THE UNIVERSITY OF HULL

Fish-based assessment of ecological health of English lowland rivers

being a Thesis submitted for the Degree of

Doctor of Philosophy

in the University of Hull

by

Md. Khalilur Rahman

B.Sc. Fisheries (Honours), Bangladesh Agricultural University, 1984
M.Sc. (Fisheries Technology), BAU, Bangladesh, 1985
M.Sc. (Inland Fisheries Management), University of Hull, UK, 1994

November 2001

786

To

My Parents,

Md. Abdul Gafur

And

Mrs. Laila Khatoon

ACKNOWLEDGEMENTS

I would like to express my sincere thanks and deepest gratitude to Dr. Ian Graham Cowx for his supervision, guidance, encouragement and advice throughout this research.

I am indebted to Dr. Jonathan Paul Harvey of the Freshwater Fisheries Laboratory, HIFI for his generous help and co-operation throughout this study.

My sincere thanks to Dr. M. A. Mazid, Director General, Bangladesh Fisheries Research Institute (BFRI), and Dr. G. C. Haldar, Chief Scientific Officer, Riverine Station, BFRI for providing me all out supports to pursue the degree.

To the Agriculture Research Management Project (ARMP-FRI Part), I am grateful for the funding which made this study possible.

My thanks also go to the Environment Agency, UK (Southeast, Midland and Northeast regions) for providing me reports, maps and historical data for this study.

My colleagues in the laboratory, Mr. Richard A. A. Noble, Mr. Andrew David Nunn, Dr. Ana Maria Pires (Portugal), Mr. Robert Trigg (Genetics laboratory) and Dr. J. H. Allen (Institute of Estuarine & Coastal Studies), I would like to thank for their creative assistance throughout this programme. I also wish to thank Mrs. Emma Doy, HIFI Secretary for her assistance in diverse ways.

Special thanks also go to my brother Dr. Md. Ataur Rahman, Associate Professor, Department of Surgery, Comilla Medical College, Bangladesh for his sustained interest in my studies.

Finally, heartiest appreciation and sincere thanks to my classmate, colleague and wife Jubaida Nasreen Akhter for her constant support, encouragement and understanding. Shiblee Nadeem Rahman and Naqvee Naveed Rahman, my two sons, were the continuous sources of love and mental strength during this critical period of my life.

ABSTRACT

Riverine fisheries in England are under pressure from a variety of activities, including increasing intensification of land–use, urbanisation, rising demands for water abstraction, pollution, proliferation of exotic species, climate change and recreational activities. As a result, the integrity of English rivers has changed. In this study, an attempt was made to measure the ecological health of 22 English lowland rivers from the Thames, Trent and Yorkshire Ouse catchments using a variety of tools. The objective was to modify the Index of Biotic Integrity (IBI) for use on English lowland rivers and compare it with existing indices.

A number of diversity indices, Margalef (D_{Mg}), Simpson (D_{Sm}) and Shannon-Wiener (H') were used to evaluate the status of fisheries in the study rivers. The Abundance / Biomass (ABC) method and computer-based multivariate analyses, UPGMA, TWINSPAN, DECORANA, were also used to evaluate the status of fish communities. In addition to these indices, the ABC method and multivariate analyses, the IBI, a multimetric index was also used to evaluate the ecological health of study rivers. The IBI is based on structural and functional attributes of fish communities and is capable of evaluating health and condition of an aquatic ecosystem. The IBI requires a reference condition with which to compare the output. In English rivers, no pristine (reference) sites were considered available, consequently best available data were used to develop a reference condition. In this study, the IBI was modified from Karr (1981), which was based on 12 metrics (community characteristics) of fish assemblages. For the study rivers, 15 metrics which described the status of the fish communities were selected to calculate the IBI. Each metric was scored on a simple scale from 0 (absence) to 5 (high quality). The sum of all the metrics (range 0 - 75) was used to assign sites to qualitative classes of biotic integrity. Six integrity classes on a continuous scale were chosen with the following class boundaries: Excellent (56 - 75), Good (42 - 55), Fair (28 - 41), Poor (16 - 27), Very Poor (1 - 15) and No Fish (0).

In the study rivers, the D_{Mg}, D_{Sm} and H' indices were unable to measure anthropogenic impacts on fish communities as all these indices were based on structural properties of fish communities. These indices also failed to take account of the presence of juveniles in the fish community in a river. Moreover, these indices were influenced by dominant species abundance and sampling strategies, giving an inaccurate assessment of the status of the fisheries. The ABC method was better at evaluating fish communities than diversity indices as the method considered fish abundance and

biomass. However, the method did not include functional components of the fish community and was over influenced by juvenile fishes. Consequently, the ABC method was not considered a good indicator of ecosystem health based on fish assemblages.

The UPGMA, TWINSPAN and DECORANA analyses, successfully grouped and separated river reaches with rich or poor fish stocks. These analyses however, did not take into account the functional attributes of the fish communities and were not sufficient to explain the status of a fishery without support from other indices.

The IBI assessed the ecological health of the middle and lower reaches of the study rivers more accurately than the other diversity indices, ABC method and multivariate analyses. The selected IBI metrics were able to evaluate many perturbations and disturbances as the metrics represented both structural and functional attributes of fish communities. The D_{Mg}, D_{Sm}, H', ABC, UPGMA, TWINSPAN and DECORANA were designed to highlight a specific attribute and lost information during calculation but the IBI included a greater variety of information and produced an appropriate index. Spearman's rank correlation indicated the IBI outputs were more similar to diversity indices than other measures, as significant relationships were found between the IBI and D_{Mg}, the IBI and D_{Sm}, and the IBI and H' at $\alpha = 0.01$ level. Significant relationships were probably due to the use of fish density and abundance in the models. However, this did not mean that all diversity indices and the IBI were similar in measuring ecological conditions of a river, rather it was probably numerical similarity. No significant relationship was found between the IBI and ABC, as the ABC index was a ratio of abundance and biomass while the IBI used absolute values of biomass and abundance separately. All the diversity indices, ABC method and multivariate analyses mentioned reinforced the view that the IBI developed in this study was an appropriate index at evaluating ecological health of the middle and lower reaches of the study rivers. The IBI, however, failed to predict the quality of the fisheries in headwater streams because of the exclusion of salmonid species, minor species and general low species diversity found in these zones. Consequently, it was identified that reference conditions and metrics chosen for the middle and lower reaches of the study rivers were not appropriate to assess the ecological health of headwaters.

The existing monitoring programmes of the Environment Agency (EA) for fishery data collection, were considered appropriate for calculating IBIs. Sampling strategies of the EA, i.e. daytime, electric fishing both in summer and winter periods irrespective of lunar cycle and breeding season were also considered acceptable to calculate the IBI.

Further research was recommended to test the IBI on a wide range of rivers to assess whether the IBI is appropriate for assessing ecological health of middle and lower reaches of rivers in all regions of the UK. Separate IBIs for headwaters, stillwaters and estuaries were proposed as these zones / waterbodies have different fish communities. Investigation should be directed at developing a simplified IBI using other cost-effective data sources if suitable resources are not available. It is also recommended that the possibility of including the IBI in wider aquatic resource monitoring programmes (e.g. WFD) be investigated. It is also recommended that the possibility of using the IBI to detect change in the pre and post implementation periods of any management action or anthropogenic activity be investigated. Research is also needed to integrate the IBI with other bioassessment methods (e.g. Habitat index, Diatom index, Microinvertebrate index, Chemical index and GQA index). For more effective application and understanding, the IBI may be built into a GIS (Geographical Information System) environment. It is suggested that a suitable computer package be developed to simplify calculations of the IBI. The interpretation should however, be carried out by the fishery manager or scientist.

CONTENTS

CONTENTS (Continued)

CONTENTS (Continued)

LIST OF TABLES

LIST OF TABLES (Continued)

LIST OF FIGURES

LIST OF FIGURES (Continued)

LIST OF FIGURES (Continued)

CHAPTER ONE

1. INTRODUCTION

1.1 DEGRADATION AND RIVERINE FISHERIES OF ENGLAND AND WALES

Degradation of water resources has long been a concern of human society both in developed and developing countries. Many English and Welsh rivers have been degraded for a variety of reasons (Cowx & Welcomme 1998). They have been enlarged, straightened and deepened for land drainage, flood relief and navigation; re-routed or covered over to accommodate urban development and major transport links; diverted to provide power for mills; dammed for hydropower schemes and public water supply reservoirs; used for the disposal and dispersal of waste effluent; modified by the transfer of water between river catchments and their water abstracted to provide water for canals, industry and agriculture (Petts 1984, EA 1998). These modifications disrupt the fabric of the aquatic system and diminish its integrity, attacking the capacity of the fish and other organisms to survive.

The total length of English and Welsh rivers and tributaries is 158,000 km (Fig. 1.1), of which 35,000 km of main river are maintained by engineering works (Petts 1988). Another 8,504 km of rivers have been severely altered (canalised, dredged, piled) by major or capital works and a further 35,500 km are managed in a lesser extent by removing aquatic plants, and bankside trees and shrubs (Petts 1988, Moss 1998). For example, during 1985/1986 the Anglian Water Authority dredged 350 km of rivers, removing many of the bankside trees and often leaving steep banks, devoid of marginal aquatic plants (Giles 1994). In the UK, the density of channelized river is 0.06 km km^{-2} (Petts 1988).

It is reported that 80% of the lowland river sites of the UK have been modified. This modification to the channel severely affects 5%, 4%, and 2% of lowland river sites in Northern Ireland, England and Wales, and Scotland respectively, whilst only 10%, 14% and 28% of lowland sites were found to be entirely unaltered (EA 1998). Thirty-eight rivers totalling 2,400 km are protected within Sites of Special Scientific Interests (SSSI) in England and Wales (EA 1998). Only 11% of rivers in the UK are classified as "natural", i.e. the flow has not been significantly regulated or modified by abstraction or discharge (Cowx 1998). However, many of the rivers have suffered some sort of flow manipulation. A total of 39%, 30% and 10% of UK rivers are affected by interbasin transfers, direct and indirect regulation, respectively (Petts 1988). There are over 14,000 licensed abstractions from non-tidal surface waters in England and Wales. The total volume of water licensed for abstraction is of the order of 40,000 Mld^{-1}, representing 23% of the total mean run-off (Solomon 1992). English Nature (1996) identified 40 low flow sites in England and Wales of which 18 are at high risk. In 1990, 34,360 km (85%) of rivers and canals in England and Wales were classified as "Good" or "Fair" which has increased to over 38,000 km (94%) in 2000 (EA 2002). The majority of low graded rivers are in the West Midlands, Greater Manchester, West Yorkshire and rural East Anglia (EA 2002).

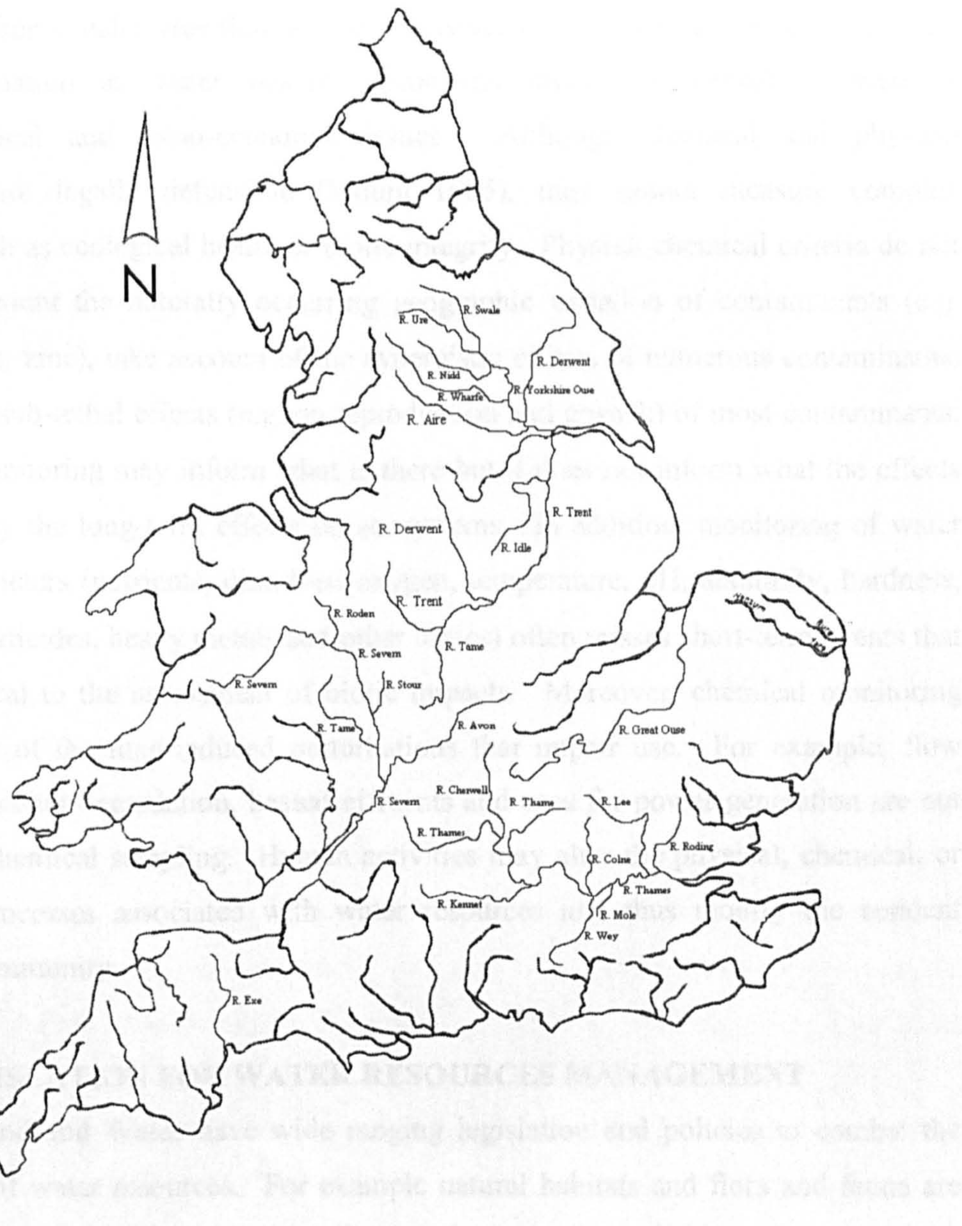


Fig. 1.1 Main river systems in England and Wales

Many factors affecting the status of inland fisheries are directly related to the degradation and loss of suitable habitat conditions (Hellawell 1988, Cowx 1998). The most demonstratable effect of degradation on biotic populations has been the conversion of salmonid rivers to cyprinid (coarse fish) rivers (e.g. River Aire) (EA, LEAP 1998a) dominated by roach (*Rutilus rutilus* (L.)), chub (*Leuciscus cephalus* (L.)) and gudgeon (*Gobio gobio* (L.)). Due to water quality and habitat degradation, coarse fish populations have declined within a number of major river systems of the UK (Cowx 2001). This has been reflected in lower anglers' catches (Cowx 1998).

Steady declines in the quality and quantity of water resources, despite massive regulatory efforts, calls attention to the inadequacies of existing methods of water quality evaluation as water resource problems involve biological as well as physicochemical and socio-economic issues. Although chemical and physical approaches are legally defensible (Mount 1985), they cannot measure complex attributes such as ecological health or biotic integrity. Physico-chemical criteria do not take into account the naturally occurring geographic variation of contaminants (e.g. asbestos, iron, zinc), take account of the synergistic effects of numerous contaminants, nor consider sub-lethal effects (e.g. on reproduction and growth) of most contaminants. Chemical monitoring may inform what is there but it does not inform what the effects are, especially the long-term effects on ecosystems. In addition, monitoring of water quality parameters (nutrients, dissolved oxygen, temperature, pH, alkalinity, hardness, ammonia, pesticides, heavy metals and other toxics) often misses short-term events that may be critical to the assessment of biotic impacts. Moreover, chemical monitoring misses many of the man-induced perturbations that impair use. For example, flow alterations, habitat degradation, heated effluents and uses for power generation are not detected in chemical sampling. Human activities may alter the physical, chemical, or biological processes associated with water resources and thus modify the resident biological community.

1.2 LEGISLATION FOR WATER RESOURCES MANAGEMENT

England and Wales have wide ranging legislation and policies to combat the degradation of water resources. For example natural habitats and flora and fauna are protected by the European Union (EU) Habitats Directive (92/43/EEC). The Freshwater Fisheries Directive (78/659/EEC) sets standards to support fish life in fresh waters, the Surface Water Abstraction Directive (75/440/EEC) controls the quality of surface water for potable supply and the Urban Waste Water Treatment Directive (91/271/EEC) sets standards for the control of toxic substances and pollution. Salmon and freshwater fishes are managed under the UK Salmon and Freshwater Fisheries Act, 1975 whilst wildlife and the countryside are protected by the UK Wildlife & Countryside Act, 1981. Water abstraction is controlled by the UK Water Resources Act, 1991. Research projects are currently being undertaken to address specific issues such as the Salmon Restoration Scheme in the River Thames.

The legislation and policies have been successful in improving or maintaining specific or individual components of an aquatic ecosystem but have sometimes failed to improve or maintain overall integrity. To aid existing legislation and policies, the EU has introduced the Water Framework Directive (WFD) (2000/60/EC) based on structural and functional components of aquatic ecosystem (EU 2000). The WFD was designed to protect, enhance, restore and defend all surface waters in the EU member states (EU 2000). According to the WFD, the status of rivers will be assessed using "quality elements", phytoplankton, macrophytes and phytobenthos, benthic invertebrate fauna and fish fauna (EU 2000, WFD Annex V 1.2.1). The status of fish fauna will be assessed using species composition, abundance, sensitive species, age structure and reproduction. Current methods of assessing ecological quality, including diversity indices, methods and techniques only include one or two characteristics of the fish fauna. A new approach to provide integrated measures of ecological health is required by the WFD (EU 2000, WFD Annex V 1.2.1). This study is the first attempt to develop fish based assessments of ecological status in UK rivers to meet the WFD.

1.3 USE OF FISH AS BIOLOGICAL INDICATORS

Many groups of organisms have been proposed as indicators of environmental quality, but no single group has emerged as the favourite of most biologists. Diatoms (Patric 1973, 1975), benthic invertebrates (Resh & Unzicker 1975, Hilsenhoff 1977, 1987, Mason 1978), macroinvertebrates (Schaeffer *et al.* 1985, Rosenberg & Reash 1993) and amphibians (Moyle *et al.* 1986, Fisher 1989) have most frequently been cited as ideal organisms for biological monitoring programmes.

Taxa other than fish (e.g. macroinvertebrates, diatoms) have been widely used in monitoring because of the availability of a theoretical substructure that allows an integrated ecological approach (Cummins 1974, Vannote *et al.* 1980, Canfield & Jones 1984). However, the use of diatoms or invertebrates as monitoring targets has major deficiencies. For example, they require specialised taxonomic expertise; they are difficult and time-consuming to sample, sort and identify; back-ground life-history information is often lacking for many species and groups; and the results obtained using diatoms and invertebrates are difficult to translate into values meaningful to the general public.

The use of fish as indicator species has also been proposed (Sprague 1973, USEPA 1977). Fish (cold blooded aquatic vertebrates, which respire by means of internal gills and swim by means of paired or unpaired fins) have numerous advantages as indicator organisms for biological monitoring programmes. Fish are present in all but the most polluted aquatic environments and many freshwater fish remain in the same general area during all seasons. Fish are sensitive to a wide array of direct stresses but are relatively long-lived (3 to 10+ years) and can provide a long-term record of environmental stress and current water resource quality. The life-history, biology and ecology of most fish species are well known and therefore, relatively easy to identify. Hence, technicians require relatively little training. Indeed, most samples can be sorted and identified at the field site, with release of study organisms after processing. Fish occupy a variety of trophic levels (omnivores, herbivores, insectivores, planktivores, piscivores). This helps to provide an integrative view of the watershed environment. Fish are a highly visible component of the aquatic community and the general public can relate to statements about conditions of the fish community. Aquatic life uses and regulatory language are generally characterised in terms of fish (e.g. EC salmonid & cyprinid designated rivers; WFD high, good and moderate status of rivers (EU 2000); fishable water, fishable and swimmable, Clean Water Act, USA). Finally, fish communities can be used to evaluate societal costs of degradation more directly than other taxa because their economic and aesthetic values are widely recognised (Fausch *et al.* 1990).

There are also a number of disadvantages of using fish as indicators. These include fish migration on diel and seasonal time scales, manpower needs to sample the communities and the selective nature of sampling. However, these are disadvantages associated with all major taxa. However, on a comparative basis, training periods for fish identification are likely to be shorter and the technology required less sophisticated than for other taxa. Field sampling may be slightly more costly, but laboratory time will be relatively small.

1.4 OBJECTIVES OF THE STUDY

To evaluate anthropogenic impacts on fish communities, numerous diversity indices (Margalef, Simpson, Shannon-Wiener indices etc.) ranging from the very simple, e.g. species richness, to very complex, e.g. Cairns SCI (Cairns 1971), have been developed. However, no single index satisfies all conditions for derivation of biotic integrity (Washington 1984), as most indices are based on attributes either from individuals or populations but not from all structural levels. Debate has been ongoing for decades on the advantages of one index over another (Hurlbert 1971, Peet 1974, Usher 1983). There are also numerous graphical techniques such as "K" dominance plots (Shaw *et al.* 1983) and ABC – Abundance / Biomass Comparison curves (Warwick 1986), which allow visual inspection of the structure of the fish communities in terms of abundance, species richness and biomass. These methods also lack integration of community attributes to measure the biotic integrity. In addition, a wide variety of computer-based multivariate methods, e.g. UPGMA, TWINSPAN, DECORANA have been developed to evaluate the status of fish communities. These methods also have limitations for the production of meaningful indices, as they usually fail to cope with the biological meaning of attributes of the fish communities. In short, indices, methods and techniques based on physical and chemical attributes of water are inadequate as surrogates for measuring biotic integrity (Karr & Dudley 1981).

The Index of Biotic Integrity (IBI), a multimetric index (Karr 1981) is however, capable of evaluating health and condition of an aquatic ecosystem. The IBI is designed to indicate the degree to which a watercourse has been impacted by pollution or morphological degradation through a measure of the health of the fish assemblages (Welcomme 2001). The IBI fulfils all the criteria needed to evaluate water resource quality for all types of habitats and ecosystems. The combination of metrics permits insights from individual, population, assemblage, ecosystem, and zoogeographic perspectives (Miller *et al.* 1988). Furthermore, there is no loss of information as the structure and calculation of IBI preserves both the original data and also provides a metric-by-metric evaluation of stream condition (Harris 1995).

The aim of this study is to develop a simple but effective measure of ecological health for English lowland rivers using the structural and functional characteristics of individual fish, population and community. This implies developing an IBI for use in English lowland rivers. As the IBI was originally developed for species-rich warmwater rivers in Midwestern States of the USA (Karr 1981), this will involve adaptation of the original concept of the IBI but modifying it to meet the requirements of English lowland rivers with lower species diversity. To achieve the aim of the study, the specific objectives are:

i. to establish a reference condition for English lowland rivers;
ii. to compare different indices to evaluate fish species diversity, richness, assemblage and abundance in English lowland rivers;
iii. to develop an IBI model for English lowland rivers;
iv. to test the IBI model for English lowland rivers.

It is anticipated the output will form the basis of a tool for measuring ecological health of English lowland rivers that can be used to meet the obligations of the UK under the WFD.

This thesis is presented in six Chapters. The first Chapter is a general introduction on degradation and anthropogenic impacts on English and Welsh riverine fisheries, legislation for water resources management, background to and justification of use of fish as bioindicators, methods available for measurement of anthropogenic impacts and objectives of the study.

In the second Chapter, the general methodology, including description of study sites, timing of survey, sampling method, data analysis, and description of study rivers, are presented. Methodologies for specific indices are described in respective Chapters.

The process of establishment of reference condition for English lowland rivers is described in Chapter three. Establishment of reference condition is crucial for calculating an IBI for a particular region. The process includes evaluation of origin and distribution of British freshwater fish, introduced fishes, classification of fishes on the basis of guild concept, habitat preference and tolerance to environmental degradation, density and biomass of fishes.

In Chapter four, the different diversity indices (Margalef, Simpson, and Shannon-Wiener indices, the ABC method, and multivariate methods (UPGMA, TWINSPAN and DECORANA)) used in this study to evaluate fish species diversity, richness, assemblage and abundance are compared. Both advantages and limitations of these indices are discussed. The Chapter describes the possibility of using these different indices, methods and techniques to measure ecological health of study rivers.

The process of IBI development is described in Chapter five. The process includes establishment and modification of IBI metrics, rating and scoring criteria of metrics, and application and testing of the IBI in English rivers. Rating and scoring criteria include evaluation of existing rating and scoring methods, defining scoring scale, integrity classes with class boundaries and these are described together with the calculation and scoring procedures of individual metrics. Finally, application and testing of the IBI in various English rivers is presented in this Chapter.

A general discussion on all aspects relating to the IBI development is presented in Chapter six. This includes critical evaluation of the suitability of the method in comparison to other available methods. On the basis of these discussions, a number of recommendations are made for future research.

CHAPTER TWO

2. GENERAL METHODOLOGY AND DESCRIPTION OF STUDY RIVERS

2.1 STUDY SITES

Three networks of English rivers from the Thames, the Midlands and the Northeast regions of the EA were selected for this study as these regions had the best available fish stock assessment data. Rivers were chosen from three major catchments, the Thames, Trent and Yorkshire Ouse depending on the availability of fishery data. The Thames catchment included five rivers, the Cherwell, Evenlode, Stort, Thame and Windrush. From the Trent catchment 15 rivers, the Anker, Blithe, Blythe, Churnet, Cole, Derwent, Idle, Mease, Penk, Sence, Soar, Sow, Tame, Tean and Trent, were included. The rivers Aire and Nidd were selected from the Yorkshire Ouse catchment (Table 2.1).

An appropriate choice of sampling sites is critical for the successful development and application of the IBI (Lyons 1992). Sites chosen for sampling should be representative of the overall habitat of the stream reach. Sampling areas should not normally include bridges, dams, mouths of the tributaries, or other atypical habitat features, unless the goal of the sampling is to characterise the influence of these atypical features on local environment. Fish assemblages in the vicinity of atypical habitat features are often not representative of the overall fish community of a stream (Lyons 1992).

In this study, all the rivers and respective sites were selected by the EA and its predecessors (The Regional Water Authorities & the National Rivers Authority, NRA) for their routine monitoring programmes. The number of sites and types of sites chosen for a particular river usually reflected the needs of the project and the scientist at the time of selection. A single site on the whole waterbody system may provide an estimate of population for that particular area of the waterbody, while a number of sites in the target area will each provide an estimate giving a clearer indication of the species diversity, assemblage structure and stock level in the river. The size of the sites is also an aspect for consideration. The EA selected the sites within the catchment in such a way that the sites represented the diversity of fish and habitats in the river.

A total of 457 sites from 22 rivers in 3 catchments were chosen for this study (Table 2.1). The number of sites on a particular river varied from 5 to 182, usually related to the length, width, depth, and habitat features of the river. Importance of the river with respect to fishery resources and other considerations also determined the number of sites on a river. Habitat variability and access difficulties also influenced the choice of location of sites (Ward *et al.* 1993, Harvey & Cowx 1996). Details of all rivers and sites are presented in Appendix 2.1.

2.2 FISH DATA COLLECTION

2.2.1 Timing of survey

Selecting the appropriate time of year for sampling is crucial. No single best period for sampling can be defined and therefore, samples should be taken at critical times in a year. In general, periods of low to moderate stream flow are preferred and the relatively variable flow conditions of early spring and late summer / autumn avoided. Karr (1981) suggested early summer as the ideal time in the USA as it is the least variable period of the year. In English rivers, spawning migrations of coarse fish usually take place in early spring and summer. This may increase or decrease the diversity spectrum of fishes. The spawning migrations of diadromous species are not considered a problem in respect of the IBI for, if diadromous species enter into the river, this will be reflected in an improved quality of the river. However, seasonal movements of resident species may affect the site specific IBIs.

Timing of the surveys was different for different catchments, due to different objectives of the surveys. Data was available for sites on the Thames catchment between 1990 and 1996, whilst the Trent catchment data covered 1989 to 1993. Data was available for the Yorkshire Ouse catchment between 1990 and 1996 (Table 2.1). These represented the most comprehensive data available from the EA during the period of the IBI development.

Survey data covered both summer and winter periods and different stages of the breeding season and lunar cycle. All sites were surveyed in the daytime. The survey interval varied from 2 months to 3 years for different rivers. Intervals between sampling specific sites, within a survey period for a particular river, were different and varied between 30 days to 90 days. The schedule of sampling for different rivers is shown in Table 2.2.

2.2.2 Sampling method

The choice of sampling method is very important as the outcome and final results may vary considerably between and within waterbodies. However, sampling methods vary, depending on the waterbody, habitat, depth, flowing condition, bottom substrate, season, size and variety of fish and nature of data to be collected. In England and Wales approximately 40% of the fisheries are in waters that cannot be sampled by netting. In this study, most selected rivers were considered large (>15 m wide and >1 m deep) (Harvey 1996). Many techniques, such as electric fishing, seine and gill netting and angler census, have been variously attempted in large rivers but none totally satisfy the requirements for assessing all aspects of fish population structure, size, distribution and seasonal variation (Kell 1991). The problem of sampling is best minimised by using a wide variety of sampling gears for the collection of data (Hay *et al.* 1996). Unfortunately, this is very costly, time consuming and labour intensive.

For this study, from the different sampling methods, electric fishing was chosen as a method of sampling fish populations in large rivers. Electric fishing is a sampling technique that has been in use for over 90 years (Vibert 1967). It is a tool used extensively to catch fish in stock assessment exercises (Cowx 1990a, Cowx & Lamarque 1990). Electric fishing is one of the least selective of fishing methods (Begenal & Tesch 1978). Electric fishing can be used in many situations, with hand-held gears being effective in small streams and small rivers and boat-mounted gears being more suited to use in large rivers and canals. The gear can also be used in stillwaters however, efficiency is reduced in large water bodies especially where depths of >5 m exists. Although considered somewhat selective to larger fish, especially to fish greater than 10 cm, electric fishing has immense potential for sampling large rivers for stock assessment purposes (Zalewski & Penczak 1981). The electric fishing method is not manpower intensive, allows the large scale removal of fish and can be used to survey long lengths of river to provide more detailed information on the community structure and population characteristics (Harvey 1996). However, the method suffers from many limitations primarily related to depth and width of the water body being surveyed, as well as factors such as water velocity, conductivity and water clarity (Zalewski & Cowx 1990).

Fish populations were sampled by the NRA and EA using different types of electric fishing gear in different catchments. However, the basic method and application were the same in all regions of the UK despite some operational differences between the Thames, Midlands and Northeast regions. In all these regions, boommounted electric fishing gear was used. Midlands used both straight multiple-anode (usually 10) (Cowx *et al* 1988) and ring arrays while Northeast used a simplified version (4 anodes) of the straight boom array. Midlands and Northeast systems required 4 personnel whilst the Thames version used 6 persons. Output for the fixed anode arrays varied.

Table 2.1 Rivers from different catchments with number of study sites

	Name of River	Length (km)	Catchment area (km^2)	Mean flow (m^3 s^{-1})	Number of sites
Thames catchment					
1	Cherwell	96	904	-	13
2	Evenlode	68	435	-	20
3	Stort	46	278	-	16
4	Thame	77	684	5.0	18
5	Windrush	73	591	-	19
Trent catchment					
1	Anker	-	368	38.34	10
2	Blithe	-	-	-	11
3	Blythe	45	162	-	9
4	Churnet	-	-	-	16
5	Cole	-	-	-	14
6	Derwent	120	1586	18.30	15
7	Idle	49	1290	15.86	5
8	Mease	-	-	-	7
9	Penk	-	-	-	11
10	Sence	-	-	-	6
11	Soar	20	1360	21.41	15
12	Sow	-	163	10.00	9
13	Tame	84	799	19.70	6
14	Tean	-	-	-	9
15	Trent	280	10,550	82.5	20
Yorkshire Ouse catchment					
1	Aire	148	1100	-	26
2	Nidd	-	1555	-	182
Total	**22**				**457**

The Thames system was powered from a 7.5 kVA generator and had a square wave pulsed DC output of 100 Hz and a 50% duty cycle. The Northeast region linear array system was powered from a 4 kVA generator with a 100 Hz, ¼- sine pulsed DC output. The Midland systems produced several outputs including 50, 100 and 300 Hz square wave at varying duty cycles and was generally powered by 4 kVA or 7.5 kVA generators. In headwaters, where the river is narrow, shallow and steep, fish were sampled by 3 or 4 personnel wading with one or two hand-held electrodes. Output was usually pulsed DC at 50 or 100 Hz from a smaller generator.

Table 2.2 Sampling periods for the study rivers in different catchment

	Name of River	Sampling period	No. of sites
Thames catchment			
1	Cherwell	February 1993 to February 1996	13
2	Evenlode	September 1992 to April 1993	20
3	Stort	October 1990 to February 1991	16
4	Thame	November 1990 to July 1991	18
5	Windrush	July 1993 to May 1994	19
Trent catchment			
1	Anker	November 1989 to March 1993	10
2	Blithe	May 1990 to May 1993	11
3	Blythe	February 1989 to March 1993	9
4	Churnet	August 1989 to July 1993	16
5	Cole	May 1989 to June 1992	14
6	Derwent	February to March 1993	15
7	Idle	February to March 1992	5
8	Mease	October 1990 to June 1991	7
9	Penk	August 1990 to September 1992	11
10	Sence	November 1989 to July 1992	6
11	Soar	May to August 1992	15
12	Sow	July 1990 to November 1991	9
13	Tame	May 1989 to June 1992	6
14	Tean	June 1990 to July 1993	9
15	Trent	January 1989 to October 1992	20
Yorkshire Ouse catchment			
1	Aire	March to April 1990	26
2	Nidd	July 1995 to August 1996	182
Total	**22**		**457**

2.2.3 Data analysis

Karr (1981) suggested that a sample from 100 m of stream is sufficient in small streams. However, large rivers should be sampled in 1 km units when electric fishing equipment is employed. In the present study, site lengths varied between 50 and 1,045 m. The sections were not always isolated by stop nets as natural features at the extremities of the sites were used as obstacles where possible. All major habitats within each site were sampled carefully to obtain a representative sample of the fish assemblage. In most cases, two or more runs were made at each site but at other sites, single runs were taken due to the small catch. In deeper waters, exceeding 1 m in depth, electric fishing was carried out from a small fibreglass boat, moving in an upstream direction. Fish stunned or immobilised by the electric current were rapidly collected with hand nets and transferred to large open plastic containers of water. All fish captured were enumerated by species and weighed (nearest g). Large numbers of small fish were batch weighed for each species. Subjective abundance was made for minor species.

After taking measurements, all the fish were returned to the river after a brief recovery period. A standard procedure was adopted to reduce handling time of the fish and, hopefully, to reduce stress effects on the fish.

2.3 DESCRIPTION OF STUDY RIVERS

2.3.1 The Thames catchment

River Cherwell

The source of the River Cherwell (Fig. 2.1) is near Charwelton (National Grid Reference, NGR: SP 521 572). The river flows in a generally southerly direction to join the River Thames in Oxford (NGR: SP 520 050), a distance of 96 km. The average gradient is 1 in 783 (EA, LEAP 2000e). The River Cherwell has 11 tributaries including several trout streams. The Oxford Canal runs parallel to the River Cherwell for much of its length, crossing over at Aynho and sharing the same channel for a distance of approximately 2 km near Shipton on Cherwell.

The River Cherwell receives the maximum consented discharge of 15,600 and 9,000 m^3d^{-1} from two major Sewage Treatment Works (STWs) (Banbury and Leicester), respectively. Moreover, river engineering, mills and land drainage practices have significantly affected habitat quality in the Cherwell catchment. In January 1995, over 15,000 fish were killed due to detergent pollution in the reach from Banbury to Clifton (EA, LEAP 2000e). The NRA reported 14 fish species during their survey in 1988. Roach, dace (*Leuciscus leuciscus* (L.)), chub and pike (*Esox lucius* L.) dominated the fauna. Total biomass ranged between 9.3 and 30.8 g m^{-2} indicating a good fishery (Lewis 1991).

River Evenlode

The source of the River Evenlode (Fig. 2.2) is near Moreton-in-Marsh (NGR: SP 175 332) from where it flows 68 km in a generally southeast direction to Cassington. The River Evenlode splits into 2 channels at Cassington Mill and both channels flow approximately 1 km to create 2 confluences with the River Thames (NGR: SP 454 094 & NGR: SP 457 098). The mean gradient of the River Evenlode is 1 in 755. The catchment area is 435 km^2, the majority of land use being agriculture (EA, LEAP 1996). The River Glyme is the main tributary while the other significant tributaries are Cornwell brook, Chadlington stream, Coldron brook and Littlestock brook.

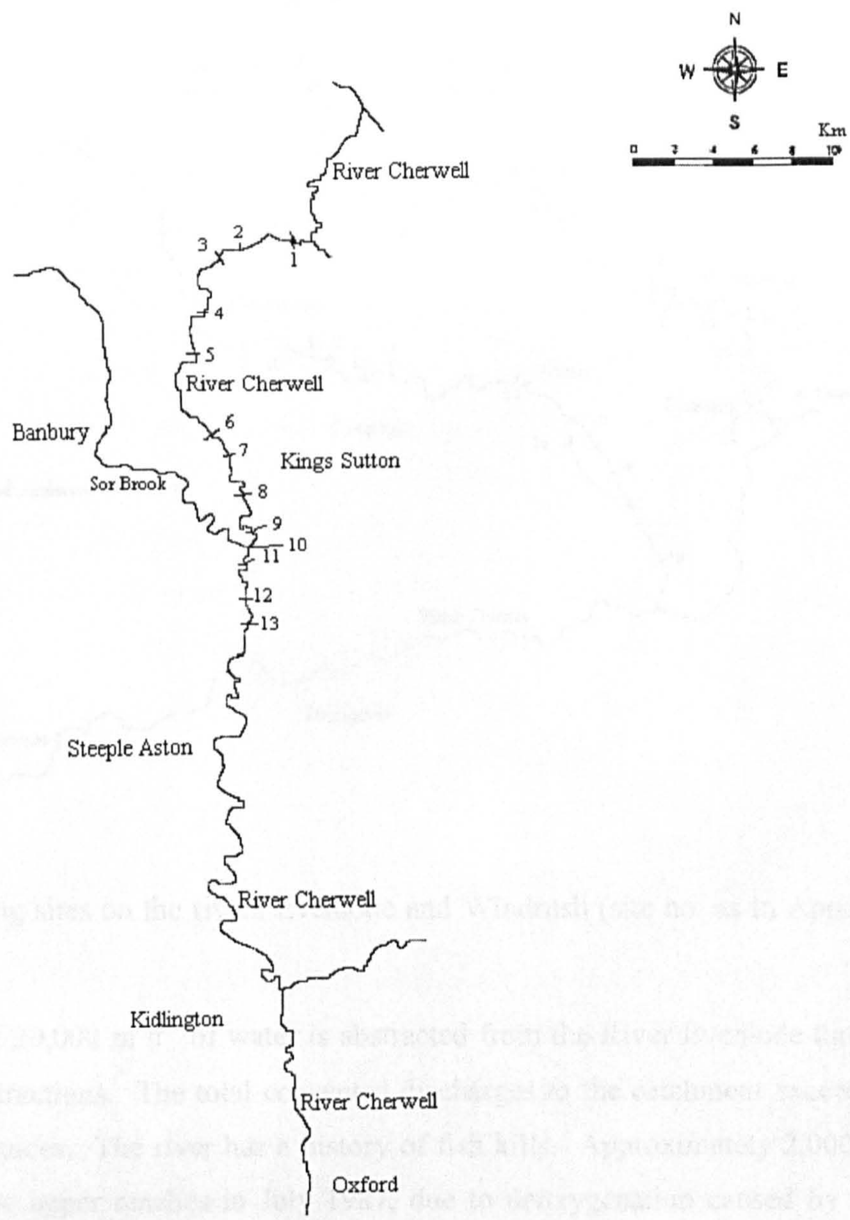


Fig. 2.1 Sampling sites on the River Cherwell (site no. as in Appendix 2.1)

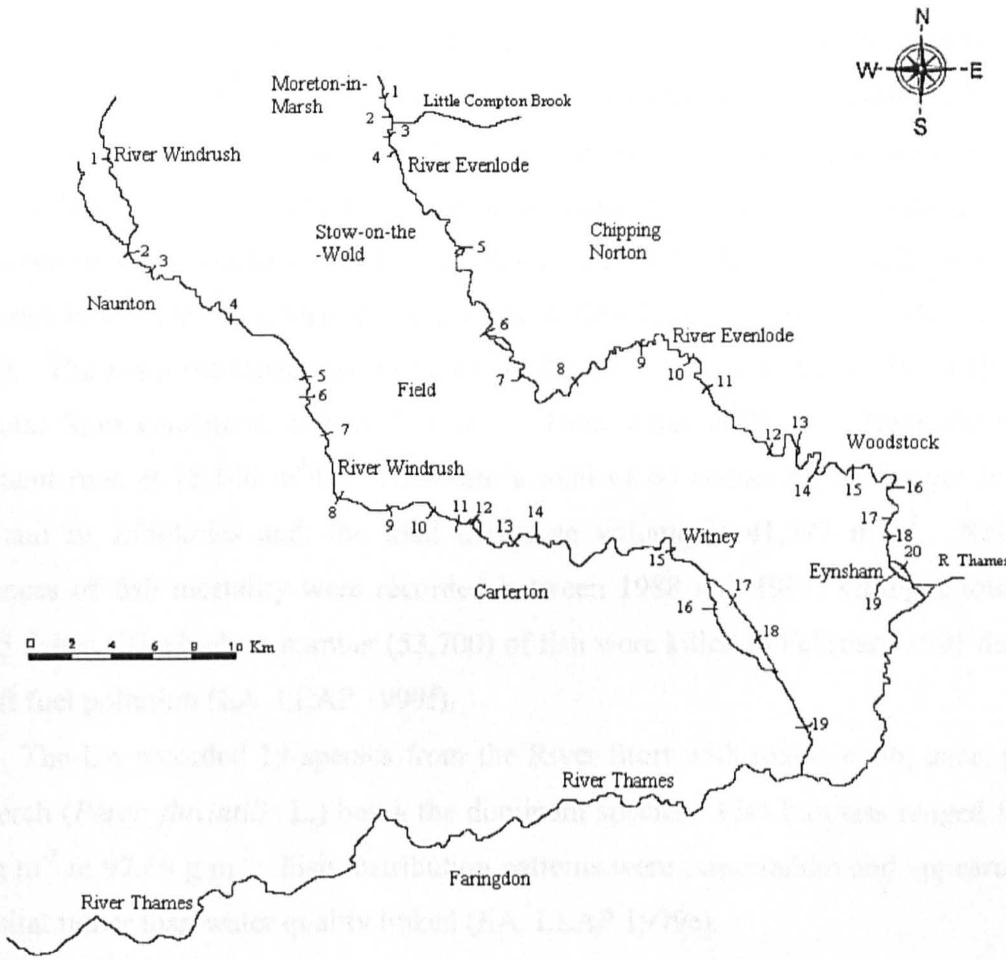


Fig. 2.2 Sampling sites on the rivers Evenlode and Windrush (site no. as in Appendix 2.1)

A total of 20,000 m^3d^{-1} of water is abstracted from the River Evenlode through 118 licensed abstractions. The total consented discharges to the catchment exceed 0.4 m^3s^{-1} from 11 sources. The river has a history of fish kills. Approximately 2,000 fish were killed in the upper reaches in July 1987, due to deoxygenation caused by farm effluent sludge (EA, LEAP 1996).

