ECOLOGICAL STUDIES ON
BENTHIC MACROINVERTEBRATE COMMUNITIES
IN RELATION TO THEIR USE IN
RIVER WATER QUALITY SURVEILLANCE

2 VOLUMES

VOLUME 1

by

CHRISTOPHER GIRTON B.Sc., M.Sc., MI.Biol.

THESIS SUBMITTED FOR THE DEGREE OF
DOCTOR OF PHILOSOPHY

THE UNIVERSITY OF ASTON IN BIRMINGHAM
JULY, 1980

ECOLOGICAL STUDIES ON BENTHIC MACROINVERTEBRATE COMMUNITIES IN RELATION TO THEIR USE IN RIVER WATER QUALITY SURVEILLANCE

Submitted by Christopher Girton to the University of Aston in Birmingham
for the Degree of Doctor of Philosophy, 1980

SUMMARY

Seven different types of colonisation substrata samplers were compared with two direct sampling methods for sampling quantitatively benthic macroinvertebrates of riffles and pools in the River Tean, Staffordshire. Differences were found in the species composition of the catches; in riffles the largest number of species was collected by cylinder sampling. In pools, colonisation substrata samplers were effective in collecting the most taxonomic groups.

Due to problems encountered in directly sampling benthic macroinvertebrates in lowland rivers, with no suitable riffles, and in the lower stretches of larger rivers,a colonisation sampler, the Standard AufwuchsUnit (S.Auf.U.) was tested at 33 stations on 17 rivers in 6 different Water Authority areas representing a wide range of river types and water qualities. Where possible, associated riffles of the same water quality were studied by direct sampling, to establish any relationship with the S.Auf.U. community. In addition, grab samples of the depositing substratum were also taken. Three replicate samples by each method were taken monthly and analysed by biotic indexes and appropriate data processing techniques.

S.Auf.U. generally collected larger numbers of taxa than cylinder or grab sampling, but were found to be influenced by the proximity of upstream riffles. A positive correlation was established between the biotic scores derived from the direct sampling of riffles and S.Auf.U. sampling in associated pools. Biotic scores calculated from S.Auf.U. data showed a general trend across a range of chemically-indicated water qualities. Computer analysis of invertebrate assemblages on S.Auf.U. to investigate degrees of similarity between stations, linked the severaly polluted stations, but these were also linked with some lowland good quality ones.

Based on effectiveness and practicality trials, S.Auf.U. are recommended for monitoring lowland rivers, whereas riffles are best sampled by direct cylinder sampling.

Biological surveillance studies on the Rivers Churnet and Trent confirmed the sensitivity of riffle communities to changes in water quality.

Key words:

Biological surveillance, macroinvertebrates, water quality, colonisation samplers, biotic indexes.

IN LOVING MEMORY OF MY FATHER NEVILLE GIRTON
WHO WAS UNABLE TO SEE THIS THESIS IN ITS FINAL
FORM. I HOPE HE WOULD HAVE BEEN PROUD OF ITS
CONTENTS AND TO HIM, MY MOTHER AND MY WIFE, HELEN,
I ADD MY VERY SPECIAL THANKS

ACKNOWLEDGEMENTS

I wish to thank Mr. H. A. Hawkes for all of his help, supervision and expert advice during the course of my post-graduate studies and for introducing me to the expanding field of hydrobiology.

Thanks should also go to Mr. G. Nickolds for technical assistance in the field and laboratory and to all research groups who helped in the identification of difficult taxonomic groups.

This project involved field work in different Water Authority rivers, and I express my appreciation for the encouragement and help received from the river biologists in the several areas, in selecting sites and providing information regarding pollution sources. Different Water Authorities kindly provided analytical data related to several of the sites studied.

The work formed part of a research contract with the Water Data Unit (Department of the Environment).

A special thanks must go to Mrs. Anne Jones for all of her excellent typing and perseverance in collating Volume I of this thesis, and also to Mrs. E. Moore and Miss Y. J. Puzey for assistance in Volume 2.

CONTENTS

VOLUME 1

VOLUME 2

APPENDICES

Page

FIGURES

Page

TABLES

Page

PLATES

CHAPTER 1

INTRODUCTION

The changes that man brings about in his environment are determined largely by his necessities, his knowledge, and his values. Such demands have often led to great stress factors being imposed on the ecosystem, which man is part of, and consequently the aquatic medium has been one of the systems affected. Efforts have been made to stem the environmental degradation that has followed, but only recently has general public awareness of the seriousness of these problems become great enough to offer hope that the tide may have been turned.

In recent years, water pollution has come to be recognised for what it is: a part of the critical problem of water resource management, and part of this system is the water quality criteria needed for aquatic life. The basic purpose of a criterion for aquatic life is to restore and maintain environmental conditions that are essential for the survival, growth, reproduction and general well being of the aquatic ecosystem. Because man relies on streams, rivers and lakes for his water supply and many of his recreational pursuits, it is essential to have some method of assessing water quality.

The word pollution, to signify contamination of water, has been defined by many workers over the years and certain misconceptions have arisen due to such definitions. To many people, water pollution means the introduction into water of anything dirty, regardless of the amounts or effects of the material introduced. This idea of pollution does not easily include the introduction of small amounts of radio-active material or toxic substances, or the causing of temperature or salinity changes. As times change, man's concepts change and words take on new meanings.

Some biologists have included in their definitions of pollution the concept of measurable change in the aquatic environment, but they have not included the concept of reduction in the value of any use of the water by man. Patrick (1950) defined pollution as 'anything which brings about a reduction in the diversity of aquatic life and eventually destroys the balance of life in a stream'. Others with similar views have stated that pollution is any influence on the stream brought about by the introduction of materials to it which adversely affects the organisms found living in the stream. According to these definitions, this change in the biological community would be evidence of pollution and no consideration would be given to whether or not any change had occurred in the value of the water to man. Legal definitions of pollution tend to be complex and often misleading in concept following the lines of the addition of something to water which changes its natural qualities so that the riparian owner does not get the natural water of the stream transmitted to him. Hynes (1960) rightly pointed out in this case, what does one mean by the natural water of a stream and what are its natural qualities? Looking at this situation, if river water were pure H_2O one could then define these natural qualities and monitor any changes in the physical and chemical conditions which take place. However, water normally contains different types of nutrients which can change depending on the season of the year and geographical location. It is evident therefore, that pollution may merely change the type of 'natural' water into another. Only when pollution is so severe as to cause changes which overstep the boundaries of normal variation, is it possible to say that the water has lost its 'natural qualities'.

To formulate objectives on water pollution, a good definition of pollution would be one of any discharge to a clean river by man, which so changes the natural qualities that its legitimate use to man is impaired by this condition. These legitimate uses would include practical ones, such as navigation, industrial, agricultural and domestic requirements; recreational uses, including fishing, boating and bathing, and also the satisfaction of man's aesthetic interests.

Water quality is often difficult to define, impossible to measure absolutely and is very subjective. Water quality criteria have been defined by Warren (1971) as 'any definite limit of variation or alteration of water quality expertly judged on the basis of scientific data, not to have some specified, usually adverse, effect on the use of water by man or on organisms inhabiting the water'. However one defines water quality like pollution, more emphasis should be placed on the specific use of the water by man.

Historically, physical and chemical data have been used almost exclusively in assessing water quality compared to biological surveillance. Physical and chemical data tend to be highly variable as chemical substances which affect the quality of water are numerous, act in a great range of concentrations and vary continuously in amount. Moreover, physico-chemical surveys indicate water conditions only at the times of sampling. Biological data, in contrast, reflect interactions of organisms both with each other and with the physical and chemical environment.

Biological surveillance has been variously defined, usually in terms of a continued programme of surveys systematically undertaken to provide a series of observations in time (Hellawell, 1978). Surveillance implies not only measurement and detection of change but also its understanding and interpretation and the development of a capacity for predicting likely future change. It cannot usefully be considered in isolation but must be accompanied by physical and chemical observations and by basic ecological and physiological research. The broad aims of biological surveillance are thus to detect temporal and spatial changes, to understand and explain the causes of these changes and to provide a capacity for predicting future change. Such a system may prove extremely valuable for detecting and measuring the ecological effects of unsuspected or intermittent pollutants, which arise through changes in water management procedures. Biological surveillance may also provide useful indications of likely sources of potential 'nuisance' organisms which could influence decisions regarding proposed water management schemes. Examples here would include certain algae which often impose problems in impounded waters, or undesirable fish species or their parasites. Consequently, biological surveillance describes the changing status and distribution of such resources and provides basic information for the conservation of rare species.

In the United Kingdom, as in many other countries, biological surveillance of rivers using macroinvertebrates has become established as an integral part of monitoring water quality. The setting up of the Water Authorities which resulted in more biologists being employed in this field, and the requirement by the Department of the Environment for a biological as well as a chemical classification in the

1970 National River Pollution Survey, both probably initiated the upsurge that took place in the use of biological monitoring for assessing river water quality. The report on the 1970 River Pollution Survey (Department of the Environment, 1972) admits for having some reservations regarding the adequacy of the biological classification system used for application to all types of river. The method, using four classes (A, B, C and D) based on the presence or absence of various invertebrate groups (considered indicative of different grades of water quality), was unsatisfactory for a variety of reasons and was not used in subsequent surveys where chemical parameters alone were used.

Although it would be wrong to assume that the poor degree of correlation between the chemical classification and the biological classification in 1970 was always attributable to the biological method, some of the anomolies were due to the biological classification adopted. The most obvious inconsistencies arose in the classification of the slow flowing rivers of East Anglia, where the biological classification indicated markedly inferior water qualities than did the chemical classification. The classification used required for a Class A river the presence of an appreciable proportion of Plecoptera and/or Ephemeroptera, Trichoptera and Amphipoda. Such communities are not found naturally in the dykes and sluggish muddy bottomed rivers of East Anglia and similar areas, however good the water quality. Such communities are typical of riffles - rapid flowing - shallow stretches with an eroding substratum, common in upland rivers. Most of the British methods using benthic invertebrates have been successfully developed in areas where the rivers were of this type (rhithron). Such methods are less successful when applied to the lowland rivers with depositing substratum (potamon). It is of interest to note that in the report of the 1970 River Pollution Survey, whereas in the rapidly-flowing Wye and tributaries (Wye River Authority) approximately 98% of the miles of river classified chemically as Class 1 were classified as Class A biologically; in the Lincolnshire River Authority area, where the rivers are more sluggish, only approximately 6% of the chemical Class 1 rivers were Class A biologically, 27% being Class B and 64% Class C. This illustrates one of the limitations of using benthic invertebrate communities for monitoring river water quality on a national basis. It also highlights the need to take into account the river type in interpreting the data from invertebrate samples in relation to water quality (Hawkes, 1975).

Although in the 1975 National River Pollution Survey a biological classification was omitted, it is the intention of the Department of the Environment to include a biological classification in the next survey, 1980. With this in mind, the Biological Monitoring Working Party (B.M.W.P.) of the Department of the Environment Standing Technical Advisory Committee on Water Quality was set up to recommend a biological classification of river water quality for use in National Pollution Surveys. Economic constraints in terms of effort available for such surveys, dictate that the method be simple, necessitating a compromise between ecological validity and logistic feasibility. The Working Party were unable to recommend a system of biological classification of river water quality but instead recommended a system to assess the biological condition of a river for National Survey purposes. A Biotic Index Score system based on the presence of selected invertebrate families was finally adopted (Department of the Environment, 1980). This method, although relatively crude, should satisfy the not very demanding requirements of a broad classification system needed for National purposes. More ecologically exacting systems can still be used for specific requirements within the Water Authorities. The proposed system goes some way to recognising the inherent differences of the invertebrate communities of lowland rivers by including families typical of such zones, such as Odonata and Hemiptera which are not taken into account in the earlier Score systems. The score attributed to some families is also related to the nature of the substratum whether eroding or depositing. Nevertheless the problems associated with the sampling of lowland rivers and relating the data to that of the upland rivers in terms of water quality remains. The B.M.W.P. Scores calculated in this thesis were derived from the final version of the fifth Department of the Environment draft produced in 1979. A modification of this system published by the D.o.E. (1980) now combines the eroding and depositing scores (Table 4.4) and drops each group's score by an order of magnitude. The results of the two systems, however, are significantly similar (99% similarity - R. Chesters, Water Data Unit, personal communication).

Related to the work of the Biological Monitoring Working Party, the Water Data Unit of the Department of the Environment initiated research to develop and evaluate a standard biological method for the surveillance of river water quality. Sampling methods applicable to all types of river were to be investigated. In this investigation, benthic invertebrates and microbial periphyton were selected for study as being the most appropriate communities for monitoring river water quality. This thesis presents the invertebrate studies related to this project.

CHAPTER 2

LITERATURE REVIEW

2.1 Biological surveillance and pollution

It is now generally accepted that benthic macroinvertebrates are useful indicators of water quality (assessment) in streams and rivers (Hellawell, 1978). Recent reviews have emphasised their importance (Hynes, 1960; Hawkes, 1962; Warren, 1971; Hellawell, 1978).

Although macroinvertebrates are rarely used in bioassay studies compared to algae and fish, they have proved extremely useful in monitoring water quality in two different ways:- community structure and as indicator organisms. The first approach to the assessment of water quality using macroinvertebrates involves determining the type of organisation of the benthic community. The utilisation of mathematical expressions, termed diversity indexes are widely used and as reported later (section 2.123) may have great potential value. There are, however, many problems inherent both in the choice of an index and in the interpretation of the meaning of the estimate of organisation or diversity index that is calculated. The second approach involves the use of macroinvertebrates as indicator organisms. Although there have been several useful reviews of the literature on the use of indicator organisms (Gaufin and Tarzwell, 1952; Hawkes, 1962) the work by Sladecek (1973a) is currently the most comprehensive work on the subject.

The tolerance of a community to stress is dependent on the nature of the receiving system, and the nature and degree to which the stress is applied. However, where the stress is sufficient to cause a response, either the numbers of individual organisms or the number of taxa, or both, will be affected. Hynes (1960) has stated that a community of macroinvertebrates in an aquatic ecosystem is very sensitive to any changes and serves as a useful tool for detecting pollution. Furthermore, Patrick (1950) stated in her paper on a biological survey in Pennsylvania that any changes occurring in an aquatic environment cause changes in composition of a benthic community. A pollutant may eliminate many macroinvertebrates; thus, those that remain may become abundant as a result of decreased competition.

The entry of pollutants into a flowing stream sets off a progressive series of physical, chemical and biological events in the downstream waters (Butcher, 1946; Bartsch, 1948) and their nature is governed by the character and quantity of the polluting substance. Domestic or industrial effluents may adversely affect natural stream life by direct toxic action, or indirectly through quantitative alterations in the character of the water or the streambed (Campbell, 1939; Biglane and Lafleur, 1954).

2.11 Historical background

Numerous stream surveys conducted during the last 50 years have illustrated the effects of pollution on the physical and chemical characteristics and the biota of receiving waters. Hawkes (1978) reviewing the literature on biological surveillance in Britain concluded that it was only in recent years that biology has played an important role in water authority assessment of stream pollution.

Before this, it was the function of the chemist to monitor the aquatic environment, with the biologist only playing a secondary role.

In Europe the earliest work based on the biological assessment of water quality was that of Kolkwitz and Marsson (1908; 1909). They described the various ecological conditions associated with the different stages of recovery during the self-purification of rivers which had been grossly polluted with putrescible organic matter as sewage. Their Saprobien system, discussed in section 2.12, was later modified in the United States by Richardson (1928), who was the pioneer of such research in that country. He studied the steady increase in the pollution of the Illinois river from 1913 to 1929 and observed the establishment of pollution zones move steadily downstream until they extended 250 miles from the source of the pollution near Chicago. Campbell (1939) and Brinley (1942) also in the United States developed, quite independently of the European system, the concept of differing zones of pollution. Although their concept was similar to that of Kolkwitz and Marsson, their zones were less rigidly defined and therefore did not correspond to the European system (Hynes, 1960).

In Britain the earliest attempts of using benthic organisms as indicators of pollution was founded on the work by Carpenter (1926). She examined the effects of lead mines on Cardiganshire trout and salmon rivers and recognised three phases of recovery from pollution based on the fauna present. Similar studies were also reported by Butcher et al. (1937) and Pentelow and Butcher (1938).

Butcher (1946) having studied the effects of sewage on the biota in small streams for twenty years, indicated that further study of this type using animals and plants, would make it possible to investigate the direct evidence of water pollution and not rely entirely on the indirect chemical analysis. The effect of pollution and repurification on the fauna and flora of the River Trent was subsequently reported by Butcher (1946). The pollution of the Trent was from the Pottery Towns of Stoke-on-Trent and this affected the river for 35 miles. In the first few miles of pollution, reflected by low dissolved oxygen and high ammoniacal nitrogen levels, the numbers and species of animals were few, while sewage fungus was abundant. The first animals to appear were tubificids approximately 4 miles below the effluent and the first stages in the recovery of the river was indicated by an increase in the numbers of tubificids and chironomids 8 to 10 miles below Stoke. As one proceeded downstream these species were replaced by an increase in the numbers of Asellus up to 24 miles below the effluent. In further stages of recovery downstream, Gammarus, molluscs and caddisfly larvae respectively reappeared.

Butcher's (1946) work on the River Tame, a major tributary of the Trent, also provided similar faunal sequences as the Trent, but repurification of the Tame, however, was considerably slower. This delayed purification was probably due to the suppression of the organisms concerned with repurification by toxic discharges in the upper Tame and tributaries (Hawkes, 1962).

Similar results were also obtained by Pentelow and Butcher (1938) and Butcher (1946) on the River Churnet, Staffs. The River Churnet at the time of Butcher's (1946) investigation was in its upper reaches organically polluted by sewage effluent resulting in a decrease in the diversity of species and an increase in the numbers of *Asellus*. Further downstream a dye works eliminated further species with a resulting increase in tubificids. 7.5 miles downstream of these effluents the river had partially recovered, the fauna being abundant and varied. The river then received a discharge from a copper works at Froghall, increasing the level of copper in the river to 1.0 $mg.l^{-1}$ and completely eliminating the fauna. Life reappeared 11 miles below the copper effluent and this only consisted of a few red chironomids and a single larva of a caddis fly.

Jones (1940, 1958) in his studies of metallic pollution of the River Ystwyth in western Wales clearly demonstrated biological succession in rivers. As the river recovered, both in time as the exposed ore became leached out, or in space as ore proceeded downstream to regions of greater dilution, many algae reappeared, together with tubificids and a few fish at intervals. In the earlier stages of the work no distinction was made between the effects of lead and zinc (Jones, 1958), but it was apparent that some forms, particularly stoneflies, e.g. *Leuctra* sp. and *Nemoura* sp., were very resistant to both and they were found living in water containing nearly 60 $mg.l^{-1}$ of zinc (Jones, 1940).

Following this period the foundation had been set for a rapid increase in interest in biological pollution control and surveillance. At first Butcher's work and ideas received little attention in

Britain (Hawkes, 1978) and only isolated studies were reported (Hawkes, 1956). This latter work examined the changes in the relative populations of dominant organisms found living in stream beds. The relative frequency of the organisms and their classification in the Saprobien system (Kolkwitz, 1950; Liebmann, 1951) were both taken into account in assessing the condition of different polluted stations. The biological methods described were developed to suit local conditions and were not necessarily applicable throughout the country. However, it was suggested that in assessing the pollution of streams, the biological as well as the chemical and physical conditions of the stream, should be taken into account.

The work of Hawkes (1956) in selecting carefully chosen sites along the length of a river, where conditions were similar, was fortunate, because these riffle regions seemed to be more sensitive to organic pollution than the silted community (Hawkes, 1962).

During the fifties biological surveillance studies in the United States were at an advanced stage compared to those in Britain. Gaufin and Tarzwell (1952) revealed during their year-round study of Lytle Creek that the quantitative and qualitative composition of an aquatic population constituted a valuable index in delineating zones of pollution in a stream. Such zones were particularly distinct during the summer months when flows were low, resulting in variations in dissolved oxygen and maximal pH values. During winter, with increased flow rate and a slower rate of microbial activity, higher concentrations of organic matter were carried into the lower sections of the stream. Gaufin and Tarzwell (1956) continuing the work on Lytle Creek found that all the species of invertebrate animals living in the septic and recovery zones were also present in limited numbers in similar microhabitats in the clean-water zones. These studies also revealed that little reliance could be placed upon the mere occurrence of a single species in a given locality as an indicator of pollution. Another reason for the lack of agreement as to the indicator value of certain aquatic macro-invertebrates was that several environmental factors other than the presence of a pollutant may affect or limit the distribution of a particular species. Such factors are described in detail in section 2.4.

Major contributions to an understanding of the biotic effects of pollution were also made by Wurtz (1955). The interpretation of stream conditions by Wurtz (1955) resulted in the development of a somewhat less complicated system for conducting stream surveys. The percentage species composition of tolerant and non-tolerant forms and the classification of organisms into four modes of life (burrowing, sessile, foraging and pelagic) in the form of a histogram resulted in a system for assessing the pollutional conditions of streams. Since this system required information as to the tolerance of the species involved (data often not available) to different types of pollution, its value as a pollution index had limited significance (Hawkes, 1962).

In Britain the work of Hynes (1959; 1960) and Butcher (1959) further recommended the use of benthic invertebrates as indicators of river pollution.

Special emphasis was placed on their value as reflecting the long term effect of an effluent and their ability to reflect the most extreme conditions of pollution. They advocated that the detection of gross pollution was far easier for the biologist to measure than it was for the chemist and was probably less time consuming. Hynes (1959) in his work made no attempt to use any particular system for the assessment of his results. He considered that when Kolkwitz drew up his Saprobien system he was rather ignorant of the larger animals and considered them of little importance. Because of this he felt that any attempt to adhere to a rigid system in the application of biological methods tended to lead to their falling into disrepute. Hynes (1959) was also critical of the system reported by Wurtz (1955) because he felt that pollution was itself very varied and different types of organic pollution produce slightly different effects, and thus produce different ecological conditions in rivers. These are reflected in differences in the species which thrive under the particular conditions.

Practical application of biological surveillance for classifying the pollutional condition of rivers in the Bristol Avon River Board Area was reported by Bielby (1960). As a modification of Kolkwitz and Marsson Saprobien system, Bielby used the method of Hawkes (1956) to classify the rivers enriched by organic pollution. He found that this method was without doubt the most readily usuable and acceptable method devised to date. He found that its great advantage for River Board work was that it was relatively simple to use and the results were easily intelligible to those who were not familiar with the subject. The work of Butcher in the Trent River Board was continued by a succession of biologists who established the Biotic Index and this was described by Woodiwiss (1964). The implications of this system discussed in section 2.12, have led to the present day methods of processing biological surveillance data in Britain.

With an increase in interest in biological surveillance apparent, further publications were documented during this period. Hawkes (1963) reported the recovery of Midland streams from gross organic pollution and revealed once again that the riffle community was remarkably sensitive to changes in the organic loading of the receiving water. Hawkes (1963) concluded that this sensitivity may be largely due to the current bringing these basic materials to the benthos. In this work comparisons were made between organic discharges and toxic industrial discharges affecting the stability of riffle communities. With toxic discharges, it was found that the chemical environment was altered in such a way that the number of species present in the community was severely restricted. Any increase in the population of the tolerant species due to reduced interspecific competition, would only be a secondary biotic effect. In contrast the effects of putrescible organic discharges, would introduce additional basic matter into the community, are principally nutritive or biotic. These result in changes in the relative populations within the community, with subsequent exploitation by tolerant species.

Brinkhurst (1965) studying a section of the River Derwent near Derby over a period of five years showed that recovery from gross organic pollution may be so rapid that some of the resulting stages in repurification may be transient or omitted, depending on a series of factors. Hynes (1960) showed that, in the Welsh Dee and in the River Lee in Hertfordshire, recovery from industrial organic matter and cyanide, respectively, could be shown to be rapid for those organisms that were mobile, but that slow-moving species took longer to re-colonise a river.

With an upsurge of interest in the freshwater environment generally over the last fifteen years; in particular power station effluents (Mann, 1965; Langford and Aston, 1972; Aston, 1973a) and domestic and industrial effluents (Hawkes and Davies, 1971; Learner et al., 1971; Williams et al., 1976) it has become apparent that there is a considerable need for the biological surveillance of river water quality. As Hawkes and Davies (1971) pointed out although some rivers should be preserved in their natural state, and all unnatural discharges to them prohibited, many rivers will need to be used for conveying our effluents. Such a practice will inevitably affect the benthic communities and simplify the stream ecosystem especially where the flow of natural diluting water is low. It should, however, be possible to manage such rivers by flow regulation and pollution control to enable them to support a benthic community beneficial to man. It is by surveillance that such objectives will be realised.

2.12 Pollution Indexes

2.121 Saprobic system

Saprobity is the state of the water quality with respect to the content of putrescible organic material, as reflected by the species composition of the community (Sladecek, 1978). A community indicates the saprobic level prevailing over a period of time sufficient for its development. This system was first proposed by Kolkwitz and Marsson (1908; 1909) for the estimation of organic pollution and has probably received the greatest recognition of indexes developed this century.

The Saprobic system was based on lists of organisms classified into different 'saprobia' groups of species associated with different degrees of organic pollution. In this system four zones were distinguished: polysaprobic, α- mesosaprobic, β-mesosaprobic and oligosaprobic, on the basis of increasing degrees of mineralisation of the organic matter. Some biologists have criticised the effects of this system because in their opinion the early biologist placed too much reliance on the tolerance classification and the mere presence or absence of particular species (Gaufin and Tarzwell, 1956; Hawkes, 1962; Edwards, 1975). The applicability of a system of indicator organisms is limited in many instances by the presence of toxic substances of industrial origin amongst organic pollutants.

These recent biologists have believed the composition of a community (biocoenose) to be more reliable as an index of environmental conditions than would be the mere presence or absence of particular species. This is particularly true of micro-organisms since life is only possible in co-existence with other species of organisms (Fjerdingstad, 1964). Thienemann (1920) according to Fjerdingstad laid down two basic principles for bio-coenotics. First, the more varying the conditions of life are in a place, the greater are the number of species in the community present at the locality. Secondly, the further the conditions of life in the locality concerned are removed from the conditions that are optimal for a majority of species, the poorer in species becomes the biocoenose, the more characteristic and the more numerous each species. Fraiz (1952-53) added a third basic principle. The more continual the development of the environmental conditions has been and the longer the organism has been subjected to uniform environmental conditions, the richer is the community in species and the greater its stability.

The system of Kolkwitz and Marsson has the disadvantage that complete and precise analyses of animal and plant communities according to their species composition are necessary. This requires a considerable amount of time and special taxonomic knowledge and needs a long training and experience. For this reason Bartsch and Ingram (1966) have stated that the method is hardly used at all in the United States, where the number of aquatic species in fresh waters is much larger than in Europe and their taxonomy has received less attention.

The Saprobien system has been modified and developed over the last half-century by many European workers including Liebmann (1951) who abandoned the designation of the four grades of the saprobic system in favour of grades of water quality based on chemical, biological and physiological criteria. He considered micro-organisms to be far more important as indicators of pollution, and as a result did not adequately consider the macro-invertebrates.

Attempts have been made to take account of the relative abundance of organisms in the use of the Saprobien system rather than the presence or absence of particular species. Pantle and Buck (1955) described a system whereby the individual species are given a qualitative numerical value depending on the saprobic zone to which the species belongs and a numerical value representing the estimated quantities in which it occurs. Similar developments were reported by Knopp (1954) and Zelinka and Marvan (1961).

2.122 Biotic indexes

The degree of tolerance of organisms and the reduction in the numbers of species by organic pollution form the basis of systems which utilise simple numerical indexes to assess the water quality of rivers. Beck (1955) developed a 'biotic index' whereby macro-invertebrates were divided into two groups on the basis of their tolerance to organic pollution. He considered that the presence or absence of the tolerant organisms (Class I) was of greater importance than the non-tolerant forms (Class II); the numbers of species of these (Class I) organisms were therefore multiplied by an arbitrary factor of 2. The biotic index was calculated as:

Biotic index = 2(n Class I) + (n Class II), where n represents the number of species. According to Beck, a stream which is grossly polluted will have an index of zero, whereas a moderately polluted stream will have a biotic index of 1 to 6. In general practice, water qualities could range from zero to 40.

A more refined system developed by the scientific staff of the former Trent River Board by Woodiwiss (1964), known as the Trent Biotic Index was derived from a consideration of two effects of pollution; the reduction in community diversity and the progressive loss of certain groups from the clean water fauna in response to organic pollution. Aquatic macroinvertebrates were sorted into 'groups' whose size depended on their ease of identification. Streams were classified according to the presence or absence of key groups and the diversity of the fauna. Clean streams were given an index of $\overline{\underline{\text{X}}}$ and this figure was reduced with increasing organic pollution, with heavily polluted streams having an index of I. No account is taken of the quantitative representation of groups, but the strength of the index is in its practicality, whereby qualitative sampling is sufficient to derive a score.

The empirical index of pollution was adapted by Graham (1965) for use in the Lothians area of Scotland, but on his six-point scale a clean stream had an index of 1 with higher values indicating increasing pollution, up to a value of 6.

Besides being the basis for other systems developed in Britain the biotic index has been adapted successfully for use in France (Verneaux and Tuffery, 1967) and South Africa (Chutter, 1972a).

The systems developed previously did not take into account relative abundance and it was not until Chandler (1970) developed a new elaborate system that some of the difficulties inherent in the Trent and Lothian systems were overcome. Chandler's 'Score' system quantified the sensitivity to organic pollution of macro-invertebrates by producing a more detailed list of species, ranked on their sensitivity to increasing deterioration in water quality. Five levels of abundance were also recognised with the score of each indicator species indicated accordingly. To calculate this index the animals are identified and counted and each group present is assigned a score. Thus species intolerant to pollution receive higher scores and these scores increase with increasing abundance of these species, while tolerant species receive lower scores which decline in value with their increasing abundance. The score values generated thus provide a continuous assessment of water quality. The biotic score is the sum of the points gained by each species present in a standard sample. No upper limit is placed on the score although values would rarely exceed 3000 (Balloch et al., 1976). Polluted streams would have values ranging from 0 to 300 (Marstrand, 1973; Balloch et al., 1976). The lower limit, in the absence of any macroinvertebrates, would be zero. The main theoretical criticism of the Score system is the subjective way in which the scores are allocated both to taxon and in relation to their abundance (Hawkes, 1978). In practice the Score system is more demanding in terms of effort compared to the Trent Biotic Index, in that quantitative samples are necessary and the abundance of the individual taxa has to be assessed.

Modifications of the Score system have been reported by Cook (1976) who suggested that an 'ideal' index should combine sensitivity to the effects of pollution with wide applicability to different water courses and should provide a continuous scale of results.

Although a number of papers have been published during recent years in which the biological estimation of water quality has been converted into numerical values, Hawkes (1962) maintains - rightly - that biological communities are complex systems and cannot be reduced to exact numerical values to be entered into neat columns alongside the analytical results.

2.123 Diversity Indexes

An alternative means of assessing water quality is to measure the structure of the community, or its diversity in terms of the number of species represented by a given number of individuals within a sample.

The strictly quantitative approach of diversity was introduced initially by Margalef (1951) who expressed species richness of a community by the formula:

$$I = \frac{S - 1}{\log N} \qquad \text{where } S = \text{Number of species}$$
$$N = \text{Number of individuals.}$$

This method, mainly used in the United States and to a lesser extent in Britain, was modified by Shannon-Wiener and took account of relative abundance according to the formula:

$$\bar{d} = - \sum_{i=1}^{t} \left(\frac{n_1}{N} \log_2 \frac{n_1}{N}\right)$$

where $\bar{d}$ = diversity index

t = number of species

n = number of individuals in each species

N = total number of individuals

Wilhm and Dorris (1968) reviewed most of the principle indexes in current use and decided the Shannon-Wiener function to be the best indicator and measure of species diversity and also to be the best measure of the complexity of animal communities; the more similar the abundance of the various species and the higher the number of species, the higher the diversity of the community. It has the considerable advantage of being theoretically independent of sample size, so long as the species in the community are represented in due proportion, thus getting round the problem of quantitative sampling which has proved a serious obstacle in fresh water benthic faunal investigations for many years. Provided organisms can be recognised as being different, precise identification is not necessary. However, the index takes no account of the different tolerances to pollution of individual species (Cook, 1976). Furthermore, large changes in the population of a single species, which may be part of its natural life cycle, can markedly alter the index value, making it unsuitable for the interpretation of results from a site sampled infrequently (Mason, 1977a).

In practice, (Wilhm, 1970) it has been found that unpolluted waters have an index value between 3 and 4, while heavily polluted waters have values of less than 1. Since low diversity is indicative of polluted conditions, one would expect the level of diversity to increase as the pollutional 'stress' decreased. However, Hawkes (1978) has pointed out that low diversity may be caused by other forms of stress. Torrential headstreams with a good water quality may have a low diversity because of the severe physical conditions found in such environments. Hawkes (1978) also points out that temporal changes in diversity at one station are more significant than spatial changes along a length of river.

Hughes (1978) taking this a further step examined factors other than pollution which could influence the value of Shannon's diversity index. He found that the sampling method, the area sampled, the time of year and the taxonomic level of identification all influenced the value to varying degrees, while depth and duration of sampling had no apparent influence on the value. Hughes recommended that these factors should be taken into account when interpreting or comparing diversity index results for benthic invertebrate samples. Additionally although diversity index values are useful indexes of community structure, the ecological significance of such values is not fully understood and they cannot therefore stand alone as indexes of environmental quality.

2.13 Evaluation of Pollution indexes

Many workers have carried out exercises to evaluate the sensitivity of different data processing methods by applying these to their data. Sladecek (1973b) comparing the British biotic indexes with the Saprobic system commented that the classifications of Woodiwiss and Chandler allowed a much more subjective evaluation than was the case with the saprobic index. He concluded that the British efforts to avoid the saprobic system had in fact only led to new modifications of it.

Marstrand (1973) compared the results of the application of Trent, Lothian and Chandler (Biotic) score indexes to sets of data and concluded that these indexes may require adaptation to local conditions, since they were devised specifically for the areas of initial application using organisms from those areas. Although Marstrand suggested that the Trent index or an adaptation of it to suit local conditions would be the most useful method of assessing polluted waters, this was not the opinion of other workers (Balloch et al., 1976; Cook, 1976; Hellawell, 1977).

Balloch et al. (1976) comparing several methods of assessment found the Chandler Score system to be the most sensitive to changes in water quality. This system was sensitive to changes in water quality associated with mild and moderate pollution levels, but they concluded that a re-assessment of the relative position of species in Chandler's tolerance list was required.

The Chandler Score, being largely determined by the numbers of taxa in a sample, is greatly influenced by sample size. To overcome this effect Jones (1973), reported by Balloch et al., (1976) proposed that the total score should be divided by the number of taxa used, thus producing an 'average' score within the range 0 - 100. Cook (1976) found that the 'averaged' Chandler Score was the most sensitive to variables influenced by pollution compared to other systems, a view confirmed by Murphy (1978). Cook (1976), also showed that the 'average' score was least likely to be influenced by seasonal changes or sample size and thus most likely to give a continuous assessment of water quality.

Murphy (1978) examining the seasonal stability of six biotic indexes found that indexes utilising a qualitative approach, e.g. Chandler Biotic Score and 'average' Chandler Biotic score, gave a far more consistent spatial discrimination between sampling stations than indexes based on community diversity. As apposed to these previous investigations Mackay et al. (1973) and Nuttall and Purves (1974) found that the Diversity index was the most useful system for coding biological findings and the least likely to produce anomalous results. It could be applied to rivers in widely separate watersheds and therefore needed no adaptation to local conditions.

Although the Biotic systems developed to date are only capable of measuring the impact of organic pollution on water quality and the diversity index toxic pollution (Hawkes, 1978), Hellawell (1978) stated that ideally, a pollution index should take notice of:

(a) the general overall trend of responses of members of the community or populations of species to the environmental stress under consideration;

(b) the differing degrees of intensity of the individual responses of faunal components or the 'indicator value' of species;

(c) the relative abundance of species or other taxa;

(d) some measure of the overall diversity of the community; at the least some recognition of the total number of species or higher taxa.

Of the indexes reviewed only the Chandler Score and the Saprobic indexes met these requirements.

2.2 General sampling methods

One of the major problems arising from the study of stream benthos is that of obtaining adequate quantitative samples. This subject has been reviewed in detail by Macan (1958), Albrecht (1959), Cummins (1962), Schwoerbel (1966), Southwood (1966), Hynes (1970) and Hellawell (1978). Much of the difficulty stems from the fact that a sampling device which is suitable for all types of habitat has yet to be developed. Cummins (1962) reviewing the literature reveals that the number of samplers is nearly proportional to the number of investigations. Considering the heterogeneity of stream benthos and wide spectrum of possible sampling objectives, the variety of methodologies is not surprising.

Techniques for sampling the benthos of rivers are affected by the animal, the nature of the substratum, the depth, the velocity of the water and the object of the study. Some of these factors can vary greatly within a short distance of the river and many of the devices available can only be operated in a limited range of conditions. Problems of comparability can therefore arise if several sampling methods are used at different sampling sites.

Since there are so many methods currently in use for sampling benthic invertebrates in rivers, it is convenient to categorise them into active and passive samplers as quoted by Hellawell (1978). Active samplers require the active participation of the operator for collecting a sample, while passive methods rely on an automatic collection of the benthos. Passive methods include the use of drift nets, artificial sub-strata and emergence traps, whereas the majority of sampling methods belong to the active group.

2.21 Active sampling

2.211 Hand collection

This method involves the lifting of individual stones and washing the animals into a hand net (Macan, 1958). The procedure is limited to sub-strata consisting of large stones and where the current is fairly strong. The method is strictly qualitative, but Macan (1958) attempted to standardise it by collecting for exactly five or ten minutes by working slowly upstream lifting stones and holding the net in such a way that anything beneath each stone was swept into it.

2.212 Kick sampling

Hynes (1961) developed a method similar to that of Macan (1958) in which a flat-edged net was held vertically against the stream bed and the area immediately upstream was disturbed by vigorous movements of the feet to dislodge the fauna into the net. This method could be standardised by 'kicking' for a fixed time period or for a set distance upstream.

An evaluation of a kicking technique was considered by Frost et al. (1971). It was found that almost 60% of the fauna collected from a site was taken at the first kick and that three kicks yielded almost 90% of the organisms secured by ten kick samples. Further investigations by Frost et al. (1971) found that some organisms bypass the net or swim from or through it, and if the net was maintained in position for any length of time, stream-drift organisms would mask the sample.

2.213 Stationary nets

Probably the most widely used sampling device for investigations of stream benthos is the Surber-type square-foot sampler (Surber, 1937). The sampler consists of two square metal frames joined at right angles to one another along one margin of each square. A long cone-shaped net is attached to the upright frame, while the other frame has no attachments and encloses an area of substratum of one square foot. Two triangular wings of netting at the lateral margins reduce the loss of material around the sides of the sampler. In operation stones are lifted by hand within the quadrat and the gravel stirred to dislodge the fauna which then floats into the net. Modifications of the general type have been made by Leonard (1939) for work in slow water and Hess (1941) for investigations in fast waters, where a screen enclosure prevents animals from moving into or out of the square foot area.

The variability of this technique and its efficiency has been examined in detail by Needham and Usinger (1956). They collected 100 samples from a relatively uniform riffle in a Californian stream and concluded that 73 samples would be required to arrive at significant figures on total numbers at a 95% confidence level and 194 samples for total wet weight of organisms at the same level of confidence. Chutter (1972b) re-appraising Needham and Usinger's data calculated that 448 samples would be required to give a sample mean within 5% of the population mean at a 95% level of confidence and that estimates within 10% and 20% would require

112 and 28 samples respectively. Chutter and Noble (1966) in their analysis of data from ten replicate Surber samples found that their number of animals per square foot followed a lognormal distribution better than a normal distribution as seen with Needham and Usinger (1956). They concluded that for normal purposes (routine surveillance) three samples of 1 square foot area would adequately describe the numbers and composition of the fauna at a site. Hales (1962) found that variation in stream bottom fauna was so great that reasonably precise estimates were impossible without excessively numerous samples. Because of this Morgan and Egglishaw (1965) in selecting a sampling method for a bottom fauna survey, discarded the Surber method because they felt that it was too time consuming. They finally settled on a kick-sampling technique as described by Hynes (1961).

Although biologists have sampled stream invertebrates with Surber samplers for many years, very few have ever determined the total number of invertebrates that escape through the mesh and around the sides of the sampler. Kroger (1972) examining this problem with five samples found that the Surber sampler was collecting only about one-fourth of the invertebrates present in a square foot area. The standing crop was under-estimated because many of the early instars crawled directly through the sampler's fine mesh and others were carried out of the sampling area in backwash created by the tapered sleeve of the net and by the resistance of the fine mesh.

2.214 Boxes and cylinders

The general procedure involved with these samplers is the enclosure of a known area of the stream bed and the removal of the animals contained within. Wilding (1940), Hess (1941) and Macan (1958) have described such samplers based on a design described by Neill (1938). Such samplers are outlined in section 2.32. Gerking (1957) sampling the fauna associated with the aquatic plants of the littoral zone (phytomacrofauna) designed a box to cut off the plants above the substratum. It consisted of two galvanised iron shells whose function was to trap stems of macrophytes for cutting and which enabled an Ekman grab to be placed within the box so that invertebrates contained within the sediment could be removed.

2.215 Push or drag nets

The use of the Surber sampler and the various sampling cylinders is limited to shallow waters, while in deeper waters either a moving net (shovel) or a metal sampling box (dredge) can be used. Such movable nets which can either be pushed or pulled through the water are not dependent on the current in the water.

Shovels described by Allen (1940) and Macan (1958) are essentially nets mounted on a metal frame with a strong cutting edge at the front. Dredges again with a cutting edge are generally pulled through the substratum, often from boats, and consequently are more adaptable than shovels. A design by Usinger and Needham (1956) was constructed with steel lines along the mouth of the sampler and this facilitated in disturbing the substratum and prevented the entry of large stones into the conical shaped net bag. This drag only caught about 25% of the animals from the area it traversed, probably because many animals adhered to the excluded stones (Hellawell, 1978).

2.216 Grab samplers

A wide variety of devices have been developed for sampling the soft sediments (silt, sand and mud) of deep rivers or depositing zones, where dredges have proved inefficient. Such grabs (see section 2.31) are designed to remove a portion of the substratum by a biting action and consequently extracting the resident benthos. Although in very soft mud there is a danger of the grab sinking too far into the substratum thereby losing part of the contents (Ekman grab), on harder substrata there are problems concerning coarse material fouling the jaws (Petersen grab). Quantitative data obtained from grabs can prove difficult and very often a number of factors influencing the efficiency of grabs are beyond the control of the operator.

2.217 Core samplers

Although there have been a number of published methods studying the vertical distribution of animals in the soft mud of lake deposits by corers (Brown, 1956; Mackereth, 1958; Elgmork, 1962), there has been little work done in streams which have hard bottoms often covered with stones (Efford, 1960). Core samplers as described by Hellawell (1978) are tubes which are pushed into the sediment and on withdrawal retain the enclosed material. Efford (1960) developed a method in which the invertebrates were fixed in their natural position with the minimum of disturbance by using liquid oxygen as a freezing agent while the corer was being inserted into the substratum. Brinkhurst et al. (1969) using transparent plastic tubes was able to obtain reliable estimates of the standing stock of benthic invertebrates inhabiting soft sediments and of their spatial distribution in lakes and rivers.

2.218 Air lifts

The use of a cylinder for enclosing a known volume of sediment and compressed air for sampling the sediment of aquatic habitats was originally developed for marine environments (Christie and Allen, 1972).

Mackey (1972) working on the River Thames at Reading, developed an air-lift based on marine experience to sample different types of substratum. The apparatus acted as a suction corer and delivered a core to a certain height above water level. Its main advantage compared to corers and grabs was that it was light, cheap to construct and easily portable from a small boat. The air-lift could be kept in position at current velocities of up to about 0.7 $m.sec^{-1}$, but above this velocity it became difficult to use and was impracticable over 1.1 $m.sec^{-1}$.

A design by Pearson et al. (1973) allowed quantitative samples to be taken from a variety of substrata at a range of water depths and which could be operated by one person. Comparison with the

Surber sampler revealed similar efficiencies for sampling gravel substratum, while on mud substratum the air-lift proved less efficient.

2.219 Diver-operated samplers

The use of SCUBA divers to assist in the quantitative sampling of deep sluggish rivers and lakes has received increasing attention over the last decade. Divers can assist in the operation of sampling equipment as well as guiding samplers to an exact area for study.

The Finnish IBP-PM group (1969) developed a series of equipment for examining the littoral biocoenoses (phytal zones) of hard substrata in the Baltic archipelagoes. They concluded that grabs in particular could be pushed into the bottom very gently, whereas with most bottom grabs the impact on the surface of the bottom is strong enough to disturb and scatter at least a part of the biologically important uppermost layer.

2.2110 Phytomacrofauna samplers

Although a number of qualitative studies have been made of the complex biotic communities formed by aquatic plants and their associated faunas, very little quantitative information is available. Macan (1949) developed a sampler similar to an Ekman grab for sampling the fauna of plants but found that it could not be used with any degree of accuracy on plants with long filaments. A review of such sampling methods is presented by Hellawell (1978).

2.22 Passive sampling

Passive sampling methods relying on a self-collecting sampler to obtain a sample of the benthos have proved successful in routine surveillance programmes. The samplers take advantage of the normal behavioural patterns of organisms (Hellawell, 1978) in addition to studying faunal life cycles.

2.221 Artificial substrata

One method of sampling the bottom fauna of lakes and rivers is to place a bag, tray or box on to or into the substratum for a given length of time. Special attention is given to artificial substrata at a later stage (sections 2.33, 2.34) as they are becoming more important in routine sampling studies.

2.222 Drift samplers

Drift is now established as a normal feature of the lotic ecosystem (Waters, 1972) and as such, devices have been developed, usually nets, which are placed in flowing water to collect animals moving actively or drifting passively in the current. Drift nets are usually selective to specific groups of invertebrates (Insecta) because they exhibit a behavioural drifting activity, but in spite of this Besch (1966) has suggested that drift methods are useful for surveys. Detailed reviews of drift methodology are given by Elliott (1970) and Hellawell (1978).

2.223 Emergence traps

Techniques for collecting aquatic insects, either as pupae rising to the water surface, or as adults which have just emerged, are used to establish the kinds, numbers and biomass of insects leaving a unit area or volume of habitat per unit time. Although only used to a limited extent in survey work, excellent reviews are presented by Edmondson and Winberg (1971) and Hellawell (1978).

2.3 Specific sampling methods

2.31 Grab samplers

The accurate quantitative sampling of the bottom fauna of rivers too deep to wade has long been a problem and although a large number of grabs have been employed by oceanographers (Longhurst, 1959), only two grabs have been used extensively by limnologists - Ekman (1911) and Petersen (1911).

The Ekman grab consists of a square or rectangular box of sheet metal with a lower opening closed by a pair of strong metal jaws that oppose each other and are closed tightly by springs. When fully pulled apart, they leave the bottom of the box fully open. The top of the box is covered by two thin, hinged, overlapping lids which are pushed open when the grab is descending through the water but which close and are held shut by water pressure while the grab is lifted to the surface. In deep waters the grab is operated by sending a messenger down a cord which releases the spring opened jaws thereby enclosing a portion of the substratum. Such a grab is specially adapted for use in medium soft deposits. The Ekman grab on a pole is used in shallower environments with harder sediments and is operated by the swivelling action of the pole to unlock the mechanism. Modifications of the basic Ekman design are reported by Edmondson and Winberg (1971).

In relatively shallow waters samples may be taken from stony or vegetation covered bottoms by the Allan (1952) grab, whilst for stream muds, Ford and Hall (1958) designed a quantitative grab which takes a core-like sample by virtue of its closing mechanism consisting of two horizontally sliding steel plates.

The Petersen grab is a veteran piece of equipment that works in sand, gravel or clay. It consists of two hinged buckets, each forming a sector of a cylinder which is lowered to the bottom on a cable in an open position. When the lowering rope slackens, a release is actuated so that when the grab is hauled the scoops close together and take a bit of sediment semi-circular in cross section. The main disadvantage of this type of grab is its weight and the hoisting apparatus necessary for its use. This type of grab operates in such a way that there is certain uncertainty about the area and the depth sampled. Also, the jaws may be kept partly open by stones or debris, and part of the sample may be lost. This latter criticism is typical of most grabs in current use (Crisp and Gledhill, 1970). The Peterson grab has therefore not been recommended for quantitative estimation of the benthos (Edmondson and Winberg, 1971).

Birkett (1958) compared the efficiencies of the Petersen and van Veen-type grabs on the basis of animal density per unit volume of sample and found that the grabs collected only 70% or less of their expected sample volume. Flannagan (1970) comparing grabs with corers, found that grabs did not give a realistic estimate of the benthos in all substrates encountered. Further comparisons between Ekman and other grabs (Howmiller, 1971; Rinne, 1978) have shown that the former method will give reasonable estimates of number of organisms per unit area.

2.32 Cylinder samplers

The introduction of cylinder samples as an active, quantitative method for sampling the bottom fauna of streams and lake margins was probably developed as a parallel method to the square-foot net and quadrat sampler of Surber (1937). Typically such samplers are open ended cylinders which can be pushed on to or into the substratum to isolate a known area. Although certain samplers were designed for specific objectives, they all had a common feature in only being suitable for use in moderately shallow water, generally less than one meter depth.

The earliest published record of cylinder sampling was documented by Neill (1938) who worked on the lower reaches of the River Don, Aberdeenshire. The cylinder, constructed of stout metal was 82 cm deep and 100 cm^2 in cross-section area, with a broad projecting outer flange 5 cm from its lower end, sharpened to provide a cutting edge. Strong handles at either side permitted the cylinder to be lifted into position on the stream bed and worked down into the substratum with a slight rotary movement, to the level of the flange (5 cm). An opening 15 cm square, guarded by a coarse grid of wire and equipped with an upward sliding door, was cut in the side of the cylinder immediately above the flange. A second opening was cut directly opposite the first to which a bag-net of bolting silk was attached. The sliding doors of both these openings were worked from the top of the apparatus. In operation the cylinder was rotated into the stream bed to a depth of 5 cm and the substratum stirred to dislodge invertebrates. Half a minute was then allowed for the coarse sand and gravel to settle before the sliding doors were opened simultaneously for three minutes, admitting the current which then carried the invertebrates into the net bag.

A more complex method of recovering dislodged fauna by cylinder sampling was described by Wilding (1940). The apparatus consisted of two separate brass cylinders, one fitting within the other, designed to enclose an area of 929 cm^2. The outer open-ended cylinder with 13 saw teeth bolted on to the bottom, was rotated into the substratum and large stones down to a depth of 8 cm were washed and removed. The inner cylinder made of perforated sheet brass (9 meshes cm^{-1}), with its valve bottom, was then plunged into the water contained in the outer cylinder. The valve was closed when the cylinder touched the substratum and thereby enclosed the water and invertebrates, which could then be removed.

Hess (1941) while studying the benthos of streams and lakes near New York, experienced difficulty in using the square-foot sampler of Surber, and to overcome these difficulties a cylindrical cage was devised. The front half of the cylinder was covered with coarse gauze (aperture 4.2 mm), while the back was covered with heavy canvas, thus enclosing an area of 929 cm^2. The circular top and bottom portions of the frame were made of strap iron, to which handles were attached. A collecting net (9 meshes cm^{-1}) was attached to the downstream side of the cylinder thus taking advantage of the flow to collect the sample.

A cylinder similar to that of Hess (1941) was designed by Waters and Knapp (1961). The frame was covered with Nitex (mesh size 471 μm) and enclosed an area of 929 cm^2. The netting satisfactorily prevented drifting organisms from entering the sampler and the collecting net of the same material was extended on the downstream side. The collecting bag was attached to the cylinder by a zipper similar to the zipper bag employed on a drag-type sampler introduced by Usinger and Needham (1956).

Rabeni and Gibbs (1978) modified the Hess cylinder for use in deep rivers which was operated by divers on the river bed. The sampler was adapted so that it could be used at any depth by adding a sleeve to the top to prevent the escape of bottom materials when the top was submerged. Further modifications and uses of the Hess type sampler have been reported elsewhere (McDaniel, 1974; Mason, 1976; Poole and Steward, 1976; Minshall and Minshall, 1977).

In the littoral region of Lake Bala, Dunn (1961) examined the bottom fauna using an open-ended galvanised iron cylinder which sampled an area of 828 cm^2. Teeth 5 cm long at the base of the cylinder facilitated the rotary action into the substratum and generally this was sampled to a depth of 6 - 7 cm.

As an aid to sampling mud and soft sediments, a 'bucket' circular quadrat was devised by Whitley (1962) for use in the depositing regions of streams and rivers. The cylinder could be sunk to a depth of 12 - 17 cm, enclosing an area of 500 cm^2 and could be modified for use in deeper waters. Garnett and Hunt (1965) while sampling invertebrates on macrophytes constructed a cylinder or tube sampler which only operated efficiently when the substratum was sandy or muddy. The sampler with a cutting blade at the base sampled an area of 0.033 m^2 and was effective amongst dense upright vegetation provided that the substratum was sufficiently flat to accommodate the base of the instrument.

As a development of cylinder sampling efficiency, Hynes (1971) introduced a foam plastic skirt, which provided additional sealing between the cylinder and the substratum and theoretically allowed sampling of rough bottoms, by filling in irregularities of the surface. Hughes (1975) comparing the efficiency of Surber sampling with that of cylinder sampling (modified Neill cylinder, area 0.05 m^2) concluded that close correlation was attained between the two methods and that previously published anxieties concerning invertebrate loss from the Surber sampler may not be as serious as once thought.

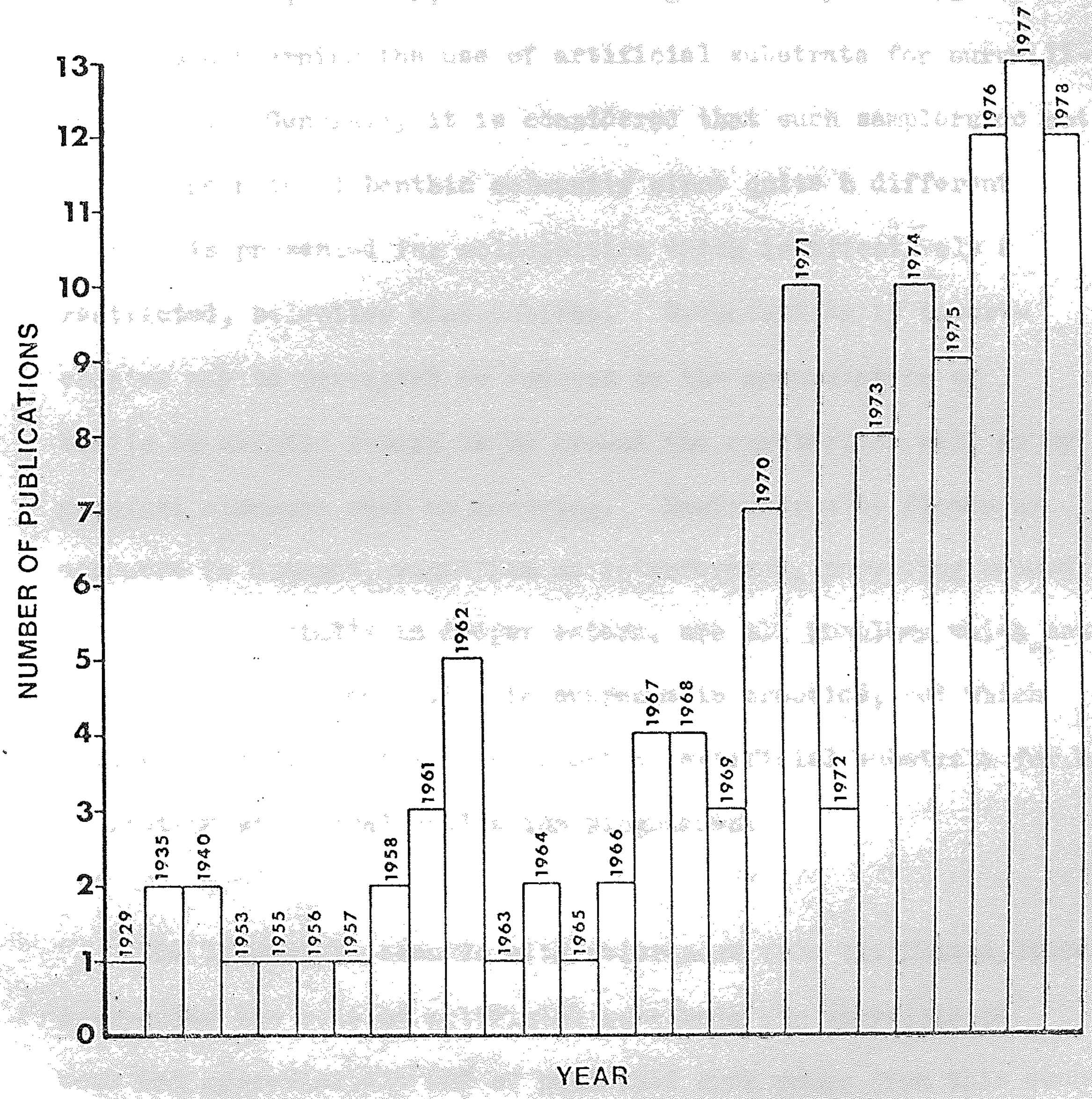
2.33 Artificial substratum samplers

The use of artificial substrata for collecting benthic macroinvertebrates has received considerable attention over the last ten years (Fig. 2.1), mainly due to the variation in results obtained by the use of conventional samplers (Hellawell, 1978). Generally such samplers can provide useful data if interpreted with care for assessing the impact of water quality on aquatic life (Arthur and Horning, 1969; Mason, et al., 1970; Pratt and Coler, 1976).

Quantitative conventional studies of macrobenthos in lentic and lotic waters are often difficult because of the tremendous variations in substratum, with corresponding contagious distribution patterns of species occurrence. The development of artificial substrata undoubtedly grew out of the realisation that all situations are not readily adaptable to these methods and that attempts had to be made to refine and quantify techniques of sampling.

The philosophy behind using artificial substrata to evaluate community structure is to provide a standardised habitat which can become colonised by the biota, thereby eliminating the heterogeneity of the natural substratum and the consequent heterogeneous distribution of benthic species. Theoretically, artificial substrata methodology provides a constant area or volume sampled; results obtained should be reproducible and comparable providing an increased procedural efficiency. The quantitative evaluation of communities in rivers where the study of natural substrata is

FIG.2.1 FREQUENCY OF AMERICAN AND WESTERN EUROPEAN PUBLICATIONS ON ARTIFICIAL SUBSTRATA FOR COLLECTING BENTHIC MACROINVERTEBRATES.



difficult or unproductive now becomes possible due to the relative ease in placing and retrieving the substrata.

There are, however, several ecological and practical problems concerning the use of artificial substrata for surveillance work. Generally it is considered that such samplers do not sample the natural benthic community since quite a different biotope is presented for colonisation which is effectively a restricted, selective microhabitat. Colonisation by invertebrates may be prevented or reduced by the accumulation of biotic or abiotic debris in or around the sampler, as well as by physical stresses such as scouring. Destruction by floods or exposure in drought, vandalism or interference, providing secure anchorage especially in deeper waters, are all problems which have proved virtually impossible to overcome in practice, but which have not restricted the development of artificial substrata for use in routine biological evaluation programmes.

The literature abounds with references from the United States, concerning the role of artificial substrata for surveillance work and approximately 60% of published work comes from this country compared to the United Kingdom and Western Europe. It appears that artificial substrata have been employed mainly in rivers and streams, and to a lesser extent in lakes and the marine environment.

Reviews using artificial substrata to study the interstitial fauna of rivers (Williams and Hynes, 1974) or general invertebrate populations (Southwood, 1966; Zillich, 1967; Khalaf, 1975; Khalaf and Tachet, 1978) are well documented in the past, although Hynes (1970) comments that such methods are unlikely to provide any measure of the population in the river bed, but could provide a source of specimens. However, in terms of assessing water quality several reviews have advocated their use (Cairns and Dickson, 1971; Beak et al., 1973; Mason, 1977b) with certain environmental monitoring manuals recommending such techniques (Slack et al., 1973; Weber, 1973).

The application, design and operation of artificial substrata to objectively study rivers in terms of surveillance has been critically reviewed by Hellawell (1978). In general practice, one category of sampler provides a plane surface of exposure, while six further groups have been designed to provide interstices for colonisation.

Terminology of artificial substrata is often confusing in the literature, because many of the techniques developed are usually colonisation samplers made up of the natural substrata and placed either on a tray or embedded in containers. For the purpose of this review such terminology remains unchanged.

2.331 Plane surface samplers

2.3311 General Invertebrates

Historically Shelford and Eddy (1929) designed the first type of artificial substratum for studying communities in swift water. Heavy meter-square pieces of concrete or rocks were fixed on the stream bed and allowed to become colonised by drift organisms. Other examples using concrete blocks (Britt, 1955; Cianficconi and Riatti, 1957) bricks (Elvins, 1962; Kovalak, 1976; 1978) and tiles (Albrecht, 1953) tend to be selective in the species colonising them. Mundie (1956) described an asbestos-cement plate which was used in sampling the sloping sides of reservoirs and which had a special cover to prevent loss of organisms during recovery. Beak et al (1973) modified Mundie's design by using a metal plate tray and found the community developing on the sampler was adequate for statistical calculations. Other selective samplers used have been plastic squares (Macan and Kitching, 1976) and polyethylene plates (Besch et al, 1967; Besch and Hofmann, 1968) and these have been designed to operate just below the water surface.

2.3312 Simuliidae

Quantitative studies on black-fly larvae (Simulium spp.) to control population densities for economic and medical reasons has received considerable attention over the last few years. Since sampling of Simulium populations attached to natural substrata has been difficult to standardise, there has been a growing tendency to place standardised artificial substratum units (SASU's) in rivers (Disney, 1972; Hall and Edwards, 1978). Such methods have included metal and plastic cones (Wolfe and Peterson, 1958) polythene sheets (Williams and Obeng, 1962; Pegel and Ruhm, 1976; Boobar and Granett, 1978) tiles (Lewis and Bennett, 1974; Hall and Edwards, 1978) and strings of leaves (Disney, 1972). Comprehensive studies of SASU methods have been reviewed by Disney (1972) and Lewis and Bennett (1974).

2.332 Embedded samplers

The use of trays embedded in the substratum and filled with 'sterilised' stones was originally introduced by Moon (1935a; b; 1940) while studying the littoral fauna of Lake Windermere. Moon's sampling devices were canvas sacking and string mesh attached to an iron frame to form a shallow bag, and filled with indigenous stones. Numerous authors have developed the tray method of sampling for use in running waters (Hughes, 1975), while limited work has continued in lakes (Dunn, 1961) and estuaries (Goddard et al., 1975).

In most studies, trays of stone have been utilised to examine the distributional ecology of benthic invertebrates (Okland, 1962; Lillehammer, 1966; Ulfstrand, 1968; Crisp and Gledhill, 1970; Allan, 1975; Mason, 1976; Sheldon, 1977;) or Drift (Townsend and Hildrew, 1976). However, some authors have exposed bricks with plant detritus (Egglishaw, 1964), gravel with artificial weed (Pearson and Jones, 1975a; b) or plant detritus (Lapchin, 1977) for colonisation by the benthos.

In an attempt to avoid the loss of animals from the tray when it is raised to the surface, Ulfstrand et al. (1974) placed a nylon net bag around the rim of the tray which was unrolled during recovery. This method was later adapted by Nilsson and Sjostrom (1977) in their study of Gammarus pulex size frequency distribution.

As a tool for examining the recolonisation mechanisms of denuded stream bottom areas by drift, Waters (1964), Williams and Hynes (1976) and Williams (1977) developed a frame trap filled with stones which eliminated contamination by other sources such as oviposition, vertical migration and upstream migration. Recently Armitage (1976; 1977) used trays of stone and synthetic carpet material to monitor changes by drift in the River Tees below Cow Green Reservoir after impoundment.

In a study of the factors affecting the microdistribution of stream benthos, Minshall and Minshall (1977) and Rabeni and Minshall (1977) used substratum-filled trays to determine the substratum preference of different species. The vertical distribution of macrobenthos within the substratum constitutes an important part of the stream ecosystem and the majority of conventional sampling techniques suffer from the shortcoming that they sample only a part of the zone occupied by the benthos. Hynes (1970) has pointed out that if samplers do not penetrate very deeply into the substratum then they will not collect much of the hyporheal. Such findings have led to the development of elaborate perforated metal cylinders (Coleman and Hynes, 1970; Radford and Hartland-Rowe, 1971; Poole and Stewart, 1976) and wooden boxes (Ford, 1962; Bishop, 1973) filled with indigenous stones and embedded deep into the substratum, often to a depth of 30 cm.

2.333 Multiple plate samplers

This type of sampler designed by Hester and Dendy (1962) was constructed of 3 mm tempered hardboard ('Masonite') and consisted of eight square plates (8 cm area) separated by seven spacers (2.5 cm area) of the same material. These plates were assembled on a long bolt and held in place by two nuts. The sampler held in suspension in midwater by a vertical wire, exposed a total surface area of 0.093 m^2 for colonisation and could be dismantled to remove attached organisms.

Dendy (1963) modified the original sampler by using nine plates, while Arthur and Horning (1969) and Mathers and Martin (1969) used six plates. Fullner (1971) added further plates with different spacings to provide several microhabitats and a total of 14 plates were used in each sampler.

The use of synthetic webbing incorporated into a multiple plate sampler was developed by McDaniel (1974), 'Masonite' plates alternating with 'conservation webbing' on a bolt to form a sampler with 21 plates. Theoretically, the 'Masonite' plates provide substrata for colonisation by the more sessile organisms and the webbing material provides habitat for the more mobile organisms.

Because the alignment of square plates is difficult, Mason et al. (1973) constructed a sampler consisting of 24 discs, separated by disc spacers, exposing an area of 0.27 m^2. Parsons and Tatum (1974) reduced this number to ten plates, while Cover and Harrel (1978) used eight plates which were smooth on one side and rough (cross-hatched) on the other; the smooth and rough sides alternating in the sampler.

2.334 Basket samplers

The commonest type of artificial substratum employed in surveillance work has been the wire-mesh baskets filled with specific media. The earliest methods were baskets filled with graded rocks (Wene and Wickliff, 1940), later adapted by Standford and Reed (1974), or brush boxes filled with sticks, grass, stones and other objects which could provide a stable environment for colonisation (Scott, 1958a; Dickson, et al, 1971). Typically such samplers are lowered on to the river bed and held in position by stakes. Recent studies have included using bags of clinker (Woodiwiss, 1978), or barbecue baskets filled with crushed limestone chips (Crowe, 1970; 1971; Bergersen and Galat, 1975; Roux, et al. 1976) rubble (Pratt and Coler, 1976), rocks (Dickson, et al., 1970; Bournaud et al., 1978) rocks and leaf detritus (Crossmanand

Cairns, 1974), granite (Rosenberg and Wiens, 1976), basalt (Minshall and Andrews, 1973), porcelain balls (Roby et al., 1978), 'conservation webbing' (Simmons and Winfield, 1971; Hocutt, 1975; Stauffer et al., 1976) and coniferous tree bark (Bergersen and Galat, 1975). In deep rivers, Gale and Thompson (1974; 1975) and Rabeni and Gibbs (1978) used SCUBA divers to collect baskets of stone and cement spheres from the bottom. In lakes Laville (1974) used baskets of stone to study the benthos, while Voshell and Simmons (1977) compared the efficiency of baskets of leaves, limestone and 'conservation webbing' to sample the fauna of reservoirs. Kathman (1978) considered, however, that geometrically different structures placed inside a polyurethane-coated wire basket would be the most suitable method for sampling the benthos.

Some workers have suspended barbecue baskets from floats in mid-water in order to sample the drifting fauna and not necessarily the adjacent benthos. In rivers some workers have used limestone chips (Mason et al., 1967, a, b; Anderson and Mason, 1968; Fullner, 1971; Mason et al., 1973), native rock (Mason et al., 1970) porcelain (Mason et al., 1973) concrete, styrofoam and wood spheres (Jacobi, 1971) and 'conservation webbing' or concrete cones (Benfield et al., 1974). In lakes and reservoirs suspended samplers have included cages filled with rocks (Henson, 1965), baskets filled with limestone chips (Kreis and Smith, 1970; Kreis et al., 1971; Prins and Black, 1971) or baskets filled with 'conservation webbing' (Prins and Black, 1971).

In recognising the problems associated with retrieving artificial substrata, Hilsenhoff (1969) devised an elaborate basket sampler with a special recovery net to reduce the loss rate of escaping invertebrates. The apparatus was constructed of galvanised iron cylinders with inner layers of expanded metal mesh set on a peg through a concrete slab. In operation the sampler was filled with rocks and dropped on to the river bed. However, the size and weight of the apparatus restricted its use in routine monitoring work. Bull (1968) overcame these problems by using a collapsible wire basket holding gravel and surrounded by a fine-mesh bag. A rim around the top of the basket kept gravel from overflowing and provided attachment for lifting ropes. The basket collapsed when lowered on to the bottom, but assumed its original shape when raised and so prevented loss or organisms.

2.335 Synthetic mesh samplers

'Conservation webbing' (3M Corporation ≠ 200) is a black, non-toxic thermoplastic mesh of nonwoven fibres pressed into sheets 1 cm. thick and used as a packing material in the United States. Simmons and Winfield (1971) tested the feasibility of using this material to evaluate how well the community collecting on this substratum mirrored the natural stream community. Prins and Black (1971) used this material placed inside barbecue baskets (Anderson and Mason, 1968) to collect the benthic fauna of a reservoir at depths of 5 and 14 m. Some workers have used the webbing to compare its efficiency with other artificial substrata for collecting invertebrates (Benfield _et al._, 1974; Crossman and Cairns, 1974), while Dickson and Cairns (1972) used it in sequential comparison studies.

2.336 Artificial weed samplers

Glime and Clemons (1971) compared two types of artificial mosses (string and plastic) with _Fontinalis_ spp. to study the species diversity of stream insects. Macan and Kitching (1972) and Macan (1976) constructed artificial _Littorella_ made from 3-strand polypropylene and artificial Carex made from lengths of whole rope on a rigid lattice of polyester mesh, to study the bottom fauna of Hodsons Tarn. Macan (1977a; b) continued using artificial Littorella to study the fauna in a moorland fishpond, while Pearson and Jones (1975a) constructed artificial weeds based on those described by Macan and Kitching (1972) to study the benthic communities of the River Hull, East Yorkshire.

2.337 Log samplers

The establishment of invertebrate communities on log substrata was investigated by Nilsen and Larimore (1973). The logs were used to study the periodicity of stream drift in riffles, pools and slowly moving shallow waters. The barkless logs used were approximately 6 cm in diameter and 23 cm in length and were selected from the study area brought down by the spring floods. The artificial logs used collected a wide selection of the fauna, but the naturally-occurring logs had greater numbers of some groups.

2.34 Efficiency of artificial substrata

In order to evaluate the different types of artificial substrata tested, it is necessary to review the requirements which an artificial substratum should meet.

First, a distinction must be made between the use of artificial substrata to measure specific biological parameters and their use to assess water pollution or water quality. For the former, it is often essential that natural bottom conditions be duplicated as closely as possible. To assess water quality, exact duplication of natural conditions is of less importance than ability to obtain replicate samples at different times and at different locations.

Secondly, the artificial substratum should represent a sufficiently complex habitat to ensure that the biological community to be sampled will be reasonably well represented. Again, it is less essential to attempt to duplicate the native habitat than it is to provide a habitat demonstrably suitable for colonisation by both pollution tolerant and pollution sensitive species.

There are two main criticisms of the literature concerning the use of artificial substrata. First, the failure to include numerical analysis of biological data and secondly, the failure to determine the reliability of the samplers. It is evident that a thorough evaluation of the performance of artificial substrata has not been made. It is essential to ascertain the time necessary for a stable community of macro-invertebrates to develop on a sampler and also the numbers of samplers needed to give an acceptable level of confidence.

2.341 Colonisation times

Optimum exposure periods for different types of artificial substrata as quoted in the literature are given in Table 2.1. From this review it is evident that a period of between four and six weeks is needed before any data can be evaluated objectively.

Some workers have been concerned with the dynamics of the colonisation process (Waters, 1964; Ulfstrand, 1968; Nilsen and Larimore, 1973; Khalaf and Tachet, 1977), others have described colonisation by individuals of particular taxa (Ulfstrand, 1968; Simmons and Winfield, 1971; Nilsen and Larimore, 1973; Pearson and Jones, 1975a; Roby et al., 1978), while certain authors have measured rates of accumulation of species (Cairns et al., 1969; Dickson and Cairns, 1972; Cover and Harrel, 1978). Ulfstrand et al. (1974) considered both the latter two aspects.

Sheldon (1977) investigating colonisation over short time periods and statistical techniques for fitting colonisation curves concluded that colonisation rates vary between taxa and depend significantly on location in the study area.

In recent studies to determine the sequence of colonisation and the time necessary for the community to reach asymptotic diversity, equilibrium models have been developed based on work by MacArthur and Wilson (1963). They proposed a model for the colonisation of island faunas and the principles necessary for the formation of stable communities. It was suggested that a new habitat would have a high rate of colonisation by new species combined with a low rate of extinction for those that became established. Later, if colonisation proceeded at a fairly constant rate, the continued influx of new species should result in a gradually increasing rate of extinction for established species and a decreasing rate of colonisation until an equilibrium is reached. This occurred when the rate of colonisation equalled the rate of extinction.

Cairns et al. (1969) tested this model by studying the colonisation of polyurethane substrata by freshwater protozoans and found that the formation and composition of protozoan communities on artificial substrata had some interactions comparable to those proposed by MacArthur and Wilson.

Dickson and Cairns (1972) again using this model, found that their data fitted the model reasonably well, but did not produce a permanent stability. However, Williams and Hynes (1977) concluded that the colonisation period in a man-made stream was in agreement with the MacArthur-Wilson model.

Table 2.1 Recommendations of colonisation rates and replication studies quoted in the literature

METHOD OF SAMPLING	AUTHOR(S)/YEAR	IMMERSION PERIODS TESTED d - days w - weeks	OPTIMUM EXPOSURE PERIOD RECOMMENDED OR FOUND	NUMBER REPLICATE SAMPLERS RECOMMENDED OR USED
PLANE-SURFACE	Mundie (1956)	12w	12w	2
	Elvins (1962)	2w	2w	1
	Kovalak (1976)	4w	4w	4
	Kovalak (1978)	4w		
EMBEDDED	Moon (1935a)	2, 4, 12w	4w	
	Dunn (1961)	4 - 5w		
	Ford (1962)	5 - 6w		
	Egglishaw (1964)	16, 22d		8
	Waters (1964)	1 - 13d, 1 - 30d	4 - 10d	
	Lillehammer (1966)	4w		
	Ulfstrand (1968)	2w	2w	
	Coleman and Hynes (1970)	1 - 7d, 14, 28d	> 28d	
	Crisp and Gledhill (1970)	4, 8, 12w	12w	20
	Radford and Hartland-Rowe (1971)	4w	4w	10
	Bishop (1973)	8, 16, 24w		8
	Ulfstrand *et al.* (1974)	2, 4, 8, 15, 32d	32d	4
	Allan (1975)	1, 2, 5, 10d	10d	3
	Pearson and Jones (1975a)	4, 7, 14, 20, 40, 90, 200d	10 - 15d	5
	Hughes (1975)	5w		> 4
	Armitage (1976)	4w	4w	1
	Mason (1976)	30d	30d	20
	Poole and Stewart (1976)	14, 28, 40, 47, 56w		2
	Townsend and Hildrew (1976)	3, 6, 9, 12d	3d	1
	Williams and Hynes (1976)	4w		
	Armitage (1977)	4w	4w	3
	Lapchin (1977)	8, 16, 22, 26d	2w	3

Table 2.1 (cotd)

METHOD OF SAMPLING	AUTHOR(S)/YEAR	IMMERSION PERIODS TESTED d - days w - weeks	OPTIMUM EXPOSURE PERIOD RECOMMENDED OR FOUND	NUMBER REPLICATE SAMPLERS RECOMMENDED OR USED
EMBEDDED (cotd)	Nilsson and Sjostrom (1977)	2, 4, 8, 17, 32d	17d	4
	Rabeni and Minshall (1977)	30d		
	Sheldon (1977)	1, 2, 3, 4, 7, 9, 14d		4
	Williams (1977)	4w		
	Minshall and Minshall (1977)	4, 5, 10w		2
MULTIPLE PLATE	Hester and Dendy (1962)	1, 2, 3w		
	Dendy (1963)	4, 8w		
	Fullner (1971)	8w		1
	Mason *et al.* (1973)	6w		3 - 4
	McDaniel (1974)	6, 13, 21d		2
	Cover and Harrel (1978)	1 - 16w	6w	3
BASKET	Wene and Wickliff (1940)	4w	4 - 6w	1
	Henson (1965)	30, 40d		
	Mason *et al.* (1967)	4, 6w	6w	
	Zillich (1967)		25d	
	Anderson and Mason (1968)	6w		
	Hilsenhoff (1969)	6, 11, 19w		1
	Crowe (1970)		3 - 4w	
	Mason *et al.* (1970)	4w		
	Dickson *et al.* (1971)	21d		5
	Fullner (1971)	8w		1
	Jacobi (1971)	6w		
	Kreis *et al.* (1971)	8, 16, 24w		2

Table 2.1 (cotd)

METHOD OF SAMPLING	AUTHOR(S)/YEAR	IMMERSION PERIODS TESTED d - days w - weeks	OPTIMUM EXPOSURE PERIOD RECOMMENDED OR FOUND	NUMBER REPLICATE SAMPLERS RECOMMENDED OR USED
BASKET (cotd)	Prins and Black (1971)	4w		5 - 6
	Simmons and Winfield (1971)	1 - 10w	6w	3
	Mason *et al.* (1973)	4, 6, 8w	8w	3
	Minshall and Andrews (1973)	8w		1
	Benfield *et al.* (1974)	30d		3
	Crossman and Cairns (1974)	32d		1
	Stanford and Reed (1974)	8w		
	Bergersen and Galat (1975)	30d		4
	Pratt and Coler (1976)	6w	6w	2
	Rosenberg and Wiens (1976)	28, 56, 132, 279, 335d		3
	Roux *et al.* (1976)	3, 4w		
	Stauffer *et al.* (1976)	5, 10, 15, 20, 25, 30d	22d	5
	Voshell and Simmons (1977)	4, 6w	6w	3
	Bournaud *et al.* (1978)	1, 3, 7, 14, 21d	3d	2
	Rabeni and Gibbs (1978)	4w		6
	Roby *et al.* (1978)	1 - 10w	2 - 4w	2
	Woodiwiss (1978)	2, 4, 6w	4 - 6w	2 - 3
SYNTHETIC WEBBING	Prins and Black (1971)	4w		5 - 6
	Dickson and Cairns (1972)	1 - 9w	7w	3
	Benfield *et al.* (1974)	30d		3
	Crossman and Cairns (1974)	32d		1
WEED	Glime and Clemons (1971)	2, 4, 6, 8, 10, 12w		1
LOG	Nilsen and Larimore (1973)	1 - 6w	> 6w	5

2.342 Replication

Needham and Usinger (1956), Chutter (1972b) and other workers have stressed the large numbers of samples required to estimate numbers of invertebrates within 10% of the mean with 95% confidence (Southwood, 1966). However, only a small number of samples will collect most of the commoner species (Hynes, 1970). It is apparent that very little information is available in the literature concerning the variability found in replication studies using artificial substrata.

The numbers of substrata employed depends largely on the purpose of the survey and on the number of man-hours that can be devoted to processing the collections. In pollution surveys Cairns and Dickson (1971) suggested that not less than three samplers were needed to describe the bottom fauna community in most cases, while Pratt and Coler (1976) considered a minimum of two was sufficient, with more permitting a statistical treatment of the data. Rosenberg and Wiens (1976) found no significant difference in numbers of macrobenthos between artificial substrata in a set of three.

Dickson *et al.* (1971) found that in unpolluted waters, four basket samplers would be required to be 95% confident that the mean number of taxa collected was within 25% of the true mean. They concluded that to obtain the same level of statistical accuracy, in comparing artificial substrata for pollution surveys, a greater number of replicates may be required at clean-water stations than at polluted-water stations.

Prins and Black (1971) were able to show that for quantitative repeatability as many as 15 limestone baskets would be required to achieve a coefficient of variability in the range of 10 - 11% (Dickson et al., 1970) while six samplers would give a correlation coefficient greater than 30 - 40%. Radford and Hartland-Rowe (1971) also arrived at a workable sampler number (10 or less) by accepting an error within 20% of the true mean and rejecting representatives of rare species with any unusual attributes.

Recently Mason et al. (1973) having comprehensively studied the factors affecting the performance of basket and multiplate samplers expected three replicate baskets would provide an estimate of the true mean number of macroinvertebrates with ± 20% of the sample mean and contain 71% of the taxa in 10 replicates. However, estimates of sampling precision based on replicate series may vary considerably depending on time, place, number of replicates and natural history of the organisms.

Table 2.1 reviews the literature objectively concerning the number of replicate samplers recommended by different authors and it is apparent that a minimum of three samples should be used in surveillance studies.

2.343 Comparison of different artificial substrata

Several authors have attempted to compare the efficiency of different types of artificial substrata with one another to establish their performance in assessing water quality. Crowe (1971) compared barbecue baskets filled with stones, with Beak (1973) trays and found that in 61% of samples, both samplers collected the same groups of animals with the same percentage composition, but the baskets collected a significantly higher population density. Fullner (1971) reported that samples from modified hardboard multiplates contained approximately the same number of individuals and taxa as baskets, but were more convenient to use in surveillance studies. On similar lines

Prins and Black (1971) compared limestone baskets to synthetic webbing baskets and found that the latter collected statistically greater number of organisms, particularly in late summer when the reservoir was stratified.

Recently Mason et al. (1973) observed baskets filled with porcelain spheres collected samples comparable in total abundance and diversity to baskets filled with limestone clips, but there were differences in the faunal composition. Hardboard multiplates collected significantly greater numbers of individuals than porcelain multiples, but differences between samples from the same substrata were not significant and the taxa recovered were similar. Basket samplers collected a greater abundance and diversity of organisms than standard hardboard multiplates but when the latter were enlarged the performances were similar. Bergersen and Galat (1975) compared bark pieces to lime-stone rocks placed in barbecue baskets and found no significant difference between the two substrata as to numbers of individuals or genera.

2.344 Comparison of artificial substrata with conventional sampling methods

One aspect of sampling which is of particular importance is that of inter-sample variability of the various sampling methods used in biological surveillance studies. Although artificial substrata have been widely employed in these procedures little attempt has been directed towards comparing them with more conventional samplers of the natural substratum. In evaluating the performance of basket samples compared to Petersen grab samples, Anderson and Mason (1968) concluded that the basket was generally more efficient in collecting aquatic insects but less so for annelids and molluscs. These findings were also reported by Voshell et al. (1977) who found significantly more individuals and taxa in three types of basket samplers compared to Ponar grab samples.

The embedded pot sampler of Coleman and Hynes (1970) generally collected over a period of 10 months, larger numbers of Chironomidae and other macroinvertebrates compared to kick samples representing a sampling efficiency at least twice that of kick sampling. Benfield et al. (1974) comparing baskets of synthetic webbing and cone-shaped concrete blocks to kick sampling, concluded that neither artificial substrata produced results comparable to net collecting. It would seem that the usefulness of any type of artificial substratum doubtless is dependent upon the physical and biological characteristics of the aquatic habitat in which it is to be placed.

On several occasions the Surber sampler has been used to compare its effectiveness in collecting macroinvertebrates with artificial substrata. Jacobi (1971) attempted to estimate the standing biomass and production of the benthos in lotic environments using baskets filled with rocks and the Surber sampler. It was found that 11 out of 16 taxa were common to both samplers and consequently concluded that baskets could be used for production estimates.

Several authors have demonstrated that artificial substrata collect most of the species detected by means of a Surber sampler (Radford and Hartland-Rowe, 1971; Roby et al., 1978) while Stanford and Reed (1974) found basket samplers collected a higher number of the most commonly occurring benthic species than did the Surber sampler in riffle areas, but in slow-moving pools both methods were equally efficient.

Simmons and Winfield (1971) found conservation webbing used in a stream collected only 63% of the species obtained by net or Surber sampling but found that certain Trichoptera and Ephemeroptera were only taken by the artificial substratum. Crossman and Cairns (1974) found that bottom basket samplers were more reliable than floating samplers as indicated by comparison of diversity indexes between the samplers and conventional sampling methods to collect benthic species. The floating substrata tended to be selectively colonised by Coleoptera, Ephemeroptera and Trichoptera and suggested that this selectivity may be used to advantage for life history studies of the colonising insects.

In their study of deep rivers, Rabeni and Gibbs (1978) modified a Hess sampler to estimate standing crop and compared this with rock-filled basket samplers. It was found that the baskets, although biased for some invertebrates, consistenly collected more individuals and taxa. Both methods showed high efficiency as measured by the coefficient of variation, which generally decreased with increasing pollution.

Hughes (1975) comparing the results obtained by four methods of sampling (Surber, cylinder, electric shock apparatus and trays) showed that artificial substrata generally captured the greatest number of individuals and species. Although the faunas of the trays were least variable compared to the other sampling methods, he did not consider that they reflected the normal community structure present in the natural substratum.

2.345 Retrieval

The possibility of loss of organisms during retrieval of artificial substrata has been reported by Zillich (1967), who described the loss of large numbers of organisms when recovering unprotected samplers. To prevent loss of organisms Wene and Wickliffe (1940) placed a hand net downstream of the sampler during retrieval, while Bull (1968), Hilsenhoff (1969), Gale and Thompson (1974) and Stanford and Reed (1974) incorporated a protection net into the design of the artificial substratum.

Mason et al. (1973) investigated the possibility of loss by enclosing samplers while they were removed from the water on two separate occasions. In 1967, a bucket with 20 mesh screened openings was placed around baskets to capture dislodged organisms, and it was found that only one mayfly was dislodged from four baskets. In 1970 50 mesh-size nets were placed around 10 multiple plate samplers. During recovery, three plates samplers contained no organisms, while seven lost 1 - 7% of the total, with an average of 2.5% for ten checks. The loss rate was found to be insignificant.

2.4 Natural factors influencing the distribution of benthic invertebrates

2.41 Current velocity

It has been shown (Ambuhl, 1959) that many invertebrates have an inherent need for current, either because they rely on it for feeding purposes, or because their respiratory requirements demand it. These are the typical rheostenic species and many workers have found that particular species are confined to fairly definite ranges of current speed as measured in the field. Ambuhl (1959) demonstrated using a flashing light and acetyl-cellulose powder that there was a considerable reduction of current velocity in the few millimeters above the surface of an object and a stagnant piece of water downstream of an object.

He studied the distribution of invertebrates in relation to the speed of the current and recorded that certain species appeared to select certain speeds in which to live. Plotting flow (measured just above the substratum) against percentage catch, the following maximum abundance levels were observed:

Simulium 80 cm sec^{-1},
Hydropsyche angustipennis 60 cm sec^{-1},
Baetis vernus 40 cm sec^{-1},
Ephemerella ignita between 10 and 25 cm sec^{-1}, and
Gammarus pulex 15 cm sec^{-1}.

Scott (1958) plotted the number of larvae per square meter against the surface velocity, the peaks of which gave Hydropsyche fulvipes maximal numbers between 40 and 50 cm sec^{-1}, Rhyacophila dorsalis between 80 and 90 cm sec^{-1} and Stenophylax stallatus between 0 and 10 cm sec^{-1}. Chutter (1969) similarly showed that the numbers of individuals of Cheumatopsyche thomasetti and C.afra did not significantly increase between 20 and 50 cm sec^{-1}, but that densities increased sharply at speeds above 50 cm sec^{-1}.

Dorier and Vaillant (1954) tested experimentally the ability of various animals to withstand various current speeds. They measured the maximum and minimum speeds in which they found certain species in nature, followed by measurements of the same species to maximum speed at which they would ascend and the maximum, catastrophic speed at which they were washed away. They concluded generally that the most tolerant animals in this respect were those which preferred torrential conditions in nature and the least tolerant were those which preferred slow water in nature. This data is summarised in Table 2.2.

Table 2.2 Reaction of different species to current (Selected from Dorier and Vaillant, 1954)

	Natural ($cm\ sec^{-1}$)	Experimental ($cm\ sec^{-1}$)	
	Max. withstood in nature	Max. against which the species will ascend	Speed at which washed away
Planaria alpina	14	140	143
Gammarus pulex	40	44	99
Chloroperla sp.	24	177	240
Rhithrogena semicolorata		125	182
Heptagenia lateralis	28	140	188
Baetis rhodani	30	102	177
Rhyacophila sp.	125	100	200
Agrion sp.	10	54	77
Simulium ornatum	114	117	240
Ancylus fluviatilis	24	109	240

Such flow preferences may be related to oxygen requirements (Philipson, 1954), feeding behaviour (Harrod, 1965) and for certain trichopteran larvae, to net-spinning activity. Philipson (1954) for example suggested that the distribution of Hydropsyche siltalai in faster flows was related to net-spinning, as larvae readily spun nets at 30 cm sec^{-1}, but would not do so in still water.

The chief filter-feeders are the net-spinning Trichoptera larvae and Simulium. The faster the current, the more food passes a fixed point in unit time, but, above a certain speed, the greater the effort required to retain a foothold and the greater the difficulty of keeping a net in operation (Hynes, 1970). Edington (1968) mapped the rate of flow and the position of the nets of Hydropsyche instabilis in a suitable area of stream. He found that this species occurred predominantly in rapids, while Plectrocnemia conspersa occurred primarily in pools. Edington additionally found that the riffle species were not present in pools, but that the pool species extended into riffles, where, in spite of the fast flow, they could probably find a place between or under stones where the current was suitable for their net-building.

Hawkes (1975) relating current velocity to the longitudinal distribution of benthic communities in river zonation and classification, concluded that this factor acted both directly or indirectly in determining the nature of the substratum as well as affecting the autecological distribution of benthic species.

2.42 Nature of the substratum

Current probably exerts its most important effect on the benthic community indirectly through its effect on the nature of the substratum. Where the flow is rapid, the bottom is composed of rock, boulders or gravel, while sluggish flows, however, allow the deposition of silt and suspended matter which produces a bed of silt, mud and detritus. Between these extremes, one finds a gradation in the size of the substratum particles.

The stability of the substratum controls the density of the constituent members of the community in addition to controlling the species present in the substratum.

Cummins (1962) advocates the measurements of the physical constitution of the sediment as an integral part of all benthic faunal surveys, because substratum particle size can serve as a common factor in benthic ecology.

The classical papers of Percival and Whitehead (1929; 1930) categorising the types of stream bed in the West Riding streams of Yorkshire found that various biotopes had a characteristic community. The following groups were classified:

1. Loose stones: Loose rounded stones of more than 5 cm diameter, forming an unstable substratum, no visible vegetation, current velocity 4 m sec^{-1}, assortment of round pebbles of diameter down to 5 mm. Ephemeroptera 33.2%, Trichoptera 31.12%, Diptera 20.81%, Plecoptera 5.14%.

2. Cemented stones: Stones as in 1 set in a matrix down to 9 mm diameter, diatomaceous growths, stable substratum. Trichoptera 39.95%, Ephemeroptera 20.45%, Coleoptera 10.3%.

3. Mixed small stones: Bed of mixed clean stones 2.5 cm to 0.5 mm diameter. No visible vegetation, unstable substratum. Trichoptera 34.4%, Coleoptera 33.8%, Ephemeroptera 13.5%.

4. Stones bearing *Cladophora*: Rounded or flattened stones 30 cm to 2.5 cm, with growths of *Cladophora* set in a matrix of particles 0.5 mm to 0.3 mm diameter, stable substratum. Diptera 40.0%, Trichoptera 15.5%, Ephemeroptera 13.0%, Naididae 12.5%.

5. Loose moss on stones: Large stones carrying a growth of moss, allows easy passage of water between the stems and prevents accumulation of detritus. In high regions this substratum may form a significant portion of a stream. Diptera 65.34%, Ephemeroptera 13.42%, Coleoptera 6.66%.

6. Thick moss on stones: Thick carpets of moss, forming dense growths which prevent the passage of water and allows the accumulation of fine detritus. Diptera 42.0%, Naididae 26.4%.

7. Potamogeton on stones: Similar to 4 but with Potamogeton perfoliatus forming dense growths. Diptera 44.03%, Gasteropoda 17.13%, Tubificidae 14.66%.

The first five divisions are characteristic of eroding reaches, while the last two divisions are possibly common in minnow reaches. The average percentage compositions of the associated fauna found in these groups indicated that the Ephemeroptera were most abundant in the loose stones, as were the Plecoptera, while Trichoptera and Gasteropods favoured stable substrata. Diptera preferred loose and thick moss, while Naids favoured the latter. Tubificids were at their highest density in Potamogeton on stones.

Maitland (1964) examined and compared the invertebrate fauna of sandy and stony substrata and found complete differences in their respective faunas. Characteristic of the sandy substratum were burrowing forms such as Tubificidae, Chironomidae and Pisidium, while Plecoptera, Ephemeroptera, Hirudinea and Gammarus were characteristic of the stony substratum.

Moon (1939) observed that as a river evolved, there was a gradual change from an eroding substratum to a depositing substratum and this was reflected by a gradual replacement of the food chains dependent on algae by those dependent on silt and detritus. He suggested that the substratum has two important effects on aquatic insects. First, the substratum is a mechanical support for the activities of the insect, offering a shelter and a foundation in turbulent water, and a firm base for the pupation of those forms which end their aquatic life with a period of quiescence. Secondly, the substratum is the surface on which the food suitable for the insect fauna grows or is deposited.

As a consequence of these investigations, the fauna of the eroding reaches can be seen to be more diverse than the depositing silted reaches and pools. Hynes (1970) generally concludes that the more complex the substratum, the more diverse is the fauna.

The silt and mud forming the bed of sluggish stretches does not form a suitable substratum for the eroding species (Hawkes, 1962). In these regions burrowing forms such as oligochaete worms, chironomid larvae and mussels are commonly found. On the mud surface *Asellus aquaticus* may be found with the predatory larva *Sialis lutaria*. The dominant species of this community are probably determined by the organic content of the substratum. In mud rich in organic matter only those organisms able to withstand low oxygen concentrations will be present, but in more inorganic silts such species as the may-flies *Ephemera* and *Caenis* are more frequent (Hawkes, 1962). In more lowland rivers the larger bivalve molluscs *Anodonta* and *Unio* supplement Pisidium and the larger species of Planorbis including P. corneus supplement Lymnaea (Hawkes, 1975).

Illies and Botosaneanu (1963), in their combined scheme for the classification of running water bodies, were able to distinguish well-defined topographical zones based on the composition of the bottom fauna communities. Of these 'rhithron' and 'potamon' were considered of major importance. However, Thorup (1966), in a critical review of studies of stream zone systems, concluded that it was not possible to construct a satisfactory classification on the basis of bottom fauna communities. Whatever the argument, the existence of a transition in faunal types from source to mouth is an ecological phenomenon of some importance in most river systems.

2.43 Presence of Macro-vegetation

Studies of animal populations in running waters have usually included some reference to the animals present on the vegetation (Percival and Whitehead, 1929; Harrod, 1964).

Percival and Whitehead (1929) found Gammarus abundant where there were stones covered with loose and thick moss, while Baetis had a similar distribution, but was also found in dense outgrowths of Potamogeton. Ephemerella while associated mainly with moss was also found amongst Cladophora and to a lesser extent Potamogeton. Ancylus fluviatilis was also most numerous on the stones with Potamogeton.

Harrod (1964) made a quantitative study of the distribution and abundance of the invertebrate fauna inhabiting four plant surfaces, Ranunculus fluitans, Callitriche platycarpa, Veronica beccabunga and Carex sp. in a chalk stream. Considerable variation in the composition of the animal populations on the plants was found. Several animals, i.e. Baetis rhodani, Rhyacophila dorsalis, chironomid larvae and Hydracarina were present on all plants, but other animals showed preferences for a particular plant. Gammarus pulex and Hydropsyche sp. showed a distinct preference for Callitriche, while Simulium ornatum completely dominated the populations of both Carex and Ranunculus. It was suggested that factors such as the morphological form of a plant, the peri-phyton on the plant surface, the chemical nature of the plants and the habits of the various animals present, may have accounted for the observed differences on the four plant species examined.

2.44 Temperature

Researchers have devoted more attention to temperature as an environmental variable for aquatic organisms than probably to any factor.

Temperature effects on chemical toxicity are complex because temperature alone may be lethal to aquatic organisms, as is the case with stenothermic organisms (narrow temperature range). A rise of 10^{o}C in accordance with the Van't Hoff rule results in the metabolic rate doubling, while the rate of a chemical reaction also increases with rising temperature. Because of this, the activity of an animal is greatly affected. The heart rate often serves as an index of metabolic or respiratory stress on the organism and at high temperatures haemoglobin has a reduced affinity for oxygen and therefore becomes less efficient in delivering oxygen to the tissues.

Temperature affects feeding, digestion and growth of organisms, as well as playing a critical role in reproduction. It also determines the amount of oxygen in solution and this will be decreased with rising temperature.

The effect of temperature as an environmental factor is often difficult to assess because it is associated with the current velocity and the nature of the substratum, cooler waters associated with shallow rapids found in headstreams.

The life histories of benthic invertebrates are largely determined by the temperature of the water, since their growth rate is a function of it. Temperature also restricts the distribution of particular species, best exemplified by the cold-water stenotherm *Crenobia alpina* (Macan, 1974b). This species is only found in temperatures below 13°C, above this it is succeeded by *Polycelis felina* extending from 13°C to 17°C. Macan (1974b) suggests that there seems to be competition between species with different thermal requirements, with current speed involved in such a relationship.

Although certain stoneflies are also restricted to the smaller upland streams, where mean annual temperatures are low, other mayfly nymphs such as *Heptagenia lateralis* are rare or absent in streams where the summer temperature exceeds 18°C (Macan, 1974b).

In Britain there is generally a gradual increase in temperature as one moves downstream, although one may experience sharp rises in temperature in the headstreams during the summer.

In lowland rivers which are used for cooling power stations, and therefore suffer temperature increases, the invertebrates are usually eurythermal and are therefore less affected by thermal discharges (Hawkes, 1978). Although temperatures may not rise above 30°C, the relative abundance of species within the community may be changed and some species may appear, e.g. the exotic worm Branchiura sowerbyi in the River Thames (Mann, 1958) or the snail Physa acuta (Hawkes, 1978).

The results of various workers attempting to classify river zones for world-wide application, using temperature, current velocity and nature of the substratum have been summarised by Hawkes (1975) for which the following classification has been developed (Illes, 1961).

1. Rhithron is defined as that part of the stream from its source down to the lowermost point where the annual range of monthly mean temperatures does not exceed 20°C. The current velocity is high and the flow volume is small. The substratum may be composed of fixed rock, stones or gravel and fine sand; only in pools and sheltered areas is mud deposited.

2. Potamon is the remaining downstream stretch of river where the annual range of monthly mean temperatures exceeds 20°C. The current velocity over the river bed is low and tends to be laminar. The river bed is mainly of sand or mud, although gravel may also be present. In the deeper pools oxygen may be depleted, light penetration limited and mud deposited.

2.45 Chemical nature of the water

The importance of the chemical nature of the water, in particular the calcium and hydrogen ion concentrations have been reported by Boycott (1936) and Moon (1939) to be responsible for certain broad differences in the density and arrangements of animal communities.

Calcium has been shown to limit the range of a number of species by creating soft and hard waters in accordance with the hydrogen ion concentration (Moon, 1939).

Boycott (1936) studying the Mollusca found that fifty percent of the species are hardly ever found in water with less than 20 $mg.l^{-1}$ Calcium. Macan (1974b) has described a range of below 1 $mg.l^{-1}$ in soft water to over 100 $mg.l^{-1}$ in hard water.

Mann (1955a; b) studied the distribution of leeches in waters of varying hardness and designated three zones:
'soft' with less than 7 $mg.l^{-1}$
'intermediate' with 7 to 24 $mg.l^{-1}$ and
'hard' with over 24 $mg.l^{-1}$.
Helobdella stagnalis was the commonest species found in hard waters, while *Erpobdella octoculata* was always numerous in soft waters.

Macan (1974b) reviewing the controversy over *Asellus*, concluded that it was found in most places where there is more than 12.5 $mg.l^{-1}$ Calcium and in a few places where there is less than 5 $mg.l^{-1}$. Macan (1974b) also commented on the distribution of *Gammarus* and showed that a concentration of 3 $mg.l^{-1}$ to be critical.

It appears that some animals are limited in range by their dependence on calcium and in general, numbers of species increase with increasing hardness.

2.5 Effect of Organic pollution on benthic invertebrate communities

The indicator concept is based upon the assumption that relatively long-lived benthic organisms reflect the environmental conditions over a long period of time. An animal that finds itself in a changing environment brought about by pollution would be called upon to do one of three things: adapt itself to the changed environment, migrate, or be destroyed. If the changed condition is favourable for those animals or plants that adapt themselves to it, then they will thrive and build high populations. This often gives evidence to the intensity of pollution and the degree of recovery.

Biological aspects of stream pollution can be considered from two separate and related points of view:

1. How pollutants change the character of the stream as a habitat for organisms.
2. The action of organisms upon the pollutant and their related distribution.

Organic pollutants may alter stream environments and thereby affect aquatic life in a number of ways. These changes may include a decrease or increase in dissolved oxygen or both, an increase in dissolves nutrients, an increase in turbidity, changes in the character of the stream bed, production of undesirable growths, an increase in stream temperatures and the addition of toxic wastes where industrial effluents are included with domestic organic wastes. It is rarely possible to distinguish between different types of organic pollution using invertebrates, but, in general, there are notable similarities in the effects of organic enrichment. The most important factors are described below:

2.51 Dissolved oxygen

The entry of putrescible organic matter, as sewage, into rivers results in a depletion in oxygen resulting in the elimination of those species requiring high dissolved oxygen concentrations. Because of the current, this depletion in dissolved oxygen known as the oxygen sag is greatest some distance below the discharge. So long as oxygen is not used up too rapidly during the oxidation of organic matter by the activities of bacteria, some improvement in the condition of the river is expected. However, in extreme cases when all the dissolved oxygen has been exhausted, self-purification will cease and septic conditions will prevail. Downing (1967) recognised the reasons for the oxygen depletion and suggested the important factors which were responsible, summarised as follows:

1. The Biochemical oxygen demand. This varies and depends on the carbonaceous content of the water and the density of bacteria.
2. Nitrification. The nitrogenous material present in the effluent is oxidised (Ammonia to Nitrates) as it passes downstream, taking oxygen out of the water.
3. Respiration of plants, muds and slimes.

In suggesting the factors for oxygen depletion, Downing (1967) also gave reasons for its recovery:

1. Reaeration from the atmosphere, depending on the depth and current.

2. Photosynthetic activity of algae and macrophytes.

The degree of oxygen depletion varies greatly, depending on the severity of the pollution. Butcher (1946) found that in the severely polluted River Trent, the oxygen concentration dropped from 107% to 0% as the effluent entered the river, the oxygen concentration in the river then recovered gradually. In the mildly polluted River Churnet (Butcher, 1946), the depletion was less severe, dropping from 83% to 35% and then a gradual recovery. Similar results have also been recorded by Gaufin and Tarzwell (1952) and Hawkes and Davies (1971).

Although it is difficult to relate species distribution to one factor alone, it has been found that the larvae of the Chironomus riparius group survive longer at low oxygen levels and are adversely affected by high oxygen levels (Fox and Taylor, 1955). Recent authors Hawkes and Davies (1971) and Gower and Buckland (1978) showed that the largest densities of C. riparius were present at the lower end of the oxygen sag curve and numbers declined as the oxygen levels elevated.

Gaufin and Tarzwell (1952; 1956) related the presence of air breathing Coleoptera, Diptera (Culex and Eristalis) and Mollusca (Physa integra) to the septic zone, where the dissolved oxygen concentration was less than 1 mg. l^{-1}, while Aston (1973b) has shown that certain tubificids are able to survive in extremely anaerobic conditions e.g. Tubifex tubifex, Branchiura sowerbyi and Limnodrilus hoffmeisteri.

2.52 Dissolved nutrients

The presence of nutrients in a well oxidised organic effluent, together with the release of nutrients during repurification in the river can indirectly affect the benthic fauna by producing luxuriant growths of algae and occasionally mosses.

The amount of ammonia produced in an effluent will depend on the type of treatment and the efficiency of nitrification in the sewage works. If the effluent is well oxidised, only small amounts of ammonia are present, the nitrogenous material mainly being in the form of nitrates and nitrites. Butcher (1946) recorded levels of 11.4 mg. 1^{-1} in the River Trent, below the effluent at Stoke which then gradually decreased as one moved downstream. In the River Churnet, however, the ammonia levels were only elevated to 1.1 mg.1^{-1}.

A sewage effluent can often contain high concentrations of orthophosphates and these together with nitrates can enrich the waters to produce eutrophic conditions. A result of eutrophication is to enhance the growths of Cladophora (Hynes, 1960; Pitcairn and Hawkes, 1973) and to a lesser extent mosses. As Percival and Whitehead (1929, 1930) have shown, these growths allow the development of large populations of specific invertebrates usually in the recovery zones of heavy organic pollution, or in mildly organically polluted waters. Gower and Buckland (1978) have shown that the Chironomidae are typical of such zones, while Hawkes (1962) regards this as the Cladophora/Asellus zone.

2.53 Suspended Solids

Suspended solids may have many different sources, varying size, and have both direct and indirect effect upon aquatic life. Such particles from organic pollution may consist of small colloidal particles which have electrically active surfaces and can absorb other ions on to their surfaces. In this way they may act as a reservoir for other chemicals which enter the water. When these colloids settle out, they often form a soft flocculent mud in which bacteria flourish. As a result, anaerobic conditions could develop in the bed of a river. This settling out on the stream bed can blanket out the clean stream fauna whilst encouraging animals which favour silted conditions.

In mildly polluted waters although the clean water species may be eliminated, filter feeders such as _Hydropsyche_ and _Simulium_ may thrive due to the increase in particulate matter present in the water (Hawkes, 1962).

2.54 Summary of effects of organic pollution on benthic invertebrates

Hawkes (1962) summarises the changes in riffle fauna co-incident with organic pollution as three major trends:

1. The progressive reduction in numbers and eventual elimination of the non-tolerant species in succession according to their degree of intolerance.

   e.g. _Rithrogena_ - _Ephemerella_ - _Gammarus_

2. The initial increase in numbers of those species which at first are able to tolerate the adverse effects of organic pollution and benefit from the increased food supply and reduced competition.

As the degree of pollution increases these also are successively reduced in numbers and eventually eliminated according to their degree of tolerance.

e.g. Baetis rhodani - Simulium ornatum - Hydropsyche - Lymnaea peregra - Erpobdella sp.

3. The invasion of the habitat by species which under natural conditions are not members of the community either because the environment is not suitable under natural conditions or because they are not able to compete successfully with members of the normal community. Such organisms are naturally members of the silted communities in sluggish stretches or ponds. As the degree of pollution increases even these are successively eliminated.

e.g. Nais - Asellus - Sialis - Chironomus riparius - Tubifex.

Thus, in the rapids both quantitative and qualitative changes occur in the nature of the community with different degrees of pollution.

In the silted reaches, changes in the nature of the community with increasing organic pollution are largely due to trends 1 and 2. (Hawkes, 1962). Since the natural silted community's major activity is primarily concerned with the breakdown of dead organic matter, it is less affected by slight organic pollution than the eroding community. Hawkes (1962) points out that at first many species will benefit by the increased food, but as the pollution increases and the oxygen concentration falls, different species will eventually decline in numbers according to their tolerance.

e.g. Ephemera - Caenis - Tanytarsus - Asellus - Sialis - Chironomus riparius - Tubifex

2.6 Taxonomic status, distribution and salinity tolerance of selected freshwater Malacostraca

2.61 Gammarus tigrinus

Gammarus tigrinus Sexton is an alien species introduced into Europe from North America and was first recorded in Britain by Sexton and Cooper (1939). It has been suggested (Spooner, 1951) that this species is closely related to the North American marine littoral species G. annulatus Smith. Clemens (1950) gives a detailed account of G. fasciatus Say, which for some time was thought to be equivalent to the European G. tigrinus (Hynes, 1954). It was shown later however (Bousfield, 1958) that G. fasciatus consisted of two forms - a freshwater form and a brackish water form. The name G. fasciatus was retained for the freshwater form and the brackish water form was called G. tigrinus as this is the type found in Britain and Europe. The detailed study of Clemens (1950) refers to the freshwater type from Lake Erie and consequently the study of Hynes (1955a) remains the most detailed account of G. tigrinus in Britain to date.

In Western Europe G. tigrinus was first recorded in the Netherlands in 1960 by Nijssen and Stock (1966) and from the Tjeukemeer in 1966 (Pinkster and Stock, 1967). Its subsequent spread and development into other parts of the Netherlands has been recorded and reviewed by Chambers (1977). In 1957, Schmitz (1960) introduced 1000 specimens purposely into saline stretches of the Rivers Werra and Weser, Germany, and since then it has successfully invaded other localities where it is now well established (Fries and Tesch, 1965; Ruoff, 1968). During this spread, it has replaced either completely or to a large extent the indigenous gammarid fauna (Chambers, 1973; 1977).

