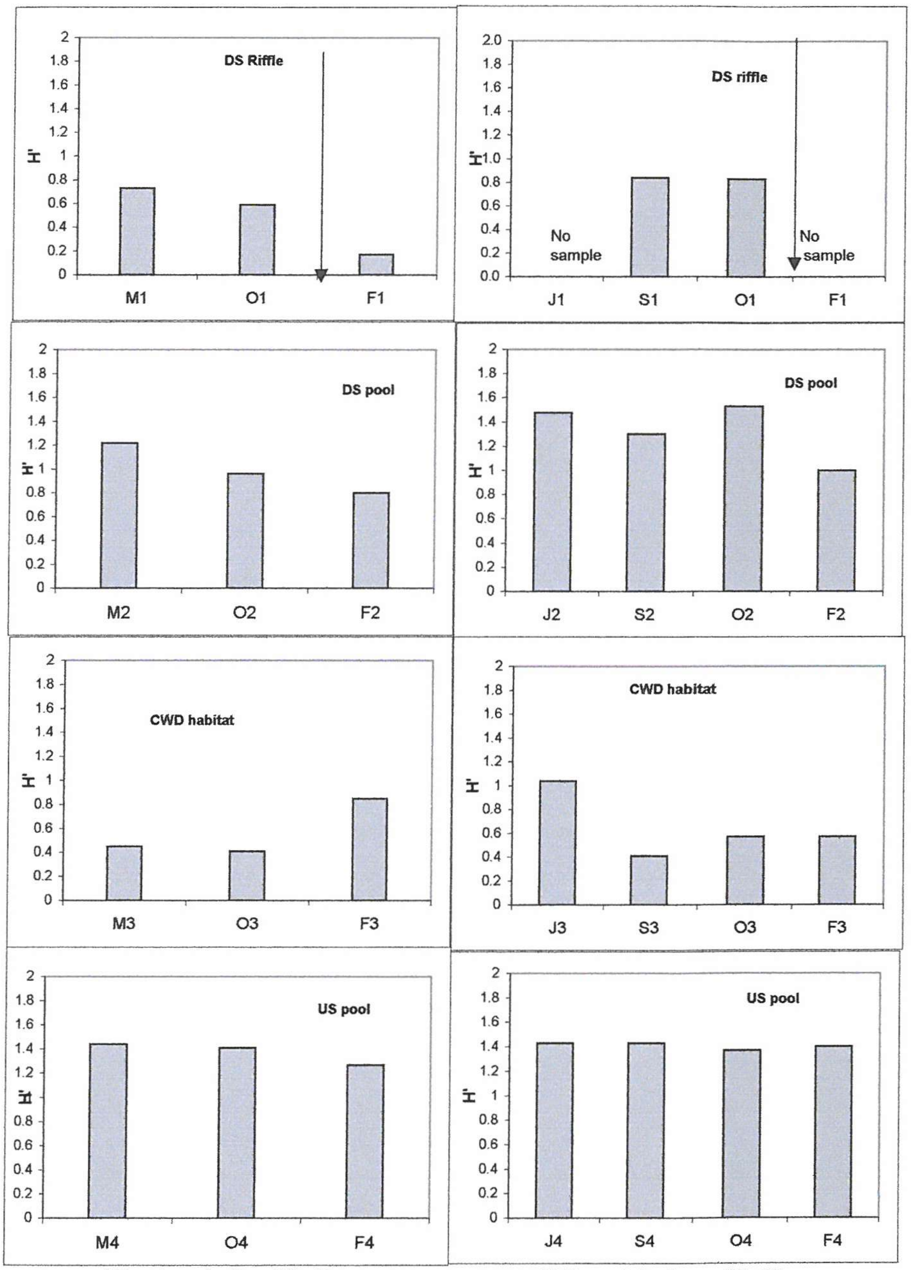
**Site 16** **a) Dam breached** **Site 22** **b) Dam retained**

Fig.4.12 **Percentage occurrence of gravel on the stream bed in habitat units at two sites on Highland Water streams in relation to CWD removal and retention and winter spates**
**(see Fig.4.10 for codes and explanation)**



**a) Dam breached Site 16** **b) Dam retained Site 22**

Fig.4.13 **Percentage area of stream bed covered by silt in habitat units of two Highland Water streams in relation to dam removal and retention after a heavy winter spate.**
**(see Fig.4.10for codes and explanation)**



**a) Dam breached** **Site 16** **b) Dam retained** **Site 22**

Fig.4.14 **Habitat diversity (as H') in habitat units of two Highland Water streams in relation to dam removal and retention after a heavy winter spate.**
**(see Fig.4.10 for codes amd explanation)**

### 4. 3. 6. Comparisons of within-reach and reach scale habitat diversity

The physical diversity of the individual channel units (riffles, pools and CWD) was compared with the overall structural diversity of the sampling reaches, i.e. the pooled length of all habitat units for that site. This was calculated from the pooled point-transect data from all the channel units for each reach. The median Shannon-Wiener (H') index for whole reaches was significantly higher (median H'= 1.36, quartiles 1.25-1.48) ($p<0.05$) than for riffles and CWD habitats (Table 4. 9) but not significantly greater than for pools. A DCA plot of the pooled data from all habitat unit measurements for each individual reach (Fig. 4.15) shows that the reaches were all very similar in overall habitat composition. Axis I and II explained 84% of the variance and the fours axes 93%. There was, however, a division of the habitat variables into two main groups based on the relative abundance of CWD, twigs, leaves, roots and silt and alternatively weed, sand, clay and undercut banks. The former group are mainly tree debris or structures, the latter not so. However, these were all variables in Highland Water streams, the catchments of which are almost entirely forested. The reason for the two groups is not therefore clear but requires further investigation.

Spearman rank correlation showed significant correlations between both habitat diversity (H') and Axis I (Coeff. = 0.527, $p=0.002$) and also Axis II (Coeff. = 0.586, $p=<0.001$) of the DCA (Fig.4.15). There was also significant correlation with the total volume of CWD matrices in the reach and maximum depth recorded (Coeff. = 0.566, $p<0.001$; Coeff. = 0.552, $p=0.014$) respectively. Axis I was heavily weighted by wood, silt and twigs, all characteristic of deeper water (Table 4.12). Thus although CWD matrices themselves showed low habitat diversity on the habitat unit scale (within-reach), taking the reach as a whole, physical diversity was closely related to the amount of CWD present (Gurnell *et al.*, 1995).

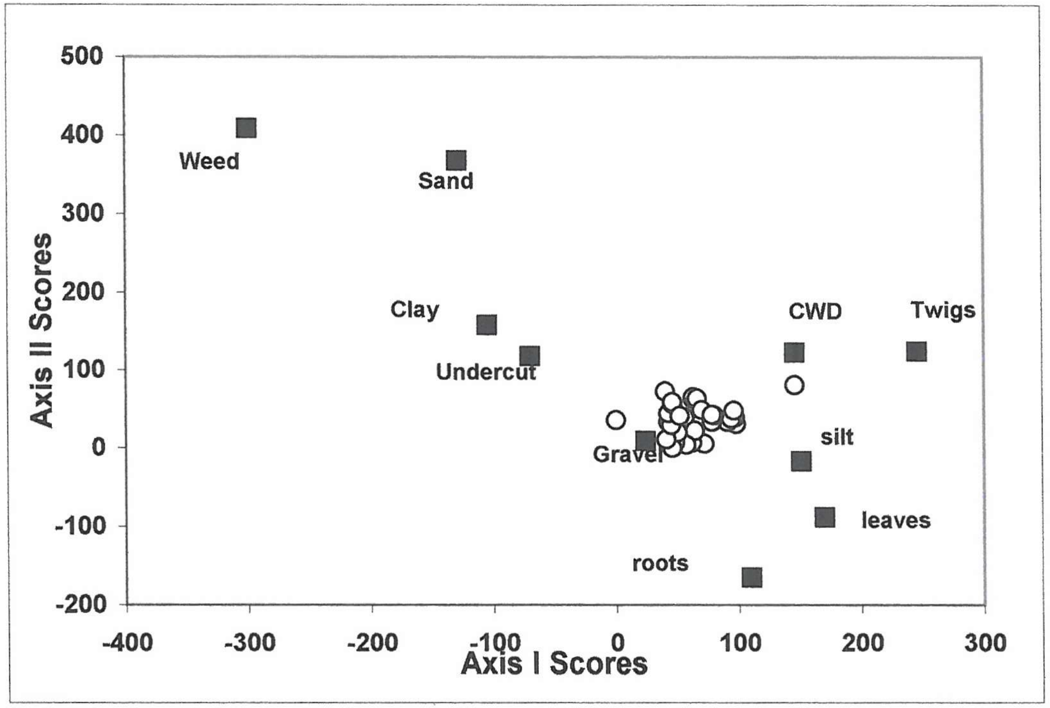


Fig. 4.15 **DECORANA plot of reaches of Highland Water streams in relation to the occurrence of physical habitat variables combined for reach scale comparisons.**
**(Open circles are reaches)**

**Table 4.12 DCA scores from analysis of physical variables and habitat diversity on the reach scale from Highland Water streams**

*(see Fig.4.15)*

| Variable | Axis I | Axis II | Axis III | Axis IV |
|---|---|---|---|---|
| CWD | 146 | 122 | 77 | 108 |
| Gravel | 23 | 9 | 54 | 73 |
| sand | -133 | 371 | 413 | 302 |
| silt | 150 | -23 | -132 | 334 |
| twigs | 246 | 123 | -62 | -112 |
| leaves | 170 | -87 | 234 | 106 |
| clay | -105 | 155 | -188 | 64 |
| undercuts | -71 | 118 | -1 | 49 |
| roots | 111 | -165 | -5 | -58 |
| weed | -302 | 411 | 251 | 262 |

### 4. 3. 7. Comparisons with streams in less wooded catchments

Some 15 channel units, mostly pools or pool-glides were sampled in the Dockens Water and Ober Water for comparisons with pools and CWD habitats in the Highland Water streams. The average length of reach sampled was approximately 25m in Dockens Water and 22m in the Ober Water, median widths ranged from 1.9-3.9m and 2.1-4m respectively. Median depths ranged from 0.2-0.62 and 0.2-0.5m. Table 4.6 shows the range of substrate compositions using the data from the two streams. The most obvious difference between the Highland Water streams and these less wooded streams is the relative scarcity of substrates most associated with riparian trees, notably submerged root matrices, leaves, twigs and woody debris. In contrast the lack of tree canopy allows strong growths of instream and marginal weed (Fig. 4.16) trailing riparian grasses and other plants. Instream weed growth covered over 60% of the substrate in some reaches. There was a greater range of depth in the Highland Water streams mainly as a result of CWD and scour effects.

Although the open streams did not show a greater number of habitat (physical) variables, the more even statistical distribution of occurrences resulted in a median habitat diversity index (H') significantly ($p<0.05$) higher than for riffles and CWD units in the wooded streams. There was no difference in physical diversity between open habitat units and wooded stream pools.

## 4. 4. DISCUSSION AND CONCLUSIONS

There was considerable within-reach variation in structural diversity in the Highland Water stream channels. Using median dimensions, riffles and pools were typically longer than CWD habitats. The short length of the CWD habitats was partly a result of isolation methods (i.e. care was taken to enclose matrices as closely as possible along the channel length), but mainly a result of the clumping (discontinuous distribution) of CWD in the channel. The isolated CWD habitats were also significantly deeper than pools with no CWD matrices. The general pattern was, however, that the *a priori* selected habitat sampling units can be

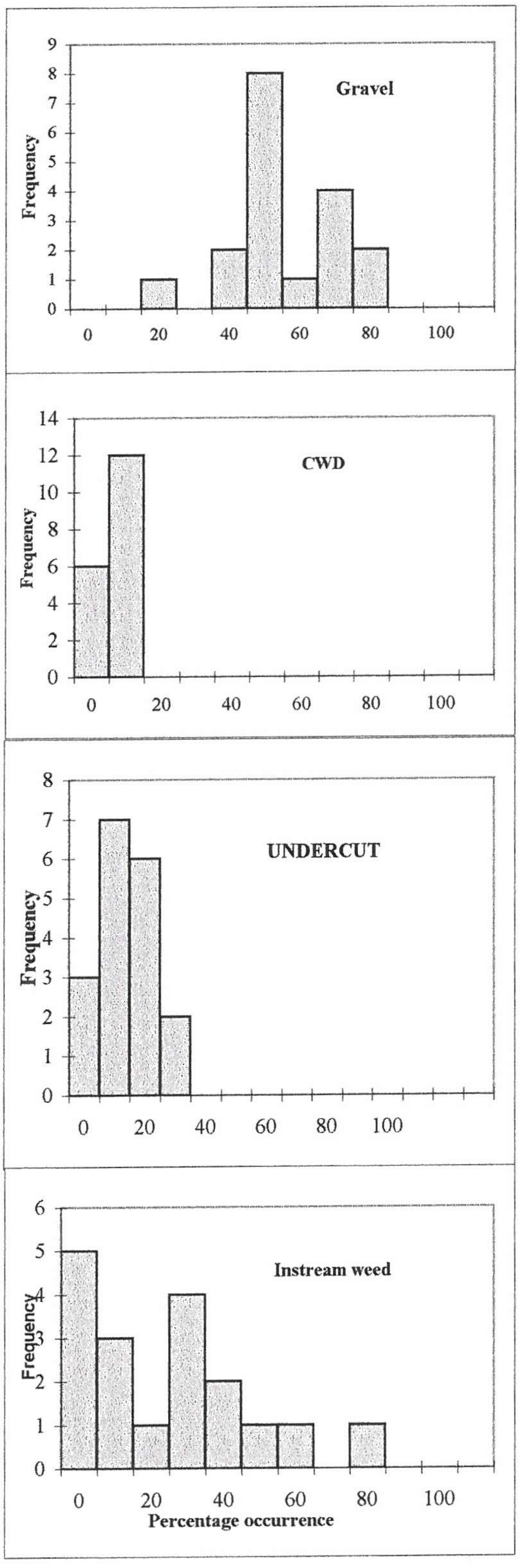


**Fig.4.16** **Frequency-distribution of physical variables and structures (as % occurrences in habitat units) in streams with reduced woodland in catchments (open)**

*(see Table 4.6 for full list)*

considered as points on a physical gradient with the general trend from eroding, shallow gravel reaches to depositing, deeper water with increasing amounts of wood. The CWD sampling units could also be regarded as isolated sub-units of pools which were originally formed by the CWD accumulation with the upstream part of the pool as a "dam-pool" the downstream part as a "plunge or scour pool" (Hawkins *et al.*, 1993). Distance downstream of the source was both a significant factor in habitat formation but was also a significant factor in CWD accumulation (Gregory *et al.*, 1993; Gurnell & Sweet 1998).

This greater depth strongly associated with CWD matrices is caused by the eroding effects of higher flows on soft substrata enhanced by the blockage resulting from the matrix itself (e.g. Bisson *et al.*, 1982; Hawkins *et al.*, 1993; Keller & MacDonald, 1995; Inoue & Nakano, 1998). Fig. 4.17 shows in diagram form a typical sequence of deepening. The initial obstruction causes both under cutting and overflow as flows increase. Entrapment of more CWD increases the blockage and consequently under cutting at moderate flow plus overflow and "plunge" or "scour" effects at higher flows. Gravels and other sediments are displaced and deposited as a transverse bar downstream of the CWD dam and ultimately the CWD matrix is left suspended or supported in a deep pool for which the gravel bar now acts as the barrier at low flows. The dam acts as an obstruction again at higher flows successively increasing the pool depth. From the literature the effects of CWD on channel formation, channel change and structural diversity, differ with many factors including gradient, substrate and discharge (see Bryant & Sedell, 1995; Gurnell *et al.*, 1995; Inoue & Nakano, 1998).

Typically pool formation is regarded as heavily dependent on CWD accumulation in some streams. Gurnell & Sweet (1998) described the association between pools and CWD dams in the Highland Water. They defined two categories of pools, namely "proximate" (at least part of the pool located within one channel width of a CWD dam) and "free" (entire pool more than one channel width from the nearest CWD dam). On this definition most of the pools sampled for this study were in the former category. The authors also conclude that the minimum distance between pools was 2 channel widths, probably the spacing

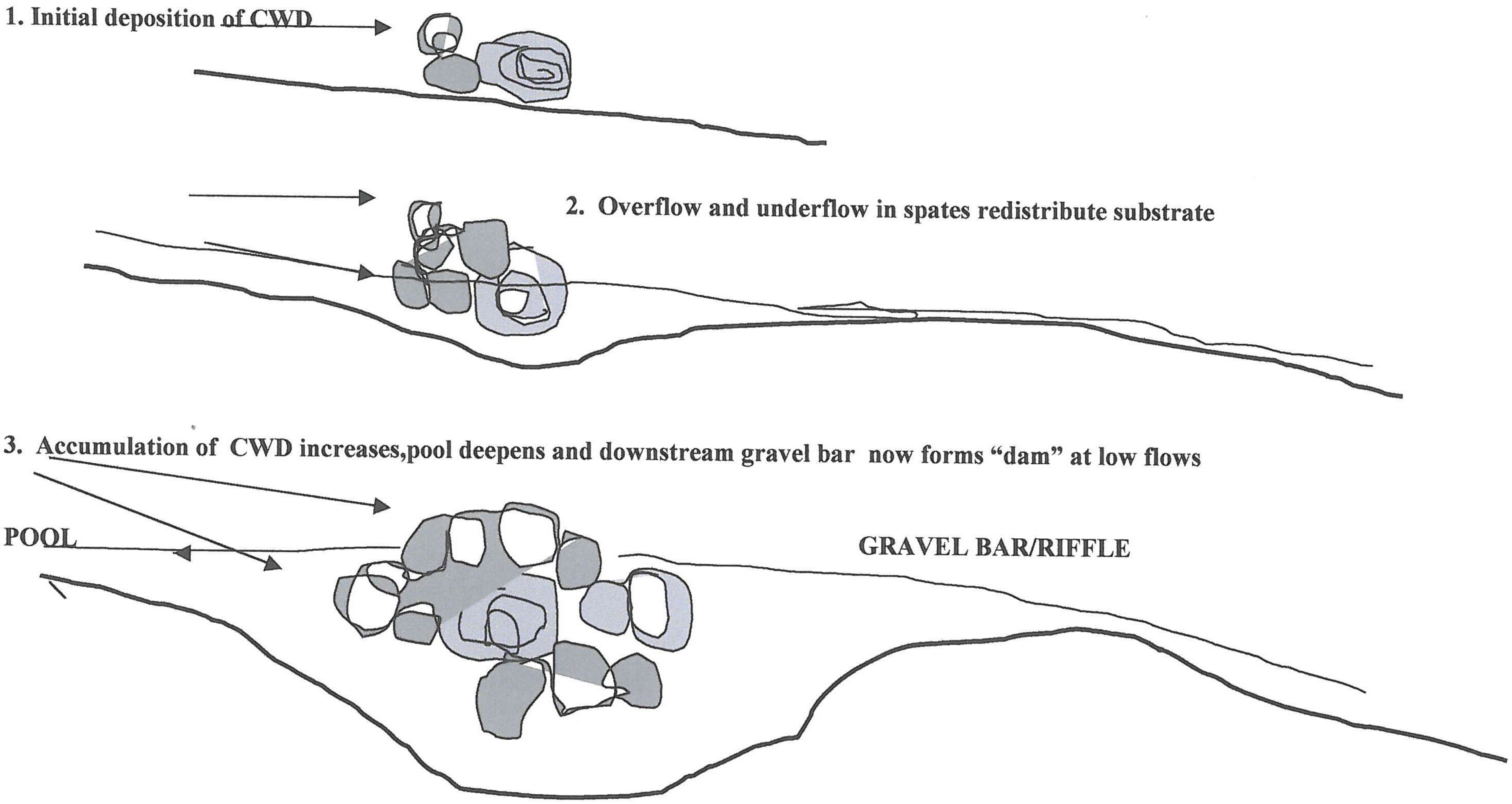


Figure 4. 17. Diagram of the deepening of a stream channel (habitat unit) in response to the accumulation of CWD .

that would occur if CWD was not removed from the Highland Water. In an earlier study, the inter-riffle distance in the Highland Water was estimated as 5-7 channel widths or 3.2 to 70.8m depending upon the distance downstream of the source (Gregory, Gurnell, Hill & Tooth, 1994). For this present study the pool lengths (inter-riffle lengths) were divided in most cases by the stop-nets and thus the median length was 11m and the range approximately 4 to 34m.

Keller & MacDonald (1995) showed that pool formation in headwater streams was caused by or strongly influenced by CWD. They also suggested that fish habitat was enhanced by the structural variety in the channel caused by CWD though no biological data were obtained. Removal of CWD decreased pool spacing from 2.5–1.6 channel widths and increased the pool area. Carlson *et al.* (1990) in contrast found that there was no difference in pool area between streams with high CWD and low CWD, though CWD was a component of the multi-variate regression relating physical variables to pool area. These authors also found pool volume greater at lower stream gradients where there was more woody debris. Keller & Tally (1979) note that debris dams are associated with and facilitate the formation of pools, providing valuable fish habitat. CWD dams affect both width and depth of stream channels and 50% of pools were associated with or caused by CWD. Fausch & Northcote (1992) noted in their streams that most pools were scour or plunge pools associated with CWD.

Gurnell *et al.* (1995) note that the precise impact of CWD on physical habitat diversity varies according to the size and energy of the stream and state that "*There is no doubt that the presence of CWD induces increased physical habitat diversity in river channels of all sizes*". The expression of physical diversity within a channel is however difficult to standardise and make comparisons because of the variety of methods and variables used in the various studies. In the Highland Water streams there was a significant relationship between amounts of CWD and habitat diversity on the reach scale but this was not so clear on the smaller, within-reach scale. The high homogeneity (low structural) diversity of riffles as habitats was previously noted by Martin-Smith (1998) in Malaysian streams.

In the Highland Water streams the highest structural diversity occurred in habitat units with moderate amounts of CWD usually partial dams or smaller full width dams. This is, of course, partly a result of the sampling technique in that the point-sampling using only dominant substrates mainly recorded CWD in the densest debris accumulations though the underlying substrates were gravel or finer sediments. The gravel riffles on the other hand did not contain other substrates and the point-contact technique gave a reasonably accurate characterisation as it did for the more diverse physical variables in the pools. Indications are that the partial removal of a dam and the subsequent changes in substrates following high discharge result in an increase in habitat diversity in some adjacent channel units despite scouring of sediments. Clearly, CWD is an important factor in determining the physical environmental gradient in the Highland Water streams. In the adjacent streams, however, with little or no CWD physical habitat diversity is very similar to that in the wooded stream pools as a result of increased marginal and instream vegetation and despite the lack of instream woody debris.

The main effect of the partial removal of a CWD dam and a subsequent spate was to allow the scouring of softer sediments from upstream and the reduction of depth in the deepest water under the dam. In comparison where a dam was retained over the same period some scouring also occurred but not to the same extent. There was generally more stability both upstream and downstream and in the CWD habitat unit where the dam was retained. Habitat diversity increased in the CWD habitat after partial removal of the timber. This preliminary study is being used to plan further studies on dam removal.

The accumulation and scouring of organic and inorganic sediments upstream of CWD dams is well reported (e.g. Gregory, Gurnell & Petts 1994; Gurnell *et al.*, 1995). Bilby (1981) demonstrated the increased export of sediments following the removal of debris dams in Hubbard Brook. Also, Shields & Smith (1992) using very similar techniques to the work in the Highland Water but on larger rivers, found that physical habitat diversity in a channelized sand-bed river was reduced where CWD had been cleared. Shannon-Wiener indices based on physical variables were 48% higher in the uncleared reaches than in the reaches where CWD had been removed. This relationship between CWD abundance and physical diversity was also obvious in the Highland Water streams but within the CWD accumulations diversity was significantly lower than in pools without CWD.

In summary, therefore, it is clear that considerable physical habitat variation occurs on the within reach scale in the wooded streams with riffles having the lowest diversity and pools with moderate amounts of CWD the highest. CWD is associated with greatest depths but despite the three dimensional matrix is assessed as a low-diversity physical habitat, mainly because of the dominance of one substrate as in riffles. There is, however, a distinction between habitat units and physical refugia in that the former can be diverse in both structure and substrate and contain a number of species, while refugia may consist of one material such as wood but be structured to provide physical niches or interstices which may be dominated by one or two species. It is thus important to distinguish between CWD as a provider of refugia and CWD as a formative agent of pools with greater habitat diversity (Gurnell & Sweet, 1998). On the reach scale CWD is clearly associated with increased habitat diversity.

The diversity of the physical habitat in the disturbed, open streams can be greater than that in the undisturbed wooded streams, because of the presence of overhanging vegetation and weed beds. Habitat associated with trees is, however, less prominent in the open lawns. The absence of CWD did not affect overall habitat diversity.

# CHAPTER 5

# ABUNDANCE AND SIZE DISTRIBUTION OF INDIVIDUAL FISH SPECIES IN RELATION TO CHANNEL STRUCTURE AND WOOD DEBRIS ACCUMULATIONS

## 5. 1. INTRODUCTION

Chapter 4 has shown that there is considerable within-reach variation in physical habitat in the low-disturbance, forested Highland Water streams. Riffles, pools and CWD habitat units as selected *a priori* showed different dimensions, habitat structures and diversities with riffles being more homogeneous than pools and CWD habitats. CWD habitat units did not show greater physical diversity than pools despite the presence of three dimensional matrices. The highest habitat diversity was associated with pools of moderate depth containing moderate amounts of CWD. Habitat diversity on the reach scale, was however, associated with increasing abundance of CWD. Physical habitat diversity in the disturbed, open streams of adjacent sub-catchments to the Highland Water was similar to that of pools in the wooded streams, The absence of CWD did not affect overall habitat diversity.

Ordination analysis of the physical variables relating to individual habitat units did not group the habitat units into the three categories on which the *a priori* selection was made (see Chapter 3). There were two major groupings which can be separated on the within-reach scale (see Chapter 4, Fig.4.7) based on a gradient of physical structure from the shallow, fast water (erosional) habitats to the deeper, slow-water (depositional) habitats. The riffles, pools and CWD habitat units are organised along the gradient depending mainly on depth, distance downstream of the source and the relative proportions of gravel and CWD recorded in the point transects. CWD is clearly a major factor, most associated with the deeper, downstream habitats.

The distribution of fish in relation to longitudinal gradients in rivers is well known both on the larger scale (e.g. Huet, 1959; Sheldon, 1968; Hynes, 1970; Hawkes, 1975) and on the reach scale (Paller, 1994; Williams *et al.*, 1996). The distribution of fish in relation to physical and structural gradients in an English chalk stream has been described by Prenda *et al.* (1997). In contrast, the abundance and community structure of fish populations in some streams has also been found to be based on discrete, definable channel units, i.e. riffles, runs, pools (e.g. Martin-Smith, 1998). This Chapter therefore aims to determine whether there are distinct and discrete within-reach differences in the spatial and temporal abundance patterns of individual fish species in the Highland Water streams which are related to definable habitat units or related to the physical gradient demonstrated in Chapter 4. The relative importance of CWD to the abundance of individual species, particularly those which are the targets of specific conservation strategies, is analysed. The role of CWD accumulations in fish abundance on the reach scale of the habitat unit is also explored.

A preliminary investigation of the within-reach and reach scale effects of the removal of a CWD dam on populations of individual species was made as a result of the partial removal of a dam in 1997/98 (see Chapter 4). The effects were compared with a reach where a dam was retained throughout the same period. Differences in stream habitat structure (e.g. Gorman & Karr, 1978; Binns & Eiserman, 1979), amounts of CWD (e.g. Dolloff, 1983; Hortle & Lake, 1983; Angermeier & Karr, 1984) and deforestation in catchments (e.g. Davies & Nelson, 1994) have all been associated with differences in fish distribution and abundance. To explore the effects on the stream and catchment scale in the New Forest comparisons of abundance between streams are also made in this chapter.

One problem was that the streams with large volumes of CWD were mainly in wooded catchments while the catchments of streams with low CWD contained significant amounts of heathland, open lawns or improved pasture in the riparian zones (Gregory *et al.*, 1993). These streams are also highly managed to maintain drainage of the lawns and pasture (see Chapter 4). It was not possible therefore to produce an orthogonal design for the comparisons. Finally, although this chapter includes some data on salmonids it deals mainly with the non-salmonid species, particularly those which have conservation implications. The salmonids are dealt with separately as a special case study in Chapter 7.

## 5. 2. DESCRIPTION OF THE AREA, SITES AND METHODOLOGY

The streams and sampling sites are shown in Appendices I, II & III and the physical descriptions and analyses are given in Chapter 4. Studies on within-stream variation in relation to CWD accumulation were carried out in the Highland Water, Bratley Water and Bagshot Gutter (see Chapter 2, Fig. 2.1 and Appendix II). Comparisons between open and wooded streams were made using data from quantitative sampling of the Ober Water and Dockens Water (see Chapter 2, Fig. 2.1) and comparisons of selected species in various streams were made using data provided by the Environment Agency (Environment Agency, Public Register, 1988-1998) (see Chapter 2, Fig.2.1). The fish fauna of New Forest streams is outlined in Chapter 2 and discussed further in Chapters 6 and 7.

Methods for measuring and analysing physical variables, and for sampling and measuring fish have been described in Chapter 3. Numerical abundance is compared on the basis of densities as numbers of individuals $m^{-2}$ of wetted surface. Biomass is compared also as $gm^{-2}$ of wetted surface. For some aspects of interpretation numbers and biomass per unit of standing volume are also used. Thus in the subsequent text "density" refers to numbers of individuals per $m^{-2}$ and "biomass" to wet weight of fish as $gm^{-2}$ unless otherwise stated.

## 5. 3. RESULTS

### 5. 3. 1. Fish densities and biomass in Highland Water streams

#### *a) Total catches and population estimates*

The reaches of the Highland Water, Bratley Water and Bagshot Gutter studied contained six species of the 20 species of fish formally recorded from New

Forest streams (Langford, 1996, see Chapter 6. Table 13). The species list and the total catches in the three habitat types in the Highland Water streams are shown in Table 5.1. Individual population estimates, standard errors and 95% confidence limits for each habitat sampling unit are given in Appendix III, together with the physical dimensions. A total of 6532 fish, with a total weight of almost 39,000g. were caught and measured. Three species (*S. trutta, P. phoxinus* and *C. gobio*) constituted 80% of the catch by number and 67% of the catch by weight, though eels *A. anguilla* were ranked second by weight. The total length of channel sampled in the Highland Water was 1704.8m. A total of 43 riffles, 80 pools and 39 CWD accumulations were sampled. This did not reflect the proportions in the stream (see Chapter 4).

Average catch efficiencies were estimated for all species from riffles, pools and CWD accumulations (Table 5.2). Kruskal-Wallis ANOVA on ranks showed that there were significant ($p<0.05$) differences between catch efficiencies for different species. Only *L. planeri* showed sufficiently low efficiencies (<25%) to invalidate the method for population estimates for most samples (Carle & Strub, 1978; Pisces Conservation Ltd., 1998). For this species most population estimates were based on total catch (see Chapter 3). The differences between efficiencies in the three habitats were not significant, though both *L. planeri* and *A. anguilla* showed more variation than the other species (Table 5.2) The highest overall efficiencies were for Salmonidae which demonstrates the relative effectiveness of this technique for this group (Cowx, 1983; Bohlin *et al.*, 1989).

### *b) Within-reach variations in fish abundance in Highland Water streams*

Within-reach variations in density and biomass of fish were compared for *a priori* habitat unit and for season. Statistical comparisons were made using the Kruskal-Wallis ANOVA on ranks and Dunn's multiple pairwise tests (see Chapter 3). Comparing the median estimated population numbers ($\text{nom}^{-2}$) and population biomass ($\text{gm}^{-2}$) of all fish from all samples over the sampling period, it is clear that there was no significant overall within-reach differences in the densities of the total fish populations in relation to the habitat sampling units (Fig. 5.1) (Kruskal-Wallis, $H = 5.576$, $df = 2$, $p = 0.062$). However, the

Table 5.1.Species composition by number and weight of the total fish catch from riffles,pools and CWD habitats in Highland Water streams from September 1996 to February 1998.

| Species | Riffles | | Pools | | CWD | | Totals | |
|---|---|---|---|---|---|---|---|---|
| | *Number* | *% comp* | *Number* | *%comp* | *Number* | *% comp* | *Number* | *% comp* |
| *Salmo trutta* Linnaeus 1758 | 433 | 26 | 472 | 12 | 200 | 22 | 1105 | 17 |
| *Phoxinus phoxinus* (Linnaeus 1758) | 180 | 11 | 1829 | 46 | 474 | 52 | 2483 | 38 |
| *Cottus gobio* Linnaeus 1758 | 934 | 56 | 745 | 19 | 89 | 10 | 1768 | 27 |
| *Lampetra planeri (Bloch 1784)* | 102 | 6 | 784 | 20 | 83 | 9 | 969 | 15 |
| *Noemacheilus barbatulus* (Linnaeus 1758) | 22 | 1 | 59 | 1 | 17 | 2 | 98 | 1.5 |
| *Anguilla anguilla (Linnaeus 1758)* | 10 | 1 | 55 | 1 | 44 | 5 | 109 | 1.7 |
| Total (all species) | **1681** | | **3944** | | **907** | | **6532** | |
| Species | Riffles/runs | | Pools/glides | | CWD habitats | | Totals | |
| | *Weight* | *% comp* | *Weight* | *%comp* | *Weight* | *% comp* | *Weight* | *%* |
| *Salmo trutta* Linnaeus 1758 | 1189.6 | 36 | 13578.4 | 60 | 5671.1 | 44 | 20579.1 | 53 |
| *Phoxinus phoxinus* (Linnaeus 1758) | 269.8 | 8 | 2494.9 | 11 | 804.4 | 6 | 3594.1 | 9 |
| *Cottus gobio* Linnaeus 1758 | 917.1 | 27 | 870.8 | 4 | 135.2 | 1 | 1955.1 | 5 |
| *Lampetra planeri (Bloch 1784)* | 270.8 | 8 | 1619.4 | 7 | 207.3 | 2 | 2114.5 | 5.5 |
| *Noemacheilus barbatulus* (Linnaeus 1758) | 95.6 | 3 | 297.2 | 1 | 141.4 | 1 | 539.2 | 1.4 |
| *Anguilla anguilla (Linnaeus 1758)* | 550.3 | 17 | 3726.1 | 1 | 5391.4 | 46 | 9731.8 | 24 |
| Total (all species) | **3291.1** | | **22415.7** | | **12995.6** | | **38702.4** | |

*Nomenclature see Maitland and Campbell, 1992*

Table 5.2. Average catch-efficiencies (as %) for all species in all habitat units in Highland Water streams. (significant differences shown by p and sig)

| TAXON | RIFFLES % | POOLS % | CWD % | p | sig |
|---|---|---|---|---|---|
| Salmonidae | 75 | 68 | 69 | *0.421* | *NS* |
| P.phoxinus | 48 | 49 | 61 | *0.146* | *NS* |
| C.gobio | 49 | 38 | 42 | *0.195* | *NS* |
| L.planeri | 14 | 25 | 23 | *0.046* | *NS* |
| N.barbatulus | 37 | 58 | 53 | *0.45* | *NS* |
| A.anguilla | 33 | 86 | 34 | *0.41* | *NS* |
| All taxa | 49 | 44 | 58 | *0.411* | *NS* |

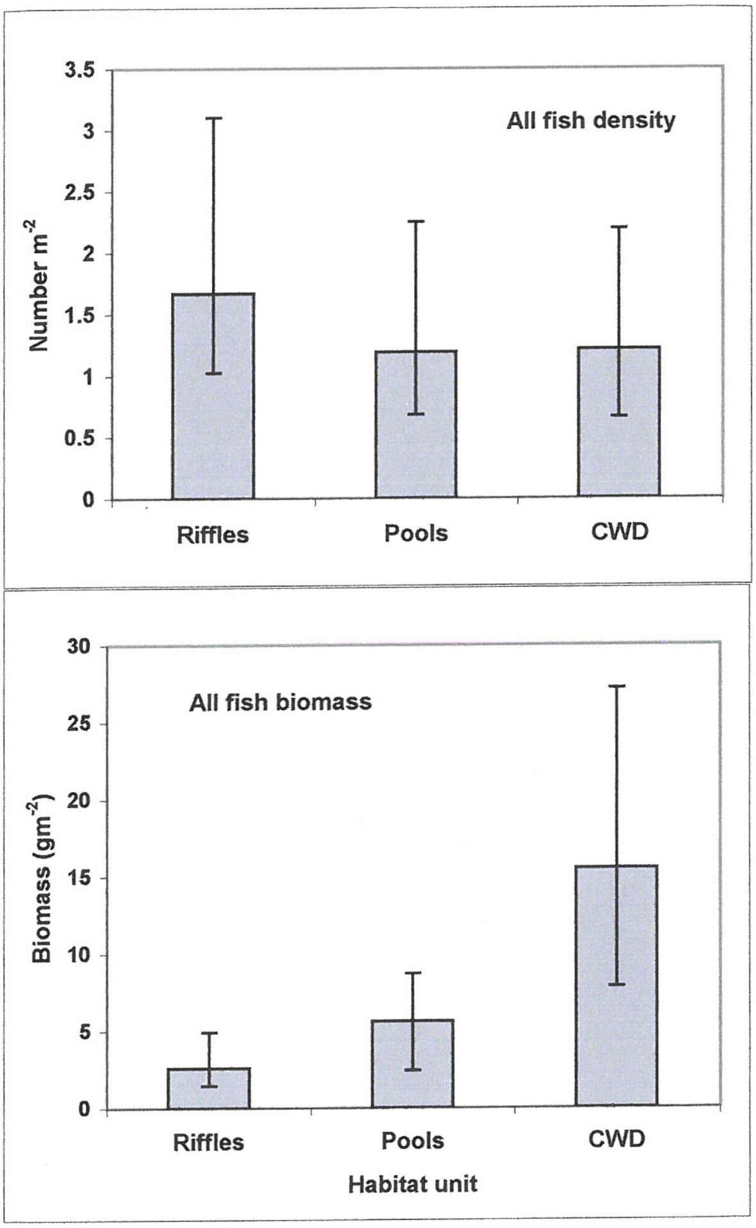


**Fig. 5.1** **Median and quartile densities and biomass of all fish from riffles,pools and CWD habitats in New Forest streams. (Data are pooled from all samples over the whole sampling period)**

difference in median biomass was very highly significant ($H = 38.72$, df= 2, $p<0.001$) (Table 5.3). Pairwise tests showed that biomass per unit area was significantly greater both in pools and CWD than in riffles ($p<0.05$) and significantly greater in CWD than pools ($p<0.05$). This indicates that the presence of matrices of woody debris did not result in larger numbers of fish overall in CWD habitat units, but suggests that the average size of fish was greater than in riffles and pools.

There were very marked differences between the overall median densities and biomass of the individual species in the three habitat unit categories (Figs. 5.2, 5.3, Table 5.3). Data for the five most abundant species are shown in the figures. Salmonidae were significantly more numerous in riffles and CWD than pools but showed significantly higher biomass in CWD than pools and in pools than riffles.

Minnow (*P. phoxinus*) and bullhead (*C. gobio*) densities showed complementary trends. The former showed highest densities in CWD habitats and pools than riffles but the difference between these two was not significant ($P>0.05$). The biomass was, however, significantly higher in CWD than pools ($p<0.05$). There were very few *P. phoxinus* found in riffle habitats and indeed the species was practically absent. The few individuals were usually in small back eddies at the edge of riffles caused by obstructions or collapsed banks.

In contrast the highest densities of *C. gobio* were in riffles and the lowest in CWD and unlike *P. phoxinus* the biomass was lower in CWD habitat units. *C. gobio* caught in pools or in CWD habitats were usually in the margins where there was an area of gravel or cobbles or in small areas of gravel or cobbles at the head of a pool where a riffle ended. No individuals were observed on wood in the CWD accumulations. There is clear evidence of marked habitat preferences by these two species.

Lampreys (*Lampetra* spp.) were most numerous in pools particularly in banks of sand mixed with silt and twigs (Fig. 5.2). They were significantly more abundant in pools than in riffles ($p<0.05$) but the difference between pools and CWD was not significant ($p>0.05$). Biomass followed the same pattern as density (Fig.5.3).

**Table 5.3** **Comparisons of overall abundance of fish species in riffles,pools and CWD habitats in Highland Water streams. (All samples, 1996-98)**

| Species | | Median | | | KW | df | p | Pairwise tests | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | Riffles | Pools | CWD | | | | | | |
| ***S.trutta*** | Density | 0.24 | 0.11 | 0.33 | 11.614 | 2 | 0.003 | **R>P** | **W>P** | W/R |
| | *Quartiles* | *0.12-0.64* | *0.05-0.34* | *0.11-0.5* | | | | | | |
| | Biomass | 0.5 | 1.56 | 5.6 | 41.491 | 2 | <0.001 | **W>R** | **W>P** | P>R |
| | *Quartiles* | *0.19-0.94* | *0.26-4.35* | *3.34-14.7* | | | | | | |
| ***P.phoxinus*** | Density | 0 | 0.33 | 0.45 | 46.196 | 2 | <0.001 | **W>R** | **P>R** | W/P |
| | *Quartiles* | *0-0.06* | *0.09-0.89* | *0.17-0.76* | | | | | | |
| | Biomass | 0 | 0.35 | 0.89 | 51.487 | 2 | <0.001 | **W>R** | **W>P** | **P>R** |
| | *Quartiles* | *0-0.05* | *0.09-0.89* | *0.46-1.44* | | | | | | |
| ***C. gobio*** | Density | 0.9 | 0.2 | 0.05 | 55.729 | 2 | <0.001 | **R>W** | **R>P** | **P>W** |
| | *Quartiles* | *0.43-1.73* | *0.06-0.4* | *0-0.15* | | | | | | |
| | Biomass | 0.93 | 0.22 | 0.07 | 44.913 | 2 | <0.001 | **R>P** | **R>W** | **P>W** |
| | *Quartiles* | *0.43-1.59* | *0.08-0.45* | *0-0.28* | | | | | | |
| ***Lampetra*** | Density | 0 | 0.3 | 0.11 | 17.671 | 2 | <0.001 | **P>R** | P/W | W/R |
| | *Quartiles* | *0-0.27* | *0.1-0.64* | *0-0.55* | | | | | | |
| | Biomass | 0 | 0.13 | 0.08 | 16.084 | 2 | <0.001 | **P>R** | P/W | W/R |
| | *Quartiles* | *0-0.9* | *0.04-0.32* | *0-0.22* | | | | | | |
| ***N.barbatulus*** | Density | 0 | 0 | 0 | 2.599 | 2 | 0.273 | NS | NS | NS |
| | *Quartiles* | *0* | *0.01* | *0* | | | | | | |
| | Biomass | 0 | 0 | 0 | 1.414 | 2 | 0.493 | NS | NS | NS |
| | *Quartiles* | *0* | *0-0.02* | *0* | | | | | | |
| ***A. anguilla*** | Density | 0 | 0 | 0.05 | 21.246 | 2 | 0.001 | **W>R** | W/P | R/P |
| | *Quartiles* | *0* | *0-0.03* | *0-0.09* | | | | | | |
| | Biomass | 0 | 0 | 3.1 | 22.253 | 2 | <0.001 | **W>R** | **W>P** | P/R |
| | *Quartiles* | *0* | *0-2.157* | *0-8.04* | | | | | | |
| ***All fish*** | Density | 1.67 | 1.19 | 1.21 | 5.586 | 2 | 0.061 | NS | NS | NS |
| | *Quartiles* | *1.03-3.10* | *0.68-2.25* | *0.66-2.19* | | | | | | |
| | Biomass | 2.62 | 5.64 | 18.7 | 38.732 | 2 | <0.001 | W>P | W>R | P>R |
| | *Quartiles* | *1.45-4.92* | *2.44-8.71* | *12.86-32.19* | | | | | | |

***Where values are given as 0 the real values are less than 0.01***
***KW= Result of Kruskal-Wallis test, Bold = sig. P<0.05***
***density as Numbers $m^{-2}$ , Biomass as $g.m^{-2.}$***

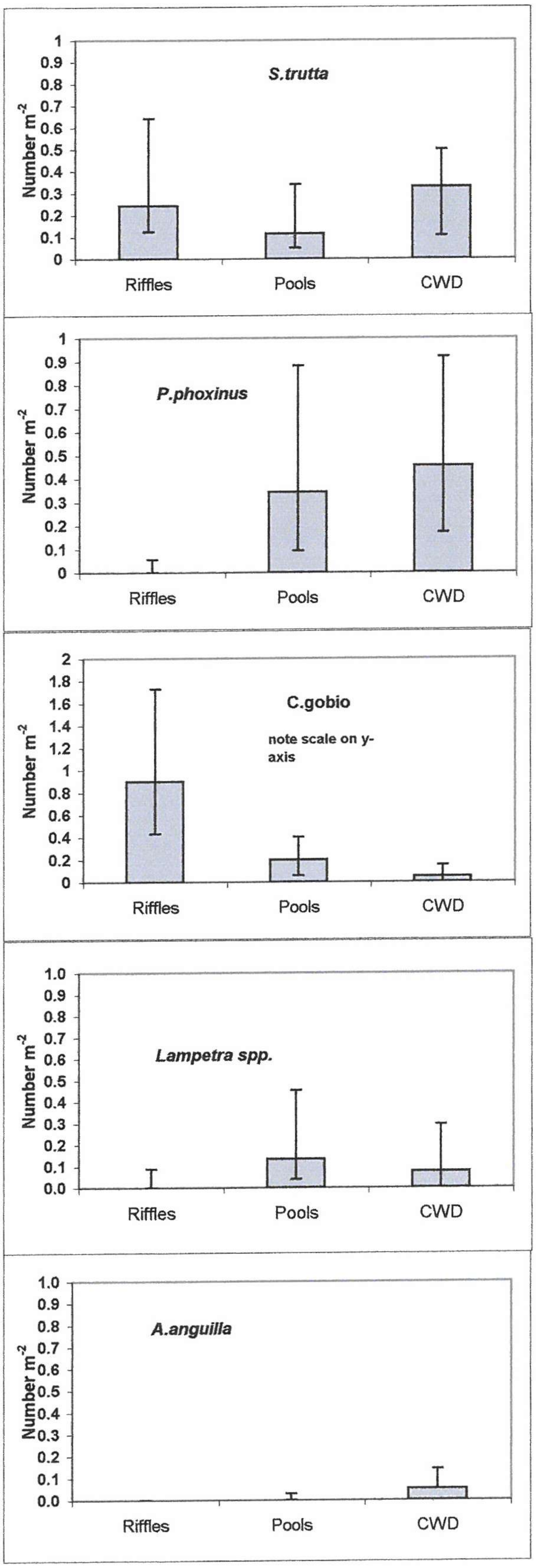


**Fig.5.2** **Median and quartile overall densities of fish species in riffles, pools and CWD habitat units in Highland Water streams.**
***(Data are from all samples taken in 1996-98)***

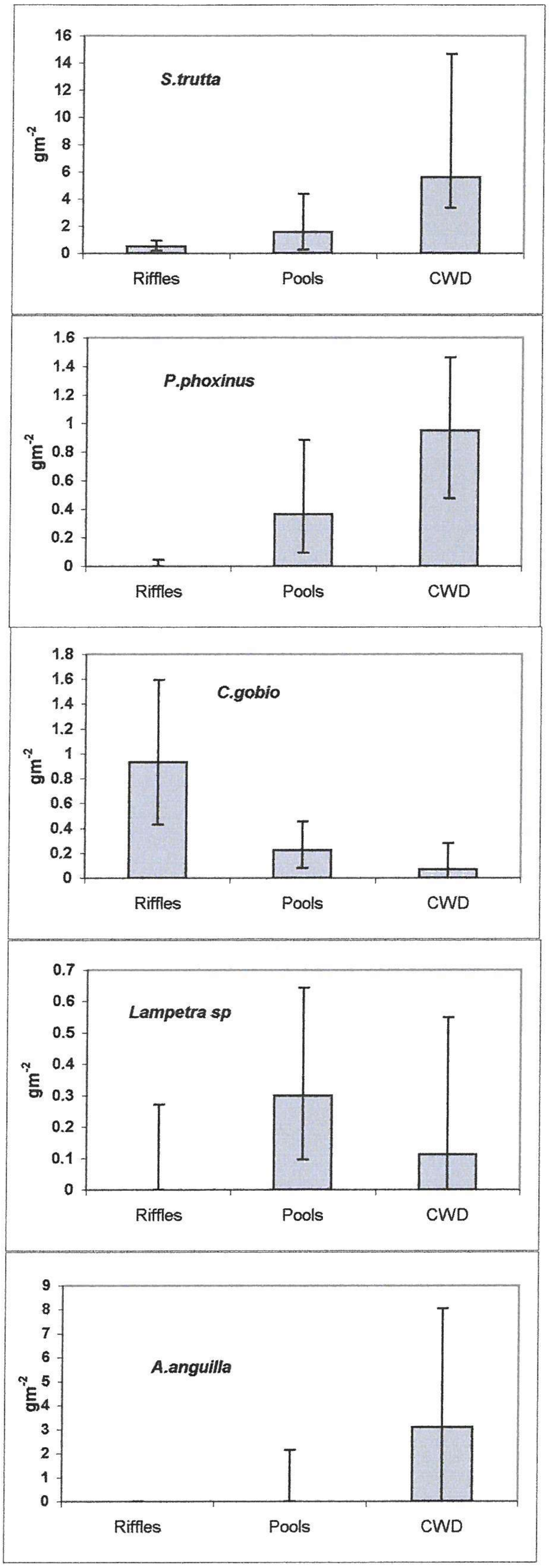


**Fig.5.3** **Median and quartile estimates of biomass (g.m$^{-2}$) in riffles,pools and CWD habitat units in Highland Water streams**

***(Note the different scales on the y-axes for the comparisons in habitats for individual species. Relative abundances of the species are shown in Table 5. 1)***

Eels (*A. anguilla*) were significantly more numerous and in greater biomass among the CWD matrices than in riffles ($p>0.05$). Densities were not significant between pools and CWD but there was a significant difference in median biomass ($p<0.05$).

Stone loaches (*N. barbatulus*) occurred relatively infrequently and in low numbers in riffles, pools and CWD accumulations. Densities as numbers $m^{-2}$ ranged from 0-0.22 in riffles, 0-0.28 in pools and 0-0.80 in CWD. Median densities were significantly greater in CWD than in the other two habitat types ($p<0.05$). Densities and biomass of species and total fish were also compared in pools upstream and downstream of the CWD accumulations. These pools equated to "*dam pools*" and "*plunge or scour pools*" (see Maddock, 1999). There were no significant differences ($p>0.05$) for either density or biomass of any species or for the total fish populations.

Because of the within-reach differences in abundance, the contributions of each habitat unit type to the total standing stock of the stream are of interest. For example, riffles comprised 23% of the reach area sampled, (Table 5.4) provided 26% of the fish numbers but less than 9% of the total biomass. In contrast CWD habitat units comprised 12% of the area sampled, 14% of the numbers and 34% of the biomass. Pools comprised 66 % of the area sampled, and provided 60% of all fish numbers and 58% of biomass. These data suggest that riffles contain a disproportionately large number of small fish relative to their area during daylight, CWD habitats contain larger fish and pools carry slightly less in both numbers and biomass than might be expected from the area sampled. If the data are compared on the basis of volume (Table 5.4) the role of riffles as habitats for small fish is enhanced with 4% by volume carrying 26% by number and 8.8% by biomass. In comparison CWD habitats carry proportionally less fish (21% volume, 14% by number, 34% by weight) and pools appear to have even lower relative holding capacity (74% volume, 60% by number, 34% by weight).

The habitat units were not sampled in proportion to their occurrence in the streams but the data from this analysis show their relative importance for different species. For example using the ratio of percentage abundance to

Table 5.4 Percentage of the total catch contributed by riffles,pools and CWD habitat units in Highland Water streams in relation to the percentage areas of each habitat sampled.

| | RIFFLES | % | POOLS | % | CWD | % | TOTAL | |
|---|---|---|---|---|---|---|---|---|
| Length (m) | 475.6 | 28.8 | 991.3 | 60.0 | 184.5 | 11.2 | **1651.4** | |
| Area ($m^2$) | 1161 | 22.8 | 3338.1 | 65.6 | 589.9 | 11.6 | **5089** | |
| Volume ($m^3$) | 58.43 | 4.2 | 1038.2 | 74.4 | 299.1 | 21.4 | **1395.7** | |
| *Total number* | | | | | | | | |
| | Number | % | Number | % | Number | % | Total | % |
| **Salmonids** | 433 | *39.2* | 472 | *42.7* | 200 | *18.1* | **1105** | **16.9** |
| ***P.phoxinus*** | 180 | *6.3* | 1829 | *64.3* | 474 | *16.7* | **2843** | **43.5** |
| ***C.gobio*** | 934 | *52.8* | 745 | *42.1* | 89 | *5.0* | **1768** | **27.1** |
| ***Lampetra*** | 102 | *10.5* | 784 | *80.9* | 83 | *8.6* | **969** | **14.8** |
| ***N.barbatulus*** | 22 | *22.4* | 59 | *60.2* | 17 | *17.3* | **98** | **1.5** |
| ***A.anguilla*** | 10 | *9.2* | 55 | *50.5* | 44 | *40.4* | **109** | **1.7** |
| **All fish** | 1681 | *25.7* | 3944 | *60.4* | 907 | *13.9* | **6532** | |
| *Catch weight in g* | | | | | | | | |
| | Biomass | % | Weight | % | Weight | % | Weight | % |
| **Salmonids** | 1189.6 | *5.8* | 13578.4 | *66.0* | 5671.1 | *27.6* | 20579.1 | **53.2** |
| ***P.phoxinus*** | 269.8 | *7.5* | 2494.9 | *69.4* | 804.4 | *22.4* | 3594.1 | **9.3** |
| ***C.gobio*** | 917.1 | *46.9* | 870.8 | *44.5* | 135.2 | *6.9* | 1955.1 | **5.1** |
| ***Lampetra*** | 270.8 | *12.8* | 1619.4 | *76.6* | 207.3 | *9.8* | 2114.5 | **5.5** |
| ***N.barbatulus*** | 95.6 | *17.7* | 297.2 | *55.1* | 141.4 | *26.2* | 539.2 | **1.4** |
| ***A.anguilla*** | 550.3 | *5.7* | 3726.1 | *38.3* | 5391.4 | *55.4* | 9731.8 | **25.1** |
| **All fish** | 3291 | *8.5* | 22415.7 | *57.9* | 12995.6 | *33.6* | 38702.4 | |

percentage area sampled as an indicator of relative importance of the habitat to the species, the figures for *P. phoxinus* are riffles, 0.27, pools, 0.98 and CWD habitats 1.43 for density showing that the CWD habitat carries a disproportionate number of minnows compared with the other habitats. In contrast, the respective ratios for *C. gobio* are 2.32, 0.64 and 0.43 confirming the complementary abundance pattern of these species. The ratios for *A. anguilla* are 0.40, 0.76 and 3.49, for *Lampetra* sp., 0.46, 1.23, and 0.74 and for *N. barbatulus*, 0.98, 0.92 and 1.49. Thus both *A. anguilla* and *N. barbatulus* showed tendencies for greater relative abundance in the CWD habitats which confirm those indicated by simple density figures.