The NRA reported 19 species from the river during their survey in 1982. The River Evenlode has a moderate to good fishery, dominated by salmonids in the upper reaches and by cyprinids in the lower reaches. Brown trout, *Salmo trutta* L. were present sporadically through its length. Fish biomass ranged between 10 and 20 gm^{-2} (EA, LEAP 1996).

River Stort

The River Stort (Fig. 2.3) rises from High Wood near Langley in Essex (NGR: TL 425 357) and flows south via Bishops Stortford (NGR: TL 490 210) to Harlow (NGR: TL 450 100). From Harlow, the course of the river veers south-westerly, past Roydon to join the River Lee at Fields Weir near Hoddesdon in Hertfordshire (NGR: TL 391 093). From there the combined river system continues to flow southwards into the River Thames. The River Stort is one of the main tributaries of the River Lee. The total distance from source to confluence is 46 km, while the total length, including tributaries is 184 km. Fourteen minor tributaries flow into the River Stort (EA, LEAP 1999e). The main tributaries are the great Hallingbury Brook and the Pincey Brook. The total Stort catchment area is 278 km^2. Total water abstraction from the Stort catchment runs at 75,140 m^3d^{-1}. There are a total of 60 consented discharges to the Stort and its tributaries and, the total discharge volume is 41,397 m^3d^{-1}. Several incidences of fish mortality were recorded between 1988 and 1991, killing a total of 54,325 fishes. The highest number (53,700) of fish were killed in February 1991 due to aircraft fuel pollution (EA, LEAP 1999f).

The EA recorded 19 species from the River Stort with roach, chub, dace, pike and perch (*Perca fluviatilis* L.) being the dominant species. Fish biomass ranged from 1.52 g m^{-2} to 97.69 g m^{-2}. Fish distribution patterns were non-random and appeared to be habitat rather than water quality linked (EA, LEAP 1999e).

River Thame

The source of the River Thame (Fig. 2.4) is near the village of Marsworth (NGR: SP 921 150) from where it flows in a generally southwest direction for 77 km to join the River Thames near Dorchester (NGR: SU 578 933). The mean gradient is 1 in 1,540, which is very shallow, compared to other rivers in the Upper Thames catchment. Bear Brook and Scotsgrove Brook are two tributaries of the River Thame (EA, LEAP 1998e). The River Thame receives consented discharges of 19,100 m^3d^{-1} and 443 m^3d^{-1} from Aylesbury and Marsworth STW, respectively. Scotsgrove Brook receives effluent from five large STWs, which also affects the water quality and quantity of the River Thame. A total of seven incidences of fish mortality were recorded between 1987 and 1991 due to pollution (EA, LEAP 1997a). Cyprinids, e.g. roach, dace, chub and barbel (*Barbus barbus* (L.)) dominate the river (Thames Water 1985). The River Thame is an excellent fishery for much of its length with a biomass range of 10 to 20 g m^{-2} (EA, LEAP 1997a).

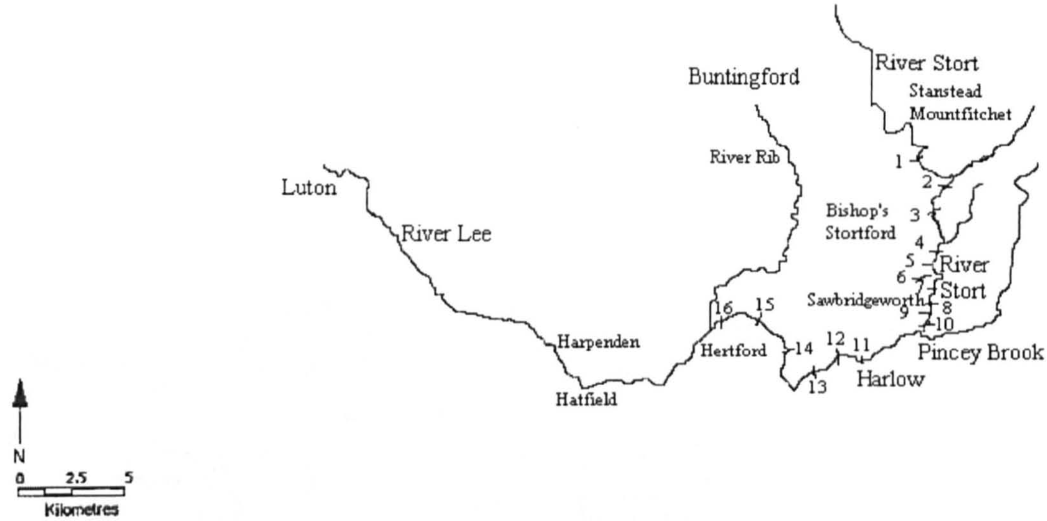


Fig. 2.3 Sampling sites on the River Stort (site no. as in Appendix 2.1)

River Windrush

The River Windrush (Fig. 2.2) originates on the Cotswold limestone approximately 4 km north of the village of Temple Guiting near Taddington (NGR: SP 094 315) from where it flows 73 km in a generally southeast direction to join the River Thames at Newbridge (NGR: SP 403 015). The river splits into 2 channels (East arm and West arm) at Witney, from where both channels flow 11 - 12 km to rejoin near Standlake. The mean gradient is 1 in 441. The main tributaries are the River Dikler, River Eye and the Sherborne Brook. The total catchment area is 591 km^2, the majority of land use being agriculture (EA, LEAP 1996). The total licensed abstraction amounts to 32,000 m^3d^{-1}, about 10% of the average available water. There are three main consented discharges to the River Windrush and its tributaries. The total discharge volume is 22,527 m^3d^{-1}. Several incidences of fish mortality due to water pollution have been recorded. The highest number (2000) consisted of minnows (*Phoxinus phoxinus* (L.)) but a large number of crayfish (*Astacus astacus* L.) and other macroinvertebrates were also killed due to pesticide pollution at Naunton (EA, LEAP 1996). The NRA (1986) recorded 19 species from the river with brown trout, chub and dace being the dominant species (EA, LEAP 1996).

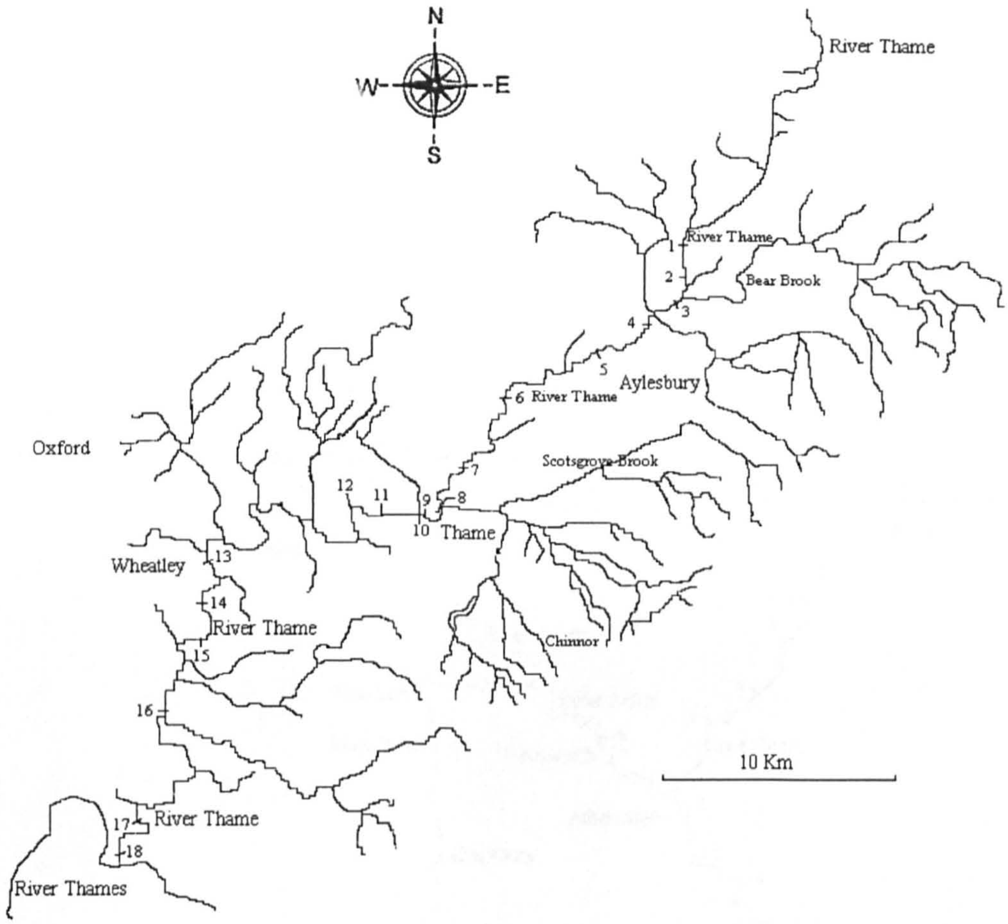


Fig. 2.4 Sampling sites on the River Thame (site no. as in Appendix 2.1)

2.3.2 The Trent catchment

River Anker

The River Anker rises around Wolvey and runs northwest through agricultural land, receives drainage from Hinckley, Bedworth, Nuneaton and Atherstone before joining the River Tame at Tamworth (Fig. 2.5). The Anker is a major tributary of the River Tame and it is itself joined by the River Sence (EA, LEAP 2000a). The catchment of the Anker is 368 km^2. The average flow is 38.34 m^{3}s^{-1} at Polesworth. The river is important as a carrier of treated effluent from several large water reclamation works (WRWs), which in the past have been the cause of fish mortality.

The River Anker has relatively low fish standing crop due to high levels of ammonia. The fish population varies greatly but few areas contained large numbers of fish. A total of eight species were recorded from the river at Fieldon Bridge and the fishery was dominated by roach and dace (EA, LEAP 2000a).

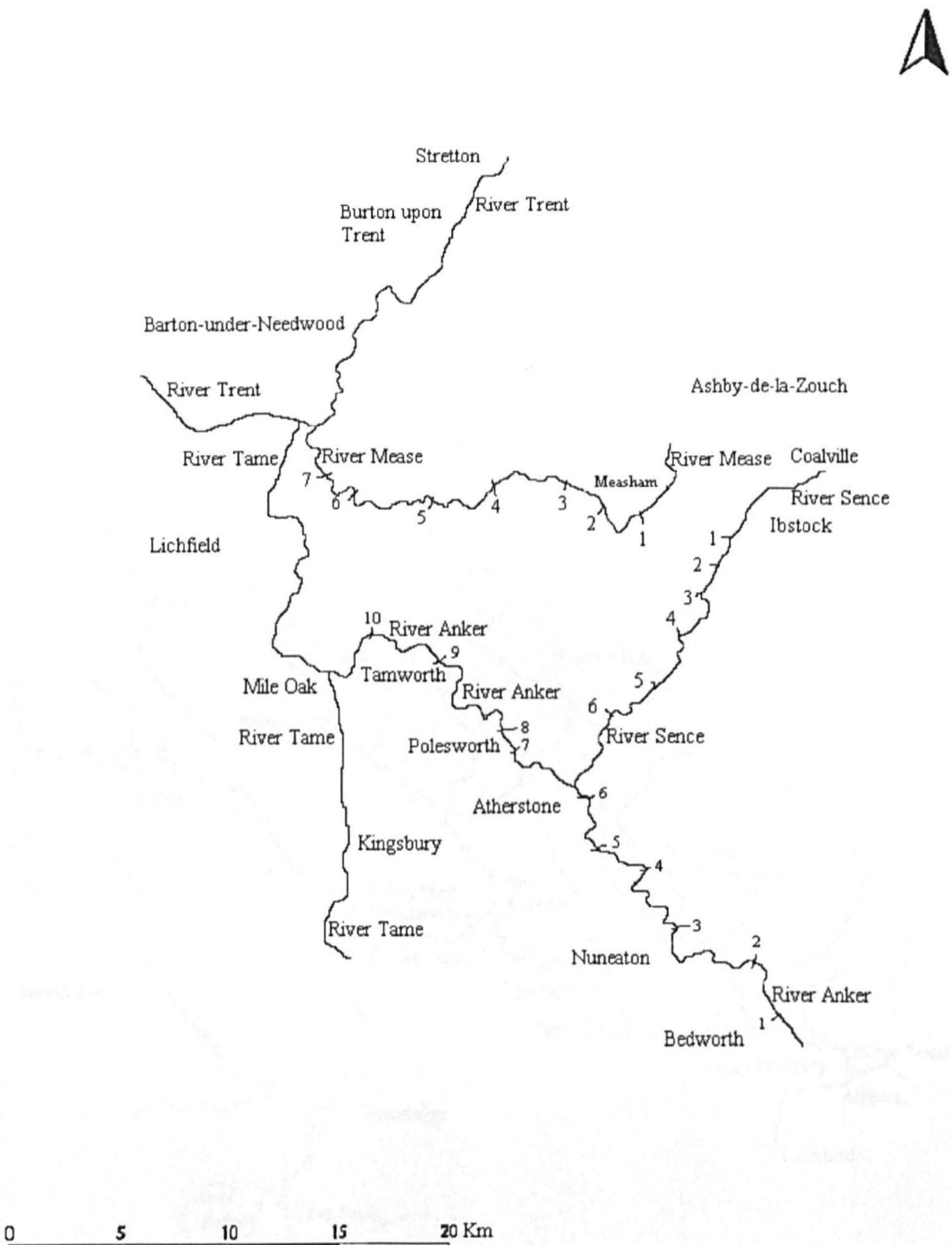


Fig. 2.5 Sampling sites on the rivers Anker, Mease and Sence (site no. as in Appendix 2.1)

River Blithe

The River Blithe rises to the east of Stoke-on-Trent and flows south-easterly to join the River Trent near King's Bromley (Fig. 2.6). The Blithe is a small river, which meanders through pasture land and has typical pool-riffle topography. In its lower reaches the river passes through Blithfield Reservoir, a 287-ha water supply reservoir. This is a public water supply river, containing good to very good quality water (EA, LEAP 1997d).

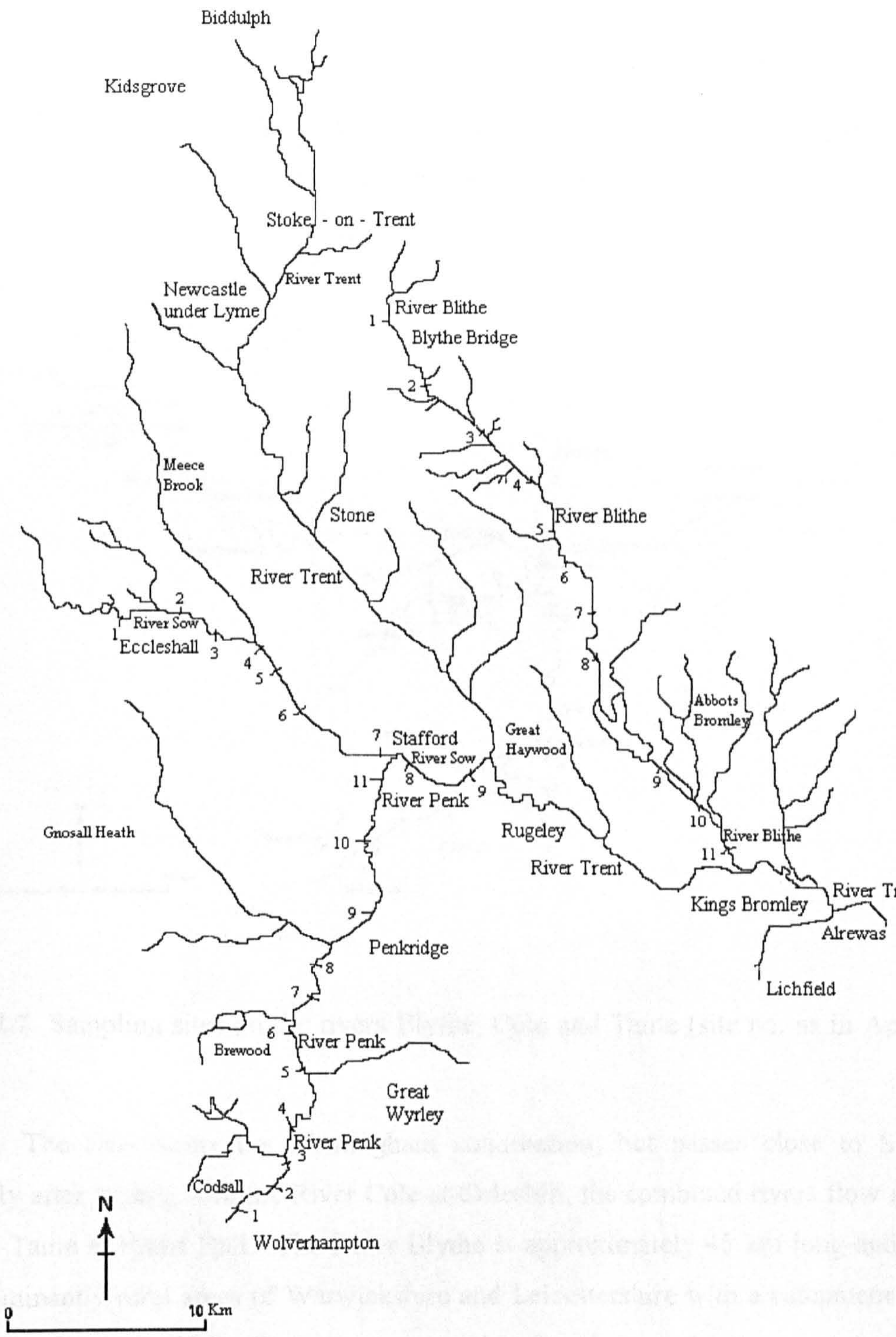


Fig. 2.6 Sampling sites on the rivers Blithe, Penk and Sow (site no. as in Appendix 2.1)

The fish community comprises eleven species and the river is dominated by chub and dace, while minnow are common. The River Blithe also holds good brown trout and grayling (*Thymallus thymallus* (L.)) populations (EA, LEAP 1997d).

River Blythe

The River Blythe is a high quality rural river that rises in the southwest of Earlswood and is used as a source of public water supply (Fig. 2.7).

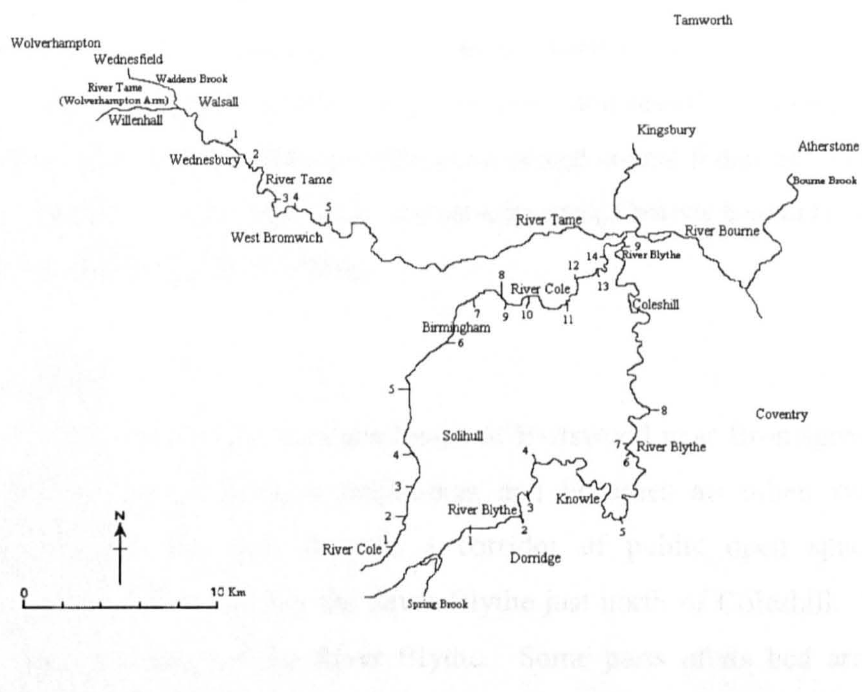


Fig. 2.7 Sampling sites on the rivers Blythe, Cole and Tame (site no. as in Appendix 2.1)

The river skirts the Birmingham conurbation, but passes close to Solihull. Shortly after joining with the River Cole at Coleshill, the combined rivers flow into the River Tame at Hams Hall. The River Blythe is approximately 45 km long and drains predominantly rural areas of Warwickshire and Leicestershire with a catchment of 162 km^2 (EA, LEAP 1998d). River terrace gravel is found along the river and deposits of alluvium are found on its floodplains (EA, LEAP 1998d). The river suffers from eutrophication problems as it receives treated effluent from many STWs. In 1996, the river suffered from 75 pollution incidents. However, the river supports prolific populations of coarse fish throughout most of its length. Coarse fish populations are dominated by roach, dace, chub and gudgeon (EA, LEAP 1998d). The River Blythe provides good angling for trout and coarse fish. Both rainbow trout (*Oncorhynchus mykiss* (Walbaum)) and brown trout are introduced annually in the middle reaches of the river to facilitate the operation of put and take fisheries (EA, LEAP 1998d).

River Churnet

The River Churnet rises north of Leek and to the east of the Roaches at a height of 459 m above sea level (Fig. 2.8). The River Churnet is one of the tributaries of the River Dove that joins at Rocester. The upper reaches are fast flowing in nature with cool and unpolluted water. Water of the upper reaches, upstream of Leek, is of good quality, while the lower reaches are of fairly good quality (EA, LEAP 1999d). The river loses water as the Caldon Canal obtains its water from the River Churnet below Leek. The river supports a small fish population, dominated by brown trout (EA, LEAP 2000c). The lower reaches provide good mixed coarse fisheries. The coarse fishery is dominated by roach, dace, chub, common bream (*Abramis brama* (L.)), and grayling are also present (EA, LEAP 1999d).

River Cole

The River Cole rises southwest of Earlswood near Bromsgrove (Fig. 2.7). The river then leaves its rural beginnings and becomes an urban river through south Birmingham, then runs through a corridor of public open space and into open countryside before joining the River Blythe just north of Coleshill. The River Cole is the main tributary of the River Blythe. Some parts of its bed are heavily silted or contain discarded domestic or industrial waste items. River terrace gravel are found along the river (EA, LEAP 1998d). The principal tributary is the Hatchfors Brook, which drains an area including Birmingham International Airport. The run-off from the M42 Motorway drains to the river. Moreover, the river receives sewage and industrial discharges, causing pollution. In 1996, 88 pollution incidents occurred in the river (EA, LEAP 1998d). The River Cole suffered from major pollution in 1970 and at that time no fish were found in the river. By 1980, five species were reported from the River Cole, which was dominated by gudgeon. In recent years the fisheries of the upper reaches have been improved by restocking with dace, chub and brown trout (EA, LEAP 1998d).

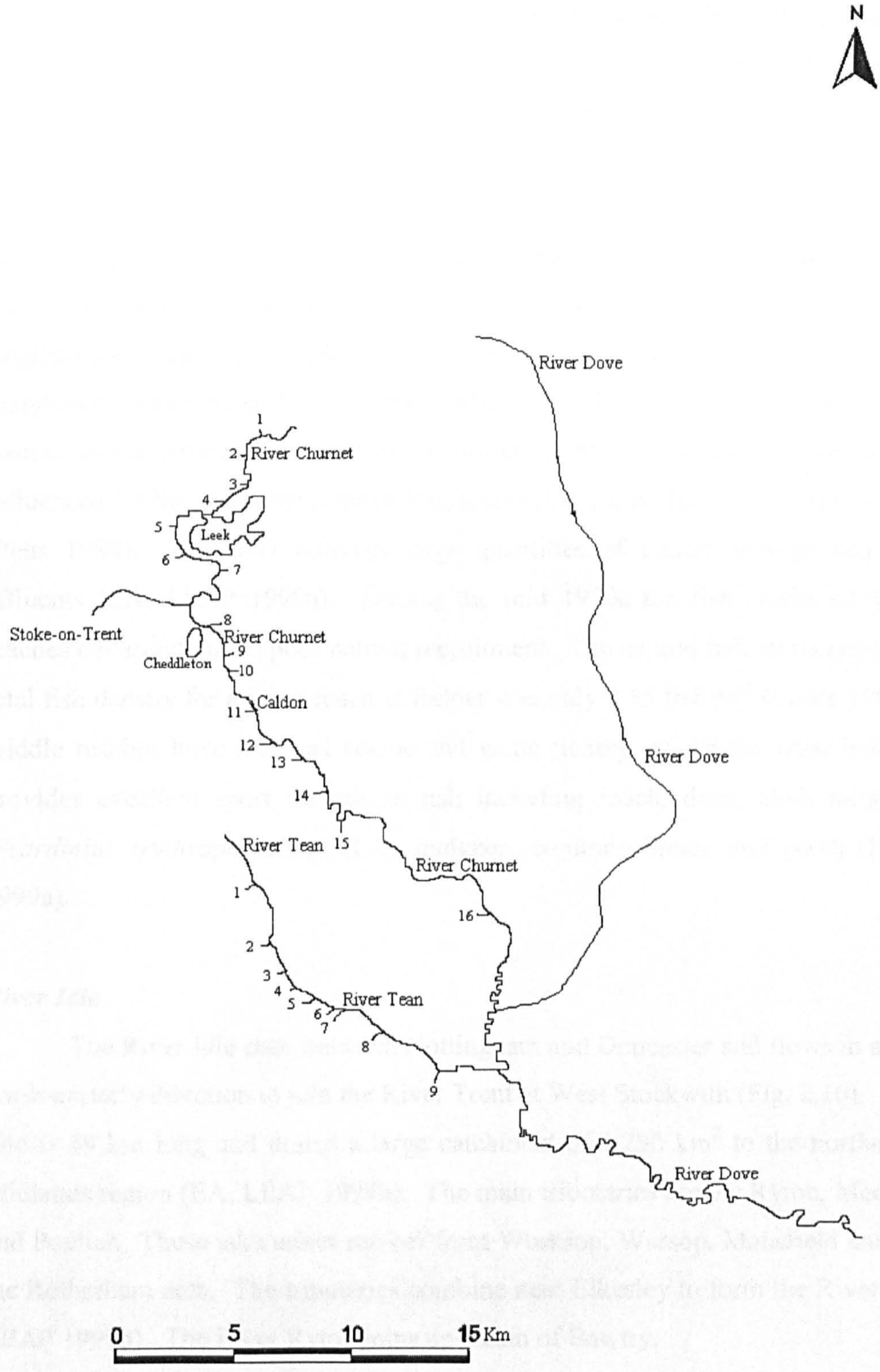


Fig. 2.8 Sampling sites on the rivers Churnet and Tean (site no. as in Appendix 2.1)

River Derwent (Derbyshire)

The River Derwent rises on Howden Moor, amongst the southern peaks of the Pennine range (Fig. 2.9). The river flows in a southeast direction for 110 km until the confluence with the River Trent (EA, LEAP 1999a). The River Derwent is a major tributary of the River Trent. The River Derwent and tributaries have a catchment of 1,586 km^2 (NERC 1996) while the Derwent alone has a drainage area of 1,200 km^2 (Severn-Trent Water Authority 1983). The rivers Ashop, Noe, Wye and Amber are the four main tributaries. The River Derwent is one of the most heavily managed rivers in England and Wales (EA, LEAP 1999a). Three large reservoirs, Howden, Derwent and Ladybower, have been built in its headwaters. In addition there are large river abstractions at Ambergate, Little Eaton and Draycott. Discharge and flow patterns are influenced by the upstream impounding reservoirs and surface or groundwater storage (Petts 1988). The river receives large quantities of treated sewage and industrial effluents (EA, LEAP 1999a). During the mid 1970s the fish stocks of the middle reaches collapsed due to poor natural recruitment. The natural fish fauna is poor and the total fish density for a 5 km reach at Belper was only 0.35 fish m^{-2} (Cowx 1990b). The middle reaches have a mixed coarse and game fishery, whilst the river below Derby provides excellent sport for coarse fish including roach, dace, chub minnow, rudd (*Scardinius erythrophthalmus* (L.)), gudgeon, common bream and perch (EA, LEAP 1999a).

River Idle

The River Idle rises between Nottingham and Doncaster and flows in a generally north-easterly direction to join the River Trent at West Stockwith (Fig. 2.10). The River Idle is 49 km long and drains a large catchment of 1,290 km^2 to the northeast of the Midlands region (EA, LEAP 1999b). The main tributaries are the Ryton, Meden, Maun and Poulter. These take urban run-off from Worksop, Warsop, Mansfield and a part of the Rotherham area. The tributaries combine near Elkesley to form the River Idle (EA, LEAP 1999b). The River Ryton joins upstream of Bawtry.

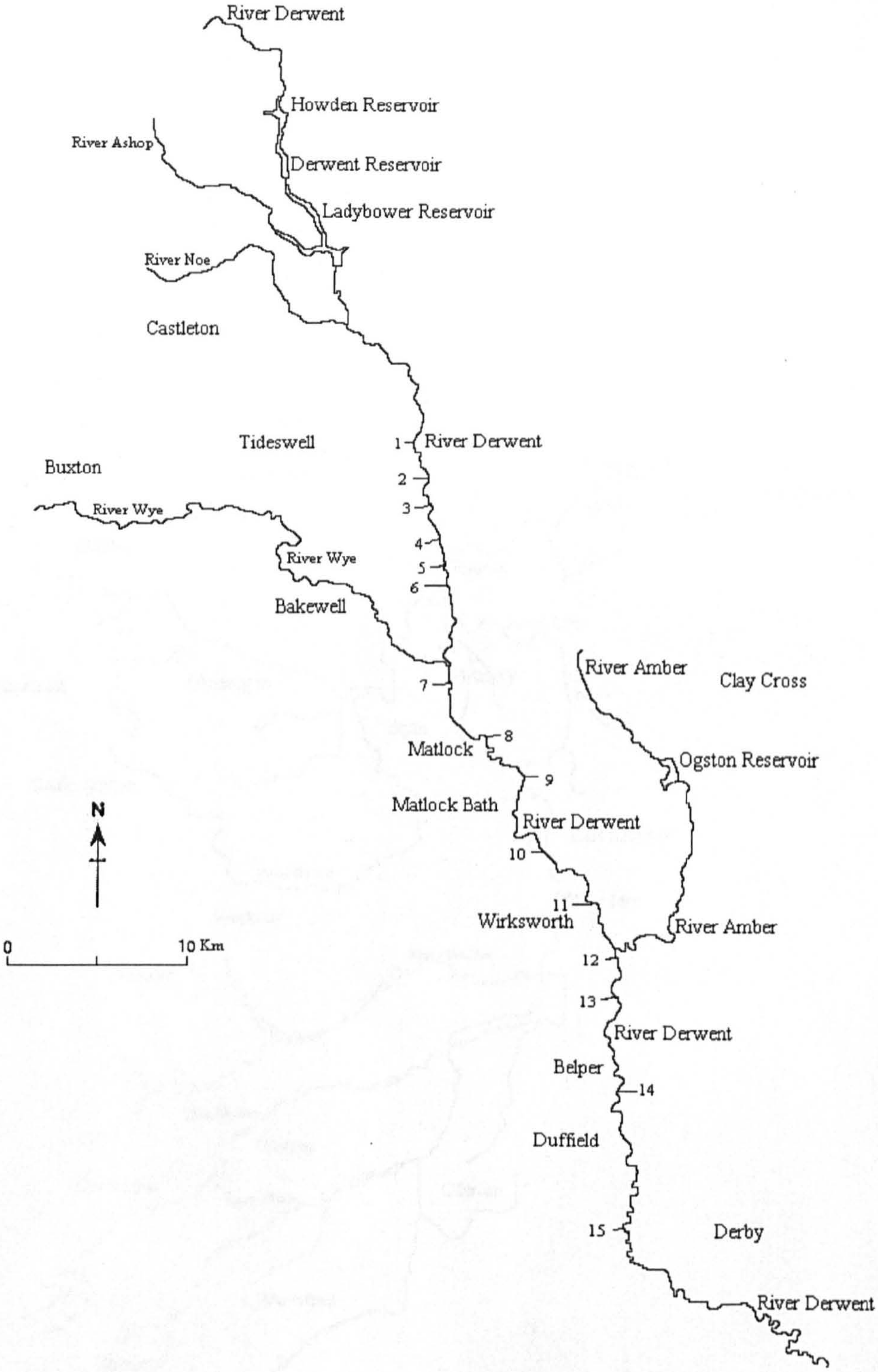


Fig. 2.9 Sampling sites on the River Derwent (Derbyshire) (site no. as in Appendix 2.1)

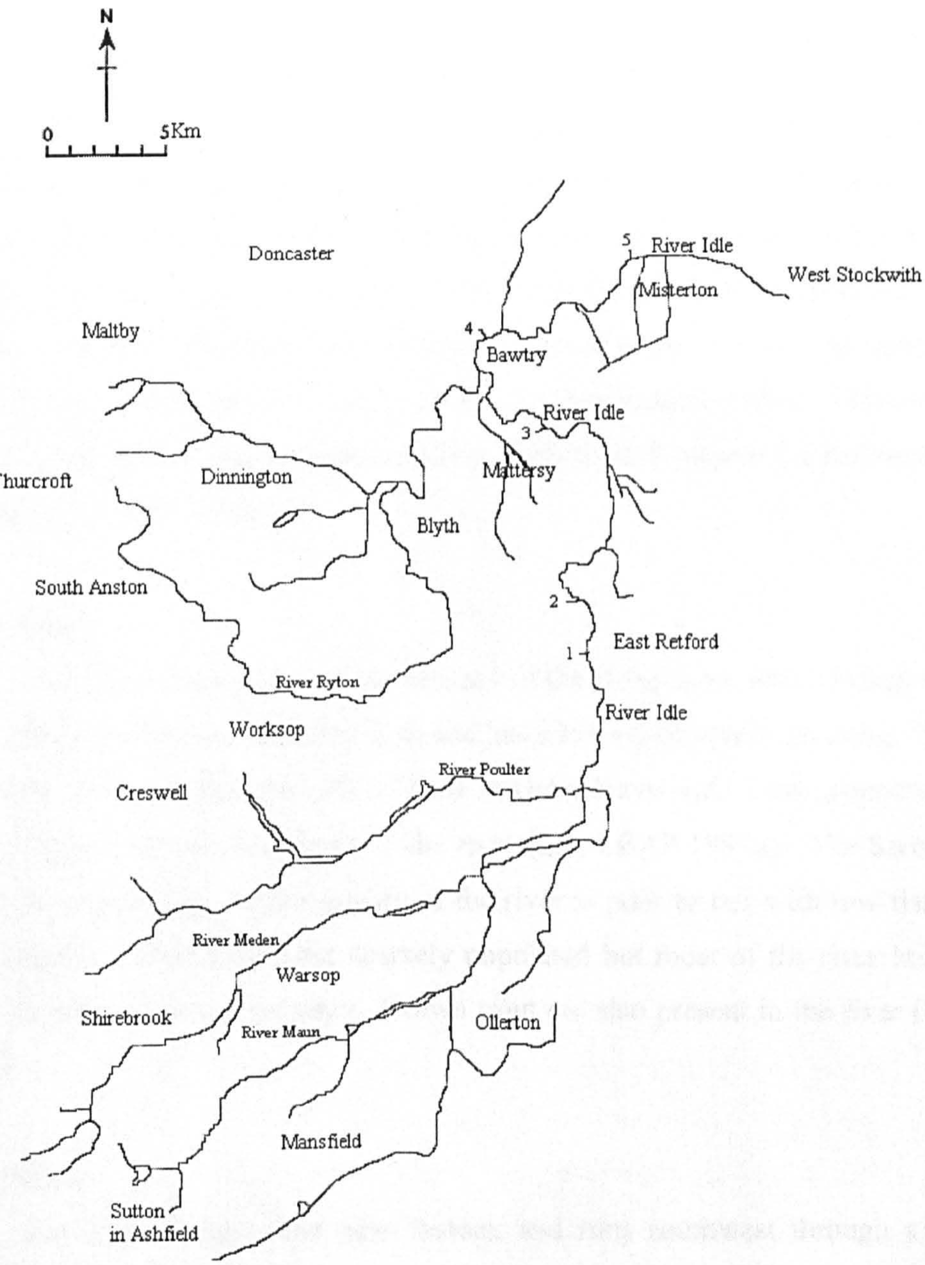


Fig. 2.10 Sampling sites on the River Idle (site no. as in Appendix 2.1)

The River Idle crosses fertile, arable land before it enters the River Trent at West Stockwith. The average flow at Mattersey (site 3) is 15.86 $m^3 s^{-1}$ (NERC 1996). The river is characterised by poor habitat structure, having fish populations of variable quality and quantity. However, the River Idle is an EC-designated cyprinid fishery, and has an ancient right of navigation from West Stockwith to Bawtry. The fish fauna is dominated by roach, common bream and eels (*Anguilla anguilla* (L.)). Other important species are perch, tench (*Tinca tinca* L.)), pike, chub and dace (EA, LEAP 1999b).

River Mease

The source of the River Mease is close to Ashby-de-la-Zouch in Leicestershire where a number of small watercourses including the Gilwiskaw Brook and Hooborough Brook come together to form the river near Measham (Fig. 2.5). The river then flows west through rural areas of Staffordshire and Warwickshire to its confluence with the River Trent near Croxall (EA, LEAP 2000a). The River Mease has retained much of its natural pool-riffle character and follows a meandering course over gravel bottom deposits (EA, LEAP 2000b). The fish fauna is dominated by chub, followed by roach and gudgeon. Other species such as spined loach (*Cobitis taenia* L.) and barbel are also present (EA, LEAP 2000a).

River Penk

The River Penk is the major tributary of the River Sow, which brings water from the north of Wolverhampton (Fig. 2.6) and has a history of severe flooding. The river is relatively clean and joins the River Trent at Great Haywood. Low groundwater levels significantly affect the baseflows of the river (EA, LEAP 1997d). The Saredon Brook is the major tributary. Water quality of the river is poor to fair with low fish stocks in some reaches. Headwaters are sparsely populated but most of the river holds a good stock of chub, dace and gudgeon. Brown trout are also present in the river (EA, LEAP 1997d).

River Sence

The River Sence rises near Ibstock and runs southwest through a rural area passing Congerstone, Sheepy Magna and Ratcliffe Culey before joining the River Anker just north of Atherstone (Fig. 2.5). The River Sence is a small meandering tributary of the River Anker with pools and shallow riffles. In the upper reaches the bed is of gravel and boulders but some siltation has taken place in the lower reaches. The river contains abnormally high levels of suspended solids, which result from mining operations near its headwaters (EA, LEAP 2000a). The River Sence has a good fishery for trout and coarse fish. Six species were reported from the river. The upper reaches are dominated by brown trout and rainbow trout while roach, dace, chub and perch were recorded from the lower reaches (EA, LEAP 2000a).

River Soar

The River Soar rises 20 km southwest of Leicester (Fig. 2.11). The river has a gentle gradient and most of its 1,360 km^2 catchment is rural, thinly populated and overlies fertile Keuper Marl, Rhaetic and Lias clays, which support beef and dairy farming. In the upper reaches above Leicester, the river is unpolluted and follows a meandering path over a gravel bed. The character of the river below Leicester was altered considerably during the 18th Century to enable boats to navigate between Nottingham and Leicester (EA, LEAP 1997c). The River Soar receives water imported from the River Dove (Higgs & Petts 1988) and it also receives sewage discharges from several sources, most notably Wanlip WRW. River flow is influenced by upstream impounding reservoirs and groundwater storage. Average flow at Littlethorpe (site 4) is 21.41 m^3 s^{-1} (NERC 1996). The river supports a good coarse fishery based on roach, dace, chub, pike, tench, common bream and gudgeon. However, the river is stocked with farm-reared fish to support angling (EA, LEAP 1998f). Minor species, including stone loach (*Barbatula barbatulus* (L.)), minnow and bullhead (*Cottus gobio* L.), are present. Native brown trout are also present in the river (EA, LEAP 1997c).

River Sow

The River Sow (Fig. 2.6) is a good quality rural river, having a catchment area of 163 km^2. The average flow at Bridgford is 10 m^3 s^{-1}. The river drains rural areas around the Staffordshire-Shropshire border, east of Stafford. The area is predominately pastureland but land drainage schemes have been undertaken to allow more arable agriculture. The River Sow meets the River Penk immediately downstream of Stafford and joins the Trent at Great Haywood (EA, LEAP 1997d). The river at Cresswell Farm is of pool-riffle character with a sand and gravel bed. Incidences of fish mortality have occurred in some reaches of the river due to low dissolved oxygen levels (EA, LEAP 1997d). The River Sow has a good coarse fishery composed of six species: viz. roach, dace, chub, gudgeon, pike and perch. Roach followed by chub and gudgeon dominate the fish community. The middle reaches are stocked annually with brown trout (EA, LEAP 1997d).