The euryhaline brackish water amphipod G. tigrinus was discovered in Britain for the first time at Droitwich in 1931 by Fox (Sexton and Cooper, 1939). It was found in abundance living in the brackish waters of the River Salwarpe, the Worcester and Birmingham Canal and in Wyken Slough near Coventry. In 1938 this species was also found in small numbers in the River Avon at Tewkesbury. It appears to thrive not only in dilute sea water but also in saline water where the proportions of the major ions are not as in the sea (Macan, 1974a). According to Hynes (1955b) this species is particularly found in localities which are mostly muddy and weedy and are much more pond-like than most Gammarus habitats. Hynes (1955b) also suggests that the distribution of G. tigrinus can be explained on the assumption that it is more tolerant of a high concentration of salt than G. pulex but cannot compete with it in fresh water.

The distribution of G. tigrinus as shown by Hynes (1955b) is restricted to the north west of England (Cheshire) and the west Midlands. It has been recorded in the marshes at the mouths of Rivers Mersey and Welsh Dee and localities near Northwich in the drainage basin of the Mersey (Hynes, 1955b) and Frodsham Marsh, Cheshire (Hynes, 1955a). A comprehensive study of gammaridae in saline parts of Cheshire (Holland, 1976b) has revealed that G. tigrinus may be invading other localities via the Trent and Mersey Canal, the Bridgewater Canal and through anglers moving fish unofficially from one water to another. In the west Midlands during 1962, Sutcliffe (1968) collected G. tigrinus from a small tributary of the River Penk at Coven, Staffordshire, but by 1964 found only a few specimens at this site.

While principally a brackish-water species, G. tigrinus has successfully invaded the fresh water of Lough Neagh, the River Bann and a stream at Ballykelly (Londonderry), Northern Ireland (Spooner, 1951; Hynes, 1955b). In England the only record at a non-saline locality is the Avon at Bretford (Bassindale, 1946; Hynes, 1951), but even here the salt content of the water is rather unusual. Only four specimens were found at this site and it has been suggested that they were strays from another habitat (Hynes, 1955b).

The establishment of G. tigrinus in coastal brackish water and in fresh waters where pollution has raised the ion content is probably due to its ability to successfully compete in waters which have wide fluctuations in saline content. The salinity tolerance range of this species as recorded in the literature is presented in Table 2.3 and it can be seen that it has been found in salt concentrations ranging from 30 - 10900 mg. l^{-1} as chloride. Unlike other invaders from brackish waters it appears that G. tigrinus has been able to occupy apparently empty niche in inland waters with a high salinity (Macan, 1974a) and it is clear that it is not physiologically confined to saline waters (Hynes, 1955b).

Table 2.3 Distribution of Gammarus tigrinus with respect to salinity as quoted in the literature

Author(s)/Year	Location	Chloride range (mg. l^{-1} as cl)
Sexton and Cooper (1939)	Wyken Slough, Coventry	662
	R.Avon, Tewkesbury	75
	R.Salwarpe, Droitwich	1026 - 1478
Bassindale (1946)	R.Avon, Bretford	83
Hynes (1955a)	Frodsham Marsh, Cheshire	287
	Wyken Slough, Coventry	290 - 533
Hynes (1955b)	Lough Neagh, N. Ireland	30
Holland (1976b)	Bridgewater canal	80 - 2300
	Trent and Mersey canal	550 - 6300
	Alt Catchment	50 - 100
	Glaze Catchment	30 - 1500
	Bollin Catchment	290
	Dane Catchment	100 - 2100
	Weaver Catchment	50 - 650
	Frodsham Marsh	Variable up to 10900

2.62 Gammarus pulex

Gammarus pulex (L.) sub-species pulex Schellenberg is the common 'fresh-water shrimp' on the mainland of Britain, occurring in both standing and running waters (Gledhill *et al.*, 1976). It is found in waters of moderately high to low mineral content but, in Britain, never where there is any marked influence of brackish water, either from estuaries or mineral springs (*Gledhill et al.*, 1976). According to Macan (1974b) it probably entered England comparatively recently, in terms of zoogeography and has now spread to most parts of the country. It is only absent from the extreme north of Scotland and from the western tip of Cornwall. Apparently as it spread across the country it drove *G. duebeni* Liljeborg out and this species now only has a small foothold on the Lizard peninsula (Cornwall), north Wales and isolated parts of Scotland (Gledhill *et al.*, 1976).

G. pulex is a eurytopic species (Hynes, 1955b), being found in streams and rivers of all types, lakes and field ponds, the latter probably introduced by anglers. It occurs both in hard and soft waters, but is not found in water with a pH consistently below 5.7. Holland (1976a) studied the distribution of this species in the catchment area of the Rivers Mersey and Weaver and found that it was almost entirely limited to the Cheshire rivers, where a chloride concentration of 50 - 100 mg. l^{1} was attained. It had a limited distribution in the Lancashire area apart from its presence in ponds and lakes where its introduction could possibly on occasions have been attributed to fishery activities.

Macan (1974b) comments that there is now general agreement that *G. pulex* occurs in rivers down to a point where it comes into contact with sea water.

2.63 Crangonyx pseudogracilis

Crangonyx pseudogracilis Bousfield is a North American species, first discovered in England by Crawford (1937) from the Metropolitan Water works at Lea Bridge. This publication stimulated Tattersall (1937) to record that it had been found in 1930 from a London suburb, but he had refrained from mentioning it since he could not justify his identification, because it was an American amphipod. By 1944 it had been recorded in the River Avon (Bassindale, 1946), and in 1951 it was found in the Midlands, Oxford, Surrey, Middlesex and Northants (Spooner, 1951; Hynes, 1951). Hynes (1955b) has suggested that this amphipod has since spread through the Midlands and other counties via the canal system, from which it is colonising nearby habitats. It has subsequently crossed land barriers for in 1960 it was found in Lake Windermere (Macan, 1974b) and the Norfolk Broads (Gledhill et al., 1976).

This species occurs in various types of habitat including ponds, canals, muddy lakes, reservoirs, stony rivers and even organically polluted water (Hynes, 1955b). It tolerates inland saline localities and has been found with G. tigrinus at Wyken Slough. Holland (1976a) characteristically found that Crangonyx favoured the slow-flowing or still-water environments and was unable to establish itself firmly in any streams and rivers in the Cheshire area. Holland (1976a) also concludes that this species is unlikely to be found in waters having a chloride level greater than 250 - 350 mg. l^{-1}.

2.64 Corophium curvispinum

Corophium curvispinum Sars var. devium Wundsch is thought to have spread from the Caspian and Black Sea regions, probably along canals

(Macan, 1974a) and was first recorded in Britain from the River Avon at Tewkesbury (Crawford, 1935). Moon (1970) recorded it in the Grand Union Canal in the Midlands and in the River Severn at Stourport in 1962 (Macan, 1974a). It has also been located in the Shropshire Union Canal (Holland, 1976a).

This tubicolous amphipod which prefers a silty environment is the only member of its genus which lives in fresh water. Moon (1970) has suggested that a second species C. spongicolum Velitchkovsky may be found in sponges on which it lives, but Gledhill et al., (1976) has reported that none has been found to date.

2.65 Asellus aquaticus

Asellus aquaticus (L.), the water hog-louse is widely distributed throughout the British Isles, although it appears to be absent from some of the Western Isles and from the most northerly parts of Scotland (Gledhill et al., 1976). Generally it is normally found inhabiting depositing sub-strata of slow flowing waters, where it feeds on decaying organic debris, although it also invades eroding sub-strata following organic pollution (Hawkes, 1962).

A.aquaticus is not found in waters with a hardness less than 5 mg. l^{-1} (Ca) (Macan, 1974b), or where there is a low total dissolved matter content (Reynoldson, 1961).

It is known that sewage inputs increase the salinity levels of receiving waters (Hynes, 1960) and it is probable that A. aquaticus can withstand a wide range of salinity conditions, while maintaining a stable reproductive community. Lagerspetz (1958) has investigated the brackish-water tolerance of A. aquaticus and has concluded that it can successfully survive and breed in salinities up to 7‰ (freshwater 0.03‰).

Holland (1976a) relating A. aquaticus distribution to physico-chemical parameters in Cheshire has shown that it can survive in oxygen levels down to 2.0 mg. l^{-1}, a view supported by Edwards and Learner (1960).

CHAPTER 3

STUDY OF SAMPLING METHODS TO COLLECT BENTHIC MACROINVERTEBRATES

3.1 Initial investigations to select sampling method

3.11 Objectives

The objectives of the research programme were to develop and evaluate for the Water Data Unit (Department of the Environment), a standard biological method for the surveillance of river-water quality, using benthic macroinvertebrates and to devise a method of processing the surveillance data.

3.12 Basis for planning investigations

Biological surveillance which is used to monitor water quality, is the repeated and standardised measurement of variables over a sufficiently long period and with sufficient frequency to allow the detection of statistically significant changes occuring within a system (N.E.R.C. report, 1976).

The premise upon which the biological surveillance of river water quality is based is that temporal changes in the biota at a given place in the river, other than those accounted for by seasonal changes or changed flow regime, reflect changes in the water quality. In this respect, it was decided to limit the scope of the investigation to benthic macroinvertebrate communities, as in riverine conditions these are more likely to be useful as indicators of general water quality flowing over them.

As reported in Chapter 2, a major problem in using benthic macroinvertebrate communities is that they are determined by factors other than water quality, especially current velocity and nature of substratum. Since these determinants differ at different points along a river, differences occur between the benthic communities at these points regardless of water quality. In using such communities therefore to indicate spatial differences in water quality along a length of river, these other determinants must be taken into account in interpreting the results. It is often difficult in practice to select ecologically similar stretches of river as sampling stations, especially when a river transforms from an upland rhithron type to a lowland potamon type. To eliminate differences due to substratum differences, the use of an artificial substratum was envisaged although it was realised that this would not eliminate differences due directly to differences in current velocity.

The investigations were therefore directed at sampling methods for the respective communities in different types of river, bearing in mind the requirements for simplicity and economy necessary for routine surveillance work.

3.13 Planned programme of Investigations

Based on the above considerations, the programme was planned as two phases:

PHASE 1 (Chapter 3) - The objective of the first phase was to select by field trials a suitable artificial substratum for sampling benthic macroinvertebrates. This would be achieved by comparing the communities developing on different possible artificial substrata in different water qualities in rapid and slow flowing conditions. A comparison would also be made between the communities of the artificial substrata tested and the natural community of the corresponding part of the river.

This preliminary investigation lasted about nine months and was carried out using the three simulated streams (earth channels) at the Checkley Hydrobiology Field station and in the adjacent River Tean upstream and downstream of the sewage works effluent.

PHASE 2 (Chapter 4) - The sampling methods selected in Phase I would be evaluated in a range of river zones and water qualities. To detect seasonal differences, this Phase would continue over a period of more than one year.

3.14 Sampling stations

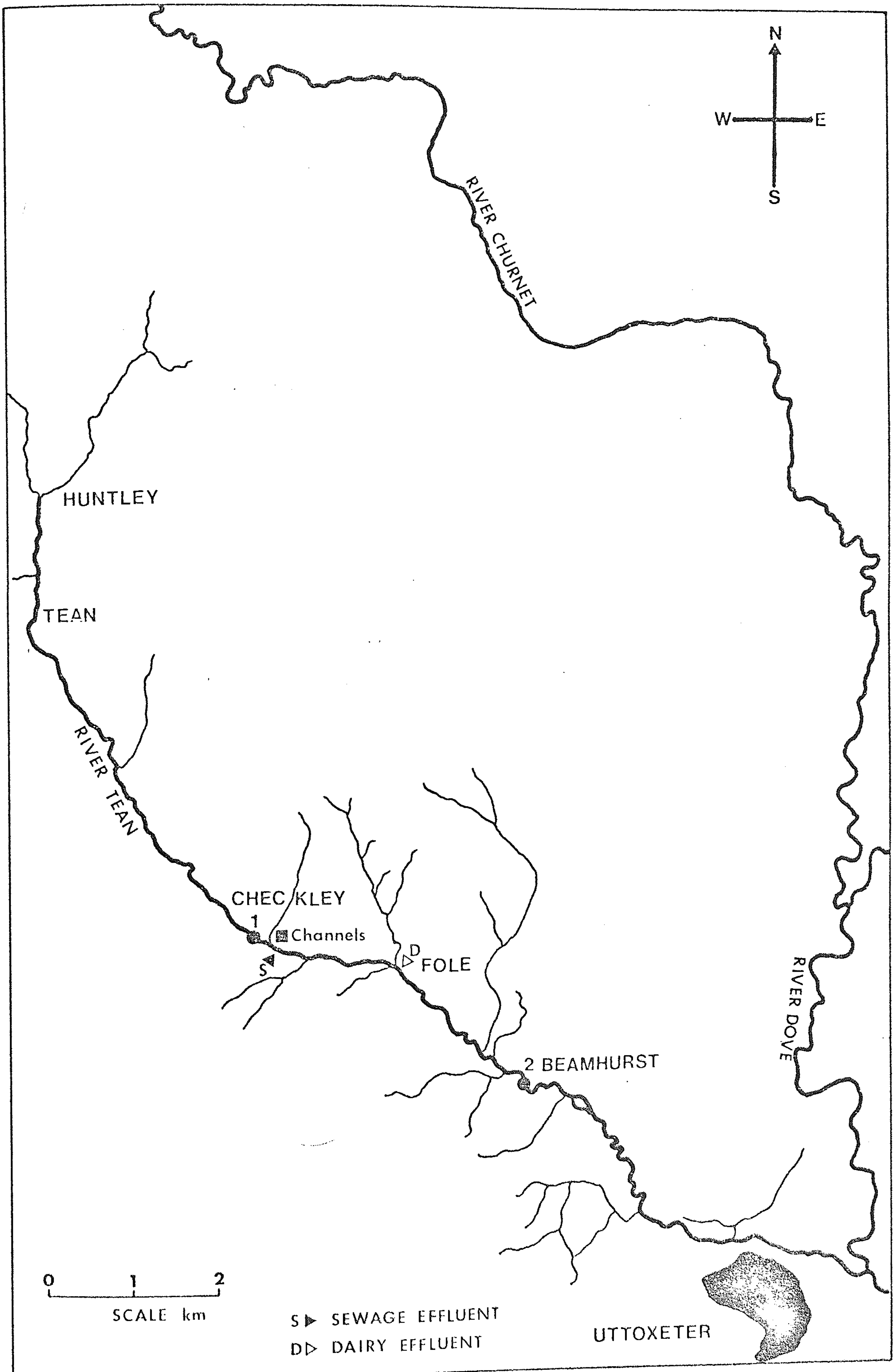
3.141 River Tean (T_1 and T_2)

The River Tean which is a tributary of the River Dove, arises as two streams N.W. and N.E. of Cheadle, at Huntley (SK 006415) about 15 km. E of Stoke-on-Trent, and flows for approximately 15 km. to its confluence with the River Dove, N of Uttoxeter (Fig. 3.1). The river flows through farm land and apart from agricultural run-off, the only sources of contamination are organic enrichment from the Blithe Valley Sewage Treatment Works at Checkley and a Creamery effluent at Fole.

Although polluted by mine wastes in the past and rendered 'fishless', it is now a fairly good quality river and is consequently an ideal river to study sampling methods, due to its isolated position along the Tean Valley in Staffordshire.

Two similar sampling stations were selected, each having a riffle

FIG.3.1 MAP OF THE RIVER TEAN AND ITS TRIBUTARIES SHOWING THE DISTRIBUTION OF SAMPLING STATIONS.



and pool reach, one upstream of the sewage effluent and the other downstream. Station 1, at Checkley (SK 028377) was about 500 metres upstream of the outfall and here a riffle (T_1R) and a pool (T_1P) were selected for study. Station 2, at Beamhurst (SK 061361) was approximately 4 km. downstream of the outfall and again a riffle (T_2R) and a pool (T_2P) were selected for investigation.

The riffle section had typically a loose, shallow, stone substratum (eroding), a water depth less than one foot and a fairly rapid flow with a turbulent surface water. The riffle habitat has typically a large number of micro-habitats and niches which support a wide diversity of animal species and consequently is an ideal habitat to study, since macroinvertebrates found here are sensitive to deteriorations in water quality. A pool section is one of a depositing substratum, with a muddy bottom, a depth greater than three foot and a slow flow.

3.142 Experimental Channels Checkley (ERR, E50R, E75R)

The earth channels investigated consisted of long channels excavated adjacent to the River Tean, each with different gradients to simulate a riffle-pool system; the riffles filled with smooth gravel (nominal size 40 ± 5 mm.). Details of the characteristics of the channels are summarised in Table 3.1 and Fig. 3.2.

FIG.3.2 PLAN OF THE EXPERIMENTAL CHANNELS AT CHECKLEY,STAFFS.

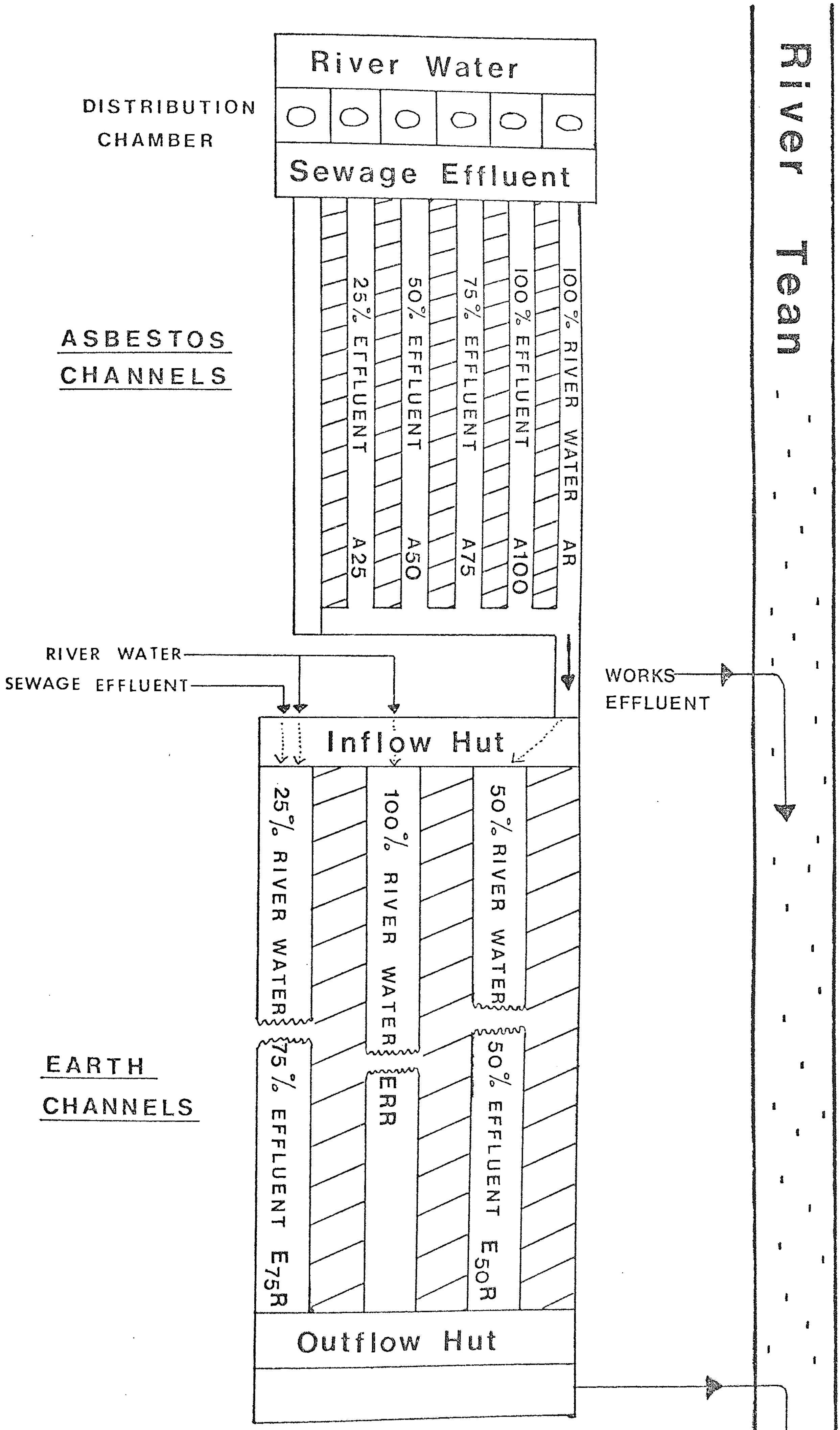


Table 3.1 Summary of the Earth Channels

Total Length	300 m	Length Riffle	90m	Length Pool	60m
Gradient Riffle	1 in 200	Width Riffle	1m	Width Pool	1.4m
Depth Riffle	15 cm.	Depth Pool	45cm	Flow	0.05 M^3/sec.

CHANNEL	CURRENT VELOCITY	WATER MIXTURE
ERR	43 cm. $sec.^{-1}$	100% River water
E50R	43 cm. $sec.^{-1}$	50% Sewage effluent 50%River water
E75R	43 cm. $sec.^{-1}$	75% Sewage effluent 25% River water

The earth channels used primarily for productivity studies and heavy metal uptake by secondary consumers, consisted of one carrying 100% river water (ERR) and one 50% sewage effluent (E50R) and the third 75% sewage effluent (E75R). It was envisaged that one riffle and one pool from each channel would be used for the initial studies; however, the pools proved unsuitable for study due to problems with excess siltation.

Asbestos-concrete channels previously used for Cladophora studies were re-layed for the purpose of this investigation at an increased gradient of 1 in 200, in five lengths each of 50 metres. Down each of these, different mixtures of sewage effluent and river water were passed (Fig. 3.2).

3.15 Methods and materials

3.151 Physico-chemical determinations

Water samples were collected from the River Tean in one litre plastic water bottles and immediately brought back to the laboratory for refrigeration. Analysis of the samples was carried out in the following five days.

The following determinations were carried out for each sampling station at intervals of two weeks:

(a) Temperature ^oC

(b) Suspended solids mg. l^{-1}

(c) Dissolved Oxygen mg. l^{-1}

(d) Dissolved Oxygen percentage saturation

(e) Biochemical Oxygen Demand (BOD_5) mg. l^{-1}

(f) Chloride mg. l^{-1}

(g) Total Alkalinity mg. l^{-1} $CaCO_3$

(h) Total Hardness mg. l^{-1} $CaCO_3$

(i) Calcium and Magnesium Hardness mg. l^{-1} $CaCO_3$

(j) Ammoniacal Nitrogen $NH_3 - N$ mg. l^{-1}

(k) Oxidised Nitrogen $NO_3 - N$ mg. l^{-1}

(l) Inorganic Phosphate mg.Pl^{-1}

Ammonia,Nitrates, and Phosphates (after preserving in the field with concentrated HCl) were determined by automatic analysis using a standard Technicon Auto-Analyser. Analysis for total inorganic phosphate was carried out following the Technicon Industrial Method 3 - 68W. The methods recommended by Chapman, Cooke and Whitehead (1967) were employed for ammonia and nitrate determinations.

Other determinations listed above were carried out as recommended in the H.M.S.O. publication 'Analysis of Raw, Potable and Waste Waters' (1972).

3.152 Benthic macroinvertebrate sampling

3.1521 Natural substrata

In the river riffles two types of routine bottom sampling techniques were employed:

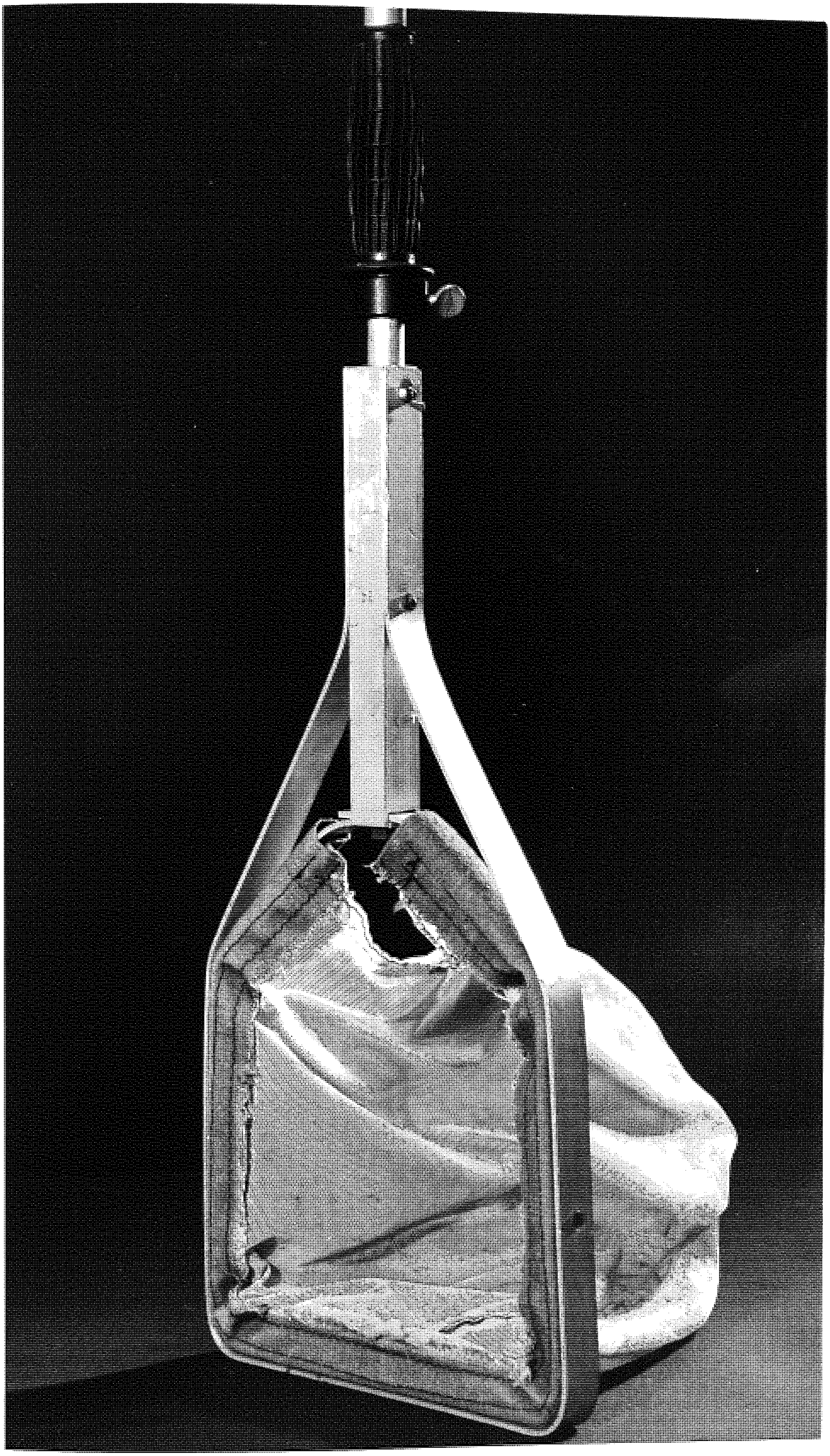
(a) Hand-net Sampling (Qualitative)

The heel-kick and stop net method of Macan (1958) consisted of holding a hand-net (10 meshes cm.$^{-1}$) against the river bed whilst the substratum immediately upstream was disturbed by kicking for 30 seconds. The macroinvertebrates thus loosened were swept into the hand net (Plate 1).

(b) Aston Cylinder Sampler (Quantitative)

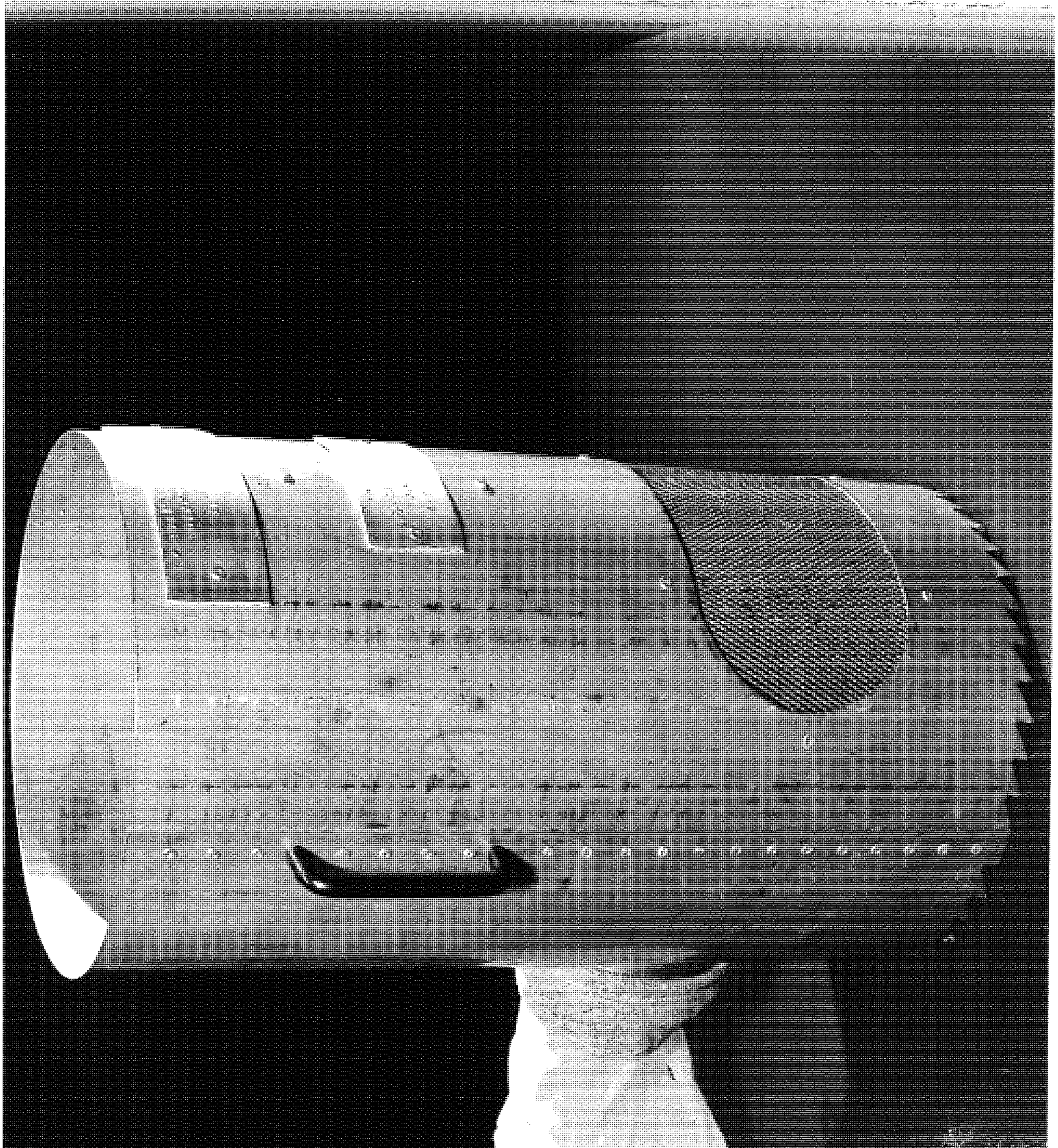
A simplified cylinder sampler was designed for quantitatively sampling the benthos of riffles in Midland streams (Hawkes and Davies, 1971). The sampler (Plate 2) is a modified Neill sampler (Neill, 1938) not having the sliding doors or the flange. It consists essentially of an open-ended cylinder constructed of 18 gauge (0.45 mm.) stainless steel having the lower edge serrated with 38 teeth each 1 cm. deep, and the upper edge is covered by a protective edging strip. Laterally positioned handles facilitate the sampler being pushed into the river bed. Water enters the aperture through an oval aperture (257 cm.^{2}) cut into one side of the cylinder towards the lower edge. This aperture is fitted with a screen con-

PLATE 1 ANTOX PROFESSIONAL HAND-NET



sisting of stainless steel coarse mesh to prevent the entry of drift organisms. Opposite this opening a second hole (diameter 113 mm.) is made with a short exit port to which a detachable collecting net is fitted. The net is 50 cm. deep and is constructed of nylon bolting cloth (15 meshes $cm.^{-1}$) having a 15 cm. deep canvas collar, which holds a draw cord, for attachment to the cylinder. The cylinder which has a depth of 50 cm. (an arm's length) encloses a cross-sectional area of 0.05 $m.^2$. With the collecting net firmly attached, the sampling position on the river bed is approached from downstream to avoid undue disturbance of the sample area. The sampler is then placed on the stream bed to enclose the area to be sampled so that the water inlet screen faces the current. The cylinder is pushed into the substratum to a depth of 7 cm., where possible, using an alternating rotary motion. The cylindrical shape of the sampler is superior to the square box samplers (Macan, 1958) in that it permits this rotary action. The collecting net is extended downstream fully open to permit an unimpeded flow of water through it. In operation one stands immediately downstream of the sampler with the feet astride of the collecting net, using both the legs to maintain the position of the sampler. The larger stones in the enclosed sample area are then worked over by hand - any attached animals being dislodged into the water flowing through the cylinder. The smaller stones and finer substratum are then distributed by turning over and stirring by hand to a depth of 5 cm. to 10 cm. depending on the substratum. Repeated stirring of the substratum ensures

PLATE 2 ASTON CYLINDER SAMPLER



that most of the organisms are removed. The water flowing through the cylinder carries the disturbed animals into the net where they are retained. After allowing time for the dislodged material to be carried into the collecting net, the net is removed. In doing so, the sample can be concentrated in the end to facilitate removal. The net is then everted to transfer the catch to bottles, care being taken to ensure all the animals in the net are removed. The sampling units were recorded at random within the site. As shown in Chapter 2, the number of sample units required will depend upon the accuracy required for the estimate of numbers per unit area depending upon the objectives of the survey. In these investigations three sampling areas were used, and the samples bulked.

3.1522 Colonisation Samplers

Two main types of colonisation samplers were examined in the initial investigations. One type was the mineral media as used in sewage biological filters (Percolating filters) and the other the artificial plastic media again as used in some sewage filters.

Three types of samplers using mineral media were tested:

(a) Slag Basket

40 pieces of slag of nominal size 40 ± 5 mm. diameter, contained in a 5 mm. mesh netlon basket 125 mm. diameter and 125 mm. high (Plate 3).

(b) Slag Bag

40 pieces of slag of nominal size 40 ± 5 mm. diameter, contained in a net bag (1 mesh $cm.^{-1}$). (Plate 4).

(c) Stone Bag

40 pieces of smooth gravel of nominal size 40 ± 5 mm. diameter, contained in a net bag (1 mesh $cm.^{-1}$).

Four types of sampler constructed of plastic filter media were tested:

(d) Biopac Unit

14 pieces of 'Biopac 50' were constructed into a cylindrical shaped sampler, using nylon straps, consisting of two layers each of seven pieces with six peripheral ones arranged around a single central piece (Plate 5). 'Biopac 50' was the trade name for this plastic filter medium made by Hydronyl (Stoke-on-Trent, Staffs.) and is now marketed as 'Actifil 50' by Norton Process Plants Ltd.

Each 'Biopac 50' piece consisted of an open cylinder 50 mm. diameter and 50 mm. long fenestrated with 12 rectangular holes each approximately 18 mm. x 11 mm. and having internally two sets of 6 radiating lamellae. Theoretically the whole unit presents a total surface area of greater than 2 $m.^{-2}$. The constructed unit is 10 cm. high and 15.5 cm. diameter. To reduce the loss of animals whilst retrieving the sampler, the lower quarter, including the base, is covered with a nylon gauze of 10 meshes $cm.^{-1}$.

(e) Biopac Bag

14 pieces of 'Biopac 50' medium contained randomly in a net bag (1 mesh cm.$^{-1}$). (Plate 6).

(f) Flocor Bag

30 pieces of 'Flocor R' medium contained randomly in a net bag (1 mesh cm.$^{-1}$). (Plate 7). 'Flocor R' was the trade name for this plastic filter medium made by I.C.I. Pollution Control Systems (Hyde, Cheshire). This medium consisted of plastic open-ended cylinders 30 mm. diameter and approximately 30 mm. long with corrugated walls.

(g) Spiral Sampler

Made by coiling a piece of netlon plastic mesh (6 mm.) 125 mm. x 500 mm. to produce a tight coil and fitted with a nylon gauze base of 10 meshes cm.$^{-1}$. (Plate 8).

In the riffles, the Slag Basket, Biopac Unit and Spiral Sampler were embedded in the natural substratum. The Biopac Bag, Flocor Bag, Slag Bag and Stone Bag were placed in small depressions in the river bed. Each sampler was securely anchored to the river bed with steel rods. For the initial stages of the programme, the middle section of the riffle was selected for study.

In the pools, all the colonisation samplers were placed on the surface of the river bed.

After the planned period of immersion, the samplers were removed individually, care being taken to prevent loss of organisms in doing so.

PLATE 3 SLAG BASKET

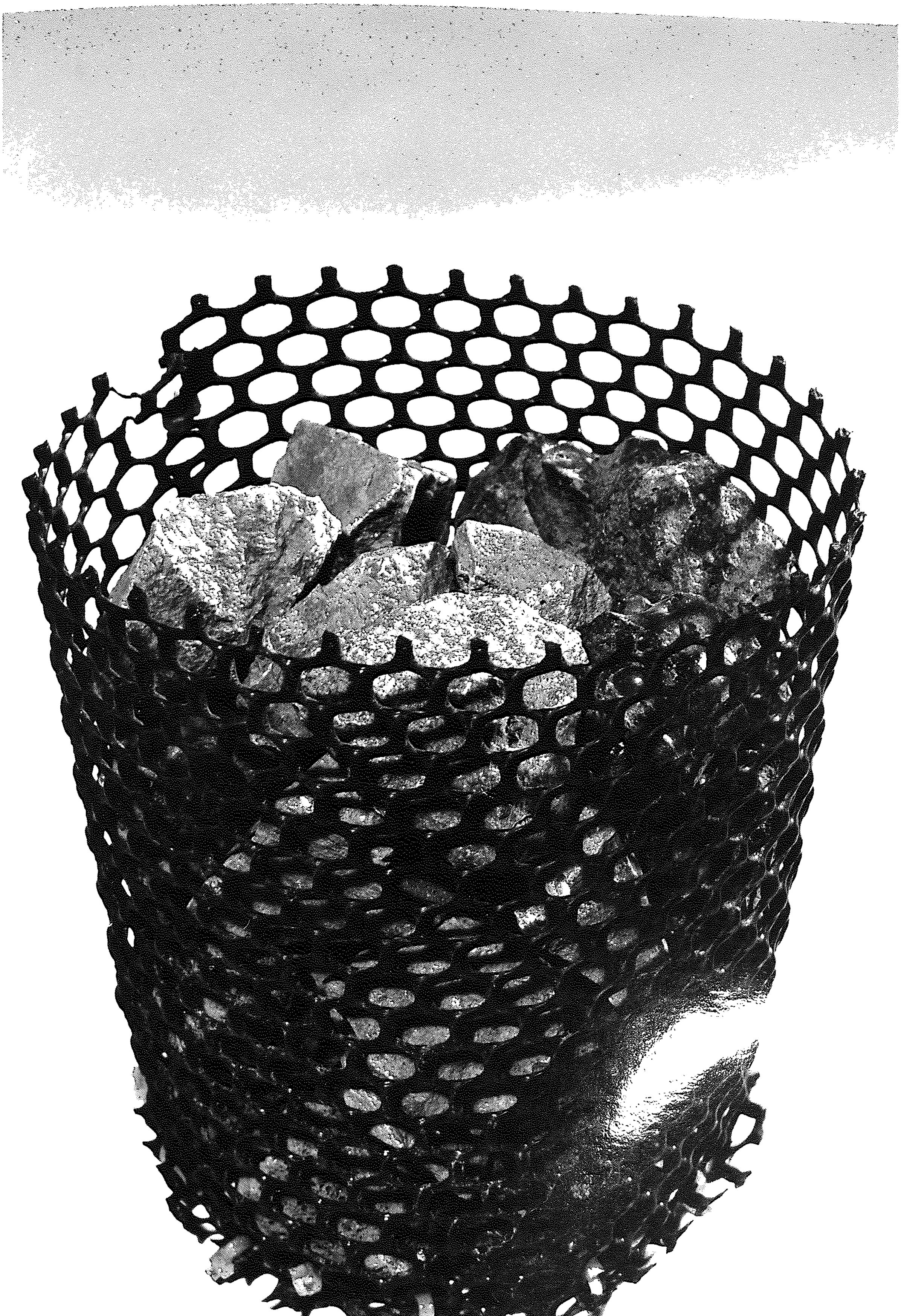


PLATE 4 SLAG BAG

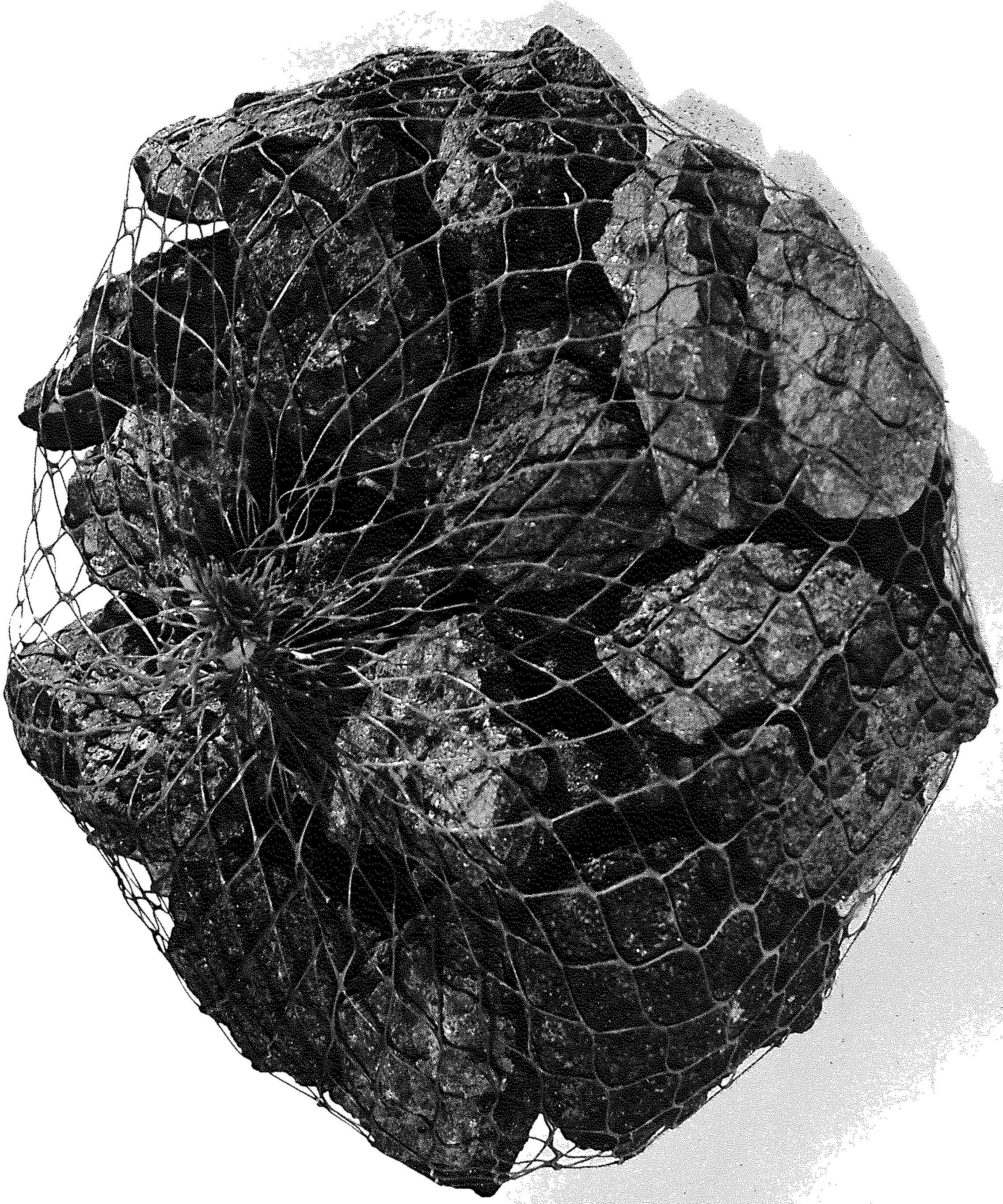


PLATE 5 BIOPAC UNIT

PLATE 6 BIOPAC BAG

PLATE 7 FLOCOR R BAG

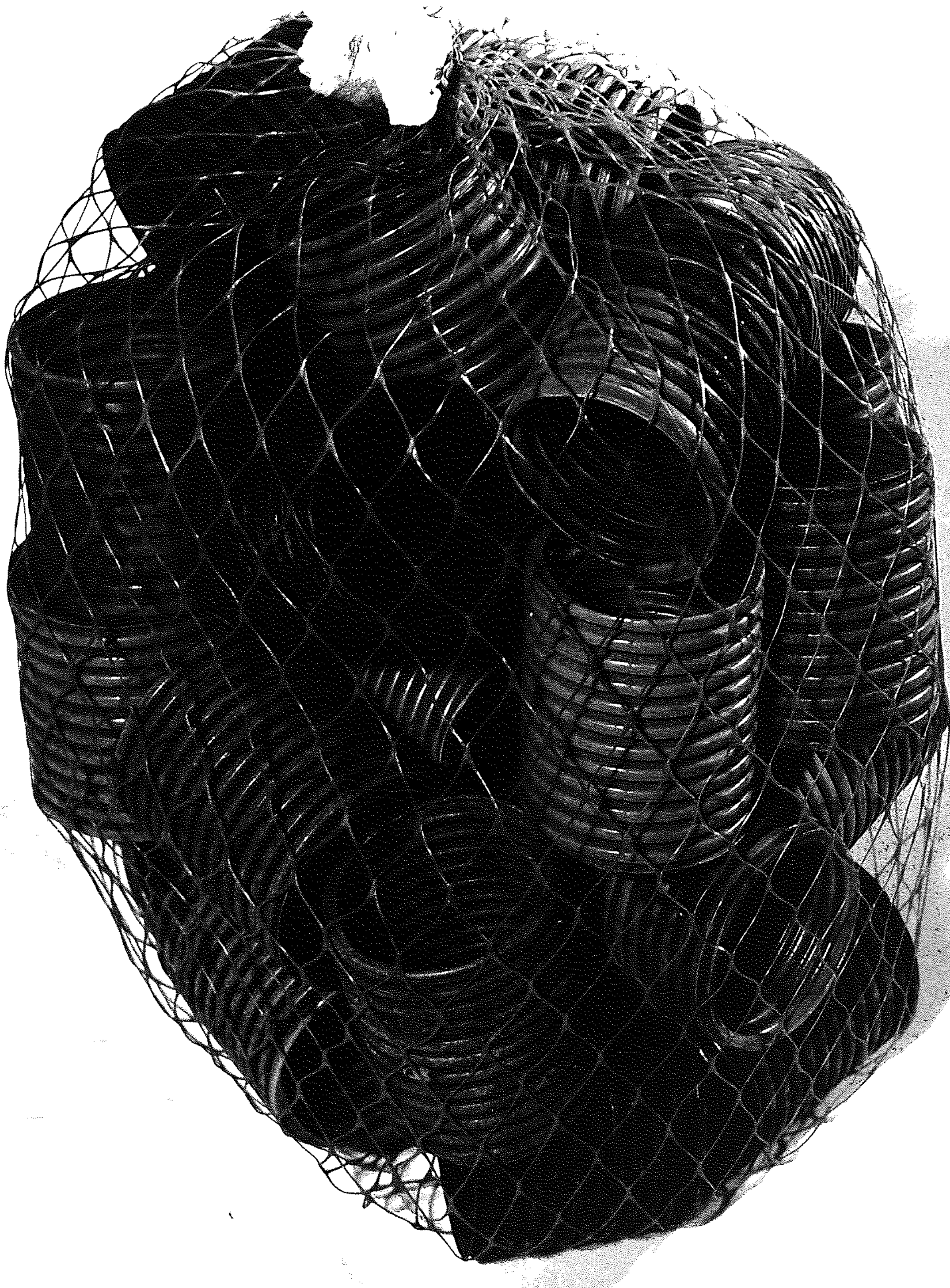
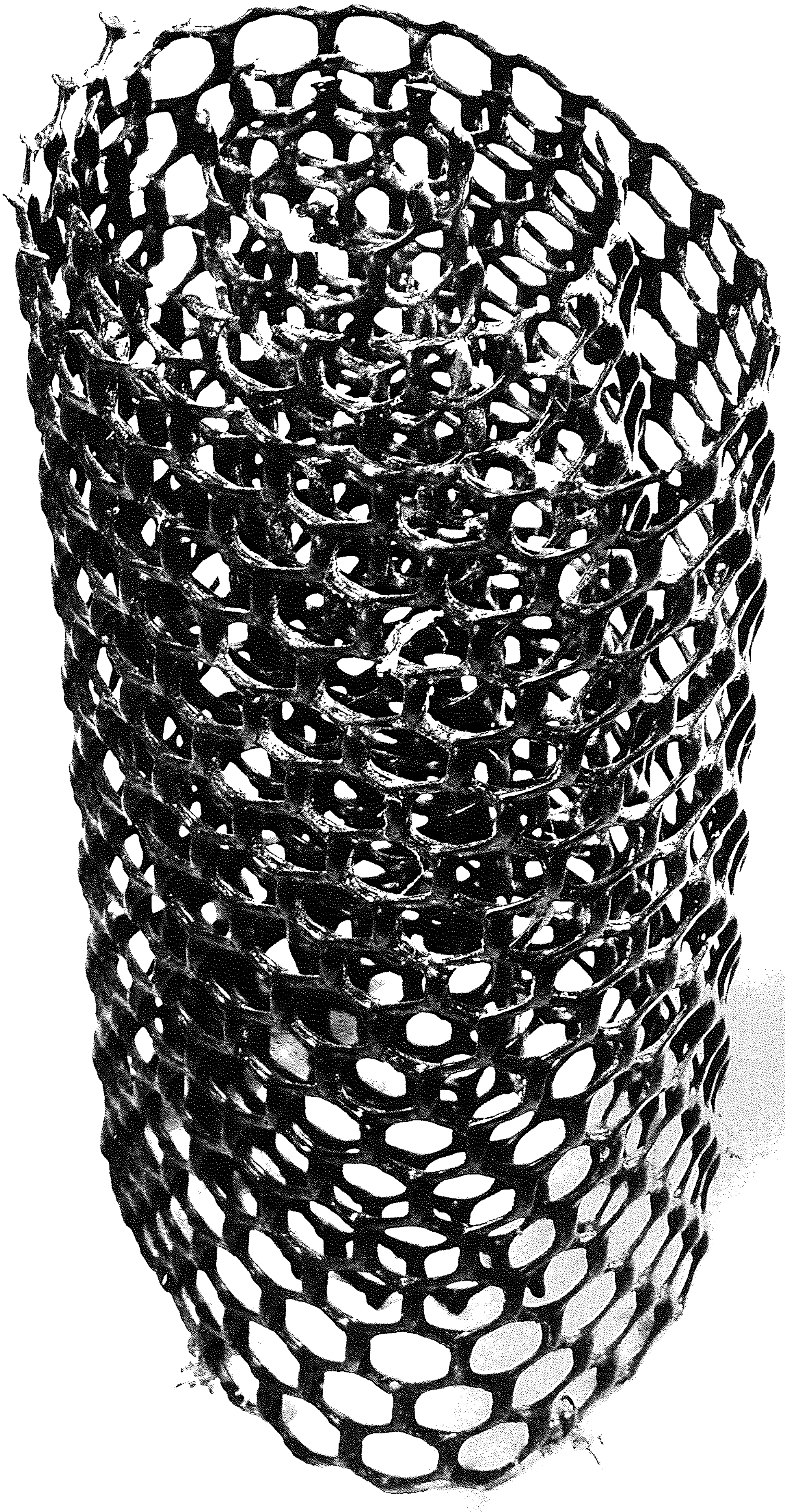


PLATE 8 SPIRAL SAMPLER



This was facilitated by using a hand net, placing it immediately downstream of the sampler and moving it under the sampler as this was lifted off the river bed, the sampler was then retrieved in the net. The whole sampler together with any animals in the net were placed in a plastic bag and sealed for transporting back to the laboratory.

3.153 Treatment and Analysis of Samples

In the laboratory the samples collected from the natural substratum were washed on a sieve of 250 micron mesh and the catch stored in 70 per cent industrial alcohol, after living Platyhelminthes and Oligochaeta had been removed and examined. The animals on the colonisation samplers were washed on to the sieve together with others found in the plastic bag. These were also stored in 70 per cent alcohol after initial examination. From these preserved samples the invertebrates were sorted, identified and counted.

Identification of macroinvertebrates was mainly to the specific level, since their taxonomy is now well documented (Table 3.2). Certain insect orders particularly the Coleoptera and Diptera are often difficult to identify to specific level and consequently reference was occasionally made to the generic or family levels. Initially it was anticipated to speciate most chironomids, however, this proved impossible due to identification problems and time taken to mount a specimen. As a consequence the majority of chironomids were taken to the sub-family/tribe level with selected examples taken to the genus/species level. Larvae were mounted for high power identification in Berlese's fluid after separating the head from the thorax, mounting

Table 3.2 Checklist of references used in the identification of benthic macroinvertebrates

MAJOR TAXA	SUB GROUPS	AUTHOR(S)
PLATYHELMINTHES		Reynoldson (1967)
OLIGOCHAETA		Brinkhurst (1971)
HIRUDINEA		Mann (1964)
CRUSTACEA		Gledhill *et al.* (1976)
PLECOPTERA		Hynes (1967)
EPHEMEROPTERA		Macan (1970)
TRICHOPTERA	General	Hickin (1967), Kimmins (1966)
	Polycentropodidae	Edington (1964)
	Psychomyiidae	Edington and Alderson (1973)
	Hydropsychidae	Hildrew and Morgan (1974)
	Rhyacophilidae	Mackereth (1954)
	Glossosomatidae	Mackereth (1956)
	Phryganeidae	Bray (1967)
	Sericostomatidae	Hiley (1972), Wallace (1977)
	Limnephilidae	Hiley (1976)
COLEOPTERA	General	Balfour-Brown (1940, 1950, 1958)
	Elminthidae	Holland (1972)
	Haliplididae	Rousseau (1919)
HEMIPTERA		Macan (1965)
MEGALOPTERA		Elliott (1977)
ODONATA		Gardner (1954)

Table 3.2 (cotd)

MAJOR TAXA	SUB GROUPS	AUTHOR(S)
DIPTERA	Tipulinae	Chiswell (1956), Brindle (1960)
	Simuliidae	Davies (1968)
	Chironomidae	Chernovskii (1949), Fittkau (1962), Bryce (1960), Bryce and Hobart (1972), Pankratova (1970), Cranston (1975, 1976, 1977)
MOLLUSCA	Gastropoda	Macan (1969)
	Bivalvia	Ellis (1962)
GENERAL CHECK LIST		Macan (1959), Mellanby (1963) Maitland (1977)

these apart at each end of a glass slide. Lundblads solution was used to dissolve muscle and clear the specimen.

Along similar lines, the oligochaetes were mounted on glass slides using polyvinyl lactophenol.

3.154 Plan of the investigations

The initial phase of the research programme involved placing the selected colonisation samplers in the River Tean and leaving them for immersion periods of 2 weeks and 6 weeks. The samplers were then re-set and left for a further 4 week and 8 week immersion period. Parallel investigations on the Checkley earth channels were also carried out involving three separate immersion periods of 4 weeks duration. The different types of colonisation samplers were not duplicated for each immersion period. At the end of each immersion period, three cylinder samples were taken from the riffles and bulked together, while a single heel-kick sample was also taken. In the pools only heel-kicks were taken, since grab sampling was unsuitable. In the Checkley channels, a single cylinder was taken. Table 3.3 gives a summary of the sampling regime.

Table 3.3 Macroinvertebrate Sampling Regime

Sampling Station	Samplers In	Samplers Out	Immersion Period
T_1R, T_1P, T_2R, T_2P	8.4.76.	21.4.76.	2 weeks
T_1R, T_1P, T_2R, T_2P	8.4.76.	19.5.76.	6 weeks
T_1R, T_1P, T_2R, T_2P	19.5.76.	16.6.76.	4 weeks
T_1R, T_1P, T_2R, T_2P	19.5.76.	14.7.76.	8 weeks
ERR, E50R, E75R	25.3.76.	21.4.76.	4 weeks
ERR, E50R, E75R	19.5.76.	16.6.76.	4 weeks
ERR, E50R, E75R	16.6.76.	14.7.76.	4 weeks

3.16 Results

3.161 Water quality

The physico-chemical data generated from the initial surveys are presented in Table 3.4.

With reference to the River Tean, the effects of the sewage discharge passing into the river produced elevated levels of most of the principle physico-chemical parameters, resulting in a deterioration in water quality. Of these parameters a slight drop in dissolved oxygen and a slight rise in BOD_5, suspended solids, ammonia, nitrates and phosphates indicated that the river at station 2 was only mildly polluted.

The water quality passing down the river water channel at Checkley was similar to that found at Tean 1 indicating that the quality of the water remained constant during transportation via the pumping systems. A progressive reduction in water quality through the other channels showed the effect of increasing the proportions of sewage effluent to river water ratios. Both the 50% and 75% channels were of a poorer quality than the water found at Tean 2.

3.162 Benthic macroinvertebrates

The detailed results of the macroinvertebrate studies are given in Annexe 1, Tables 1 - 7.

The numbers of macroinvertebrate species and the total numbers of organisms taken in the riffles and pools at the two stations on the River Tean above and below the effluent, after different immersion periods and by different sampling methods are presented in Figs. 3.3 -

Table 3.4 Mean, minimum and maximum physico-chemical conditions for the stations on the River Tean and Checkley channels

	STATIONS				
Parameter	T_1	T_2	ERR	E50R	E75R
Temperature °C	14.3 (11.5 - 16.5)	15.3 (12.0 - 18.5)	15.4 (12.8 - 18.5)	16.1 (12.8 - 18.5)	16.5 (14.0 - 19.0)
Susp. Solids mg. l^{-1}	5.9 (1.0 - 14.0)	11.6 (5.0 - 18.5)	5.7 (1.0 - 13.5)	15.9 (7.5 - 32.0)	17.9 (7.0 - 33.5)
Diss. Oxygen mg. l^{-1}	9.3 (5.9 - 13.3)	6.6 (3.7 - 8.9)	10.3 (7.0 - 14.0)	9.5 (8.1 - 11.0)	8.8 (8.1 - 9.8)
Diss. Oxygen % satn.	76.8 (61 - 88)	53.8 (39 - 79)	94.0 (87 - 103)	88.7 (73 - 102)	89.3 (84 - 100)
Chloride mg. l^{-1}	43.9 (36 - 59)	59.2 (53 - 72)	44.2 (35 - 61)	61.7 (52 - 69)	73.6 (61 - 90)
Alkalinity mg. l^{-1}	134 (100 - 151)	162 (130 - 177)	133 (92 - 151)	128 (95 - 139)	126 (100 - 145)
BOD_5 mg. l^{-1}	3.2 (1.1 - 7.0)	5.4 (4.75 - 6.1)	2.1 (1.8 - 2.8)	9.0 (4.2 - 15.9)	8.9 (2.9 - 14.2)
NH_3 - N mg. l^{-1}	2.2 (1.7 - 2.8)	2.4 (1.9 - 3.0)	2.2 (1.7 - 3.2)	2.3 (1.6 - 2.77)	2.7 (1.6 - 2.77)
NO_3 - N mg. l^{-1}	2.5 (1.3 - 5.1)	4.9 (2.5 - 7.7)	2.6 (1.5 - 5.1)	7.8 (4.8 - 12.0)	11.5 (5.2 - 16.0)
PO_4 - P mg. l^{-1}	0.75 (0.04 - 1.75)	3.3 (1.95 - 4.4)	0.74 (0.05 - 1.75)	4.8 (3.6 - 5.6)	7.5 (6.0 - 9.0)
Total Hdns. mg l^{-1}	239 (176 - 270)	288 (218 - 332)	242 (176 - 276)	250 (205 - 282)	247 (222 - 266)
Ca^{2+} Hdns. mg. l^{-1}	174 (148 - 184)	186 (152 - 202)	176 (144 - 192)	161 (116 - 178)	173 (142 - 168)
Mg^{2+} Hdns. mg. l^{-1}	64 (28 - 90)	102 (66 - 140)	66 (32 - 92)	89 (82 - 106)	85 (54 - 101)

3.6. The data for the three Checkley experimental channels after four weeks immersion are given in Figs. 3.7 - 3.9.

As a means of comparing the sampling methods faunistically in a qualitative investigation based on presence or absence, Sorensen's (1948) Quotient of similarity was calculated. The coefficient of similarity was obtained by using the formula:

$$S = \frac{2c}{a+b}$$

where a = no. of species in community A

b = no. of species in community B

c = no. of species common to both communities.

Total similarity between 2 samples, therefore, gives a value of unity, whereas total dissimilarity gives a value of zero.

In the River Tean riffles, similarity values were calculated comparing the species found on each type of colonisation sampler to the sum of the species collected by cylinder and heel-kick sampling (Figs. 3.3 and 3.5). In the River Tean pools this value was calculated between the colonisation samplers and the heel-kick (Figs. 3.4 and 3.6), while in the Checkley channels the cylinder sampler was used (Figs. 3.7 - 3.9). Inter-sample variability between the different types of colonisation samplers was not considered.

Considering the data from the riffle upstream of the effluent (T_1R)(Fig. 3.3), fewer species were taken by most of the colonisation samplers than by direct sampling (cylinder or heel-kick). Apart from the stone bag and spiral sampler which consistently collected

FIG.3.3 TOTAL NUMBERS OF SPECIES, NUMBERS OF ORGANISMS AND SORENSEN'S SIMILARITY COEFFICIENT, TAKEN BY EACH SAMPLING METHOD IN THE RIVER TEAN 1 RIFFLE.

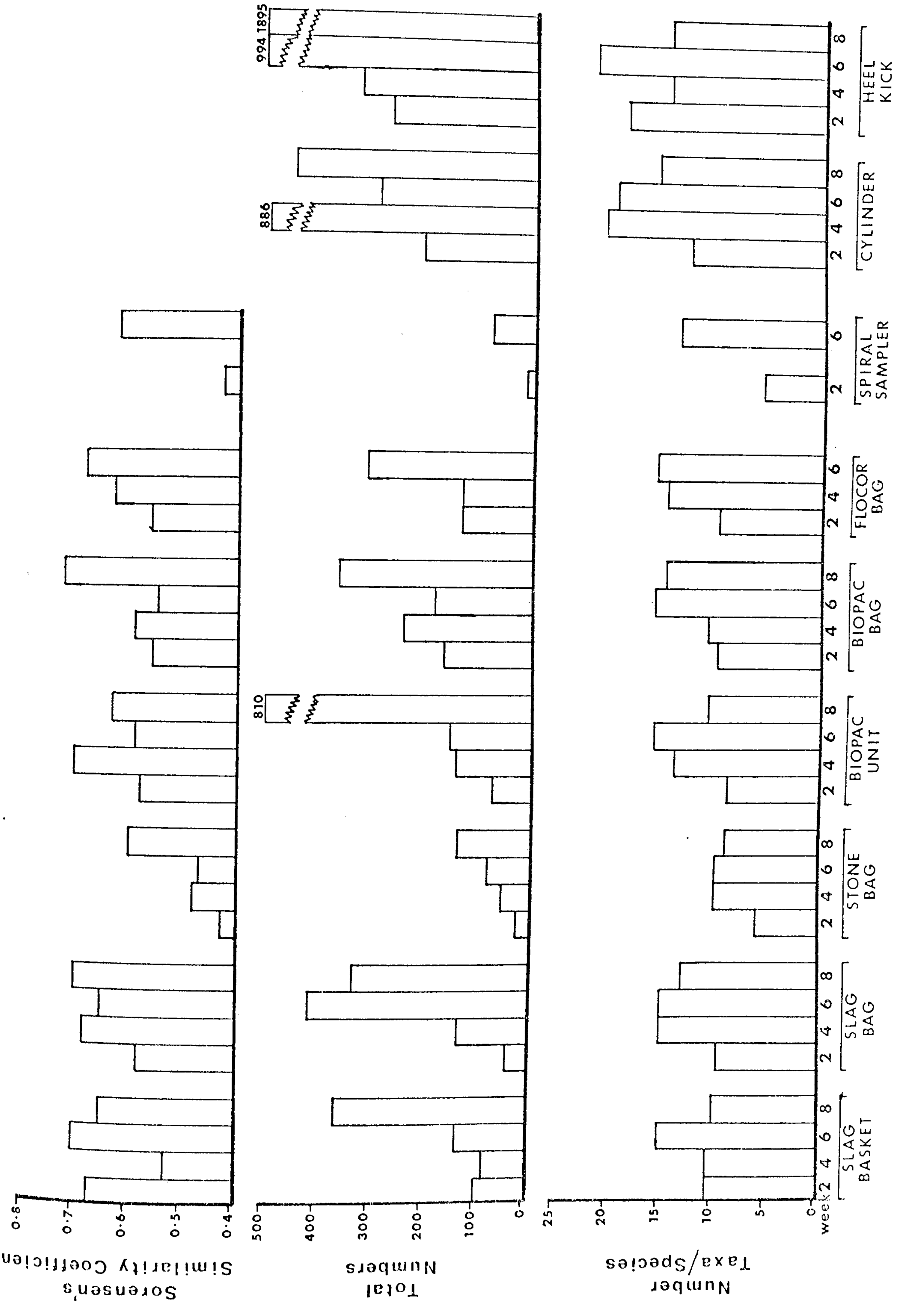


FIG.3.4 TOTAL NUMBERS OF SPECIES, NUMBERS OF ORGANISMS AND SORENSEN'S SIMILARITY COEFFICIENT, TAKEN BY EACH SAMPLING METHOD IN THE RIVER TEAN 1 POOL.

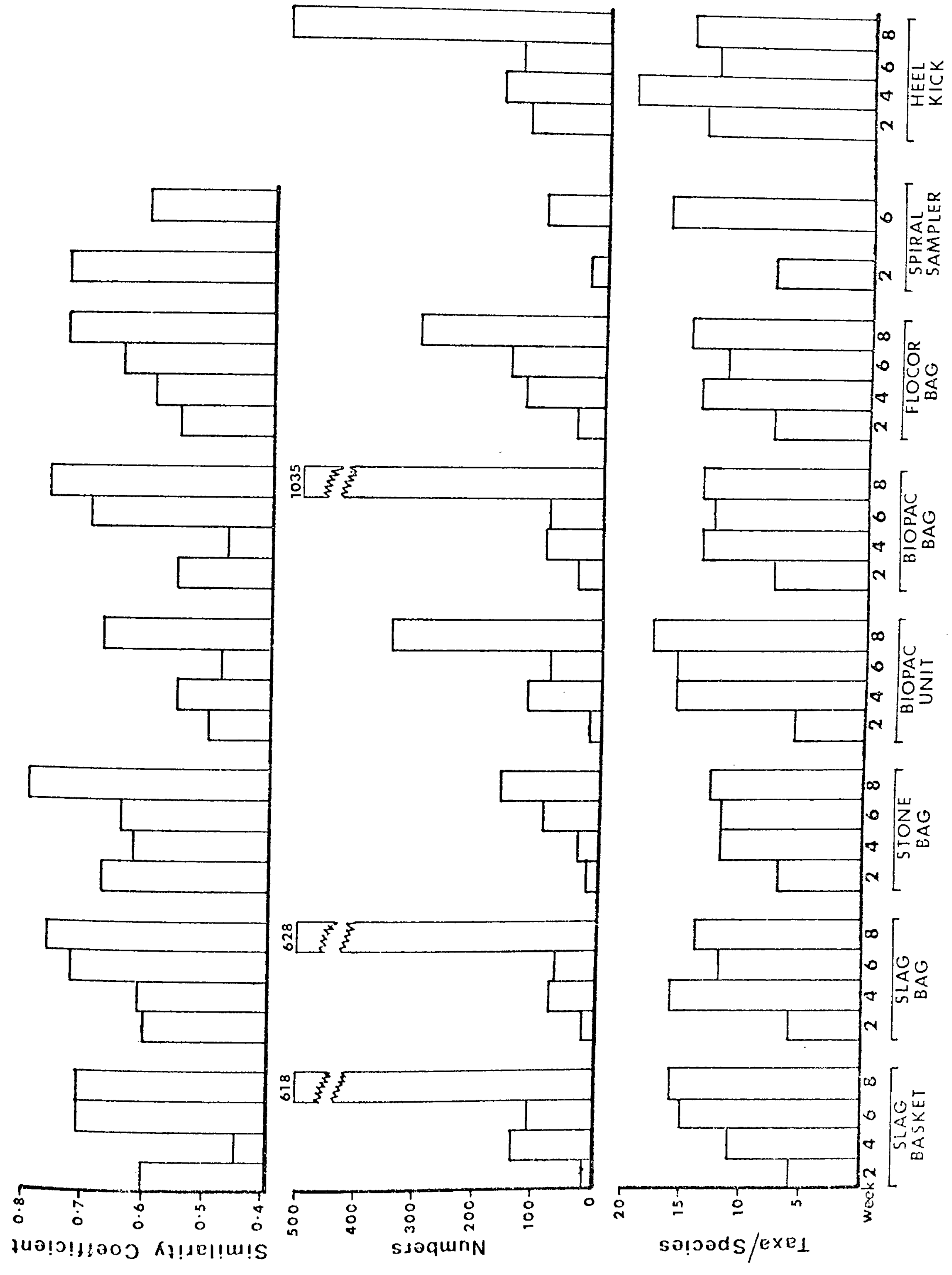


FIG.3.5 TOTAL NUMBERS OF SPECIES, NUMBERS OF ORGANISMS AND SORENSEN'S SIMILARITY COEFFICIENT, TAKEN BY EACH SAMPLING METHOD IN THE RIVER TEAN 2 RIFFLE.

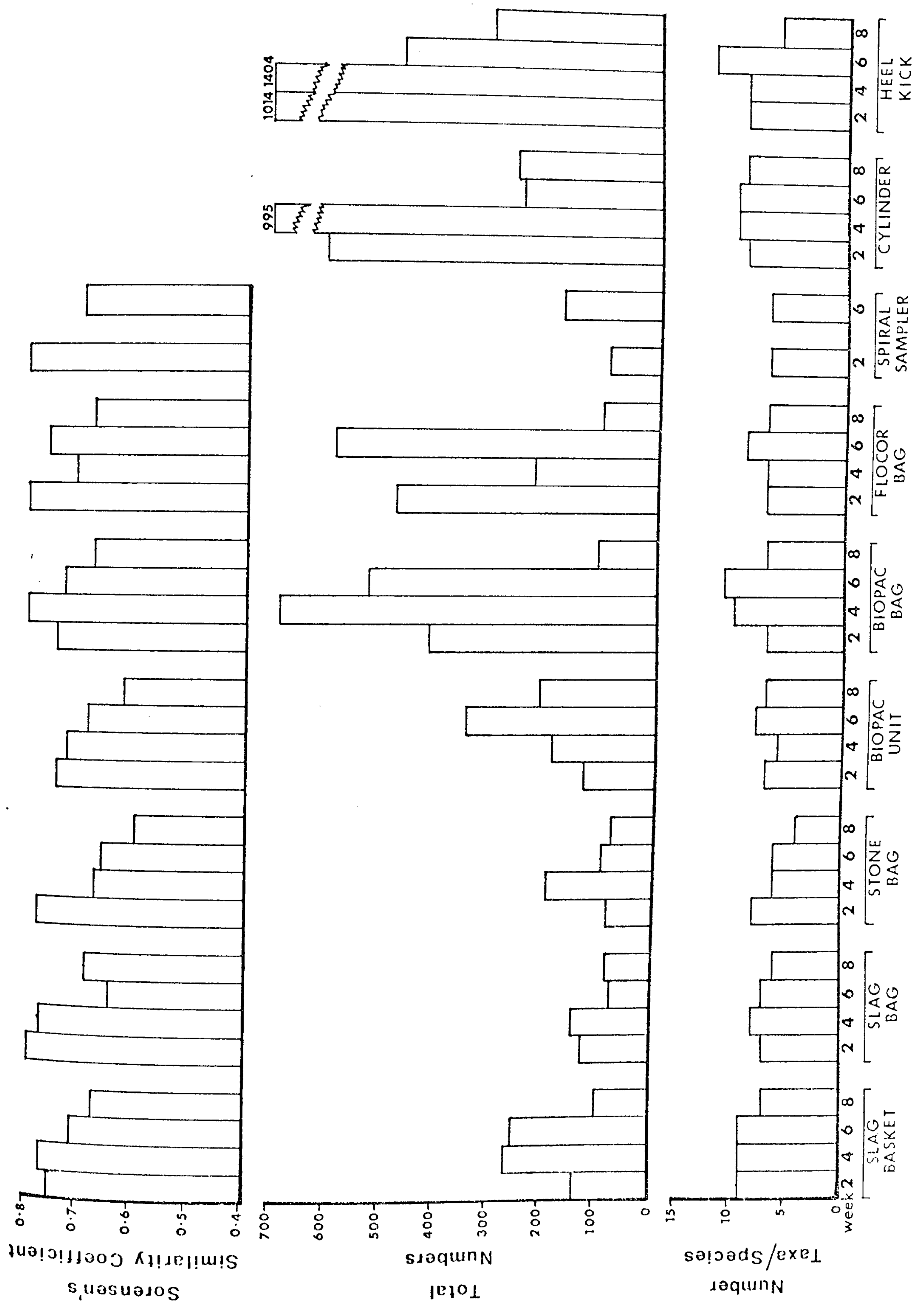
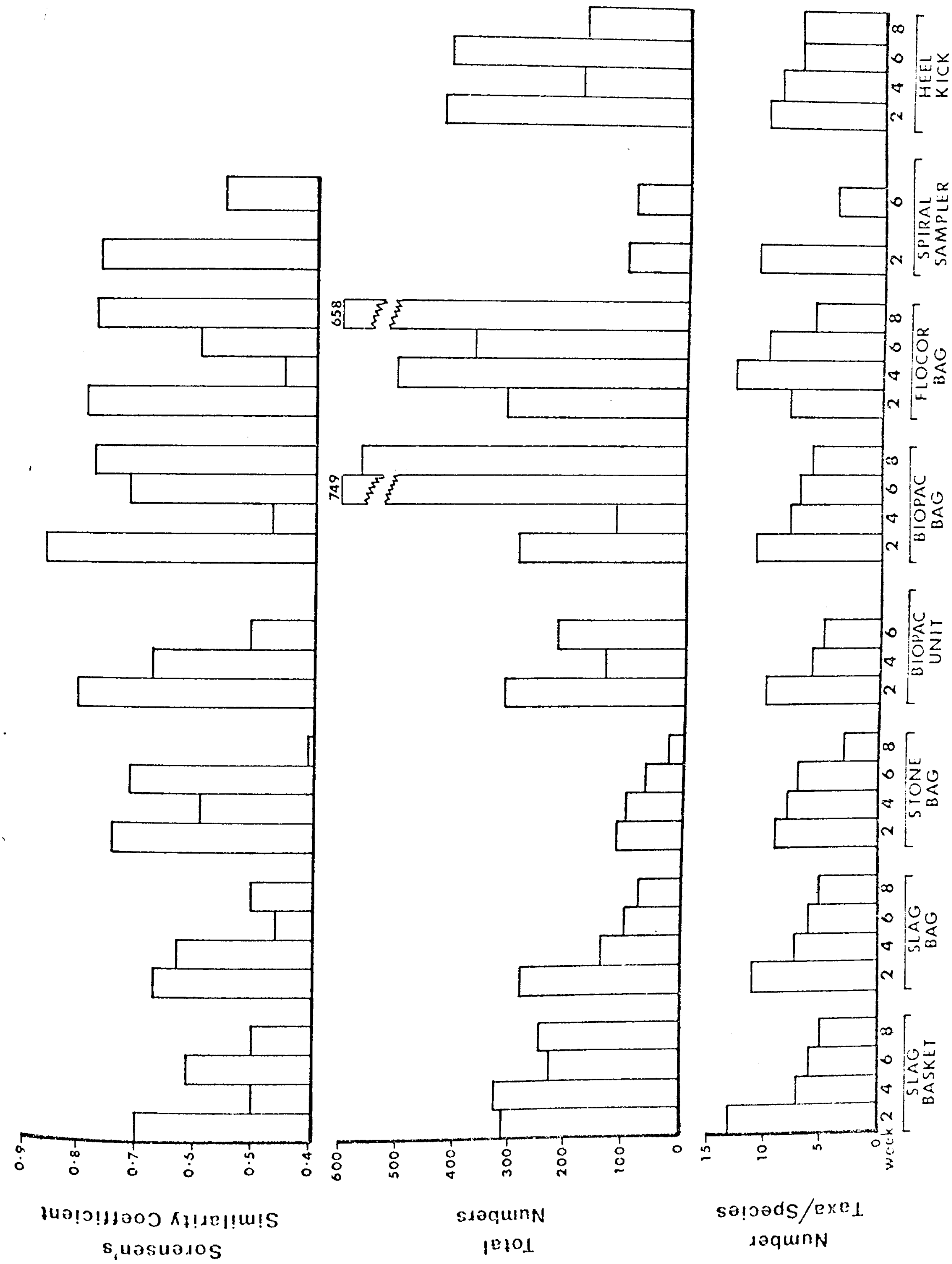


FIG.3.6 TOTAL NUMBERS OF SPECIES,NUMBERS OF ORGANISMS AND SORENSEN'S SIMILARITY COEFFICIENT,TAKEN BY EACH SAMPLING METHOD IN THE RIVER TEAN 2 POOL.



fewer species than the others, the other colonisation samplers showed no marked differences in performance. This was also apparent in the total numbers of individuals collected. In the riffle downstream of the effluent (T_2R)(Fig. 3.5) it is significant that all methods of sampling showed a reduction in the numbers of macro-invertebrate species indicating the deterioration in water quality. Although in some cases fewer species were taken on the colonisation samplers than by direct sampling, the differences were less marked than in the upstream riffle. Again the stone bag gave the lowest number of species.

From the results of the faunistic relationship based on Sorensen's coefficient at the upstream riffle, it is apparent that although the slag basket and slag bag had slightly higher similarity values than the other colonisation samplers, it is clear that 50% of the time a 60% similarity to the natural substrata was attained. In the downstream riffle, all the colonisation samplers had similar coefficient values compared to the natural substrata, but accomplished a 72% similarity 50% of the time, a rise again reflected by the deterioration in water quality.

In the river pool upstream of the effluent (T_1P)(Fig. 3.4) after an immersion period of four weeks or more, most of the colonisation samplers collected only slightly fewer species than by direct sampling and one, the Biopac unit, collected on average more species. In the pool downstream of the effluent (T_2P)(Fig. 3.6) all samplers collected fewer species than from the upstream pool indicating the deterioration in water quality. In this instance there was little difference between the different samplers and the direct sampling.

An examination of the Sorensen values from the upstream pool showed that most of the colonisation samplers had similar performances, although the Biopac unit on the whole produced lower similarity values compared to the natural substrata. This is not surprising since the Biopac unit collected a greater number of species than found by direct sampling. Again in the upstream pool 50% of the time all the colonisation samplers showed a 64% similarity to the natural substrata by direct sampling. In the downstream pool, this value was 68%.

The results from the natural river riffles were largely substantiated by the results from the riffle zones of the experimental channels at Checkley. In the river water channel (ERR)(Fig. 3.7) fewer species were taken by most of the colonisation samplers than by cylinder sampling, although on two occasions the slag bag and Biopac unit recovered larger numbers. The results from the 50% effluent channel (E50R)(Fig 3.8) reflect the deterioration in water quality, by a reduction in the number of species and an increase in the numbers of individuals compared to the river water channel. The plastic artificial media tended to collect larger number of species than by cylinder sampling. In the 75% effluent channel (E75R)(Fig. 3.9) most colonisation samplers collected fewer species than by cylinder sampling, although the results here were more anomolous.

Sorensen similarity quotients from all three experimental channels indicated that the slag bag and Biopac unit tended to collect species

FIG.3.7 TOTAL NUMBERS OF SPECIES,NUMBERS OF ORGANISMS AND SORENSEN'S SIMILARITY COEFFICIENT,TAKEN BY EACH SAMPLING METHOD IN THE CHECKLEY RIVER RIFFLE.

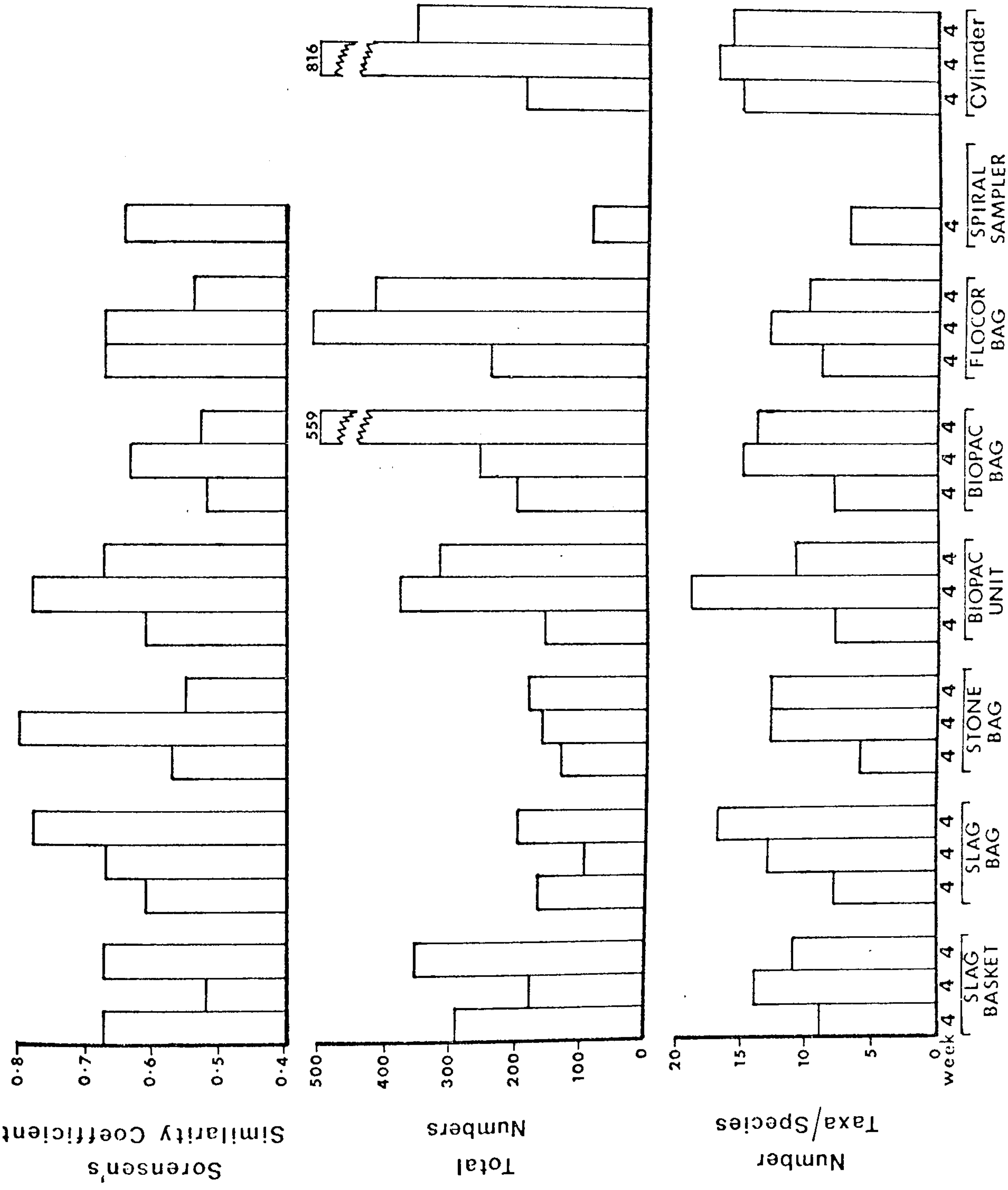


FIG.3.8 TOTAL NUMBERS OF SPECIES, NUMBERS OF ORGANISMS AND SORENSEN'S SIMILARITY COEFFICIENT, TAKEN BY EACH SAMPLING METHOD IN THE CHECKLEY 50% EFFLUENT RIFFLE.

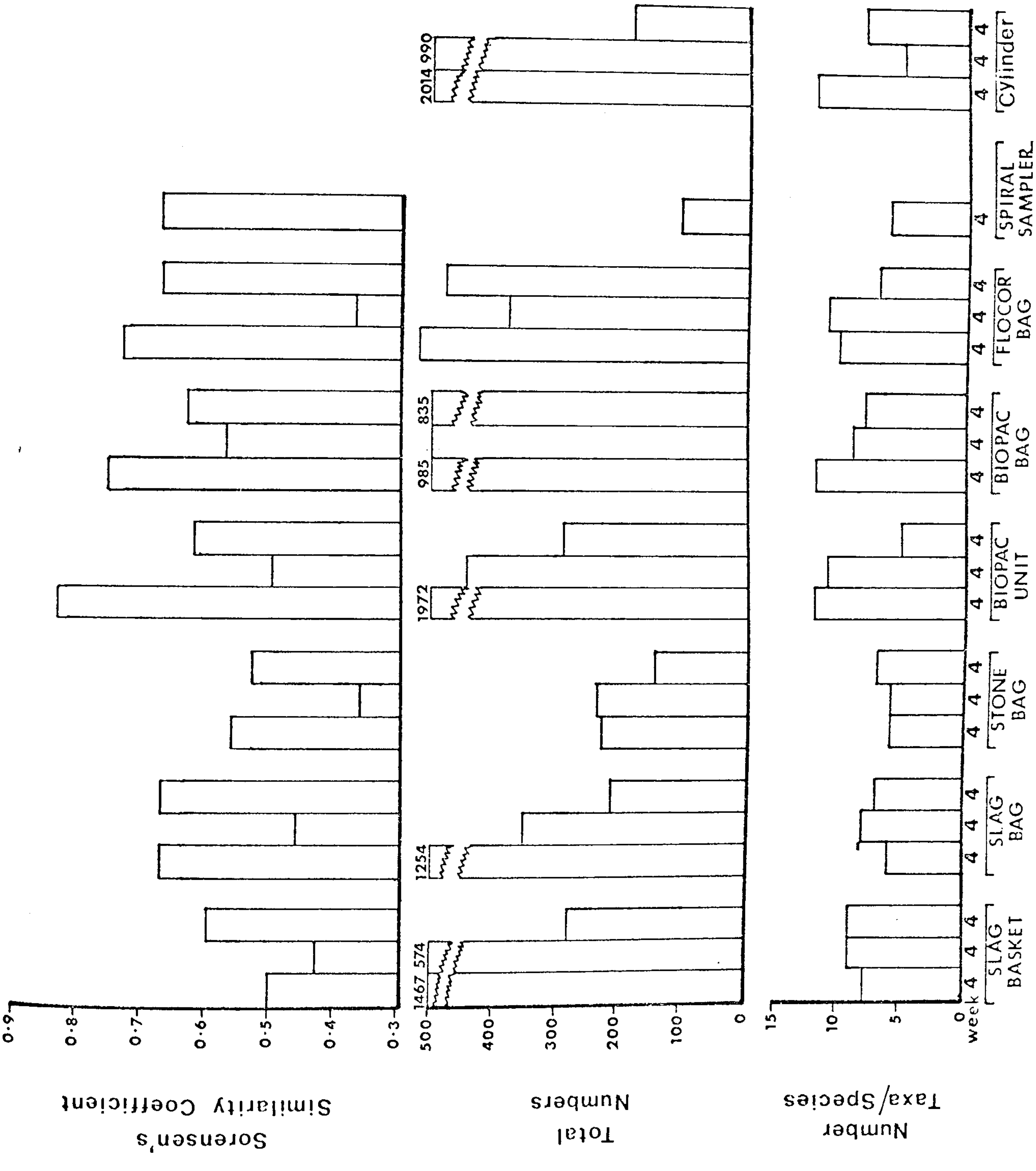
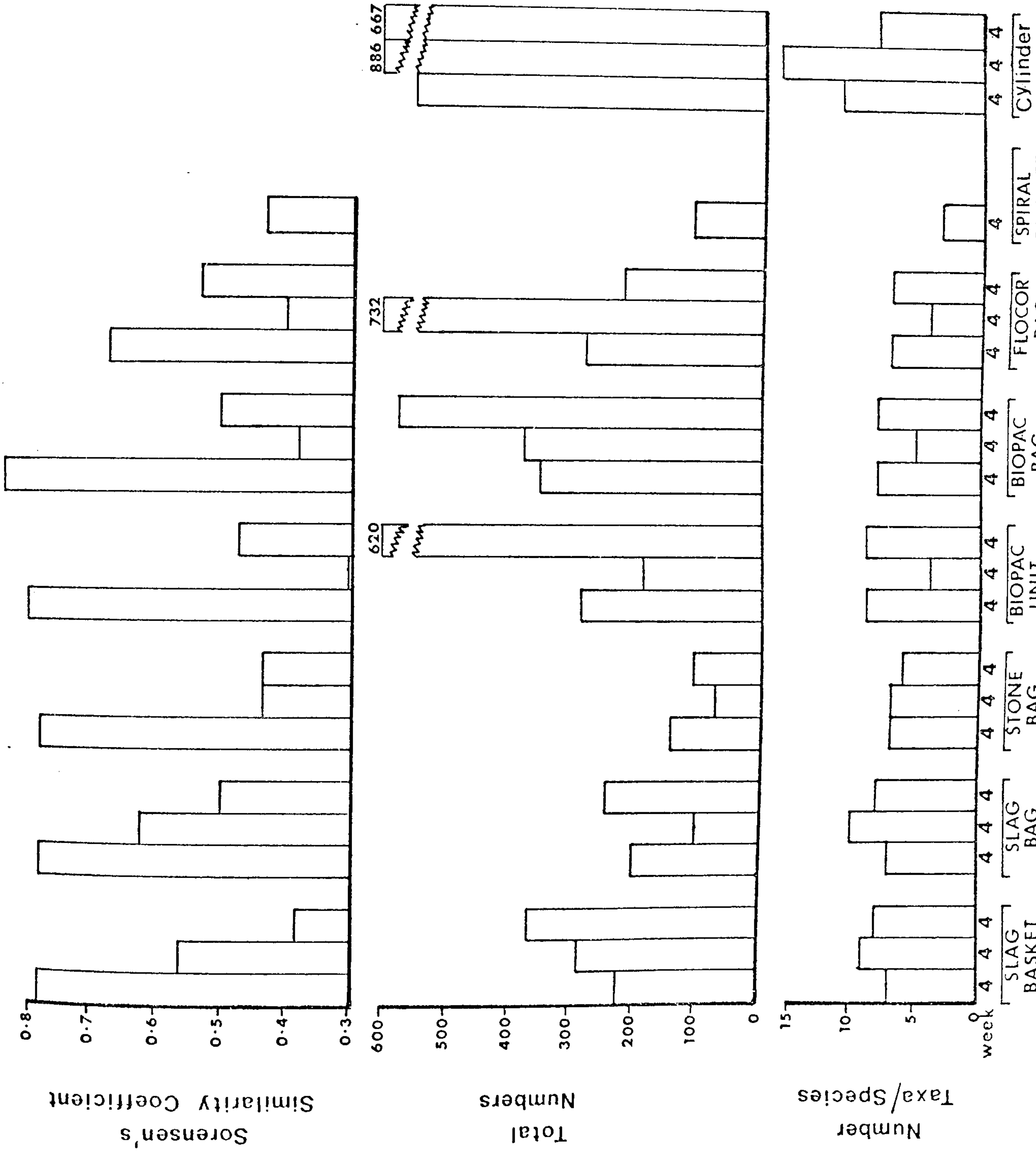


FIG.3.9 TOTAL NUMBERS OF SPECIES,NUMBERS OF ORGANISMS AND SORENSEN'S SIMILARITY COEFFICIENT,TAKEN BY EACH SAMPLING METHOD IN THE CHECKLEY 75% EFFLUENT RIFFLE.



similar to the cylinder sampler than the other colonisation samplers.

Fig. 3.10 evaluates the data of the 2, 4, 6 and 8 week immersion periods expressed as percentages of the total number of species taken by all of the sampling methods at the various stations in the River Tean. These percentages for the experimental channels are given in Fig. 3.11.

In the Tean 1 riffle the highest percentage of species were collected by the cylinder and heel-kick sampling techniques. These samplers showed no marked differences in performance and of the colonisation samplers the slag bag and Biopac unit collected slightly higher percentages than the others. In the downstream riffle again the direct sampling techniques proved the most successful.

In the upstream pool the Biopac unit tended to collect the greatest percentage of the common species over the different time periods studied, while in the downstream pool the flocor bag produced similar results. In the polluted pool excessive growths of the blanket weed Cladophora tended to entangle around some of the samplers, particularly the Biopac units and this resulted in irregular results.

From the River Tean work it is apparent that an immersion period of two weeks was not sufficient for colonisation by benthic invertebrates. It would appear that at least a four week period of immersion was necessary, although practical problems could arise as seen in the polluted pool.

The sampling methods studied in the Checkley channels further supplemented the data obtained from the River Tean riffles.

FIG.3.10 THE NUMBERS OF SPECIES TAKEN BY EACH SAMPLING METHOD, REPRESENTED AS A PERCENTAGE OF THE TOTAL NUMBER OF SPECIES FOUND AT EACH SAMPLING STATION FOR EACH IMMERSION PERIOD.

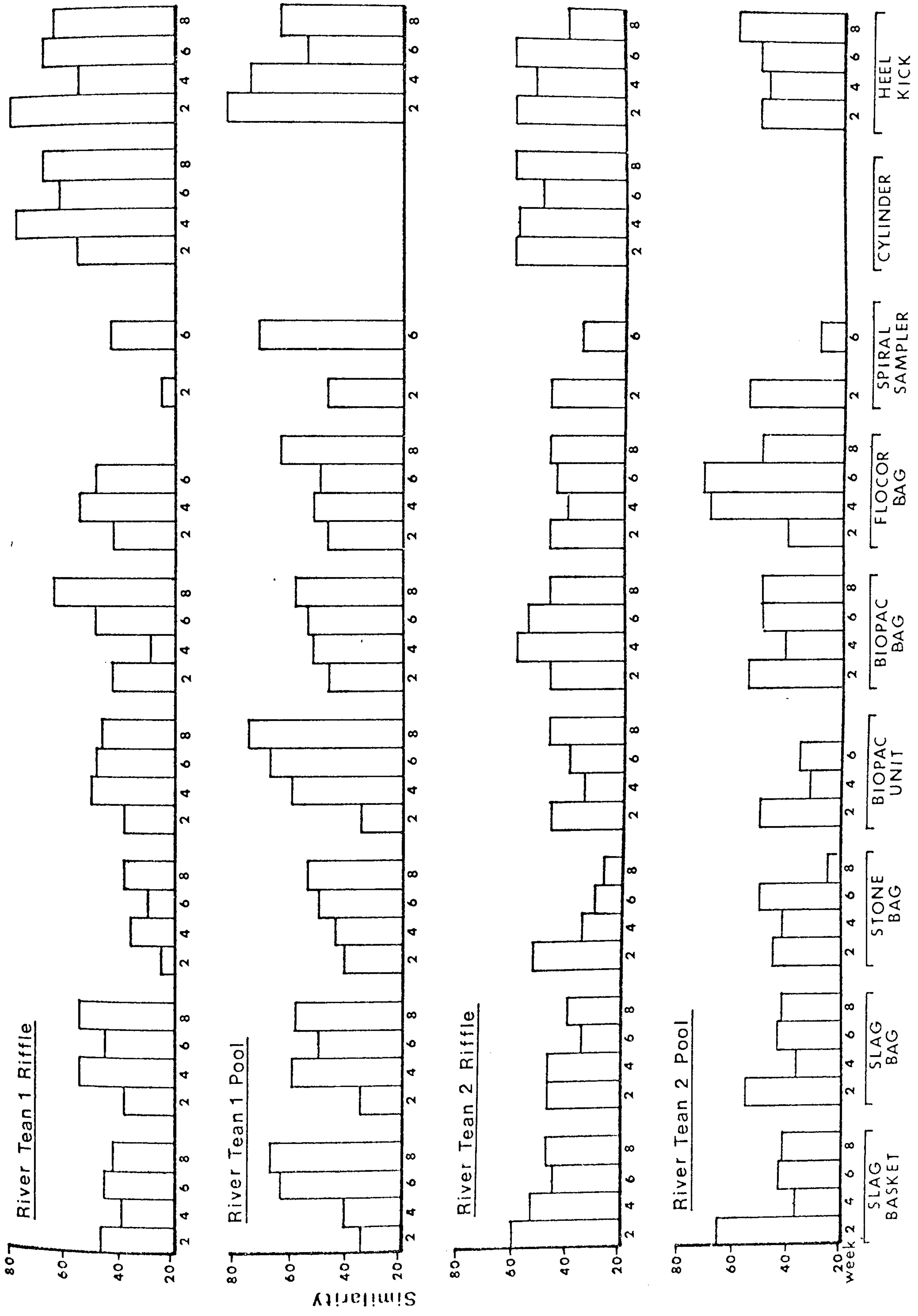
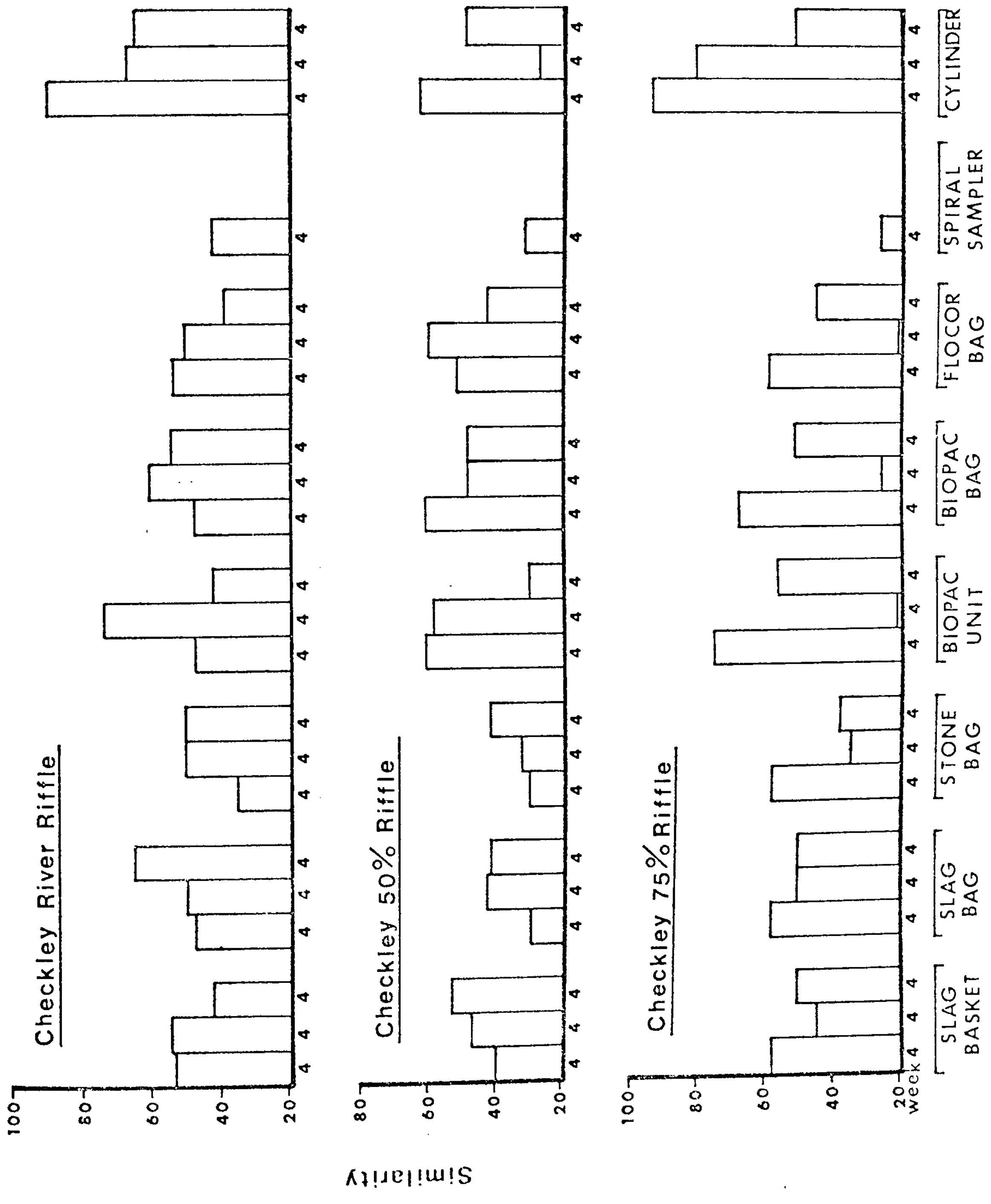


FIG.3.11 THE NUMBERS OF SPECIES TAKEN BY EACH SAMPLING METHOD, REPRESENTED AS A PERCENTAGE OF THE TOTAL NUMBER OF SPECIES FOUND AT EACH SAMPLING STATION FOR EACH IMMERSION PERIOD.



Generally, the cylinder sampler collected the highest proportion of the species present in all 3 channels. Of the colonisation samplers the Biopac unit collected the highest percentage of species.

The species composition of the major taxa, represented as a percentage for each method of sampling are shown in Figs. 3.12 - 3.14. In the upstream riffle (Fig. 3.12) the majority of the artificial substrata were dominated by Chironomidae, Mollusca and Gammarus pulex and occasionally Oligochaeta, Ephemeroptera and Trichoptera. The Chironomidae were composed particularly of the sub-family Orthocladiinae, a group mainly found on hard substrata, but as the silt content of the colonisation samplers accumulated with time it was noticeable that the Tanypodinae species increased in numbers. Although it is often difficult to show trends with such data, it is clear that chironomid numbers increase with lengthening immersion periods. Large populations of Mollusca appeared to colonise the colonisation samplers within a short period of time, particularly Ancylus fluviatilis and to a lesser extent Potamopyrgus jenkinsi. The former example is surprising since limpets would not appear to be capable of colonising exposed areas with great speed. Of the active swimming species found in the upstream riffle, G.pulex, Baetis rhodani and Ephemerella ignita should have colonised the colonisation samplers in the shortest period of time, however their numbers only built up slowly with respect to time. The direct sampling methods collected low numbers of chironomids (mainly Orthocladiinae) and oligochaetes (Tubifex tubifex and Lumbriculus variegatus) and large populations of A. fluviatilis, G.pulex, B.rhodani and E.ignita.

In the upstream pool (Fig. 3.12) the species present on the colonisation samplers were generally similar to those found in the riffle, although attached forms were now becoming more apparent. The Chironomidae were dominated by Tanypodinae species (*Macropelopia* spp., Pentaneurini) and to a lesser extent Tanytarsini (*Microspecta* spp.) and Chironomini. The Orthocladiinae group were severely restricted due to the presence of silt and organic debris. *G.pulex* was common on all of the colonisation samplers, but again was slow to colonise the different types of substrata under investigation. With an increase in deposited allochthonous material present in the pool reaches it was of no surprise to find *Asellus aquaticus* colonising the colonisation samplers. However, only small numbers were found together with water mites (Hydracarina) and *T.tubifex*. The Mollusca were dominated by *P.jenkinsi* reaching the highest densities after eight weeks immersion. This was related to a sudden increase in the number of juvenilles present in the pool. Compared to the riffle zone, the pool reach did not support a well developed Ephemeropteran population. Occasionally *E.ignita* was found on the colonisation samplers, probably related to drift from the upstream riffles.

The major difference between the riffle and pool zones was the development of a trichopteran population on the colonisation samplers. Of the cased caddis *Stenophylax* sp. was the dominant species, while uncased caddis were represented by *Polycentropus flavomaculatus* and *Cyrnus trimaculatus*. Leeches particularly *Glossiphonia complanata* and *Piscicola geometra* were also common on the colonisation samplers

FIG.3.12 PERCENTAGE COMPOSITION OF MAJOR TAXA REPRESENTING DIFFERENT SAMPLING METHODS AT DIFFERENT IMMERSION PERIODS, RIVER TEAN 1.

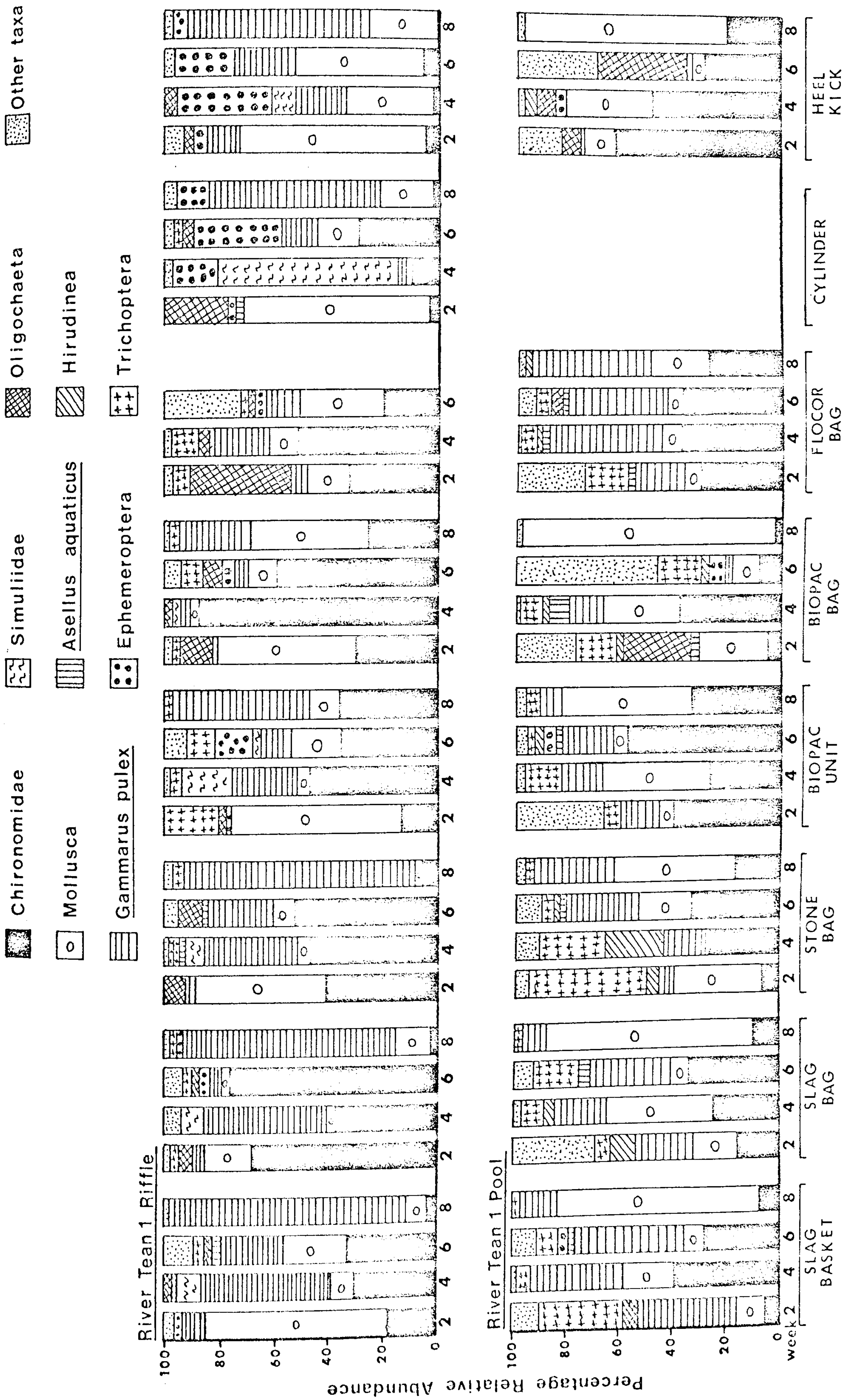


FIG.3.13 PERCENTAGE COMPOSITION OF MAJOR TAXA REPRESENTING DIFFERENT SAMPLING METHODS AT DIFFERENT IMMERSION PERIODS, RIVER TEAN 2.