### *c) Seasonal variations*

Two-way analysis of variance using season and habitat (General Linear Model (GLM)) was unsuccessful for most fish abundance data because even after log transformation variances were not homogeneous and even after ranking, the data were not acceptable for calculating interactions between season and habitat. For individual species both the variance and the low frequencies of occurrence in samples from habitat types prevented the GLM analysis. The data were therefore compared separately for both season and habitat using the Kruskal-Wallis and Dunn's multiple pairwise tests on the monthly data combined into seasonal totals. (Table 5.5). Densities and biomass were compared initially using the data for all samples in each *a priori* selected habitat units over the sampling period.

Seasonal density and biomass estimates are given for all species and all habitat units in Table 5.5. The seasonal and habitat unit data have been compared using separate Kruskal-Wallis and Dunn's tests and there is no estimate of any season and habitat interaction. In the following text each species is dealt with separately

*S. trutta* (all Salmonidae)

There were significant seasonal and habitat differences in density (Table 5.5). The densities of salmonids were significantly higher ($p<0.05$) in riffles in summer than in pools. In autumn and winter densities were higher in CWD

**Table 5.5** **Median density and biomass of fish in habitat units and seasons in Highland Water streams in the New Forest 1996-98**

**Table 5.5.** **Part 1**

*Salmonidae*

| ***Density*** | **Riffles** | **Pools** | **CWD** | **Significant at p <0.05** |
|---|---|---|---|---|
| Autumn | 0.19 | 0.09 | 0.28 | **W>P** |
| Winter | 0.11 | 0.08 | 0.29 | **W>P** |
| Spring | 0.85 | 0.43 | 0.49 | NS |
| Summer | 0.9 | 0.35 | 0.63 | **R>P** |
| ***Significance*** | <0.001 | <0.001 | NS | |
| ***Biomass*** | | | | |
| Autumn | 0.46 | 1.26 | 6.29 | **W>R,W>P** |
| Winter | 0.4 | 0.69 | 9.66 | NS |
| Spring | 0.48 | 0.91 | 6.9 | **W>R,W>P** |
| Summer | 1.3 | 1.39 | 5.43 | **W>R** |
| ***Significance*** | NS | ***<0.001*** | NS | |

*P.phoxinus*

| ***Density*** | **Riffles** | **Pools** | **CWD** | **Significant at p<0.05** |
|---|---|---|---|---|
| Autumn | 0.17 | 0.41 | 0.53 | **W>R** |
| Winter | 0.05 | 0.8 | 0.54 | NS |
| Spring | 0 | 0.09 | 0.41 | **W>R** |
| Summer | 0 | 0.3 | 0.43 | NS |
| ***Significance*** | **0.005** | **0.005** | **0.005** | |
| ***Biomass*** | | | | |
| Autumn | 0.02 | 0.43 | 1.06 | W>R, P>R |
| Winter | 0.17 | 0.76 | 1.2 | NS |
| Spring | 0 | 0.09 | 1.1 | **W>R** |
| Summer | 0 | 0.92 | 0.64 | NS |
| ***Significance*** | NS | NS | NS | |

*C.gobio*

| ***Density*** | **Riffles** | **Pools** | **CWD** | **Significant at p <0.05** |
|---|---|---|---|---|
| Autumn | 0.87 | 0.17 | 0 | **R>P>W** |
| Winter | 0.81 | 0.14 | 0 | R>W |
| Spring | 0.78 | 0.24 | 0.1 | **R>W, R>P** |
| Summer | 2.84 | 1.35 | 0.08 | NS |
| ***Significance*** | NS | **0.002** | NS | |
| ***Biomass*** | | | | |
| Autumn | 0.88 | 0.22 | 0 | R>P>W |
| Winter | 0.87 | 0.14 | 0 | **R>W** |
| Spring | 0.84 | 0.23 | 0.26 | **R>P** |
| Summer | 4.01 | 1.56 | 0.09 | NS |
| ***Significance*** | NS | **0.009** | NS | |

**Table 5. 5 .** **Part 2**

*Lampetra sp*

| ***Density*** | **Riffles** | **Pools** | **CWD** | **Significant at p <0.05** |
|---|---|---|---|---|
| Autumn | 0 | 0.1 | 0 | NS |
| Winter | 0.87 | 0.19 | 0 | **R>W** |
| Spring | 0.04 | 0.07 | 0.08 | NS |
| Summer | 0.04 | 0.18 | 0.12 | NS |
| ***Significance*** | NS | NS | NS | |
| ***Biomass*** | | | | |
| Autumn | 0 | 0.29 | 0 | NS |
| Winter | 0.42 | 0.49 | 0.51 | NS |
| Spring | 0.06 | 0.17 | 0.23 | NS |
| Summer | 0.08 | 0.31 | 0.11 | NS |
| ***Significance*** | NS | NS | NS | |

*N.barbatulus*

| ***Density*** | **Riffles** | **Pools** | **CWD** | **Significant at p <0.05** |
|---|---|---|---|---|
| Autumn | 0 | 0 | 0 | NS |
| Winter | 0.9 | 0 | 0 | NS |
| Spring | 0 | 0 | 0 | NS |
| Summer | 0 | 0 | 0 | NS |
| ***Significance*** | Ns | NS | NS | |
| ***Biomass*** | | | | |
| Autumn | 0 | 0 | 0 | NS |
| Winter | 7.3 | 0 | 0 | NS |
| Spring | 0 | 0 | 0 | NS |
| Summer | 0 | 0 | 0 | NS |
| ***Significance*** | NS | NS | NS | |

***NB, medians are shown only to three decimal points actual medians may be less than 0.001***

*A.anguilla*

| ***Density*** | **Riffles** | **Pools** | **CWD** | **Significant at p <0.05** |
|---|---|---|---|---|
| Autumn | 0 | 0 | 0.22 | NS |
| Winter | 0 | 0 | 0.07 | NS |
| Spring | 0 | 0.02 | 0.07 | **W>R** |
| Summer | 0 | 0.05 | 0 | NS |
| ***Significance*** | NS | NS | NS | |
| ***Biomass*** | | | | |
| Autumn | 0 | 0 | 0.51 | NS |
| Winter | 0 | 0 | 5.63 | NS |
| Spring | 0 | 0.63 | 4.24 | **W>R** |
| Summer | 0 | 2.4 | 0 | NS |
| ***Significance*** | *NS* | ***0.045*** | *NS* | |

**Table 5.5.** **Part 3**

*All fish (total)*

| ***Density*** | **Riffles** | **Pools** | **CWD** | **Significant at p <0.05** |
|---|---|---|---|---|
| Autumn | 1.24 | 1.22 | 0.95 | **P>W,P>R** |
| Winter | 2.36 | 1.07 | 1.73 | NS |
| Spring | 1.75 | 1.15 | 1.38 | NS |
| Summer | 3.41 | 2.67 | 1.79 | NS |
| ***Significance*** | NS | **0.016** | NS | |
| ***Biomass*** | | | | |
| Autumn | 1.81 | 5.05 | 14.2 | **P>W,P>R** |
| Winter | 4.19 | 4.51 | 31.9 | **W>P** |
| Spring | 2.63 | 6.1 | 12.7 | **W>R** |
| Summer | 5.01 | 7.6 | 16.2 | NS |
| ***Significance*** | NS | NS | NS | |

***Density as numbers m $^{-2}$, biomass as g m $^{-2}$***

***R = riffles, P = pools, W = CWD, P>R = pools greater than riffles (significant)***

habitats. There was no significant difference in spring. There were no significant differences in seasonal density in riffles ($p<0.05$) despite the recruitment of young fish and the higher median values for spring and summer. Low numbers of samples most likely masked any differences. In pools, however, median densities showed significant seasonal variation ($p<0.001$) with the highest values in spring and summer. There was no significant seasonal variation in densities in the CWD habitats.

Biomass was significantly different between habitats in autumn, spring and summer ($p<0.001$) with CWD showing higher values than riffles and pools. The differences in winter were not significant despite showing the same pattern as the other seasons. The data for salmonids are discussed in more detail in Chapter 7.

*P. phoxinus*

Densities and biomass of minnows varied significantly with habitat in autumn ($p <0.001$) and spring ($p <0.001$) with higher densities in CWD habitats than in riffles. Densities were significantly higher ($p<0.05$) in CWD than riffles in spring and autumn. Biomass was also significantly higher in pools and CWD habitats ($p<0.05$) than in riffles in autumn but in spring the only significant difference was between CWD and riffle habitats. There was no seasonal variation in biomass in riffles mainly because of the very sporadic occurrence and low numbers. Densities varied significantly with season in both pools ($p = 0.005$) and CWD habitats ($p = 0.005$).

*C. gobio*

Densities and biomass varied significantly with both habitat and season. There were significant differences in the densities between habitat types in autumn ($p < 0.001$), winter ($p = 0.020$) and spring ($p = 0.002$) but not in summer ($p = 0.12$). The lack of significance in summer was most likely a function of the low numbers of samples. The highest densities were in the riffles in all seasons and the lowest mostly in the CWD habitats. Biomass followed the same patterns.

There was a significant seasonal variation in density (p = 0.002) and biomass (p= 0.009) in the pools. Both were 4-5 times higher in summer than in the other seasons. The median densities were also higher in riffles in summer but the differences were not significant (p = 0.518) probably as a result of low sample numbers. There was no evidence of a shift in the balance of abundance during low summer flows despite the shallow depth and decrease of area of the riffles.

*Lampetra* sp. *(planeri)*

The only significant difference in density or biomass was in winter when riffles showed higher densities than pools or CWD. There were no other significant seasonal differences in either density or biomass in either of the habitats. As Table 5. 5 shows the abundance was typically low in all habitats in all seasons with most median values calculated as below 0.001. From the summed data for all the sampling period there was evidence of an overall difference with the highest numbers and biomass in pools (see Fig.5.2, 5.3).

*N. barbatulus*

Stone loach densities and biomass were typically low in all habitats with occasional higher values. There were no significant differences in either density or biomass in either habitat type in either season. Further there were no significant seasonal variations in either habitat. The highest median biomass was 7.3gm$^{-2}$ in riffles in winter but this was from a particularly large and unusual sample.

*A. anguilla*

There were significant differences between the density and biomass of eels in spring though abundance was generally low except in CWD habitats. The highest biomass values were in CWD habitats in spring.

*All fish*

Comparison of the total numbers and biomass of all fish summed for each season of the sampling period (Table 5.5) shows that there were consistent spatial differences in biomass but less temporal variation. Significant differences in density between habitats only occurred in autumn ($p < 0.001$) where the median number of fish in pools was greater than CWD and riffles respectively (Table 5.5) though this was not sufficient to bias the total sample. In comparison significant differences in total biomass occurred between habitat types in autumn ($p<0.001$), winter ($p = 0.033$) and spring ($p = 0.027$) but not in summer ($p=0.466$).

The highest median biomass values were in the CWD habitat units in all months but low numbers of samples in summer were the probable cause of the non-significant difference. Only pools showed a significant seasonal variation and that only in total fish density ($p = 0.016$). Here the median density in summer was more than twice that of the other seasons though some of this may have been due to the reduction in pool area in the drier periods. Biomass was not significantly higher ($p = 0.318$) than other seasons though it was the highest of the four values.

### *d) Size-distributions*

There were significant differences in size-distributions of all species in catches on the within-reach scale in the three habitat units. Descriptive statistics for the size distributions are given in Table 5.6. Salmonids are dealt with in greater detail in Chapter 7 but as Table 5.7 shows, fish were on average significantly larger ($p<0.05$) in CWD habitats than in pools or riffles. The length-frequency distributions (Fig. 5.4) show that higher proportions of the catch were below 7cm in the riffles and pools than in the CWD. Modal lengths were 3cm in both riffles and pools and 10.8cm in the CWD. Medians were 3.5, 6.4 and 10.8cm TL. Kurtosis and skewness were similar for the riffle and pool distribution (Table 5.6) but there was a marked shift in the size distribution in CWD habitats toward

**Table 5.6. Descriptive statistics for size-distributions of all individuals of all species caught in Highland Water streams. New Forest. 1996-98**

| RIFFLES | *S.trutta* | *P.phoxinus* | *C.gobio* | *Lampetra** | *N.barbatulus* | *A.anguilla** |
|---|---|---|---|---|---|---|
| Mean | 4.7 | 5.4 | 4.1 | 2.6 | 7.8 | 55 |
| Std.Error | 0.14 | 0.22 | 0.04 | 0.15 | 0.45 | 13 |
| Median | 3.5 | 5.7 | 4.2 | 2.6 | 7.5 | 4.64 |
| Mode | 3 | 4.4 | 4 | 2.4 | 10.7 | NA |
| Std.Dev. | 8.3 | 1.82 | 1.23 | 1.3 | 2.16 | 42.5 |
| Variance | 24.9 | 3.31 | 1.26 | 1.7 | 4.7 | 180.9 |
| Kurtosis | 17.91 | 0.62 | -0.69 | 0.7 | -1.19 | 1.1 |
| Skewness | 3.6 | -0.36 | 0.08 | -0.02 | -0.02 | 1.21 |
| Number | 409 | 70 | 933 | 79 | 25 | 12 |
| **POOLS** | | | | | | |
| Mean | 8.6 | 4.6 | 4.4 | 2 | 7.7 | 69 |
| Std.Error | 0.31 | 0.1 | 0.04 | 0.05 | 0.36 | 6.52 |
| Median | 6.4 | 4.7 | 4.4 | 2 | 8.5 | 59.4 |
| Mode | 3 | 2 | 4 | 1 | 9 | 85 |
| Std.Dev. | 6.7 | 1.8 | 1.01 | 1.22 | 2.82 | 47.9 |
| Variance | 44.31 | 3.3 | 1.01 | 1.51 | 7.95 | 2292.9 |
| Kurtosis | 18.89 | -0.84 | 0.02 | -0.19 | -1.43 | 8.73 |
| Skewness | 3.5 | 0.16 | 0.29 | 0.41 | -0.2 | 2.4 |
| Number | 463 | 312 | 704 | 508 | 56 | 43 |
| **CWD** | | | | | | |
| Mean | 11.9 | 5.5 | 4.6 | 2.2 | 9.6 | 135 |
| Std.Error | 0.43 | 0.15 | 0.13 | 0.14 | 0.39 | 49.2 |
| Median | 10.8 | 5.7 | 4.6 | 2.4 | 9.7 | 71.7 |
| Mode | 10.8 | 6 | 5.5 | 2.3 | 10.5 | NA |
| Std.Dev. | 6 | 2.05 | 1.14 | 1.04 | 1.67 | 326.3 |
| Variance | 35.8 | 4.2 | 1.3 | -1.07 | 2.78 | 106435.9 |
| Kurtosis | 0.14 | 0.6 | 0.72 | -0.27 | 0.52 | 40.91 |
| Skewness | 0.61 | -0.92 | -0.17 | -0.16 | -0.72 | 6.3 |
| Number | 186 | 220 | 84 | 53 | 19 | 52 |

***Number = number measured, * = weight in g, others as Total length in cm***

Table 5.7 Comparisons of individual lengths/weights of fish caught from riffles,pools and CWD habitats in Highland Water streams

*a) S.trutta (length)*

| | n | Median | 25% | 75% |
|---|---|---|---|---|
| Riffles | 409 | 3.5 | 3 | 5.5 |
| Pools | 463 | 6.4 | 3.7 | 12.2 |
| CWD | 198 | 10.8 | 7 | 16.5 |

*H=285.05,df =2, (p=0.001)*

*W>P>R (p<0.05)*

*b)P.phoxinus (length)*

| | n | Median | 25% | 75% |
|---|---|---|---|---|
| Riffles | 70 | 5.7 | 4.4 | 6.7 |
| Pools | 312 | 4.8 | 2.8 | 6.2 |
| CWD | 196 | 5.7 | 4.6 | 7 |

*H = 24.812, df =2, p <0.001*

*W>P, R>P, R/P NS ( p<0.05)*

*c) C.gobio (length)*

| | n | Median | 25% | 75% |
|---|---|---|---|---|
| Riffles | 933 | 4.2 | 3.2 | 4.9 |
| Pools | 705 | 4.4 | 3.7 | 5 |
| CWD | 84 | 4.6 | 3.8 | 5 |

*H = 36.485, df =2, P<0.001*

*W>R,P>R, P/R NS*

*d) Lampetra (weight)*

| | n | Median | 25% | 75% |
|---|---|---|---|---|
| Riffles | 79 | 2.6 | 1.7 | 3.6 |
| Pools | 508 | 2 | 1.1 | 2.9 |
| CWD | 53 | 2.5 | 1.5 | 2.9 |

*H = 13.688, df = 2, p =0.001*

*R>P, R/W, W/P NS.*

*e) N.barbatulus* (length)

| | n | Median | 25% | 75% |
|---|---|---|---|---|
| Riffles | 23 | 7.5 | 6.1 | 9.6 |
| Pools | 61 | 8.5 | 4.7 | 9.9 |
| CWD | 18 | 9.7 | 8.7 | 10.5 |

*H = 6.942, df=2, p =0.031*

W>P, W/R. P/R NS

*f) A.anguilla*

| | n | Median | 25% | 75% |
|---|---|---|---|---|
| Riffles | 11 | 46.4 | 19.3 | 77.8 |
| Pools | 55 | 59.4 | 39.4 | 85 |
| CWD | 45 | 71.7 | 38.1 | 119 |

*H=3.969,df=2, (p=0.137) NS*

*lengths as TL in cm, Weight in g*

*n=number of fish measured*

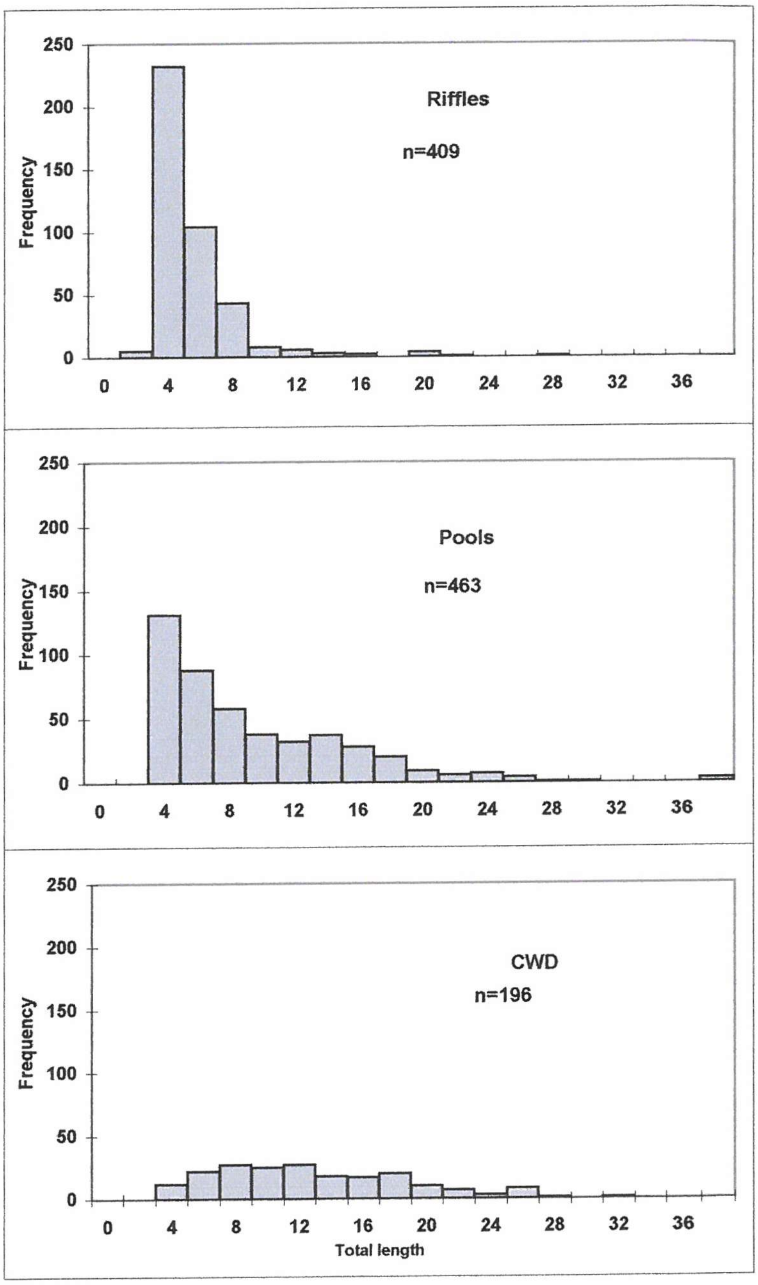


**Fig.5.4** Length-frequency distributions of all Salmonidae from riffles,pools and CWD habitat units in Highland Water streams New Forest. 1996-98. (n= number of fish measured)

normality and to a negative kurtosis. The ranges of sizes and sample variance were greatest in the pool samples.

The overall median length of minnows (*P. phoxinus)* was also significantly greater in CWD (5.7cm) and riffles (5.7 cm) than in pools (4.7 cm) (Table 5.7) mainly because the proportion of small fish in pools was higher (Fig. 5.5). Modal lengths were 4.4 cm, 2 cm and 6 cm for riffles, pools and CWD distributions respectively (Table 5.6). Length range was lower in riffles than pools or CWD. The length frequency distributions for riffles and CWD were both negatively skewed while that for pools was positive and biased toward larger fish.

The median length of bullheads (*C. gobio*) was significantly greater in the CWD and pool habitats than in the riffles (Table 5. 7) again because of the relatively higher frequency of larger fish (Fig.5.6). Modal lengths were 4, 4 and 5.5 cm TL in riffles, pools and CWD habitats respectively. Length-frequency distributions were positively skewed for riffles and pools but negatively for CWD though all three showed negative kurtosis. Because of the lack of very small fish there is some evidence that the electric fishing technique was selective for the individuals over 2cm.

Lampreys (*Lampetra* sp) were on average larger in riffles (Median = 2.6g) than in pools (2.0g) or CWD (2.2g) (Table 5.7). There was a relatively higher frequency of smaller individuals in pool samples. (Fig.5.7). Modal weight was 2.4g in riffles, 1.0g in pools and 2.3g in CWD habitats. Smaller lampreys appeared to be associated with silt and sand deposition whether in pools or the margins of other habitats. There was a positive skew to the pool distribution (skewness = 0.41) and negative skew to the riffle and CWD distributions (-0.02, and -0.16 respectively) (Table 5.6).

The median length of stone loach (*N. barbatulus*) in the CWD habitat units was significantly greater than in riffles or pools (Table 5.7). Small fish were relatively more abundant in pools (Fig. 5.8). In both riffles and pools the fish showed a bimodal length-frequency distribution which was not evident in the CWD habitat. Modal lengths were greater in riffles (10.7 cm) and CWD (10.5 cm) than

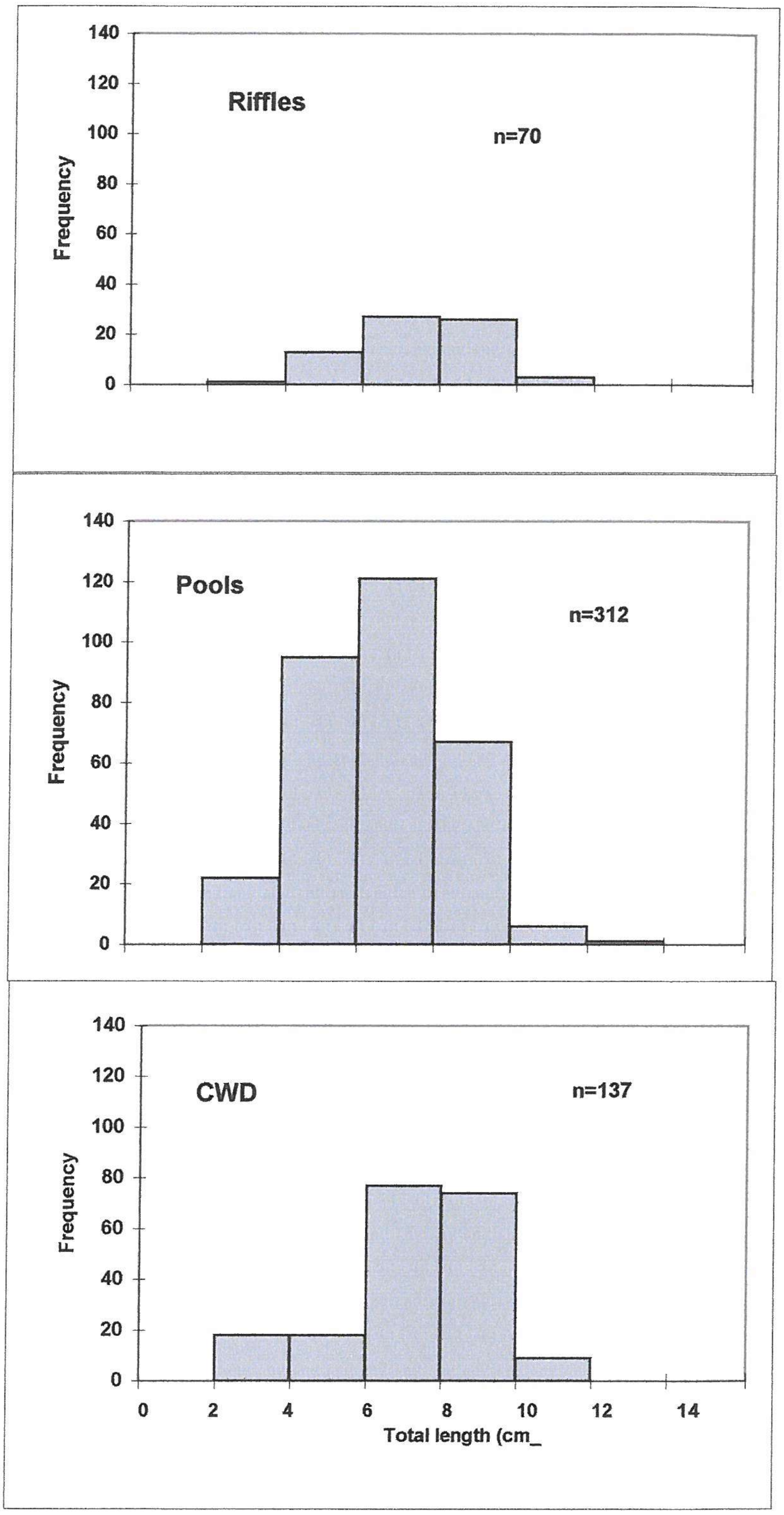


**Fig.5.5** **Length-frequency distributions for all minnows (*P.phoxinus*) from riffles, pools and CWD habitat units in Highland Water streams. 1996-98.** ***(n= number of fish measured)***

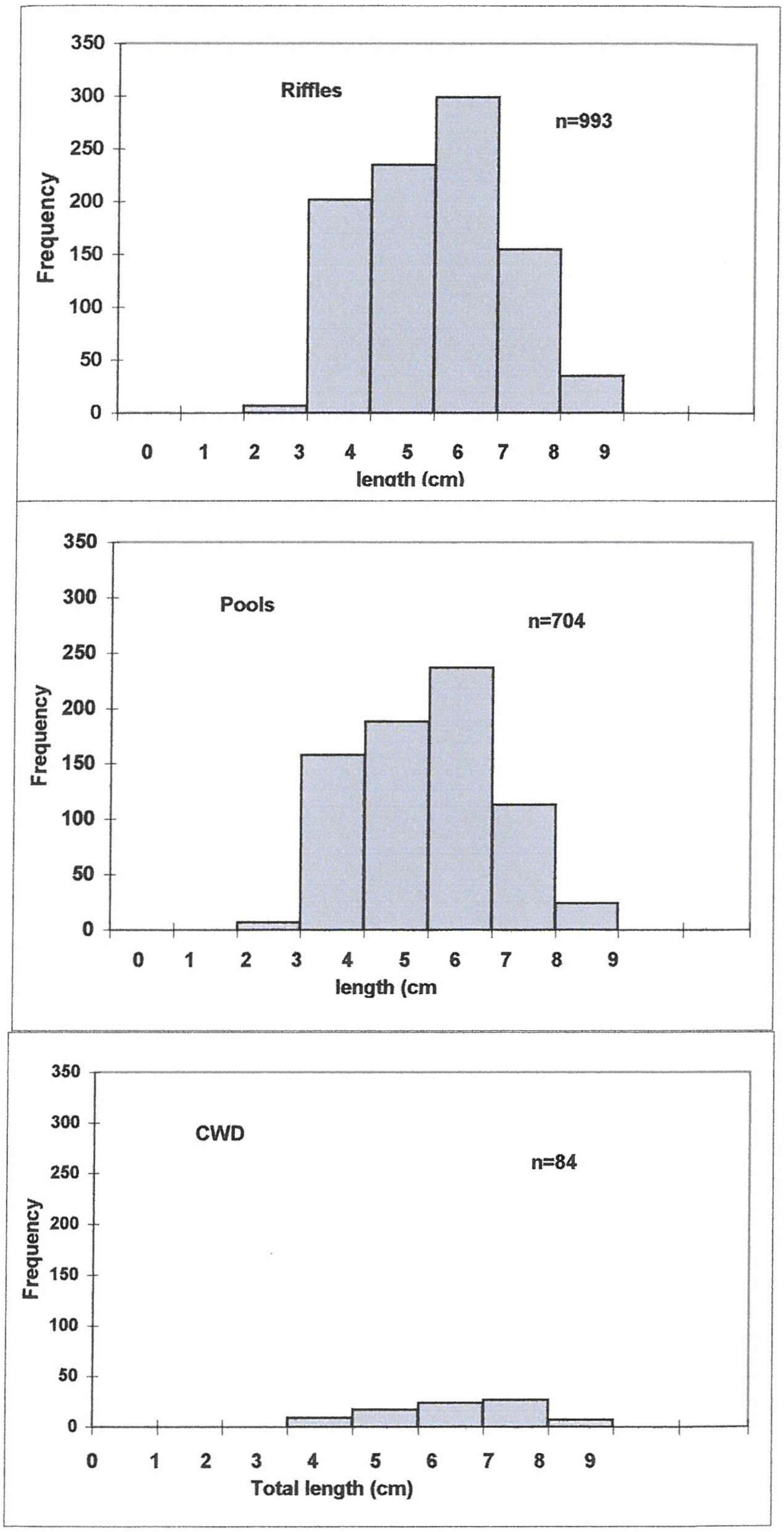


**Fig.5.6** **Length-frequency distributions of all bullheads (*C.gobio*) from riffles, pools and CWD habitat units in Highland Water streams. New Forest, 1996-98**

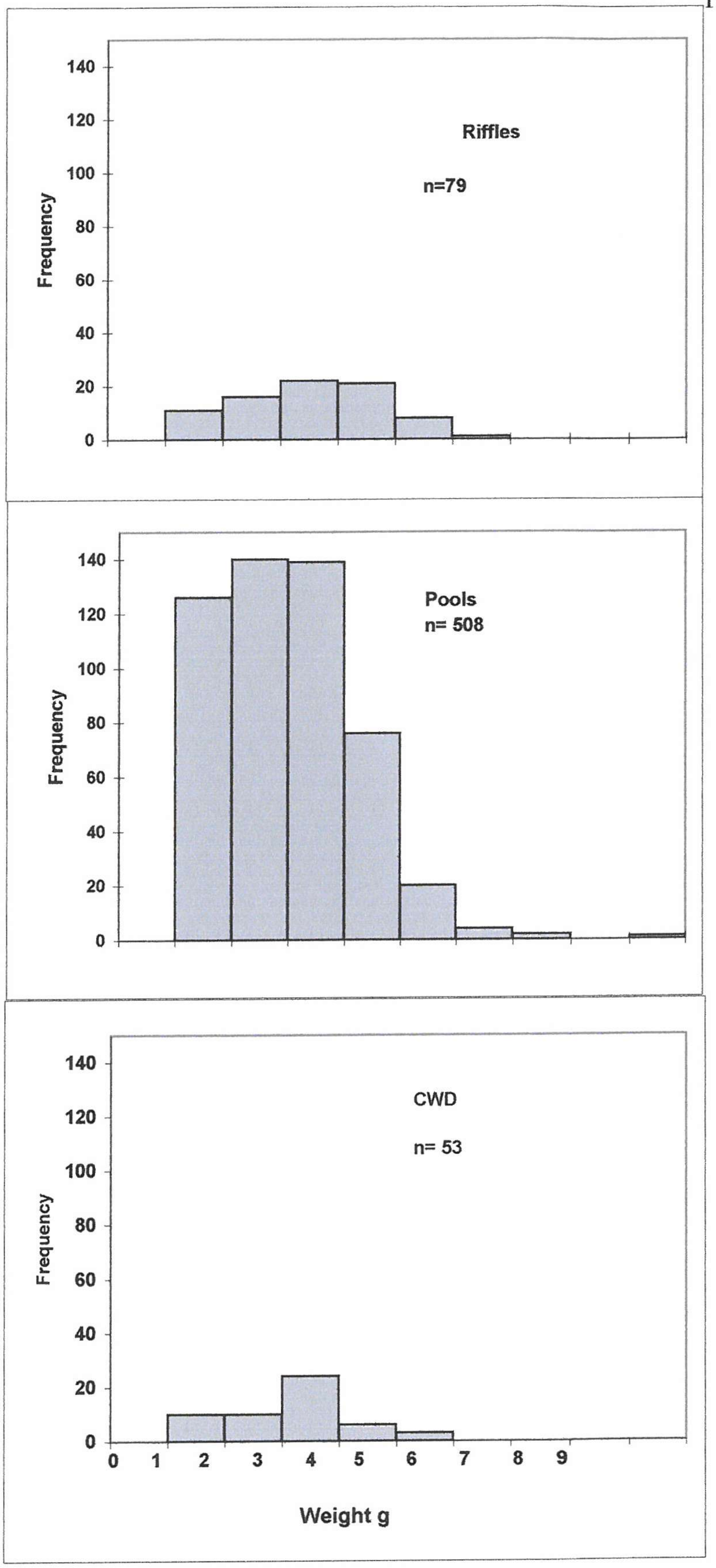


**Fig. 5.7** Weight-frequency distributions for all lampreys (*Lampetra* sp.) from riffles, pools and CWD habitat units in Highland Water streams. New Forest. 1996-98

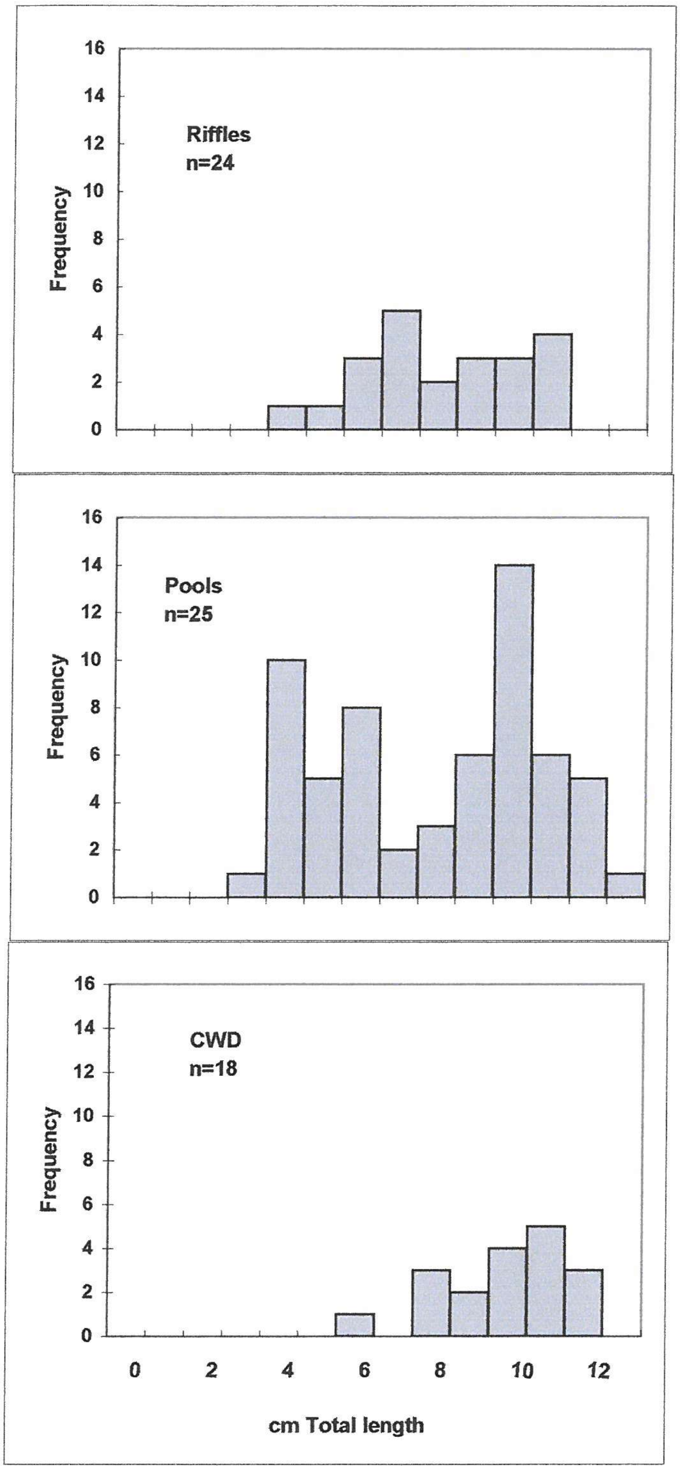


**Fig.5.8** **Length-frequency distributions for all stone loach (*N. barbatulus*) from riffles,pools and CWD habitat units in Highland Water streams New Forest. 1996-98**

in pools (9 cm) as a result of this relative abundance of smaller individuals in pools. Both the sample variance and standard deviation were greatest in pools. Very few individuals less than 6cm long were caught in the CWD accumulations.

There was no significant difference between the median weights of eels (*A. anguilla*) from riffles, pools and CWD habitats (Table 5.7) though they did increase from 46.4g in riffles to 59.4g in pools and 71.7g in CWD accumulations. There were, however, clear differences in the range of sizes between riffles and the other two habitat types (Fig.5.9). Eels in riffles and pools were mainly less than 120g. The size distribution in all three habitat types was essentially bimodal but this was clearer in the CWD habitats. Individuals larger than 160g were relatively scarce in riffles and pools but more common in CWD accumulations. The largest eel was over 2kg in weight, caught in a pool habitat. All distributions were positively skewed (Table 5.6). The largest values for skewness, kurtosis and sample variance were for the CWD frequency distribution.

There were marked overlaps in size distributions of all species in the various habitats but some identifiable trends which were apparently associated with the accumulation of CWD. The main trend was the tendency for larger individuals of most species to be caught in the CWD accumulations though pools carried the greatest range of sizes. The possible reasons for the differences in size distributions are discussed later in this chapter.

### 5. 3. 2. Species abundance in relation to physical structure and CWD

The physical variation in the stream channel was mostly explained by DCA Axes I & II using physical variables (see Chapter 4, Table 4.11). To explain the relationship between the physical variables and the abundance of individual species Spearman rank correlation was applied to densities and biomass of each species and the category scores from Axis I and Axis II of the DCA on the physical variables from Chapter 4. Axis I is heavily weighted positively by depth, CWD, silt and twigs and distance downstream (depositional) with a strong negative component for percentage gravel and width-depth ratio

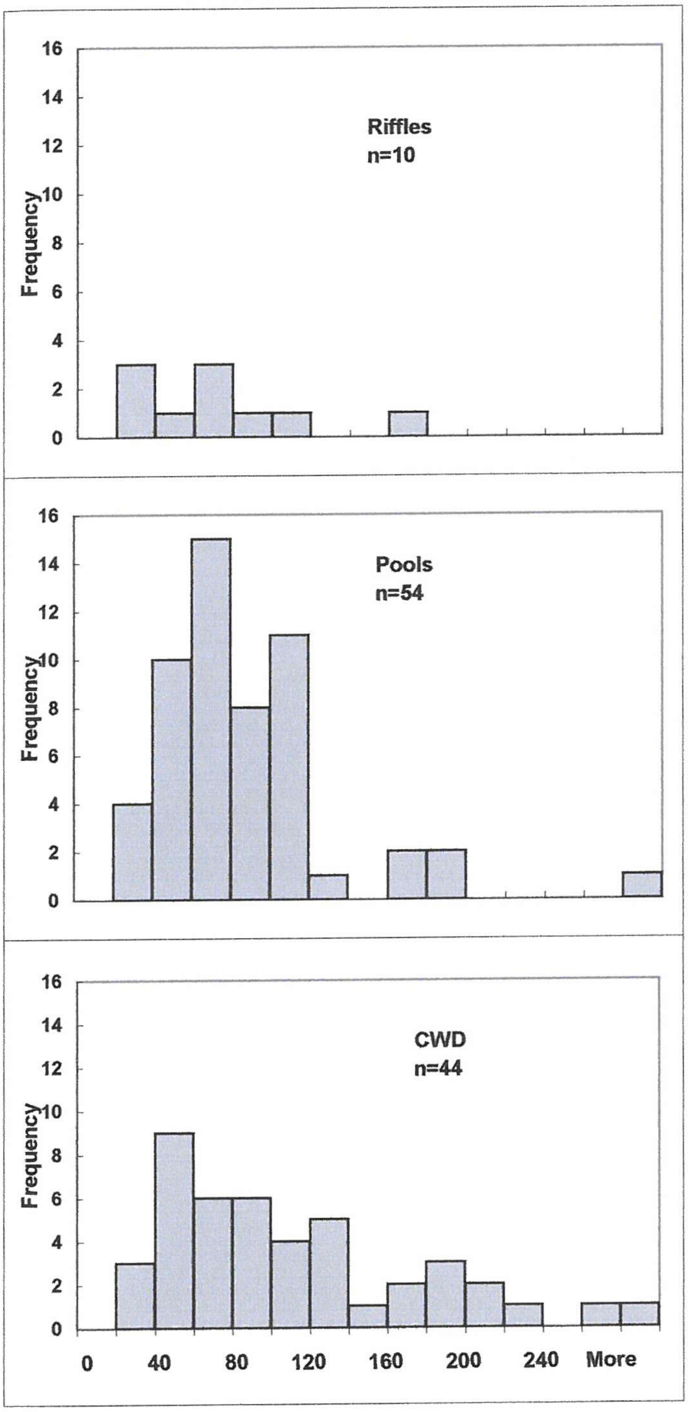


**Fig. 5.9** **Weight-frequency distributions of all *A.anguilla from* riffles,pools and CWD habitat units in Highland Water streams. New Forest, 1996-98**

(erosional). Axis II was heavily weighted positively by softer sediments and organic materials such as leaves and twigs. The strong negative weighting was by the rarer features and substrates, twigs, clay and sand.

Results of the Spearman rank correlation calculations are shown in Table 5. 8. There was no correlation between the axes and salmonid density but the correlations with biomass were significant ($p < 0.001$, Axis I; $p = 0.002$ Axis II). This is discussed in more detail in Chapter 7. Significant positive correlations also occurred between Axis I and the densities of minnows, lampreys and eels. In contrast, there was a significant negative correlation with Axis I and densities of bullheads. For biomass data significant positive correlations were between Axis I, minnows and eels and negative correlations between Axis I and bullheads.

Although there was strong cross correlation between the physical variables (see Chapter 4) gravel, CWD and depth were important components of DCA Axis I. Using the calculated volumes of CWD in each habitat unit (see Chapter 3) Spearman rank correlation showed that there were also highly significant correlations between wood volumes, gravel occurrence, maximum depth and fish abundance on the within-reach scale (Table 5. 9). The highest positive correlations were salmonid biomass with CWD, minnow density and biomass with CWD and density and biomass with depth. The strongest negative correlations were minnow density and biomass with gravel and with depth, and bullhead density and biomass with CWD and depth. Thus there were clear differences on the within-reach scale in the abundance of species which could be related to both substrate composition and depth of the individual habitat units.

### 5. 3. 3. Effects of dam removal and retention on fish abundance

Chapter 4 included a preliminary analysis of the effects of the removal and retention of CWD dams on the physical structure and substrates in Highland Water streams following a series on winter spates. Fish were sampled during the same period (May 1997-February 1998) on the same dates as the physical measurements were made. The CWD dam at Site 16 was breached in December 1997 and this was followed by heavy spates (see Chapter 3 Fig. 3.2, Chapter 4

Table 5.8 Spearman rank coefficients for the correlation of fish density and biomass to Axis I and Axis II of the DCA on physical variables at the habitat unit scale (see Chapter 4, Table 4.11 for Axis component scores)

Density

| Species | AXIS I | AXIS II |
|---|---|---|
| ***S.trutta*** | -0.007 | -0.04 |
| ***P.phoxinus*** | ***0.534*** | **-0.279** |
| ***C.gobio*** | ***-0.569*** | **0.24** |
| ***Lampetra sp*** | **0.234** | -0.011 |
| ***N.barbatulus*** | 0.068 | 0.005 |
| ***A.anguilla*** | ***0.379*** | -0.09 |
| ***All fish*** | -0.11 | 0.076 |

*Biomass*

| Species | AXIS I | AXIS II |
|---|---|---|
| ***S.trutta*** | ***0.511*** | **-0.279** |
| ***P.phoxinus*** | **0.293** | -0.186 |
| ***C.gobio*** | ***-0.434*** | 0.078 |
| ***Lampetra sp*** | 0.187 | 0.113 |
| ***N.barbatulus*** | 0.136 | 0.057 |
| ***A.anguilla*** | **0.432** | -0.11 |
| ***All fish*** | ***0.419*** | **-0.276** |

***bold italics, p<0.001, bold, p<0.05, Ordinary type, not significant***

**Table 5. 9 Spearman rank coefficients for the correlation of fish abundance with the primary physical variables on the within-reach scale in Highland Water streams. New Forest. 1996-98**

| Species | CWD |  | Gravel |  | Max.depth |  |
|---|---|---|---|---|---|---|
|  | Density | Biomass | Density | Biomass | Density | Biomass |
| ***S.trutta*** | -0.07 | ***0.49*** | -0.05 | ***-0.42*** | ***-0.34*** | ***0.36*** |
| ***P.phoxinus*** | ***0.48*** | **0.25** | ***-0.51*** | ***-0.56*** | ***0.55*** | ***0.58*** |
| ***C.gobio*** | ***-0.49*** | ***-0.33*** | ***0.55*** | ***0.5*** | ***-0.65*** | ***-0.62*** |
| ***Lampetra*** | **0.19** | 0.15 | **-0.22** | **-0.19** | 0.12 | 0.12 |
| ***N.barbatulus*** | 0.09 | 0.05 | -0.03 | -0.03 | 0.03 | 0.04 |
| ***A.anguilla*** | ***0.3*** | **0.43** | ***-0.39*** | ***-0.39*** | **0.19** | **0.22** |
| ***All fish*** | -0.12 | ***0.44*** | 0.08 | ***-0.39*** | -0.34 | ***0.29*** |

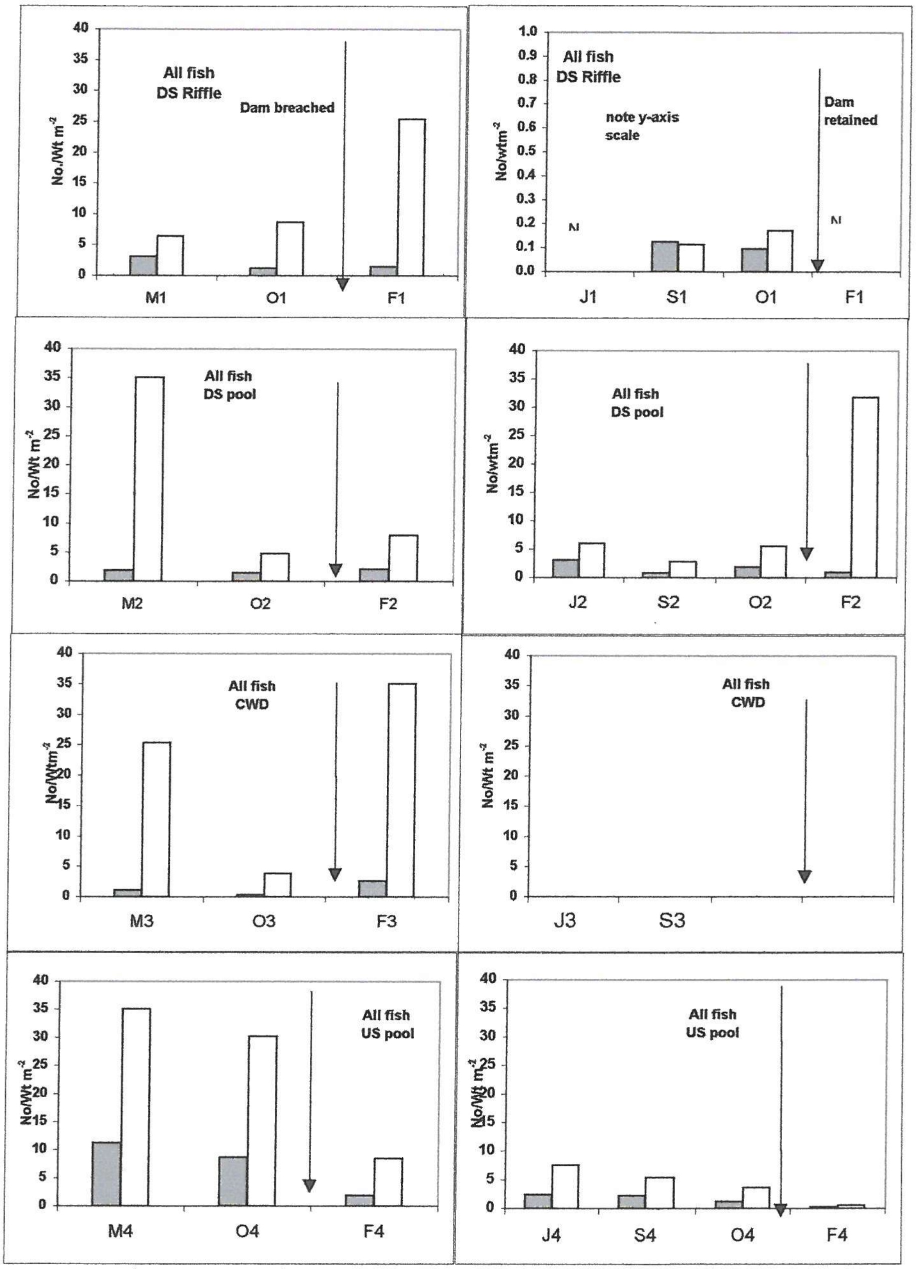
***Significance, bold italics- p<0.001, bold-p =<0.05, ordinary type not significant***

Fig. 4.9). During the same period the CWD dam at site 22 remained in position despite the spates. Both sites comprised four habitat units namely, a downstream riffle, downstream pool, CWD habitat unit and an upstream pool (see Chapter 4. Fig. 4.9). The overall effects of the spates were to cause the removal of softer sediments, (more pronounced where the dam was breached) and to reduce the depth of water in the CWD habitat of the breached dam. There were fewer marked changes in physical variables at Site 22 where the dam was retained.