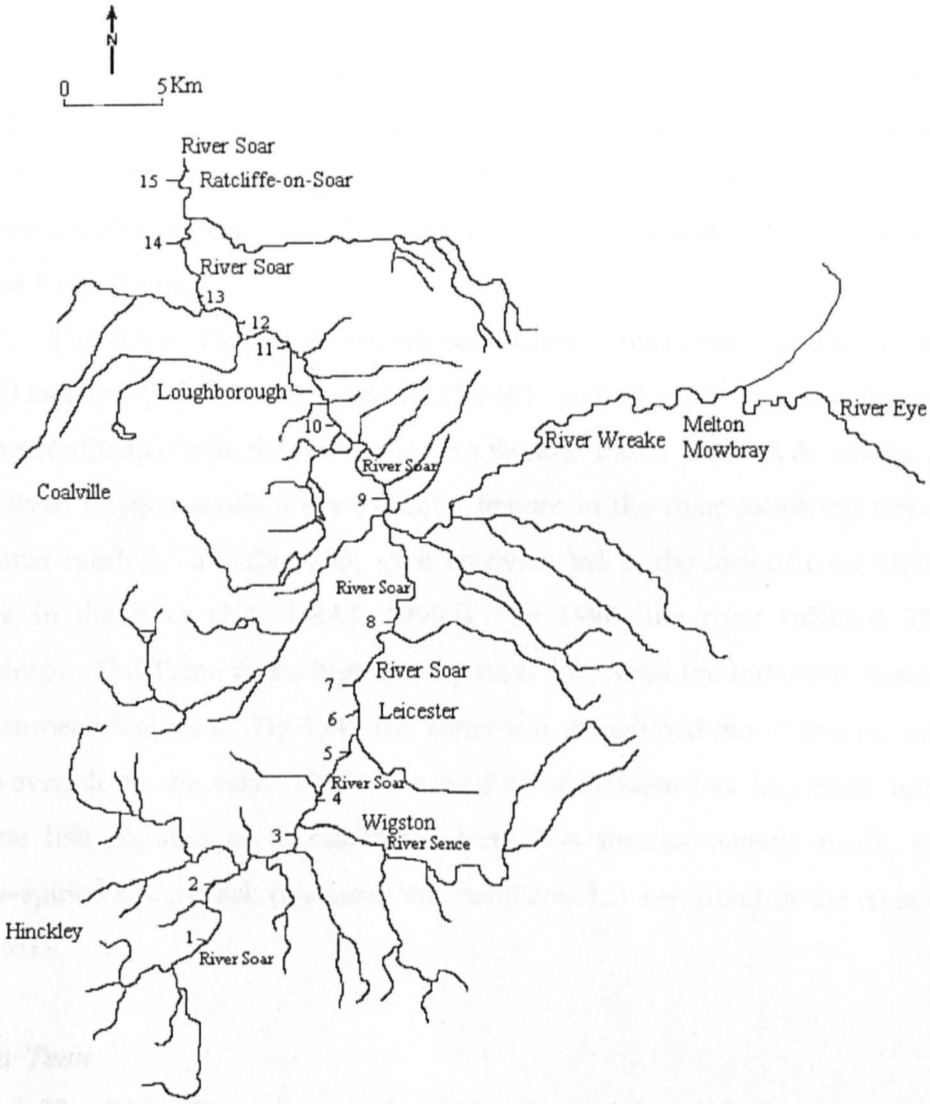


Fig. 2.11 Sampling sites on the River Soar (site no. as in appendix 2.1)

River Tame

The River Tame rises in the Black Country with the Wolverhampton and Oldbury arms of the river joining at Bescot (Fig. 2.7). The river flows eastward to the north of Birmingham City centre. Below Lea Marston lakes, the river flows north through Tamworth before joining the River Trent near Wychnor, just north of Lichfield (EA, LEAP 2000a). The River Tame, a tributary of the River Trent was considered the dirtiest river in England (OU 1972). The river is 84 km long with a catchment of 799 km^2 (NERC 1996). The River Tame is fast-flowing, contains boulders on its bed, and carries a high sediment load. The rivers Anker, Blythe and Rea are the main tributaries of the River Tame.

The River Tame receives treated effluent from Birmingham. Annual average BOD and ammonia concentrations of 25 mg l^{-1} and 12 mg l^{-1} respectively were recorded at the confluence with the River Trent in the late 1980s (Brewin & Martin 1988). Low dissolved oxygen levels are a common feature in the river following heavy, localised summer rainfall. In July 1995, such an event led to the loss of over 90% of the fish stock in the river (EA, LEAP 1998d). In 1996, the river suffered 325 pollution incidents. The Tame was a high quality rural river until the Industrial Revolution had a disastrous effect on it. By 1945 the Tame was so polluted that it was devoid of all life. However, from the early 1980s, the quality of effluent has improved sufficiently for coarse fish populations to establish. Very few species, namely roach, gudgeon and three-spined stickleback (*Gasterosteus aculeatus* L.) are found in the river (EA, LEAP 1998d).

River Tean

The River Tean rises north of Cheadle and flows from just east of Stoke-on-Trent and passes near Uttoxeter before entering the River Dove (Fig. 2.8). The River Tean is a major tributary of the River Dove. Most of the river catchment overlies carboniferous limestone and Keuper Marl giving the river its fast, cool and mainly unpolluted characteristics. The middle reaches receive treated effluent from different sources (EA, LEAP 2000c). The Tean is a small river, which supports a poor fish community throughout its length. Brown trout is the principal species while grayling, bullheads, stone loach, minnows and sticklebacks are also present (EA, LEAP 1999d).

River Trent

The Trent (Fig. 2.12) is the principal river catchment of the East Midlands of England and the third largest river in Britain (OU 1972). The River Trent rises as a spring on Biddulph Moor in North Staffordshire (NGR: SJ 896 548) and meets its confluence with the Humber Estuary at Trent Falls (NGR: SE 865 233) (Cowx 1991). The river initially flows through Stoke-on-Trent where historically it received considerable industrial and domestic effluent from Potteries (Lester 1975). The river is 280 km long, draining, with its 2,200 km of tributaries, an area of 10,550 km^2 (Templeton & Churchward 1990). Below Stoke-on-Trent, the Trent is fast flowing with a gravel bed and a pool/riffle topography. In its middle reaches the character of the Trent has been altered by construction of weirs and by-passes to facilitate navigation. Below Gainsborough the Trent receives water from the large catchment of the River Idle, Eau and Torne, and its associated channels. The river is used for a whole range of water-based pursuits and was once probably the most famous coarse fishery in England (EA, LEAP 1997d). Discharge and flow are influenced by the impounding reservoirs upstream. The average flow at Nottingham and Colwick is 82.50 and 76.82 $m^3 s^{-1}$, respectively (NERC 1996).

The River Trent has many problems, e.g. pollution (industrial, domestic and thermal), river engineering works and alleged poor fisheries, which often confront fisheries managers (Cowx 1990). Annual nitrate load from agricultural and other sources is 30 kg ha^{-1} which has increased at a rate of 455 t yr^{-1}. Water temperatures occasionally exceeded 30°C although this is now rare because of decommissioning of power stations (Templeton & Churchward 1990). The water quality of the River Trent along its entire length has improved during the last twenty years. The river now supports a wide variety of coarse fish together with some trout. Salmon, which were once plentiful, have returned to the lower reaches and are now being stocked in the River Dove. Barbel, which were absent from the river during the 1940s, 1950s and 1960s, are now recolonising the river (Jacklin 1996). Twelve species including both brown trout and rainbow trout, were recorded from Colwick. Over the whole of the river, chub and gudgeon were the dominant species present although roach and bleak (*Alburnus alburnus* (L.)) were also numerous (Templeton & Churchward 1990). The upper and middle reaches of the river were used in this study.

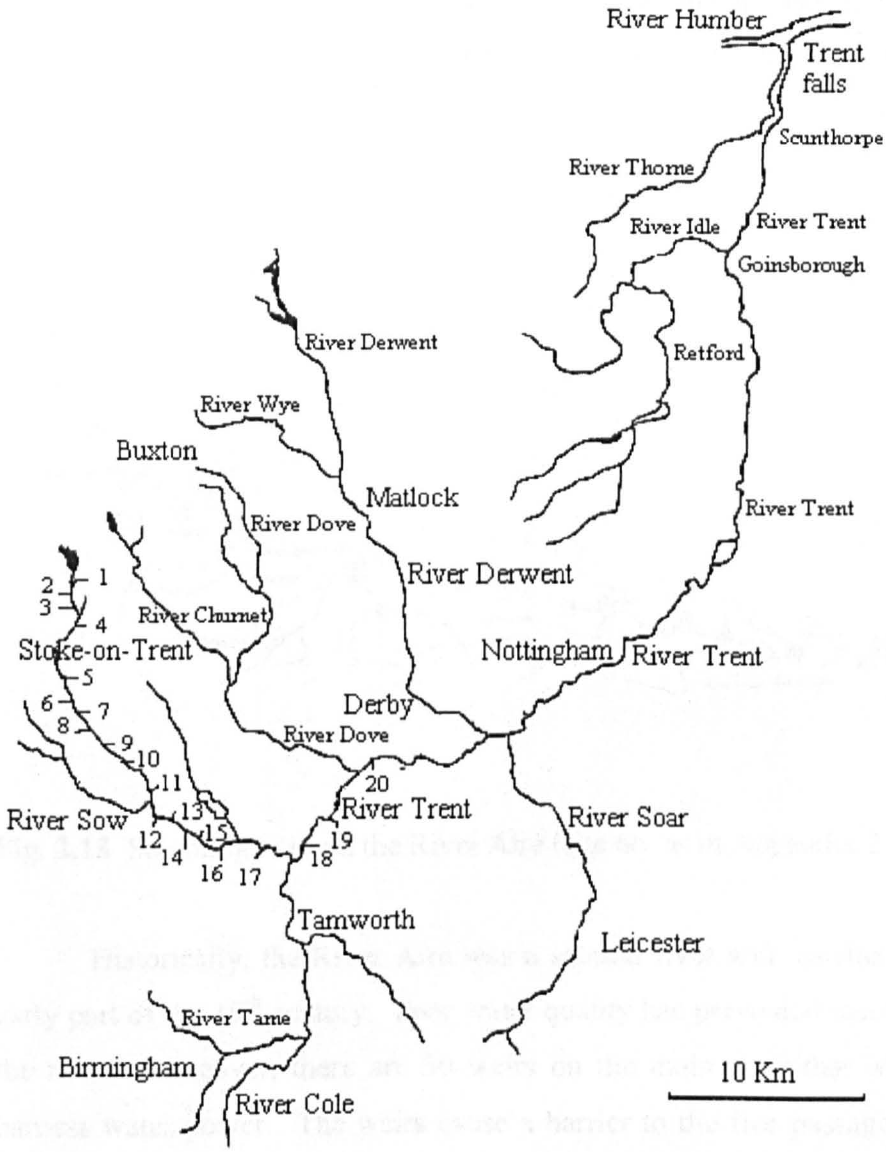


Fig. 2.12 Sampling sites on the River Trent (site no. as in Appendix 2.1)

2.3.3 The Yorkshire Ouse catchment

River Aire

The River Aire (Fig. 2.13) rises on the limestone moorland around Malham (NGR: SD 901 627). The main river flows for 148 km to its confluence with the Yorkshire Ouse at Goole (NGR: SE 745 227). The last 26 km of the river, upstream of its confluence, are tidal. The river is fed by a number of becks such as Kirkby, Hetton, Otterburn, Crosber, Earby, Broughton, Holme, Ninelands, Bridgehouse, Bradford, Eastbrook, Gill, Oulton, Fryston, Eshton, Stirton, Wyke, Little, Gledhow, Sheepscar, Thorlby, Winterburn, Meanwood and Cononley. The catchment covers an area of approximately 1,100 km^2 (EA, LEAP 1998a). Within the catchment 18 impounding reservoirs are operated for public water supply. There are 40 STWs discharging into the river (Atkinson 1994).

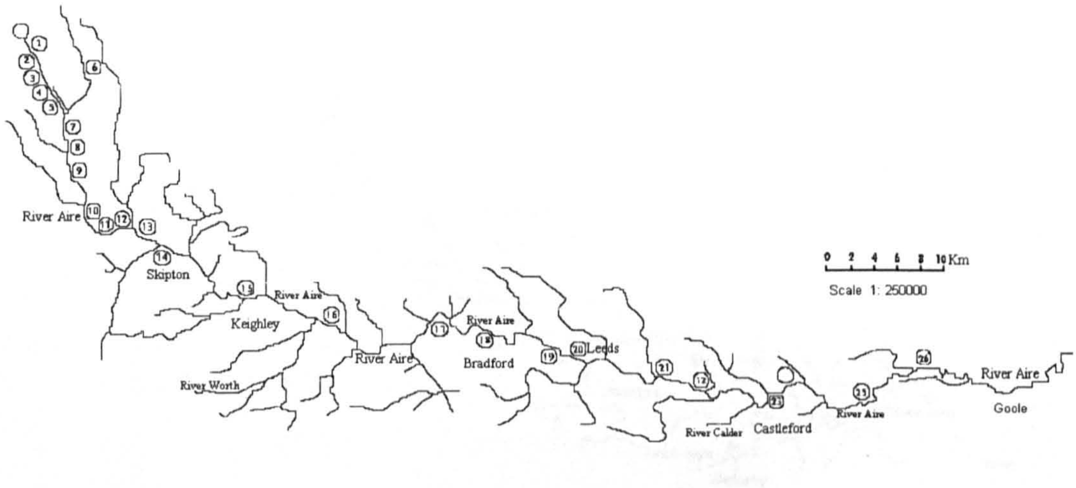


Fig. 2.13 Sampling sites on the River Aire (site no. as in Appendix 2.1)

Historically, the River Aire was a salmon river with catches recorded into the early part of the 19th century. Poor water quality has prevented them from returning to the river. Moreover, there are 30 weirs on the main river that were constructed to harness water power. The weirs cause a barrier to the free passage of both trout and coarse fish species (LEAP, EA 1998a). The river supports a mixed fishery of both trout and coarse fish species. The headwaters are stocked with farm-reared trout (EA, LEAP 1998a).

River Nidd

The River Nidd (Fig. 2.14) rises to the east of Great Whernside, the highest point in the catchment at 704 m Above Ordnance Datum (AOD). The river flows southeast through a steep sided valley, before turning east below Birstwith. Below Knaresborough, the river takes a meandering course across the Vale of York, joining the River Ouse at Nun Monkton (EA, LEAP 1998b). The Howstean, Ramsgill, Fell, Darley, Tang, Thornton, Oak and Rampart Becks flow into the river. The River Crimple is the major tributary. The total catchment area is about 1,555 km^2. The land use is predominantly rural, accounting for over 90% of the total area. The River Nidd receives consented discharges of treated sewage from 51 sources. The number of licensed groundwater abstractions is 270 while spring and surface water abstractions are 341. Snowfall is a significant source of water during winter months. The largest artificial inputs of water to the river are from the two Harrogate STWs (LEAP, EA 1998b).

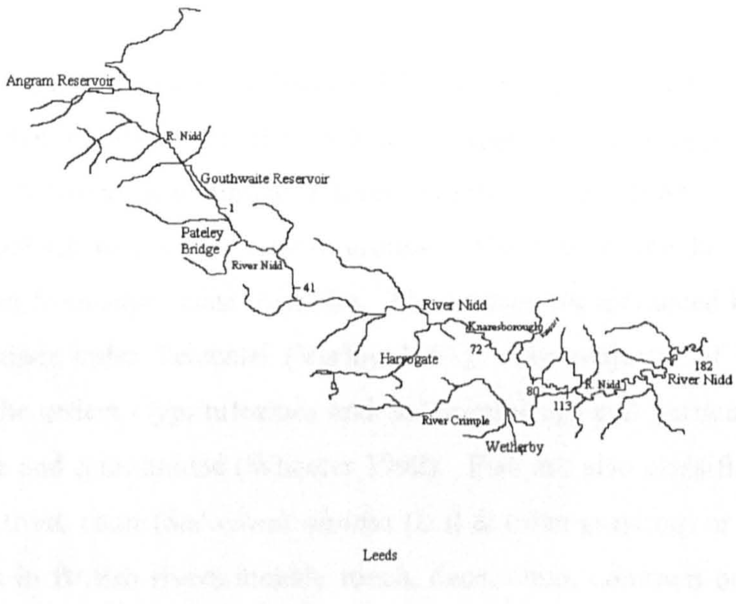


Fig. 2.14 Sampling sites on the River Nidd (site no. as in Appendix 2.1)

Fish populations are generally of a high quality. Species distribution follows the classic zonation associated with changes in river width and gradient. Trout inhabit the upper reaches, grayling appear further down, then riverine cyprinids, such as barbel, chub, dace and finally fish characteristic of slow flows, such as gudgeon, perch, ruffe (*Gymnocephalus cernuus* (L.)), pike, common bream and roach. Bullhead, minnow and stone loach are found in the upper reaches of the river. However, the headwaters upstream of Angram reservoir are virtually fishless, possibly due to low pH (LEAP, EA 1998b).

CHAPTER THREE

3. BRITISH FRESHWATER FISH AND REFERENCE CRITERIA

3.1 INTRODUCTION

3.1.1 British freshwater fish

The zoogeography of British freshwater fish species was determined by the post-glacial dispersal characteristics of the individual species, hydrological, physical features, climatic conditions and human-induced events (Varley 1967). The British freshwater fishes belong to two zoological groups. The first is the lampreys, very primitive vertebrates from super class Agnatha. The second, the advanced bony fishes, belonging to the super order Teleostei (Varley 1967). The majority of British fish species belong to the orders Cypriniformes and Salmoniformes and particularly to the families Cyprinidae and Salmonidae (Wheeler 1992). Fish are also classified either as game (e.g. salmon, trout, charr (*Salvelinus alpinus* (L.)) & often grayling) or coarse fish. Coarse fish species in British rivers include roach, dace, chub, common bream, silver bream (*Blicca bjoerkna* (L.)), barbel, rudd, tench, common carp (*Cyprinus carpio* L.), bleak, gudgeon, pike, perch, ruffe and pikeperch (*Stizostedion lucioperca* (L.)) (Cowx 2001). For freshwater fish, the British fauna is poor compared with that of continental Europe, which is itself poor compared with North America, Africa or Asia. The main reason for the limited British fauna is that the glaciations during the Pleistocene Era virtually eliminated the fauna that was living in Britain (Varley 1967). Before the Ice Age, the British Isles were joined with the continent of Europe and the River Thames was a tributary of the River Rhine. The impacts of glaciations can be seen in the distribution and diversity of freshwater fishes in Britain. Rivers in southern and eastern England have more species, and are more similar to continental Europe than rivers in northern and western England (Varley 1967). The freshwater fish, which are now found in the rivers, are the result of subsequent immigrations from the rivers of northwestern Europe (the rivers Elbe, Weser, Rhine, Scheldt & Meuse) in the rivers Thames and Great Ouse. The eastern rivers of Britain tend therefore to have a richer variety of species than those of the north and west (Giles 1994).

Giles (1994) reported 42 native and 12 introduced freshwater fish species in Great Britain. A historical series of fish extinctions have taken place in Britain due to anthropogenic activities (Maitland 1972) (Table 3.1). In addition, the following riverine species are considered to be either actually or potentially nationally threatened: spined loach, *Cobitis taenia* L., burbot, *Lota lota* L., twaite shad, *Alosa fallax* (Lacepede), allis shad, *Alosa alosa* (L.), houting (anadromous), *Coregonus oxyrinchus* (L.), smelt (anadromous), *Osmerus eperlanus* (L.), river lamprey, *Lampetra fluviatilis* (L.), brook lamprey, *Lampetra planeri* (Bloch) and sea lamprey, *Petromyzon marinus* L. (NRA 1994b).

Table 3.1. Extinction of fish from different British freshwaters (after Maitland 1972)

Year of Extinction	Species extinct	Waterbody from which extinct
1830s	Arctic charr, *S. alpinus*	Loch Leven
1840s	Allis shad, *A. alosa*	River Severn
1850s	Twaite shad, *A. fallax*	River Thames
1900s	Burbot, *L. lota*	River Foss
1910s	Vendace, *Coregonus albula* (L.)	Castle Loch
1920s	Smelt, *O. eperlanus*	Rostherne Mere
1920s	Burbot, *L. lota*	River Trent
1950s	Arctic charr, *S. alpinus*	Loch Dungeon
1960s	Arctic charr, *S. alpinus*	Loch Grannoch
1960s	Smelt, *O. eperlanus*	River Forth
1960s	Burbot, *L. lota*	Rivers Derwent, Waveney & Cam
1970s	Vendace, *O. eperlanus*	Mill Loch
1980s	Arctic charr, *S. alpinus*	Llyn Peris

Water temperature is one of the most important factors limiting distribution and diversity of fish in the British Isles. Except for alien species, most UK species are stenotherms, sensitive to eurythermal conditions (Wheeler 1969). British freshwater fish are divided into three main breeding groups. Most of the salmonids spawn in autumn or winter when the water is cold; many cyprinids, such as common carp, common bream and tench, spawn in summer when water temperature is near its maximum; the third group, including grayling, perch, pike, bullheads and cyprinids such as minnows, barbel and chub spawn in spring when the water is beginning to warm up (Varley 1967).

Distribution, growth, spawning success and recruitment of fish are also influenced by anthropogenic activities (Cowx 2001). British rivers were used and abused during the Industrial Revolution, resulting in loss of natural habitat and fauna (Petts 1984). It is thus important to know the distribution and diversity of British freshwater fish species when establishing the reference condition. Development of an

IBI requires clear definition of fish species that actually reflect ambient environmental conditions based on residency. British freshwater fish species may be classified as follows:

*a. **Resident indigenous:*** naturally occurring native species populating suitable aquatic habitats, e.g. roach, dace and chub.

*b. **Resident naturalised:*** well-established non-native species populating suitable aquatic habitats, e.g. common carp, pikeperch.

*c. **Nonresident transient:*** non-populating fish species found to occur in unsuitable aquatic habitats, e.g. sunbleak (*Leucaspius delineatus* (Heckel)).

*d. **Nonresident stocked:*** non-populating fish species introduced for a recreational fishery only, e.g. wels (*Silurus glanis*) L.

3.1.2 River zonation and fish distribution

Huet (1949 & 1959) divided the course of a river into four zones on the basis of river gradient, width and water temperature (Fig. 3.1). In this classification it was proposed that "in any given geographical area, river or stretches of river of like breadth, depth and slope have near identical biological characteristics and very similar fish populations". The four main zones are as follows (Huet 1954):

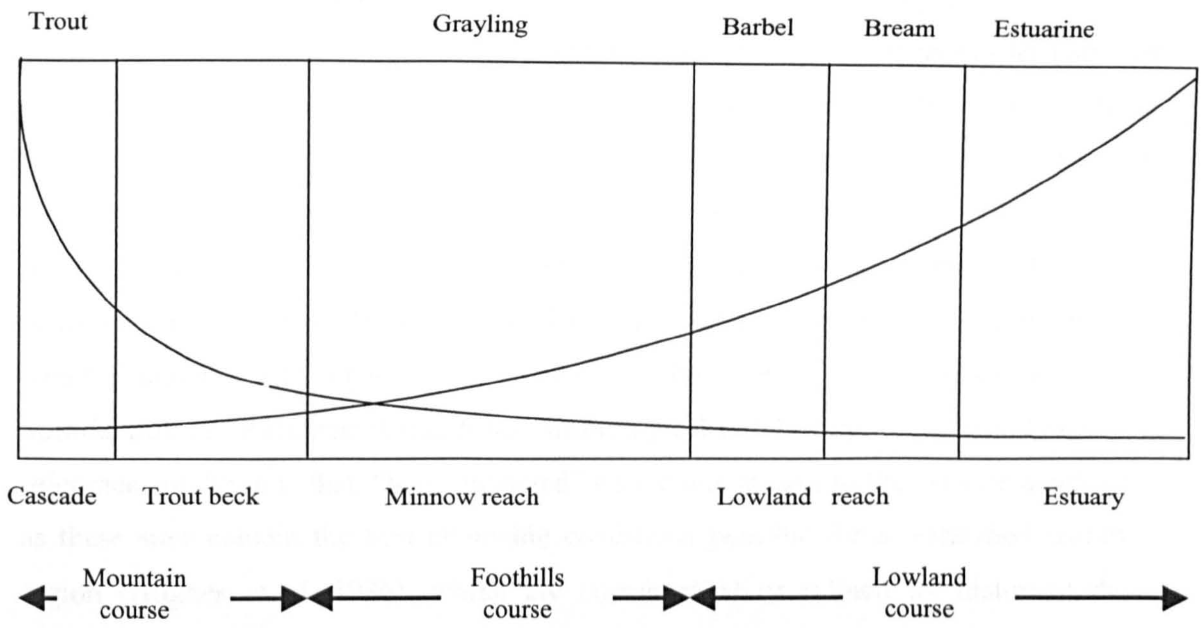
a. The "trout zone", which is small, very steep, often torrential and usually very cold (Fig. 3.1). Even in summer the temperature is low and does not exceed 10°C, but water is clear and well oxygenated. The bottom consists of pebbles or coarse gravel. Trout and salmon are the dominant species but stone loach, bullhead and minnow are also found here.

b. The "grayling zone", which is slightly wider and deeper than the trout zone but steep, still torrential and rocky (Fig. 3.1). The water is still well oxygenated and clear, but somewhat warmer than in the trout zone. Trout, salmon and grayling live in the open water but the bullhead may shelter among the stones. Running water cyprinids, e.g. barbel, chub and dace are also present. Complementary cyprinids, e.g. roach or rudd, and predators, e.g. perch, pike and eel may be present.

c. The "barbel zone", which is still fairly swift but in which there are patches of mud and silt in places protected from the current and a few rooted plants, such as the water buttercup (*Ranunculus fluitans* Lam.), are able to grow (Fig. 3.1). In summer the water may reach temperatures around 20°C. Running water cyprinids are the dominant species but trout and grayling may still be present. Complementary cyprinids and predators are fairly abundant. Still water cyprinids, e.g. tench, common bream, silver bream and common carp are present occasionally.

d. The "bream zone", which is very gentle, slow flowing and meandering with a muddy bottom and many rooted plants (Fig. 3.1). The river is wider and deeper with shallows and backwaters. The water tends to be cloudy with suspended materials and in summer temperature exceeds 20°C. Coarse fish species, mainly still water cyprinids, complementary cyprinids and predators are the main species but running water cyprinids may also be present. The bream zone is a temporary refuge for many migratory fish such as sea lamprey, sturgeon (*Acipenser sturio* L.), shad, and also sea fishes such as mullet (*Liza ramada* (Risso)), bass (*Ambloplites rupestris* (Rafinesque)) and flounder (*Pleuronectes flesus* L.) (Muus 1971).

The natural distribution of fishes of many English and Welsh rivers follows the river zonation patterns (Cowx 2001). However, distribution patterns of fish species may be affected by anthropogenic activities and indicator fish species may move or disappear from any zone. Consequently, fish populations become imbalanced in terms of structural and functional characteristics. It is thus important to know the distribution patterns of British freshwater fish species when establishing reference conditions.



Zone	1	2	3
Mean monthly temperature	Rises to 20°C		Rise to over 20°C
Substrate	Rocks, stones, gravel EROSIONAL		Fine gravel, sand, silt accumulation DEPOSITIONAL
Flow	Fast turbulent		Variable to stable, water turbid
Oxygen	High to saturated		Variable to deficient
Faunal community	Low to increasing species diversity	High diversity	Diversity maintained
Biocenosis	Plecoptera, Trichoptera, Ephemeroptera, Coleoptera, Diptera	Ephemeroptera, Trichoptera, Diptera	Trichoptera, Diptera, Mollusca, Crustacea, Hirudinea, Hemiptera, Odonata, Tubificid worms
Trophic group	Shredders	Collectors, grazers	Grazers, filter feeders
Class	⟵ RHITHRON ⟶		⟵ POTAMON ⟶
Dominant and common associated fish species	*Salmo trutta* *Salmo salar*	*Thymallus* *Barbatula* *Phoxinus* *Cottus* *Barbus*	*Rutilus* *Leuciscus cephalus* *Leuciscus leuciscus* *Abramis* *Cyprinus* *Esox*

Fig. 3.1 Zonation of rivers to illustrate species associations (adapted from Huet 1949 and Cowx 2001)

3.1.3 Reference condition

Establishment of a reference condition is the prerequisite to apply an index of biotic integrity to a water body. The reference condition helps to compare the situation with the present and historical conditions of a site. The majority of English rivers have been altered for various reasons (Petts 1984) hence, pristine sites are rare. In any event, the biologist cannot evaluate biological integrity effectively by any method without first addressing the question, "What should fish communities look like in this ecoregion or country under least impacted conditions?" The answer to this question is the introduction of "Reference Conditions" in biological monitoring. The hypothesis of a reference condition is that, "least impacted" sites come closest to the pristine conditions as these sites contain the best attainable conditions possible for a watershed within a region (Hughes *et al.* 1986), which are corroborated or refined by historical data, paleoecological data, quantitative models and expert judgement (Hughes *et al.* 1998).

The evaluation of any aquatic assemblage is based on a comparison between the observed condition and the expected condition. The expected condition represents the "biological potential" for that particular site, as defined by a regional standard or reference condition (Hughes *et al.* 1986, Hughes 1995). Reference condition is usually defined as a river in its natural state with naturally occurring biota. Usually, a natural site is rich with flora and fauna according to the geomorphological conditions. Properly defined reference conditions provide a reasonable benchmark for comparison to measure the degree of water quality or habitat degradation (Hughes *et al.* 1986). Generally, reference sites in lotic waters support fish assemblages dominated by top carnivores and benthic invertivores. Generalist feeders may occur at low densities in reference stream fish assemblages (Lyons *et al.* 1996). Diversity and density of the fauna varies with the geographical location, ecological and meteorological conditions of the site. Naturally, fish species diversity in temperate regions is less than tropical regions. Due to lack of unimpacted natural sites in English rivers, historical data, regional ichthyological texts, published reports and scientific papers were used to establish a reference condition. Historical data on ecology and biology of fish communities are invaluable to establish a reference condition. Deviation from the natural condition will indicate the severity of change in community structure of the aquatic biota.

Any alteration of habitat has an adverse effect on the ecosystem and it may become imbalanced in producing energy, maintaining integrity and sustaining the community structure of fish populations. Degradation due to physical alteration, pollution and anthropogenic activities have altered and reduced breeding and nursery grounds which results in poor spawning success and recruitment of fish populations. As a result, diversity of fish fauna has changed in the rivers of England. As there is no pristine site in most English rivers therefore, reference condition needs additional information.

3.2 MATERIALS AND METHODS

3.2.1 Source of information

Pristine aquatic habitats, having naturally occurring fauna, are rare in English rivers. Therefore, historical data, ichthyological texts, reports, scientific papers and expert judgement were used to define the reference conditions. Establishment of a reference condition required information on origin, habitat, diversity, density, trophic level, reproductive guild, and tolerance to degradation of fishes. Yarrell (1836), Cuvier & Valenciennes (1839), Gunther (1862), Day (1880-84), Regan (1917), Jenkins (1925), Carpenter (1928), Norman (1943), Hodgson (1945), Varley (1967), Muus (1967), Wheeler (1969, 1983 & 1992), Maitland (1972), Hawksworth (1974), Pitcher & Hart (1982), Lelek (1987), Wootton (1990), Maitland & Campbell (1992), Nelson (1994), Giles (1994), Jobling (1995), Miller & Loates (1997), Cowx & Welcomme (1998) and Simon (1999) and, reports / scientific papers published by Hartley (1947), Wheeler & Maitland (1973), Wheeler (1974, 1977 & 1992), Maitland (1979), EIFAC (1984), Holcik (1984 & 1991), Welcomme (1984, 1988 & 1995), Oberdorff & Hughes (1992), Winfield *et al.* (1994), Cowx *et al.* (1995), Didier & Kestemont (1996), Belliard *et al.* (1997), Boet *et al.* (1999), Kestemont *et al.* (2000), Cowx & Godkin (2000) and Cowx (1997 & 2001) were consulted and used to organise, structure and establish a reference condition.

3.2.2 Guild concept

The concept of the guild was developed to simplify analysis and assist in the prediction of community change (Austen *et al.* 1994). Root (1967) defined guilds in the ecological sense as "a group of species that exploit the same class of environmental resources in a similar way." Guilds were developed based on reproduction, feeding, habitat use and morphology. One strength of the guild approach is that it simplifies analysis of the community by providing an operational unit between the individual species and the community as a whole (Root 1967). Species are grouped based on some degree of overlap in their niches regardless of taxonomic relationships.

Several guilds were used in this study to classify fish species according to their habitat preference, feeding habit, reproductive strategy and tolerance to water quality degradation. Each species collected was assigned to appropriate ecological guilds based on existing knowledge from the scientific literature. Fish display a wide range of feeding habits. They occupy many trophic roles from detritivores to secondary carnivores. However, it is rare for fish to specialise in one particular food category throughout their entire life cycle. There is often a correlation between morphological traits and trophic role because morphology determines how a fish can feed. Generally body shape, mouth morphology, teeth, gill rakers and the structure of the alimentary canal are important to diet selection. Fish were grouped into different trophic guilds as outlined by Kushlan (1976), Keenleyside (1979), Grossman *et al.* (1982), Schlosser (1982a), Dill (1983), Angermeier & Karr (1983), Keast (1985), Berkman & Rabeni (1987), Bayley (1988) and Goldstein & Simon (1999). For the present study, each species was assigned to a trophic guild according to feeding habits (Cowx 2001) and the following definitions were used to categorise the species.

Planktivores: adult diet consists of more than 75% zooplankton and / or phytoplankton (Lyons *et al.* 1995). Fish, having fine gill-rakers and elongated pharyngeal teeth, prefer inertial sucking of water containing food. They have no stomach but have an elongated, undifferentiated intestine (Goldstein & Simon 1999).

Herbivores: adult diet consists of more than 75% plant material (Lyons *et al.* 1995). Fish have terminal or subterminal mouth with bony slashing jaw for clipping and tearing aquatic vegetation / weed. In most cases, the digestive tract is as long or longer than the total length of the individual (Goldstein & Simon 1999).

Omnivores: adult diet consists of more than 25% plant material and more than 25% animal material (Schlosser 1982b). They are also called "generalists" as they take food from a wide range of flora and fauna (Leonard & Orth 1986).

Insectivores / Invertivores: adult diet consists of more than 75% insects (Lyons *et al.* 1995). Fish with terminal or supraterminal mouth, take aerial, drifting or swimming insects and invertebrates. Invertivores compose the largest and perhaps the most diverse trophic class. They include species that feed on the smallest midge, to species that consume large molluscs (Goldstein & Simon 1999).

Benthivores: adult diet consists of more than 75% benthic organisms (Goldstein & Simon 1999). Fish have ventro-terminal, sometimes a highly protractile mouth that are used to vacuum-clean. They have file-like teeth that comb and sort small organisms.

Molluscivorse: adult diet consists of more than 75% molluscs, and bivalves (clams and snails). This group possesses heavy dentine or pharyngeal teeth and strong, powerful jaws. The stout, flattened, molariform pharyngeal teeth act as a crushing / grinding mechanism. They gather shells and feed on the soft parts of the body (Goldstein & Simon 1999).

Piscivores: adult diet consists of more than 75% fish (Lyons *et al.* 1995, Goldstein & Simon 1999). Fish have a wide mouth aperture with needle-like teeth and a strong jaw with marginal and palatal bones. They are capable of capturing active, mobile prey, inclusive of larger invertebrates. They pursue a prey by stalking, chasing, ambushing or lying-in-wait approach (Simon & Emery 1995).

Fish were also classified according to habitat utilisation (Schlosser 1982b, Bain *et al.* 1988, Leonard & Orth 1988, Lobb & Orth 1991 & Mann 1996). Fish have the most diverse forms of reproduction. Some fishes produce large numbers of small eggs and others produce few eggs of large diameter. They show different spawning behaviour and use diverse spawning grounds. On the basis of ontogeny, spawning behaviour and the place of egg deposition, Balon (1975, 1980) classified fish into 33 groups known as "Reproductive guilds". In this study, reproductive guilds of fish were assigned according to the classification proposed by Balon (1975) and the concept modified by Chadwick (1976), Balon *et al.* (1977), Balon (1978 & 1981a, b), Mahon (1984), Berkman & Rabeni (1987), Bruton & Merron (1990), Oberdorff & Hughes (1992), EIFAC (1993), Boet *et al.* (1999) and Cowx (2001). The following definitions were adopted for this study.

Lithophils: Fish spawn exclusively on gravel, rocks, stones, rubble or pebbles. Spawning success depends on the availability of suitable sized and clean gravel. Hatchlings are photophobic (Balon 1975).

Phytophils: Fish spawn especially on plants, leaves and roots of live or dead vegetation. Larvae of this group are not photophobic (Balon 1975).

Phytolithophils: Fish deposit eggs in relatively clear water habitats on submerged plants, if available, or on other submerged items such as logs, gravel and rocks. Larvae exhibit photophobia like lithophils (Balon 1975).

Psammophils: Fish spawn on roots or grass above sandy bottom or on the sand itself. Larvae are not photophobic (Balon 1975).

Fishes were also classified according to their tolerance to water quality to calculate IBI for English lowland rivers (Davies 1977, Alabaster & Lloyd 1982, NRA 1994b, Mann 1996 and Cowx 2001).

3.3 RESULTS

3.3.1 Establishment of metric expectation criteria

Generally British rivers harbour 45 to 50 fish species (Wheeler 1983, Moss 1988). The River Thames (Winfield *et al.* 1994) and the Yorkshire Ouse (Burnett *et al.* 1978) catchments contain 35 fish species each, while about 40 species were reported to inhabit the Trent catchment (Braddock 1977). Using fish distribution data (Wheeler 1969 & 1983, Cowx 1998) and collection records from 1975 - 1986 (Dearsley & Reeves 1977, Saxby & Lewis 1982, Severn-Trent Water Authority 1983, Thames Water 1985 & 1986), the number of species common enough to be collected with a thorough sampling method was estimated. This was done to establish expected values for species richness and composition, and fish abundance and biomass metrics. Historically rare species, (e.g. burbot) or large-bodied fish (e.g. huchen, *Hucho hucho* or *Silurus glanis* L.) (Winfield *et al.* 1994) were not included in the estimates.

Available literature indicates that 41 exotic species (Table 3.2) have been introduced across the country within the last century (Wheeler & Maitland 1973, Cowx 1997, Cowx & Godkin 2000). On the basis of total freshwater fish species, the maximum expectation criterion was fixed for this study. Maximum expected values for different ecological categories are presented in Table 3.3. Although situated in different locations, a generalised value for each category was considered for the three catchments (The Thames, Trent and Yorkshire Ouse). Among the 91 species (50 native and 41 exotic), only 30 species are found in many UK rivers (Table 3.4).

3.3.2 Geographic origin and distribution

The majority of coarse fishes found in the study rivers are indigenous to the fresh waters of Britain (Table 3.4). Among the 41 exotic species (Table 3.2) common carp and goldfish (*Carassius carassius* (L.)) were introduced into England from the Continent several centuries ago while pikeperch was introduced into isolated still waters within Britain in 1878 (Wheeler 1974). Now pikeperch has self-sustaining populations, predominantly in eutrophic waters of East Anglia and the West Midlands (Hickley 1986). Other introduced species that have become established, often at only one locality (Varley 1967, Muus 1971, Wheeler 1974, Maitland 1979, Lelek 1987, Winfield *et al.* 1994, Miller & Loates 1997, Cowx 1997, Cowx & Godkin 2000) are listed in Table 3.2.