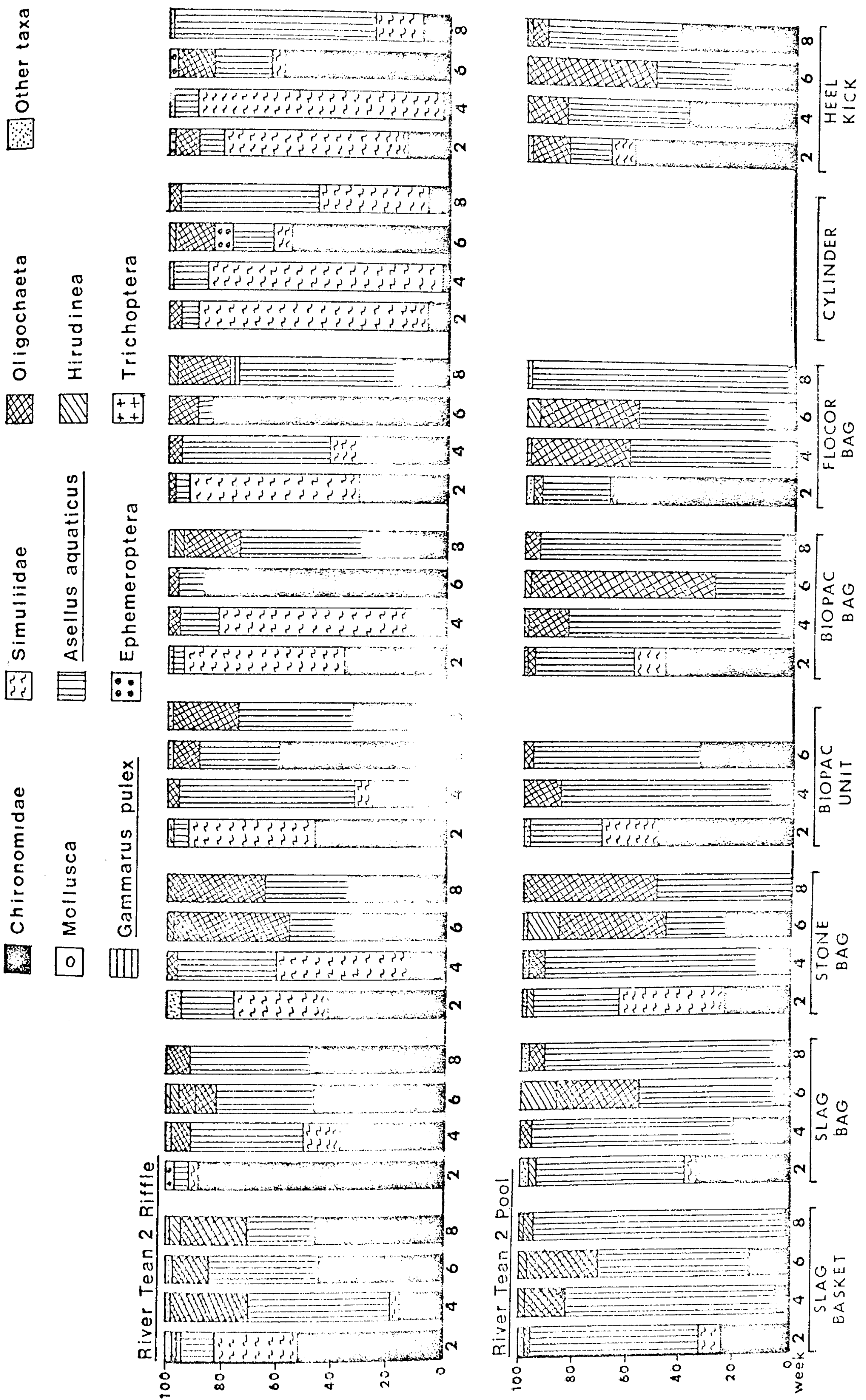
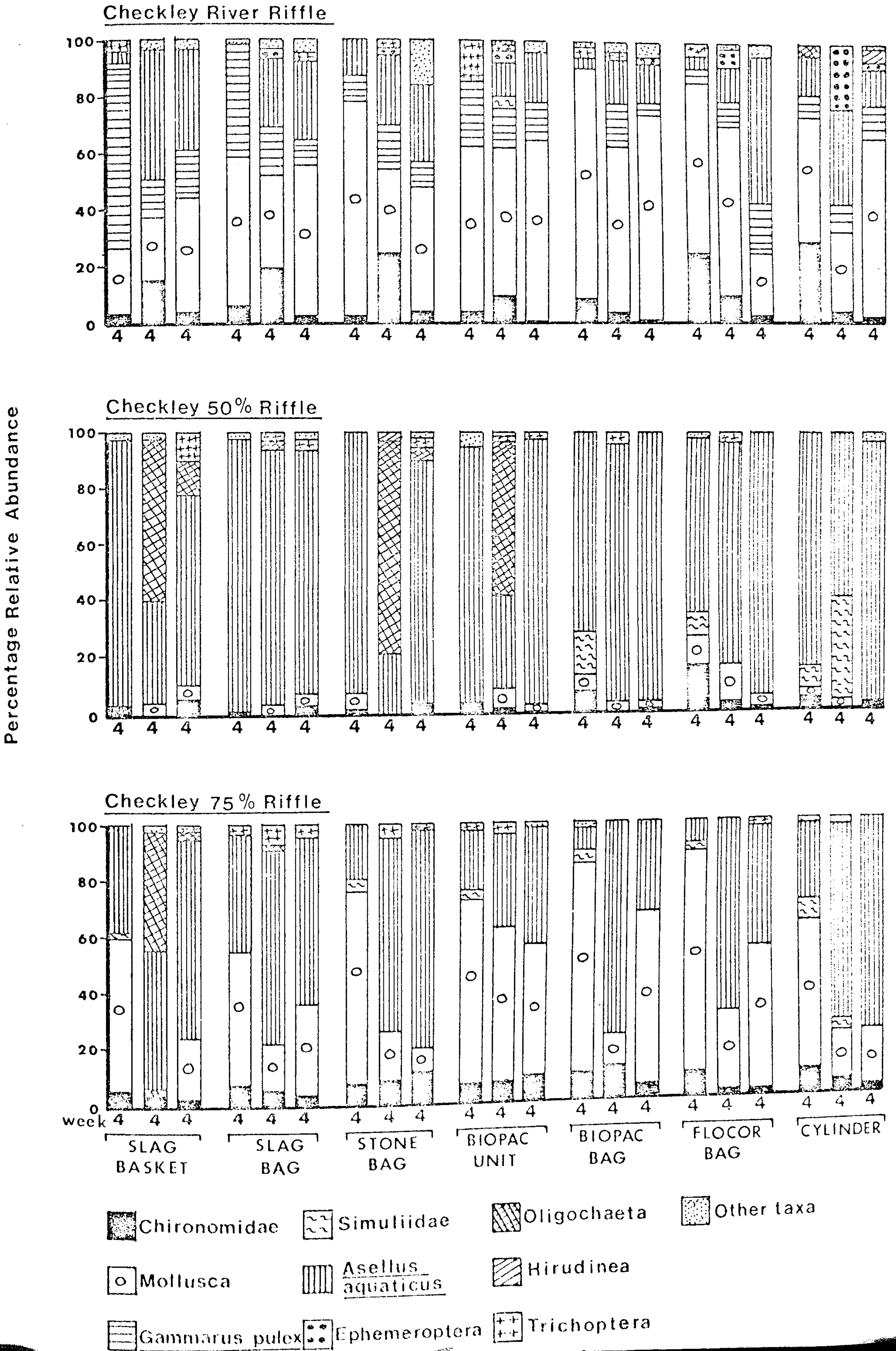


FIG.3.14 PERCENTAGE COMPOSITION OF MAJOR TAXA REPRESENTING DIFFERENT SAMPLING METHODS AT DIFFERENT IMMERSION PERIODS,CHECKLEY CHANNELS.



which provided a hard stable substratum for colonisation by these species.

In the riffle downstream of the effluent (Fig. 3.13) Tubificidae, *A.aquaticus* and Chironomidae were the dominant species on the colonisation samplers together with a population of blackfly larvae *Simulium ornatum* during the two and four week immersion periods. Chironomidae were dominated primarily by Tanytarsini and to a lesser extent Tanypodinae, *Prodiamesa olivacea* and Orthocladiinae. The community of the downstream riffle revealed by colonisation samplers and direct sampling was typical of mildly polluted rivers. It is significant that all methods of sampling showed a similar community structure, which was also apparent in the downstream pool.

In the polluted pool (Fig. 3.13) chironomids were again reduced in percentage composition compared to the riffle, as seen in the upstream reaches and numbers were mainly represented by the silt loving types Tanypodinae and Tanytarsini. Chironomini and *Chironomus riparius* were also represented on the artificial substrata. The dominant species in the pool were *A.aquaticus* and the Tubificid *Limnodrilus hoffmeisteri*. Again leeches were found on the colonisation samplers, particularly *Erpobdella octoculata* and *G.complanata*. In both the downstream riffle and pool, Trichoptera, Ephemeroptera and Mollusca were severely reduced.

In the Checkley experimental channels (Fig. 3.14) similar faunal patterns were observed between the colonisation samplers and cylinder sampler. The deterioration in water quality was marked by a reduction in the numbers of G.pulex, B.rhodani, E.ignita, P.jenkinsi and A.fluviatilis and an increase in the numbers of T.tubifex, A.aquaticus, S.ornatum, C.riparius and Lymnaea peregra.

3.17 Discussion

The results of this study emphasise the importance of using the same sampling method when carrying out biological surveillance investigations. Beak et al. (1973) pointed out that the evaluation of water quality by the use of benthic invertebrates requires the collection of comparable samples at different times and locations.

The use of introduced substrata in the study of benthic invertebrates has only occasionally been used to advantage as an experimental tool in monitoring environmental conditions in rivers. Although the technique is still in a developmental stage and needs further testing under a variety of conditions, the results of this exercise have demonstrated that colonisation samplers may be of some value in assessing water quality under specific conditions. On this premise, the selection of a suitable artificial media would be based not only on the community colonising it, but for management purposes, the practicality of such a sampler.

In terms of water management, the prime purpose of a colonisation sampler is not necessarily to represent the natural stream bottom substratum, but rather to present a standardised habitat at locations which may otherwise be different in terms of natural topography so that changes in the community colonising the standard substratum will reflect changes in water quality.

As stated earlier the selection of a sampling technique would be made on the basis of both practicability in the field situation and performance. In the riffles the largest number of species were recorded by direct sampling which is also the most practicable method of sampling this biotope. In shallow riffles problems were experienced with colonisation samplers such as siltation, fouling with drifting algal mats and vandalism. For these reasons it would seem logical to recommend direct sampling as a method of examining riffle biocoenoses. The colonisation samplers employed in the present study were found unsuitable for sampling the riffle habitat.

The colonisation samplers examined in this investigation presented several problems which had not been discussed adequately previously. In riffles and pools the number of organisms collected per sampler presumably depended on both the exposure period and the amount of detritus either in the form of food or silt, which accumulated in or around the samplers. There was a distinct difference in the quantity of silt which built up in the samplers from the riffle and pool habitats. In theory the riffle section of a river can accumulate silt, but it is the presence of spates which controls this accumulation. However, the colonisation samplers tended to trap and shelter this material, resulting in increased levels with time. In the pools this deposition was less dramatic and levels in the colonisation samplers built up slowly. The burying of samplers, particularly in the riffles, removed any control of substratum type and surface area. In addition the accumulation of silt in buried and partially buried samplers significantly increased the laboratory sorting time per sample. It was evident that colonisation by Chironomidae was related to silt accumulation, a phenomenon reported by other workers Anderson and Mason (1968), Simmons and Winfield (1971), Mason _et al._ (1973), Rosenberg and Wiens (1976), and Roux _et al._ (1976). Large numbers of Trichoptera, although not associated with silt, have also been reported previously (Anderson and Mason, 1968; Mason _et al._, 1973; Roby _et al._, 1978).

The one advantage of using colonisation samplers is to standardise the biotope sampled. At some 'riffle' sampling stations several biotopes may be present and if these differ at the different stations being compared, this will affect the results regardless of any differences in water quality. This effect can, however, usually be minimised by the selection of similar biotopes at each station rather than collecting from all the biotopes available at each station.

In the pools however, some colonisation samplers performed equally well as direct sampling in terms of the numbers of species collected. Although in these preliminary studies the pools used were relatively shallow and could be sampled directly by 'kick-heeling' in waders, the direct sampling of the depositing substratum in deeper waters is much more difficult. It was therefore concluded that for pool reaches and deeper rivers difficult to sample directly, the use of colonisation samplers should be considered.

Of the colonisation samplers tested the spiral sampler was abandoned after the 2 and 6 week test as being impracticable as it silted up too readily and was structually unstable. The others were all of similar performance, with the exception of the stone bag and it was on the basis of practical experiences with the samplers in pool conditions that the Biopac unit and the slag bag were selected for further testing.

3.2 Further tests with colonisation substrata for benthic invertebrate sampling in pool reaches

3.21 Objectives

Before using the colonisation samplers selected from Phase 1 of the investigations on a country-wide survey further validation tests were carried out on the two media selected. Biopac units and slag bags were studied to determine:

1. the sequence of colonisation of aquatic macro-invertebrates, and
2. their assessment of water quality in Midland rivers.

3.22 Methods

The two sampling stations T_1P and T_2P on the River Tean above and below the sewage effluent were again used for the intensive part of this study (Fig. 3.1). Initially in the pools at each station nine Biopac units and nine slag bag samplers were positioned on to the substratum 1m. apart using 8 mm. steel rods. To provide replicate samples three of each sampler were removed after an immersion period of 2, 4 and 6 weeks. At the end of each sampling period a 30 second heel-kick sample was taken from the pool. For comparison with direct sampling techniques the associated riffles of the same water quality (T_1R, T_2R) were also sampled using the Aston cylinder sampler. Three samples were taken across the width of the river, i.e. left side, middle and right side. In addition to this investigation, the Rivers Dove and Blithe in Staffordshire were also sampled using a single Biopac unit and slag bag in their pools. The samplers were immersed for a period of one month and again three cylinder samples were taken from associated upstream riffles for comparison. The River Dove (SK 113338) and River Blithe (SK 016358), although good quality rivers were selected to test the effect of fast flows, silting and vandalism on the efficiency of the artificial samplers.

As a final conclusion to this work, the two stations on the River Tean were sampled every month after the sequential study until the start of the Phase 2 work in March, 1977. The sampling programme is summarised in Table 3.5.

Table 3.5 Macroinvertebrate Sampling Regime

Sampling station	Samplers In	Samplers Out	Immersion Period
T_1P, T_2P	11.8.76.	25.8.76.	2 weeks
T_1P, T_2P	11.8.76.	8.9.76.	4 weeks
T_1P, T_2P	11.8.76.	27.9.76.	6 weeks
Dove, Blithe	11.8.76.	8.9.76.	4 weeks
T_1P, T_2P	27.10.76.	25.11.76.	4 weeks
T_1P, T_2P	25.11.76.	22.12.76.	4 weeks
T_1P, T_2P	22.12.76.	19.1.77.	4 weeks

3.23 Results

The detailed results of the macroinvertebrate studies are given in Annexe 1, River Tean sequential study (Tables 8 - 9) River Dove and Blithe (Table 10) and River Tean monthly studies (Tables 11 - 12).

3.231 River Tean sequential study

Although the sequential testing was interfered with by the loss of the six weeks samples due to adverse weather conditions, the remaining results are presented in Table 3.6

From the results it appears that the Biopac units require a period of four weeks immersion in good quality water to collect a representative fauna, while slag bags appear to stabilise more rapidly in such waters. Faunal colonisation trends similar to the initial pool investigations were again observed. Chironomid numbers tended to build up slowly in the Biopac units, but were more abundant in the slag bags. Trichoptera favoured the Biopac units together with P.jenkinsi. The large numbers of P.jenkinsi after only two weeks would tend to confirm Heywood and Edwards' (1962) findings that this mollusc is an extremely active species, travelling at speeds up to 3 cm. min^{-1}. The presence of G.pulex, A.fluviatilis and L.peregra on both substrata again confirms previous findings.

The Biopac units stabilised more rapidly in the poorer quality water below the effluent. This was also observed in the earlier tests (Fig. 3.6). Both samplers again reflected the different water qualities by the number of species collected.

Replication of the samplers increased the number of species taken in both the upstream and downstream stations. The standard error calculated for the colonisation samplers showed that there was little difference between the two sampling methods. This value decreased below the effluent due to decreased species diversity. Direct riffle sampling also emphasised this trend. Due to the small numbers of some

		POOL							ASSOCIATED RIFFLE		
		BIOPAC UNIT			SLAG BAG			HEEL KICK	CYLINDER		
STATION	IMMERSION PERIOD (Weeks)	TOTAL NUMBER (3)	MEAN (3)	STANDARD ERROR (3)	TOTAL NUMBER (3) * (2)	MEAN (3) * (2)	STANDARD ERROR (3) * (2)	TOTAL NUMBER (30 sec)	TOTAL NUMBER (3)	MEAN (3)	STANDARD ERROR (3)
NUMBER SPECIES											
TEAN 1	2	13	8.3	0.33	17	12.0	0.58	12	25	13.3	2.4
	4	15	11.0	1.5	16	10.7	0.33				
	6				14*	10.0*	1.0*		20	13.0	2.0
TEAN 2	2	8	4.7	0.33	8	5.3	0.67	6	14	8.0	1.15
	4	7	4.3	1.2	6	3.0	0.58				
	6								11	7.0	1.0
TOTAL NUMBERS											
TEAN 1	2	1215	405	49.2	550	183	27.7	449	1109	370	47.2
	4	1279	426	122.8	373	124	7.8				
	6				372*	186*	2.0*		897	299	34.2
TEAN 2	2	1755	585	120.0	2082	694	99.1	319	1585	528	144.0
	4	416	139	15.1	1089	363	29.9				
	6								666	222	67.7

Table 3.6 River Tean Sequential macroinvertebrate data

species found on the colonisation samplers, it would seem that at least three samples need to be taken.

3.232 River Blithe and River Dove study

In the River Blithe the Biopac unit collected 18 species after four weeks immersion and collected a similar number of species to those found by direct sampling of the riffle. Although three cylinder samples were taken, yielding 27 species, 13 of these were similar to the Biopac unit. The slag bag was less effective in this river, collecting 14 species. Of these 11 were similar to the cylinder and Biopac unit. Of the major groups found on the Biopac unit, the Hirudinea, Trichoptera, Coleoptera and Mollusca were particularly common.

In the River Dove where the samplers for test purposes had been positioned in a deep fast flowing stretch (velocity $>$ 1 m. sec.$^{-1}$), particular problems were encountered. The Biopac unit was found to have been forced up its anchoring rod so as to be situated above the bed of the river, where it had collected large amounts of drifting *Ranunculus*. This may have been the cause of the low number of species taken at this station.

3.233 River Tean monthly studies

The sampling programme using the two types of colonisation samplers was continued in the River Tean for three months using a four-week immersion period. The results of this investigation are summarised in Table 3.7. Although the colonisation samplers collected a similar number of species after four weeks immersion in the upstream pool, the mean of three Biopac units was always higher than three slag bags.

		POOL							ASSOCIATED RIFFLE		
		BIOPAC UNIT			SLAG BAG			HEEL KICK	CYLINDER		
STATION	IMMERSION PERIOD (Weeks)	TOTAL NUMBER (3)	MEAN (3)	STANDARD ERROR (3)	TOTAL NUMBER (3)	MEAN (3)	STANDARD ERROR (3)	TOTAL NUMBER (30. sec)	TOTAL NUMBER (3)	MEAN (3)	STANDARD ERROR (3)
NUMBER SPECIES											
TEAN 1	4	14	11.0	0.58	13	9.0	0.58	17	18	12.0	28.9
	4	13	9.3	0.88	12	5.7	1.45	12	17	14.0	12.4
	4	14	9.7	1.2	13	6.7	0.88	18	11	9.0	0.0
TEAN 2	4	13	11.3	0.88	10	8.0	0.58	10	15	9.3	0.88
	4	13	8.0	1.15	6	3.7	0.33	10	13	10.7	0.33
	4	11	8.3	1.33	7	5.3	0.33	6	13	8.0	1.53
TOTAL NUMBERS											
TEAN 1	4	281	94	12.1	130	43	5.8	245	497	166	39.4
	4	181	60	14.2	141	47	34.0	32	292	97	5.8
	4	203	68	24.5	74	25	6.2	216	272	91	19.9
TEAN 2	4	997	332	115.0	731	244	40.2	1280	1229	410	157.0
	4	891	297	24.0	680	227	97.4	1013	1003	334	38.7
	4	1121	374	33.1	442	147	18.2	2015	675	225	61.9

Table 3.7 River Tean Macroinvertebrate data taken after a succession of monthly immersion periods

This was also the case in the downstream pool where the Biopac units generally collected a larger number of species. Similar standard errors to those given in Table 3.6 were again observed, with higher values recorded upstream.

Faunal trends related to water quality were again observed together with temporal variations in the species composition. P. jenkinsi numbers decreased on the onset of winter together with numbers of L. peregra and Trichoptera in the upstream pool. In general most pool species were present throughout the entire year.

Downstream the numbers of Chironomidae were reduced during the winter months, although the Tanypodinae were the dominant group.

3.24 Discussion

The animals recolonising an area of denuded stream substratum are thought to come from four main sources:

(a) drift,
(b) upstream migration of benthos within the water,
(c) migration from within the natural substratum,
(d) aerial sources, i.e. oviposition (Williams and Hynes, 1976).

In riffles Townsend and Hildrew (1976) reported that in a small stream 82% of the colonisation of experimentally introduced substrata was by drift and that colonisation took place in currents as low as 5 cm sec^{-1}. Williams and Hynes (1976) followed the sequence of invertebrate species colonising a denuded stream substratum and found that drift contributed to 41.4% of this total. This compared to 28.2% by air, 18.2% by upstream migration and 19.1% by movement up from within the substratum.

In the present investigation, colonisation samplers placed in pools of clean water quality would probably become colonised by species from the littoral zone, natural substratum and drift. Such colonisation was slow and probably never reached asymptotic stability. In pools of poor water quality colonisation samplers, particularly the Biopac unit became colonised quickly and became stabilised more rapidly than those in cleaner waters. This was probably due to the fact whereas in polluted waters the fauna colonising the colonisation sampler is restricted to those species associated with pools, those colonising it in pools with good quality water are also derived from drift from upstream riffles and their colonisation takes longer.

In this section of the work, both the colonisation samplers tested proved useful and practicable and on the basis of performance neither was preferable. The need for National survey purposes however for a standard sampler would best be met by the Biopac units. The interpretation of slag type would probably be different in various parts of the country.

At this stage of the investigations, it was appreciated that the community of invertebrates found on colonisation samplers positioned in the pool reaches of rivers is not that of the natural depositing substratum typical of such reaches. Species such as Hirudinea, Trichoptera and Mollusca which require firm solid surfaces for attachment are found on colonisation samplers, but rarely in the natural depositing substratum. Burrowing forms however, such as worms and tube forming Chironomidae, are more common in the natural substratum.

The community colonising the colonisation sampler is best described as the 'Aufwuchs' of the early German hydrobiologists. This refers to all those organisms that are firmly attached to a hard substratum but do not penetrate into it. One could therefore regard the Biopac unit as a 'Standard Aufwuchs Sampling Unit' (S.A.S.U.). Although 'S.A.S.U.' was at first applied to the Biopac Unit, these initials were subsequently found to have been used by earlier workers (Disney, 1972; Hall and Edwards, 1978) for a 'Standard Artificial Substratum Unit' which in the latter case was developed from work on Simulium in rivers on the Ivory Coast.

As a result of the existing nomenclature the Biopac unit was defined as a 'Standard Aufwuchs Unit' (S.AuF.U.) as used in pools to assess water quality.

3.3 General Conclusions

(1) The benthic invertebrate communities of riffles are sensitive indicators of river water quality.

(2) For monitoring river water quality in rivers where suitably situated riffles are present, as in the rhithron zones, the use of the natural riffle benthic invertebrates is recommended.

(3) The riffle community is best sampled directly either by heel-kick sampling or by quadrat sampling such as the cylinder sampler.

(4) Pools, especially with deep waters are, for biological monitoring purposes, most conveniently sampled by colonisation samplers.

(5) For lowland rivers, with no suitable riffles and in the lower stretches (potamon zone) of larger rivers which are difficult to sample directly, the use of colonisation samplers particularly the S.Auf.U. sampler, appear useful.

(6) Using colonisation samplers, the effect of differences in natural substratum is reduced. The S.Auf.U. therefore standardises and simplifies sampling. Delicate species such as flatworms are less damaged and sample processing is simplified.

(7) The S.Auf.U. usually contain negligible amounts of extraneous material, permitting quick laboratory processing. This depends on the length of time a sampler is immersed. For practical and theoretical reasons, a four-week immersion period was found suitable. A minimum of three samples was found suitable per sampling site.

(8) The S.Auf.U. is easy to install and the collections can be made by persons of varying experience and training.

(9) The S.Auf.U. is durable, corrosion resistant, re-usable and convenient to handle. It is relatively inexpensive to construct (60 pence) and when installed is inconspicuous, yet easily recoverable. The mesh base ensures that most macroinvertebrates are not lost during retrieval.

(10) The main disadvantage is that unless sites are carefully selected, the samplers may be lost due to vandalism and

fishermen. However, a firm attachment generally ensures that samplers are not lost during torrential spates.

(11) Finally, although they may prove useful as a method for monitoring water quality, the S.Auf.U. samplers do not sample the natural depositing substratum invertebrate community.

CHAPTER 4

FIELD TRIALS ON THE USE OF THE STANDARD AUFWUCHS UNIT (S.Auf.U.) SAMPLER FOR SLOW FLOWING LOWLAND RIVERS AND THE POTAMON ZONES OF LARGER RIVERS

4.1 Objectives

The aim of the research programme was to consider the use of the Standard Aufwuchs Unit (S.Auf.U.) as a reliable and acceptable method for sampling deep rivers for the purpose of biological surveillance.

For the purpose of devising a method of processing the surveillance data, the recommendations of the Biological Monitoring Working Party of the Freshwater Monitoring Group (Department of the Environment Standing Technical Advisory Committee on Water Quality set up to recommend a biological classification of river water quality for use in National River Pollution surveys) was to be taken into consideration.

4.2 Introduction

As discussed in Chapter 2, benthic invertebrates are the most commonly used group of organisms in the biological monitoring of river water quality (Hellawell, 1978). One difficulty in using benthic communities is that they are also influenced by the characteristics of the river, e.g. whether rapidly flowing over eroding substratum or slow flowing over a depositing substratum. Furthermore the benthic communities of the deeper rivers are difficult to sample. For this reason, the benthic communities of riffles - shallow, rapidly flowing stretches of rivers - are most commonly used in biological surveillance in Britain. Where such riffle stretches are available along a river, the river water quality can be monitored by the biological surveillance of these riffles.

Difficulties arise, however, in lowland (potamon) zones of rivers and in lowland rivers, both slow-flowing and fast-slowing in which no riffle stretches are present.

The results of Phase 1 of these investigations (Chapter 3) showed that whereas there were no advantages and some disadvantages in using colonisation samplers for sampling riffles, their use in pools had advantages. By providing a solid, firm, substratum they were colonised by species not found in the natural mud substratum. The community could therefore be best described as the 'Aufwuchs' of the German hydrobiologists.

It was considered in Phase 1 that the species found on such units of artificial medium (S.Auf.U.) after a given period of immersion would be influenced by the river water quality. Thus such units, although not sampling the natural benthic community, could be used to monitor water quality in rivers not having suitable riffles.

The trials outlined in this Chapter were therefore carried out to investigate this possibility (Phase 2).

4.3 Selection of Sampling Sites

River sites were selected with the co-operation of different Water Authority biologists to cover a wide range of river types and water qualities. Initially 36 sites on 18 rivers in six different Water Authority areas were selected for study, but early in the programme this was reduced to 33 sites on 17 rivers, due to unforseen vandalism problems. These sites are listed in Table 4.1. Where possible associated riffles were also studied in order to establish a relationship between the S.Auf.U. community and the riffle community in the same water quality. The distribution of the sampling sites is shown in Figure 4.1.

4.4 Methods and materials

4.41 Physico-chemical

Because of the intensity of the sampling programme, which often involved sampling different rivers three times a week over a period of one year, physico-chemical analysis was reduced as compared to the work-load presented in Chapter 3. For example, because of the nature of the BOD_5 test requiring a five-day incubation period, it was impossible to arrange sampling programmes around this analysis. However, the resulting lack of physico-chemical data was supplemented by data supplied by the respective Water Authorities.

4.42 Macroinvertebrate

Due to periods of very heavy rain and consequent high river levels, work on most rivers was delayed until March, 1977.

At each pool site, three S.Auf.U. samplers were positioned on the river bed and were recovered after a period of four weeks immersion. Where possible, the associated riffles of the same water quality, upstream or downstream of the pools (depending on proximity) were also sampled by taking three cylinder samples across the riffle. In addition, where practicable, three 'grab' samples of the natural depositing substratum were taken for comparing the 'Aufwuchs' community with the natural benthic community. At specific

Table 4.1 Details of Sampling sites studied

Water Authority	River/stations	Site	O.S. Reference	Section of River Studied	
				R	P
Yorkshire	Foss	Strensall	SE 62505	R	P
	Ouse *	Naburn	SE 597442		P
	Calder 1	Brighouse	SE 134228	R	P
	Calder 2	Horbury	SE 279181		P
	Don	Conisbrough	SK 509995		P
North-West	Weaver 1	Hankelow Mill	SJ 658450	R	P
	Weaver 2	Windy Arbour	SJ 657544		P
	Weaver 3	Church Minshull	SJ 667608	R	P
	Weaver 4	Weaver Hall	SJ 670644		P
Anglian	Nene 1	Whiston	SP 845618		P
	Nene 2	Stanwick	SP 967712	R	P
	Nene 3	Wansford	TL 084997		P
	Nene 3	Water Newton	TL 104997	R	
Wessex	King's Sedge-moor Drain	Bawdrip	ST 341391		P
	South Drain	Huntspill	ST 367430		P
	Avon 1 *	Bathford	ST 785668		P
	Avon 2	Twerton	ST 726648		P
Welsh	Wye 1	Hereford	SO 532390		P
	Wye 2	Ross-on-Wye	SO 592256		P

continued over

Table 4.1 Details of Sampling sites studied (cotd)

Water Authority	River/stations	Site	O.S. Reference	Section of River Studied	
				R	P
Severn-Trent	Severn 1	Bewdley	SO 788754	R	P
	Severn 2	Saxon's Lode	SO 863387		P
	Severn 3	The Mythe	SO 889336		P
	Blythe	Castle Farm	SP 207888	R	P
	Blythe	Blythe Bridge	SP 211897		P
	Dove	Egginton	SK 267273	R	P
	Derwent 1	Allestree	SK 358398	R	P
	Derwent 2	Draycott	SK 444327		P
	Trent 1	King's Bromley	SK 126176	R	P
	Trent 2	Catton Hall	SK 205155	R	P
	Trent 3	Gunthorpe	SK 682437		P
	Trent 4	Kelham	SK 776557	R	P
	Mease	Croxall	SK 192140	R	P
	Tame 1	Water Orton	SP 175915		P
	Tame 1	Castle Bromwich	SP 147903	R	
	Tame 2	Elford	SK 190103	R	P
	Churnet 6	Cheddleton Mill	SJ 973526	R	P
	Churnet 10	Alton	SK 072426	R	P
	Tean 1	Checkley	SK 028377	R	P
	Tean 2	Beamhurst	SK 059361	R	P

Key:

R = Riffle

P = Pool, deep slow-flowing or deep, fast-flowing waters

* = Eliminated due to vandalism

FIG. 4.1 MAP OF ENGLAND AND WALES SHOWING THE DISTRIBUTION OF RIVERS STUDIED.

sites where the pool substratum consisted of gravel, a 30-second heel kick was preferred to the grab sample.

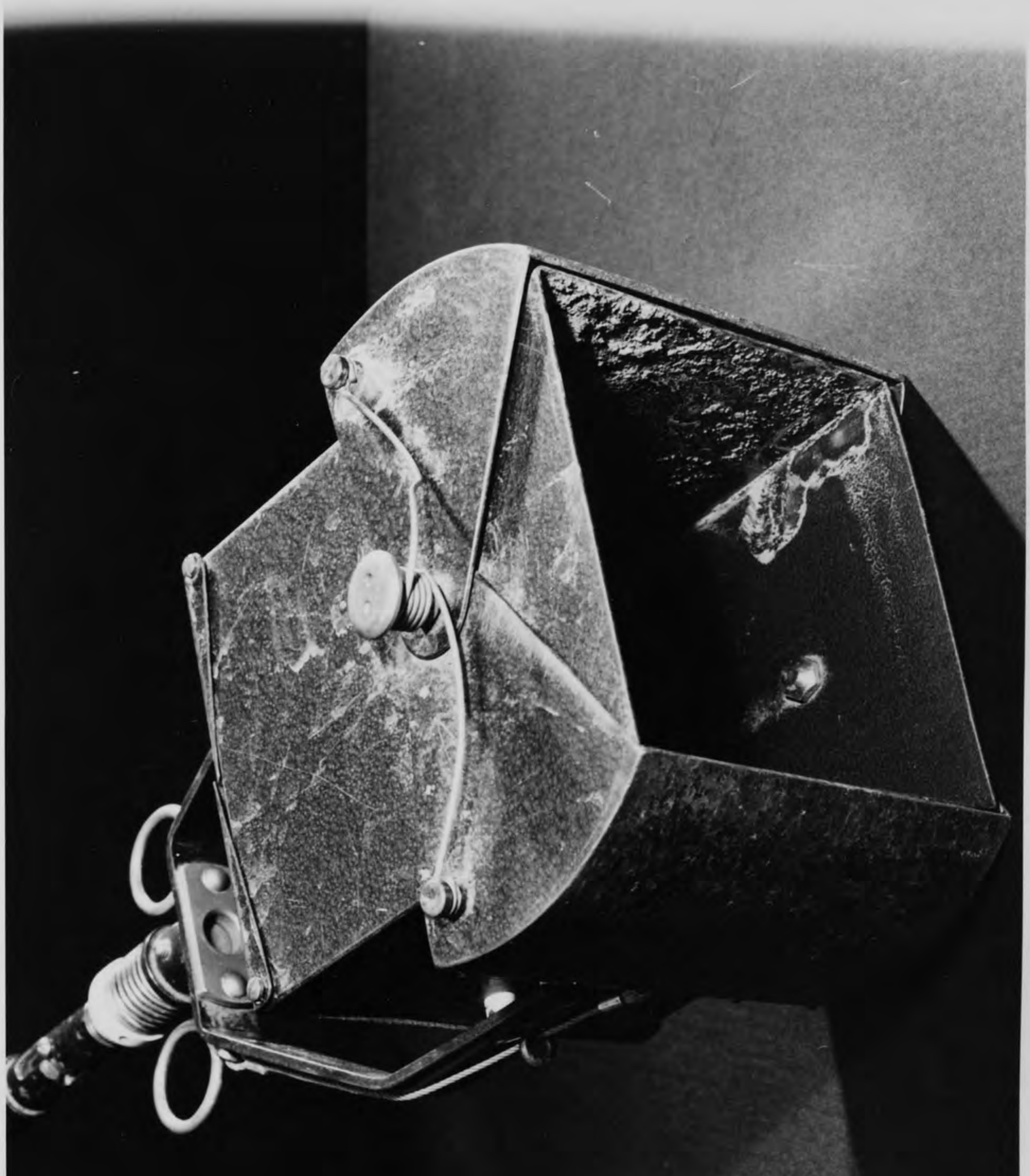
The number of occasions the river sites were sampled differed according to their proximity and suitability. In some cases, nine samples were taken throughout the year. Except for the distant Yorkshire and Wessex rivers and the River Nene, which were sampled every two months, all sites were sampled monthly. The dates for the sampling periods for the different rivers are given in their respective sub-sections.

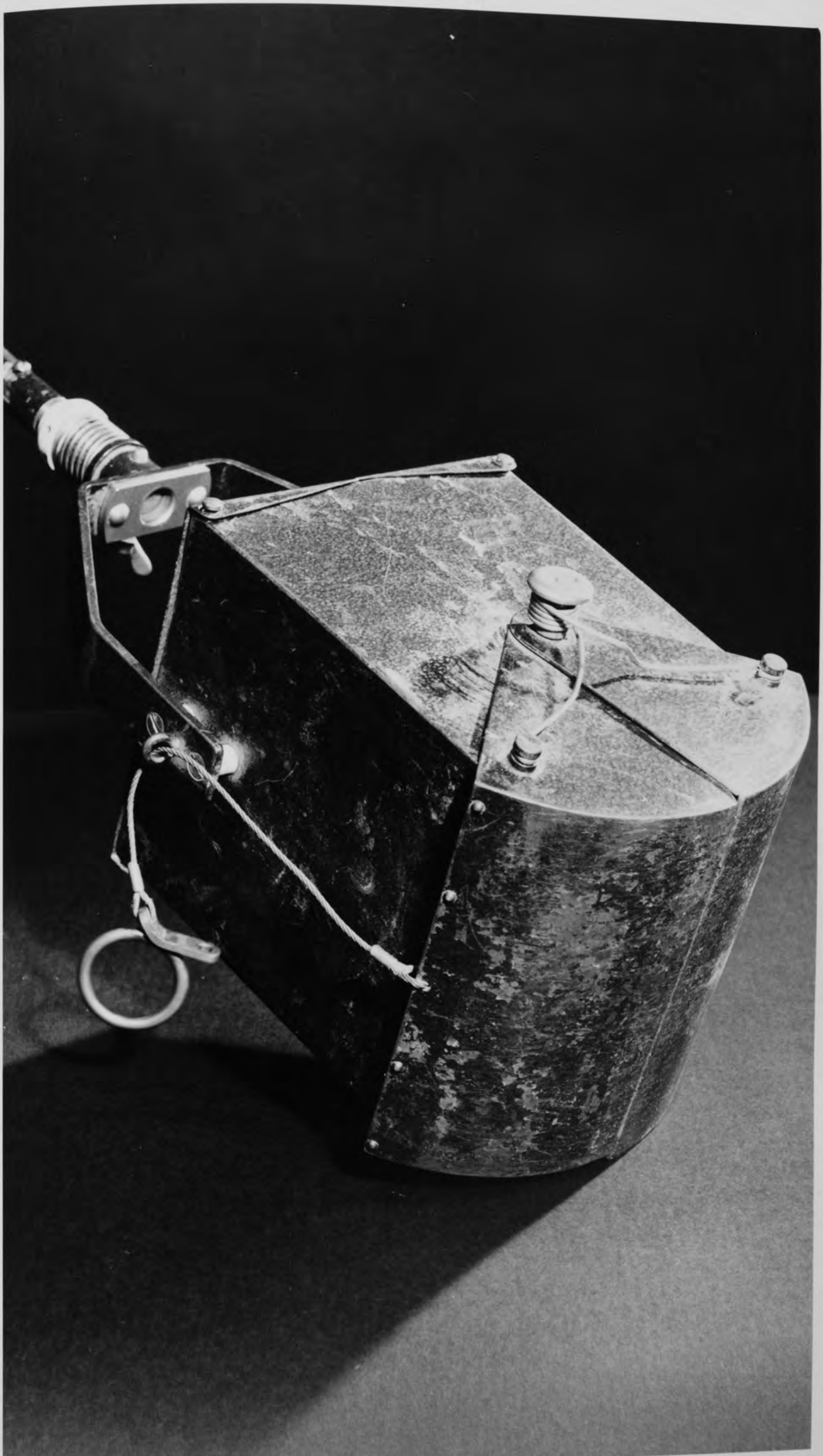
The methods of sampling used in these investigations were those presented in Phase 1 of the programme. The Ekman grab (Plates 9 and 10) used in this investigation was designed specifically for sampling soft sediments and is operated by means of a rod. Penetration into the substratum is assisted by pushing on the rod and the grab is closed by rotating the rod to release the jaw springs. This samples an area of 0.02 m^{-2}.

The S.Auf.U. samplers were generally anchored towards the banks of the river but in the main river flow, using 8 mm. diameter, six foot steel rods. Rubber bungs above and below the unit also secured the unit to the rod and prevented any upward movements due to the force of the current. In very deep waters they were weighted with a house brick and suspended using 3 mm. nylon cord so as to rest on the river bed.

At the time of sampling current velocities were measured using an F2 portable OTT meter. In the pools, velocities were taken at the level of the S.Auf.U. samplers, while in the riffles these were

PLATE 9 EKMAN GRAB(OPEN POSITION)





taken in the main flow of the river just above the substratum.

In most cases, identification of the fauna was to the specific level, but as described in Chapter 3, the Chironomidae were mainly taken to the sub-family/tribe level.

As a means of evaluating the data accumulated from each river system, the rivers were categorised into two main groups for examination; those with a riffle-pool system and those stretches which were remote from riffles (Potamon). These sub-divisions will be analysed and discussed in sequence, (Plate 11).

4.5 Processing of results to indicate Water Quality

Apart from examining the biological data in detail to show spatial and temporal patterns of species distribution, one can also show biological trends associated with water quality.

In evaluating different data processing techniques devised to produce an index or grade of water quality one is faced with the dilemma of not having an objective value against which one can test any derived values. Since biological methods are intended to complement physico-chemical analysis, one cannot merely use any single physico-chemical determinand as a measure. If one could, there would be no point in developing biological methods!

The only practical approach is to compare biological values, indexes etc., derived by different data processing techniques, for different stations in relation to known polluting discharges and their dilution, and in light of as wide a range of determinands as possible and known water uses. Comparisons can be made at different stations along the same river system in relation to known discharges. Com-

PLATE 11 EXAMPLES OF RIVER TYPE STUDIED

FIGURE J

EXAMPLES OF SAMPLING STATIONS USED IN THE SURVEY, ILLUSTRATING DIFFERENT RIVER TYPES

A. Tributary stream of River Churnet Head stream

B. River Mease Riffle Rhithron

C. River Weaver Station 1 Pool Rhithron

D. River Nene Station 2 Lowland river

E. South Drain, Wessex Drainage dyke

F. River Severn The Mythe Potamon

TABLE 4.2 THE ASSESSMENT OF THE 'BIOTIC INDEX' AS USED BY THE TRENT RIVER BOARD(WOODIWISS,1964).

Clean

Organisms in order of tendency to disappear as degree of pollution increases

		Total number of groups present				
		0-1	2-5	6-10	11-15	16+
Plecoptera nymph present	More than one species	-	7	8	9	10
	One species only	-	6	7	8	9
Ephemeroptera nymph present	More than one species*	-	6	7	8	9
	One species only*	-	5	6	7	8
Trichoptera larvae present	More than one species†	-	5	6	7	8
	One species only†	4	4	5	6	7
Gammarus present	All above species absent	3	4	5	6	7
Asellus	All above species absent	2	3	4	5	6
Tubificid worm and/or Red Chironomid larvae present	All above species absent	1	2	3	4	-
All above types absent	Some organisms such as Eristalis tenax not requiring dissolved oxygen may be present.	0	1	2	-	-
		Biotic Index				

Polluted

*Baetis rhodani excluded †Baetis rhodani(Ephem.)is included in this section for the purpose of classification

The term 'Group'used for purpose of the biotic index means any one of the species included in the following list of organisms or sets of organisms.

Each known species of Platyhelminthes(flatworms)
Annelida(worms excluding genus Nais)
Genus Nais(worms)
Each known species of Hirudinae(leeches)
Each known species of Mollusca(snails)
Each known species of Crustacea(hog louse,shrimps)
Each known species of Plecoptera(stone-fly)
Each known genus of Ephemeroptera
(may-fly,excluding Baetis rhodani)

Baetis rhodani(mayfly)
Each family of Trichoptera(caddis-fly)
Each species of Neuroptera larvae(alder-fly)
Family Chironomidae(midge larvae except Chironomus Ch.thummi)
Chironomus Ch.thummi(blood worms)
Family Simuliidae(black-fly larvae)
Each known species of other fly larvae
Each known species of Coleoptera(Beetles and beetle larvae)
Each known species of Hydracarina(water mites)

TABLE 4.3 BIOTIC INDEX BY THE 'SCORE' SYSTEM(CHANDLER,1970).

Groups present in sample		Increasing abundance P	F	C	A	V
		Points scored				
Each species of	Planaria alpina					
	Taeniopterygidae					
	Perlidae,Perlodidae	90	94	98	99	100
	Isoperlidae,Chloroperlidae					
Each species of	Leuctridae,Capniidae					
	Nemouridae(excl.Amphinemura)	84	89	94	97	98
Each species of	Ephemeroptera(excl.Baetis)	79	84	90	94	97
Each species of	Cased caddis,Megaloptera	75	80	86	91	94
Each species of	Ancylus	70	75	82	87	91
-	Rhyacophila(Trichoptera)	65	70	77	83	88
Genera of	Dicranota,Limnophora	60	65	72	78	84
Genera of	Simulium	56	61	67	73	75
Genera of	Coleoptera,Nematoda	51	55	61	66	72
-	Amphinemura(Plecoptera)	47	50	54	58	63
-	Baetis(Ephemeroptera)	44	46	48	50	52
-	Gammarus	40	40	40	40	40
Each species of	Uncased caddis(excl.Rhyacophila)	38	36	35	33	31
Each species of	Tricladida(excl.P.alpina)	35	33	31	29	25
Genera of	Hydracarina	32	30	28	25	21
Each species of	Mollusca(excl.Ancylus)	30	28	25	22	18
-	Chironomids(excl.C.riparius)	28	25	21	18	15
Each species of	Glossiphonia	26	23	20	16	13
Each species of	Asellus	25	22	18	14	10
Each species of	Leech(excl.Glossiphonia.Haemopis)	24	20	16	12	8
-	Haemopis	23	19	15	10	7
-	Tubifex sp.	22	18	13	12	9
-	Chironomus riparius	21	17	12	7	4
-	Nais	20	16	10	6	2
Each species of	Air breathing species	19	15	9	5	1
	No animal life			0		

P ;1-2 F;3-10 C;11-50 A;51-100 V;100+ (NOS. PER 5 MIN. KICK SAMPLE)

TABLE 4.4 BIOLOGICAL MONITORING WORKING PARTY(B.M.W.P.) SCORE 1980

FAMILIES	SCORE	
	Eroding	Depositing
Siphlonuridae,Heptageniidae,Leptophlebiidae Ephemerellidae,Potamanthidae,Ephemeridae	80	100
Taeniopterygidae,Leuctridae,Capniidae, Perlodidae,Perlidae,Chloroperidae		
Aphelocheiridae		
Phryganeidae,Molannidae,Beraeidae,Goeridae, Odontoceridae,Leptoceridae,Lepidostomatidae, Brachycentridae,Sericostomatidae		
Astacidae	60	80
Lestidae,Agriidae,Gomphidae,Aeshnidae, Corddulegasteridae,Corduliidae,Libellulidae		
Psychomyiidae,Philopotamidae		
Caenidae	50	70
Nemouridae		
Rhyacophilidae,Polycentropodidae,Limnephilidae		
Neritidae,Viviparidae,Ancylidae	40	40
Hydroptilidae		
Unionidae		
Corophiidae,Gammaridae		
Platycnemididae,Coenagriidae		
Mesovelidae,Hydrometridae,Gerridae,Nepidae, Naucoridae,Notonectidae,Pleidae,Corixidae	30	30
Haliplidae,Hygrobiidae,Dytiscidae,Gyrinidae, Hydrophilidae,Clambidae,Helodidae,Dryopidae, Elminthidae,Chrysomelidae,Curculionidae		
Hydropsychidae		
Tipulidae,Simuliidae		
Planariidae,Dendrocoelidae		
Baetidae	20	20
Sialidae		
Piscicolidae		
Valvatidae,Hydrobiidae,Lymnaeidae,Physidae, Planorbidae,Sphaeriidae	10	10
Glossiphoniidae,Hirudidae,Erpobdellidae		
Asellidae		
Chironomidae	5	5
Oligochaeta(whole class)	1	1

parisons can also be made between different rivers to assess the nationwide applicability of the method.

In the first instance comparisons will be made using four data processing techniques, namely Trent Biotic Index (Woodiwiss, 1964; Table 4.2), Chandler Score System (Chandler, 1970; Table 4.3), B.M.W.P. Score System (Table 4.4) and the Diversity Index (Wilhm and Dorris, 1968). At a later stage comparisons will be made to see if the systems are applicable and acceptable across the country in different Water Authority areas involving different river types.

4.6 Processing of sampling methods by similarity coefficients

As described in Chapter 3, several methods have been derived which compare community species lists between two samples. Such techniques are generally used as a basis for comparing one station temporally or several stations spatially. In this investigation such methods were used to detect biological discontinuities between the different samplers under examination.

In a qualitative approach Sorensen's similarity coefficient was calculated as shown in section 3.162, for comparing S.Auf.U. data with Grab and cylinder data.

Quantitative comparisons taking into account relative abundance of the species were also calculated using the method of Czekanowski (1913). Comparable measures of abundance were determined using the expression:

$$C_Z = \frac{2W}{A + B}$$

where W = the sum of the lesser measures of abundance of each species common to both communities, A = the sum of measures of abundance in community A,

B = the sum of measures of abundance in community B.

With both methods, computation of data resulted in values between one (total similarity) and zero (total dis-similarity) being obtained.

4.7 Rhithron-riffle-pool case studies

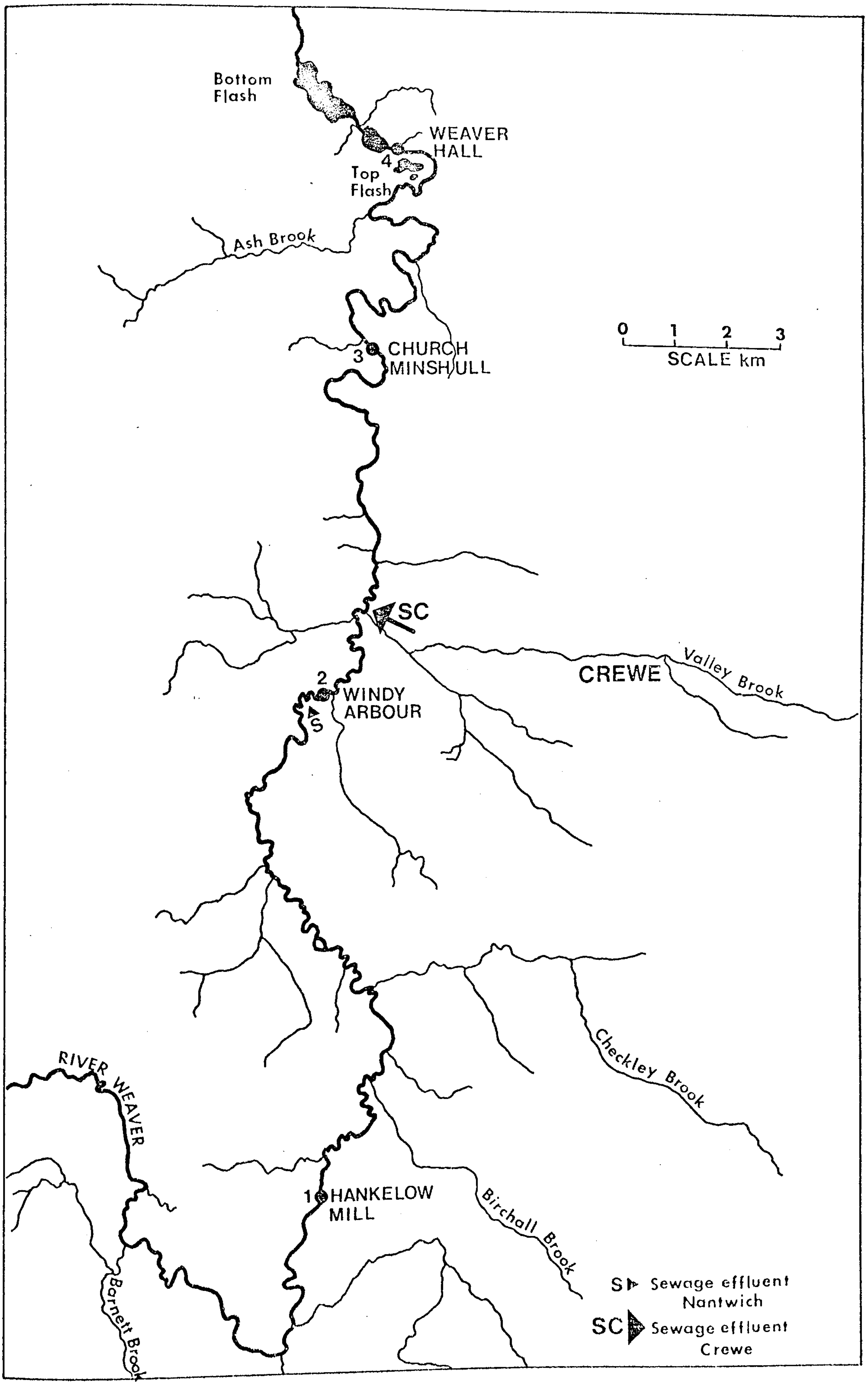
4.71 River Weaver

4.711 Sampling stations

The River Weaver rises in the south-west plains of lowland Cheshire and flows north, eventually draining into the Mersey Estuary near Northwich. Most of the region is covered superficially with inter-bedded deposits of sand, alluvium and boulder and laminated clays. The river flows through the thinly populated and largely agricultural parts of Cheshire, although salt extraction is practised in mid-Cheshire.

The present study area approximately 23 km. long was situated between Audlem and Winsford (Figure 4.2). Station 1 at Hankelow Mill was upstream of the major discharges, while Station 2 at Windy Arbour was downstream of the small sewage works serving Nantwich. Between Stations 2 and 3 a major discharge of sewage and industrial effluent from Crewe enters the Weaver via Valley Brook and consequently affects the quality of the river at Church Minshull (Station 3). Station 4 at Weaver Hall was situated in between the Top and Bottom Flash (deep, wide, lake-like section of the river) in a section where some self-purification from pollution may take place.

FIG. 4.2 MAP OF THE RIVER WEAVER, CHESHIRE, SHOWING THE DISTRIBUTION OF SAMPLING SITES.



The bottom of the river consisted mainly of silt and sand. Loose stones and gravel were uncommon and areas suitable for direct quantitative sampling were restricted. The nature of the substratum and current velocities experienced at each station are given in Table 4.5.

Table 4.5 The mean and range of current velocities and nature of the substratum found in the River Weaver

	Riffle		Pool	
Station	Nature of Substratum	Current Velocity cm.sec^{-1}	Nature of Substratum	Current Velocity cm.sec^{-1}
1	Gravel and Silt	74.5 (37.0 - 96.7)	Silt	23.6 (17.9 - 34.1)
2			Silt	8.4 (3.8 - 12.4)
3	Stones and Gravel	74.5 (35.8 - 104.2)	Mud + Silt	8.3 (Negl. - 14.4)
4			Detritus	Negl.

The riffle at Station 1 was approximately 50 m. downstream of the pool, while at Station 3, the riffle was situated below a weir, 50 m. downstream of the pool. The nearest upstream riffle was 1 km. from the pool.

The sampling programme commenced at the end of March (29.3.77) and lasted for nine months until December (6.12.77).

4.712 Results and discussion

Although some small sewage effluent discharges occur above Stations 1 and 2, the major discharge is that from Crewe which enters between Stations 2 and 3. The chemical data are presented in Annexe 2 (Tables 13a - c), Annexe 3 (Table 22) and summarised in Table 4.6 which shows the means and ranges of determinands made by Aston and as supplied by the North West Water Authority. These reflect the change in water quality between Station 2 and 3 shown by a decrease in dissolved oxygen and increase in BOD_5 and ammonia. Generally the analyses carried out by the two independent bodies show similar trends, considering water samples were taken at different time periods. The river appears only to have recovered slightly by Station 4.

Table 4.6 Physico-chemical data expressed as mean and range for the River Weaver (mg. l^{-1})

Station	Analysis	S.S.	D.O.	BOD_5	Ammonia	Nitrate
1	Aston	5 (0- 9)	8.3 (6.1-11.2)		1.4 (0.5-3.2)	6.7 (5.2- 9.1)
	WA	14 (5- 38)	9.6 (7.5-12.0)	3.8 (1.9- 6.5)	0.6 (0.1-2.1)	7.4 (4.5-14.0)
2	Aston	16 (2- 79)	10.2 (8.0-13.0)		1.3 (0.8-2.3)	9.2 (6.6-14.4)
	WA	19 (3- 61)	10.2 (8.3-12.3)	4.9 (2.3-12.0)	0.7 (0.1-2.7)	8.7 (6.5-14.5)
3	Aston	11 (5- 27)	7.2 (4.9-10.3)		3.7 (1.9-6.7)	7.5 (5.4-12.2)
	WA	21 (2-103)	7.8 (4.1-11.2)	7.7 (2.9-24.0)	2.7 (1.1-4.8)	7.6 (5.4-15.0)
4	Aston	9 (3- 18)	6.3 (3.2-10.2)		3.2 (1.8-4.9)	8.1 (6.0-11.8)

The detailed macroinvertebrate data collected from the River Weaver are presented in Annexe 4 (Tables 31 - 41). The number of taxa taken by S.Auf.U. and direct sampling of the depositing substratum of the pools and by cylinder sampling in associated riffles at Stations 1 and 3 are presented in Figure 4.3. These show the mean number of taxa collected, together with the minimum for one sample and the cumulative maximum for three samples (S.Auf.U. and cylinder only). From this data it can be seen that higher numbers of taxa were taken on the S.Auf.U. than by direct sampling of the pools, the difference being greatest at Stations 2 and 4 where no associated riffles were present. Generally both methods of sampling in the pools showed similar temporal trends, which were also supported by direct sampling from the nearby riffles. Typically the highest number of taxa were obtained from Station 1, with a slight reduction at Station 2 due to mild organic pollution. Numbers fell dramatically at Station 3 but started to increase at Station 4 as a result of self purification.

An analysis of similarity coefficients to compare sampling methodology are presented in Figure 4.4. In almost every case Sorensen's values were considerably higher than Czekanowski values. At Weaver 1 where the current velocity was highest in the pools compared to downstream sites, the greatest similarity was obtained between S.Auf.U. and direct sampling of the mud. However, in the vicinity of riffles, as recorded at Weaver 1 and 3, a comparison between S.Auf.U.and direct sampling of riffles revealed a very similar level of similarity. Also in the riffle/pool situation, Sorensen and Czekanowski similarities were closely linked compared

FIG. 4.3 NUMBERS OF TAXA COLLECTED MONTHLY ON S.AUF.U. IN POOLS AND DIRECT SAMPLING OF POOLS AND ASSOCIATED RIFFLES , RIVER WEAVER.

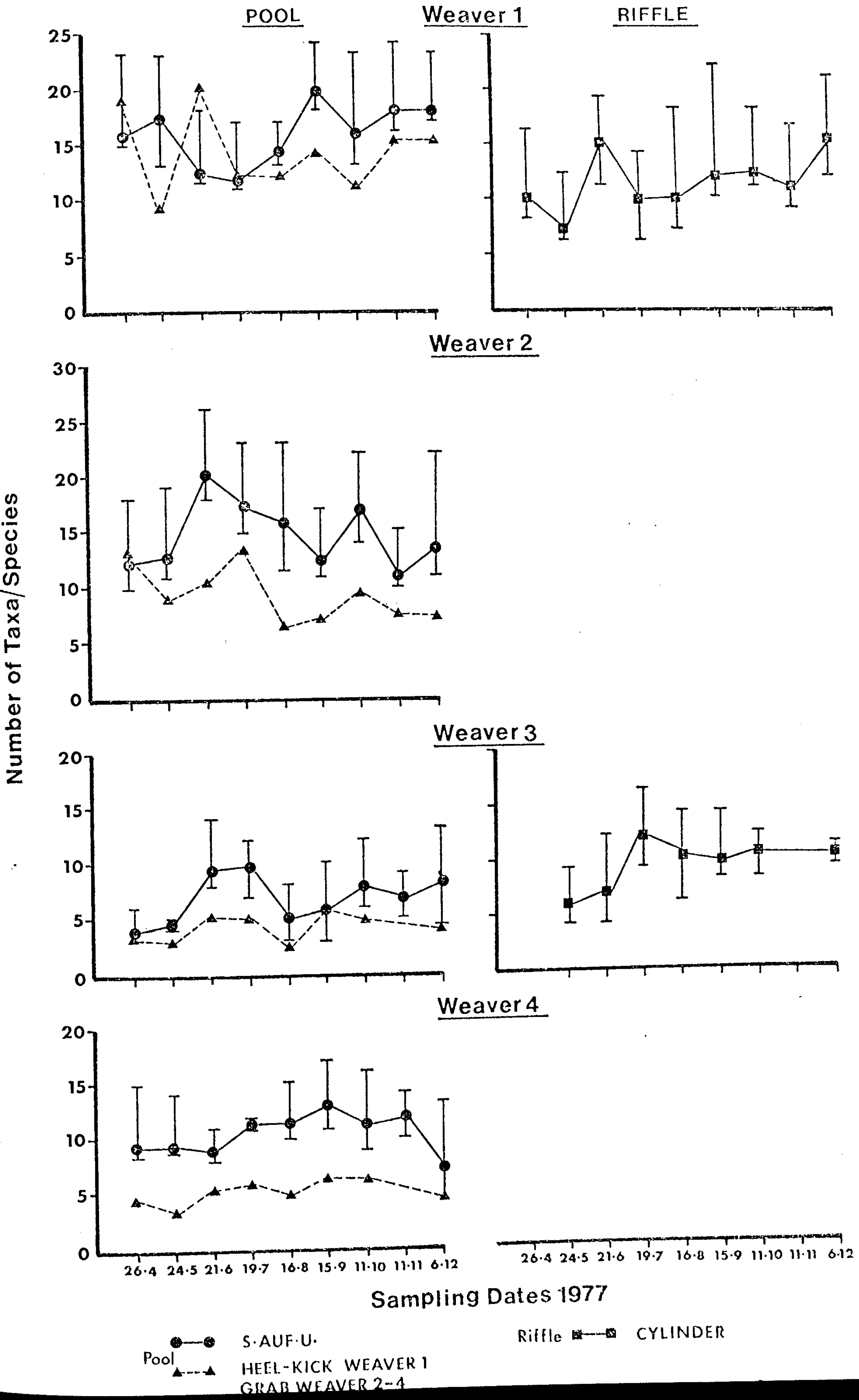
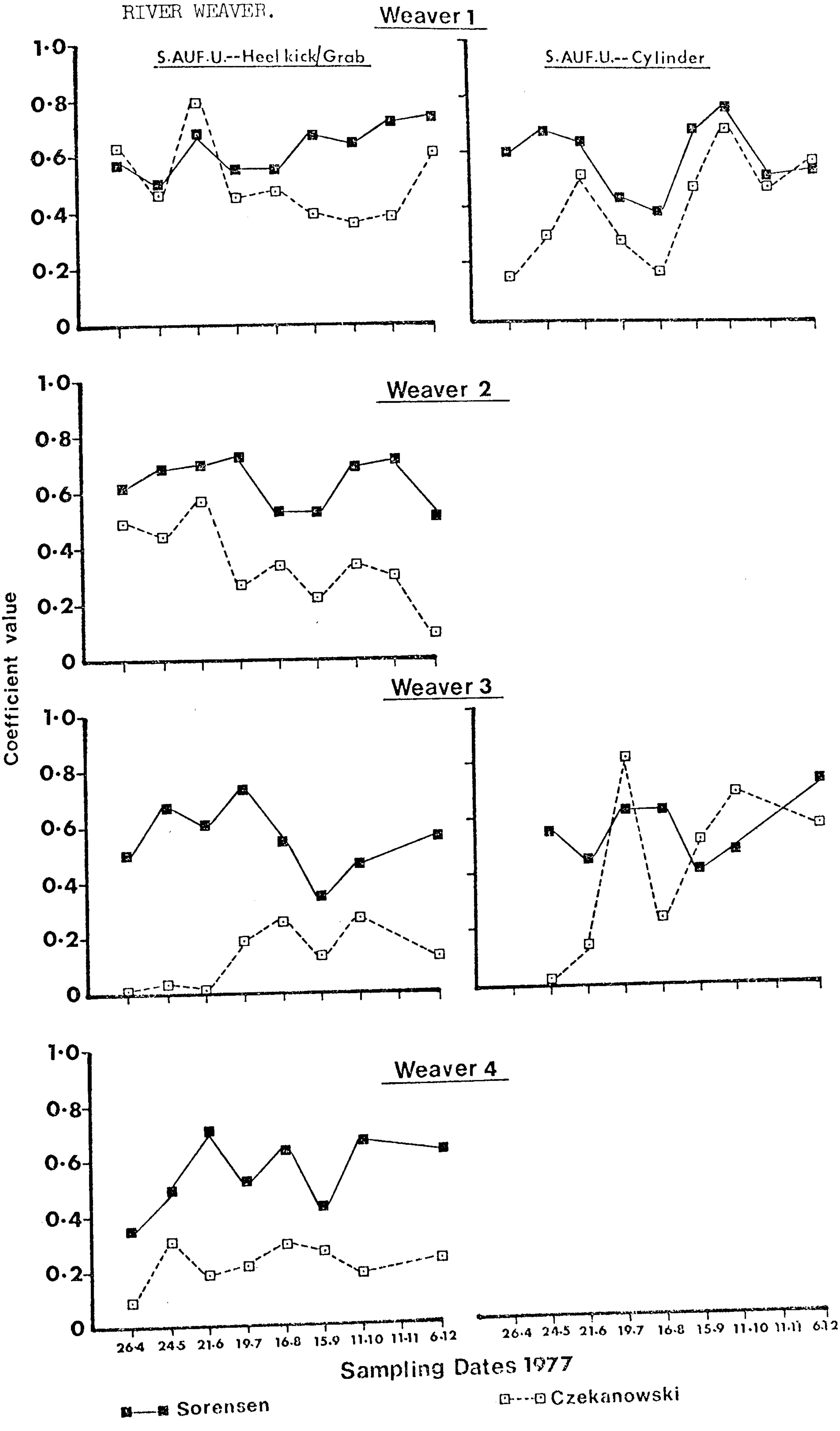


FIG.4.4 COMPARISONS OF SAMPLING BY S.AUF.U. WITH DIRECT SAMPLING OF POOLS AND ASSOCIATED RIFFLES USING SIMILARITY COEFFICIENT ANALYSIS RIVER WEAVER.

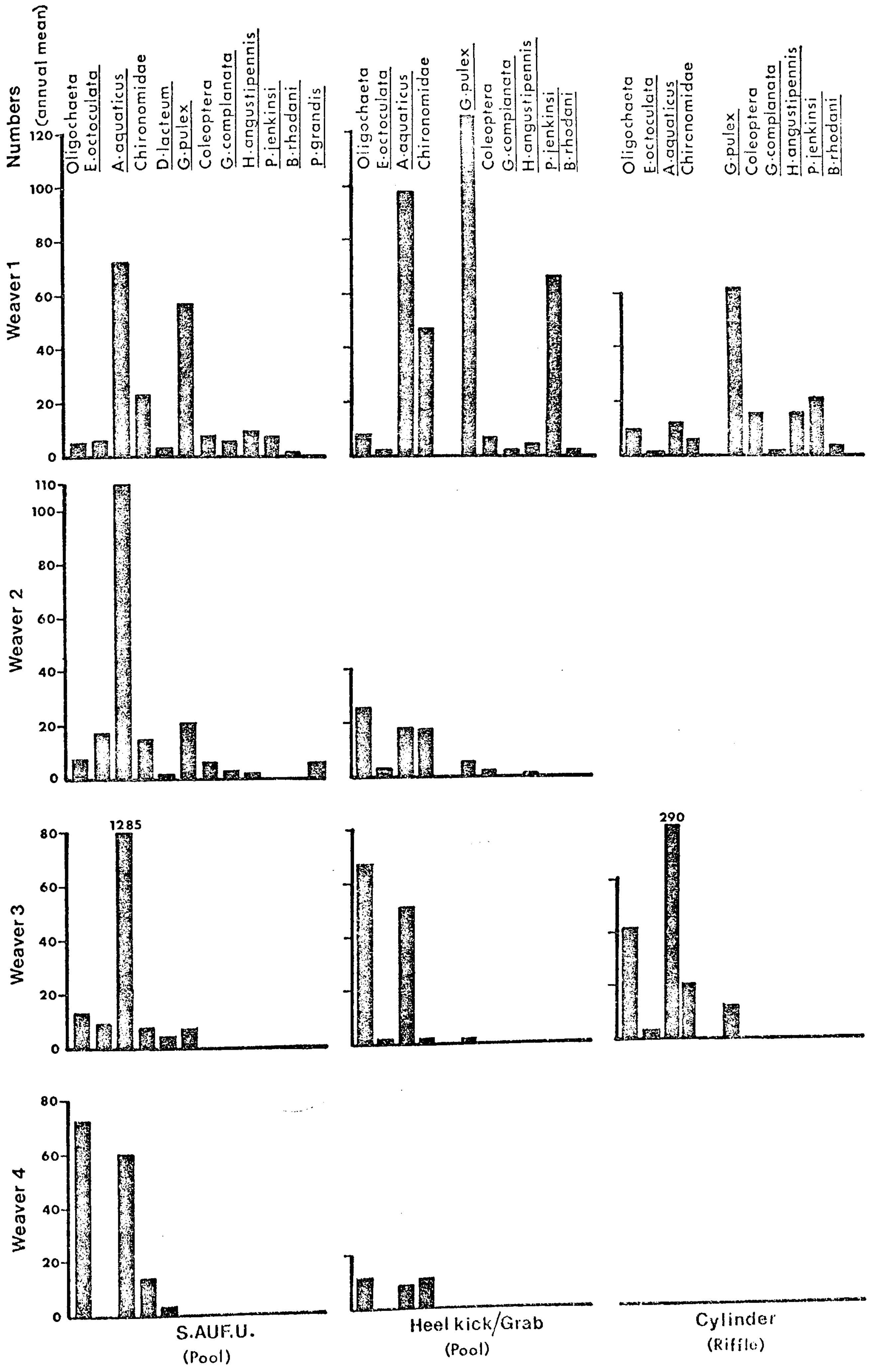


to some downstream pool sites. In the pools S.Auf.U. tended to collect a greater diversity of species and higher numbers compared to direct sampling by grab and as a consequence large discrepancies were obtained between sampling methodologies and types of similarity analyses.

The relative abundance of the major taxa, as indicated by the different sampling methods, at the four stations are shown in Figure 4.5. These values are calculated as the mean of three samples taken and represented as an annual mean for the different sampling methodologies. Both the density of particular taxa and the taxa present, as indicated by direct sampling of the pool substratum, indicate a marked deterioration between Station 1 and 2, whereas the known effluent discharges would suggest the most marked change occurred between 2 and 3 as supported by the chemical data (Table 4.6). The S.Auf.U. samplers, however, indicate a slight change between 1 and 2 and the most marked change occurring between 2 and 3 as expected. These differences are also illustrated by riffle sampling, where species are known to be sensitive to changes in water quality.

In the upstream zones of the River Weaver, as indicated by Station 1, one finds a reasonable diversity of species with large numbers of Gammarus pulex. Asellus aquaticus is common in the pool sections of the river, although only occasional in the riffle sections, indicating a slight background organic enrichment, probably from agricultural run off. Baetis rhodani, Hydropsyche angustipennis, Coleoptera (Dytiscidae and Haliplus) and Potamopyrgus jenkinsi are present in both pools and riffles. At Station 2, after receiving organic effluent from Nantwich, the numbers of G.pulex drastically

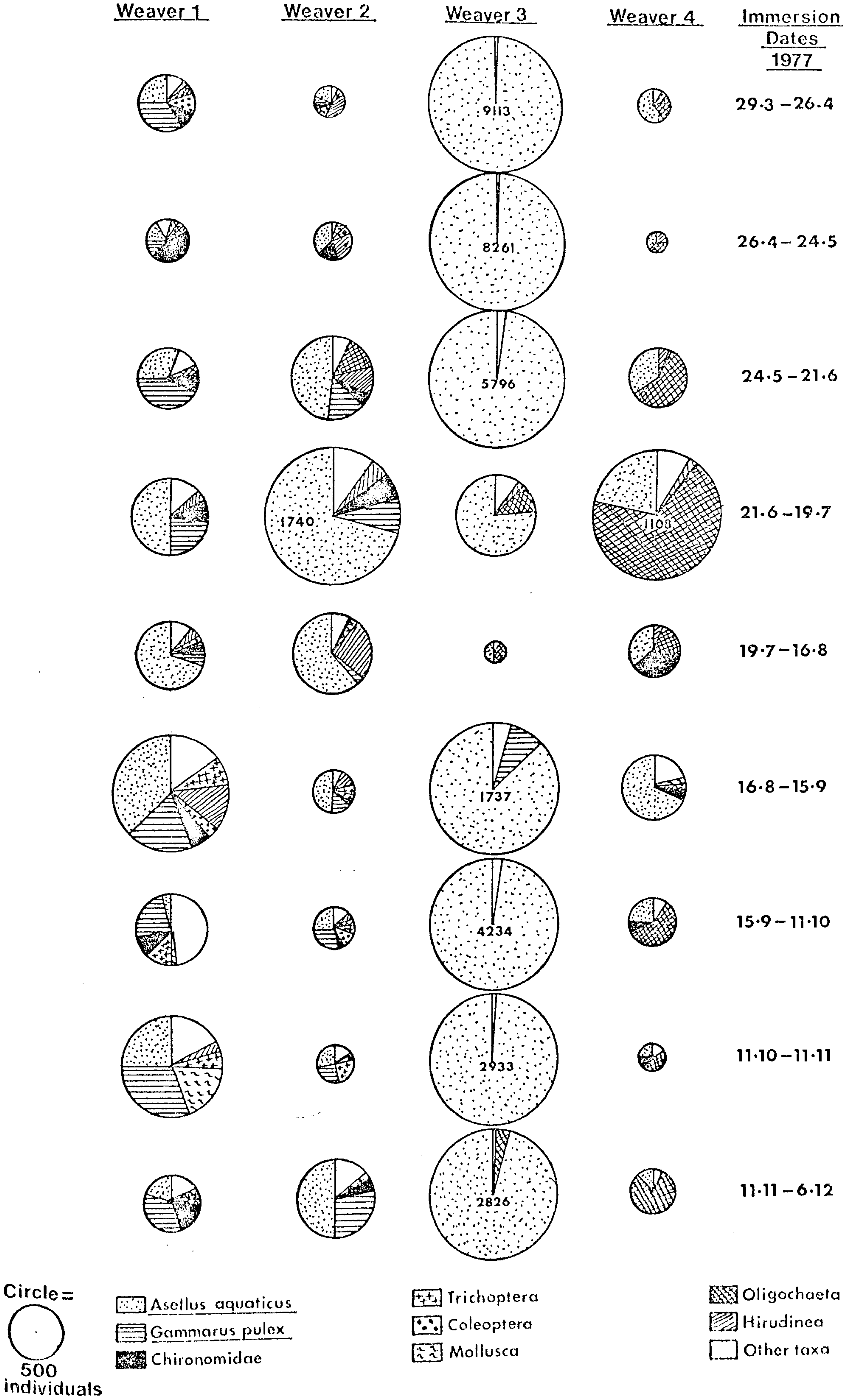
FIG.4.5 RELATIVE ABUNDANCE OF MAJOR TAXA AT FOUR STATIONS ON THE RIVER WEAVER AS INDICATED BY DIFFERENT SAMPLING METHODS.



reduce, while *A.aquaticus* becomes the dominant species. With S.Auf.U. sampling one finds little change in the community structure, other than a reversal of the *G.pulex*:*A.aquaticus* ratio. The cased caddis *Phryganea grandis* becomes common as a result of increased macrophytic cover and *Sialis lutaria* is found occasionally together with the leech *Erpobdella octoculata* as a result of slight nutrient enrichment. The reduction in diversity recorded by grab sampling at Station 2 was as a result of sampling inefficiency caused by macrophytes jamming the jaws of the grab. At Station 3 downstream of the industrialised effluent of Crewe, all sampling methods reflected the change in water quality. *A.aquaticus* was totally dominant and Oligochaete numbers increased particularly in the pool benthos. At Station 4, the numbers of Oligochaetes increased as a result of detrital sediment and not further deteriorations in water quality.

The percentage composition of selected taxa collected by S.Auf.U. sampling over a nine-month sampling period is presented in Figure 4.6. From this data analysis it is evident that the species composition varies within an annual cycle. In the upstream zones detritivores, i.e. *G.pulex* and *A.aquaticus* and clinging forms, i.e. Trichoptera and Mollusca make up the largest proportion of the community, while in the downstream polluted zones detritivores such as *A.aquaticus* and Oligochaeta account for the largest proportion. The greatest variation in abundance as shown by the pie charts is observed at Weaver 3, where a sudden population 'crash' occurs between July and August. This is accounted for by a dramatic increase in the percentage cover and density of the blanket weed *Cladophora*, resulting in a smothering of the samplers. As the weed cover decreases during September, then the numbers of individuals increase again to relatively high levels.

FIG.4.6 PERCENTAGE COMPOSITION OF SELECTED TAXA COLLECTED BY S.AUF.U. IN RIVER WEAVER.



Although it is extremely difficult to prevent the fouling of colonisation samplers, it can be one of their major faults.

The seasonal variation of particular species are presented in Figures 4.7 - 4.10. The highest numbers of G.pulex (Figure 4.7) recorded by all sampling methods at most sites occurred between June and July. At Weaver 3, increased numbers on S.Auf.U. were recorded during September as a result of heavy rain and subsequent scouring of upstream sites. The resulting specimens deposited downstream in the polluted zones reflected the past catastrophic condition. Generally, each sampling method reflected similar seasonal trends with respective responses to water pollution. The seasonal variation of A.aquaticus (Figure 4.8) collected by S.Auf.U. was typically similar to the other sampling methods, although larger numbers were generally collected. The most irregular results were obtained at Weaver 3, due to fouling of the samplers by Cladophora. Aufwuchs 'loving' species such as Erpobdella octoculata and Glossiphonia complanata were typically collected in larger numbers on S.Auf.U. than found by direct sampling of the benthos (Figure 4.9). The numbers of E.octoculata varied according to the strength of the organic effluents recorded along the Weaver, while the numbers of G. complanata fell immediately below Nantwich, indicating its intolerance to organic pollution. Chironomids as observed in Chapter 3 and so often recorded in the literature (in abundance on colonisation samplers), were characteristically recorded in equal abundance on S.Auf.U. and in the benthos. Again numbers recorded on S.Auf.U. appeared to follow broadly the seasonal variation observed on the benthos. The numbers of Tanytarsini appeared to decline as a result of Crewe sewage, while Orthocladiinae and Chironomini appeared to be unaffected by water quality.

FIG. 4.7 SEASONAL VARIATION OF GAMMARUS PULEX IN RIVER WEAVER.

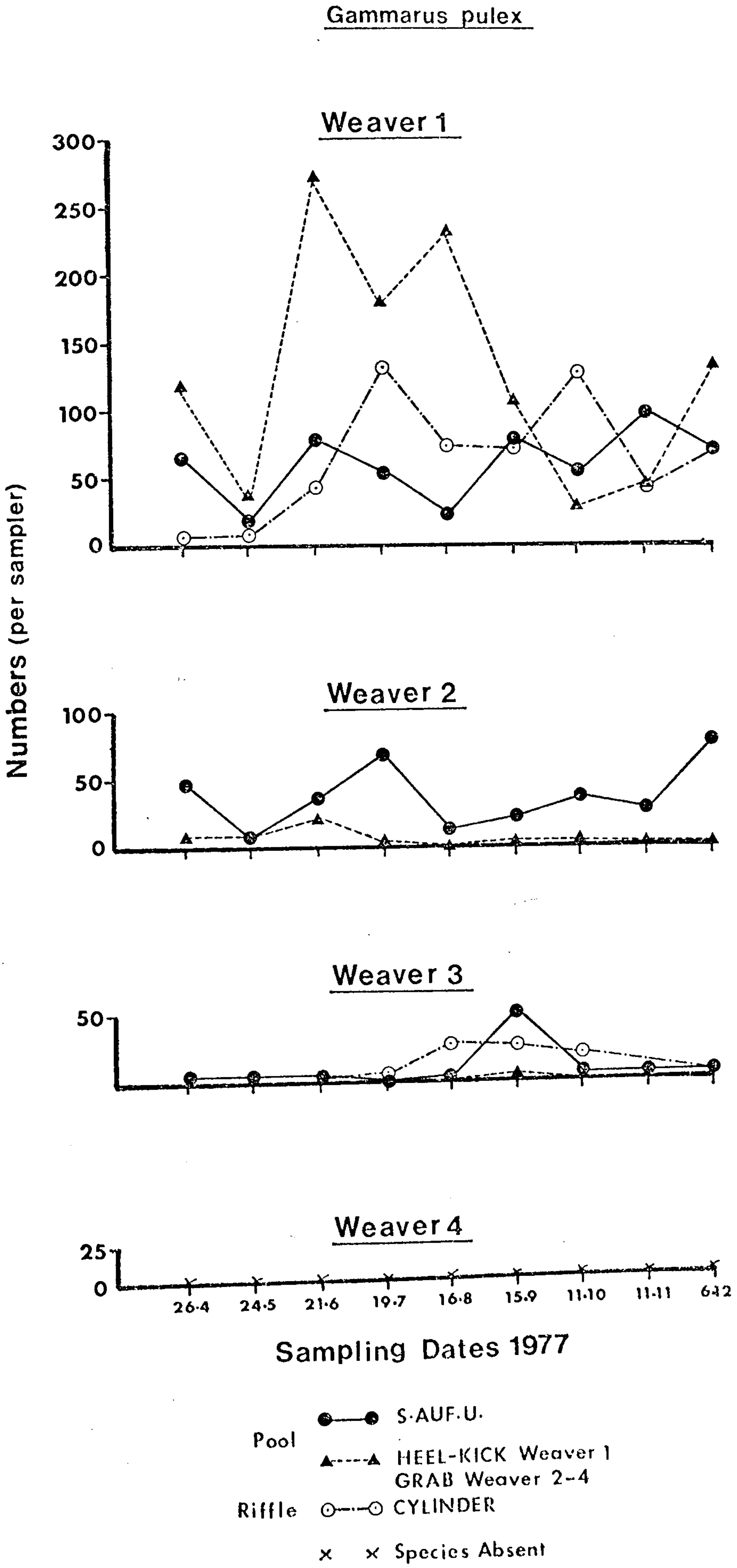


FIG. 4.8 SEASONAL VARIATION OF *ASELLUS AQUATICUS* IN RIVER WEAVER.

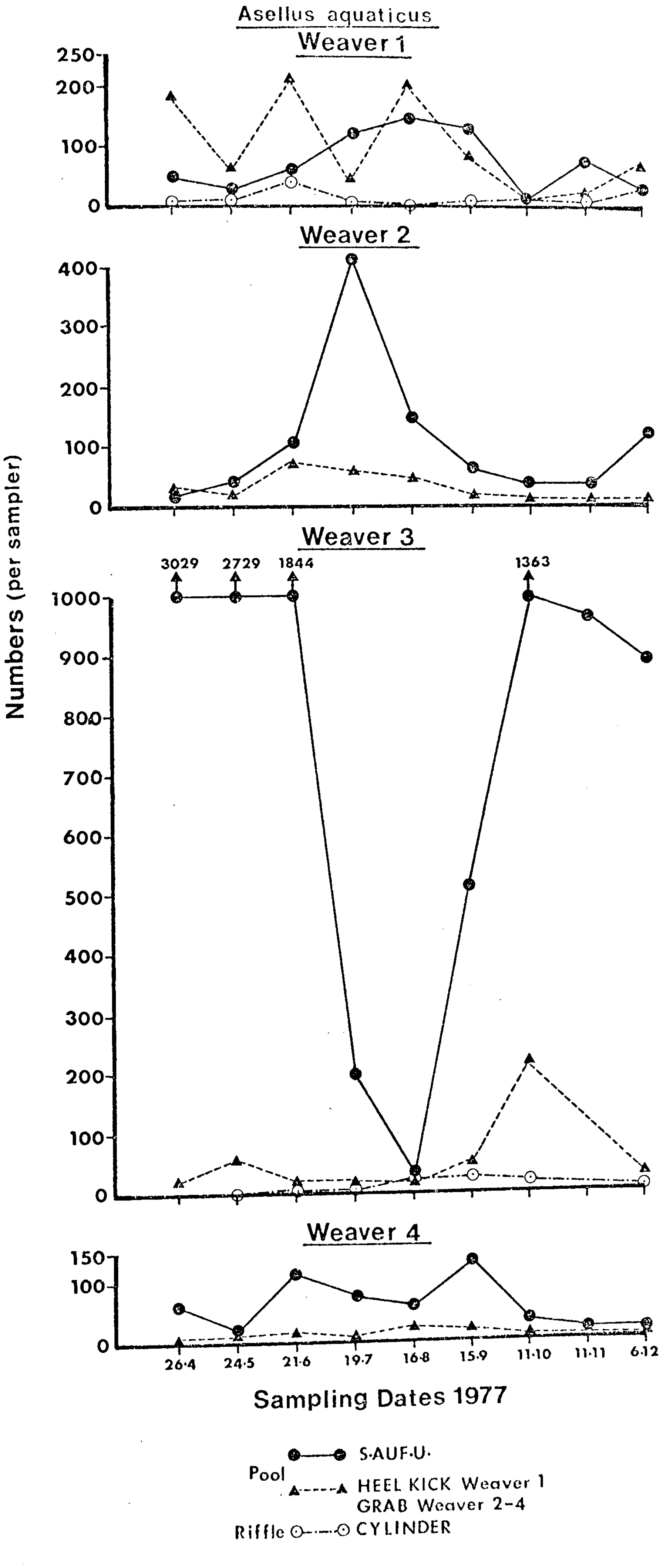


FIG.4.9 SEASONAL VARIATION OF HIRUDINEA IN RIVER WEAVER.

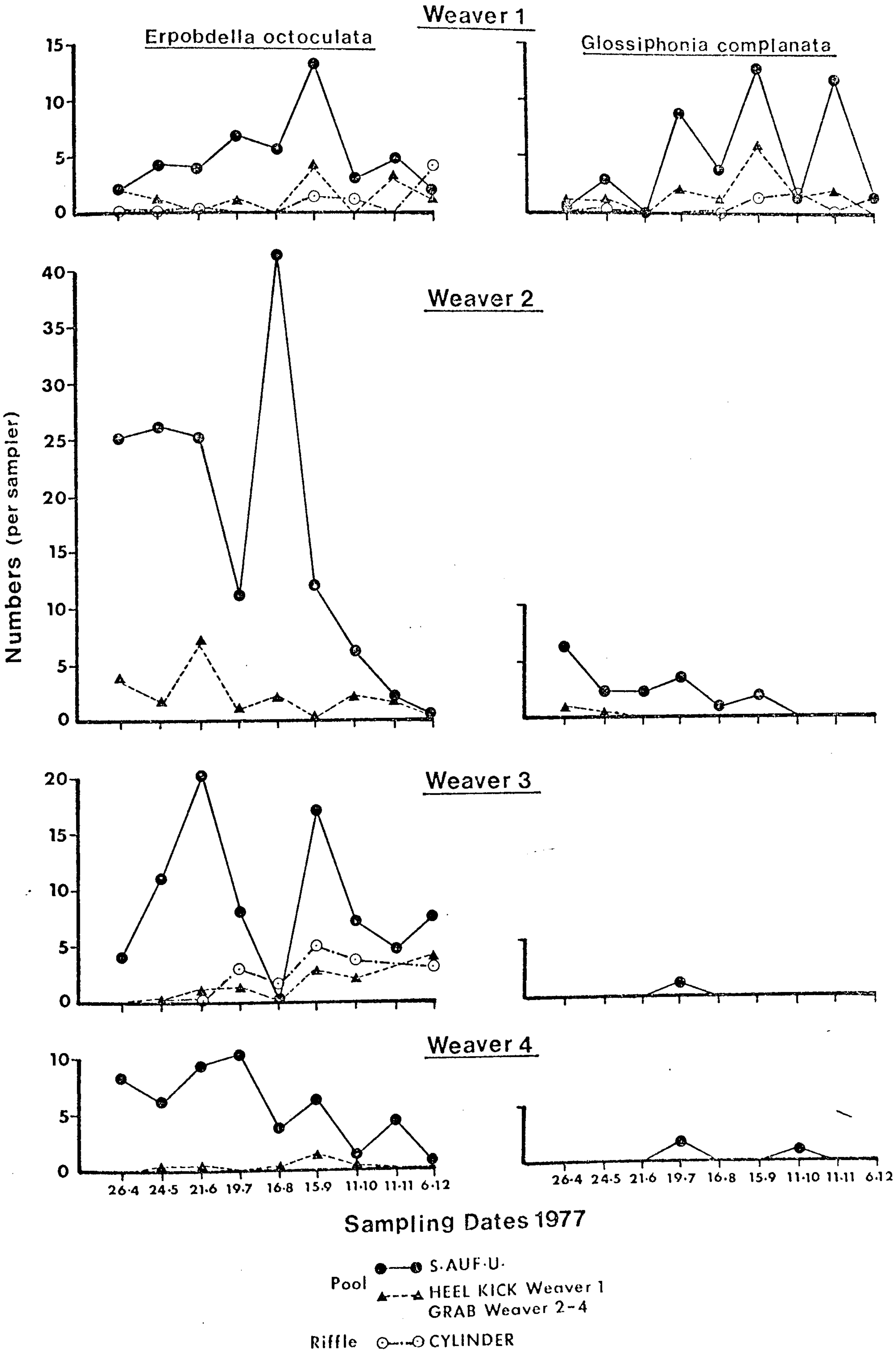
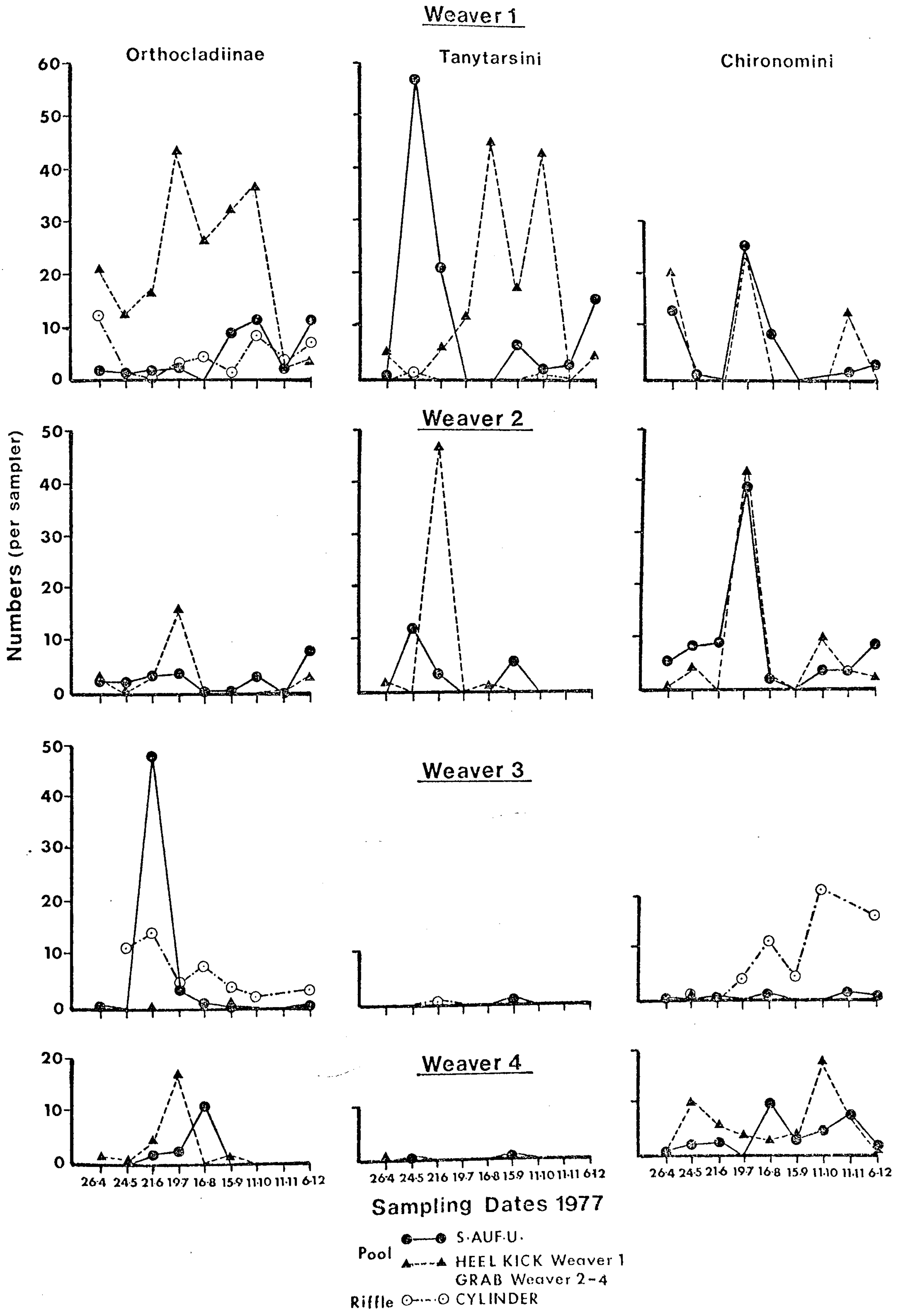


FIG.4.10 SEASONAL VARIATION OF CHIRONOMIDAE IN RIVER WEAVER.



The differences observed in water quality at each station on the Weaver can be assessed by calculating the scores obtained by the four pollution indexes. The seasonal variation obtained by the various sampling methodologies are shown in Figures 4.11 - 4.14. These show the tremendous variation obtained by the various sampling methods recorded throughout the season and emphasise the need to sample at least twice per year in order to obtain some type of representative score. At each station all the pollution indexes, with the occasional exception of the diversity index, appeared to show similar seasonal trends with their respective sampling method. In the depositing zones, S.Auf.U. sampling produced higher scores than by direct sampling of the benthos, while comparable results were obtained between S.Auf.U. and cylinder sampling. These differences are best reflected in the mean scores calculated at each station to show trends associated with water pollution (Figure 4.15). In these graphs the biotic and diversity indexes reflect the general decline in water quality between Stations 1, 2 and 3, while only the Chandler and B.M.W.P. scores reflect the slight improvement to Station 4. The Trent biotic index appears to be insensitive, while the diversity index over-reacts and indicates a vast improvement.

On the whole S.Auf.U. sampling was completely adequate for biologically surveying and monitoring the water quality of the River Weaver. The B.M.W.P. Score (depositing substratum score) worked well in association with the S.Auf.U.methodology and could be considered as an acceptable index for a Nationwide survey. In the Weaver, unfortunately riffles were not available at all the sites studied. In such rivers, although typically rhithron, the use of S.Auf.U. would

FIG. 4.11 SEASONAL VARIATION OF INDEXES AND SCORES OBTAINED BY DIFFERENT SAMPLING METHODS FROM RIVER WEAVER 1.

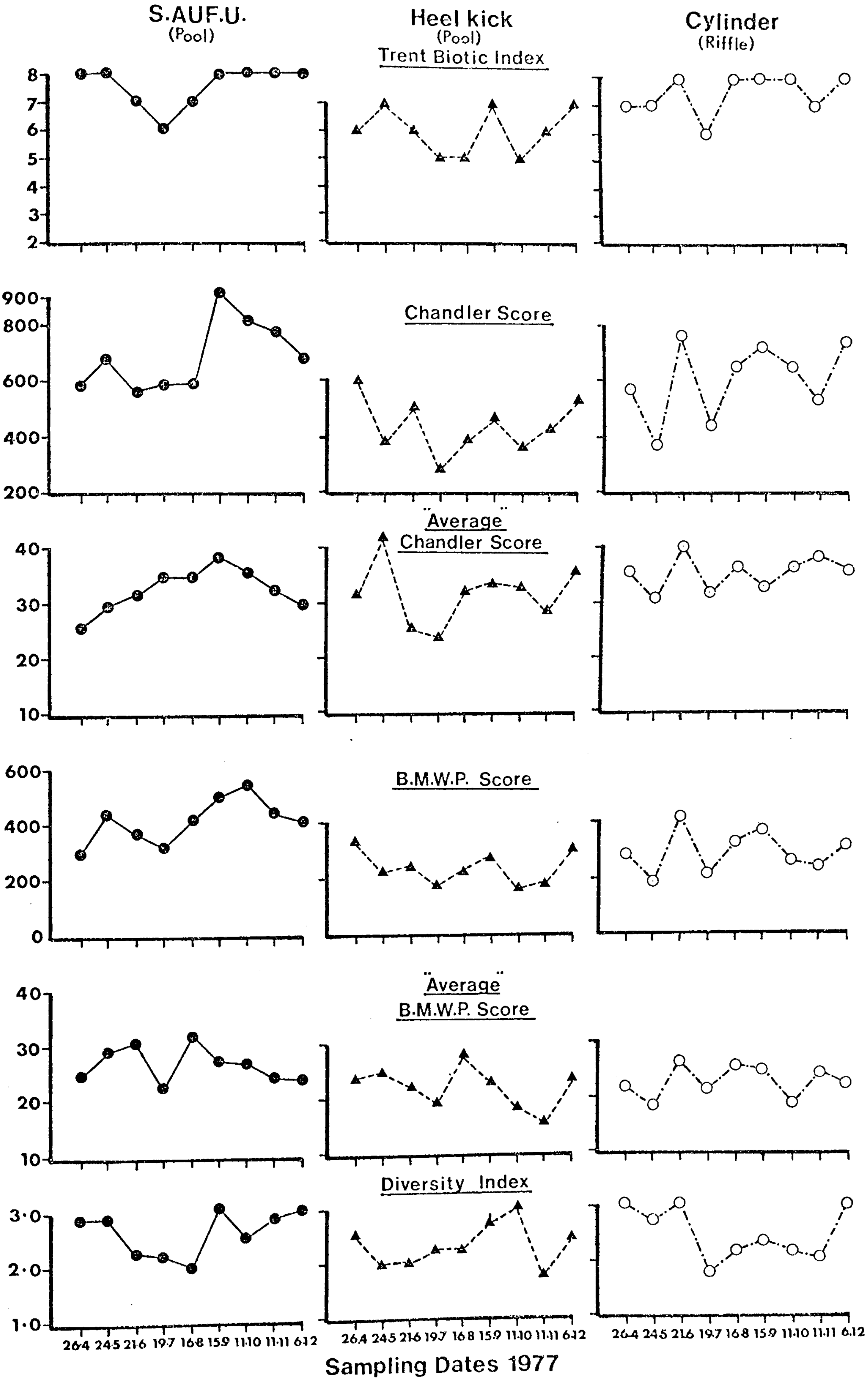


FIG. 4.12 SEASONAL VARIATION OF INDEXES AND SCORES OBTAINED BY DIFFERENT SAMPLING METHODS FROM RIVER WEAVER 2.

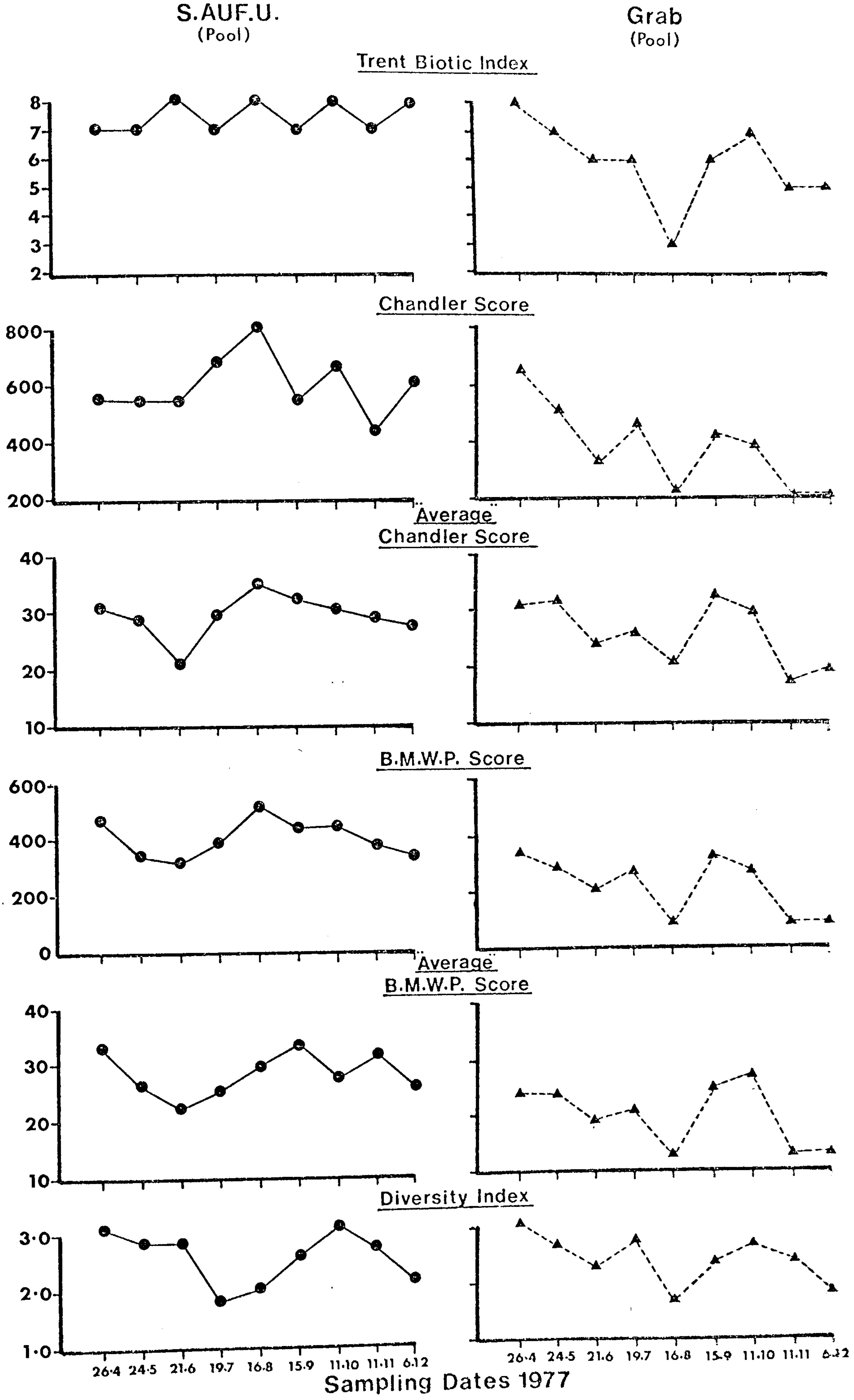


FIG.4.13 SEASONAL VARIATION OF INDEXES AND SCORES OBTAINED BY DIFFERENT SAMPLING METHODS FROM RIVER WEAVER 3.

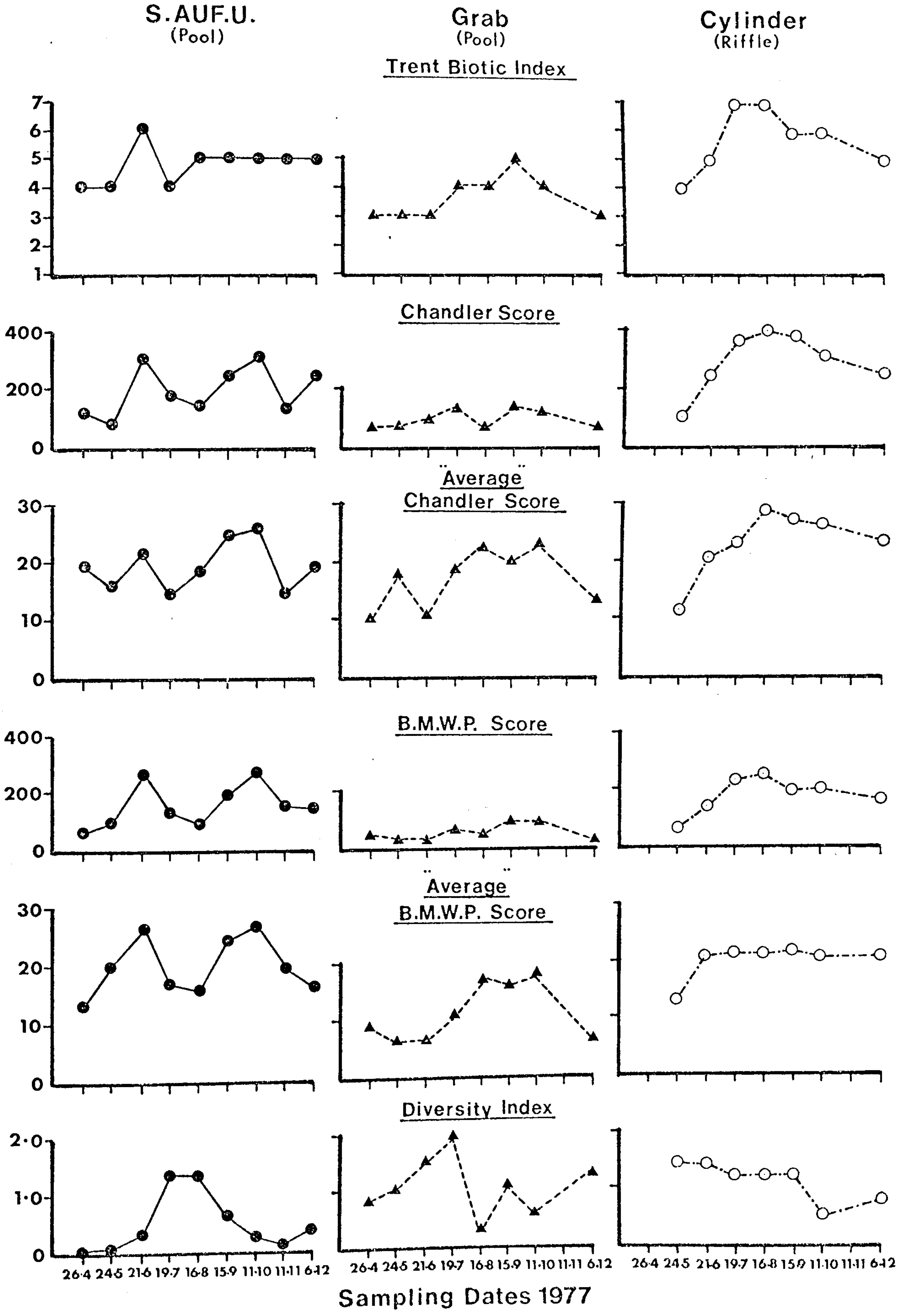


FIG. 4.14 SEASONAL VARIATION OF INDEXES AND SCORES OBTAINED BY DIFFERENT SAMPLING METHODS FROM RIVER WEAVER 4.