There were also changes in the within-stream distribution and abundance of fish during the period though it is likely that some changes were seasonal and unrelated to the dams. Also, the period between the spates and sampling was up to several weeks and the very immediate effects of removal were not assessed. Differences between before and after breaching therefore involve a longish period of redistribution and equilibration

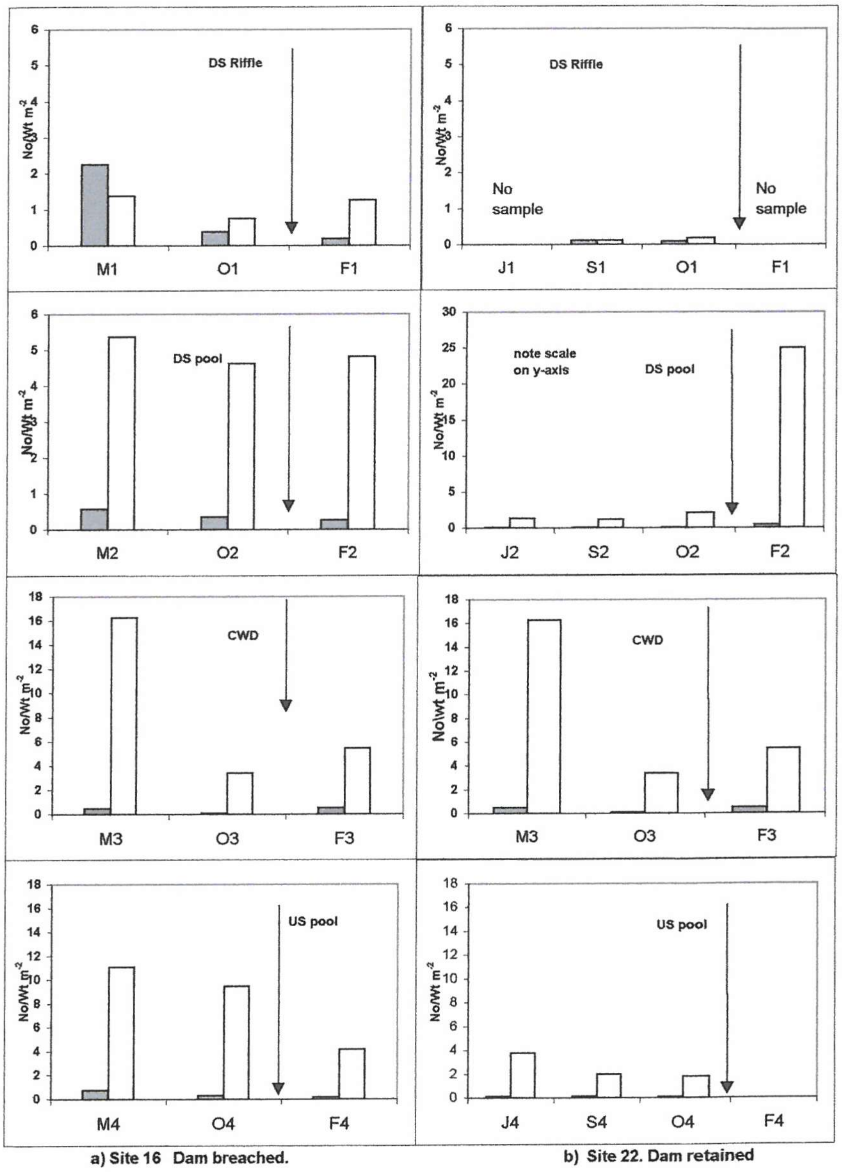
The abundance (density and biomass) of all fish (summed for all species) was greater below the breached dam (Site 16) and in the CWD habitat after the spates but both the upstream and downstream pools showed an overall decline (Fig. 5.10). Fish abundance declined, however, in both habitat units between May and October prior to the breaching and the spates possibly because of natural mobility and mortality. Declines in total abundance also occurred where the dam was retained (Site 22) though these were in the CWD habitat and in the upstream pool. Thus total fish abundance declined in both upstream pools despite the differences in dam permanence. The decline in salmonid density in the downstream riffle at Site 16 was most likely a result of natural mortality (Elliott, 1994) and the increased biomass in the downstream pool at Site 22 between October and February was because of one large sea-trout (Fig. 5.11). No salmonids were recorded from the upstream pool of Site 22 in February 1998 though they were still present at Site 16.

Density of minnows was lower upstream of the dam at Site 16 after it was breached but the same decline occurred at Site 22 (Fig. 5.12). At Site 16 the upstream pool was reduced in size and parts of it near the dam were reduced to a shallow riffle. At Site 22 the spate also resulted in a reduction of the size of the



**a) Site 16 Dam breached.** **b) Site 22. Dam retained**

Fig. 5.10 **Population density and biomass of all fish from two reaches of Highland Water streams before and after a CWD dam was breached and winter spates occurred.**
**Figures 5.11-5.16 refer to the same sites and the same period.**
**(open columns = biomass, shaded = density . Scales are matched where possible at least in pairs of samples)**
**Habitat units identified by letter and number (see Chapter 4, Fig. 4.9 for full explanation)**
**The vertical arrows on Figs. 10-16 indicate the period in which the dam was breached at Site 16.**
**Months, M= May, O = October, F= february, J= July, S = September.**
***habitat units, 1 = DS riffle, 2 = DS pool, 3 = CWD habitat, 4 = US pool.*** ***(eg. M4 = May sample, US pool)***

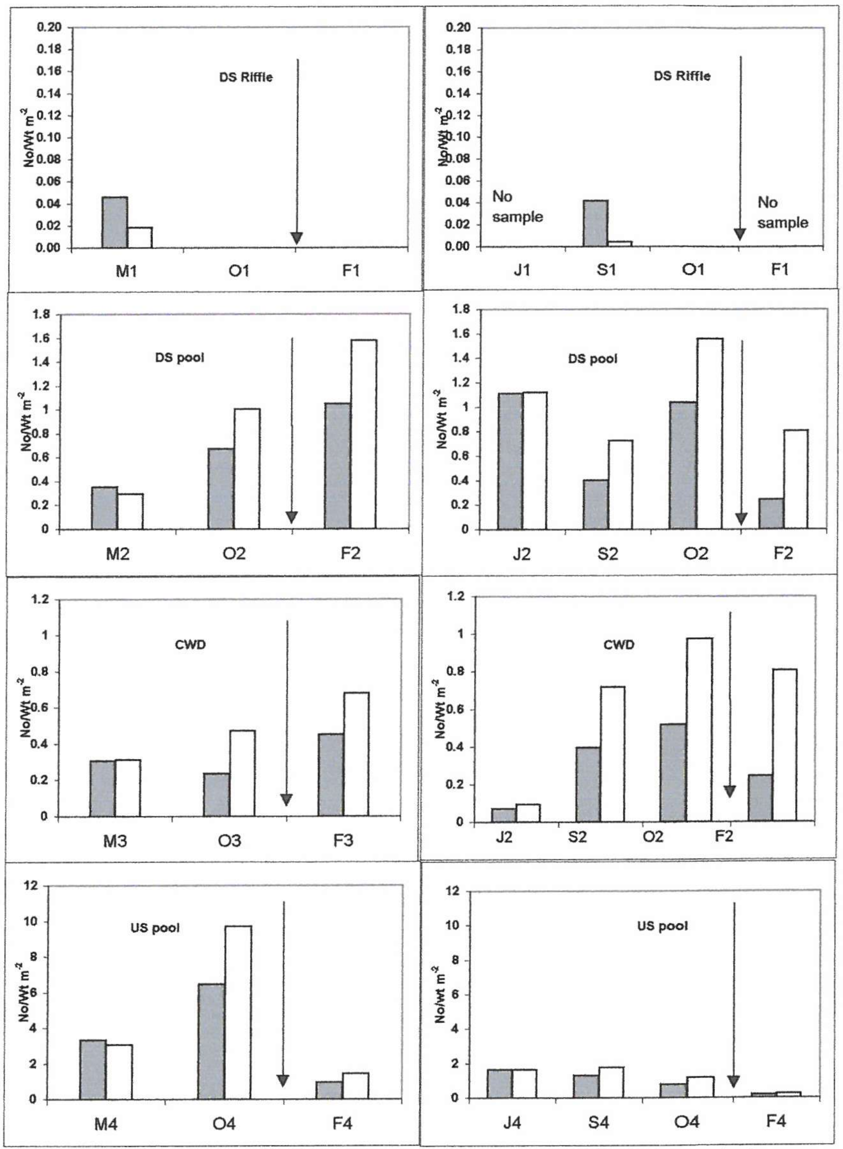


a) Site 16 Dam breached.

b) Site 22. Dam retained

**Fig.5.11 Population density and biomass of Salmonidae in two reaches of Highland Water streams before and after winter spates and dam breaching at Site 16.**

(see Fig.5.10 for key to codes and scales)



a) Site 16 Dam breached. b) Site 22. Dam retained

**Fig.5.12** **Population density and biomass of minnows (*P.phoxinus*) at two sites on Highland Water streams.**

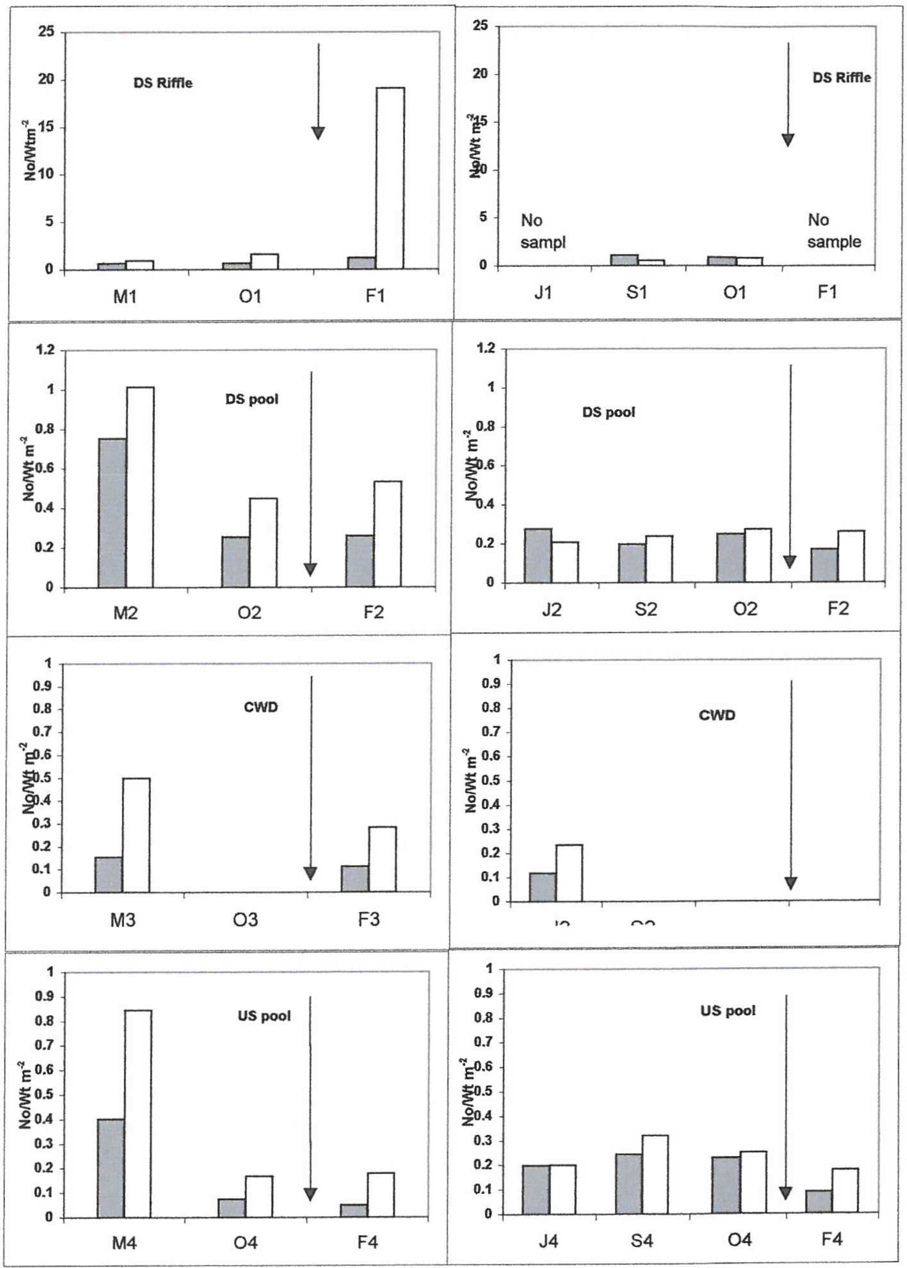
*(see Fig. 5.10 for sample codes and scale notes)* *(Scales are comparable for each pair of samples)*

upstream pool and an encroachment of a gravel riffle. Thus the reduced density of minnows was most likely related to a reduction in pool habitat. There was an increase in the density of minnows in the CWD habitat and downstream pool at Site 16 but not at Site 22 but there was no parallel increase in pool area.

Both the density and biomass of bullheads (*C. gobio*) were relatively stable in all habitat units before and after the breaching and the spates at both sites (Fig. 5.13) though there were some changes before these events. Lampreys (*Lampetra* spp) were less abundant in both upstream pools after the spates but there were increases in the CWD habitats suggesting some displacement and redistribution at the scale of the habitat unit (Fig. 5.14). The changes were probably related to the scouring of the softer sediments caused in both habitat units as a result of the spate though there was evidence of a gradual decline before the spates began. Abundance in the downstream pools showed relative stability.

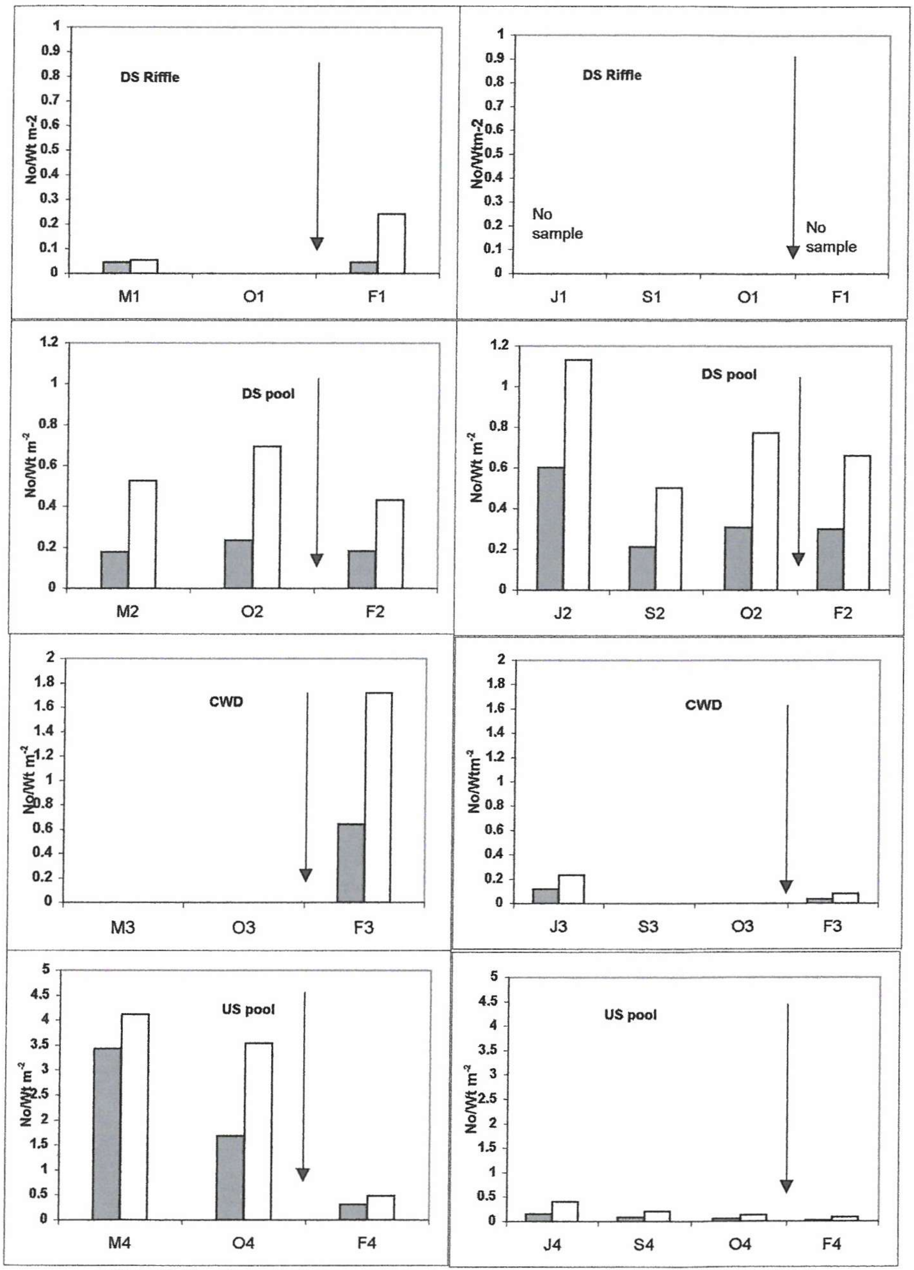
The abundance of stone loach (*N. barbatulus*) was relatively stable in the downstream pools at both Site 16 and 22 (Fig. 5.15). Both density and biomass fluctuated in the CWD habitats but there was no apparent pattern. There was an increase in the upstream pool at Site 16 but not at Site 22. The fluctuations in abundance of eels (*A. anguilla*) showed no obvious trends (Fig. 5.16). As with the stone loach, eels were sporadic in occurrence at all sites.

The effect of the spates and dam removal was also examined on the reach scale by comparing the data for all fish species for all four habitat units at each site. Total fish densities and biomass for the whole reach (all four habitat units combined) showed no marked effects of either dam removal or retention. There was a decrease in density at Site 16 from 4.4 fish $m^{-2}$ in May 1997 to $2.9m^{-2}$ in October and $2.0m^{-2}$ after dam removal. Total biomass estimates were 18.9, 11.7 and 14.3 $gm^{-2}$ respectively. At Site 22 where the dam was retained, density values for the whole reach for July, September, October and February were 2.6, 5.2, 4.7 and 1.7 fish $m^{-2}$. Biomass values were 9.9, 28.8, 22.8 and $35.9gm^{-2}$ respectively. The pattern for both sites was similar in that densities decreased slightly but biomass increased after the spates. There was evidence, therefore, of some redistribution of fish on the within-reach scale but little effect on the reach



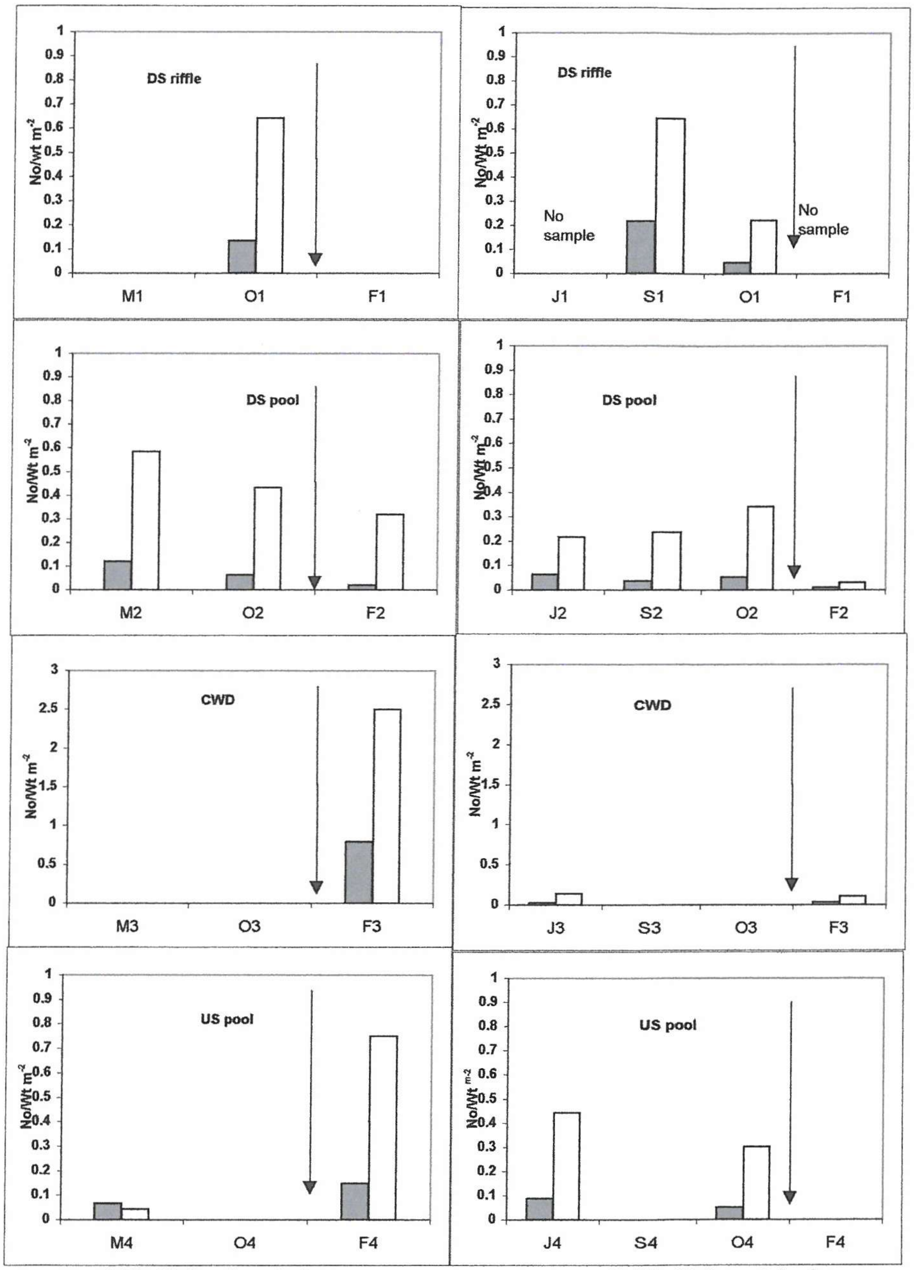
**a) Site 16 Dam breached.** **b) Site 22. Dam retained**

**Fig.5.13 Population density and biomass of bullheads (*C.gobio*) from two reaches on Highland Water streams.** **(See Fig.5.10 for sample codes and scale notes)**



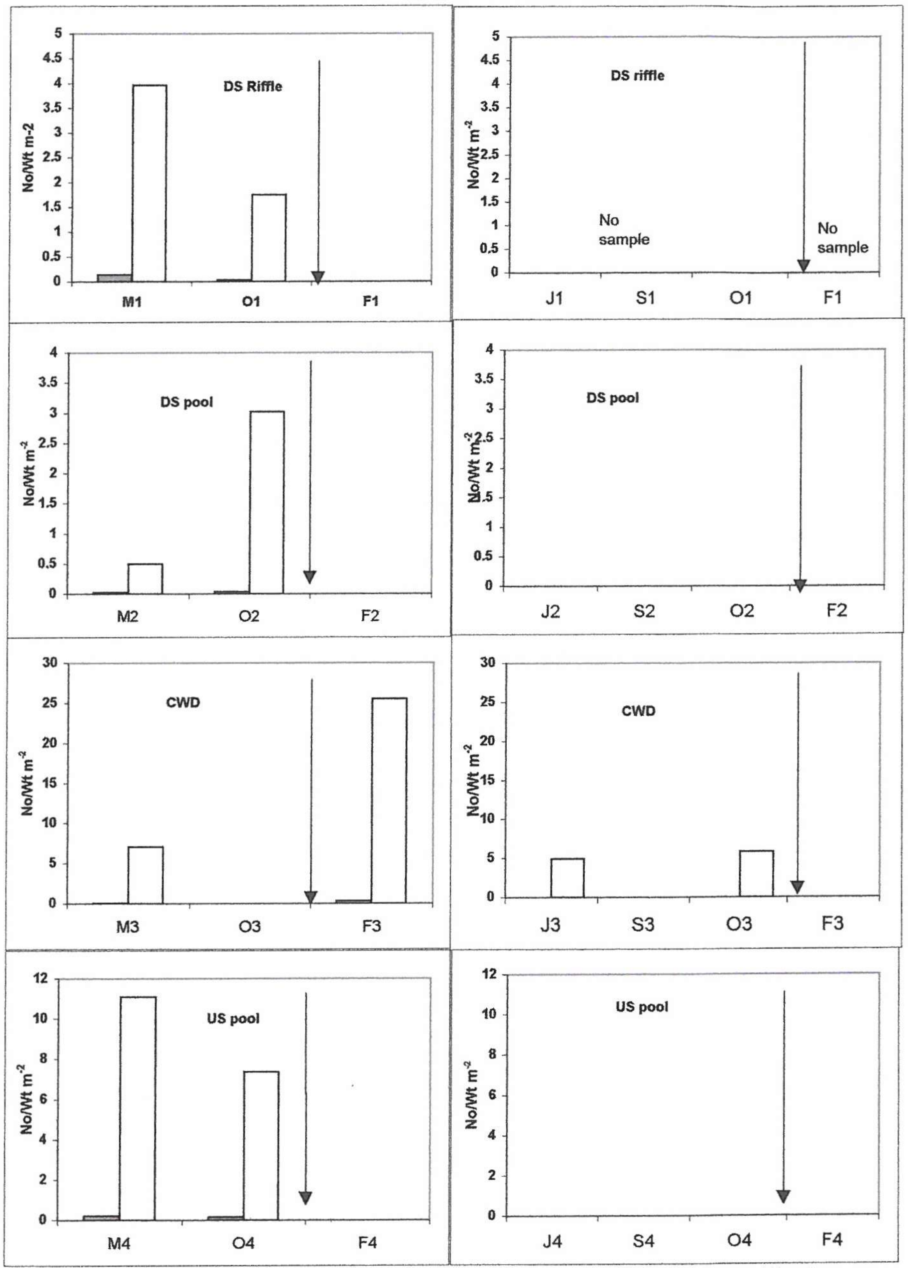
**a) Site 16 Dam breached.** **b) Site 22. Dam retained**

**Fig.5.14** **Population density and biomass of lampreys *(Lampetra sp (planeri?)* at two sites on Highland Water streams.** ***(See Fig. 5.10 for sample codes and scale notes)***



**a) Site 16 Dam breached.** **b) Site 22. Dam retained**

**Fig.5.15** **Population density and biomass of stone loach (*N.barbatulus*) at two sites on Highland Water streams.** ***(See Fig.5.10 for sample codes and scale notes)***



a) Site 16 Dam breached. b) Site 22. Dam retained

**Fig.5.16** **Population density and biomass of eels (*A. anguilla*) at two sites on Highland Water streams.** ( See Fig.5.10. For sample codes and scale notes) scale either where the dam was removed or retained. Following this preliminary analysis further studies of the removal of CWD dams on fish are in progress (Langford & Baker, *unpublished data*)

## 5. 3. 4. Fish abundance in relation to physical variables and CWD on the reach scale

Using the scores from Axis I and Axis II of the DCA of physical variables summed for each reach (see Chapter 4 Table 4.12), Spearman rank correlation (Table 5. 10) showed significant negative relationships between Axis I and densities of *Salmo* spp. ($p<0.001$), *C. gobio* ($p <0.001$) and biomass of *C. gobio* ($p <0.001$). There was also a significant relationship between Axis II and densities of *N. barbatulus* ($p = 0.006$). A significant positive correlation was shown by Axis I and densities of *P. phoxinus* ($p=0.42$). Axis I was positively weighted toward the depositional variables (twigs, leaves, silt, CWD) and negatively toward the primary erosional variable (gravel). Axis II was weighted toward the rarer variables, notably roots, weed and sand. Apart from this species, the increasing trend toward depositional habitats was not correlated with increases in fish density of biomass per unit area. Indeed the reverse was indicated for densities of *Salmo* spp. and both densities and biomass of *C. gobio.*

It has been shown in this chapter that the relative abundance of CWD and gravels is important in relation to fish abundance at the within-reach scale (habitat-units). To test their relative importance on the reach-scale, Spearman rank correlation analysis was carried out on reach density and biomass data using estimated total volumes of CWD and proportions of gravel in each reach. Reach data were calculated by summing point-contact data for all the habitat units in a sampling reach divided by the total wetted area of the reach. There were no significant correlations between the volumes of CWD (overall submerged matrix volume including interstices) in any reach and the population density of most species though the density of *N. barbatulus* was significantly correlated ($p<0.05$) (Table 5. 11). There were, however, significant negative correlations with densities of *Salmo* sp. ($p <0.001$) and *C. gobio* ($p<0.001$). There were also significant negative correlations with the biomass of *P. phoxinus* ($p=0.018$) and biomass of

**Table 5.10 Spearman rank coefficients for the correlation between fish abundance and Axis I and Axis II of the DCA on physical variables on the reach scale for Highland Water streams**

| **Species** | **Axis I** | **Axis II** |
|---|---|---|
| DENSITY | | |
| ***S.trutta*** | ***-0.802*** | -0.19 |
| ***P.phoxinus*** | **0.412** | 0.3 |
| ***C.gobio*** | ***-0.632*** | -0.199 |
| ***Lampetra*** | 0.144 | 0.198 |
| ***N. barbatulus*** | 0.0799 | **0.526** |
| ***A.anguilla*** | **0.0117** | -0.21 |
| BIOMASS | | |
| ***S.trutta*** | 0.091 | 0.0161 |
| ***P.phoxinus*** | -0.23 | -0.206 |
| ***C.gobio*** | ***-0.618*** | -0.14 |
| ***Lampetra*** | 0.271 | 0.167 |
| ***N. barbatulus*** | 0.129 | 0.283 |
| ***A.anguilla*** | 0.253 | -0.028 |

***bold italics, p<0.001, bold, p<0.05, ordinary type, p>0.05 (NS).***

Table 5.11 **Spearman rank coefficients for the correlation of fish abundance with the total amounts of CWD and gravel substrate in reaches of Highland Water streams. New Forest. 1996-98**

| Species | CWD | | Gravel | |
|---|---|---|---|---|
| | Density | Biomass | Density | Biomass |
| *S.trutta* | ***-0.74*** | -0.05 | 0.01 | -0.17 |
| *P.phoxinus* | 0.23 | **-0.33** | -0.24 | 0.19 |
| *C.gobio* | ***-0.6*** | ***-0.59*** | 0.07 | 0.05 |
| *Lampetra* | 0.1 | 0.21 | -0.18 | -0.09 |
| *N.barbatulus* | **0.35** | 0.15 | -0.05 | -0.12 |
| *A.anguilla* | **-0.32** | -0.09 | -0.03 | -0.13 |

***Significance: bold italics,p<0.001, bold, p<0.05***
***ordinary type, not significant, p>0.05***

*C. gobio* ($p < 0.001$). There was no overall correlation between the total abundance of fish and the amount of CWD in a reach either for density or biomass ($p > 0.05$). There were no significant positive or negative correlations between any species and the amounts of gravel recorded in a reach despite the importance of this substrate at the habitat unit level.

### 5. 3. 5. Comparisons with other New Forest streams

The streams used for the comparisons were Dockens Water and Ober Water, both with reduced riparian tree cover and low CWD loadings (see Chapter 3 and Chapter 4). For these comparisons only reaches sampled in the same seasons were used. This included 21 reaches in the wooded streams and 8 reaches in the more open streams. As Chapter 4 shows, the average widths and depths of the reaches compared were similar in the two types of stream, though wooded streams showed some greater depth variation mainly as a result of the effects of larger CWD accumulations. The reaches sampled in the open streams were mainly between 1 and 4km from the sources approximately in the same relative positions as the sites in the Highland Water streams.

The relative proportions of the various land uses in the riparian zones are shown in Chapter 4 (Table 4.2). The difference in tree cover between the streams is clear. The streams are described as " wooded" and "open" in the following text and in the figures and tables. For the comparisons the fish data from the individual habitats units were pooled to provide reach data. Reach data were considered to be more realistic for comparisons between streams so that the effects of within-reach variations would be smoothed out. Riffle-pool-CWD habitat units were not as clearly defined in the more open streams especially where regular maintenance is carried out.

The data for all fish species, summed, were analysed using data from individual streams (Fig. 5.17). There was no significant difference in mean fish numbers ($F= 0.985$, $p = >0.05$) but initial tests showed that there were significant differences in fish biomass between streams (KW, $H = 13.74$, $df= 4$, $p= 0.008$). Replicates were too few to carry out pairwise tests. Therefore the data for the two

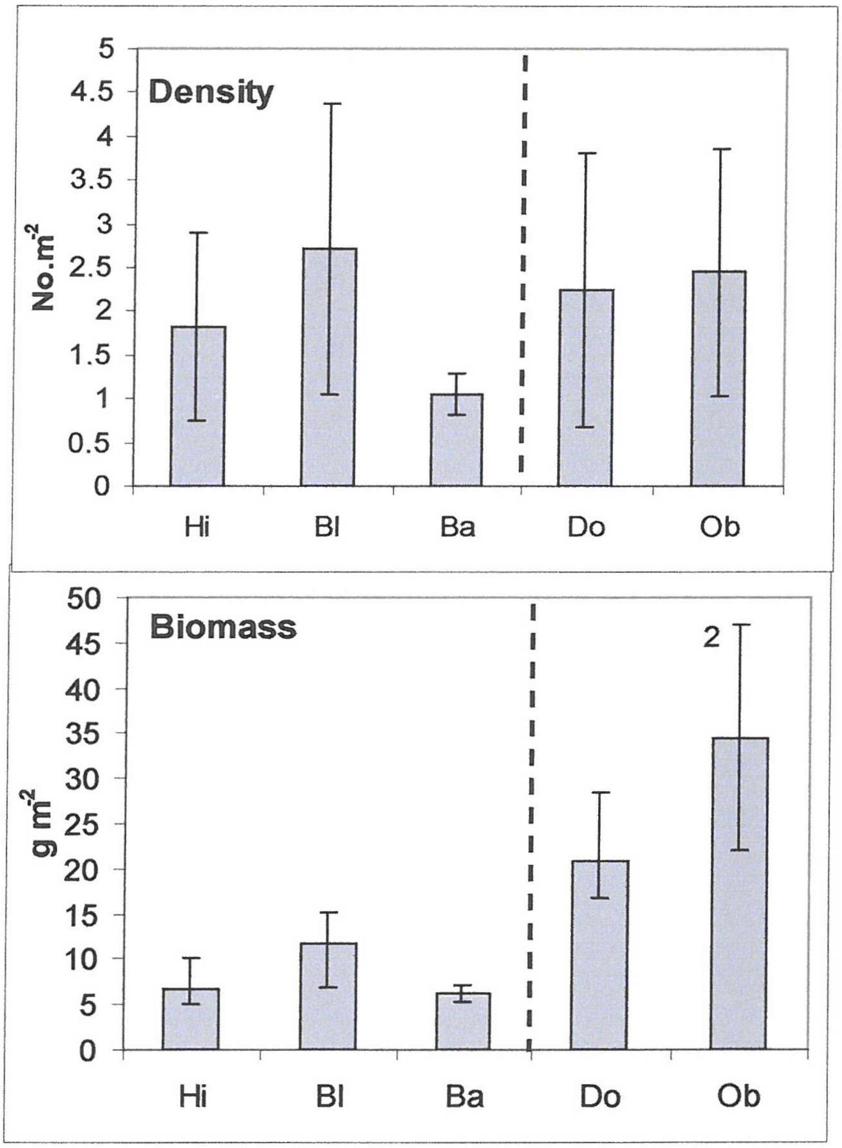


**Fig. 5.17 Comparisons of fish density and biomass (all fish summed) in wooded and open streams in the New Forest.**

***Numbers denote number of reaches sampled***

***Hi= Highland, Bl = Bratley, Ba = Bagshot, Do = Dockens, Ob = Ober***

***n = Hi 21, Bl,4, Ba,2, Do,6, Ob,2***

types of streams were pooled to give two groupings. This improved replication sufficiently for statistical comparisons.

These comparisons confirmed that there was no significant difference ($p>0.268$) in fish density between the two types of stream (Fig. 5.18, Table 5.12). In contrast total fish biomass (as $gm^{-2}$) was significantly different overall ($p<0.05$) with the open streams carrying an average of almost $22gm^{-2}$ compared with $7.5gm^{-2}$ for the wooded streams. There were also apparent differences between the densities and biomass of the six common individual species (Fig. 5.19, Table 5.12) though only the abundance of *N. barbatulus* was significantly different for both ($p=0.03$, 0.01). Open streams contained a significantly higher biomass of *P. phoxinus*, ($p=0.01$) *N. barbatulus* ($p=0.03$) and *A. anguilla* ($p=0.007$). Salmonid abundance was not significantly different ($p>0.05$). The higher abundance of *N. barbatulus* was mostly a result of high densities at one site on Dockens Water (Mann & Orr, 1969; Mann, 1971; Langford, 1996).

In addition to the six species of fish in the wooded streams, samples from the open streams also contained additional species, viz. Chub, (*Leuciscus cephalus*), sticklebacks, (*Gasterosteus aculeatus*) and roach, (*Rutilus rutilus*). The distribution of these species is discussed further in Chapter 6. In the open stream, the addition of *L. cephalus* to the population was probably responsible for the significant increase in biomass. The fish were typically large but in low numbers. The median number $m^{-2}$ was 0.1 (quartiles 0.08-0.122) and the median biomass as $g\ m^{-2}$ was 6.69 (quartiles 3.28-7.26). This biomass was considerably greater than any other species recorded during the programme and the species is clearly very successful in the unshaded New Forest streams. Sticklebacks (*G. aculeatus)* on the other hand were numerous but with very low biomass. The highest numbers were caught in Dockens Water, with median density of $0.21m^{-2}$ (0-0.385) and biomass of $0.1gm^{-2}$ (0-0.17). Further species have been discovered in the upper reaches of the Ober Water in the most recent samples including bream (*Abramis brama* (Linnaeus 1758)) and rudd (*Scardinius erythrophthalmus* (Linnaeus 1758)) (Quinlan, *unpublished data*).

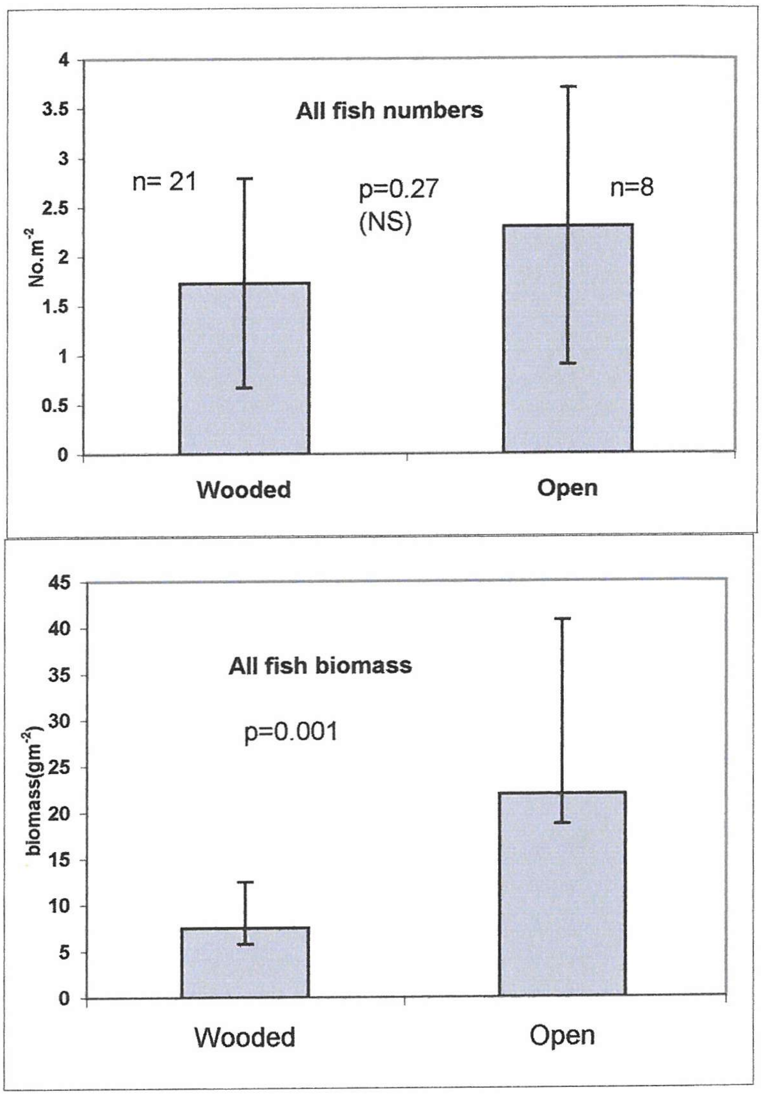


Fig.5.18 Densities and biomass of all fish (summed) from comparable samples in open and wooded New Forest streams. 1996-2000

Table 5.12 **Comparisons of the density and biomass of all fish species in wooded and open streams in the New Forest. (wooded streams are Highland,Bagshot and Bratley Waters open are Ober Water, Dockens Water)**

| | | | WOODED | | | OPEN | | | |
|---|---|---|---|---|---|---|---|---|---|
| **Species** | **Test** | **Variable** | **Mean/Med** | **25%(SD)** | **75%** | **Mean/Med** | **25%** | **75%** | **p(sig)** |
| ***S.trutta*** | M-W | density | 0.119 | 0.08 | 0.53 | 0.16 | 0.09 | 0.17 | 0.49(NS) |
| ***S.trutta*** | t | biomass | 3.82 | 1.92 | 4.92 | 5.28 | 1.09 | 8.02 | 0.671(NS) |
| ***P.phoxinus*** | M-W | density | 0.72 | *0.572* | <> | 1.15 | *1.01* | <> | 0.171(NS) |
| ***P.phoxinus*** | M-W | biomass | 0.551 | 0.03 | 0.113 | 1.2 | 0.57 | 2.44 | 0.01 |
| ***C.gobio*** | M-W | density | 0.215 | 0.08 | 0.4 | 0.04 | 0 | 0.28 | 0.28 (NS) |
| ***C.gobio*** | M-W | biomass | 0.212 | 0.11 | 0.8 | 0.081 | 0 | 0.4 | 0.90(NS) |
| ***Lampetra*** | t | density | 0.26 | 0.26 | <> | 0.23 | 0.23 | <> | 0.91 (NS) |
| ***Lampetra*** | t | biomass | 0.53 | 0.45 | <> | 0.52 | 0.51 | <> | 0.95 (NS) |
| ***N.barbatulus*** | M-W | density | 0 | 0 | 0.04 | 0.05 | 0.01 | 1.4 | 0.03 |
| ***N.barbatulus*** | M-W | biomass | 0 | 0 | 0.01 | 0.15 | 0.04 | 2.25 | 0.02 |
| ***A.anguilla*** | t | density | 0.02 | 0.02 | <> | 0.05 | 0.03 | <> | |
| ***A.anguilla*** | M-W | biomass | 1.41 | 0.5 | 2.5 | 3.61 | 2.53 | 5.82 | 0.007 |
| ***L.cephalus*** | NA | density | NA | NA | NA | 0.103 | 0.08 | 0.122 | NA |
| ***L.cephalus*** | NA | biomass | NA | NA | NA | 6.69 | 3.28 | 7.26 | NA |
| ***G.aculeatus*** | NA | density | NA | NA | NA | 0.021 | 0 | 0.385 | NA |
| ***G.aculeatus*** | NA | biomass | NA | NA | NA | 0.1 | 0 | 0.07 | NA |
| ***Allfish*** | t | density | 1.73 | 1.06 | <> | 2.3 | 1.4 | <> | 0.268 (NS) |
| ***Allfish*** | M-W | biomass | 7.53 | 5.8 | 12.5 | 21.95 | 18.7 | 40.84 | <0.001 |

***t= t test, M-W = Mann-Whitney U test, NA = not applicable, <> = no SD value in previous column, significance level = p =0.05***

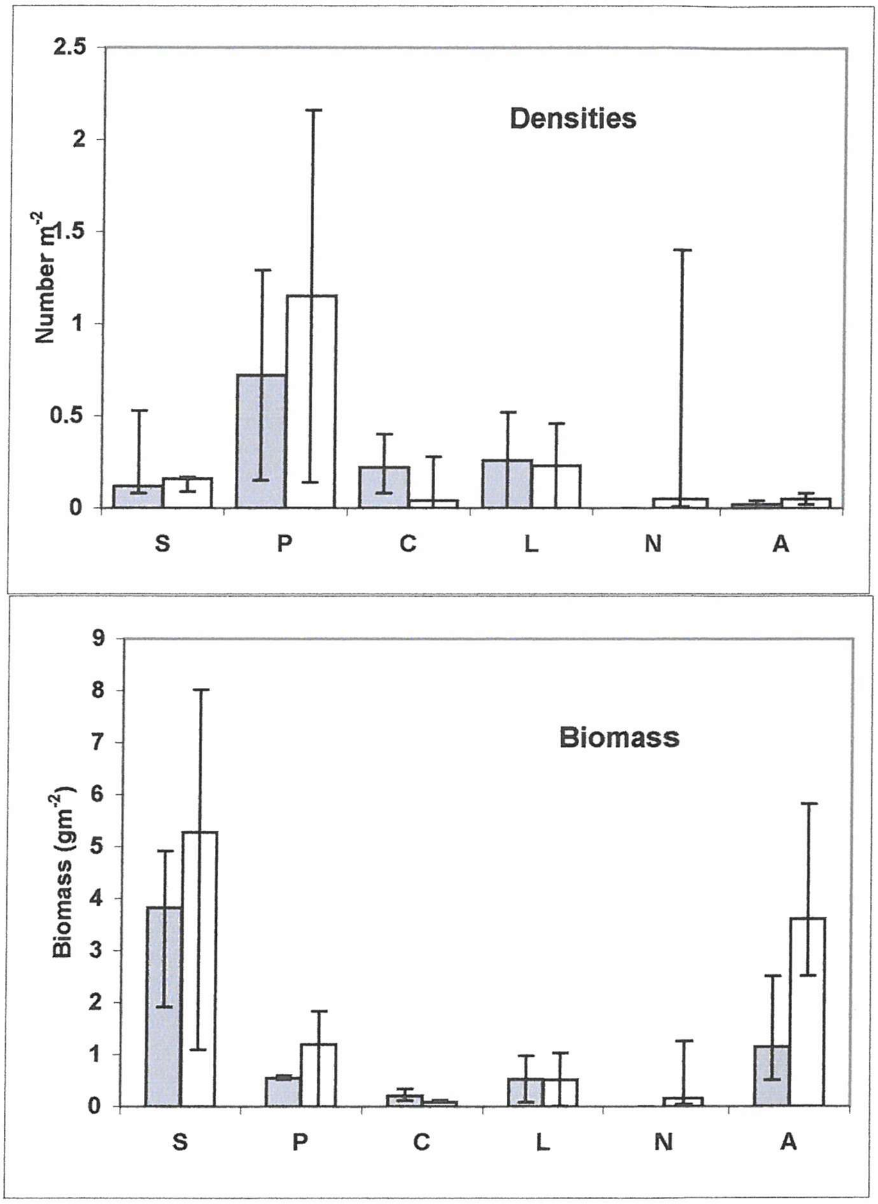


**Fig.5.19 Comparisons of density and biomass of fish species common to both wooded and open streams in the New Forest.** ***n= 21 wooded, n=7 open***
***S= S.trutta, P =P.phoxinus, C = C.gobio, L = Lampetra sp., N = N.barbatulus, A = A.anguilla*** ***(Significance of differences is shown in Table 5.12)***

Clearly the open streams carried a higher biomass of fish than the shaded streams in total together with additional species but very few of the species common to both stream types were significantly more or less abundant than in the wooded streams. Thus the abundance of *L. cephalus* and *G. aculeatus* as additional species does not seem to be associated with decreased abundance of other species particularly *S. trutta* and *P. phoxinus* as compared with the wooded streams. The apparent replacement of *C. gobio* by *N. barbatulus* in one reach of Dockens Water when the two were found to coexist lower down in other New Forest streams (Jones, *personal communication*) and in streams in other regions (e.g.Welton *et al.*, 1983, 1991) is the subject of continuing research (Quinlan, *unpublished data*)

### 5. 3. 6. Effects of using volume for within-stream abundance comparisons

Because fish live in a three dimensional environment, the within-reach scale abundance data for the Highland Water streams were also compared on the basis of densities and biomass per unit of standing volume (Table 5.13). The data are compared with the results for areal abundance data in Table 5.3. In the Highland Water streams, the significance of differences in total densities and biomass of fish were different between areal and volumetric estimates though the ranking was the same. The results of Kruskal-Wallis tests were $H=34.1$, $df=2$, $p<0.001$ for volume-based comparisons of densities in riffles, pools and CWD habitats. For biomass the results were $H=7.515$, $df=2$, $p=0.023$. The difference in numbers was not significant for area but it was for volume. Conversely, the differences in biomass were significant for area but not for volume.

For the separate species the analyses by area and volume were broadly comparable for most species (Table 5.13). The main differences were that by area salmonid densities were significantly higher in CWD than in pools but by volume there was no significant difference ($p<0.05$). The differences in salmonid biomass between pools and CWD were significant for both area and volume ($p<0.05$). Thus it can be deduced that the CWD matrices were not carrying more individuals despite the three-dimensional structure but the individuals were larger.

**Table 5.13 Results of Kruskal-Wallis and Dunns**
**Multiple pairwise tests of abundance of species in relation to volume.**

| Species | | KW | *df* | p | Pairwise tests | | |
|---|---|---|---|---|---|---|---|
| ***S. trutta*** | density | 33.079 | 2 | p<0.001 | ***R>P**** | ***R>W*** | P/W |
| | biomass | 7.479 | 2 | p=0.024 | ***W>P**** | P/R | R/W |
| ***P. phoxinus*** | density | 28.325 | 2 | p<0.001 | ***P>R**** | ***W>R**** | P/W |
| | biomass | 30.215 | 2 | p<0.001 | ***W>R**** | ***P>R*** | W/P |
| ***C. gobio*** | density | 68.876 | 2 | p<0.001 | ***R>W**** | ***R>P**** | ***P>W**** |
| | biomass | 68.983 | 2 | p<0.001 | ***R>W**** | ***R>P**** | ***P>W**** |
| ***Lampetra*** | density | 12.367 | 2 | p=0.002 | ***P>R**** | ***P>R*** | P/R |
| | biomass | 16.213 | 2 | p<0.001 | ***P>R*** | ***P>W*** | W/R* |
| ***N.barbatulus*** | density | 2.537 | 2 | p=0.281 | NS | NS | NS |
| | biomass | 2.228 | 2 | p=0.328 | NS | NS | NS |
| ***A.anguilla*** | density | 13.053 | 2 | p=0.001 | ***W>R**** | P/W* | P/R* |
| | biomass | 14.893 | 2 | p<0.001 | ***W>R**** | P/W | P/R* |

**R=riffles P=pools W =CWD habitat units** ***Bold italics = significant (p<0.05)***
**** indicates same result as for areal data. Agreement is about 50% including NS values***

Eels were showed a greater biomass CWD in relation to area but not by volume again indicating that there were not higher densities in the matrices but that individuals were larger. Lampreys, minnows and bullheads data showed the same patterns of abundance in relation to within-stream habitat units whether area or volume were used. It is clear therefore that using a volumetric basis for assessing abundance can clarify the relative effects of structure though areal data are generally acceptable for most comparisons on the within-reach scale.

## 5. 4. DISCUSSION AND CONCLUSIONS

### 5. 4. 1. Within-stream variations in abundance in relation to CWD

There were clear variations in abundance and size-distribution of individual fish species at the within-reach scale in the Highland Water streams shown from using *a priori* selection of habitat units. There were also wide variations in the abundance of any species with the habitat unit categories. These were related to the variability in substrate and structure within these habitat units as selected for sampling (see Chapter 3). For example, a "riffle" as isolated could contain elements of pool habitats such as small backwaters which acted as refugia for individuals of non-riffle species. Thus although the distribution of some species on the micro-scale may therefore be largely deterministic (Grossman *et al*, 1982) on the scale of the habitat unit and higher, the diversity of habitat structure together with mobility, density pressures and hydraulic effects results in a diversity of species in any habitat unit.

Overall, the biomass per unit area of all fish in CWD habitat units was greater by a factor of 4 over riffles and 3 over pools, though densities were not significantly different. The positive within-reach effects on abundance applied to salmonids (*S. trutta*), minnows (*P. phoxinus*) and eels (*A. anguilla*). In contrast bullheads (*C. gobio*) were significantly less abundant in both pools and CWD habitats than riffles and lampreys (*Lampetra* sp.) were most abundant in pools though the differences between pools and CWD were not significant. The stone loach (*N.*

*barbatulus*), though sporadic in all three habitat units, was found in greatest densities in CWD. In contrast *C. gobio* clearly was at its maximum densities in riffles. Its strongly negative correlation with Axis I of the DCA both as biomass and density indicates that shallow water, high proportions of bare gravel and upstream locations with little CWD form its major habitat.

There were seasonal differences between the abundance of species in the individual habitat units and thus the differences between the habitat units differed with season. The interactions between season and habitat could not be calculated because of the high variances. It is clear, however, that the overall differences in abundance for some species calculated using all the data for the sampling period were influenced by seasonal differences in some species. The reasons for the seasonal variations are not established but are most likely a combination of recruitment of young into catches and redistribution of older fish between the habitat units.

Significantly in the Highland Water streams, the abundance of the two species protected as Annex II species under European legislation (European Communities 1992), *C. gobio* and *L. planeri* was not positively correlated with CWD and its associated variables at the within-reach scale. In fact the former showed a highly significant negative correlation. Whilst this relationship is predictable from the known ecological data on *C. gobio* (Crisp, 1963; Green, 1975; Maitland & Campbell, 1992) it is important for management to confirm that this species does not benefit directly from CWD accumulation.

Lampreys (*Lampetra* sp.) were most abundant in pools and this was most likely related to the presence of mixed fine sediment habitats (silt, sand and small twig deposits) found in these habitat units (see Chapter 4) which is their preferred habitat (Maitland & Campbell, 1992). The greatest densities occurred in sandy silt deposits at Site 16 on the Bratley Water, upstream of a large CWD dam and at Site 22 on the Highland Water in a similar deposit downstream of a large CWD dam. The abundance of this species (genus) was indirectly related to the presence of CWD where CWD was the main pool-forming factor.

Eels were significantly more abundant overall in the CWD habitats than elsewhere in the Highland Water streams both numerically and by weight. The relative abundance of larger eels in the CWD habitats may have significant influence on the distribution and abundance of potential prey species, including minnows, trout, bullheads and stone loach (Maitland & Campbell, 1992).

The largest densities of the stone loach (*N. barbatulus*) in the Highland Water streams occurred in the CWD accumulations and the lowest in riffles though there was a wide range of abundance in all habitats. The complementary abundance pattern with *C. gobio* would suggest some spatial segregation but the occurrence of stone loach was so sporadic that it was impossible to reach firm conclusions. Where the two species occur together in chalk streams they are segregated mainly by feeding habits (Welton *et al*., 1983, 1991) but in the New Forest streams there would appear to be some spatial segregation probably as a result of lower food resources. Mann (1971) found no stone loach at a site on the upper Dockens Water and this was confirmed in this work. However, both species were found at sites further downstream during these studies and recent samples in the Ober Water (Quinlan, *personal communication*) have contained both stone loach and bullheads in equal abundance at some sites.