Table 3.2 Fish species introduced in the UK (Reproductive success : + = established, - = not established, ? = unknown)

Sl. No	Family / Scientific name	Common name	Origin	Trophic guild	Reproductive status	Year of first introduction	Reason for introduction
	1. Cyprinidae						
1	*Cyprinus carpio* L.	Common carp	Asia	Omnivore	+	1200s	Sports fishing
2	*Carassius auratus* (L.)	Goldfish	Asia	Omnivore	+	?	Sports fishing
3	*Ctenopharyngodon idella* (Cuvier & Valenciennes)	Grass carp	Asia	Herbivore	+	1949	Weed control
4	*Hypophthalmicthys molitrix* (Valenciennes)	Silver carp	Asia	Planktivore	-	1953	Bloom control
5	*Aristichthys nobilis* (Richardson)	Bighead carp	Asia	Planktivore	+	1954	Bloom control
6	*Vimba vimba* (L.)	Zahrte	Europe	Benthivore	?	1957	Sports fishing
7	*Aspius aspius* (L.)	Asp	Europe	Carnivore	+	1984	Sports fishing
8	*Chondrostoma nasus* (L.)	Nase	Europe	Omnivore	+	?	Sports fishing
9	*Rhodeus sericeus* (Pallas)	Bitterling	Europe	Invertivore	+	1920s	Ornamental
10	*Leucaspius delineatus* (Heckel)	Sunbleak	Europe	Omnivore	+	1980	Sports fishing
11	*Pseudorasbora parva* (Temminck & Schlegel)	False Harlequin	Europe	Insectivore	+	1960	Sports fishing
12	*Leuciscus idus* (L.)	Ide	Europe	Invertivore	+	1874	Sports fishing
13	*L. souffia* Risso	Soufie	Europe	Invertivore	?	?	Sports fishing
	2. Siluridae						
14	*Siluris glanis* L.	Wels	Europe	Carnivore	+	1881	Scientific interest
	3. Ictaluridae						
15	*Ictalurus melas* (Rafinesque)	Black Bullhead	USA	Carnivore	+	1871	Sport fishing
16	*I. punctatus* (Rafinesque)	Channel catfish	USA	Carnivore	+	1968	Sport fishing
17	*I. nebulosus* (LeSueur)	Brown Bullhead	USA	Carnivore	+	1885	Sport fishing

	4. Percidae						
18	*Stizostedion lucioperca* (L.)	Pikeperch	Europe	Carnivore	+	1878	Sports fishing
19	*S. vitreum* (Mitchill)	Walleye	USA	Carnivore	-	1925	Sport fishing
	5. Cichlidae						
20	*Tilapia zillii* (Gervais)	African Cichlid	Africa	Omnivore	+	1925	Scientific curiosity
21	*Oreochromis mossambicus* (Peters)	Tilapia	Africa	Omnivore	+	1962	Scientific curiosity
22	*O. aureus* Steindachner	Tilapia	Africa	Detritivore	-	?	Aquaculture
	6. Salmonidae						
23	*Oncorhynchus mykiss* (Walbaum)	Rainbow trout	USA	Carnivore	+	1882	Aquaculture
24	*O. gorbuscha* (W.)	Pink salmon	USA	Carnivore	+	1939	Aquaculture
25	*O. tschawytscha* W.	Chinook salmon	USA	Carnivore	-	1877	Aquaculture
26	*Hucho hucho* (L.)	Huchen	Europe	Carnivore	-	1888	Scientific curiosity
27	*Salvelinus fontinalis* (Mitchill)	Brook trout	USA	Carnivore	+	1869	Aquaculture
28	*S. namaycush* (W.)	Lake trout	USA	Carnivore	+	1888	Aquaculture
	7. Centrarchidae						
29	*Lepomis gibbosus* (L.)	Pumpkinseed Sunfish	USA	Molluscivore	+	1885	Scientific curiosity
30	*L. cyanellus* Rafinesque	Green Sunfish	USA	Carnivore	?	?	Scientific curiosity
31	*Micropterus salmoides* (Lacepede)	Large mouth bass	USA	Carnivore	+	1877	Aquaculture
32	*M. dolomieui* (Lacepede)	Small mouth bass	USA	Carnivore	+	1873	Aquaculture
33	*Ambloplites rupestris* (Rafinesque)	Rock bass	USA	Invertivore	+	1930	Aquaculture
	8. Poeciliidae						
34	*Poecilia reticulata* Peters	Guppy	USA	Larvivore	+	1963	Biological control
35	*Gambusia affinis* (Baird & Girard)	Mosquito fish	USA	Larvivore	+	1921	Biological control

	9. Clariidae						
36	*Clarias batrachus* (L.)	Walking catfish	Asia	Detritivore	-	?	Scientific curiosity
37	*C. gariepinus* (Burchell)	Sharp tooth catfish	Africa	Carnivore	-	1974	Scientific curiosity
	10. Cobitidae						
38	*Misgurnus fossilis* (L.)	Weather fish	Europe	Molluscivore	?	?	Accidental
	11. Mugilidae						
39	*Mugil cephalus* L.	Grey mullet	Asia	Planktivore	?	?	Accidental
	12. Coregonidae						
40	*Coregonus clupeaformis* (Mitchill)	Lake Whitefish	Europe	Planktivore	-	1881	Scientific curiosity
	13. Umbridae						
41	*Umbra krameri* W.	European mud minnow	USA	Larvivore	+	1925	Biological control

Table 3.3 Historical and ecological basis of metric expectation values for English rivers

Criteria	Expected number	Reference / Table / Species
Total native species in UK	50	See Moss (1988) & Table 3.6
Total introduced fish in UK	41	See Table 3.2
Most common species in UK	30	See Table 3.4
Maximum expected number of common native species in lowland rivers	18	See Table 3.4 & Section 3.3.2
Maximum expected exotic species in lowland rivers	10	Common carp, pikeperch, rainbow trout, bitterling, black bullhead, goldfish, sunbleak, ide, wels and asp
Tolerance		
Intolerant species	9	Rainbow trout, brown trout, Atlantic salmon, grayling, barbel, minnow, chub, dace and bleak
Tolerant species	15	Common carp, crucian carp, tench, roach, common bream, silver bream, 3-spined stickleback, 10-spined stickleback, rudd, gudgeon, pike, perch, pikeperch, ruffe and eel
Habitat guild		
Limnophilic (Typically vegetation preferring species)	14	Common carp, crucian carp, tench, roach, common bream, silver bream, 3-spined stickleback, 10-spined stickleback, rudd, pike, perch, pikeperch, ruffe and eel
Rheophilic species	10	Rainbow trout, brown trout, Atlantic salmon, grayling, barbel, minnow, chub, dace, bleak and gudgeon
Trophic guild		
Omnivores	10	Common carp, crucian carp, tench, roach, common bream, silver bream, rudd, chub, 3-spined stickleback and 10-spined stickleback
Invertivores	6	Grayling, barbel, minnow, dace, bleak and gudgeon
Piscivores	8	Rainbow trout, brown trout, Atlantic salmon, pike, perch, pikeperch, ruffe and eel
Water-column species	14	Rainbow trout, brown trout, Atlantic salmon, minnow, chub, dace, bleak, 3-spined stickleback, 10-spined stickleback, rudd, pike, perch, pikeperch and ruffe
Benthic species	10	Grayling, barbel, common carp, crucian carp, roach, tench, common bream, silver bream, gudgeon and eel
Reproductive guild		
Phytophilic species	7	Common carp, crucian carp, tench, rudd, perch, pikeperch and ruffe
Phytolithophilic species	5	Roach, common bream, silver bream, pike and bleak
Lithophilic species	8	Rainbow trout, brown trout, Atlantic salmon, grayling, barbel, minnow, chub and dace
Total gravel spawners	13	Rainbow trout, brown trout, Atlantic salmon, grayling, barbel, minnow, chub, dace, roach, common bream, silver bream, pike and bleak
Psammophils	1	Gudgeon
Nest builders	2	3-spined stickleback and 10-spined stickleback
Number of long-lived species in Britain	22	See Table 3.9
Number of long-lived species used in this study	2	Chub and common bream (Section 3.3.7 & Table 3.9)

Table 3.4 Classification of common freshwater fish species in English rivers into different guilds

Family English Name / Species	Origin	Tolerance	Habitat	Trophic guild	Water column / benthic species	Reproductive guild	Remarks
Cyprinidae							
Roach, *Rutilus rutilus* (L.)	N	T	LAV	O	BS	PL	Selected for RC
Dace, *Leuciscus leuciscus* (L.)	N	IS	RGS	I	WC	L	Selected for RC
Chub, *Leuciscus cephalus* (L.)	N	IS	RGS	O	WC	L	Selected for RC
Common bream, *Abramis brama* (L.)	N	T	LAV	O	BS	PL	Selected for RC
Silver bream, *Blicca bjoerkna* (L.)	N	T	LAV	O	BS	PL	Selected for RC
Rudd, *Scardinius erythrophthalmus* (L.)	N	T	LAV	O	WC	PH	Selected for RC
Barbel, *Barbus barbus* (L.)	N	IS	RSG	I	BS	L	Selected for RC
Tench, *Tinca tinca* (L.)	N	T	LMAV	O	BS	PH	Selected for RC
Common carp, *Cyprinus carpio* L.	NN	T	LMAV	O	BS	PH	Discarded
Bleak, *Alburnus alburnus* (L.)	N	IS	RSG	I	WC	PL	Selected for RC
Gudgeon, *Gobio gobio* (L.)	N	T	RSG	I	BS	PS	Selected for RC
Minnow, *Phoxinus phoxinus* (L.)	N	IS	RGS	I	WC	PL	Selected for RC
Crucian carp, *Carassius carassius* (L.)	N	MT	LMAV	O	BS	PH	Selected for RC
Goldfish, *Carassius auratus* (L.)	NN	MT	LAV	O	BS	PH	Discarded
Esocidae							
Pike, *Esox lucius* L.	N	T	LAVP	P	WC	PL	Selected for RC
Percidae							
Perch, *Perca fluviatilis* L.	N	T	LAVP	P	WC	PH	Selected for RC
Pikeperch, *Stizostedion lucioperca* (L.)	NN	T	LSGAV	P	WC	PH	Discarded
Ruffe, *Gymnocephalus cernuus* (L.)	N	T	LAVP	P	WC	PH	Selected for RC
Thymallidae							
Grayling, *Thymallus thymallus* (L.)	N	MI	RPGWO	I	BS	L	Discarded
Anguillidae							
Eel, *Anguilla anguilla* (L.)	N	T	LSM	P	BS	PS	Selected for RC

Cobitidae							
Spined loach, *Cobitis taenia* L.	N	T	LMSS	I	BS	PS	Discarded
Stone loach, *Barbatula barbatulus* (L.)	N	T	RGS	I	BS	PS	Discarded
Cottidae							
Bullhead, *Cottus gobio* L.	N	T	RSGS	I	BS	L	Discarded
Gasterosteidae							
Three-spined stickleback, *Gasterosteus aculeatus* L.	N	T	LAP	O	WC	NBP	Selected for RC
Ten-spined stickleback, *Pungitius pungitius* (L.)	N	T	LMAP	O	WC	NBP	Selected for RC
Petromyzonidae							
River lamprey, *Lampetra fluviatilis* (L.)	N	T	RSPGS	PA	BS	L	Discarded
Brook lamprey, *Lampetra planeri* (Bloch)	N	T	RSPGS	PA	BS	L	Discarded
Salmonidae							
Rainbow trout, *Oncorhynchus mykiss* (Walbaum)	NN	MI	RSPGS	P	WC	L	Discarded
Brown trout, *Salmo trutta* L.	N	MI	RSPGS	P	WC	L	Discarded
Atlantic salmon, *Salmo salar* L.	N	MI	RSPGS	P	WC	L	Discarded

RC = Reference condition, N = Native, NN = Non-native, T = Tolerant, MT = More Tolerant, IS = Intolerant species, MI = More Intolerant, O = Omnivore, I = Insectivore, P = Piscivore, PA = Parasitic, BS = Benthic Species, WC = Water Column Species, PH = Phytophils, L = Lithophils, PL = Phyto-lithophils, PS = Psammophils, NBP = Nest Builder Parental Care, LAV = Limnophilic Aquatic Vegetation, RGS = Rheophilic Gravel Stone, RSG = Rheophilic Sand Gravel, LMAV = Limnophilic Mud Aquatic Vegetation, LAVP = Limnophilic Aquatic Vegetation Plant, LSGAV = Limnophilic Sand Gravel Aquatic Vegetation, RPGWO = Rheophilic Pebble Gravel Well-Oxygenated, LSM = Limnophilic Sand Mud, LMSS = Limnophilic Mud Sand Stone, RSGS = Rheophilic Sand Gravel Stone, LAP = Limnophilic Aquatic Plant, LMAP = Limnophilic Mud Aquatic Plant, RSPGS = Rheophilic Sand Pebble Gravel Stone.

Most alien species are confined in isolated areas (Wheeler 1977) and only 10 species, i.e. common carp, pikeperch, rainbow trout, bitterling, black bullhead, goldfish, sunbleak, ide, wels and asp are found in some English rivers (Table 3.3). These were included in the "maximum expected number" for English rivers. This follows Wheeler & Maitland (1973) and Wheeler (1992). Details of distribution of coarse fishes of the UK were summarised by Wheeler (1969), Maitland (1972) and Cowx (2001) (Table 3.5).

A total of 27 species are considered common in many rivers of England and Wales (Table 3.4) and 3 exotics, i.e. common carp, goldfish and pikeperch are found in some rivers more frequently than the remaining 7 exotic species listed in Table 3.3. Roach, dace, chub, common bream, perch and pike are widely distributed and dominant species in the rivers of England. However, a maximum of 18 species can be caught from a site in lowland rivers (Table 3.4). Therefore, these species were considered for establishing reference conditions for English lowland rivers (Table 3.4). They were from 16 genera and 5 families. The 12 species discarded were: common carp, pikeperch, goldfish, grayling, spined loach, stone loach, bullhead, river lamprey, brook lamprey, rainbow trout, brown trout and Atlantic salmon (Table 3.4).

Common carp, goldfish and pikeperch were discarded as they are exotic species. They are naturalised but are only occasionally found in different rivers usually at low density (Winfield & Nelson 1991). Grayling was not included in the reference condition for lowland rivers because it prefers fast flowing waters with moderate to high gradient and is rare in the lowland stretches (Cowx & Welcomme 1998, Cowx 2001). Spined loach are rare in English waters and inhabit only a few rivers in Eastern England (Wheeler 1983). Stone loach prefer upland streams with fast-flowing conditions, having rock and stony beds. They usually do not migrate to the lowland reaches for feeding or breeding purposes. Bullheads, river lamprey and brook lamprey are rare in English lowland rivers and usually live in headwaters (Maitland 1972). Rainbow trout, brown trout and Atlantic salmon are typically found in headwaters with fast-flowing conditions, high gradients and high DO levels. Except as part of migration route, Atlantic salmon normally do not utilise the lowland stretches of a river for feeding or reproduction (Miller & Loates 1997).

3.3.3 Habitat

Each coarse fish species has preferred habitat requirements, which result in changes in community structure along the upstream-downstream gradient of a river

(Wheeler 1969). These habitat requirements have long been recognised and used to classify different zones in a river (Hawkes 1975), grouping different fish species with similar habitat preferences. It is widely acknowledged that the size of individuals, their vitality, and the spatial distribution of species are dependent on the quantity and quality of their habitat (Karr 1991). Generally, species composition and population structure are changed as a result of habitat degradation due to physical, chemical or biological alterations. The decline of pike and tench appear to be the most noticeable fishery impact as boat traffic increases in a river (Wheeler 1974).

Cyprinids are eurytopic and are present from the grayling to the bream zone (Fig. 3.1). They predominate in the warmer stretches, i.e. the barbel and bream zones as they are able to tolerate high summer temperatures and relatively low oxygen conditions. Huet (1949, 1959) divided cyprinids into three groups according to their tolerance to stream velocity:

a. running water cyprinids: barbel, chub and nase;
b. accompanying cyprinids: roach, rudd and dace;
c. still water cyprinids: carp, tench and bream.

Predators such as pike, perch and eel increase in dominance from the grayling to the bream zone but they generally thrive in the lowland stretches of rivers (Fig. 3.1). Migratory species, such as shad (*Alosa alosa* (L.), *A. fallax* (Lacepede)), may occasionally be encountered in lowland rivers (Wheeler 1969).

Table 3.5 Distribution and preferred habitat of coarse fishes in UK rivers

Species	UK distribution	Preferred habitat characteristics
Roach	Throughout UK but limited in south-west England, Wales and Scotland	Lowland rivers, bankside vegetation or open water
Dace	Throughout UK but limited in south-west England, Wales and Scotland, absent from Northern Ireland	Upper / middle reaches, gravel substrate, obstacles (rocks, etc.), fast flow
Chub	Throughout UK but limited in south-west England, Wales and Scotland, absent from Northern Ireland	Upper / middle reaches, gravel substrate, obstacles (rocks, etc.), fast flow
Common bream	Throughout UK but limited in south-west England, Wales and Scotland	Lowland reaches, slow flow, deep backwaters, vegetated areas, mud / silt substrate
Silver bream	Central, eastern and southern England only	Lowland reaches, slow flow, deep backwaters, vegetated areas, mud / silt substrate
Rudd	Throughout England and Wales, Northern Ireland, and parts of Scotland	Mainly still waters, slow flowing lowland rivers associated with littoral macrophyte stands
Barbel	Throughout England, although restricted in the north and southwest.	Middle reaches, fast flow, high oxygenation, gravel substrate, vegetation and obstructions
Tench	Throughout England, although restricted in the north and southwest. Present in east Wales and Northern Ireland	Lowland reaches, backwaters, mud / silt substrate
Common carp	Throughout the UK, except in northern Scotland	Mainly still waters, slow flowing lowland rivers, vegetation, occasionally in brackish water
Bleak	Throughout England except south-west, absent from Scotland and Northern Ireland	Middle reaches, clear-flowing water, sand / gravel substrate
Gudgeon	Throughout England except south-west,	Lower reaches, slow flow, Sand / gravel substrate

	absent from Scotland, restricted in Wales and Northern Ireland	
Minnow	Throughout the UK	Upper / middle reaches, cool, clean fast flowing waters, gravel / stone substrate
Pike	Widespread throughout the UK	Middle and lower reaches, slow-flowing to moderately-flowing, emergent vegetation, silt substrate
Perch	Widespread throughout the UK	Lowland reaches, slow-flowing, occasionally moderately -flowing shallow water with emergent and submerged vegetation, moderately productive water bodies
Ruffe	South, central and eastern England, a few sites in Wales and Scotland	Lowland reaches, still and slow flowing habitats, weed substrate
Pikeperch	Introduced, present in central and eastern England	Lowland reaches and large still waters, prefer shallow, turbid, oxygenated waters, hard substrate
Eel	Widespread throughout the UK	Middle / lowland reaches, Moderate to slow flowing, soft bottom with sand / mud
Three spined stickleback	Throughout England, Wales and Scotland, absent from Northern Ireland	Lowland reaches, slow flow, occasionally in brackish water

This theoretical classification, however, shows inconsistencies with observed data (Cowx 2001), mainly because man has impacted on rivers (Woolland *et al.* 1977). Cowx (2001) summarised the preferred habitat of coarse fish in UK rivers (Table 3.5).

In this study, six species were identified as rheophilic, these were minnow, barbel, dace, chub, gudgeon and bleak (Table 3.4). They prefer to live in a habitat with high flow conditions and clear water, using this habitat both for breeding and feeding purposes. The remaining 12 species (roach, rudd, tench, common bream, silver bream, crucian carp, 3-spined stickleback, 10-spined stickleback, perch, ruffe, eel and pike) were considered limnophilic (Table 3.4). They prefer to live, feed and reproduce in a habitat with slow flowing to stagnant conditions. This classification is based on Mann (1996) and Cowx (2001). However, in English lowland rivers the maximum expected numbers of rheophilic and limnophilic species are 10 and 14, respectively (Table 3.3).

Both water-column and benthic species are abundant in English lowland rivers. Of the 18 common species under consideration, 10 (dace, chub, rudd, bleak, minnow, pike, perch, ruffe, 3-spined stickleback and 10-spined stickleback) are water-column species (Table 3.4). They prefer to live and feed in the water column. These species usually do not go to the bottom to search for food. The remaining eight species (barbel, roach, common bream, silver bream, tench, crucian carp, gudgeon and eel) were considered bottom-dwelling species (Table 3.4). They prefer to live on or near to the bottom, from where they take food and usually do not go to the surface for feeding purposes. This classification is based on Mann (1996) and Cowx (2001), who described preferred habitat of British fishes. The maximum expected numbers of water-column and benthic in English lowland rivers species are 14 and 10, respectively (Table 3.3).

3.3.4 Diversity and density of fish

The total area of the fresh waters of England and Wales is little more than 340 square miles and it has been estimated that this could produce some 2000 t of fish in an average year (Norman 1943). Naturally, English lowland rivers have low density and diversity of fish fauna in comparison to tropical rivers. It has been reported that 50 freshwater fish species are found in Great Britain (Moss 1988) while the number is 260 in the freshwaters of Bangladesh (Rahman 1989) and 2400 in the Amazon (Goulding 1980). Moss (1988) reported that 50 freshwater fish species were from seven orders with the highest number of species belong to the order Cypriniformes (20) followed by Salmoniformes (14) and Perciformes (11), respectively. Norman (1943) reported 45

British freshwater fish while Regan (1917) tabulated only 22 species, and showed all occur in Yorkshire, and nearly all in the rivers Trent, Ouse, and in the Norfolk Broads. Table 3.6 lists freshwater fish species diversity in British rivers. There is, however, a marked diminution in the number of species towards the north of England and a similar decrease in the number occurs from east to west, with a number of species missing from Wales, west of the Severn basin (Norman 1943). Glaciation eliminated pre-existing lakes and rivers and the period afterwards, when the waterways of Britain were connected with those of mainland Europe, i.e. before sea levels rose to isolate the islands, was short (Moss 1988).

Members of the family Cyprinidae are widely distributed and abundant in the lowland rivers of Britain and is represented by 12 species (Table 3.4). Percidae is the second most important family in the English rivers, containing three species. Only pike and eel represent the family Esocidae and Anguillidae, respectively. The three-spined stickleback represents the family Gasterosteidae. This follows Wheeler (1992), who detailed the taxonomic classification of fishes of the British Isles. Some species like burbot have been extinct from the lowland rivers of England (Maitland 1972, Burnett *et al.* 1978) due to the effects of land drainage operations and the loss of adequate weed cover (Marlborough 1970). They were not included when establishing a reference condition.

Density of fish in natural waters depends on biotic and abiotic factors. Suitability of habitat and food resources determine the density of fish populations at a particular site on a river. Moreover, density of fishes greatly varies with season, lunar cycle and geographical location. There is a paucity of information on density and production of coarse fish populations in large rivers (Williams 1965, Cooper & Wheatley 1981, Cowx 1988), primarily because of the inherent problems of sampling such systems (Cowx 1996). Number of individuals in a sample depends on several factors such as sampling gear, efficiency and type of gear, time and season of sampling, type and nature of the habitat, temperature and the target species. A range of 0 - 200 fish 100 m^{-2} was recorded from 16 lowland rivers. Therefore, this catch range was taken as standard reference for a site on an English lowland river. Production of individual coarse fish varies considerably according to the waterbody, ranging from 0.01 to 11.70 g m^{-2} yr^{-1} in different European rivers (Table 3.7).

Table 3.6 Diversity of freshwater fishes in UK rivers

Area / River	Total species	Exotic species	Family	Reference
UK rivers	50	-	20	Moss 1988
UK rivers	47	13	18	Maitland 1972
Great Britain rivers	42	13	12	Giles 1994
River Severn	39	-	-	Giles 1994
River Thames	36	-	-	Giles 1994
River Hampshire Avon	33	-	-	Giles 1994
River Great Ouse	30	-	-	Giles 1994
River Tweed	16	-	-	Giles 1994
River Annan	14	-	-	Giles 1994
UK rivers	43	11	15	Wilson & Turnbull 1993
Great Britain rivers	40	6	13	Varley 1967
River Welland, Anglia	23	-	-	Leeming 1967
River Trent, Midlands	22	-	-	Cowx 1990
Rivers of Anglian Region	53	3	-	Vallipuram & Wortley 1990
River Derwent, Yorkshire	35	-	-	Burnett *et al.* 1978
River Thames (lowland reach)	21	-	-	Naismith & Knights 1994
River Dee, Cheshire	35	-	15	Hodgson 1993
River Trent	40	-	-	Braddock 1977
River Exe	20	1	11	Cowx 1980

Total biomass of fish varies widely in UK rivers due to e.g. size, location and degree of degradation. Historical data for 32 UK rivers, ranging from large to small in size, shows that total fish biomass varies from 0.00 to 128.10 g m^{-2}. Some 38% of rivers contained a biomass ranging from 18 to 47 g m^{-2} with an average of 32.50 g m^{-2} (Table 3.8). After analysing nation-wide data for UK rivers, irrespective of size and location, Cowx *et al.* (1995) concluded that an excellent fishery in the lowland rivers of England should have more than 30 g m^{-2} of fish standing crop. The NRA Thames Region (1994b) classified rivers into four groups: <10 g m^{-2}, 10 - 15 g m^{-2}, 15 - 20 g m^{-2} and >20 g m^{-2}. The Water Research Centre (WRc 1990) stated that the EEC designated Cyprinid rivers should have a minimum of 20 g m^{-2}. After considering all these recommendations, a biomass of 35 g m^{-2} was taken as a desirable biomass reference condition for English lowland rivers.

Table 3.7 Ranges of production for coarse fishes in European rivers (from Mathews 1971, Hickley & Bailey 1982, Mann & Penczak 1986 and Mann 1991)

Species	**Production (g m^{-2} yr^{-1})**
Roach, *Rutilus rutilus*	0.20 - 17.8
Dace, *Leuciscus leuciscus*	0.23 - 2.6
Chub, *Leuciscus cephalus*	0.01 - 9.4
Common bream, *Abramis brama*	0.15 - 160
Silver bream, *Blicca bjoerkna*	0.03 - 0.12
Barbel, *Barbus barbus*	0.60 - 0.43
Bleak, *Alburnus alburnus*	0.40 - 52.8
Gudgeon, *Gobio gobio*	0.84 - 11.7
Pike, *Esox lucius*	0.15 - 0.52
Perch, *Perca fluviatilis*	0.01 - 2.03
Pikeperch, *Stizostedion lucioperca*	0.09 - 42.40
Crucian carp, *Carassius carassius*	0.13
Common carp, *Cyprinus carpio*	12.0
Minnow, *Phoxinus phoxinus*	0.01 - 7.0
Rudd, *Scardinius erythrophthalmus*	4.3
Tench, *Tinca tinca*	0.94 - 2.0
Asp, *Aspius aspius*	0.37 - 2.73
Nase, *Chondrostoma nasus*	0.06 - 83.1
Sunbleak, *Leucaspius delineatus*	0.09
Schneider, *Alburnoides bipunctatus*	0.002 - 0.32
Barbel, *Barbus cyclolepis*	20.5
Roach, *Rutilus arcasii*	0.68 - 1.42

3.3.5 Trophic guild / level

Fish are typically present even in the smallest streams and in all but the most polluted waters (Karr *et al.* 1986). They occupy positions throughout the aquatic food web and thus provide an integrated view of watershed conditions. Fish communities generally include species that represent a variety of trophic levels such as planktivores, herbivores, omnivores, insectivores, benthivores, molluscivores and piscivores. They take foods from both the aquatic and terrestrial environments (Wootton 1990).

Table 3.8 Ranges of total fish biomass recorded from UK rivers

River	Biomass (g m^{-2})	Authority
Thames catchment		
Cherwell	0.30 – 44.30	Killingbeck *et al.* 1996
Windrush	2.10 – 93.00	Killingbeck *et al.* 1994
Thame	7.20 – 78.10	Cowx *et al.* 1995
Stort	1.52 – 97.69	Pilcher *et al.* 1991
Evenlode	4.80 – 59.80	Killingbeck *et al.* 1993
Dun	12.90 – 57.80	Preston *et al.* 1996
Enborne	11.80 – 37.90	Preston *et al.* 1996
Lambourn	5.40 – 45.80	Preston *et al.* 1996
Kennet	5.80 – 101.00	Butterworth *et al.* 1990
Severn-Trent catchment		
Anker	1.30 – 5.70	Severn-Trent Water Authority 1983
Blithe	10.60 – 16.80	Severn-Trent Water Authority 1983
Blythe	2.20 – 4.80	Severn-Trent Water Authority 1983
Cole	0.00 – 1.20	Severn-Trent Water Authority 1983
Churnet	1.24 – 24.80	Severn-Trent Water Authority 1983
Derwent (Derby)	7.34 – 42.56	Cowx *et al.* 1995
Dove	12.70 – 18.30	Severn-Trent Water Authority 1983
Devon	18.60 – 60.00	Severn-Trent Water Authority 1983
Mease	11.60 – 46.80	Severn-Trent Water Authority 1983
Penk	1.35 – 5.80	Cowx *et al.* 1995
Sence	4.80 – 89.80	Severn-Trent Water Authority 1983
Soar	7.30 – 17.20	Severn-Trent Water Authority 1983
Sow	2.30 – 29.30	Severn-Trent Water Authority 1983
Tame	1.00 – 6.80	Severn-Trent Water Authority 1983
Tean	4.58 – 5.18	Severn-Trent Water Authority 1983
Trent	1.30 – 89.80	Severn-Trent Water Authority 1983
Wye	7.70 – 26.80	Gee *et al.* 1978
Yorkshire Ouse catchment		
Idle	2.00 – 9.50	Severn-Trent Water Authority 1983
Nidd	0.20 – 34.24	EA, LEAP 1998b
Anglian region		
Ancholme	5.00 – 10.00	O'Hara & Williams 1991
Wensum	15.00 – 40.00	Punchand *et al.* 2000
Great Ouse/Bedford Ouse	3.80 – 43.58	Edevane 1994
Cam	0.60 – 128.10	Cowx *et al.* 1995
Average (n = 32)	**5.45 – 42.89**	

The majority of cyprinid species of English rivers are omnivorous (Table 3.4) but show a preference for benthic invertebrates and / or plant material depending on the habitat and food availability (Cowx 2001). Adult crucian carp and common bream, tend to be exclusively benthivorous while adult pike, perch, ruffe and eel are predominantly piscivorous. Silver bream and chub are facultative piscivores (Cowx 2001). Roach and rudd are omnivores, taking macrophytes and insects, while dace, barbel, tench and gudgeon prefer benthic invertebrates. Bleak and minnow are two invertivores having a preference for aerial insects. For the reference condition, 9 species (crucian carp, tench, roach, common bream, silver bream, rudd, chub, 3-spined stickleback and 10-spined stickleback), 5 (barbel, minnow, dace, bleak and gudgeon) and 4 (pike, perch, ruffe and eel) were considered as omnivore, invertivore and piscivore, respectively (Table 3.4). The maximum expected number of species classified as omnivore, invertivore and piscivore are 10, 6 and 8, respectively (Table 3.3).

3.3.6 Reproductive guild

The reproductive styles developed by fish species are strongly dependent upon the habitat characteristics of a river (Schiemer *et al.* 1991). Spawning in the coarse fish species is seasonal and is linked to both temperature (Baras 1995) and photoperiod (de Vlaming 1972).

English lowland rivers are dominated by phytophils (6 species) followed by phytolithophils (5 species) (Table 3.4) (Mann 1996, Cowx & Welcomme 1998, Cowx 2001). For the present study, both phytolithophils and lithophils were grouped under gravel spawner. For the reference condition, the total number of gravel spawners are 9 (minnow, barbel, chub, dace, roach, common bream, silver bream, pike and bleak) (Table 3.4). Gudgeon, a psammophilic species, uses sand as spawning substrate. Sticklebacks build nests and offer parental care to the offspring. The maximum expected numbers of gravel spawner are 13 (Table 3.3).

3.3.7 Long-lived species

The life span of fish ranges from 1 to 100 years (Muus 1971). The age limit of carps is 10 - 100 years, but most fish die within 7 to 10 years (Muus 1971). Life-history data for major coarse fish in UK rivers suggest they live for 3 to 40 years (Table 3.9). The females of roach, rudd, and probably of most British coarse fish, generally live longer than the males (Cowx 2001). For this study, chub and common bream were chosen as long-lived species to establish the reference condition. The chub normally lives for 12 to 14 years (Hellawell 1971, Mann 1976). Chub usually live in fast-flowing or in lowland stretches of a river, having a home range of ≈4 km (Langford 1979). Being an omnivore, it consumes a variety of foodstuffs. Therefore, chub are able to give information on habitat degradation, trophic alteration and reproductive interactions over a relatively long period.

Table 3.9 Life spans of freshwater fish of the UK

Species	Maximum life-span (range) in years	Maximum size (L∞) (cm)	Authority
Common carp, *C. carpio* L	40 (12-15)	102	Muus (1971)
Goldfish, *C. auratus* (L.)	30 (10-13)	45	Pincher (1947)
Dace, *L. leuciscus* (L.)	10 (4-5)	30	Mann (1974)
Chub, *L. cephalus* (L.)	25 (10-12)	61	Mann (1976a)
Bleak, *A. alburnus* (L.)	9 (3-5)	20	Williams (1967)
Roach, *R. rutilus* (L.)	12 (4-8)	53	Wheeler (1983)
Rudd, *S. erythrophthalmus* (L.)	17 (8-10)	45	Pincher (1947)
Bream, *A. brama* (L.)	20 (8-12)	80	Muus (1971)
Silver bream, *B. bjoerkna* (L.)	19 (5-7)	35	OVB (1988)
Barbel, *B. barbus* (L.)	25 (8-12)	136.6	Philippart (1987)
Tench, *T. tinca* (L.)	14 (6-9)	71	Giles *et al.* (1990)
Gudgeon, *G. gobio* (L.)	7 (3-4)	20	Wheeler (1983)
Minnow, *P. phoxinus* (L.)	7 (3-4)	12	Wheeler (1969)
Eel, *A. anguilla* (L.)	50 (10-15)	100	Muus (1971)
Pike, *E. lucius* L.	30 (10-14)	150	Muus (1971)
Perch, *P. fluviatilis* L.	10 (5-6)	51	Wheeler (1983)
Pikeperch, *S. lucioperca* (L.)	20 (8-10)	130	Svardson & Molin (1968)
Ruffe, *G. cernuus* (L.)	6 (3-4)	30	Muus (1971)
Three-spined stickleback, *G. aculeatus* L.	4 (1-2)	11	Pennycuik (1971)
Grayling, *T. thymallus* (L.)	14 (5-8)	60	Muus (1971)
Atlantic salmon, *S. salar* L.	13 (4-6)	150	Pincher (1947)
Brown trout, *S. trutta* L.	18 (4-6)	140	Pincher (1947)
Stone loach, *B. barbatula* (L.)	7 (2-3)	15	Muus (1971)
Bullhead, *C. gobio* L.	5 (3-4)	18	Wheeler (1983)
Ten-spined stickleback, *P. pungitius* (L.)	3 (1-2)	7	Wheeler (1983)
Mirror carp, *C. carpio*	47 (12-15)	-	Pincher (1947)
Bitterling, *R. sericeus* (Pallas)	5 (2-3)	9	Wheeler (1983)
Ide, *L. idus* (L.)	9	40	Muss (1971)
Sturgeon, *Acipenser sturio* L.	100 (30-40)	600	Muus (1971)
Twaite shad, *A. fallax* (Lacepede)	25 (8-9)	55	Muus (1971)
Allis shad, *A. alosa* (L.)	25 (9-10)	70	Muus (1971)
Arctic charr, *S. alpinus* (L.)	12 (5-8)	80	Muus (1971)

Common bream, is a hardy fish, living in lowland reaches and has a life span of 12 - 16 years (Cowx 1983). Common bream is also able to provide community information on habitat, trophic and reproductive guilds. Being an omnivore, the species may also be used to indicate the change of food base from plant and animal origin. Poor spawning success of bream usually indicates a quantitative and qualitative loss of macrophytes and aquatic vegetation. Other long-lived species like common carp (30 years), barbel (25), pike (25), pikeperch (20), silver bream (19) and tench (14) were not considered suitable to establish a reference condition. Common carp is an exotic fish and occasionally found in some rivers of the UK while barbel lives on or near bottom, feeding on benthos in the fast-flowing stretches. Pike is highly piscivorous, lives in the lowland areas, while pikeperch is an alien, having a high predatory habit and lives in the lowland stretches. Silver bream and tench also live in the lowland areas and rarely move out of the reach for feeding or breeding. With this in mind, chub and common bream were selected to represent long-lived fish group to establish a reference condition for English rivers. A density range of 0 - 55 individuals 100 m^{-2} of chub and common bream were reported from 16 English rivers. However, 0 - 25 individuals 100 m^{-2} were reported from 78% of sampling sites and this range was taken as reference for English lowland rivers (Severn-Trent Water Authority 1983, Butterworth *et al.* 1990, Pilcher *et al.* 1991, Lewis *et al.* 1992, Killingbeck *et al.* 1993, 1994 & 1996, Atkinson 1994, Edevane 1994, Preston *et al.* 1996).

3.3.8 Tolerance capacity

Each fish species has characteristic tolerances for water quality, habitat and other conditions. They have specific requirements for breeding, feeding, growth, recruitment and survival. Indeed, within each family, sets of species may be ranked for their tolerances. Some species are tolerant to the degradation of habitat while some are not. A single species may be highly tolerant of one form of pollution but intolerant to another.

In English rivers, Atlantic salmon and brown trout are less tolerant to poor water quality than cyprinids. Many coarse fish species are tolerant of low oxygen conditions; common bream survive 20%, roach 14% and common carp <14% O_2 saturation (Cowx *et al.* 1995). Normally dissolved oxygen does not act as a limiting factor for their distribution. In contrast, the Arctic charr as well as the North American brook trout and lake trout are believed to be among the fish species that are most sensitive to degradation of their natural environment.

Salinity is an important factor regulating fish distribution and abundance in the lower reaches of many English rivers. Bleak, common bream, common carp, gudgeon, pike, roach and tench, are all able to tolerate 15.5 to 18.5 gl^{-1} salinity (Cowx *et al.* 1995). Water temperature is one of most important factors affecting the growth and distribution of freshwater fish. Water temperature is not only important in terms of absolute tolerance of each species, but it also determines many aspects of the life history of coarse fish (Cowx 2001). However, fish can tolerate sudden changes in temperature, both increases and decreases, within certain limits. Each fish species has upper lethal limits of temperature on which they were grouped into three categories as follows (Varley 1967):

a. Fish with upper lethal temperatures below 28°C have optimum temperatures for growth between 7 and 17°C and spawn at temperatures up to 10°C, e.g. freshwater salmonids and grayling. These species are stenothermal as they can tolerate a narrow range of water temperature.

b. Fish with upper lethal limits between 28 and 34°C have optimum temperatures for growth between 14 and 23°C and spawn only at temperatures of 10°C or more, e.g. roach, gudgeon, common bream, pike, perch and ruffe. These species are able to tolerate a medium range of water temperature and are called mesothermal.

c. Fish with upper lethal temperatures above 34°C have optimum temperatures for growth between 20 and 28°C and spawn only when the water temperature is 15°C or higher, e.g. common carp, crucian carp, goldfish, tench, rudd, pikeperch and silver bream. These species are called eurythermal as they can tolerate a wide range of water temperature.

According to the limit of upper lethal temperature, the most tolerant species is goldfish (41°C) and the least tolerant species is grayling (24.1°C) (Varley 1967). Tolerance capacity of a species to pollution and environmental degradation depends on its genetic and physiological characters. Moreover, tolerance capacity varies with the nature and type of degradation (Varley 1967, Horoszewicz 1973). Tench, common carp, crucian carp, goldfish and 3-spined stickleback are highly tolerant of poor water quality whilst chub and barbel are less tolerant. Dace and grayling are classified as intolerant (Cowx *et al.* 1995).

In this study, five intolerant species (barbel, minnow, chub, dace and bleak) were considered to be sensitive to habitat degradation and poor water quality in English rivers. Of the 18 species, 13 tolerant species including the 3 most tolerant and 10 moderately tolerant forms were recorded (Table 3.4). Dace followed by minnow, barbel, chub and bleak were recorded as highly intolerant species. They are highly sensitive to habitat degradation and water quality. The most tolerant forms were crucian carp, tench, 3-spined stickleback and 10-spined stickleback, while gudgeon, silver bream, common bream, roach, rudd, pike, perch, ruffe and eel were designated as moderately tolerant forms (Table 3.4). The maximum expected numbers for intolerant and tolerant species are 9 and 15, respectively (Table 3.3).

In terms of key water quality parameters, such as ammonia, dissolved oxygen, suspended solids, the major freshwater species are categorised according to their tolerance as follows (Fig. 3.2) (Cowx 2001):

High

Tench

Common bream, gudgeon

Roach, rudd, pike

Increasing tolerance

Chub

Perch

Dace

Barbel

Grayling, trout, salmon

Low

Fig. 3.2 Tolerance of fish to environmental degradation

According to Karr (1981), who developed an IBI for Mid-western USA, the percentage of intolerant forms should not be greater than 10% of the total species identified in an ecoregion. However, the total number of species including intolerant and tolerant forms varies according to the geographical location and position, region, climate and waterbody. The Mid-western USA, a warm water region and the birth place of IBI, has a high species diversity. Conversely, rivers of England, a temperate system, have a comparatively low species diversity. Therefore, it is expected that the number of intolerant forms in England will be higher than warm water regions, although total number of species is low. A high percentage of intolerant species was used by Mundahl & Simon (1999) to develop an IBI for coldwater streams. In this study, intolerant forms represent 28% of total species recorded, and this was accepted to develop a reference condition for the English rivers. The summary of the reference condition is presented in Table 3.10.

Table 3.10 Summary of the reference condition proposed for English lowland rivers

Criteria	Reference number	Table / Species
Maximum expected number of common native species in lowland rivers	18	See Table 3.4 & Section 3.3.2
Maximum expected exotic species in lowland rivers	10	Common carp, pikeperch, bitterling, rainbow trout, black bullhead, goldfish, sunbleak, ide, wels and asp
Tolerance		
Intolerant species	5	Barbel, minnow, chub, dace and bleak
Tolerant species	13	Crucian carp, tench, roach, common bream, silver bream, 3-spined stickleback, 10-spined stickleback, rudd, gudgeon, pike, perch, ruffe and eel
Habitat guild		
Limnophilic (Typically vegetation preferring species)	12	Crucian carp, tench, roach, common bream, silver bream, 3-spined stickleback, 10-spined stickleback, rudd, pike, perch, ruffe and eel
Rheophilic species	6	Barbel, minnow, chub, dace, bleak and gudgeon
Water-column species	10	Minnow, chub, dace, bleak, 3-spined stickleback, 10-spined stickleback, rudd, pike, perch and ruffe
Benthic species	8	Barbel, crucian carp, roach, tench, common bream, silver bream, gudgeon and eel
Trophic guild		
Omnivores	9	Crucian carp, tench, roach, common bream, silver bream, rudd, chub, 3-spined stickleback and 10-spined stickleback
Invertivores	5	Barbel, minnow, dace, bleak and gudgeon
Piscivores	4	Pike, perch, ruffe and eel
Reproductive guild		
Phytophilic species	5	Crucian carp, tench, rudd, perch, ruffe
Phytolithophilic species	5	Roach, common bream, silver bream, pike and bleak
Lithophilic species	4	Barbel, minnow, chub and dace
Total gravel spawners	9	Barbel, minnow, chub, dace, roach, common bream, silver bream, pike and bleak
Psammophils	1	Gudgeon
Nest builders	2	3-spined stickleback and 10-spined stickleback
Abundance		
Number of individuals of long-lived species used in this study (No. 100 m^{-2})	≥25	Chub and common bream (Section 3.3.7 & Table 3.9)
Number of individuals in sample (No. 100 m^{-2})	≥200	Section 3.3.4
Biomass for English lowland rivers (g m^{-2})	≥35	Section 3.3.4

CHAPTER FOUR

4. FISH SPECIES ASSEMBLAGE, DIVERSITY, DENSITY AND ABUNDANCE IN THREE CATCHMENTS

4.1 INTRODUCTION

In this chapter the status of fisheries in study rivers is evaluated using different indices. Structural and functional components of fish communities were used to evaluate the status of fish communities. Fish communities are formed by either direct and/or indirect interactions of all fish species, irrespective of taxonomic identity in a defined area or habitat (Angermeier & Karr 1983). All the fish species in an area constitute an assemblage (Dionne & Karr 1992). Resource partitioning is a common characteristic of fish assemblages (Moyle & Senanayake 1984). Food and habitat are the two resources that most commonly seem to be partitioned by coexisting species. Diurnal and seasonal rhythms, temperature, dissolved oxygen, BOD and suspended solids also influence fish assemblages. Structurally complex habitats usually have higher species richness than more homogeneous habitats. Abiotic factors, such as disturbance, predation and variable recruitment play a role in preventing an assemblage from reaching an equilibrial species richness determined by interspecific competition (Wootton 1990).

The concept of species richness, the number of species in a sample from a community, is relatively old, although it continues to be the most basic attribute of ecological communities measured (Karr 1981, Scott *et al.* 1987). Stressed fish communities show changes in community structure, i.e. changes in the number of species (richness) and their relative abundance (evenness). Environmental degradation is shown to change from "diverse" communities consisting of many species that are relatively equally abundant, to "simple" assemblages dominated by a few species (Fausch *et al.* 1990) (Fig. 4.1). Fish species richness varies as a function of region, stream size, elevation and stream gradient. Upstream lower-order sections with high gradient usually have lower species richness than downstream sections (Leonard & Orth 1986, Karr 1991). Tropical systems have high species richness compared to temperate systems, but both systems can be classified as healthy (Hocutt 1981).