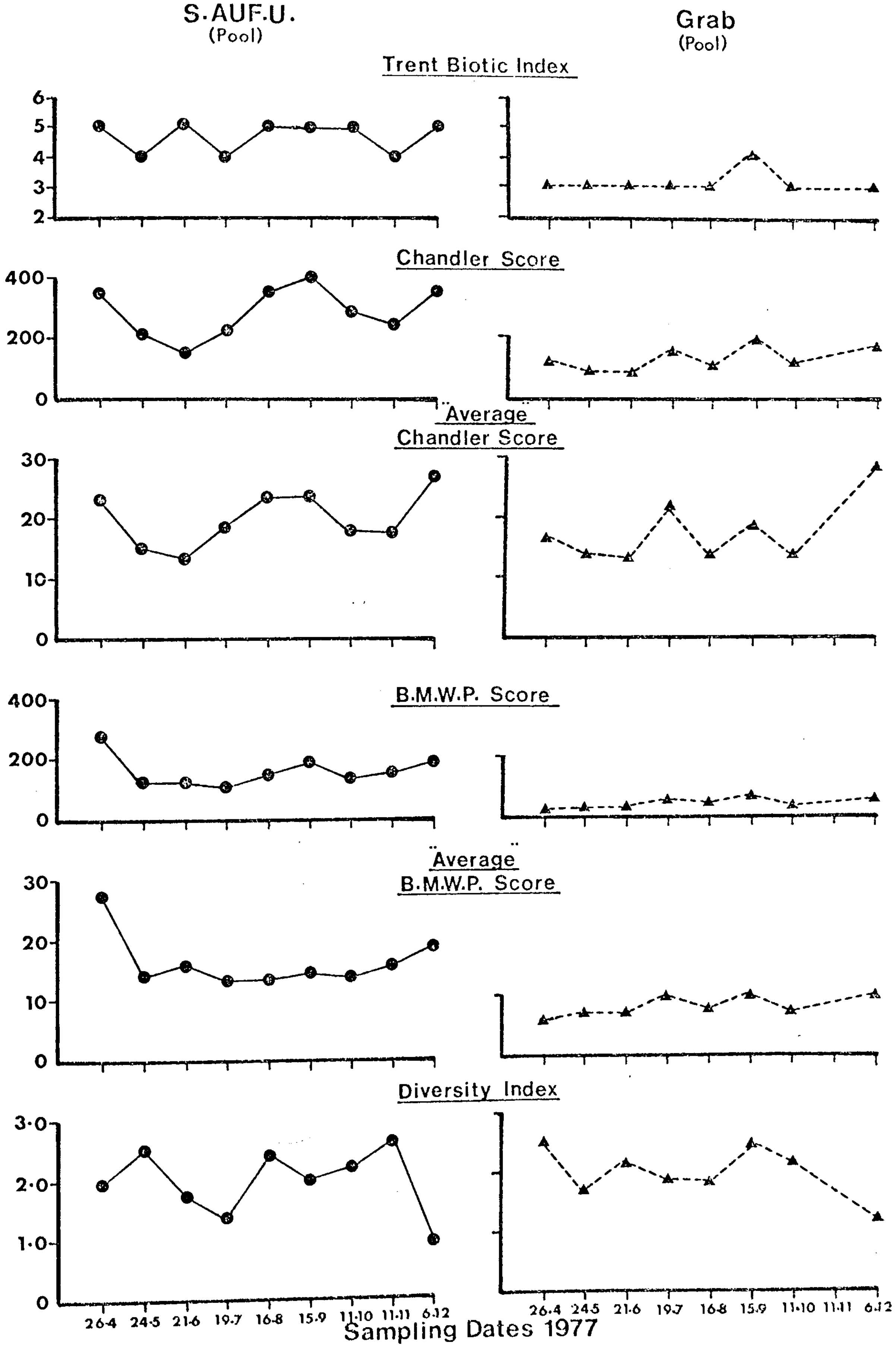
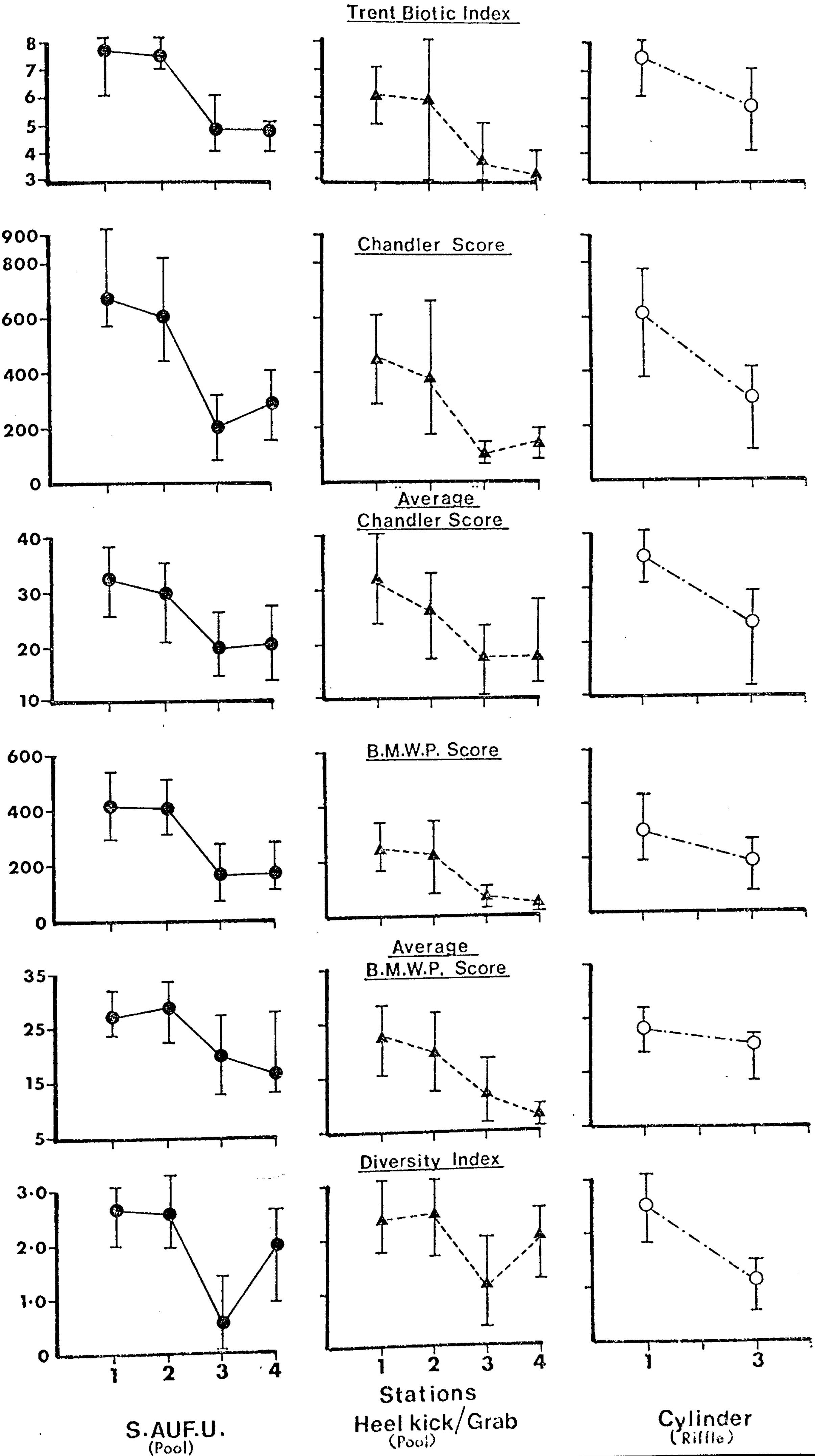


FIG.4.45 COMPARISON OF INDICES AND SCALES DERIVED FROM DATA OBTAINED BY DIFFERENT SAMPLING METHODS FROM RIVER WEAVER.



enable more detailed surveillance to be carried out than the use of the limited riffle sites available.

4.72 River Trent

4.721 Sampling stations

The Trent which rises to the north of Stoke and eventually flows into the Humber estuary, was examined in the West Midlands north of Birmingham and N.E. of Nottingham. During its passage through the Midlands, it receives a multitude of complex effluents from general urbanisation and industry and a whole series of thermal discharges.

In this investigation, four stations were studied on the River Trent (Figure 4.16) covering a length of approximately 42 km. Station 1 at King's Bromley was situated on a side branch of the main river, although clearly well downstream of any pollutional impact from Stoke-on-Trent. Between Stations 1 and 2, the pollutional load of the river increases due to the entry of the River Tame, while Stations 3 and 4 were situated in the lower potamon stretches of the river where some recovery had taken place.

The substratum of the Trent consisted mainly of gravel in the upper reaches, while silt and clay were a common feature of the lowland sites. These features are summarised in Table 4.7.

The sampling programme started at the beginning of April for Trent 1 and 2 (May for Trent 3 and 4) and continued until the middle of December, 1977.

FIG.4.16 MAP OF RIVER TRENT AND MAJOR TRIBUTARIES,MIDLANDS,SHOWING THE DISTRIBUTION OF SAMPLING SITES.

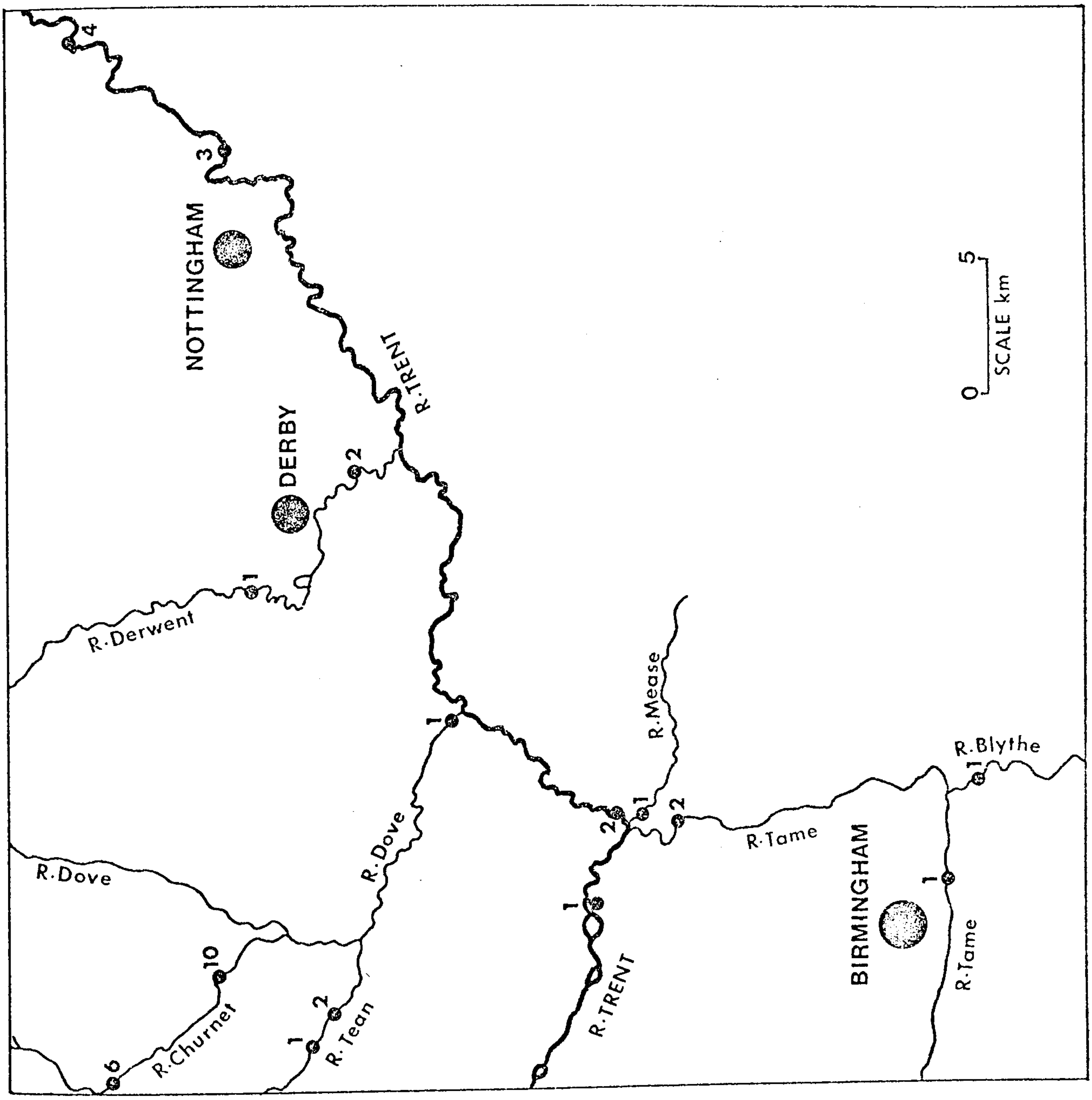


Table 4.7 The mean and range of current velocities and nature of the substratum found in the River Trent

Station	Riffle		Pool	
	Nature of Substratum	Current Velocity cm.sec^{-1}	Nature of Substratum	Current Velocity cm.sec^{-1}
1	Gravel	49.4 (15.6-102.1)	Silt + Clay	29.0 (Negl.-76.8)
2	Gravel	57.3 (32.6- 84.2)	Gravel	23.8 (5.3-54.7)
3			Silt + Gravel	Negl.
4	Silt + Gravel	41.5 (20.7-59.3)	Silt + Clay	4.1 (Negl.-16.4)

4.722 Results and Discussion

The chemical data presented in Annexe 2 (Table 14) and Annexe 3 (Table 23) and summarised in Table 4.8 reflect the overall water quality and pollution encountered in the Trent. The effect of the Tame between Stations 1 and 2 is to substantially decrease the dissolved oxygen and raise the levels of BOD_5, suspended solids, ammonia and nitrate. The difference between Station 2 and Stations 3 and 4 reflects the gradual improvement in water quality over the 30 km. distance between them.

The numbers of taxa taken monthly by the different sampling methods are shown in Figure 4.17. In this figure the results are presented both as the means of the three sampling units (S.Auf.U. and cylinder) and as the total numbers of taxa taken in all three units for each sampling method.

Table 4.8 Physico-chemical data expressed as mean and range for the River Trent (mg.l^{-1})

STATION	Analysis	S.S.	D.O.	BOD$_5$	Ammonia	Nitrate
1	ASTON	13.8 (1 - 27)	9.9 (8.9-11.2)		1.69 (0.7-5.1)	8.5 (6.7-10.6)
	WA	25.0 (1.8- 84)	10.3 (7.3-15.1)	4.3 (1.8- 9.6)	0.3 (0.1-0.8)	9.4 (7.0-12.0)
2	ASTON	21.6 (10 - 52)	7.2 (6.2- 9.1)		2.92 (1.0-7.0)	14.1 (10.2-26.8)
	WA	33.0 (10 -256)	7.6 (4.3-10.2)	10.2 (4.0-17.2)	1.3 (0.3-2.6)	12.5 (10.0- 19.2)
3	WA	26.0 (8 - 74)	9.1 (7.5-10.6)	6.4 (3.8-15.8)	0.7 (0.3-1.5)	9.4 (6.8-12.0
4	WA	29.0 (8 -178)	9.1 (7.1-11.5)	6.2 (2.5- 9.8)	0.5 (0.1-1.1)	10.1 (7.5-15.7)

The detailed macroinvertebrate data collected from the River Trent are presented in Annexe 4 (Tables 42 - 48).

At Stations 1 and 2, above and below the entry of the River Tame respectively, the mean numbers of taxa taken on the S.Auf.U. in the pools were very similar to the number taken by cylinder from the riffles, even showing similar seasonal variations. Direct sampling of the pools by heel kick was possible at only two Stations, 2 and 3, the pools being too deep at 1 and 4 for direct sampling. At Station 2 where there was an associated riffle, the numbers of taxa obtained by direct sampling of the pools and accumulated by S.Auf.U. was very similar, following similar seasonal trends. However, at the lower Station 3 where there was no riffle, larger numbers were taken by S.Auf.U. than by direct heel kick sampling. At Station 4, problems were encountered with unpredictable rising water levels, which made retrieval of S.Auf.U. impossible at times. Vandalism due to 'juvenile fisherman' also caused a problem and finding a site inaccessible from the general public in this reach was almost impossible. The results obtained from this site were therefore extremely sparse. Generally the number of taxa taken by S.Auf.U. was similar to Station 3, and reflect the improvement in water quality compared to the station below the Tame.

All methods reflected the deterioration in water quality between Stations 1 and 2 due to the entry of the River Tame in the reduction in numbers of taxa followed by a recovery at the lower stations.

Similarity coefficients (Figure 4.18) calculated to compare S.Auf.U. sampling in pools with cylinder sampling in riffles at Stations Trent 1 and 2,revealed an annual mean value of 69% by Sorensen's method and 50% by Czekanowski's method. On the whole, both methods followed similar seasonal patterns. A comparison of S.Auf.U. with heel kick sampling at Trent 2 and 3 produced high mean values at Trent 2 (66% Sorensen, 54% Czekanowski), but much lower values at Trent 3 (54% Sorensen, 21% Czekanowski). In the upper sites, the poolssections are incluenced considerably by drift from upstream riffles, resulting in higher coefficient values, while in the lowland sites, where riffles are sparse, such an influence is markedly reduced.

FIG.4.17 NUMBERS OF TAXA COLLECTED MONTHLY ON S.AUF.U. IN POOLS AND DIRECT SAMPLING OF POOLS AND ASSOCIATED RIFFLES,RIVER TRENT.

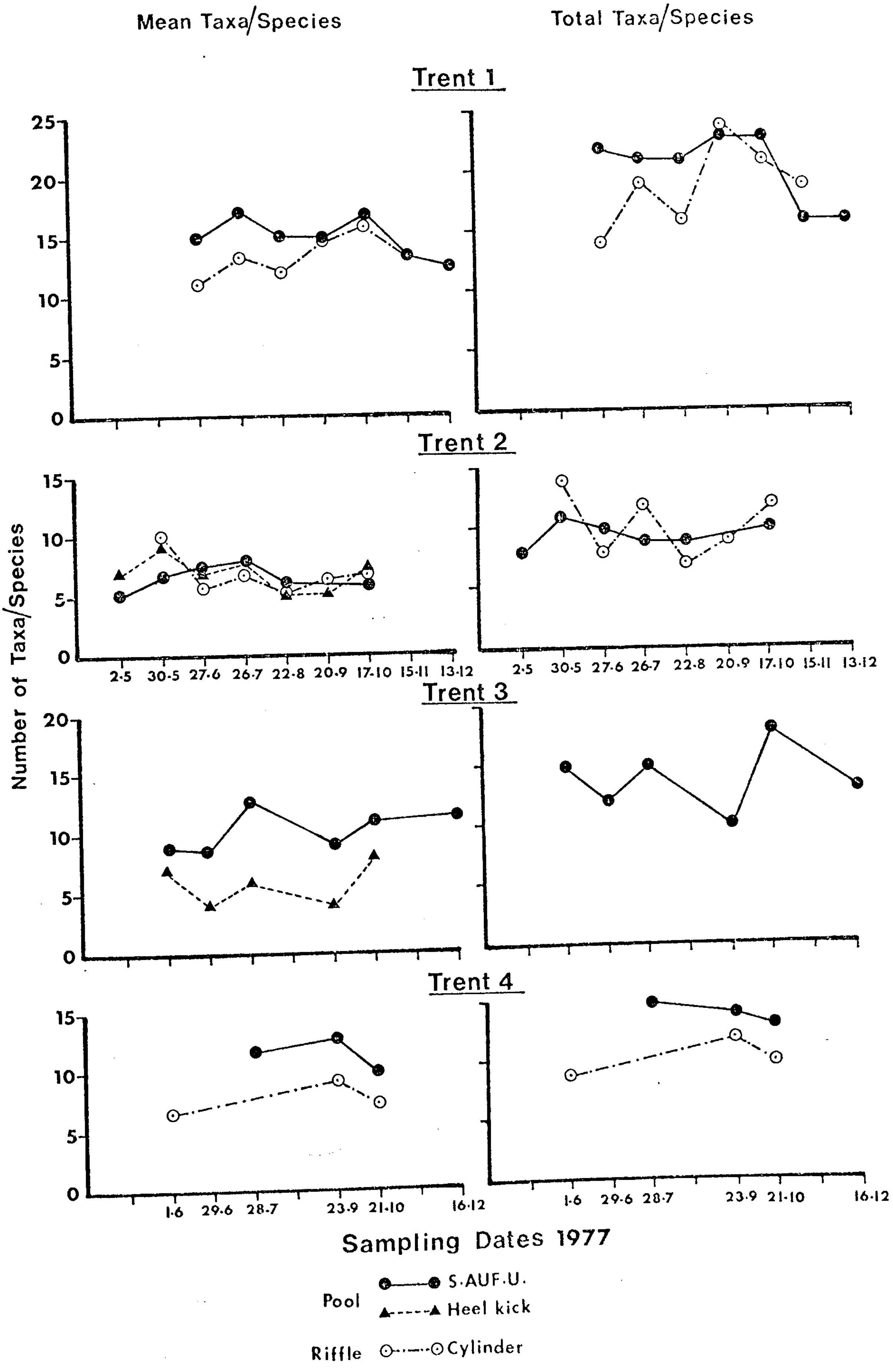
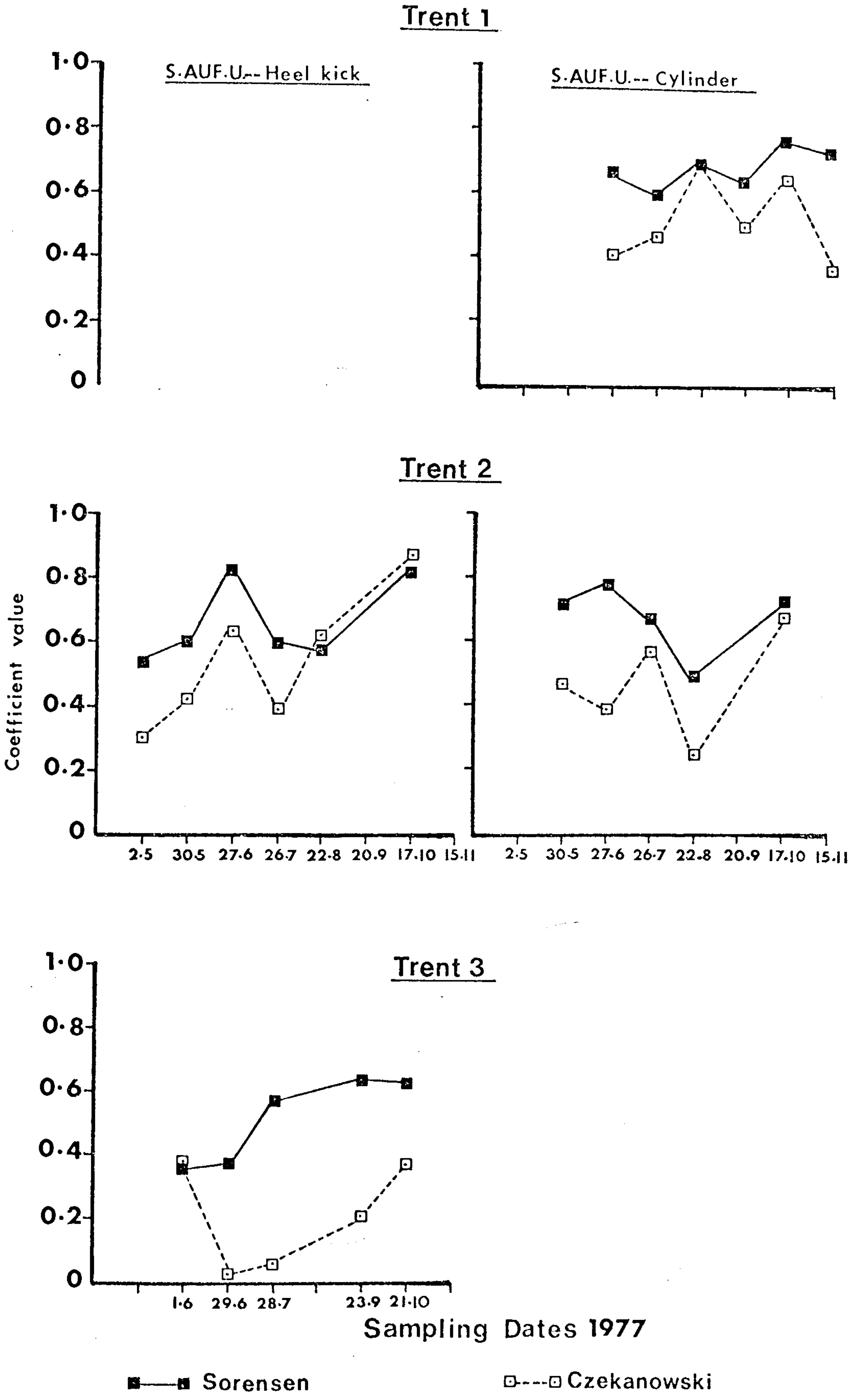


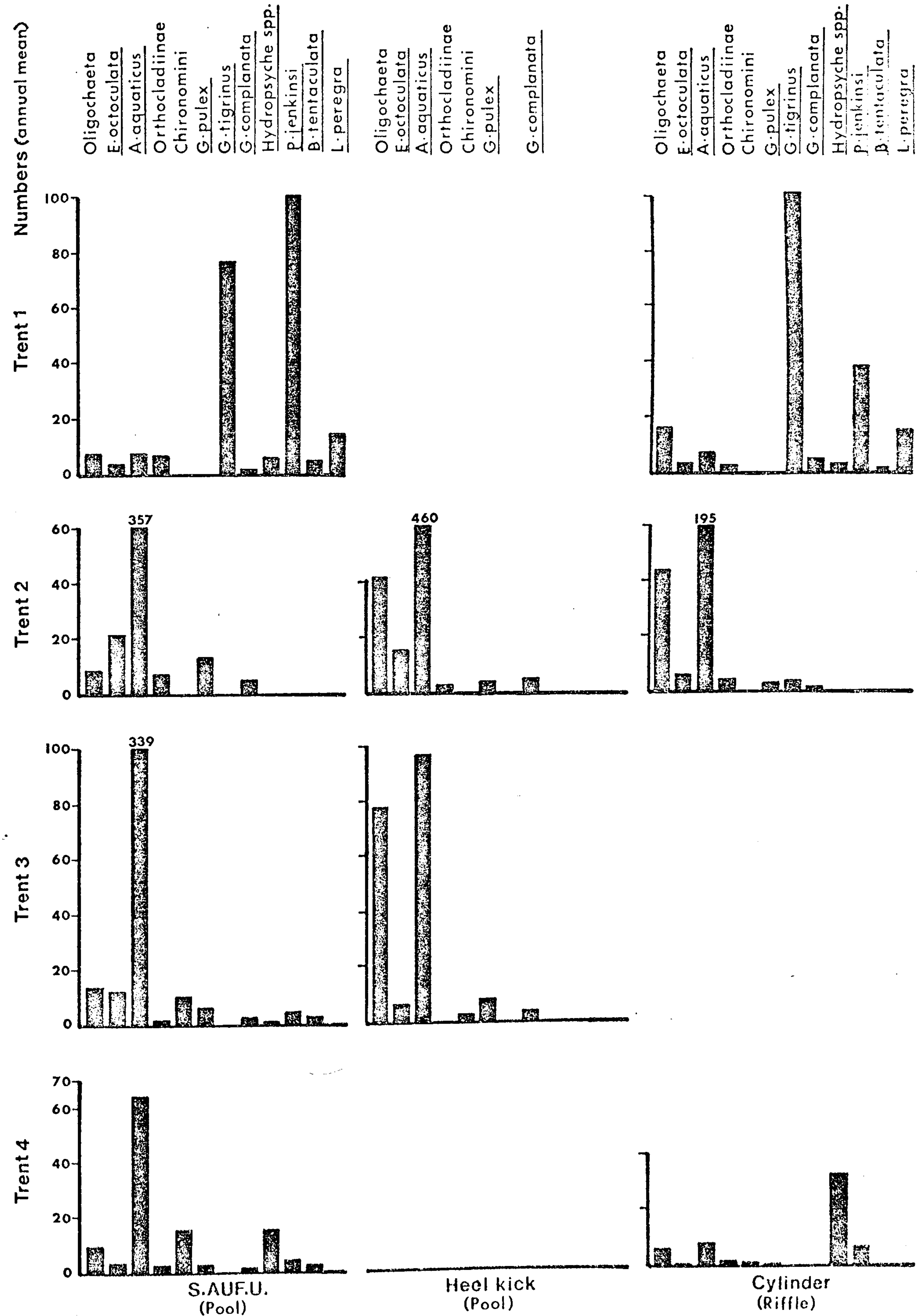
FIG. 4.18 COMPARISONS OF SAMPLING BY S.AUF.U. WITH DIRECT SAMPLING OF POOLS AND ASSOCIATED RIFFLES USING SIMILARITY COEFFICIENT ANALYSIS, RIVER TRENT.



At Trent 3, S.Auf.U. sampling tended to collect larger numbers of taxa and individuals than heel-kick sampling of pools due to the general nature of the substratum found at this site. S.Auf.U. were usually colonised by attached forms such as Platyhelminthes, Hirudinea and Mollusca which were rarely recorded by heel-kick sampling, and tended to concentrate higher numbers of Tubificidae, Asellus aquaticus and Chironomini. With such data processing techniques, the success of S.Auf.U. sampling in lowland conditions is high-lighted.

Figure 4.19 shows the distribution of the major taxa recorded at the four stations as indicated by the different sampling methods. S.Auf.U. was the only method applicable at all four stations. There was a close similarity between the fauna of the S.Auf.U. in the pools and the riffle fauna as sampled by cylinder and also in the faunistic changes which took place between Stations 1 and 2. In each case there was a reduction in the numbers of taxa the same more tolerant taxa remaining. At Trent 1, both sampling methods recorded large densities of Gammarus tigrinus and Potamopyrgus jenkinsi throughout most of the year, while other taxa with lower densities reached their height during the summer months. These minor groups were dominated by Lymnaea peregra, Physa fontinalis, tubificids, Orthocladiinae and some of the Hirudinea. The unstable nature of the substratum in the pool sections, made S.Auf.U. a suitable colonising habitat for Hydropsyche spp. and occasionally other Trichoptera. With both sampling methods, A.aquaticus was never recorded as a dominant species. With the entry of the River Tame, A.aquaticus became the dominant species at Trent 2, a situation recorded by all sampling methods. The numbers of tubificids and naids increased together with the

FIG. 4.19 RELATIVE ABUNDANCE OF MAJOR TAXA AT FOUR STATIONS ON THE RIVER TRENT AS INDICATED BY DIFFERENT SAMPLING METHODS.

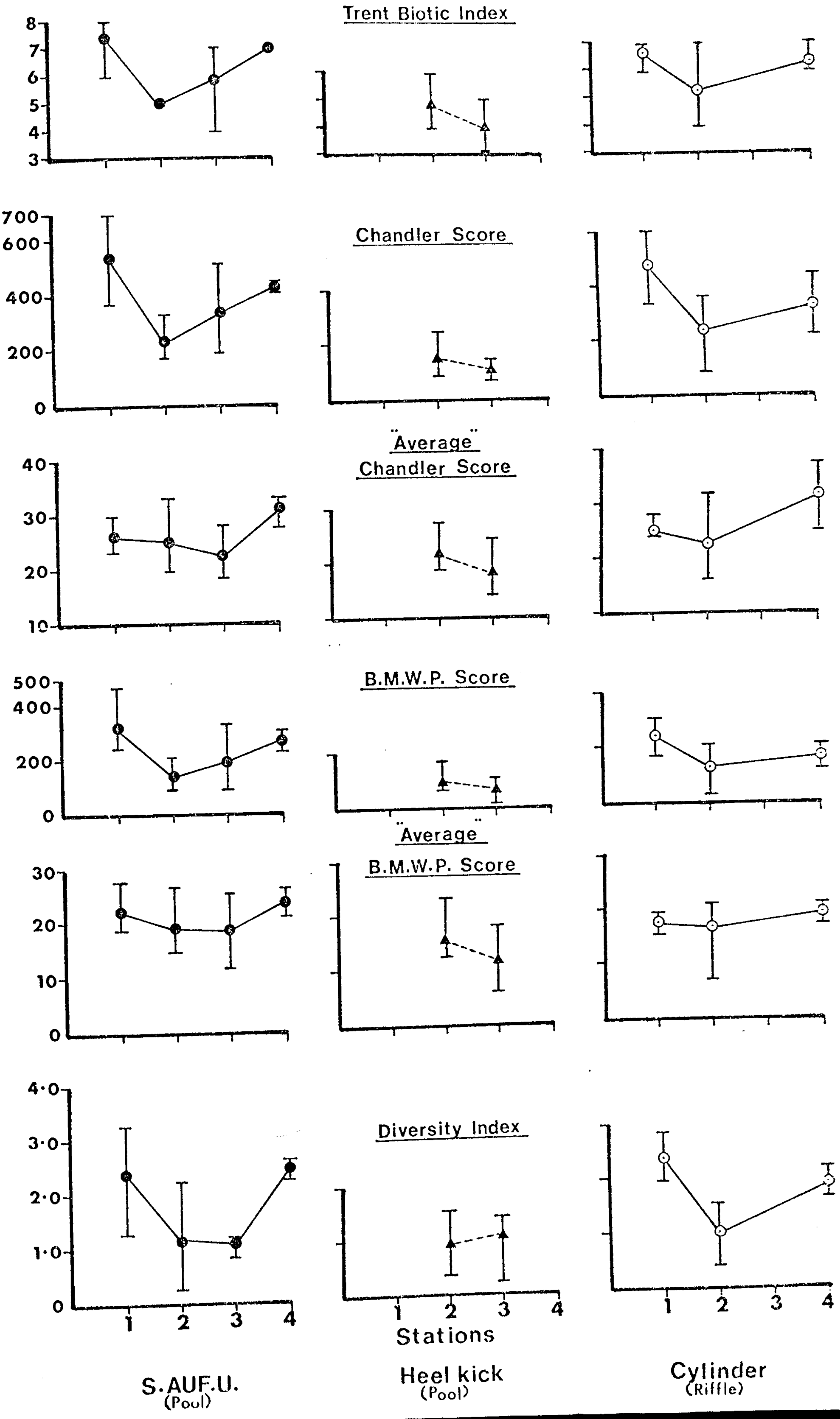


leeches Eropbdella octoculata and Glossiphonia spp.. The numbers of G.tigrinus decreased, although G.pulex increased slightly probably as a result of the entry of the River Mease 1 km. upstream. With every sampling method , there was a significant decrease in the species diversity.

With the improved conditions in the lower stations, the displaced taxa returned, this being most evident in the case of the S.Auf.U.. A.aquaticus although still recorded in similar proportions at Trent 3, as Trent 2, the silty nature of the substratum prevented any significant decrease in abundance, a condition also observed with the tubificids. The improvement in water quality at this site was reflected in the numbers of G.pulex, Mollusca and Trichoptera collected on the S.Auf.U. At Station 4, although problems were encountered with sampling methodology, the S.Auf.U. method recorded an increase in species diversity as a result of improving water quality. This was typified by an increase in the numbers of Ephemeropteran species and numbers of Hydropsyche spp. and Ancylus fluviatilis.

The changes associated with pollutional impact and recovery are clearly represented when the data are processed to give indexes and scores (Figure 4.20). S.Auf.U. and cylinder sampling demonstrate in most cases the sequence of events occurring along the length of the Trent. Heel-kick sampling, however, appears to indicate a deterioration between Trent 2 and 3 by all biotic score methods, except the diversity index. In the case of the B.M.W.P. Score, the recovery in the lower stations was best reflected by the S.Auf.U. data. The 'Average' score values calculated for Chandler and B.M.W.P. appear to be too insensitive to changes in water quality and pollution.

FIG. 4.20 COMPARISON OF INDEXES AND SCORES DERIVED FROM DATA OBTAINED BY DIFFERENT SAMPLING METHODS FROM THE RIVER TRENT.



4.73 River Tame

4.731 Sampling stations

The River Tame is probably one of the dirtiest rivers in England and as seen in section 4.72, has a dramatic impact on the pollutional load of the River Trent. The Tame is a fairly rare river because it is heavily polluted from its source in the Black Country and flows through Birmingham (Figure 4.16) to its confluence with the Trent. The Tame receives discharges from sewage treatment along its length, the major one coming from Minworth east of Birmingham, and during periods of dry weather 86% of the river can be made up of treated effluent. The Tame also suffers from motorway run off, waters percolating from industrial storage areas and waste tips and general industrial effluents.

Two stations were selected for study on this polluted tributary of the River Trent. Tame 1 at Castle Bromwich was 1 km. upstream of Birmingham's sewage works effluent at Minworth, where the river had slightly improved from the gross organic pollution received in the upper reaches. Tame 2 at Elford was 20 km. downstream of the Minworth effluent and some distance upstream of its confluence with the River Trent.

The substratum of Tame 1 consisted mainly of loose unstable gravel, which was constantly being scoured, while Tame 2 was relatively stable and allowed silt to accumulate (Table 4.9).

At times of high flow, the riffle sections were impossible to sample, while the pool sections, although rapidly flowing were constantly accessible. The sample programme consequently continued from April until December, 1977.

Table 4.9 The mean and range of current velocities and nature of the substratum found in the River Tame

Station	Riffle		Pool	
	Nature of Substratum	Current Velocity cm.sec^{-1}	Nature of Substratum	Current Velocity cm.sec^{-1}
1	Loose Gravel	45.1 (23.6-74.8)	Loose Gravel	48.9 (20.3-95.0)
2	Gravel	34.2 (19.1-60.5)	Silt + Clay	3.4 (Negl.-16.4)

4.732 Results and Discussion

The chemical data presented in Annexe 2 (Table 15) and Annexe 3 (Table 24) are summarised in Table 4.10. This reflects the overall poor water quality encountered along the length of the study area.

Table 4.10 Physico-chemical data expressed as mean and range for the River Tame (mg.l^{-1})

Station	Analysis	S.S.	D.O.	BOD_5	Ammonia	Nitrate
1	ASTON	24.6 (16- 35)	7.2 (6.5-8.4)		5.83 (1.8-8.9)	10.7 (7.8-13.6)
	WA	42.0 (13-256)	7.6 (3.3-9.6)	12.1 (5.5-33)	5.5 (1.6-10)	7.6 (0.2-11)
2	ASTON	23.6 (10- 46)	6.8 (5.2-8.8)		3.72 (1.7-7.5)	14.4 (5.1-25.6)
	WA	21.0 (21.0)	8.9 (8.9)	10.5 (10.5)	3.7 (3.7)	0.7 (0.7)

The chemical analysis shows a moderate oxygen content present at both sites which would suggest only slight oxygen depletion occurring in the river. Suspended solids and BOD_5 are slightly higher in the upper sections than further downstream, but it is the abnormally high ammoniacal nitrogen levels persisting at Tame 1 which indicates grossly polluted conditions. At Tame 2 the river appears to have slightly improved as indicated by lower ammonia levels and the fact that at least 8 coarse fish species including Dace (_Leuciscus leuciscus_) and Chub (_Leuciscus cephalus_) are known to have now become established there.

The detailed macroinvertebrate data collected from the River Tame are presented in Annexe 4 (Tables 49-52).

The number of taxa taken each month by different sampling methods at the two stations is shown in Figure 4.21. At both stations the mean number of taxa taken by S.Auf.U. and heel-kick sampling (Tame 1) in the pools were similar to the number taken by cylinder from the riffles. However, an examination of the total number of taxa taken by three replicate samples shows that S.Auf.U. collected higher number of taxa than cylinder sampling. At Tame 1 this increase was mainly due to _L.peregra_ and occasionally Hirudinea, while at Tame 2 Coleoptera and Odonata made up the numbers.

Similarity coefficients at Tame 1 appeared to increase throughout the season and were generally much higher than values recorded at Tame 2. Again the abundance of particular species on the S.Auf.U. resulted in low Czekanowski values being recorded in comparison with heel-kick and cylinder sampling.

FIG. 4.21 NUMBERS OF TAXA COLLECTED MONTHLY ON S.AUF.U. IN POOLS AND DIRECT SAMPLING OF POOLS AND ASSOCIATED RIFFLES WITH SIMILARITY COEFFICIENTS , RIVER TAME.

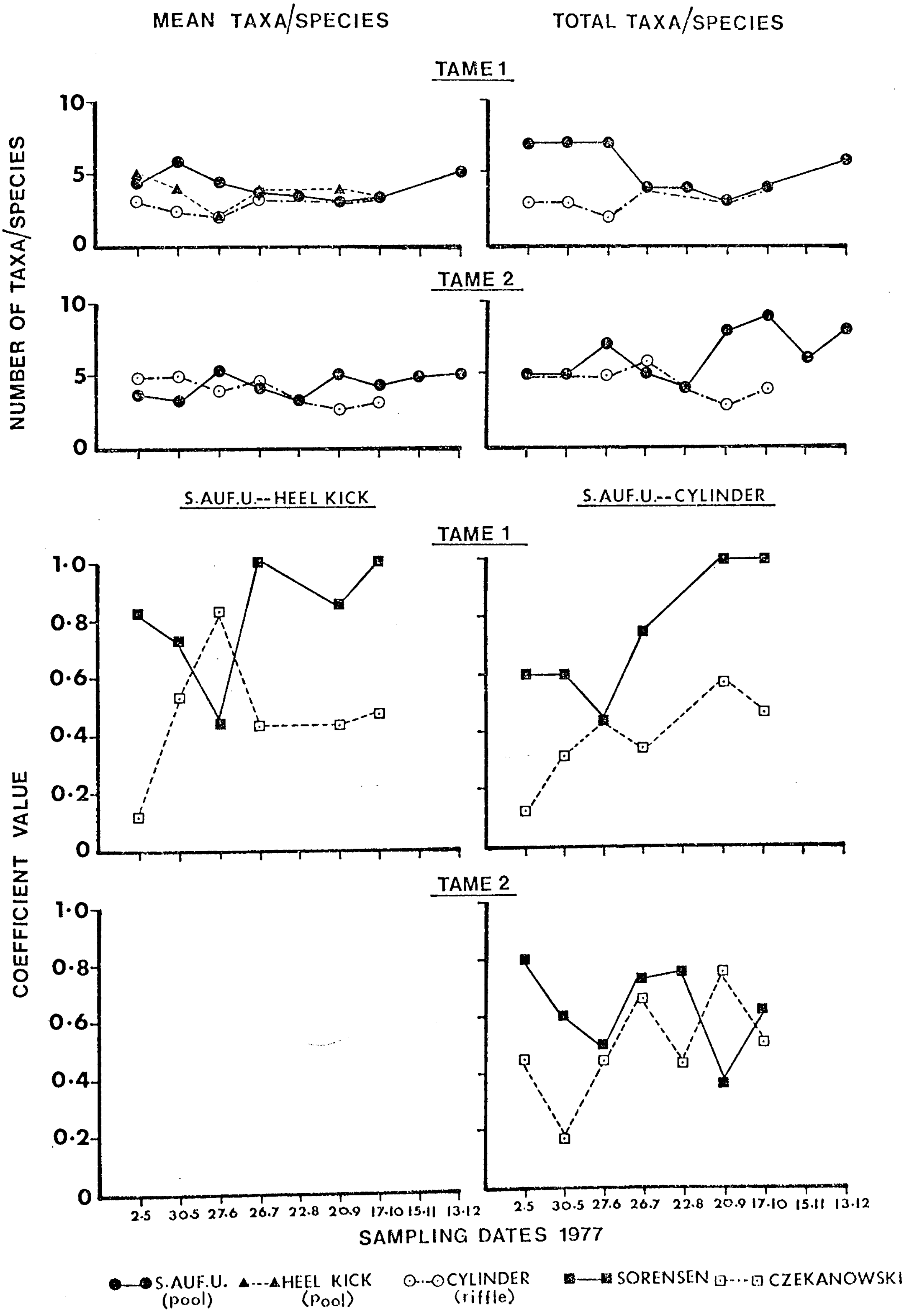


Figure 4.22 compares the relative abundance of the common taxa at the two stations as illustrated by the different sampling methods. At station 1, although similar species are present, whereas Tubificidae were dominant in the natural substrata of both pool and riffle, A.aquaticus was the dominant species in the S.Auf.U.. This was probably related to the unstable nature of the top layer of the substratum. While tubificids can penetrate into the lower hyporheic zone, Asellus would seek cover in more stable areas and consequently the S.Auf.U. would serve this purpose. At Tame 2, with the relative improvement in water quality both S.Auf.U.and cylinder sampling of the riffle recorded the appearance of the leech Erpobdella octoculata, a substantial increase in the Asellus population and a corresponding marked decrease in Tubificidae. Although an increase in the numbers of A.aquaticus is often associated with a deterioration in water quality, it is often recorded in a zone which is recovering from gross organic pollution. The pool at Station 2 was too deep to be sampled by conventional methods and as a consequence the S.Auf.U.acted as a suitable monitoring tool. By comparing S.Auf.U. with cylinder sampling similar changes in community structure are observed in the Tame.

The extent to which these changes are measurable is observed in the preceeding calculated biotic and diversity indexes. Figures 4.23 and 4.24 show the seasonal variation occurring at the two sites. Although the Tame is severely polluted in its upper reaches, one can detect changes using these indexes. This is particularly reflected by the Chandler score and the diversity index. The species contributing to this variation are only incidental species occurring

FIG. 4.22 RELATIVE ABUNDANCE OF MAJOR TAXA AT TWO STATIONS ON THE RIVER TAME AS INDICATED BY DIFFERENT SAMPLING METHODS.

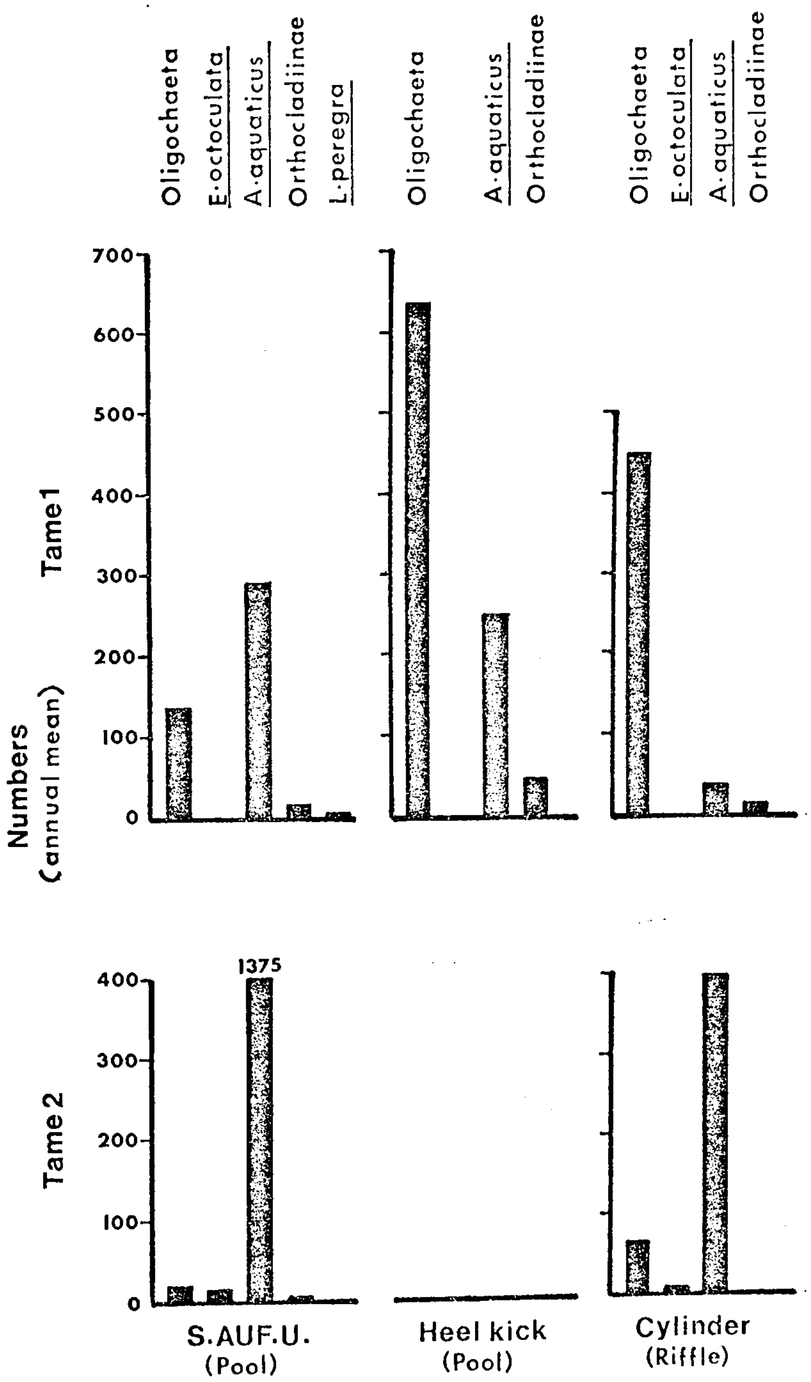


FIG.4.23 SEASONAL VARIATION OF INDEXES AND SCORES OBTAINED BY DIFFERENT SAMPLING METHODS FROM RIVER TAME 1.

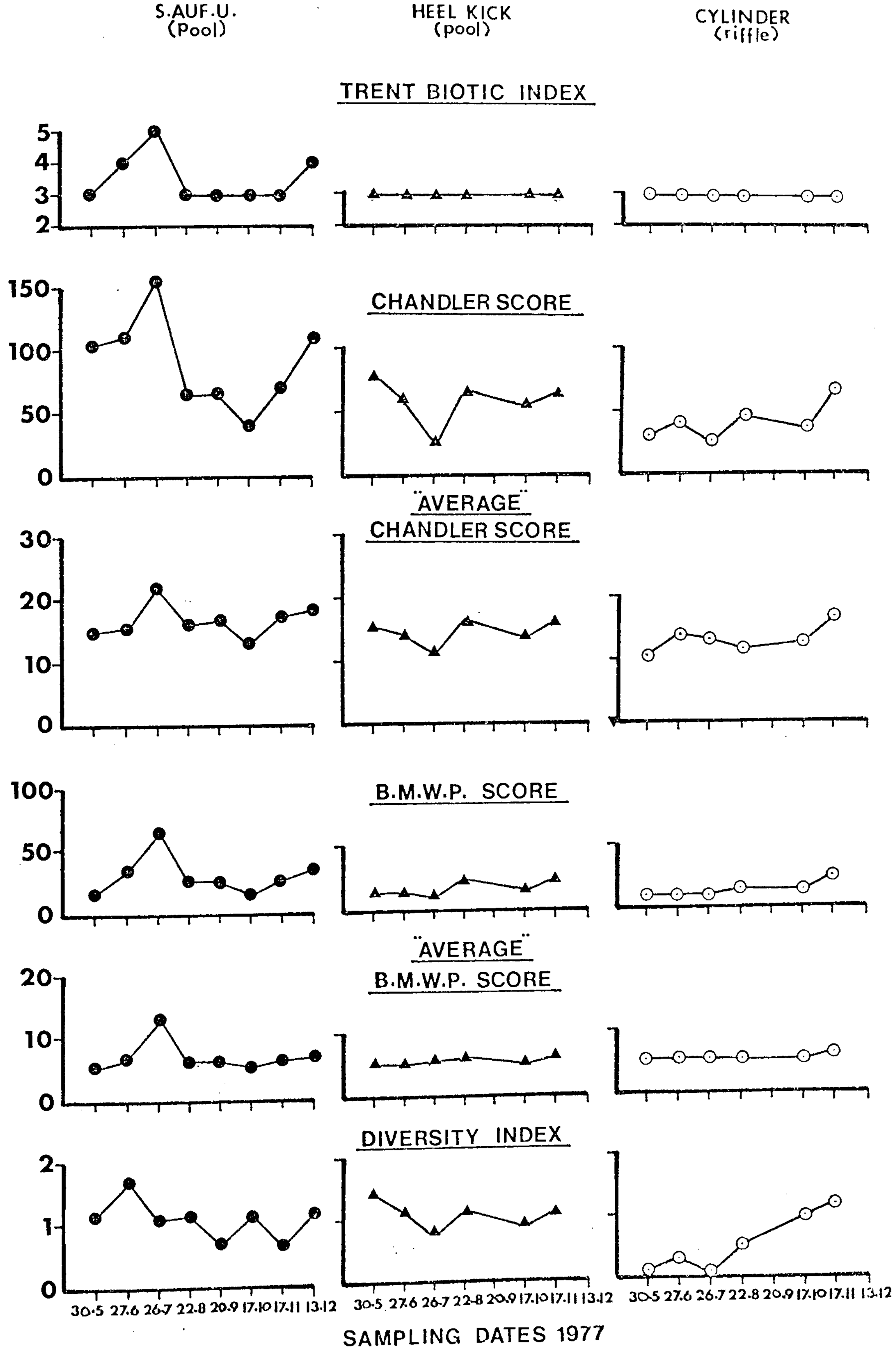


FIG. 4.24 SEASONAL VARIATION OF INDEXES AND SCORES OBTAINED BY DIFFERENT SAMPLING METHODS FROM RIVER TAME 2.

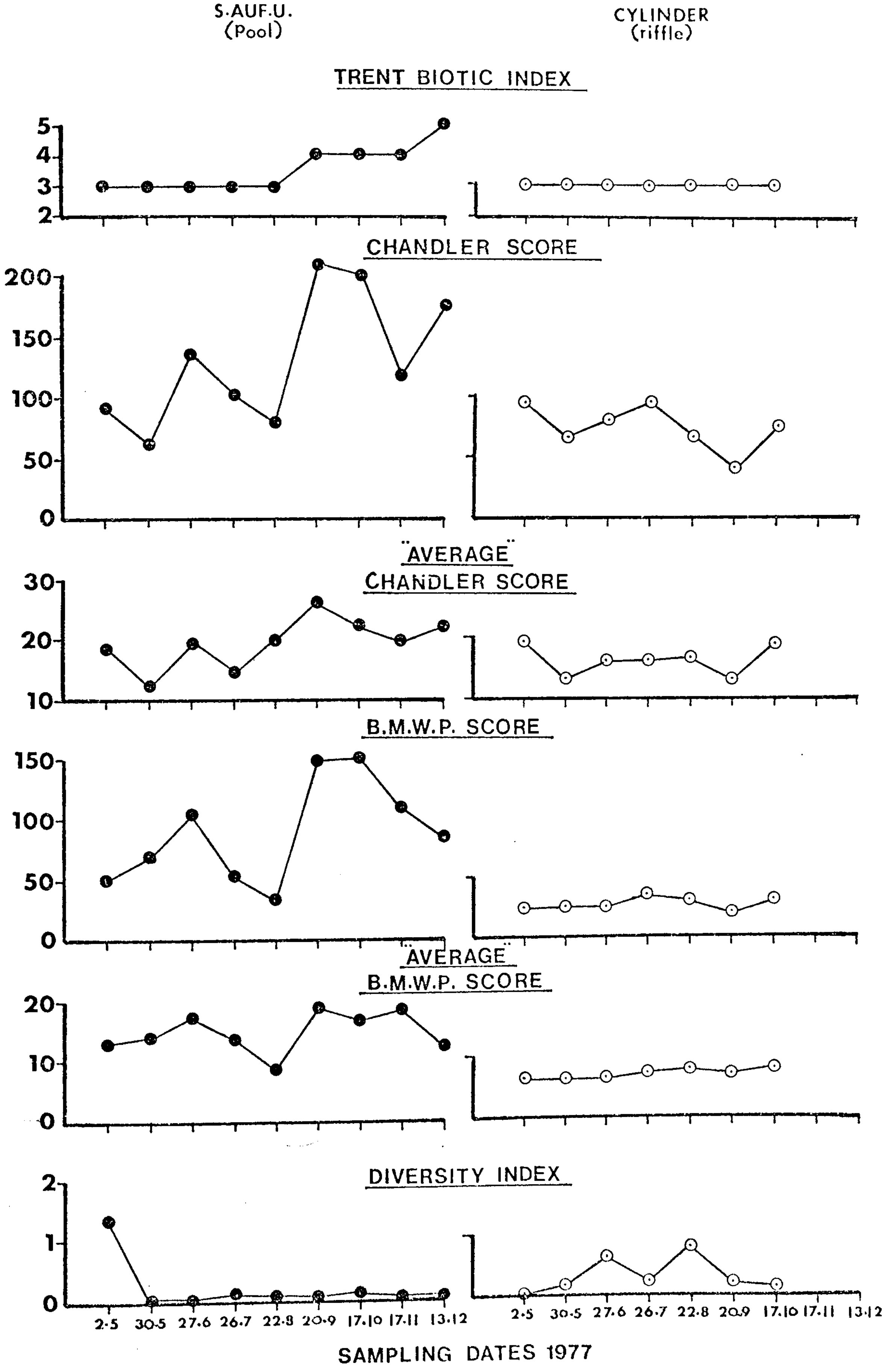
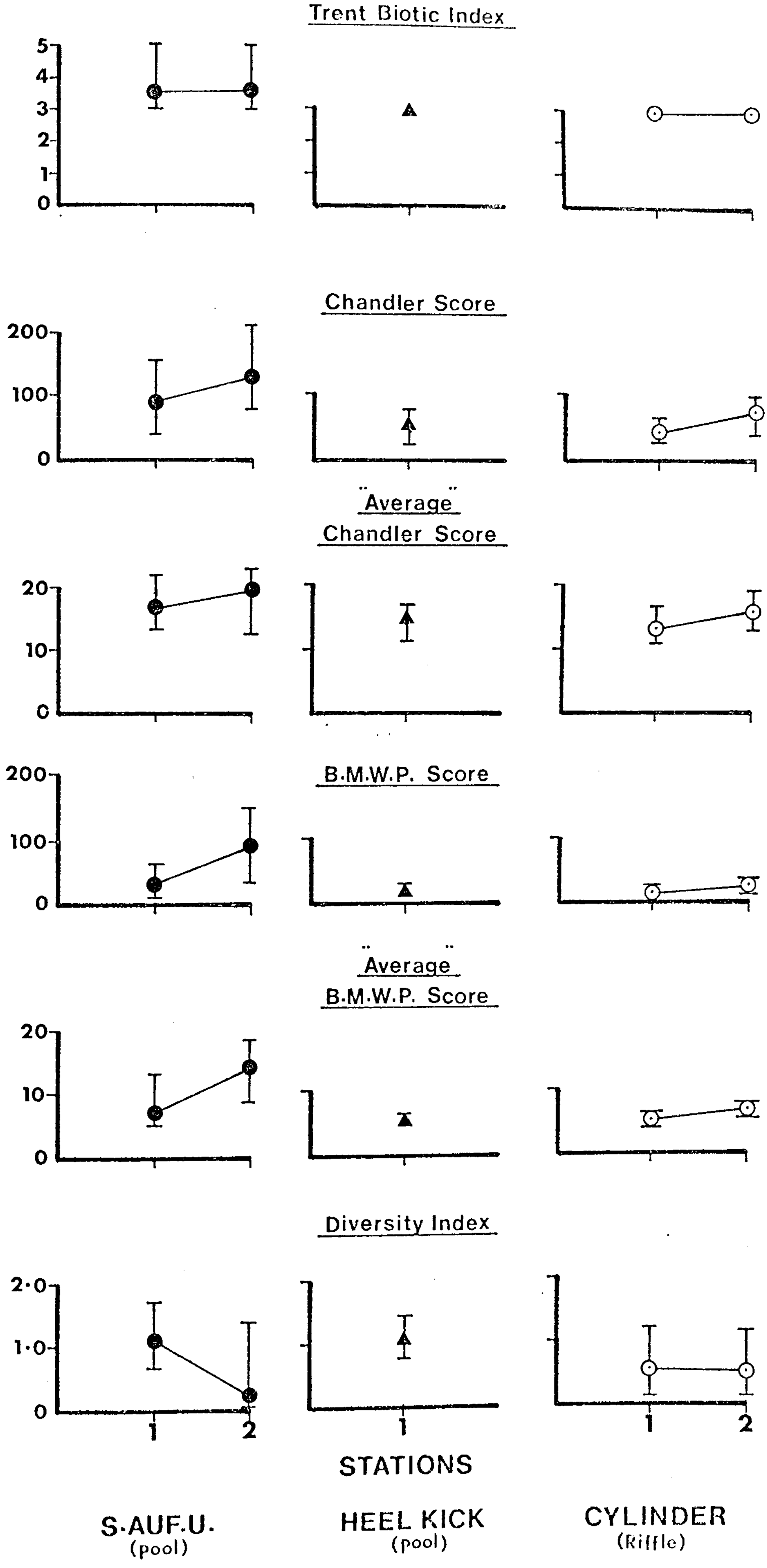


FIG. 4.25 COMPARISON OF INDEXES AND SCORES DERIVED FROM DATA OBTAINED BY DIFFERENT SAMPLING METHODS FROM RIVER TAME.



occasionally in samples, possibly as a result of drift/sample variation/season of the year. In the lower section of the Tame, where some recovery has taken place, this variation is more marked, but it is more likely to be due to the seasonal variation of established species than chance due to drift or sample variation. Again the Chandler and B.M.W.P. score reflect this variation, while the sensitive diversity index is suppressed due to the high abundance of one species and low numbers of other species.

In Figure 4.25, the annual mean scores reflect the improvement in water quality recorded at Tame 2. The Trent Biotic Index, however, does not indicate this improvement since it does not take into consideration relative abundance. This factor was observed both by S.Auf.U and cylinder sampling. The S.Auf.U. data from the pools showed this difference between stations more clearly than data derived from the riffle community. The reduction in the diversity index shown by the S.Auf.U. data as explained above was due to the overwhelming dominance of A.aquaticus at Station 2.

4.74 River Mease

4.741 Sampling station

The River Mease situated to the N.E. of Birmingham flows for approximately 15 km. before joining the River Trent, just downstream of the entry of the River Tame. The upper sections of the river receive minor inputs from small sewage works and the river is of particular interest to anglers, since it supports a wide variety of coarse fish.

The station selected for study (Figure 4.16) supported a diverse community of aquatic macrophytes, particularly Ranunculus spp. and these were prevalent in both pools and riffles. The substratum of the riffle consisted mainly of hard compact gravel and was scoured by an annual mean current velocity of 57.6 cm. sec^{-1} (range 31.6 - 77.8 cm. sec^{-1}). The pool selected for study was 50 m. upstream of the riffle, with a similar substratum, but an annual mean current velocity of 23.7 cm. sec^{-1} (range 12.8 - 33.3 cm. sec^{-1}).

The sampling programme commenced at the beginning of April and continued monthly until December. Some difficulties were experienced in placing and retrieving S.Auf.U. samplers during the summer months due to angling activities and as a consequence, the sampling routine was suspended during July and August. Direct sampling of the downstream riffle was, however, continued during this period.

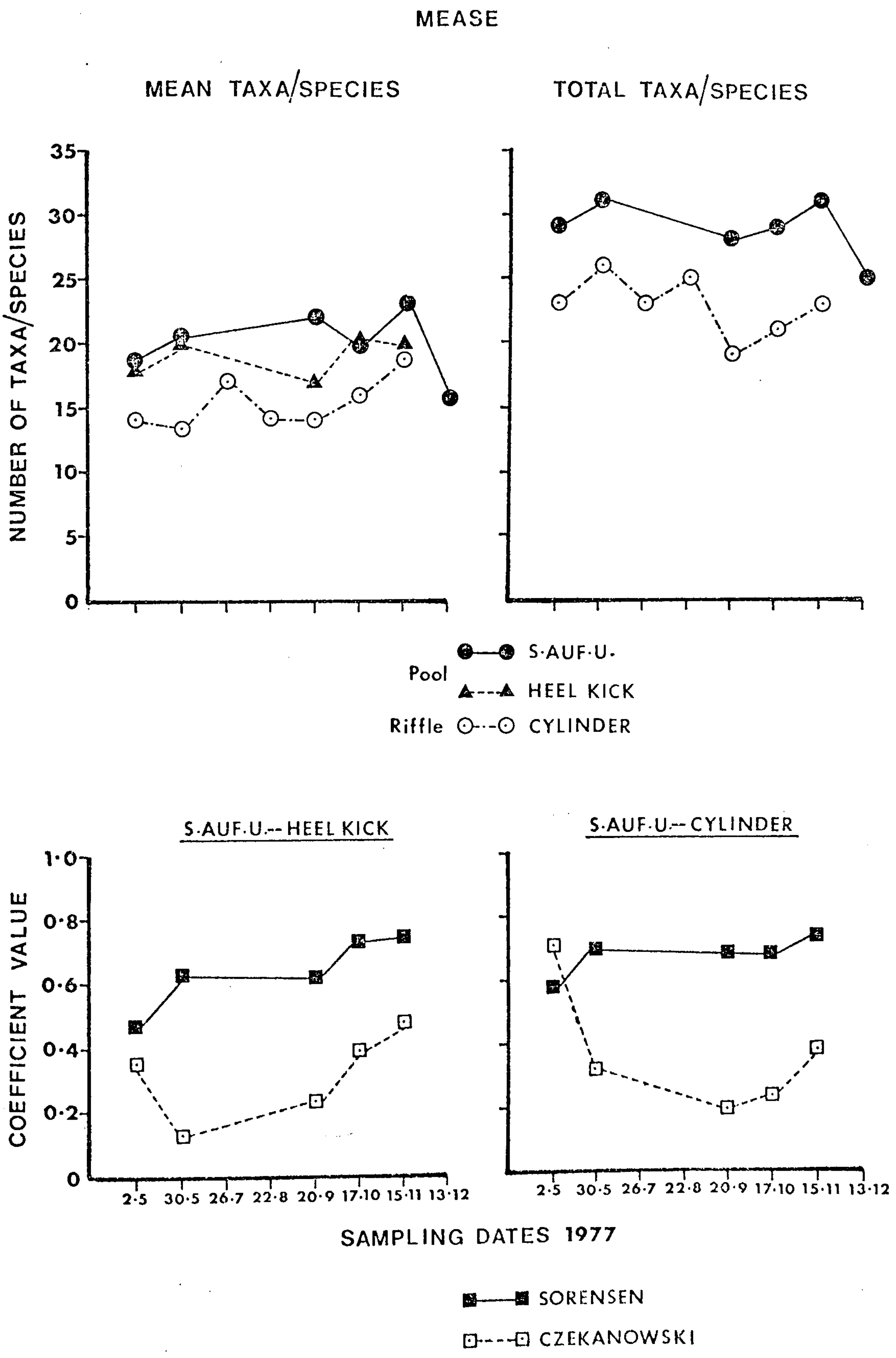
4.742 Results and Discussion

The physico-chemical data presented in Annexe 2 (Table 16) and Annexe 3 (Table 25) and summarised in Table 4.11, indicate that the River Mease generally has a good water quality, thus capable of supporting a wide variety of fish. The water is well oxygenated and has low levels of BOD_5 and ammonia.

The detailed macroinvertebrate data are presented in Annexe 4 (Tables 53 - 54).

The numbers of taxa collected monthly by the three sampling methods (Figure 4.26) showed similar trends as recorded previously from other rhithron rivers. Typically, S.Auf.U. collected the greatest number of taxa compared to direct sampling of the riffle and pool, but there was little difference in their seasonal variations. Sorensen's similarity

FIG. 4.26 NUMBERS OF TAXA COLLECTED MONTHLY ON S.AUF.U. IN POOLS AND DIRECT SAMPLING OF POOLS AND ASSOCIATED RIFFLES WITH SIMILARITY COEFFICIENTS , RIVER MEASE.



coefficient revealed a high correlation of taxa similarity between S.Auf.U. and direct sampling of the pool (annual mean 64%) and riffle (annual mean 68%). Czekanowski's coefficient was less convincing, based on relative abundance, but higher levels of similarity were recorded here compared to other rivers (S.Auf.U.- pool annual mean 32%; S.Auf.U. - riffle annual mean 37%).

Table 4.11 Physico-chemical data expressed as mean and range for the River Mease (mg. l^{-1})

Station	Analysis	S.S.	D.O.	BOD_5	Ammonia	Nitrate
1	ASTON	14 (5- 38)	10.4 (8.0-13.2)		1.88 (0.85-6.5)	8.9 (4.8-13.9)
	WA	30 (1-382)	10.3 (5.9-15.2)	2.4 (0.5-6.3)	0.2 (0.1 -0.6)	10.3 (5.2-20.8)

The relative abundance of the major taxa again showed direct relationships between the three sampling methods tested (Figure 4.27). All three sampling techniques were able to collect similar faunal groups, very often with similar abundances, and there appeared to be little difference between the riffle-pool communities. In the pool, larger numbers of Oligochaeta, *A.aquaticus*, *C.moesta* and Chironomidae were collected compared to the riffle. The riffle community, however, had only slighly higher abundances of *G.pulex* and *P.jenkinsi* compared to the pool. In essence, the stony substratum present in both the riffle and pool accounted for similar faunal assemblages being found. In addition to the species presented in Figure 4.27, S.Auf.U. also collected small numbers of Damsel-fly larvae (*Agrion* and *Coenagrion*), *Centroptilum luteolum*, *Sialis lutaria*, *Valvata* spp. and *Planorbis* spp..

FIG. 4.27 RELATIVE ABUNDANCE OF MAJOR TAXA COLLECTED FROM THE RIVER MEASE AS INDICATED BY DIFFERENT SAMPLING METHODS.

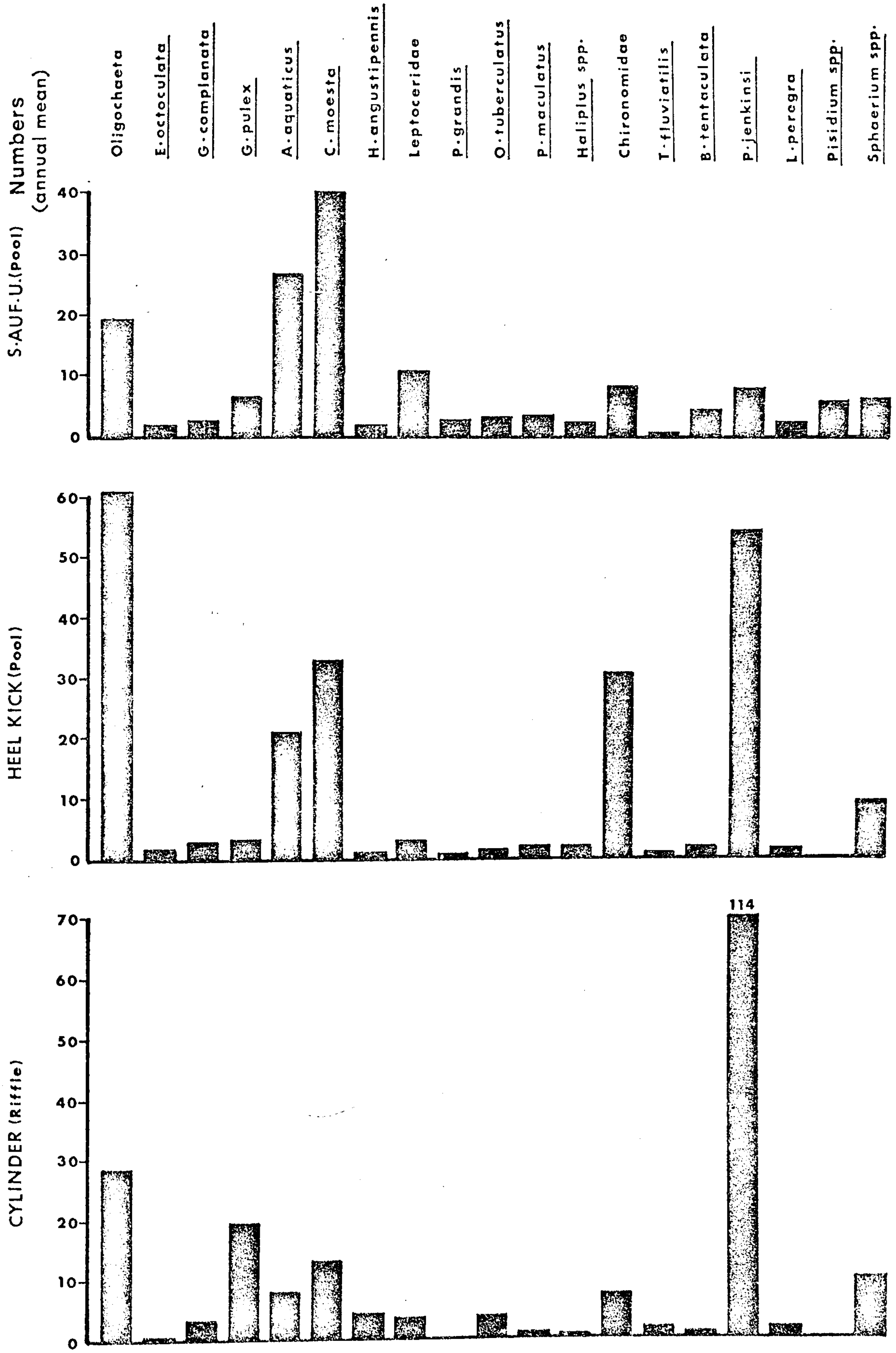
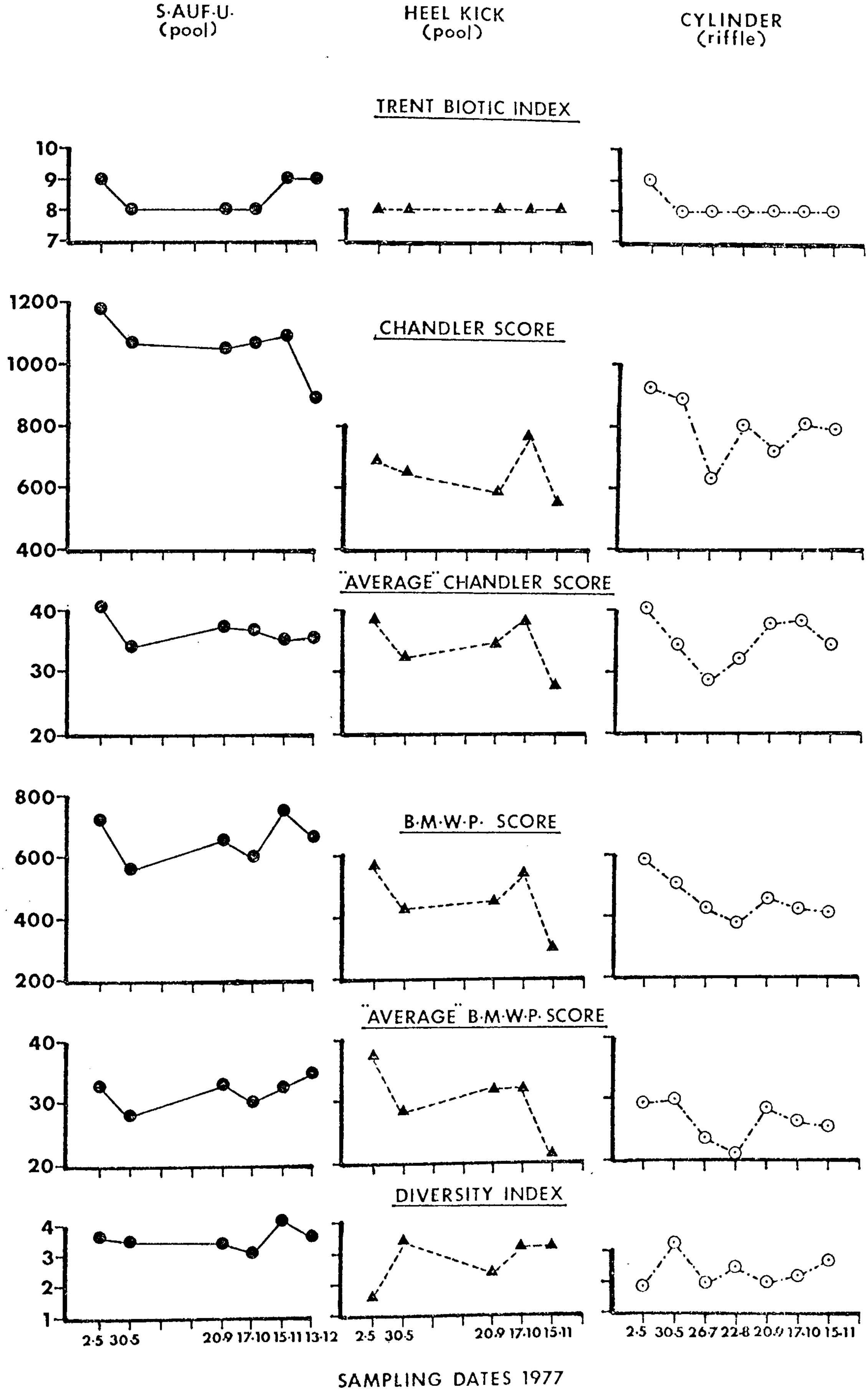


FIG. 4.28 SEASONAL VARIATION OF INDEXES AND SCORES OBTAINED BY DIFFERENT SAMPLING METHODS FROM RIVER MEASE.



The various biotic and diversity indexes calculated from the River Mease data are presented in Figure 4.28 on a seasonal basis. Of the scores derived, the Trent Biotic Index produced the smallest seasonal divergence, while the Chandler Score produced the greatest seasonal variation. Generally, all the indexes tested showed similar monthly trends.

The S.Auf.U. scores were generally less varied than those obtained from riffle sampling and this was probably due to the stable environment provided by the colonisation sampler. As on previous occasions, S.Auf.U. typically generated higher scores than the other sampling methods.

4.75 River Tean

4.751 Sampling stations

As described in Chapter 3, the River Tean is a small tributary of the River Dove and its only source of contamination are organic enrichment from the Blithe Valley Sewage Treatment Works at Checkley and a Creamery effluent at Fole (Figure 4.16).

The sampling stations were those used in Phase I, upstream of the Checkley effluent and downstream at Beamhurst. The substratum of both riffle and pool at both stations consisted of hard compact gravel; their current velocities are summarised in Table 4.12.

Samples of macroinvertebrates were first collected mid-way through March and continued on a monthly basis until mid-December. Direct sampling of the pool and riffle benthos was possible on every occasion except during December when the river was high. S.Auf.U. sampling was possible on every occasion regardless of the river flow.

Table 4.12 The mean and range of current velocities recorded in The River Tean (cm. sec^{-1})

Station	Riffle	Pool
1	73.9 (44.6-112.2)	17.9 (5.7-39.1)
2	77.2 (63.1-123.5)	22.7 (10.5-36.3)

4.752 Results and discussion

The physico-chemical data generated during this survey are presented in Annexe 2 (Tables 17a - b) and Annexe 3 (Table 25) and summarised in Table 4.13. The water quality deteriorated only slightly between Stations 1 and 2, possibly indicating some recovery at the lower station. The most significant feature was a decrease in dissolved oxygen and an increase in BOD_5 between the two sites. There was an insignificant increase in the levels of ammonia between the two sites.

The detailed macroinvertebrate data collected from the River Tean are presented in Annexe 4 (Tables 55 - 60). The number of taxa collected by the three sampling methods are presented in Figure 4.29. From this data, it can be seen that the highest numbers of taxa were taken on the S.Auf.U. at Tean 1, than by direct sampling of the associated riffle and pool. At Tean 2, there was little difference between the various sampling techniques. At Both stations similar seasonal trends were observed by all samplers, the largest numbers being recorded during June. The numbers of taxa collected by S.Auf.U. during this phase were similar to the numbers collected during phase I of the work.

Table 4.13 Physico-chemical data expressed as mean and range for the River Tean (mg. l^{-1})

Station	Analysis	S.S.	D.O.	BOD_5	Ammonia	Nitrate
1	ASTON	17 (2-102)	10.35 (9 -12.1)	4.1 (1.7-6.8)	1.24 (0.5-3)	4.4 (1.7-6)
	WA	19 (4- 61)	10.7 (8.5-11.8)	2.8 (1.4-6)	0.3 (0.1-0.7)	4.5 (3.4-5.7)
2	ASTON	14 (7-33)	7.24 (4.4-11)	11.3 (6.4-14)	1.26 (0.7-2.4)	8.0 (2.9-11.6)
	WA	35 (4-427)	9.7 (7.5-11.1)	5.1 (2.2-12)	0.5 (0.1-1.8)	8.8 (4.7-12.6)

It can be seen from Figure 4.29 that the levels of similarity based on Sorensen's values between S.Auf.U.and direct sampling were higher at Tean 2 than at Tean 1. Using Czekanowski's coefficient this increase was less evident, although the values increased slightly at Tean 2.

The numbers of individual species, are shown in Figure 4.30, and show the effect of organic enrichment on the benthic communities present in the Tean. Although the numbers of individuals fluctuate more widely over the season than the numbers of taxa, the mean annual abundance reflects the overall community change. An examination of the major taxa collected by the three sampling methods showed that pollution sensitive species, particularly *E. ignita*, *Stenophylax* spp. *A. nervosa*, *P. flavomaculatus* and certain Mollusca (*L. peregra* and *P. jenkinsi*) were eliminated by the organic effluents entering the

FIG. 4.29 NUMBERS OF TAXA COLLECTED MONTHLY ON S.AUF.U. IN POOLS AND DIRECT SAMPLING OF POOLS AND ASSOCIATED RIFFLES WITH SIMILARITY COEFFICIENTS , RIVER TEAN.

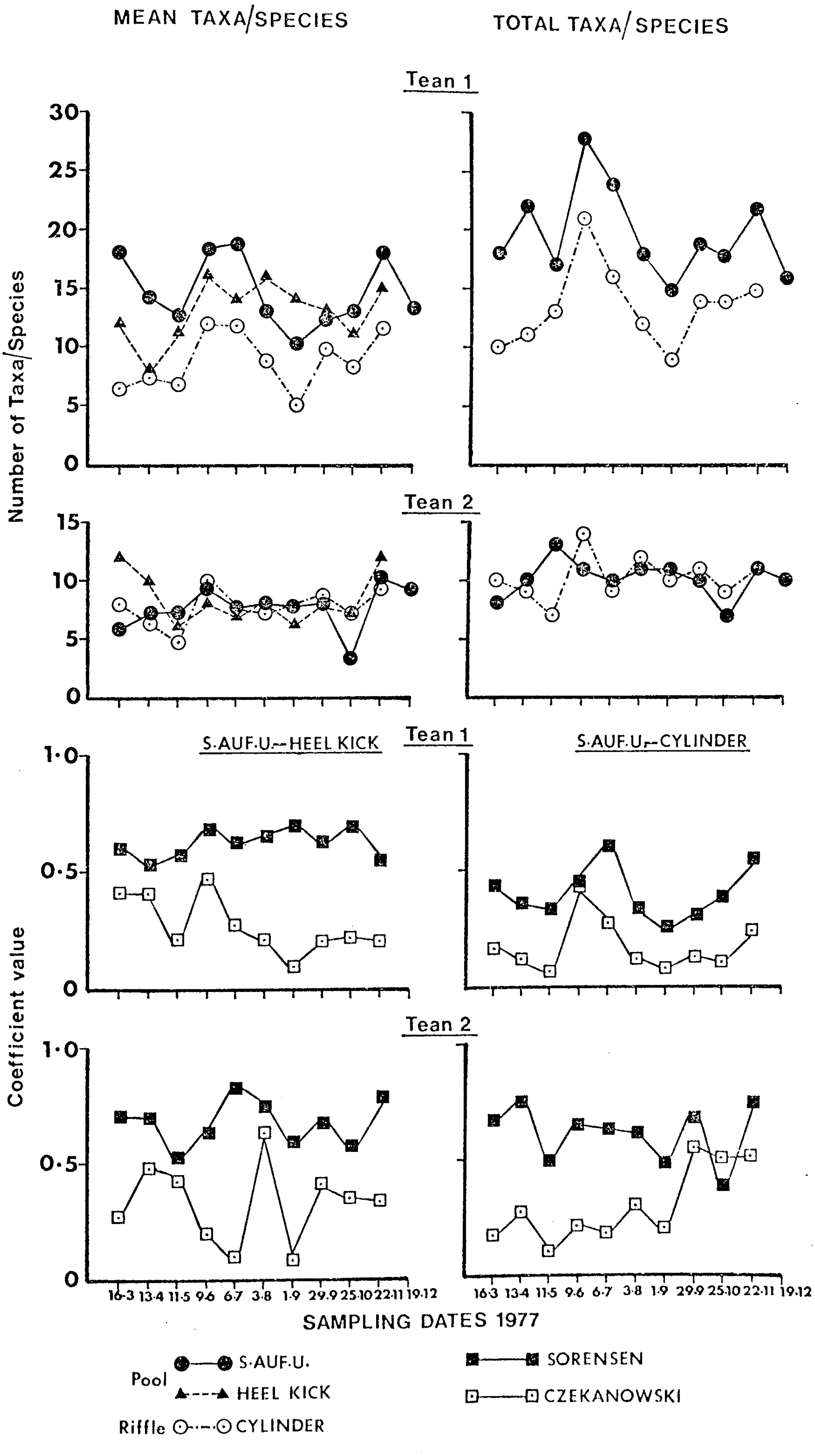
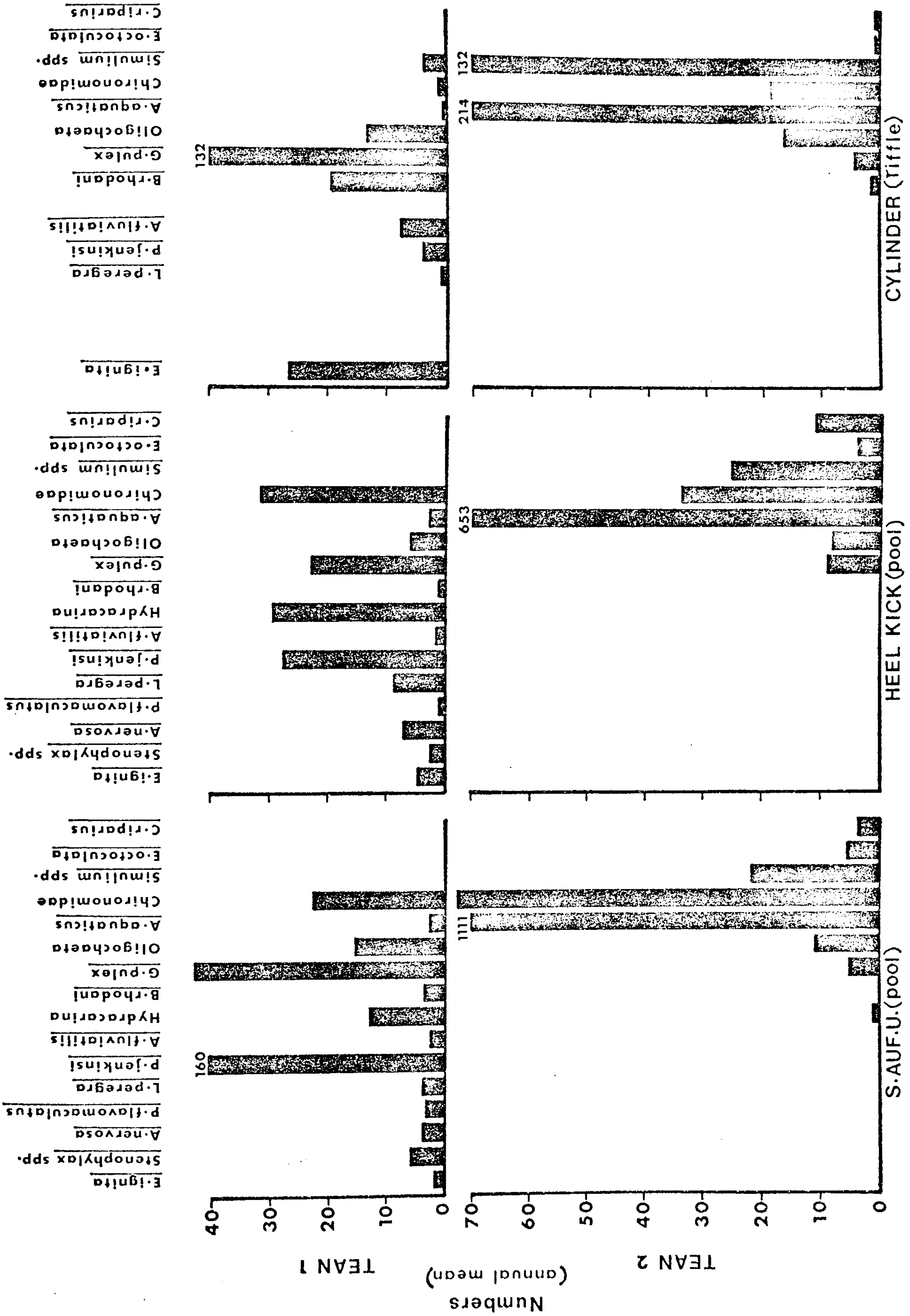


FIG. 4.30 RELATIVE ABUNDANCE OF MAJOR TAXA COLLECTED FROM THE RIVER TEAN AS INDICATED BY DIFFERENT SAMPLING METHODS.

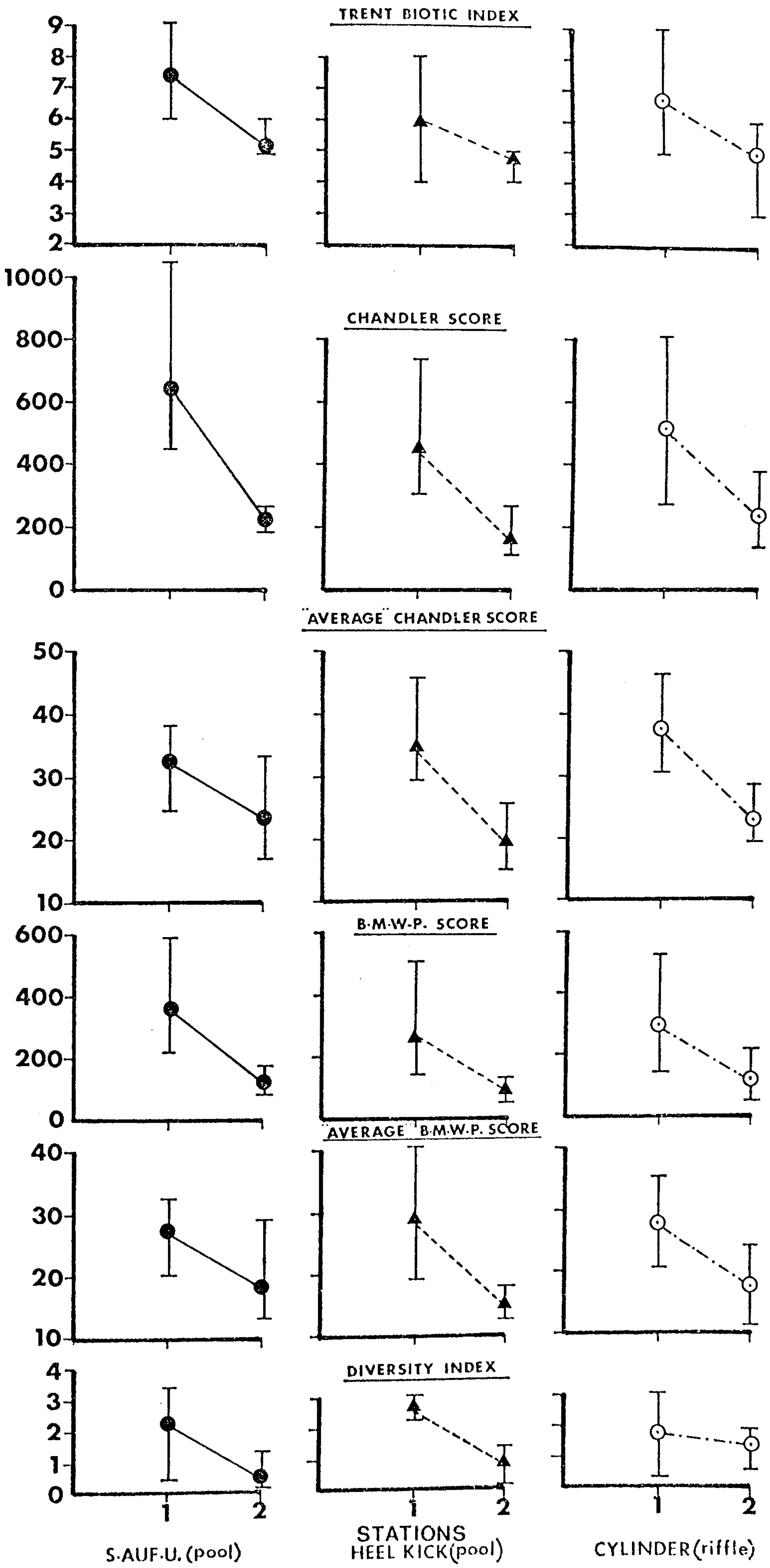


Tean. Numbers of *G. pulex* and *B. rhodani* were also reduced in quantity from the downstream riffle zone. The relative abundance of pollution tolerant species increased below the various effluents so that the downstream pool and riffle zones were dominated by *A. aquaticus*, *Simulium* spp., and Chironomidae. The latter species were mainly of the sub-family Chironominae (Chironomini and Tanytarsini) including the *Chironomus riparius* group. The numbers of Oligochaeta also increased at Tean 2, with *Tubifex tubifex* replacing *Lumbriculus variegatus* and *Rhyacodrilus coccineus* from the upstream clean zone. With the increase in organic enrichment the leech *Eropbdella octoculata* replaced the insensitive leeches *Glossiphonia complanata* and *Piscicola geometra* from the upper sites.

Thus, an examination of the three sampling techniques at the two stations suggests that they are all responding to the changes in water quality recorded in the Tean. The S.Auf.U. sampler was capable of recording data when the river was in high flow and as a consequence could be relied upon when other sampling techniques were impracticable.

The various biotic and diversity indexes calculated for the River Tean are shown in Figure 4.31. All values showed a decline between Stations 1 and 2, due to organic enrichment from Checkley and Fole. The greatest difference was obtained using the Chandler score and the least using the Trent Biotic Index and diversity index. In the clean zone S.Auf.U. sampling produced the highest scores, but in the polluted zone all sampling methods produced comparable scores. Smaller seasonal trends were also recorded in this zone compared to wide fluctuations recorded upstream.

FIG. 4.31 COMPARISONS OF INDEXES AND SCORES DERIVED FROM DATA OBTAINED BY DIFFERENT SAMPLING METHODS FROM THE RIVER TEAN.



Thus, in the River Tean the data produced from physico-chemical analysis and data obtained by macroinvertebrate sampling, clearly reflected the deterioration in water quality obtained between the two stations. S.Auf.U. sampling in pools, although influenced by species from upstream riffles clearly correlated well with cylinder sampling from riffles and produced data which was comparable with findings from Phase I.

4.76 River Churnet

4.761 Sampling stations

The River Churnet which flows south and S.E. around the edge of the Derbyshire Peak District and joins the River Dove 45 km. from its source (Figure 4.16) was studied in detail over a fourteen month period during 1977 and 1978. This investigation was specifically concerned with sampling riffles as a suitable habitat for the biological surveillance of upland rivers. The data generated from this study is reported in detail in Chapter 5. During this investigation two of the fourteen stations which had suitable pool zones upstream of the riffle stations were examined on a monthly basis by S.Auf.U. sampling. Station 6 at Cheddleton Mill was situated a few km. below a chemical and sewage works at Leek. This station was probably the most polluted site along the length of the Churnet. Station 10 at Alton was situated in the recovery zone of the river, well away from any major effluents.

The substratum of the pool at Cheddleton consisted mainly of mud, while the pool at Alton consisted of gravel and silt. Other features of the sampling stations are summarised in Table 4.14.

Table 4.14 The mean and range of current velocities and nature of the substratum found in the River Churnet

Station	Riffle		Pool	
	Nature of Substratum	Current Velocity $cm.sec^{-1}$	Nature of Substratum	Current Velocity $cm.sec^{-1}$
6	Gravel	73.2 (47.5-121)	Mud	Negl.
10	Gravel	94.1 (42.1-136)	Gravel + Silt	Negl.

4.762 Results and Discussion

The physico-chemical data presented in Annexe 2 (Tables 18a-h) and Annexe 3 (Tables 26a-c) and summarised in Table 4.15, reflect the improvement in water quality by the time one reaches Station 10.

Some discrepancies were found in the results between the two independent sets of water analysis, particular DO, BOD_5 and ammonia, but their overall trends indicated an improvement in water quality. The DO levels in the lower section of the river pointed to well saturated water conditions, mainly associated with a high current velocity and good turbulence.

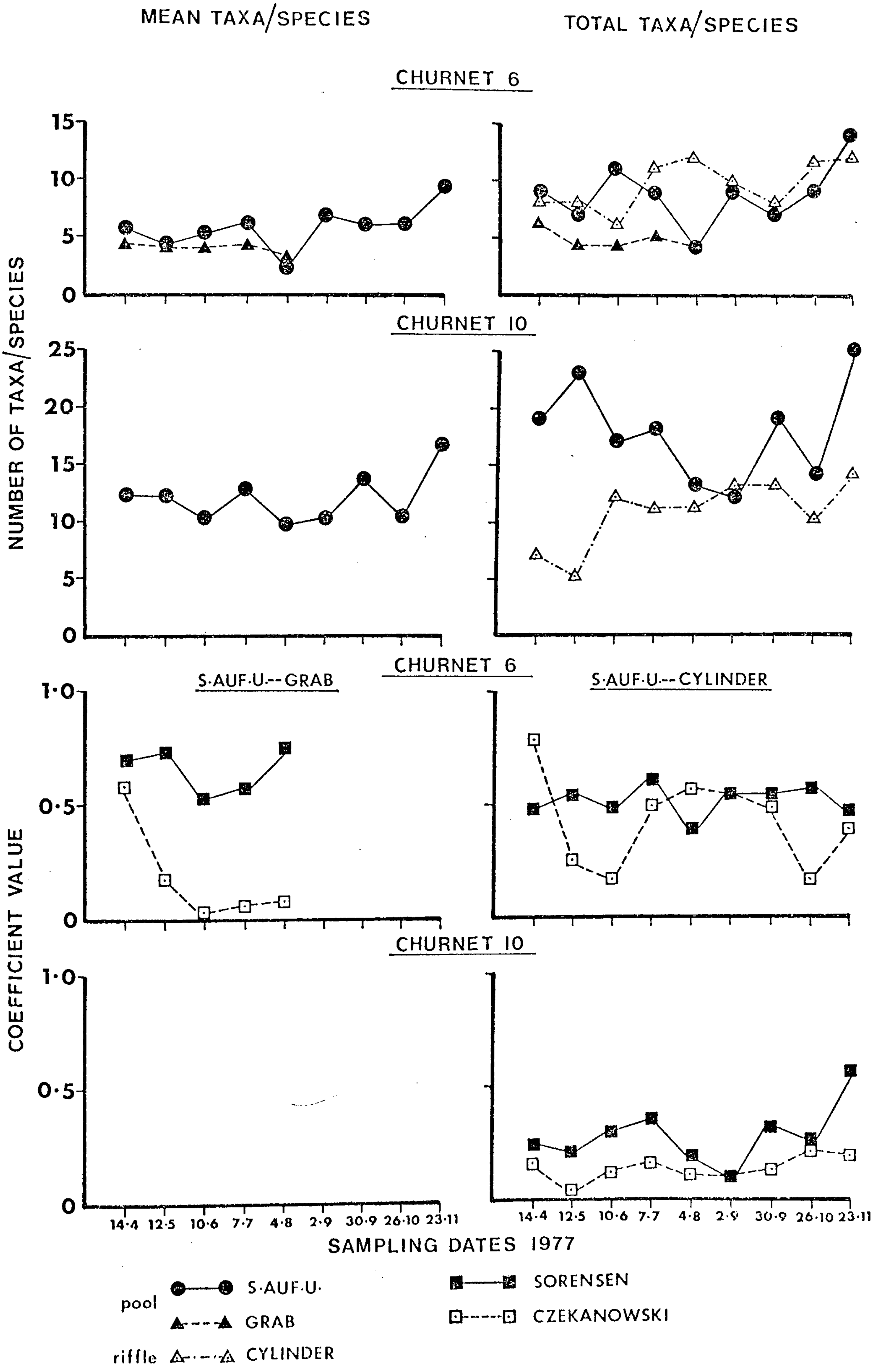
The macroinvertebrate data generated from this study are presented in Annexe 4 (Tables 61 - 65). The number of taxa taken by S.Auf.U. and by direct sampling of the pool and riffle communities are shown in Figure 4.32. An examination of the mean number of taxa collected by S.Auf.U. revealed that there was little seasonal variation detectable at either station. The total number of taxa accumulated by three S.Auf.U. was, however, more varied, particularly at station 10. During the summer period when flows were low, the number of taxa collected on S.Auf.U. was at its lowest. The taxa mainly eliminated were those found in the riffle zones, e.g. Isoperla grammatica, Ecdyonurus venosus, Ephemerella ignita, Baetis rhodani, Rhyacophila dorsalis and Hydropsyche spp. Any increases in flow would ultimately result in scouring of the riffle communities with inevitable deposition in the pool zones. In the polluted section where species of the riffle and pool zones were similar, such scouring effects would be of less consequence to the community structures.

Table 4.15 Physico-chemical data expressed as mean and range for the River Churnet (mg. l^{-1})

Station	Analysis	S.S.	D.O.	BOD_5	Ammonia	Nitrate
6	ASTON	11 (5- 22)	6.4 (2.7-10.5)	13.4 (3.3-33)	1.54 (1 -2.25)	3.0 (1.3-4.5)
	WA	13 (5- 38)	9.1 (6 -11.3)	4.7 (2.9-7.2)	1.1 (0.3-2.3)	3.6 (2.3-4.6)
10	ASTON	8 (1- 12)	10.4 (9.5-11.9)	3.5 (1.9-8.4)	1.21 (0.9-2.25)	4.1 (1.6-6.2)
	WA	19 (3-127)	10.4 (9.3-11.5)	3.4 (0.7-5.8)	0.4 (0.1-1.0)	4.1 (3.2-5.3)

At Station 6, the number of taxa collected by S.Auf.U. and cylinder sampling was very similar, with comparable seasonal variations. The lowest number of taxa were collected by grab sampling of the pool benthos, where little seasonal variation was observed. At Station 10, S.Auf.U. nearly always collected larger numbers of taxa than cylinder sampling, the numbers being made up of Platyhelminthes, Tubificidae, Hirudinea, Limnephilidae, Chironomidae and Mollusca, which were not recorded in riffles.

FIG. 4.32 NUMBERS OF TAXA COLLECTED MONTHLY ON S.AUF.U. IN POOLS AND DIRECT SAMPLING OF POOLS AND ASSOCIATED RIFFLES WITH SIMILARITY COEFFICIENTS , RIVER CHURNET.

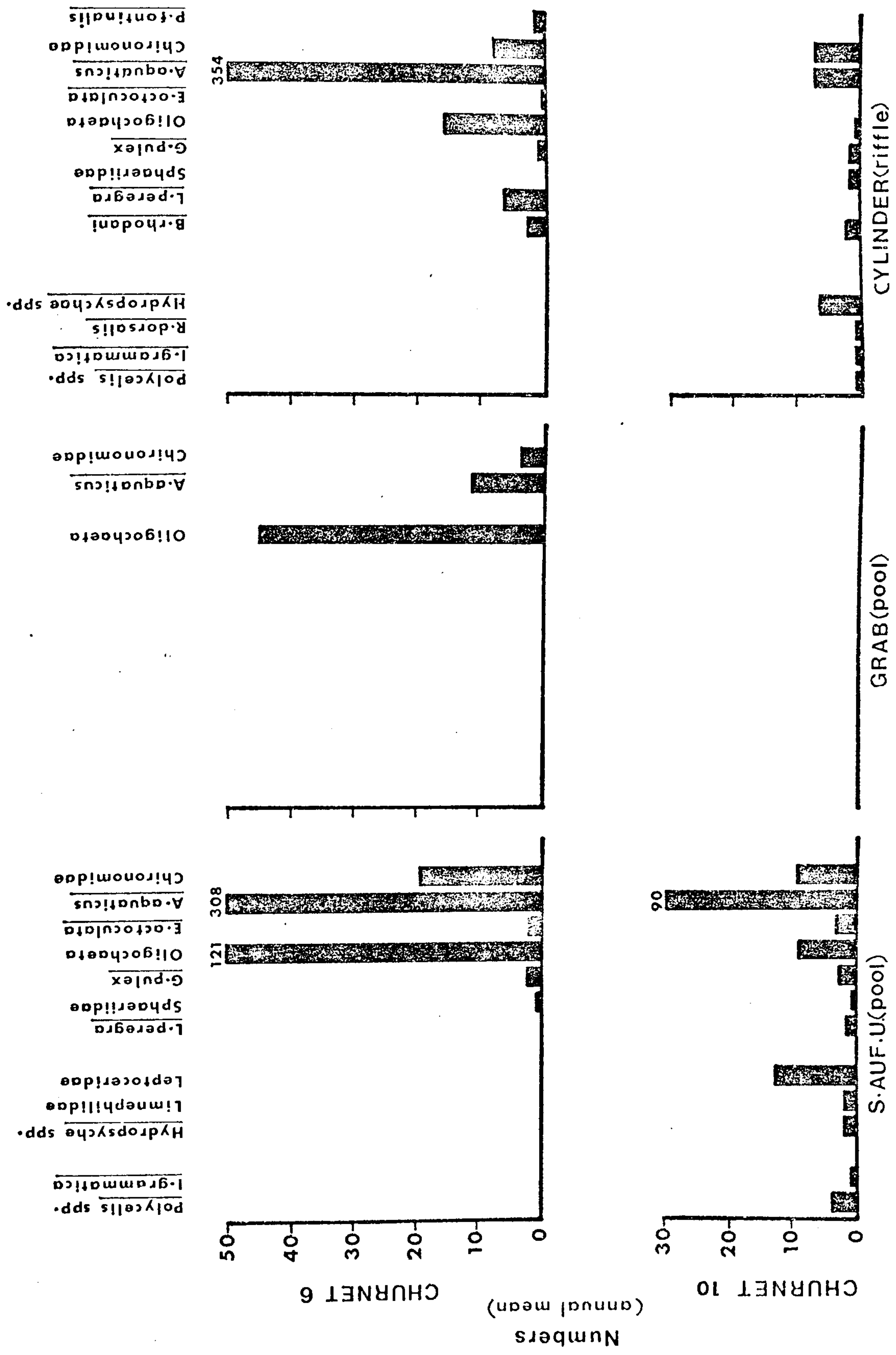


The similarity of species composition of the various sampling techniques is best reflected in Figure 4.32 where Sorensen's coefficient displayed a higher level of affinity in the polluted section than in the clean section. This was also the case with Czekanowski's values between S.Auf.U.and cylinder samples. There was, however, a low level of affinity between S.Auf.U. and grab sampling of the polluted pool using Czekanowski's coefficient since the numbers of particular species varied greatly with the two sampling methods. Grab sampling was particularly difficult at Station 6 because the deposited mud contained a high percentage of extraneous material (twigs, roots, boulders), thus resulting in inefficient collections.

The annual mean relative abundance of the major taxa is presented in Figure 4.33. From this data it is evident that the benthic species intolerant of organic pollution dominate the community structure present at Station 10. These species have replaced the pollution tolerant species dominating the community at Station 6.

At Station 6, A.aquaticus was the principle organism collected by S.Auf.U. and cylinder sampling. Recorded as mean annual numbers per single sampling unit, both methods of sampling were of similar proportions. Higher numbers of the Oligochaeta Tubifex tubifex were found in the pool section compared to the riffle and this was particularly the case with grab sampling. Chironomids on S.Auf.U. were mainly from the sub-family Tanypodinae (Procladius spp. and Pentaneurini) and occasionally from the Chironomus riparius group. The species recorded from the nearby riffle community were particularly made up of the sub-family Orthocladiinae, presumably because of the hard substrata present. Other species recorded from both pool and riffle were G.pulex, E.octoculata and Sphaeriidae.

FIG. 4.33 RELATIVE ABUNDANCE OF MAJOR TAXA COLLECTED FROM THE RIVER CHURNET AS INDICATED BY DIFFERENT SAMPLING METHODS.



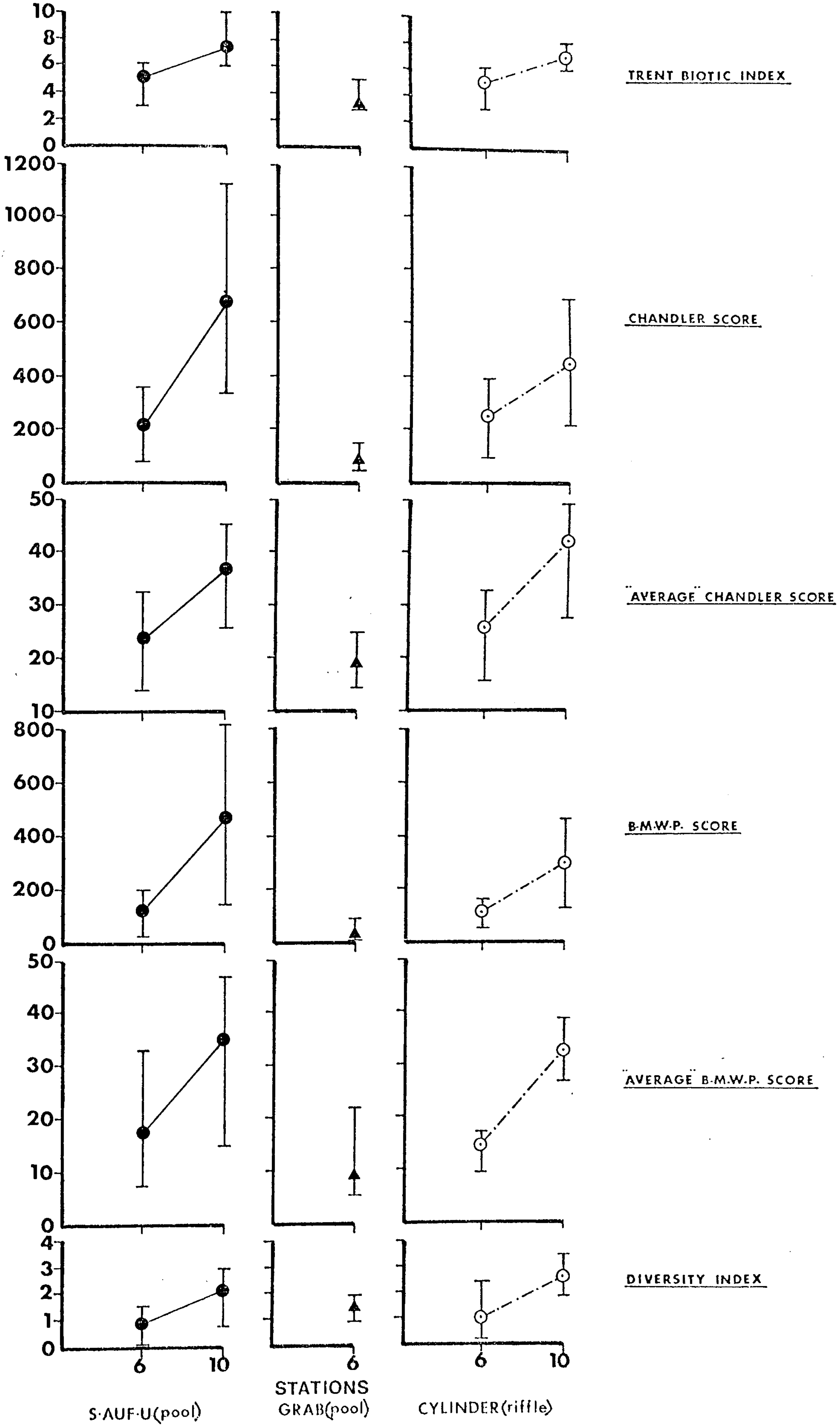
At Station 10, although A.aquaticus was the dominant organism collected on S.Auf.U., its numbers were greatly reduced compared to the polluted zone. An increase in the diversity of the faunal composition was apparent at this station, reflected by an increase in the number of taxa and a decrease in their densities. As mentioned previously, the fauna of the S.Auf.U. was greatly influenced by the presence of upstream riffles and this was apparently noticeable during the summer months when flows were low and diversity low. However, over most of the season attached forms such as Limnephilidae and Leptoceridae were present on S.Auf.U.. Additional species present on S.Auf.U. were Polycelis spp., G.pulex, Prodiamesa olivacea and L.peregra.

The riffle community although not dominated by one particular species, was well represented by Hydropsyche spp. and typical clean riffle species, e.g. I.grammatica, E.venosus, E.ignita, B.rhodani, R.dorsalis and G.pulex.