*P. phoxinus* was clearly an inhabitant of the slower current velocities and deeper water which is in accordance with its known habitats in other small trout streams (e.g. Frost, 1943; Maitland & Campbell, 1992). The highly significant difference between biomass per unit area of pools and CWD indicate that the average size of minnows was larger in the latter habitat and this was confirmed by the size-distribution. Salmonids also showed the same pattern (see Chapter 6).

The effects of the partial removal of a dam and a subsequent spate were compared at the within-reach scale with the effects in another reach where the dam was retained. Marked changes in sediments, habitat diversity and fish abundance and diversity upstream of the dam and within the CWD accumulation occurred at both sites. From these observations it would appear that the effects of removal of an impounding structure plus the effects of a subsequent spate scoured sediments and reduced the pool depth and area upstream (see Chapter 4). The populations of minnows and lampreys upstream were also reduced at both sites though there was some re-distribution to the remaining CWD accumulation. The overall indication was that both the physical structure and the fish communities changed at both sites but there was no consistent difference which could be attributed solely to the removal or retention of a CWD dam. Clearly this preliminary observation does not provide sufficient data to confirm trends but there are indications of the movement from a depositional to a more erosional community following the reduction in depth and removal of sediment deposits. Few data exist on the effects of dam removal on fish distribution though Fausch & Northcote (1992) estimated the net loss of salmonids as 8 kg (or 5 times the standing crop) through the year in 1990 from a 332m reach of a small British Columbian stream following the clearance of CWD accumulations.

In the Highland Water streams there was no net loss in either of the reaches as a whole despite the within-stream redistribution of species. The displacement and recolonisation of fish in relation to debris dam removal is a potential subject for further research (Langford & Baker, *unpublished data*).

### 5. 4. 2. Reach-scale abundance and CWD

On the reach scale using the pooled habitat unit data, the strong negative correlation between the abundance of *C. gobio* and CWD was confirmed. The strong negative correlation of salmonid densities with CWD matrix volumes was a result of the size-related distribution of salmonids (Langford & Hawkins, 1997, see also Chapter 7). The smaller fish prefer shallow gravels to pools or CWD (see Chapter 7) as in other streams (e.g. Egglishaw & Shackley, 1982; Kennedy & Strange, 1982; Heggenes, 1988a) and are generally in larger numbers than the larger fish. The strong correlation of minnow biomass with CWD was again because of the size-related distribution with larger fish occurring in the CWD habitats (Langford & Hawkins, 1997). In contrast with the habitat unit scale data there was no significant relationship between the abundance of eels and CWD.

Accumulations of CWD (debris dams) clearly provide habitats where some species are more abundant than elsewhere usually in terms of biomass. The combination of CWD and depth, which are themselves directly correlated as described in Chapter 4, is perhaps the primary factor determining the abundance of species such as *A. anguilla, P. phoxinus* and the extent of within-stream variation. Clearly also, *C. gobio,* with its requirement for shallow water and bare gravels would be at a disadvantage where CWD and deeper water dominated the channel.

### 5. 4. 3. Indirect effects of CWD on fish abundance

The fact that pool volume is directly related to the abundance of fish has been noted by several other authors (e.g. Angermeier & Schlosser, 1989; Inoue & Nakano, 1998) but strict comparisons with the Highland Water streams are difficult because of the differences in methods. Angermeier & Schlosser (1989) found that the relationship between the abundance of individual species and pool volume differed between streams in different regions. Whereas species in the USA streams showed few significant correlations, species in Panama streams showed a higher percentage of significant correlations. The reasons for the differences were not fully explained. Angermeier & Karr (1984) using manipulation techniques and simulated CWD structures in Jordan Creek in Illinois (USA), similar in size to the Highland Water, found that more large fish and more individuals occurred where CWD was not cleared from the stream reach. In multiple reach experiments with artificial CWD structures the results were variable for species, some being more abundant than others in manipulated reaches but with some inconsistencies within the categories and between years. In a larger river (75m wide) Thévenet & Statzner (1999) used non-isolated 50m zones to compare areas with and without CWD. Clearly, the CWD was not generally in the form of dams as in the Highland Water streams and was more scattered. Although six of the species collected showed highest abundances where CWD (LWD in their terminology) was present there was no significant difference between the catches.

The data relating to CWD shown by Braaten & Berry (1997), working in a South Dakota stream, were also equivocal. They showed clear differences in fish abundance between riffles, pools, runs and "woody habitats" but the differences varied with species and with time of year. Woody habitats generally contained more large fish and did not show temporal changes in habitat use. None of the 19 species collected was consistently associated with a habitat type though some species showed significant differences between habitat types in one of the two sampling periods. In the stream studied small fish were most abundant in riffles and woody habitats in early summer but not later in the year. Small fish sheltered in riffles and woody habitat during high discharges. In contrast to the Highland Water streams, and other studies (e.g. Egglishaw & Shackley 1982; Kennedy & Strange 1982; Schlosser, 1987), Braaten & Berry (1997) concluded that the riffles and woody habitats were the most important habitats for small fish.

Hortle & Lake (1983) showed clear differences between the abundance of fish in channelized and unchannelized reaches of the Bunyip river in Australia which they related to the presence of CWD (snags). Eels (*Anguilla australis*) were the most clearly associated with CWD and the largest individuals of this species and *S. trutta* occurred where CWD was most abundant. These results compare with those from the New Forest streams.

To date no other studies have isolated CWD accumulations from other habitat units in small streams and the precise mechanisms which relate CWD to fish abundance are not well known. In the New Forest streams the close correlation of depth, CWD matrix volume and distance downstream of the source are clearly the factors which combine to influence the abundance of most species either positively or negatively. The role of CWD matrices as direct refugia or as secondary creators of deeper pools is difficult to interpret though some clarification may come from using three dimensional criteria for estimating densities and biomass (see Chapter 6). Hortle & Lake (1983) note that expressing fish abundance in terms of surface area may mask the effect of significant increases in depth.

Although as a generality it can be assumed that fish clearly live in a three-dimensional environment (Angermeier & Schlosser, 1989) the significance of this may differ with species. For example it is unlikely that several species of fish with different functional positions in the assemblage would maintain feeding or breeding territories based on depth in fast flowing waters less than 10cm deep such as in the New Forest streams. In such habitats territories are most likely to be based on area as in salmonids (Elliott 1994) and cottids (Welton *et al.*, 1983, 1991; Maitland & Campbell, 1992) and therefore abundance based on two dimensional units would be applicable. Also, for species which burrow into the substrate such as lampreys, the limiting factor in their distribution and abundance will be the area of suitable substrate. In this case the third dimension may be depth into the substrate rather than the water above it.

However CWD may be an important factor here as the deposition of sediments is a function of the creation of pools which in turn may be created by CWD accumulations (Gurnell *et al.*, 1995; Gurnell & Sweet, 1998). For eels the number of crevices may be the vital factor and this may be determined by depth and structural heterogeneity of the channel which will include three dimensional CWD matrices and structures such as root wads.

For species such as *S. trutta* and *A. anguilla*, CWD as a major creator of pools may be the vital factor for larger individuals (e.g. Egglishaw & Shackley, 1982; Hortle & Lake, 1983; Inoue & Nakano, 1998) rather than the presence of a physical matrix. However, analysis based on volume of the habitat units indicated that the CWD matrices did not carry more individuals per unit volume than pools but that the proportion of larger individuals increased. Thus the presence of large feeding salmonids and large eels, both of which may be piscivorous (Maitland & Campbell, 1992) could influence the distribution of potential prey species such as minnows, bullheads, stone loach and small salmonids.

The significance of the patchiness of CWD accumulations is different for cottids and, as Chapter 7 shows, smaller salmonids than for other species. Both of these inhabit the bare gravels and are less abundant in the deeper water and CWD habitats in the Highland Water streams. Thus any excessive pool formation and excessive CWD which results in the reduction of riffles will reduce the available habitat for these species. However, the scour caused by CWD also produces gravel bars downstream which eventually form new riffles though the period between new riffle formation and colonisation by fish is not yet known. There is clearly a threshold riffle-pool area ratio for the optimal survival of the species. This is also a potential topic for future research.

### 5. 4. 4. Effects of reduced tree canopy and low CWD loading on fish abundance

The open streams represent a considerable amount of disturbance from the original wooded state though in the New Forest much of this disturbance occurred several hundred years ago (Tubbs, 1968,1986). Disturbance in relation to fish populations, based on logging and tree clearance has been studied in other streams in other countries (e.g. Burton & Odum, 1945; Davies & Nelson, 1994) but mainly where it occurred relatively recently. The disturbance of the New Forest streams has clearly led to an increase in fish abundance mainly as a result of local immigrant species. The importance of CWD in the abundance of fish in such streams appears to be low. Indeed, there was very little CWD in the reaches sampled in the open New Forest streams. There was, however, a significant amount of instream and marginal vegetation at most sites in contrast to the wooded sites. Instream cover was reduced because of the lack of CWD accumulation and because of the lack of riparian trees which produced submerged root wads in wooded streams. Trailing vegetation and undercut banks however, provided more cover. The effects of land-use changes on community structure and species diversity are discussed in Chapter 6.

The significantly larger fish biomass per unit area indicated a more productive environment (e.g. Bilby & Bisson, 1992) though the densities were not significantly different. Of the species common to both wooded and open streams those which showed a significant increase were minnows (*P. phoxinus*) (biomass only), stone loach (*N. barbatulus*) and eels (*A. anguilla*) (biomass only). The main difference between the two stream types was the appearance of well established populations of additional species in open streams not found in the heavily shaded Highland Water streams. For example chub (*L. cephalus*) were numerically as abundant as salmonids in open streams and much more abundant by biomass. This biomass together with the eel biomass caused the significant increase in biomass in open over wooded streams. Thus despite the high CWD loadings the overall standing stock in the wooded streams is lower than in the cleared streams. The effects of disturbance on species invasions and communities are discussed in Chapter 6.

There were very slight chemical differences between the open and wooded streams (see Chapter 4) and these were not considered of sufficient magnitude to affect the fish populations significantly. In comparison there were a number of physical factors more likely to determine the status of the fish populations in unshaded streams as compared to shaded streams. The most obvious of these are light and temperature (e.g. Brown & Krygier, 1967, 1970). Deforestation in some regions has been shown to cause temperature rises of up to 8.8° C as a result of direct insolation (e.g. Brown, 1970; Smith, 1972; Ringler & Hall, 1975). Moorland streams and streams with banks cleared of vegetation also show higher mean and maximal temperatures than forested streams (Ormerod *et al.*, 1987). In the open streams of the New Forest, the tree canopy was removed many years ago. The streams have had time therefore to develop a stable physical regime. The absence of shade has allowed stable growth of instream and marginal vegetation. In Dockens Water and the Ober Water these growths of instream vegetation can cover up to 60% of the stream bed in some shallower reaches. This vegetation has allowed the establishment of additional species such as the chub which has increased the biomass of fish in such streams (see Chapter 6).

The effect of the clearance of trees has been to transform the stream from a relatively unproductive, cool allochthonous energy driven system to a more productive, warmer and autochthonous energy driven system. The increased productivity is shown in the fish abundance by the addition of species which can exploit the higher temperatures and vegetation. Chub, for example spawn on weed beds and the small fish feed on planktonic organisms (Maitland & Campbell, 1992). Sticklebacks also use plant material for nest building and are tolerant of higher temperatures (Maitland & Campbell, 1992; Langford, 1990). These and other phytophilous species can clearly utilise the resources created by the increased light to considerable advantage, though as the data so far show, none of the other species appear to be reduced in abundance when compared with the wooded streams. Further research into the differences between the fish populations of wooded and non-wooded streams is in progress (Quinlan, *unpublished data*).

# CHAPTER 6

# THE COMPOSITION AND DIVERSITY OF FISH COMMUNITIES IN RELATION TO CHANNEL STRUCTURE, COARSE WOODY DEBRIS AND LAND USE IN FORESTED STREAMS

## 6. 1. INTRODUCTION

The ultimate composition of the fish fauna of any stream system depends on the overall suitability of the stream habitat to accommodate the relevant life-history stages of all the species both spatially and temporally. A major factor in this accommodation is the differential use by the various species of the physical structure and resources of the stream, including substrata, depth, cover and variations in current velocity and flow (e.g. Gorman & Karr, 1978; Binns & Eiserman, 1979; Egglishaw & Shackley, 1982; Heggenes, 1988a; Prenda *et al.*, 1997; Watkins *et al.*, 1997; Martin-Smith, 1998). Whether the processes involved in the coexistence of species are essentially deterministic or stochastic has been the subject of much discussion and study (e.g. Gorman & Karr, 1978; Moyle & Vondracek, 1985; Prenda *et al.*, 1997). Grossman (1982) concluded that deterministically structured communities should show limited morphological similarity between species, segregation on the basis of space or diet, persistent composition over many generations and the ability to recover following disturbance. Communities controlled by stochastic processes would show mainly reverse characteristics, namely morphological similarity among species, less spatial and dietary separation and variable species composition over relatively short periods. Where the physical habitat is stable, with low levels of disturbance over periods of up to several thousand years, habitat partitioning by the coexisting species would be expected to have evolved through deterministic processes to a state of equilibrium, with clearly defined spatial segregation and food partitioning (Angermeier, 1987; Schlosser, 1987; Prenda *et al.*, 1997). Conversely, where major disturbance has occurred in a stream system, whether through human activity or otherwise, this evolved pattern of segregation might be broken down with resulting overlaps in habitat use by the coexisting species (see Prenda *et al.*, 1997). At the same time species richness and diversity may increase as a result of species invading mainly from downstream, (Burton & Odum, 1945; see Gotelli & Graves, 1996).

The Highland Water streams of the New Forest represent a relatively undisturbed system (Sear & Arnell, 1997) in that the catchment has been continuously forested for at least several thousand years (Tubbs, 1968,1986). Thus despite some significant physical changes to the channel itself (see Chapter 2) and some planting of conifers among the deciduous trees, the consistent energy source for the stream ecosystem has been allochthonous material. In contrast, the deforested streams such as the Ober Water and Dockens Water, represent highly disturbed systems, and it can be deduced that a significant proportion of their energy source is autochthonous. Although the major disturbance (deforestation) most likely began more than 2000 years ago, regular maintenance and clearance of streams for lawn drainage has continued over many centuries and may, therefore, represent a pattern of frequent unnatural disturbance (Tubbs, 1968, 1986).

Most natural or semi-natural streams comprise a series of recognisable morphological discontinuities, of which the most well used as a basis for habitat studies are riffles and pools, both sub-divisions of the fast-water and slow-water habitats described by Bisson *et al.* (1982) and Hawkins *et al.* (1993) (see Chapters 4 & 5). For the within-stream studies in the Highland Water the channel units were sub-divided further by isolating the CWD accumulations from the adjacent channel units. These CWD habitat units were then sampled and the data treated separately (see Chapters 3, 4 and 5). Ordination of the sampled habitat units showed them to be organised along a physical gradient based on their dimensions, structures and substrates (see Chapter 4). The trend was from shallow, gravel-dominated riffles (erosional) to deep pools dominated by CWD accumulations and finer sediments (depositional). Distance from the source was also a significant factor in determining the location of the habitat unit on the gradient.

References to the longitudinal zonation of fish communities in running waters have been noted in Chapter 5. Fish assemblages clearly characteristic of or even exclusive to defined geomorphic units such as riffles and pools in both temperate and tropical systems have been described (e.g. Gorman & Karr, 1978; Gilliam *et al.*, 1993; Martin-Smith, 1998). In others the physical gradients of streams have been paralleled by gradients in the composition of the fish assemblages (e.g. Williams *et al.*, 1996; Prenda *et al.*, 1997). There is a known within-stream variation in assemblages in some streams often on a very small scale which is related to the small scale variation in physical structure (Schlosser, 1985).

In Chapter 5 it has been shown that there are highly significant differences in fish abundance on the within-reach scale in the Highland Water streams which can be related to the physical discontinuities, including CWD accumulations. Also, there are clear and significant differences in fish abundance between the heavily forested (wooded) and less-forested (open) streams. Distribution and abundance are different facets of the same phenomenon (Andrewartha & Birch, 1954; Andrewartha 1970) and it follows that the community composition in any habitat will also be closely related. Given the spatial scale of the variations in species abundance it should also follow that similar within-reach scale variations in both diversity and composition of fish assemblages may be identified.

In this Chapter, the fish abundance data are therefore used to examine the extent of physical habitat partitioning by fish species and community structure within the wooded stream system on the within-reach scale with particular reference to the role of CWD accumulations. The analysis will also show the extent to which discrete communities can be distinguished on the within-reach scale in relation to the *a priori* selection of sampling units, namely riffles, pools and CWD habitats or whether the communities of the habitat units show a gradient which can be related to the physical gradient (see Chapter 4). Subsequently habitat segregation of species and community composition of wooded and open streams are also compared in relation to catchment differences and differences in the stream channels. Species richness and diversity are also compared in relation to the within-stream habitat units and between open and wooded streams. The analyses will show the extent to which physical habitat and long-established disturbance affects the structure and stability of the stream communities. The data form the ecological background on which future management of the habitat and communities may be based.

## 6. 2. DESCRIPTION OF SITES AND METHODS

The visual selection and initial classification of sampling reaches, sampling units, and sampling methods are described in detail in Chapter 3. The streams and sites are described in Chapter 4. Physical data were analysed separately from the fish community data to provide the background for the analysis of the fish population and community data (see Chapter 4). The community structure analysis is based on comparisons of *a priori* habitat units, ordination methods (see Kent & Coker, 1992; Southwood & Henderson, 2000) using Detrended Correspondence Analysis (DCA), physical variables as proportional occurrences and fish abundance data as numbers $m^{-2}$. DCA was used because the technique is robust and applicable to various types of abundance data and generally requires no data transformation (Townsend & Pierson, 1988; Kent & Coker, 1992). A series of DCA analyses show the development of the community models and their relationships with the physical variables.

Diversity is expressed using the Shannon-Wiener index (H'), the equitability index (J) and Simpson's D (see Magurran, 1988). The Shannon-Wiener index is sensitive to both species richness and the evenness of composition while Simpson's D is weighted toward the most abundant species and is less sensitive to species richness. For further interpretation, rank-abundance plots (Magurran, 1988) are also used to compare the communities on the within-reach scale. Species associations are compared using both presence-absence data ($\chi^2$ test) and quantitative density data (Spearman rank correlation).

## 6. 3. RESULTS

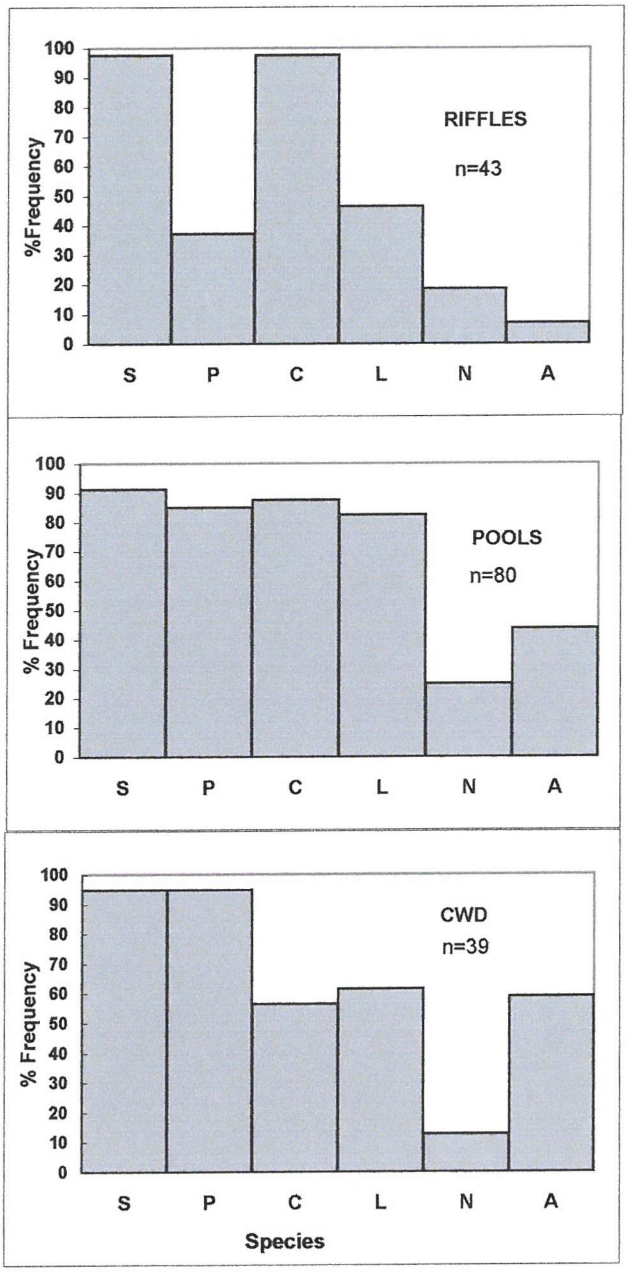
### 6. 3. 1. Species composition of assemblages in the various habitats

The percentage frequencies of occurrence of each species in riffles, pools and CWD habitats are shown in Fig.6.1. The composition of the total catch in riffles, pools and by number and weight are shown in Figs. 6.2 and Fig. 6.3. To assess the differences in distribution of species, the percentage occurrence in each category of habitat unit was compared with the percentage occurrence in reaches (habitat units combined for each site) (Table 6.1). *S. trutta* and *C. gobio* occurred in all reaches ie.100% of the reach catches, *P. phoxinus* and *Lampetra* sp. occurred in 90% and 92% respectively, *A. anguilla* in 80% and *N. barbatulus* in 43%. *P. phoxinus* and *A. anguilla* occurred less frequently in riffles than pools or CWD habitats. *N. barbatulus* was by far the least common species.

There were apparent differences between the quantitative compositions of the assemblages of each habitat unit. Numerically, *C. gobio* and *S. trutta* were ranked highest in riffles, while *P. phoxinus* and *S. trutta* were most numerous in pools and CWD. From the totalled catches from all reaches, *P. phoxinus* and *C. gobio* were numerically dominant (see Chapter 5, Table 5.1). Expressed as biomass, *S. trutta* and *C. gobio* were most abundant in riffles, *S. trutta* and *A. anguilla* in pools and *A. anguilla* and *S. trutta* in CWD. In the total catches *S. trutta* and *A. anguilla* together comprised 77% of the total biomass but less than 19% of the numbers (see Chapter 5, Table 5.1).

### 6. 3. 2. Fish assemblages, channel structure and CWD accumulation

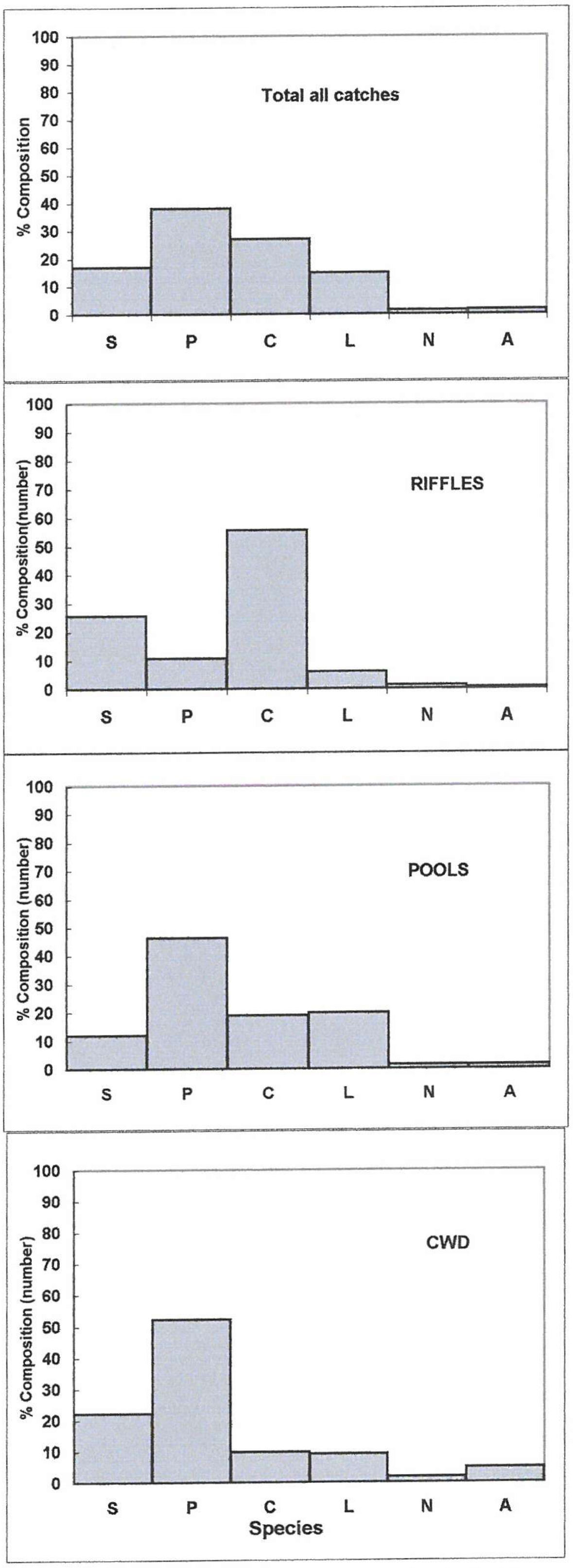
A series of analyses were carried out using fish species abundance data and the physical variables first to determine the association between the species



*% Frequency in all reaches, S= 100, P=90, C=100, L=93, N=43,A=80*

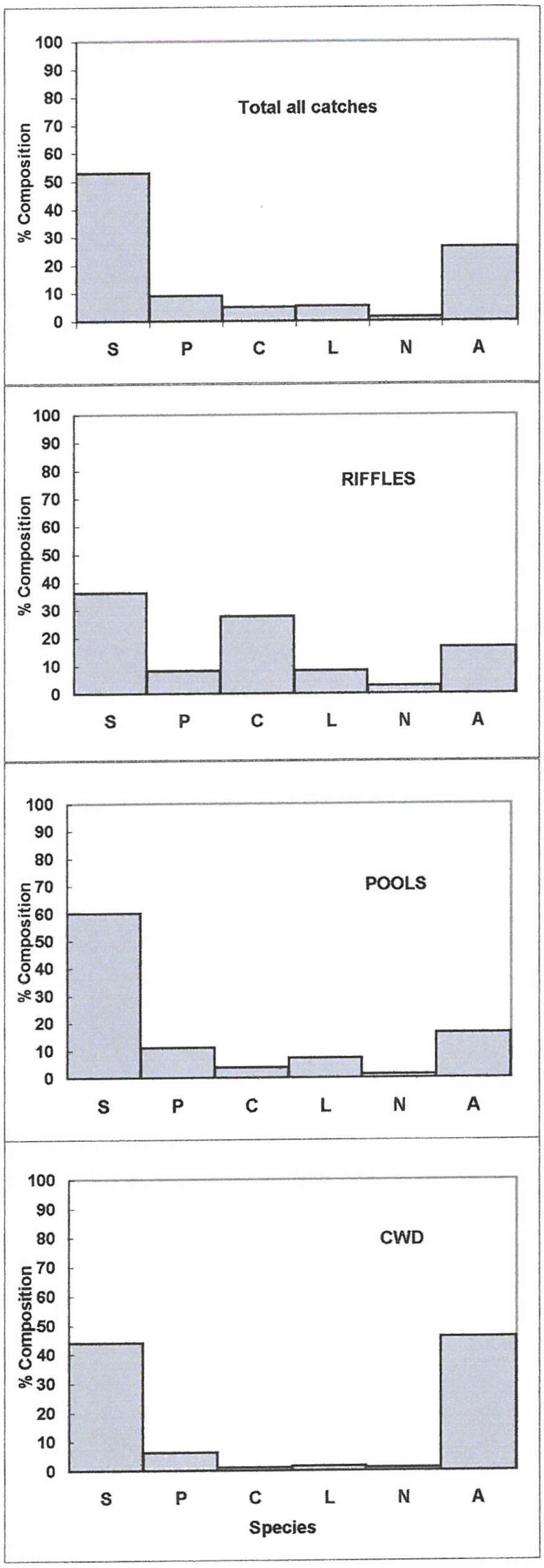
*S=S.trutta, P = P.phoxinus, C = C.gobio, L = Lampetra, N= N.barbatulus*
*A = A.anguilla.*

**Fig.6.1 Percentage frequencies of occurrence of each species of fish in riffles,pools and CWD habitats in Highland Water streams. New Forest. 1996-98**



*S=S.trutta, P=P.phoxinus, C= C.gobio, L= Lampetra spp. N=N.barbatulus, . A= A.anguilla*

**Fig.6.2 Percentage composition by number of totalled fish catches from riffles, pools and CWD habitats in Highland Water streams. compared with the composition of the total catch for all habitats. New Forest. 1996-98**



*S=S.trutta, P=P.phoxinus, C= C.gobio, L= Lampetra spp. N=N.barbatulus, A= A.anguilla*

**Fig.6.3 Percentage composition by weight of totalled fish catches from riffles, pools and CWD habitats in Highland Water streams compared with the composition of the total catch for all habitats. New Forest, 1996-98**

**Table 6.1 Percentage frequency of occurrence of fish species in habitat types, compared with frequency in reaches in Highland water streams.**

| Species | Riffles | Pools | CWD | Reaches |
|---|---|---|---|---|
| S.trutta | 98 | 91 | 95 | 100 |
| P.phoxinus | 37 | 85 | 95 | 90 |
| C.gobio | 98 | 88 | 56 | 100 |
| Lampetra sp. | 47 | 83 | 62 | 93 |
| N.barbatulus | 19 | 25 | 13 | 43 |
| A.anguilla | 7 | 44 | 59 | 80 |
| *Samples (N)* | ***43*** | ***80*** | ***39*** | ***31*** |

NB. Note low frequencies of *A.anguilla* in riffles and low overall frequencies of *N.barbatulus.*

themselves, and secondly between each individual species and the structure of the habitat unit. DCA using the pooled density data (nos.m$^{-2}$) confirmed that there was no clear distinction between assemblages that could be equated with three types of habitat unit, namely riffles, pools and CWD (Fig. 6.4). Indeed, the species data show the species widely spaced on the DCA plot with the exception of *N. barbatulus* and *Lampetra* sp. *N. barbatulus* was relatively scarce in Highland Water streams often found singly or in very small numbers in depositional habitats. *Lampetra* sp. burrows well into softer sediments (Maitland & Campbell, 1992) and thus although the two appear spatially close, they are most likely spatially separated by the difference in their behaviour.

The results of the DCA indicate that there is a clear spatial habitat separation of the four species based on their relative abundance in habitat unit samples. There is a broadly identifiable division between a "slow-water" group and a "fast-water" group, the former based on *P. phoxinus*, *N. barbatulus*, *Lampetra* sp. and *A. anguilla,* the latter on *C. gobio* and *Salmo* sp. The distance between *Salmo* and *Cottus* on Axis II indicates that the association between the two is not close and that *Salmo* sp. is a more generalist species (e.g. Prenda *et al.*, 1997). For this analysis all salmonids of all sizes were used in the analysis.

The habitat units showed a clear horizontal trend on Axis I (Fig.6.4) based on the quantitative species composition, from samples dominated by high *P. phoxinus* abundance to those with high *C. gobio* abundance. The habitat groups around *C. gobio* comprised mainly riffles and pools, those around *P. phoxinus* and its associated species mostly pools and CWD habitats. *Salmo* spp. showed a significant influence in the ordination with a trend toward the upper portion of Axis II. As with the physical variables the indication was of a more or less continuous gradient of within-reach communities rather than separation into discrete habitat groups.

Axis I of the DCA was most heavily weighted by *C. gobio* and in the opposing direction by *P. phoxinus* (Table 6.2). In Axis II the dominant components were *Salmo* sp. and *A. anguilla* with *Lampetra* sp. as the opposing component. Spearman rank correlation coefficients for Axes I and II of the DCA in relation

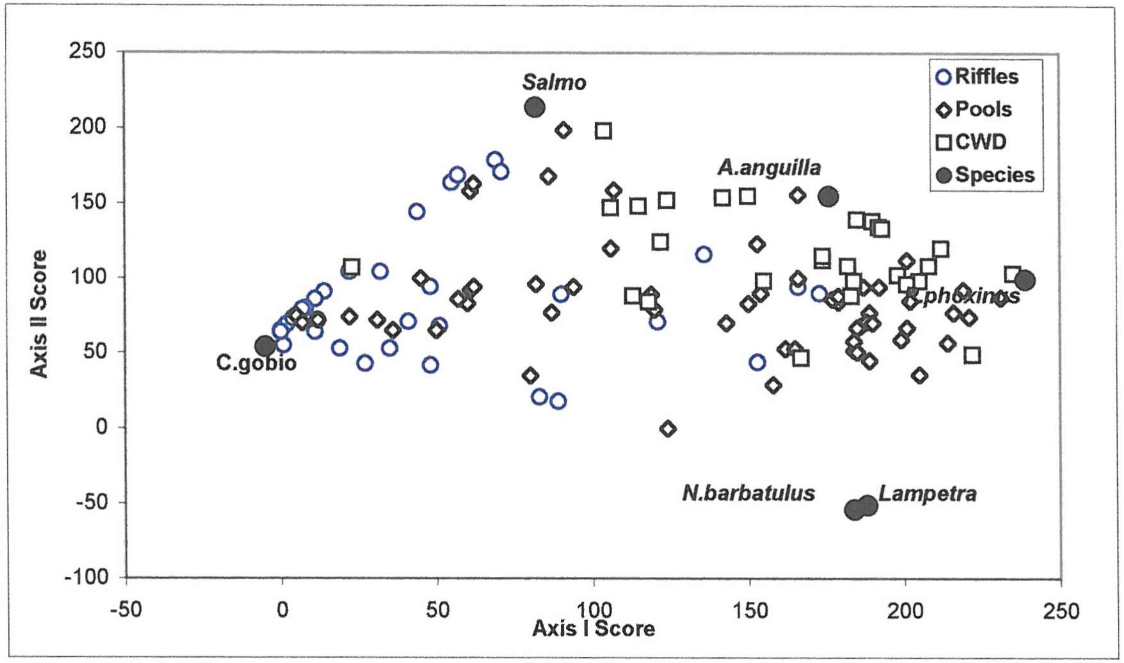


**FIG.6.4** **DECORANA plot of riffle, pool and CWD communities sampled in Highland Water streams in relation to fish species ordination. New Forest 1996-98.**

*(NB. The closer the community is to a species, the more dominant the species was in the sample) The relative abundance of a species in a sample is inversely proportional to its distance between the sample location and the species location on the ordination. The distance between the species is a measure of their association*

**Table 6.2. Species scores from the DCA of fish species and and habitat unit samples from Highland Water streams (see Fig.6.4)**

| | Axis I | Axis II | Axis III | Axis IV |
|---|---|---|---|---|
| Species scores = | 0.5631 | 0.1619 | 0.09416 | 0.06973 |
| S.trutta | 82 | 214 | 176 | 112 |
| P.phoxinus | 239 | 99 | 0 | 0 |
| C.gobio | -5 | 54 | 77 | 32 |
| Lampetra | 188 | -51 | 89 | 163 |
| N.barbatulus | 184 | -54 | 365 | -219 |
| A.anguilla | 176 | 155 | 270 | 293 |

***Axes 1-4 explain 88% of the variation in the assemblage composition***

to species abundance as numbers and biomass are shown in Chapter 5, (Tables 5. 8). The strong positive correlations for *P. phoxinus* and *A. anguilla* numbers and negative correlations for *C. gobio*, confirm these as dominant components determining the distribution of sample sites along Axis I. The strong correlations of Axis II are for *Lampetra* sp. and *Salmo* sp. and the relative locations of these are shown on Fig. 6.4. Axes 1-4 explain some 88% of the variation in the composition of the samples from the habitat units. Axis I and II cumulatively explain over 72% of the variation.

Association analysis using only presence and absence data (Table 6.3) showed significant positive associations ($p<0.05$) between species in the samples which were generally confirmed by the quantitative data (Table 6.4). No significant negative associations ($p>0.05$) are shown by the association analysis but Spearman rank analysis shows significant negative correlation between *P. phoxinus* and *C. gobio* and *C. gobio* and *A. anguilla.*

The pairs of species which showed significant positive correlation were *S. trutta/C. gobio, P. phoxinus/N. barbatulus, P. phoxinus/A. anguilla*, and *Lampetra/N. barbatulus*. As will be seen in the following section and from Chapter 7 the division of the salmonids into size-classes shows that the strongest habitat correlation is between the smallest salmonids and *C. gobio* in riffles. There is a marked negative association between the distribution of the older salmonids and *C. gobio.*

The other associated species pairs are characteristic of the deeper habitat units with *A. anguilla* and *N. barbatulus* most abundant in CWD habitats.

Both the association analysis and DCA give clear indications of an assemblage in which the species have clear within-stream spatial habitat partitioning.

**Table 6.3 Association analysis of species at the within-reach (habitat unit) scale for Highland Water streams (chi-squared,presence/absence only) New Forest. 1996-98**

| | *S.trutta* | *P.phoxinus* | *C.gobio* | *Lampetra* | *N.barbatulus* |
|---|---|---|---|---|---|
| ***S.trutta*** | | | | | |
| ***P.phoxinus*** | -0.93 | | | | |
| ***C.gobio*** | 0.31 | -2.84 | | | |
| ***Lampetra*** | 0.36 | ***13.91*** | ***8.8*** | | |
| ***N.barbatulus*** | 0.39 | *7.73* | 2.7 | *7.42* | |
| ***A.anguilla*** | 3.31 | ***13.7*** | -0.62 | ***4.86*** | -0.001 |

***bold = $p<0.05$***

**Table 6.4 Spearman rank coefficients for the correlation of fish abundances in habitat units based on densities as numbers $m^{-2.}$ Highland Water.New Forest 1996-98**

| *Species* | *S.trutta* | *P.phoxinus* | *C.gobio* | *Lampetra* | *N.barbatulus* |
|---|---|---|---|---|---|
| *S.trutta* | | | | | |
| *P.phoxinus* | -0.149 | | | | |
| *C.gobio* | ***0.333*** | ***-0.364*** | | | |
| *Lampetra* | 0.082 | 0.45 | -0.01 | | |
| *N.barbatulus* | -0.081 | ***0.348*** | 0.039 | ***0.287*** | |
| *A.anguilla* | 0.127 | **0.242** | **-0.17** | 0.149 | 0.016 |

***Bold italic= $p<0.001$, bold =$P<0.05$, ordinary type NS ,$p>0.05$***

### 6. 3. 3. Effects of salmonid size differentiation on species associations

In the following chapter (Chapter 7), where the salmonids are dealt with separately as a special case, it is shown that, as in other streams, salmonids of different size groups have different habitat preferences (e.g. Egglishaw & Shackley, 1982; Milner *et al.*, 1985). For the next stage in this community analysis four size-groups of salmonids were used. These are coded on Fig. 6.5 as follows, Sa7=<7cm TL, Sa715=7-15cm, Sa152 =15-20cm, Sa2=20+cm. This produced a clearer division of the community into three groups than that shown in Fig.6.4. Larger salmonids (Sa152, Sa2) separated out from the other slow-water species/size-groups (*N. barbatulus, A. anguilla, P. phoxinus, L. planeri* and Sa715) while *C. gobio* and the smallest salmonids (Sa7) also formed a separate grouping. The habitat unit samples did not separate out into three groups though CWD units were mostly grouped toward the larger salmonids, *A. anguilla* and *P. phoxinus* while riffles were mainly grouped toward the smaller salmonids and *C. gobio.* The division into the four size groups showed the salmonids to be more specialist than generalist depending on size.

Table 6.5, shows the highly significant associations shown (on presence/absence data) between *C. gobio* and the smallest salmonids (Sa7) and positive but not significant associations between the following pairs, *N. barbatulus*/Sa715, *A. anguilla*/Sa715, Sa715/Sa512, Sa715/Sa2 and Sa152/Sa2. As in the previous section, the use of abundance data and Spearman rank correlation confirmed much of the association analysis coefficients (Table 6.6) with significant positive correlations between the abundance of *Lampetra* sp./*P. phoxinus, N. barbatulus/P. phoxinus, A. anguilla/P. phoxinus*, all three larger size-groups of salmonids and *P. phoxinus*, *Lampetra* sp./*A. anguilla, A. anguilla*/Sa715 and *A. anguilla*/Sa152. There were also significant correlations between the larger salmonid groups,

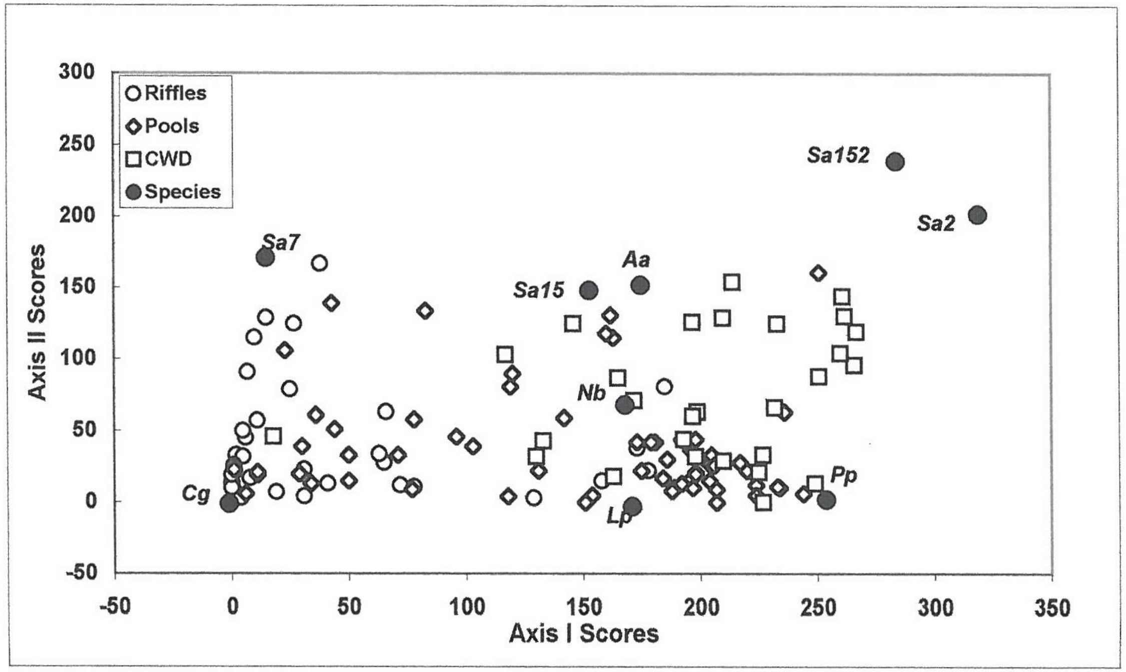


**Fig. 6.5** **DECORANA plot of riffle,pool and CWD communities sampled in Highland Water streams in relation to the ordination of fish species and four salmonid size groups. New Forest. 1996-98**
***(Salmonids: Sa7 = <7cmTL, Sa715 = 7-15cm, Sa152 = 15-20cm, Sa2 = >20cm)***
***Cg = C.gobio, Lp =Lampetra sp., Nb = N.barbatulus, Aa = A.anguilla, Pp = P.phoxinus)***
***(See Fig.6.4. For explanatory notes on ordination)***

**Table 6.5 Coefficients of association of species using four size-groups of salmonids and presence/absence data from habitat unit samples in Highland Water streams. New Forest, 1996-98**

| *Species/Group* | *P.phoxinus* | *C.gobio* | *Lampetra sp.* | *N.barbatulus* | *A.anguilla* | *Sa7* | *Sa715* | *Sa152* |
|---|---|---|---|---|---|---|---|---|
| ***C.gobio*** | -2.026 | | | | | | | |
| ***Lampetra sp.*** | **15.53** | **4.583** | | | | | | |
| ***N. barbatulus*** | **7.399** | 1.599 | **5.464** | | | | | |
| ***A.anguilla*** | **13.06** | -1.235 | 3.066 | 0.05302 | | | | |
| ***Sa7*** | **-8.567** | **22.46** | 0.3948 | 1.244 | -0.5572 | | | |
| ***Sa715*** | **14.04** | 0.2195 | **10.04** | 2.56 | **3.849** | 0.09328 | | |
| ***Sa152*** | **17.8** | -2.722 | 1.108 | 0.1537 | 3.208 | -1.965 | **7.496** | |
| ***Sa2*** | **7.399** | -0.2086 | 0.2228 | -0.01376 | 0.4485 | -1.95 | **8.649** | **12.63** |

***Sa = salmonid size groups (Sa 7= <7cm TL, Sa 715 = 7-15cm, Sa152 = 15-20cm, Sa2 = 20cm +)***
***bold signifies significance (p<0.05)***

**Table 6.6 Spearman rank coefficients for abundance of fish species using density data from habitat units in Highland Water streams. New Forest.1996-98**

| **Species/group** | *C.gobio* | *Lampetra sp* | *N.barbatulus* | *A.anguilla* | *Sa7* | *Sa715* | *Sa152* | *Sa2* |
|---|---|---|---|---|---|---|---|---|
| ***P.phoxinus*** | ***-0.365*** | ***0.444*** | ***0.39*** | **0.284** | ***-0.425*** | ***0.404*** | ***0.335*** | **0.249** |
| ***C.gobio*** | | -0.0319 | -0.0162 | **-0.193** | 0.588 | -0.0563 | ***-0.4*** | **-0.276** |
| ***Lampetra sp*** | | | **0.244** | **0.222** | -0.046 | **0.288** | 0.0325 | 0.0634 |
| ***N.barbatulus*** | | | | -0.0298 | -0.109 | 0.0568 | 0.00825 | -0.0499 |
| ***A.anguilla*** | | | | | -0.0437 | **0.267** | **0.192** | 0.114 |
| ***Sa7*** | | | | | | 0.0307 | **-0.299** | **-0.275** |
| ***Sa715*** | | | | | | | **0.231** | **0.267** |
| ***Sa152*** | | | | | | | | ***0.384*** |

***(see Table 6.5 for explanation of salmonid size-group codes)***
***bold italics =p<0.001, bold = p<0.05***
***Association analysis and Spearman-rank correlation mostly show similar correlations***

though there was a significant negative correlation between the abundance of the smallest salmonids (Sa7) and the larger size-groups of salmonids. There were significant negative correlations between the abundance of the smallest salmonids (Sa7) and *P. phoxinus* and a significant positive correlation between the abundance of *C. gobio* and the smallest salmonids (Sa7). The lack of apparent agreement between the results of association analysis and DCA using abundance data are discussed later in this chapter.

The DCA plot (Fig.6.6) shows the relationships between species and habitat variables in more detail. It indicates, however, that the deeper waters show two physical gradients with associated fish species, one grouping branching toward the higher scores on Axis II, comprising silt, organic sediments and *Lampetra* sp. with the alternative gradient branching along Axis I toward higher abundance of CWD with greater abundance of larger salmonids. On this plot, salmonids in the 1+ group (Sa715) (7-15cmTL) are grouped with stone loach (*N. barbatulus*) and associated with moderate depth.

The indication is therefore of three general habitat guilds though the separation between the two guilds in the slow water habitats is less distinct than between the fast and slow-water guilds. The three guilds are as follows, Guild 1 comprising, *C. gobio,* and small salmonids (Sa7), grouped with gravel, clay and shallow water. Guild 2 comprises mid-sized salmonids (Sa715), *P. phoxinus, N. barbatulus, A. anguilla* and *Lampetra* sp. associated with silt and organic sediments in deeper water, whilst Guild 3 comprises the largest larger salmonids (Sa152, Sa2) more associated with woody debris (CWD). Of the second guild, *Lampetra* sp. is most closely associated with the sediments whilst the other species/groups appear to be more generally associated with the habitat type.

When the data from the individual habitat units are superimposed on the species/variables plot most of the CWD habitat units are clustered near Guild 3 while Guild 2 is more generally associated with a mixture of pools and CWD. Riffles are more clustered toward Guild 1. The habitat units are not clearly separated into identifiable groups based on both fish species and physical variables but the trends towards given physical features are clear.

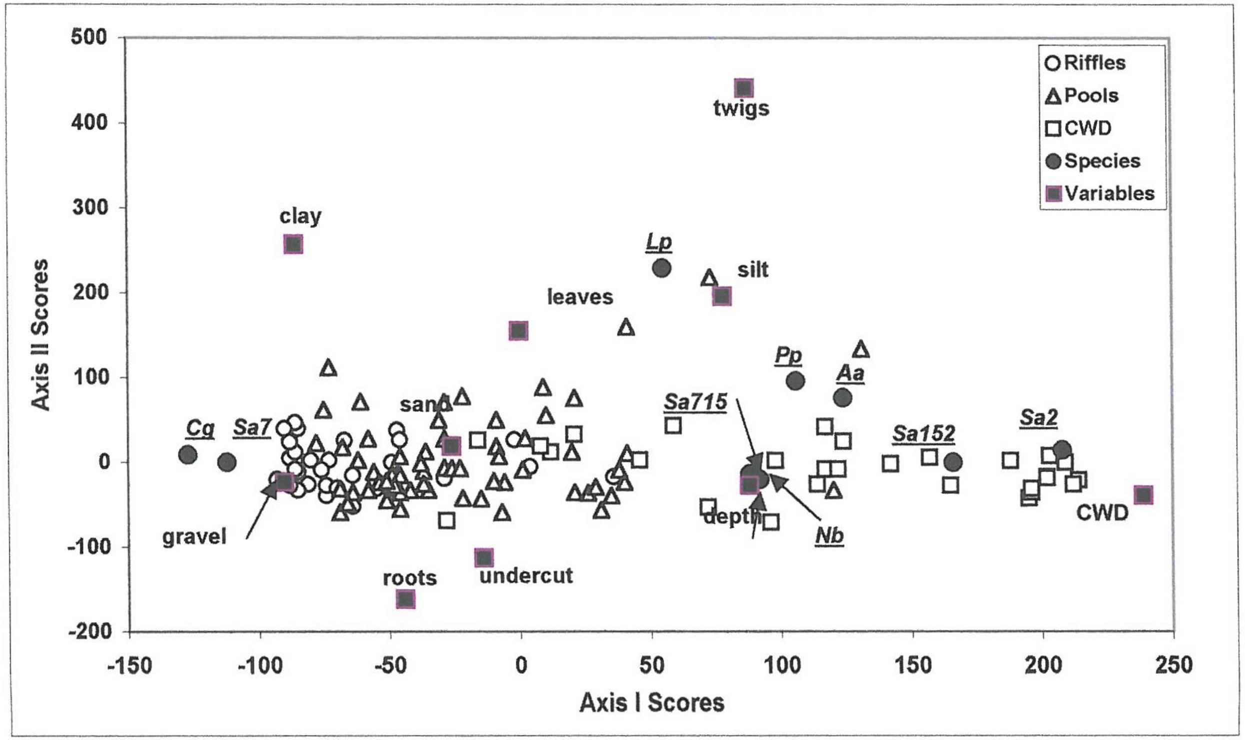


Fig.6.6. **DECORANA plot of both physical variables and fish in riffles, pools and CWD habitats in Highland Water streams 1996-98**

***(Association between physical variables is inversely proportional to the distance between them)***

***(Distance between sample location and species/variables is inversely proportional to their relative "abundance" in that sample/habitat unit)***

Salmonid size-groups:- Sa7=<7cmTL, Sa715= 7-15cm, Sa152=15-20cm, Sa2=20+cm

*Other species:-* *Cg=C,gobio, Lp=Lampetra sp.,Aa=A.anguilla, Nb=N.barbatulus,Pp=P.phoxinus*

The conclusion is that the fish communities and physical variables of the individual habitat units are both located along a continuum from the shallow gravelly riffles dominated by small salmonids and bullheads to the deeper waters dominated by the remaining species/groups. Ordination of the deeper water habitat units indicates two basic gradients with characteristic sediments and fish species, one based on increasing silt and organic materials, the other on increasing CWD. Both have characteristic fish species or species size-groups which are not, however, exclusive to that habitat.

Spearman rank coefficients for Axis I of the DCA of fish communities and of Axis I of the DCA using physical variables showed highly significant correlations (Coeff. = 0.631, $P<0.001$). This clearly confirmed that there was a strong relationship between the composition of the fish community and the physical structure of the channel with increasing proportions of species characteristic of depositional habitats correlated with increasing depth, softer sediments and amounts of CWD. Areas of bare gravel, abundance of CWD, increasing depth and distance from the source are the main physical determinands. *P. phoxinus, A. anguilla, C. gobio* and salmonids (small and large) are the main fish determinands of community structure. The evidence for three habitat guilds based on physical variables indicates that the fish community is well established and stable with relatively little effect of disturbance despite alterations to the stream channel over many years. Within the guild structure there is also some segregation of the individual species/size groups indicating physical habitat partitioning within the guild habitat.

There is therefore a broadly definable model which gives an ecological basis for the management and prediction of changes in the wooded streams and the effects of alterations in the amounts of CWD present in given reaches. The effects on channel depth, changes in species abundance and to some extent community structure and composition can be predicted within limits on very small spatial scales from the data.

### 6. 3. 4. Species diversity, habitat diversity and CWD

The Shannon-Wiener (H') diversity index and Simpson's D may be relatively insensitive to changes with such small numbers of species (Magurran, 1988) but they are relatively robust and can be used as reliable indicators of change in quantitative community structure even in communities of low species richness. In addition rank-log abundance plots (Magurran, 1988) can aid the interpretation of differences between the diversities of communities. *A priori* comparisons of the mean fish community diversities (Shannon-Wiener H', J' and Simpson's D) from riffles, pools and CWD accumulations on the within-reach scale in the Highland Water streams (Table 6.7) showed that there were significant differences between the three *a priori* habitat types ($p<0.001$). *A posteriori* pairwise tests showed that both pools and CWD habitats had significantly higher mean diversities than riffles ($p<0.001$). However, there were no significant differences between diversities in pools and CWD ($p< 0.994$). Equitability (J') showed the same pattern as did the Simpson's D index. The median number of species was 3 for the riffles and 4 for the pools and CWD. Rank-abundance plots (Fig. 6.7) showed no clear differences between the habitat units, though the pools again showed a more even distribution of the species abundance. Thus although the deeper water habitats, pools and CWD both showed higher fish species diversities than riffles the structure of the communities were similar in principle. The higher diversity indices reflect the more evenly distributed abundances in pools. As Chapter 4 showed, riffles also showed lower physical habitat diversity than pools and CWD.