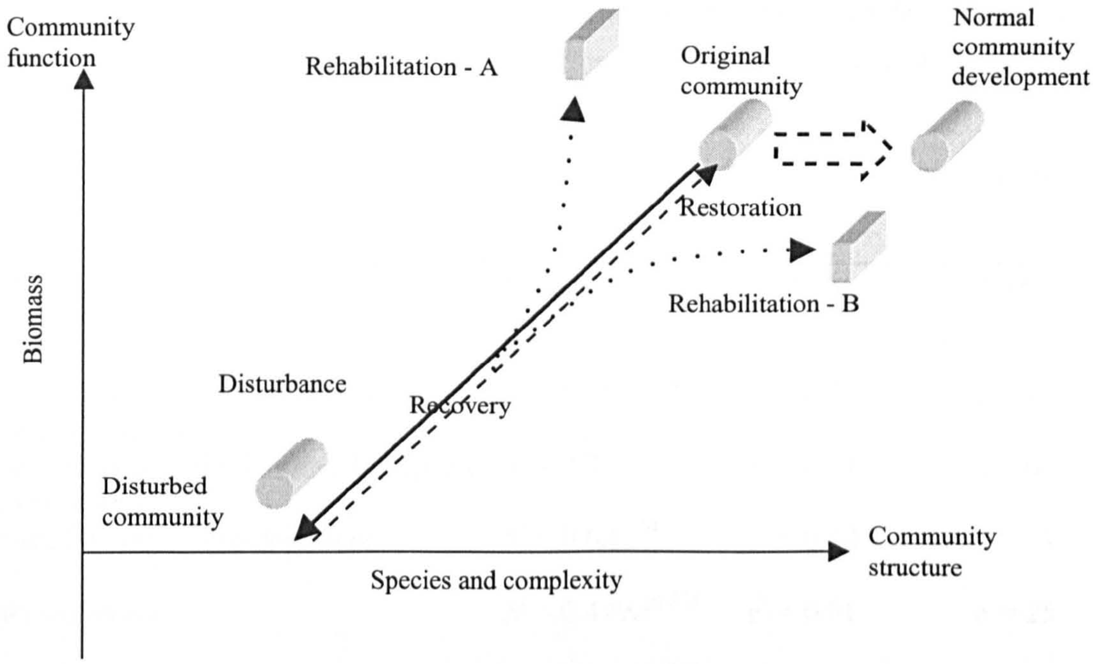


Fig. 4.1 Recovery pathways of a disturbed fish community (after Cowx 1999)

Fish assemblages in rivers are highly complex and environmental change produces impacts on community structures (Welcomme 2000). Fish species richness in a river is strongly correlated with its basin area (Oberdorff *et al.* 1995) described by:

$$N = \mathrm{f}A^{\mathrm{b}}$$

or:

$$\log_e N = \log_e \mathrm{f} + \mathrm{b} \log_e A$$

where, N is the number of species, A is the basin area (km^2) and f and b are parameters determined empirically. The exponent b is typically less than 1.0, indicating a decrease in the rate at which species richness increases with area. Oberdorff *et al.* (1995) derived a global relationship for 292 rivers:

$$\log \textit{Number of species} = 0.478 + \log \textit{drainage basin area (km}^2\textit{)},\ r^2 = 0.439$$

The correlation can be improved considerably if individual continental data sets are analysed separately (Welcomme 1985). There is a remarkably small effect of latitude on the overall distribution of fish in rivers. The species numbers in individual river systems range from tens in small basins to over 2000 in the Amazon (Goulding

1980). Usually considerable numbers of species are present in the lower reaches of most rivers. Relationships between species richness and basin area for a group of 45 rivers from South America, Africa, Asia and Europe are shown in Table 4.1.

Table 4.1 Correlation between fish species richness and basin area (after Welcomme 1979 & 1985)

Rivers	**Relationship**	**r^2 value**	**Number of rivers**
South American, African, Asian and European rivers	$N = 0.297A^{0.477}$	$r^2 = 0.77$	n= 45
North flowing Siberian and European Soviet rivers	$N = 2.76A^{0.19}$	$r^2 = 0.91$	n = 6
South flowing European rivers	$N = 0.6A^{0.14}$	$r^2 = 0.72$	n = 11
African rivers	$N = 0.449A^{0.434}$	$r^2 = 0.91$	n = 25
South American rivers	$N = 0.169A^{0.552}$	$r^2 = 0.95$	n = 11
Greek rivers	$N = 2.319A^{0.24}$	$r^2 = 0.94$	n = 12
Portuguese rivers	$N = 1.786A^{0.19}$	$r^2 = 0.92$	n = 12

The size of British rivers is small by world standards. Species diversity and richness are also low (Petts 1988) because of geological isolation. Therefore, relationships between species richness and basin area do not suit English River systems.

Species diversity indices are mathematical expressions which use three components of community structure, namely species richness (number of species present), evenness (uniformity in the distribution of individuals among the species) and abundance (total number of organisms present) to describe the response of a community to the quality of its environment (Metcalfe-Smith 1994). Undisturbed environments are characterised by high diversity indices or richness, an even distribution of individuals among the species, and moderate to high counts of individuals (Ghetti & Bonazzi 1977, Kolavak 1981, Mason *et al.* 1985). The purpose of measuring a community's diversity is usually to judge its relationship to other community properties such as productivity and stability (Pielou 1975) or to the environmental conditions to which the community is exposed.

4.2 MATERIALS AND METHODS

Data from 457 sampling sites in 22 rivers in 3 different catchments were used in this study (Table 2.1). All sites were sampled by electric fishing (Section 2.2.2). Data on fish abundance and species composition were obtained from regional offices in the Thames, Midlands and Northeast regions of the Environment Agency, UK. Data were sorted, arranged and analysed by different methods to obtain outputs for the different indices.

4.2.1 Diversity indices

Diversity may be expressed as species richness, evenness or equitability. Species richness can be expressed as the number of species in a sample or habitat, or could be expressed more usefully as species richness per unit area. The use of indices of community diversity is based upon the concept that the structure of normal communities will be changed by perturbations in the environment and the degree of change in community structure may be used to assess the intensity of the environmental stress. A number of indices have been developed to measure diversity (Magurran 1988). However, all indices are not equally suitable to measure diversity of all types of organism. Each index has specific character, which differentiates it from other indices. Therefore, considerable differences exist between the indices (Hellawell 1977).

Diversity measures are used extensively to gauge the adverse effects of pollution and environmental disturbance (Magurran 1988). Although there is considerable disagreement about which index or model is the most sensitive indicator of damage, the general picture that emerges is that polluted or stressed environments experience a shift from a log normal pattern of species abundance, to an increase in dominance of a small number of species and a decrease in species richness. Species diversity measures are divided into three main categories (Magurran 1988):

i. Indices for species richness

The number of species is the first and oldest concept of species diversity and called species richness. Species richness is best expressed as number of species per m^2, which is the most commonly used measure of species richness (Spellerberg 1991). Species richness is expressed, for example, as number of species of organisms/100 individuals (Homer 1976). The simple index for species richness is as follows.

Margalef's diversity index (D_{Mg})

Margalef's index (D_{Mg}) is intuitively simple but sensitive to sample size and sampling effort. The index is a measure of mean population size and uses the form $S - 1$ (Number of species recorded minus 1), rather than S, giving a value of zero if there is only one species (Margalef 1951). Furthermore, the index is misleading because it fails to take account of abundance patterns. Margalef's index is expressed as:

$$D_{Mg} = (S - 1) / \log_e N$$

where,

D_{Mg} is the Margalef's diversity index,
S is the number of species,
N is the total number of individuals.

ii. Indices for dominance measures

Simpson's diversity index (D_{Sm})

Assemblage structure of fish communities is essentially unpredictable. Relative abundance of fish can be explained by Simpson's diversity index. The index is only sensitive to the abundance of the more plentiful species in a sample (Whittaker 1965). Simpson's diversity index is a nonparametric measure of diversity, which is derived from probability theory. In circumstances where it is not practical or not possible to determine the total number of individuals then a random sample has to be used (Simpson 1949). The Simpson's index (D_{Sm}) is based on random sampling and is sensitive to sample size and dominance (Pielou 1969). Simpson's index is calculated as:

$$D_{Sm} = \sum_{i=1}^{S} P_i^2$$

where, $(P_i^2) \sim (N_i / N_T)^2$

D_{Sm} is the Simpson's diversity index (a measure of the probability that two randomly sampled individuals belong to different species),
P_i is the relative (proportional) abundance of the ith species,
N_i is the number of the ith species and
N_T is the total number of individuals.

The index is sometimes used in the following form:

$$D_{Sm} = 1 - \sum P_i^2$$

Simpson's index is heavily weighted towards the most abundant species in the sample and less sensitive to species richness or rare species. For example, if a single species dominates a community (so that the community's diversity is low), then Simpson's diversity index value will be high. While numerous species that all are fairly evenly present, will produce a low index value.

This problem is avoided by the use of a reciprocal form of Simpson's index. This ensures that the value of the index increases with increasing diversity. Simpson's index ranges in value from 0 (low diversity) to a maximum of 1 ($1 - 1/S$), where S is the number of species.

iii. Indices for information statistics

Shannon-Wiener diversity index (H')

The main objective of information indices is to measure the amount of *order* or *disorder* contained in the system (Shannon & Wiener 1948). The Shannon-Wiener index (H'), based on the proportional abundance of species, provides an alternative approach to the measurement of diversity (Magurran 1988). The H' measures the diversity per individual in a multi-species population. The index takes into account the evenness of the abundance of species (Peet 1974). The H' assumes that individuals are randomly sampled from an infinitely large population (Pielou 1975). The index also assumes that all species are represented in any sample. A substantial source of error comes from a failure to include all species from the community in the sample (Peet 1974) and thus estimates of H' should always be accompanied by estimates of their standard error. This error increases as the proportion of species represented in the sample declines, i.e., it is strongly influenced by species richness. The H' estimates the diversity of the unsampled, as well as sampled, portion of the community (Magurran 1988). The Shannon-Wiener index is simple in computation and calculation but difficult to interpret. The H' is calculated as:

$$H' = - \sum P_i \ln P_i$$

Where, $(P_i) \sim (N_i / N_T)$

H' is the Shannon-Wiener index of diversity,
P_i is the relative abundance of the ith species,
N_i is the number of the ith species and
N_T is the total number of individuals.

In this index natural $\log_e$ (ln) is used because this gives information in binary digits (i.e. $\log_{10}$ of 100 is 2 while $\log_e$ of 100 is 4.6). The value of the H' usually ranges between 1.5 and 3.5 and rarely exceeds 4.5 (Margalef 1972).

4.2.2 Relative abundance (*Ra*)

Relative abundance (*Ra*) measures the relative proportions of different species in the community. Relative abundance is calculated as:

$Ra = (Ni \ / \ Tn) \times 100$

Where,

Ni is the number of individuals of species i,

Tn is the total number of all species.

4.2.3 Abundance / Biomass Comparison (ABC) index

The Abundance / Biomass Comparison, or ABC, method was proposed and applied by Warwick (1986) as a technique for detecting pollution effects on marine macrobenthic communities. The ABC index is based on sound ecological principles instead of on statistical properties such as the log-normal distribution. The distribution of numbers of individuals among species differs from the distribution of biomass between species when influenced by a disturbance (Warwick 1986). This difference can be shown easily by k-dominance plots (Lambshead *et al.* 1983). The ABC index is sensitive to all kinds of stress or disturbance (Meire & Dereu 1990). This index is calculated as the average of the difference between cumulative biomass and abundance:

$$ABC\ index = \Sigma(B_i - A_i) \ / \ N$$

Where:

B_i is the percentage dominance for biomass of species i (ranked from the highest to the lowest biomass);

A_i is the percentage dominance for abundance of species i (ranked from the most to the least abundant species);

N is the total number of species.

The index is negative in heavily stressed situations, near zero in moderately stressed situations and positive in unstressed situations (Meire & Dereu 1990). The number of times the cumulative percentage dominance for biomass is higher than the cumulative percentage dominance for abundance can be totalled and expressed as the percentage of the total number of species minus one (cumulative biomass dominance).

This percentage gives an idea whether the biomass curve lies just above, just below, or intersects the abundance curve when the ABC index is close to zero.

When the community approaches equilibrium, the biomass becomes increasingly dominated by one or a few species with large sized individuals, each represented by few individuals. The numerical dominants are smaller species. Hence, when plotted as k-dominance curves, 'numerical diversity' is greater than 'biomass diversity', so that the line for abundance lies well below the line for biomass, since one species forms a much larger proportion of the total biomass than it does of the total numbers (Fig. 4.2a). The k-selected species are characterised by a large body, long lifespan, delayed maturity and reproduction, few offspring, parental care, trophic specialisation and a population size which is fairly constant in time and close to the carrying capacity of the environment (Pianka 1970). Under stress (natural physical and biological or pollution-induced disturbance), large competitive dominants should be eliminated and biomass and abundance curves will be close together and crossing one or several times (Fig. 4.2b). Under severe disturbance, fish communities become increasingly dominated by one or a few small species (usually highly tolerant species) and few larger species are present. Hence 'numerical diversity' is lower than 'biomass diversity' (Fig. 4.2c).

Application of the technique to several data sets showed that it is a sensitive indicator of natural, physical and biological disturbance as well as pollution-induced disturbance over space and time (Warwick 1986, Warwick *et al.* 1987, Warwick & Ruswahyuni 1987). Coeck *et al.* (1993) tested the ABC method on distribution patterns of number and biomass among fish species in regulated and non-regulated lowland rivers in Belgium. They stated that the ABC method is applicable to the assessment of disturbance in fish communities in rivers and that the method gives information about both pollution and physical disturbance.

They further concluded that the ABC method is a useful instrument to assess the status of a fish community before and after river channel works or natural and human induced river regulations. However, Meire & Dereu (1990) strongly advised the application of this method to as diverse as possible a range of data sets to further evaluate its applicability as an indicator of environmental stress.

4.2.4 Multivariate analysis of the fish assemblage

An "assemblage" represents a phylogenetic subset of a community, while a community is the entire biological component of an ecosystem (Fauth *et al.* 1996). Fish assemblages, which are groups of species that co-occur in the same area, are structured by local, regional and historical processes operating at various spatial and temporal scales (Tonn 1990). Multivariate techniques can analyse many fish assemblage variables simultaneously, which summarise complex data sets and present the results in an easily communicable fashion.

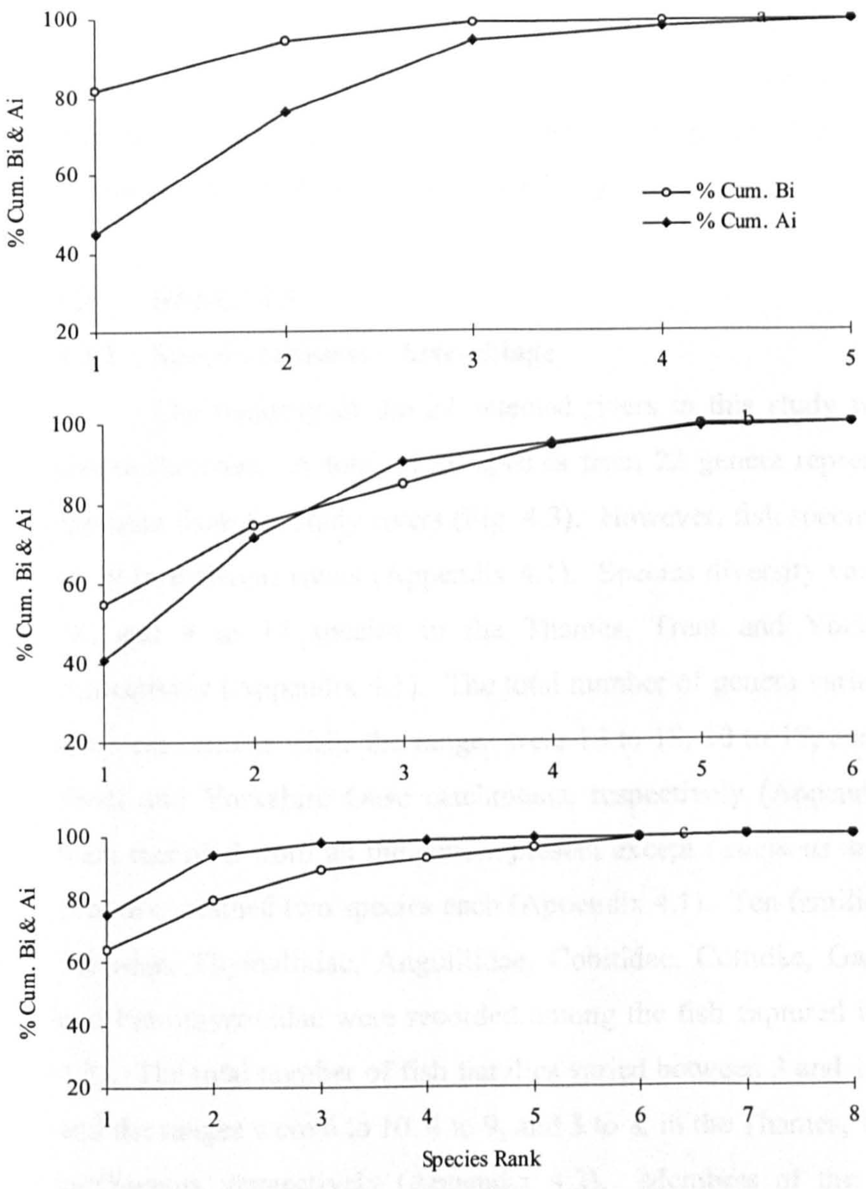


Fig. 4.2 Hypothetical K-dominance curves for species abundance and biomass, showing unstressed (a), moderately (b) and heavily stressed (c) conditions (after Warwick 1986)

Percent similarity of sites on the basis of fish abundance and biomass was determined by cluster analysis through UPGMA (Unweighted-Pair Groups Method

Analysis (Sokal & Michener 1958)). The Multi-Variate Statistical Package 1998, version MVSP 3.0, was used to obtain dendrograms for abundance and biomass (Kovach 1998). The UPGMA is based on the average distance between samples with recalculation of linkage distance after each successive linkage. TWINSPAN (Two-Way INdicator SPecies ANalysis) (Hill 1979a) and DECORANA (DEtrended CORrespondence ANAlysis) (Hill 1979b) outputs were obtained using the Community Analysis Package (CAP) programme (Pisces Conservation Ltd, 1999). The TWINSPAN programme classifies the sites and constructs an ordered two-way table from a sites-by-species matrix, while the DECORANA programme with its "down-weighting" option, gives a low "weight" to a species that occurs in a few sites and minimises their influence in the assemblage.

4.3 RESULTS

4.3.1 Species richness / Assemblage

The majority of the 22 selected rivers in this study were found to have good coarse fisheries. A total of 24 species from 22 genera representing 10 families were captured from the study rivers (Fig. 4.3). However, fish species richness ranged from 9 to 19 in different rivers (Appendix 4.1). Species diversity varied from 14 to 19, 11 to 18, and 9 to 17 species in the Thames, Trent and Yorkshire Ouse catchments, respectively (Appendix 4.1). The total number of genera varied between 8 to 18 in the three catchments while the ranges were 13 to 18, 10 to 17, and 8 to 16, in the Thames, Trent and Yorkshire Ouse catchments, respectively (Appendix 4.1). Single species were recorded from all the genera present except *Leuciscus* and *Carassius*. These two genera contained two species each (Appendix 4.1). Ten families, Cyprinidae, Esocidae, Percidae, Thymallidae, Anguillidae, Cobitidae, Cottidae, Gasterosteidae, Salmonidae and Petromyzonidae were recorded among the fish captured in these rivers (Appendix 4.2). The total number of fish families varied between 3 and 10 in the three catchments and the ranges were 6 to 10, 4 to 9, and 3 to 8, in the Thames, Trent and Yorkshire Ouse catchments, respectively (Appendix 4.2). Members of the family Cyprinidae were caught in all selected rivers (100% of distribution). Distribution of different fish families ranged between 27 and 100% of selected rives (Appendix 4.2). Species diversity including the number of genera and families, decreased from south to north of England, in agreement with Wheeler (1977), who stated, in Britain the number of freshwater fish species decreases from south to north and from east to west.

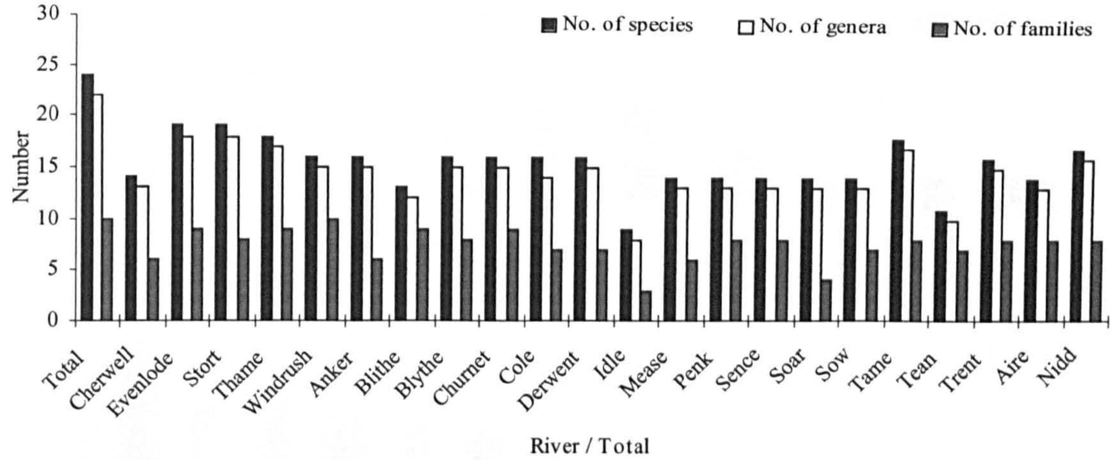


Fig. 4.3 Number of species, genera and families found in study rivers

Chub, dace and roach were found to be present in all 22 rivers studied (100% of distribution), but no pikeperch, spined loach or 10-spined stickleback were caught (Appendix 4.1). Minor species; minnow, stone loach, bullhead and three-spined stickleback were found in 82 to 86% of study rivers. Three exotic species; rainbow trout, common carp and goldfish were recorded from the Thames and Trent catchments, whilst only rainbow trout was caught from the Yorkshire Ouse catchment. Exotic species were present in 5% (goldfish) to 41% (rainbow trout) of the study rivers (Appendix 4.1). Percent similarity of sites based on UPGMA dendrograms was considered at 60% level for all the study rivers as most authors used this level to show the similarity of sites (Wolda 1981).

4.3.2 Results for fish diversity & density, Margalef, Simpson, Shannon-Wiener diversity indices and ABC method

The Thames catchment

River Cherwell

Ten species of coarse fish and four minor species (Appendix 4.1) from 13 genera and six families (Appendix 4.2) were recorded from the 13 sites on the River Cherwell. Eurytopic species such as pike, chub, dace and roach were the most widespread species in the river, being present in > 65% of the sites (Fig. 4.4a). Poor species diversity was recorded at West Farndon Mill and Tramroad Industrial Estate (sites 1 & 5), but was high in the lower reaches between Millhouse Farm and, Angel & Greyhound Meadows (sites 8 - 13) (Appendix 4.3).

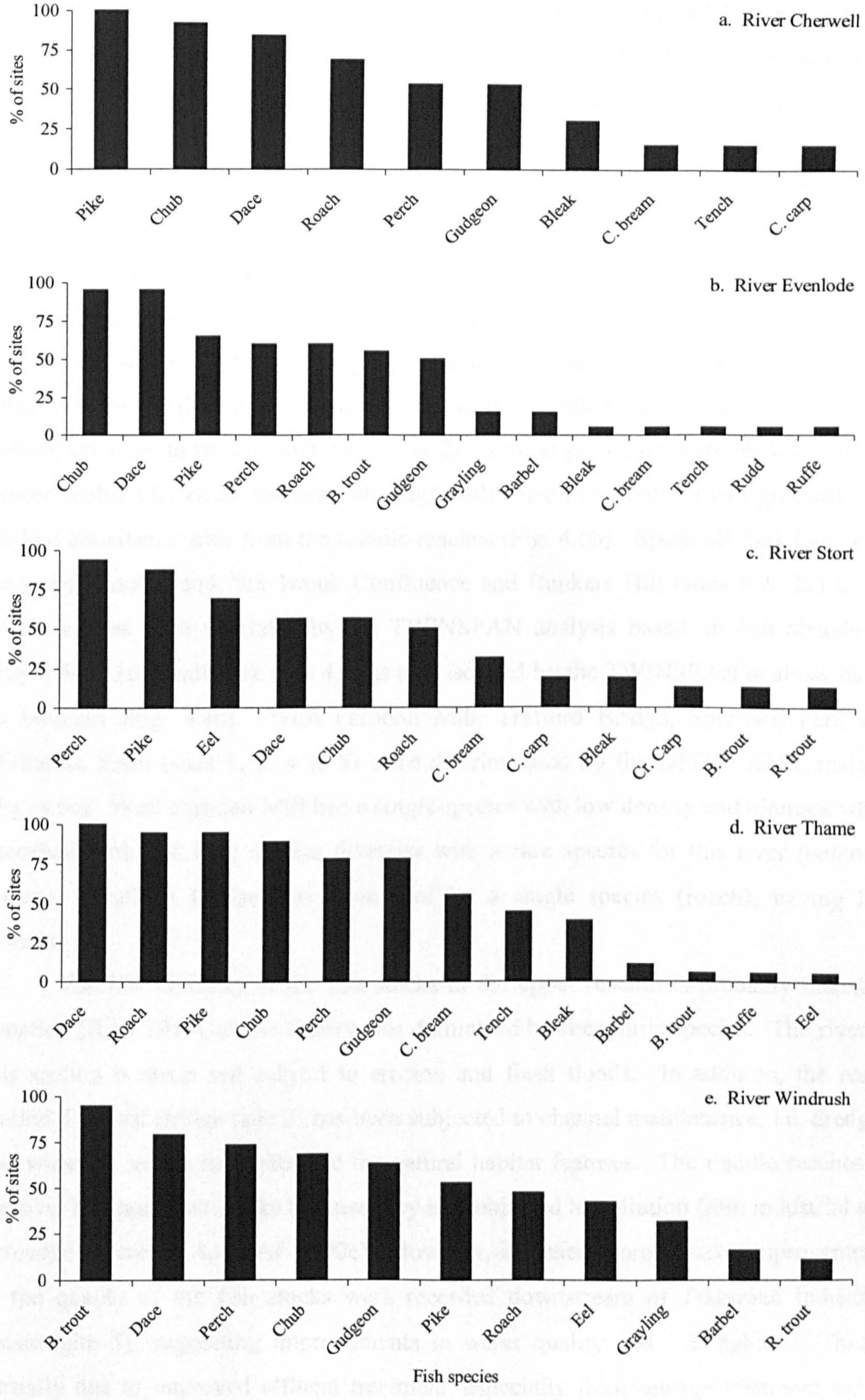


Fig. 4.4 Percentage of sites in the Thames catchment containing major fish species

Fish density was low to moderate and ranged between 0.2 and 13.5 fish 100 m^{-2}, with a mean of 5.65±4.73 fish 100 m^{-2} (Fig. 4.5a). Highest fish density was recorded at Slat Mill (site 3), where seven species were recorded, but density was very low at West Farndon Mill and Trafford Bridge (sites 1 & 2) in the upper reaches and Tramroad Industrial Estate, Footbridge near M40 and Twyford Mill (sites 5, 6 & 7) in the middle reaches.

These observations on the diversity, density and assemblage were supported by the various diversity indices (Figs 4.5b, c, d & e). The ABC index also discriminated Sor Brook Confluence (site 9) as poor. Cluster analyses, based on fish density (Fig. 4.6a) grouped the sites with poor abundance in the middle reaches (sites 5, 6 & 7) and isolated the most upstream reach (sites 1 & 2). Similar groupings were obtained for the cluster analysis based on biomass, although Millhouse Farm (site 8) was grouped with the low abundance sites from the middle reaches (Fig. 4.6b). Spiceball Park (site 4) in the upper reaches and, Sor Brook Confluence and Bunkers Hill (sites 9 & 12) in the lower reaches were separated by the TWINSPAN analysis based on fish abundance (Fig. 4.6c). Spiceball Park (site 4) was also isolated by the TWINSPAN analysis based on biomass (Fig. 4.4d). West Farndon Mill, Trafford Bridge, Spiceball Park and Millhouse Farm (sites 1, 2, 4 & 8) were discriminated by the DECORANA analysis (Fig. 4.6e). West Farndon Mill had a single species with low density and biomass while Spiceball Park had high species diversity with a rare species for this river (common bream). Trafford Bridge was dominated by a single species (roach), having low biomass.

The low diversity of the fish stocks in the upper reaches is probably linked to zonation (Huet 1959), as the fishery was dominated by rheophilic species. The river in this section is steep and subject to erosion and flash floods. In addition, the reach around Trafford Bridge (site 2) has been subjected to channel maintenance, i.e. dredged and widened, which has destroyed the natural habitat features. The middle reaches of the river had poor fish stocks because they are subjected to pollution from industrial and domestic sources (EA, LEAP 2000e). However, in general, progressive improvements in the quality of the fish stocks were recorded downstream of Tramroad Industrial Estate (site 5), suggesting improvements in water quality and / or habitat. This is partially due to improved effluent treatment, especially from sewage treatment works (STWs), but also the expected improved diversity in the lower reaches predicted by river continuum theory (Naiman *et al.* 1988). Abbreviations for site characteristics e.g. Er., Lc., Dg., etc. are presented in glossary.

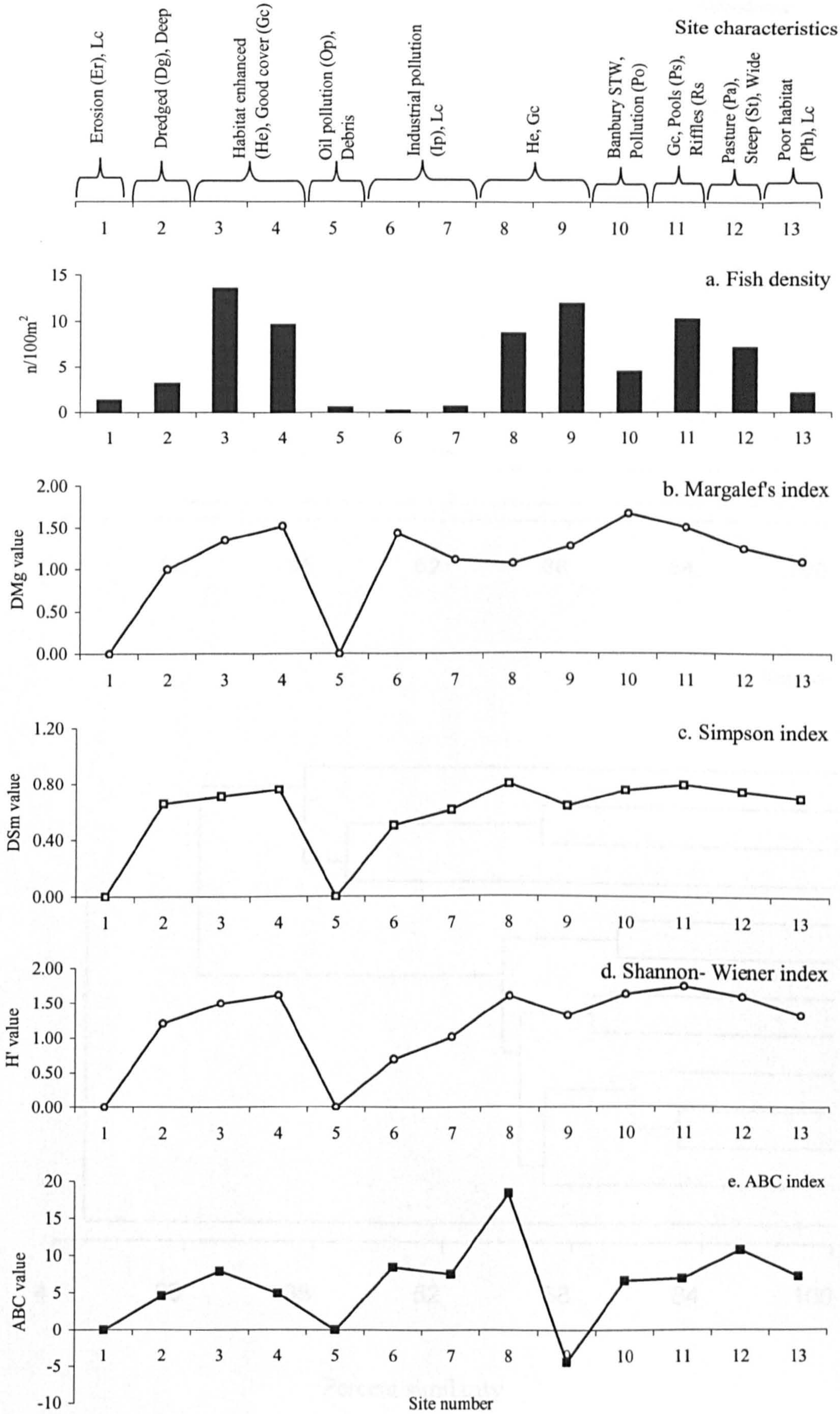


Fig. 4.5 Site characteristics and variations in different indices for the River Cherwell

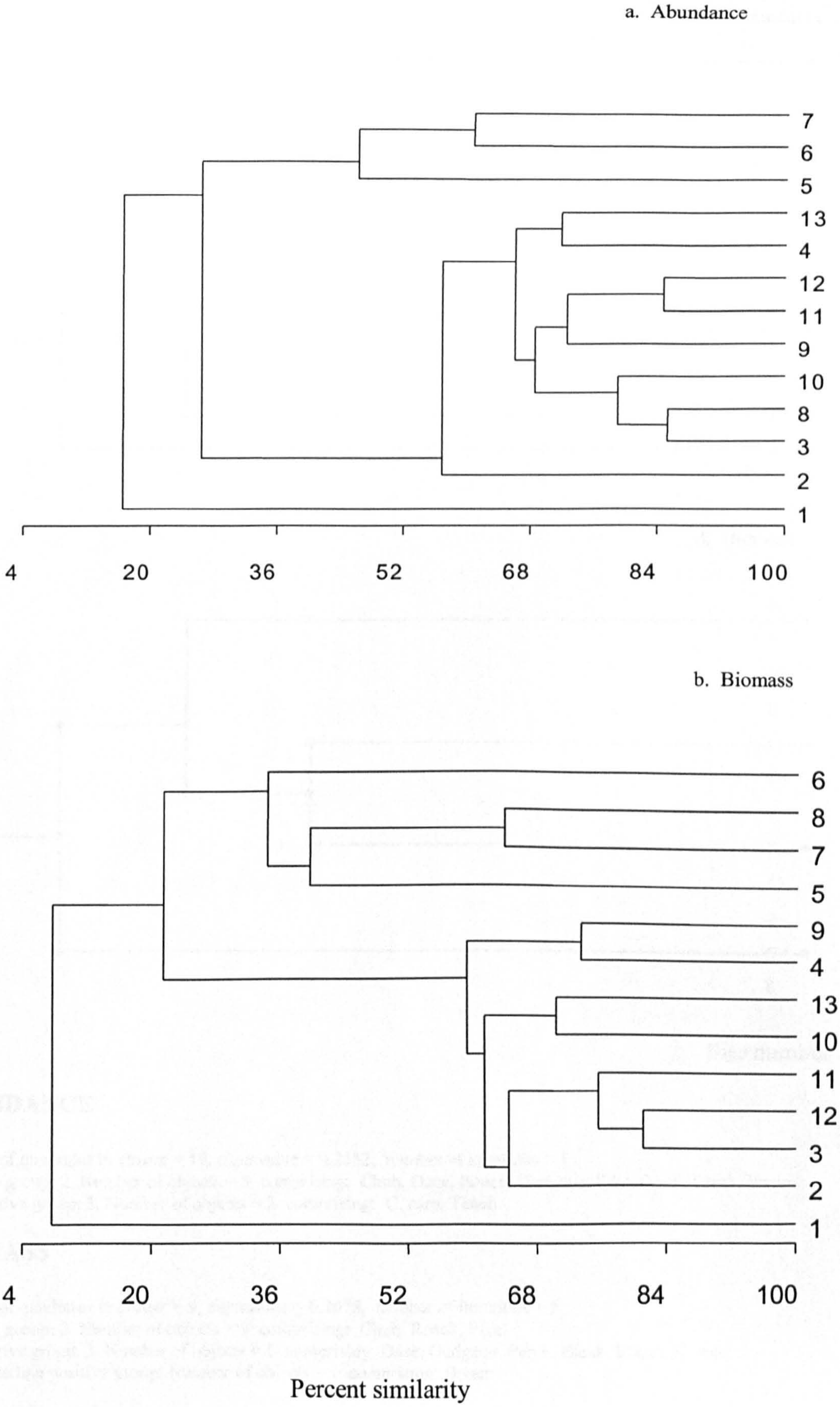
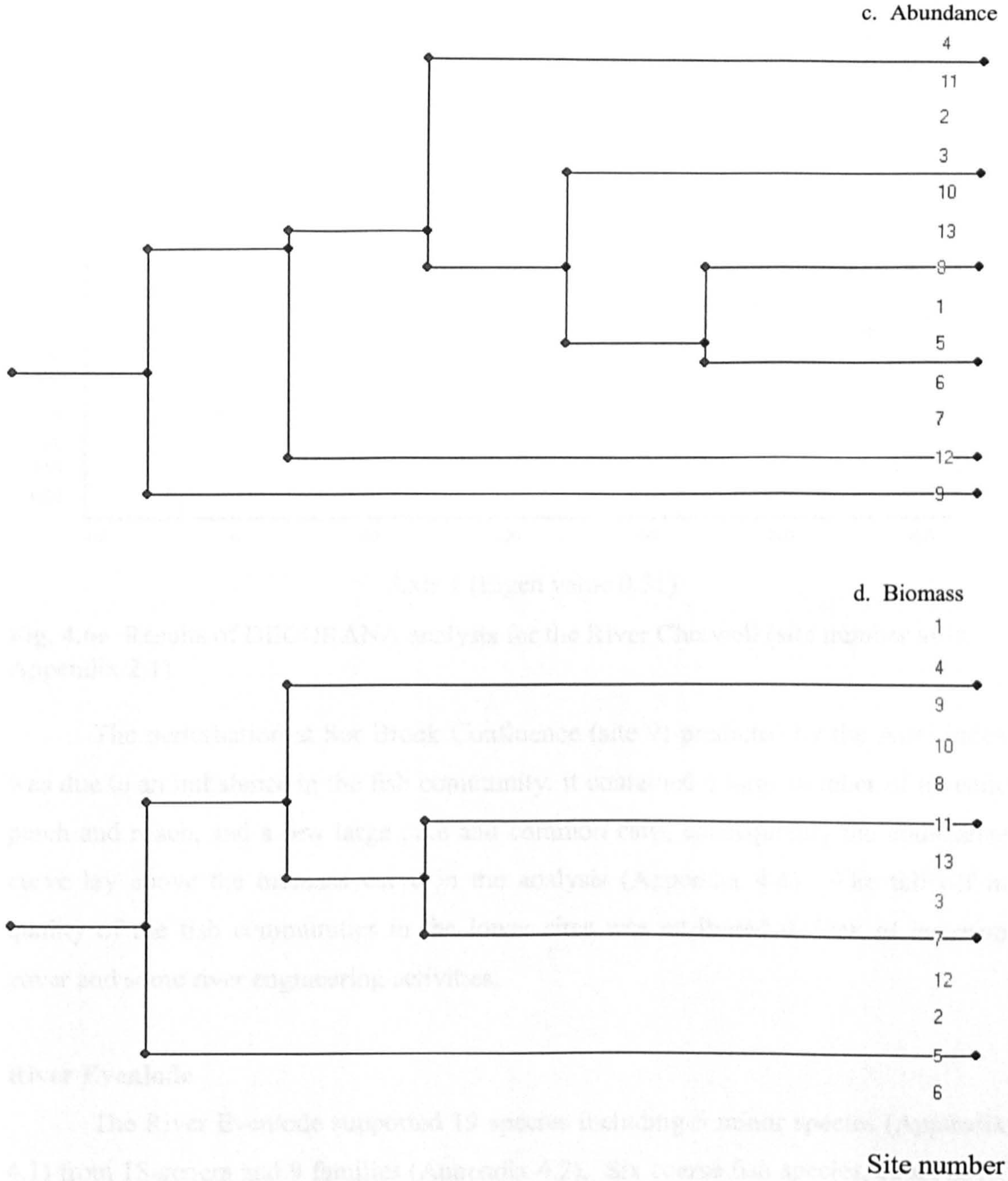


Fig. 4.6 Dendrograms based on UPGMA for the River Cherwell (site number as in Appendix 2.1)



ABUNDANCE

Number of quadrates in cluster = 10, eigenvalue = 0.2582, number of iterations = 5
Negative group: 2, Number of objects = 8 comprising: Chub, Dace, Roach, Gudgeon, Pike, Perch, Bleak, Bream
The positive group: 3, Number of objects = 2 comprising: C. carp, Tench

BIOMASS

Number of quadrates in cluster = 9, eigenvalue = 0.2678, number of iterations = 5
Negative group: 2, Number of objects = 3 comprising: Chub, Roach, Pike,
The positive group: 3, Number of objects = 6 comprising: Dace, Gudgeon, Perch, Bleak, Bream, C. carp
The borderline positive group: Number of objects = 1 comprising: Bream

Fig. 4.6 (Continued) Results of TWINSPAN for the River Cherwell at one level of division (site number as in Appendix 2.1)

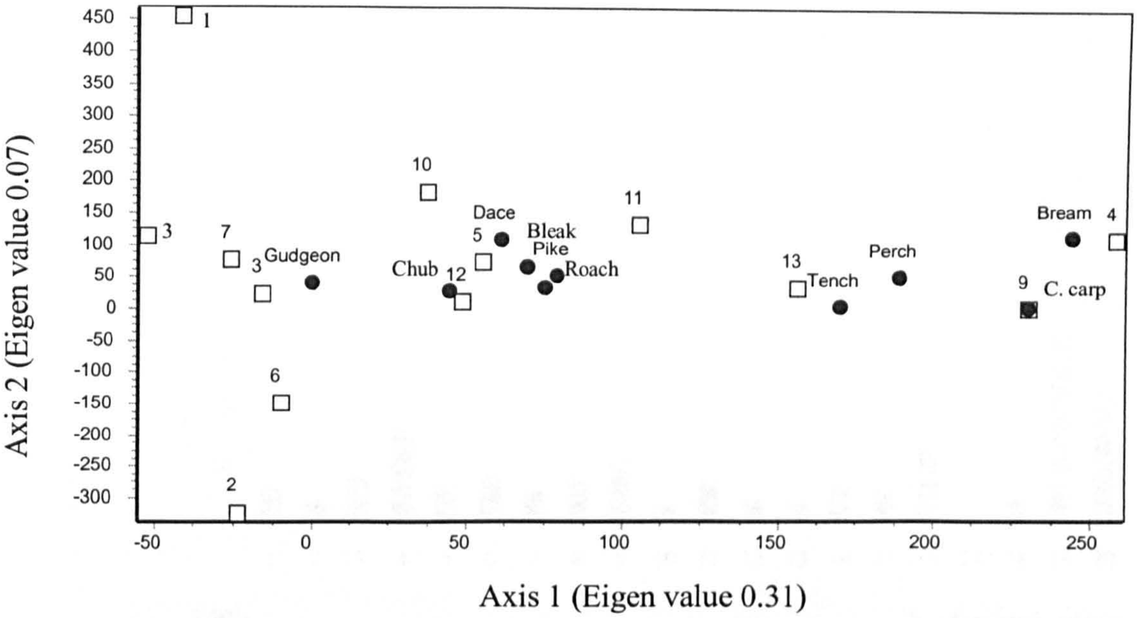


Fig. 4.6e Results of DECORANA analysis for the River Cherwell (site number as in Appendix 2.1)

The perturbation at Sor Brook Confluence (site 9) predicted by the ABC index was due to an imbalance in the fish community: it contained a large number of juvenile perch and roach, and a few large pike and common carp, consequently the abundance curve lay above the biomass curve in the analysis (Appendix 4.4). The fall off in quality of the fish communities in the lower sites was attributed to lack of instream cover and some river engineering activities.