The various indexes and scores determined from the Churnet data are presented in Figure 4.34. These reflect the overall improvement in water quality observed between the two sites. All scores increased between Station 6 and 10, with the greatest seasonal variances being recorded by the Chandler and B.M.W.P. scores.

The mean values derived by S.Auf.U. and cylinder sampling at Stations 6 and 10 were very similar for all scores, although the seasonal variances recorded by S.Auf.U. were generally greater. The values obtained for grab sampling were particularly low, but this was associated with the inadequacies of the sampling method. Of the scores

FIG. 4.34 COMPARISON OF INDEXES AND SCORES DERIVED FROM DATA OBTAINED BY DIFFERENT SAMPLING METHODS FROM THE RIVER CHURNET.



analysed, the Chandler score probably correlated the nearest to that reflected by water quality physico-chemical analysis.

4.77 River Dove

4.771 Sampling station

The River Dove (Figure 4.16) descends about 450 m. along its whole length of some 70 km. forming a natural boundary between Staffordshire and Derbyshire. In its upper reaches, the river is set between limestone hills forming part of the Peak District National Park and in this section there are only two discharges from sewage treatment works serving two villages.

South of the National Park, the river valley flattens out and eventually the Rivers Churnet and Tean flow into the Dove. In this lower section, the river receives sewage effluent from Uttoxeter, just downstream of the confluence with the River Tean and other small discharges from small sewage works. The Dove finally enters the Trent at Newton Solney, just below Burton-on-Trent. Before the confluence with the Trent, water is abstracted at Stretton at the rate of 90 million litres of water per day to the Staunton Harold Reservoir which serves the city of Leicester.

The pool section studied in the Dove was situated just upstream of the water intake in a region excavated purposely for water abstraction. The substratum consisted mainly of large boulders and hence made grab sampling of the benthos impossible. The use of colonisation samplers was therefore the only method of sampling suitable for this site. The samplers were placed near the bank of the river in a region where there was a slight current velocity of annual mean 18.0 cm. sec^{-1} (range 12.8 - 27.8 cm. sec^{-1}). The riffle site of the same water quality was situated 250 m. downstream of the pool with an annual mean current velocity of 69.4 cm. sec^{-1} (range 54.7 - 85.3 cm. sec^{-1}).

4.772 Results and Discussion

The physico-chemical data derived from this river were restricted to those obtained from the Water Authority and presented in Annexe 3 (Table 25). For most of the year, the river is well oxygenated and has low BOD_5 and ammonia levels. Low nitrate and phosphate levels indicate the low nutrient status of the river which therefore appears to be an excellent source for water abstraction.

The macroinvertebrate data collected from the pool and riffle are presented in Annexe 4 (Tables 66 - 67). From the data presented in Figure 4.35, it appears that the number of taxa collected by S.Auf.U. and cylinder sampling was similar, and as observed previously, S.Auf.U. collected the greater numbers. The similarity of species collected by the two sampling methods was rather low in the Dove. Using Sorensen's coefficient, an annual mean of 39% was attained, while Czekanowski's value was much lower at 18%.

The relative abundance of the major taxa recorded in the Dove is presented in Figure 4.36. Chironomids were the dominant organisms present in both pool and riffle, divided between the sub-families Orthocladiinae and Chironomini. A.aquaticus was only taken by S.Auf.U. sampling from the pool region together with A.splendens, Leptoceridae, T.fluviatilis and B.tentaculata. In the riffle, typical species found were B.rhodani, E.ignita, Taeniopteryx nebulosa,

FIG.4.35 NUMBERS OF TAXA COLLECTED MONTHLY ON S.AUF.U. IN POOLS AND DIRECT SAMPLING OF POOLS AND ASSOCIATED RIFFLES WITH SIMILARITY COEFFICIENTS , RIVER DOVE.

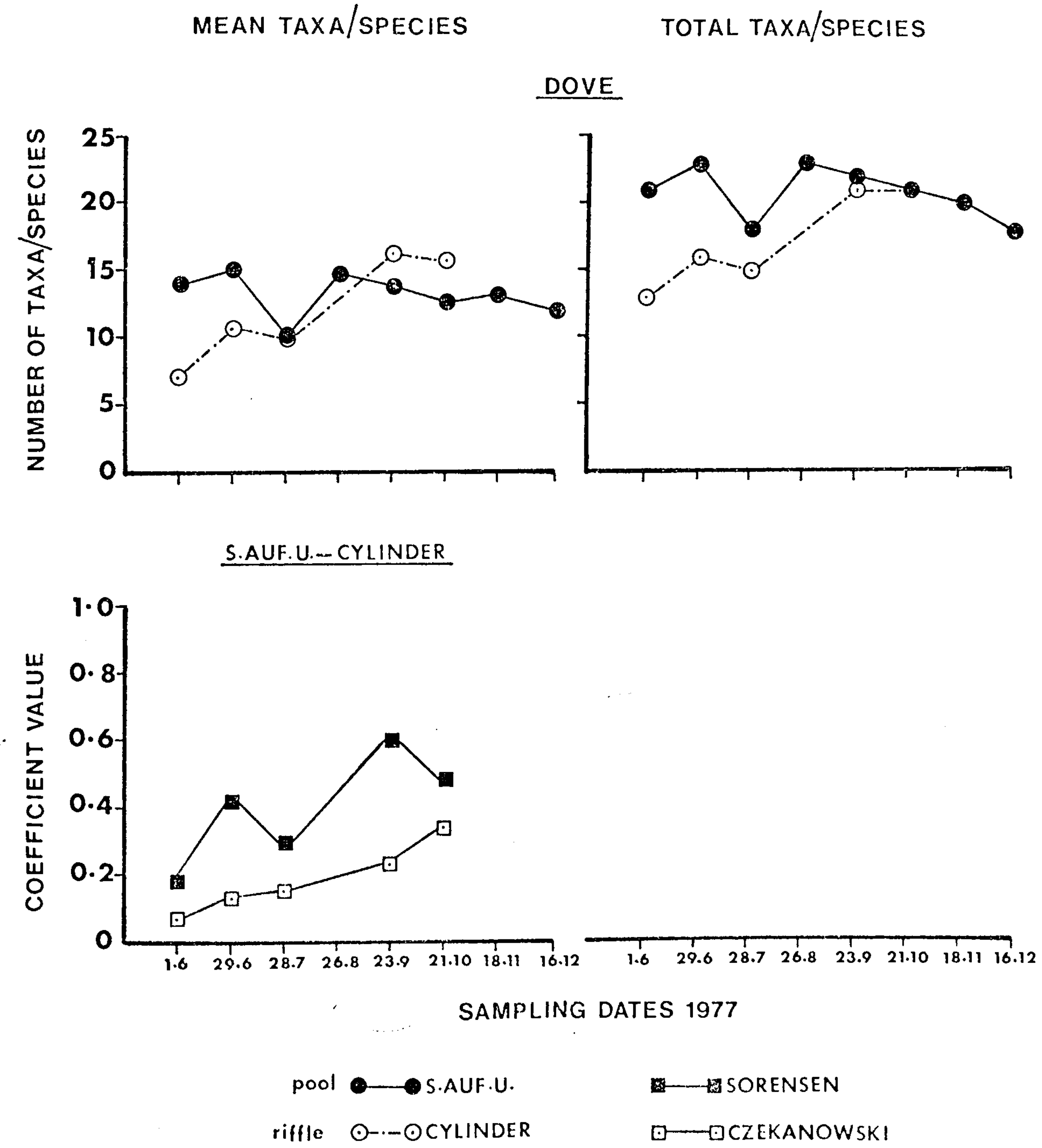


FIG. 4.36 RELATIVE ABUNDANCE OF MAJOR TAXA COLLECTED FROM THE RIVER DOVE AS INDICATED BY DIFFERENT SAMPLING METHODS.

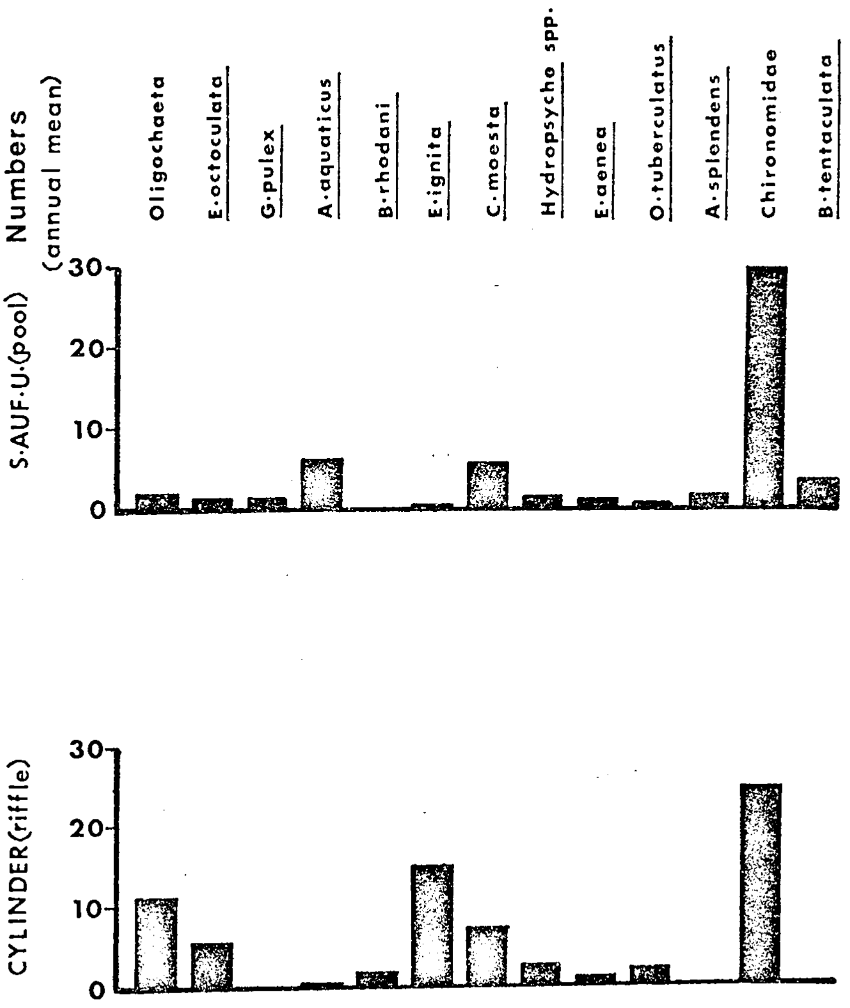
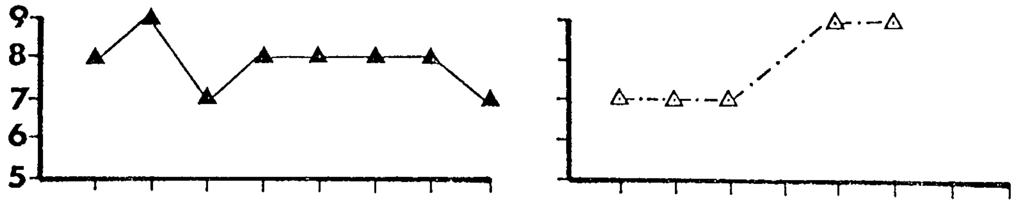


FIG. 4.37 SEASONAL VARIATION OF INDEXES AND SCORES OBTAINED BY DIFFERENT SAMPLING METHODS FROM THE RIVER DOVE.

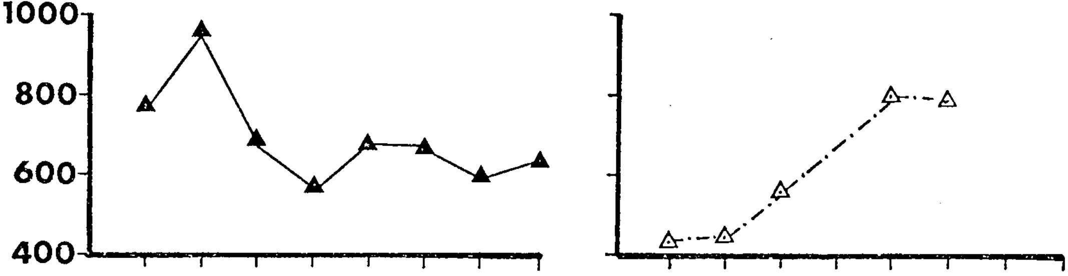
S·AUF·U.
(pool)

CYLINDER
(riffle)

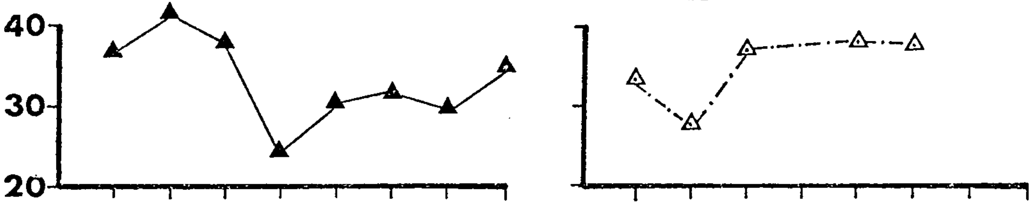
TRENT BIOTIC INDEX



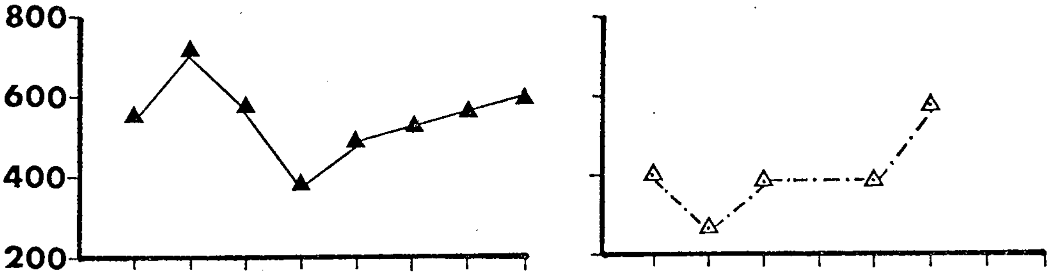
CHANDLER SCORE



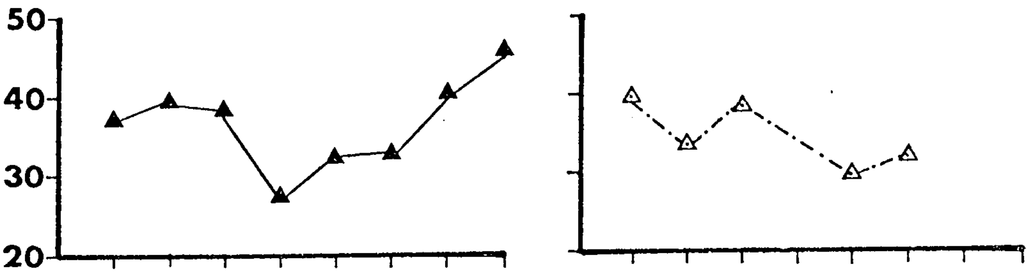
"AVERAGE" CHANDLER SCORE



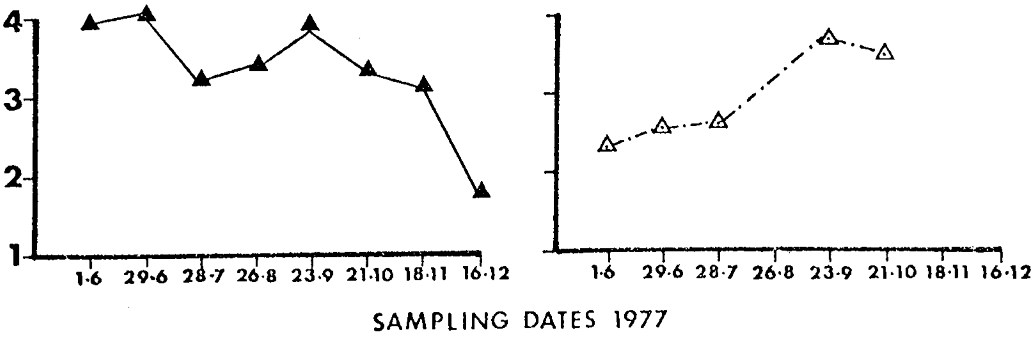
B.M.W.P. SCORE



"AVERAGE" B.M.W.P. SCORE



DIVERSITY INDEX



E.octoculata and H.stagnalis. Elminthid beetles, C.moesta and Hydracarina were found in both pool and riffle. Although there was a large diversity of species taken by both sampling methods, no one species was recorded in abundance.

The calculated biotic and diversity indexes shown in Figure 4.37 reflect the overall water quality status of the river. Both methods of sampling recorded relatively high scores with all indexes, and showed that the river was well suited for water abstraction.

4.78 River Derwent

4.781 Sampling stations

The River Derwent was studied in the lower section of its course upstream and downstream of Derby before it flowed into the River Trent, upstream of Nottingham (Figure 4.16). Station 1 at Allestree consisted of a silty pool of negligible flow 50 m upstream of a riffle section which had an annual mean current velocity of 37.9 cm. sec^{-1} (range 19.5 - 60.1 cm. sec^{-1}). Station 2 at Draycott was downstream of Derby and received large quantities of domestic and industrial effluents. The river in this section consisted mainly of a pool reach and was only possible to sample using colonisation samplers. The riffle sites were inaccessible in this reach and were therefore not included in the survey.

4.782 Results and Discussion

The chemical data are presented in Annexe 3 (Table 24) and summarised in Table 4.16.

Table 4.16 Physico-chemical data expressed as mean and range for the River Derwent (mg. l^{-1})

Station	Analysis	S.S.	D.O.	BOD_5	Ammonia	Nitrate
1	WA	16 (6 - 46)	10.8 (9 - 13)	2.4 (1.5-4.4)	0.2 (0.1-0.3)	4.1 (3 -6.4)
2	WA	13 (7 - 31)	8.8 (5.1 - 11.4)	3.9 (0.6-7.5)	0.4 (0.1-0.8)	4.8 (3.2-7.1)

Compared to other rivers examined, receiving domestic or industrial waste water inputs, the water quality status of the River Derwent deteriorated only slightly after receiving the majority of Derby's industrial effluents between Stations 1 and 2. Levels of BOD_5, ammonia and nitrate increased only slightly with an associated small decrease in dissolved oxygen. Large growths of the blanket weed Cladophora and the occasional appearance of sewage fungus in the riffle at Station 1, suggested an organic input present above the upper station.

The macroinvertebrate data are presented in Annexe 4 (Tables 68 - 70) and summarised in Figure 4.38 to show the number of taxa collected by each sampling method. At Station 1 all sampling methods used collected relatively small numbers of taxa, although of similar proportions and gradually increased in numbers throughout the season. Downstream of Derby the number of taxa collected by S.Auf.U. was slightly higher than found upstream suggesting a suppression of fauna in the upstream cleaner waters.

FIG.4.38 NUMBERS OF TAXA COLLECTED MONTHLY ON S.AUF.U. IN POOLS AND DIRECT SAMPLING OF POOLS AND ASSOCIATED RIFFLES , RIVER DERWENT.

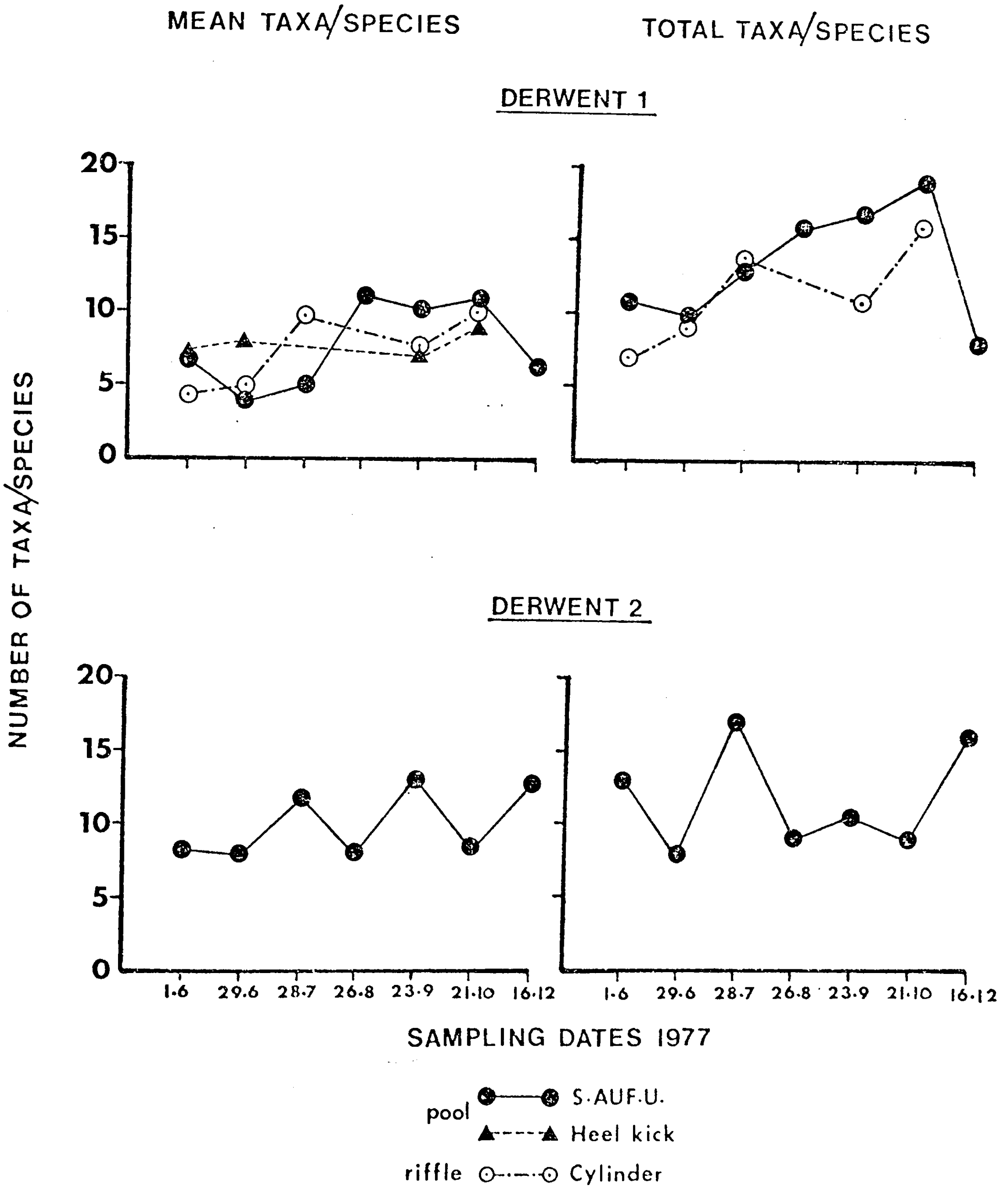


FIG. 4.39 RELATIVE ABUNDANCE OF MAJOR TAXA COLLECTED FROM THE RIVER DERWENT AS INDICATED BY DIFFERENT SAMPLING METHODS.

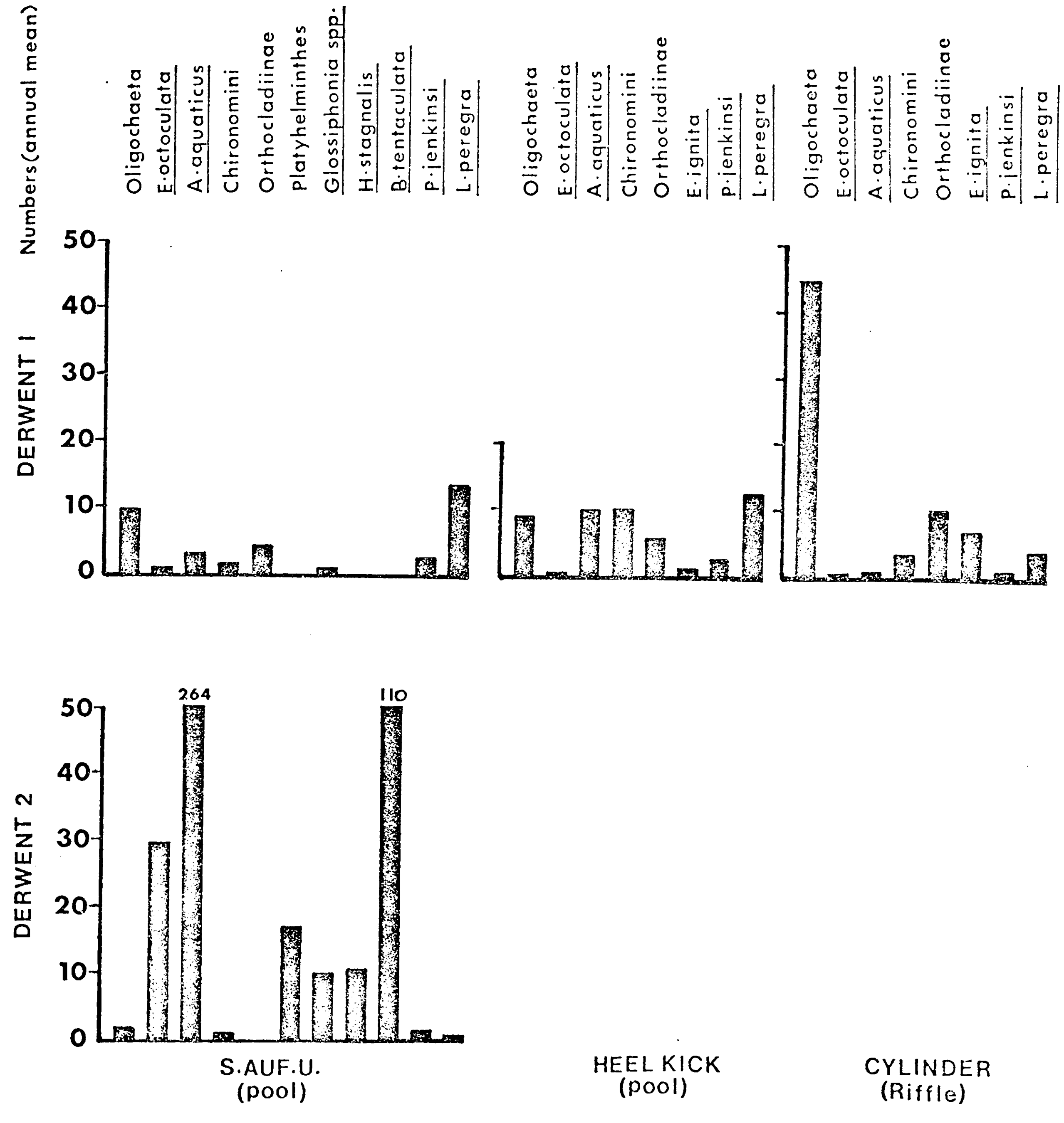
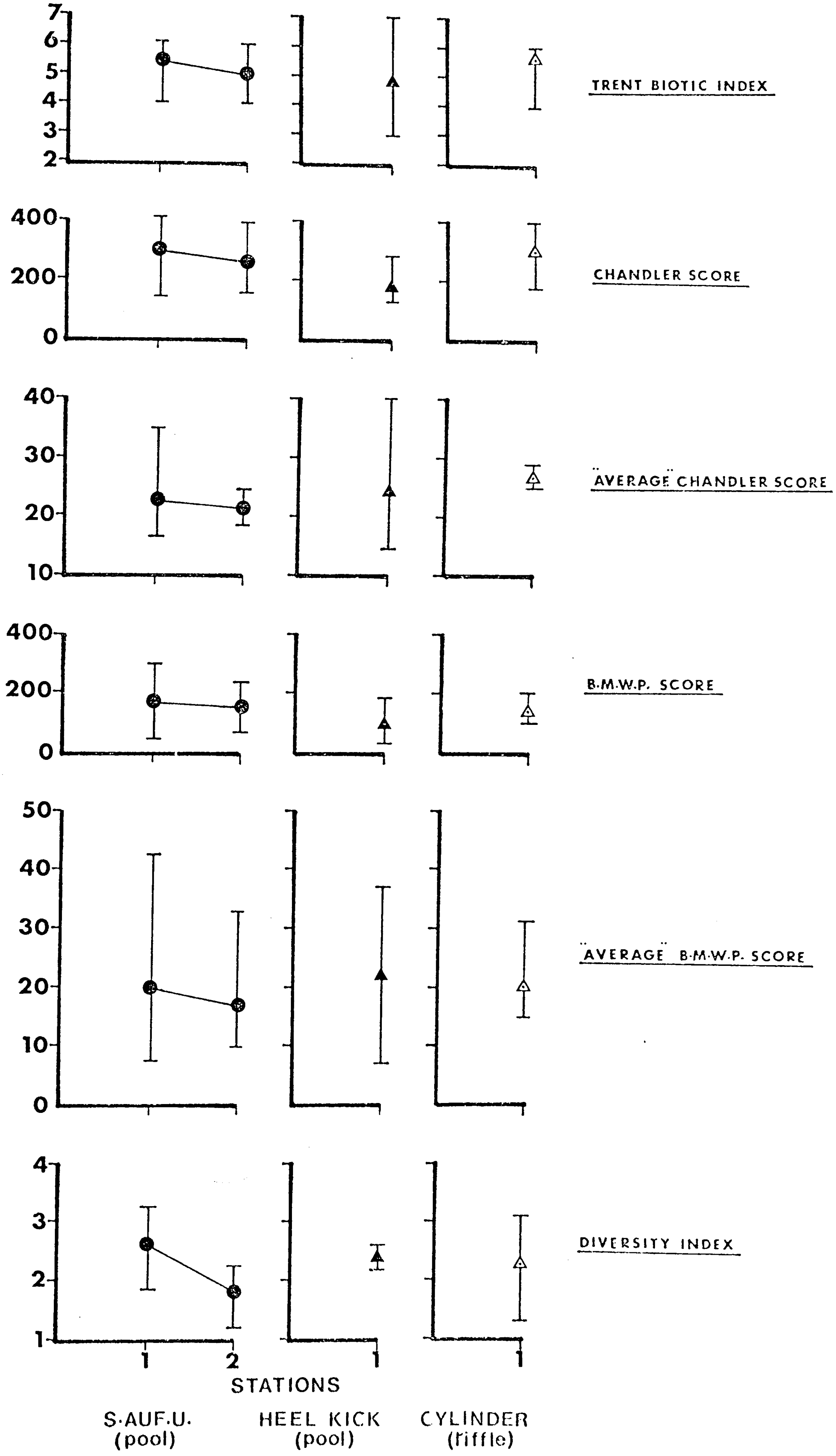


FIG.4.40 COMPARISON OF INDEXES AND SCORES DERIVED FROM DATA OBTAINED BY DIFFERENT SAMPLING METHODS FROM THE RIVER DERWENT.



An examination of the relative abundance of major taxa (Figure 4.39) recorded at Station 1 revealed a low diversity of taxa collected by all sampling methods. Although the riffle zone was dominated by Tubificidae, Chironomidae and occasionally Ephemerella ignita, there was an apparent lack of pollution-sensitive species. Taxa recorded on the S.Auf.U. samplers were mainly made up of Tubificidae, Chironomidae and the molluscs Lymnaea peregra and Potamopyrgus jenkinsi. Downstream of Derby, the species composition colonising S.Auf.U. changed to a community dominated by Asellus aquaticus, Erpobdella octoculata, Glossiphonia spp., Helobdella stagnalis and the mollusc Bithynia tentaculata. Platyhelminthes such as Polycelis tenuis and Dendrocoelum lacteum were also recorded towards the latter part of the year.

Although the number of taxa increased below the entry of Derby's effluent, the values calculated as biotic indexes and scores (Figure 4.40) indicated a slight deterioration in water quality between Stations 1 and 2, as revealed by the chemical data. The greatest decrease in value was obtained using the diversity index due to the presence of large numbers of A.aquaticus and B.tentaculata.

4.79 Rivers Foss, Calder and Don

4.791 Sampling stations

The rivers studied in Yorkshire were based around the heavily industrialised areas of Huddersfield, Sheffield and Rotherham and in the agricultural areas around York.

The River Foss located to the north of York is a small river with a mud and silty substratum receiving minor inputs of organic wastes from small village sewage works and agricultural run-off. The river flows into a lowland river - the Yorkshire Ouse, which was initially examined in these investigations but found unsuitable to sample using the sampling stations selected due to problems with vandalism and fluctuating river levels.

The River Calder located to the south of Bradford and Leeds was sampled upstream and downstream of Huddersfield's sewage effluent. Calder 1 at Brighouse was sited near a textile mill and received large quantities of carpet manufacturing wastes. Calder 2 at Horbury was sited downstream of an area receiving large quantities of carpet and dye wastes and domestic and industrial effluents from Huddersfield. Both stations consisted of long pool stretches with hard compact gravel and negligible flow.

The River Don sampled at Conisbrough was downstream of Sheffield and Rotherham in an area rich in coal deposits. The river as a consequence was heavily polluted from domestic and industrial wastes and had a substratum consisting entirely of mud, silt and fine coal deposits. The section of the river studied was used extensively by barges which caused considerable wave action and heavy erosion of the river banks. Natural river flow was negligible along this stretch.

Due to distances involved in sampling these stations from Birmingham, only three sampling visits were possible between May and August, 1977, using immersion periods of four, eight and four weeks respectively for S.Auf.U. sampling. Natural benthic sampling by grab or cylinder sampling was difficult at most stations.

4.792 Results and Discussion

The chemical data obtained from the Yorkshire Water Authority are presented in Annexe 3 (Table 27) and summarised in Table 4.17.

Table 4.17 Physico-chemical data expressed as mean and range for the Rivers Foss, Calder and Don (mg. l^{-1})

Station	Analysis	S.S.	D.O.	BOD_5	Ammonia	Nitrate
Foss	WA	21 (6 -69)	9.9 (6.1-12.4)	2.5 (1.6- 3.7)	0.1 (0.1- 0.2)	14.3 (6.6-22.1)
Calder 1	WA	21 (4 -54)	6.2 (2.3-10.7)	4.9 (1.7- 8.4)	1.0 (0.1- 2.5)	3.3 (1.7- 5.7)
Calder 2	WA	16 (9 -24)	4.4 (1.6- 8.8)	8.3 (4.8-10.6)	7.2 (1.5-11.7)	4.5 (2.5- 8.0)
Don	WA	28 (10-83)	7.6 (4.3-11.8)	9.1 (3.6-22.1)	11.6 (2.9-19.6)	6.1 (4.6- 8.5)

Of the four rivers studied, the River Foss was considerably cleaner than the others with higher levels of dissolved oxygen. The use of agricultural fertilisers in the area is reflected in the levels of nitrates recorded compared to the other rivers. The water quality of the River Calder deteriorates significantly as a result of putrescible organic wastes and industrial effluents entering the river from the Huddersfield area, resulting in low dissolved oxygen levels and high BOD_5 and ammonia levels. The River Don although severely polluted, has a higher minimum dissolved oxygen content than Calder 2 and probably receives fewer toxic waste effluents than the Calder.

The macroinvertebrate data collected during the summer months is presented in detail in Annexe 4 (Tables 71 - 74). The number of taxa collected by S.Auf.U. at the three rivers is shown in Figure 4.41. Larger numbers of taxa were taken from the River Foss compared to the Rivers Don and Calder suggesting a suppression of fauna in these heavily polluted sites. There was a small decrease in the number of taxa colonising S.Auf.U. at Calder 2 compared to Calder 1, confirming the deterioration in water quality between these sites. The River Don, however, although heavily polluted supported a more diverse community compared to the two Calder stations. Although an eight-week immersion period was used during this study, it had little effect on the colonisation of additional taxa with respect to time.

The relative abundance of major taxa are depicted in Figure 4.42. The River Foss community was dominated by Asellus aquaticus, Tubificidae, Erpobdella octoculata and other Hirudinea. The presence of Gammarus pulex, Amphinemura sulcicollis, Caenis moesta and Leptoceridae confirmed that the river was only slightly organically enriched. The River Calder, however, showed signs of a 'stressed' community grossly affected by toxic and industrial wastes. Calder 1 supported a sparse community consisting mainly of Hirudinea (E.octoculata and Glossiphonia heteroclita) and Chironomidae (particularly Chironomini) with the occasional presence of A.aquaticus. Calder 2 was dominated by A.aquaticus, Chironomini and Chironomus riparius reflecting the degree of pollution. Oligochaetes were lacking in this community, probably due to the hard substratum and minimal accumulation of sediments. The River Don although dominated by Tubificidae, A.aquaticus and Chironomidae (Orthocladiinae) also supported a smaller community containing Hirudinea, Mollusca (Lymnaea peregra and Physa fontinalis) and

FIG.4.41 NUMBERS OF TAXA COLLECTED MONTHLY ON S.AUF.U. IN POOLS , FROM YORKSHIRE RIVERS.

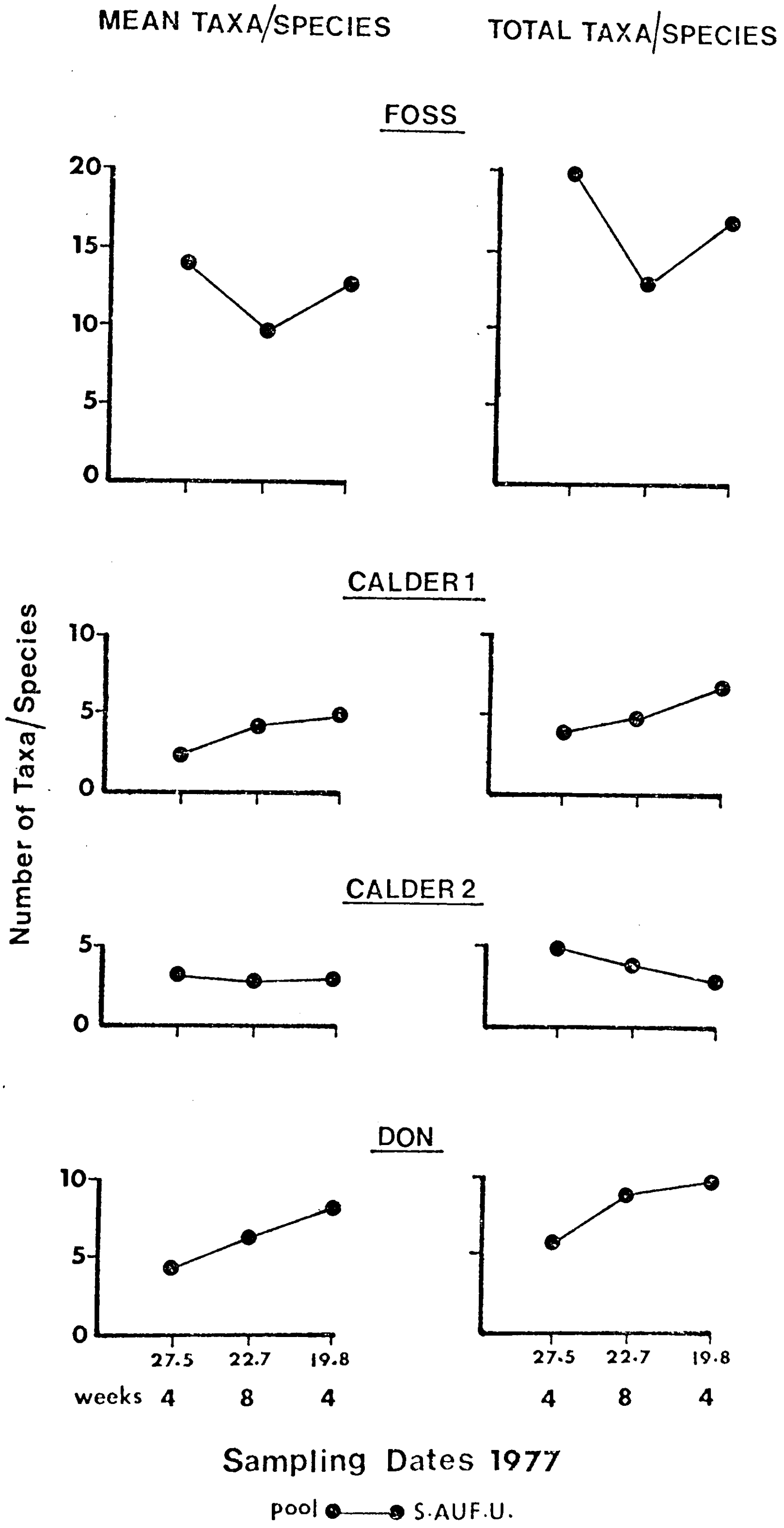


FIG. 4.42 RELATIVE ABUNDANCE OF MAJOR TAXA COLLECTED BY S.AUF.U. SAMPLING FROM YORKSHIRE RIVERS.

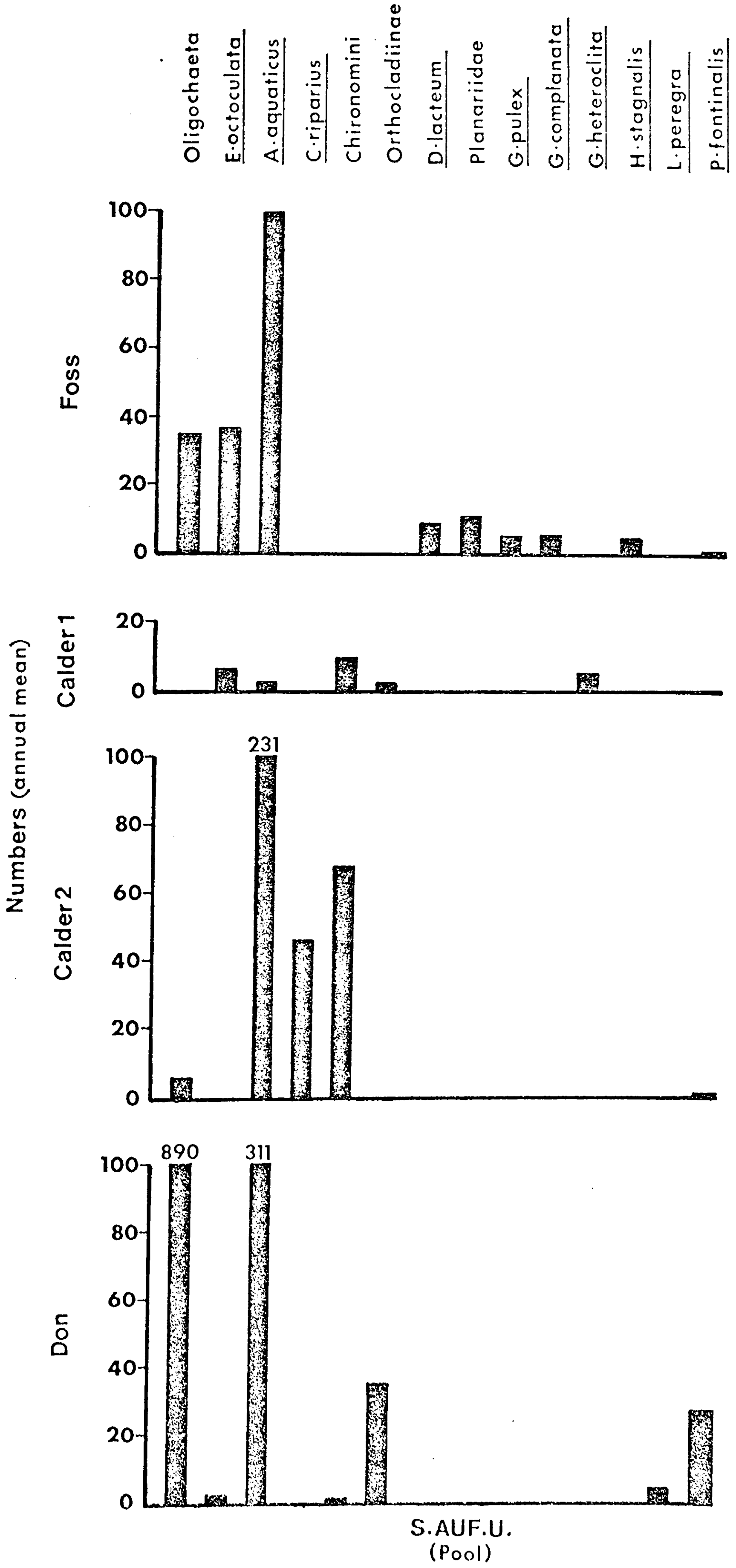


FIG.4.43 PERCENTAGE COMPOSITION OF SELECTED TAXA COLLECTED BY S.AUF.U. FROM YORKSHIRE RIVERS.

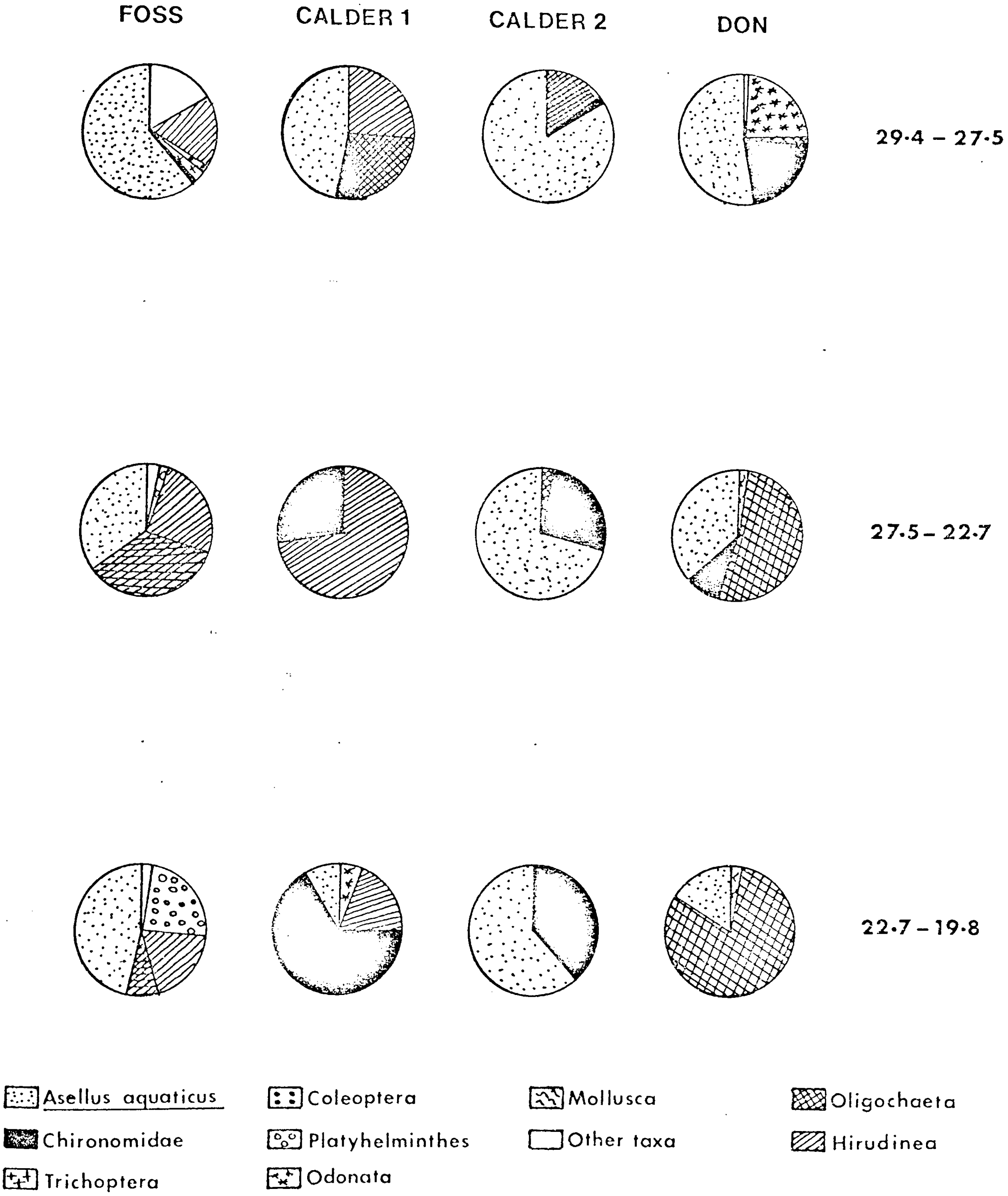
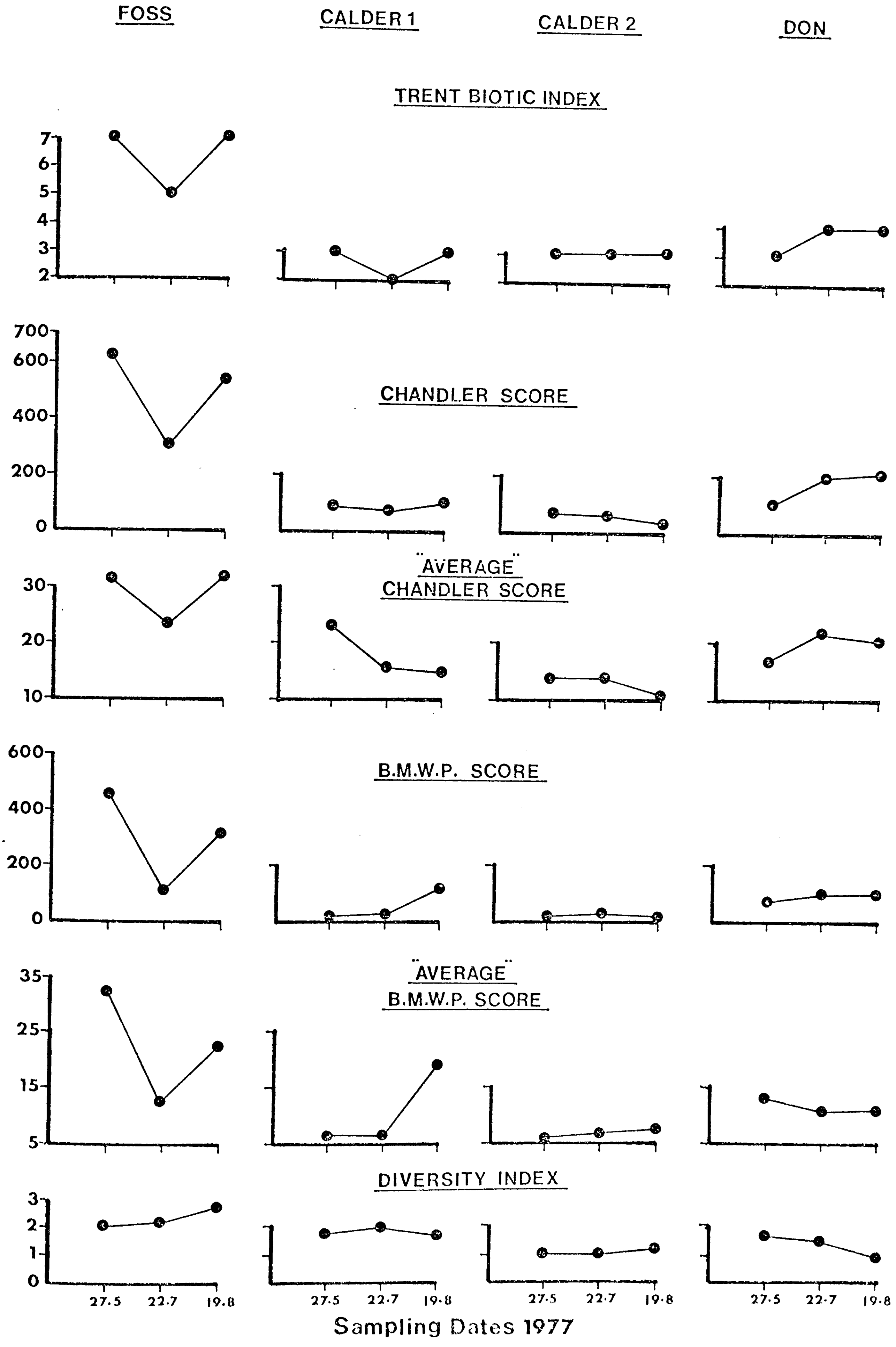


FIG. 4.44 SEASONAL VARIATION OF INDEXES AND SCORES OBTAINED BY S.AUF.U. SAMPLING FROM YORKSHIRE RIVERS.



Odonata (Ischnura elegans). The greater availability of oxygen in this river compared to the River Calder would explain the additional species present.

The percentage composition of major taxa collected by S.Auf.U. at each river for the three sampling periods are represented in Figure 4.43. From this data it is evident that the community inhabiting S.Auf.U. varies considerably throughout the season and makes interpretation of data difficult if only one spot sample is to be considered in a sampling season.

The differences observed between the various rivers are clearly represented when the data are processed to give indexes and scores (Figure 4.44). The River Foss typically produces higher scores than the more polluted Yorkshire rivers, although as seen with Figure 4.41, has a large variation throughout the season. The polluted rivers, as seen previously, produced low constant scores throughout the season. With respect to the River Calder, although the species composition is drastically different at the two stations, there is little difference in the scores produced at either station. This emphasises the insensitivity of all score systems at the polluted end of the spectrum compared to variations obtained with non-polluted rivers. Typically the River Don also falls into this insensitive zone producing scores very similar to the River Calder even though a larger number of taxa are represented in this river.

4.710 River Wye

4.7101 Sampling stations

The Wye catchment area which includes most of Herefordshire and

Powys, together with parts of Worcestershire, Gwent and Gloucestershire covers an area of approximately 4,183 km. The catchment is predominantly rural and in the west, the higher land is given to stock raising whereas the eastern parts are characterised by mixed farming, dairying and horticulture.

Difficulties were encountered in establishing sampling stations along this river due to problems arising out of riparian ownership in association with salmon angling. The two stations selected were Wye 1 one kilometre downstream of Hereford sewage works, treating domestic and cider brewery wastes, and Wye 2, forty kilometres downstream of Station 1 at Ross-on-Wye. Both pool stations had a mud and silt substratum with a negligible current velocity flowing through them (< 3 cm. sec^{-1}), and an abundance of aquatic macrophytes. S.Auf.U. sampling was only possible at these stations due to the depth of the river and difficulties experienced in taking grab samples. Permission was not granted to sample the immediate upstream riffle zones.

4.7102 Results and Discussion

The chemical data obtained from the Welsh Water Authority are presented in Annexe 3 (Table 28) and summarised in Table 4.18.

The chemical data associated with the River Wye reflect its status as a Class I river with high dissolved oxygen levels and low levels of pollutants.

Table 4.18 Physico-chemical data expressed as mean and range for the River Wye (mg. l^{-1})

Station	Analysis	S.S.	D.O.	BOD_5	Ammonia	Nitrate
1	WA	22.5 (4- 99)	10.4 (9.4-12.5)	2.2 (0.7-5.7)	0.1 (0.01-0.23)	1.3 (0.8-1.76)
2	WA	29.4 (4-145)	10.9 (8.4-13.0)	1.6 (0.4-3.3)	0.09 (0.01-0.26)	3.3 (1.0-6.9)

The detailed macroinvertebrate data are presented in Annexe 4 (Tables 75 - 76). The number of taxa collected by S.Auf.U. (Figure 4.45) indicates very little difference between the two stations with negligible seasonal variation at the lower station. Such a relationship is also observed with the faunal composition present at each station, (Figure 4.46) where a well established pool community develops on the S.Auf.U. Although the species composition was similar at both stations, larger numbers of *Asellus aquaticus*, Chironomini, *Gammarus pulex*, Oligochaeta and *Dugesia tigrina* were present at the lower station. In both pools, it is interesting to note that few riffle species were taken on S.Auf.U. as seen with previous riffle-pool studies. The percentage composition and abundance of major taxa recorded by S.Auf.U. over the sampling period (Figure 4.47) indicates a diverse fauna present at both stations which varies considerably thoughout the season. This variation was partly the result of macrophytes collecting around the samplers, restricting colonisation and the natural seasonal fluctuation of particular species.

FIG.4.45 NUMBERS OF TAXA COLLECTED MONTHLY ON S.AUF.U. IN POOLS, RIVER WYE.

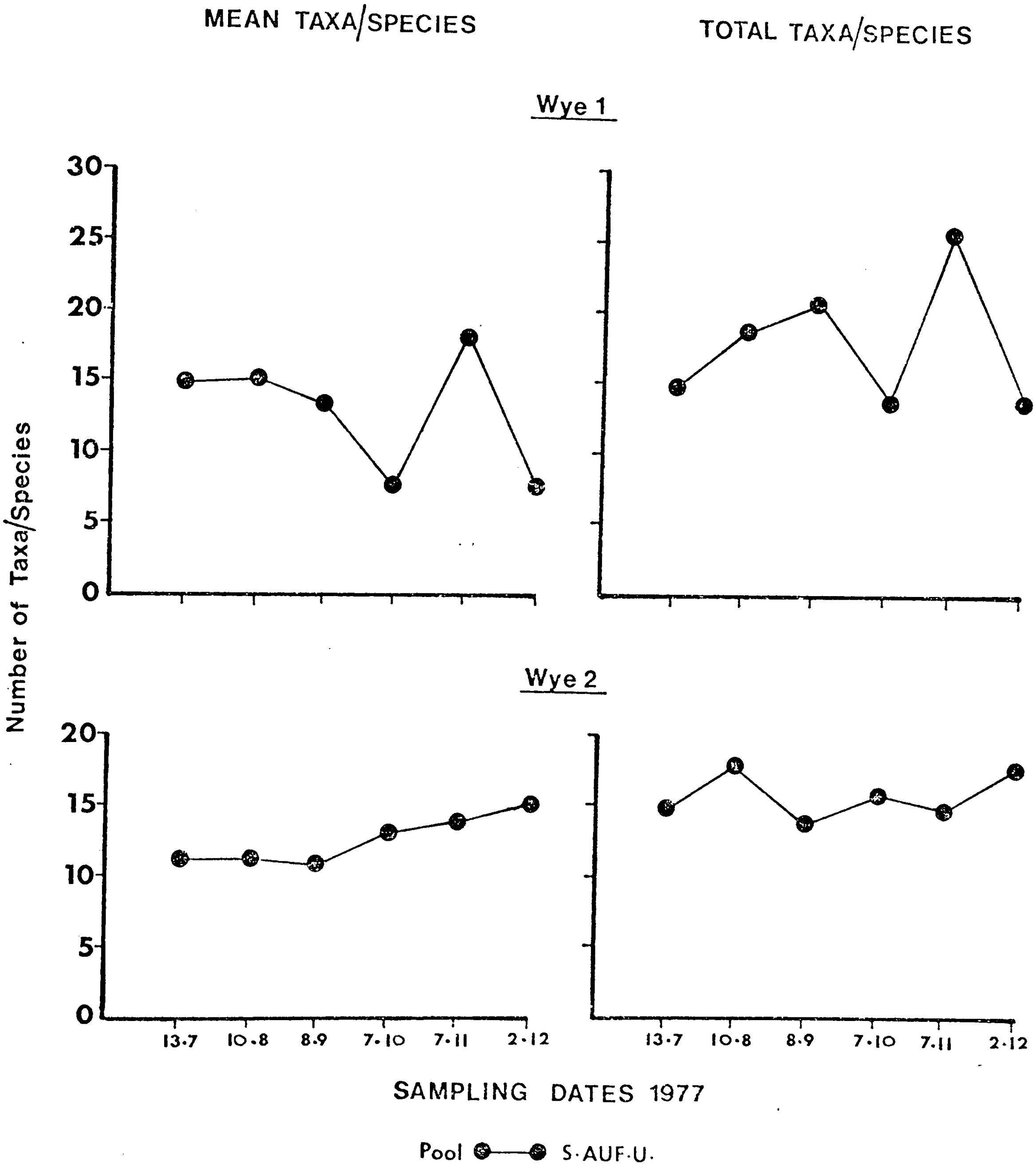


FIG. 4.46 RELATIVE ABUNDANCE OF MAJOR TAXA COLLECTED FROM RIVER WYE AS INDICATED BY S.AUF.U. SAMPLING.

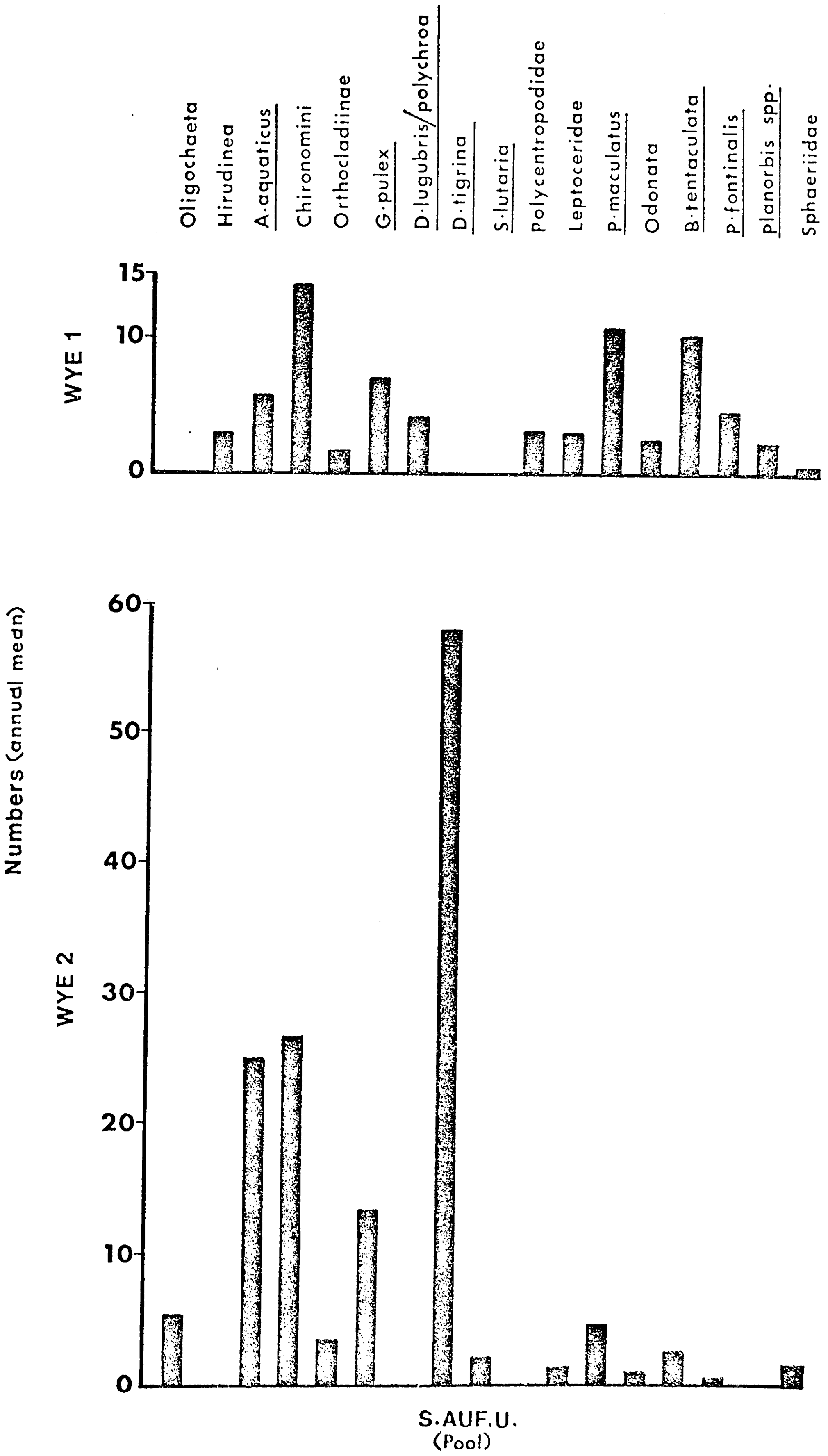


FIG. 4.47 PERCENTAGE COMPOSITION OF SELECTED TAXA COLLECTED BY S.AUF.U. FROM RIVER WYE.

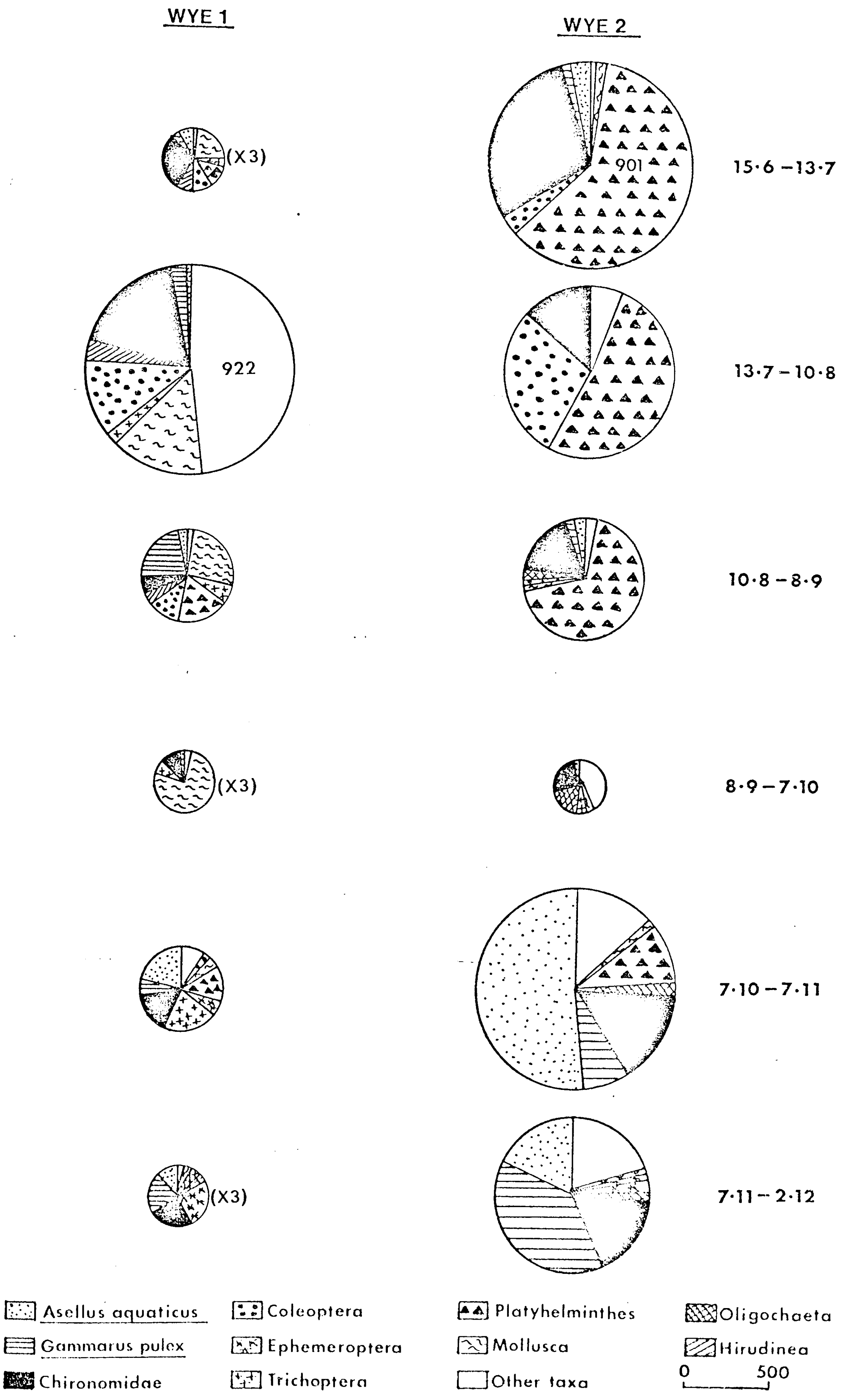
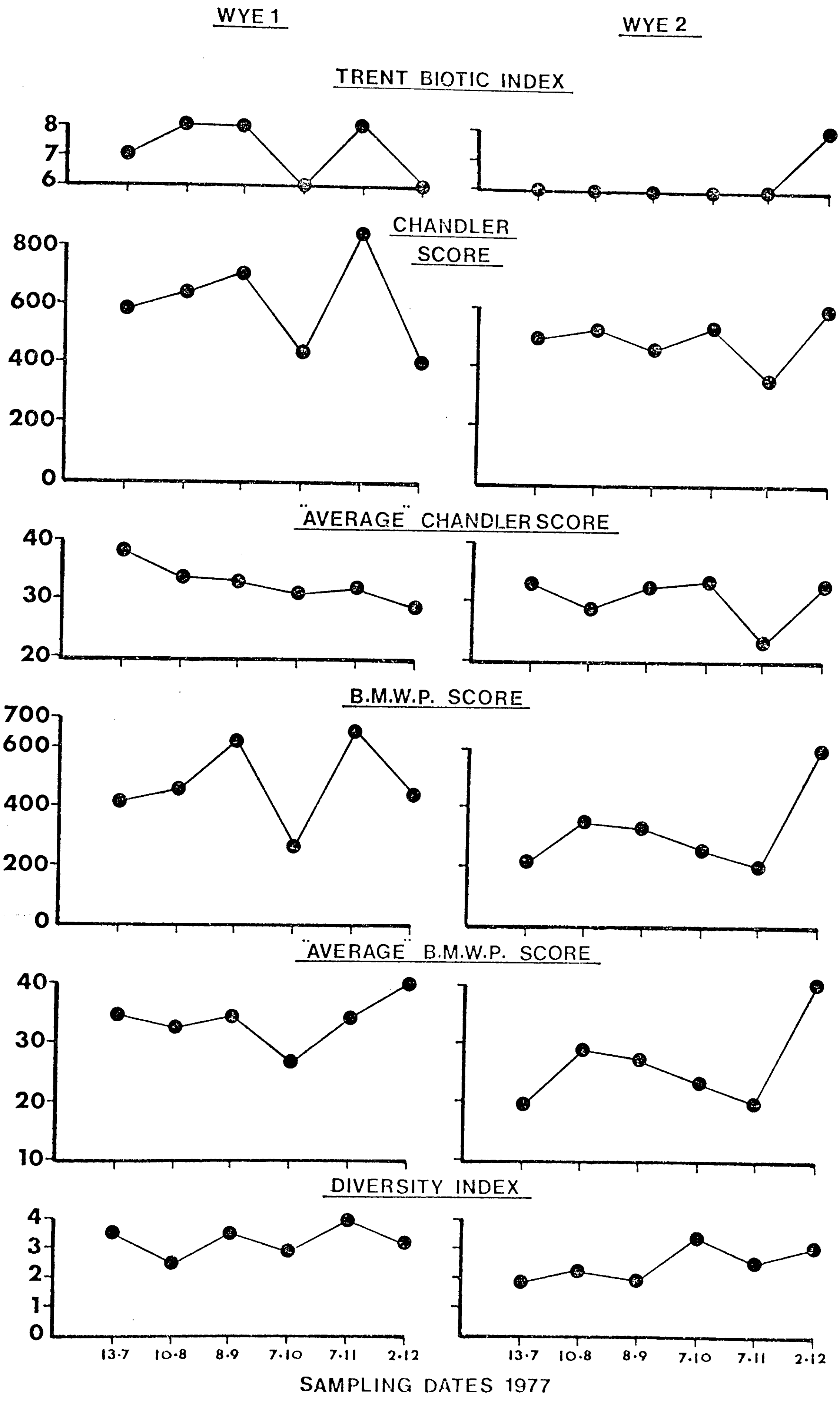


FIG. 4.48 SEASONAL VARIATION OF INDEXES AND SCORES OBTAINED BY S.AUF.U. SAMPLING FROM RIVER WYE.



The indexes and scores shown in Figure 4.48 reflect a slight deterioration in water quality between the two stations. Lower scores recorded at Station 2 were inevitably the result of substratum type affecting index values rather than water quality effects.

In biological surveillance studies one cannot be too selective in choosing sampling stations which are to be of some value in monitoring a particular problem. In the case of the River Wye such selection would certainly have been useful in these trial studies in order to objectively assess the adequacy of the sampling method with respect to water quality.

4.8 Potamon-lowland River case studies

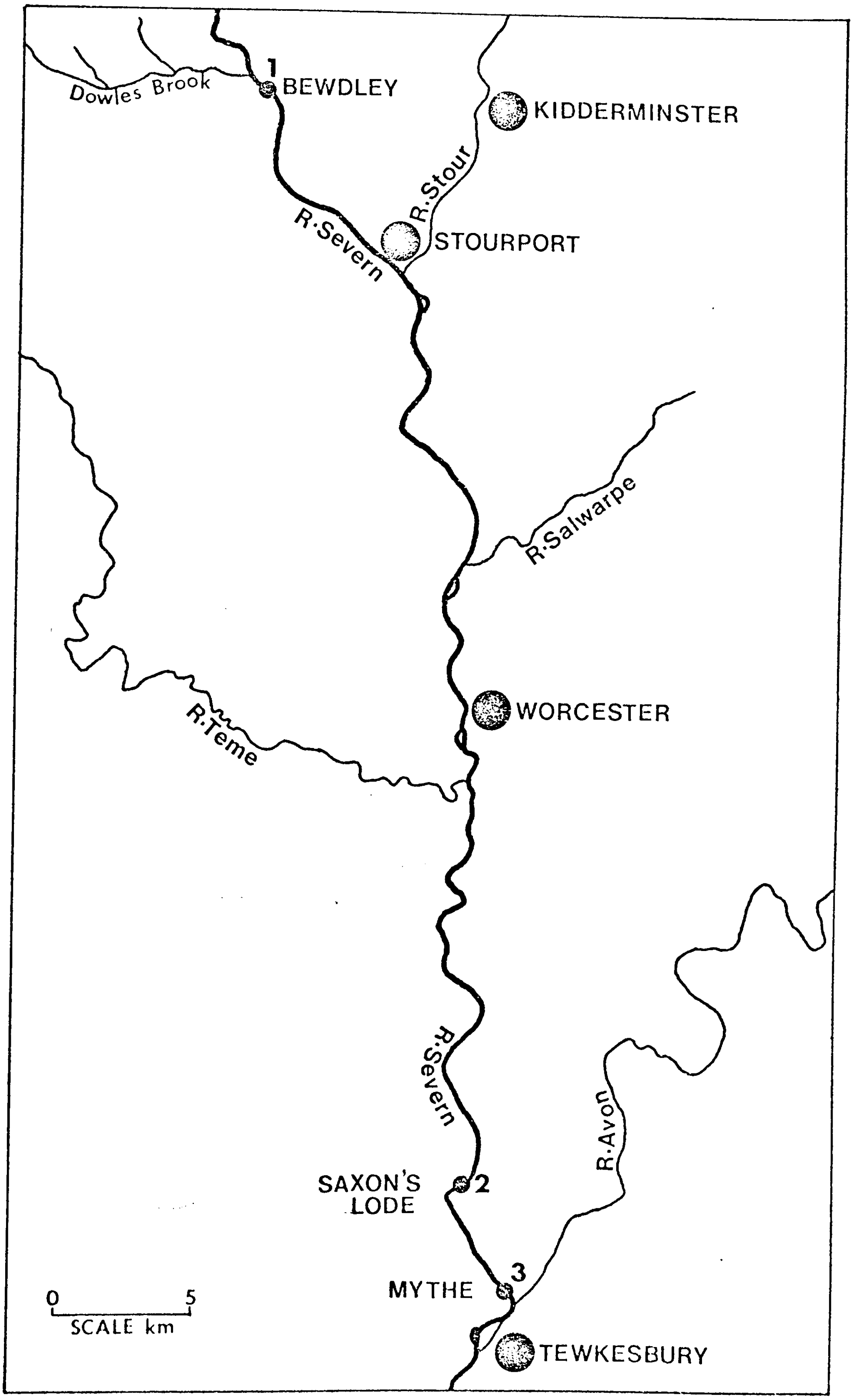
4.81 River Severn

4.811 Sampling stations

The River Severn which rises approximately 2 km. from the source of the River Wye, flows south-east passing through Shropshire, Worcestershire and Gloucestershire before entering the sea via the Bristol Channel.

The river was studied in the lower section of its course and for these investigations, three stations were selected as an example of a better quality larger river (Figure 4.49). The uppermost station, Severn 1, was situated at Bewdley where the lowermost natural riffle on the river was known to occur. Here the data from the riffle sampled by Aston cylinder sampler and from S.Auf.U. in a pool some 200 m. downstream, were compared. Grab sampling of the natural pool community was impossible at this station due to the presence of large boulders at the monitoring site. The other two stations studied (Severn 2 and 3) were some 43 and 48 km. downstream of the top station,

FIG. 4.49 MAP OF RIVER SEVERN , SHOWING THE DISTRIBUTION OF SAMPLING SITES.



in the potamon zone of the river where the substratum was predominantly silty with zones of dense clay and gelatinous mud. Near the banks of the river negligible current velocities were recorded, while in the middle of the river faster flows were known to occur, resulting in an extremely unstable substratum continually changing forming zones of compact gravel and silt. Thus such changes were probably due to an interaction of flow, heavy suspended solid loads and dredging activities. Sampling of the natural depositing community was therefore exceptionally difficult.

Sampling by S.Auf.U. was carried out on a monthly basis from May until December, 1977, with natural benthic sampling conducted at the same time when possible.

4.812 Results and Discussion

Chemical data supplied by the Severn-Trent Water Authority for stations Severn 1 and 3 are presented in Annexe 3 (Table 29) and summarised in Table 4.19.

Table 4.19 Physico-chemical data expressed as mean and range for the River Severn (mg. l^{-1})

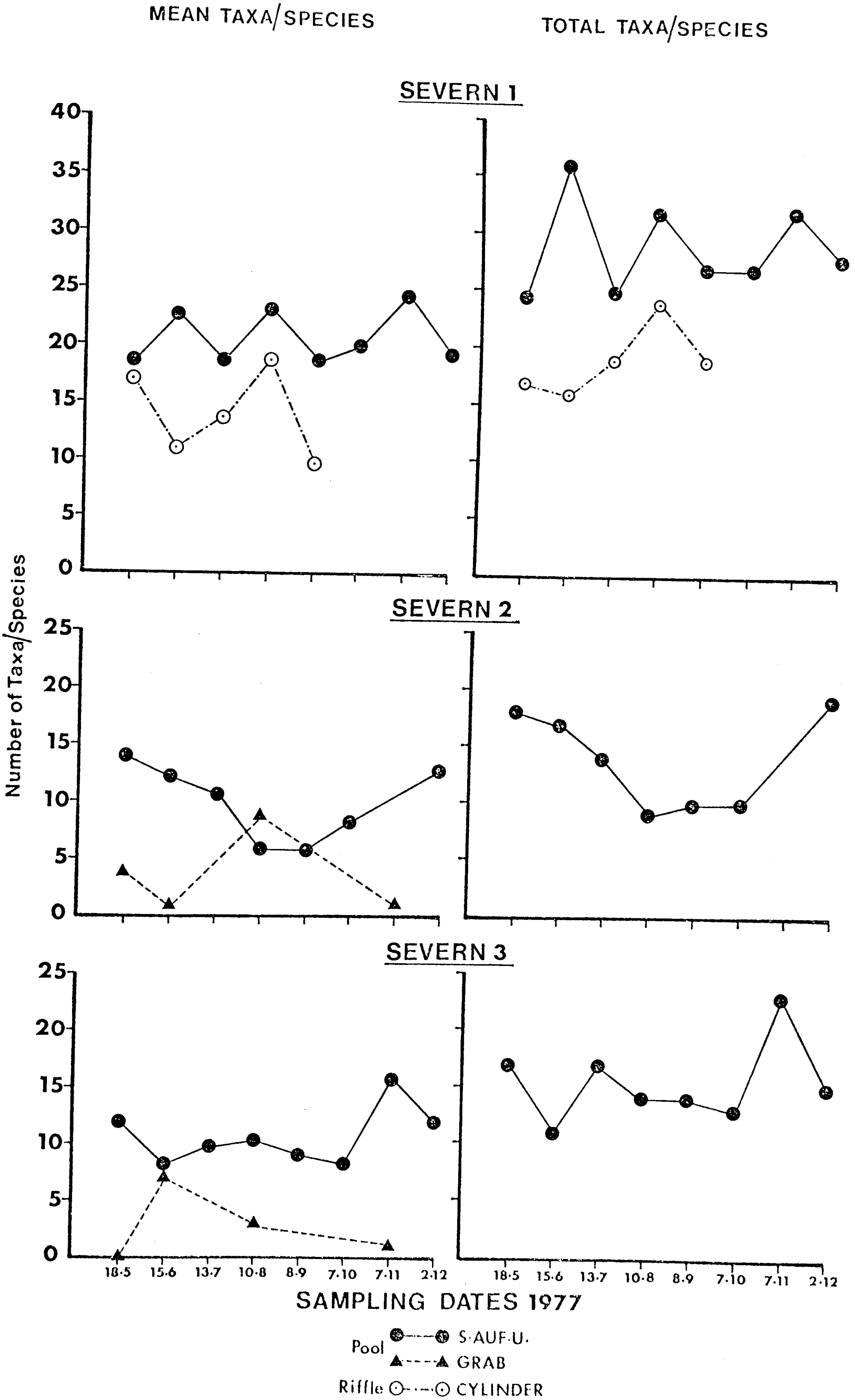
Station	Analysis	S.S.	D.O.	BOD_5	Ammonia	Nitrate
1	WA	33 (6-180)	11.4 (9.4-15.6)	2.5 (1.0-5.5)	0.1 (0.1 -0.4)	4.6 (1.5-9.6)
2	WA	35 (8-151)	11.7 (11.1-12.8)	2.8 (1.3-7.0)	0.2 (0.01-0.5)	5.5 (1.6-9.2)

The water quality as shown in Table 4.19 was little different at Station 1 and Station 3 where it is abstracted for public supply. The Severn is generally considered a good quality river and is classified as IB along this length. The river is extremely turbid from Station 1 to the sea due to a high suspended solid load and this will ultimately have a direct effect on the fauna found in the river.

The macroinvertebrate data arising from this study are presented in Annexe 4 (Tables 77 - 80). Figure 4.50 compares the number of taxa found on the S.Auf.U. after four weeks immersion in the pool with numbers taken by cylinder sampling of the upstream riffle at Station 1 and grab sampling of the benthos at Stations 2 and 3 (data kindly supplied by the Craig Goch Research Team, Malvern, of Central Water Planning Unit/Severn-Trent W.A. taking nine Petersen grabs at each station and bulking the data). At Station 1 it can be seen that on all the occasions when it was possible to sample the riffle, i.e. in the summer months, the numbers of taxa recorded was always less than those found on the S.Auf.U. in the downstream pool. At the two lower stations (2 and 3), the numbers of taxa on the S. Auf.U., although usually appreciably greater than those found in the natural depositing substratum as taken by grab sampler, were considerably lower than those found on the S.Auf.U. at Station 1 immediately below the riffle.

A comparison of the incidence of the more common taxa (Figure 4.51) found in the riffle and in the S.Auf.U. revealed that the S.Auf.U. taxa included almost all the taxa found in the riffle, e.g. Baetis rhodani, Caenis moesta, Ephemerella ignita, Hydropsyche spp., Gammarus pulex and Chironomidae, but also taxa more associated with depositing sub-

FIG. 4.50 NUMBERS OF TAXA COLLECTED MONTHLY ON S.AUF.U. IN POOLS AND BY DIRECT SAMPLING OF POOLS AND ASSOCIATED RIFFLES , RIVER SEVERN.



stratum which were not represented in the riffle, e.g. Corophium curvispinum, Asellus aquaticus, Sphaerium spp., Ephemera danica and Tubificidae. The majority of taxa taken on the S.Auf.U. were typical inhabitants of the marginal vegetation attracted to the hard surface of the S.Auf.U., e.g. most Hirudinea and Mollusca, Trichoptera, Coleoptera and Odonata. It would therefore appear that the S.Auf.U. at this station were colonised both from the upstream riffle by drift and from the natural substratum of the pool including the littoral vegetation.

At the lower two stations comparison of the more common taxa at these stations with those found at Station 1 (Figure 4.51) showed clearly that the difference in the numbers of taxa was accounted for by the absence of riffle species from the lower stations and reduced migration of taxa from marginal vegetation which was considerably sparse. The majority of taxa from these stations were of a depositing type, e.g. Oligochaeta, Chironomidae, C.curvispinum with a small community of molluscs from the littoral region, e.g. Theodoxus fluviatilis, Viviparus viviparus, Bithynia tentaculata and Potamopyrgus jenkinsi. These results again indicate the importance of the proximity of riffles in contributing to the taxa found on the S.Auf.U. and hence to the resultant biotic scores.

The effects of these differences in community structure present in the River Severn are reflected in the calculated biotic scores. At Station 1, the Trent Biotic Index produced similar values for S.Auf.U. and cylinder sampling, while the higher number of taxa on the S.Auf.U. in association with relative abundance elevated the Chandler Score compared to the cylinder, although this difference

FIG. 4.51 RELATIVE ABUNDANCE OF MAJOR TAXA COLLECTED ON S.AUF.U. AT THREE STATIONS ON THE RIVER SEVERN AND AS COLLECTED BY DIRECT SAMPLING OF THE UPSTREAM RIFFLE AT STATION 1.

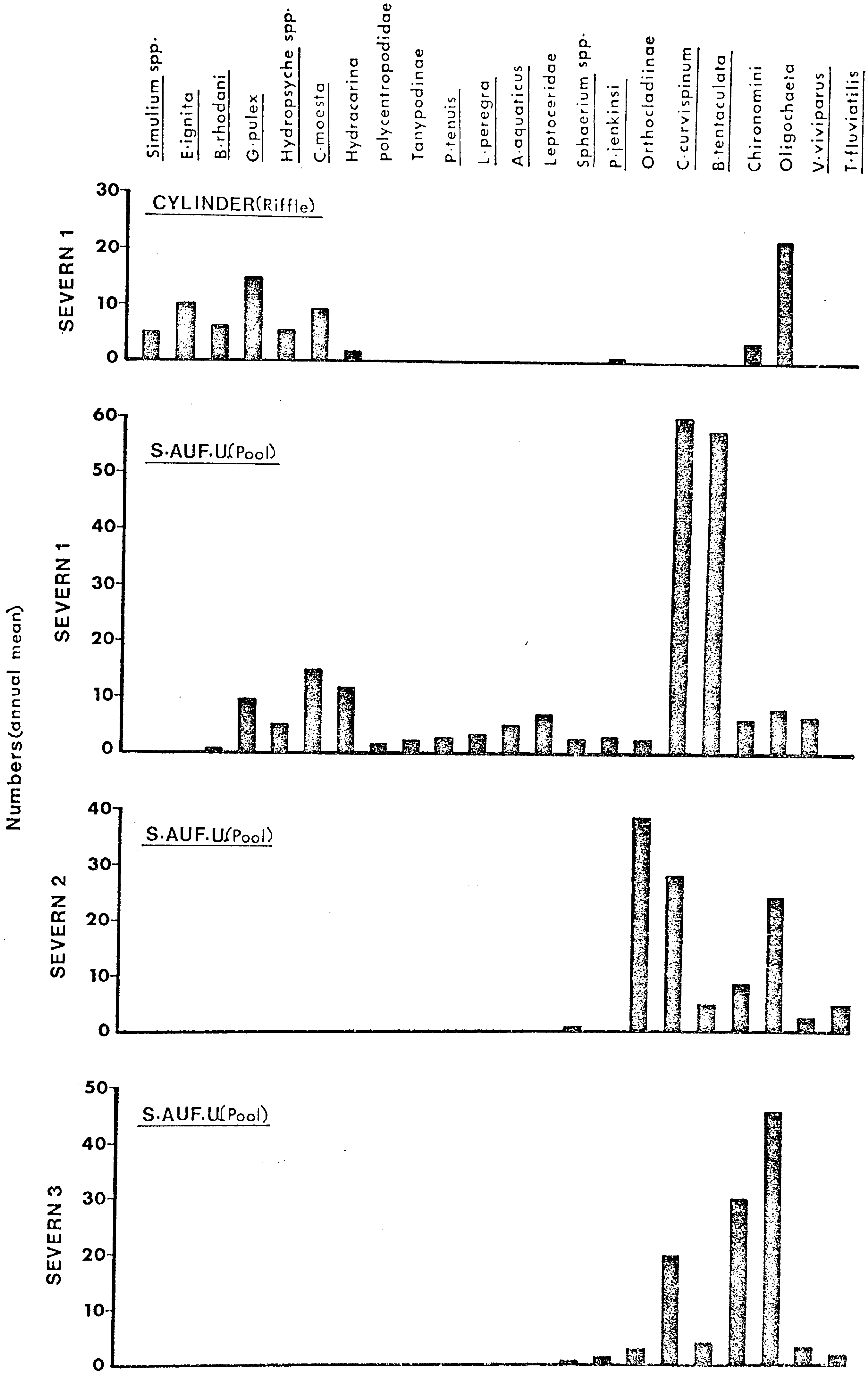
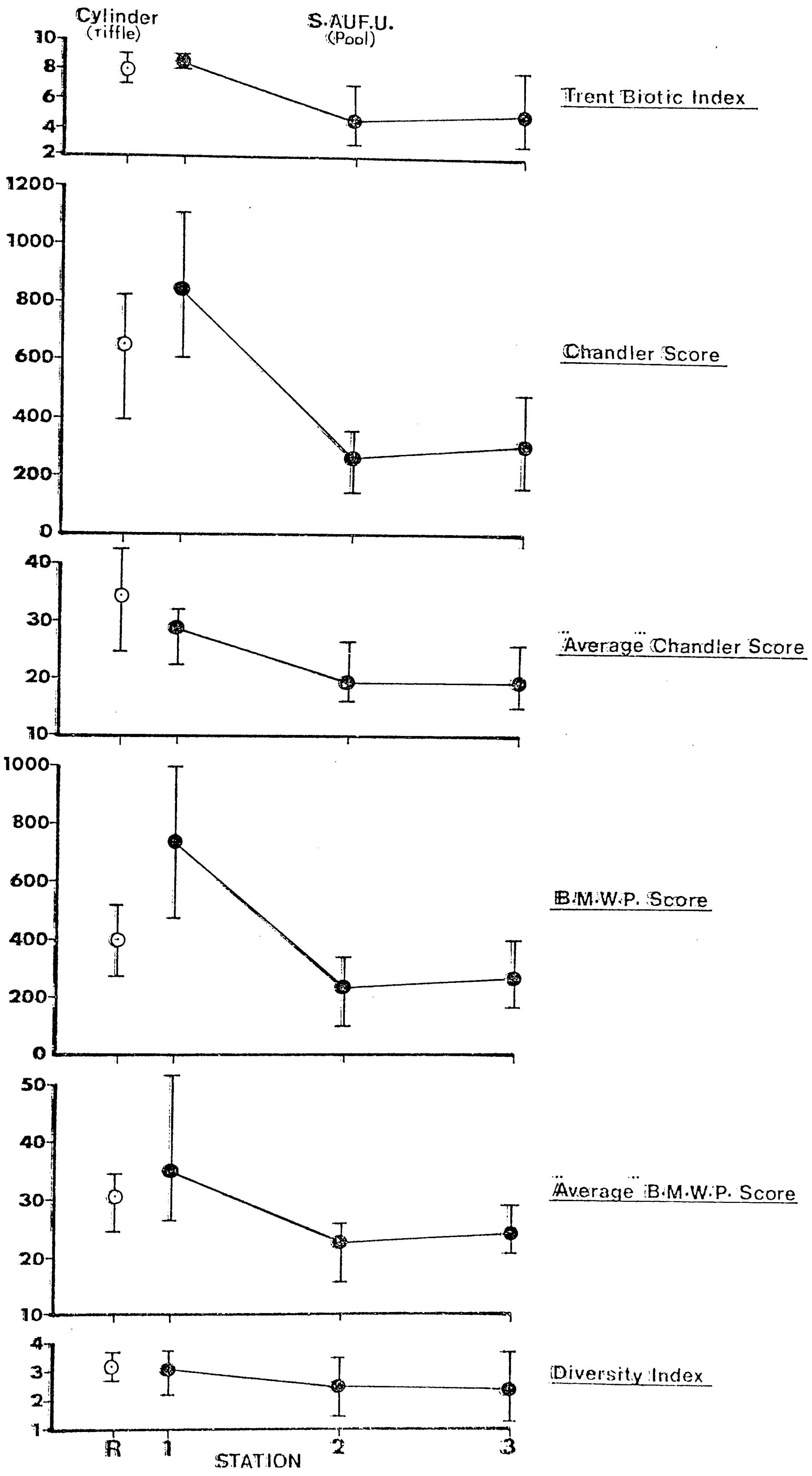


FIG. 4.52 COMPARISON OF INDEXES AND SCORES DERIVED FROM DATA OBTAINED BY DIFFERENT SAMPLING METHODS FROM THE RIVER SEVERN.



was reduced by 'averaging' the Chandler Score. There was an even greater difference in the B.M.W.P. Score probably because depositing substrata taxa, not contributing to the Chandler Score, do contribute to the B.M.W.P. Score. Again, 'averaging' the score, markedly reduced the difference. The different scores for Stations 2 and 3 were appreciably lower than those for Station 1, and since there was no evidence of any significant differences in water quality, it must be concluded that the differences were due to differences in the natural fauna in the rhithron and potamon zones. This would need to be accounted for in any index or score system for S.Auf.U. data.

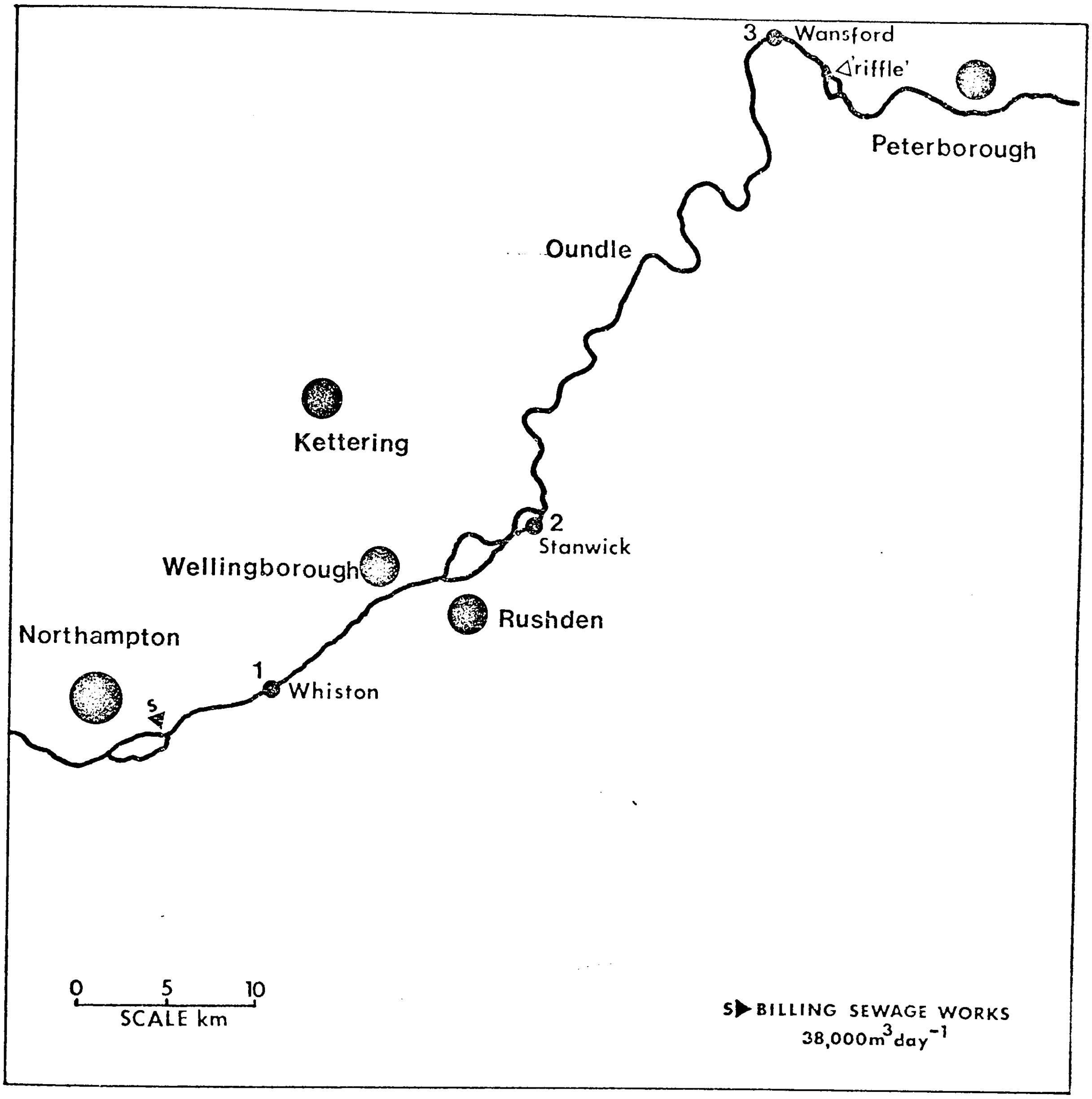
4.82 River Nene

4.821 Sampling stations

The River Nene located between Northamptonshire and Huntingdonshire was selected for study as a typical lowland river with no good riffles, although at times of low flow, there were some shallows which could be sampled by the heel-kick method. Three stations were selected for study between Northampton and Peterborough representing a range of water qualities (Figure 4.53). In terms of volume, the major discharge was above Station 1 at Whiston, where on average 38,000 m^3 of effluent was pumped into the river each day from Billing sewage works. Numerous smaller discharges entered the river between Stations 1 and 2, but few were found below Station 2. At Station 3, water was abstracted for the Peterborough area.

Sampling by S.Auf.U. was carried out on a rotating four and eight-week immersion period due to distances involved in travelling from Birmingham. Grab samples of the natural depositing substratum were also taken on each visit. The substratum of the river was a

FIG. 4.53 MAP OF RIVER NENE , SHOWING THE DISTRIBUTION OF SAMPLING SITES.



mixture of clay and silt with negligible current velocities.

4.822 Results and Discussion

The physico-chemical data resulting from this study are presented in Annexe 2 (Table 19) and Annexe 3 (Table 30) and summarised in Table 4.20.

The chemical data indicates very little difference between Stations 1 and 2 other than the BOD_5 which shows some sign of deterioration in water quality. The river appears to be well oxygenated and has fewer suspended solids compared to the River Severn. A reduction in BOD_5 and ammonia and an increase in dissolved oxygen between Stations 2 and 3 indicates some recovery in the downstream waters. Nitrate levels throughout the river were high but constant.

The macroinvertebrate data arising from this study are presented in Annexe 4 (Tables 81 - 86). Comparisons of the numbers of taxa taken by S.Auf.U. and by direct grab sampling (Figure 4.54) shows that more taxa were taken by S.Auf.U. on all occasions. Both methods of sampling indicated the deterioration in water quality between Stations 1 and 2 by the reduction in taxa present. The S.Auf.U., however, most clearly indicated the improved conditions considered to exist at Station 3. Grab sampling proved difficult at this station due to the presence of compact clay in the substratum. The efficiency of sampling technique using similarity coefficients (Figure 4.55) between S.Auf.U. and grab sampling revealed high similarity values in the polluted stations compared to the non-polluted station where S.Auf.U. tended to collect greater numbers of taxa.

Table 4.20 Physico-chemical data expressed as mean and range for the River Nene (mg. l^{-1})

Station	Analysis	S.S.	D.O.	BOD_5	Ammonia	Nitrate
1	ASTON	10 (4-14)	10.9 (8.8-13.8)		1.1 (0.75-1.35)	10.0 (5.8-12.7)
	WA	15 (5-29)	9.1 (1.6-12.7)	6.9 (4.1-13.5)	1.3 (0.03-4.0)	13.8 (10.5-16.7)
2	ASTON	12 (6-20)	10.4 (7.9-12.4)		1.43 (0.85-2.1)	12.0 (7.1-15.6)
	WA	15 (5-31)	9.0 (4.7-12.3)	10.5 (2.0-27.0)	1.29 (0.45-2.5)	13.4 (11.7-17.7)
3	ASTON	16 (6-34)	11.7 (9.6-15.1)		0.96 (0.5-1.25)	10.5 (7.0-14.0)
	WA	17 (2-78)	11.2 (7.5-14.3)	4.4 (1.2-12.5)	0.31 (0.05-2.4)	13.6 (6.8-19.6)

The change in the faunal composition between Station 1 and 2 was clearly demonstrated by both sampling methods (Figure 4.56). At Station 1, S.Auf.U. sampling revealed a fauna dominated by pollution-tolerant species present in reasonably high numbers, e.g. Tubificidae, *Erpobdella octoculata*, *Helobdella stagnalis*, *Asellus aquaticus*, Chironomini and Mollusca. At Station 2 with a slight deterioration in water quality, the numbers of *A.aquaticus* increased while other species recorded previously decreased drastically in numbers. The reduction in diversity between Station 1 and 2 was also revealed by

FIG. 4.54 NUMBERS OF TAXA COLLECTED EVERY FOUR AND EIGHT WEEKS ON S.AUF.U. IN POOLS AND BY DIRECT SAMPLING OF POOLS , RIVER NENE.

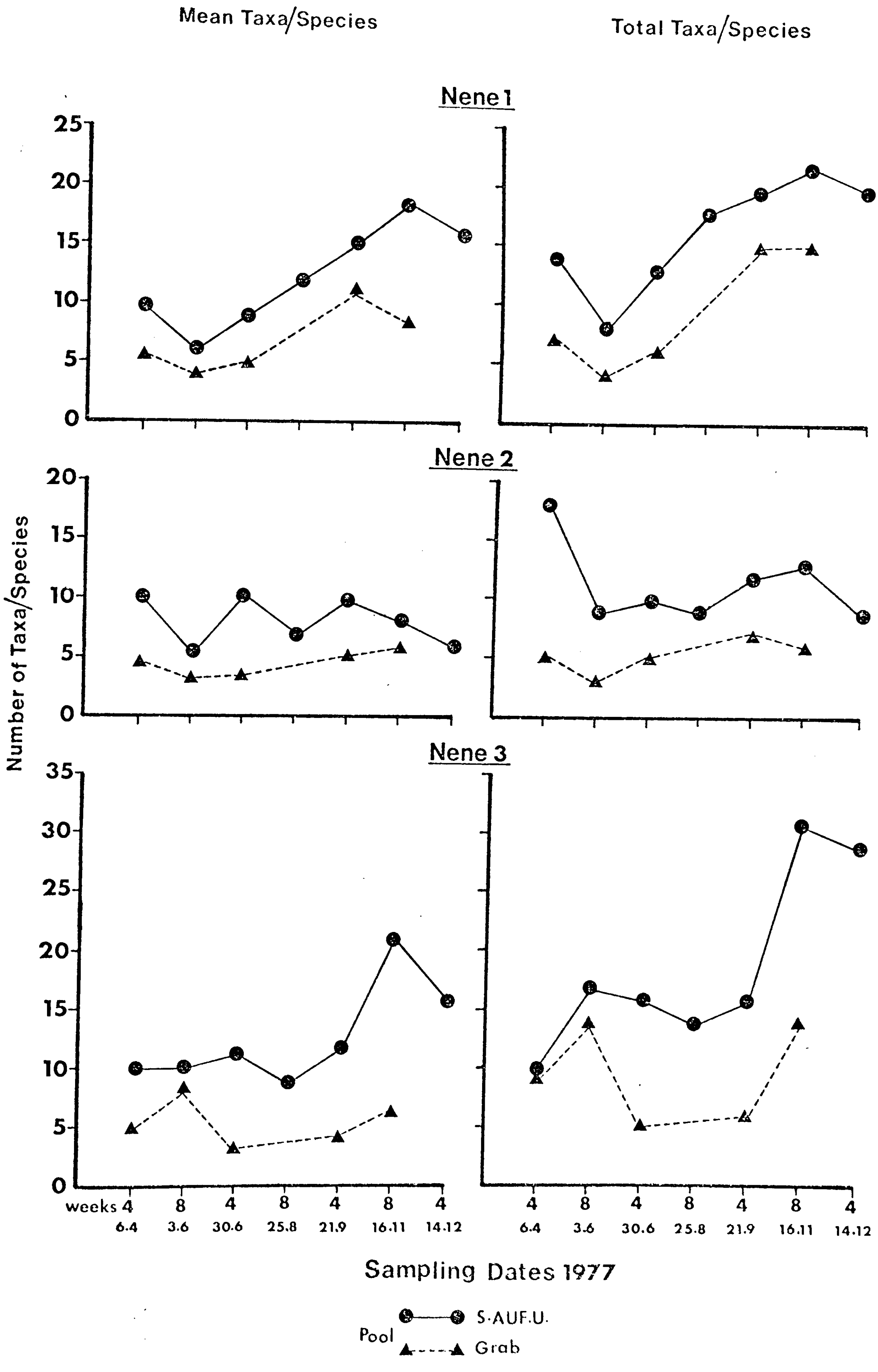


FIG. 4.55 COMPARISON OF S.AUF.U. AND GRAB SAMPLING IN POOLS BY SIMILARITY COEFFICIENT ANALYSIS , RIVER NENE.

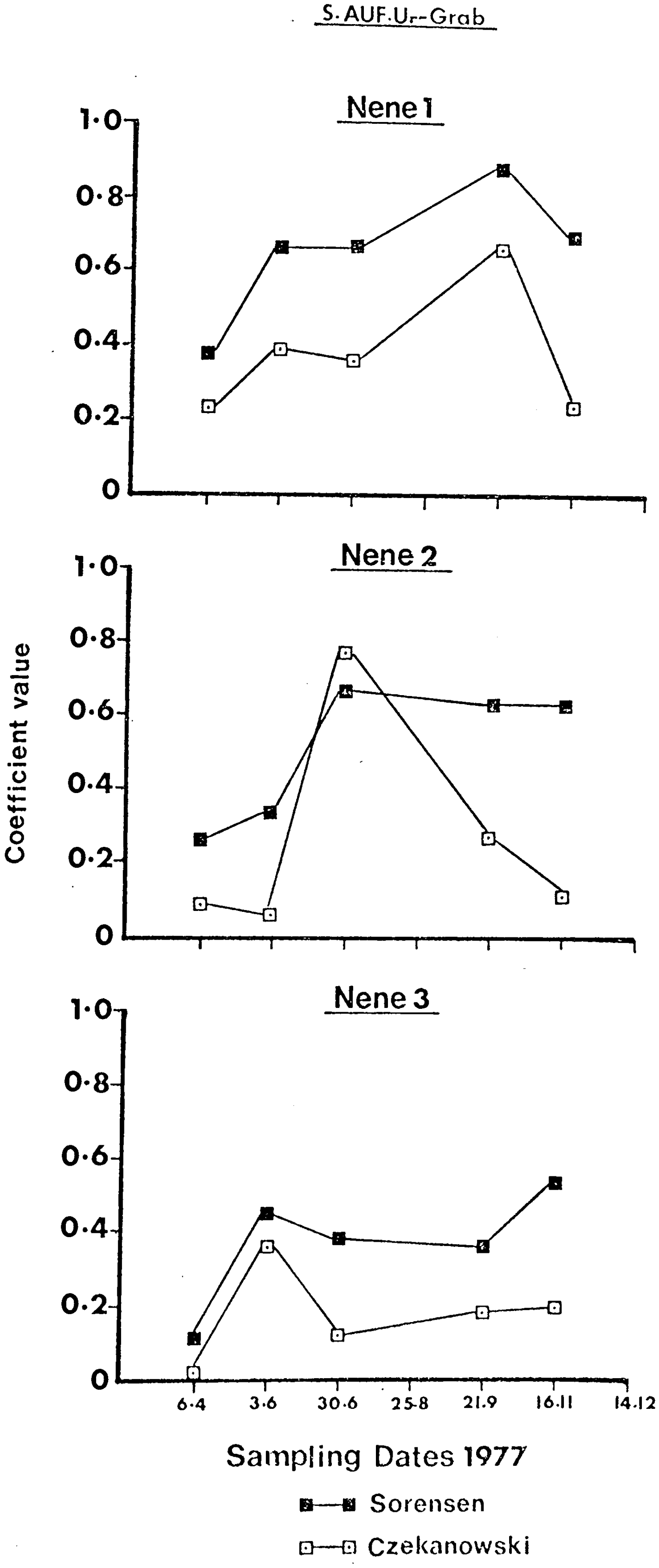
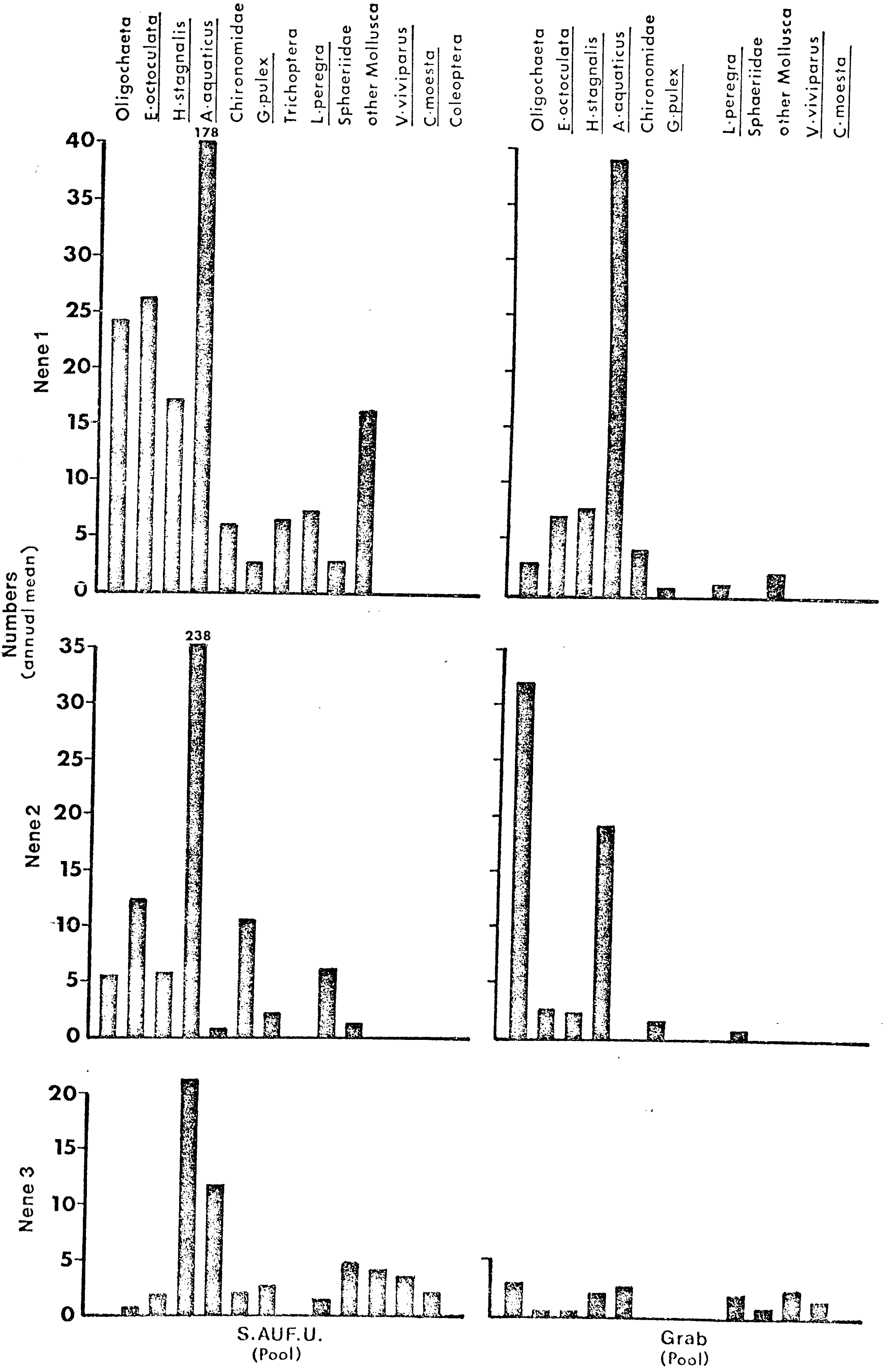


FIG. 4.56 RELATIVE ABUNDANCE OF MAJOR TAXA AT THREE STATIONS ON THE RIVER NENE AS INDICATED BY DIFFERENT SAMPLING METHODS.



grab sampling with an increase in the numbers of Tubificidae in the sediments. The recovery in water quality at Station 3 was more clearly indicated by S.Auf.U. sampling than grab sampling. The numbers of *A.aquaticus* were greatly reduced as seen by S.Auf.U., while numbers of Trichoptera (*Polycentropus flavomaculatus*, Leptoceridae), *Caenis moesta*, Coleoptera (*Platambus maculatus*), *Viviparus viviparus*, *Baetis rhodani* and *Cloeon dipterum* increased. Grab sampling proved rather inadequate for detecting these changes.