Least squares regression of the Shannon-Wiener indices (H') and habitat diversity on the within-reach scale for Highland Water streams (Fig. 6.8) indicated a significant linear relationship ($p= 0.002$). The non-normality of the equitability indices (J') did not allow viable tests for significance.

The relationship between diversity and physical structure of the habitat was also tested using Spearman rank correlation for the Shannon-Wiener index (H'), Simpson's D index, Equitability (J) and Axes I and II of the DCA on physical

**Table 6.7** Mean diversity indices for habitat unit samples of fish from Highland Water streams (Median for non-normalised data)

| Index | Habitat Unit | Mean | Std.deviation |
|---|---|---|---|
| H' | Riffles | 0.734 | 0.365 |
|  | Pools | 1.018 | 0.294 |
|  | CWD | 0.998 | 0.357 |

*F = 11.25 df = 2 p<0.001*
*(Tukey) Pools>Riffles, CWD>Riffles, p=0.05*

|  |  | Median | Quartiles |
|---|---|---|---|
| Simpsons D | Riffles | 1.807 | 1.464-2.419 |
|  | Pools | 2.547 | .9006-3.094 |
|  | CWD | 2.672 | 1.953-3.492 |

KW *H=17.72, df=2, p<0.001*
*df=2, CWD>Riffles, Pools>Riffles, Pools/CWD p>0.05 (NS)*

|  |  | Mean | Std.Deviation |
|---|---|---|---|
| J' | Riffles | 0.41 | 0.203 |
|  | Pools | 0.568 | 0.164 |
|  | CWD | 0.557 | 0.199 |

*F=11.247, df=2, p<0.001*
*Tukey test* *Pools>Riffles, CWD>Riffles, CWD/P Pools/CWD p>0.05 (NS)*

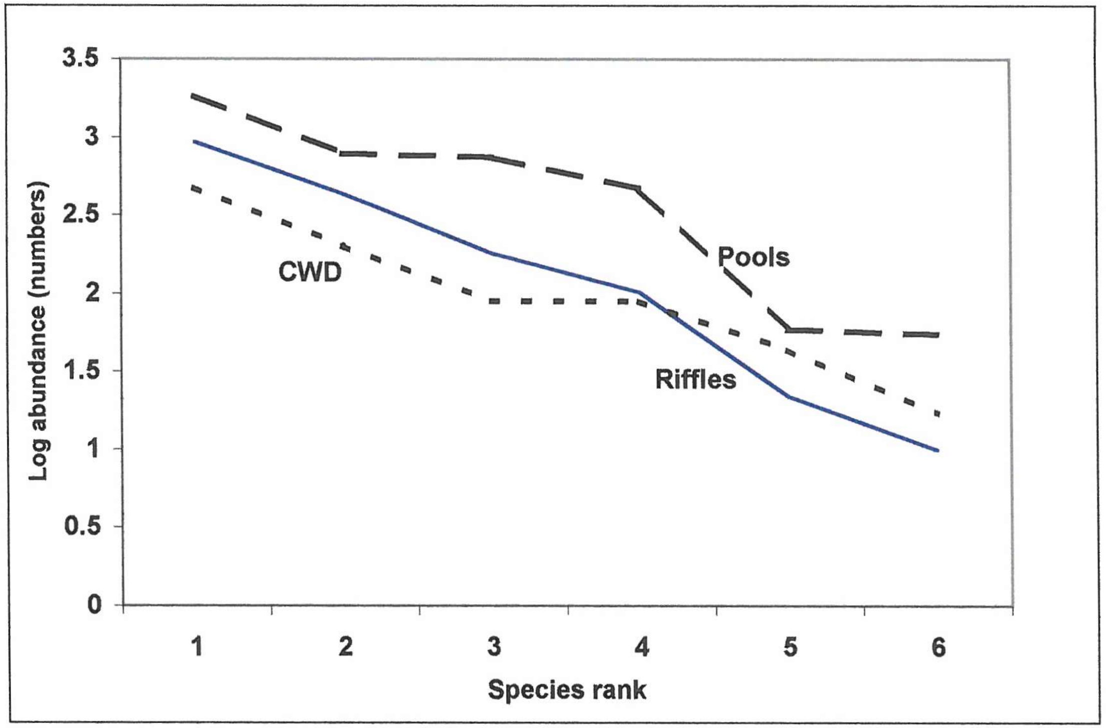
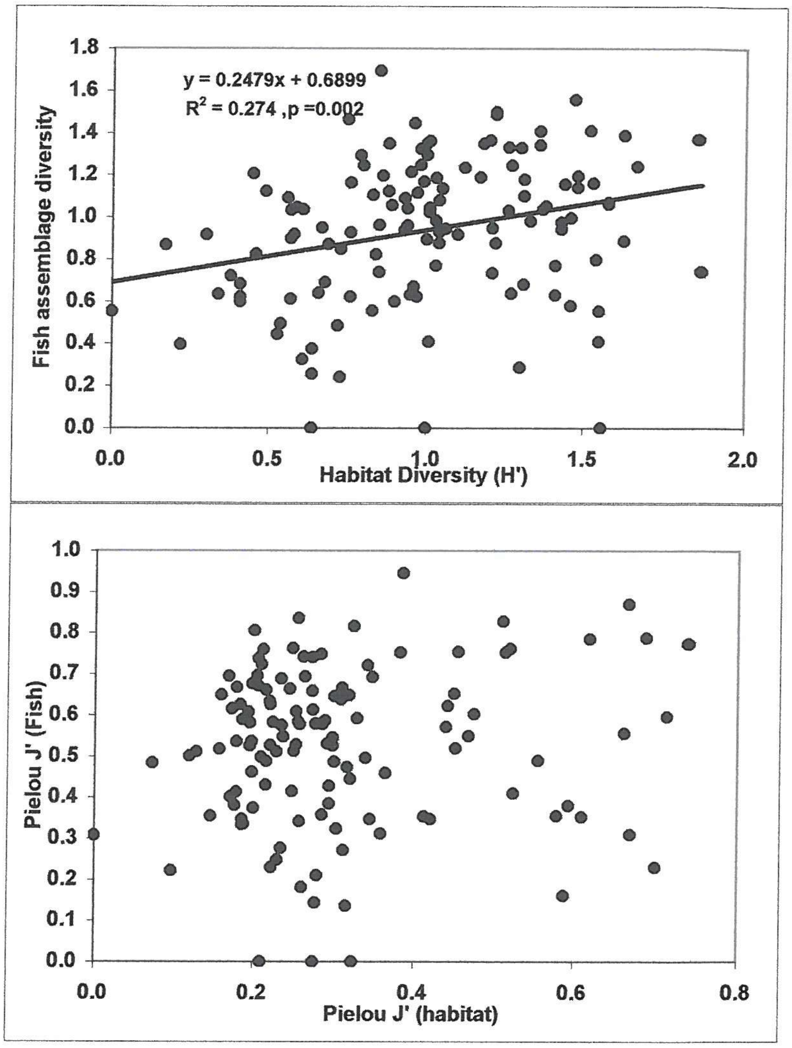


Fig.6. 7 Rank abundance plots for fish species in samples from riffles,pools and CWD habitat units in Highland Water streams. Pools show less variations in abundance of the different species. (see Fig. 5.2 for proportional abundances)



*For H' normality and variance tests passed.*

*H' (fish) = H' (Habitat) x 0.25 +0.69 ( p=0.002)*

R2 = 0.075

**Fig. 6. 8** **Least squares regression plots of habitat diversity (H') and fish species diversity (H') and for (J') for habitat units in Highland Water streams. New Forest 1996-98**

variables (Table 6.8). There were significant correlations with Axis I for both H' and D ($p = 0.002$, and $p = 0.007$ respectively). The indication was therefore of increasing fish diversity with increasing stream size, depth and CWD volumes and a negative correlation with decreasing depth and gravel substrates. Correlation coefficients for selected separate physical variables (Table 6.9) showed significant correlations of fish diversity (H') with maximum depth ($p<0.001$) and with the volumes of CWD in the debris accumulations (CWD) ($p<0.001$) and significant negative correlations with the proportion of gravel (arcsine transformed) in the substrate. Simpson's D index was correlated less strongly with habitat diversity ($p = 0.018$).

There was also a significant negative correlation between diversity (H') and Axis I of the DCA using fish data (Table 6.10). Axis I was dominated by *C. gobio* and *S. trutta*, both species of riffle habitats. This confirmed further the low diversity of riffle communities in the Highland Water streams. There were no significant seasonal variations in diversity (Table 6.11) and the differences between the habitats were not dependent on season.

Scaling diversity up to the level of the reach, it was predicted that the overall diversity of the fish community (using densities in pools, riffles and CWD summed for each sampling site) would be significantly greater than those of the individual habitat units. The mean diversity (H') for the reach data for the Highland Water streams ($n = 31$) was 1.14 (+/- 0.256 SD). This was significantly different from those of riffles (0.73) and CWD habitat units (1.0) but not significantly different ($p>0.05$) from that of the pools alone (see Table 6.7) despite the increased length of the sampled reach. This is a clear reflection of the general paucity of the fish fauna in the Highland Water streams and an indication that the stream has reached stable species richness and diversity limits in its present state. The overall indication is of a highly structured, well established and stable fish community, primarily regulated more by deterministic than stochastic processes. The species show clear indications of basic spatial habitat segregation even within three small guilds which themselves are based on habitat structure and substrate type. Juvenile salmonids clearly show different primary habitat preferences to the older fish as in other streams (Egglishaw & Shackley,

Table 6.8 **Fish species diversity in relation to physical habitat variables in the Highland Water streams. (Axis I & II from DCA of physical variables)**

| Diversity Index | Axis I | Axis II |
|---|---|---|
| H' | **0.28** | -0.17 |
| D | **0.241** | -0.155 |
| J' | **0.28** | -0.17 |

***H' Shannon-Wiener Index, D=Simpson's D Index***
***J' =equitability index. Bold = p <0.05***

Table 6.9 **Spearman rank correlation coefficients for fish diversity indices and selected physical variables . Highland Water st**

| Index | Distance | Depth | CWD | % gravel |
|---|---|---|---|---|
| H' fish | 0.149 | **0.243** | ***0.265*** | -0.229 |
| J' fish | 0.149 | **0.243** | ***0.265*** | **-0.229** |
| Sim D | 0.108 | 0.184 | 0.248 | -0.188 |

***Bold italics = p<0.001, Bold = p <0.05***
***depth = maximum in habitat unit***

Table 6.10 **Fish species diversity in relation to species composition. Correlation (Spearman rank) with Axis I & Axis II of DCA on fish populations in the Highland Water streams.**

| Fish diversity | Axis I | Axis II |
|---|---|---|
| H' | ***-0.364*** | 0.073 |
| D | **-0.285** | 0.146 |
| J' | ***-0.364*** | 0.073 |

***Bold italics = p<0.001, Bold = p <0.05***

**Table 6.11 Interactions between season and habitat for fish diversity in habitat units of the Highland Water streams.**

| Index | Habitat<br>(p) | Season<br>(p) | Interaction | Sig. |
|---|---|---|---|---|
| H' | **0.018** | 0.246 | 0.262 | NS |
| J' | **0.018** | 0.246 | 0.262 | NS |
| D | **0.044** | 0.505 | 0.406 | NS |

***There were significant ($p<0.05$)differences between habitat units***
***NB. There are no differences between seasons and no interaction.***
***Diversity differences between habitat do not depend on seasons.***

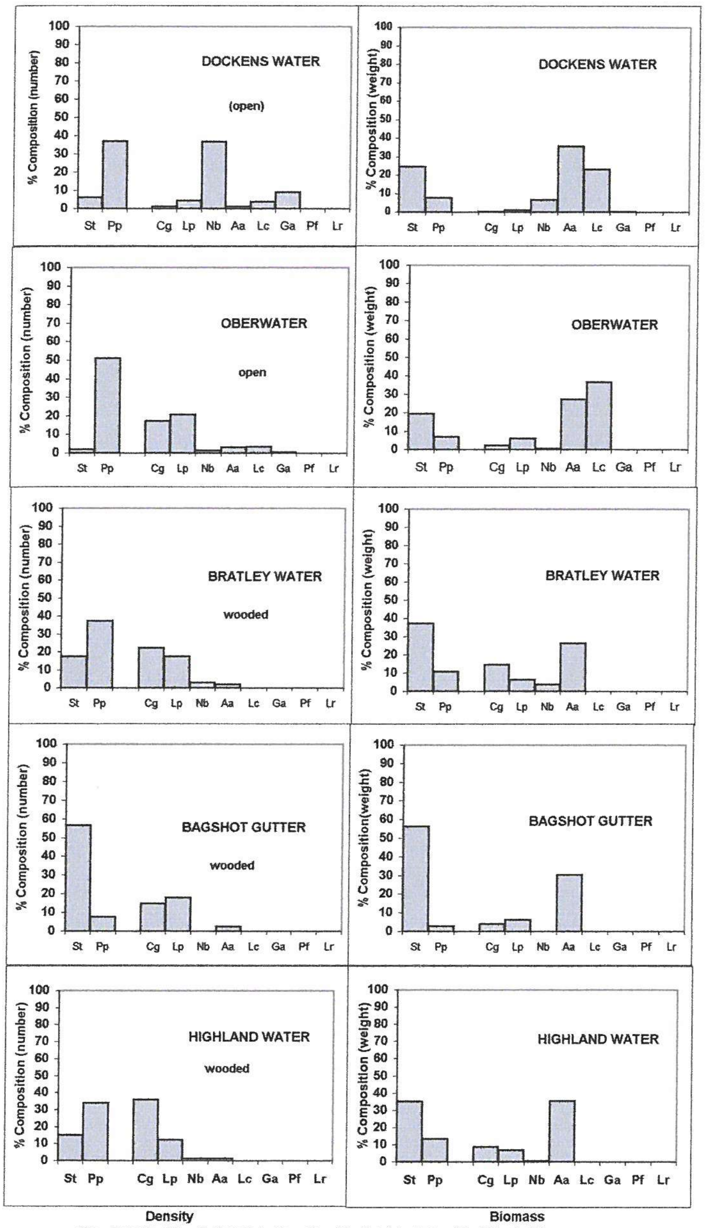
1982; Milner *et al.*, 1985) and behave practically as a separate species. Riffles, pools and CWD habitats selected for sampling do not, however, separate out clearly based on their fish communities mainly as a result of the physical heterogeneity within the isolated individual habitat units and stochastic processes which distribute fish outside their primary habitat.

## 6. 4. COMPARISON WITH OTHER NEW FOREST STREAMS

### 6. 4. 1. Community composition

Preliminary comparisons were made between the composition and diversities of fish communities in adjacent streams with different land uses in the catchments. Communities of streams with high proportions of woodland in the catchment and high CWD loadings (Highland Water, Bratley Water, Bagshot Gutter) were compared with those streams with reduced proportions of woodland and low CWD loadings (Dockens Water, Ober Water) (see Chapter 4). The main difference in the physical habitat, apart from the lack of CWD, was the higher incidence of instream weed which, in turn, was a result of less shade. In some open stream reaches instream weed covered over 60% of the substratum. For the statistical comparisons only samples of the wooded and open streams taken in similar seasons were used, mainly November/December and July to September to reduce seasonal bias.

Differences in quantitative species composition and percentage composition by number and biomass of the fish catches were apparent (Fig. 6.9). *P. phoxinus* was numerically dominant in the Ober Water. In Dockens Water, sticklebacks, *G. aculeatus* occurred in larger numbers than elsewhere. In the Highland Water and Bratley Water the catches were numerically dominated by *P. phoxinus* and *C. gobio* and by salmonids and *A. anguilla* by weight. There was an obvious difference in species richness between these streams in that the open streams, Ober Water (8) and Dockens Water (8) contained more species than the wooded streams, namely Highland Water (6), Bratley Water (6) and Bagshot Gutter (5). (The most recent data from July and August 2000, (Langford & Quinlan,



*(St = S.trutta, Pp = P.phoxinus, Cg= G.gobio, Lp=Lampetra, Nb=N.barbatulus, Aa = A.anguilla, Lc=L.cephalus, Ga=G.aculeatus,Pf=P.fluviatilis,Lr=L.rutilus*

**Fig.6.9 Percentage species composition of fish assemblages by number and biomass from five streams sampled in the New Forest during 1996-2000. Data are from population densities and biomass estimates**

*unpublished data*) has extended the number of species caught in the Ober Water in these studies to 11). The difference in species richness and composition between the Ober Water and the Highland Water is remarkable in that the two have a common stem, the Lymington River, (see Chapter 2, Fig. 2.1) and almost a common confluence point. Indeed, during two years of sampling in the Highland Water streams no fish other than the six species listed were recorded even in the lower parts, whilst the invading species were clearly resident in various size ranges and well established in the Ober Water even in the upper reaches. The invasion by a large omnivore *L. cephalus* did not appear to affect the standing stock of Salmonidae as the numbers and biomass densities were not significantly different between open and wooded streams (see Chapter 5).

Multidimensional scaling (MDS) using the Sorensen similarity coefficient (see Southwood & Henderson, 2000) (Fig. 6.10) shows a clear separation of five fish species characteristic of the wooded streams from the three additional species found only in the open streams. *N. barbatulus,* was common to both but being more abundant overall in open streams it is separated out singly from the two main groups. DCA using both species and habitat unit (sample) data shows that some Dockens Water reaches contained communities which were different from those of the other streams (Fig. 6.11) but the Ober Water communities had many similarities with wooded streams. The two groups of fish species are not separated out as clearly as in the MDS plot. Indeed there is evidence on the DCA plot (Fig.6.11) of three species-habitat groups, namely, *C. gobio* and *Salmo* spp. characteristic of the undisturbed, wooded streams, *Lampetra* sp. and *A. anguilla* relatively common and moderately abundant in both types of stream and *L. cephalus, N. barbatulus, R. .rutilus* and *G. aculeatus* most common and abundant in the open streams. *P. phoxinus* is perhaps the most widespread species after *Salmo* sp. but was more abundant numerically in the open streams.

From the DCA plot of species it is clear that there is a general gradient between relatively high abundance of *C. gobio* and *Salmo* sp. and high abundance of *G. aculeatus* and occurrence of *L. rutilus* indicative of a trend from a low disturbance to a high disturbance community. The reaches also follow a general gradient from left to right based on the composition of the fish fauna and do not

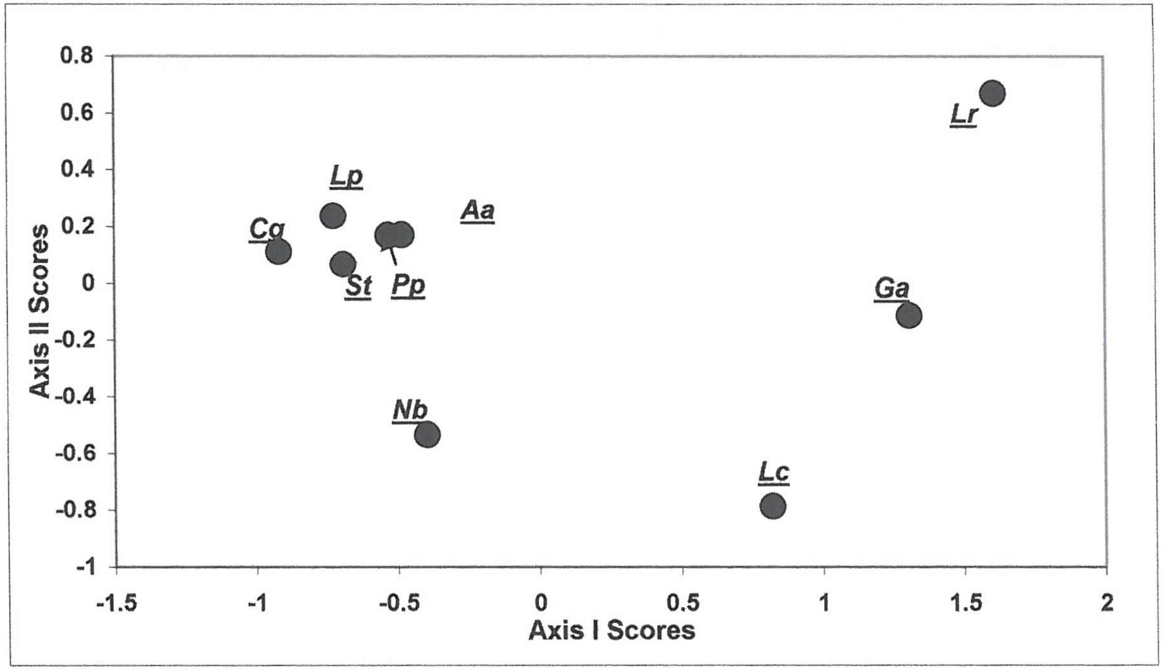


Fig.6.10 Multi-dimensional scaling ordination of species of fish using density to show communities of open and wooded streams in the New Forest. (Abundance data from summation of within-reach data) (For key to species, see Fig. 6.9)

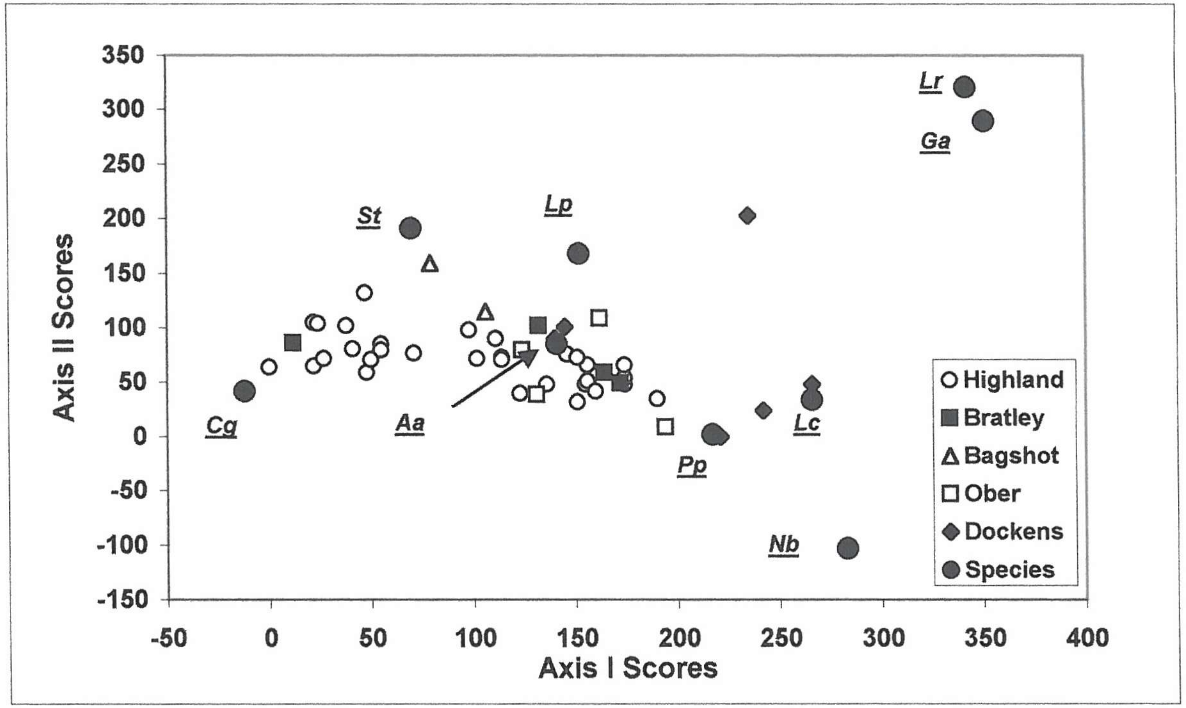


Fig.6.11 DECORANA plot of samples of fish from reaches of wooded and open streams in the New Forest in relation to the abundance of each species in the sample. The nearer a sample point is to a species on the ordination, the more abundant the species was in the sample. The similarity of the distribution of the species is inversely proportional to the distance between them on the ordination.
*The samples are distributed on a continuum from wooded to open but the open streams are predictably located more toward the locations of the invading species, see text)*
*(Wooded, Highland, Bratley, Bagshot; open, Dockens, Ober) (Key to species, Fig. 6.9)*

separate out into two discrete groups despite the apparent differences in species composition (Fig. 6.11). Several of the Dockens Water samples are separated out by DCA mainly because of the relatively high abundance of *N. barbatulus* and the presence of *G. aculeatus* and *L. rutilus*. Although the Ober Water has a chub (*L. cephalus*) population unlike the Highland Water streams the communities do not separate out, mainly because the species is not numerically abundant. The Ober Water also contains significant numbers of *C. gobio* unlike Dockens Water which increases the similarity of the community to that of the wooded streams. Further physical and biological data are being gathered in a continuing research programme on the biology of the species and the processes involved in the establishment and persistence of the disturbance communities (Langford & Quinlan, *unpublished data).* Axis I of the DCA is heavily weighted by the additional species in the open streams with *C. gobio* as the opposing species (Table 6.12). Axis II was also positively weighted by the additional species but *N. barbatulus* was the opposing species with high negative weighting.

### 6. 4. 2. Observations on individual species in New Forest streams

Table 6.13 shows the species recorded from 26 sites on 13 New Forest streams using a combination of data from the studies described here and from Environment Agency surveys. There are essentially two groups of species based on these distribution data, namely those that are common and widely distributed (e.g. salmonids *S. trutta,* minnows *P. phoxinus,* bullheads *C. gobio*) and those that are less common and mainly in the lower reaches of streams (e.g. sticklebacks *G. aculeatus,* chub *L. cephalus,* roach *L. rutilus,* perch *P. fluviatilis*)

Minnows (*P. phoxinus*) occurred throughout the stream systems and were observed even in very small, temporarily flooded lawn drainage channels during high discharges. In one case they were swimming among flooded terrestrial vegetation. This exploratory behaviour to exploit new resources has been discussed by Garner *et al.* (1998).

Lampreys (*Lampetra* sp.) were also common to most streams. All individuals were not identified to species for this present work or in the data provided by the

Table 6.12 Species scores from DCA of data from open and wooded streams in the New Forest

| NAME | AXIS I | Axis II |
|---|---|---|
| Species scores = | 0.4319 | 0.1435 |
| S.trutta | 70 | 191 |
| P.phoxinus | 217 | 2 |
| C.gobio | -12 | 42 |
| Lampetra | 152 | 168 |
| N.barbatulus | 283 | -103 |
| A.anguilla | 141 | 85 |
| L.cephalus | 266 | 34 |
| G.aculaeatus | 351 | 290 |
| L.rutilus | 342 | 321 |

***Open stream species have the higher scores in Axis I, with C.gobio as the negative influence***

**Table 6.13** **Distribution of Fish species in New Forest streams** **(Additional data from the Environment Agency)**
**(Plus Mann, 1971, Mann and Orr, 1969**

| | S.salar | S.trutta | S.trutta* | C.gobio | P.phoxinus | N.barbatulus | G.aculeatus | A.anguilla | Lampetra sp | L.cephalus | L.leuciscus | L.rutilus | P.fluvialilis | S.erythrophthalmus | A.brama | T.thymallus | E.lucius | C.carpio | P.flesus |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Bartley Water (Busketts) | | | * | * | * | * | | * | * | | | | | | | | | | |
| Bartley Water (Costicles) | | * | * | * | * | * | | * | * | | | | | | | | | | |
| Bartley Water (Ashurst) | | * | * | * | * | * | | * | * | | * | * | | | | | | | |
| Dark Water (Gatewood Bridge) | | * | * | * | * | * | | * | * | | | | | | | | | | |
| Dark Water (Exbury) | | * | * | * | * | * | | * | * | | | | | | | | | | * |
| Beaulieu River (Decoy farm) | | | * | * | * | * | | * | * | | | | | | | | | | |
| Beaulieu River (Ipley) | | * | * | * | * | * | | * | * | | | | * | | | | | | |
| Beaulieu (North gate) | | * | * | * | * | * | | * | * | | | | | | | | | | * |
| Bratley Water (Upper) | | * | | * | * | * | | * | * | | | | | | | | | | |
| Highland Water (Warwickslade) | | * | * | * | * | * | | * | * | | | | | | | | | | |
| Blackwater (Rhinefield) | | | * | * | * | * | | * | * | * | | | | | | | | | |
| Ober Water (Puttles Bridge) | | * | * | * | * | * | | * | * | * | | * | * | | | | | | |
| Ober Water (Mill Lawn) | | * | * | * | * | * | * | * | * | * | | * | * | * | * | | | | |
| The Weirs (Culverley) | | | * | * | * | * | | * | * | | | | | | | | | | |
| Lymington River (Bolderford) | | | * | * | * | * | | * | * | | | | | | | | * | | |
| Lymington River (Roydon) | | * | | * | * | * | | * | * | | | | | | | | | | |
| Lymington River (Haywood Bridge) | | | * | * | * | * | | * | * | | * | | | | | | | | * |
| Avon Water (Wootton) | | * | * | * | * | * | | * | * | | | | | | | | | | |
| Avon Water (Stud farm) | | * | * | * | * | * | | * | * | | | | | | | | | | |
| Avon Water (Gordleton) | | * | * | * | * | * | | * | * | | | * | * | * | | | | | |
| Avon Water (Efford) | | * | * | * | * | * | | * | * | | * | * | * | | | | | | |
| Danes Stream (Lower) | | * | * | * | * | * | | * | * | | * | | | | | | | | |
| Dockens Water (Upper) | ? | * | * | * | * | * | | * | * | * | * | * | | | | * | * | | |
| Dockens Water (Upper) | | * | | | * | * | * | * | * | * | | * | | | | | | | |
| Linford Brook (Lower) | | * | * | | | | | | | * | * | * | | | | | | | |
| Linford Brook (Middle) | | * | | | * | * | * | * | | | | | | | | | | | |
| Latchmoor Brook (Upper) | | * | * | * | * | | | * | * | | | | * | | | | | | |

Environment Agency. Adult lampreys identified were all *L. planeri* but it was not possible to distinguish between the larvae of these and those of *L. fluviatilis* and therefore for the analyses all lampreys are grouped as the genus *Lampetra* sp. On many occasions individuals were observed to be emerging from small areas of silt at the extreme margins of the channel even at the edge of riffle reaches where small backwaters occurred. The largest catches came from banks of sandy silt in pools where there were also accumulations of small twig and leaf debris.

The coexistence of bullheads (*C. gobio*) and stone loach (*N. barbatulus*) and their competition for space and food has been the subject of research for some years (e.g. Welton *et al.*, 1983, 1991). In New Forest streams the distribution of both suggests a complex pattern of habitat partitioning which may vary with the stream. Stone loach (*N. barbatulus*) were common to most streams but very rarely abundant. They were typically found in the margins of Highland Water streams often among twigs or leaves deposited in small back eddies. *C. gobio* was very abundant in these wooded streams. The exception to the typical distribution and abundance of *N. barbatulus* was recorded in Dockens Water. Here at the uppermost site, (Holly Hatch) *N. barbatulus* was recorded at densities of 1 to 1.5 $m^{-2}$ mostly among instream weed beds in mid-channel. This was far greater than at any other site or in any other New Forest stream. There were no bullheads (*C. gobio*) recorded in this reach. A similar distribution was also found in studies in 1968 (Mann & Orr, 1969; Mann, 1971) where the same reach was sampled. Downstream from this reach in Dockens Water subsequent sampling in 1999 and 2000 recorded both *C. gobio* and *N. barbatulus* though in very small numbers. Both species have also been recorded in regular surveys in the lower reaches (Environment Agency, Public Register, 1988-1998). In the Ober Water both species were found at most sites. The most recent data (Langford & Quinlan, *unpublished data*) indicate that in the wooded reaches both exist in more or less equal numbers but there may be some lateral spatial segregation in the channel. In the open reaches, *N. barbatulus* dominates and the young fish are particularly numerous in the weed beds. Further research on these species is in progress.

Dace *L. leuciscus*), roach *(L. rutilus*), pike (*E. lucius*), perch *(P. fluviatilis*), bream (*A. brama*) and rudd (*S. erythrophthalmus*) are generally restricted to the lower reaches of New Forest streams, mostly where the rivers run through improved pastureland and where the channels are larger and contained significant instream weedbeds. The exceptions are the open reaches of deforested streams. In the upper reaches of Dockens Water *L. rutilus* was recorded in 1998, while in 2000 samples in the upper part of the Ober Water contained *L. rutilus, S. erythrophthalmus* and *P. fluvialis,* together with bream (*Abramis brama,* (Linnaeus 1758)) and one hybrid specimen believed to be a roach/bream hybrid (Langford & Quinlan, *unpublished data*).

The chub (*L. cephalus*) has a sporadic distribution in the streams but occurs in abundance in a few streams penetrating almost the full length to the smaller headwaters. The distribution of this species in two streams (Dockens Water and the Ober Water) is discussed in more detail later in this chapter in contrast to its absence from the Highland Water. This absence is notable because the species occurs in both the Ober Water and Black Water which are like the Highland Water tributaries of the Lymington River.

Confirmed records of sticklebacks (*G. aculeatus*) are relatively rare in New Forest streams and in the distribution studies the species was recorded only in the less wooded streams. No mention is made of this species in the Environment Agency data from the Lymington River catchments and those draining eastwards though the records are not considered fully comprehensive. The only site where the species was very abundant was in the upper reaches of Dockens Water (see Chapter 5).

Salmonids are ubiquitous in New Forest streams. As Fig. 6.9 shows, however, they are rarely dominant numerically. The exception in this work was the Bagshot Gutter, a very small stream with a virtually 100% wooded catchment where they comprised 55% of the numbers and biomass of the population. In most wooded stream reaches sampled, eels (*A. anguilla*) dominated the biomass. In the open streams the dominant species by weight was chub (*L. cephalus*).

### 6. 4. 3. Comparisons of diversity between streams

As shown above species richness clearly varies between streams within the New Forest drainage system (Table 6.13) ranging from 6 to 11 species depending upon the stream and the location of the sampling points. The lowest reaches of most streams usually contained most species as might be expected, though as we have already seen there were very obvious differences in species composition even between the upper reaches of streams with different riparian land uses.

Fig. 6.12 shows the diversities of the fish assemblages (H') recorded from the streams sampled for this study. There were no significant differences between the overall diversities of the fish communities between the individual wooded (undisturbed) and open (disturbed) streams ($p > 0.05$) despite the larger number of species in both Dockens Water and the Ober Water. The imbalance in the number of samples suggests that the tests must be interpreted with caution. To improve the replication, data from the wooded and open streams were combined and compared as two groups. This gave a significant difference between diversities (H', J' D) (Fig. 6.13). The wooded streams showed significantly lower diversity and equitability than the open streams (H' = 1.47 and 1.42 respectively ($t = -2.636$, $df = 44$, $p = 0.012$) $D = 2.736, 3.616$, ($t = 2.697$, $p = 0.01$;) $J' = 0.498, 0.618$, ($t = 2.636$, $df = 44$, $p = 0.012$). The habitat diversity (H') and fish diversity indices (H') from the open stream habitats were combined with those from the wooded streams in a least squares regression (Fig. 6.14). There was some overlap of points but the regression was significant with some of the open streams showing the highest habitat and fish diversities.

Chemically, the five streams are little different (see Chapter 4, Fig. 4.2) but there is evidence of higher conductivities in the Highland Water. No temperature data were available for comparisons. The effects of historical disturbance in the open streams (see Chapter 2) including deforestation of the catchments, pasture drainage and stream channel maintenance has produced higher species richness, and higher species-diversity, both classic effects of habitat disturbance. The higher overall production in the open streams indicated by fish biomass (see

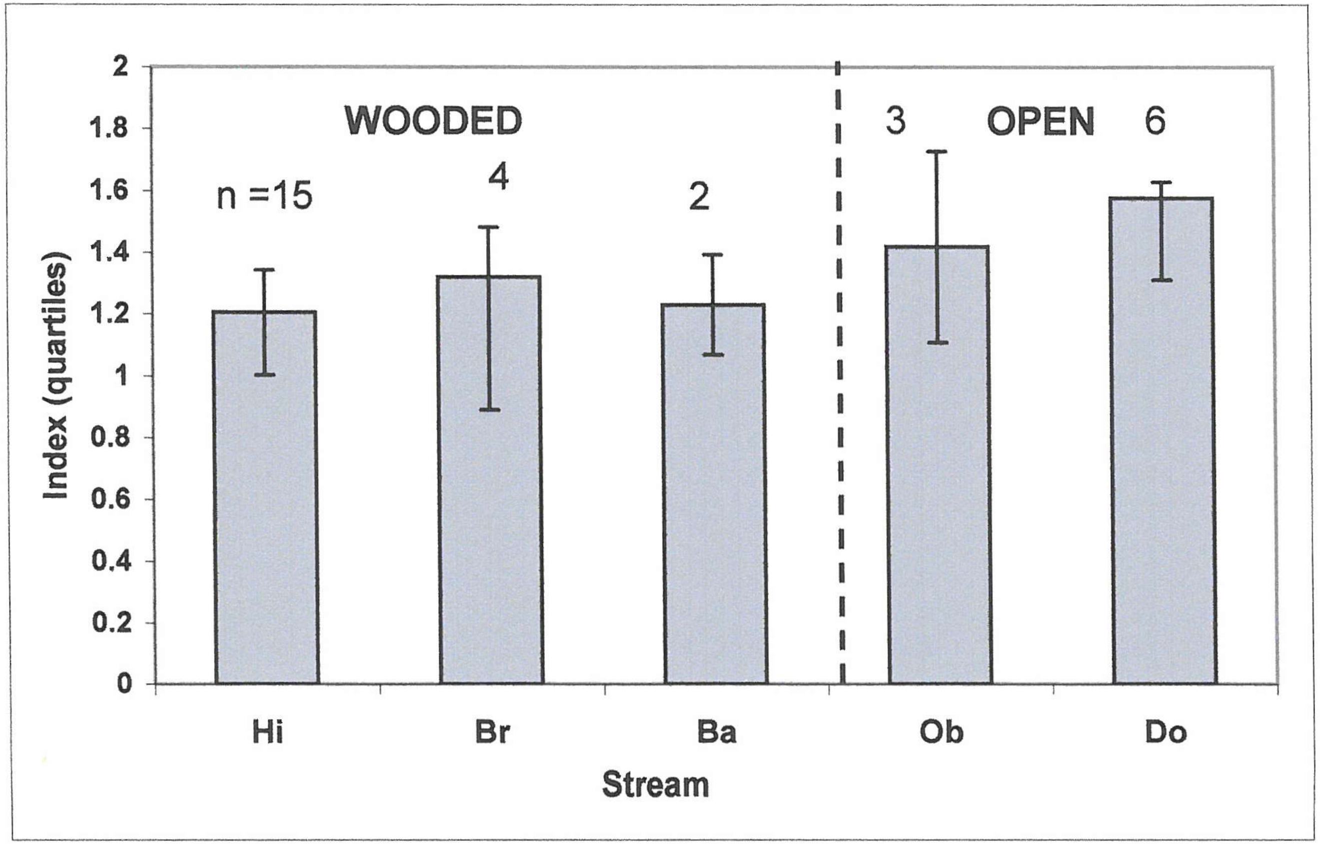


**Fig.6.12** Median and quartile Shannon-Wiener indices for fish communities of five New Forest streams

*Hi=Highland, Br = Bratley, Ba = Bagshot, Ob = Oberwater, Do = Dockens water*

KW test H= 9.097, df = 2, p =0.154 (NS)

*Figures above columns denote number of samples*

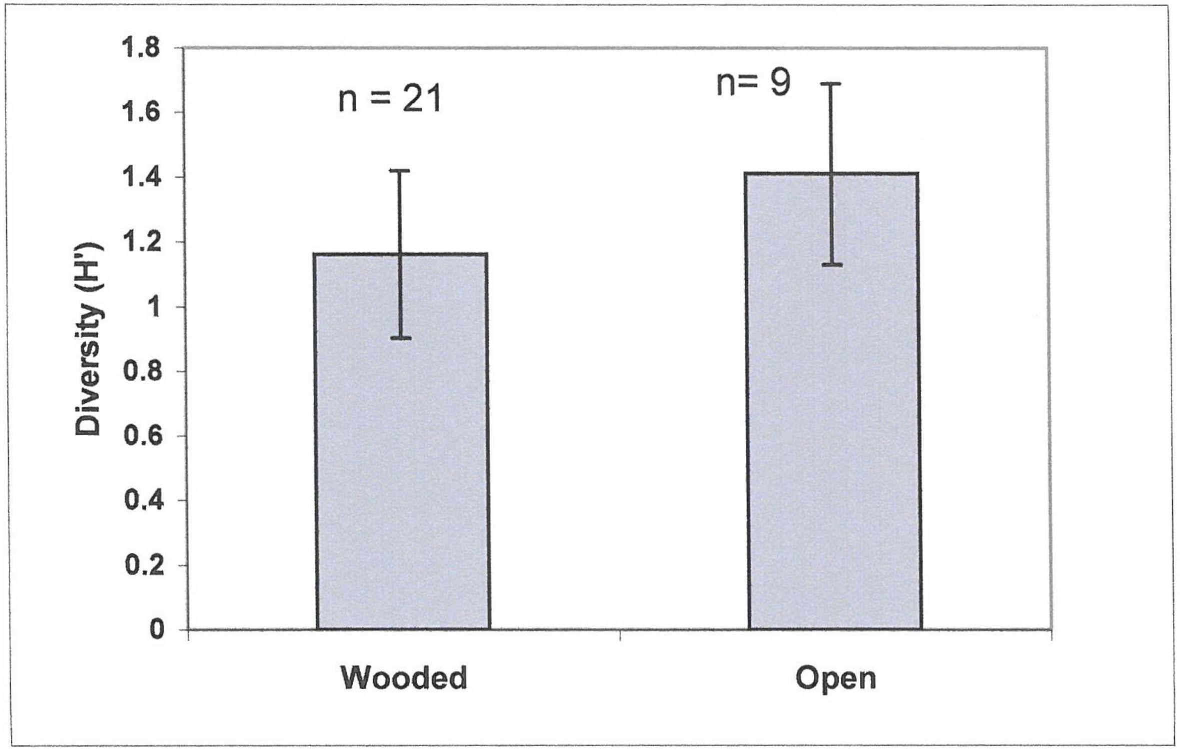


**Fig.6.13** Median and quartile Shannon-Wiener diversity indices for fish communities of open and wooded streams. Data are from pooled samples for each stream type. (see Fig.6.12)

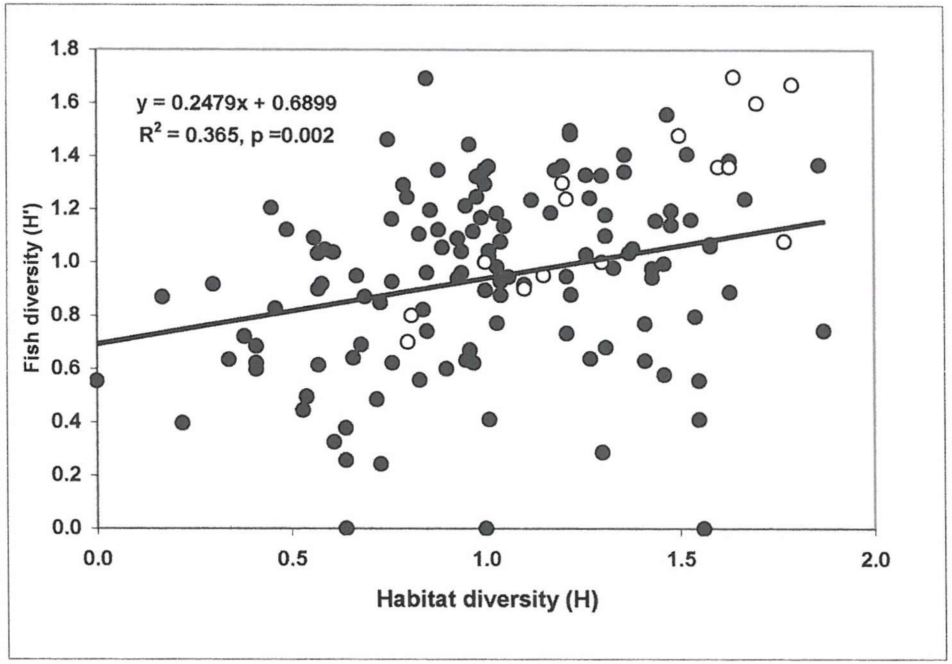


**Fig.6.14** **Least squares regression of habitat diversity and fish diversity (H') using data from habitat units in open (open circles) and wooded (filled points) New Forest streams. 1996-2000**

Chapter 5) is an effect of disturbance and the shift from an allochthonous energy driven system to one driven by autochthonous energy.

## 6. 5. DISCUSSION AND CONCLUSIONS

### 6. 5. 1. Distribution

The most widely distributed species in New Forest streams are *S. trutta, C. gobio, P. phoxinus, N. barbatulus, A. anguilla* and *Lampetra* sp. Salmon, *Salmo salar* have been reported from Dockens Water (Mann, 1971; Environment Agency, Public Register, 1988-1998) but not with verification from any other streams. Although brown trout and adult sea-trout (*S. trutta*) are reported from most streams (Environment Agency, Public Register, 1988-1998) the proportion of brown trout to juvenile sea-trout among the fish under 25cm (TL) is not generally recorded. Sampling in various streams for this study produced a small number of fish which could be readily identified from appearance and scale reading as brown trout. These were mostly fish over 25cm (TL). Many salmonids in the 15-20cm length range caught from January to May had the silvery appearance and delicate scale attachment of smolts. Fish of less than 15cm (TL) were not readily identifiable as either migratory or non-migratory forms. Thus, for most of the analyses, all Salmonidae were considered as one species (see Chapter 7). This group was ubiquitous throughout the stream systems, from the lower reaches to the uppermost where stream widths were less than 1m. Observations by the author and others during 1996-1999 reported adult sea-trout spawning in the upper reaches of the Highland Water, Ober Water, Dockens Water and Avon Water where the water was less than 1m wide and 10cm deep. Eels (*A. anguilla*) were also common and abundant and often comprised the highest biomass in a reach.

### 6. 5. 2. Habitat structure and fish communities

Both the physical structure of the Highland Water stream habitat and the fish community structure form a continuum based on the transition from erosional habitats to depositional habitats (e.g. Prenda *et al.*, 1997). The habitat units and their communities are located at intervals along this continuum depending upon their composition and diversity. The basis of the continuum, however, is a division of the fish assemblage into three groups or guilds associated with different aspects of the physical habitat. The guilds can be primarily divided into two, namely one characteristic of the shallow, gravelly riffles (*C. gobio*, small salmonids <7cm) the other characteristic of the deeper habitats (*N. barbatulus, Lampetra, P. phoxinus, A. anguilla,* larger salmonids). This second guild can be further divided into two groups, namely a "soft substrates" group and a "depth/cover" group. The former (*N. barbatulus, P. phoxinus, A. anguilla, Lampetra* and salmonids 7-15cm TL) were associated with habitat units containing high proportions of silt, leaves and twigs, the latter (larger salmonids, 15-20cm and 20+cm) associated with deep water, roots, undercuts and CWD accumulations. These also included sites with greater distances from the source. The functioning of juvenile fishes as a separate species from adults was previously noted by Moyle & Vondracek (1985) in a small Californian stream.

The separation into the guilds and the wide spacing of species on DCA plots indicates a highly structured, stable low-disturbance community in the Highland water streams, probably with most features of the original fish community of the Forest streams. Such communities are rare in the UK (Maitland & Campbell, 1992). The within-stream variations in community composition and the overall structuring of the stream community can be partly directly related to the accumulation of CWD in that it forms a refuge for the largest salmonids. The presence of these may limit the abundance of other species because they exclude other fish through territoriality or predation (Prenda *et al.*, 1997). The indirect effects of CWD are mostly related to its influence in the formation of pools (Gurnell *et al.*, 1995). The riffle community was the most clearly defined on the basis of both composition and diversity though even with this there were variations.

Andrewartha (1970) noted that "*distribution was* (sic) *merely another aspect of abundance*" and although the two are dealt with separately in this thesis, the within-stream variability in abundance is clearly related to the within-stream variations in diversity and assemblage structure on the habitat sampling unit scale. Despite the fundamental identification of the three broadly definable fish/habitat guilds the practical problems in isolating and sampling habitat units meant that various combinations of species and physical variables were included in each unit. The composition of the habitat units therefore formed a continuum rather than clearly defined groups when viewed on the reach scale and larger. Despite this, the evidence is that there is an underlying deterministic fish distribution on the microhabitat scale in the undisturbed Highland Water streams. Factors based on fish mobility and density and the isolation and size of sampling units producing the apparent continuum on the habitat-unit, reach and stream scales.

There were significant within-reach variations in the diversity of fish populations, pools showed the highest diversities and riffles the lowest. There was a significant linear relationship between habitat diversity and community diversity on the channel-unit scale in the Highland Water streams. As Chapter 4 showed habitat diversity was at its maximum in moderately deep habitats with moderate amounts of CWD and this also applied to fish species diversity. Notably the overall habitat diversity of the reaches (data from all habitat units pooled) was not significantly higher than that of the separate pools but the diversity of the fish assemblage was significantly higher.

The diversity and structuring of fish assemblages in streams based on physical gradients has been reported over many years (e.g. Shelford, 1911; Huet, 1959; Sheldon, 1968; Hynes, 1970). Also increases in stream order have been reported as related to species richness (Evans & Noble, 1979; Paller, 1994). Further, where streams suffer discontinuities either seasonally (Capone & Kushlan, 1991) or as a result of physical barriers (Turnpenny *et al.*, 1987; Martin-Smith, 1998) reaches or channel-unit scale habitats may show specific assemblages. That such assemblages occur and that there are consistent within-stream variations in abundance and community composition even in streams with contiguous habitats indicate that some species show very strong preferences for specific physical habitat whilst others are more flexible in their habitat preferences, i.e. generalists. It is likely, however, that species may have considerable flexibility and can adapt habitat preferences in different streams. For example Prenda *et al.* (1997) describe *S. trutta, P. phoxinus, C. gobio* and *Lampetra* as generalists in a chalk stream because of their lack of affinity with physical habitat types. In contrast, in the Highland Water streams *C. gobio* was strongly associated with riffles, *S. trutta* varied in habitat preference with size while *Lampetra* and *P. phoxinus* were both most associated with depositional habitats.

Prenda *et al.* (1997) identified three habitat guilds associated with a physical continuum. They concluded that the species had partitioned the habitat to minimize potential competition. The physical gradient was essentially from erosional to depositional habitats and was similar in principle to that in the Highland Water streams though the chalk stream did not have significant accumulations of CWD. The difference between the relatively abundant food supply in a chalk stream and the less abundant food in a New Forest stream may, however, increase the pressure on species to be selective and competitive for both space and food where the latter is less abundant.

In a tributary of the Cimarron River, three assemblages were identified definable on a longitudinal gradient of stream size and current velocity and substrate (Williams *et al.*, 1996). Although there was also strong element of downstream distance in the Highland Water the primary gradient axis was dominated by depth and amounts of CWD which also showed strong co-linearity. There was also a significant downstream component in the distribution of CWD in New Forest streams particularly the Highland Water and this distribution clearly has significant effects on the fish community. On the stream scale, species richness and diversity were greater in New Forest streams with less riparian tree cover and CWD than in the wooded streams with high CWD loadings, even in adjacent streams of similar size.

Such disturbance effects have been known for many years. For example changes in temperature in small streams as a result of human activities such as forest clearance and impoundment are well known to cause changes in fish and invertebrate assemblages (e.g. Hall & Lantz, 1969; Hynes, 1970; Langford, 1983). The temperature increases in summer resulting from forest clearance were reported to have caused the displacement of trout in streams of the Smoky Mountains in the USA. The fish moved upstream to cooler and more shaded reaches after clearfelling (Burton & Odum, 1945). Hynes (1970) also quotes Hobbs (1948) and Boussu (1954) who showed that trout were less abundant in densely shaded streams than in more open streams. The effects on fish abundance have been discussed in Chapter 5 and are discussed further in Chapter 8. The increases in abundance and biomass were a result of the increased light penetration and the shift from autochthonous to allochthonous energy sources.

Two conclusions arise from the community analysis. Firstly, the community of the Highland Water streams is highly structured and is probably a pristine lowland forest stream community with no introduced species and no species invading as a result of disturbance. It meets important criteria for a deterministic community (Grossman, 1982) in that first, the species are mostly morphologically dissimilar, only *P. phoxinus* and *S. trutta* being similar in form. Secondly, there is clear spatial segregation either in relation to substrate or channel structure, to the extent that small salmonids (<7cm TL) and *P. phoxinus*, though morphologically similar, show distinctly dissimilar habitat preferences, the former mainly shallow riffles, the latter deeper pools. *C. gobio, N. barbatulus, A. anguilla* and *Lampetra* sp. separate out either because of morphological differences (see Maitland & Campbell, 1992) or because of habitat preferences. Connell (1980) noted that low diversity was probably also characteristic of communities which had evolved to reduce competition. The persistence of the Highland Water community is not known as no records exist further back than a few decades (Environment Agency, Public Register, 1988-1998) and the dietary habits are as yet not fully investigated. The indication from the ordination analysis is that the channelized upper reaches with more shallow, riffle areas do not contain different species from the more sinuous lower reaches, though there are differences in relative abundance with distance downstream of the source.

The Highland Water community is clearly part of a typical forested stream ecosystem with the main energy sources being allochthonous materials from the woodland. The stability of the community probably relies on dense shade, low summer temperatures, low disturbance and lack of instream weed. The effects of artificial channelization cannot be determined fully.