River Evenlode

The River Evenlode supported 19 species including 5 minor species (Appendix 4.1) from 18 genera and 9 families (Appendix 4.2). Six coarse fish species, chub, dace, pike, perch, roach and gudgeon, and one game fish species, e.g. brown trout, were present in more than 50% sites (Fig. 4.4b). Fish species diversity was poor between Evenlode and Goose Eye Farm (sites 1 - 17). Diversity was high in the lower reaches between upstream of A40 and Mill Stream Cassington (sites 18 - 20) (Appendix 4.3). With few exceptions, fish density was low and varied from 0.70 to 33.50 fish 100m^{-2} with a mean of 8.97±8.09 fish 100m^{-2} (Fig. 4.7a). Nine species were recorded from Canal Stream Cassington (site 19) which supported the highest fish density.

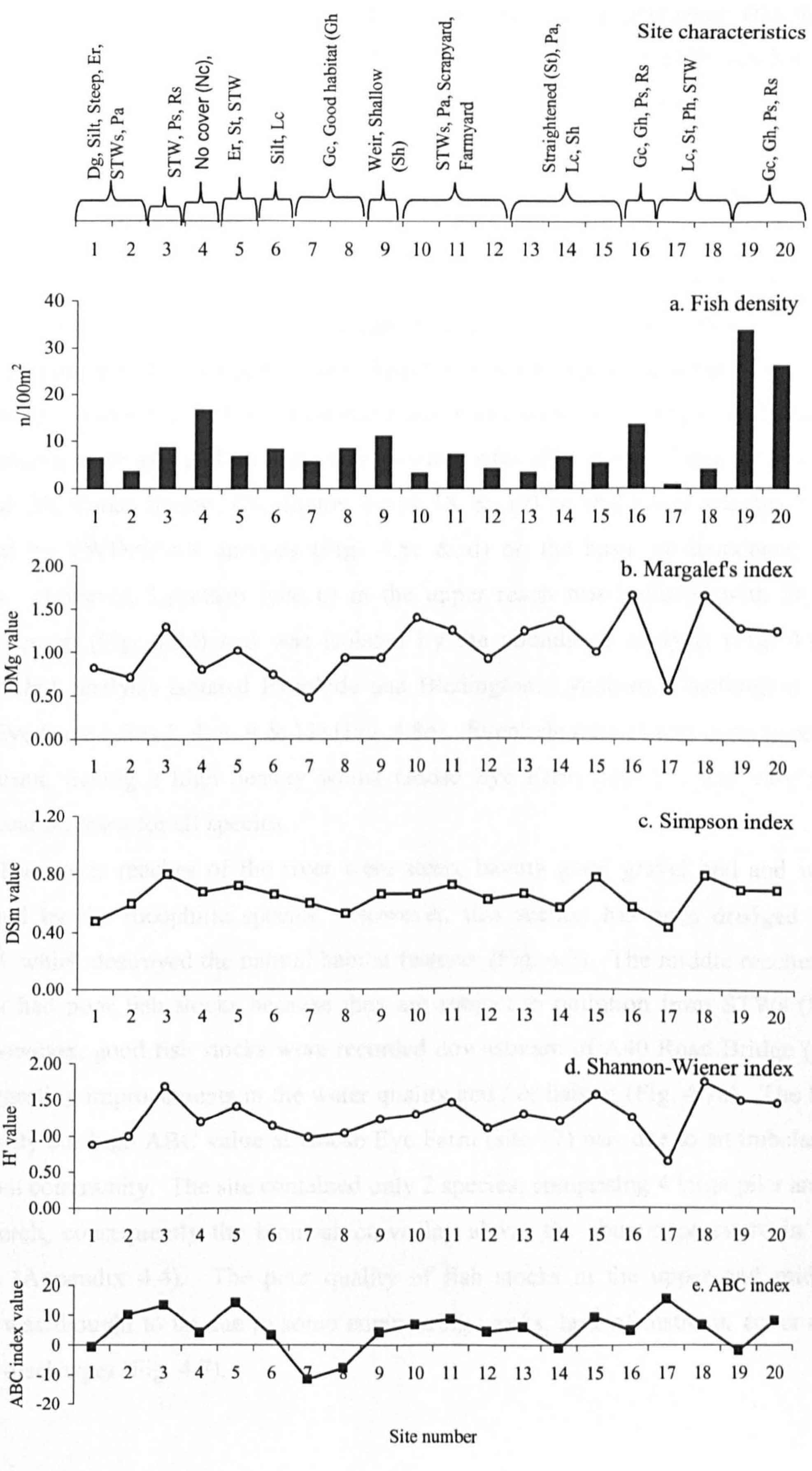


Fig. 4.7 Site characteristics and variations in different indices for the River Evenlode

Fish density was very low at Oddington (site 2) in the upper reaches, Charlbury, Ashford Mill and Lower Riding Farm (sites 10, 12 & 13) in the middle reaches and Goose Eye Farm and Upstream A40 (sites 17 & 18) in the lower reaches.

Various indices reflected these results on fish diversity, density and assemblage (Figs 4.7b, c, d & e). The ABC index separated Shipton-under-Wychwood and Ascott-under-Wychwood (sites 7 & 8) as poor as compared with the rest of the sites. Cluster analysis, based on fish density (Fig. 4.8a), grouped the sites with poor abundance in the upper reaches (sites 1 & 2) and isolated the extreme lowland reaches (sites 16 & 17). Similar groupings were obtained for the cluster analysis based on biomass. Evenlode and Oddington (sites 1 & 2) from the upper reaches and sites 16, 17, 18, 19 & 20 in the lower reaches were grouped with the low biomass sites (Fig. 4.8b). Upstream of the A40 and the Canal Stream Cassington (sites 18 & 19) in the lower reaches were separated by TWINSPAN analysis (Figs 4.8c & d) on the basis of abundance and biomass. However, Lyneham (site 6) in the upper reach was included with higher biomass group (Fig. 4.8d) and was isolated by the abundance analysis (Fig. 4.8c). DECORANA analysis isolated Evenlode and Bledington, Lyneham, Chadlington and Goose Eye Farm (sites 1, 4, 6, 9 & 17) (Fig. 4.8e). Evenlode (site 1) was dominated by brown trout, having a high density whilst Goose Eye Farm (site 17) had very low density and biomass for all species.

The upper reaches of the river were steep, having good gravel bed and were dominated by the rheophilic species. However, this section has been dredged and widened, which destroyed the natural habitat features (Fig. 4.7). The middle reaches of the river had poor fish stocks because they are subject to pollution from STWs (Fig. 4.7). However, good fish stocks were recorded downstream of A40 Road Bridge (site 18), suggesting improvements in the water quality and / or habitat (Fig. 4.7a). The low fish density but high ABC value at Goose Eye Farm (site 17) was due to an imbalance in the fish community. The site contained only 2 species, comprising 4 large pike and 2 small perch, consequently the biomass curve lay above the abundance curve in the analysis (Appendix 4.4). The poor quality of fish stocks in the upper and middle reaches was thought to be due to some engineering works, lack of instream cover and effluent discharges (Fig. 4.7).

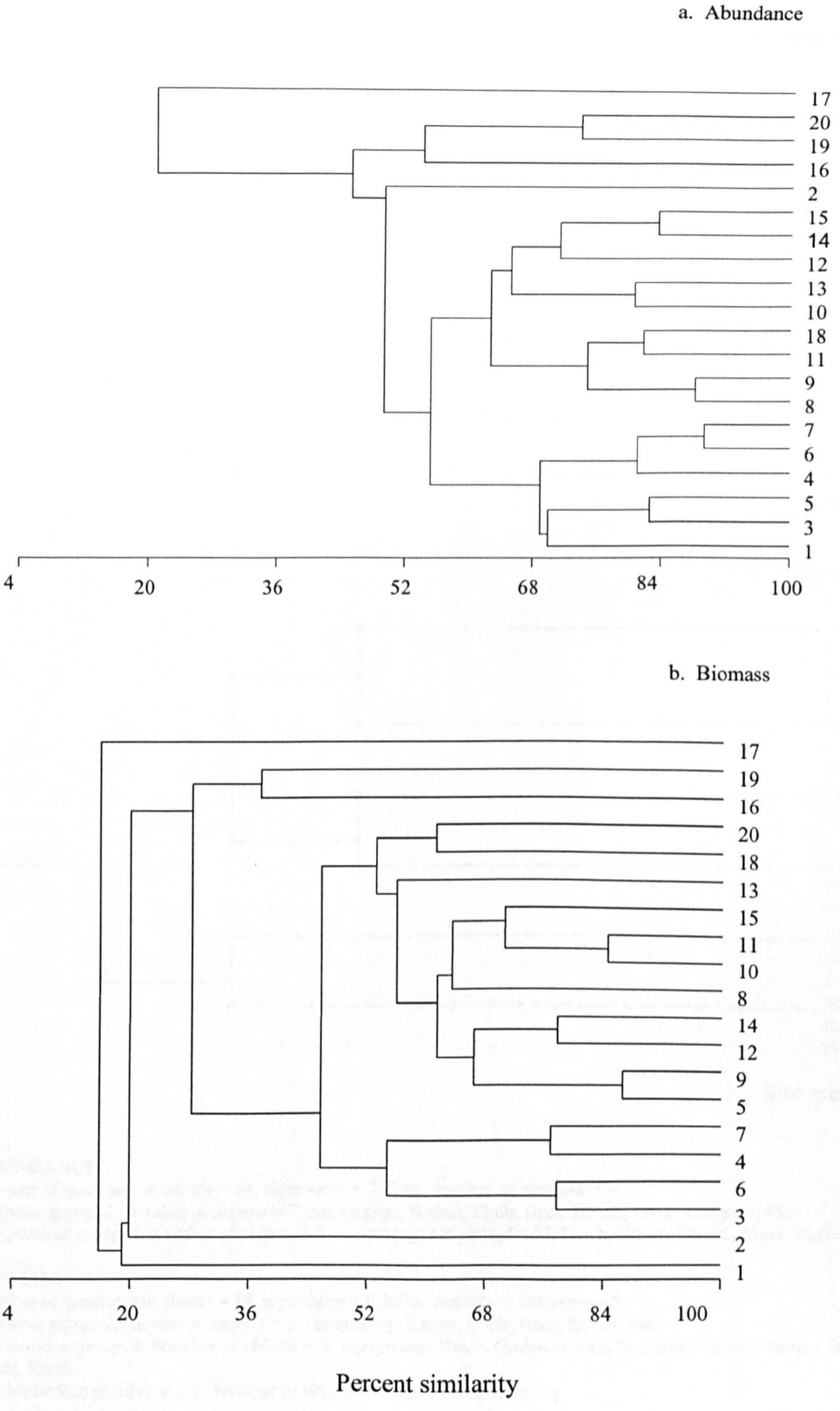
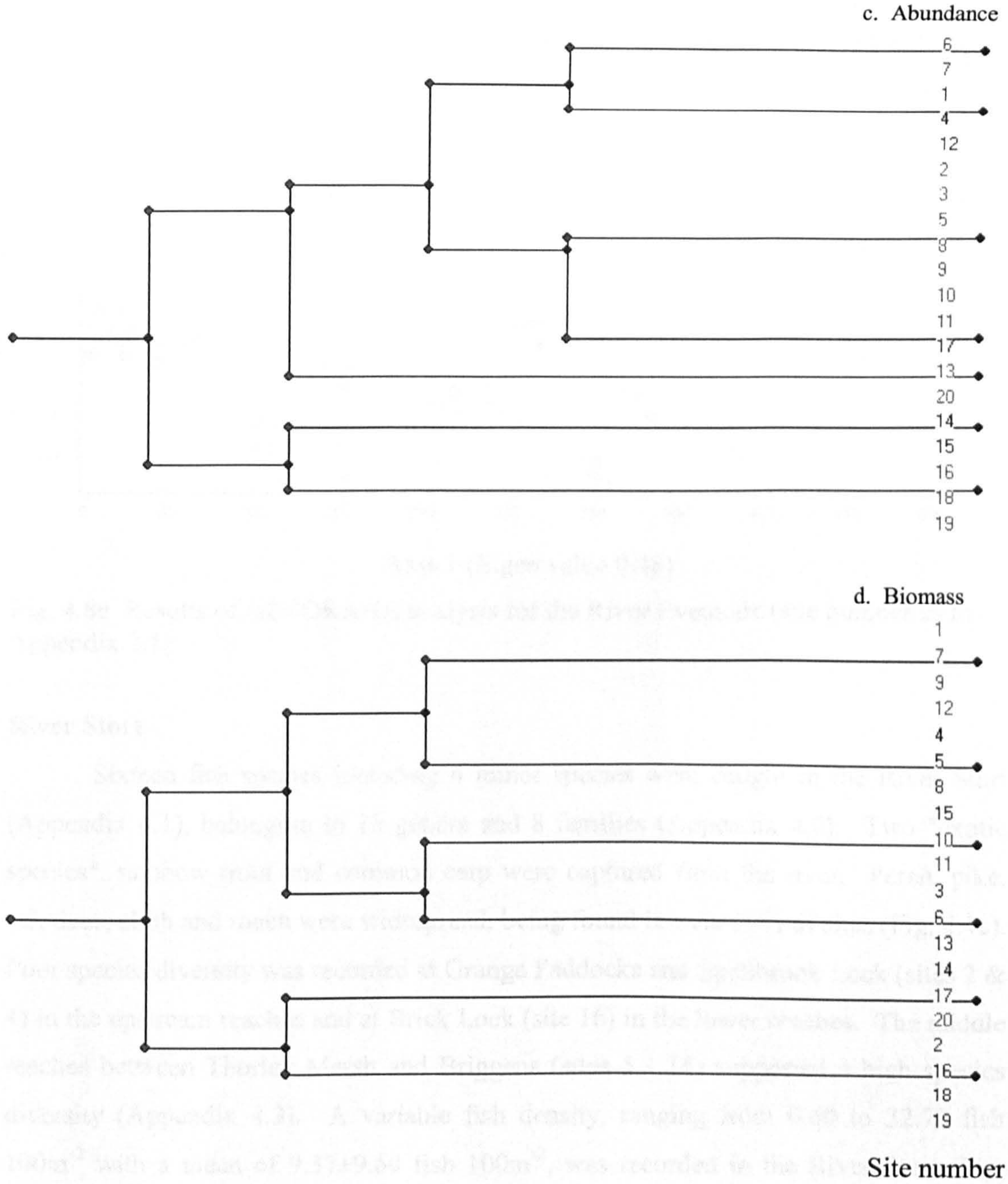


Fig. 4.8 Dendrograms based on UPGMA for the River Evenlode (site number as in Appendix 2.1)



ABUNDANCE
Number of quadrates in cluster = 14, eigenvalue = 0.3734, number of iterations = 4
Negative group: 2, Number of objects = 7 comprising:, B.trout, Chub, Dace, Roach, Perch, Gudgeon, Pike.
The positive group: 3, Number of objects = 7 comprising: Grayling, Rudd, Tench, Bream, Barbel, Bleak, Ruffe.

BIOMASS
Number of quadrates in cluster = 14, eigenvalue = 0.3676, number of iterations = 5
Negative group: 2, Number of objects = 5 comprising: B.trout, Chub, Dace, Roach, Pike.
The positive group: 3, Number of objects = 9 comprising: Perch, Gudgeon, Grayling, Rudd, Tench, Bream, Barbel, Bleak, Ruffe.
The borderline positive group: Number of objects = 1 comprising: Grayling

Fig. 4.8 (Continued) Results of TWINSPAN for the River Evenlode at one level of division (site number as in Appendix 2.1)

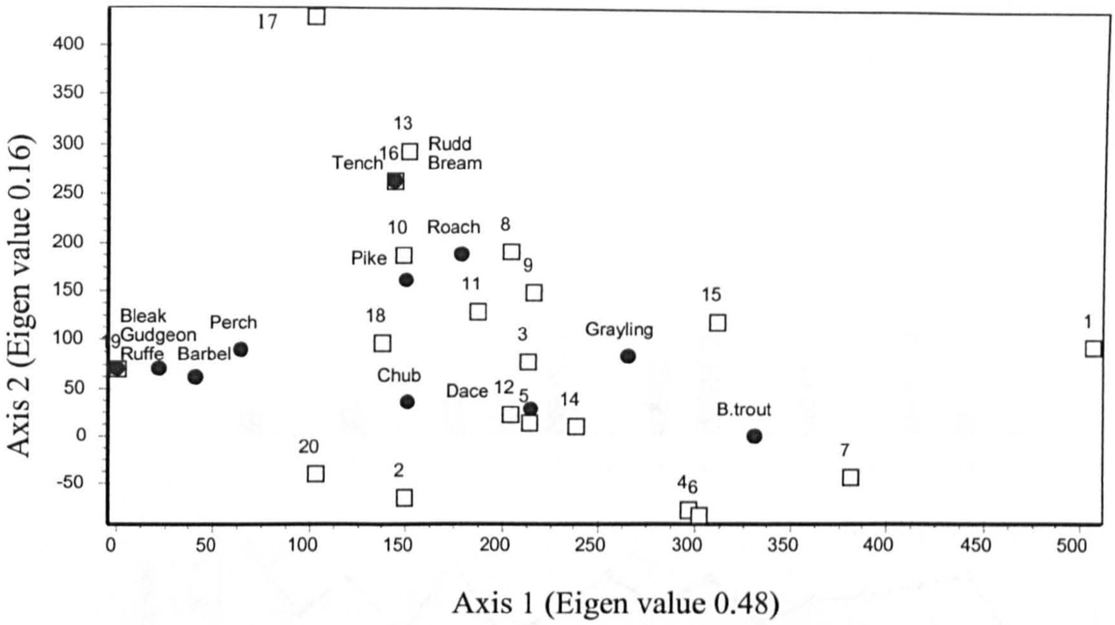


Fig. 4.8e Results of DECORANA analysis for the River Evenlode (site number as in Appendix 2.1)

River Stort

Sixteen fish species including 4 minor species were caught in the River Stort (Appendix 4.1), belonging to 15 genera and 8 families (Appendix 4.2). Two "exotic species", rainbow trout and common carp were captured from the river. Perch, pike, eel, dace, chub and roach were widespread, being found in over 50% of sites (Fig. 4.4c). Poor species diversity was recorded at Grange Paddocks and Spellbrook Lock (sites 2 & 4) in the upstream reaches and at Brick Lock (site 16) in the lower reaches. The middle reaches between Thorley Marsh and Briggens (sites 5 - 14) supported a high species diversity (Appendix 4.3). A variable fish density, ranging from 0.60 to 32.70 fish $100m^{-2}$ with a mean of 9.37±9.69 fish $100m^{-2}$, was recorded in the River Stort (Fig. 4.9a). Highest fish density was recorded at Briggens (site 14), based on eight species, but density was very low at Grange Paddocks and Spellbrook Lock (sites 2 & 4) in the upper reaches, at Sawbridgeworth and Harcamlow Way (sites 8 & 11) in the middle reaches and the A414 Harlow Road and Brick Lock (sites 13 & 16) in the lower reaches.

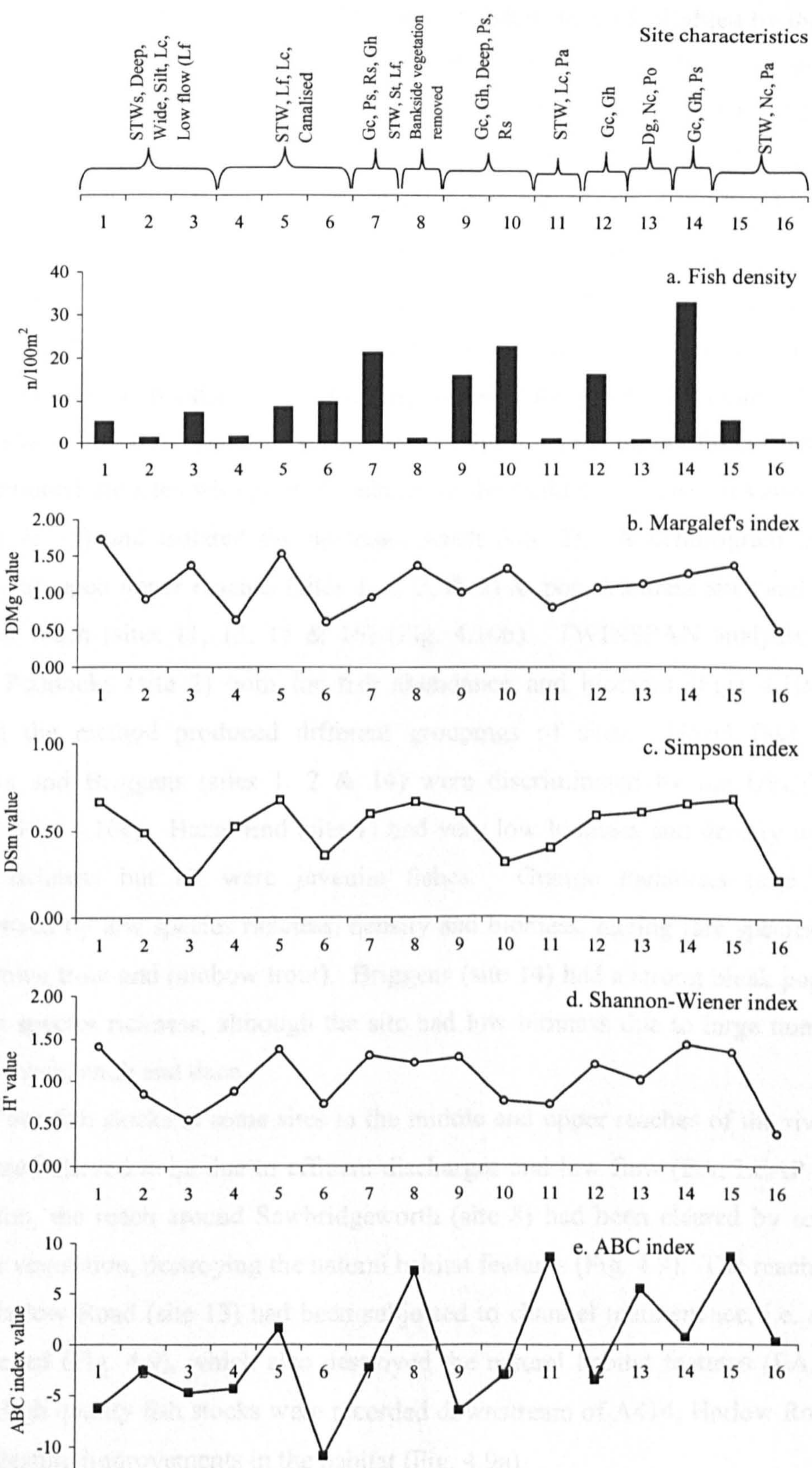
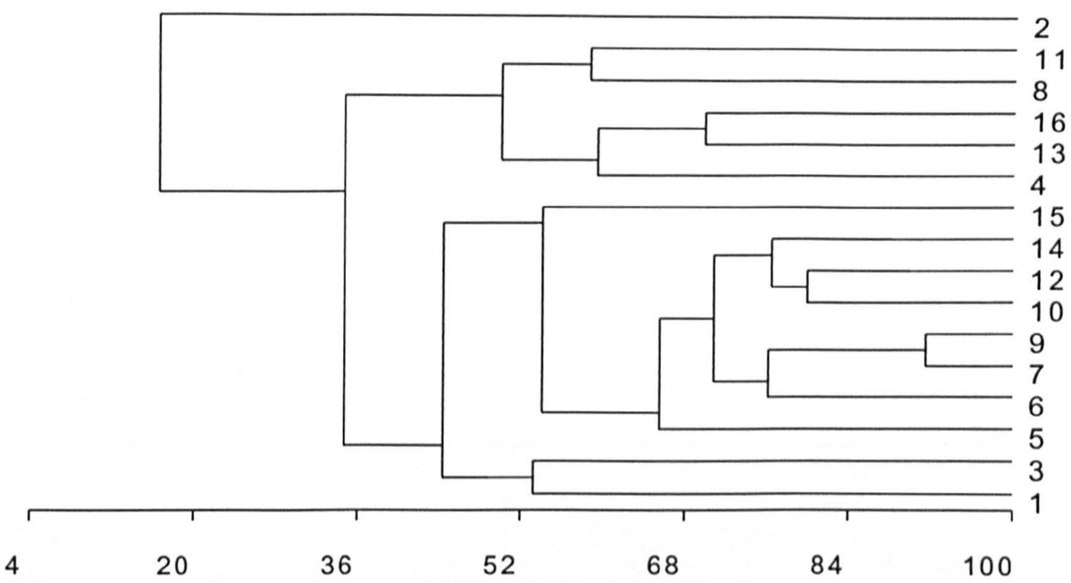


Fig. 4.9 Site characteristics and variations in different indices for the River Stort

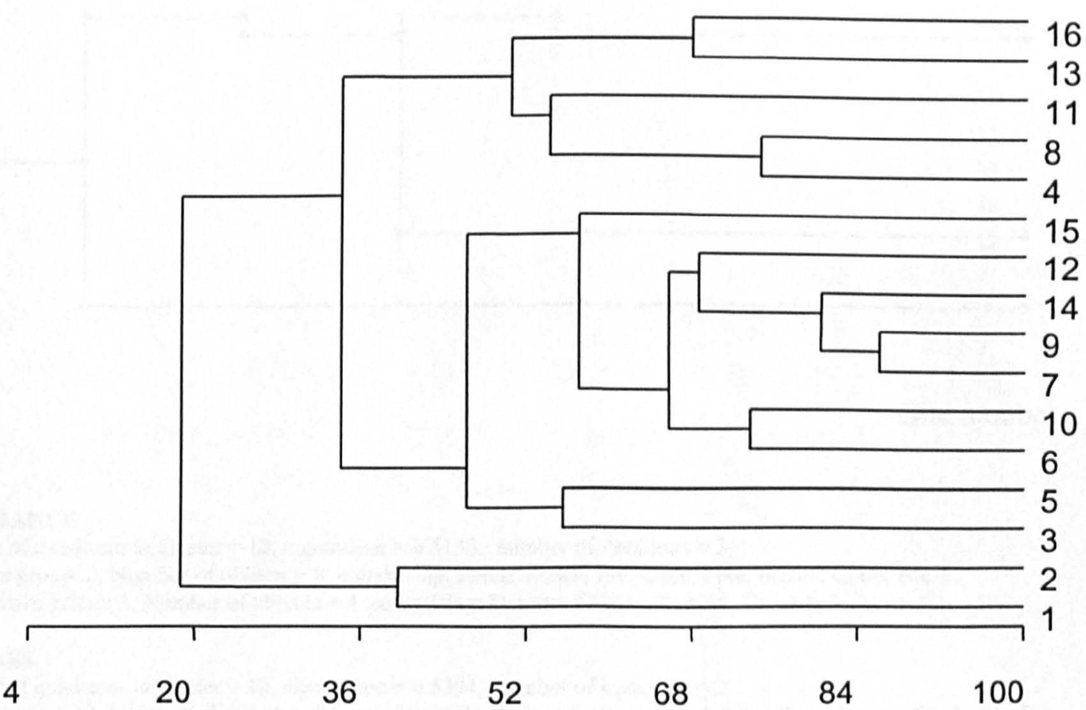
Variations in the diversity and density of fishes were highlighted by the various diversity indices (Figs 4.9b, c, d & e). The ABC index indicated the imbalance in the fish community in the upper (sites 1, 2, 3 & 4) and middle (sites 7, 9, 10 & 12) reaches of the river. The ABC index discriminated Briggens (site 14) as poor, although the site contained the highest fish density. This high fish density but low ABC index value at Briggens (site 14) was due to a high number of juveniles, having low biomass. Consequently, the biomass curve was close to the abundance curve in the analysis (Appendix 4.4). The ABC index was high but fish density was low at Sawbridgeworth (site 8), due to low abundance and biomass, consequently the biomass curve lay above the abundance curve (Appendix 4.4). Cluster analysis, on the basis of fish density (Fig. 4.10a) grouped the sites with poor abundance in the middle and lower reaches (1, 3, 4, 5, 8, 11 & 15) and isolated the upstream reach (site 2). A dendrogram based on biomass, grouped upper reaches (sites 1, 2, 3, & 5) as poor biomass sites and isolated the lower reach (sites 11, 13, 15 & 16) (Fig. 4.10b). TWINSPAN analysis isolated Grange Paddocks (site 2) both for fish abundance and biomass (Figs 4.10c & d), although the method produced different groupings of sites. Hazel End, Grange Paddocks and Briggens (sites 1, 2 & 14) were discriminated by the DECORANA analysis (Fig. 4.10e). Hazel End (site 1) had very low biomass and density with high species richness but all were juvenile fishes. Grange Paddocks (site 2) was characterised by low species richness, density and biomass, having rare species for the river (brown trout and rainbow trout). Briggens (site 14) had a strong bleak population and high species richness, although the site had low biomass due to large numbers of juvenile roach, chub and dace.

Poor fish stocks at some sites in the middle and upper reaches of the river (Fig. 4.9a) were believed to be due to effluent discharges and low flow (EA, LEAP 1999e). In addition, the reach around Sawbridgeworth (site 8) had been cleared by removing bankside vegetation, destroying the natural habitat features (Fig. 4.9). The reach around A414, Harlow Road (site 13) had been subjected to channel maintenance, i.e. dredged and widened (Fig. 4.9), which also destroyed the natural habitat features (EA, LEAP 2001). High quality fish stocks were recorded downstream of A414, Harlow Road (site 13), suggesting improvements in the habitat (Fig. 4.9a).

a. Abundance

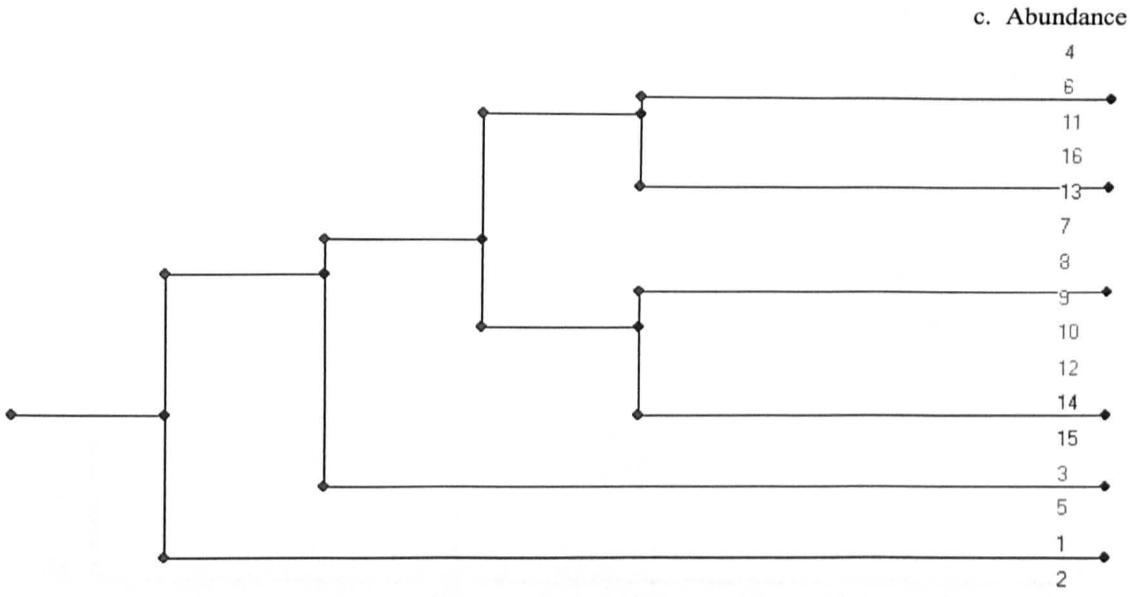


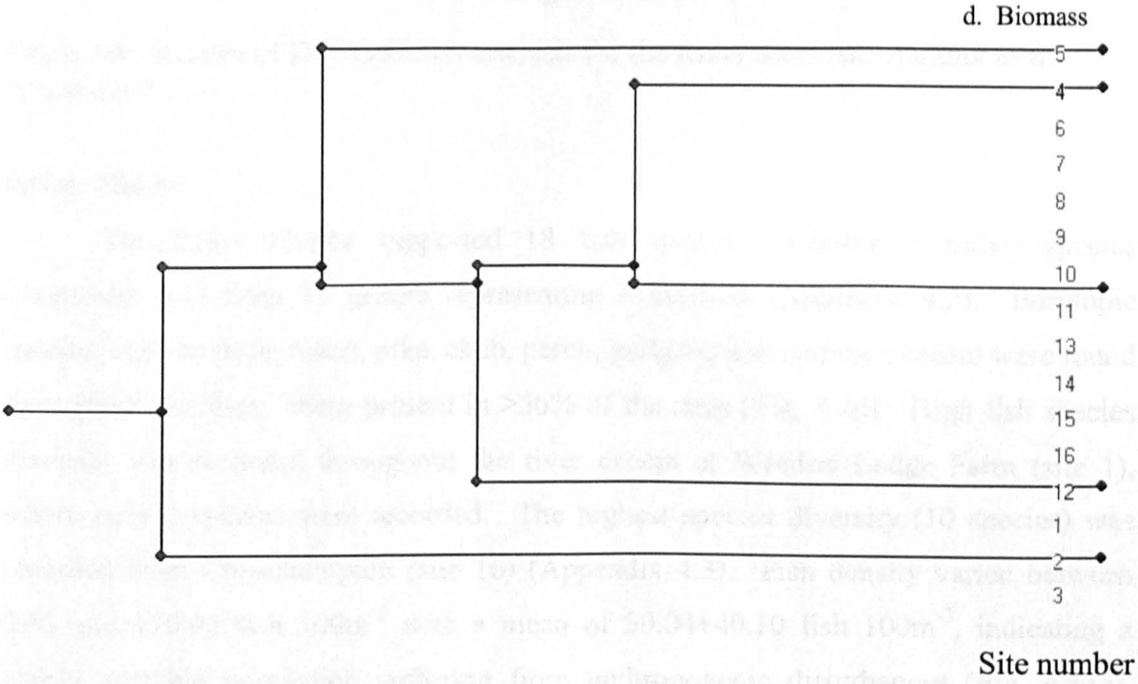
b. Biomass



Percent similarity

Fig. 4.10 Dendrograms based on UPGMA for the River Stort (site number as in Appendix 2.1)





ABUNDANCE
Number of quadrates in cluster = 12, eigenvalue = 0.5135, number of iterations = 3
Negative group: 2, Number of objects = 8 comprising: Perch, Roach, Eel, Dace, Pike, Bream, Chub, Bleak.
The positive group: 3, Number of objects = 4 comprising: B. trout, C. carp, R. trout, Cr. carp.

BIOMASS
Number of quadrates in cluster = 12, eigenvalue = 0.5394, number of iterations = 3
Negative group: 2, Number of objects = 9 comprising: Perch, Roach, C. carp, Eel, Pike, Dace, Bream, Chub, Bleak.
The positive group: 3, Number of objects = 3 comprising: B. trout, R. trout, Cr. carp.

Fig. 4.10 (Continued) Results of TWINSPAN for the River Stort at one level of division (site number as in Appendix 2.1)

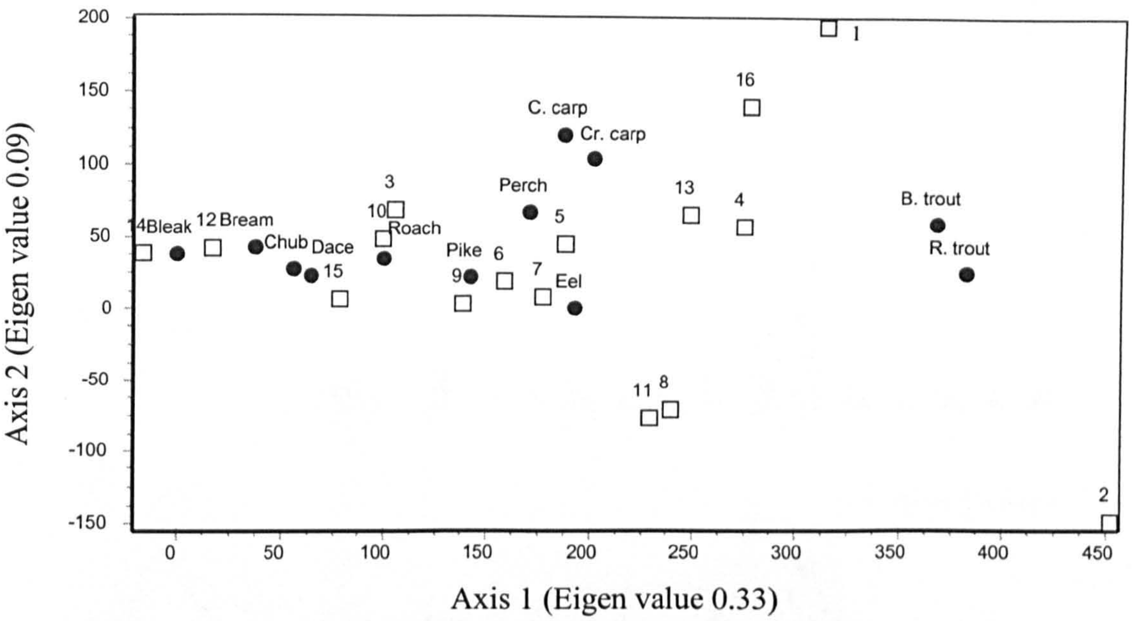


Fig. 4.10e Results of DECORANA analysis for the River Stort (site number as in Appendix 2.1)

River Thame

The River Thame supported 18 fish species including 5 minor species (Appendix 4.1) from 17 genera representing 9 families (Appendix 4.2). Eurytopic species, such as dace, roach, pike, chub, perch, gudgeon and common bream were found throughout the river, being present in >56% of the sites (Fig. 4.4d). High fish species diversity was recorded throughout the river except at Weedon Lodge Farm (site 1), where only 2 species were recorded. The highest species diversity (10 species) was recorded from Chiselhampton (site 16) (Appendix 4.3). Fish density varied between 0.40 and 170.90 fish $100m^{-2}$ with a mean of 30.04±40.10 fish $100m^{-2}$, indicating a highly unstable population suffering from anthropogenic disturbances (Fig. 4.11a). Highest fish density was recorded from Cuddesdon (site 13), where nine species were recorded but density was low at Weedon and Lower Hartwell (sites 1 & 4) in the upper reaches and Shabbington West Arm and Shabbington East Arm (sites 9 & 10) in the middle reaches.

These observations on the diversity, density and assemblage were supported by the various diversity indices (Figs 4.11b, c, d & e). Downstream of the A41 Bridge, Lower Hartwell and Eythrope (sites 2, 3 & 4) in the upper reaches and Waterstock, Cuddesdon and Coddesdon Mill Channel (sites 12, 13 &14) in the lower reaches were identified as poor by the ABC index (Fig. 4.11e).