The percentage composition of major taxa collected by S.Auf.U. for each station throughout the sampling season are presented in Figure 4.57. Again as seen previously with other rivers, the species composition varies within an annual cycle. In the polluted zone detritivores dominated by *A.aquaticus* and to a lesser extent Oligochaeta and *Gammarus pulex* made up the largest proportion of the community, often in large numbers, while in the cleaner zone their dominance was reduced in extent. Chironomids, Trichoptera and Ephemeroptera were now the dominant taxonomic groups, although *A.aquaticus* was still present in association with the silty depositing substratum.

The indexes and scores shown in Figure 4.58 reflect the differences discussed previously, i.e. a reduction in water quality between Stations 1 and 2, with recovery taking place by Station 3. Using B.M.W.P. Score the differences between the stations are more clearly indicated by the S.Auf.U. data.

FIG.4.57 PERCENTAGE COMPOSITION OF SELECTED TAXA COLLECTED BY S.AUF.U. IN THE RIVER NENE.

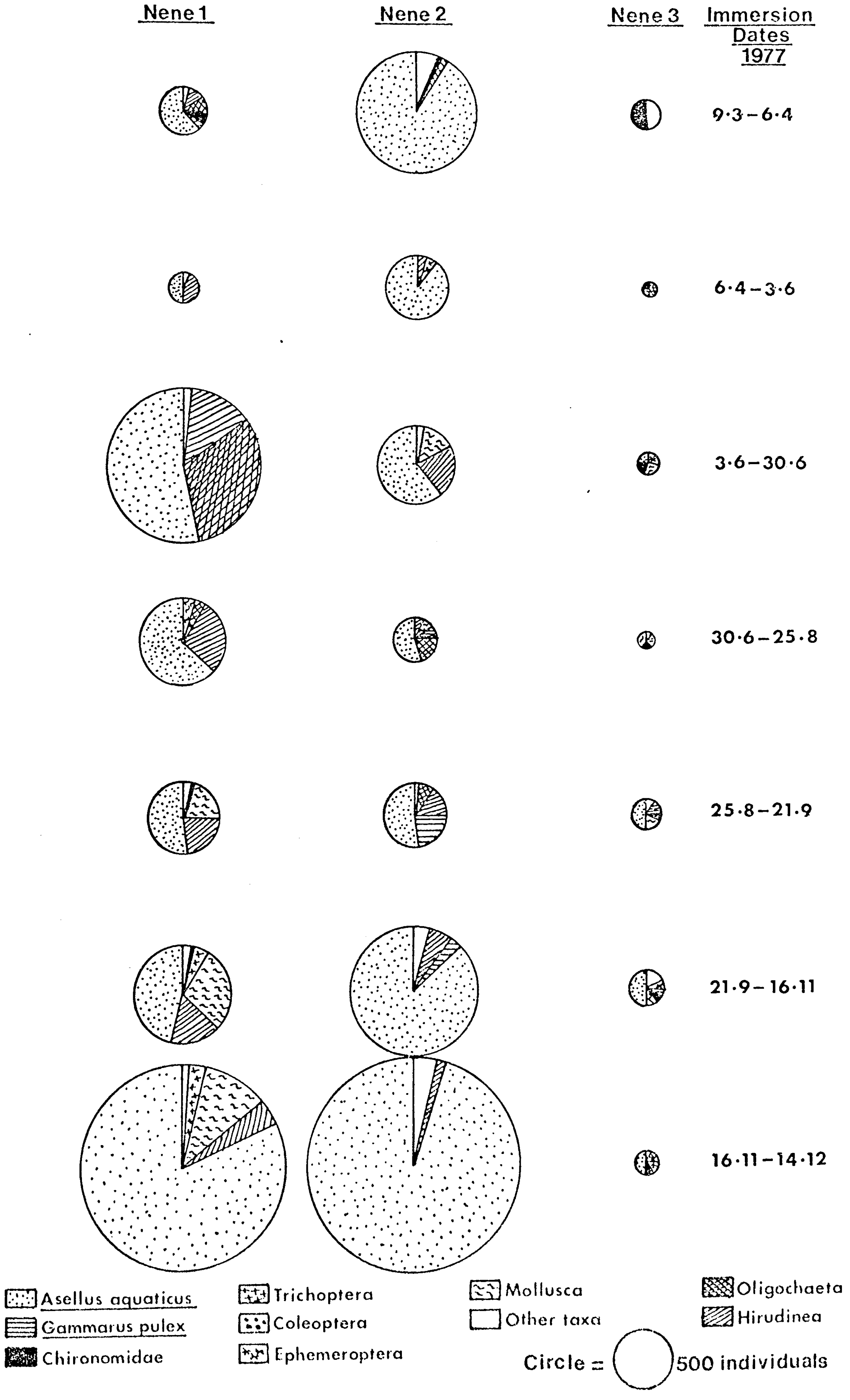
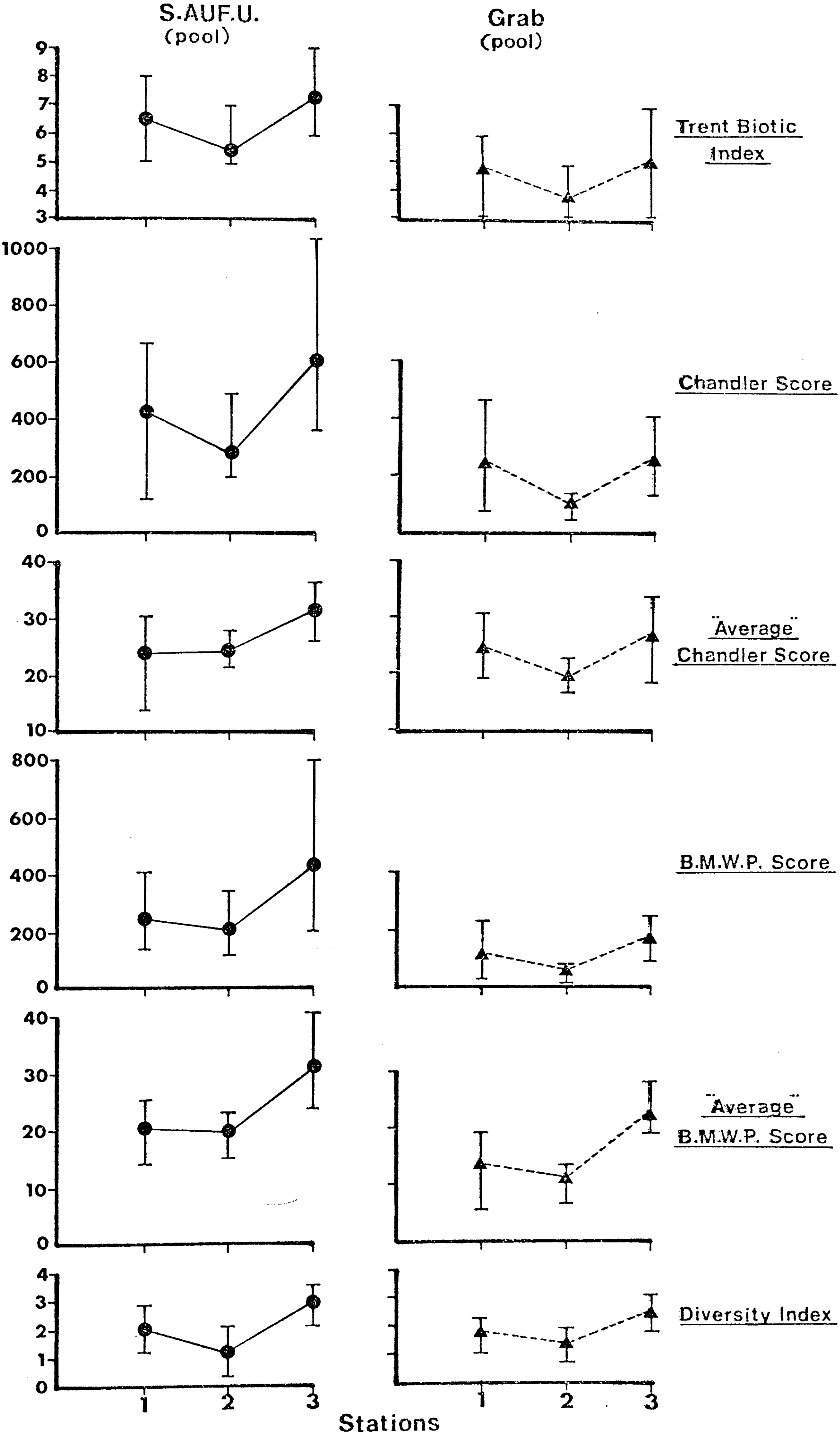


FIG.4.58 COMPOSITION OF INDEXES AND SCORES DERIVED FROM DATA OBTAINED BY DIFFERENT SAMPLING METHODS FROM THE RIVER NENE.



4.83 River Avon and Somerset Drains

4.831 Sampling stations

Further examples of potamon zones were chosen in the Wessex Water Authority area where riffle zones were limited, except in periods of low flow. One station was chosen on the River Avon at Twerton (ST 726648) near Bath, where the river was very deep (>4.5 m) and slow flowing with a hard gravel substratum. However, when in full spate, the river was known to have a current velocity greater than 1 m $sec.^{-1}$. The other stations selected were located in the drainage area of the Somerset flats. The stations examined were the South Drain near Huntspill (ST 367430) and the King's Sedgemoor Drain at Bawdrip (ST 341391). Water levels and flow of the drains were controlled by pumping stations resulting in near static conditions. During the summer months, the surface of the drains were covered with dense growths of Duckweed (Lemna). The substratum of the South Drain consisted entirely of Peat and silt, while the King's Sedgemoor Drain was composed mainly of hard compact gravel.

Sampling by S.Auf.U. was carried out on a rotating four and eight-week immersion period due to the distances involved in travelling from Birmingham. Sampling of the natural benthic community was only possible at the South Drain using a single heel-kick technique.

4.832 Results and Discussion

The chemical data are presented in Annexe 2 (Tables 20 - 21) and Annexe 3 (Table 29) and summarised in Table 4.21.

The limited physico-chemical data available suggested that the River Avon had a moderate water quality, while the King's Sedgemoor Drain compared to the South Drain was of inferior water quality due to the presence of higher levels of ammonia.

Table 4.21 Physico-chemical data expressed as mean and range for the River Avon and Somerset Drains (mg. l^{-1})

Station	Analysis	S.S.	D.O.	BOD_5	Ammonia	Nitrate
Avon	WA	28 (3-230)		3.7 (1.6-10.8)	0.4 (0.08-1)	6.6 (4.7-12.5)
South Drain	ASTON	12 (6- 17)	8.8 (4.3-14.3)		0.86 (0.5-1.2)	2.4 (0.05-5.4)
King's Sedgemoor Drain	ASTON	10 (2- 26)	10.2 (7.1-15.7)		2.3 (0.75-6.2)	2.6 (0.4- 4.9)

The macroinvertebrate data arising from this study are presented in Annexe 4 (Tables 87 - 90). The numbers of taxa taken on the S.Auf.U. (Figure 4.59) in the River Avon were somewhat higher than those in the potamon zone of the River Severn (Figure 4.50) probably the result of a hard substratum supporting a wider diversity of species compared to the softer sediments in the River Severn. Since this was the first set of data collected from the River Avon around the Twerton area due to previous difficulties in sampling, the S.Auf.U. sampler proved successful in obtaining a benthic sample. An eight-week immersion period generally collected a larger number of taxa than a four-week immersion period, suggesting such a time period may be more suitable for the larger rivers. Of the two drains, the numbers of taxa were less in the King's Sedgemoor Drain than in the South Drain (Figure 4.59). Where it was possible to take a heel-kick sample of the natural substratum in the South Drain, fewer taxa were recorded

FIG. 4.59 NUMBERS OF TAXA COLLECTED EVERY FOUR AND EIGHT WEEKS ON S.AUF.U. IN A LOWLAND RIVER (AVON) AND SOMERSET DRAINS (SOUTH DRAIN AND KING'S SEDGEMOOR DRAIN).

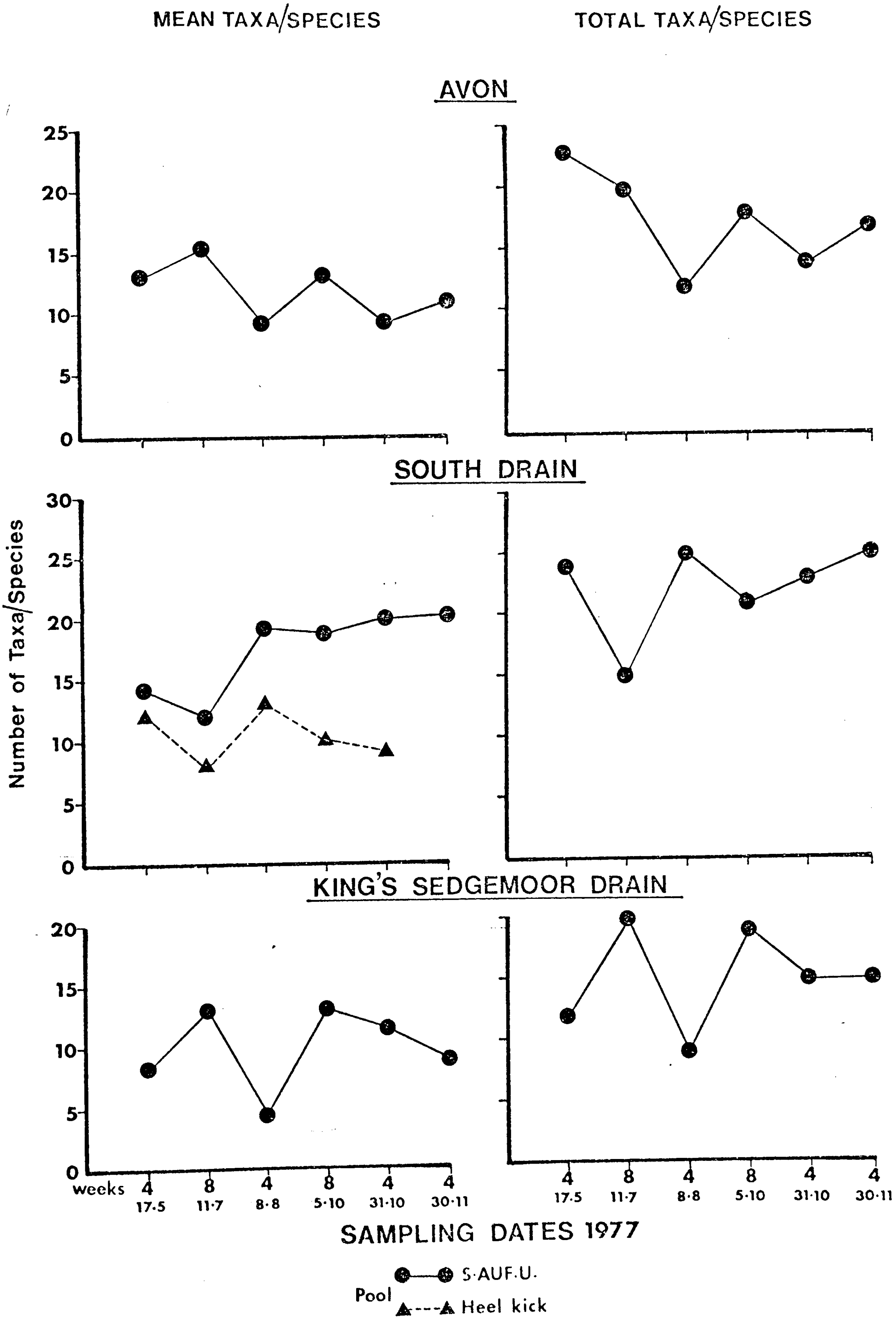


FIG.4.60 RELATIVE ABUNDANCE OF MAJOR TAXA COLLECTED ON S.AUF.U. IN THE RIVER AVON AND SOMERSET DRAINS.

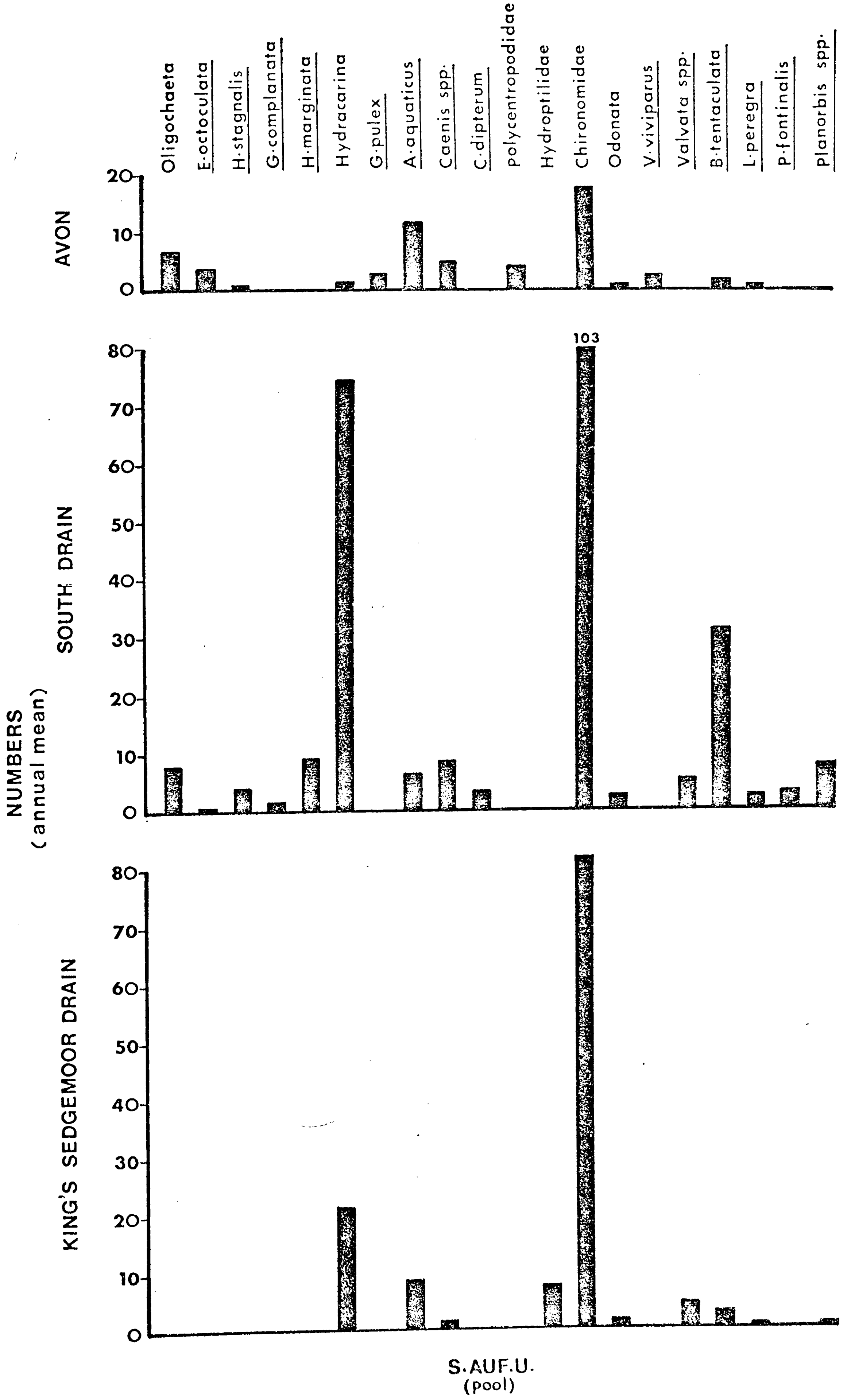
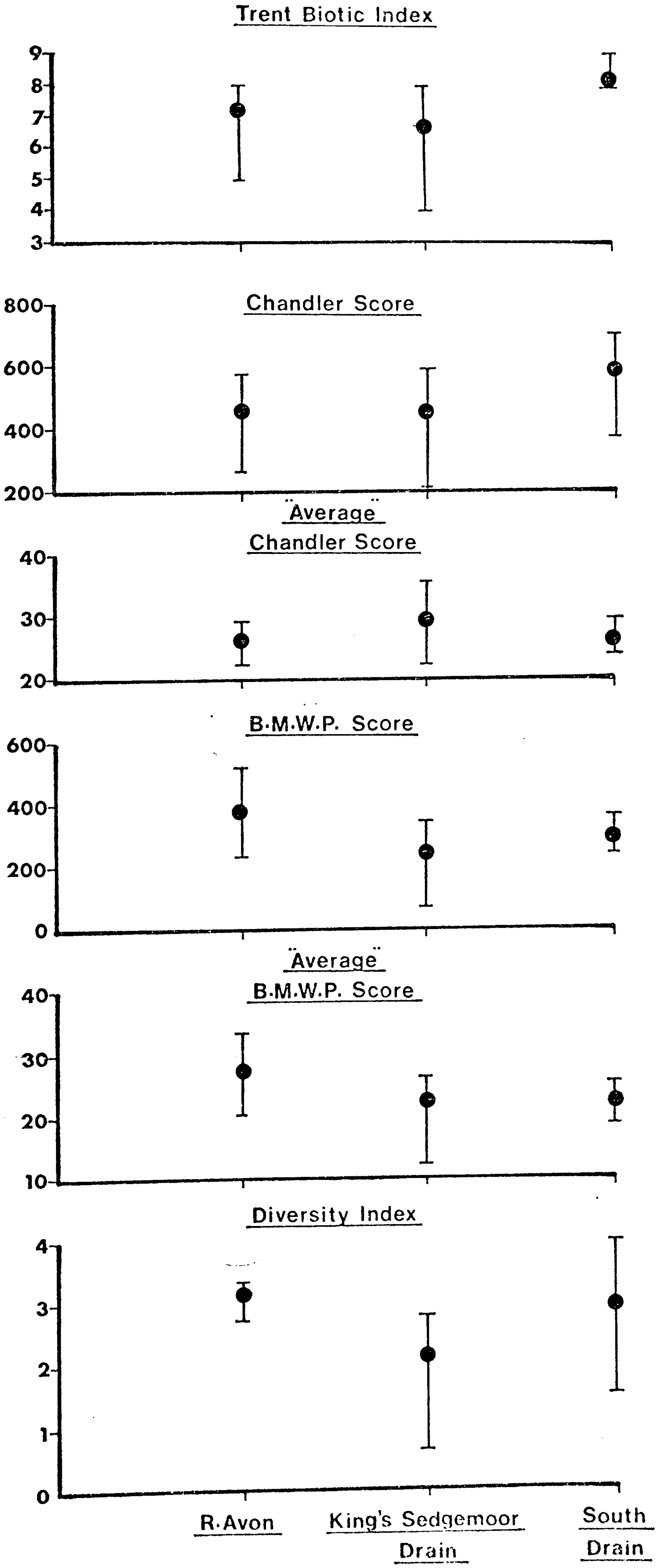


FIG. 4.61 COMPARISON OF INDEXES AND SCORES DERIVED FROM S.AUF.U. DATA IN THE RIVER AVON AND SOMERSET DRAINS.



than on the corresponding S.Auf.U. sampler on all occasions. In the King's Sedgemoor Drain, an eight-week immersion period again proved the most successful, while in the South Drain fairly stable results were obtained throughout the season.

The relative abundance of the major taxa are presented in Figure 4.60. The River Avon appears to support a fairly wide diversity of fauna typical of a very mildly polluted river. The taxa are mainly pool species, although the majority as seen in many other lowland rivers are attracted to the hard surface of the S.Auf.U.. Asellus aquaticus and Chironomini were the dominant taxa although only recorded in low numbers. The South Drain supported a larger number of Taxa than the River Avon with many species associated with lentic waters, e.g. Cloeon dipterum, Plea leachi, Ilyocoris cimicoides, Sigara sp., Corixa sp., Acroloxus lacustris and other Coleoptera, Trichoptera and Mollusca. Generally the species found in the South Drain were representative of a clean water community. This was also the case with the King's Sedgemoor Drain although the species diversity in this dyke was greatly reduced. The fauna was dominated by Chironomidae and Hydracarina and occasionally lentic species.

The respective biotic scores (Figure 4.61) were on the whole higher than those found in the lower sections of the River Severn, probably attributed to more stable flows and less physical stress.

4.9 Replication tests with S.Auf.U.

According to Elliott (1971) the distribution of many benthic invertebrate species is frequently contagious (clumped), a large variation is encountered in sampling natural populations and a small number of samples is statistically inaccurate. Reliable estimates can, therefore, often be made only after considerable sampling effort, but in most studies a compromise must be reached between accuracy and expenditure of effort (Hellawell, 1978). Thus in surveillance studies, the number of samples needed to be taken depends largely on the purpose of the survey (e.g. qualitative vs. quantitative) and on the number of man-hours that can be devoted to processing the collections. As a consequence, it was decided that field tests would be conducted to determine the normal variability in the macro-invertebrates collected by replicate bottom S.Auf.U. samplers placed in pools. More S.Auf.U. per station would permit a statistical treatment of the data.

On different occasions throughout the year, ten S.Auf.U. samplers were placed at different stations being used in the field trials. The data arising from this study are presented in Annexe 4 for the following rivers: River Tean 1 (Table 55), River Weaver 1 (Table 32), River Nene 1 (Table 81), South Drain (Table 88) and River Severn 1 (Table 78). On one occasion, 20 S.Auf.U. were used in a similar test in the River Severn 1, the results of which are presented in Annexe 4 (Table 91). After the standard 4-week immersion period, the number of taxa found on each sampler were determined. By using the mean of ten randomly selected sequences of the samples, the cumulative number of taxa were plotted against the number of samples. These data are presented graphically in Figures 4.62 and 4.63. Using the same ten randomly selected sequences, the mean cumulated B.M.W.P. Scores were also plotted against number of samplers and these are presented in Figures 4.64 and 4.65. From these graphs, the numbers of taxa and the respective B.M.W.P. Scores derived from three

S.Auf.U. compared with those derived from 10 or 20 S.Auf.U., were expressed as a percentage and these are given in Table 4.22.

Table 4.22 Percentage number of taxa and B.M.W.P. Score represented by 3 S.Auf.U.

RIVER	NUMBER OF SAMPLERS	NUMBER OF TAXA % represented by 3 S.Auf.U.	B.M.W.P. SCORE % represented by 3 S.Auf.U.
Tean 1	10	78.0	77.7
Weaver 1	10	78.3	84.6
Nene 1	10	67.3	62.9
South Drain	10	88.3	82.6
Severn 1 (1977)	10	72.1	75.9
Severn 1 (1978)	20	66.4	52.8

The optimum number of samples required for a species list will depend on the diversity and dispersion of the fauna and a host of other variables including time, season, depth and invertebrate drift. The data used in plotting Figures 4.62 and 4.63 show that in most instances over half the species present were collected in the first sample, but species were still being added by the tenth, and in one case the twentieth sample as the curves approached asymptotic levels. Such a situation was also seen when processing the data by the B.M.W.P. Score system (Figures 4.64 and 4.65). The number of taxa represented by 3 S.Auf.U. when derived from 10 was found to vary between 67 and 88% for the five rivers and for the B.M.W.P. Score between 63 and 85%. From this exercise it would appear that triplicate

FIG.4.62 MEAN CUMULATIVE NUMBER OF TAXA TAKEN BY DIFFERENT NUMBERS OF S.AUF.U. AT DIFFERENT STATIONS.

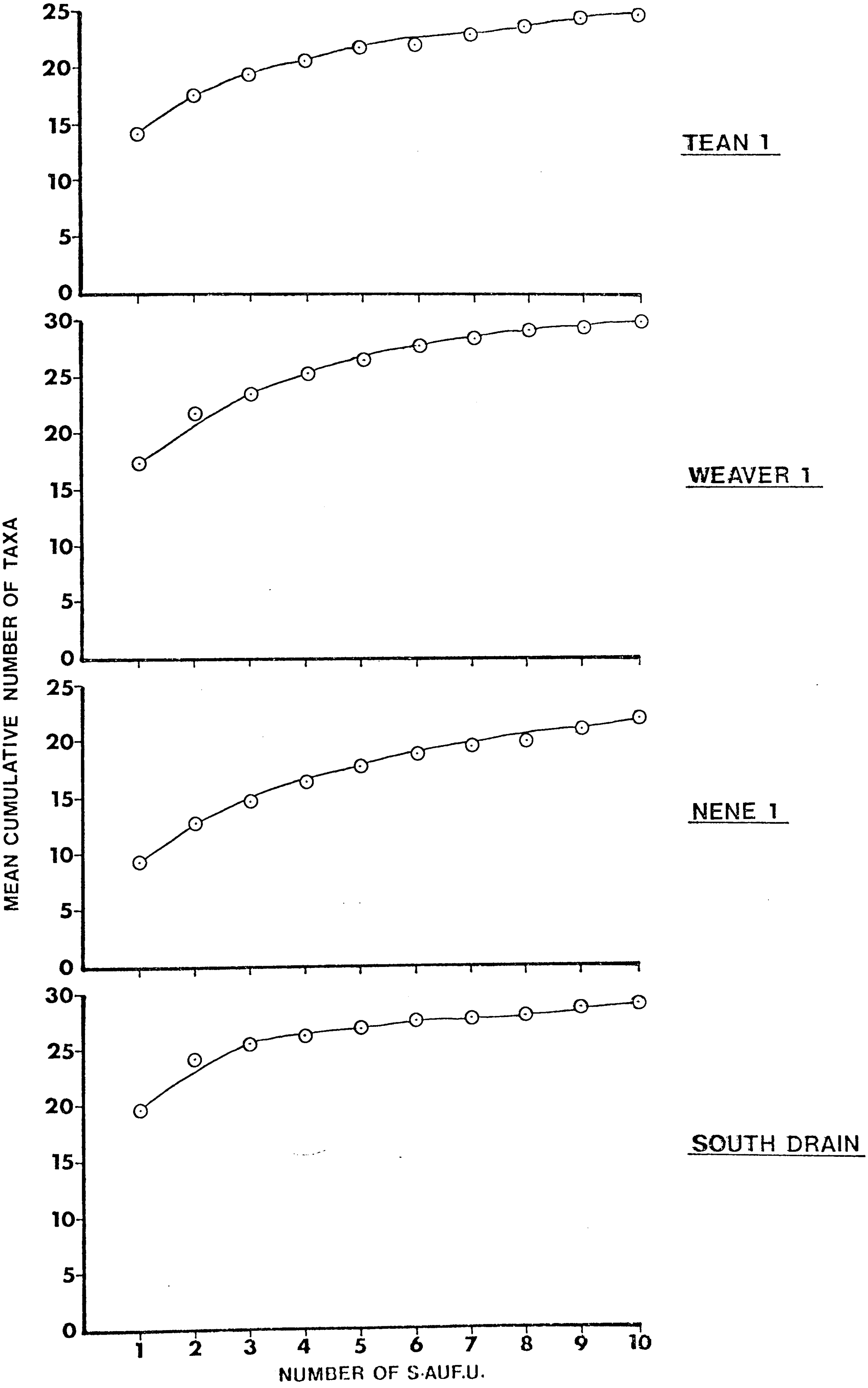


FIG.4.63 MEAN CUMULATIVE NUMBER OF TAXA TAKEN BY DIFFERENT NUMBERS OF S.AUF.U. IN THE RIVER SEVERN(STATION 1).

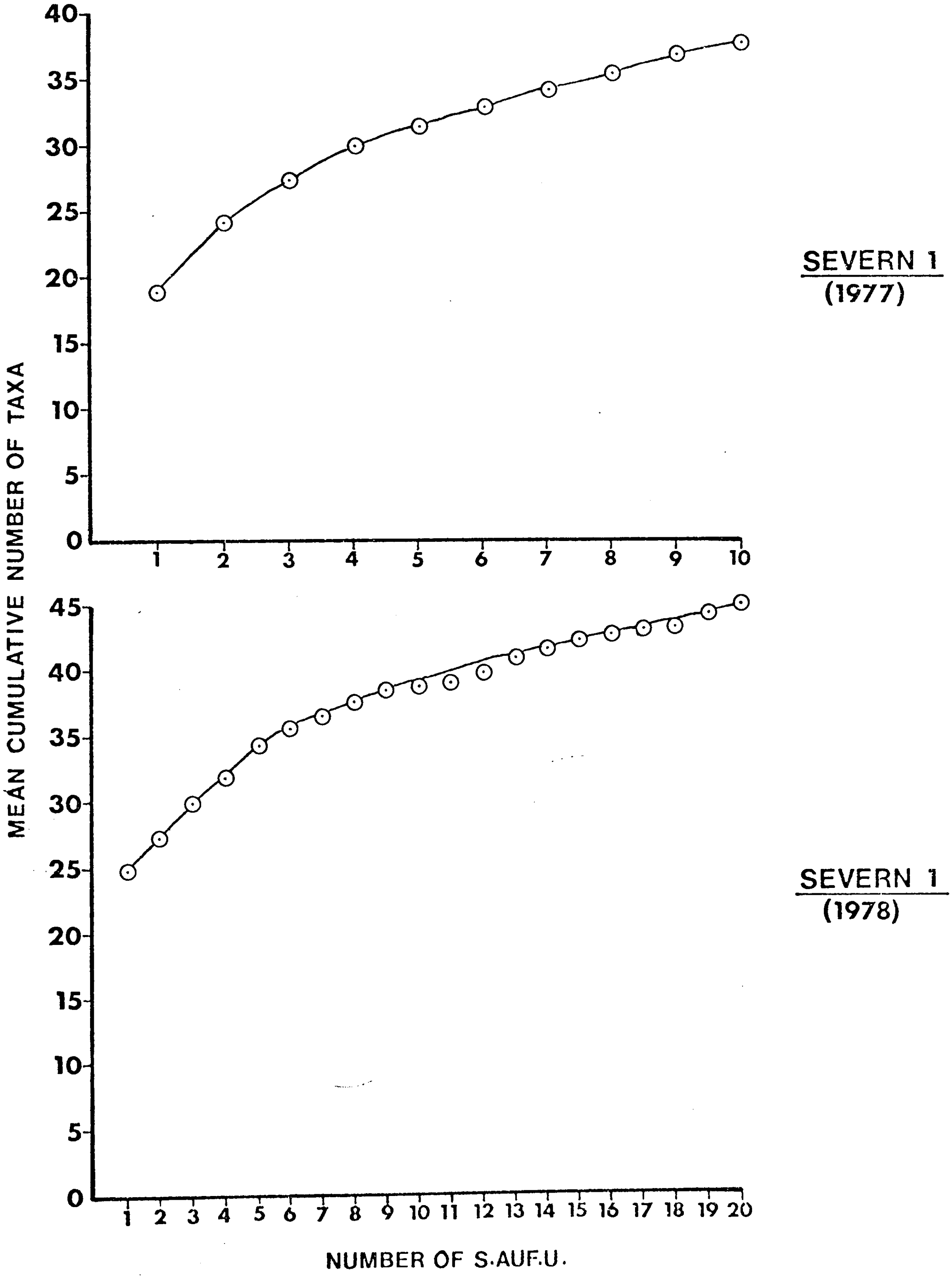


FIG.4.64 MEAN CUMULATIVE B.M.W.P. SCORE DERIVED FROM DIFFERENT NUMBERS OF S.AUF.U. AT DIFFERENT STATIONS.

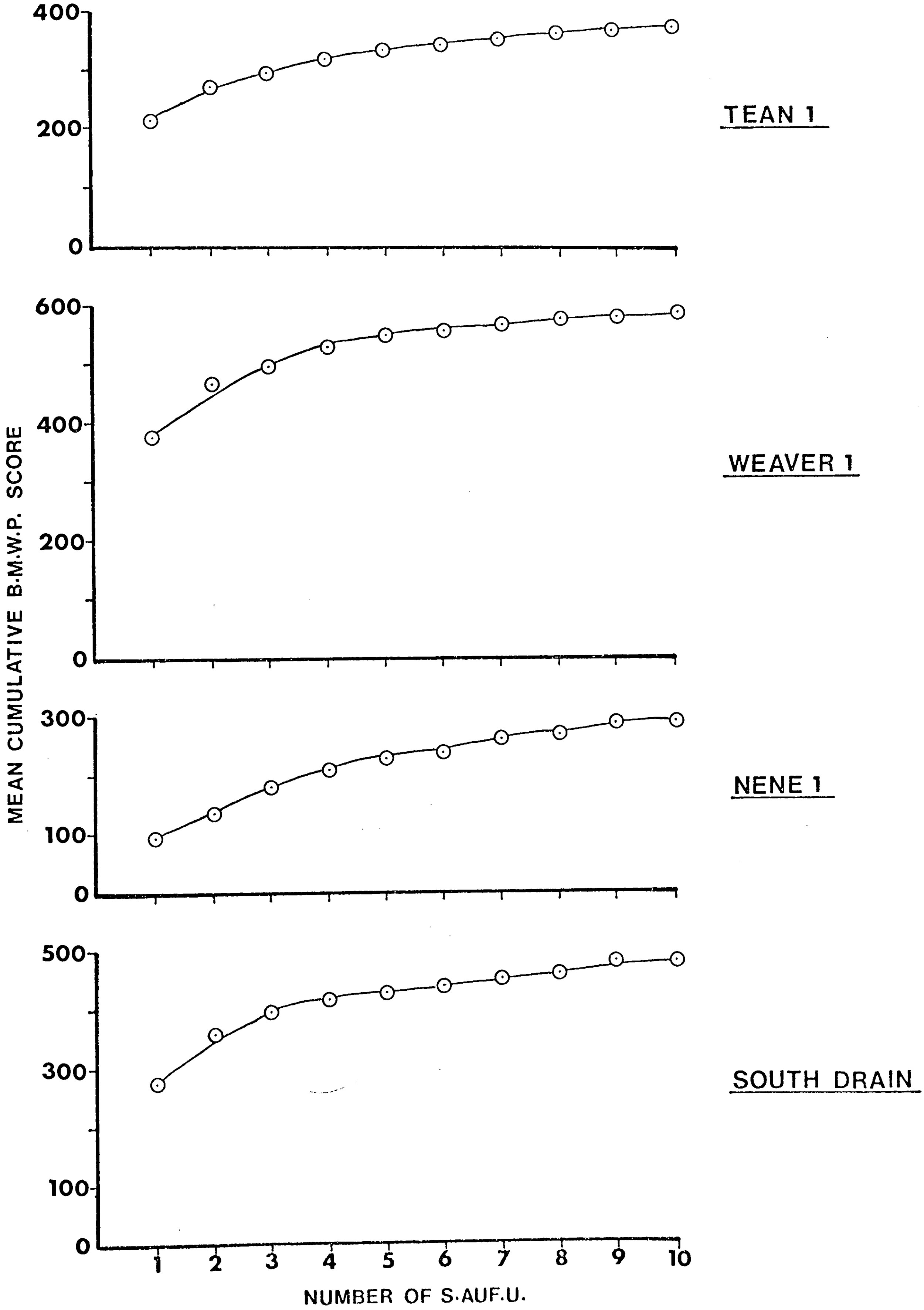
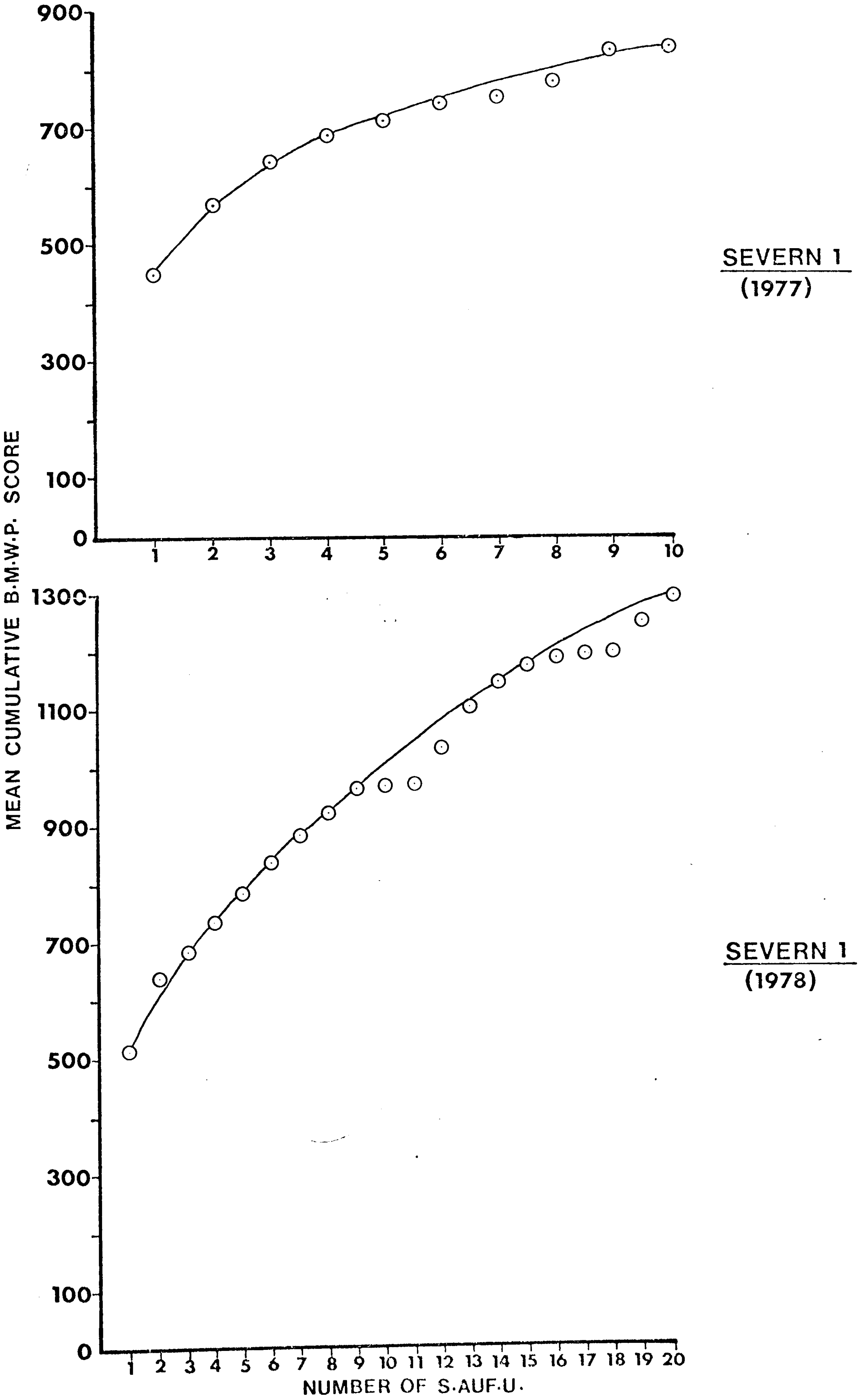


FIG.4.65 MEAN CUMULATIVE B.M.W.P. SCORE DERIVED FROM DIFFERENT NUMBERS OF S.AUF.U. IN THE RIVER SEVERN(STATION 1).



samples were collecting a large proportion of the taxa needed to provide a reasonably accurate measure of water quality. Larger sample numbers would only probably collect the rarer, widely dispersed species.

Another closely related measure of the reliability of a sampling method is to determine the number of replicates needed to be within a defined level of accuracy. Sample size can be calculated for a specified degree of accuracy knowing that the higher the level of accuracy required the greater the number of samples need to be taken. One can determine such a requirement using the following formula quoted by Hellawell (1978) from Southwood (1966):

$$n \approx \left(\frac{tS}{D\bar{x}}\right)^2$$

When n = number of sampling units

t = Student's t value for the required probability (95% confidence level gives a t value of approximately 2)

S = standard deviation

D = level of relative error required (i.e. 10% = 0.1; 20% = 0.2)

$\bar{x}$ = arithmetic mean

Using the number of taxa derived from sample replicates from the previous five rivers, the following statistical results were derived, presented in Table 4.23.

Table 4.23 Statistical results from replication trials on S.Auf.U.

River	No. of Samplers	Taxa Mean	Std. Dev.	Std. Error	95% C.L.	Coeff. Var.(%)	NO.ofSAMPLES 10% Error	20% Error
Tean 1	10	12.7	2.11	0.67	1.31	16.6	11.0	2.8
Weaver 1	10	16.5	3.37	1.07	2.09	20.4	16.7	4.2
Nene 1	10	9.1	2.02	0.64	1.25	22.2	19.7	4.9
South Drain	10	18.0	2.71	0.86	1.68	15.1	9.0	2.3
Severn 1 (1977)	10	19.1	3.21	1.02	1.99	16.8	11.3	2.8
Severn 1 (1978)	20	22.0	3.26	0.73	1.42	14.8	8.78	2.2

By accepting a standard error equal to 20% of the mean (a reasonable error in most bottom samples, Elliott (1971)), it was found that on average three replicates would provide an estimate of the true mean at the 95% confidence limits. This agrees with the majority of data presented in Table 2.1 and also by the cumulative number of taxa method (Table 4.22). In the overall study described in Chapter 4, three sample replicates were only taken due to the number of sampling stations and rivers examined. With a reduced sampling work-load one could possibly increase the number of replicates taken at each station if a greater level of accuracy is required. However, in most biological surveillance studies related to water quality monitoring, such a requirement is not always necessary, especially when one is considering Water Authority biological surveillance, or industrial surveillance programmes where time is of the essence.

4.10 Colonisation of S.Auf.U. by macroinvertebrates

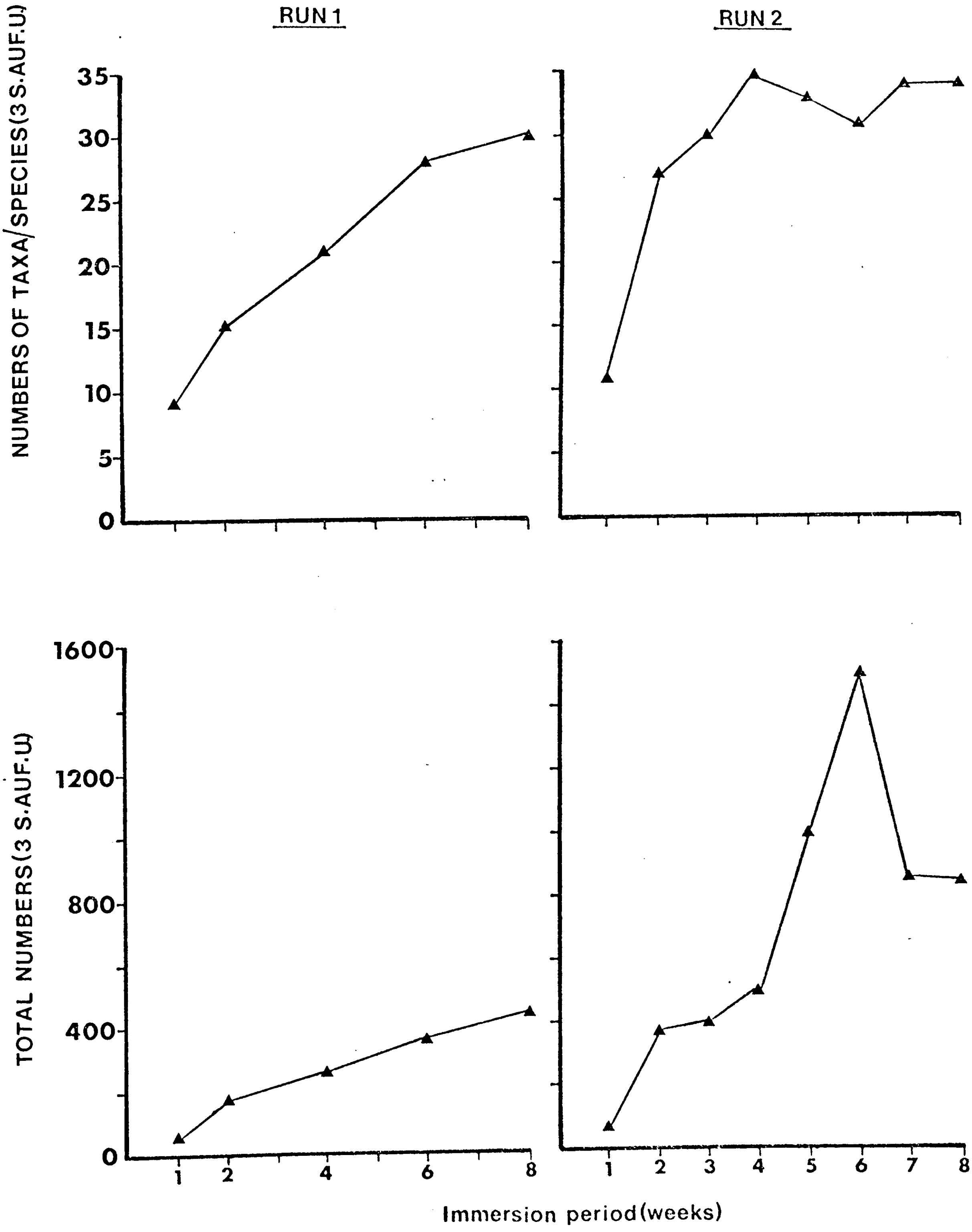
4.101 Sequential studies on colonisation by S.Auf.U.

The dynamics of macroinvertebrate colonisation of artificial substrata in streams has formed the basis of many studies concerning the proficiency of sampler performance. The purpose of this study was primarily an attempt to fit colonisation of S.Auf.U. to the equilibrium model proposed by MacArthur and Wilson (1963) for island faunas and secondly an attempt to determine trends of colonisation for most abundant aquatic invertebrates.

The sequential colonisation of S.Auf.U. was studied in the River Severn pool at Bewdley (Severn 1). Two consecutive runs were carried out between April and June, 1978, when flows in the river were relatively low. In the first run, 15 units were placed in the river and 3 were removed after 1, 2, 4, 6 and 8-week immersion periods. In the second run, 24 units were used, 3 being removed at weekly immersion intervals for 8 weeks. The data for these surveys are presented in Annexe 4 (Tables 92 - 93). After the respective periods of immersion the numbers of taxa and numbers of individuals in each taxa were determined. The total numbers of taxa and individuals found in the 3 S.Auf.U. after different immersion periods are shown in Figure 4.66. In the first run, the numbers of taxa increased throughout the 56-day period, while in the second run, the maximum number of taxa had appeared after 28 days immersion. With the total number of individuals an exponential growth rate was seen in the first run, while in the second run, maximum numbers were found after 42 days.

The total number of species found at each sampling date, the number of 'new species', the number of recurring species, and the number of

FIG.4.66 SEQUENTIAL COLONISATION OF TAXA AND INDIVIDUALS ON S.AUF.U. IN RIVER SEVERN , STATION 1.



species eliminated are shown in Tables 4.24 and 4.25 respectively, for the two runs. 'New species' were those that had never been recorded before during the run. Recurring species were considered to be those that were previously eliminated and subsequently became re-established. The number of species eliminated was found by adding the number of 'new species', plus those recurring, to the total number present at the preceding sampling date and then subtracting the current total number from this figure. For example, at day 28 (Table 4.25), there were 35 species present, of which 8 were 'new' and 0 were recurring. If all of the species present at day 21 had remained on the S.Auf.U. until day 28, then there should have been 38 species at day 28. Therefore, 3 species were eliminated between the two dates.

Colonisation rates and extinction rates were calculated from the data presented in Tables 4.24 and 4.25. The number of 'new' species plus the number of recurring species divided by the time in days between sampling periods equals the colonisation rate expressed in species per day. Extinction rate was determined by dividing the number of species eliminated by the days between sampling periods. This also was expressed in species per day.

The relationship between period in days and the colonisation rate and extinction rate are presented in Figure 4.67 for runs 1 and 2 respectively. The 'cross-over' points of colonisation and extinction according to MacArthur and Wilson (1963) represent the stabilisation level.

Table 4.24 Total number of macroinvertebrate species present, and parameters required for calculating colonisation and extinction rates (Run 1)

Day	Total no. of species	'New' species	Recurring species	Species eliminated	Colonisation rate	Extinction rate
7	9	9	0	0	1.28	0
14	15	7	0	1	1.00	0.14
28	21	10	1	5	0.78	0.36
42	28	13	1	7	1.00	0.5
56	30	3	4	5	0.5	0.36

Table 4.25 Total number of macroinvertebrate species present, and parameters required for calculating colonisation and extinction rates (Run 2)

Day	Total no. of species	'New' species	Recurring species	Species eliminated	Colonisation rate	Extinction rate
7	11	11	0	0	1.57	0
14	27	16	0	0	2.29	0
21	30	6	0	3	0.86	0.43
28	35	8	0	3	1.14	0.43
35	33	3	2	7	0.71	1.0
42	31	4	3	9	1.0	1.28
49	34	2	4	3	0.86	0.43
56	34	1	4	5	0.71	0.71

The pattern of colonisation of S.Auf.U. from the first run indicated an established community after approximately 49 days, with the maximum number of species still not reached by sampling day 56 (Figure 4.66). In the second run, where a larger number of immersion periods were studied, a stabilised community was found after approximately 43 days, with the maximum number of species being found on sampling day 28 (Figure 4.66). Assuming MacArthur and Wilson's model relates to colonisation and extinction rate phenomena, then the data seems to indicate that only a temporary stable community existed for the macroinvertebrates on the colonisation substrata. This view being derived mainly from the wide scatter of data shown in Figure 4.67 (Run 2). It is probable that this instability is a result of species drifting downstream from the nearby riffle, interfering with the pool community, colonising S.Auf.U..

Dickson and Cairns (1972) used this model to assess the suitability of '3M Corporation webbing' for collecting quantitative data on freshwater invertebrates in riffles. Their data fitted the model reasonably well, but, possibly owing to a lack of diversity of habitats on the substrata, community interactions did not produce permanent stability. This was also the conclusion reached by Stauffer _et al._ (1976) and Williams and Hynes (1977).

Experience in Phase 1 of these investigations (Chapter 3) suggested an immersion period of 4 weeks based on the effectiveness and practicality of the sampler when used in the field. The longer the period of immersion, the greater the chance of it becoming fouled with drifting debris, silted up, or lost by being washed away or vandalised.

FIG.4.67 COLONISATION AND EXTINCTION CURVES OF MACROINVERTEBRATES FROM S.AUF.U. IN THE RIVER SEVERN , STATION 1.

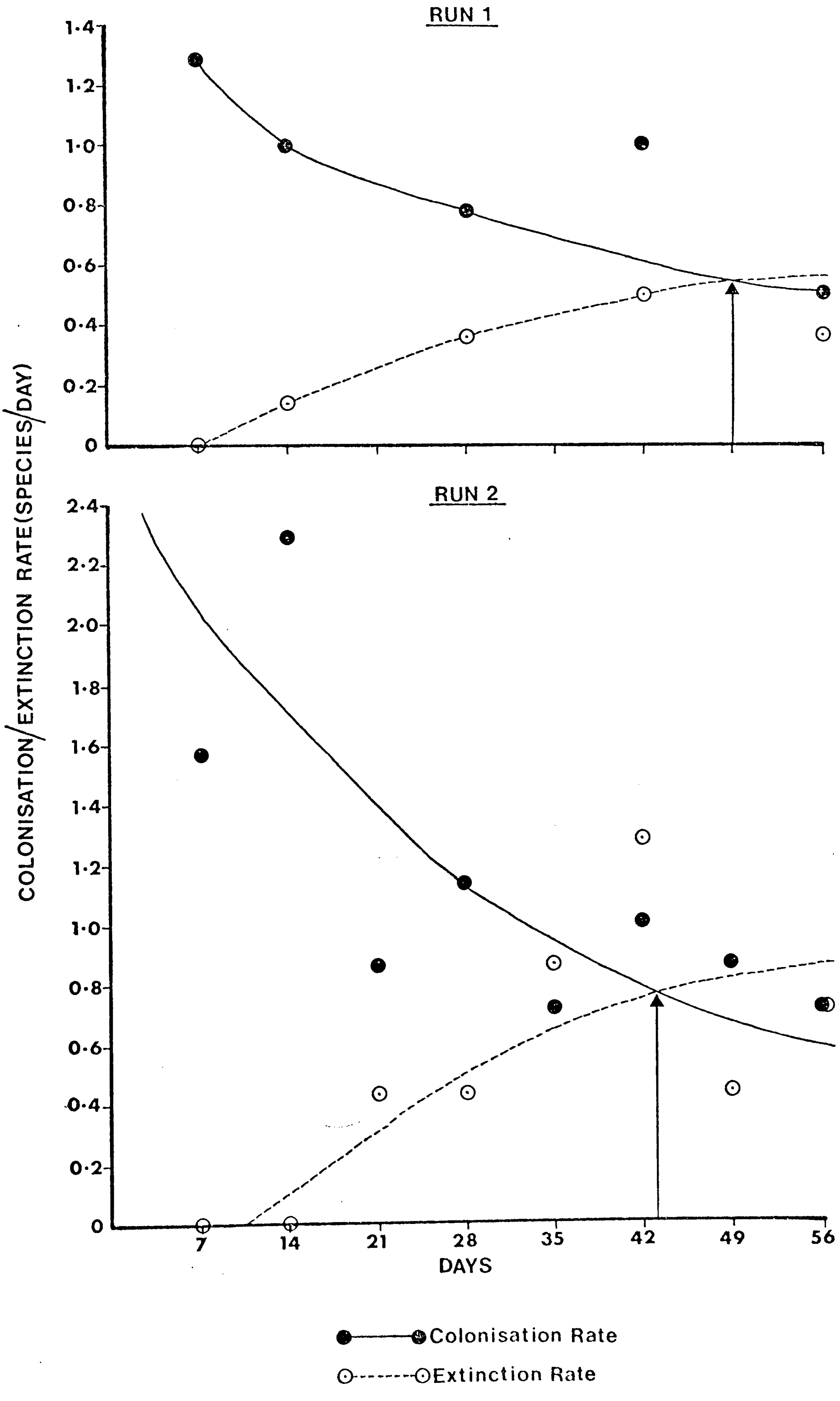


FIG.4.68 SEQUENTIAL STUDY OF MAJOR TAXA COLONISING S.AUF.U. IN THE RIVER SEVERN , STATION 1.

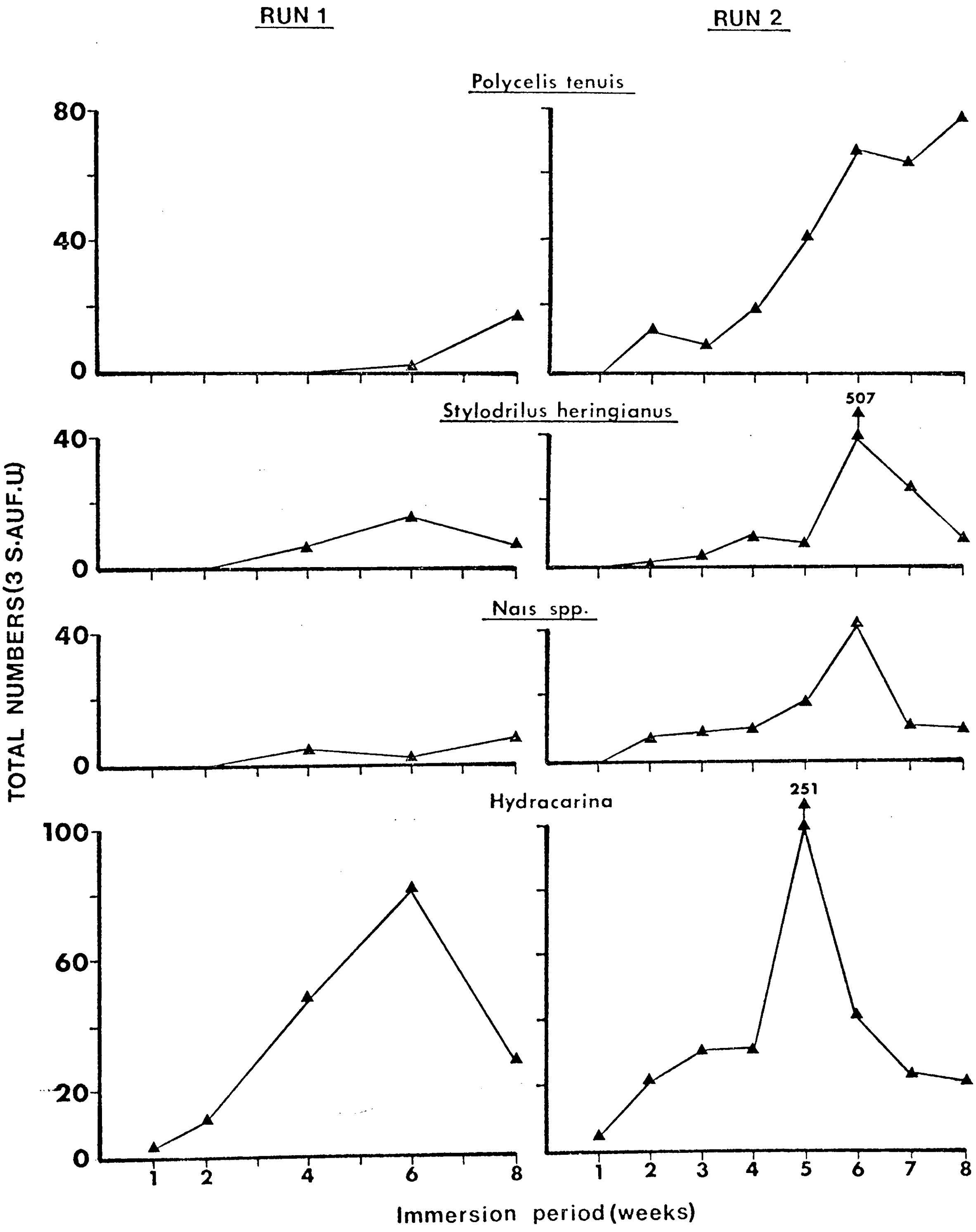


FIG. 4.69 SEQUENTIAL STUDY OF MAJOR TAXA COLONISING S.AUF.U. IN THE RIVER SEVERN , STATION 1.

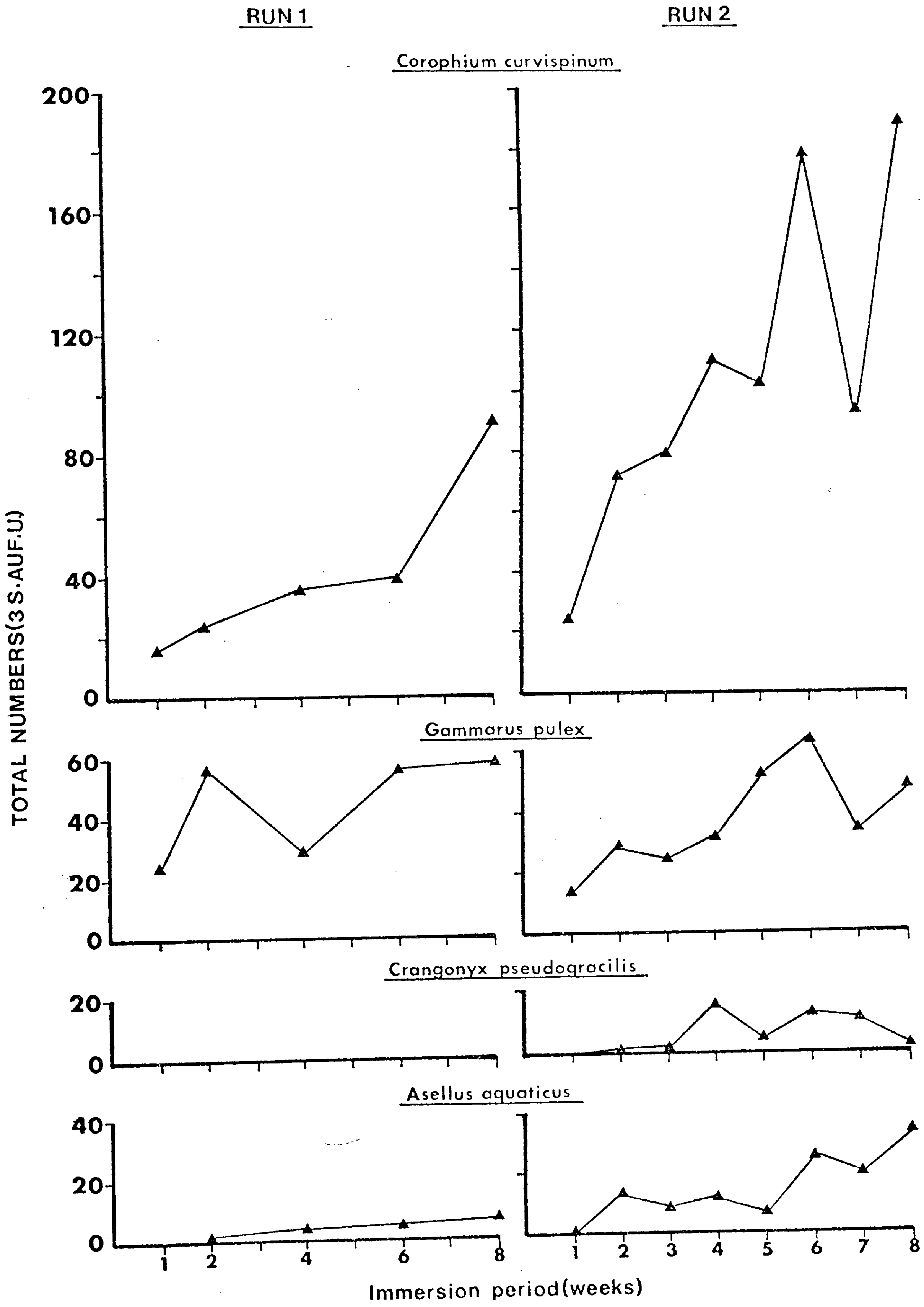


FIG. 4.70 SEQUENTIAL STUDY OF MAJOR TAXA COLONISING S.AUF.U. IN THE RIVER SEVERN , STATION 1.

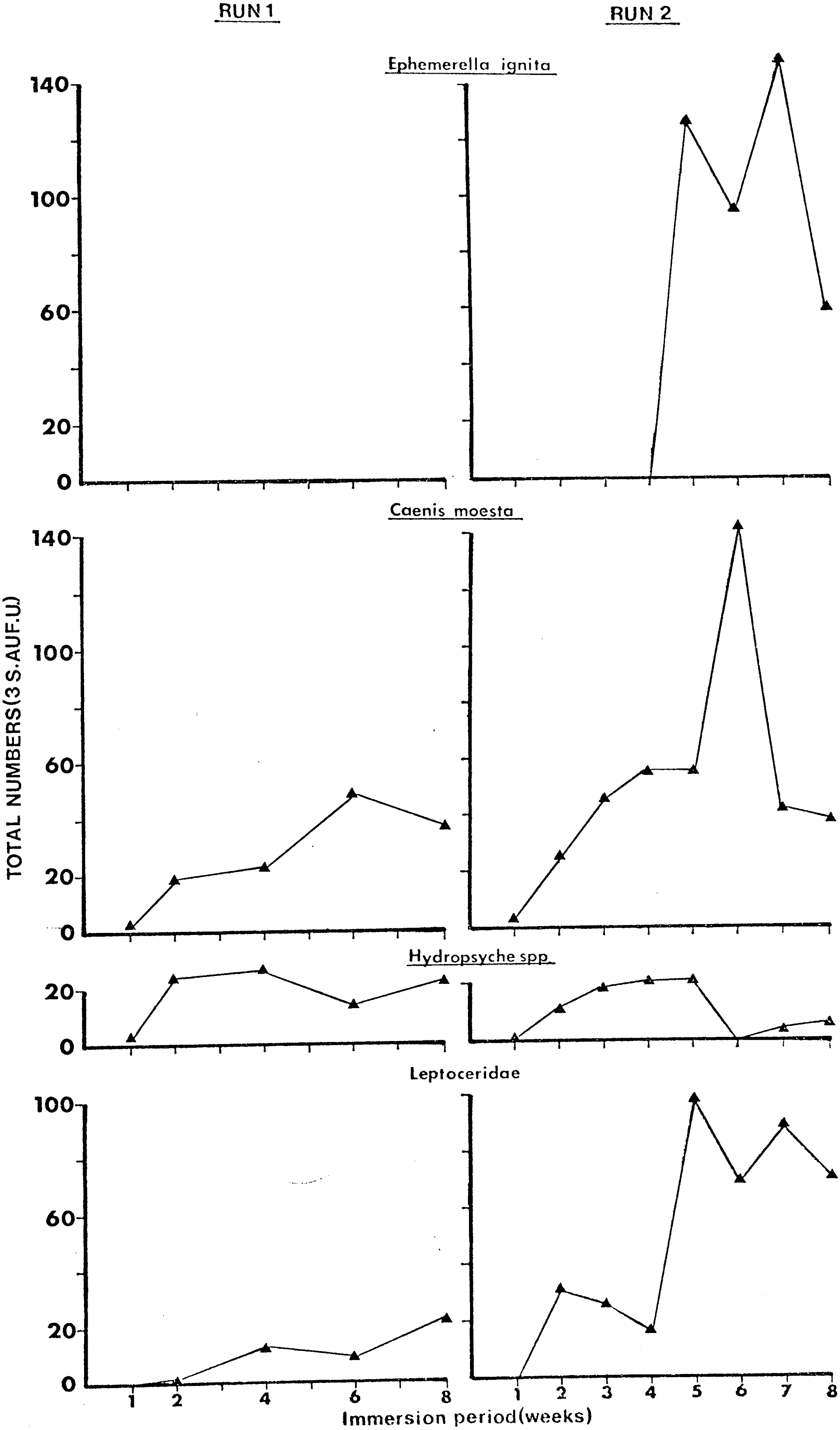
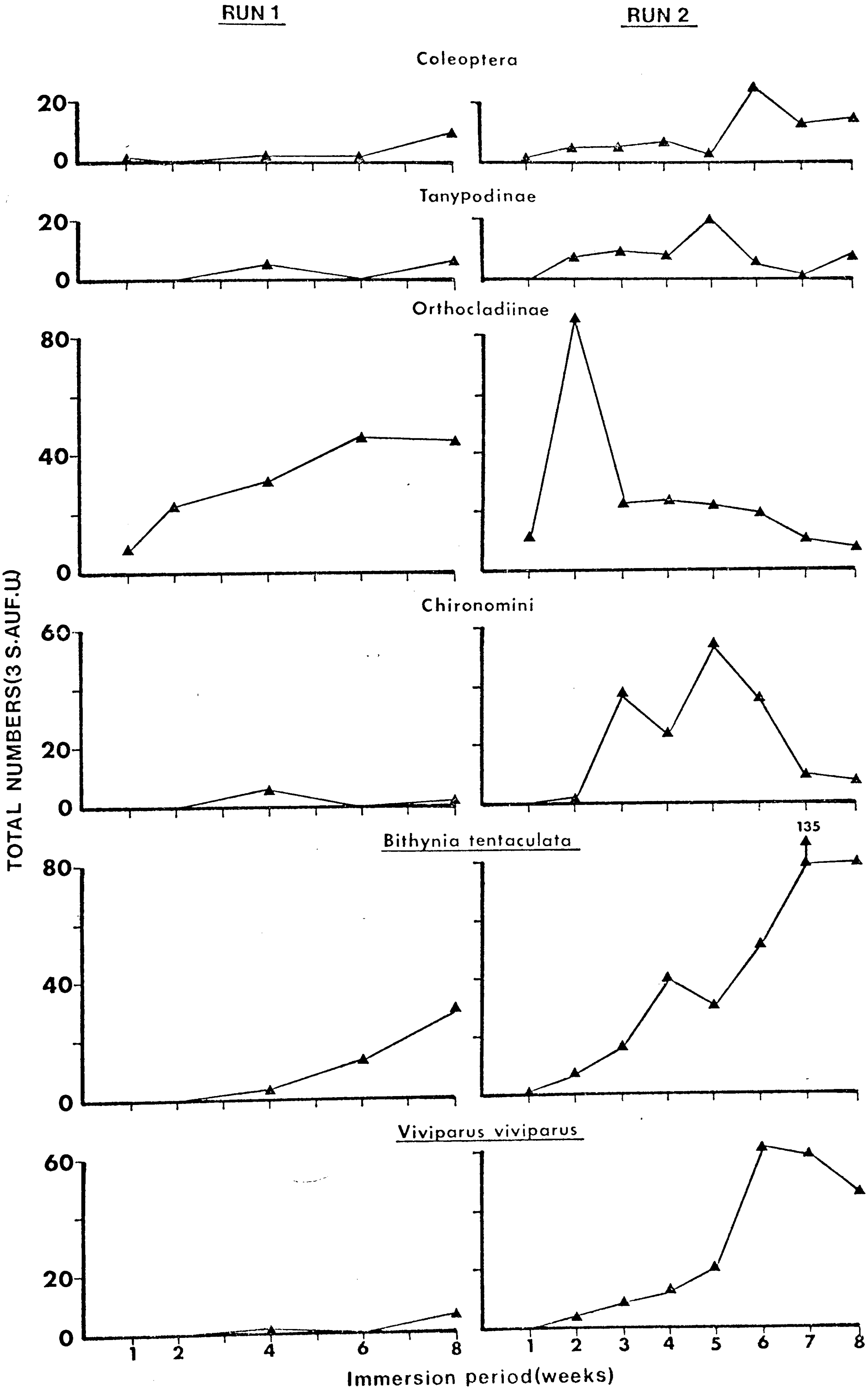


FIG.4.71 SEQUENTIAL STUDY OF MAJOR TAXA COLONISING S.AUF.U. IN THE RIVER SEVERN , STATION 1.



Although this study was conducted downstream of a large riffle, it is suggested that a 'more stable' community might become established in the potamon region of lowland rivers which do not usually become contaminated with riffle species, except possibly via tributaries.

The numbers of individuals of different species colonising S.Auf.U. are presented in Figures 4.68 - 4.71. The total number of organisms on S.Auf.U. fluctuated during the colonisation period with maximum numbers usually occurring around the 42-day mark. A gradual colonisation occurred with most species up to the maximum. The majority of species colonising the S.Auf.U. were pool species and their numbers probably increased in the S.Auf.U. as a result of increased silt and detrital matter accumulating within the samplers. Initial colonisation within the first couple of weeks was probably due to species using the units for shelter and protection. Generally the changing substratum conditions were accompanied by a successive exchange of species demonstrating their capability of active habitat selection.

4.102 Colonisation of S.Auf.U. in relation to proximity of upstream riffles

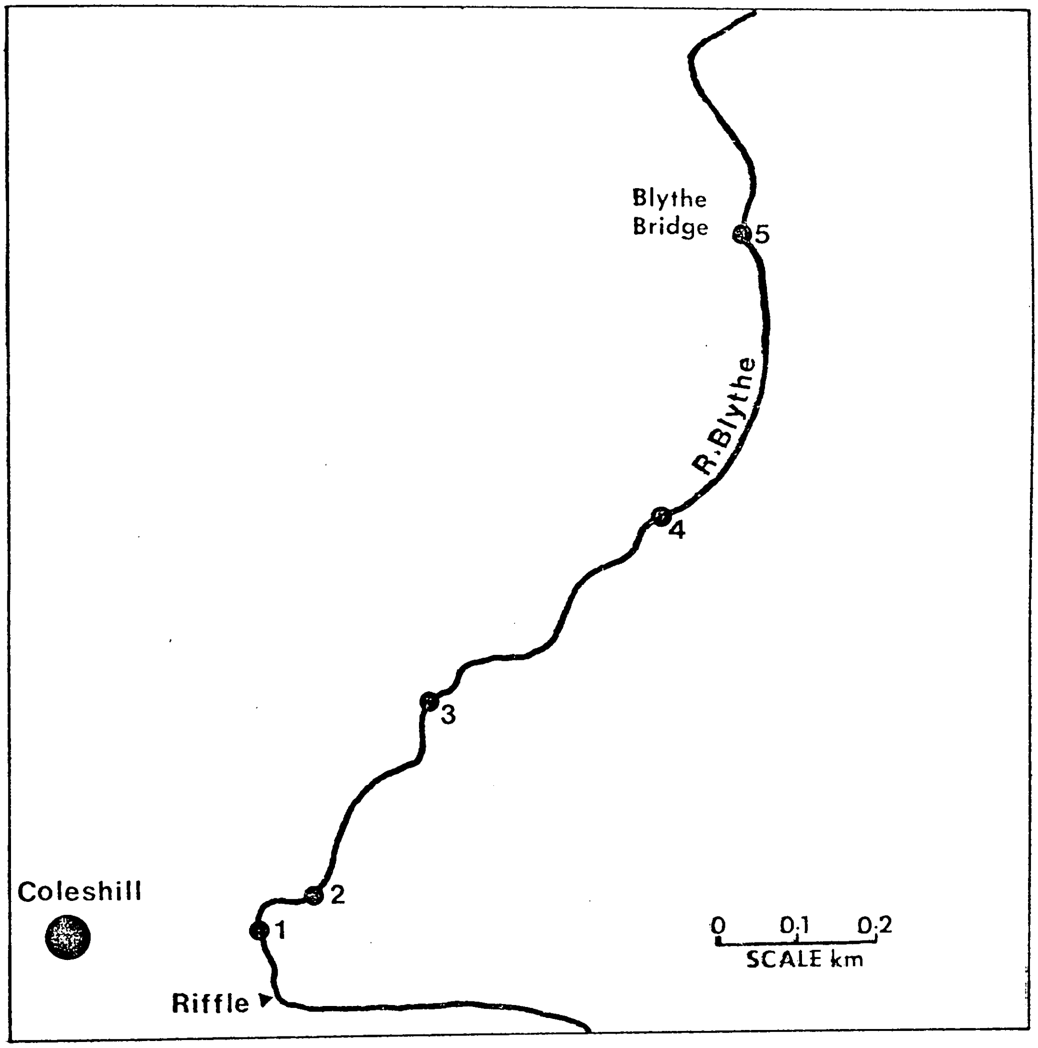
Several authors have studied the mechanisms of colonisation and the relationships between invertebrate drift and colonisation of new substrata (usually trays of natural substrata) in running water habitats (Waters, 1964; Townsend and Hildrew, 1976; Williams and Hynes, 1976). Re-colonisation studies on running water benthos have shown that animals quickly re-appear in the affected areas (Waters, 1964). In Townsend and Hildrew's (1976) study on colonisation, they distinguished between movements of invertebrates brought about by crawling over artificial substrata, as opposed to drifting and evaluated the role that drift played in the colonisation of new areas of stream bed exposed both experimentally and naturally. They found, of movements involved in the colonisation of experimentally introduced substrata, 82% were by drifting.

Some of the results of the case studies reported in sections 4.7-4.8 suggested that the numbers of taxa colonising the S.Auf.U. compared with the number present in the natural depositing substratum at the station, were influenced by the proximity of upstream riffles. To investigate the influence of riffle species on the colonisation of S.Auf.U. in pools, a stretch of the River Blythe (W. Midlands) was selected at Coleshill where there was a riffle followed by a slow-flowing stretch of over 1.5 km. with no riffles.

At five stations downstream of the riffle, three S.Auf.U. were positioned at varying intervals as depicted in Figure 4.72. They were removed after four weeks immersion and the fauna of each determined. Ekman grab samples of the natural depositing substratum were also taken in triplicate for comparison. At the beginning and end of the S.Auf.U. immersion period, the upstream riffle was sampled by cylinder sampler, taking three samples across the river. The macro-invertebrate data arising from this study are presented in Annexe 4 (Tables 94 - 95).

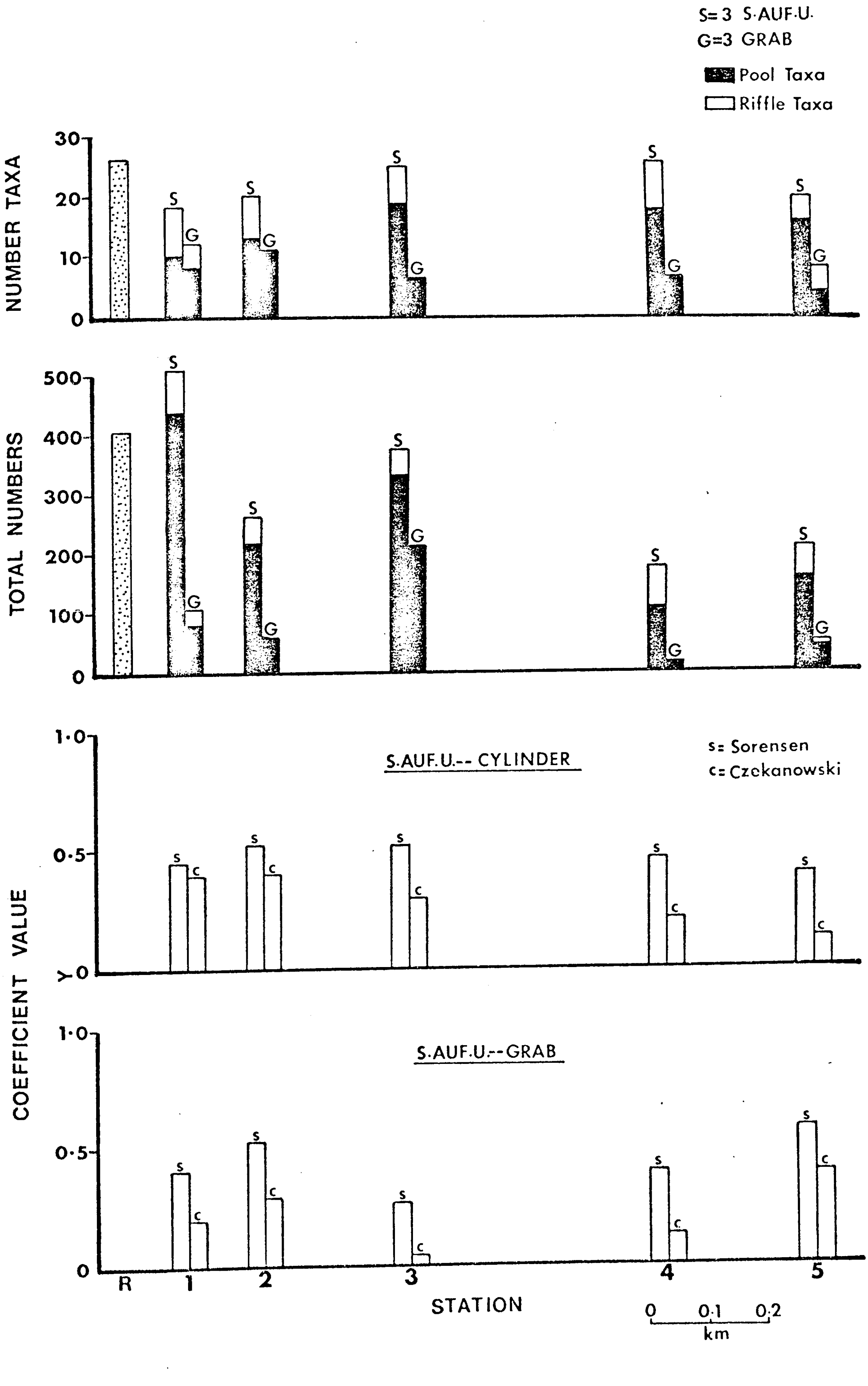
The numbers of taxa collected by S.Auf.U. and the total number of individuals varied considerably as one moved downstream from the riffle (Figure 4.73). The majority of taxa recorded by S.Auf.U. were typical of a depositing community, although *Gammarus pulex* normally associated with riffle regions was found on the S.Auf.U. at all the pool stations. Similarity calculations between S.Auf.U. and the natural community of the upstream riffle showed a gradual

FIG. 4.72 SKETCH MAP SHOWING POSITIONS OF S.AUF.U. IN RELATION TO RIFFLES IN THE RIVER BLYTHE.



SITE	DISTANCE DOWNSTREAM FROM RIFFLE(km)
1	0.06
2	0.19
3	0.44
4	0.88
5	1.13

FIG.4.73 NUMBERS OF TAXA, INDIVIDUALS AND SIMILARITY COEFFICIENT VALUES IN RELATION TO PROXIMITY OF UPSTREAM RIFFLE, RIVER BLYTHE.



decline as one moved downstream away from the riffle. Similar calculations between S.Auf.U. and grab sampling of the natural depositing community produced low similarity values since S.Auf.U. collected larger numbers of taxa than by grab sampling.

The relative abundance of the major taxa are shown in Figures 4.74 - 4.75. Some species which were common in the riffle such as Caenis moesta and Hydropsyche angustipennis showed a marked decline in numbers found on the S.Auf.U. with distance below the riffle. Both species persisted in the natural substratum for shorter distances below the riffle. The Tanytarsini on the S.Auf.U. showed a similar type of distribution but persisted also in the natural substratum. They are typical depositing substratum species which makes their distribution on the S.Auf.U. in relation to the riffle difficult to explain. Nais elinguis also showed a distribution on the S.Auf.U. in relation to distance below the riffle but was not recorded from the riffle itself. This may have been due to the mesh size of the cylinder sample used in riffle sampling. Other taxa such as the Hydracarina, Orthocladiinae and G.pulex, although more associated with the S.Auf.U. than with the natural depositing substratum, showed no relationship with distance below the riffle. The Tubificidae showed as a close an association with the natural substratum and therefore no relationship with distance below the riffle.

The effect of proximity of the upstream riffle on the colonisation of the S.Auf.U. affected the B.M.W.P. Score as shown in Figure 4.76. The S.Auf.U. score was nevertheless much higher than the score derived from grab sample data from the natural substratum at the same station.

FIG. 4.74 DISTRIBUTION OF TAXA ON S.AUF.U. IN NATURAL DEPOSITING SUBSTRATUM IN RELATION TO PROXIMITY OF UPSTREAM RIFFLE, RIVER BLYTHE.

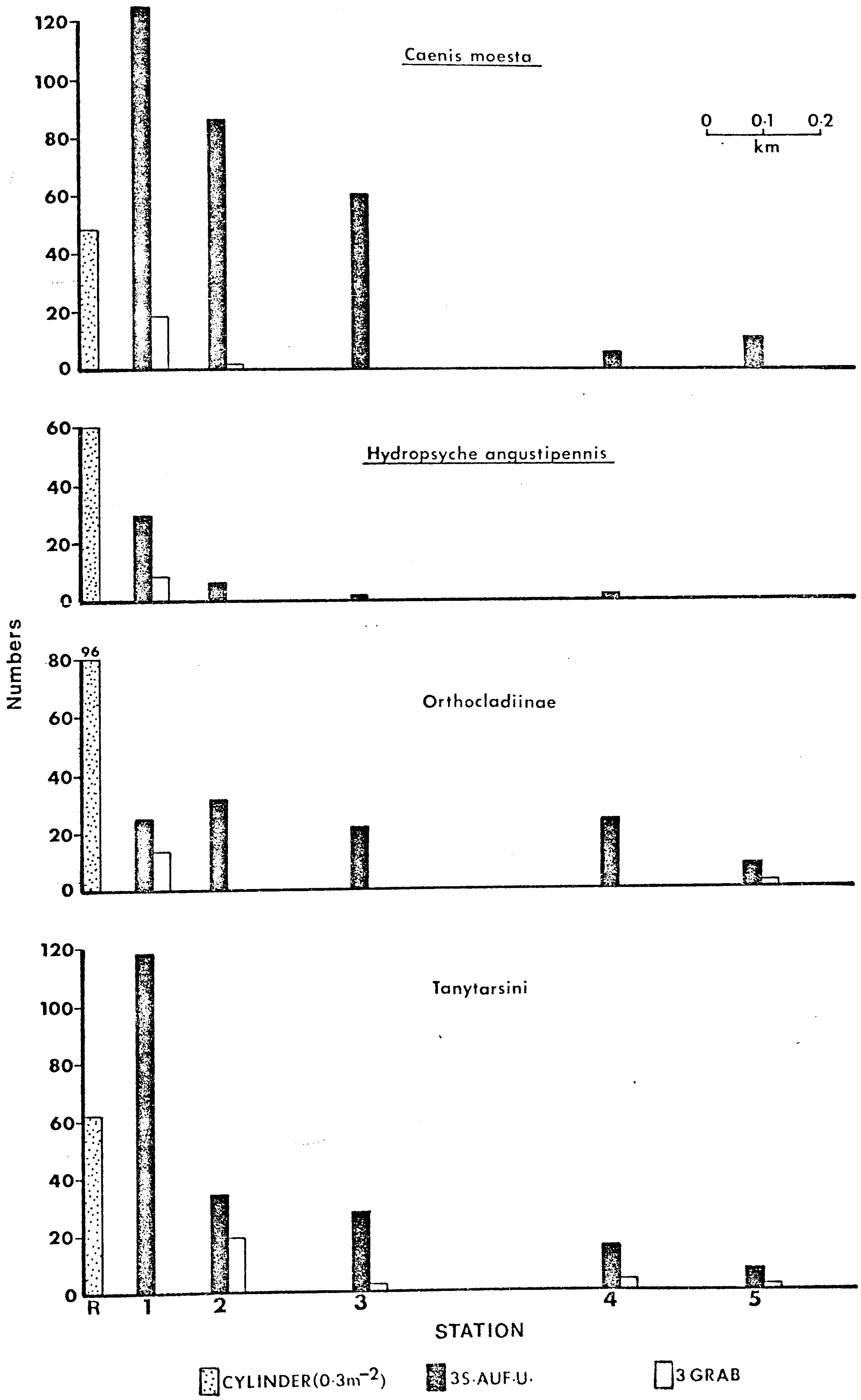


FIG.4.75 DISTRIBUTION OF TAXA ON S.AUF.U. IN NATURAL DEPOSITING SUBSTRATUM IN RELATION TO PROXIMITY OF UPSTREAM RIFFLE, RIVER BLYTHE.

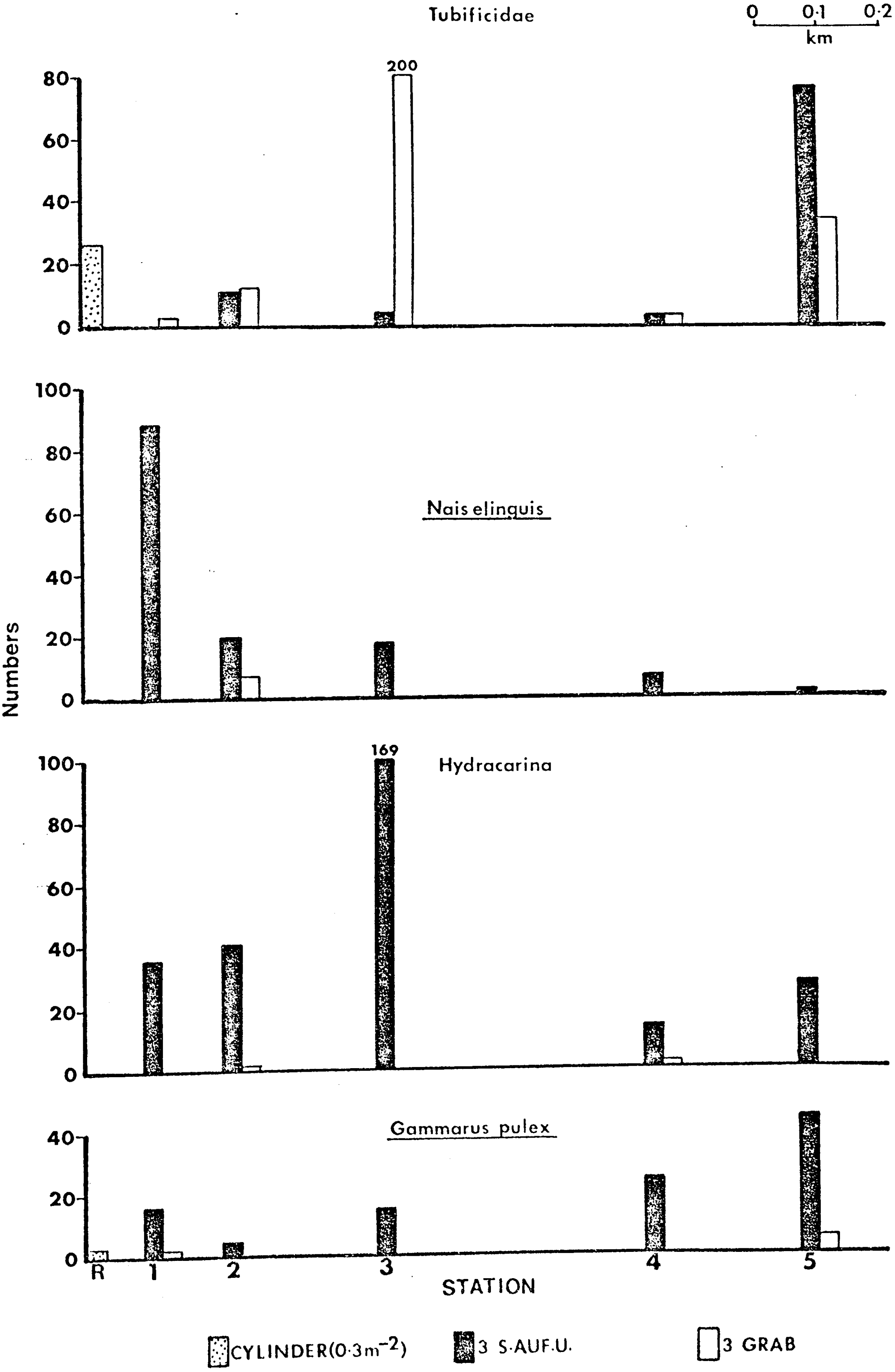
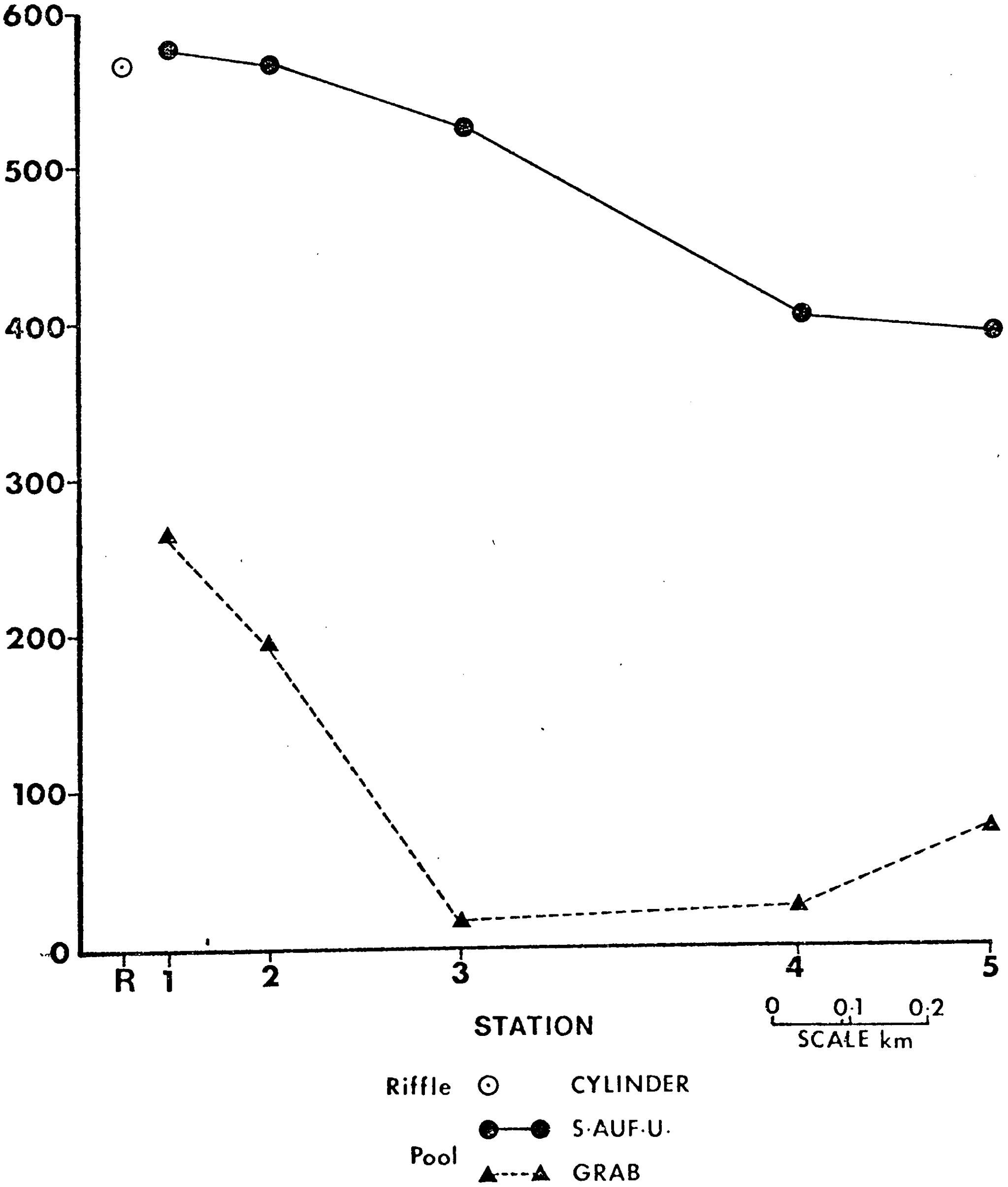


FIG.4.76 B.M.W.P. SCORES DERIVED FROM DATA OF S.AUF.U. PLACED AT DIFFERENT DISTANCES DOWNSTREAM OF RIFFLE AND FROM NATURAL DEPOSITING SUBSTRATUM SAMPLES, RIVER BLYTHE.



The effect of colonisation of S.Auf.U. in pools from upstream riffles will primarily be determined by the general flow of the river. In the River Blythe although the riffle was relatively slow-flowing (20 cm. sec^{-1}) and the pool reach with negligible flow (< 3 cm. sec^{-1}) the results indicate that even in the slowest sections of a slowly-flowing river, there exists a great potential for colonisation and for benthic re-distribution in general. The colonisation of S.Auf.U. was therefore by invertebrate drift and by random movements across the natural bottom substratum. On the basis of this investigation, riffle influence declines rapidly in the upper zone of the depositing reach.

4.11 General Discussion

A major problem in developing a standard method for the biological monitoring of river water quality is the natural differences which occur in communities of different river types independent of water quality. A further problem is that of sampling benthic communities in deep waters, especially when associated with different substrata. This research project investigated the suitability of different sampling methods for supplying biological data needed for surveillance purposes, involving different river types. Special attention was therefore paid to the possible use of colonisation samplers. Experience of a large number of published investigations has shown that colonisation samplers are an effective method of biological sampling for the evaluation of water pollution. It has been found that for purposes of evaluating water pollution, it is not essential that the samplers collect a community identical to the naturally occurring one, provided it is broadly representative, has low variability and is reproducible at different times and locations.

In this study, 765 S.Auf.U. were tested in a wide range of river types and water qualities, including extreme conditions, and the number recovered was in the region of 95%. Of the 5% lost, 3% were actively vandalised, while 2% were lost through high flows. The samplers proved highly durable and efficient at collecting benthic communities often when natural benthic sampling of riffles and pools was impossible.

The colonisation samplers generally captured the greatest number of animals and species compared to natural sampling techniques which could be regarded as a more efficient method. The types of community colonising S.Auf.U. were predominatnly depositing species representing most major groups. Many species used the sampler as a hard substratum for colonisation whilst others used the unit for probable protection and shelter. One factor which contributes to the variability among replicate bottom samples is the loss of organisms during collection and retrieval. The S.Auf.U. samplers on the whole lost very few organisms during retrieval, thus adding to the overall success of the sampler. The dominant organisms collected on the S.Auf.U. throughout this study are in general agreement with previously published work and these are presented in Table 4.26.

The suitability of the data generated by the different sampling methods tested, for biological surveillance purposes, were assessed by applying the score system recommended by the Biological Monitoring Working Party (B.M.W.P.) of the Department of the Environment and by other recognised methods. The B.M.W.P. system, although less taxonomically demanding and easier to calculate than previously developed Biotic Indexes, provided data which was comparable to these systems. For national river classification purposes, such a score system would seem adequate.

Table 4.26 Dominant macroinvertebrates collected on S.Auf.U. during Phase II in a wide range of river types and water qualities

Taxa	Occasions recorded (max.235)	Stations recorded (max.33)	Taxa	Occasions recorded (max.235)	Stations recorded (max.33)
A.aquaticus	210	32	*P.flavomaculatus*	45	13
E.octoculata	159	30	*S.lutaria*	43	13
Orthocladiinae	150	33	*C.moesta*	42	12
Chirononomini	148	31	Dytiscidae	38	19
G.pulex	148	27	*M.nigra*	38	11
Tanypodinae	109	24	*Pisidium* spp.	37	16
L.peregra	108	27	*C.puella*	36	17
G.complanata	106	30	*D.lugubris*	34	19
B.tentaculata	104	20	*A.splendens*	34	11
Hydracarina	98	21	*B.rhodani*	30	13
T.tubifex	91	20	*V.macrostoma*	29	11
P. jenkinsi	75	20	*P.geometra*	28	11
H.stagnalis	73	20	*A.fluviatilis*	28	11
S.corneum	67	21	*Nais* spp.	27	15
L.hoffmeisteri	63	19	*P.barbatus*	26	9
Haliplus spp.	62	22	*P.pennipes*	25	13
P.maculatus	59	18	*L.variegatus*	24	10
Ceratopogonidae	58	23	*G.heteroclita*	23	13
P.tenuis	58	22	*P.grandis*	23	10
D.lacteum	56	17	*P.carinatus*	23	9
H.angustipennis	53	17	*V.viviparus*	23	5
P.fontinalis	51	19	*C.curvispinum*	23	3
N.elinguis	49	21	*H. marginata*	22	8
Tanytarsini	47	14	*T.fluviatilis*	22	5

4.111 Comparisons of the performance in monitoring water quality of S.Auf.U. in pools and the natural community of related riffles

Data from related pools and riffles at 15 stations from a range of river types and water qualities were processed to produce B.M.W.P. Scores. The relationship between the scores derived from the S.Auf.U. data from pools and those derived from the corresponding riffle sample data by cylinder sampling are shown in Figures 4.77 - 4.78. In one figure (4.77) the S.Auf.U. data is processed using the depositing substratum score, while in Figure 4.78 the eroding substratum score is used. In both cases there is a positive correlation ($P < 0.001$) between the score derived from riffle data and S.Auf.U. data from the pools. Both had a high correlation coefficient for such data ($r = 0.79$ and $r = 0.81$, respectively). The regression equation suggests a conversion factor for deriving riffle scores from S.Auf.U. derived ones. This, however, is only applicable in rhithron rivers where riffle species are available to colonise the S.Auf.U. by drift.

Similar regression equations were also applied to S.Auf.U. and cylinder data using the Chandler Score (Figure 4.79), Trent Biotic Index (Figure 4.80) and Diversity Index (Figure 4.80). The former two again showed high correlation coefficients ($r = 0.80$ and $r = 0.82$ respectively)while the latter Diversity Index showed lower values ($r = 0.71$). This was probably related to large numbers of particular taxa, (e.g. A.aquaticus) dominating the S.Auf.U. community.

FIG. 4.77 RELATIOSHIPS BETWEEN B.M.W.P. SCORES DERIVED FROM CYLINDER RIFFLE SAMPLERS AND S.AUF.U. IN ASSOCIATED POOLS OF SIMILAR WATER QUALITY(USING DEPOSITING SCORE-FIG.4.4-FOR S.AUF.U.).

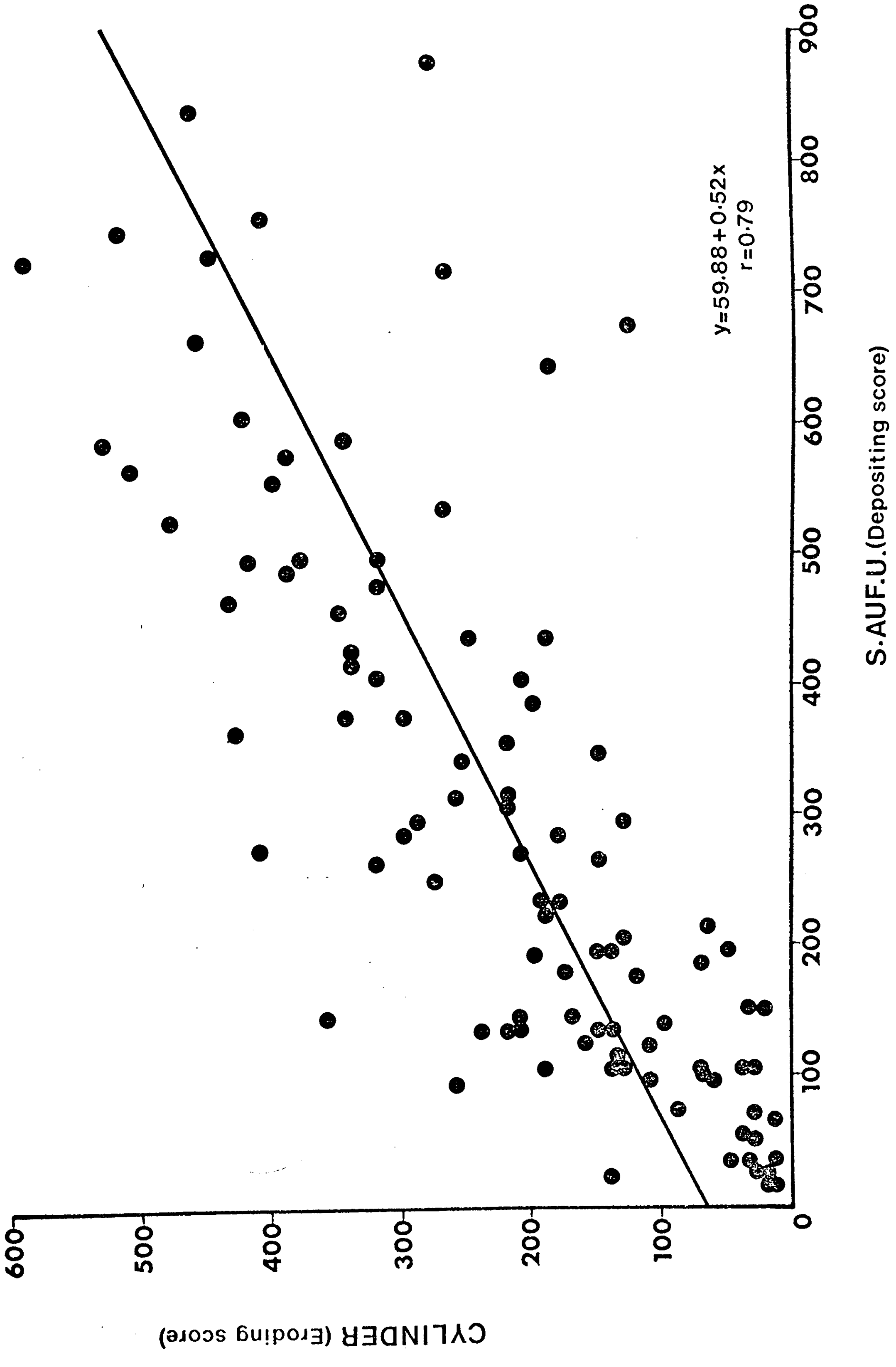


FIG.4.78 RELATIONSHIPS BETWEEN B.M.W.P. SCORES DERIVED FROM CYLINDER RIFFLE SAMPLERS AND S.AUF.U. IN ASSOCIATED POOLS OF SIMILAR WATER QUALITY(USING ERODING SCORE-FIG.4.4-FOR S.AUF.U.).