The removal of the canopy and riparian woodlands over many centuries has led to the destruction of this pristine community in streams adjacent to the Highland Water. A replacement community has developed as part of an ecosystem fed by higher primary production. The evidence here is that in the open (disturbed) streams, the criteria of a stochastically controlled, disturbance community are partly met in that species with similar morphology coexist (e.g. larger cyprinids) and spatial segregation is not as clear as for the wooded streams. Further studies of these contrasting communities are in progress (Langford & Quinlan, *unpublished data*)

It follows from the comparisons that further clearfelling, which allowed greater light penetration, weed growth and higher levels of primary productivity, would eventually allow the invasion of species in to the Highland Water streams from the adjacent open streams and would destabilise the pristine community. It is also evident that further accumulation of CWD in the upper reaches and the reduction of shallow riffles could shift the guild species characteristic of slow deep water upstream, where they would replace the assemblages of gravel riffles. There are of course gravel riffles and pools further downstream in these rivers but to date the quantitative composition of these downstream reaches is unknown. The optimal area of gravel riffles to CWD and pool habitat is also as yet unknown. Provisional models on which the predictions of effects of changes in the stream channels might be based are outlined in Chapter 8.

# CHAPTER 7

# THE DISTRIBUTION AND ABUNDANCE OF SALMONIDAE IN RELATION TO WOODY DEBRIS IN LOWLAND FORESTED STREAMS

## 7. 1. INTRODUCTION

There have been many studies of the relationships of Salmonidae to the physical structure of streams in which the presence of coarse woody debris (CWD) has been perceived as an important factor (e.g. Binns & Eiserman 1979; Bovee, 1982; Milner, 1983; Murphy *et al.*, 1984; Heggenes & Saltveit, 1990; Heggenes *et al.*, 1990; Ault & White, 1994; Inoue *et al.*, 1997; Inoue & Nakano, 1998; Urabe & Nakano, 1998; Eklov *et al.*, 1999). Much of the information on the relationship between physical habitat and salmonids has been reviewed by Milner *et al.* 1985, Heggenes 1988a, and Korman *et al.* 1994. Despite the amount of research to date no universally acceptable model has been developed which can predict salmonid abundance over a wide range of stream conditions (Elliott, 1994) though some models are broadly applicable to streams within limited regional or physical limits (e.g. Bovee, 1982; Milner *et al.*, 1985; Korman *et al.*, 1994; Milhous, 1999).

Even the most widely used physical modelling methods, using incremental flows (IFIM) and habitat simulation (PHABSIM) (Bovee, 1982; Orth & Maughan, 1982; Stalnaker *et al.*, 1995; Bovee *et al.*, 1998) have serious limitations, particularly based on the known relationships between the physical habitat and fish abundance (Mathur *et al.*, 1985; Elliott, 1994). In most models, the fundamental concept is based on some linear or log-linear relationship between abundance and physical structure but there is little evidence that this is universally true (Elliott, 1994).

It is, however, clear that Salmonidae utilise different parts of the stream habitat at different stages in their life-history (Frost & Brown, 1967; Elliott, 1994) and that there is marked size-segregation with variables such as depth current velocity and cover (see Egglishaw & Shackley, 1982; Kennedy & Strange 1982; Heggenes, 1988a). The biomass of salmonids has also been indirectly related to the effects of CWD on pool sizes and volumes (e.g. Inoue & Nakano, 1998) and the importance of CWD for creating and maintaining salmonid habitat has been stressed in several reviews (e.g. Bisson *et al.*, 1982; Bryant, 1985; Bisson *et al.*, 1987; Bryant & Sedell, 1995). Most of the data have been collected on the reach, channel–unit scale (pools, runs, riffles) or micro-habitat scale (Heggenes, 1988a) and relatively little attention has been given to the separate, direct effects of CWD on the distribution abundance and size-segregation of Salmonidae in relation to accumulations of CWD.

Of the six species found in the Highland Water in the New Forest, *S. trutta* is the most widespread and the only commercially exploited species, mainly as sea-trout. It has been studied more than the other species present as the references in this and previous chapters show (Chapters 5, 6) and thus more comparable data on habitat relationships are available than for the other species. There are also known to be clear differences between the within-reach abundance and habitat preferences of different size groups (e.g. Egglishaw & Shackley, 1982; Milner *et al.*, 1985) and these differences have clear implications for the management of salmonid populations in streams where channel structure is very heavily influenced by CWD accumulation. Further, no data have been published from studies where CWD accumulations have been isolated from other habitat units on the within-reach scale to determine the direct effects on distribution and abundance. For these reasons this chapter deals with salmonids as a separate special case-history.

The overall aim was to quantify the direct and indirect effects of CWD accumulations on the spatial and temporal abundance of Salmonidae on the within-reach or mesohabitat scale (Maddock, 1999) and to determine the role of CWD in their post-larval life-history, particularly effects on size-segregation. On the stream scale, data from adjacent streams are used to compare effects of low CWD loadings and reduced tree canopy on salmonid populations. The data were intended to be used for the future management of Salmonidae in New Forest streams and to provide guidance for the management of CWD in forested streams in other regions.

## 7. 2. DESCRIPTION OF THE SITES AND METHODS

The main sampling sites for the within-reach studies were in the Highland Water, Bratley Water and Bagshot Gutter in the southern central area of the New Forest (see Chapter 2, Fig.2.1) and the locations and dimensions of each sampling unit are given in Appendix I. The streams are described in detail in Chapter 4 and are referred to in this text collectively as the Highland Water streams. The sampling period was from September 1996 to February 1998 with additional sampling at intervals until February 2000. Sampling in 1996-98 was typically at monthly intervals using replicate sites, though river flows did not always allow access at the chosen times. The hydrograph for 1996-98 is shown in Chapter 3 (Fig.3.2) where sampling dates are also indicated.

Full details of the methods of site, selection, sampling and data analysis are given in Chapter 3. The main difference between this and studies in other streams is that the CWD accumulations were isolated from the channel-units in which they were located, generally pools, and treated as a separate habitat unit. Definitions are given in Chapter 3. Habitat sampling units were selected *a priori* on the basis of visually recognisable fast and slow water channel units (riffles and pools) and CWD accumulations. The physical habitat structure and diversity of each habitat unit was analysed in relation to CWD and its relevance to fish (Chapter 4). The physical differences between the habitat units were quantified and compared using dimensions and the occurrence of substrates and structural features. Two basic habitat types were identified namely shallow riffles and deep-water habitats (Hawkins *et al.*, 1993). The second of these was sub-divided into pools (without CWD accumulations) and CWD habitat units themselves (see Chapter 3, Fig. 3.1).

The physical data from each habitat unit were analysed using detrended correspondence analysis (DCA) and showed that each habitat unit was located along a physical continuum which ranged from shallow gravel-based riffles to the deeper pools in which most CWD accumulations occurred (Chapter 4). This basic erosional to depositional gradient was subsequently used to determine the factors influencing spatial variations in abundance and community structure of the fish populations on the within-reach and reach scales (Chapters, 5, 6). Chapters 5 and 6 include some data on the salmonids but concentrate mainly on the whole fish assemblage. Temporal data on fish abundance and size are compared initially on the basis of the monthly samples but to improve replication pairwise comparisons are made on the basis of pooled seasonal data (see Chapter 3).

The Highland Water system is known to carry high loadings of CWD relative to other streams in the New Forest (Gregory *et al.*, 1985, 1993) (see Chapter 4). To determine the effects of CWD on the stream-scale preliminary comparisons were made with adjacent streams, namely the Ober Water and Dockens Water. Both are streams with low CWD loadings though they also have differences in the amounts of woodland in the catchments and riparian canopy. Descriptions of the streams are given in Chapter 4.

All individual salmonids were weighed and total and standard lengths measured. Total length was measured from snout to the tip of the dorsal part of the caudal fin with the tail spread in the swimming position. As with the abundance data for other species the salmonid abundance in the three habitat types are initially compared on the basis of numbers or biomass per unit area which is the dimension most commonly used (e.g. Bohlin *et al.*, 1989). Because fish live in a three dimensional environment comparisons of abundance are also made on the basis of numbers or biomass per unit volume for interpretation.

## 7. 3. RESULTS

### 7. 3. 1. Relative abundance of salmonids in catches

Salmonidae represented approximately 17% of the total numbers and 53% of the total weight of all fish caught in the Highland Water streams (see Chapter 5, Table 5.1). Of the 1105 salmonids caught only 7 (<0.5%) were positively identified as brown trout. All of these were over 3 years old but it was not possible to separate indigenous fish from those which had been introduced to downstream reaches in previous years (Lappin, *personal communication*). The majority of Salmonidae were considered to be juvenile sea-trout among which smolts were identified in catches between February and April. For this chapter all Salmonidae are considered as a single taxon irrespective of their migratory habit.

### 7. 3. 2. Size distribution

The length-frequency distributions of all salmonids caught in riffles, pools and CWD habitats in each monthly sampling are shown in Fig. 7.1 and the median and quartile lengths shown in Fig. 7.2. Although there is a very large overlap in size ranges, the figure indicates the differences in the size of fish found on average in the three habitat types. Because of the multi-modal size-frequency distributions such simplistic comparisons must be considered with caution. Median lengths were significantly lower in riffles than in pools for 8 out of 12 samples (Table 7.1) and lower than CWD reaches for 7 of 12 months. The differences in median lengths between pools and CWD were significant for only 2 of 12 months. The pooled data from all samples for all dates from each of the three habitat types showed highly significant differences between median lengths (Fig.7.3) (KW = 35.4, df = 2, $p<0.001$).

The median monthly standard deviation (Table 7.2), size-range and sample variance were significantly ($p <0.001$) smaller in riffles than in pools and CWD but there was no difference between pools and CWD. The size distributions of the riffle samples were more skewed than either of the other habitats and the kurtosis of the riffle sample distribution was greater than those of the other

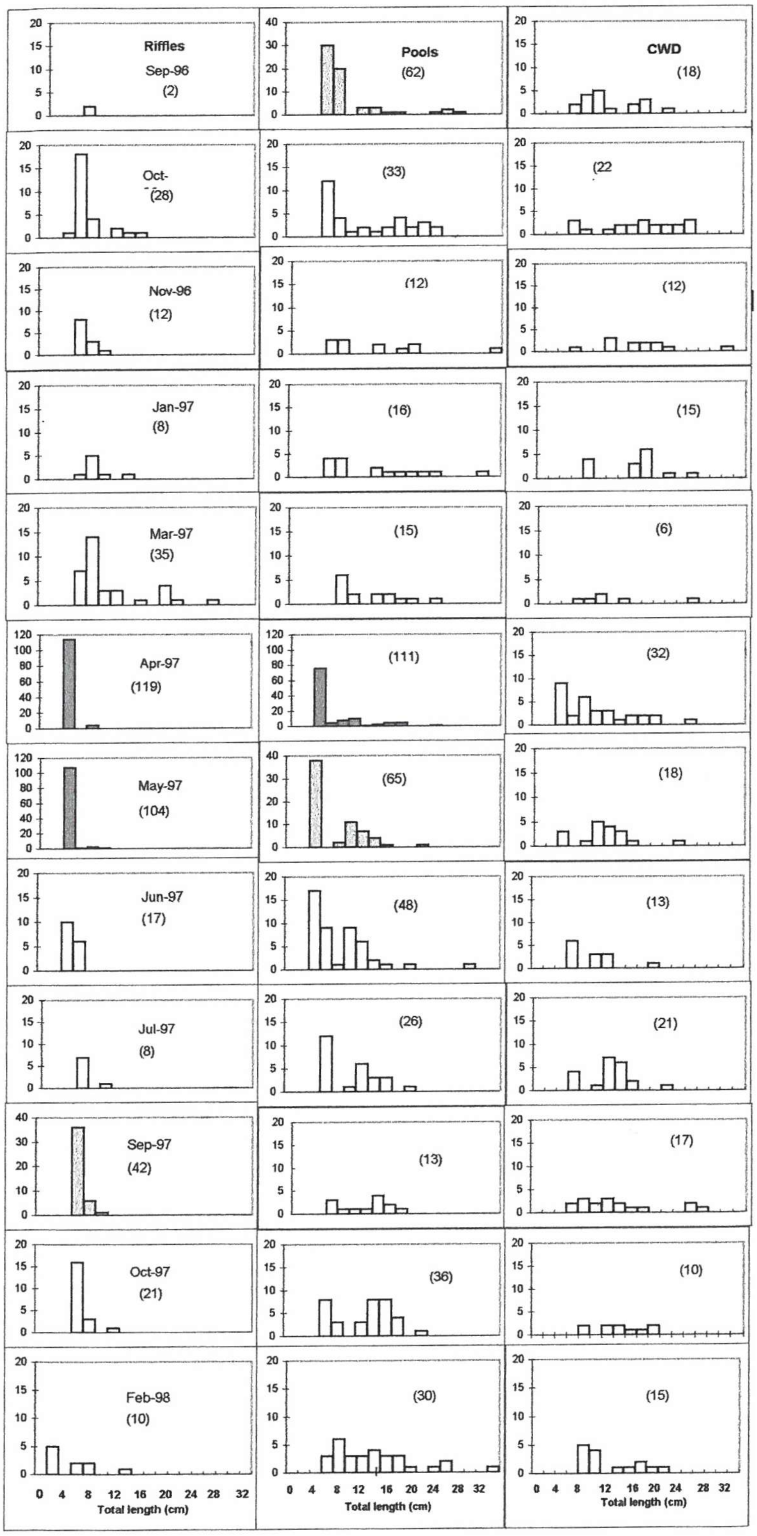


**Fig.7.1** Length-frequency distributions of all Salmonidae caught from riffles, pools and CWD habitats in New Forest streams,September 1996 - February 1998.
*( ) = number of fish measured  Shaded columns indicate changed scale on y-axis*

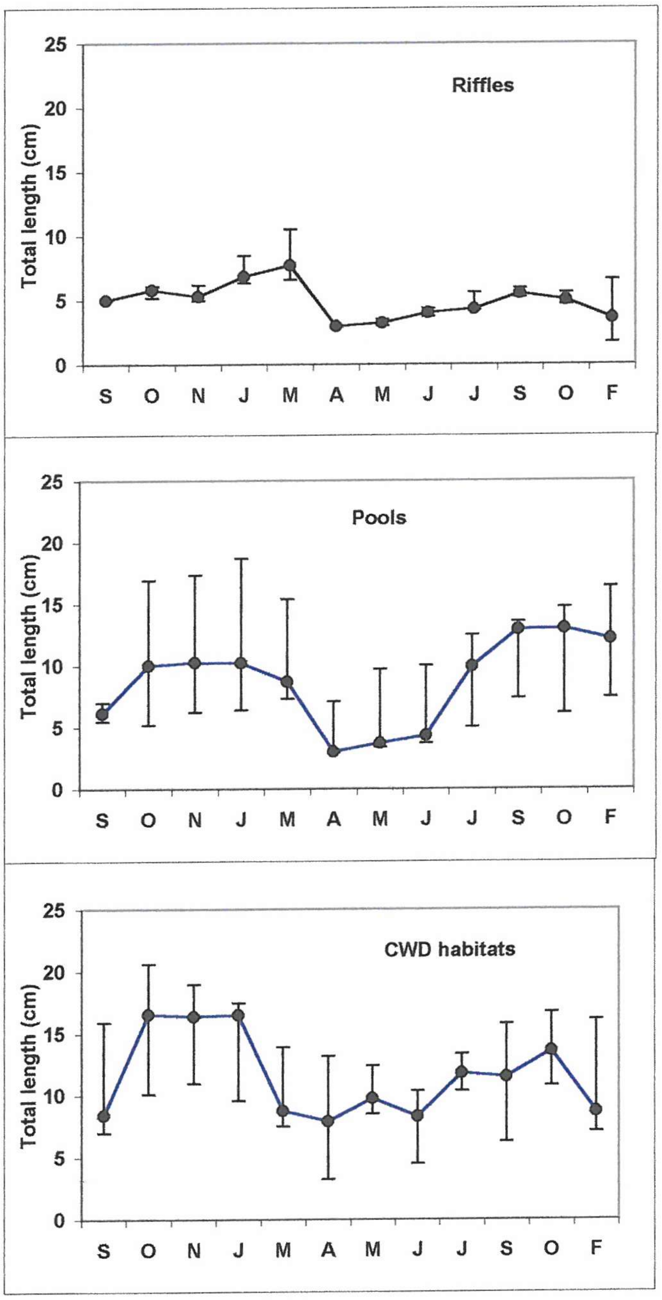


**Fig. 7.2. Median and quartile lengths of all Salmonidae caught from riffles,pools and CWD habitat units each month in Highland Water streams, 1996-98**

*(no samples were taken in December 1996, February, 1997, August, 1997, November, December 1997, Januaty, 1998 because of spates in the streams)*

**Table 7.1. Results of comparisons of lengths of all Salmonidae caught in each month from riffles, pools and CWD habitat units in Highland Water streams. The data were analyzed using the Kruskal-Wallis analysis of variance on ranks *(a-priori)* and Dunn's *a posteriori* pairwise tests. Figure 7.2 shows the median and quartile lengths.**

| Month | Riffles/Pools | Riffles/CWD | Pools/CWD |
|---|---|---|---|
| September 1996 | NS | < | < |
| October 1996 | NS | < | NS |
| November 1996 | < | < | NS |
| January 1997 | NS | NS | NS |
| March 1997 | NS | NS | NS |
| April 1997 | < | NS | < |
| May 1997 | < | NS | NS |
| June 1997 | < | NS | NS |
| July 1997 | < | < | NS |
| September 1997 | < | < | NS |
| October 1997 | < | < | NS |
| February 1998 | < | < | NS |

*< or > indicates that the median length of the fish caught from the first of the habitat pair is less or greater than that of the fish from the second ($p<0.05$) NS indicates that the difference is not significant ($p > 0.05$)*

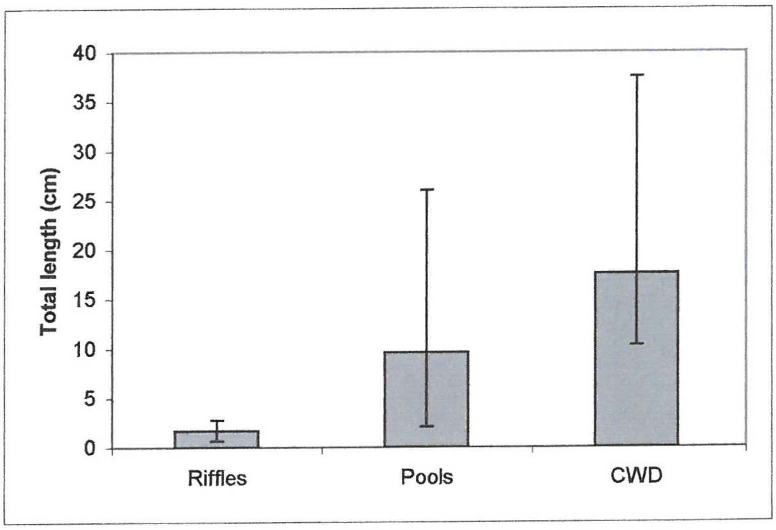


**Fig 7.3** **Median and quartile total lengths of all Salmonidae from all samples taken from Highland Water streams, 1996-98**

Table 7.2 Comparison of measures of variation in the data from analysis of lengths and length-frequencies of Salmonidae from Highland Water streams

| *Habitat* | *Riffles/Pools* | *Riffles/CWD* | *Pools/CWD* |
|---|---|---|---|
| Measure | p | p | p |
| **SD** | 0.001 (<) | 0.001 (<) | NS |
| **Svar** | <0.05 (<) | <0.05(<) | NS |
| **Skewness** | 0.02 (>) | NS | NS |
| **Kurtosis** | NS | <0.05 (>) | NS |
| **Range** | <0.05 (<) | p<0.05 (<) | NS |

*(<) = measure greater for fist named habitat than second*
*(>) = measure for second named habitat greater than first*
*a-priori test was Kruskal-Wallis, a posteriori, Dunn's pairwise test*

habitat types. Clearly the smaller fish showed a clear preference for the riffles. Both pools and CWD habitat units contained a wider size range of salmonids but on average the fish were larger in CWD than pools.

Samples from the riffles were dominated by fish under 7cm total length and by February/March the median length of the previous year class was approximately 7 cm TL. Apart from March 1997 and February 1998 the frequency distributions for the riffles were unimodal and biased toward small fish, mostly under 1 year old. Larger fish only occurred where the riffle sampling units included areas of deeper water (see Chapter 3). In pools without CWD accumulations the fish over 7cm were more numerous in every month than in riffles. The frequency distributions were multi-modal but still biased toward smaller fish in most months. In the three months following recruitment from the post larval stage (April, May and June) the proportions of fish < 7cm long in the pools were larger than other months. Overall, pools contained fish over the almost whole size range found in the Highland Water streams. The CWD habitats, in contrast, showed more evenly distributed size ranges but with less bias toward the smaller fish. Samples from CWD habitat units showed bias toward the smaller fish only in April 1997 when the numbers of small fish were at their greatest in all habitats.

Four size groups of salmonids were defined for further analysis, namely <7cmTL, 7-15cm TL, 15-20cm TL and 20+cm TL. No fish were aged for this analysis and the size-groups were defined from a visual analysis of the length-frequency distributions of all fish caught from all habitats each month (Fig.7.4) and from previously published data (Mann, 1971). The groups were selected to represent the four post larval age/life-history groups, namely 0+, 1+, 2+ and larger smolts/adults. Table 7.3 shows the composition of the catches in relation to specific size-groupings. Salmonids under 7cm TL (0+group) constituted 78.8%, 39.2% and 24.5% of the total numbers caught in riffles, pools and CWD habitats respectively. Conversely, fish over 20cmTL constituted 1.8 %, 5.8 and 10.7% of the respective total catches. Compared with the size-distribution of the totalled catches of which fish under 7cm comprised 51.7% and fish over 20cm 4.8% of the catch, it is clear that the bias is toward small fish (<7cm) in the shallow riffles and large fish (15cm+) in the CWD habitat units. The 7-20cm TL size-groups

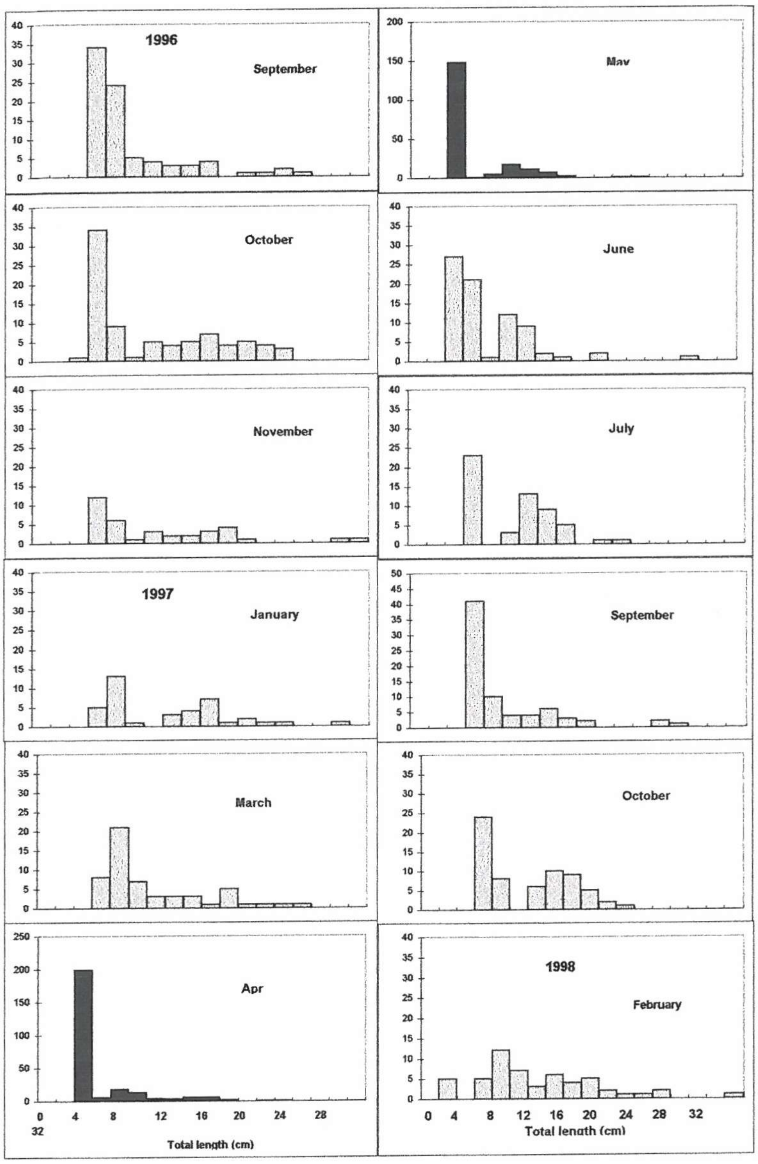


Fig.7.4. **Length-frequency distributions of all Salmonidae caught from Highland Water streams each month. 1996-98**
***(black shaded columns are where the y-axis scale is different to allow for new recruits)***

**Table 7. 3. Percentage size-composition of all Salmonidae caught from riffles, pools and CWD habitat units in Highland Water streams in comparison with the total size-composition from all samples**

| Size-group TL cm | Riffles | Pools | CWD | All |
|---|---|---|---|---|
| < 7.0 | 78.8 | 39.2 | 24.5 | 51.7 |
| 7.0 – 15 | 8.3 | 28.9 | 42.3 | 23.5 |
| 15.1 – 20.0 | 1.2 | 8.8 | 18.9 | 7.8 |
| 20.1 + | 1.8 | 5.8 | 10.7 | 4.8 |

summed, were almost twice as abundant in relative terms in the CWD habitats than in the pools with little or no CWD. Almost 30% of the salmonids from the CWD habitats were more than 15cm TL compared with under 15% from pools and 3% from riffles. Pools with small to moderate amounts of timber debris but not three dimensional matrices were inhabited by the whole range of sizes. $\chi^2$ tests showed that the three habitats contained significantly different size-distributions, (riffles/pools, $p = 0.238$, pools/CWD, $p = 0.213$, riffles/CWD, $p = 0.213$).

### 7. 3. 3. Population Density

There was a clear monthly variation in salmonid densities in riffles, pools and CWD habitat units (Fig. 7.5). The maximum abundance occurred in the period from April to August. This was the period when the new recruits to the catches appeared. Two way ANOVA on ranks (General Linear Model) for the population density data showed significant differences in mean values between habitats ($F = 13.7$, $df = 2$, $p<0.001$) and between months in each habitat ($F = 8.68$, $df = 11$, $p<0.001$) but there was no significant interaction between date and habitat ($F = 1.036$, $df = 22$, $p = 0.43$) (Table 7.4). Thus any differences in density between the habitat unit types were consistent and not dependent on the month of sampling. Pairwise tests were not conclusive for some months because low sample numbers reduced the power of the tests.

Therefore the statistical tests were carried out on the data combined into seasonal groups (Fig. 7.6). Two-way analysis of variance on the ranked seasonal data (General Linear Model) confirmed that there was a significant difference in the mean densities between the different habitat unit types ($p<0.001$) irrespective of the differences between seasons ($p<0.001$) and there were no significant interactions between season and habitat type ($p = 0.43$, $p = 0.39$, $p = 0.43$ respectively). *A posteriori* pairwise analysis (Table 7.5) showed, however, that the differences between habitats were not consistently significant ($p <> 0.05$) throughout the year. In winter salmonid densities in the CWD habitats were significantly greater than in the pools but the differences between pools and

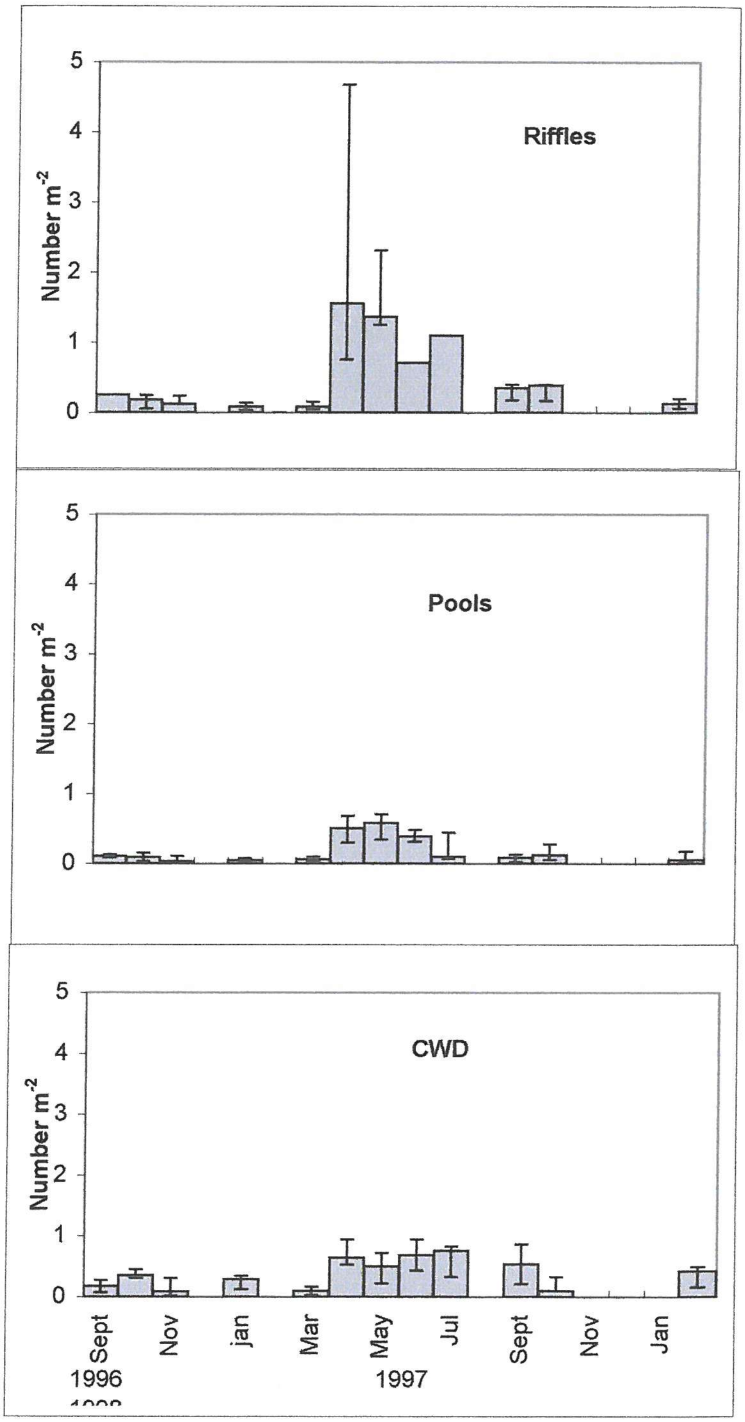


**Fig.7.5** **Median and interquartile estimated population densities of all Salmonidae from riffles, pools and CWD habitat units in Highland Water streams.1996-98 (as numbers m$^{-2}$)**

**Table 7.4. Results of the two-way analysis of variance on ranks of population densities (as numbers $m^{-2}$) of Salmonidae from riffles, pools and CWD habitat units of Highland Water streams.**
(General Linear model)

| **Source of Variation** | **DF** | **SS** | **MS** | **F** | **P** |
|---|---|---|---|---|---|
| Hab unit | 2 | 4064.555 | 7032.278 | 15.254 | <0.001 |
| Date | 11 | 125523.700 | 11411.245 | 10.220 | <0.001 |
| Hab unit x Date | 22 | 26158.463 | 1189.021 | 1.065 | 0.394 |
| Residual | 126 | 140692.164 | 1116.604 | | |
| Total | 161 | 354137.000 | 2199.609 | | |

NB. The differences between habitat types are independent of month of sampling. For pairwise comparisons see seasonal analysis, Table 7.4.

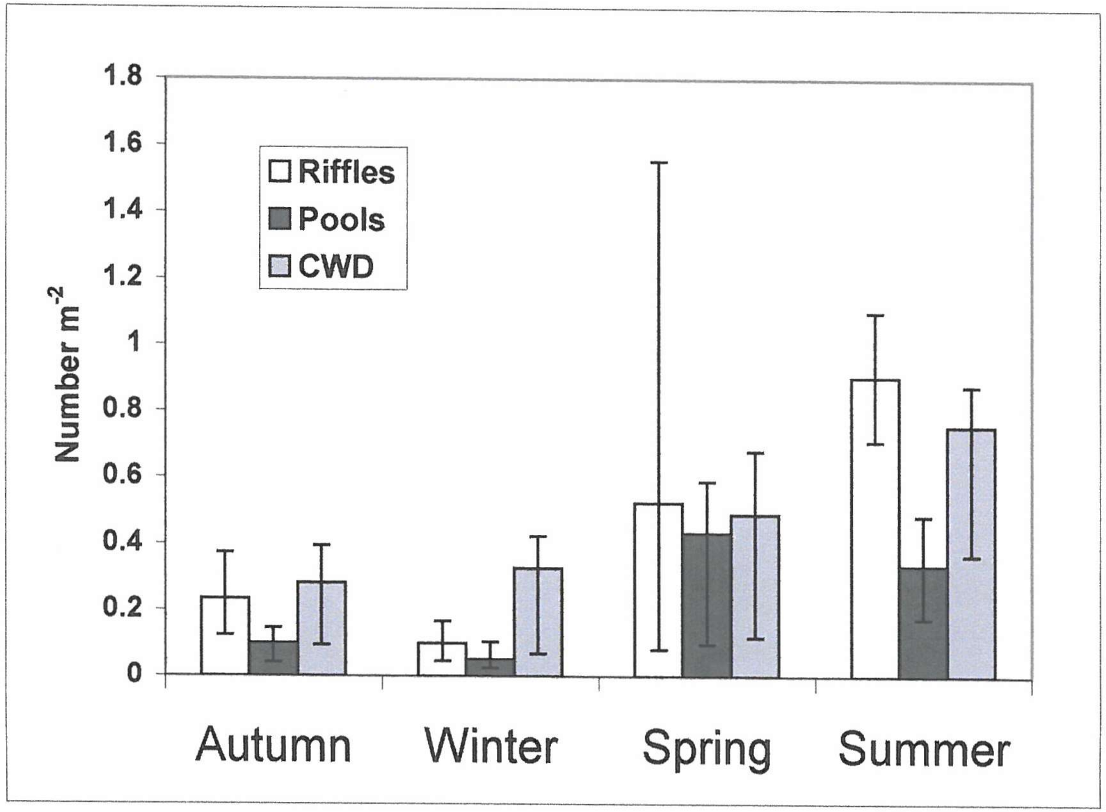


Fig.7.6 **Seasonal estimated population densities of all Salmonidae in riffles, pools and CWD habitat units in Highland Water streams, 1996-98**

**Table 7.5 Results of comparisons of estimated population densities of all Salmonidae in riffles,pools and CWD habitats in defined seasons. Highland Water 1996-98 (significant at $p<0.05$)**

| Winter | Spring | Summer | Autumn |
|---|---|---|---|
| **W>P** | NS | **R>P** | **R>P** |
| R/W | NS | R/W | R/W |
| R/P | NS | P/W | P/W |

W= CWD habitat units, R=riffles, P= pools

*Bold = significant, NS = not significant, a-priori-Kruskal-Wallis, a posteriori- Dunn's test* riffles and between CWD and riffles were not significant. In spring there was no significant difference between densities in any of the habitats. In both summer and autumn only densities in riffles were significantly greater than in the other two habitats. Overall, using the pooled data for the whole sampling period (Fig. 7.7) median densities per unit area were significantly different (Kruskal-Wallis, $H' = 11.074$, $df = 2$, $p = 0.004$) in the three habitat unit types being higher in riffles and CWD habitats than in pools ($p < 0.05$).

It has already been shown that the percentage composition of size-groups differed significantly between riffles, pools and CWD habitat units. These were confirmed by the actual density differences. One-way analysis of variance by ranks (Kruskal-Wallis test) showed that the density of salmonids under 7cm was significantly different overall in the three habitat types ($p<0.001$). Pairwise analysis (Dunn's multiple pairwise test) showed that densities in the riffles were significantly greater than in pools and CWD reaches and that densities in pools and CWD reaches were not significantly different ($p<0.05$) (Table 7.6). Fish in the 7-15cm and 15-20cm ranges were in significantly higher densities ($p<0.05$) in the CWD reaches than in the pools or riffles but densities in riffles and pools were not significantly different ($p>0.05$). The numbers of fish over 20cmTL were generally small but an analysis of the data showed that the densities were significantly greater in the CWD reaches than in the riffles but that there was no significant difference between the other two habitats. The data are as yet sparse but the indications are that the larger fish utilise both deeper water habitats but with a tendency for the largest fish to use the CWD habitat units in preference to the pools. This is analysed further in a later section of this chapter.

Thus the differences in overall density of salmonids in the three habitat types are clearly a function of the densities of the four size groups and the apparent redistribution of the fish over 7cm TL. The 0+ group clearly dominated the riffles from April to August and smaller numbers of these also occurred in the pools and CWD habitats. In the autumn and winter densities were lower in riffles and pools than in the CWD habitats and size segregation between pools and CWD habitats was not as marked. The indication was, therefore, that salmonids over 7-9cm

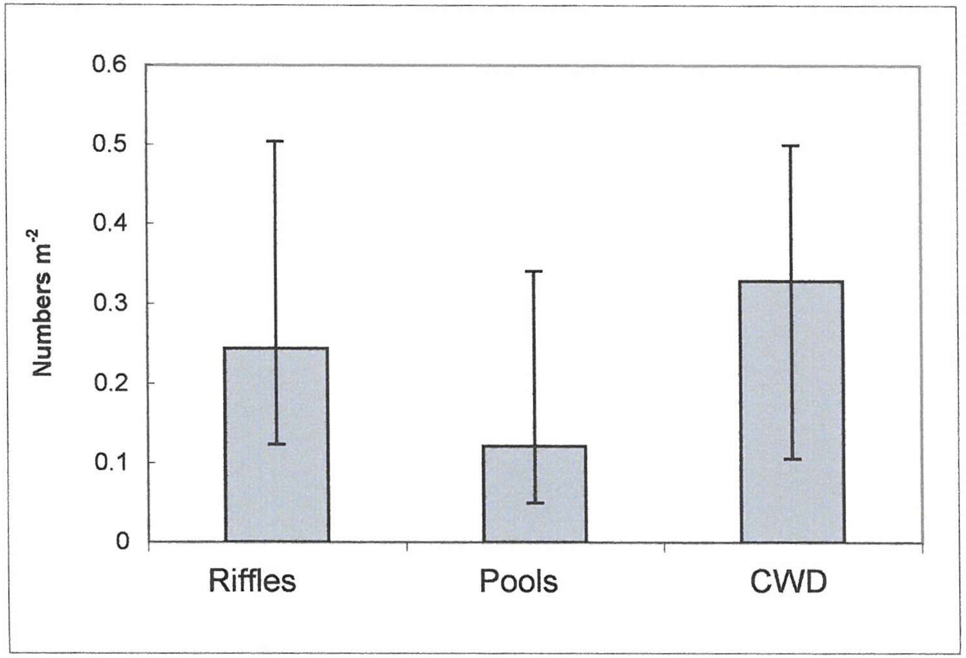


Fig.7.7 Median and quartile densities of all Salmonidae (as numbers $m^{-2)}$ from riffles pools and CWD habitat units in Highland Water streams. 1996-98

**Table 7.6 Results of comparisons of the estimated population densities of four size-groups of Salmonidae in riffles, pools and CWD habitats in Highland water streams. (Data pooled from all samples)**

| <7cm | 7-15cm | 15-20cm | 20+cm |
|---|---|---|---|
| **R>W** | **W>R** | **W>R** | **W>R** |
| **R>P** | **W>P** | **W>P** | **W/P** |
| W/P | P/R | P?R | P/R |

*(see Table 7.5 for explanatory notes)*

long move into the deeper water or into suitable cover, probably during early spring as the new recruits appear in riffles.

Spearman rank correlation showed a significant correlation (Coeff. = 0.356, n = 162, $p < 0.001$) between the density of fish per $m^{-2}$ and the estimated volume of CWD in habitat units (see Chapter 3). In contrast, there was no significant correlation between densities and CWD volumes when the fish data were expressed as numbers per $m^{-3}$ ($p = 0.949$). This suggested that the three dimensional CWD matrix itself did not provide additional physical refugia for higher numbers of individuals in a habitat unit compared with pools without a CWD matrix.

### 7. 3. 4. Biomass

The variation in biomass between months did not follow the same pattern as for density (Fig. 7.8). There was an absence of clear peaks in riffles, pools and CWD habitat units. Two-way analysis of variance on the ranked data (General Linear Model) (Sigma-Stat v2.03) showed significant differences between the habitat units ($p<0.001$) but no significant differences between months (Table 7.7). There was no evidence of interaction between months and habitat types. After pooling the biomass data for four seasons to improve replication (Fig. 7.9) the statistical tests confirmed that the differences between habitat types were dependent on season and that the differences were not consistent throughout the year. In spring and summer only the differences between CWD and riffles were significant ($p<0.05$) (Table 7.8). No significant differences were found at all in winter. In contrast, in autumn, the estimated biomass of Salmonidae in CWD sections was significantly greater than in the pools ($p<0.001$).

The median overall population biomass estimated from all pooled samples from each habitat unit type (as $gm^{-2}$ of wetted area) (Fig. 7.10) was significantly greater in the CWD sections ($p<0.001$) than in pools or riffles. Biomass in pools was in turn greater than in riffles. The overall differences in biomass are clearly mainly a function of the differences in autumn and this is most likely a

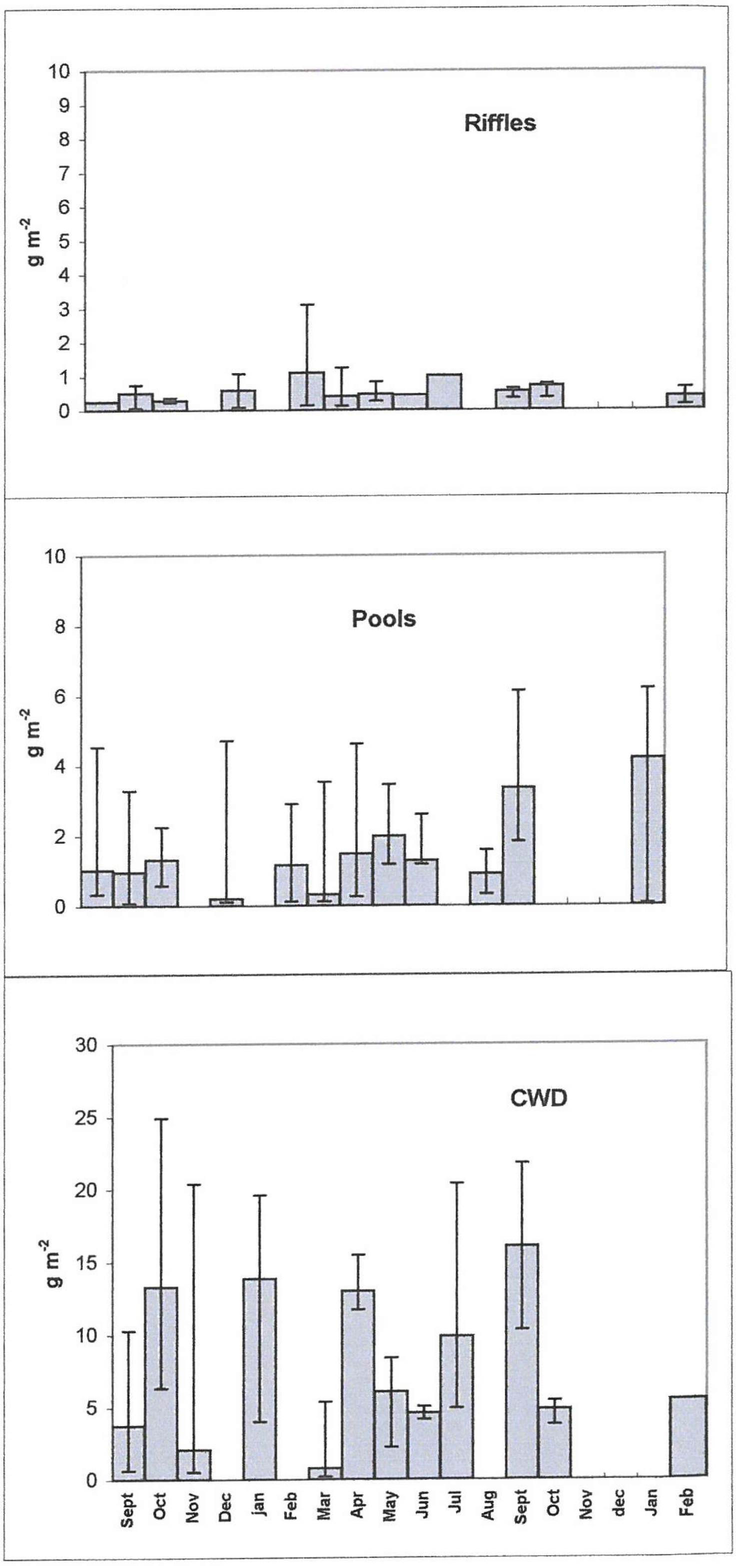


**Fig.7.8 Median and quartile estimated biomass (as g.m-2) of all Salmonidae from riffles, pools and CWD habitats in Highland Water streams. 1996-98**

(note different scales on y-axes)

**TABLE 7. 7 Results of the two-way analysis of variance on ranks of population densities (as numbers per unit volume) of Salmonidae from Riffles, pools and CWD reaches of Highland Water streams**

(General Linear Model)

Normality Test: Passed $(P > 0.200)$

Equal Variance Test: Passed $(P = 0.684)$

| Source of Variation | DF | SS | MS | F | P |
|---|---|---|---|---|---|
| Hab unit | 2 | 64665.193 | 32332.596 | 33.732 | <0.001 |
| Date | 11 | 96242.393 | 8749.308 | 9.128 | <0.001 |
| Hab unit x Date | 22 | 23086.755 | 1049.398 | 1.095 | 0.361 |
| Residual | 126 | 120771.113 | 958.501 | | |
| Total | 161 | 354110.000 | 2199.441 | | |

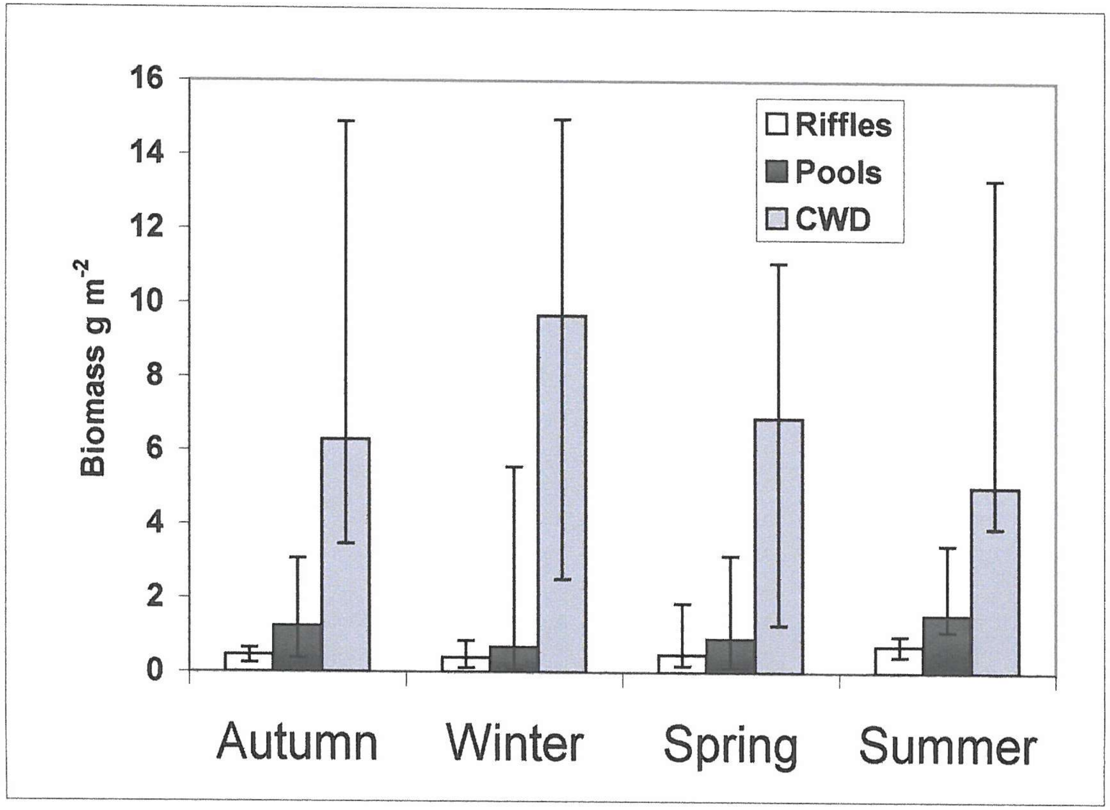


Fig.7.9 Seasonal estimated median and quartile biomass ( as g $m^{-2}$) ( of all Salmonidae in riffles,pools and CWD habitat units in Highland Water streams. 1996-98

**Table 7.8 Results of comparisons of estimated population biomass of all Salmonidae from riffles,pools and CWD habitat units in Highland Water streams (within-reach variation)**

| Winter | Spring | Summer | Autumn |
|---|---|---|---|
| NS | **W>R** | **W>R** | **W>R** |
| NS | **W>P** | P/R | **W>P** |
| NS | R/P | W/P | R/P |

*R =riffles, P = pools, W =CWD habitat units. Significance ,p<0.05*
*a-priori, Kruskal-Wallis, a-posteriori, Dunns pairwise test.*

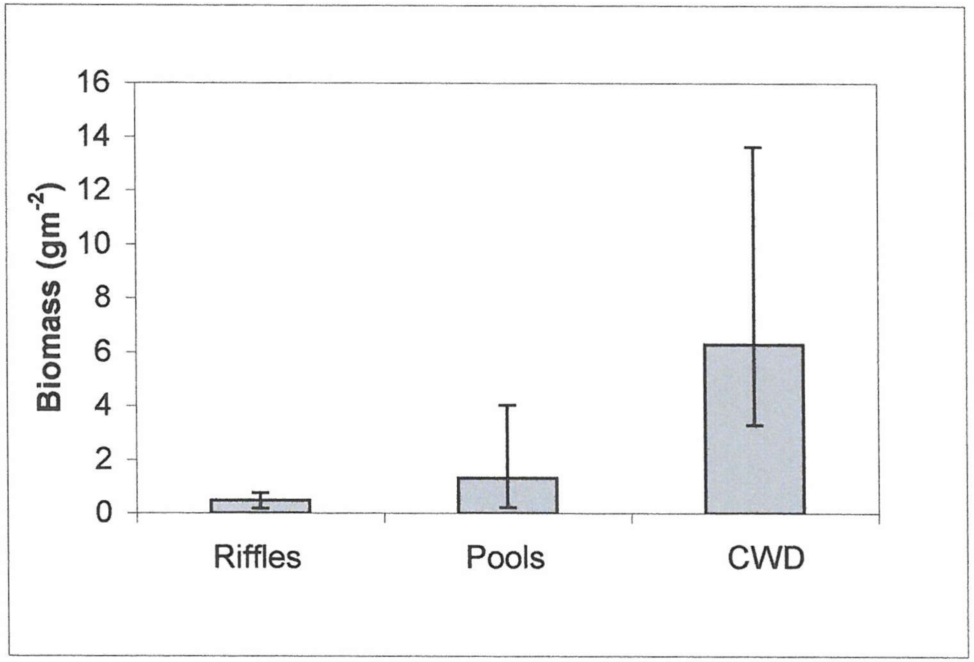


**Fig.7.10** **Median and quartile biomass of all Salmonidae (asgm$^{-2}$) from riffles, pools and CWD habitat units in Highland Water streams, 1996-98**

consequence of larger fish migrating into the river system and the dispersal of fish over 7cmTL into the deeper water habitats.

Comparing the two deeper habitats it seems clear, therefore, that where there was a matrix of CWD in deeper water the overall carrying capacity of an area of stream channel in terms of biomass was significantly greater than pools with no CWD. The overall medians and quartiles for the whole period expressed as biomass per unit standing volume were 8.7g (3.5-15.4), 6.5g (1-15.3) and 16.4 g (6.4-35.5) The CWD habitats carried on average up to 5 times the biomass of pools and up to 20 times the biomass of the riffles per $m^{-2}$ and twice that of the riffles and 2.5 times that of pools per $m^{-3}$. Clearly this difference is a function of the size segregation and the distribution of the different size groups of salmonids in the three habitat types throughout the year. In some samples, particularly in autumn and winter individual large sea-trout caused considerable bias in overall biomass estimates.