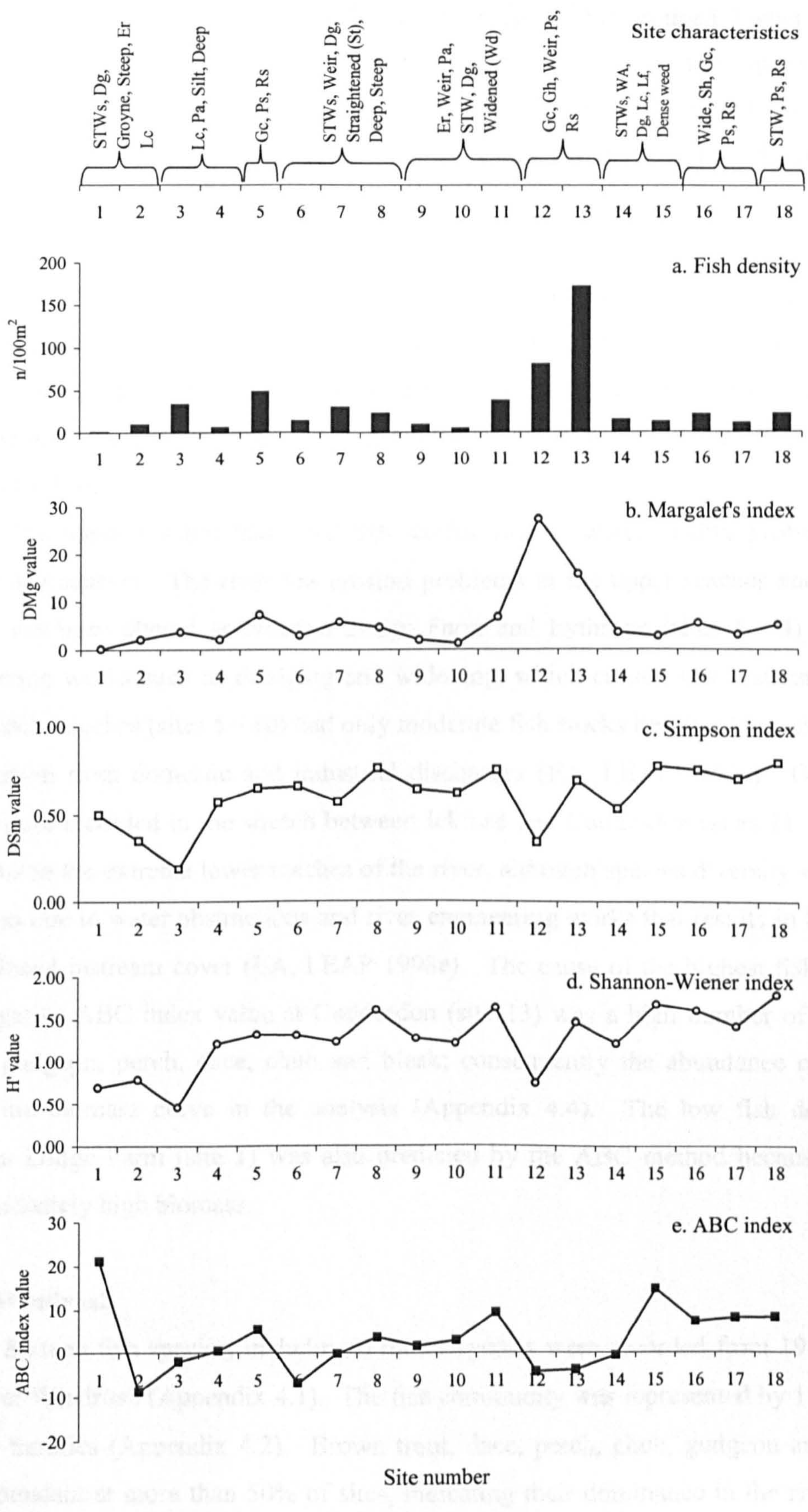


Fig. 4.11 Site characteristics and variations in different indices for the River Thame

Cluster analysis, based on fish biomass (Fig. 4.12a) grouped Lower Hartwell and Eythrope (sites 3 & 4) with high biomass sites and isolated the most upstream reach (site 1) as poor. Eythrope (site 4) was separated by the TWINSPAN analysis, although the site was grouped with the lower reaches (sites 16 & 18) (Fig. 4.12b). The DECORANA analysis based on biomass isolated Lower Hartwell, Nether Winchendon, U/S Notley Abbey, Shabbington East Arm and Dorchester (sites 3, 6, 7, 10 & 18) from the remaining sites (Fig. 4.12c). Lower Hartwell (site 3) was characterised by moderate species richness and low biomass due to significant numbers of juvenile roach while Nether Winchendon (site 6) was isolated due to low biomass and density. Dorchester (site 18) had low density (Fig. 4.11a) and biomass but had a strong barbel population (Appendix 4.3).

The upper reaches had poor fish stocks, due to water quality problems and habitat degradation. The river has erosion problems in the upper reaches and natural habitat has been altered at Weedon Lodge Farm and Eythrope (sites 1 - 4) by river engineering works such as dredging and widening, which caused low instream cover. The middle reaches (sites 5 - 10) had only moderate fish stocks because they are subject to pollution from domestic and industrial discharges (EA, LEAP 1997a). Good fish stocks were recorded in the stretch between Ickford and Cuddesdon (sites 11 - 13) but were low in the extreme lower reaches of the river, although species diversity was high. This was due to water abstractions and river engineering works that results in low flow and reduced instream cover (EA, LEAP 1998e). The cause of the highest fish density and negative ABC index value at Cuddesdon (site 13) was a high number of juvenile roach, gudgeon, perch, dace, chub and bleak; consequently the abundance curve lay above the biomass curve in the analysis (Appendix 4.4). The low fish density at Weedon Lodge Farm (site 1) was also predicted by the ABC method because of the proportionately high biomass.

River Windrush

Sixteen fish species, including 5 minor species were recorded from 19 sites on the River Windrush (Appendix 4.1). The fish community was represented by 15 genera and 10 families (Appendix 4.2). Brown trout, dace, perch, chub, gudgeon and roach were abundant at more than 50% of sites, indicating their dominance in the river (Fig. 4.4e). Very poor fish diversity was observed upstream between Kineton and Upstream of A429 (sites 1 - 4), but density was high in the middle and lower reaches between D/S Dikler Confluence and Standlake STW (sites 5 - 19) (Appendix 4.3).

a. Biomass

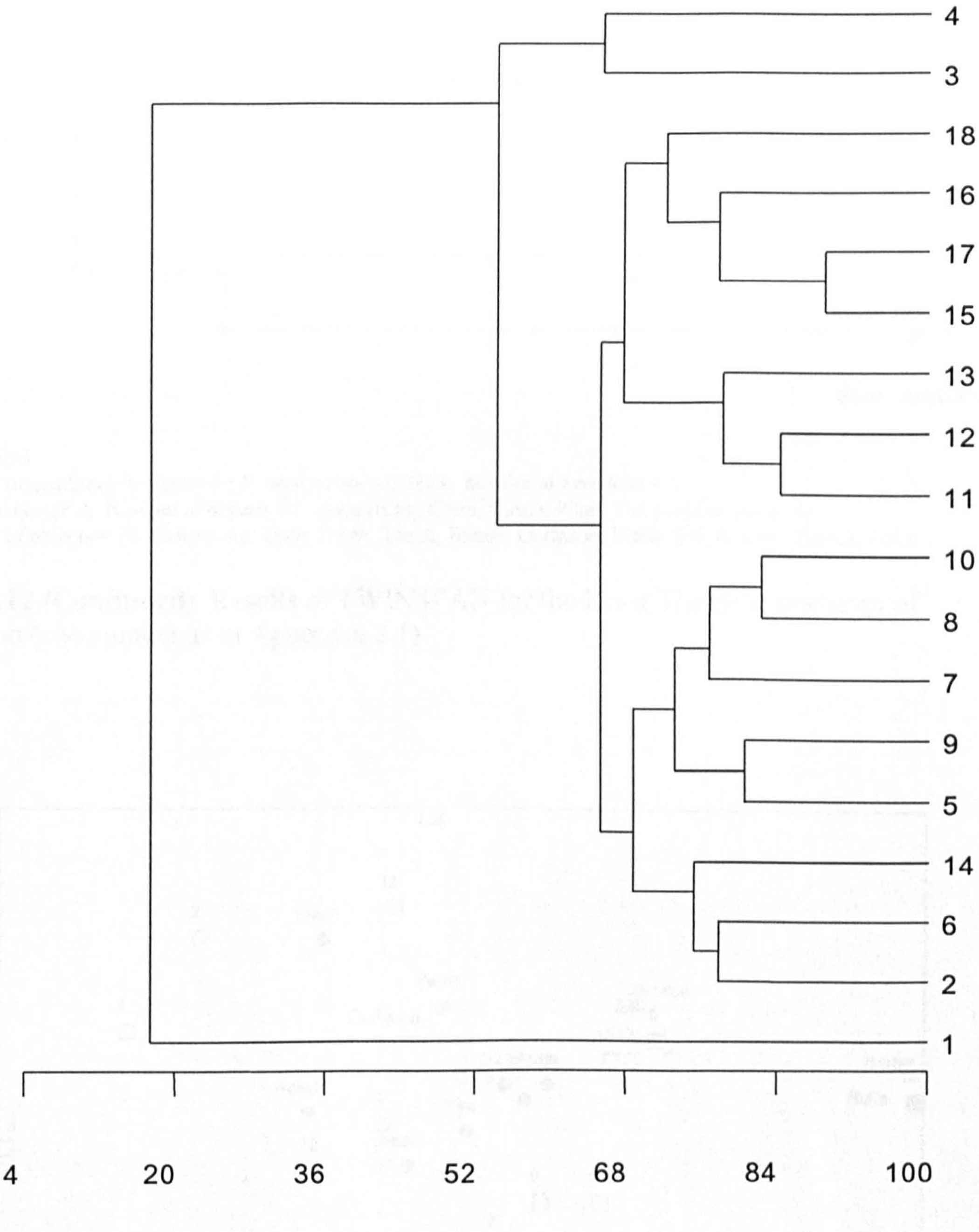
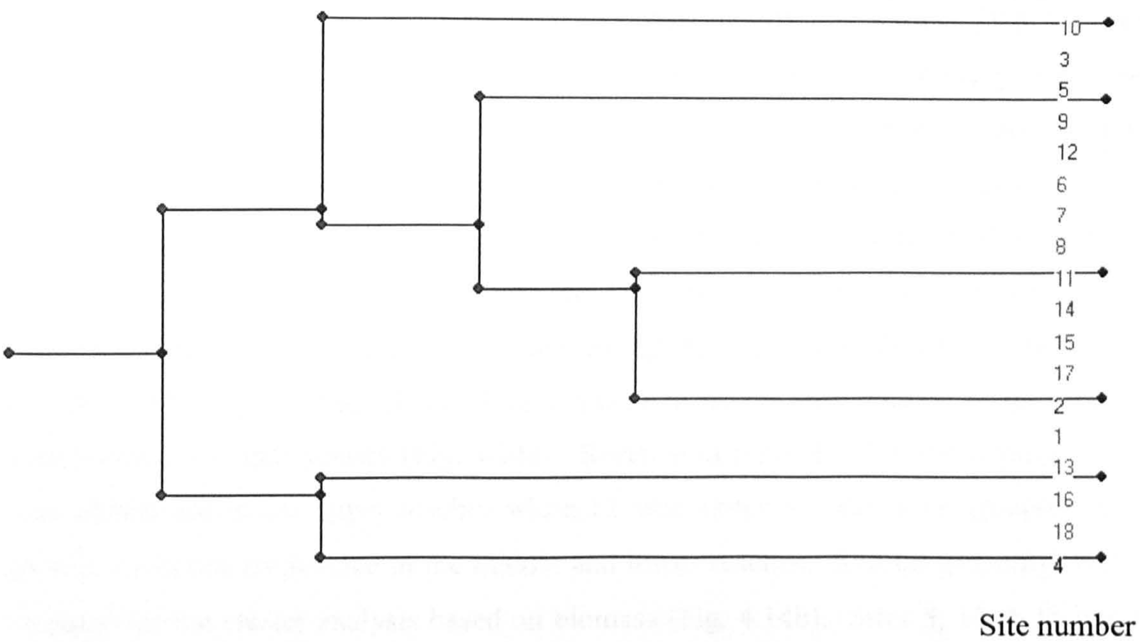


Fig. 4.12 Dendrograms based on UPGMA for the River Thame (site number as in Appendix 2.1)



BIOMASS
Number of quadrates in cluster = 13, eigenvalue = 0.3323, number of iterations = 6
Negative group: 2, Number of objects = 3 comprising: Chub, Roach, Pike. The positive group: 3,
Number of objects = 10 comprising: Dace, Perch, Tench, Bream, Gudgeon, Bleak, Eel, B. trout, Barbel, Ruffe.

Fig. 4.12 (Continued) Results of TWINSPAN for the River Thame at one level of division (site number as in Appendix 2.1)

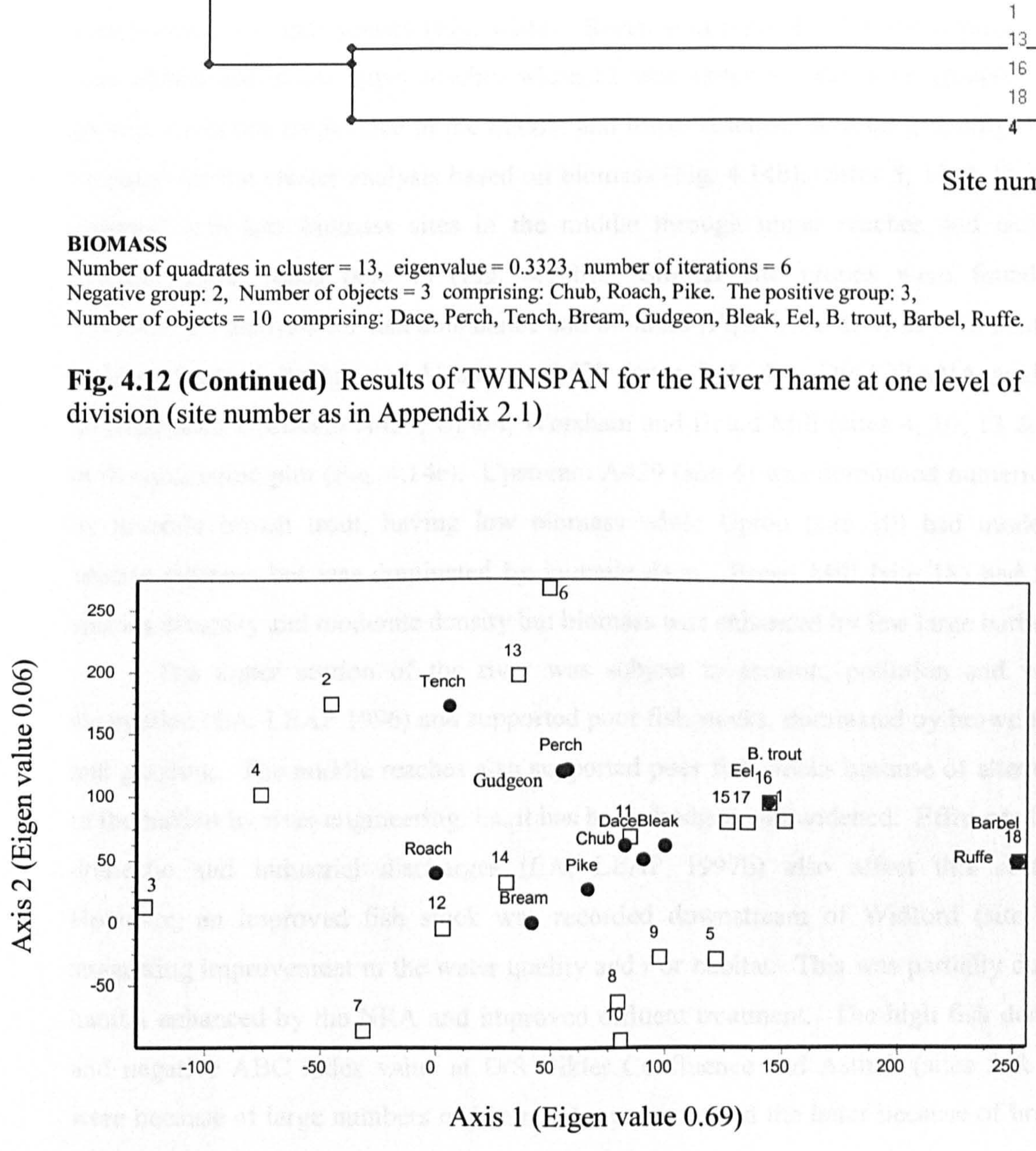


Fig. 4.12c Results of DECORANA analysis for the River Thame (site number as in Appendix 2.1)

Brook lamprey was observed in the river (Appendix 4.1). Fish density was low to moderate and varied between 3.30 and 21.40 fish $100m^{-2}$ with a mean of 9.74 ± 5.49 fish $100m^{-2}$, indicating a fluctuating fish stock (Fig 4.13a). Fish density was low between Sherborne Common and Widford (sites 7 - 11) in the middle reaches and the highest fish density was at Asthall (site 12), where seven species were recorded.

The patterns in fish diversity, density and assemblage were supported by various indices (Figs 4.13b, c, d & e). However, the ABC index discriminated U/S A429, D/S Dikler Confluence, Barrington Park, Little Barrington, Asthall and Worsham (sites 4, 5, 8, 9, 12 & 14) as poor (Fig. 4.13e). Two distinct site groups were formed by the cluster analysis based on fish density (Fig. 4.14a). Seven sites (sites 1 - 7) were grouped with poor abundance in the upper reaches while 12 sites (sites 8 - 19) were grouped with good to moderate abundance in the middle and lower reaches. Similar groupings were obtained for the cluster analysis based on biomass (Fig. 4.14b). Sites 5, 10 & 14 were grouped with low biomass sites in the middle through upper reaches and isolated extreme upper reach (site 1) (Fig. 4.14b). Similar site groups were found by TWINSPAN analysis for fish abundance and biomass (Figs 4.14c & d) as both analysis isolated Harford Bridge and Upstream A429 (sites 3 & 4). DECORANA analysis discriminated Upstream A429, Upton, Worsham and Bread Mill (sites 4, 10, 13 & 18) in the ordination plot (Fig. 4.14e). Upstream A429 (site 4) was dominated numerically by juvenile brown trout, having low biomass while Upton (site 10) had moderate species richness but was dominated by juvenile dace. Bread Mill (site 18) had high species diversity and moderate density but biomass was enhanced by few large barbel.

The upper section of the river was subject to erosion, pollution and water abstraction (EA, LEAP 1996) and supported poor fish stocks, dominated by brown trout and grayling. The middle reaches also supported poor fish stocks because of alteration in the habitat by river engineering, i.e. it has been dredged and widened. Effluents from domestic and industrial discharges (EA, LEAP 1997b) also affect this section. However, an improved fish stock was recorded downstream of Widford (site 11), suggesting improvement in the water quality and / or habitat. This was partially due to habitat enhanced by the NRA and improved effluent treatment. The high fish density and negative ABC index value at D/S Dikler Confluence and Asthall (sites 5 & 12) were because of large numbers of juvenile brown trout and the latter because of brown trout, gudgeon, chub and dace. As a result, the abundance curve lay above the biomass curve in the analysis (Appendix 4.4). On the other hand, the highest index value, but moderate fish density, was found at New Mill (site 15).

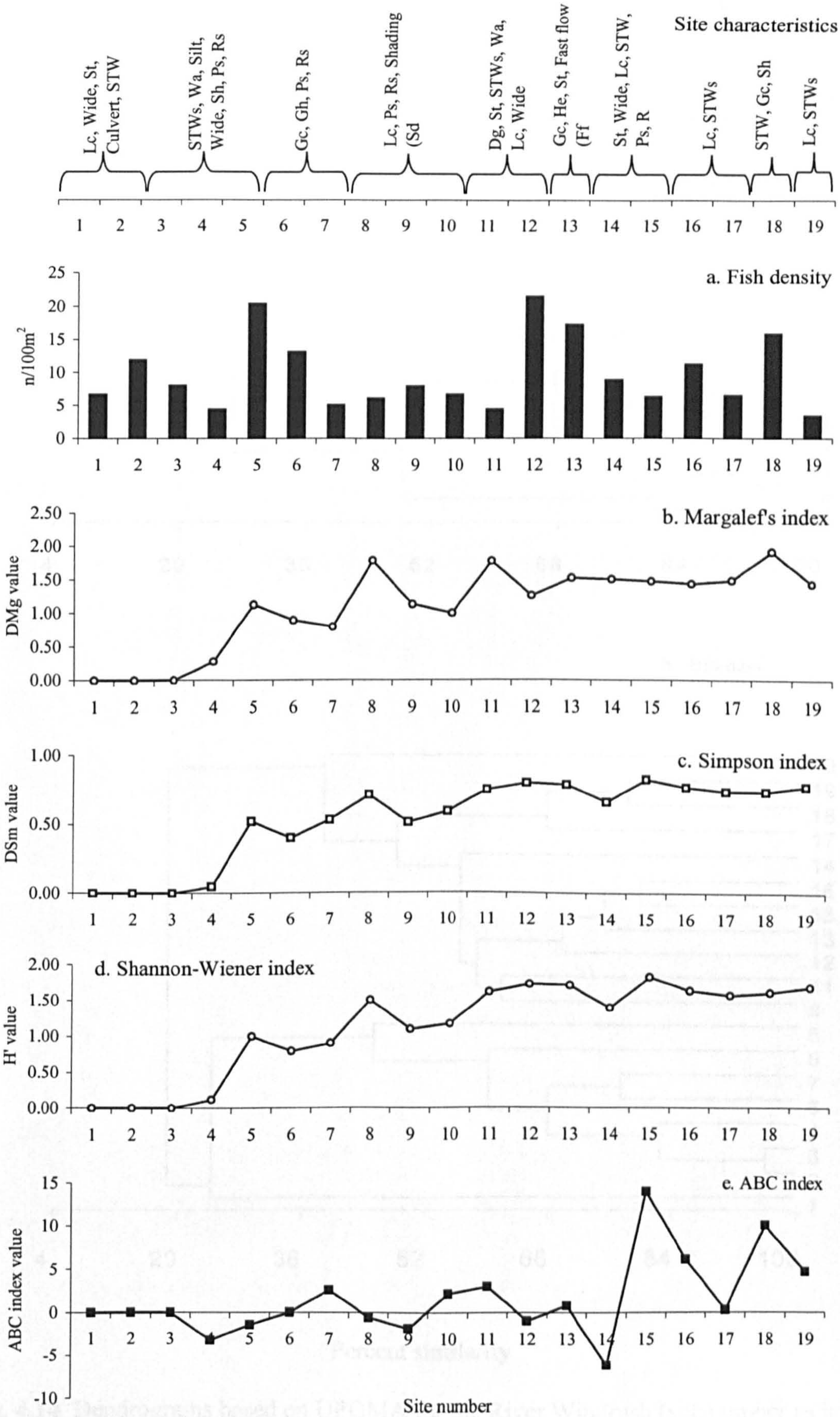
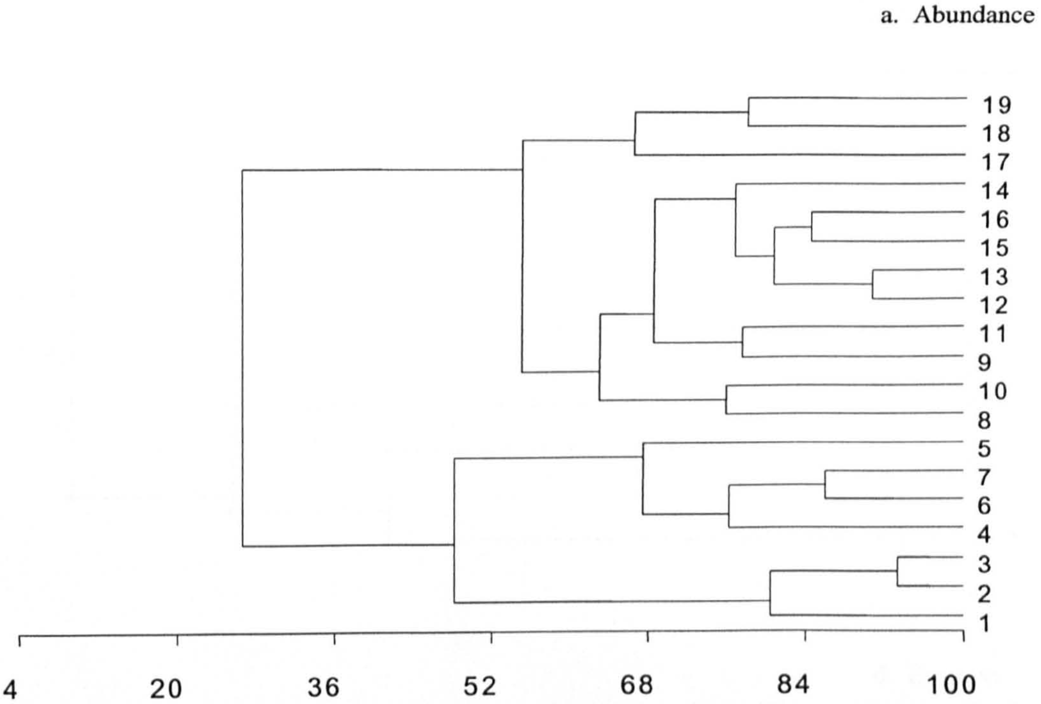


Fig. 4.13 Site characteristics and variations in different indices for the River Windrush



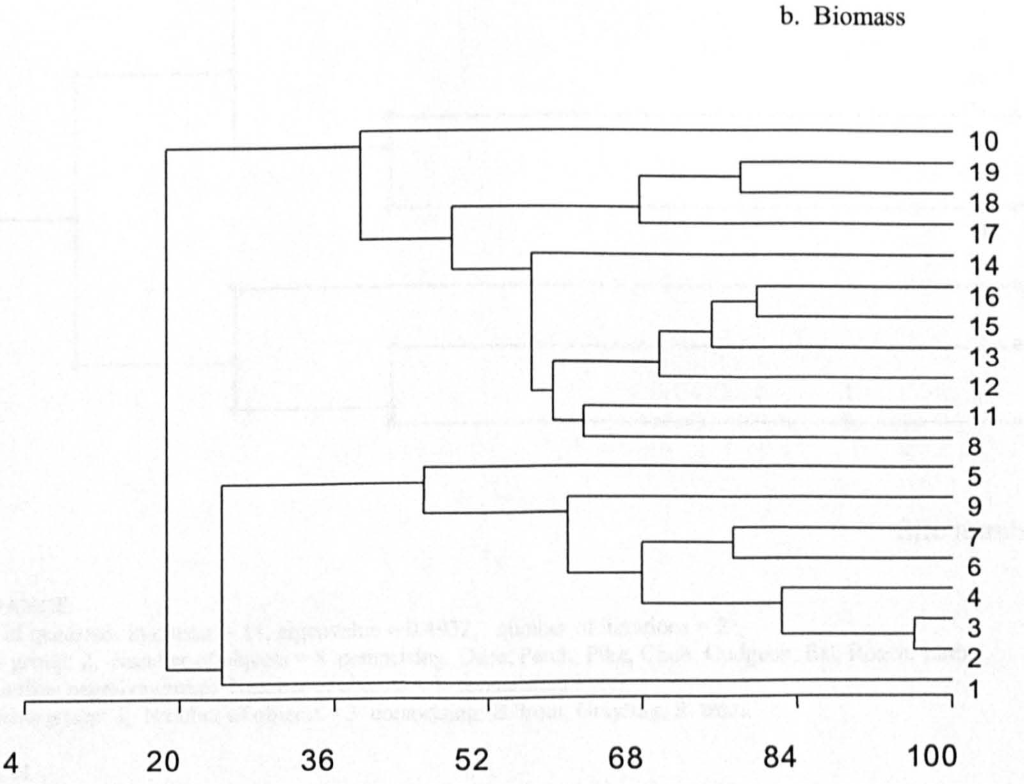
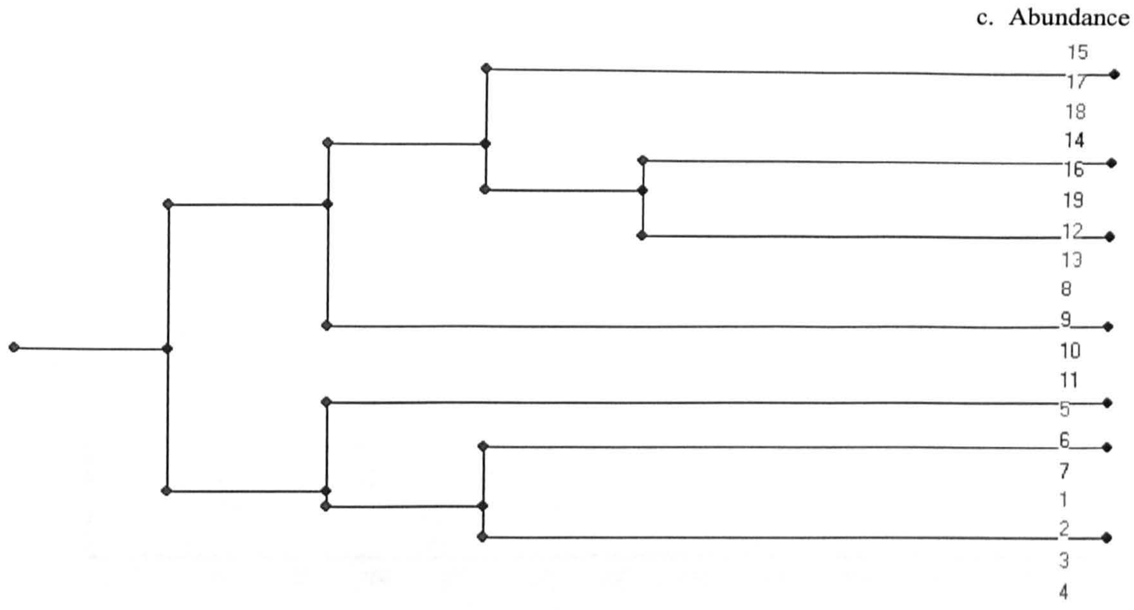
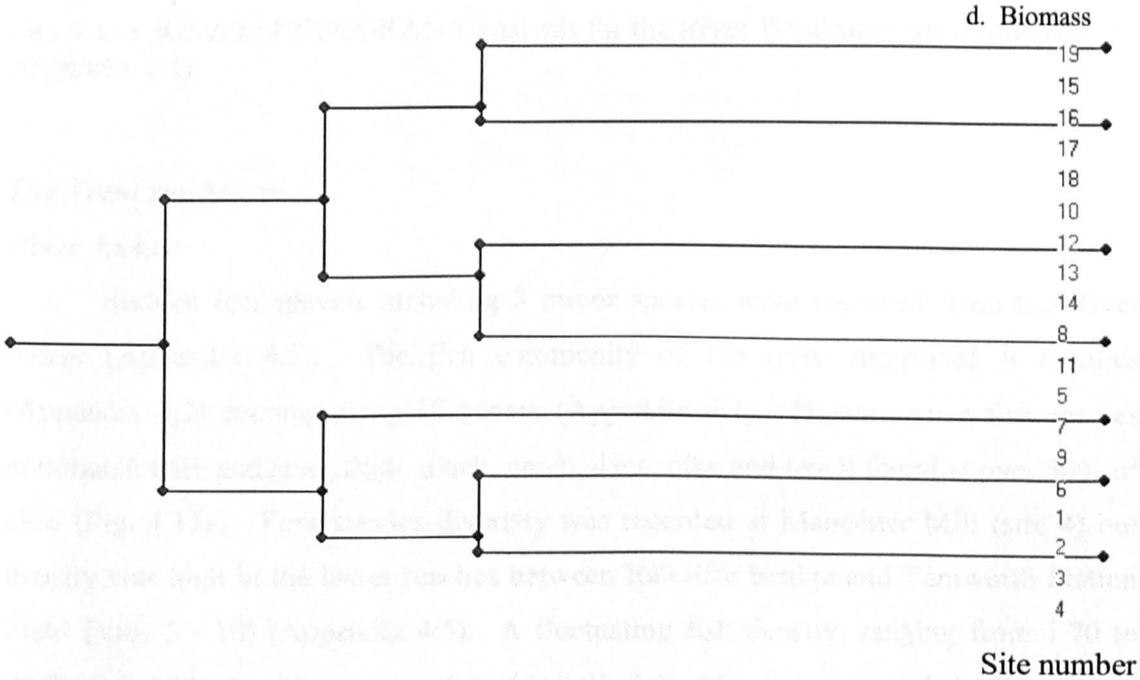


Fig. 4.14 Dendrograms based on UPGMA for the River Windrush (site number as in Appendix 2.1)





ABUNDANCE
Number of quadrates in cluster = 11, eigenvalue = 0.4932, number of iterations = 2
Negative group: 2, Number of objects = 8 comprising: Dace, Perch, Pike, Chub, Gudgeon, Eel, Roach, Barbel.
The borderline negative group: Number of objects = 1 comprising: Perch.
The positive group: 3, Number of objects = 3 comprising: B. trout, Grayling, R. trout.

BIOMASS
Number of quadrates in cluster = 11, eigenvalue = 0.5191, number of iterations = 2
Negative group: 2, Number of objects = 8 comprising: Dace, Pike, Perch, Chub, Gudgeon, Eel, Roach, Barbel.
The positive group: 3, Number of objects = 3 comprising: B. trout, Grayling, R. trout.

Fig. 4.14 (Continued) Results of TWINSPAN for the River Windrush at one level of division (site number as in Appendix 2.1)

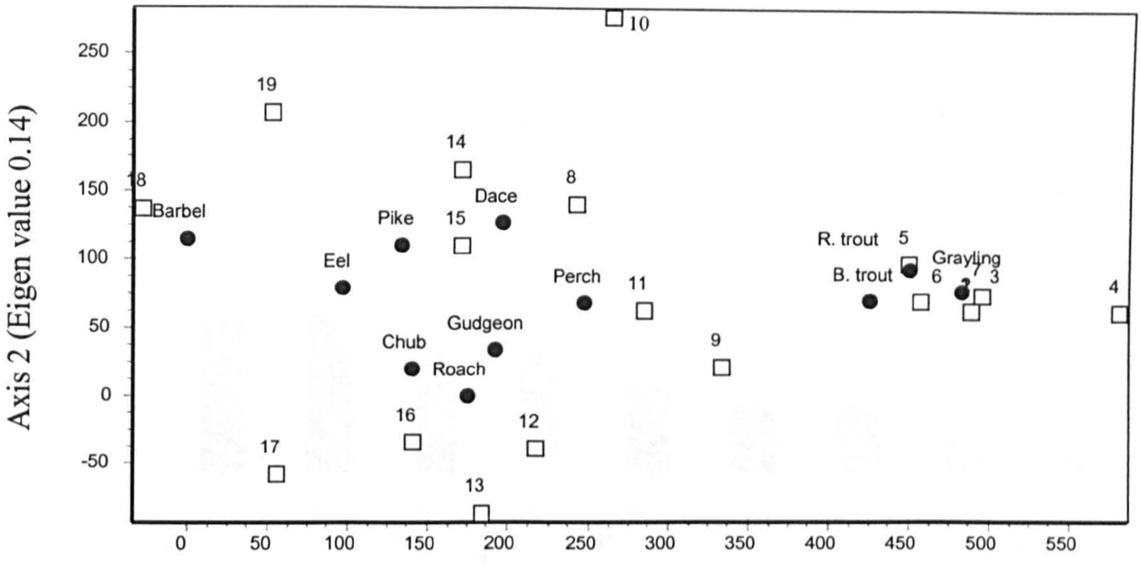


Fig. 4.14e Results of DECORANA analysis for the River Windrush (site number as in Appendix 2.1)

The Trent catchment

River Anker

Sixteen fish species including 3 minor species were recorded from the River Anker (Appendix 4.1). The fish community of the river comprised 6 families (Appendix 4.2) encompassing 15 genera (Appendix 4.1). Native coarse fish species dominated with gudgeon, chub, roach, perch, dace, pike and tench found at over 50% of sites (Fig. 4.15a). Poor species diversity was recorded at Mancetter Mill (site 4) but density was high in the lower reaches between Ratcliffe Bridge and Tamworth Station Field (sites 5 - 10) (Appendix 4.5). A fluctuating fish density, ranging from 1.70 to 33.90 fish $100m^{-2}$ with a mean of 10.12±9.05 fish $100m^{-2}$ was recorded (Fig. 4.16a). Highest fish density was at Ratcliffe Bridge (site 5), where 7 species were recorded but density was very low at U/S Tamworth Cowells Farm (site 9) in the lower reaches and Mancetter Mill (site 4) in the upper reaches.

The characteristics of fish stocks based on diversity and density were supported by various diversity indices (Figs 4.16b, c, d & e). The ABC index identified Woodford Bridge, Ratcliffe Bridge, Fieldon Bridge, Polesworth-1 and Polesworth-2 (sites 3, 5, 6, 7 & 8) as poor.

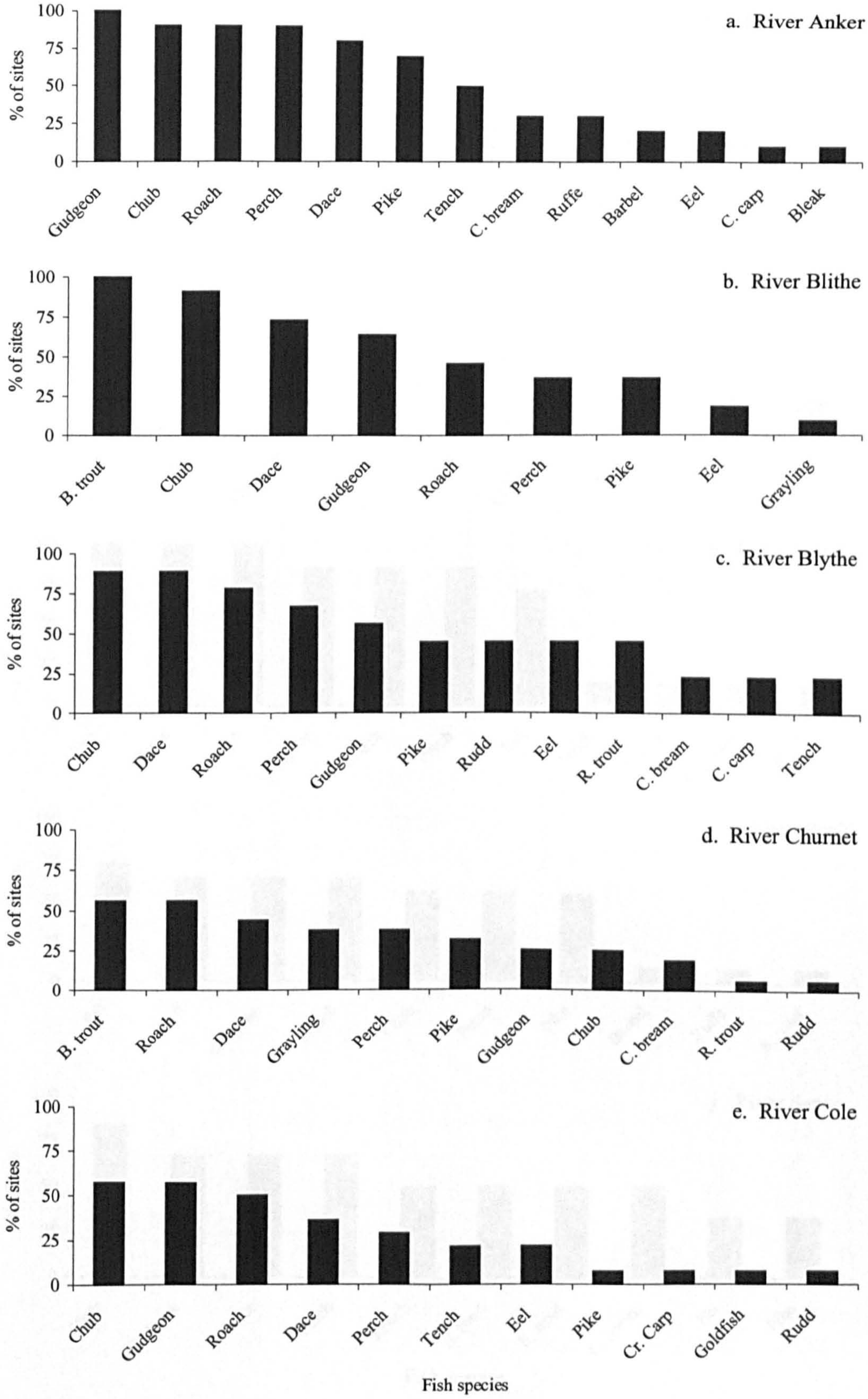


Fig. 4.15 Percentage of sites in the Trent catchment containing major fish species

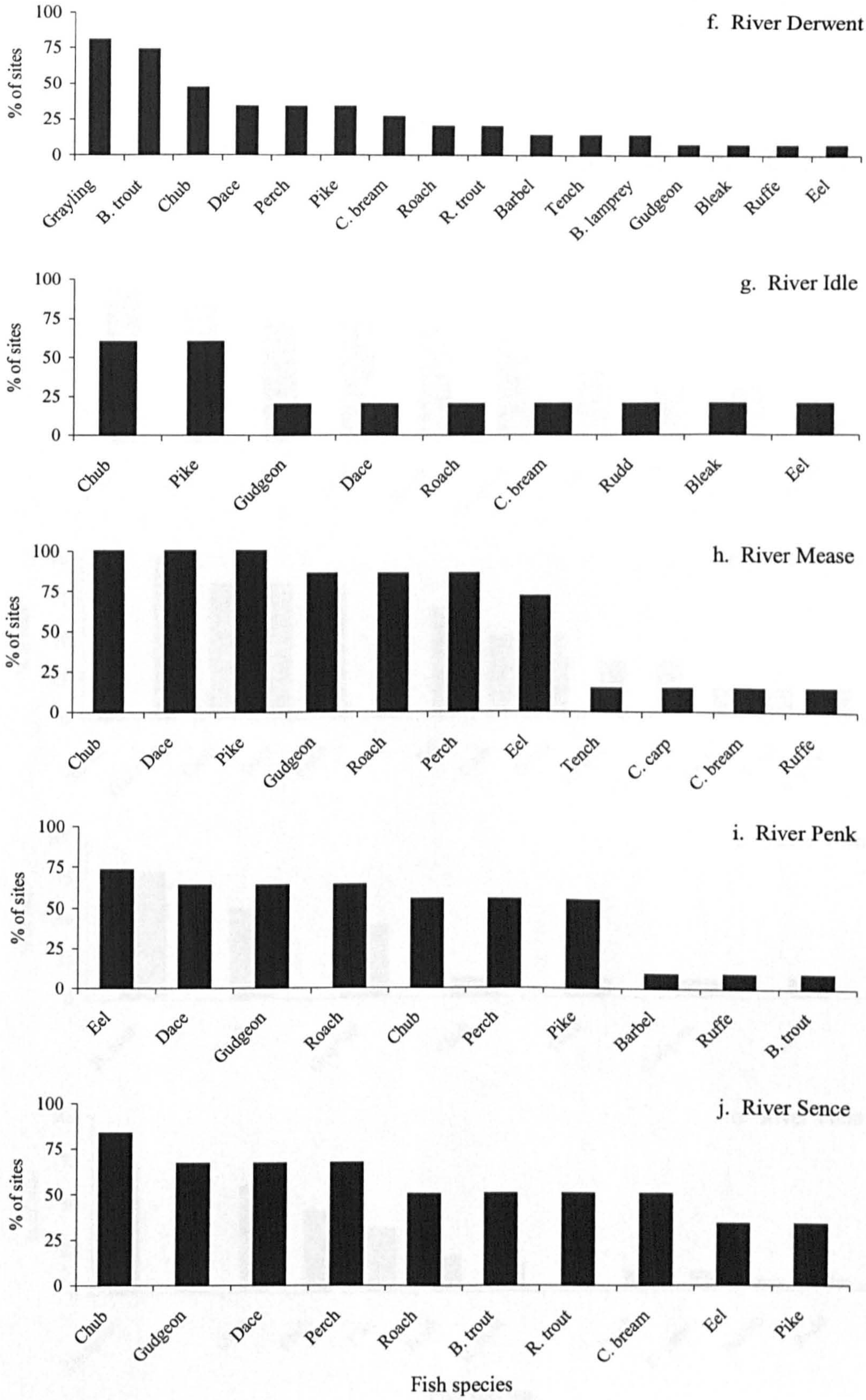


Fig. 4.15 (Continued) Percentage of sites in the Trent catchment containing major fish species