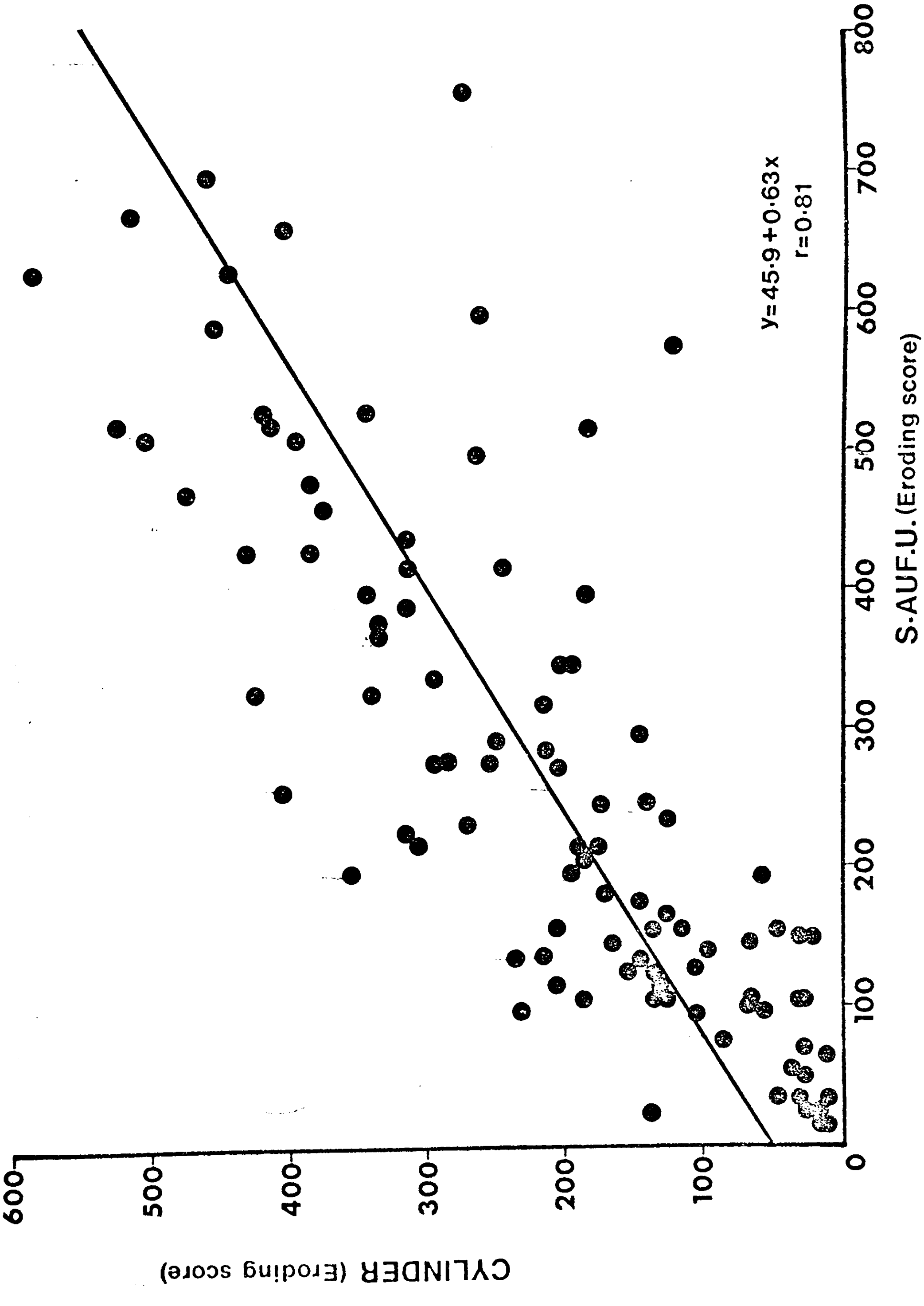


FIG.4.79 RELATIONSHIPS BETWEEN CHANDLER SCORES DERIVED FROM CYLINDER RIFFLE SAMPLES AND S.AUF.U. IN ASSOCIATED POOLS OF SIMILAR WATER QUALITY.

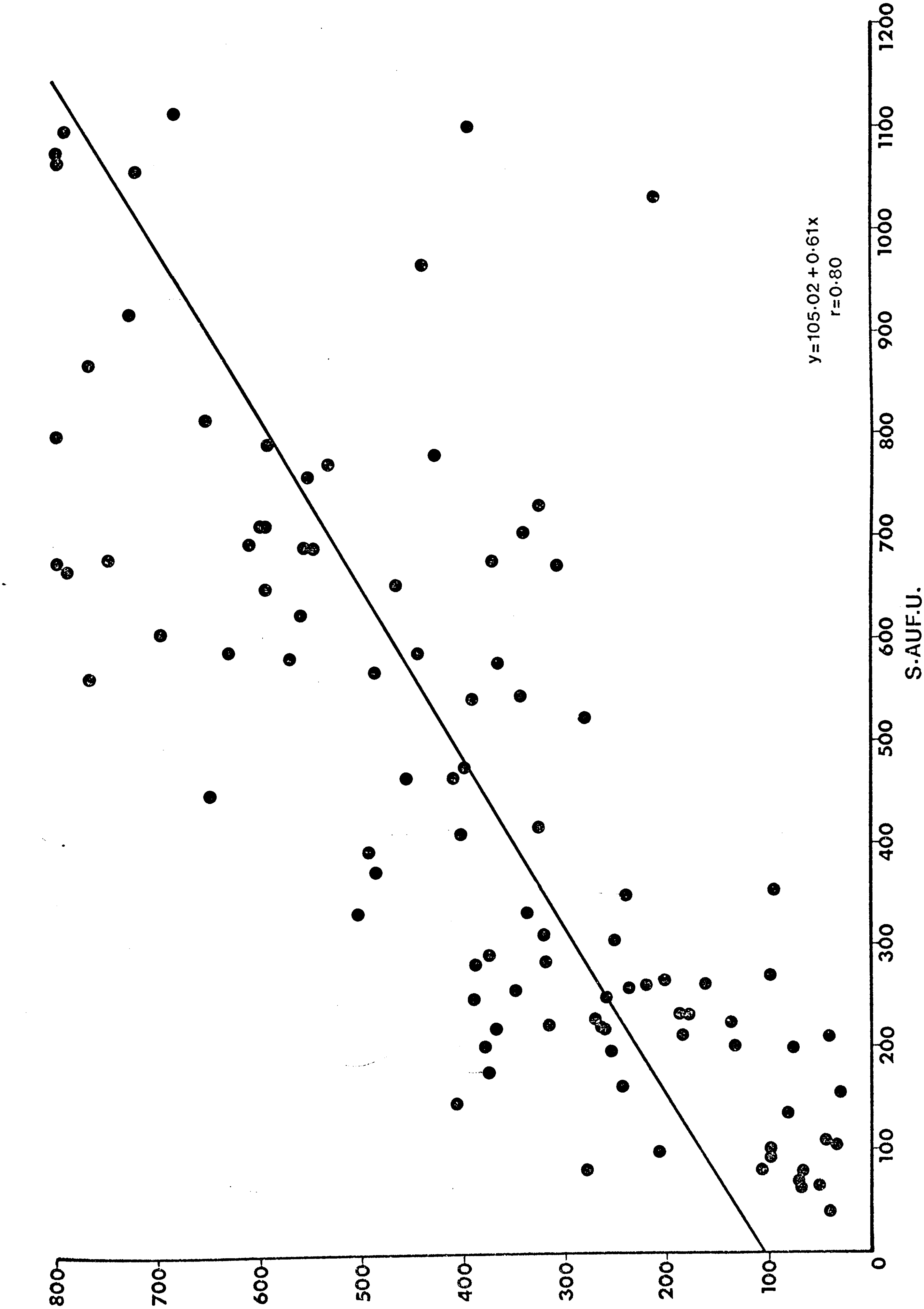
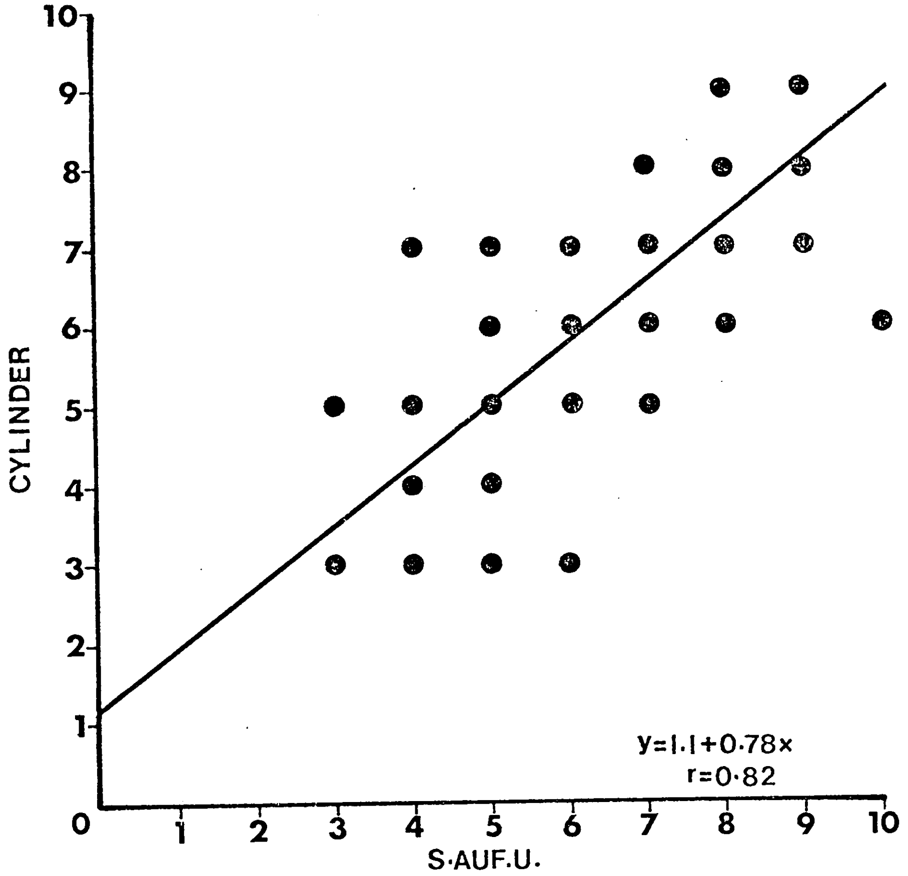
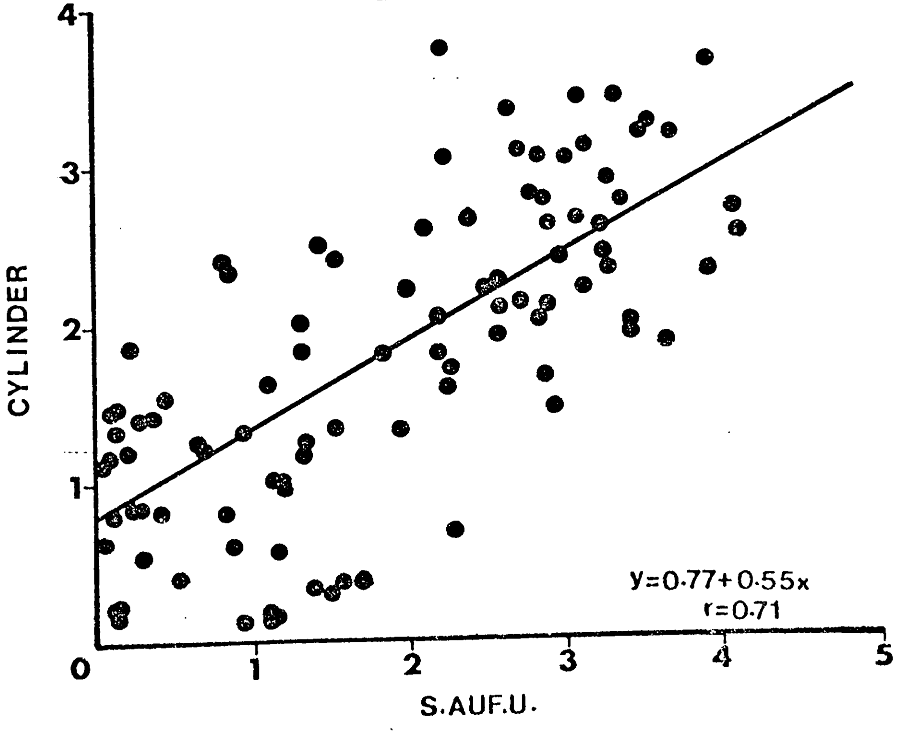


FIG. 4.80 RELATIONSHIPS BETWEEN TRENT BIOTIC INDEX AND DIVERSITY INDEX DERIVED FROM CYLINDER RIFFLE SAMPLERS AND S.AUF.U. IN ASSOCIATED POOLS OF SIMILAR WATER QUALITY.

TRENT BIOTIC INDEX



DIVERSITY INDEX



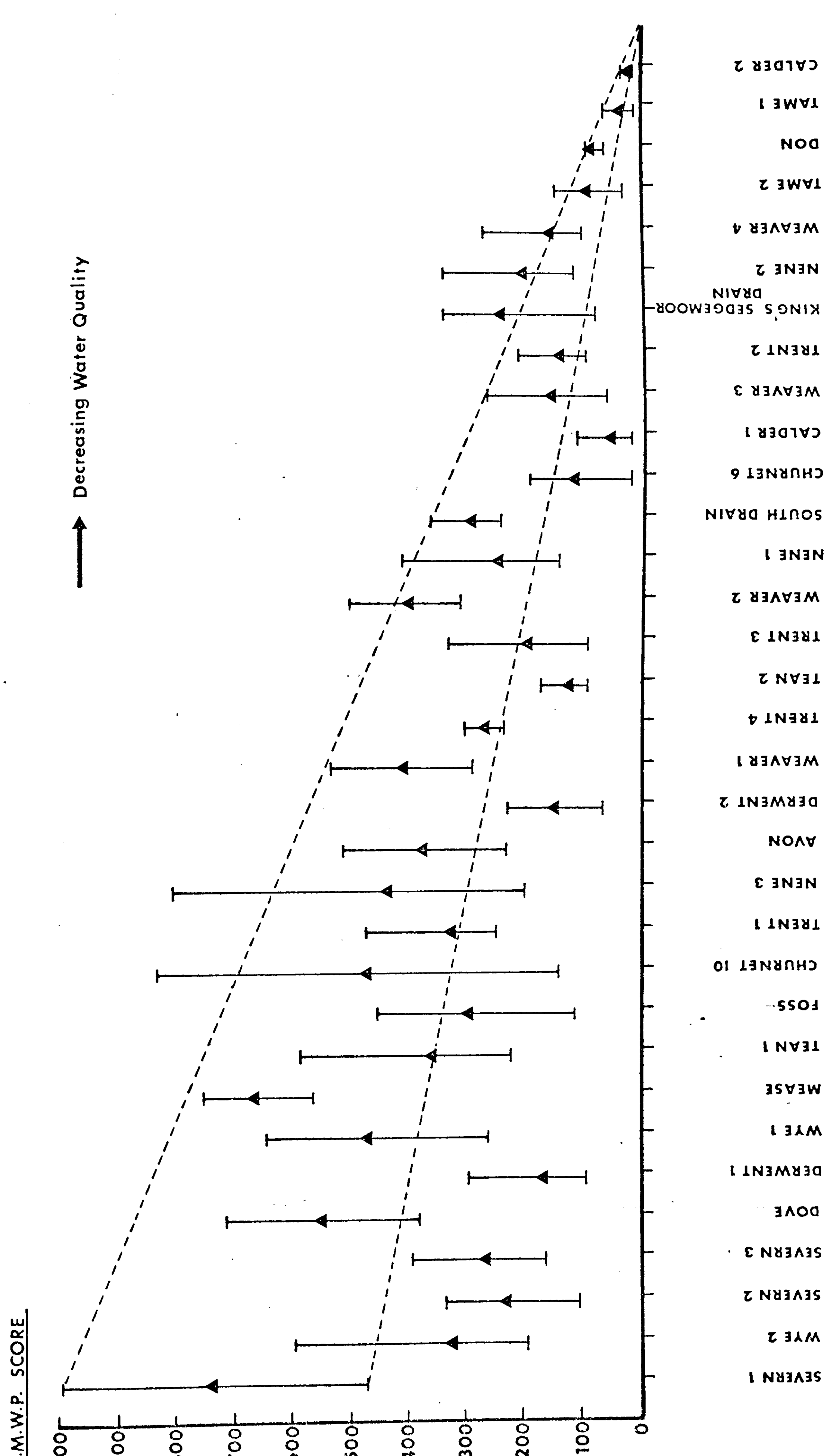
4.112 Comparison of S.Auf.U. data from a range of water qualities and river types

During the survey period (Phase II, 1977) S.Auf.U. samples were taken monthly from 33 stations covering a range of water qualities. Some stations were pools in the rhithron zones, others were situated in the potamon zone and others in slow flowing lowland rivers and drainage dykes. To assess the usefulness of S.Auf.U. data in indicating water quality in a range of river types, the data were processed to produce B.M.W.P. Scores. To investigate any similarities between stations, dendrograms were also prepared using A.U. Clustan Analysis.

4.1121 Comparison of B.M.W.P. Scores derived from S.Auf.U. data

The 33 stations from which regular chemical data were available were arbitarily arranged in order of water quality based on different chemical determinands (minimum dissolved oxygen, BOD_5 and ammonia), a knowledge of the effluent discharges and water use. The seasonal ranges of B.M.W.P. scores for these stations are shown in Figure 4.81 in which the stations are arranged from good water quality on the left to most polluted on the right. A major drawback to this approach is the validity of the sequence selected since this was made in some cases on scanty information. Nevertheless, as seen in Figure 4.81, there was a general trend in the scores across the range of water qualities. There were, however, some marked discontinuities in this general trend which need consideration. The two stations representing a good quality potamon stretch, i.e. Severn 2 and 3 produced a score much lower than expected and this was related to the nature of the substratum and the absence of upstream riffles.

FIG.4.81 COMPARISON OF B.M.W.P. SCORES(MONTHLY AVERAGE AND YEARLY RANGE) DERIVED FROM S.AUF.U. DATA FROM A RANGE OF RIVER TYPES AND WATER QUALITIES.



Other stations which although not true potamon, were not associated with nearby riffle zones (Table 4.1) also produced scores lower than expected, e.g. Wye 2, Trent 3 and Foss. Both stations on the River Derwent produced scores considerably less than expected from the available chemical data. Reference to the basic biological data for Derwent 1 shows that there was on most occasions a significant suppression of the invertebrate fauna at this station, this being restricted to oligochaet worms, leeches, Asellus, chironomid larvae and snails. Ephemeroptera, Odonata and Trichoptera were generally notably absent. The community of organisms is therefore indicative of appreciably inferior water quality condition than indicated by the limited chemical data available. An intermittent toxic discharge could possibly be the cause of these low scores. Churnet 6 and Calder 1 also support invertebrate communities indicative of inferior conditions than suggested by the chemical data. Churnet 6 is one of the most polluted stations on the River Churnet (Chapter 5) which is reflected in the low scores. Again intermittent toxic discharges were probably the cause of the suppressed community at Calder 1.

These significant discrepancies in the biological and chemical data, although detracting from the correlation, highlight the value of biological data in indicating possible pollutional sources not detected by routine chemical sampling.

Figures 4.82 - 4.83 present the Trent Biotic Index and Chandler Score prepared in a similar way to the B.M.W.P. system. Again, although in some cases biological data indicated inferior conditions than those indicated by chemical data, they rarely indicated better conditions.

FIG. 4.82 COMPARISON OF TRENT BIOTIC INDEX VALUES (MONTHLY AVERAGE AND YEARLY RANGE) DERIVED FROM S.AUF.U. DATA FROM A RANGE OF RIVER TYPES AND WATER QUALITIES.

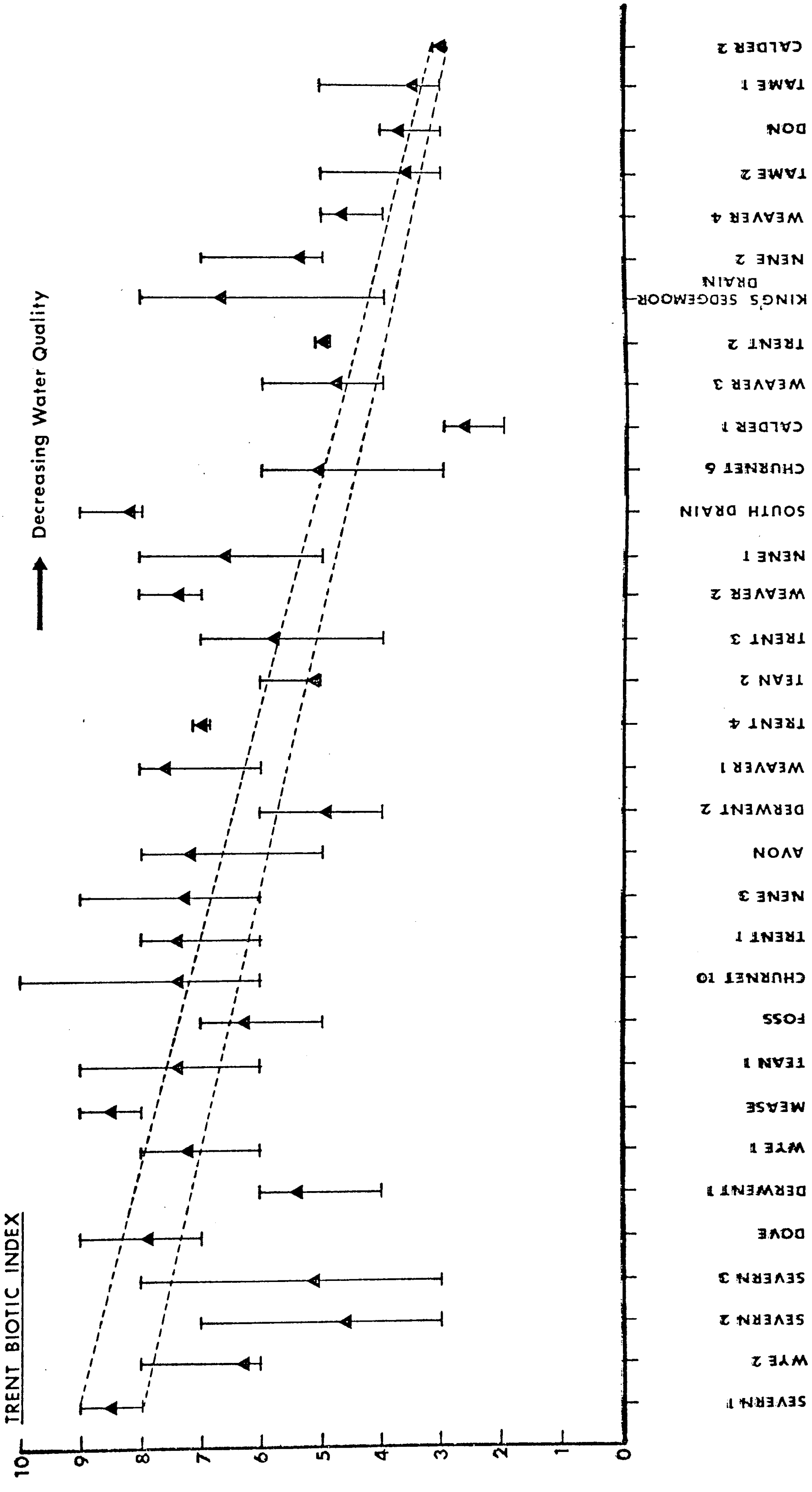
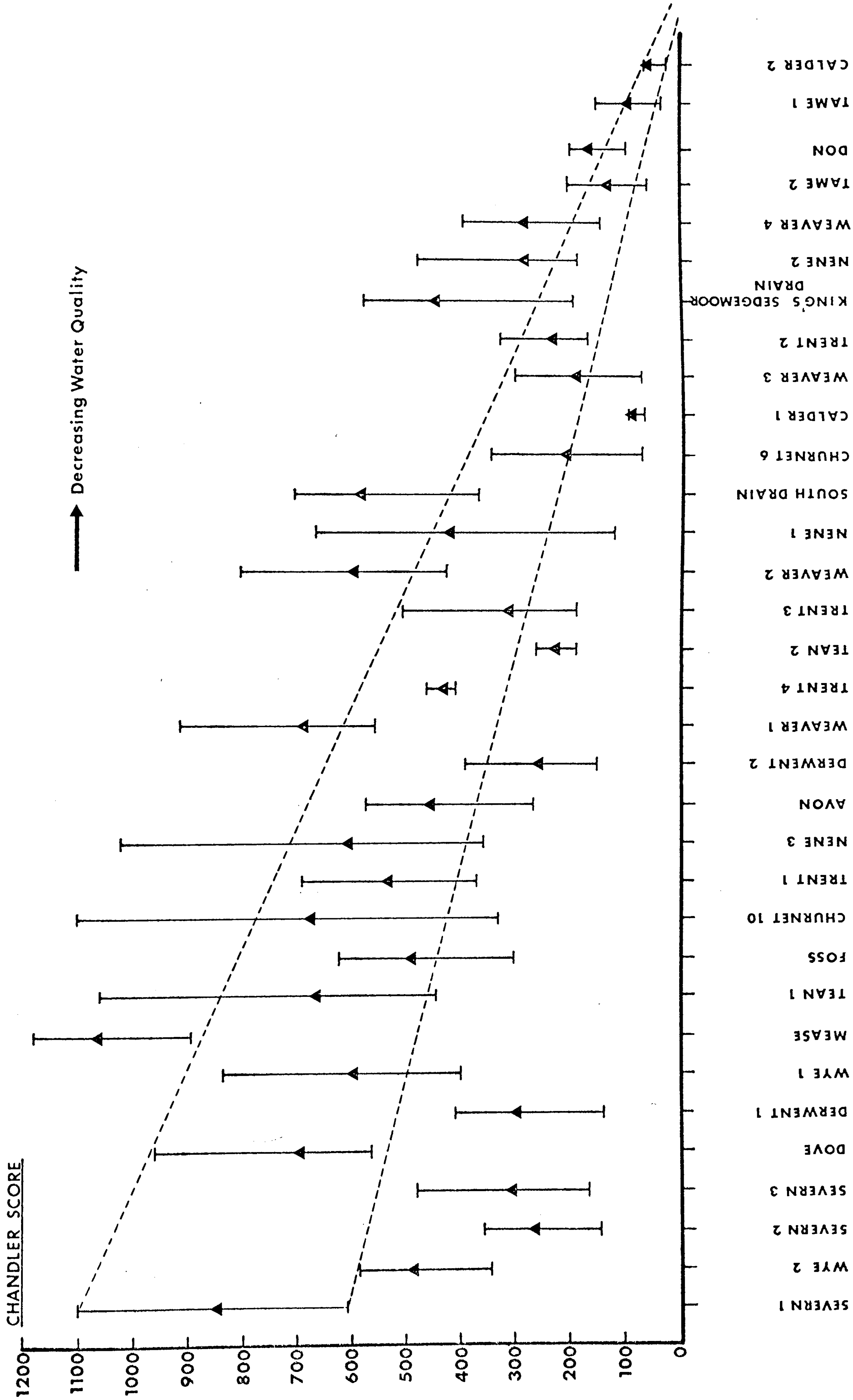


FIG. 4.83 COMPARISON OF CHANDLER SCORES (MONTHLY AVERAGE AND YEARLY RANGE) DERIVED FROM S.AUF.U. DATA FROM A RANGE OF RIVER TYPES AND WATER QUALITIES.



4.1122 Comparison of Invertebrate Assemblages on S.Auf.U. at different stations

The level of similarity between the invertebrate assemblages colonising S.Auf.U. at the different stations were examined to establish whether stations of similar water quality were more closely associated than those of different water qualities and river types. Based on the presence or absence of particular taxa and their relative abundance, using 55 taxa at 29 stations for May's data and 55 taxa at 30 stations for August's data, similarity dendrograms using a single linkage method of clustering were prepared using 'A.U. Clustan Analysis' (FORTRAN language) on an ICL George III computer.

The actual process of clustering is extremely straightforward. At its simplest level it consists of arbitarily selecting a level on the scale of similarity coefficients and collecting together those samples and stations with the required level of affinity. At the highest level only a few samples would be grouped. As the level of affinity is lowered more groups would be formed and some groups would coalesce. Thus M variables are measured for each of N objects (samples) and the similarity between them computed, so that large quantities of data can be objectively analysed and interpreted. It must be emphasised that cluster techniques are not absolute but can provide relative estimates of association within each analysis. Several different methods of analysis need to be used, ideally demonstrating consistent trends in grouping when different analyses are performed on the same data. While there are recognised problems in the interpretation of the results displayed by these techniques, the approach seems to be receiving increased attention. Although clustering techniques used in this investigation are not intended to produce an index of water quality, they do allow a reasonable assessment to be made particularly if sampling stations containing fauna representative of a clean river and a polluted river of similar type to the others under investigation are included as reference points.

The results of quantitative data computed for May, 1977 are presented in Figure 4.84. Although, as discussed below, some of the stations of similar water quality are found to be more closely related, in general there is no clear clustering of stations into the different ranges of water quality as assessed in the sequential presentations (Figures 4.81 - 4.83). There is, however, a closer relationship between the groupings of the stations and their respective B.M.W.P. Scores. All three methods of presentation, ie. nearest neighbour, furthest neighbour and group average, indicate the closest relationship between Calder 1 and Calder 2. Although these two stations were separated on the sequence based on chemical determinands (Figure 4.81) as discussed earlier, Calder 1 was probably misplaced as indicated by the B.M.W.P. Scores. Closely linked with the Calder stations were Churnet 6 and Derwent 1 (Figure 4.84). Both these stations gave B.M.W.P. Scores lower than expected from the chemical data (Figure 4.81). Other polluted stations, e.g. Tame 1 and Nene 2, although in the same grouping are less closely associated. Equally closely related to these, however, were Nene 3 and Severn 3 both good quality river stretches from which water is abstracted for public supply. They are, however, both stations on lowland rivers not under the influence of local riffles. It would appear therefore that although this treatment of the results showed associations between the more polluted stations, these were also linked with some lowland good quality stations, in which the fauna was similarly restricted but for different reasons.

FIG.4.84 DENDROGRAMS SHOWING DEGREE OF SIMILARITIES BETWEEN INVERTEBRATE ASSEMBLAGES COLONISING S.AUF.U. AT DIFFERENT STATIONS IN A RANGE OF RIVER TYPES AND WATER QUALITIES(MAY,1977) QUANTITATIVE DATA.

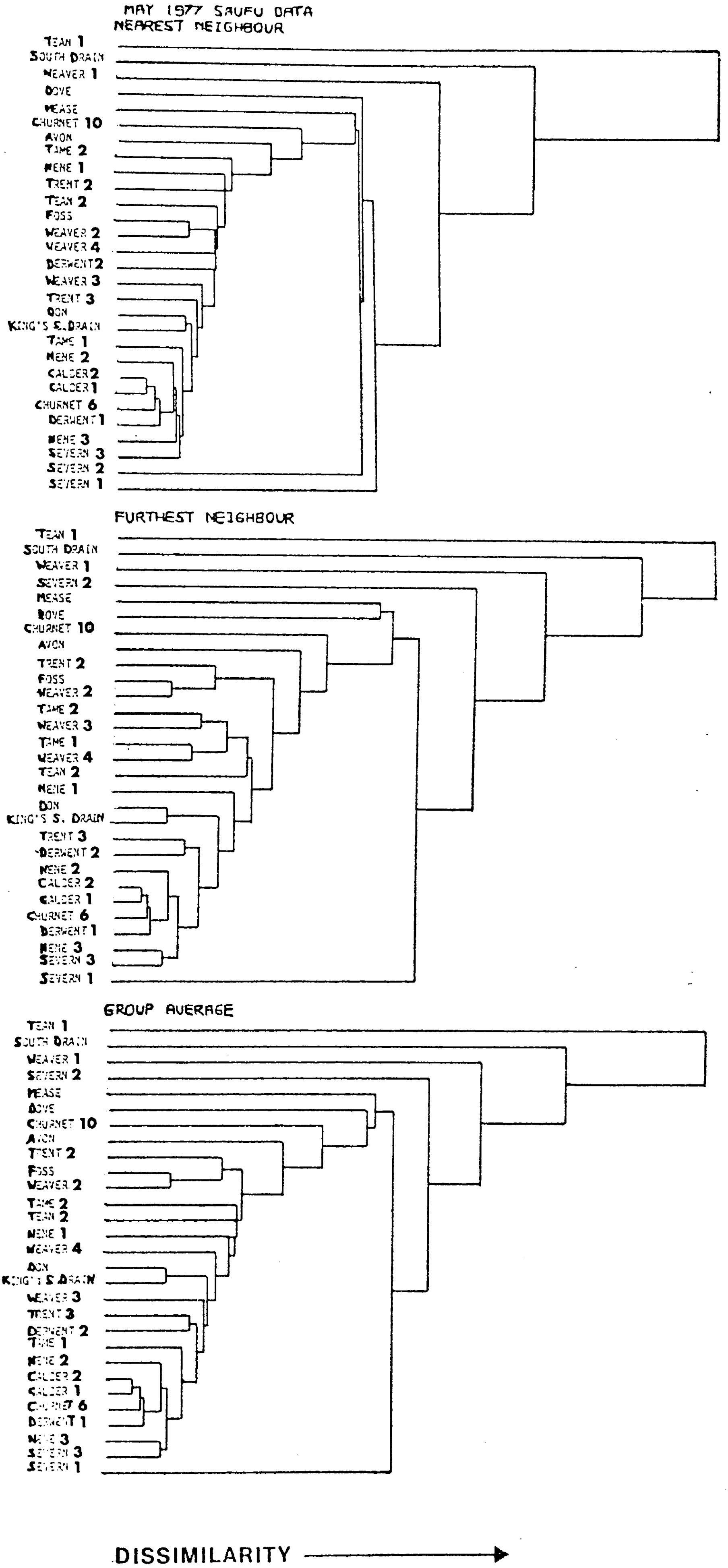


FIG.4.85 DENDROGRAMS SHOWING DEGREE OF SIMILARITIES BETWEEN INVERTEBRATE ASSEMBLAGES COLONISING S.AUF.U. AT DIFFERENT STATIONS IN A RANGE OF RIVER TYPES AND WATER QUALITIES (MAY, 1977) BINARY DATA.

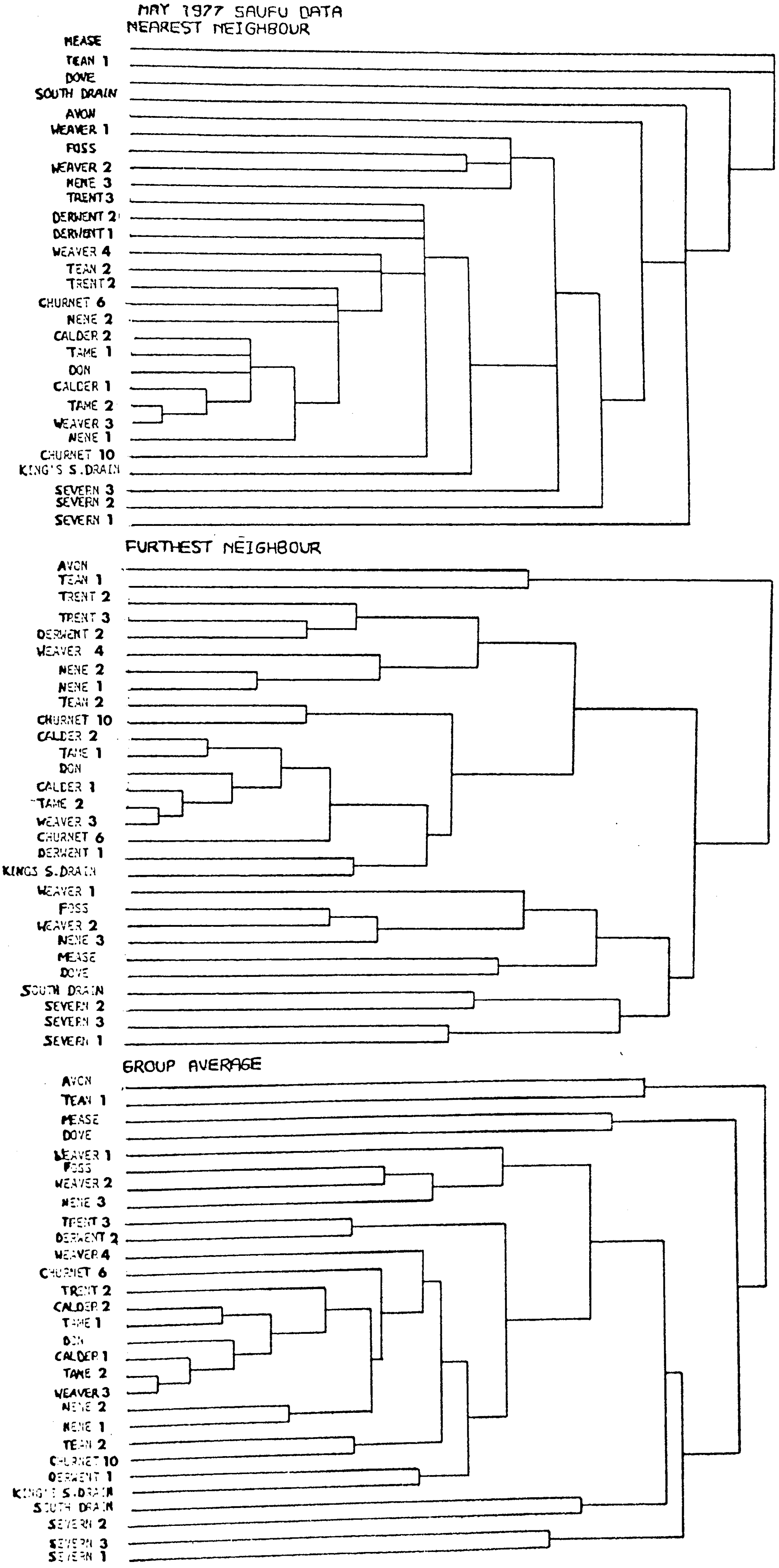


FIG.4.86 DENDROGRAMS SHOWING DEGREE OF SIMILARITIES BETWEEN INVERTEBRATE ASSEMBLAGES COLONISING S.AUF.U. AT DIFFERENT STATIONS IN A RANGE OF RIVER TYPES AND WATER QUALITIES (AUGUST, 1977) QUANTITATIVE DATA.

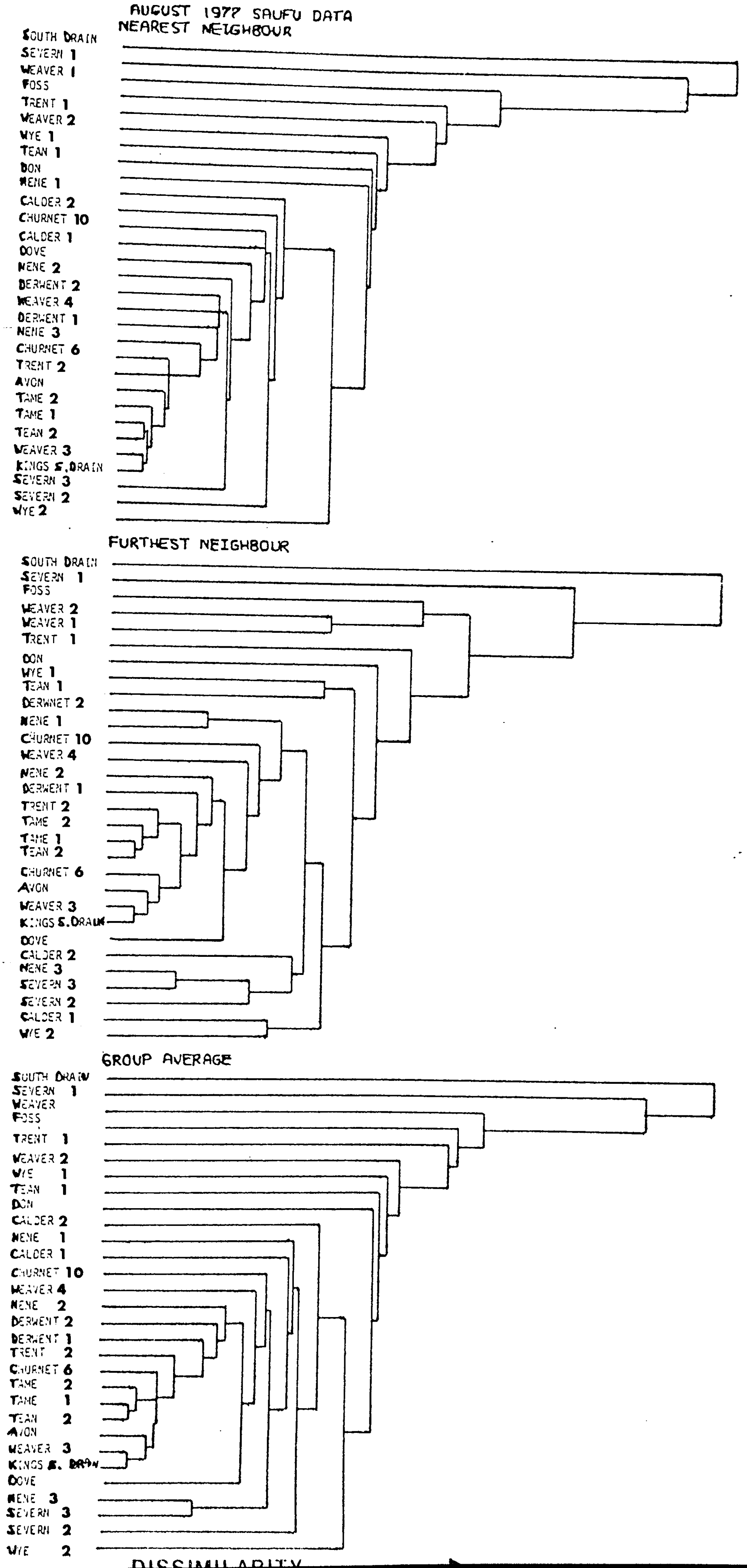
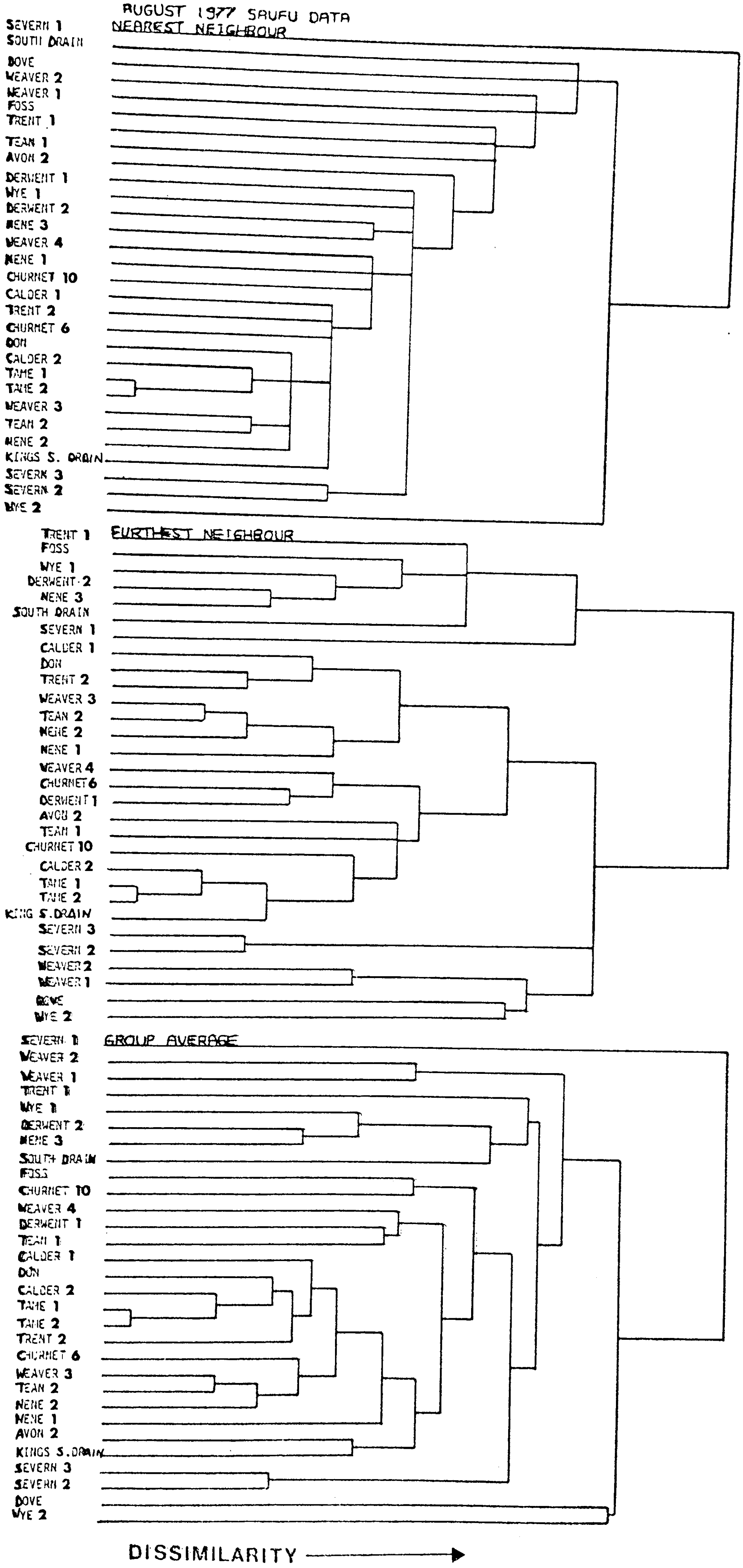


FIG.4.87 DENDROGRAMS SHOWING DEGREE OF SIMILARITIES BETWEEN INVERTEBRATE ASSEMBLAGES COLONISING S.AUF.U. AT DIFFERENT STATIONS IN A RANGE OF RIVER TYPES AND WATER QUALITIES(AUGUST,1977) BINARY DATA.



The dendrograms indicate that at the polluted stations water quality is the over-riding factor determining the invertebrate assemblages on the S.Auf.U. which therefore show a high degree of similarity. At the non-polluted and mildly polluted stations other different factors assume importance resulting in a high degree of dissimilarity even between stations of similar water quality.

Processing the results on a binary basis (present or absent) more clearly linked the polluted stations and separated them from the lowland rivers (Severn 2 and 3) with which they were linked when quantitative data was considered (Figure 4.85). Similar conclusions were also evident from the August data (Figure 4.86 - 4.87) although seasonal variation was now affecting the distribution of sampling stations in the dendrogram.

From the data presented, clustering techniques must be interpreted with great caution since different processing techniques appear to alter the presentation of assemblages which influence the general classification of the sampling stations.

However, by using species assemblages or entire communities of benthic invertebrates to indicate environmental disturbances in freshwater ecosystems together with the indicator concept one can define criteria for assessing water quality and pollution.

The results of this study emphasise the importance of using the same sampling method when carrying out biological surveillance studies. If it is desirable to standardise one particular type of sampling technique as local conditions and preference of workers vary, it is essential that the acceptable types meet certain requirements of reproducibility, variability and retrievability. If further work is continued on the performance of S.Auf.U. samplers in lowland rivers to assess water quality their future in this role would seem to be assured. The manner in which they are used, however, will differ with different surveillance objectives. However, the problems of applying a biological classification to the widely differing types of rivers in England has yet to be overcome satisfactorily.

4.12 Conclusions, Recommendations and Future work

4.121 Conclusions

Based on the results of this survey (Phase II) and the data obtained from earlier work (Phase I), the following conclusions can be made.

4.1211 Sampling Riffles

In good quality riffles, direct sampling by heel-kick using a hand net or by cylinder sampler produced more species than were taken by any of the colonisation samplers. In the riffles of polluted water the difference between direct sampling and colonisation sampling was less than in good quality water. Besides collecting fewer species difficulties were experienced in using colonisation samplers in riffles such as their fouling with drifting algal mats, accumulation of debris and inorganic material, and vandalism. Since riffles are readily sampled, it is therefore recommended that riffles be sampled directly. There was little difference between the number of species taken by a 30 second heel-kick sample and the number taken using a cylinder sampler. Since in biological surveillance it is advantageous to sample similar biotopes at the stations being compared and that at different stations different mosaics of biotopes may exist, the use of a cylinder enables the selected biotope to be sampled more readily than by timed heel-kick or distance heel-kick sampling which may range through a number of biotopes.

It is therefore recommended that for biological surveillance purposes, riffle benthic invertebrate communities should be sampled directly using a quadrat sampler such as the cylinder sampler.

The application of this method is illustrated by the intensive survey of the River Churnet (Chapter 5). This survey confirmed the sensitivity of riffle communities to changes in water quality and therefore their usefulness in biological surveillance.

It is recommended therefore that for rivers with suitably positioned riffles in relation to both discharges and public use, biological surveillance is best carried out by direct sampling of the riffles using a quadrat type sampler.

4.1212 Sampling pools

After four weeks immersion in the pools the colonisation samplers collected, on average, as many species as taken by direct sampling and in some cases (Biopac 50 unit - Phase I) more.

Although in the initial trials (Phase I) shallow pools were used which could be sampled directly by hand net, in deeper pools and in lowland rivers with no riffles, sampling the benthos is difficult. It was therefore concluded that the use of colonisation samplers for pools and other non-riffle stretches should be further investigated. Apart from the bags of smooth gravel which were colonised by fewer species than the pitted slag, there was no marked differences in the performance of the different samplers tested and the selection of the

Biopac 50 Unit for further testing was made on the basis of practicability in the field situation and the fact that it was more readily standardised (Phase I).

4.1213 Comparison of riffle samples and S.Auf.U. samples from associated pools

(a) In cases where associated riffles were available for comparison the S.Auf.U. data responded in a similar way to that from direct sampling of the riffle to known changes in water quality.

(b) Comparing related pools and riffles at 15 stations from a range of river types and water qualities there was a positive correlation between the Scores derived from riffle data and S.Auf.U. data from the pools.

(c) It was concluded therefore that in a pool-riffle type river in the rhithron zone sampling by S.Auf.U. in the pools would provide biological surveillance data comparable with that from associated upstream riffles. Although where riffle sites are suitably situated they would provide a more suitable biotope for biological surveillance purposes, in some situations where riffle sites are not always available the use of S.Auf.U. in pools could prove useful. In such cases the S.Auf.U. data should be processed using the eroding substratum score in deriving the B.M.W.P. Score.

4.1214 Comparison of S.Auf.U. samples and samples taken directly from non-riffle substrata

The relative number of taxa taken by the two methods was greatly influenced by the proximity of the upstream riffle. Where there was a nearby upstream riffle the numbers taken were similar. Where no riffle was present appreciably higher numbers were taken on the S.Auf.U. than from the natural substratum. These differences were reflected in the corresponding scores derived.

This effect of upstream riffles on the S.Auf.U. assemblages was confirmed by studies on the River Blythe and the River Severn. Consideration of the individual taxa colonising S.Auf.U. reveals that they do not sample the natural depositing substratum fauna. A study of the methods of colonisation suggests they are colonised both from the natural depositing substratum and from drift from upstream riffles and possibly from nearby solid hard surfaces including littoral macrophytes.

In lowland rivers without riffles S.Auf.U. provides large number of taxa on which to base a biological surveillance method than does the natural substratum. The score produced from such data, however, would need to be adjusted to be comparable with scores derived from riffle data. To derive a suitable score more data from more potamon stations in a range of water qualities is required.

4.1215 Comparison of B.M.W.P. Scores derived from S.Auf.U. data with Chemical Classification

Comparison of the means and seasonal ranges of the B.M.W.P. Scores with that expected from the chemical data showed that although there was a general trend in the scores across the range of chemically-indicated water qualities, there were some marked discontinuities. Some of these were the lowland and potamon rivers which provided a reduced number of taxa to colonise the S.Auf.U. Others, however, indicated that the true water quality was not indicated by the limited chemical data available. It is significant that where there was a discrepancy, not explained by the physical conditions of the station, the biological data indicated inferior conditions to the chemical data. Such cases highlight the usefulness of complementary chemical and biological data in surveillance work.

4.1216 Comparison of Invertebrate Assemblages on S.Auf.U. at the different stations

Computer analysis of the basic S.Auf.U. quantitative data to investigate the degrees of similarity between stations, showed that whereas the severly polluted stations were more closely linked than the other stations, they were also linked with some lowland good quality stations, in which the fauna was similarly restricted but for different reasons. At the seriously polluted stations, water quality is the over-riding determinant. At non-polluted and mildly polluted stations, other different factors assume importance resulting in dis-similarity. Processing the results on a binary (present or absent) basis more clearly linked the polluted stations and separated them from the lowland rivers (Severn 2 and 3) with which they were linked when quantitative data were considered.

4.122 Recommendations for sampling rivers with no suitably situated riffles for water quality surveillance purposes

4.1221 In the absence of suitable riffle sites, the use of colonising samplers such as S.Auf.U. is recommended. In processing the data to produce B.M.W.P. Scores, consideration must be given to the presence of local upstream riffles. In a riffle-pool type river the S.Auf.U. data should be processed using the eroding score. In lowland rivers with no riffles, the score will need to be adjusted by a factor determined by further experience and investigation.

4.1222 Based on effectiveness and practicality and the results of sequential studies on colonisation, 4 weeks immersion period is recommended.

4.1223 Replication tests showed that by using S.Auf.U. in triplicate between 75% and 85% of the B.M.W.P. Score produced by 10. S.Auf.U. was achieved. It is recommended therefore that a minimum of three S.Auf.U. should be used for biological surveillance studies.

4.123 Future work

Based on the conclusions drawn from this study, it is highly recommended that a comprehensive investigation should be conducted on the biological surveillance of lowland rivers, using benthic macroinvertebrates. Such a study would need to involve the investigation of sampling techniques (including the comparison of S.Auf.U. with grab, dredge and air-lift sampling), sorting methods, identification procedures linked to increasing ones knowledge of their benthic ecology and data processing techniques in a wide range of lowland river types (possibly including estuaries) and water qualities. The latter section would be in the form of producing a recognised water quality index using benthic macroinvertebrates which would take into account the very often low diversity and sparse benthic community of such rivers. It will not be until such research is carried out that lowland rivers can be successfully included in water quality index categorisation.

CHAPTER 5

BIOLOGICAL ASSESSMENT OF WATER QUALITY IN THE RIVER CHURNET, STAFFORDSHIRE, USING RIFFLE BENTHIC INVERTEBRATE COMMUNITIES

5.1 Introduction

A typical response from benthic communities subjected to a catastrophic environmental disturbance is a significant decline in both species diversity and numbers. After a period free from disturbance, or reduced pressure, diversity and abundance usually increase to levels that prevailed before the catastrophe.

The recovery and restoration of bottom fauna communities from the effects of stress may be rapid or slow, depending on several factors, such as:

(1) severity and duration of the stress,

(2) re-colonisation of the damaged area by appropriate aquatic organisms, or

(3) the residual effect of the stress or associated materials.

Greatest damage usually occurs when the stress is highly toxic, although a chronic stress of low toxicity may also cause significant damage. After damage has occurred, the re-colonisation of the damaged area varies, depending on several factors, such as seasonal influence, drift, stress survivors and ovipositing adults.

Butcher (1946) examined biologically various Midland rivers to assess their pollutional status. One of these rivers, the River Churnet was studied in detail because of the presence of three large effluents which severely damaged the ecological balance of the river. A sewage works and dye works in the upper sections of the river reduced a diverse fauna consisting mainly of Gammaridae, Ephemeroptera and Trichoptera, to a community comprising only of Tubificidae and Chironomini. After a slight recovery when Asellus, Hirudinea and Mollusca returned to the river, the entry of a copper works' effluent at Froghall completely exterminated the fauna for over ten miles and had an equally striking effect on the non-polluted River Dove.

The River Churnet was therefore chosen principally to compare the present conditions with those described by Butcher (1946), as revealed by a survey carried out during the late 1930's, and also to examine the use of riffles in the biological surveillance of river water quality.

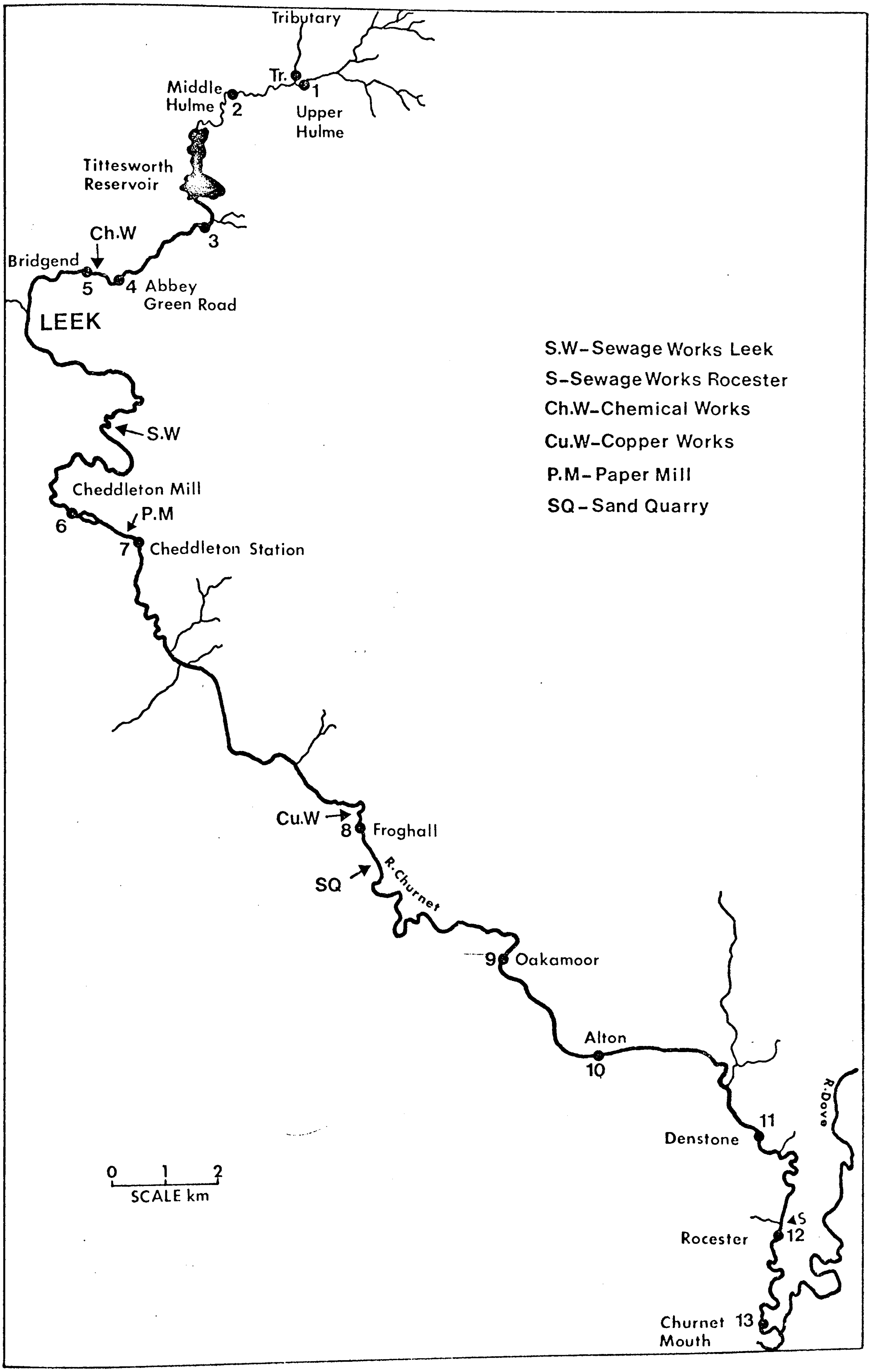
5.2 Description of the River

The River Churnet rises in moorland on Blackshaw Moor in Staffordshire and flows south and south-east around the south-western edge of the Derbyshire Peak District and joins the River Dove near Rocester, about 45 km. from its source. In this distance the river falls about 92 m., an average of 2.25 m/km.

The river rises in an area of millstone grit and gradually descends, passing through regions of Bunter deposits, coal measures, Keuper marl and sandstone, and finally alluvium. Thus, the river passes from an area of relatively soft water to harder water.

The majority of the land in the watershed is agricultural,

FIG.5.1 MAP OF THE RIVER CHURNET SHOWING SAMPLING STATIONS IN RELATION TO MAJOR EFFLUENT DISCHARGES.



although the steep sides of the Churnet valley are wooded. About 5 km. downstream from its source, the river flows into Tittesworth reservoir which provides 27 million litres of water per day for the population of North Staffordshire. Most of the inhabitants in the area live in and around the town of Leek, in the upper quarter of the river's length.

The principal industries in the area and their resulting effluents are depicted in Fig. 5.1. A chemicals factory and a number of textile, dye works and sewage works are sited in Leek, paper mills at Basford (Cheddleton) and copper processing works at Froghall contribute to the pollutional load of the river. Sand is quarried and refined near Oakamoor for glass manufacture and at Rocester agricultural and earth-moving machinery is made. The small sewage works at Rocester finally discharges treated sewage to the river before it finally joins the River Dove at Churnet mouth. Thus eight sewage works in the area treat a total of about 3.5 mgd sewage. In addition, about 5 mgd of industrial effluent enters the river from three major sources: dye-works and paper manufacture waste waters after biological treatment, and copper processing waste waters after precipitation with lime.

5.3 Sampling stations

Fourteen stations were selected for study at suitable riffles and related to the different discharges (Fig. 5.1). A summary of the different sites examined are presented in Table 5.1

Table 5.1 List of Sampling sites examined R. Churnet

Station No.	Site	O.S. Reference	Distance downstream from site 1 (km.)
Tr.	Upper Hulme	SK 013607	-
1	Upper Hulme	SK 012606	-
2	Middle Hulme	SJ 993606	1.4
3	Below Tittesworth Reservoir	SJ 993581	3.9
4	Abbey Green Road, Leek	SJ 978572	6.2
5	Bridgend, Leek	SJ 972572	7.0
6	Cheddleton Mill	SJ 973526	14.8
7	Cheddleton Station	SJ 982521	16.1
8	Froghall	SK 026468	24.2
9	Oakamoor	SK 053443	29.6
10	Alton	SK 072426	32.6
11	Denstone	SK 101412	36.8
12	Rocester	SK 104391	39.9
13	Churnet mouth	SK 101376	42.8

5.4 Methods of sampling

The macroinvertebrate populations were sampled by means of a cylinder sampler described in detail in Chapter 3. Three samples (each 0.05 m^{-2}) were taken across the river at each station and the contents bulked before being preserved with 4% formaldehyde for later examination, sorting, identification and counting. The fourteen stations selected were sampled monthly over a period of fourteen months during 1977 and 1978.

Water samples were also taken at each site for subsequent analysis and the current velocity measured just below the water surface using a small Ott current meter.

5.5 Results

5.51 Current velocity

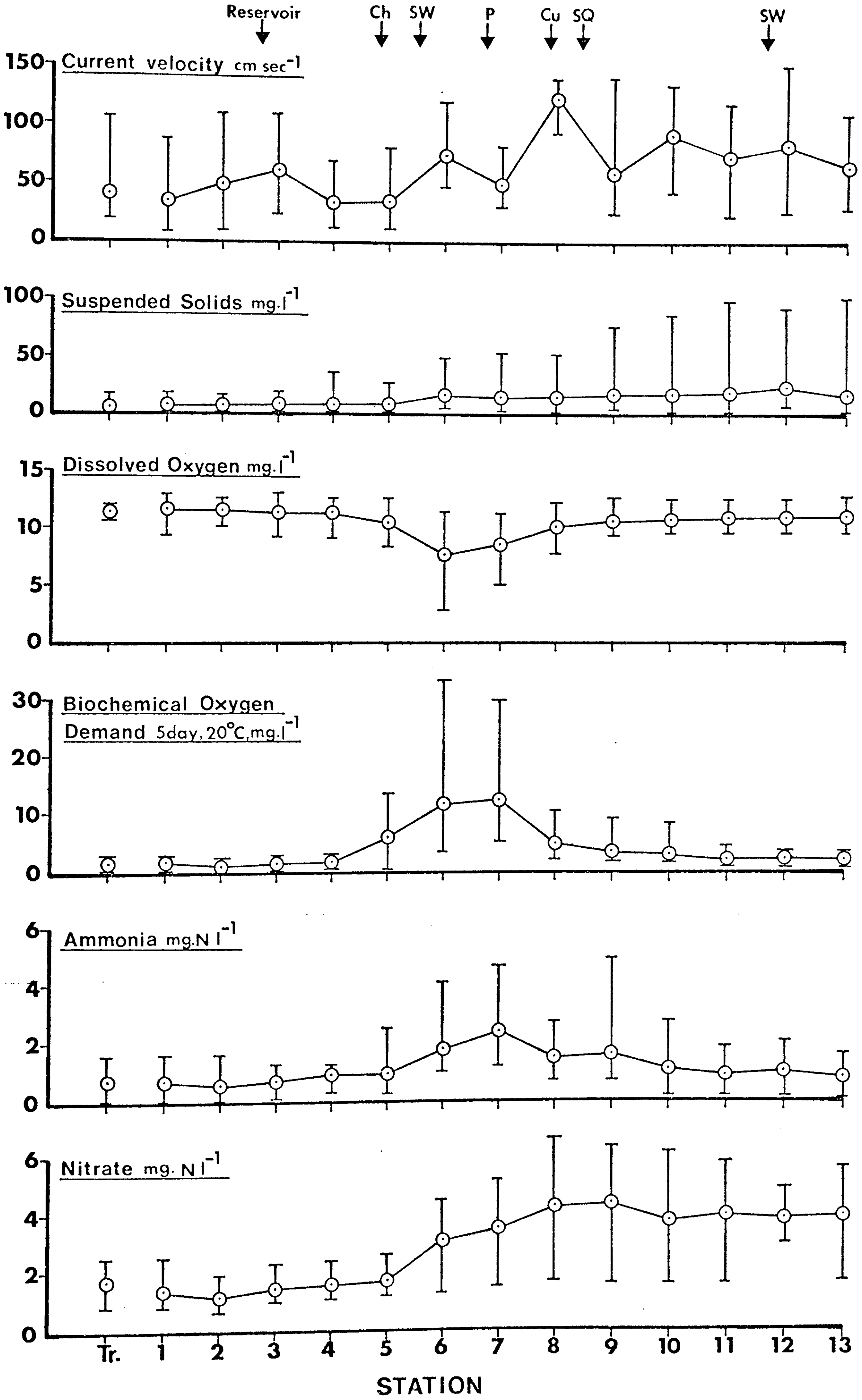
The water velocity measured at each station is presented in Annexe 2 (Table 18a) and summarised in Fig. 5.2. It is apparent that the current velocity gradually increases as one moves down-stream and an examination of the annual mean data suggests that only station 8 below Froghall was likely to have any catastrophic influence on the benthic macroinvertebrate community. Certainly the recordings made by Dorier and Vaillant (1954) on maximum tolerance levels recorded in nature and the laboratory were not exceeded to any significant level. The only possible animal to be affected would be G. pulex if all possible shelter was eliminated.

The day-time oxygen levels recorded over the sampling season at all sampling stations suggested that oxygen saturation would not be a limiting factor to the species survival. However, as pointed out by Hawkes and Davies (1971) low night-time dissolved oxygen levels accounted for the absence of some species, such as G. pulex, in some organically enriched waters, where day-time oxygen conditions were satisfactory.

5.52 Water Quality

The water quality characteristics reported for 1977 and 1978 are presented in Annexe 2 (Tables 18 a - h) and Annexe 3 (Table 26 a - c) and summarised in Fig. 5.2.

FIG.5.2 CURRENT VELOCITY AND CHEMICAL DATA IN RELATION TO EFFLUENT DISCHARGES INTO RIVER CHURNET(1977-1978).



Although the number of samples upon which the tables are based is small, certain generalisations can be made about the overall water quality of the Churnet. The average pH value for all stations lay in the range 6.7 to 7.3 throughout the study; it was generally lowest at Station 2, rising as the water moved downstream. The lowest annual temperatures, as expected, were found in the headwaters, but the greatest fluctuation between stations occurred at Station 6 with the entry of sewage effluent at Leek, increasing the temperature on average by $2^{o}C$. There was a steady increase in total hardness from 75 to 97 mg. l^{-1} as $CaCO_3$ in the upper stations, but the levels increased from 133 to 145 mg. l^{-1} as $CaCO_3$ in the few stations immediately downstream of Station 6. The levels increased again at Station 9 to 182 mg. l^{-1} as $CaCO_3$ below the sand quarry and remained about this level to the confluence with the Dove. The total hardness levels found in the Churnet consisted mainly of calcium hardness, and as a percentage of the total this increased from 65% in the headwaters to 81% in the lower section. Total Alkalinity also increased with the entry of sewage effluent from Leek, ranging from 23 to 41 mg. l^{-1} $CaCO_3$ in the upper section, to between 64 and 77 mg. l^{-1} $CaCO_3$ in the lower section.

The means and seasonal ranges of other important water quality criteria are presented in Fig. 5.2 for the fourteen stations examined on the Churnet. These indicate that the upper stretch of the river as far as Station 4, upstream of Leek, and the Tributary stream 1, were of good water quality. The concentrations of dissolved oxygen were greatest at these stations, while levels of suspended solids, BOD_5, ammonia and nitrate were very low.

At Station 5, below the discharge from the chemical works at Leek, the increase in BOD_5 and resultant decrease in dissolved oxygen indicated an organic discharge, but the absence of any increase in nitrogen, either as ammonia or nitrate, suggested that this was organic matter other than sewage effluent.

The entry of the treated sewage effluent below Leek was indicated by all physico-chemical parameters at Station 6. Dissolved oxygen fell progressively to a minimum mean of 7.7 mg. l^{-1}, while BOD_5 increased to a mean of 11.9 mg. l^{-1}, ammonia to a mean of 1.8 mg. l^{-1} and nitrate to a mean of 3.1 mg. l^{-1}. The discharge from the paper mill at Station 7 had less effect with BOD_5, ammonia and nitrate increasing only slightly. Levels of dissolved oxygen, however, started to increase at this station. At Station 8, any discharge from the copper works which in early years had a marked toxic effect on this river, with levels between 1 and 2 mg. l^{-1} Cu being recorded (Butcher, 1946), now had little effect and a marked increase in dissolved oxygen and decrease in BOD_5 and ammonia indicated considerable recovery. Analytical data from atomic absorption spectrophotometry (Table 5.2) indicated a slight increase in the copper concentrations at Station 8. Other heavy metals showed little increase in concentration along the length of river.

Below Station 9, the data (Fig. 5.2) showed little evidence of changes in the water quality parameters determined along the remainder of the river which could be regarded as being in good condition. Dissolved oxygen levels remained at a constant level approximately 1 mg. l^{-1} below the levels in the upper section, while levels of BOD_5 and ammonia did not decrease to any significant degree. Nitrate levels were generally three times higher in the lower section compared to levels recorded in the upper stretches. Phosphate levels throughout the river were extremely low.

Table 5.2 Means and seasonal ranges of heavy metal concentrations $\mu g. l^{-1}$ at four stations on R. Churnet (1977-78)

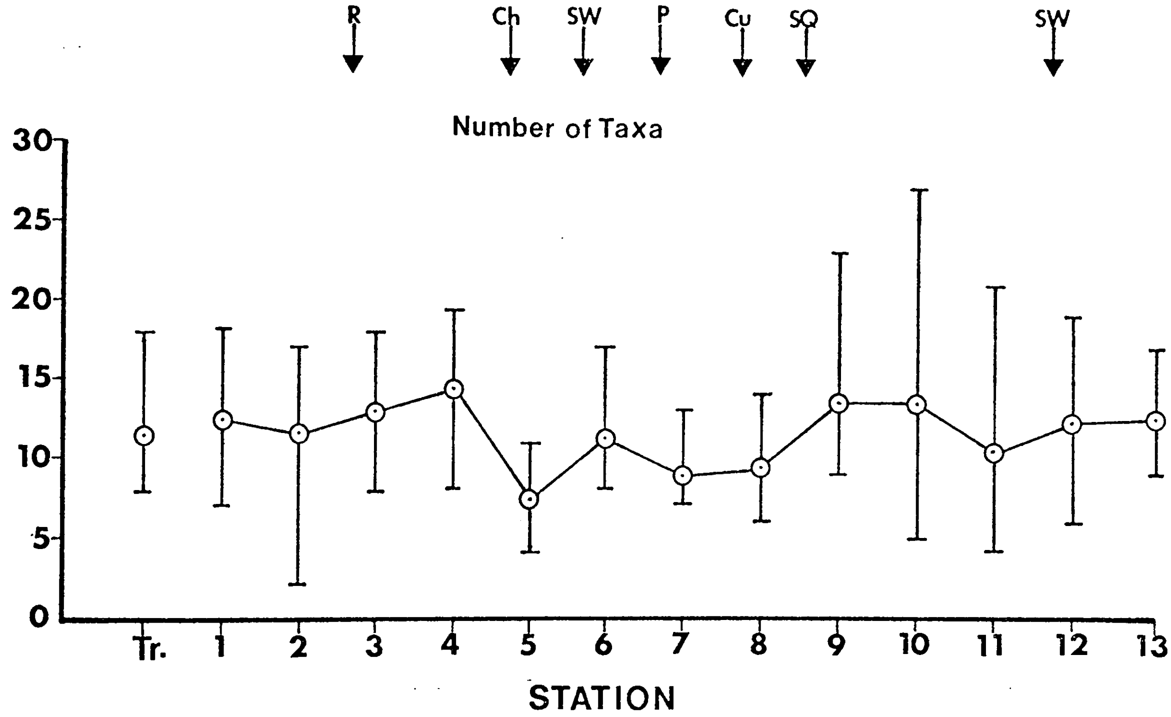
STATION	3	6	8	10
Cadmium	2 (1 - 2)	2 (1 - 2)	2 (1 - 2)	2 (1 - 2)
Chromium	10 (0 - 10)	10 (0 - 10)	10 (0 - 10)	10 (0 - 10)
Lead	14 (0 - 20)	20 (20)	20 (20)	20 (10 - 30)
Copper	10 (2 - 24)	13 (4 - 26)	29 (12 - 60)	25 (14 - 44)
Nickel	16 (5 - 50)	10 (10 - 15)	10 (10 - 15)	17 (10 - 50)
Zinc	34 (0 - 80)	39 (0 -110)	30 (10 - 50)	30 (0 - 50)

5.53 Benthic macroinvertebrates

The detailed results of the benthic macroinvertebrate surveys are presented in Annexe 5 (Tables 96 - 109).

The mean numbers of taxa with seasonal ranges recorded for each station are shown in Fig. 5.3. In the upper two stations, including

FIG.5.3 NUMBERS OF TAXA (MONTHLY MEAN AND RANGE) RECORDED AT DIFFERENT STATIONS ALONG THE RIVER CHURNET IN RELATION TO KNOWN DISCHARGES (1977-1978).



the tributary stream numbers of taxa were generally low, probably due to the physical stress of the river. There was, however, a marked reduction in the numbers of taxa at Station 5 below the discharge from the chemical works at Leek followed by a recovery at Station 6 below the sewage effluent. Below the paper mill the numbers declined again and remained at this level at Station 8, possibly as a result of the high current velocities. In the lower reaches downstream of Froghall, associated with the improved conditions, the numbers of taxa again increased often to levels higher than in the upstream unpolluted zones.

The mean annual distributions of the individual major taxa in relation to the known effluent discharges are shown in Figs. 5.4 - 5.6. Species whose abundances were never greater than one at the fourteen sampling stations were not depicted in these graphs.

The relative abundance of species represented in the upper section of the river, above Tittesworth reservoir, were mainly made up of the stoneflies *Amphinemura sulcicollis*, *Leuctra* spp. (particularly *L. hippopus* during the low temperature winter months) and *Isoperla grammatica*; the mayfly *Rhithrogena semicolorata* and the caseless caddis *Rhyacophila dorsalis*. Chironomids were represented by Orthocladiines and *Diamesa* spp., particularly where the stones bore well-developed growths of filamentous algae and moss. The distribution of *Polycelis felina* (Fig. 5.4) which was mostly restricted to the tributary stream and occasionally station 1 was probably determined by non-water quality factors e.g. low annual temperatures. Below Tittesworth reservoir, the abundance of *Diamesa* spp. dramatically increased possibly as a result of sufficient particles in suspension swept out of the reservoir to support this density. Numbers of the mayflies *Baetis rhodani*, *R. semicolorata* and *Ephemerella ignita* also increased. In these upper regions it is interesting to note that *Hydropsyche* spp., *Potamophylax* spp. and *Glossosoma* spp. were also recorded, but in low numbers.

Most taxa present upstream were eliminated or markedly reduced at Station 5 downstream of the chemical works at Leek. Notable exceptions, however, were *Brillia longifurca* and the silt-loving Chironomini which were most abundant at this station and the Lumbriculidae which increased in numbers. The mayfly *B. rhodani* although slightly reduced at Station 5 exhibited a marked degree of tolerance compared with other species. The distribution of *Brillia modesta* and *B. longifurca* in the River Churnet is of considerable interest. *B. modesta* is generally considered a stream species and *B. longifurca* an inhabitant of ponds and lakes (Thienemann, 1944), but in the Churnet, while *B. modesta* occurred in all the upstream sites and the tributary, *B. longifurca* was only present downstream from Station 5 and a few other regions. Similar situations of *B. longifurca* replacing *B. modesta* downstream of an organic effluent have also been recorded by Besch and Hoffman (1968) in the River Steinach (Germany), Davies (1971) in the River Cole and Learner *et al.*, (1971) in the River Cynon.

The effects of the discharge of the sewage effluent between Stations 5 and 6 was masked by the effects of the upstream intermittent discharge from the chemical works above Station 5. However, some significant changes in the fauna were evident, *Asellus aquaticus*, absent upstream, appeared in abundance and completely overwhelmed

Gammarus pulex as the dominant Crustacean. The levels of A. aquaticus were particularly related to the seasonal growths of Cladophora spp. and mosses; the smallest populations occurring during the winter months when food supply and cover were low. The lumbriculid worms Lumbriculus variegatus which increased in numbers below the chemical works (Station 5) and were common in the upstream waters, were replaced by tubificids at Station 6, especially Tubifex tubifex . Several other species, absent at Station 5, re-appeared below the sewage effluent, these included Eropbdella octoculata, G. pulex, Simulium spp., Lymnaea peregra and Physa fontinalis. It is immediately evident from the data obtained that the effect of the sewage effluent on the benthic communities was to increase the overall level of secondary production compared to the station situated immediately below the chemical works.

Because of the marked effects on the fauna of these two discharges any effects of the discharges downstream were masked. The paper mill effluent upstream of Station 7, probably added to the organic load of the river, while the discharges from the copper works and sand quarry appeared to have little effect. This was indicated by the chemical data shown in Fig. 5.2.

Downstream of the paper mill the river gradually recovered, However, at Station 7 levels of BOD_5 and ammonia remained high. The river is generally more eutrophic as indicated by the nitrate results and the dissolved oxygen levels are on the whole somewhat lower than in the upper reach (Fig. 5.2).

Below the paper mill, numbers of A. aquaticus and Tubificidae decreased, while the levels of Erpobdella octoculata, L. peregra,

FIG.5.4 DISTRIBUTION OF MAJOR TAXA ALONG THE RIVER CHURNET IN RELATION TO KNOWN EFFLUENT DISCHARGES.

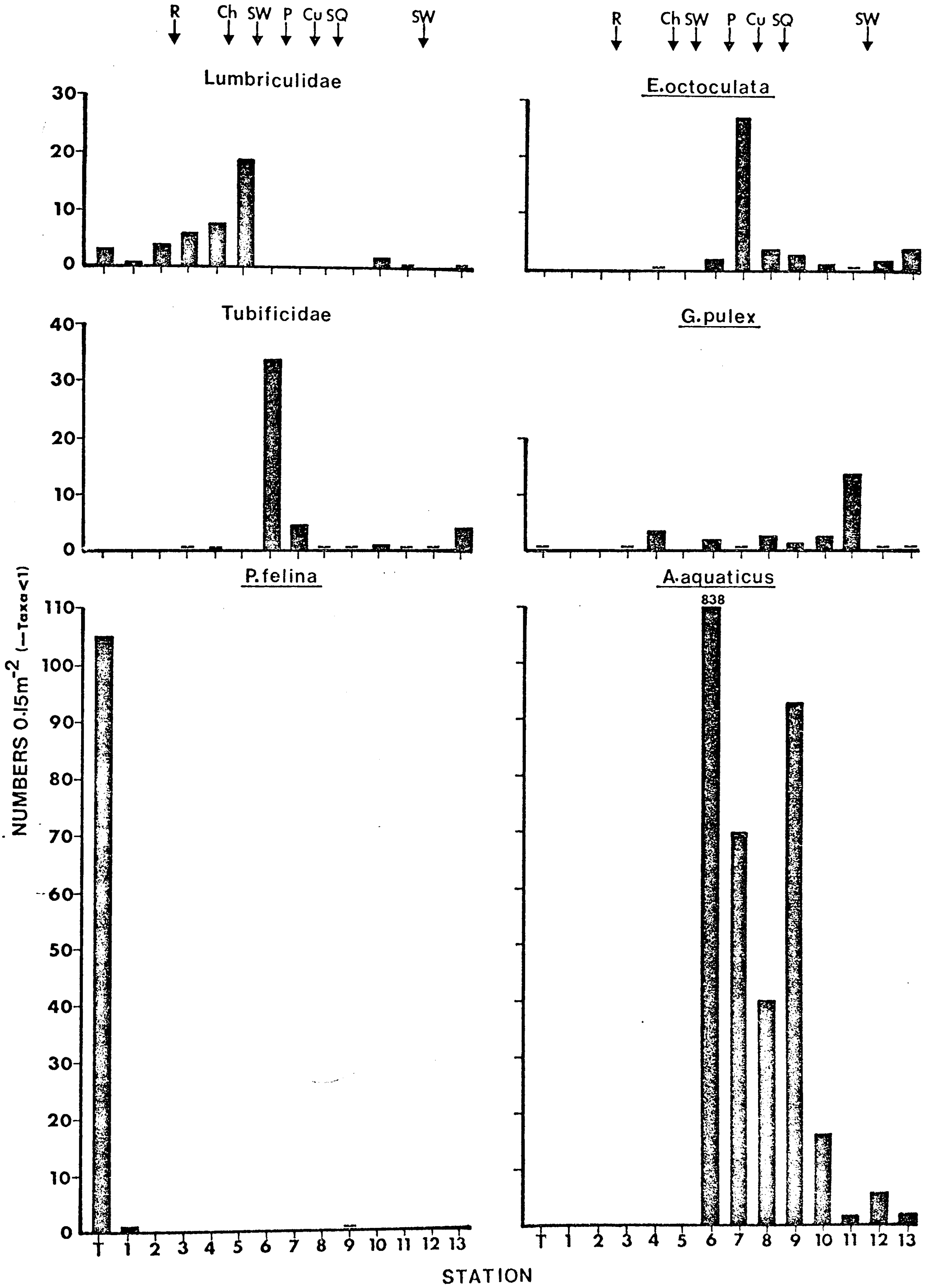


FIG.5.5 DISTRIBUTION OF MAJOR TAXA ALONG THE RIVER CHURNET IN RELATION TO KNOWN EFFLUENT DISCHARGES.

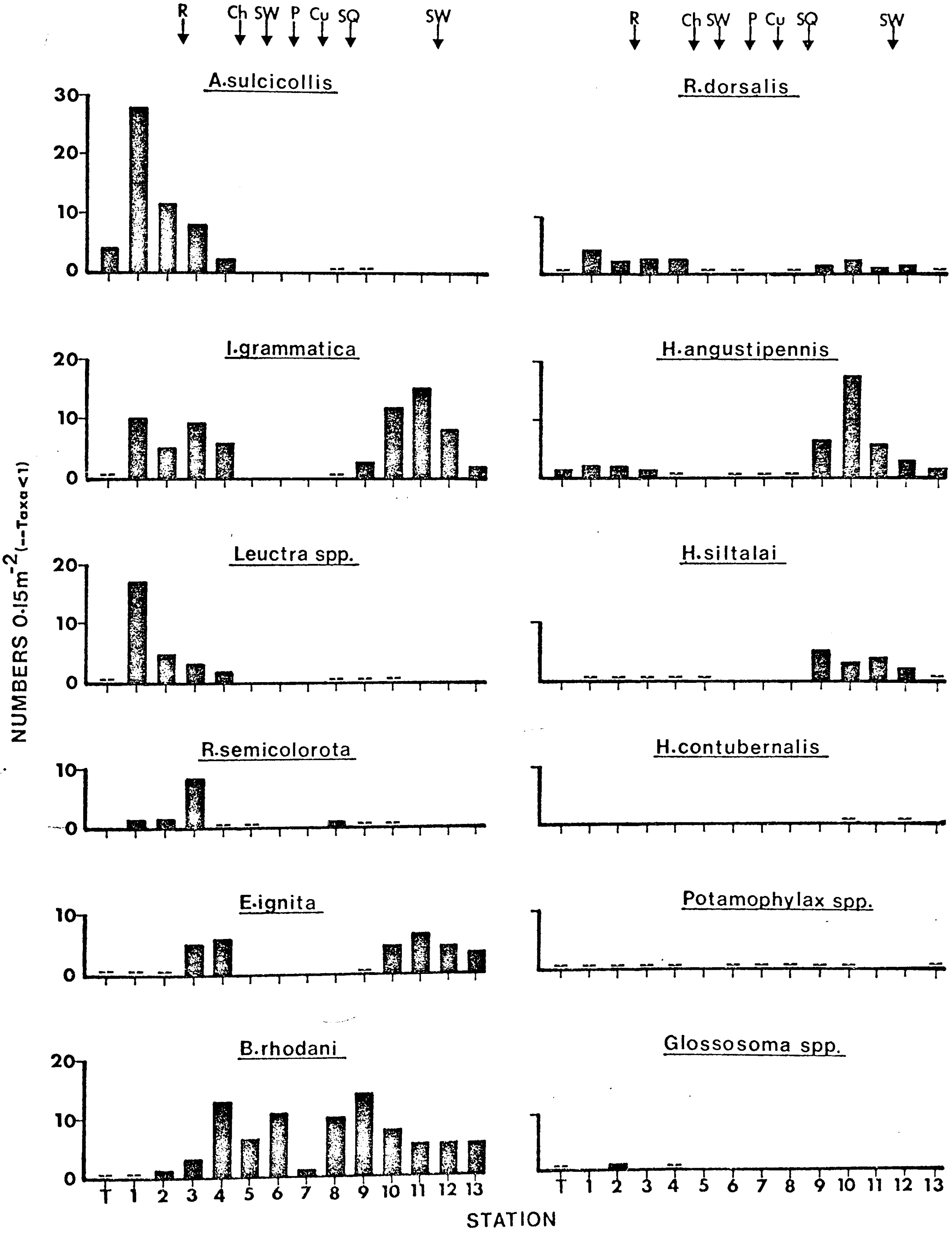
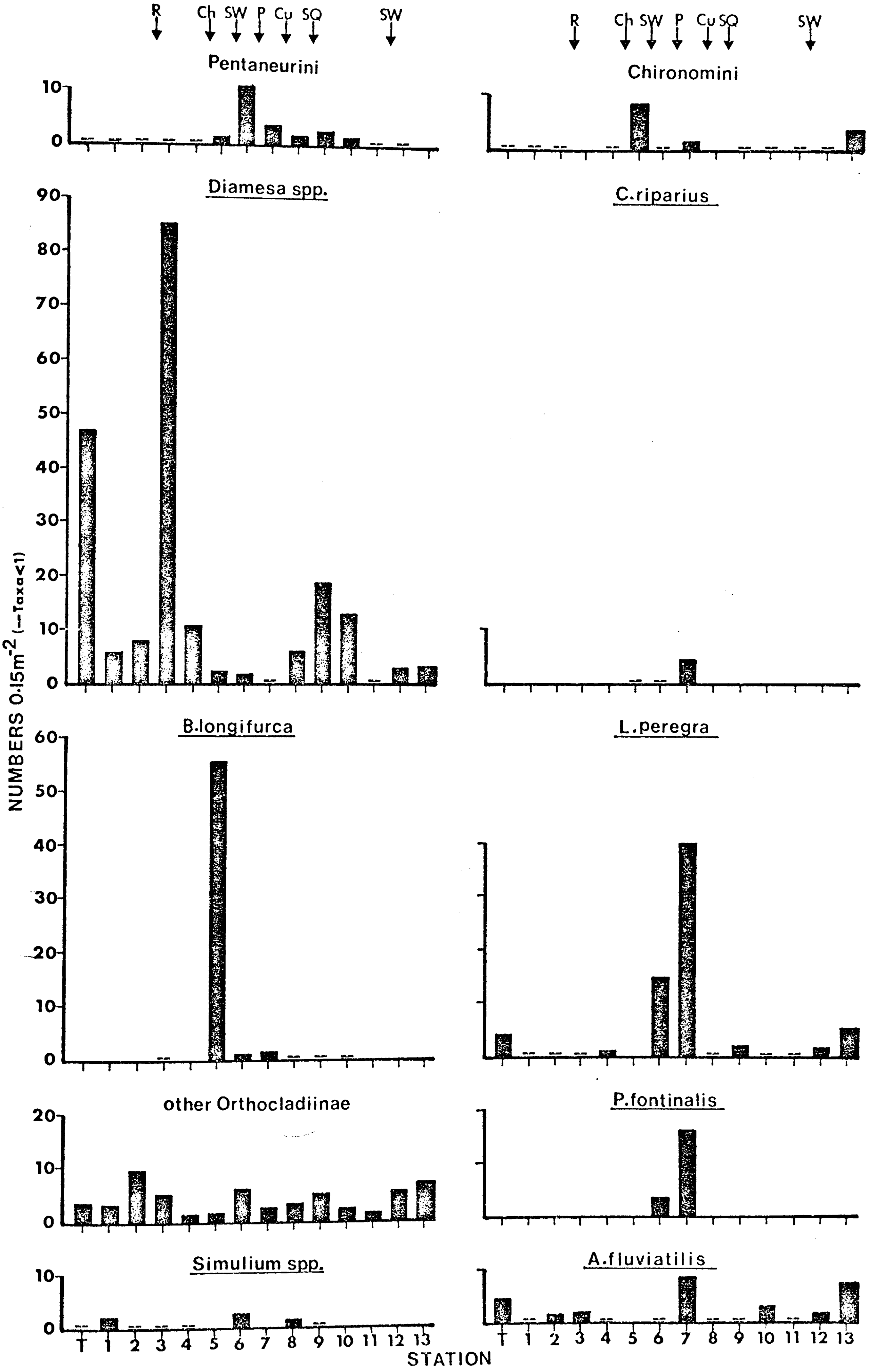


FIG.5.6 DISTRIBUTION OF MAJOR TAXA ALONG THE RIVER CHURNET IN RELATION TO KNOWN EFFLUENT DISCHARGES.



P. fontinalis and *Ancylus fluviatilis* increased. Hynes (1960) comments that the presence of molluscs in streams is favoured by organic enrichment and cites *L. peregra* and *P. fontinalis* as characteristic examples. During the summer months sewage fungus formed massive growths of slimy plumes which colonised all submerged surfaces. The appearance of sewage fungus also led to an increase in the populations of *Chironomus riparius*, but these never reached high proportions.

Further downstream associated with the improvements in water quality, some species, but not all, which were suppressed by the pollutional discharges in the middle reach, became re-established in the lower reaches, e.g. *l. grammatica*, *E. ignita*, *B. rhodani*, *R. dorsalis*, *H. angustipennis*, *H. siltalai*, *G.pulex*, *Diamesa* spp. and *A. fluviatilis*. This was particularly the case at Stations 10 and 11.

The discharge of sewage effluent at the lower end of the river, between Stations 11 and 12, which had little effect on the chemical parameters monitored (Fig. 5.2) also had little effect on the benthic fauna except for a suppression of *G. pulex*, a slight reduction in the numbers of *Hydropsyche* spp., *l. grammatica* and *E. ignita* and associated increase in *A. aquaticus*, *E. octoculata*, *L. peregra* and *A. fluviatilis*.

There was an increase in the percentage cover of *Ranunculus* spp. in the lower section of the Churnet before it flowed into the River Dove. Associated with the macrophytic cover and deposition of silt, there was an increase in the abundance of *Pisidium* spp. and *Atherix ibis*, together with an increase in the numbers of *L. peregra*, *A. fluviatilis* and *Potamopyrgus jenkinsi*.

The level of similarity between the invertebrate assemblages inhabiting the riffle zones at each station on the Churnet were examined to establish whether stations of similar water quality were more closely associated than those of different water qualities. Based on the presence or absence of particular taxa and their relative abundance, similarity dendrograms using a single linkage method of clustering were prepared for May and August, 1977 using 'A.U. Clustan Analysis' (FORTRAN language) on an ICL George III computer.

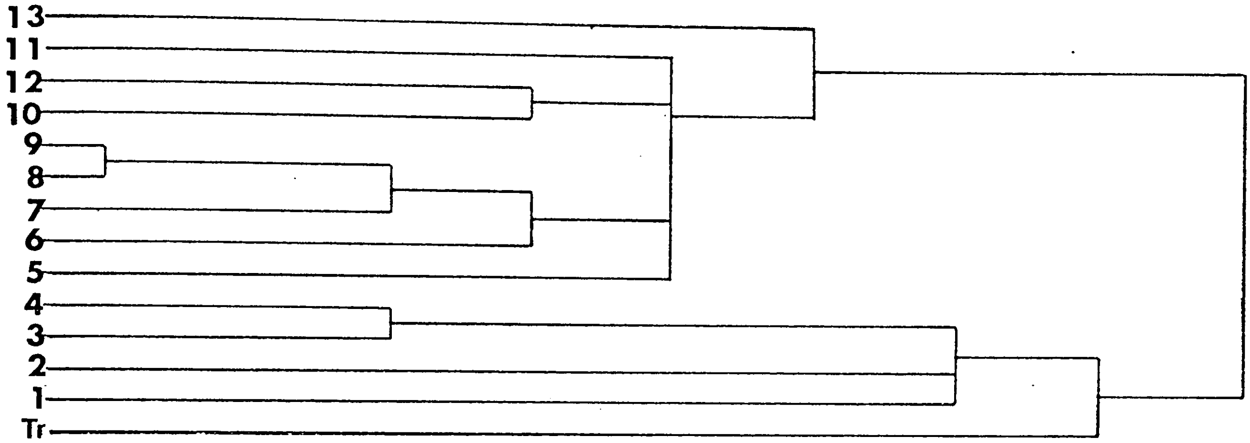
Cluster analysis is the name given to various procedures whereby a set of individuals or units is divided into two or more assemblages or sub-groups (clusters) on the basis of a set of characteristics which they share. All types of cluster analysis are based on the objective assessment of association between groupings within a similarity matrix, which provides a full listing of similarity of association co-efficients between each site and each other site. The techniques used were Nearest Neighbour, Furthest Neighbour and Group Average based on the 14 stations using 30 different taxa. Presence or absence was presented in the calculations as binary data, while relative abundance used numerical data.

The results derived from this analysis are presented in Figures 5.7 a - d. Cluster strategies are complex and it has been pointed out that vastly dissimilar results may be obtained from the same data by the application of different methods of cluster formation.

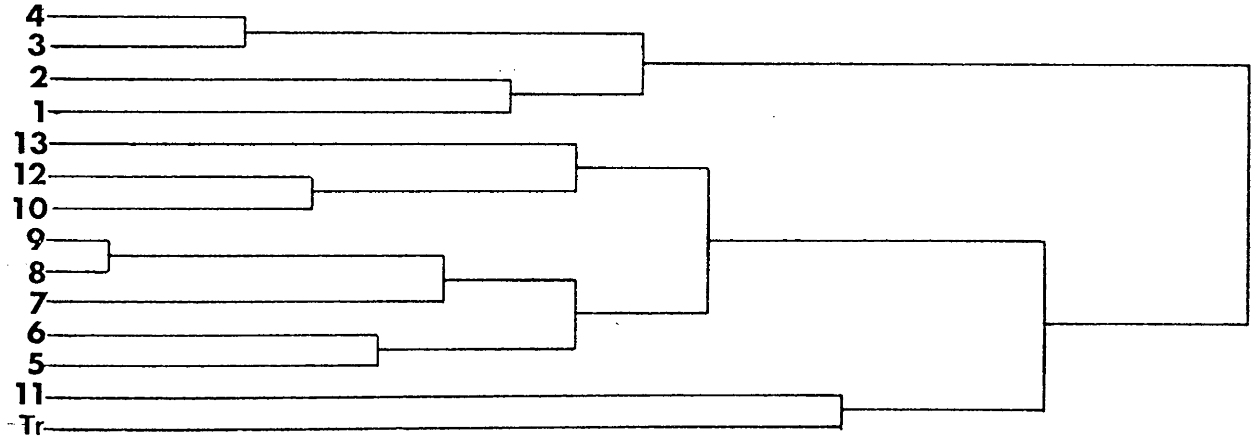
From the various figures, it is apparent that dendrogram structures are altered by the application of different clustering techniques to the same data matrix. A comparison of the various cluster methods

FIG.5.7a DENDROGRAMS SHOWING FAUNISTIC RELATIONSHIPS BETWEEN STATIONS ON THE RIVER CHURNET , MAY 1977 , USING BINARY DATA.

CNTMAY 1977 CYL DATA
NEAREST NEIGHBOUR



FURTHEST NEIGHBOUR



GROUP AVERAGE

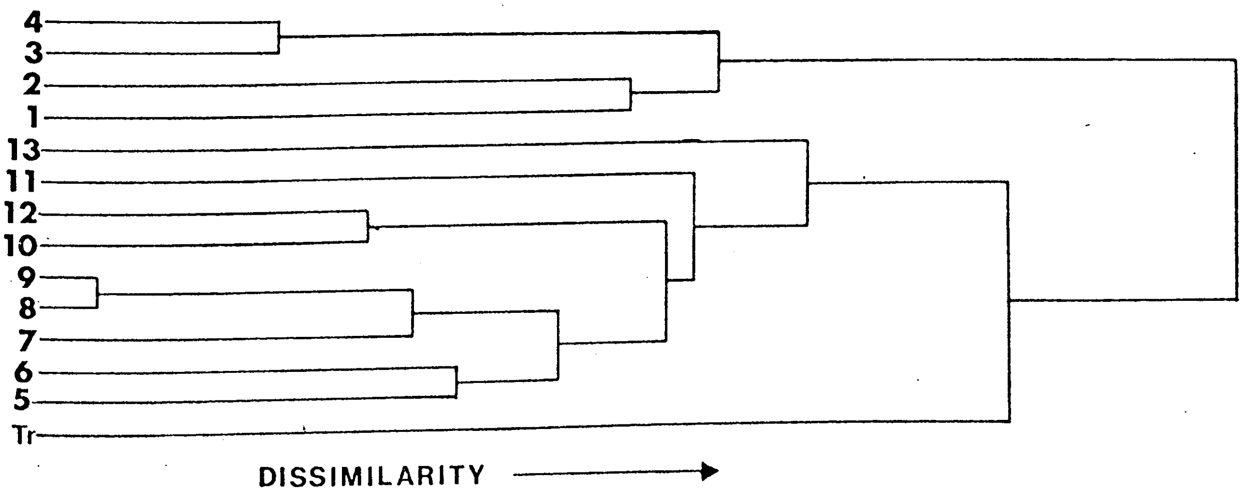


FIG.5.7b DENDROGRAMS SHOWING FAUNISTIC RELATIONSHIPS BETWEEN STATIONS ON THE RIVER CHURNET , MAY 1977 , USING QUANTITATIVE DATA.

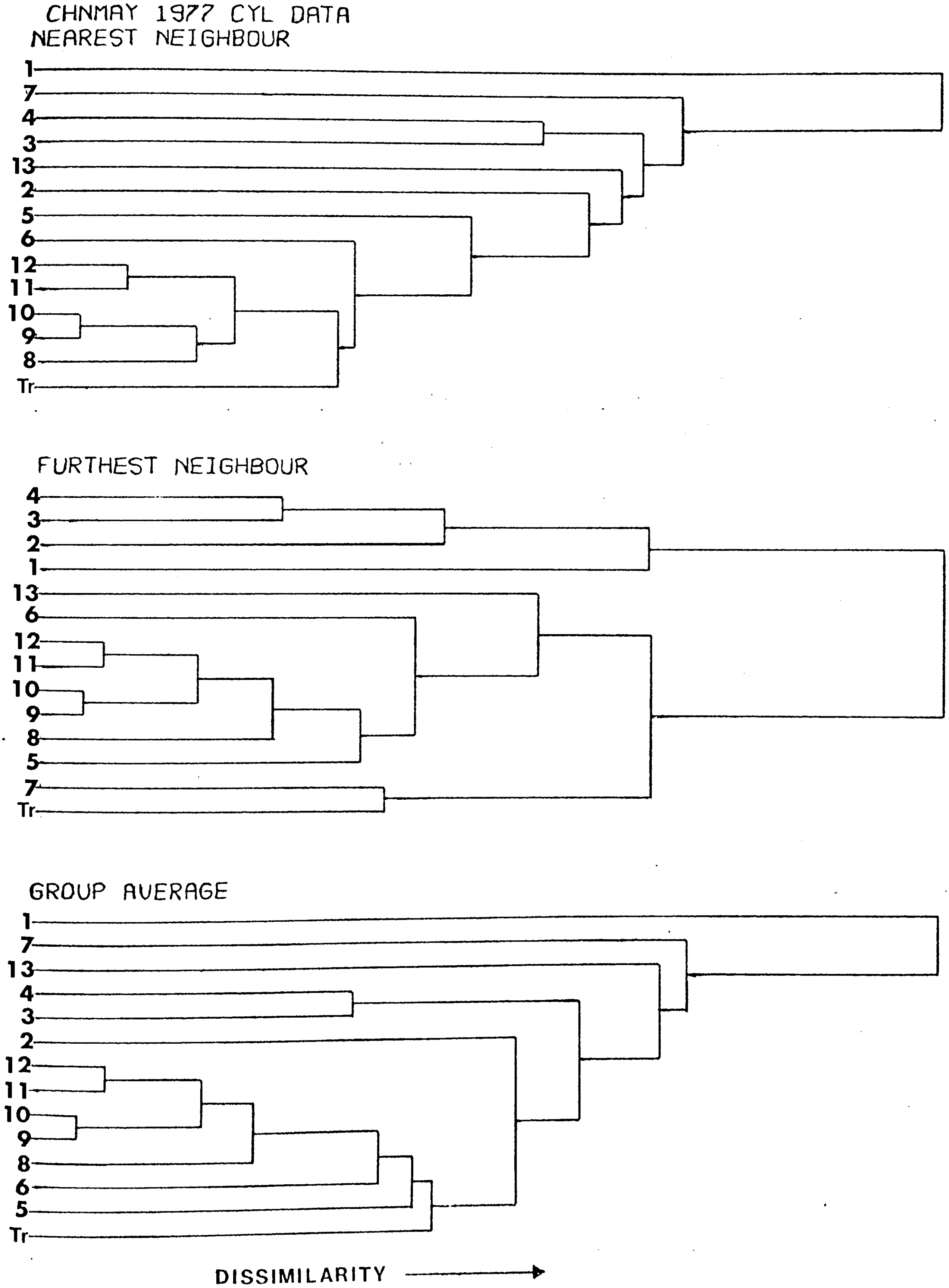


FIG.5.7c DENDROGRAMS SHOWING FAUNISTIC RELATIONSHIPS BETWEEN STATIONS ON THE RIVER CHURNET , AUGUST 1977 , USING BINARY DATA.

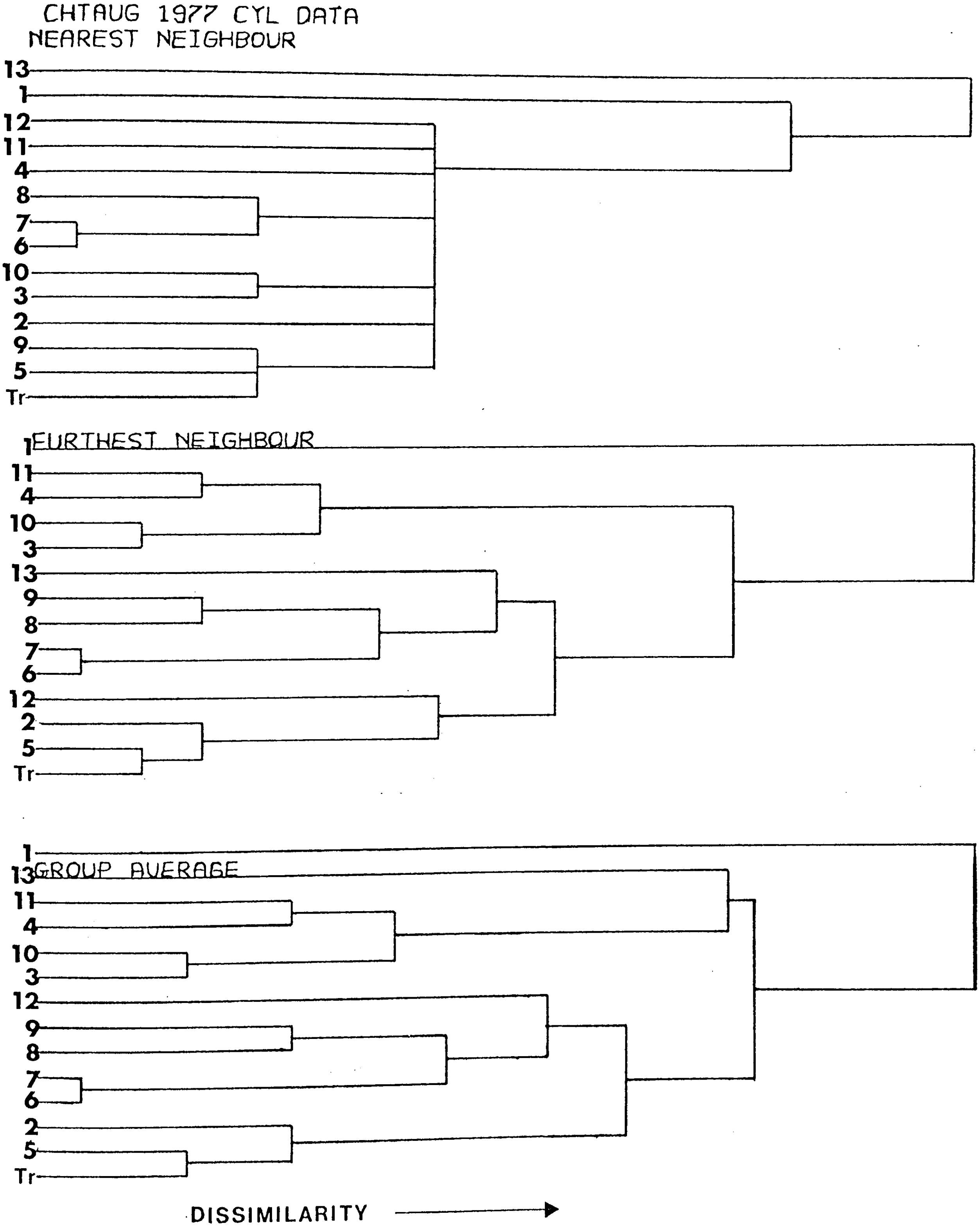
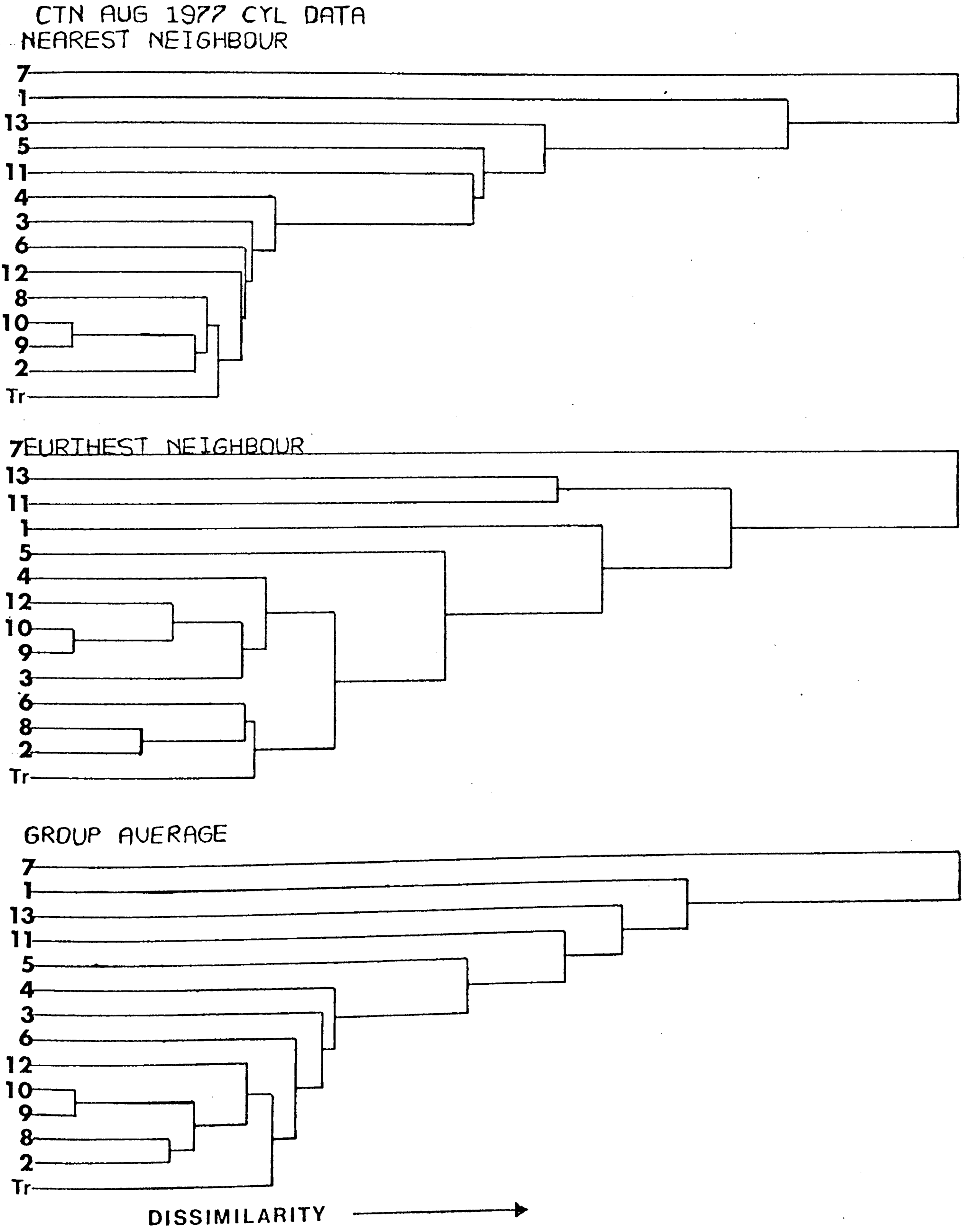


FIG.5.7d DENDROGRAMS SHOWING FAUNISTIC RELATIONSHIPS BETWEEN STATIONS ON THE RIVER CHURNET , AUGUST 1977 , USING QUANTITATIVE DATA.