Spearman-rank correlation showed significant correlation between biomass expressed as $gm^{-2}$ and the estimated volume of CWD (Coeff. = 0.3345, $p < 0.001$, $n = 162$). The significant co-linearity between depth and the estimated volume of CWD in a habitat unit (see Chapter 4) indicated that either depth or volume could be a significant predictor of biomass but combining the two did not improve the prediction significantly. Median depth showed a greater degree of correlation (Coeff.= 0.401, $p <0.001$) with biomass of salmonids than estimated volume of the habitat unit (Coeff. = 0.334, $p < 0.001$)

### 7. 3. 5. Size-related segregation and channel structure

In Chapter 5 it has been shown that the abundance of the various species in the Highland Water streams could be related to a combination of primary physical variables namely depth, gravel, CWD and distance from source. DCA of the four size groups in relation to the physical variables showed clear separation with groups on the basis of size (Fig. 7.11). The three size-groups of fish over 7cm TL labelled SaA715 (7-15cm TL), Sa152 (15-20cmTL) and Sa2 (>20cmTL) were grouped closely together with physical variables representing deeper water and

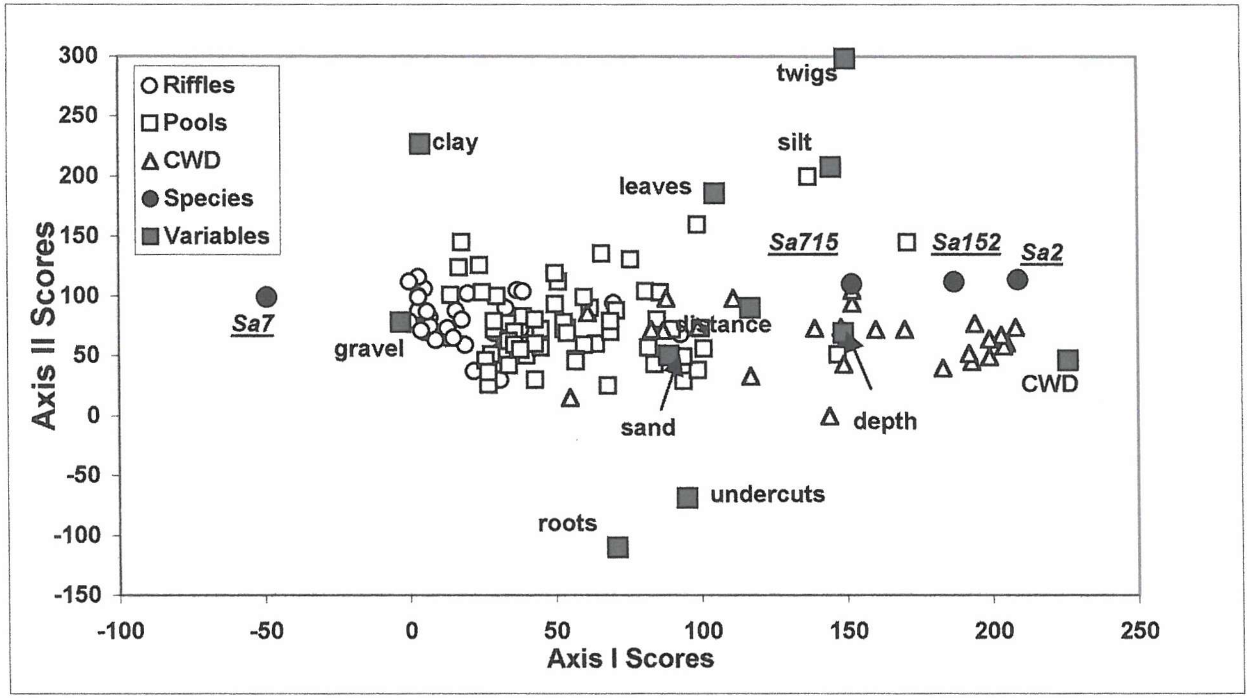


**Fig. 7.11** **DECORANA plot of abundance of four size-groups of Salmonidae in relation to physical variables measured in Highland Water streams, 1996-98.**

***(Open symbols are locations of habitat units relative to the abundance of the fish, physical dimensions and the relative composition of the substrate)***

***(Sa7 = Salmonids TL <7cm, Sa715 = 7-15 cm, Sa152 = 15-20cm, Sa2 = >20cm)***

high volumes of CWD. The largest fish were most closely associated with the higher occurrence of CWD. The adjacent group of physical variables also included distance from the source, finer sediments and non-wood cover. The smaller salmonids (Sa7 = <7cmTL) were grouped almost at the extreme opposite end of the gradient, closely associated with high proportions of gravel and shallow water in the habitat units. It is also clear that small salmonids respond almost as a separate species from the larger groups as far as habitat selection is concerned (see Chapter 6).

Spearman-rank correlations between the densities of the four salmonid size groups and Axis I and II of the DCA on physical variables showed significant negative correlations for Axis I and the smallest fish and significant correlations for the other size groups (Table 7.9). Three size-groups showed significant positive correlations with Axis II but not the largest fish. Axis I explained 62% of the variation in the physical habitat, while Axis I and II together explain some 80% of the variation (Table 7.10). Both Axes were weighted toward the deeper water variables (silt, depth, twigs, leaves, wood) but in Axis II non-CWD cover structures (roots, undercuts) were strong negative influences and gravel had a higher weighting than CWD.

Spearman rank correlation showed that the maximum length of individual fish was highly significantly related to Axis I of the DCA on physical variables (Coeff. = 0.539, $p < 0.001$, $n = 119$). There was also a highly significant negative relationship with Axis II (Coeff. = -0.327, $p<0.001$, $n = 119$). Fig. 7.12 shows the maximum total lengths of fish from individual habitat units in relation to the Axis I scores. Axis I is heavily weighted toward the deeper water with high CWD volumes while gravel is more important and CWD less important in Axis II (Table 7.10).

### 7. 3. 6. Comparisons with other New Forest streams

Data on abundance of salmonids from a range of streams are shown in Fig. 7.13 (Environment Agency, Public Register, 1988-1998). The range of median densities is wide from $0.01m^{-2}$ in the Lymore Brook to >0.1 fish $m^{-2}$ in the Dark

### Table 7.9 Spearman rank correlation coefficients for DCA Axis I and II of physical variables and size groups of salmonids from Highland Water streams

| Groups | Axis I | Axis II |
|---|---|---|
| **<7cm** | ***-0.442*** | **0.279** |
| **7-15cm** | ***0.432*** | ***-0.328*** |
| **15-20cm** | ***0.358*** | ***-0.22*** |
| **20+cm** | **0.217** | -0.113 |

**bold italics = p<0.001, bold, p<0.05**

### Table 7.10 Component scores for physical variables measured in habitat units in Highland Water streams

| Variable | Axis I | Axis II | Axis III | Axis iV |
|---|---|---|---|---|
| W/D | -5 | 123 | 119 | 139 |
| maxD | 169 | 81 | 82 | 18 |
| GRAVEL | -3 | 102 | 87 | 45 |
| SAND | 95 | 90 | 256 | 432 |
| SILT | 165 | 255 | 161 | -51 |
| TWIGS | 168 | 330 | -62 | 119 |
| LEAVES | 126 | 230 | 356 | 35 |
| CLAY | 10 | 250 | -182 | 4 |
| UNDER | 118 | -62 | -49 | 251 |
| ROOTS | 96 | -110 | 44 | -153 |
| WOOD | 225 | 30 | 29 | 80 |
| DISTANCE | 139 | 111 | 89 | 93 |

***(see Fig.7.11 for DECORANA plot)***

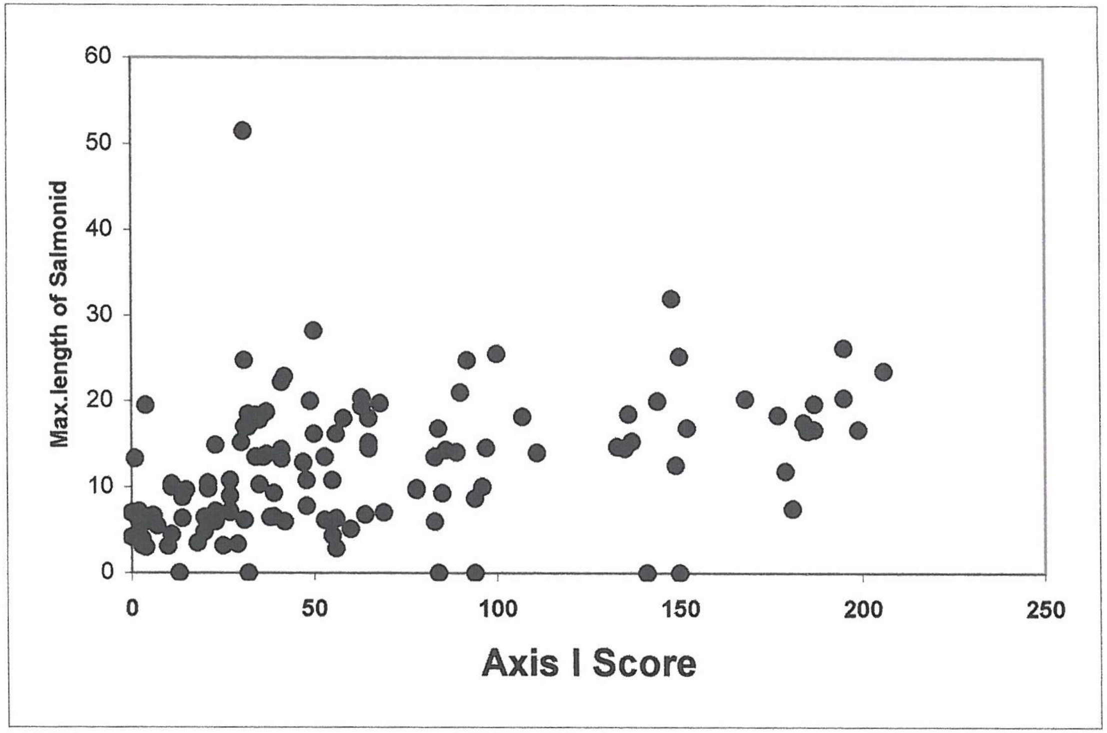


Fig.7.12 **Scatter plot of maximum length of salmonid in a habitat unit in relation to component scores of Axis I on physical variables (see Chapter 4)**

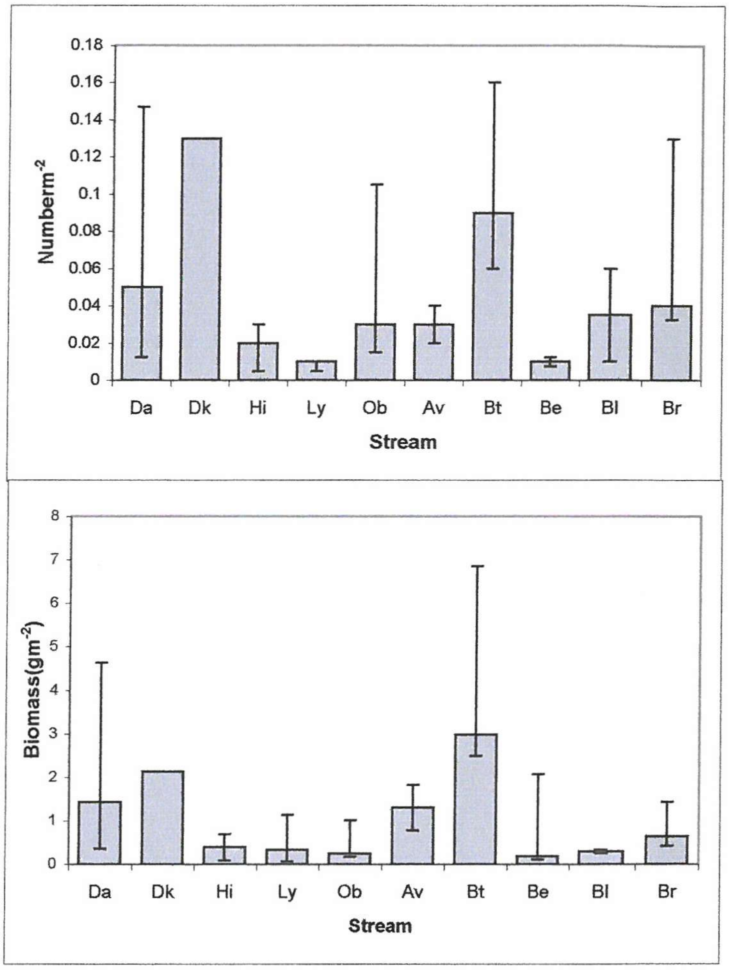


**Fig.7.13** **Median and quartile densities and biomass of Salmonidae (as g.m$^{-2}$) from selected New Forest streams. Data from Environment Agency Surveys, 1993-1998**

***Da=Danes stream, Dk = Dark Water, Hi = Highland Water, Ly= Lymington River,***
***Ob= Ober Water, Av =Avon Water, BT = Bartley Water, Be = Beaulieu River***
***Bl = Blackwater, Br = Bratley Water (see Fig.2.1. For map)***
***Numbers of reaches sampled=, Da,3,Dk,2,Hi,4, Ly4, Ob,3, Av,2, Bt,3, l Be,5,Bl,2  Br,3.***

Water. Most of the EA data originate from single surveys at different sites on the stream but include lengths of stream from 50-400m. The Highland Water is ranked third lowest for salmonid densities and biomass per unit area of the ten streams sampled. The median densities were 0.18 fish $m^{-2}$ and $0.39gm^{-2}$ compared with 0.17 fish and 3.28 $gm^{-2}$ for the overall average in this study. Clearly the factor of nine difference in biomass is related to the higher number of very large fish found in the Highland Water through the year in contrast to the number found in single EA surveys. Both the Danes Stream and Bartley Water show higher biomass of salmonids despite poorer water quality (see Chapter 4).

The EA data also show that reaches of the Ober Water, one of the open streams (see Chapters 5 & 6), carries higher densities but not necessarily higher biomass of salmonids than the Highland Water. Statistical comparisons between the five streams surveyed for this study showed that there was a significant difference between the median salmonid populations ($p = 0.016$) (Table 7.11a) but pairwise tests were not powerful enough to show which pairs were significantly different as a result of low replication. There was no significant difference between median biomass estimates ($p = 0.89$) (Table 7.11b). Comparing the pooled data from wooded and open streams (Table 7.11c,d) showed no significant differences in salmonid densities or biomass between the two groups ($p = 0.10$, $p = 0.497$), though those of the open streams were lower. The data are not fully conclusive because of the low sample numbers from the open streams but the indications are that there were no differences between the densities and biomass of salmonids despite the reductions in tree cover and CWD loadings. Further data would be needed to confirm this. The greater abundance in the open streams (See Chapters 5, 6) was therefore a result of the increases in the standing stock of other species.

Other quantitative data from New Forest streams are scarce. Mann & Orr (1969) found numerical densities of trout at $0.75m^{-2}$ in Dockens Water in August 1967 compared with 0.02 and 0.37 $m^{-2}$ for July and August 1998. In circumneutral upland streams in northern England densities of salmonids in August were found to vary with stream from 0-0.73 fish $m^{-2}$ with a mean value of 0.1 (+/- 0.14) (95% CL) (Crisp, 1993). In a Scottish stream Gardiner & Shackley (1991) found densities of 1.3 to $2.42m^{-2}$. Overall fish production in such streams is much

Table 7.11 **Results of statistical tests on salmonid densities and biomass in five New Forest streams (see text for explanation)**

a) Density (Kruskal-Wallis)

| Stream | N | Median | 25% | 75% |
|---|---|---|---|---|
| Highland | 15 | 0.0856 | 0.0667 | 0.197 |
| Bratley | 4 | 0.285 | 0.27 | 0.785 |
| Bagshot | 2 | 0.519 | 0.325 | 0.713 |
| Dockens | 6 | 0.123 | 0.0654 | 0.17 |
| Ober | 3 | 0.0334 | 0.0253 | 0.145 |

*H = 12.127 with 4 degrees of freedom. (P = 0.016)*

b) Biomass (ANOVA)

| Stream | N | Mean | Std Dev | SEM |
|---|---|---|---|---|
| Highland | 15 | 5.437 | 6.219 | 1.606 |
| Bratley | 4 | 4.328 | 2.662 | 1.331 |
| Bagshot | 2 | 3.118 | 0.995 | 0.703 |
| Dockens | 6 | 4.219 | 3.413 | 1.394 |
| Ober | 3 | 2.638 | 4.212 | 2.432 |

*F=0.263 P=0.89 (NS)*

POOLED DATA FROM TWO STREAM TYPES

c) Wooded vs Open (density)

| | N | Median | 25% | 75% |
|---|---|---|---|---|
| Wooded | 21 | 0.119 | 0.0782 | 0.525 |
| Open | 9 | 0.0893 | 0.0307 | 0.173 |

*T= 103.00 p=0.10 (NS)*

d) Wooded vs Open (biomass)

| | N | Median | 25% | 75% |
|---|---|---|---|---|
| Wooded | 21 | 3.822 | 1.92 | 4.929 |
| Open | 9 | 1.687 | 0.342 | 6.865 |

*T=124, p=0.497, (NS)*

*Density as numbers* $m^{-2}$, *biomass as* $gm^{-2}$

lower than in the richer chalk streams (Mann, 1971). However, comparisons between streams and regions on the basis of different years may be meaningless without some knowledge of the variations from year to year (Elliott, 1994).

## 7. 4. DISCUSSION AND CONCLUSIONS

There were significant within-reach differences in size distribution, population densities and biomass of salmonids at the habitat unit scale in the Highland Water streams which were independent of the seasonal variations. In general, the pools without identifiable CWD accumulations showed the lowest densities of salmonids. Peak densities in riffles were up to an order of magnitude greater than in either of the other habitat types mainly as a result of the recruitment of 0+ fish into the catches in April. CWD matrices did not contain significantly higher overall densities than the pools without CWD but the biomass was significantly greater whether expressed as $gm^{-2}$ or $gm^{-3}$. There were also clear size-related differences in abundance indicating clear habitat segregation by salmonids of different size groups.

Detrended correspondence analysis (DCA) applied to the abundance data for four size groups indicated that 0+ group salmonids (< 7cm TL) clearly occupied shallow gravel riffles in preference to the deeper waters and could be considered essentially as a separate species from the larger salmonids as far as habitat partitioning was concerned. This clearly has implications for the effects of any alterations in habitat structure on the different stages in the life cycle. There was clear correlation between the abundance of the size-groups and physical variables (see also Chapters 5, 6) as expressd by DCA Axis I. There were no significant effects of reduced canopy and low CWD loadings on salmonids in adjacent streams in the New Forest, despite the increased overall species richness and total abundance of fish. Further research on the effects of land use and fish populations is in progress.

There is a general consensus, summarised by Gurnell *et al.* (1995) that the higher the density of CWD, the greater the density or biomass of fish will be present in a stream. Whist this appears to be true on the within-stream scale in some New Forest streams, other published data do not show equivocal conclusions for either salmonid or non-salmonid fish. The causes are mainly the variability in the methodology, variations in the types of stream systems and in the aims of the particular programmes (e.g. Gorman & Karr, 1978; Hortle & Lake 1983; Heggenes, 1988a; Braaten & Berry, 1997; Inoue & Nakano, 1998). Such variations have occurred in many studies of physical habitat and the problems with habitat measurements and selection are discussed in a recent review by Maddock (1999). Unlike many of the earlier studies, this work in the New Forest aimed to determine the direct use of CWD accumulations by quantifying the densities, biomass and size distribution of Salmonidae within and outside definable CWD structures. Thus sampling units were based on sub-divisions of the geomorphological channel-unit scale divisions of streams (Hawkins *et al.*, 1993), a category between macro- and microhabitat scale (Heggenes, 1988a; Maddock, 1999). This use of small scale sampling units was more likely to show discontinuities than units on the large reach scale and higher scales (Bohlin *et al.*, 1989; Angermeier & Smogor, 1995).

Possible reasons for within-reach variations in recorded abundance of Salmonidae include sampling bias, refuge from predators or competitors, optimal foraging position, adverse flow conditions, increased food resources, and biased sampling or catch efficiencies (e.g, Bjornn, 1971; Fausch & White, 1981; Egglishaw & Shackley, 1985; Cunjak & Power, 1986; Nielsen, 1986; Gatz *et al.*, 1987; Heggenes, 1988b,c; Bardonnet & Heland, 1994). In New Forest streams the CWD matrices contain large predators such as eels and larger feeding salmonids, viz. brown trout (Maitland & Campbell, 1992), though the largest salmonids caught were mostly inward, non-feeding migrants. The diet of the smaller salmonids has not yet been investigated but the stomachs of two brown trout caught contained smaller salmonids (Langford, *unpublished data*). Other predators are mainly mink or herons.

The possibility of fright-bias (Bovee, 1982; Thévenet & Statzner, 1999) (see Chapter 3) was not specifically investigated but a consistent positive preference for CWD over the other available refugia would stress the importance of the

CWD. It is, however, unlikely that fish would always prefer the CWD matrix rather than the nearest cover structure. There were no observations of movements of larger salmonids to or from habitat units during sampling but some movement within a habitat unit was observed where fish were not immobilised by the electric current during the fishing runs. Thévenet & Statzner (1999) conclude that fright bias was up to 3.4% in the larger river in which they worked even when sampling units were not enclosed by stop nets. It is likely to be lower where the sampling units are isolated in smaller streams such as those studied here. Catch efficiencies differed between riffles and the other two habitats but there was no difference between pools and CWD habitats.

One of the limitations to the study which could cause bias was that all sampling was done during daylight though this is the case with most published work. The diel variations in fish movements are well documented (e.g. Helfman, 1993) and a number of species are known to make nocturnal movements from day to night habitats (e.g. Frost & Brown, 1967; Clough & Ladle, 1997; Baade & Fredrich, 1998). It is, therefore, likely that the use of cover by large salmonids in the Highland Water was essentially a daylight phenomenon and that similar sampling during the hours of darkness might illustrate a much wider habitat use. This is discussed further in Chapter 8.

Many of the published studies on the relationships of fish to their physical habitat have used reaches of streams defined by length measurements, for example 50m (e.g. Milner, 1983; Milner *et al.*, 1985) or 100m (Moore & Gregory, 1988) or reaches of varying lengths with mixed habitat structure (e.g. Binns & Eiserman, 1979; Fausch & Northcote, 1992; Inoue & Nakano, 1998). Thus the direct role of specific accumulations of CWD as fish habitat or refugia are difficult to interpret and complex analysis has to be applied to establishing some degree of quantitative relationship between the amounts of CWD and the abundance of salmonids (e.g. Heggenes, 1988a; Inoue & Nakano, 1998). There is strong evidence from this work in the New Forest that the importance of CWD for salmonids between 7 and 20cm TL is most likely as a pool-former than as a 3-dimensional refuge as in other streams (Fausch & Northcote, 1992; Inoue *et al.*, 1997; Inoue & Nakano 1998; Urabe & Nakano, 1998). However, for the under-yearling fish (<7cm) the formation of pools and the consequential reduction in riffle habitats is likely to reduce their preferred habitat and total numbers significantly leading ultimately to the reduction of the populations. On the other hand, the strong association between the largest fish and CWD habitat shown by the DCA (Figure 7.11) indicates that this habitat is of importance, most likely as refugia for upstream migrants and larger smolts. The presence of larger, aggressive territorial salmonids might be expected to deter other species and smaller salmonids from occupying the CWD habitat and Prenda *et al.* (1997) note that this was apparent in some chalk streams.

Heggenes (1988a) concluded from his review of the literature that it is not possible to rank the physical variables in stream habitats such that they apply universally to the prediction of salmonid stock densities and biomass, although water depth, water velocity, substrate and cover have been the most studied variables and may well be the most important. Similarly the role of CWD as cover and as habitat modifier has not been universally quantified and it is likely that its function differs with the type and character of the stream. For example, in streams at higher altitudes (e.g. Binns & Eiserman, 1979; Egglishaw & Shackley, 1982; Heggenes & Saltveit, 1990) where rocks and boulders, fast turbulent water or overhanging banks provide shelter from currents, salmonids might be less dependent upon large woody debris as cover than fish in lower gradient streams and rivers with substrata of smaller gravels, sand or silt (Angermeier & Karr, 1984; Braaten & Berry, 1997). In the streams of the New Forest, where the substrata includes relatively few natural particles over about 120mm diameter, the largest instream cover structures are the CWD matrices, though tree roots and undercut banks also provide cover. Ultimately for the survival of the salmonid populations in the longer term the important factor is the optimal proportions of riffle, pool and CWD habitat in any length of stream. This is discussed further in Chapter 8.

The direct use of CWD as a refuge has been well demonstrated in a recent study (Harvey *et al.*, 1999). The authors showed that radio-tagged salmonids caught near or within CWD accumulations moved less than those caught in reaches without CWD. Further, most fish tracked consistently also used the CWD accumulations as shelter from floods and were not displaced. In many studies using direct observation (e.g. Heggenes 1988a; Heggenes *et al.*, 1990) salmonids have been observed using CWD items as refugia or shelter from currents though large rocks, boulders, tree-roots and undercut and overhanging banks were as important in most streams.

In the New Forest streams, water depth and the estimated overall volume of CWD in a matrix were significant indicators of the maximum size of salmonids which may be present in a habitat unit. Depth alone has also been used as a significant predictor of fish size present in a habitat by other authors (e.g. Egglishaw & Shackley, 1982; Milner *et al.*, 1985; Heggenes, 1988a; Harvey & Stewart, 1991).

Depth and CWD volume are correlated because of the scouring of substrates from beneath CWD formations, particularly during spates. This increases depth both within and immediately downstream of the CWD accumulation (Gregory, 1992). The consequence is that the CWD matrix is eventually in or near a deepened pool (Gurnell & Sweet, 1998). The pool is typically maintained by a gravel bar formed downstream of the CWD matrix by the displaced substrate material. The gravel bar in turn will form the basis of a shallow riffle which will subsequently be used by the smaller salmonids.

The importance of CWD as a factor determining the population densities and biomass of salmonids in streams is at times difficult to determine. For example, in comprehensive studies of streams in Northern Hokkaido, Japan, both total cover and CWD cover showed positive significant correlations with density of juvenile salmonids (Inoue & Nakano, 1998). However, water temperature and depth were considered as a major limiting factors (Inoue *et al.*, 1997) and the relative importance of depth and cover including CWD varied considerably among the reach and stream types. Bryant (1985) also found that densities of under-yearling coho salmon (*Oncorhynchus kisutch*) were 6-7 times greater in gravelly side channels where CWD was present but there was no direct relationship between the amount of wood and fish abundance. In this case the CWD acted indirectly to create protected side channels where young fish could shelter. In contrast, Fausch & Northcote (1992) showed that the densities of salmonids were related to pool volumes which in turn were related to the density of CWD in British Columbian streams and they concluded that the removal of CWD reduces fish habitat and production because of the loss of sinuosity and pool volume rather than the loss of direct refugia.

In the New Forest streams there was a clear and significant difference in both median densities and median biomass $m^{-2}$ of salmonids between riffles, pools and CWD habitats, but it was not possible to predict with significance, using least squares regression, either potential density or biomass from either depth or volume of CWD because of the difficulties of homogenizing variance in the data. The significance of the difference between salmonid abundance in the habitat units also varied with the units of measurement used. For example, there was a significant correlation between density and the estimated volume of CWD in a habitat when the data were calculated as numbers $m^{-2}$ but the correlation was not significant when the data were calculated as numbers $m^{-3}$. The reason was that extrapolation to volume so exaggerated the data for the shallow riffles that the difference between the riffle and CWD biomass estimates was obscured. However, the CWD habitats still showed a significantly higher biomass than the pools without CWD ($p < 0.001$) indicating that the increase was real and not a result of statistical artefact or extrapolation.

Despite the clear effects on salmonid size distribution and biomass the overall effect of CWD loading on all life-history stages of the salmonid population is as yet difficult to quantify. The higher biomass in CWD matrices than in non-CWD pools and the preponderance of fish over 1 year old in CWD matrices suggest that this habitat enhances the carrying capacity for fish over 1 year old. In contrast, the formation of deepening pools and potential depositing habitats caused by the establishment of full CWD dams destroys any shallow riffles immediately upstream (Gregory *et al.*, 1993). These riffles are salmonid spawning sites and nursery habitats for the under-yearling fish and the number of these under-yearling survivors will influence the number of pre-smolt fish available to use the CWD matrices (e.g. Elliott, 1994). The riffles are also inhabited by species such as *C. gobio* and used as spawning sites for *L. planeri* but *L. planeri* is also most abundant in the areas of silt and twigs in pools (see Chapters 5, 6).

The re-establishment of gravel riffles downstream of full dams will subsequently produce replacement for the upstream riffles but there may be some inertia in the process and the newer riffles may be unstable for some time after formation. The prediction of the potential effect of non-clearance of CWD from streams is therefore vital to the maintenance of the salmonid population. If, as estimated, the streams contain only 6-7% of the CWD that would be there without clearance (Gregory *et al.*, 1993), the impact of up to 15 times more CWD on the riffle and pool sequence would be the determining factor in the survival of the salmonid population of New Forest streams. Gurnell & Sweet (1998) also show that pool spacing stabilized at about 4 channel widths, compared with 2 channel widths for channels where CWD was not removed. This implies significant reduction in riffle lengths (inter-pool length) and could have serious implications for spawning and the post larval population size of salmonids apart from other species such as *C. gobio.* Lower post-larval populations will clearly eventually lead to lower overall populations. The need for the establishment of optimal CWD loadings for the survival of salmonid populations in the New Forest streams is clear and this almost certainly applies to all managed forest stream systems.

The differences in salmonid biomass between open and wooded streams reflects the increase in overall productivity in these streams where the lack of canopy has led initially to increases in primary productivity. It is clear that in the open streams on the New Forest the presence of a large omnivorous species such as the chub (*L. cephalus*) has not suppressed the numbers or biomass of Salmonidae though further investigation is intended to determine the degree of spatial and dietary competition by these species.

# CHAPTER 8

# GENERAL DISCUSSION: METHODOLOGY, TIMBER DEBRIS AND FISH IN FORESTED STREAMS, MANAGEMENT OPTIONS AND FUTURE RESEARCH

## 8. 1. SYNTHESIS OF MAJOR FINDINGS AND AIMS OF CHAPTER

My research has shown that the fish community of a relatively undisturbed lowland forest stream is highly structured on the within-reach scale and that the six species in the assemblage show significant variations in abundance and habitat preferences on the same scale. Accumulations of woody debris (CWD) clearly influence the distribution and abundance of species significantly on both the within-reach and stream scales. The main direct influence of CWD is the provision of increased habitat for larger salmonids (S. *trutta*) and, to a lesser extent, larger eels (*A. anguilla*). The main indirect influences are the effects on channel structure, depth and substrates, all of which affect the distribution of individual fish species within the community. There was a higher total biomass of fish associated directly with CWD accumulations on the within-reach scale though individual species show contrasting patterns of abundance.

In adjacent streams, where deforestation, grazing and channel maintenance for drainage has occurred over many hundreds of years, the fish community is more diverse, with a higher species richness and is less clearly structured than in the forested streams. There is considerable immigration of species more characteristic of the lower reaches of rivers and a higher total biomass per unit area. The overall effect of the disturbances is the destruction and replacement of the "low-disturbance" fish community. There is also a shift from an ecosystem with low productivity fed by allochthonous energy sources to a higher productivity ecosystem fed by autochthonous energy sources. Immigrant fish species from lower reaches, mainly larger Cyprinidae, form a significant part of the biomass in the disturbed streams. The two separate ecosystems appear to be sustained in adjacent streams despite them being tributaries of the same river.

In the undisturbed, forested stream system, three habitat guilds have been identified, based on species and size-groups. These can be associated with the physical structure of the stream channels, including substrates and CWD accumulations. The evidence from the spatial variations in abundance, species association and ordination of fish data and physical data is that the main processes involved in the distribution of the species in the low disturbance streams are deterministic. The community also meets several of the criteria suggested for deterministic communities, namely low diversity (Connell, 1980) clear spatial habitat segregation and low morphological similarity among the species (Grossman, 1982; Grossman *et al.*, 1982; Moyle & Vondracek, 1985). In contrast, the communities of the disturbed streams show less clarity of spatial separation, higher diversity and more morphological similarity among the resident species.

The assemblage of the forested Highland Water streams is probably that of the original pre-disturbance stream system in the region's lowland forest. It is unlikely to have changed significantly in species-composition as a result of human activities though some changes in relative abundance may have occurred as a result of the physical alterations to the stream channel. Despite the habitat preferences of the species, the quantitative composition of the stream assemblage does not show segregation into discrete categories which can be clearly associated with geomorphological channel units. The communities of the habitat units vary along a continuum on the reach and stream scales which is related to a continuum of physical variables on a gradient from shallow, erosional habitats to deeper, depositional habitats. Habitats with high CWD abundance form the extreme points of the depositional gradient. There is a significant relationship between habitat diversity and fish community diversity on the within-reach scale in the forested streams though on the reach scale diversity does not differ significantly from that of individual pool habitats. The abundance of individual species on the reach scale is mainly dependent upon the relative amounts of CWD and bare gravel in the reach.

As with most other similar research, however, there are limitations to the methodology, data analysis and interpretation that must be considered before accepting conclusions from the results and these are discussed briefly later in this chapter.

A provisional model of the potential effects of changes in CWD abundance on the fish populations is also proposed and finally the wider relevance of CWD accumulation to the management of forested streams and fish populations is discussed. The relevance of the data to the general ecology of fish communities and their conservation is then discussed prior to an outline of the implications of the research conclusions for management of streams of the New Forest.

Studies of fish in running waters have always been subject to a number of limitations related to sampling design and methods (e.g. Cowx, 1983, 1990; Bohlin *et al.*, 1989) and data analysis. In any research of this kind where the natural habitat is sampled rather than using manipulations, choices on scale, timing, frequency and replication will influence the precision and the validity of the data. These choices are often dictated to some extent by external factors, for example stream discharge which prevents access for sampling for periods of any year. Choices of the most appropriate data analysis methods can also influence the conclusions particularly where the data are such that normalisation of the data and homogeneity of variances is difficult to achieve despite various methods of transformation.

## 8. 2. LIMITATIONS TO THE STUDIES

### 8. 2. 1. The scale of the observations, sampling effort and replication

Among the many factors which influence the estimation and description of populations or communities of fish in any habitat are the sampling effort and the spatial scale of the observations (Angermeier & Smogor, 1995; Maddock, 1999). For example observations on the small scale are more likely to reveal discontinuity in distribution whereas larger scale observations are more likely to show continuous patterns. Cummins (1992) also concluded that " *the information gained from intensive small-scale studies, scaled up to broad area and regional* *scale using techniques such as remote sensing or GIS* (Geographical Information Systems) *is the most promising tool for stream and river conservation and management*".

Clearly the choice of sampling unit is likely to influence data collection and the choice must reflect the objectives of the research. Where management data are required, the scale of the "managerial units" is vital (Underwood & Chapman, 1998). Given that CWD is patchily distributed in the stream systems studied here (Gregory *et al.*, 1993) and that the scale of management units is typically at reach or sub-reach level (Cowx & Welcomme, 1998), the scale of the primary sampling units for this research was also set at sub-reach (channel-unit) level (Bisson *et al.*, 1982; Hawkins *et al.*, 1993). In many studies such units would be termed "mesohabitats" to define the scale (e.g. Maddock, 1999). This could equate with "habitat-unit" used as the sampling unit in this work though Maddock (1999) noted that the definitions of mesohabitat vary in different studies and that "*there is no definitive mesohabitat typology available.*" This may also apply to many of the geomorphological categories used for habitat classification and therefore the use of any terms may be criticised unless there are strict definitions attached. In this study, each of the *a priori* sampling unit types (habitat units) is defined and described as fully as possible (see Chapter 3). Even so, analysis shows that the habitat units sampled do not separate out clearly into those categories as selected prior to sampling but form a continuum based on their physical structure and dimensions. The main reason is that the habitat units, as isolated by transverse stop nets across the full width of the channel, will almost always contain elements of other habitat units. For example *a priori* selected and isolated "riffles" may contain small pockets of deeper water or a small silted backwater or an undercut bank which will provide refugia for fish not characteristic of the faster, shallow water and gravels that occupy the greater part of the isolated habitat unit.

Comparing the New Forest stream habitat units with those listed by Maddock, (1999) riffles equate with his "low-gradient" riffles, pools equate mostly with the "plunge-pools" (downstream of debris dams) and dam pools (upstream of debris dams). There is no category comparable in Maddock's list with the CWD habitat units described for the New Forest streams. The unit was, however, a major factor in the analysis of the distribution, abundance and structure of the fish communities and provided data on the direct and indirect importance of CWD matrices to species and to the community. For comparisons between streams on the larger scale, data from sub-channel unit samples were combined to provide larger samples from reach-scale up to catchment level.

In the New Forest streams the restricted species richness of the assemblages and the small size of the streams meant that a relatively large sampling effort was unlikely to miss any species in the sampling reaches except those with extremely sporadic occurrence spatially and temporally. Indeed the species recorded in the streams described here were those recorded by previous authors and in other investigations (Mann & Orr, 1969; Mann, 1971; Environment Agency, Public Register, 1988-1998). In streams with greater species richness (e.g. Williams *et al.*, 1996; Martin-Smith, 1998) sampling effort needs to be tested to ensure that the species richness is accurately reflected. This generally does not apply to UK streams (e.g. Ibbotson *et al.*, 1994; Watkins *et al.*, 1997) or forested streams of first or second orders where species richness is typically low (e.g. Shelford, 1911; Huet, 1959; Hawkes, 1975). Angermeier & Smogor (1995) also concluded that the sampling effort necessary to characterise fish community structure in their streams was inversely related to population density. Their streams contained 21 –31 species in contrast to the six found in the Highland Water streams.

Detailed analysis by Gurnell & Sweet (1998) on both reach and stream scales found that most debris dams were associated with pools in the Highland Water and most, though not all, pools were associated with debris dams. In most cases the CWD habitat unit could be considered as a continuation of the pool habitat on the channel-unit scale. Usually if CWD was found in a riffle, the wood was newly deposited and mostly single logs or large branches. Separating out the CWD habitat unit allowed the testing of the hypothesis that CWD itself was a direct cause of discontinuity in the abundance of the fish populations and that CWD formed a direct refuge for fish. Isolation of CWD accumulations from their adjacent habitat units has not been described in other similar studies on CWD and fish (e.g. Angermeier & Karr, 1984; Inoue & Nakano, 1998) and it is therefore difficult to make realistic comparisons with much of the published data. The delineation of sampling areas has provided much material for discussion but generally, the division of sampling areas into small units provides better replication for statistical analysis and gives better precision (e.g. Bohlin *et al.*, 1989).

One further limitation was poor replication in the analysis of the effects of the removal and retention of CWD dams. Each dam has different characteristics for example, the amounts of wood, orientation of pieces, channel size and relationship to other habitat units. In this study only two CWD dams were included and the data must be regarded as preliminary. However both reaches had similar overall habitat units, the work was over a similar period and there were contrasts in that one dam was breached and one retained. The data are comparable but clearly statistical analysis was not possible because of lack of replication. Further work on dam removal and retention is in progress (Langford & Baker, *unpublished data*)

Restoration work proposed for the Highland Water (Holzer, *personal communication*) will be based on the manipulation of reaches and sub-reach habitat units mainly introductions of CWD and the re-instatement of sinuosity to the pre-drainage works conditions. Many of the old meanders are clearly visible in the catchment and may be reconnected. The intention is to increase habitat complexity and structural diversity and presumably to increase biological diversity as has been attempted by restoration works in other running waters (e.g. Cowx & Welcomme, 1998). Present management of CWD in New Forest streams is also based on the sub-reach scale in that some dams are removed or destroyed by human activity either to improve drainage or to improve upstream access for fish (Langford, 1996). Thus to provide data for application to management, sub-reach scale biological data scaled up to meet required reach and stream-scale criteria are necessary.

The distribution and abundance of fish on whole stream and catchment scales is, as has been described in Chapter 5 & 6, probably a result of factors outside the stream channel. For example the effects of historical deforestation and stream maintenance for drainage in the streams adjacent to the Highland Water streams have been the destruction of the original fish community and its replacement by a higher diversity, higher abundance community as a result of increased productivity. This clearly has significant implications for future riparian canopy management.

### 8. 2. 2. Fish sampling

#### *a) Fright and avoidance*

One of the limitations of quantitative work on fish in running waters is the efficiency of electric fishing and much has been written on the techniques and the subsequent methods of calculating population sizes. Most aspects of the sampling techniques have been subject to scrutiny and change over the past three decades. (Cowx, 1983; Bohlin *et al.*, 1989; Cowx, 1990). Even so, for small streams with a diverse physical structure electric fishing is the major universally usable method of fish capture (Cowx, 1990). One of the lesser acknowledged problems of any methods is the effect of fright-bias (Bovee, 1982; Thévenet & Statzner, 1999) and little has been done to calculate the extent or even the existence of this phenomenon in most habitats (Cowx, 1983, 1990; Bohlin *et al.*, 1989). There was a possibility that fright-bias led to spurious results in the New Forest streams in that fish may have been driven into or out of refugia either during the setting of the nets or the fishing operations. This possibility is also pertinent to all other such studies using the methodology throughout the world yet it is rarely noted in the literature (e.g. Bohlin *et al.*, 1989; Cowx, 1990)

Very few studies using electric fishing have attempted to calculate fright-bias effects or even allow for such effects and most studies clearly accept that for some species, particularly salmonids, fright bias is not significant (e.g. Heggenes, 1988a). Usually, these fish would be expected to move to the nearest cover and remain there until caught or take avoiding action until they have no escape route. Thévenet & Statzner (1999) in their studies in a French river concluded that fright bias was not a significant factor affecting catches near timber debris in the large river that they sampled whether they used 50m$^2$ enclosures or nonenclosed, point-abundance sampling. Their estimate was that 3.4% of fish were driven from the areas they were sampling but this was in a large (75m wide) river where boat-based electric fishing was used.

Where blocking nets (stop nets) are used across the entire channel width in a small stream as in the New Forest, the initial activity in the stream is during the setting of the nets and this might be expected to disturb fish near that point. Once set, the nets clearly impede (though they may not completely prevent) free movement to and from the enclosed sampling area. This criterion of non-immigration or emigration is one of the conditions for the use of the population estimation methods (Carle & Strub, 1978; Cowx, 1983; Bohlin *et al.*, 1989).

The increase in the abundance of fish in the CWD accumulations might therefore have been the result of a significant fright bias movement to the wood matrices. However, the data showed that although there was a greater overall biomass of fish in the CWD habitat units, there were generally no significant increases in density, except for large salmonids. The evidence is therefore that fright bias toward CWD was not a significant factor in the daytime distribution of fish on the habitat-unit scale.

The effects of short term movements and behaviour of the fish were not investigated for this study and this remains an area for further research. It is well known that many species can make extensive movements over short periods (e.g. Langford *et al.*, 1979; Welton *et al.*, 1991; Clough & Ladle, 1997; Lucas & Frear 1997). Sampling in daylight also limits full interpretation of habitat use in that many species become active at night and seek refuge from predators during daylight (Maitland & Campbell, 1992; Roussel & Bardonnet, 1997). Thus their daylight distribution may be more discontinuous than during the hours of darkness. Fish may also move between refugia but this work did not determine the extent of such movements or whether individual fish used the same refuge for long periods (e.g. Clough & Ladle, 1997).

Harvey *et al.* 1999 found that radio-tagged salmonids moved less when caught and replaced in pools with CWD accumulations or other cover structures than those caught and replaced in pools without cover. Some fish also clearly used CWD accumulations as shelter during a bankfull flood. Clough & Ladle (1997) showed a variation on this site fidelity for dace (*Leuciscus leuciscus* L.) in that radio-tagged fish used clearly defined night-time and daytime habitats, returning almost exactly to their previous locations within the stream channel.

Unlike chalk streams, in the author's experience sightings of salmonids in New Forest streams are rare outside the spawning season when fish may be seen cutting redds in shallow riffles. Larger salmonids, eels and loaches almost certainly would already be sheltering from mammalian or avian predators in the daytime and would not be affected significantly. Highly mobile, shoaling species such as the minnow were observed moving ahead of the operator during the electric fishing but little is known about their movements during net setting. Both bullheads and small salmonids (<7cm) were generally observed being stunned just ahead of the electrode in riffles and few moved very far before they were caught.

### *b) Differential catch efficiencies*

The initial prediction was that catch efficiencies within the CWD matrices would be much lower than in the less structurally complex habitats. The technique of inserting the electrode between timbers in the matrix and capturing the immobilised fish (see Chapter 3) would seem to be more difficult than fishing in the riffles or pools with little CWD. However, observations *in-situ* suggested that relatively few immobilised fish were not caught once seen and catch efficiencies were not significantly different overall in the three habitat types. The major source of variation was the very small numbers of fish in some sampled units. In many of the habitat units the catches of some species were very small and sometimes single salmonids were the only catch. These small catches meant that population estimates and catches were the same or very similar in many habitat units and that few or no fish were sometimes caught on the first run. All the average catch efficiencies except those of *L. planeri* were above the threshold of 25% which the population estimation model requires (Carle & Strub, 1978). The numbers used for comparisons of *L. planeri* were mainly catch data.

### 8. 2. 3. Data analysis and reporting

The representation of physical or structural diversity in stream channels has presented problems for many years (Elser, 1968; Binns & Eiserman, 1979; Milner *et al.*, 1985). To date there has been no standard methodology for measuring structural or physical diversity of stream habitats and the mixture of subjective and objective variables has led to a diversity of measures which have been reviewed by other authors (e.g. Milner *et al.*, 1985; Hegennes, 1988a; Maddock, 1999). Clearly a single habitat diversity or complexity index is a useful tool if it is a viable concept, though the variables used for the derivation and the methods of interpretation of such indices vary as much as the measurement methods on which they are based.

Diversity indices (Magurran 1988) have been used over a number of years to express physical diversity with the most popular being Simpson's D (e.g. Gorman & Karr, 1978) or Shannon-Wiener H' (e.g. Binns & Eiserman, 1979) but each variable can be used in several ways to derive the index. Thus individual measurements (e.g. depth or current velocity) from a habitat unit can be used separately in frequency categories to produce diversity indices or they can be used in conjunction with other variables such as cover or substrate to produce an overall habitat suitability or habitat quality index (e.g. Binns & Eiserman, 1979). The problem here is that all the variables have different measurement units and relate to different parts of the habitat. The advantage is that if the method is standardised the results can be comparable for a variety of habitat units at least within one stream continuum. The comparisons of structural or habitat diversity in themselves are viable for comparing stream channels. The problems arise when these indices are used to predict fish diversity and abundance.

Methods which use a mix of data sources and a mixture of qualitative, semi-quantitative and quantitative for producing models have been used for many years (e.g. Binns & Eiserman, 1979), as have instream flow models (Bovee, 1982; Milhous, 1999). One unusual alternative was to assess fish abundance from physical data based on a questionnaire for which observers would be trained to assess the criteria in a standardised method (Milner *et al.*, 1985). The conclusion from trials of this method was that the method would perform as well as other published methods for salmonids, although it was clear that it had only limited general use. For fish not targeted by anglers and for conservation data the method has very little use.

Given the diversity of methods and the diversity of variables, together with the questioning of the fundamental concept that fish abundance may not be directly related to structural diversity (Elliott, 1994) it is only reasonable to assume that the search for a universal model is fruitless. This may even apply to a single taxon such as Salmonidae for which much research has been done to date. The problem is that if fish diversity and abundance are not related to structural diversity (see Elliott, 1994) the use of habitat diversity or quality index would be based on the wrong fundamental premise. As shown, while there may be some relationships on the within-reach scale, factors affecting the distribution and abundance of fish on the stream and catchment scale in the New Forest are more related to the nature of the broader catchment than the stream channel itself.

Despite the lack of a universally applicable model, locally acceptable physical habitat models, even if only applicable to one stream or a few streams in a catchment, may have considerable value, particularly if they are based on standardised measurements and use multivariate methods of analysis (Milner *et al.*, 1985; Elliott, 1994). The categories used must, of course, be standardised and used consistently. The co-linearity of many physical variables (see Binns & Eiserman, 1979; Milner *et al.*, 1985) has encouraged the use of ordination methods (see Kent & Coker, 1992) which may be more relevant to the expression of relationships between physical, geomorphological and biological data. Detrended correspondence analysis (DCA) has produced a clear representation of the trends in fish abundance and community structure on both within-reach and reach scales in these studies. The method is robust in that it can use measurements of variables based on different scales and units to produce satisfactory two-dimensional ordinations.

## 8. 3. FISH ABUNDANCE AND COMMUNITY STRUCTURE IN RELATION TO COARSE WOODY DEBRIS AND STREAM MANAGEMENT

### 8. 3. 1. Relevance of CWD to fish community structure

Cummins (1992) concluded that the predictable shift from the allochthonous material driven ecosystem to an autochthonous system could probably provide a *"major underlying organising principle in the conservation of running waters"* in the face of spatial and temporal changes in land use and climate change. Such alterations of stream ecosystems from allochthonous to autochthonous as tree cover is removed have been observed in forest streams particularly in the USA where clearfelling and logging have removed the riparian vegetation (e.g. Murphy *et al.*, 1986; Bilby & Bisson, 1992; Davies & Nelson, 1994). In the streams studied in the New Forest the clear effects of a shift from the pristine, wooded allochthonous to disturbed, open, autochthonous systems are reflected in the changes in the fish community. Species richness and fish abundance increases as woodland decreases and the immigrant fish species are more phytophilous. There is clearly a penetration upstream of species more characteristic of the downstream reaches. Productivity is also clearly related to the degree of riparian canopy and woodland (e.g. Bilby & Bisson, 1992). The main factor ultimately determining fish species richness and species composition of any New Forest stream is therefore the degree of cover by riparian trees, so much so that even adjacent and confluent streams with different amounts of tree cover can differ in species richness, abundance and community composition. The contributions from allochthonous materials decline markedly after riparian trees are removed. For example, Bilby & Bisson (1992) found that the annual input to a stream from an established forest was five times that from an area where riparian trees had been removed. In contrast autochthonous contributions were 75% greater from the clear-felled stream than from the shaded stream. However, the fish production was greater in the open stream where the fish fed on autochthonous material in spring and summer.

The amount of CWD in the streams is also a function of the amount of riparian woodland in the catchment (Gregory, 1992; Gregory *et al.*, 1993). Thus, as far as the fish are concerned, the presence of CWD is a secondary habitat factor on the stream scale but has considerable influence on the abundance of fish at sub-reach (habitat unit) and reach scales. The role of CWD in structuring the fish community is not fully clear but the evidence is that it has a direct role in increasing the refugia available for larger salmonids and perhaps eels. It also has a major indirect role in the distribution of substrates and channel features which in turn form the physical basis for the fish community structure.

CWD is therefore important to the deterministic processes structuring the fish community in these low-disturbance forested streams, but no species is dependent upon CWD for its survival. The influence of CWD on communities at the within-reach and reach scale in the more highly disturbed streams in the UK is largely unknown despite its being used for potential fishery enhancement projects (e.g. Cowx & Welcomme, 1998). Clearly the relatively undisturbed, lowland streams of the New Forest have a similar restricted species-richness and diversity to upland low-order streams (Hawkes, 1975; Moyle & Vondracek, 1985; Turnpenny *et al.*, 1987; Crisp, 1993). It is well known that small streams in many regions typically contain fish communities with low species richness and diversity (Hawkes, 1975; Moyle & Vondracek, 1985) though how far this is a result of size or other factors in unclear. For example in the River Tarrant and Devils Brook both small, disturbed, non-forested lowland chalk streams in the southern UK within 40km of the New Forest, 8 species were listed as common (Mann & Orr, 1969; Mann, 1971) six of which were also common to the Highland Water. These were *S. trutta, P. phoxinus, C. gobio, N. barbatulus, L. planeri* and *A. anguilla*. Salmon, (*S. salar)*, pike, (*Esox lucius*) and sticklebacks (*G. aculeatus*) were also recorded but no mention of larger cyprinids were made in these streams. The additional species were also common to the disturbed streams of the New Forest, notably the Ober water and Dockens Water. The original fish fauna of these small chalk streams is unknown. Further comparisons of the fish communities of small lowland streams are necessary to determine the mechanisms which structure the communities.

Channelization of the Highland Water streams does not appear to have caused the loss of any species of the pristine community but, based on the abundance data, the reduction of CWD and pools (Gurnell & Sweet, 1998) in the channelized reaches would, predictably, have altered the relative abundance of species on the reach scale. There was some evidence of different water chemistry between the open and wooded streams of the Forest proper but this was relatively small. There was very little difference in overall channel dimensions though depth and width variations in the wooded streams were on average greater mainly because of CWD accumulations and structures associated with riparian trees. There were no data on temperatures from the unshaded areas, though studies in other regions (e.g. see Langford, 1983, 1990; Ormerod *et al.*, 1987; Crisp, 1997) show a pattern of higher summer temperatures in unshaded streams than in shaded streams. Davies & Nelson, (1994) also recorded significant correlations ($p<0.05$) between open stream length, riffle length and temperature.

### 8. 3. 2. Relevance of CWD to fish abundance

All six species in the Highland Water showed discontinuous patterns of distribution and abundance on the within-reach (habitat unit) scale, much of which can be directly or indirectly related to CWD accumulation. Thus the manipulation of CWD as part of stream management practices can be predicted to have measurable effects on the species populations. At the same time species abundance was variously correlated with the gradient of combined physical variables from the erosional to the depositional of which CWD was only one component. The relative importance of CWD depends very much on the species and is summarised below.

*A. anguilla* and older salmonids showed both higher numerical densities and higher biomass in the denser CWD accumulations in the Highland Water though they were also found in pools with no CWD. In contrast the densities and biomass of *C. gobio* are clearly negatively correlated with Axis I of the DCA, in turn heavily influenced by CWD abundance. Thus densities of *C. gobio* were highly positively correlated with shallow gravel riffles and for this species, increased pool formation which could occur with increased CWD (Gurnell &

Sweet, 1998) would result in decreases in abundance and a reduced distribution. CWD as a pool-forming influence would clearly need to be managed to maintain sufficient gravel riffles for this species to sustain significant populations.

Older salmonids were distributed in deeper habitat units of the Highland Water streams, with the maximum size significantly related both to water depth and amounts of CWD in an accumulation. The largest fish were clearly closely associated with the deepest water and CWD. It is likely that these larger fish, mostly upstream migrants or large smolts, use the CWD habitat units as direct refugia, at least during the day. Alternatives are other cover structures such as root wads, overhanging vegetation or undercut banks. Minnows (*P. phoxinus*) are essentially more limnophilic species and were rarely found in riffles (Roussel & Bardonnet, 1997). The relationship to CWD is weak except in that CWD accumulation may increase pool area (Gurnell & Sweet, 1998), the preferred habitat. The brook lamprey, *L. planeri*, was most abundant in pool habitats with suitable sand-silt deposits and was scarcer in the dense CWD accumulations and riffles. This species spawns in gravel riffles (Maitland & Campbell, 1992) but no sampling was done in the spawning period and the species was not observed in its spawning phase on the gravels. The preferred habitats are pools with large sandy silt deposits sometimes with amounts of finer woody debris such as twigs or partly decayed leaves.

### 8. 3. 3. Effects of CWD management in New Forest streams: A provisional model

It is clear from these studies that the variable abundance of CWD in the streams studied affects the distribution, abundance and diversity of the fish populations on both the within-reach (habitat unit) and the reach scale. Effects of CWD on the stream and catchment scales using adjacent streams for comparisons are obscured by other major factors, mainly deforestation and channel maintenance. From the fish data and from geomorphological data collected over many years by other authors (see Gurnell *et al*, 1985; Gurnell & Sweet, 1998 for references) I propose a simple provisional model of the effects of changing CWD abundance and potential changes in channel structure on the fish populations is proposed

**Table 8.1** **Median and quartile densities of fish from riffles, pools and CWD habitats in Highland Water streams used to estimate effects of CWD management on fish populations.**

| Species | RIFFLES | | | POOLS | | | CWD | | |
|---|---|---|---|---|---|---|---|---|---|
| | *Median* | *25%* | *75%* | *Median* | *25%* | *75%* | *Median* | *25%* | *75%* |
| *S.trutta** | 0.24 | 0.12 | 0.64 | 0.11 | 0.05 | 0.34 | 0.33 | 0.11 | 0.5 |
| *C.gobio* | 0.9 | 0.43 | 1.73 | 0.2 | 0.06 | 0.26 | 0.05 | 0 | 0.15 |
| *P.phoxinus* | 0.01 | 0 | 0.05 | 0.33 | 0.09 | 0.89 | 0.45 | 0.17 | 0.92 |
| *Lampetra sp.* | *0.001* | 0 | 0.08 | 0.3 | 0.1 | 0.64 | 0.08 | 0 | 0.22 |
| *N. barbatulus* | *0.001* | 0 | 0.002 | *0.001* | 0 | 0.002 | 0.01 | 0 | 0.01 |
| *A. anguilla* | *0.001* | 0 | 0.001 | *0.001* | 0 | 0.03 | 0.05 | 0 | 0.09 |

** <7cm = 79% of riffle totals, 39% of pool totals, 25% of CWD totals, used for these preliminary calculations* using a hypothetical 100m reach of the Highland Water. A median width of 3m is assumed which represents the Highland Water in its middle reaches. Fish abundance data for each habitat type are the overall median and quartile densities calculated for each species (Table 8.1) using the data from all the samples in that habitat type over the whole sampling period from September 1996 to February 1998. Medians were used instead of means as they were the values used for statistical comparisons (see Chapter 5).

Gregory *et al.* (1993) concluded that only 7% of the natural CWD loading remained in New Forest streams as a result of management in the 1980s. Given that there were variations between streams and that the Highland Water system showed higher than average loadings it can be predicted that the natural CWD loading could be at least 10 times that of the late 1980s. In more recent years, however, less CWD removal has occurred and there has been a movement toward the more natural state, though dams are still destroyed or removed for management purposes or by floods.

There is a clear relationship between the structure of the channel and CWD accumulation (Gurnell *et al.*, 1995) and it follows from the fish abundance data that this relationship will affect fish populations. One of the problems with devising my model was, however, that the geomorphological effects of CWD as described are not always fully clear. For example, in geomorphological studies various terms are used which are not always clearly defined. For example "pool-spacing" might be interpreted as "riffle lengths" though the inter-pool habitats are not generally described in terms of dimension or substrate (e.g. Gurnell *et al.*, 1995). Conversely inter-riffle distance (IRD) might be interpreted as "pool-lengths" though it may include glides or runs. For the fish studies the faster waters included riffles and runs and the slower waters pools and glides in addition to CWD habitats. However, as the ordination showed, there were not clear definitions between the three *a priori* habitat unit types in practice. Although Gurnell & Sweet (1998) defined pools associated with CWD and those not so associated in terms of distance from an accumulation, it is difficult to compare these with the actual accumulations isolated for fish sampling. In devising the following model, therefore, there are a number of assumptions related to the geomorphological effects of CWD which may be refined in later versions.

First, IRD is regarded as "*pool length*", irrespective of any variations in interpretation and pool-spacing is regarded as "*riffle-length*". The mean length of the isolated CWD habitat units used for fish sampling was approximately 5m (see Chapter 4) and this is the length used for estimating the lengths of CWD accumulations in the model. Because IRD and pool-spacing tend to be expressed as numbers of stream widths, the same method is used for the fish model. The unknown factor is the effect of CWD accumulation on riffle formation. At lower CWD densities, sediment movement during spates can create plunge bars which may become riffles. The effect of high densities, with very little of the channel not containing CWD is difficult to predict but at the highest densities space for riffles may become the limiting factor. Also, it is assumed for the model that all CWD is in the form of full-width or partial dams with no allowance for individual pieces outside such formations.

Gurnell & Sweet (1998) showed that pool spacing in the Highland Water was reduced from about 10 to 4 stream widths as a result of an increase in the number of CWD dams per 100m from 0 to 12. They also suggested that this would decrease to 2 stream widths if the CWD was left completely unmanaged and this implies that the natural density of debris dams would be $16.100m^{-1}$. Fig.8.1 shows the changes in the area of riffle (gravel), pool (with no CWD) and CWD habitat in a hypothetical 100m reach of the Highland Water in relation to the increase in CWD accumulation. The overall effect is to reduce riffle area from about $90m^2$ to $18m^2$, pool area (without CWD) from 200 to $30m^2$ with a corresponding increase in CWD habitat from 0 to $240m^2$. The effect on the total numbers of fish in the 100m reach is negligible in that the predicted numbers of fish altered little between the extremes (Fig. 8.2). The median values show an overall change from 281 to 280 fish though the lowest estimate was 270 fish as CWD dams began to increase. There were corresponding levels of change using the upper and lower quartile densities.

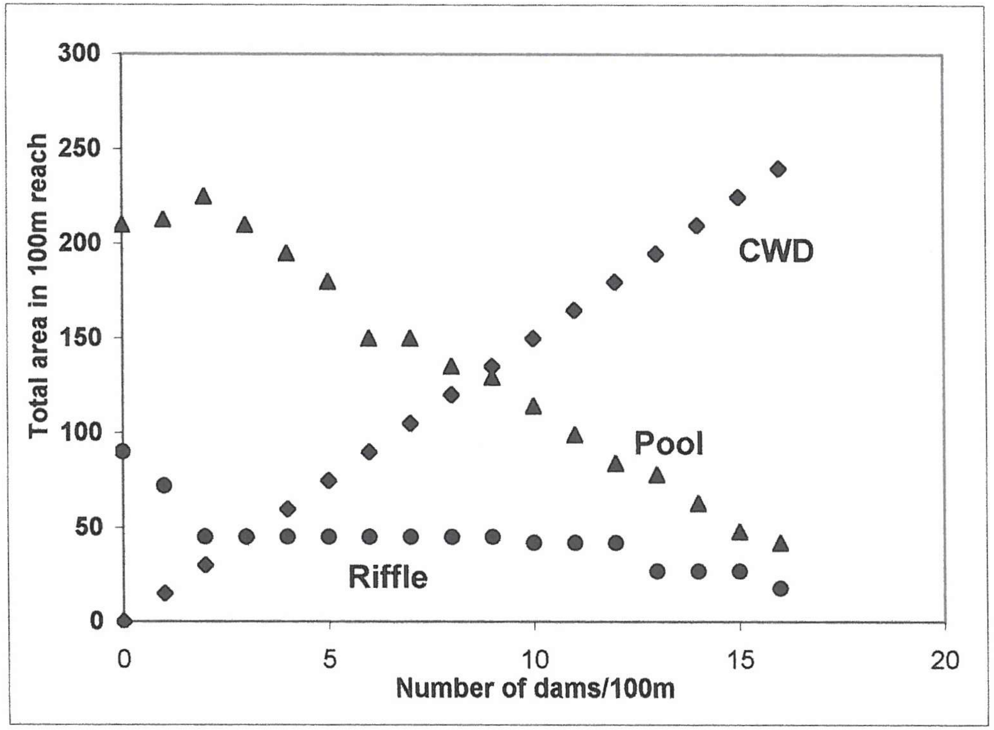


Fig. 8.1 Changes in the areas of riffles,pools and CWD habitat in a hypothetical 100m reach of a forested stream (Highland Water) in relation to the increase in the number of debris dams. (stream width=3m)

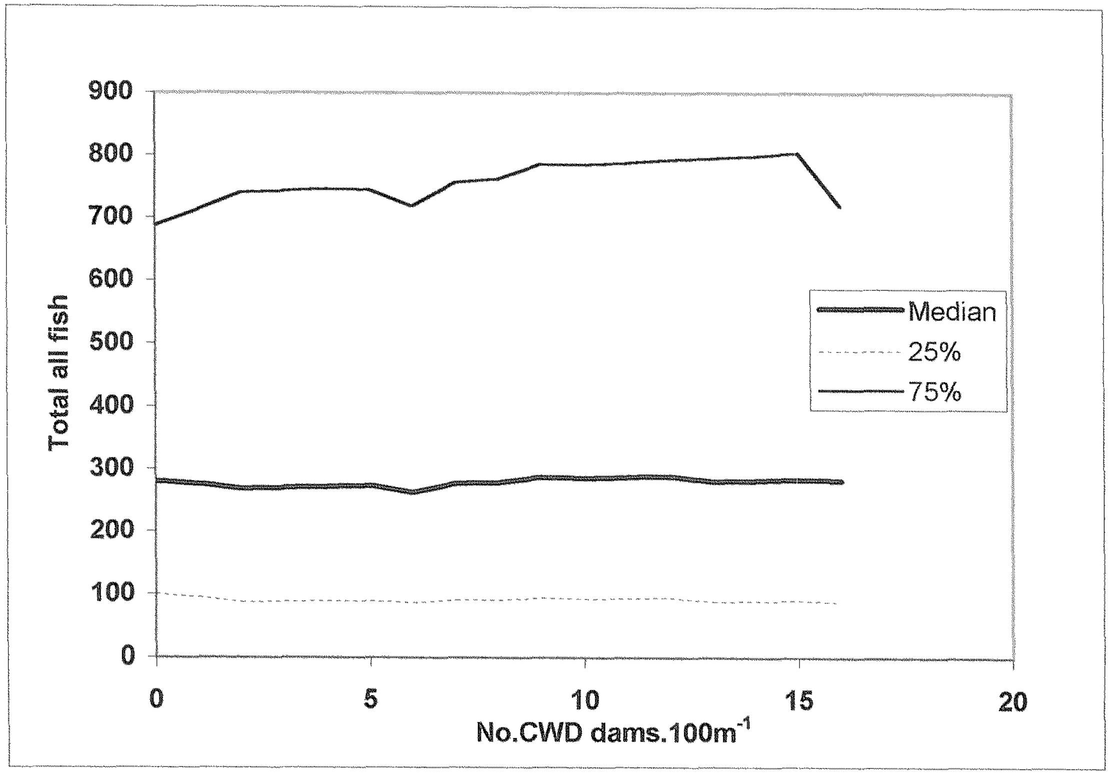


**Fig. 8.2 Estimated median and quartile numbers of all fish in a hypothetical 100m reach of a New Forest stream (wooded) in relation to the number of CWD dams.**

*(see text for explanation of source data)*

In contrast, the predicted numbers of each individual species in the hypothetical reach changed markedly. For example the median numbers of small salmonids (< 7 cm TL) decreased initially but stabilised at 22 (Fig.8.3). The median numbers of fish over 7cm were predicted to increase as the number of dams increased. The median estimate rose from 22 to 55 fish $100m^{-1}$. However, the population of larger fish must eventually be dependent on the numbers of small fish and must therefore reach a lower equilibrium which is not predicted by the model. Some refinement of the model to allow for this can be made from future studies using natural mortality rates, density dependent factors (e.g. Elliott, 1994) and seasonal fluctuations.

The median predicted number of *C. gobio* declined from approximately 120 to 40 in the 100m reach as CWD abundance increased (Fig. 8.4). This is clearly significant for the conservation of this Annex II species and for the management of the CWD in the stream. If the lower quartile densities are considered, the numbers per 100m reduce to levels of around 10-20. Predicted median *P. phoxinus* populations increased from approximately 80 to equilibrate at about 120 per 100metre reach (Fig. 8.4). Higher populations could exceed 250 per 100m reach. The second Annex II species (*L. planeri*) represented as *Lampetra* on Fig. 8.4 showed an initial predicted increase in the median population but a decline as CWD increased. The overall trend is a reduction in populations whatever the starting point. The overall change in median numbers from about 30 individuals to 24 may not be significant, particularly as population estimates were difficult and the species is very patchily distributed. Predictions for *N. barbatulus* and *A. anguilla* both showed a small increase in numbers as CWD increased from 0 to 16 per 100m reach. The median numbers for *N. barbatulus* in the model rose from 3 to 5 (quartiles 0.01 - 6) and those for *A. anguilla* from 3 to 3.2 (quartiles 8-25). Because the species were scarce and infrequent in samples the predictions may be subject to larger error than for the more common and abundant species.

From preliminary estimates of biomass changes, the model indicates a potential rise from about 0.6 kg of all fish to over 2.6kg per 100m reach (Fig.8.5). This threefold increase would be accounted for by salmonids (mainly larger) and eels

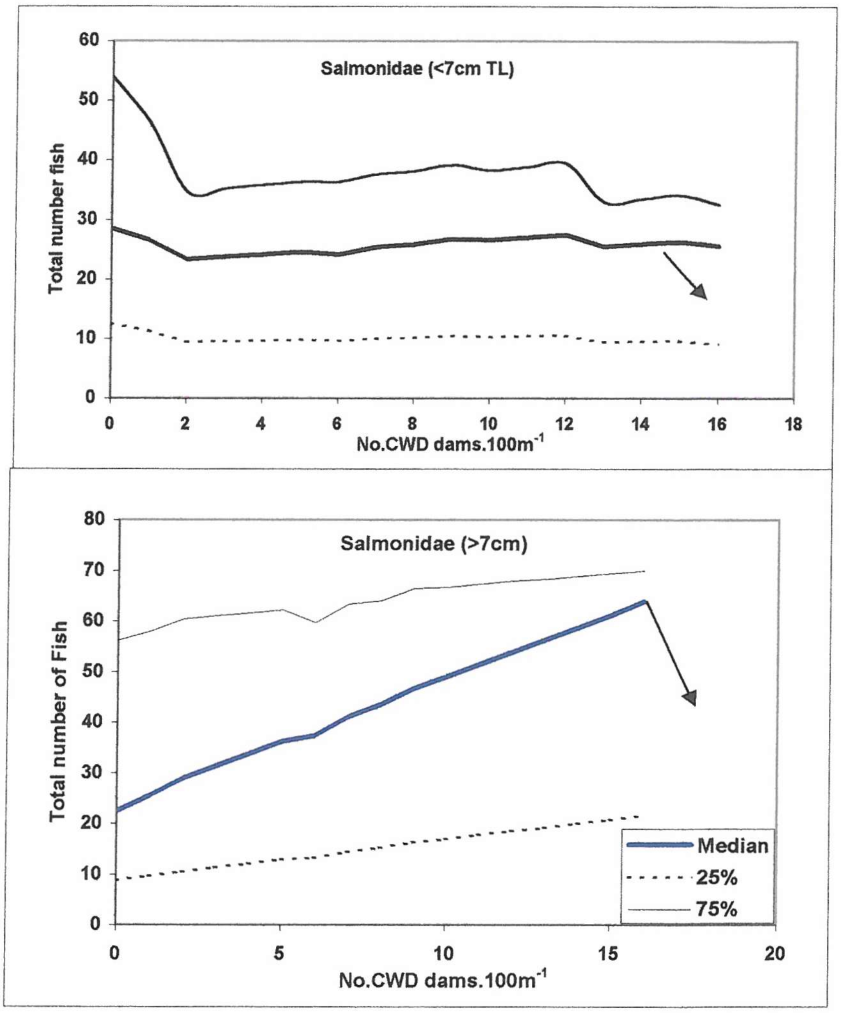


**Fig.8.3** Estimated median and quartile numbers of Salmonidae (<7cm and >7cm) in a hypothetical 100m reach of a New Forest (wooded)stream in relation to the number of CWD dams.
(see text for explanation of source data)

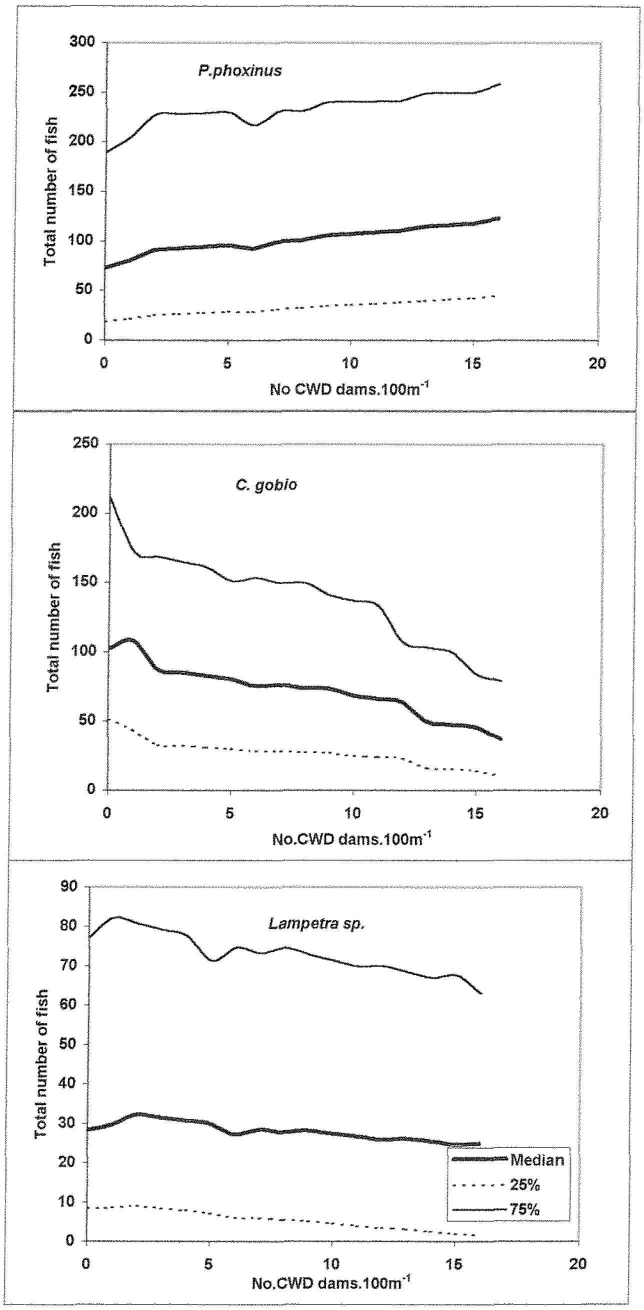


Fig.8.4 Estimated median and quartile numbers of three species of fish in a hypothetical 100m reach of a New Forest stream (wooded) in relation to the number of CWD dams.

(see text for explanation of source data)

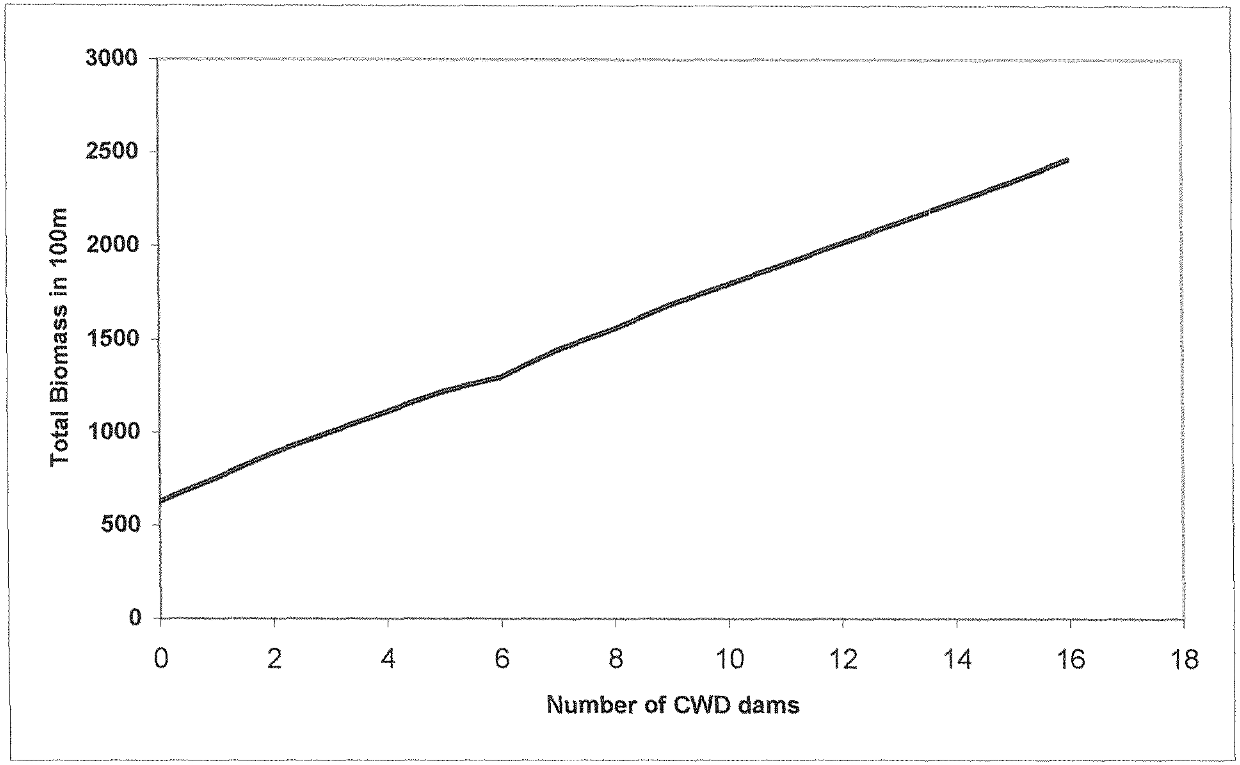


Fig.8.5 Median estimated biomass of all fish in a hypothetical 100m reach of a New Forest stream (wooded) in relation to the number of CWD dams present.
(Biomass data are median values of all fish summed for all samples, 1996-98)

which would comprise approximately 57% and 30% of the total biomass respectively. Estimates suggest that *C. gobio* and *N. barbatulus* would suffer the largest decrease in biomass terms both by an estimated 60%. *P. phoxinus* would increase from about 80g to 230g per 100 metre reach and the biomass of *Lampetra* sp. would remain relatively unchanged. Clearly, therefore the model suggests a highly significant increase in fish biomass on the reach scale with a large scale increase in CWD dams. However, analysis of reach-scale data in Chapter 5 indicates no significant correlation (Table 5.11) between the biomass of any species and the total volume of CWD in a reach. The differences may be a result of the geomorphological assumptions and the assumed form of the CWD accumulations.

Thus the model is as yet provisional and further work is in progress to refine and improve the data. There are clear indications of possible reductions in populations of salmonids, *C. gobio* and *L. planeri* if CWD increases significantly provided that the assumptions of geomorphological change are valid. At the same time total fish biomass increases, mainly of eels and salmonids, could be significant. However, the protection of the populations of species protected by the European Habitats Directive (*C. gobio, L. planeri*) (Boon, 1995) is dependent on maintaining the population numbers and all stages of the life-history and thus biomass is relatively unimportant. This simple model can be improved significantly with more detailed geomorphological data and using seasonal and natural mortality data and taking factors such as density dependence into account. The key physical factor is the effect of more CWD on the relative areas of riffle and pool habitat.

### 8. 3. 4. Management of New Forest streams for fish

Habitat management is interference. In most industrialised or agricultural regions it generally constitutes the replacement of one, sometimes long-established, disturbed system by another disturbed system albeit with the premise that the new system may be more "natural" or "beneficial". Mellquist (1992) notes that management, as a principle, has "*never been and never will be particularly precise*", and Cummins (1992) adds that "*the concept of management is one of* *the more arrogant human notions*". Boon (1992) indeed noted that theoretical topics are too little applied to river management and that conservation management can be ineffective simply because the requirements of the community are not known. Despite these observations the urge to manage habitats to produce communities or ecosystems, which are essentially human creations, persists.

In rivers, management is usually aimed at conservation or for some feature related to human benefit, for example angling, aesthetics, drainage or navigation. Management of any fish habitat may be aimed at the protection or enhancement of target species or communities (e.g. Maitland & Campbell, 1992; Cowx & Welcomme, 1998) or, more frequently, at increased convenience and access for fishing. Conservation management is more likely based on the human perception of what is natural or to attain a particular state identified from historical information. Often this bears no relation to the "pristine" state. For example, restoration of chalk streams is often aimed at re-creating conditions more like the age of mills or navigation (i.e. channelization), than the most likely pristine, pre-historic state of forested, anastomosed meandering channels in wet floodplains. Elliott (1995), questioned the need for any management of rivers and expressed some sympathy with the view that all is needed is to "*ensure a high water quality and the fish will look after themselves*". Unfortunately the drastic physical changes in catchments and in river channels over thousands of years, together with demands for land drainage and angling facilities, dictates that management is a self-perpetuating process.

Most of the management of forested streams in the past has been to maintain or improve land drainage and lower the level of the water table to attain the optimum growth and survival of the tree crop. This is, however, changing and in the future, some proposals for the New Forest aim to manage the streams for both in-stream and floodplain ecology (Holzer, *personal communication*). For increases in forest floor vegetation some trees, particularly conifers have been removed to enhance light penetration. At the same time riparian coniferous trees have also been removed along some reaches particularly along the Bratley Water and Highland Water. The aim is to replace these with deciduous trees such as beech and oak.

With regard to the fish populations the management of the riparian forest is significantly more important than managing the stream channel. Therefore, retaining the riparian forest and full shade is the key to the preservation of the low-disturbance, probably pristine fish community of the Highland Water streams. Such communities are rare in lowland Britain. However, Ratcliffe (1977) suggests that the fish fauna of the Ober Water may be a more suitable case for conservation despite its high level of disturbance but as Chapter 6 shows, high-disturbance communities maybe more common than low-disturbance communities in the New Forest and probably in the UK generally. Thus, although there is a general tendency to aim conservation programmes at retaining species richness or diversity, in the case of the New Forest and other lowland forested streams the rarity of the pristine community may be more important.

As far as physical changes to the stream channel are concerned it is important to note the predictions of the simple model that the overall effect of increasing the areas of deeper water by introducing CWD or increasing impoundment may be to reduce populations of *C. gobio* and to a lesser extent *L. planeri.* The balancing effect may be, however, that increased sinuosity and better water retention may create areas of gravel and increased retention may improve the flow of water over them in summer. In recent years periods of summer drought have dried out areas of gravel in much of the upper Highland Water and its associated streams. The relationship between community composition and the physical gradient from erosional to depositional habitats provides a suitable general model on which to base the management of the streams. Essentially the more the habitats develop toward the deeper depositional structure the more the community of these habitats will encroach.

For the salmonid populations all three habitat units, riffles, pools and CWD accumulations, have some relevance for the various stages of the life-history. Riffles are the main habitat for the 0+group, while pools and CWD habitats are preferred by the older groups. Riffles in New Forest streams are generally so shallow that larger predators such as larger feeding salmonids and eels are excluded either because of physical access or lack of refugia. The CWD habitat itself, however, appears to be an extension of the pool habitat but with the deepest water and this is clearly relevant for the largest fish, typically the upstream migrating salmonids. This relationship between largest fish and CWD in deep water seems a common pattern in streams (e.g. Hortle & Lake, 1983; Harvey & Stewart, 1991). The critical factor for salmonids is the diversity of the habitat on the reach scale, including a balance between optimal areas of clean gravel for spawning and for 0+fish habitat, optimal pool or deeper glide areas for the older fish and the optimal areas of deepest water for the largest smolts and upstream migrants.

Ultimately, the decision about the removal or retention of CWD and its management in the streams of the New Forest depends on the management targets. If the main aim is to preserve species diversity in the forested streams, accumulations of CWD are probably only important in that the pool-riffle ratio is increased which in turn will increase the diversity of substrates and structures. If the target is to protect the protected Annex II (European Communities, 1992) species the frequency of CWD dams must be optimised to retain a balance of shallow riffles for *C. gobio* and pools with areas of suitable silty-sand substrate for *L. planeri.* Both species, however, spawn in the gravels as do the salmonids.

For the salmonids the total areas of shallow gravel for spawning and 0+ habitat must be sufficient to provide a large enough number of survivors to pass to the subsequent age/size group in the deeper waters. At the same time there must be deep pools, preferably with CWD matrices for the largest upstream migrants. However, if the numbers of recruits to the 0+ age group are reduced the salmonid population will decline to a stable but lower equilibrium level, including the number of inward migrants. *N. barbatulus* is sporadic and clearly occupies micro-habitats in the deeper water and CWD whilst *A. anguilla* obviously finds the CWD habitat a usable direct refuge. From the point of view of total standing biomass a large increase in CWD would seem to be of great benefit to eels and larger salmonids but again not to the designated Annex II species.

The immediate direct effects of removing CWD dams on the fish at the reach scale appears to be negligible though the data are as yet preliminary. There was evidence of redistribution at the within-reach scale following a spate but there were distinct similarities between a reach where a dam was retained and a reach where a dam was partially removed.

There is a perception that the typical debris dam may act as a barrier to migrating salmonids (Lappin, 1991; Langford, 1996). This has not been addressed directly in this programme but there is ample indirect evidence from the data obtained that salmonids have spawned in the upper reaches of the Highland Water both from observations of spawning females in the small headwaters and from the densities of 0+ fish in the upper reaches of both the Highland Water and Bratley Water at sites 21 and 16 respectively (Appendix II). This debris dams may not act as major barriers to upstream migrants.

The absence of CWD and the absence of canopy from the adjacent streams does not reduce either species richness, diversity or standing stock, in fact the reverse. The shift from an allochthonous fed to an autochthonous fed system clearly results in increases in all three as has been observed in other streams (Murphy *et al.*, 1986). However, the result is a replacement of a low-diversity pristine community by a high-diversity, high abundance disturbance community. This is also a potential result of tree clearing in the Highland Water if temperatures rise as has been recorded in many other streams and instream vegetation increases as in the Ober Water and Dockens Water. The specific consequences of the absence of CWD from the unshaded streams are unclear and probably masked by the other changes.

Large, unmanaged amounts of CWD are not in themselves essential to maintain the fish community in either open or wooded streams though they are part of the contribution from the riparian trees and are significant components of the channel modification and structuring. On the within-reach scale CWD accumulation causes directly significant variations in abundance and species composition.

### 8. 3. 5. CWD and fish management in other streams

The implications of the structural diversity of the stream channel for the maintenance of fish populations has been discussed by many authors (e.g. see Boon, Calow & Petts, 1992; Cowx & Welcomme, 1998). In most reviews the importance of instream timber debris for enhancement of fish habitat has been stressed (e.g. Gurnell *et al.*, 1995). This research in the New Forest has shown clearly, however, that CWD accumulation is not the primary factor influencing the structure of the fish community of forest streams but is very much a secondary factor which can influence both the overall abundance of fish and of some species on the channel unit and reach scale. This is very much in agreement with studies in other regions (e.g. Bryant, 1985 in North America; Inoue & Nakano, 1998 in Japan). The secondary effects on pool formation can, however, have detrimental effects on riffle species or life-history stages which depend on riffles. Thus it cannot be concluded that the presence of unmanaged CWD accumulations, though in themselves probably the original state in rivers, is of advantage to all species or to the overall species diversity. Indeed, the more disturbed and managed streams contain more species and a greater overall abundance of fish than the relatively undisturbed streams.

The important factor for management is the detailed knowledge of the habitat requirements of each species in the community and the factors which structure the community. Elliott (1995) noted that " *such knowledge is often lacking, especially at the quantitative level*". As with other habitat management, there is no universally applicable strategy. Each stream system must be managed on an individual basis at whatever level is decided. The ecological information should come from a detailed study at the scale of the channel unit or smaller. One of the limitations to the acquisition of ecological knowledge for stream management is the resource requirement, namely time and money. However, the possibility of more efficient management with better knowledge of the working of the ecosystems which may also provide the benefits of targeting resources, must be acknowledged.

## 8. 4. FUTURE RESEARCH

There is still a need to understand better the fundamental mechanisms and processes which determine the structure of stream ecosystems particularly in regions of the world where there are some low-disturbance communities remaining. In Britain such areas are rare and the New Forest is a unique example of an area with a stable, long-established socio-economic infra-structure which has low disturbance in some catchments over the past 1000 years or more (Tubbs, 1968, 1986; Langford, 1996). In addition the presence of high-disturbance catchments in the immediate vicinity and contrasting catchments with a short distance, gives scope for detailed comparisons and predictive studies which can form the fundamental basis for wider predictions of the effects of channel structure and land uses on fish communities. Whilst there are funds from various sources for the physical alteration of river habitats, the resources for pre-alteration and post-alteration research and monitoring are generally less available. To provide the framework data, there is, therefore, a need to co-ordinate a programme of research at least on the national scale using a selection of different rivers which represent the range of ecological types and degrees of disturbance. In each river, exactly the same programme would be followed, using within-reach and reach scale methodology to provide the fundamental data.

On the local scale, proposals for massive new developments on the fringes and within easy range of the Forest, demands for more space for recreation plus the urge to manage and alter habitats before the full ecological implications are understood, threaten the ecosystems of the low-disturbance streams, particularly those which are perceived as "damaged". Because of the long-term stability of the catchment uses, the potential pressures and the proposed management strategy for the catchments, the New Forest provides a particularly significant location for research. This will provide both the ecological basis for future management of the streams and the applied data for assessing the effects of management activities. This will include a major opportunity for measuring the geomorphological and ecological effects of stream restoration and rehabilitation simultaneously on low-disturbance communities and will have application on both national and international scales.

The limited species richness and the differences between adjacent streams also offers the opportunity for fundamental ecological studies *in-situ* of competition between species for space and resources, particularly species such as chub (*L. cephalus*) or other cyprinids and salmonids (*S. trutta*) or bullheads (*C. gobio*), young salmonids and stone loach (*N. barbatulus*). In particular there are the opportunities to determine the potential flexibility of habitat selection, resource partitioning and community organisation among fish in adjacent and confluent streams with differing levels of catchment disturbance and channel alteration.

As far as the specific effects of CWD are concerned, more work on the within-reach and reach effects of removal and retention of dams is required to determine the spatial and temporal scales of the changes caused by CWD management. The effects of CWD accumulations on habitat use and both small-scale fish movements and larger-scale migrations using marked or radio-tagged fish is also needed. This would determine the precise role of the CWD as a refuge for large salmonids and as a potential block to migrants. The proposed, simple model, of the potential effects of increased, unmanaged CWD in the streams requires validation with further data, particularly from studies on the reach scale using manipulation of the CWD loadings.

Given the clear effects of CWD management and disturbance in the New Forest streams a programme of large scale research is required to compare the role of CWD and channel structure on the ecology of streams of different regions, altitudes and sizes. On the national scale, the effects of restoration and management in different areas are not well known, except for a few locations, and the fundamental research needed to establish both the ecological and economic consequences of such operations is long overdue.

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**Appendix 1.** **List of all sampled reaches showing the location, total reach length and total catch of each species in the summed habitat unit samples.**
***ID Code shows the site number followed by the repeat visit number, e.g. 1.2 = 2nd sampling visit to site 1.***

| River | Date | ID Code | OS Ref. | Length | *S.trutta* | *P.phoxinus* | *C.gobio* | *Lampetra* | *N.barbatulus* | *A.anguilla* | *L.cephalus* | *G.aculeatus* | *L.rutilus* | *Total* |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Highland** | Sep-96 | 1.1 | SU 26950775 | 33.3 | 31 | 78 | 13 | 25 | 3 | 0 | **0** | **0** | **0** | ***150*** |
| **Highland** | Sep-96 | 2.1 | SU26920768 | 29.6 | 5 | 47 | 3 | 2 | 0 | 1 | **0** | **0** | **0** | ***58*** |
| **Highland** | Sep-96 | 3.1 | SU26810785 | 32.1 | 23 | 133 | 6 | 24 | 0 | 1 | **0** | **0** | **0** | ***187*** |
| **Highland** | Sep-96 | 4.1 | SU26750790 | 35.1 | 10 | 91 | 14 | 64 | 0 | 1 | **0** | **0** | **0** | ***180*** |
| **Highland** | Sep-96 | 5.1 | SU26700850 | 32.7 | 14 | 60 | 6 | 4 | 0 | 3 | **0** | **0** | **0** | ***87*** |
| **Highland** | Oct-96 | 1.2 | SU 26950775 | 30.7 | 7 | 43 | 14 | 12 | 3 | 2 | 0 | 0 | 0 | ***81*** |
| **Highland** | Oct-96 | 6.1 | SU26520811 | 49.1 | 16 | 37 | 10 | 15 | 1 | 2 | 0 | 0 | 0 | ***81*** |
| **Highland** | Oct-96 | 7.1 | SU25500850 | 43.2 | 19 | 8 | 37 | 5 | 1 | 3 | 0 | 0 | 0 | ***73*** |
| **Highland** | Oct-96 | 8.1 | SU25450865 | 51.3 | 19 | 18 | 44 | 7 | 0 | 7 | 0 | 0 | 0 | ***95*** |
| **Highland** | Oct-96 | 9.1 | SU25250890 | 39.3 | 16 | 22 | 77 | 4 | 0 | 2 | 0 | 0 | 0 | ***121*** |
| **Highland** | Nov-96 | 10.1 | SU28270880 | 25.7 | 4 | 35 | 24 | 4 | 0 | 4 | 0 | 0 | 0 | ***71*** |
| **Highland** | Nov-96 | 11.1 | SU25030933 | 34.2 | 17 | 8 | 106 | 7 | 0 | 4 | 0 | 0 | 0 | ***142*** |
| **Highland** | Nov-96 | 12.1 | SU25450870 | 50.0 | 13 | 27 | 29 | 12 | 0 | 1 | 0 | 0 | 0 | ***82*** |
| **Highland** | Jan-97 | 13.1 | SU26870785 | 68.1 | 15 | 155 | 60 | 18 | 7 | 5 | 0 | 0 | 0 | ***260*** |
| **Highland** | Jan-97 | 4.2 | SU26750790 | 33.4 | 13 | 121 | 17 | 32 | 0 | 0 | 0 | 0 | 0 | ***183*** |
| **Highland** | Jan-97 | 5.2 | SU26700850 | 40.3 | 12 | 123 | 6 | 16 | 2 | 5 | 0 | 0 | 0 | ***164*** |
| **Highland** | Mar-97 | 14.1 | SU25000950 | 40.0 | 10 | 4 | 54 | 7 | 0 | 1 | 0 | 0 | 0 | ***76*** |
| **Highland** | Mar-97 | 3.2 | SU26810785 | 29.9 | 8 | 35 | 8 | 13 | 0 | 4 | 0 | 0 | 0 | ***68*** |
| **Highland** | Mar-97 | 6.2 | SU26520811 | 64.5 | 16 | 48 | 45 | 32 | 1 | 3 | 0 | 0 | 0 | ***145*** |
| **Highland** | Mar-97 | 8.2 | SU25450865 | 52.0 | 20 | 38 | 41 | 42 | 0 | 5 | 0 | 0 | 0 | ***146*** |
| **Highland** | Apr-97 | 12.2 | SU25450870 | 67.9 | 76 | 55 | 47 | 13 | 0 | 2 | 0 | 0 | 0 | ***193*** |
| **Highland** | Apr-97 | 9.2 | SU25250890 | 44.1 | 145 | 7 | 84 | 2 | 0 | 3 | 0 | 0 | 0 | ***241*** |
| **Highland** | May-97 | 10.2 | SU28270880 | 30.3 | 26 | 17 | 62 | 9 | 0 | 3 | 0 | 0 | 0 | ***117*** |
| **Highland** | May-97 | 14.2 | SU25000950 | 20.0 | 54 | 0 | 73 | 0 | 0 | 0 | 0 | 0 | 0 | ***127*** |
| **Highland** | May-97 | 17.1 | SU25250888 | 33.4 | 41 | 18 | 112 | 15 | 0 | 1 | 0 | 0 | 0 | ***187*** |
| **Highland** | Jun-97 | 18.1 | SU26950770 | 27.0 | 19 | 57 | 50 | 31 | 0 | 7 | 0 | 0 | 0 | ***164*** |
| **Highland** | Jun-97 | 19.1 | SU25240791 | 41.1 | 36 | 31 | 157 | 28 | 2 | 7 | 0 | 0 | 0 | ***261*** |
| **Highland** | Jun-97 | 20.1 | SU24800998 |  | **HABITATS** | **MEASURED** | **BUT** | **SAMPLES** | **NOT USED** | **BECAUSE** | **OF** | **HIGH WEIR** | **BLOCKING** | ***ACCESS*** |
| **Highland** | Jun-97 | 21.1 | SU25240791 | 14.3 | 24 | 0 | 36 | 2 | 0 | 0 | 0 | 0 | 0 | ***62*** |
| **Highland** | Jul-97 | 22.1 | SU27408685 | 56.6 | 24 | 215 | 33 | 89 | 13 | 1 | 0 | 0 | 0 | ***375*** |
| **Highland** | Jul-97 | 7.2 | SU25500850 | 30.4 | 35 | 3 | 56 | 9 | 0 | 2 | 0 | 0 | 0 | ***105*** |
| **Highland** | Sep-97 | 13.2 | SU26870785 | 47.2 | 27 | 107 | 35 | 34 | 15 | 2 | 0 | 0 | 0 | ***220*** |
| **Highland** | Sep-97 | 14.3 | SU25000950 | 40.0 | 27 | 0 | 153 | 0 | 0 | 0 | 0 | 0 | 0 | ***180*** |
| **Highland** | Sep-97 | 22.2 | SU27408685 | 63.8 | 18 | 103 | 60 | 32 | 9 | 0 | 0 | 0 | 0 | ***222*** |
| **Highland** | Oct-97 | 22.3 | SU27408685 | 62.0 | 20 | 176 | 53 | 42 | 10 | 1 | 0 | 0 | 0 | ***302*** |
| **Highland** | Feb-98 | 22.4 | SU27408685 | 60.1 | 8 | 56 | 20 | 42 | 2 | 0 | 0 | 0 | 0 | ***128*** |
| **Highland** | Feb-98 | 5.3 | SU26700850 | 41.1 | 13 | 117 | 13 | 64 | 3 | 2 | 0 | 0 | 0 | ***212*** |

Numbers of each species of fish caught in the whole reach are shown for each sampling date

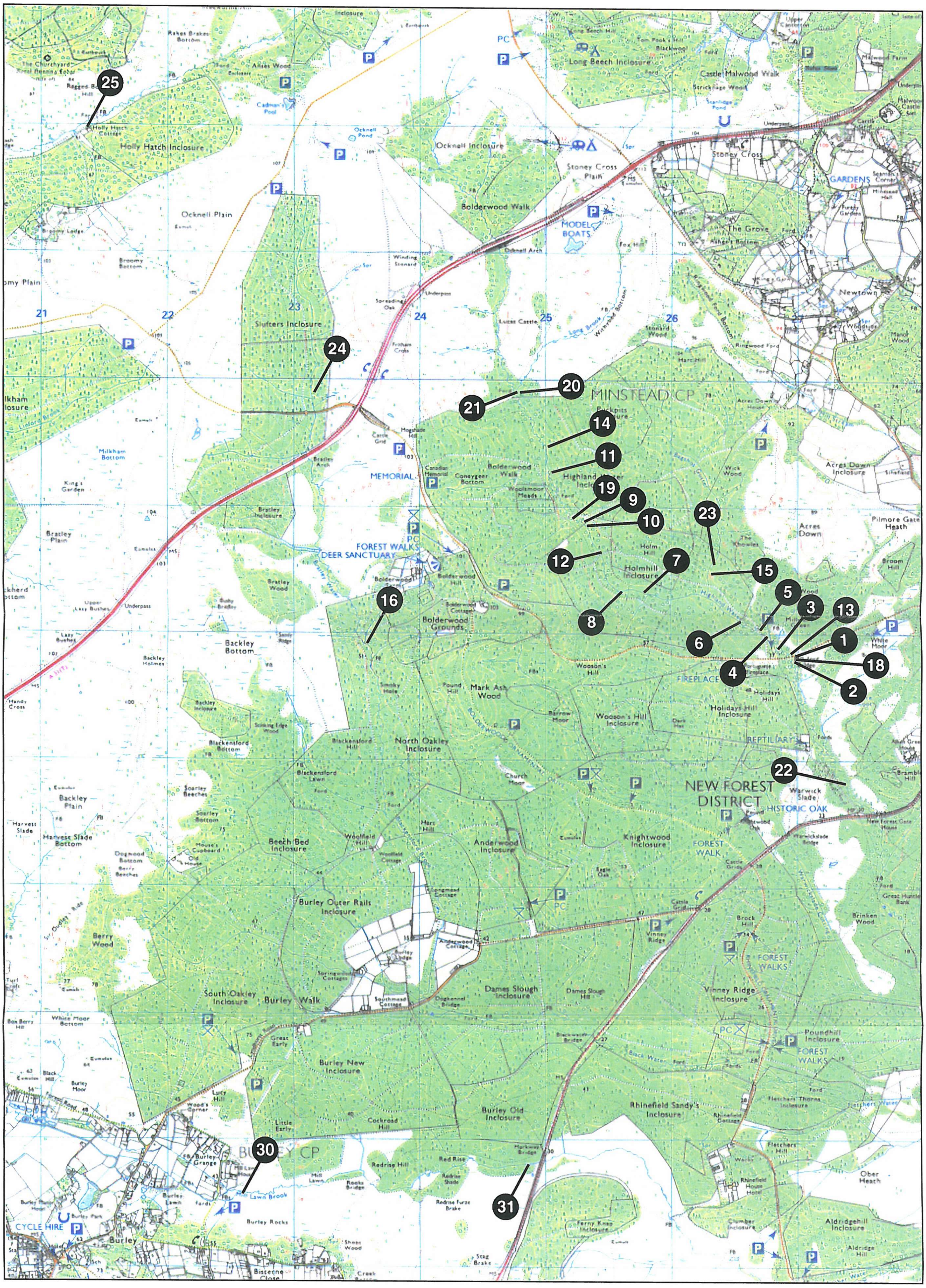
**Appendix 1** **Part 2**

| | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Bratley** | May-97 | 16.1 | SU23600792 | 42.5 | 83 | 103 | 45 | 105 | 6 | 10 | 0 | 0 | 0 | ***352*** |
| **Bratley** | Oct-97 | 16.2 | SU23600792 | 47.9 | 28 | 199 | 29 | 53 | 6 | 6 | 0 | 0 | 0 | ***321*** |
| **Bratley** | Feb-98 | 16.3 | SU23600792 | 44.0 | 30 | 76 | 14 | 26 | 14 | 4 | 0 | 0 | 0 | ***164*** |
| **Bratley** | Jul-98 | 24.1 | SU23200980 | 37.9 | 26 | 0 | 62 | 0 | 0 | 0 | 0 | 0 | 0 | ***88*** |
| **Bagshot** | Apr-97 | 15.1 | SU26400855* | 50.2 | 66 | 2 | 15 | 21 | 0 | 3 | 0 | 0 | 0 | ***107*** |
| **Bagshot** | Oct-97 | 23.1 | SU26350850* | 26.4 | 17 | 10 | 7 | 7 | 0 | 1 | 0 | 0 | 0 | ***42*** |
| **Dockens** | Jul-98 | 25.1 | SU21451200 | 37.9 | 7 | 223 | 0 | 0 | 58 | 4 | 14 | 40 | 1 | ***347*** |
| **Dockens** | Aug-98 | 25.2 | SU21451200 | 24.4 | 3 | 85 | 0 | 0 | 53 | 2 | 12 | 48 | 0 | ***203*** |
| **Dockens** | Aug-98 | 29.1 | SU19451125 | 48.1 | 29 | 13 | 0 | 9 | 0 | 1 | 5 | 0 | 0 | ***57*** |
| **Dockens** | Nov-98 | 25.3 | SU21451200 | 39.0 | 2 | 121 | 0 | 2 | 45 | 2 | 13 | 28 | 0 | ***213*** |
| **Dockens** | Nov-99 | 32.1 | SU16350850 | 54.0 | 28 | 35 | 18 | 24 | 0 | 13 | 14 | 0 | 0 | ***132*** |
| **Dockens** | Feb-00 | 33.2 | SU18601045 | 55.0 | 23 | 25 | 6 | 22 | 2 | 2 | 4 | 0 | 0 | ***84*** |
| **Ober** | Aug-98 | 31.1 | SU24950390 | 82.0 | 6 | 478 | 66 | 152 | 9 | 14 | 24 | 1 | 0 | ***744*** |
| **Ober** | Nov-98 | 30.2 | SU22650360 | 38.3 | 17 | 52 | 61 | 51 | 5 | 17 | 6 | 5 | 0 | ***197*** |
| **Ober** | Nov-98 | 31.2 | SU24950390 | 39.6 | 5 | 23 | 23 | 0 | 0 | 2 | 12 | 0 | 0 | ***60*** |
| **TOTAL FOR ALL SAMPLES** | | | | **2161.0** | **1251.0** | **3538.0** | **2004.0** | **1229.0** | **270.0** | **166.0** | **104.0** | **122.0** | **1.0** | **8657.0** |

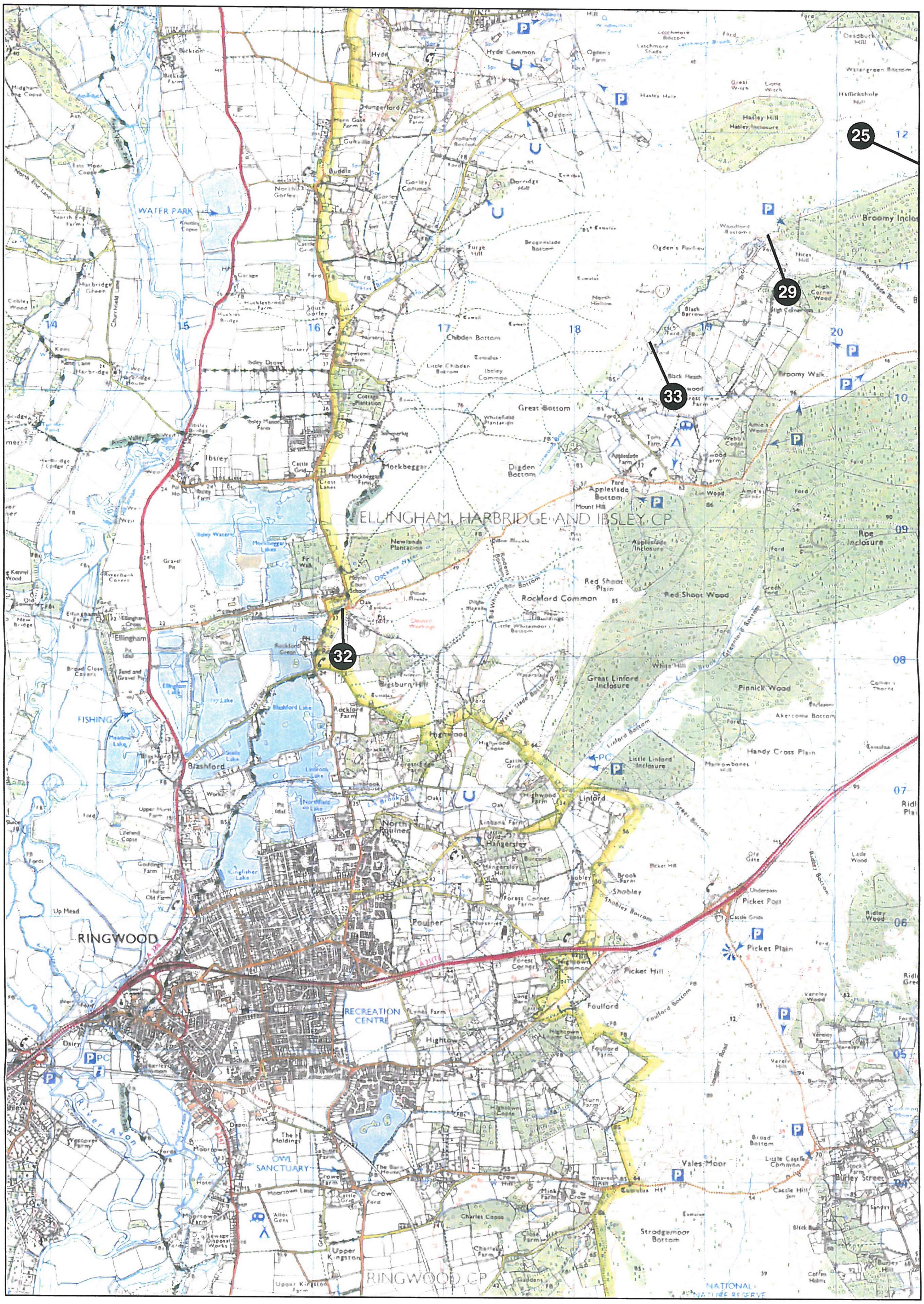
**Numbers of each species of fish caught in the whole reach are shown for each sampling date**

**APPENDIX II.** Appendix II comprises two OS maps (1:25000) of parts of the New Forest showing the locations of the sites (reaches) used for this research. Each site is labelled with its ID number (see Appendix 1). The stream and overall sampling areas are summarised diagrammatically in Fig. 2.1.

**APPENDIX III**. Appendix III is a 1.4 Mb computer disc containing the raw fish catch data for each successive sub-sample from each habitat unit at each site on the Highland Water streams during 1996-98. The format is MS Excel spreadsheet.



APPENDIX II. Map of parts of the New Forest in Southern England showing locations of sites used for sampling fish populations. Part I



APPENDIX II. Map of parts of the New Forest in Southern England showing locations of sites used for sampling fish populations. Part II