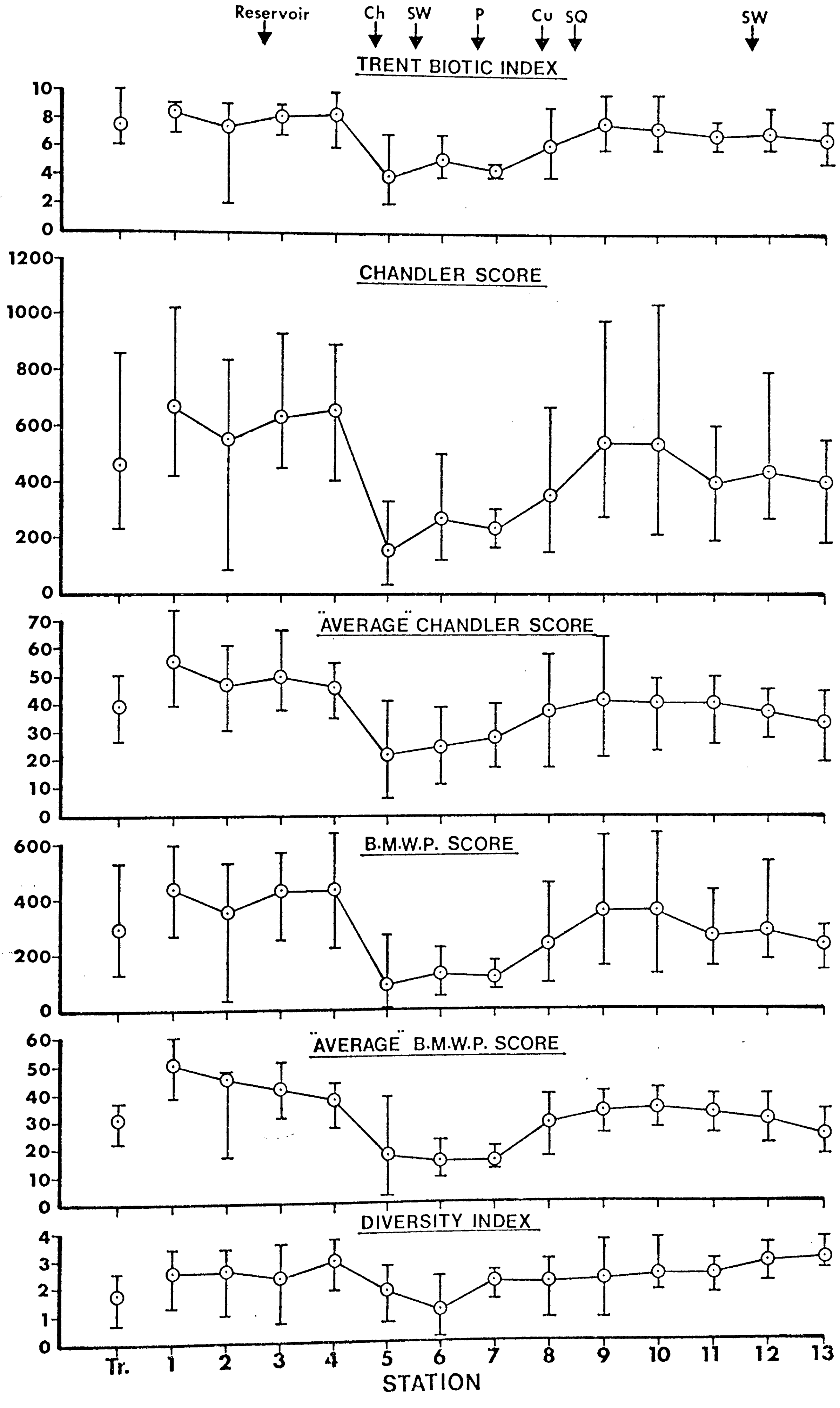


for each month and data level shows that the overall grouping of the stations remains similar for each method. This emphasises the need to detect consistent trends in groupings when the analyses are performed on the same data by a variety of clustering methods.

The May dendrograms using binary and quantitative data generally do not appear to show clear clustering of stations into the different ranges of water quality as assessed by physico-chemical analysis and macroinvertebrate distribution patterns. The closest relationships using numerical data, mainly occurred between stations at Oakamoor (St. 9) and Alton (St. 10) and Denstone (St. 11) and Rocester (St. 12) in the lower section of the river. In August, closer similarity relationships were observed between stations of similar water quality thus forming distinctive dendrogram patterns. It would appear therefore that although this treatment of the results showed associations between the data, seasonal temporal variation would be responsible for significant changes in the types of dendrograms produced, thus affecting the interpretation of water quality and pollution effects on benthic communities.

The responses of the different taxa to the discharges to the river are reflected in the data when processed to produce indexes and scores (Fig. 5.8). When considering the known discharges (Fig. 5.1), the chemical data (Fig. 5.2 and Table 5.2) and the responses of individual taxa (Figs. 5.4 - 5.6), the general pollutional condition along the length of the river can be deduced. The upper reach (Stations 1 - 4) represents a good quality non-polluted river. The discharge from the chemical works, sewage works and probably the paper works, markedly reduces the quality of the river (Stations 5 - 8); this later recovers

FIG.5.8 COMPARISION OF INDEXES AND SCORES(ANNUAL MEAN AND RANGE)IN THE RIVER CHURNET,SHOWING RESPONSES TO DIFFERENT DISCHARGES(1977-1978).



in the lower reaches but is more eutrophic than upstream of the discharges. The fluctuations in the indexes and scores can be considered in respect of this pollutional pattern along the length of the river.

The headstream tributary stream (Tr.) although of good water quality, was a typical mountainous stream where the arduous physical environment resulted in a low species diversity and consequent lowered scores. The four stations upstream of Leek showed similar annual means, reflected by all scores and indexes, with values substantially higher than the tributary stream.

The Trent Biotic Index, Chandler Score and B.M.W.P. Score all reflected the general pollutional condition along the length of the river. All three systems produced results which followed similar trends, decreasing markedly at Station 5 below the chemical works, recovering slightly below the sewage works (Station 6) and decreasing slightly again below the paper mill works effluent (Station 7). There then followed a gradual recovery to Station 9 with a small decline along the rest of the river. 'Averaging' the scores produced slightly different spatial patterns. The 'averaged' Chandler score showed a gradual recovery from Stations 5 to 9 with no response to the sewage effluent or the paper mill effluent. The 'averaged' B.M.W.P. score showed a slight deterioration below the sewage effluent with recovery taking place up to Station 8. Both 'averaged' scores showed a significant reduction along the upper stretch of the river from Stations 1 to 4. Such a decrease in the 'averaged' scores may have been accounted for by the increase in the numbers of taxa along this stretch of the river associated with the more suitable physical environment (Fig. 5.3). The Wilhm and Dorris (1968) diversity index exhibited a somewhat different detailed pattern, in that there was a continual decline below the sewage effluent followed by an immediate recovery at Station 7 and a more gradual recovery over the remainder of the river. The marked decrease at Station 6, not reflected by the other index systems other than the 'averaged' B.M.W.P. score, was due to the large populations of A. aquaticus rather than to a decrease in numbers of taxa present, which in fact increased (Fig. 5.3).

All the scores and indexes showed marked seasonal fluctuations due to marked changes in the seasonal incidence of individual taxa, spatial contagious distributions, as well as changes in the pollutional conditions of the effluents. In the upper non-polluted reaches (Stations 1 to 4) the scores were usually suppressed during the summer months of July to September, but at the other stations no regular seasonal patterns emerged. It is clear that, despite some modification by other factors, pollution was the dominant factor in determining the composition of downstream communities in the Churnet.

5.6 Discussion

Except for a single survey conducted by Pentelow during May, 1938, (Butcher, 1946) no significant published investigation of the invertebrate fauna of the River Churnet has been undertaken. The results of this investigation show that the water quality and pollutional status of the Churnet has improved significantly during the last 40 years, particularly downstream of the copper processing works at Froghall, where there is now a well established macroinvertebrate community.

From the present survey it is clear that in the River Churnet there is a dramatic difference in the macroinvertebrate communities upstream and downstream of the chemical works (Station 5) at Leek, with increasing numbers of worms (Lumbriculidae) and a change in the chironomid species and then a sudden change in the community structure downsteam of the sewage outfall. Although most of the pollution sensitive species present in the headwaters do not return to the river in the lower sections, some species such as I. grammatica, E. ignita, R. dorsalis and Hydropsyche spp. have been able to re-establish themselves in the recovery zone. One explanation for the absence of many stoneflies, mayflies and caddisflies in the lower waters is the lack of a stable, hard substratum in these downstream waters.

The results of this surveillance exercise clearly demonstrate the sensitivity of the riffle communities to different discharges, different taxa responding in different ways. A knowledge of the ecology of these taxa and their responses to different types of pollution enables the basic surveillance data to be interpreted directly to evaluate the pollutional condition of the river to detect any changes.

Processing the data to produce indexes and scores, produces a general pictorial representation of the pollutional conditions. Different systems, however, give differences in detail in relation to specific discharges. This is probably accounted for by the different aspects of the community structure which are made use of in the different systems, e.g. the presence or absence of individual taxa, the numbers of taxa present, the populations or relative abundance of taxa present (Hawkes, 1977). Since these features of the community structure are affected differentially by different types of discharge, all the indexes and scores are not likely to respond in the same way to different types of discharge. It follows therefore that no one system of expressing biological surveillance information will be sensitive to the many ecological effects of different pollutants.

As reviewed in Chapter 2, many attempts have been made to compare several of the indexes and scores described above. However, indexes based on the ratios of readily recognisable taxa could prove to be a simple yet effective way of following subtle changes in water quality as one species replaces another at the point of overlap of their environmental tolerances. Although there are few examples of such a technique, Hawkes and Davies (1971) have suggested that the ratio of <u>Gammarus</u> to <u>Asellus</u> may prove a useful indicator of organic enrichment. Most indicator systems of assessing water quality are based on the fact that different species and communities are associated with different degrees of degradation of organic matter. By using biological indicators rather than abusing their use, one could conceivably develop new approaches and indexes, without contributing additional confusion to the wealth of indexes and modification of established indexes produced to date. Other workers (Brinkhurst, 1966) have proposed that the number of individual species of tubificid present, combined with the proportion of <u>Limnodrilus hoffmeisteri</u> to all other species, may also be a useful index, since the abundance of this species increases with increase in organic pollution.

Using the types of tolerance ranges suggested by Hawkes (1978) for taxa associated with organic pollution, the following ratios derived from the River Churnet data could possibly be incorporated into an index for assessing water quality and pollution:

Gammarus : *Asellus*
Glossiphonia : *Erpobdella*
Lumbriculidae : Tubificidae
Perlodidae : Nemouridae
Heptageniidae : *Baetis*
Rhyacophila : *Hydropsyche*
Orthocladiinae : Chironomini
Brillia modesta : *Brillia longifurca*
Ancylus : *Lymnaea*

i.e. $$\frac{\text{Numbers of One Species Intolerant Organic Pollution}}{\text{Numbers of One Species Tolerant Organic Pollution}}$$

Such a system at present, however, would only be possible using a limited number of taxa. Until more information is available on the life-cycles of different species and the tolerance limits of all stages of the life cycle of each species to parameters such as dissolved oxygen and ammonia concentration and the complex properties of the bottom substratum, it is unwise to place any rigid quantitative values on such observations.

In using riffle communities for water quality monitoring, it is necessary to standardise as far as practicable the other ecological factors involved. Riffles although having certain characteristics in common - the presence of an eroding substratum and shallow fast-flowing water - may have other features differing. The use of a cylinder sampler, as used in this exercise, enables the sampling of specific biotopes, thus facilitating the comparison of similar biotopes at the different stations. Different riffle sites may also provide other different biotopes, e.g. weed beds and mud banks. Sampling by the heel-kick method with a hand-net it is more difficult to ensure the sample is derived from the same biotope. With this type of sampling therefore, the data generated although probably containing a higher number of species, reflect more the number of different biotopes sampled at a particular site.

From this survey, it is recommended that for biological surveillance purposes, riffle benthic invertebrate communities should be sampled directly by using a quantitative quadrat sampler such as the cylinder sampler described in Chapter 3. The application of this method to the intensive study on the River Churnet confirms the sensitivity of riffle communities to changes in water quality and therefore their use in biological surveillance.

For national survey purposes the use of the B.M.W.P. Score is recommended since it adequately reflects subtle changes in organic pollution and is easy to use and apply. For more intense surveys, the Chandler Score, as illustrated by many authors (Balloch et al., 1976; Cook, 1976), appears to be the most sensitive to changes in water quality, but in practice the Score System is more demanding in terms of effort in that quantitative samples are called for and the abundance of the individual taxa has to be assessed.

CHAPTER 6

STUDY ON THE INVASION OF RIFFLE COMMUNITIES IN THE UPPER TRENT AND TRIBUTARIES BY GAMMARUS TIGRINUS SEXTON

6.1 Introduction

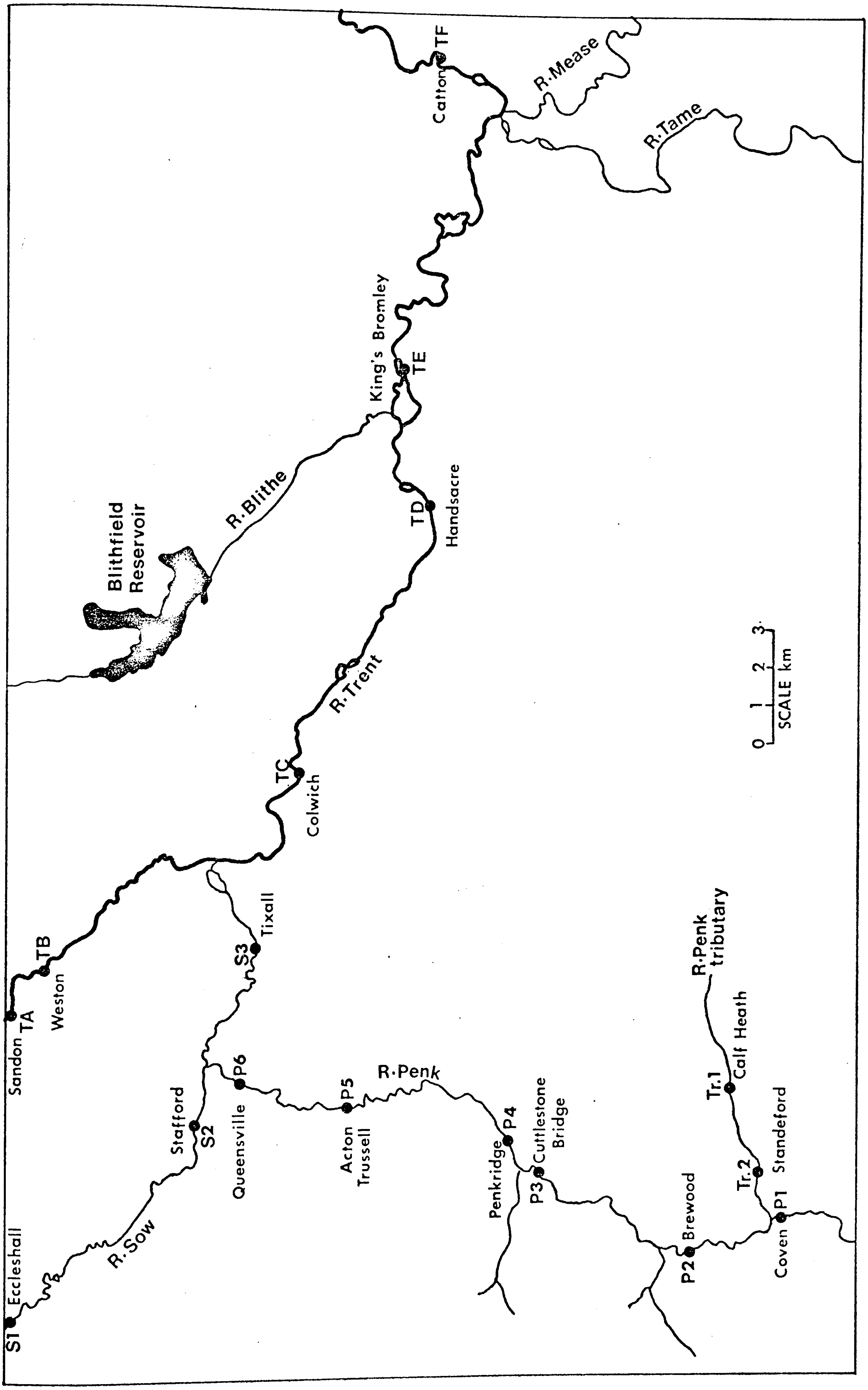
According to Hynes (1970) one interesting feature about inland saline waters is that the ordinary brackish-water coastal animals never seem to invade them. Although there are few records of such invasions in the literature, the scattered examples observed are unique in that the species in question have successfully survived, competed and spread in their new environment. Such examples are illustrated by the sudden spread of the introduced North American amphipod Gammarus tigrinus Sexton in England (Hynes, 1955b) and its immediately successful introduction into the River Werra, Germany (Schmitz, 1960).

During 1976 an extensive biological monitoring survey of benthic macroinvertebrates inhabiting riffles of the upper Trent in the proximity of its confluence with the River Tame revealed the presence of Gammarus tigrinus at King's Bromley (SK 126176) upstream of the River Tame. This species had not been recorded previously from this reach of the River Trent (Davies, 1971) and consequently its distribution in this area of the Trent system was studied in view of the recent observations by Holland (1976b).

6.2 Methods

Gammarus tigrinus was initially obtained from King's Bromley underneath stones and amongst Potamogeton and Ranunculus in the main flow of the river. Initial identification was based on the presence of very distinct dark bands on the thoracic and epimeral segments.

FIG.6.1 MAP OF UPPER TRENT AND TRIBUTARIES SHOWING POSITION OF SAMPLING STATIONS USED TO ESTABLISH DISTRIBUTION OF *GAMMARUS TIGRINUS*.



Confirmation of the species was made using the key of Gledhill et al. (1976), the description by Sexton and Cooper (1939) and consultation with Dr. D. W. Sutcliffe of the Freshwater Biological Association.

The objective of the survey was to establish the exact range of this species in the Trent catchment area and to relate its distribution with known salinity concentrations, interspecific competition with other malacostracan species and tolerance to major putrescible organic discharges.

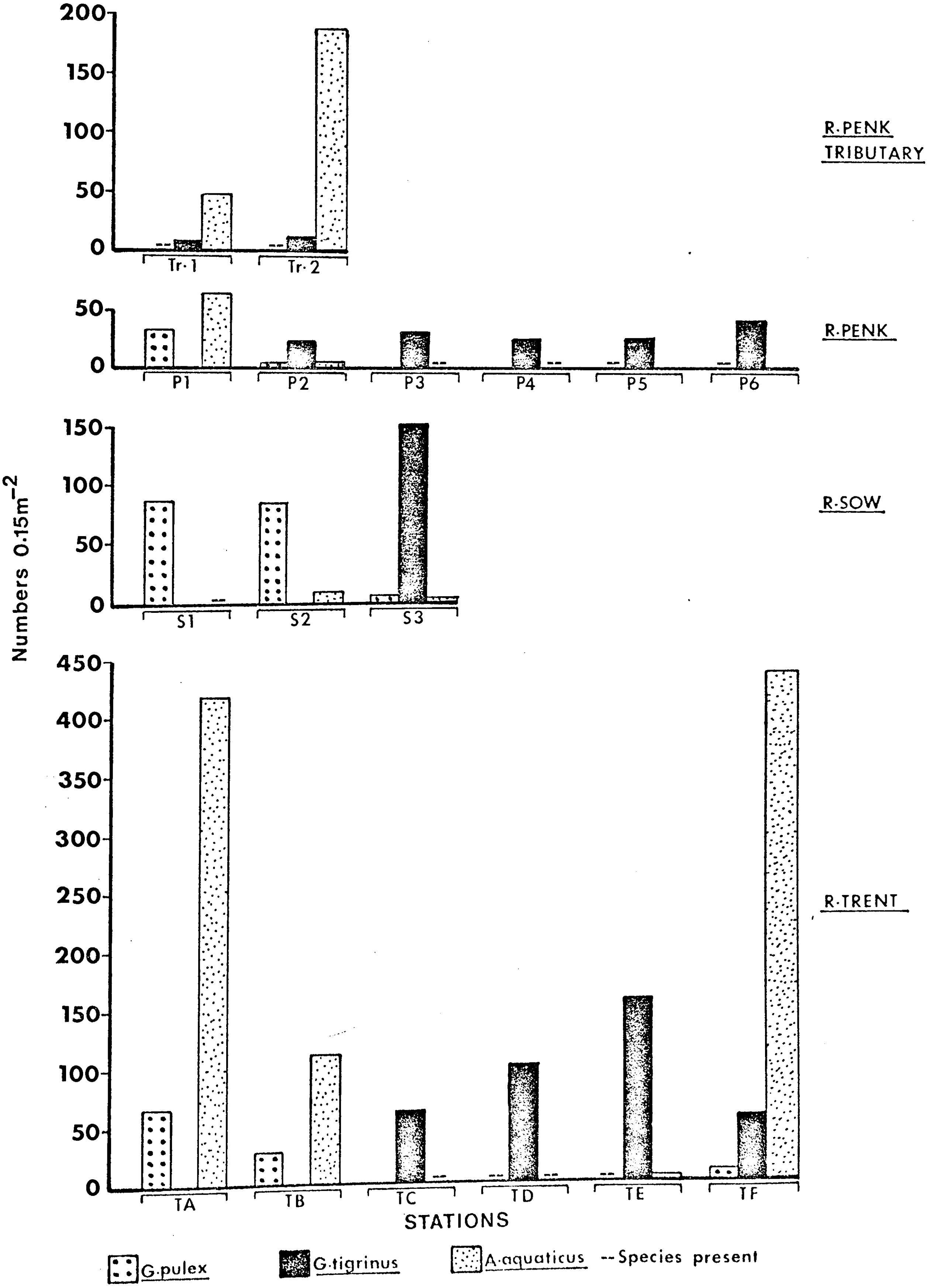
17 stations were selected, situated on the main river and tributaries (Figure 6.1), although 12 were only sampled on a regular basis. 3 Aston cylinder samples (bulked) were taken monthly throughout the spring and summer 1978, representing an area of 0.15 m^{-2} per month. Water samples were collected on each sampling visit and subjected to physico-chemical analysis as described in Chapter 3. The sampling stations studied in this investigation are presented in Table 6.1.

6.3 Results

The detailed macroinvertebrate data generated from this study are given in Annexe 6, Tables 110 - 114.

The distribution of *G. tigrinus* in relation to *G. pulex* and *A. aquaticus* as revealed by the five monthly surveys is shown in Figure 6.2. It can be seen that the distribution of *G. tigrinus* appears to originate from a small tributary of the River Penk, downstream of Coven. This tributary starts in the coal-mining foothills of Cannock, Staffordshire, and flows under the Stafford-Worcester canal before joining the River Penk at Standeford. Although *G. tigrinus* was not abundant in the tributary, it is apparent

FIG.6.2 DISTRIBUTION OF GAMMARUS TIGRINUS, G.PULEX AND ASELLUS AQUATICUS IN THE UPPER TRENT AND TRIBUTARIES (1978).



that this species has spread from the area and colonised the River Penk downstream of this tributary to the River Sow, the River Sow downstream of the River Penk to the River Trent and in the River Trent downstream of the River Sow to the entry of the River Tame. In most cases G. tigrinus typically replaced the indigenous populations of G. pulex which were mainly found in the upstream zones. Chloride concentrations of the River Trent and associated tributaries shown in Figure 6.3, suggest that salinity is a major factor accounting for the distribution of G. tigrinus, although current velocity and deteriorations in water quality are of equal importance. From this data one could conclude that G. tigrinus tolerates a salinity range of 100 - 420 mg. l^{-1} , while G. pulex withstands a range of 34 - 140 mg. l^{-1}. Since the distribution of brackish-water gammarids in the area is being described for the first time, full locality details are given in Table 1.

The relative abundance of G. tigrinus in relation to the other Crustacea - G. pulex and A. aquaticus at the 17 different stations studied, for the five months of the study period are shown in Figure 6.4. In the small tributary stream of the River Penk which appeared to be the main source of the high salinity (Figure 6.3), G. tigrinus was present in low densities and not abundant (Tr. 1 and Tr. 2), Asellus being the dominant crustacea with G. pulex occasionally present. The stream was well oxygenated (D.O. $>$ 12 mg. l^{-1}) and had a low oxygen demand (BOD $<$ 1.2 mg. l^{-1}) with low ammonia levels (0.5 mg. l^{-1}). In the River Penk above this tributary at Coven (P1), G. pulex and A. aquaticus were both common, while G. tigrinus was not recorded. The water quality at this station was similar to

FIG.6.3 SALINITY CONCENTRATIONS OF THE UPPER TRENT AND TRIBUTARIES (mg.Cl.l^{-1}).

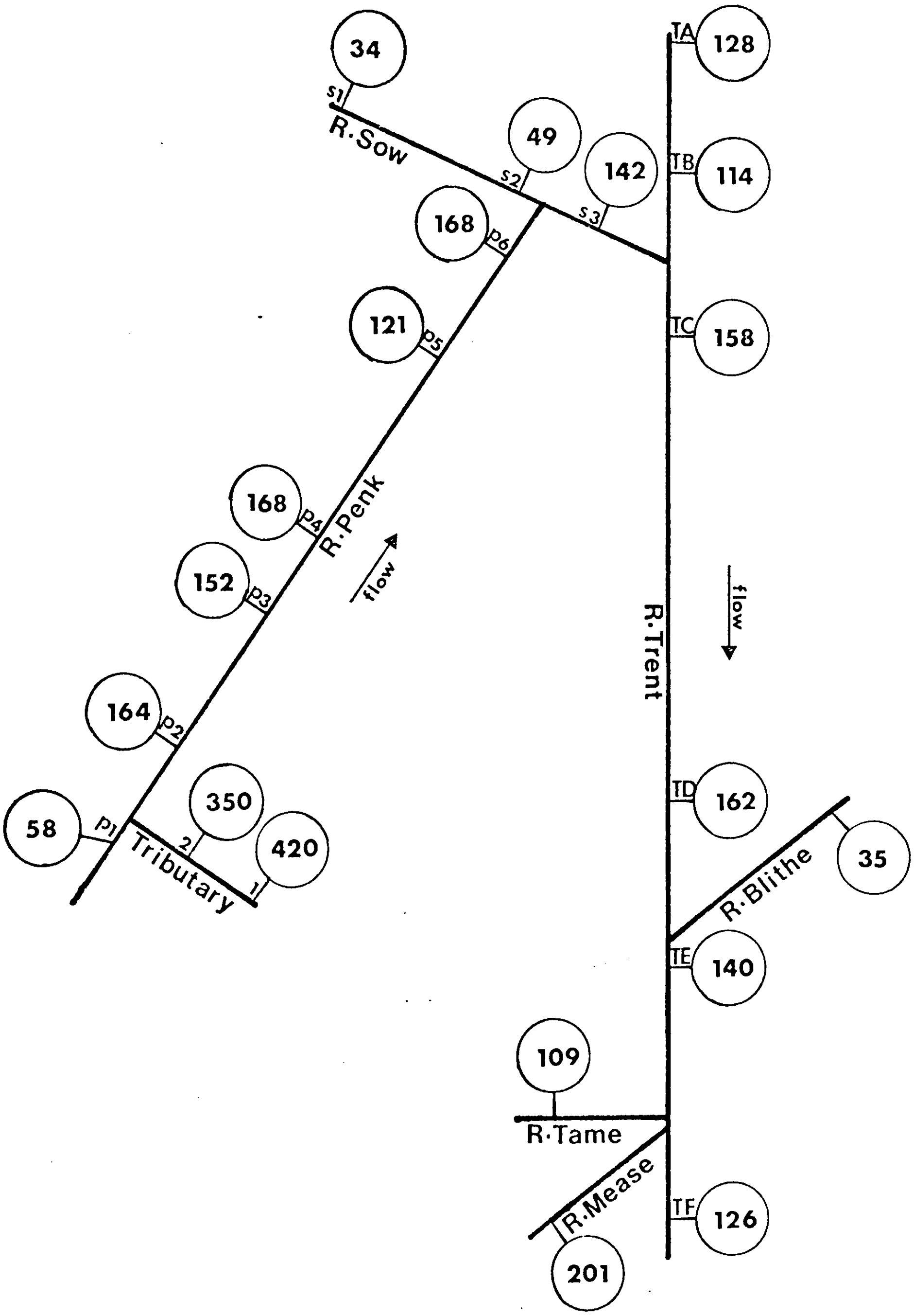
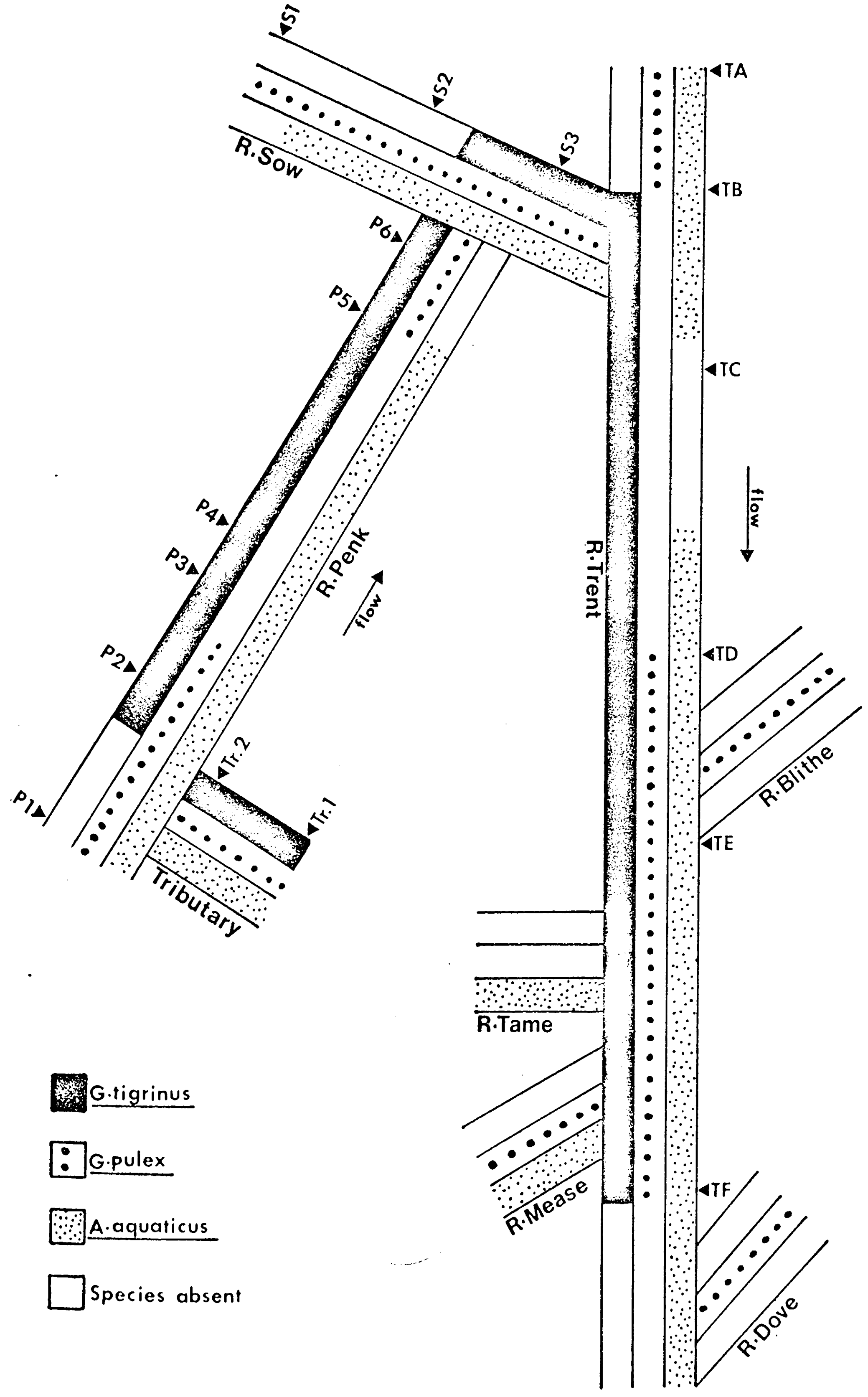


Table 6.1 Distribution of brackish-water *G. tigrinus* and fresh-water Malacostraca in the River Trent and tributaries

(* indicate *G. tigrinus* present)

River/Site		Locality	Grid Ref.	chloride (mg.l^{-1} as cl.) Mean and range
TRENT	TA	Sandon	SJ 946289	128
	TB	Weston	SJ 968271	114 (102 - 140)
	TC	Colwich	SK 019204 *	158 (148 - 173)
	TD	Handsacre	SK 093167 *	162 (146 - 188)
	TE	King's Bromley	SK 126176 *	140 (124 - 156)
	TF	Catton	SK 205155 *	126 (80 - 178)
SOW	S1	Eccleshall	SJ 835296	34
	S2	Stafford	SJ 929229	49 (46 - 52)
	S3	Tixall	SJ 975215 *	142 (122 - 186)
PENK	P1	Coven	SJ 903072	58 (56 - 60)
	P2	Brewood	SJ 895093 *	164 (158 - 170)
	P3	Cuttlestone Brd.	SJ 915138 *	152
	P4	Penkridge	SJ 923145 *	168
	P5	Acton Trussell	SJ 932190 *	121 (92 - 150)
	P6	Queensville	SJ 975215 *	168
TRIB/PENK	Tr.1	Calf Heath	SJ 937084 *	420
	Tr.2	Standeford	SJ 914077 *	350 (336 - 364)

FIG.6.4 RELATIVE ABUNDANCE OF G.TRIGRINUS, G.PULEX AND A.AQUATICUS AT DIFFERENT STATIONS ON THE RIVER TRENT AND TRIBUTARIES.



the tributary, other than salinity levels, and supported a fairly diverse fauna consisting mainly of tubificids, chironomids and bivalve molluscs. Below the entry of the tributary, where salinity levels increased, G. tigrinus became the dominant Crustacea (P2 - P6), with numbers increasing as one moved downstream. The water quality showed no significant signs of deterioration and maintained a similar diverse community as upstream, with the addition of Hydropsyche angustipennis, Hydroptila tineoides, Limnephilidae and riffle beetles Elmis aenea. Towards the lower end of the River Penk it is worth noting that the substratum changed to a silty depositing type compared to an eroding type upstream. It is apparent that this has led to an increase in the percentage cover of submerged aquatic macrophytes, e.g. Callitriche spp. and Potamogeton spp.

In the River Sow below its confluence with the River Penk, G. tigrinus replaced G. pulex as the dominant Crustacea (S3). Again the confluence of the River Sow with the River Penk resulted in minor changes in water quality, apart from salinity and it is significant that a similar succession of species composition occurred, repetitious of the Upper Penk.

In the River Trent above the entry of the River Sow (TA and TB) both A. aquaticus and G. pulex were common with the former dominant. Generally the water quality was poor compared to the Penk and Sow, representing mildly polluted conditions as a result of effluents from Stoke-on-Trent upstream. As a comparison, the following mean values were recorded: D.O. 8.8 mg. l^{-1}, BOD 2.4 mg. l^{-1} and ammonia 1.7 mg. l^{-1}. Below the River Sow where salinity levels increased, with D.O. 9.5 mg. l^{-1}, BOD 2.0 mg. l^{-1} and ammonia 1.0 mg. l^{-1}, G. tigrinus became the dominant species while the other two

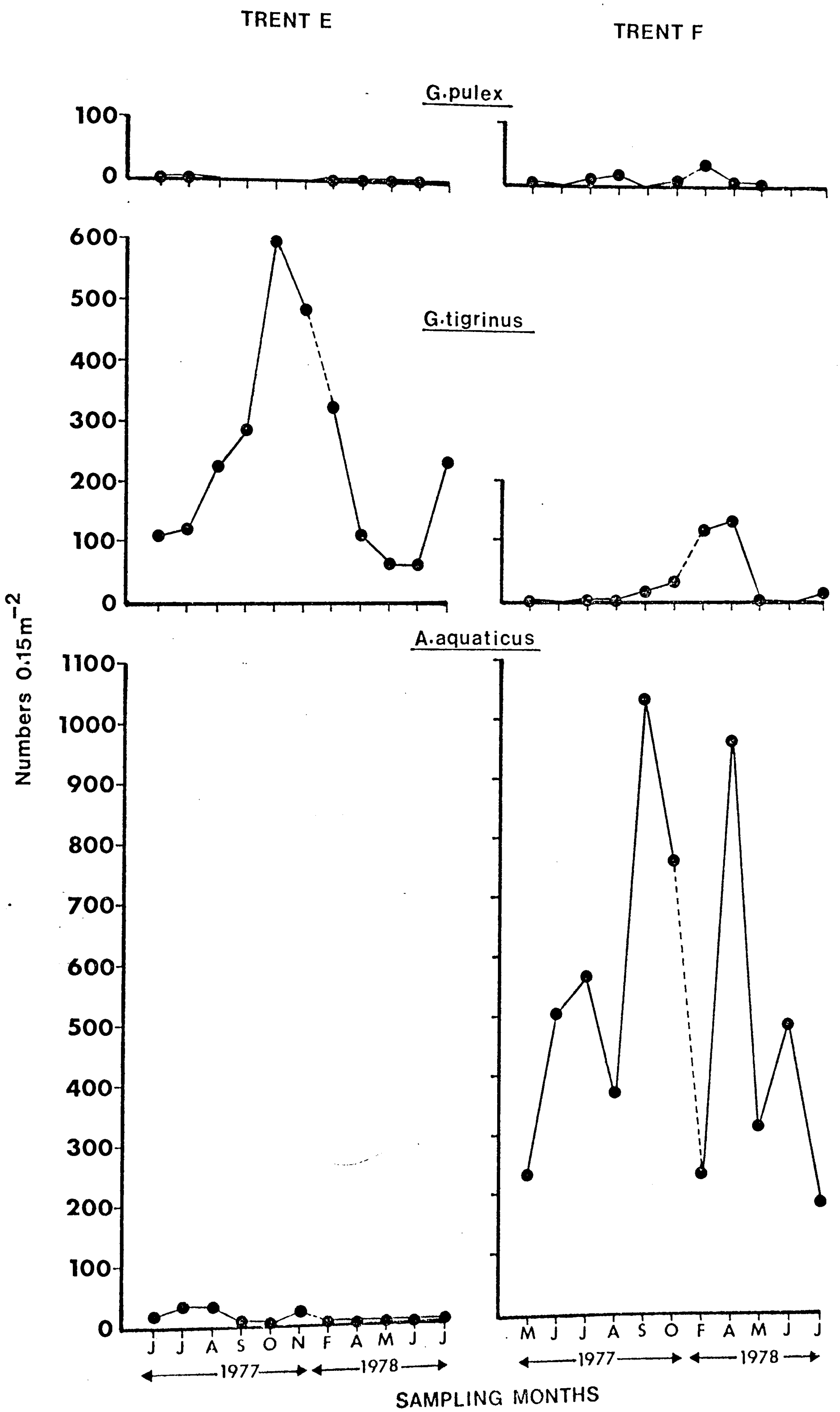
Crustacea were present in markedly reduced numbers (TC, TD, TE). In a similar fashion to the River Penk, the numbers of G. tigrinus increased as one moved downstream, associated with increased macrophytic cover and sediment accumulation. The overall species diversity also improved in the lower sections with the addition of Gastropod molluscs, particularly Valvata macrostoma, Bithynia tentaculata, Potomopyrgus jenkinsi and Ancylus fluviatilis. At station TF, however, below the entry of the polluted River Tame, although G. tigrinus persisted in reduced numbers, A. aquaticus clearly became dominant and there was a slight increase in the numbers of G. pulex, although these may have been due to drift from the River Mease. It is important to note that as a consequence of the River Tame, the water quality in the River Trent often resulted in levels similar to the following being recorded:

D.O. 6.7 mg. l^{-1}, BOD 20.3 mg. l^{-1}, ammonia 4.5 mg. l^{-1}.

A drastic reduction in the species diversity also led to an increase in the numbers of tubificids and Erpobdella octoculata. These results besides showing the relationship between the distribution of G. tigrinus and salinity as well as water quality, suggest there are interspecific competitive effects between the three Crustacea in this situation.

An examination of the seasonal relative abundance of these Crustacea above and below the confluence of the River Tame further exemplifies the competitive nature of these species. Where G. tigrinus is found in large numbers, one only finds a relatively minor population of G. pulex and A. aquaticus. Although low numbers of G. tigrinus are found in the spring, the reproductive cycle of the other species does not allow them to exploit the formers' dominance (Figure 6.5).

FIG.6.5 RELATIVE SEASONAL ABUNDANCE OF G.PULEX, G.TIGRINUS AND A.AQUATICUS AT TWO STATIONS ON THE RIVER TRENT, ABOVE(E) AND BELOW(F) THE ENTRY OF THE RIVER TAME.



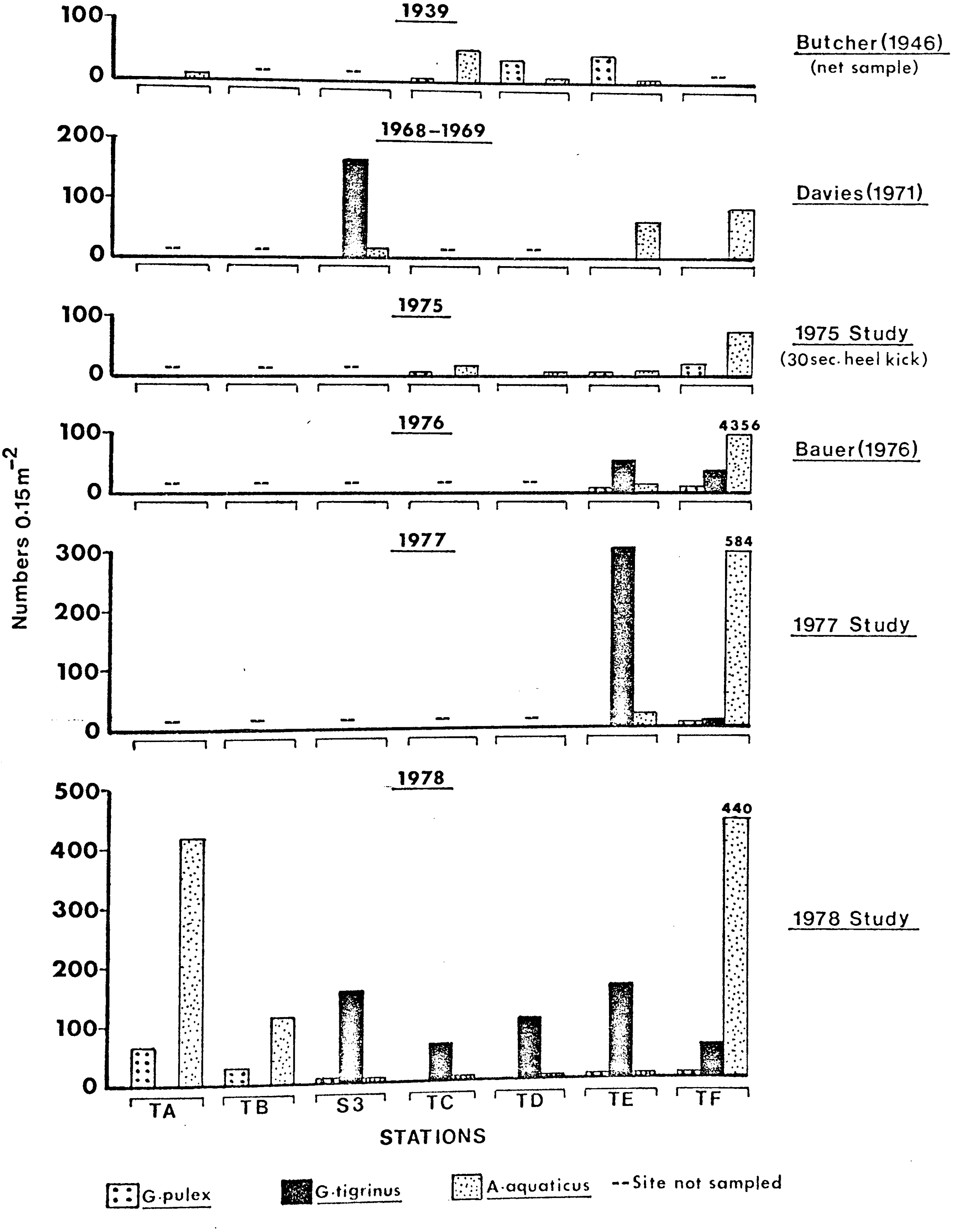
Above the Tame, the highest densities of G. tigrinus appear to be present in the late autumn, with low numbers recorded in spring/ early summer. Below the Tame, the reverse is found with the maximum density available in early spring. It is highly probable that this reflects a false situation whereby large numbers of G. tigrinus have been flushed downstream due to the high flows recorded during winter. It is likely in reality that only a small population consisting of the most tolerant individuals can survive in the relatively poor water quality encountered downstream. In this instance A. aquaticus takes over dominance, with the largest population occurring in late summer.

6.4 Discussion

Since G. tigrinus is only a recent immigrant into this country, it has become a most interesting case-study for the dispersal of a species, because its progress can be measured over the comparatively short time interval of 48 years (Sexton and Cooper, 1939; Hynes, 1955b; Holland, 1976b).

Reference to past records in the West Midlands (Figure 6.6) suggest that G. tigrinus is a relatively recent invader of this stretch of the River Trent. Although the records are not complete for all the stations used in the study, Butcher (1946) did not record this species from stations TA, TC, TD, TE which he used in his studies. The upper Trent during this period was seriously polluted from the Pottery towns around Stoke-on-Trent, ecologically typified by tubificids and chironomids. In the re-purified recovery zone (TD, TE), G. pulex was the dominant species with A. aquaticus detected in low numbers.

FIG.6.6 THE OCCURRENCE OF G.TIGRINUS IN THE UPPER TRENT AND TRIBUTARIES (1939-1978).



Evidence for the existence of G. tigrinus in the vicinity of the River Trent is provided by Sutcliffe (1968) who collected specimens from a small tributary of the River Penk at Coven, presumably the tributary examined in this study. Apparently G. tigrinus was readily available and common in 1962, but less so in 1963, with only a few specimens found in 1964. Following this note by Sutcliffe, more positive evidence of invasion is provided by Davies (1971) who recorded it in the River Sow (S3) but not in the Trent (TE and TF). Subsequent records up to 1975 did not record it from the Trent (TC, TD, TE, TF), although G. pulex was present in low numbers. During 1976, however, its presence in the Trent at stations TE and TF was revealed by Bauer (1976) and this was confirmed by subsequent surveys. It may be significant that this was at the time of an extremely dry summer with low river flows. Although the upper stations of the Trent to the entry of the River Sow were not examined in 1976, it is probable that G. tigrinus would have been recorded.

Hynes (1955b) commenting on the anomalous distribution of G. tigrinus in terms of competition with G. pulex suggests that a facultative freshwater organism is unable to maintain itself in the presence of a strict freshwater organism. The results of this study suggest that G. tigrinus can successfully compete with G. pulex to the extent of almost complete dominance under changed conditions.

Previous considerations of the life cycle of G. tigrinus have shown it to be a rapid growing and breeding species (Hynes, 1955a; Chambers, 1977). According to Chambers (1977) the females reach sexual maturity in about four weeks and the populations have a rapid turnover rate. Hynes (1955a) observing temperature as a factor affecting sexual maturity recorded growth rates up to maturity taking six weeks during a normal summer period. *G. pulex* on the other hand required about three months to reach a similar stage. *G. tigrinus* females were also able to breed at a size 2 mm. smaller than *G. pulex*. Consequently as pointed out by Chambers (1977), the combination of rapid growth rate, early onset of sexual maturity and high fecundity, for populations of a similar size, have meant that *G. tigrinus* has a greater reproductive potential than *G. pulex*. This ability to breed quickly has enabled it to compete with and largely replace the former gammarid fauna in a short time. Although salinity is probably one of the major factors restricting the distribution of *G. pulex*, interspecific competition with *G. tigrinus* cannot be overlooked.

Finally, the invasion of a new species as exemplified here by *G. tigrinus* needs to be taken into account when using macroinvertebrate communities in biological surveillance work. Besides adding to the species list such invaders may through interspecific competition change the relative abundance of other species. In such circumstances biological surveillance systems, in terms of data interpretation, could well be affected.

REFERENCES

Albrecht, M. L. (1953) Die plane und andere flamingbach. Z. Fisch. (N.F.) 1, 389 - 476.

Albrecht, M. L. (1959) Die quantitative Untersuchung der Bodenfauna fliessender Gewasser (Untersuchungsmethoden und Arbeitsergebnisse). Z. Fisch. (N.F.) 8, 481 - 550.

Allan, I. R. H. (1952) A hand-operated quantitative grab for sampling river beds. J. Anim. Ecol. 21, 159 - 160.

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

Allen, K. R. (1940) Studies on the biology of the early stages of the salmon (Salmo salar). I. Growth in the River Eden. J. Anim. Ecol. 9, 1 - 23.

Ambuhl, H. (1959) Die Bedentung der Stromung als okologischer Faktor. Schweiz. Z. Hydrol. 21, 133 - 264.

Anderson, J. B. and Mason, W. T. (1968) A comparison of benthic macroinvertebrates collected by dredge and basket sampler. J. Wat. Pollut. Control Fed. 40 (2), 252 - 259.

Armitage, P. D. (1976) A quantitative study of the invertebrate fauna of the River Tees below Cow Green reservoir. Freshwat. Biol. 6, 229 - 240.

Armitage, P. D. (1977) Development of the macroinvertebrate fauna of Cow Green reservoir (Upper Teesdale) in the first five years of its existence. Freshwat. Biol. 7, 441 - 454.

Arthur, J. W. and Horning, W. B. (1969) The use of artificial substrates in pollution surveys. Am.Midl.Nat. 82, 83 - 89.

Aston, R. J. (1973a) Field and experimental studies on the effects of a power station effluent on Tubificidae (Oligochaeta, Annelida). Hydrobiologia 42 (2 - 3), 225 - 242.

Aston, R. J. (1973b) Tubificids and water quality: a review. Environ. Pollut. 5, 1 - 10.

Balfour-Browne, F. (1940) British water beetles. Vol. 1. London. Ray Society, 375 pp.

Balfour-Browne, F. (1950) British water beetles. Vol. 2. London. Ray Society, 394 pp.

Balfour-Browne, F. (1958) British water beetles. Vol. 3. London. Ray Society, 210 pp.

Balloch, D., Davies, C. E. and Jones, F.H. (1976) Biological assessment of water quality in three British rivers: The North Esk (Scotland), The Ivel (England) and the Taff (Wales).Wat. Pollut. Control 75 (1), 92 - 114.

Bartsch, A. F. (1948) Biological aspects of stream pollution. Sewage Works Journal 20, 292 - 302.

Bartsch, A. F. and Ingram, W. M. (1966) Biological analysis of water pollution in North America. Verh. int. Verein. theor. angew. Limnol. 16, 786 - 800.

Bassindale, R. (1946) New records of Eucrangonyx gracilis S. I. Sexton and Gammarus tigrinus Sexton. Nature 157, 485.

Bauer, M. J. (1976) A comparison of the relative merits of Biopac and bags of slag as a technique for sampling pools. Unpublished M.Sc. Project Report, University of Aston in Birmingham.

Beak, T. W., Griffing, T. C. and Appleby, A. G. (1973) Use of artificial substrate samplers to assess water pollution. Biological methods for the assessment of water quality, ASTM STP 528, American Society for testing and materials, 227 - 241. J. Cairns and K. L. Dickson Eds.

Beck, W. J. (1955) Suggested method for reporting biotic data. Sewage Ind. Wastes 27 (10), 1193 - 1197.

Benfield, E. F.,Hendricks, A. C. and Cairns, J. (1974) Proficiencies of two artificial substrates in collecting stream macroinvertebrates. Hydrobiologia 45 (4), 431 - 440.

Bergersen, E. P. and Galat, D. L. (1975) Coniferous tree bark: a lightweight substitute for limestone rock in barbeque basket macroinvertebrate samplers. Wat. Res. 9, 729 - 731.

Besch, W. (1966) Driftnetzmethode und biologische fliesswassersuntersuchung. Verh. int. Verein. theor. angew. Limnol. 16, 669 - 678.

Besch, W. and Hoffman, W. (1968) La macrobenthos sur des substrats de polyethylene dans les eaux courantes 2. La Steinach, une riviere de la zone a truite. Annls. Limnol. 4, 235 - 263.

Besch, W., Hoffman, W. and Ellenberger, W. (1967) Das makrobenthos auf polyethylene substratum in Fliessgewassern 1. Die Kinzig, ein Fluss der unteren Salmoniden under oberen Barbenzone. Annls. Limnol. 3, 331 - 367.

Bielby, G. H. (1960) The biology of river pollution with reference to the Bristol Avon river board area. J. Inst. Sew. Purif. 3, 298 - 306.

Biglane, K. E. and Lafleur, R. A. (1954) Biological indices of pollution as observed in Louisiana streams. Bill - La. Engng.Sta.43,1-7.

Birkett, L. (1958) A basis for comparing grabs. J. Cons. perm. int. Explot. Mer. 23, 202 - 207.

Bishop, J. E. (1973) Observations on the vertical distribution of the benthos in a Malaysian stream. Freshwat. Biol. 3, 147 - 156.

Boobar, L. R. and Granett, J. (1978) Evaluation of polyethylene samplers for black fly larvae (Diptera : Simuliidae), with particular reference to Maine species. Can.J. Zool. 56, 2245 - 2248.

Bousfield, E. L. (1958) Fresh-water amphipod crustaceans of glaciated North America. Can. Fld. Nat. 72, 55 - 113.

Bournaud, M., Chavanon, G. and Tachet, H. (1978) Colonisation by macroinvertebrates of artificial stony substrates suspended in the River Rhone. Verh. int. Verein. theor. angew. Limnol. 20, 1485 - 1493.

Boycott, A. E. (1936) The habitats of freshwater Mollusca in Britain. J. Anim. Ecol. 5, 116 - 186.

Bray, R. P. (1967) The taxonomy of the larvae and pupae of the British Phryganeidae (Trichoptera). J. Zool. Lond. 153, 223 - 244.

Brindle, A. (1960) The larvae and pupae of the British Tipulinae (Diptera: Tipulidae) Trans. Soc. Br. Ent. 14 (3), 63 - 114.

Brinkhurst, R. O. (1965) Observations on the recovery of a British river from gross organic pollution. Hydrobiologia 25, 9 - 51.

Brinkhurst, R. O. (1966) The Tubificidae (Oligochaeta) of polluted waters. Verh. int. Verein. theor. angew. Limnol. 16, 854 - 859.

Brinkhurst, R. O. (1971) A guide for the identification of British aquatic Oligochaeta. (2nd Edn.) Scient. Publs. Freshwat. biol. Ass. No. 22, 55 pp.

Brinkhurst, R. O., Chua, K. E. and Batoosingh, E. (1969) Modifications in sampling procedures as applied to studies on the bacteria and tubificid oligocheates inhabiting aquatic sediments. J. Fish. Res. Bd. Canada 26 (10), 2581 - 2593.

Brinley, F. J. (1942) Biological studies, Ohio River pollution survey I. Biological zones in a polluted stream. Sewage Works Journal 14, 147 - 159.

Britt, N. W. (1955) New methods for collecting bottom fauna from shoals or rubble bottoms of lakes and streams. Ecology 36, 524 - 525.

Brown, S. R. (1956) A pistol sampler for surface sediments of lake deposits. Ecology 37, 611 - 613.

Bryce, D. (1960) Studies on the larvae of the British Chironomidae (Diptera) with keys to the Chironomidae and Tanypodinae. Trans. Soc. Br. Ent. 14 (2), 19 - 62.

Bryce, D and Hobart, A. (1972) The biology and identification of the larvae of the Chironomidae (Diptera) Introduction and key to sub-families. Entomologist's Gaz. 23, 175 - 217.

Bull, C. J. (1968) A bottom fauna sampler for use in stony streams. Progve. Fish Cult. 30 (2), 119 - 120.

Butcher, R. W. (1946) The biological detection of pollution. J. Inst. Sew. Purif. 2, 92 - 97.

Butcher, R. W. (1959) Biological assessment of river pollution. Proc. Linn. Soc. Lond. 170, 159 - 165.

Butcher, R. W., Longwell, J. and Pentelow, F. T. K. (1937) Survey of the River Tees, 3. The non-tidal reaches, chemical and biological. J.Inst. Sew. Purif. 2, 92 - 102.

Cairns, J. and Dickson, K. L. (1971) A simple method for the biological assessment of the effects of waste discharges on aquatic bottom-dwelling organisms. J. Wat. Pollut. Control Fed. 43, 755 - 772.

Cairns, J., Dahlberg, M. L., Dickson, K. L., Smith, N. and Waller, W. T. (1969) The relationship of fresh-water protozoan communities to the MacArthur-Wilson equilibrium model. Amer. Natur. 103, 439 - 454.

Campbell, M. S. A. (1939) Biological indicators of intensity of stream pollution. Sewage Works Journal 11, 123 - 127.

Carpenter, K. E. (1926) The lead mine as an active agent in river pollution. Ann. appl. Biol. 13, 395 - 401.

Chambers, M. R. (1973) Notes on the gammarid fauna of the Frisian Lake District following the invasion of the alien amphipod Gammarus tigrinus Sexton. Bull. zool. mus. Univ. Amsterdam 3 (1), 1 - 6.

Chambers, M. R. (1977) The population ecology of Gammarus tigrinus Sexton in the reed beds of the Tjeukemeer. Hydrobiologia 53 (2), 155 - 164.

Chandler, J. F. (1970) A biological approach to water quality management. Wat. Pollut. Control 69 (4), 415 - 422.

Chapman, B., Cooke, G. H. and Whitehead, R. (1967) Automated analysis: The determination of Ammoniacal, nitrous and nitric nitrogen in river waters, sewage effluents and trade effluents. Wat. Pollut. Control 66, 185 - 190.

Chernovskii, A. A. (1949) Identification of larvae of the midge family, Tendipedidae. Translation by E. Lees, 1961. National lending library, Boston Spa, Yorkshire. Opred. Faune SSSR 31, 1 - 186.

Chiswell, J. R. (1956) A taxonomic account of the last instar larvae of some British Tipulinae (Diptera: Tipulidae). Trans. R. ent. Soc. Lond. 108, 409 - 484.

Christie, N. and Allen, J. (1972) A self-contained diver-operated quantitative sampler for investigating the macrofauna of soft sediments. Trans. R. Soc. S.Afr. 40 (4), 299 - 307.

Chutter, F. M. (1969) The distribution of some stream invertebrates in relation to current speed. Int. Revue ges. Hydrobiol. Hydrogr. 54 (3), 413 - 422.

Chutter, F. M. (1972a) An empirical biotic index of the quality of water in South African streams and rivers. Wat. Res. 6, 19 - 30.

Chutter, F. M. (1972b) A reappraisal of Needham and Usinger's data on the variability of a stream fauna when sampled with a Surber sampler. Limnol. Oceanogr. 17, 139 - 141.

Chutter, F. M. and Noble, R. G. (1966) The reliability of a method of sampling stream invertebrates. Arch. Hydrobiol. 62, 95 - 103.

Cianficconi, F. and Riatti, M. (1957) Impiego di pietre artificiali per l'analisi quantitativa delle colonizzazioni faunistiche dei fondi potamici. Boll. Pesca Piscic. Idrobiol. 12, 299 - 335.

Clemens, H. P. (1950) Life cycle and ecology of Gammarus fasciatus Say. Contr. Stone Lab. Ohio Univ. 12, 1 - 63.

Coleman, M. J. and Hynes, H. B. N. (1970) The vertical distribution of the invertebrate fauna in the bed of a stream. Limnol. Oceanogr. 15, 31 - 40.

Cook, S. E. K. (1976) Quest for an index of community structure sensitive to water pollution. Envrion Pollut. 11 (4), 269 - 288.

Cover, E. C. and Harrel, R. C. (1978) Sequences of colonisation, diversity, biomass and productivity of macroinvertebrates on artificial substrates in a freshwater canal. Hydrobiologia 59 (1), 81 - 95.

Cranston, P. S. (1975) Corrections and additions to the list of British Chironomidae (Diptera). Entomologist's mon. Mag. 110, 87 - 95.

Cranston, P. S. (1976) Chironomidae. In Kloet, G.S. and Hinks, W. D. A. check list of British Insects. 2nd edition, revised, Volume 2, Part 5. Diptera and Siphonaptera. pp. 12 - 21. Royal Entomological Society, London. 139 pp.

Cranston, P. S. (1977) A provisional key to the genera of the larvae of British Chironomidae. British Museum (Natural History), London.

Crawford, G. I. (1935) Corophium curvispinum, G. O. Sars, Var. devium, Wundsch, in England. Nature, Lond. 136, 685 - 686.

Crawford, G. I. (1937) An amphipod, Eucrangonyx gracilis S. I. Smith, new to Britian. Nature, Lond. 139, 327.

Crisp, D. T. and Gledhill, T. (1970) A quantitative description of the recovery of the bottom fauna in a muddy reach of a mill stream in Southern England after draining and dredging. Arch. Hydrobiol. 67 (4), 502 - 541.

Crossman, J. S. and Cairns, J. (1974) A comparative study between two different artificial substrate samplers and regular sampling techniques. Hydrobiologia 44, 517 - 522.

Crowe, J. M. E. (1970) The use of rock-filled basket samplers to survey the aquatic invertebrates of the Saskatchewan River. Paper presented at 18th annual meeting of the mid-west benthological society, Notre Dame University.

Crowe, J. M. E. (1971) The use of two types of artifical substrates to sample the macro-invertebrates of the Rat river 1970. Paper presented at 19th annual meeting of the mid-west benthological society, Notre Dame University, 24 - 26.

Cummins, K. W. (1962) An evaluation of some techniques for the collection and analysis of benthic samples. With special emphasis on lotic waters. Am. Midl. Nat. 67 (2), 477 - 504.

Czekanowski, J. (1913) Zarys metod statystycznyeh (Die Grundzuge der statischen Metoden) Warsaw.

Davies, L. (1968) A key to the British species of Simuliidae (Diptera) in the larval, pupal and adult stages. Scient. publs. Freshwat. biol. Ass. No. 24, 125 pp.

Davies, L. J. (1971) Some factors influencing the distribution of macro-invertebrates in riffles of organically polluted streams. Unpublished Ph.D. thesis, University of Aston in Birmingham.

Dendy, J. S. (1963) Living food for aquatic animals. Turtox News 41, 258 - 259.

Department of the Environment (1972) Report of River Pollution survey of England and Wales. 1970, H.M.S.O., London.

Department of the Environment (1980) Assessment and presentation of the biological qualities of rivers in Great Britain. Final report of the Biological Monitoring Working Party of Standing Technical Advisory Committee on Water Quality.

Dickson, K. L. and Cairns, J. (1972) The relationship of fresh-water macro-invertebrate communities collected by floating artificial substrates to the MacArthur-Wilson Equilibrium model. Am. Midl. Nat. 88 (1), 68 - 75.

Dickson, K. L., Arnold, J. C. and Cairns, J. (1970) The use of a basket-type artificial substrate sampler for stream pollution monitoring. Assoc. S.E. Biol. Bul. 17, 40.

Dickson, K. L., Cairns, J. and Arnold, J. C. (1971) An evaluation of the use of basket-type artificial substrate for sampling macrobenthic invertebrate organisms. Trans. Amer. Fish. Soc. 100 (9), 553 - 559.

Disney, R. H. L. (1972) Observations on sampling pre-imaginal populations of blackflies (Dipt, Simuliidae) in West Cameroon. Bull. ent. Res. 61, 485 - 503.

Dorier, A. and Vaillant, F. (1954) Observations et experiences relatives a la resistance au courant de divers invertebres aquatiques. Trav. Lab. Hydrobiol. Piscic. Univ. Grenoble 45/46, 9 - 31.

Downing, A. L. (1967) The effects of pollution on the oxygen balance of rivers. Wat. Waste Treatm. J. 11, 360 - 364.

Dunn, D. R. (1961) The bottom fauna of Llyn Tegid (Lake Bala), Merionethshire. J. Anim. Ecol. 30, 267 - 281.

Edington, J. M. (1964) The taxonomy of British polycentropid larvae (Trichoptera) Proc. zool. Soc. Lond. 143 (2), 281 - 300.

Edington, J. M. (1968) Habit preferences in net-spinning caddis larvae with special reference to the influence of water velocity. J. Anim. Ecol. 37, 675 - 692.

Edington, J. M. and Alderson, R. (1973) The taxonomy of British psychomyiid larvae (Trichoptera). Freshwat. Biol. 3, 463 - 478.

Edmondson, W. T. and Winberg, G. G. (1971) A manual on methods for the assessment of secondary productivity in freshwaters. IBP handbook No. 17, Blackwell Scientific Publications, Oxford and Edinburgh, 358 pp.

Edwards, R. W. (1975) A strategy for the prediction and detection of effects of pollution on natural communities. Schweizerische Zeitschrift fur Hydrologie 37 (1), 135 - 143.

Edwards, R. W. and Learner, M. A. (1960) Some factors affecting the oxygen consumption of Asellus. J. exp. Biol. 37 (4), 706 - 718.

Efford, I. E. (1960) A method of studying the vertical distribution of the bottom fauna in shallow waters. Hydrobiologia 16, 288 - 292.

Egglishaw, H. J. (1964) The distributional relationship between the bottom fauna and plant detritus in streams. J. Anim. Ecol. 33, 463 - 476.

Ekman, S. (1911) Neue apparate zur qualitativen und quantitativen erforschung der Bodenfauna der Seen. Int. Revue ges. Hydrobiol. Hydrogr. 3, 553 - 561.

Elgmork, K. (1962) A bottom sampler for soft mud. Hydrobiologia 20, 167 - 172.

Elliott, J. M. (1970) Methods of sampling invertebrate drift in running water. Annls. Limnol. 6, 133 - 159.

Elliott, J. M. (1971) Some methods for the statistical analysis of samples of benthic invertebrates. Scient. Publs. Freshwat. biol. Ass. No. 25, 144 pp.

Elliott, J. M. (1977) A key to the larvae and adults of British freshwater Megaloptera and Neuroptera. Scient. Publs. Freshwat. biol. Ass. No. 35, 52 pp.

Ellis, A. E. (1962) British freshwater bivalve Mollusca. Linnean Society of London Synopses of the British Fauna. No. 13, 92 pp.

Elvins, B. J. (1962) Investigation of the animal population in polluted streams. J. Inst. Sew. Purif. 6, 569.

Finnish IBP - PM Group (1969) Quantitative sampling equipment for the littoral benthos. Int. Revue ges Hydrobiol. Hydrogr. 54, 185 - 193.

Fittkau, E. J. (1962) Die Tanypodinae (Diptera,Chironomidae). Die Tribus Anatopyniini, Macropelopini und Pentaneurini. Abh. Larvalsyst. Insekten 6, 1 - 453.

Fjerdingstad, E. (1964) Pollution of streams estimated by benthonic Phytomicro-organisms: 1. A saprobic system based on communities of organisms and ecological facts. Int. Revue ges. Hydrobiol. Hydrogr. 49 (1), 63 - 131.

Flannagan, J. F. (1970) Efficiencies of various grabs and corers in sampling freshwater benthos. J. Fish. Res. Bd. Can. 27 (10), 1691 - 1700.

Ford, J. B. (1962) The vertical distribution of larval Chironomidae (Diptera) in the mud of a stream. Hydrobiologia 19, 262 - 272.

Ford, J. B. and Hall, R. E. (1958) A grab for quantitative sampling in stream muds. Hydrobiologia 11, 198 - 204.

Fox, H. M. and Taylor, A. E. R. (1955) The tolerance of oxygen by aquatic invertebrates. Proc. Roy. Soc. Lond. B. 143, 214 - 225.

Fraiz, H. (1952/1953) Dauer und Wandel der Lebensgemeinschaften. Schr. Ver. Verbreit. naturw. Kenntnisse, Wien. 93, 27 - 45.

Fries, G. and Tesch, F. W. (1965) Der einfluss des massenvorkommens von Gammarus tigrinus Sexton auf fische und niedere tier welt in der Weser. Archiv. fur Fischerei Wissenschaft 16, 133 - 150.

Frost, S., Huni, A. and Kershaw, W. E. (1971) Evaluation of a kicking technique for sampling stream bottom fauna. Can. J. Zool. 49, 167 - 173.

Fullner, R. W. (1971) A comparison of macroinvertebrates collected by basket and modified multiplate samplers. J. Wat. Pollut. Control Fed. 43 (3), 494 - 499.

Gale, W. F. and Thompson, J. D. (1974) Placement and retrieval of artificial substrate samplers by SCUBA. Progve. Fish Cult. 36 (4), 231 - 233.

Gale, W. F. and Thompson, J. D. (1975) A suction sampler for quantitatively sampling benthos on rocky substrates in rivers. Trans. Amer. Fish Soc. 2, 398 - 405.

Gardner, A. E. (1954) A key to the larvae of the British Odonata. Entomologist's Gaz. 5, 157 - 171, 191 - 223.

Garnett, P. A. and Hunt, R. H. (1965) Two techniques for sampling freshwater habitats. Hydrobiologia 26, 114 - 120.

Gaufin, A. R. and Tarzwell, C. M. (1952) Aquatic invertebrates as indicators of stream pollution. Publ. Hlth. Rep. Wash. 67 (1), 57 - 64.

Gaufin, A. R. and Tarzwell, C. M. (1956) Aquatic macro-invertebrate communities as indicators of organic pollution in Lytle Creek. Sewage Ind. Wastes 28 (7), 906 - 924.

Gerking, S. D. (1957) A method of sampling the littoral macrofauna and its application. Ecology 38, 219 - 226.

Gledhill, T., Sutcliffe, D. W. and Williams, W. D. (1976) A revised key to the British species of Crustacea: Malacostraca occurring in fresh water. Scient. Publs. Freshwat. biol. Ass. No. 32, 71 pp.

Glime, J. M. and Clemons, R. M. (1971) Species diversity of stream insects on Fontinalis spp. compared to diversity on artificial substrates. Ecology 53 (3), 458 - 464.

Goddard, C. I., Goodwin, M. H and Greig, L. A. (1975) The use of artificial substrates in sampling estuarine benthos. Trans. Amer. Fish. Soc. 104 (1), 50 - 52.

Gower, A. M. and Buckland, P. J. (1978) Water quality and the occurrence of Chironomus riparius Meigen (Diptera: Chironomidae) in a stream receiving sewage effluent. Freshwat. Biol. 8, 153 - 164.

Graham, T. R. (1965) A biological index of pollution. Annual report, Lothians Purification Board 16 - 26.

Hales, D. C. (1962) Stream bottom sampling as a research tool. Proc. Utah, Acad. 39, 84 - 91.

Hall, R. O. and Edwards, A. J. (1978) Observations on the settling of Simulium damnosum larvae on artificial substrates, in the Ivory coast. Hydrobiologia 57 (1), 81 - 84.

Harrod, J. J. (1964) The distribution of invertebrates on submerged aquatic plants in a chalk stream. J. Anim. Ecol. 33, 335 - 348.

Harrod, J. J. (1965) Effect of current speed on the fans of the larva of Simulium ornatum. Hydrobiologia 26, 8 - 12.

Hawkes, H. A. (1956) The biological assessment of pollution in Birmingham streams. J. Inst. Munic. Engrs. 82, 452 - 463.

Hawkes, H. A. (1962) Biological aspects of river pollution. In River pollution 2. Causes and Effects. Edt. L. Klein. 311 - 432.

Hawkes, H. A. (1963) Effects of domestic and industrial discharges on the ecology of riffles in Midland streams. Inst. J. Air Wat. Poll. 7, 565 - 583.

Hawkes, H. A. (1974) Water quality: biological considerations. Chemistry and Industry 990 - 1000.

Hawkes, H. A. (1975) River zonation and classification. River Ecology Ed. B. A. Whitton, 312 - 374, Blackwell, Oxford.

Hawkes, H. A. (1977) Biological classification of rivers: conceptual basis and ecological validity. Biological monitoring of inland fisheries. Ed. J. S. Alabaster, 55 - 67, Appl. Sc. Pub. Lond.

Hawkes, H. A. (1978) Invertebrates as indicators of river water quality. International symposium on biological indicators of water quality. University of Newcastle upon Tyne 12 - 15 September, 1978.

Hawkes, H. A. and Davies, L. J. (1971) Some effects of organic enrichment on benthic invertebrate communities in stream riffles. The Scientific management of animal and plant communities for conservation, Eds. E. Duffey and A. S. Watt, 271 - 293, Blackwell, Oxford.

Hellawell, J. M. (1977) Change in natural and managed ecosystems: detection, measurement and assessment. Proc. R. Soc. Lond. B. 197, 31 - 57.

Hellawell, J. M. (1978) Biological surveillance of rivers. Water Research Centre, Medmenham, Bucks. 332 pp.

Henson, E. B. (1965) A cage sampler for collecting aquatic fauna. Turtox News 43, 298 - 299.

Hess. A. D. (1941) New limnological sampling equipment. Spec. Publs. Limnol. Soc. Am. 6, 1 - 5.

Hester, F. E. and Dendy, J. S. (1962) A multiple plate sampler for aquatic macroinvertebrates. Trans. Amer. Fish. Soc. 91 (4), 420 - 421.

Heywood, J. and Edwards, R. W. (1962) Some aspects of the ecology of Potamopyrgus jenkinsi Smith. J. Anim. Ecol. 31, 239 - 250.

Hickin, N. E. (1967) Caddis larvae: Larvae of the British Trichoptera. Hutchinson, London, 476 pp.

Hildrew, A. G. and Morgan, J. C. (1974) The taxonomy of the British Hydropsychidae (Trichoptera). J. Ent. B. 43 (2), 217 - 229.

Hiley, P. D. (1972) The taxonomy of the larvae of the British Sericostomatidae (Trichoptera). Entomologist's Gaz. 23, 105 - 119.

Hiley, P. D. (1976) The identification of British limnephilid larvae (Trichoptera) Syst. Ent. 1, 147 - 167.

Hilsenhoff, W. L. (1969) An artificial substrate device for sampling benthic stream invertebrates. Limnol Oceanogr. 14 (3), 465 - 471.

H.M.S.O. (1972) Analysis of Raw, Potable and Waste waters. H.M.S.O. publ. London.

Hocutt, C. H. (1975) Assessment of a stressed macroinvertebrate community. Water Resources Bulletin 11 (4), 820 - 835.

Holland, D. G. (1972) A key to the larvae, pupae and adults of the British species of Elminthidae. Scient. Publs. Freshwat. biol. Ass. No. 26, 58 pp.

Holland D. G. (1976a) The distribution of the freshwater Malacostraca in the area of the Mersey and Weaver River Authority. Freshwat. Biol. 6, 265 - 276.

Holland, D. G. (1976b) The inland distribution of brackish-water Gammarus species in the area of the Mersey and Weaver River Authority. Freshwat. Biol. 6, 277 - 285.

Howmiller, R. P. (1971) A comparison of the effectiveness of Ekman and Ponar grabs. Trans. Amer. Fish. Soc. 100, 560 - 564.

Hughes, B. D. (1975) A comparison of four samplers for benthic macroinvertebrates inhabiting coarse river deposits. Wat. Res. 9, 61 - 69.

Hughes, B. D. (1978) The influence of factors other than pollution on the value of Shannon's Diversity index for benthic macroinvertebrates in streams. Wat. Res. 12 (5), 359 - 364.

Hynes, H. B. N. (1951) Distribution of British freshwater Amphipoda. Nature, Lond. 167, 152 - 153.

Hynes, H. B. N. (1954) Identity of Gammarus tigrinus Sexton 1939. Nature, Lond. 174, 563.

Hynes, H. B. N. (1955a) The reproductive cycle of some British freshwater Gammaridae. J. Anim. Ecol. 24, 352 - 387.

Hynes, H. B. N. (1955b) Distribution of some freshwater Amphipoda in Britain. Verh. int. Verein. theor. angew. Limnol. 12, 620 - 628.

Hynes, H. B. N. (1959) The use of invertebrates as indicators of river pollution. Proc. Linn. Soc. Lond. 170, 165 - 169.

Hynes, H. B. N. (1960) The biology of polluted waters. Liverpool University Press, 202 pp.

Hynes, H. B. N. (1961) The invertebrate fauna of a Welsh mountain stream. Arch.Hydrobiol. 57, 344 - 388.

Hynes, H. B. N. (1967) A key to the adults and nymphs of British stoneflies (Plecoptera). Scient. Publs. Freshwat. biol. Ass. No. 17, 90 pp.

Hynes, H. B. N. (1970) The ecology of running waters. Liverpool University Press, 555 pp.

Hynes, H. B. N. (1971) Benthos of flowing water. In W. T. Edmondson and G. G. Winberg (Eds.). A manual on methods for the assessment of secondary productivity in freshwaters. IBP handbook No. 17, pp 66 - 74, Blackwell, Oxford.

Illies, J. (1961) Versuch einer allgemein biozonotischen Gleiderung der Fliessgewasser. Int. Revue ges. Hydrobiol. Hydrogr. 46, 205 - 213.

Illies, J. and Botosaneanu, L. (1963) Problemes et methodes de la classification et de la zonation ecologique des eaux courentes, considerees surtout du pont de vue faunistique. Mitt. int. Verein. Limnol. 12, 1 - 57.

Jacobi, Z. G. (1971) A quantitative artificial substrate sampler for benthic macroinvertebrates. Trans. Amer. Fish.Soc. 100 (1), 136 - 138.

Jones, F. H. (1973) Quantitative changes in the benthic macroinvertebrate communities of the River Taff and the relationship between plant detritus and invertebrate numbers. Unpublished M.Sc. Project Report, University of Aston in Birmingham.

Jones, J. R. E. (1940) A study of the zinc-polluted River Ystwyth in North Cardiganshire, Wales. J. Appl. Biol. 27, 368 - 378.

Jones, J. R. E. (1958) A further study of the zinc-polluted River Ystwyth. J. Anim. Ecol. 27, 1 - 14.

Kathman, D. (1978) Artificial substrate sampler for benthic invertebrates in ponds, small lakes and reservoirs. Progve. Fish Cult. 40 (3), 114 - 115.

Khalaf, G. (1975) Utilisation de substrats artificiels, en eau courante, pour l'etude de la repartition et de la dynamique de colonisation des macroinvertebres benthiques. These Doct. Spec. Lyon. 91 pp.

Khalaf, G.and Tachet, H. (1977) The dynamics of macroinvertebrate colonisation of artificial substrates in a stream. Annls. Limnol. 13 (2), 169 - 190.

Khalaf, G.and Tachet, H. (1978) Un probleme d'actualite: revue de travaux en matiere d'utilisation des substrats artificiels pour l'echantillonnage des macroinvertebres des eaux courantes. Bull. Ecol. 9, 29 - 38.

Kimmins, D. E. (1966) A revised check-list of the British Trichoptera. Entomologist's Gaz. 17, 111 - 120.

Knopp, H. (1954) Ein neuer weg zur Darstellung biologische Vorfluteruntersuchungen, erlautert an einen Gutelangsschnitt des Maines. Die Wasserwirtsch 45 (1), 9 - 15.

Kolkwitz, R. (1950) Ecology of the saprobes. The relation of aquatic organisms to their environment. Schr Reihe Ver. Wasserhyg. No. 4, Piscator-Verlag, Stuttgart.

Kolkwitz, R. and Marsson, M. (1908) Okologie der pflanzlichen Saprobien. Ber. dtsch. bot. Ges. 26, 505 - 519.

Kolkwitz, R. and Marsson, M. (1909) Okologie der tierischen Saprobien. Int. Revue ges Hydrobiol. Hydrogr. 2,126 - 152.

Kovalak, W. P. (1976) Seasonal and diel changes in the positioning of Glossosoma nigrior Banks (Trichoptera: Glossosomatidae) on artificial substrates. Can. J. Zool. 54 (9), 1585 - 1594.

Kovalak, W. P. (1978) Diel changes in stream benthos density on stones and artificial substrates. Hydobiologia 58 (1), 7 - 16.

Kreis, R. D. and Smith, R. L. (1970) A method of suspending multiple 'basket samplers' in reservoirs. Progve. Fish Cult. 32 (3), 182 - 183.

Kreis, R. D., Smith, R. L. and Moyer, J. E. (1971) The use of limestone-filled basket samplers for collecting reservoir macroinvertebrates. Wat. Res.5, 1099 - 1106.

Kroger, R. L. (1972) Underestimation of standing crop by the Surber sampler. Limnol. Oceanogr. 17, 475 - 478.

Lagerspetz, K. (1958) The brackish-water tolerance of some fresh water crustaceans. Verh. int. Verein. theor. angew. Limnol. 13, 718 - 721.

Langford, T. E. and Aston, R. J. (1972) The ecology of some British rivers in relation to warm water discharges from power stations. Proc. R. Soc. Lond. B. 180, 407 - 419.

Lapchin, L. (1977) Utilisation of artificial substrata in the study of benthic populations of invertebrates. Preliminary results obtained in a salmonidae stream in Brittany. Ann. Hydrobiol. 8 (1), 33 - 44.

Laville, H. (1974) Utilisation de substrats artificiels pour l'etude de la faune macrobenthique de la zone littorale rocheuse de lacs de montagne. Annls. Limnol. 10 (2), 163 - 172.

Learner, M. A., Williams, R., Harcup, M. and Hughes, B. D. (1971) A survey of the macro-fauna of the River Cynon, a polluted tributary of the River Taff (South Wales). Freshwat. Biol. 1, 339 - 367.

Leonard, J. W. (1939) Comments on the adequacy of accepted stream bottom sampling technique. Trans. 4th N.Amer. Wildl. Conf., 288 - 295.

Lewis, D. J. and Bennett, G. F. (1974) An artificial substrata for the quantitative comparison of the densities of larval Simuliidae (Diptera) populations. Can. J. Zool. 52 (6), 773 - 775.

Liebmann, H. (1951) Handbuch der Frischwasser - und Aberwasserbiologie. Vol. 1, R. Oldenbourg, Munchen.

Lillehammer, A. (1966) Bottom fauna investigations in a Norwegian river. The influence of ecological factors. Nytt. Mag. Zool. 13, 10 - 29.

Longhurst, A. R. (1959) The sampling problem in benthic ecology. Proc. N.Z. ecol. Soc. 6, 8 - 12.

Macan, T. T. (1949) Survey of a moorland fishpond. J. Anim. Ecol. 18, 160 - 186.

Macan, T. T. (1958) Methods of sampling the bottom fauna in stony streams. Mitt. int. Verein. theor. angew. Limnol. 8, 1 - 21.

Macan, T. T. (1959) A guide to freshwater invertebrate animals. Longmans, London, 118 pp.

Macan, T. T. (1965) A revised key to the British water bugs (Hemiptera-Heteroptera). Scient. Publs. Freshwat. biol. Ass. No. 16, 78 pp.

Macan, T. T. (1969) A key to the British fresh - and brackish-water gastropods with notes on their ecology. Scient. Publs. Freshwat. biol. Ass. No. 13, 44 pp.

Macan, T. T. (1970) A key to the nymphs of the British species of Ephemeroptera with notes on their ecology. Scient. Publs. Freshwat. biol. Ass. No. 20, 68 pp.

Macan, T. T. (1974a) Freshwater invertebrates. Systematics association special vol. No. 6. 'The changing flora and fauna of Britain', edited by D. L. Hawksworth, 1974, 143 - 155. Academic Press, London and New York.

Macan, T. T. (1974b) Freshwater Ecology. Longmans, London, 343 pp.

Macan, T. T. (1976) A twenty-one-year study of the water-bugs in a moorland fishpond. J. Anim. Ecol. 45, 913 - 922.

Macan, T. T. (1977a) A twenty-year study of the fauna in the vegetation of a moorland fishpond. Arch. Hydrobiol. 81 (1), 1 - 24.

Macan, T. T. (1977b) The fauna in the vegetation of a moorland fishpond as revealed by different methods of collecting. Hydrobiologia 55 (1), 3 - 15.

Macan, T. T. and Kitching, A. (1972) Some experiments with artificial substrata. Verh. int. Verein. theor. angew. Limnol. 18, 213 - 220.

Macan, T. T. and Kitching, A. (1976) The colonisation of squares of plastic suspended in midwater. Freshwat. Biol. 6, 33 - 40.

MacArthur, R. H. and Wilson, E. O. (1963) An equilibrium theory of insular zoogeography. Evolution 17, 373 - 387.

Mackay, D. W., Soulsby, P. G. and Poodle, T. (1973) The biological assessment of pollution in streams. Ass. Riv. Auth. Year Book and Dir., London, 189 - 197.

Mackereth, J. S. (1954) Taxonomy of the larvae of the British species of the genus Rhyacophila (Trichoptera). Proc. R. ent. Soc. Lond. A, 29, 147 - 152.

Mackereth, J. C. (1956) Taxonomy of the larvae of the British species of the sub-family Glossosomatinae (Trichoptera). Proc. R. ent. Soc. Lond. A, 31, 167 - 172.

Mackereth, F. J. H. (1958) A portable core sampler for lake deposits. Limnol. Oceanogr. 3, 181 - 191.

Mackey, A. P. (1972) An air-lift for sampling freshwater benthos. Oikos 23, 413 - 415.

Maitland, P. S. (1964) Quantitative studies on the invertebrate fauna of sandy and stony substrats in the River Endrick, Scotland. Proc. Roy. Soc. Edinb. B, 68, 277 - 301.

Maitland, P. S. (1978) A coded checklist of animals occurring in freshwater in the British Isles. Inst. Terr. Ecol. Publ. 76 pp.

Mann, K. H. (1955a) The ecology of the British freshwater leeches. J. Anim. Ecol. 24, 98 - 119.

Mann, K. H. (1955b) Some factors influencing the distribution of freshwater leeches in Britain. Verh. int. Verein. theor. angew. Limnol. 12, 582 - 587.

Mann, K. H. (1958) Occurrence of an exotic Oligochaeta *Branchiura sowerbyi* (Beddard 1892) in the River Thames. Nature, Lond. 182, 732.

Mann, K. H. (1964) A key to the British freshwater leeches with notes on their ecology. Scient. Publs. Freshwat. biol. Ass. No. 14, 50 pp.

Mann, K. H. (1965) Heated effluents and their effects on the invertebrate fauna of rivers. Proc. Soc. Wat. Trt. and Exam. 14, 45 - 53.

Margalef, R. (1951) Diversidad de especies en las commundades naturales. P. Inst. Biol. Appl. 9, 5 - 28.

Marstrand, P. K. (1973) Using biotic indices as a criterion of in-river water quality. Ass. Riv. Auth. Year Book and Dir., London, 182 - 188.

Mason, C. F. (1977a) The performance of a diversity index in describing the zoobenthos of two lakes. J. Appl. Ecol. 14, 363 - 367.

Mason, C. F. (1977b) Abibliography of biological surveillance methodology for macroinvertebrates in running waters. Wata Data Unit Technical Memorandum No. 14, 14 pp.

Mason, J. C. (1976) Evaluating a substrate tray for sampling the invertebrate fauna of small streams, with comment on general sampling problems. Arch. Hydrobiol. 78, 51 - 70.

Mason, W. T., Anderson, J. B. and Morrison, G. E. (1967a) A limestone-filled, artificial substrate sampler-float unit for collecting macroinvertebrates in large streams. Progve. Fish Cult. 29, 74.

Mason, W. T., Anderson, J. B.,Kries, R. D. and Johnson, W. C. (1967b). Collecting macroinvertebrates in a polluted stream using rock-filled samplers. Report No. 28 Applications and Development U.S. Dept. Int. F.W.P.C.A., 23 pp.

Mason, W. T., Anderson, J. B., Kreis, R. D. and Johnson, W.C. (1970) Artificial substrate sampling, macroinvertebrates in a polluted reach of the Klamath River, Oregon. J. Wat. Pollut. Control Fed. 42, R315 - R328.

Mason, W. T., Weber, C. I., Lewis, P. A. and Julian, E.C. (1973) Factors affecting the performance of basket and multiplate macroinvertebrate samplers. Freshwat. Biol. 3, 409 - 436.

Mathers, K. C. and Martin, M. R. (1969) A new multiple-plate sampler for collecting macroinvertebrates of the stream biocies. Trans. Illinois st. Acad. Sci. 62 (3), 331 - 334.

McDaniel, M. D. (1974) Design and preliminary evaluation of an improved artificial substrate sampler for aquatic macroinvertebrates. Progve. Fish Cult. 36 (1), 23 - 25.

Mellanby, H. (1963) Animal life in freshwater. A guide to freshwater invertebrates. Methuen and Co. Ltd., London, 308 pp.

Minshall, G. W. and Andrews, D. A. (1973) An ecological investigation of the Portneuf River, Idaho: a semiarid-land stream subjected to pollution. Freshwat. Biol. 3, 1 - 30.

Minshall, G. W. and Minshall, J. N. (1977) Microdistribution of benthic invertebrates in a rocky mountain (U.S.A.) stream. Hydrobiologia 53 (3), 231 - 249.

Moon, H. P. (1935a) Methods and apparatus suitable for an investigation of the littoral region of oligotrophic lakes. Int. Revue ges Hydrobiol. Hydrogr. 32, 319 - 333.

Moon, H. P. (1935b) Flood movements of the littoral fauna of Windermere. J. Anim. Ecol. 4, 216 - 228.

Moon, H. P. (1939) Aspects of the ecology of aquatic insects. Trans. Soc. Brit. Ent. 6 (3), 39 - 49.

Moon, H. P. (1940) An investigation of the movements of freshwater invertebrate faunas. J. Anim. Ecol. 9, 76 - 83.

Moon, H. P. (1970) Corophium curvispinum (Amphipoda) recorded again in the British Isles. Nature, Lond. 226, 976.

Morgan, N. C. and Egglishaw, H. J. (1965) A survey of the bottom fauna of streams in the Scottish Highlands. Part 1. Composition of the fauna. Hydrobiologia 25, 181 - 211.

Mortimer, C. H. (1956) The oxygen content of air-saturated freshwaters, and aids in calculating percentage saturation. Mitt. d. Intern. Ver. f. Limnol. 6, 1 - 20.

Mundie, J. H. (1956) A bottom sampler for inclined rock surfaces in lakes. J. Anim. Ecol. 25, 429 - 432.

Murphy, P. M. (1978) The temporal variability in biotic indices. Environ. Pollut. 17 (3), 227 - 236.

Needham, P. R. and Usinger, R. L. (1956) Variability in the macrofauna of a single riffle in Prosser Creek, California, as indicated by the Surber sampler. Hilgardia 24 (14), 383 - 409.

Neill, R. M. (1938) The food and feeding of the brown trout (Salmo trutta L.) in relation to the organic environment. Trans. R. Soc. Edinb. 59 (2), 481 - 520.

N.E.R.C. Report (1976) Biological monitoring and surveillance of rivers. Handbook, N.E.R.C. London.

Nijssen, H. and Stock. J. H. (1966) The amphipod, Gammarus tigrinus Sexton, 1939, introduced in the Netherlands (Crustacea). Beaufortia 13, 197 - 206.

Nilsen, H. C. and Larimore, R. W. (1973) Establishment of invertebrate communities on log substrates in the Kastaskia River, Illinois. Ecology 54, 366 - 374

Nilsson, L. M. and Sjostrom, P. (1977) Colonisation of implanted substrates by differently sized Gammarus pulex (Amphipoda). Oikos 28, 43 - 48.

Nuttall, P. M. and Purves, J. B. (1974) Numerical indices applied to the results of a survey of the macroinvertebrate fauna of the Tamar catchment (South-west England). Freshwat. Biol. 4, 213 - 222.

Okland, J. (1962) Litt om teknikk ved innsampling of konservering av ferskvannsdyr. Fauna 15 (2), 69 - 92.

Pankratova, V. Ya (1970) Larvae and pupae of midges of the subfamily Orthocladiinae of the U.S.S.R. Partially translated as F.B.A. Transl. No. 54 and No. 64.

Pantle, R. and Buck, H. (1955) Die biologische Uberwachung der Gewasser und die Darstellung der Ergebnisse. Gas-und Wasserfach 96, 604.

Parsons, D. S. and Tatum, J. W. (1974) A new shallow water multiple-plate sampler. Progve. Fish Cult. 36 (3), 179 - 180.

Patrick, R. (1950) Biological measure of stream conditions. Sewage Ind. Wastes 22, 926 - 938.

Pearson, R. G. and Jones, N. V. (1975a) The colonisation of artificial substrata by stream macroinvertebrates. Progress in Water Technology 7 (3/4), 497 - 504.

Pearson, R. G. and Jones, N. V. (1975b) The effects of dredging operations on the benthic community of a chalk stream. Biol. Conserv. 8, 273- 278.

Pearson, R. G., Litterick, M. R. and Jones, N. V. (1973) An air-lift for quantitative sampling of the benthos. Freshwat. Biol 3, 309 - 315.

Pegel, V. M. and Ruhm, W. (1976) Verusche zur besiedlung kunstlicher substrate durch praionaginale stadien von Simuliiden unter besonderer berucksichtigung von boophthora erythrocephala de Geer (Simuliidae, Dipt). Zeitschrift fur Angewandte Entomologie 82 (1), 65 - 71.

Pentelow, F. T. K. and Butcher, R. W. (1938) Observations on the condition of Rivers Churnet and Dove in 1938. Rep. Trent Fish. Bd. App. 1.

Percival, E. and Whitehead, H. (1929) A quantitative study of the fauna of some types of stream bed. J. Ecol. 17, 282 - 314.

Percival, E. and Whitehead, H. (1930) Biological survey of the River Wharfe. 2. Report on the invertebrate fauna. J. Ecol. 18, 286 - 302.

Petersen, C. G. J. (1911) Valuation of the sea. 1. Animal life of the sea, its food and quantity. Rep. Dan. biol. Stn. 20, 1 - 76.

Pinkster, S. and Stock, J. H. (1967) Range extension in 1966 of the alien amphipod, Gammarus tigrinus Sexton, 1939, in the Netherlands. Beaufortia, 14, 81 - 86.

Philipson, G. N. (1954) The effect of water flow and oxygen concentration on six species of caddis fly (Trichoptera) larvae. Proc. zool. Soc. Lond. 124, 547 - 564.

Pitcairn, C. E. R. and Hawkes, H. A. (1973) The role of phosphorus in the growth of Cladophora. Wat. Res. 7, 159 - 171.

Poole, W. C. and Stewart, K. W. (1976) The vertical distribution of macrobenthos within the substratum of the Brazos River, Texas. Hydrobiologia 50 (2), 151 - 160.

Pratt, J. M. and Coler, R. A. (1976) A procedure for the routine biological evaluation of urban run-off in small rivers. Wat. Res. 10 (11), 1019 - 1025.

Prins, R. and Black, W. (1971) Synthetic webbing as an effective macrobenthos sampling substrate in reservoirs. In reservoir fisheries and limnology, G. E. Hall, Ed., Americal Fisheries Society special publ. 8, 203 - 208.

Rabeni, C. F. and Gibbs, K. E. (1978) Comparison of two methods used by divers for sampling benthic invertebrates in deep waters. J.Fish. Res. Bd. Can. 35 (1), 332 - 336.

Rabeni, C. F. and Minshall, G. W. (1977) Factors affecting microdistribution of stream benthic insects. Oikos 29, 33 - 43.

Radford, D. S. and Hartland-Rowe, R. (1971) Surface and subsurface sampling of benthic invertebrates in two streams. Limnol. Oceanogr. 16, 114 - 120.

Reynoldson, T. B. (1961) Observations of the occurrence of Asellus (Isopoda, Crustacea) in some lakes of northern Britain. Verh. int. Verein. theor. angew. Limnol. 14, 988 - 994.

Reynoldson, T. B. (1967) A key to the British species of freshwater triclads. Scient. Publs. Freshwat. biol. Ass. No. 23, 28 pp.

Reynoldson, T. B. and Bellamy, L. S. (1970) The status of Dugesia lugubris and D. polychroa (Turbellaria, Tricladida) in Britain. J.Zool. Lond. 162, 157 - 177.

Richardson, R. E. (1928) The bottom fauna of the Middle Illinois River 1913 - 1925; its distribution, abundance, variation and index value in the study of stream pollution. Bull. Ill. Nat. Hist. Surv. 17, 387 - 475.

Rinne, J. N. (1978) Standing crops of Chironomidae and Tubificidae in two desert reservoirs, Central Arizona, Hydrobiologia 57 (3), 217 - 224.

Roby, K. B., Newbold, J. D. and Erman, D. C. (1978) Effectiveness of an artificial substrate for sampling macroinvertebrates in small streams. Freshwat. Biol. 8,1 - 8.

Rousseau, E. (1919) Larves D'Haliplides D'Europe. Annales Biologie Lacust 9, 269 - 278.

Rosenberg, D. M. and Wiens, A. P. (1976) Community and species responses of Chironomidae (Diptera) to contamination of freshwaters by crude oil and petroleum products, with special reference to the Trail River, Northwest Territories. J. Fish. Res. Bd. Can. 33, 1955 - 1963.

Roux, A. L., Tachett, H. and Neyron, M. (1976) Structure et fonctionnement des ecosystemes du Haut-Rhone Francais 3. Une technique simple et peu onereuse pour L'etude des macro-invertebres benthiques des grands fleuves. Bull. Ecol. 7 (4), 493 - 496.

Ruoff, V. K. (1968) Experimentelle untersuchung uber den in die wesser eingeburgerten amerikanischen Bachflohkrebs *Gammarus tigrinus* Sexton. Arch. fur. Fisch. Wissenschaft 19, 134 - 158.

Schmitz, W. (1960) Die einburgerung von *Gammarus tigrinus* Sexton auf dem euripaischen Kontinent. Arch. Hydrobiol. 57 (1 - 2), 223 - 225.

Schwoerbel, J. (1966) Methoden der hydrobiologie (Susswasserbiologie). Franckh'sche verlagschandlung, Stuttgart, 207 pp.

Scott, D. (1958) Ecological studies on the Trichoptera of the River Dean, Cheshire. Arch. Hydrobiol. 54, 340 - 392.

Scott, D. C. (1958a) Biological balance in streams. Sewage Ind. Wastes 30 (9), 1169 - 1173.

Sexton, E. W. and Cooper, L. H. N. (1939) On a new species of *Gammarus* (*G. tigrinus*) from Droitwich district. J. Mar. Biol. Assoc. 23, 543 - 551.

Sheldon, A. L. (1977) Colonisation curves: application to stream insects on semi-natural substrates. Oikos 28, 256 - 261.

Shelford, V. E. and Eddy, S. (1929) Methods for the study of stream communities. Ecology 10, 382 - 391.

Simmons, G. M. and Winfield, A. (1971) A feasibility study using conservation webbing as an artificial substrate in macroinvertebrate studies. Virginia J. Science 22 (2), 52 - 59.

Slack, K. V., Averett, R. C., Gresson, P. E. and Lipscomb, R. G. (1973) Methods for collecting and analysis of aquatic biological and microbiological samples. Techniques of water-resources investigations of the U.S. geological survey. Chapter A4 Book 5 (Laboratory Analysis) Government punting office, Washington, D.C.

Sladecek, V. (1973a) System of water quality from the biological point of view. Arch. Hydrobiol. Beih. Ergebn. Limnol. 7, 1 - 218.

Sladecek, V. (1973b) The reality of three British biotic indices. Wat. Res. 7, 995 - 1002.

Sladecek, V. (1978) Continental systems for the assessment of river water quality. International symposium on biological indicators of water quality. University of Newcastle upon Tyne 12 - 15 September, 1978.

Sorensen, T. (1948) A method of establishing groups of equal amplitude in plant sociology based on similarity of species content and its application to analyses of the vegetation on Danish commons. K. Danske Vidensk. Selsk. 5, 1 - 34.

Southwood, T. R. E. (1966) Ecological methods with particular reference to the study of insect populations. Chapter 6. Absolute population estimates by sampling a unit of habitat - Freshwater habitats, 158 - 173. Methuen London, 391 pp.

Spooner, G. M. (1951) Distribution of British freshwater amphipoda. Nature, Lond. 167, 530.

Stanford, J. A. and Reed, E. B. (1974) A basket sampling technique for quantifying riverine macrobenthos. Wat. Res. Bull. 10 (3), 470 - 477.

Stauffer, J. R., Beiles, H. A., Cox, J. W., Dickson, K. L. and Simonet, D. E. (1976) Colonisation of macrobenthic communities on artificial substrates. Revista de Biologia 10 (1 – 4), 49 – 61.

Surber, E. W. (1937) Rainbow trout and bottom fauna production in one mile of stream. Trans. Amer. Fish. Soc. 66, 193 – 202.

Sutcliffe, D. W. (1968) Sodium regulation and adaptation to fresh water in gammarid crustaceans. J. exp. Biol. 48, 359 – 380.

Tattersall, W. M. (1937) Occurrence of Eucrangonyx gracilis. Nature, Lond. 139, 593.

Thienemann, A. (1920) Die Grundlagen der Biozonotik und Monards faunistische Prinzipien. Festschr. f. Zschokke No. 4, 1 – 14, Basel.

Thienemann, A. (1944) Bestimmungstabellen fur die jetst bekannten larven und pupen der Orthocladiinen. Archwm. Hydrobiol. Ryb. 39, 551 – 664.

Thorup, J. (1966) Substrate type and its value as a basis for the delimitation of bottom fauna communities in running water. Spec. Publs. Pymatuning Lab. Fld. Biol. 4, 59 – 74.

Townsend, C. R. and Hildrew, A. G. (1976) Field experiments on the drifting, colonisation and continuous redistribution of stream benthos. J. Anim. Ecol. 45, 759 – 772.

Ulfstrand, S. (1968) Benthic animal communities in Lapland streams. A field study with particular reference to Ephemeroptera, Plecoptera, Trichoptera and Diptera Simuliidae. Oikos. Suppl. 10, 1 – 120.

Ulfstrand, S., Nilsson, L. M. and Stergar, A. (1974) Composition and diversity of benthic species collectives colonising implanted substrates in a South Swedish stream. Ent. Scand. 5, 115 – 122.

Usinger, R. L. and Needham, P. R. (1956) A drag-type riffle-bottom sampler. Progve. Fish Cult. **18**, 42 - 44.

Verneaux, J. and Tuffery, A. (1967) Une methode zoologique practique de determination de la qualite biologique des eaux courantes. Ann. Sci. Univ. Besancon (3). Zool. fasc. **3**, 79 - 90.

Voshell, J. R. and Simmons, G. M. (1977) An evaluation of artificial substrates for sampling macrobenthos in reservoirs. Hydrobiologia **53** (3), 257 - 269.

Wallace, I. D. (1977) A key to larvae and pupae of *Sericostoma personatum* (Spence) and *Notidobia ciliaris* (Linne) (Sericostomatidae: Trichoptera) in Britain. Freshwat. Biol. **7**, 93 - 98.

Warren, C. E. (1971) Biology and water pollution control. W. B. Saunders Co., Philadelphia, London, Toronto, 434 pp.

Waters, T. F. (1964) Recolonisation of denuded stream bottom areas by drift. Trans. Amer. Fish Soc. **93**, 311 - 315.

Waters, T. F. (1972) The drift of stream insects. Ann. Revue Ent. **17**, 253 - 272.

Waters T. F. and Knapp, R. J. (1961) An improved stream bottom fauna sampler. Trans. Amer. Fish Soc. **90**, 225 - 226.

Weber, C. I. (1973) Biological field and laboratory methods for measuring the quality of surface waters and effluents. U.S. E.P.A. Cincinnati Ohio Publ.

Wene, G. and Wickliff, E. L. (1940) Modifications of the stream bottom and its effect on the insect fauna. Can. Ent. **72**, 131 - 135.

Whitley, L. S. (1962) New bottom sampler for use in shallow streams. Limnol. Oceanogr. **7**, 265 - 266.

Wilding, J. L. (1940) A new square-foot aquatic sampler. Spec. Publs. Limnol. soc. Amer. No. 4, 1 - 4.

Wilhm, J. L. (1970) Range of diversity index in benthic macroinvertebrate populations. J. Wat. Pollut. Control Fed. 42, R221 - R224.

Wilhm, J. L. and Dorris, T. C. (1968) Biological parameters for water quality criteria. Bio Science 18, 477 - 481.

Williams, D. D. (1977) Movements of benthos during the recolonisation of temporary streams. Oikos 29, 306 - 312.

Williams, D. D. and Hynes, H. B. N. (1974) The occurrence of benthos deep in the substratum of a stream. Freshwat. Biol. 4, 233 - 256.

Williams, D. D. and Hynes, H. B. N. (1976) The recolonisation mechanisms of stream benthos. Oikos 27, 265 - 272.

Williams, D. D. and Hynes, H. B. N. (1977) Benthic community development in a new stream. Can. J. Zool. 55, 1071 - 1076.

Williams, R., Williams, P. F., Benson-Evans, K., Hunter, M. D. and Harcup, M. F. (1976) Chemical and biological studies associated with the recovery of the River Ebbw Fawr (1970 - 1974). Wat. Pollut. Control 75, 428 - 444.

Williams, T. R. and Obeng, L. (1962) A comparison of two methods of estimating changes in Simulium larval populations, with a description of a new method. Ann. trop. Med. Parasit. 56, 359 - 361.

Wolfe, L. S. and Peterson, D. G. (1958) A new method to estimate levels of infestations of blackfly larvae (Diptera: Simuliidae). Can. J. Zool. 36, 863 - 867.

Woodiwiss, F. S. (1964) The biological system of stream classification used by the Trent River Board. Chemistry and Industry 443 - 447.

Woodiwiss, F. S. (1978) The Trent Biotic Index-Macroinvertebrates in biological surveillance. Elaboration of the Scientific Bases for Monitoring the quality of Surface Water by Hydrobiological Indicators. Report of the First U.K./U.S.S.R. Seminar held at Valdai, U.S.S.R., 12 - 14 July, 1976, 58 - 81.

Wurtz, C. B. (1955) Stream biota and stream pollution. Sewage Ind. Wastes 27, 1270 - 1278.

Zelinka, M. and Marvan, P. (1961) Zur prazisierung der biologischen klassifikation der Reinheit fliessender gewasser. Arch. Hydrobiol. 57 (3), 389 - 407.

Zillich, J. A. (1967) Responses of Lotic insects to artificial substrate samplers. Unpublished M.S. Thesis, University of Wisconsin, Madison, U.S.A.