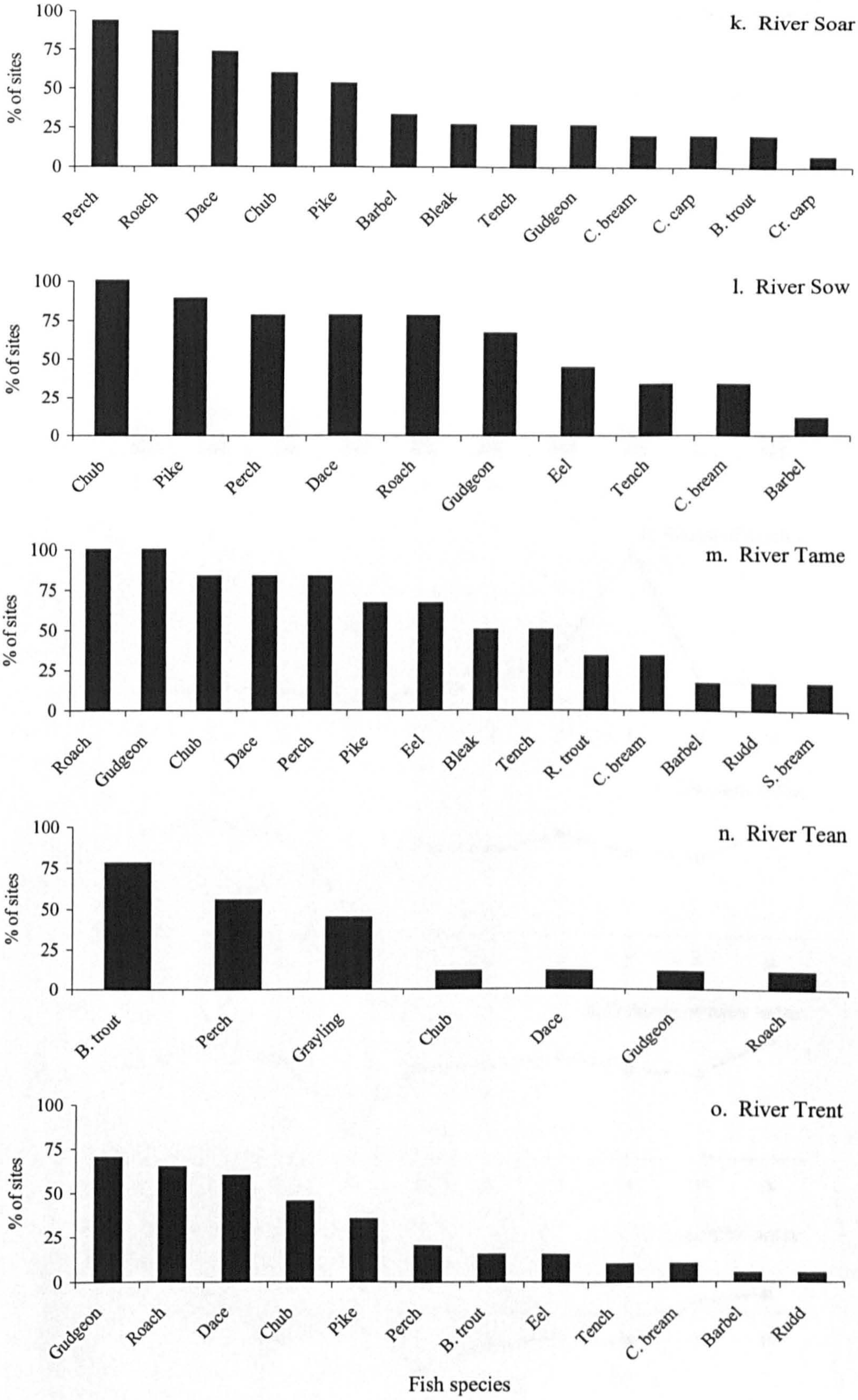


Fig. 4.15 (Continued) Percentage of sites in the Trent catchment containing major fish species

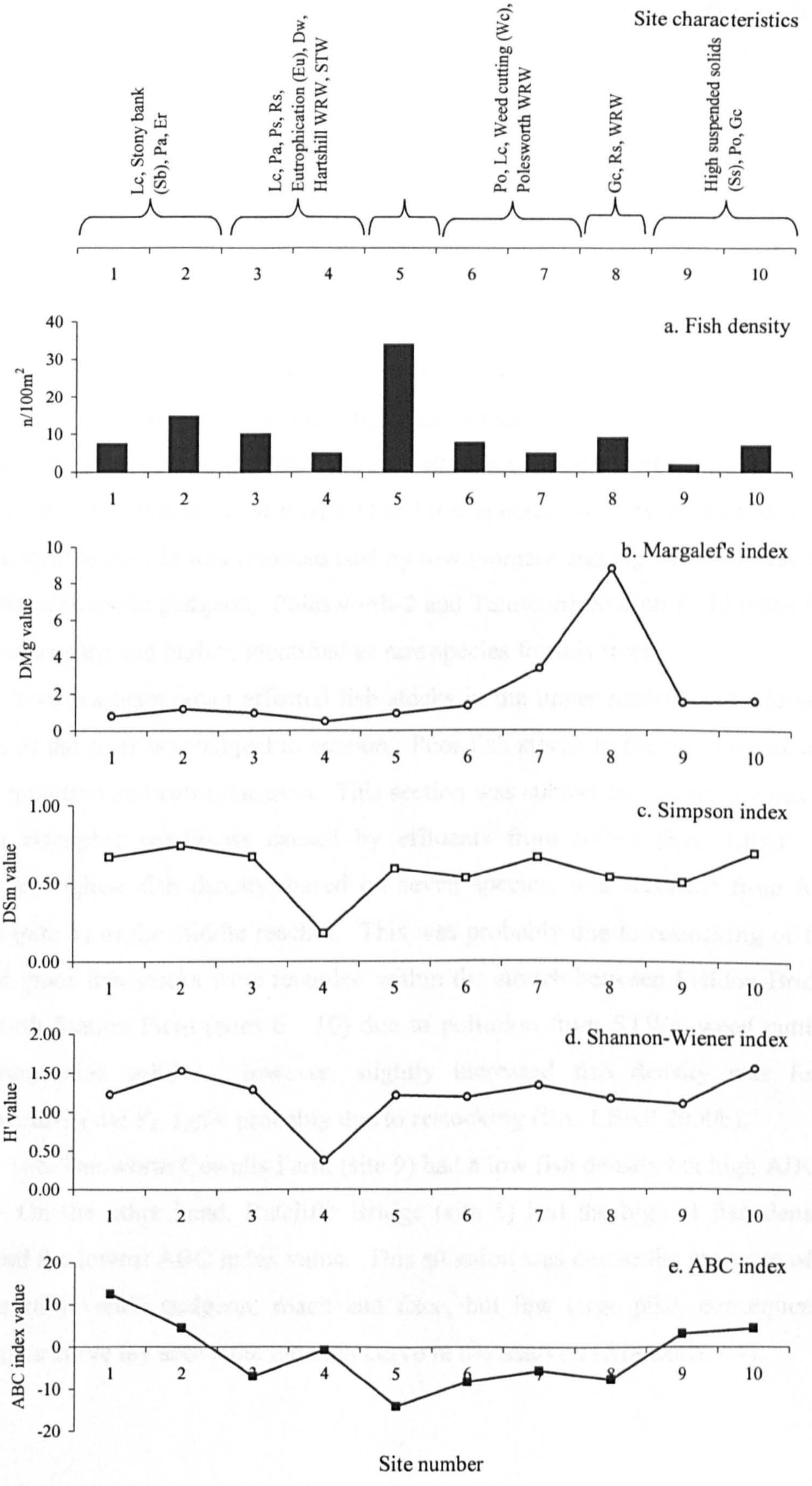


Fig. 4.16 Site characteristics and variations in different indices for the River Anker

Similar dendrograms, based on fish abundance and biomass, were obtained from cluster analysis. In both dendrograms, Mancetter Mill, Polesworth-1 and U/S Tamworth Cowells Farm (sites 4, 7 & 9) were grouped by poor abundance and biomass (Figs 4.17a & b). However, Weddington and Tamworth Station Field (sites 1 & 10) were grouped with poor biomass sites and isolated from the remaining sites (Fig. 4.17b). Ratcliffe (site 5) in the middle reach and U/S Tamworth Cowells Farm (site 9) in the lower reach were separated by the TWINSPAN analysis, based on fish abundance (Fig. 4.17c). The TWINSPAN for biomass also isolated Ratcliffe from the remaining sites but grouped U/S Tamworth Cowells Farm with other high biomass sites (Fig. 4.17d). The DECORANA analysis dispersed Mancetter Mill, Ratcliffe, Polesworth-2 and Tamworth Station Field (sites 4, 5, 8 & 10) towards the periphery of the ordination plot (Fig. 4.17e). Mancetter Mill (site 4) had low species diversity, density and biomass while Ratcliffe (site 5) was characterised by low biomass and high density, due to large numbers of juvenile gudgeon. Polesworth-2 and Tamworth Station Field (sites 8 & 10) had crucian carp and barbel, identified as rare species for this river.

Low instream cover affected fish stocks in the upper reaches. In addition, this section of the river was subject to erosion. Poor fish stocks in the middle reaches were due to pollution and eutrophication. This section was subject to excessive weed growth due to eutrophic conditions caused by effluents from STWs (EA, LEAP 2000a). However, highest fish density, based on seven species, was recorded from Ratcliffe Bridge (site 5) in the middle reaches. This was probably due to restocking of fish. In general, poor fish stocks were recorded within the stretch between Fieldon Bridge and Tamworth Station Field (sites 6 - 10) due to pollution from STWs, weed cutting and high suspended solids. However, slightly increased fish density was found at Polesworth-2 (site 8), again probably due to restocking (EA, LEAP 2000b).

U/S Tamworth Cowells Farm (site 9) had a low fish density but high ABC index value. On the other hand, Ratcliffe Bridge (site 5) had the highest fish density but produced the lowest ABC index value. This situation was due to the presence of a high number of juvenile gudgeon, roach and dace, but few large pike, consequently the abundance curve lay above the biomass curve in the analysis (Appendix 4.4).

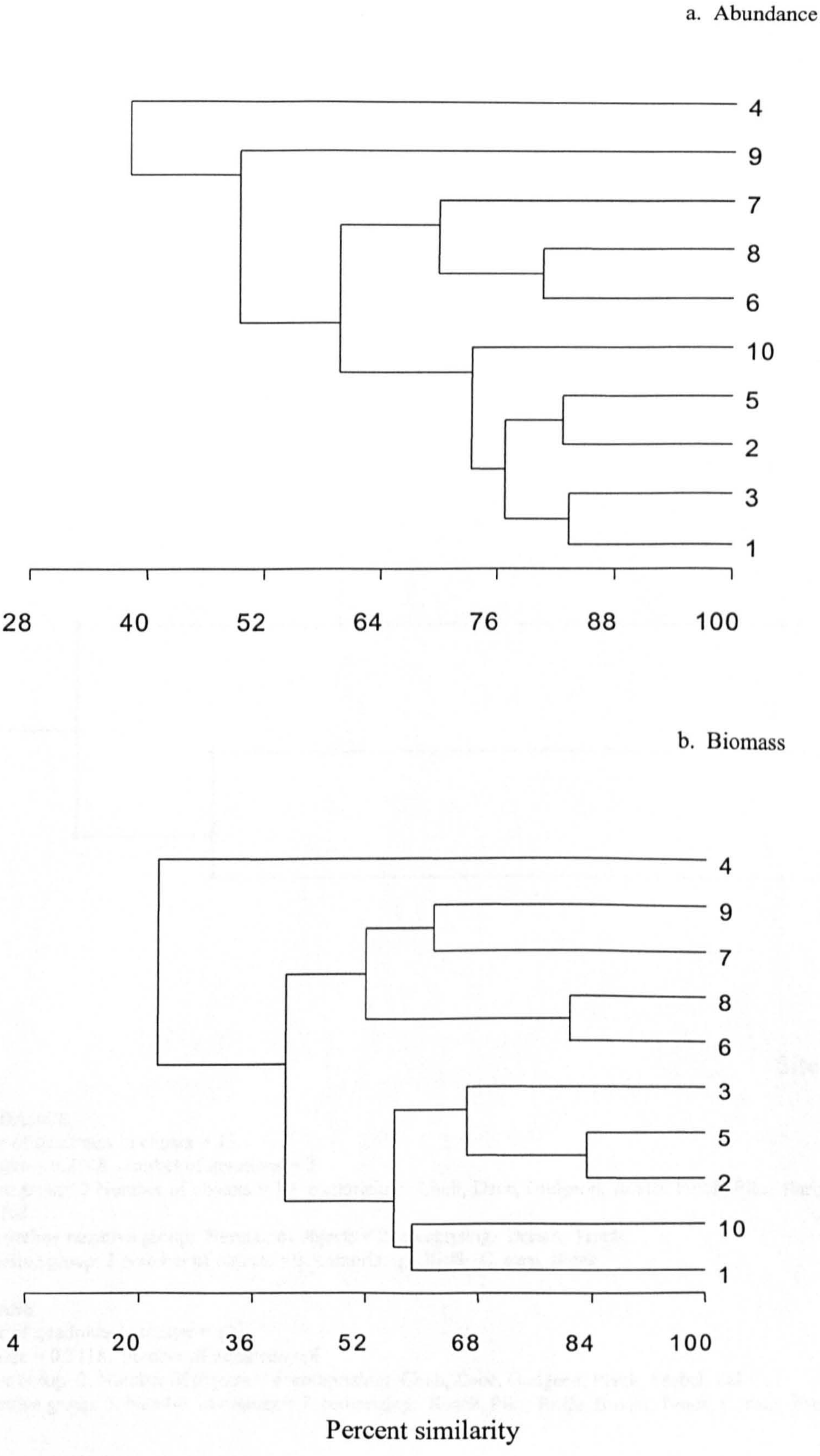
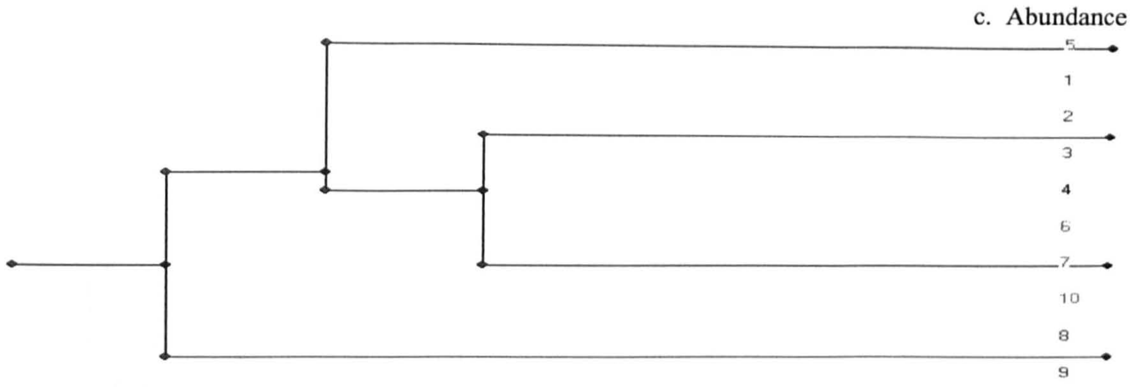
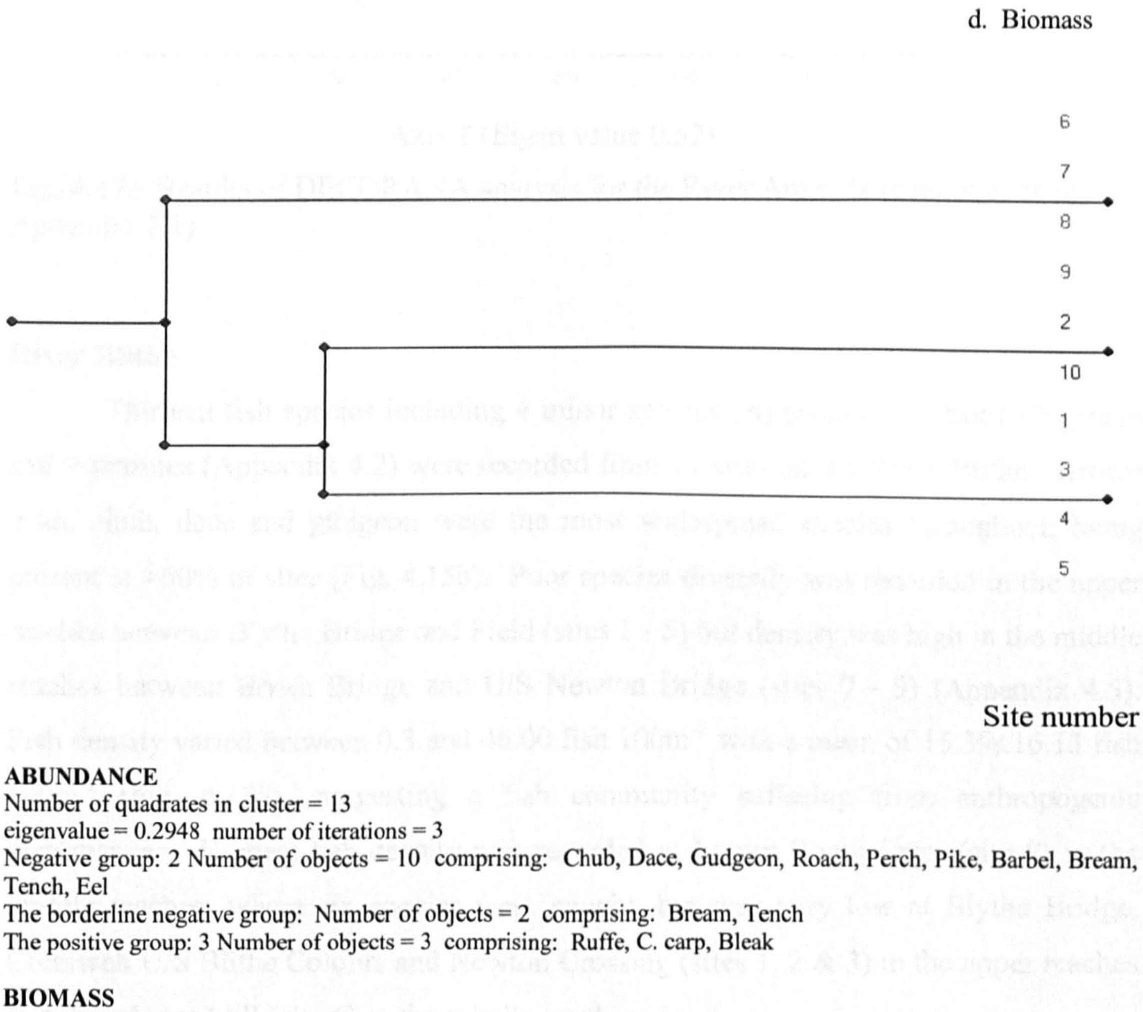


Fig. 4.17 Dendrograms based on UPGMA analysis for the River Anker (site number as in Appendix 2.1)





ABUNDANCE
Number of quadrates in cluster = 13
eigenvalue = 0.2948 number of iterations = 3
Negative group: 2 Number of objects = 10 comprising: Chub, Dace, Gudgeon, Roach, Perch, Pike, Barbel, Bream, Tench, Eel
The borderline negative group: Number of objects = 2 comprising: Bream, Tench
The positive group: 3 Number of objects = 3 comprising: Ruffe, C. carp, Bleak

BIOMASS
Number of quadrates in cluster = 13
eigenvalue = 0.3118, number of iterations = 4
Negative group: 2, Number of objects = 6 comprising: Chub, Dace, Gudgeon, Perch, Barbel, Eel
The positive group: 3, Number of objects = 7 comprising: Roach, Pike, Ruffe, Bream, Tench, C. carp, Bleak

Fig. 4.17 (Continued) Results of TWINSPAN for the River Anker at one level of division (site number as in Appendix 2.1)

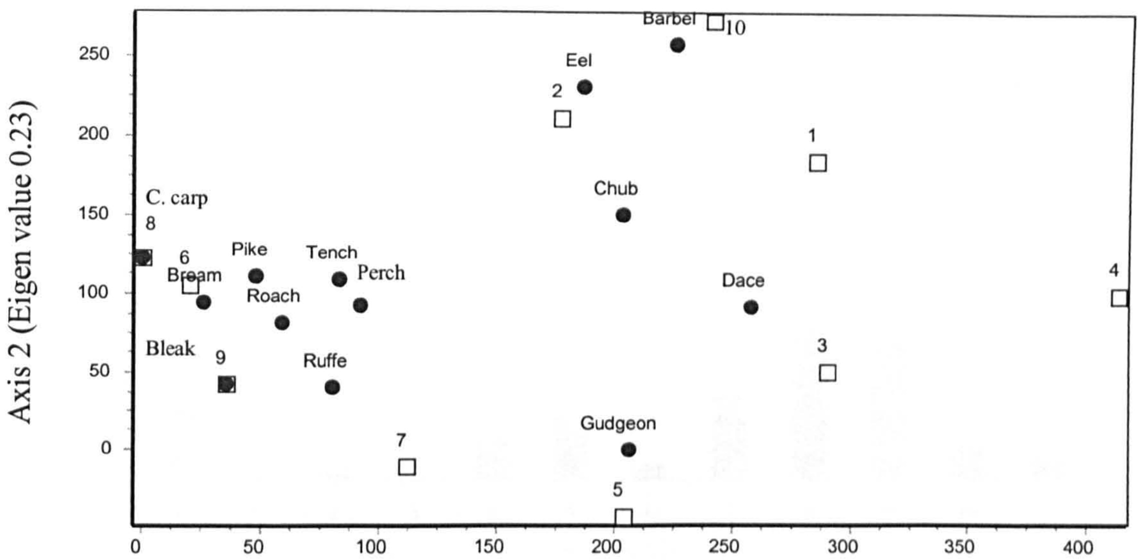


Fig. 4.17e Results of DECORANA analysis for the River Anker (site number as in Appendix 2.1)

River Blithe

Thirteen fish species including 4 minor species (Appendix 4.1) from 12 genera and 9 families (Appendix 4.2) were recorded from 11 sites on the River Blithe. Brown trout, chub, dace and gudgeon were the most widespread species throughout, being present at >60% of sites (Fig. 4.15b). Poor species diversity was recorded in the upper reaches between Blythe Bridge and Field (sites 1 - 5) but density was high in the middle reaches between Booth Bridge and U/S Newton Bridge (sites 7 - 9) (Appendix 4.5). Fish density varied between 0.3 and 46.00 fish 100m^{-2} with a mean of 15.39±16.13 fish 100m^{-2} (Fig. 4.18a), suggesting a fish community suffering from anthropogenic disturbances. Highest fish density was recorded at Lower Booth Farm (site 8) in the middle reaches, where six species were caught, but was very low at Blythe Bridge, Cresswell U/S Blithe Colours and Newton Crossing (sites 1, 2 & 3) in the upper reaches and Burnthurst Mill (site 6) in the middle reaches.

These observations on fish diversity, density and assemblage were supported by the various diversity indices (Figs 4.18b, c, d & e). The ABC index discriminated Booth Bridge, Lower Booth Farm and U/S Newton Bridge (sites 7, 8 & 9) as poor from other sites although these sites had a high fish density.

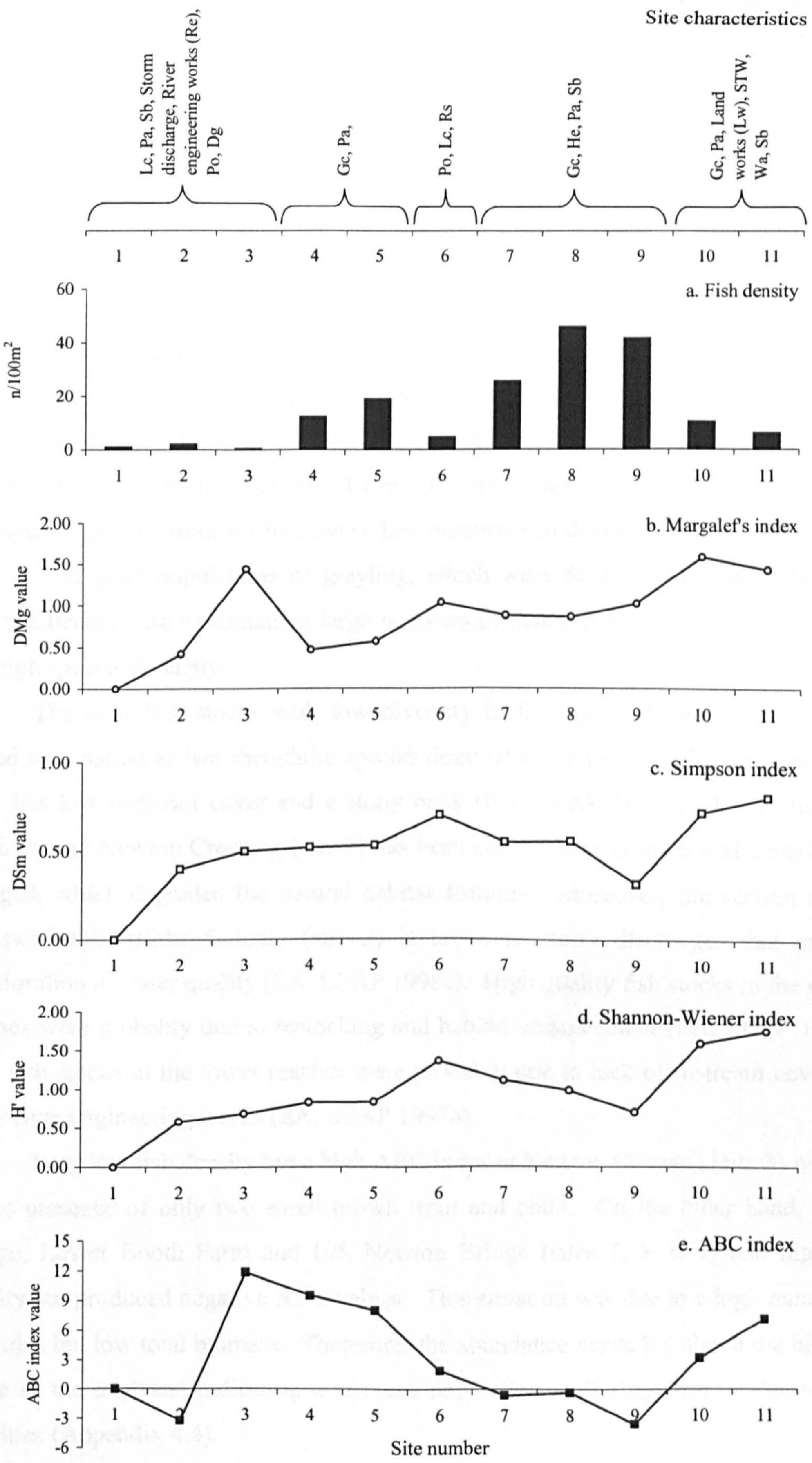


Fig. 4.18 Site characteristics and variations in different indices for the River Blithe

Two dendrograms, each having two distinct site groups, were obtained from cluster analysis, based on fish density and biomass, respectively (Figs 4.19a & b). In both analyses, Blythe Bridge, Cresswell U/S Blithe Colours and Newton Crossing (sites 1, 2 & 3) from the upper reaches and Priory Farm and Hamstall Ridware (sites 10 & 11) from the lower reaches were grouped as low abundance and biomass sites, respectively. However, Burnthurst Mill (site 6) was included with high abundance and biomass sites, although the site supported relatively low fish density. The TWINSPAN analysis for fish abundance isolated Newton Crossing and Booth Bridge (sites 3 & 7) from the remainder (Fig. 4.19c). Booth Bridge (site 7) was also separated by the TWINSPAN for biomass (Fig. 4.19d). DECORANA analysis discriminated Blythe Bridge, Burnthurst Mill, Booth Bridge, Lower Booth Farm and U/S Newton Bridge (sites 1, 6, 7, 8 & 9) in the ordination plot (Fig. 4.19e). Blythe Bridge (site 1) was characterised by rare species (brown trout for this river), low biomass and density while Burnthurst Mill (site 6) had good populations of grayling, which were found at this site only. U/S Newton Bridge (site 9) contained large numbers of juvenile dace, having low biomass but high species diversity.

The poor fish stocks with low diversity in the upper reaches were probably linked to zonation as two rheophilic species dominated the fishery. This section of the river has low instream cover and a stony bank (EA, LEAP 1997a). In addition, the reach around Newton Crossing (site 3) has been subjected to channel maintenance, i.e. dredged, which degraded the natural habitat features. Moreover, the section around Cresswell U/S Blithe Colours (site 2) is prone to storm discharges that cause a deterioration in water quality (EA, LEAP 1998c). High quality fish stocks in the middle reaches were probably due to restocking and habitat enhancement (EA, LEAP 1998c). Low fish stocks in the lower reaches were probably due to lack of instream cover and some river engineering works (EA, LEAP 1997a).

Very low fish density but a high ABC index at Newton Crossing (site 3) was due to the presence of only two small brown trout and chub. On the other hand, Booth Bridge, Lower Booth Farm and U/S Newton Bridge (sites 7, 8 & 9) had high fish density but produced negative ABC values. This situation was due to a high number of juveniles but low total biomass. Therefore, the abundance curve lay above the biomass curve in the analysis, indicating a stressed population, suffering from anthropogenic activities (Appendix 4.4).

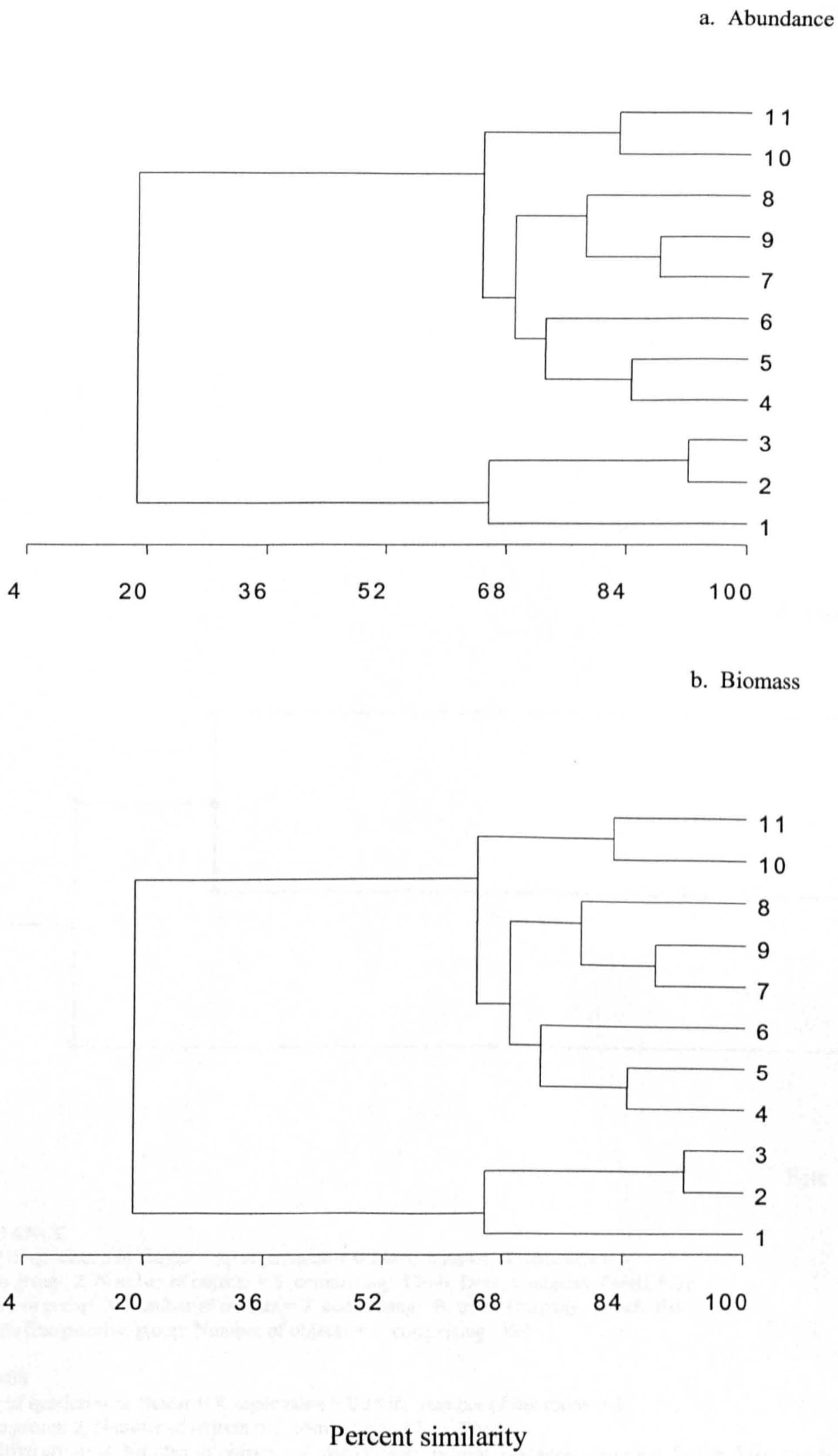
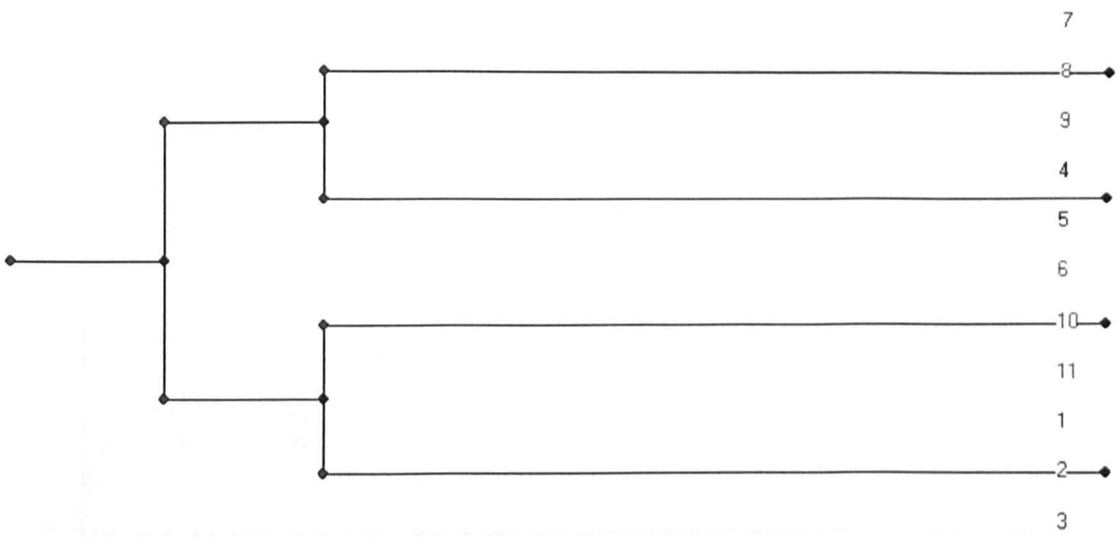
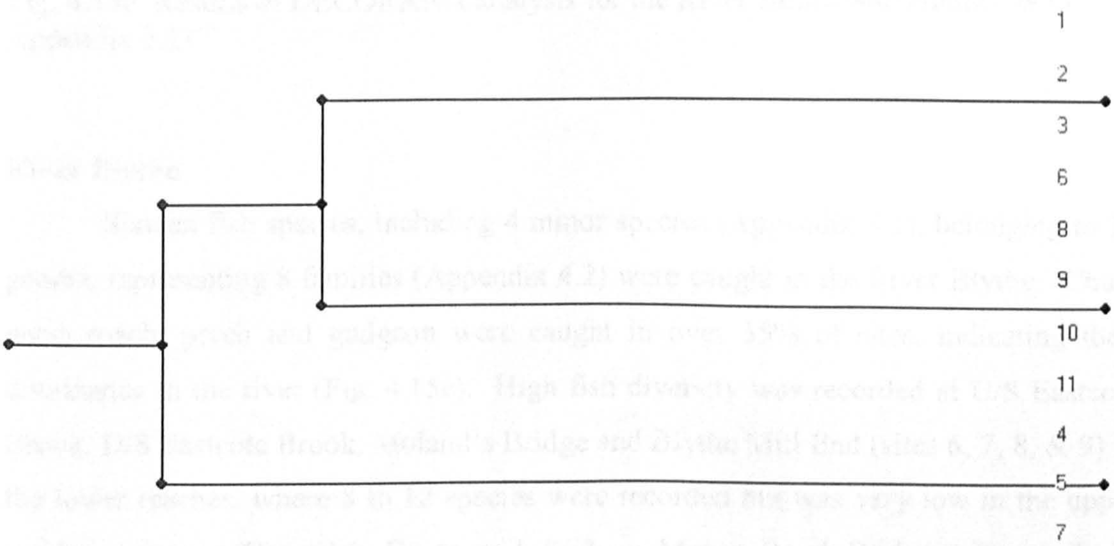


Fig. 4.19 Dendrograms based on UPGMA analysis for the River Blithe (site number as in Appendix 2.1)

c. Abundance



d. Biomass



Site number

ABUNDANCE
Number of quadrates in cluster = 9, eigenvalue = 0.2486, number of iterations = 5
Negative group: 2, Number of objects = 5 comprising: Chub, Dcae, Gudgeon, Perch, Pike
The positive group: 3, Number of objects = 4 comprising: B. trout, Grayling, Roach, Eel
The borderline positive group: Number of objects = 1 comprising: Eel

BIOMASS
Number of quadrates in cluster = 9, eigenvalue = 0.3836, number of iterations = 3
Negative group: 2, Number of objects = 2 comprising: Chub, Dace
The positive group: 3, Number of objects = 7 comprising: B.trout, Gudgeon, Grayling, Roach, Pike, Perch, Eel
The borderline positive group: Number of objects = 1 comprising: Grayling

Fig. 4.19 (Continued) Results of TWINSPAN for the River Blithe at one level of division (site number as in Appendix 2.1)

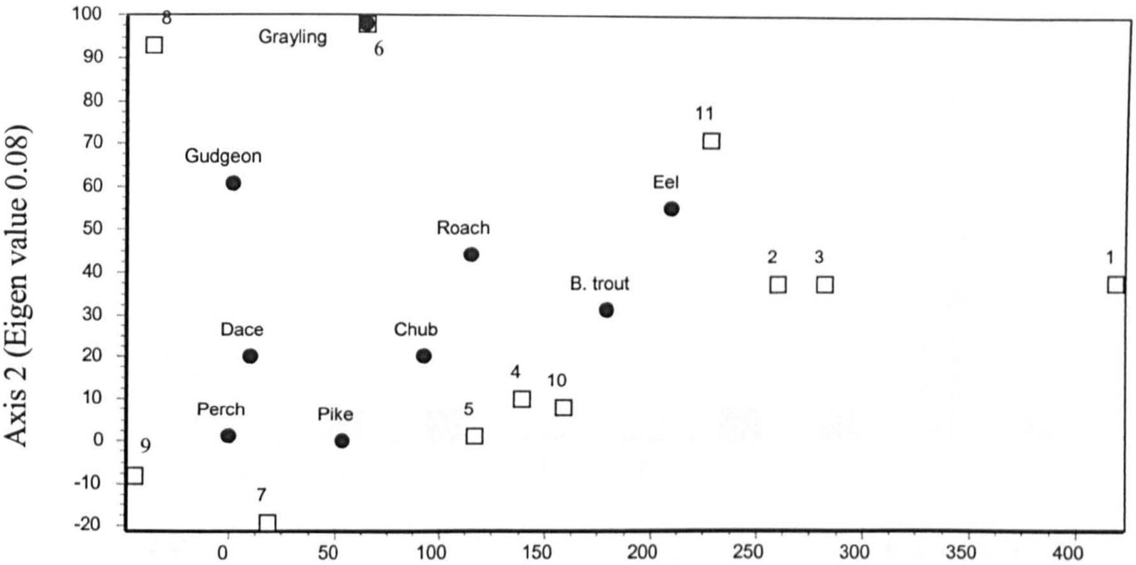


Fig. 4.19e Results of DECORANA analysis for the River Blithe (site number as in Appendix 2.1)

River Blythe

Sixteen fish species, including 4 minor species (Appendix 4.1), belonging to 15 genera, representing 8 families (Appendix 4.2) were caught in the River Blythe. Chub, dace, roach, perch and gudgeon were caught in over 55% of sites, indicating their dominance in the river (Fig. 4.15c). High fish diversity was recorded at U/S Eastcote Brook, D/S Eastcote Brook, Moland's Bridge and Blythe Mill End (sites 6, 7, 8, & 9) in the lower reaches, where 8 to 12 species were recorded but was very low in the upper reaches between Cheswick Green and Widney Manor Road Bridge (sites 1 & 2) (Appendix 4.5). Fish density ranged between 0 and 77.1 fish $100m^{-2}$ with a mean of 18.04 ± 23.22 fish $100m^{-2}$ (Fig. 4.20a), indicating a disturbed fish community. No fish were recorded at Cheswick Green (site 1) while the highest fish density was recorded at Moland's Bridge (site 8), where eight species were captured.

These results on fish diversity, density and assemblage were supported by the various diversity indices (Figs 4.20b, c, d & e). The ABC index discriminated Moland's Bridge (site 8) as poor (Fig. 4.20e). Cluster analysis, based on fish density grouped the sites with high abundance in the lower reaches (sites 6, 7, 8 & 9) and isolated the most upstream reaches (sites 1 & 2) as poor (Fig. 4.21a).

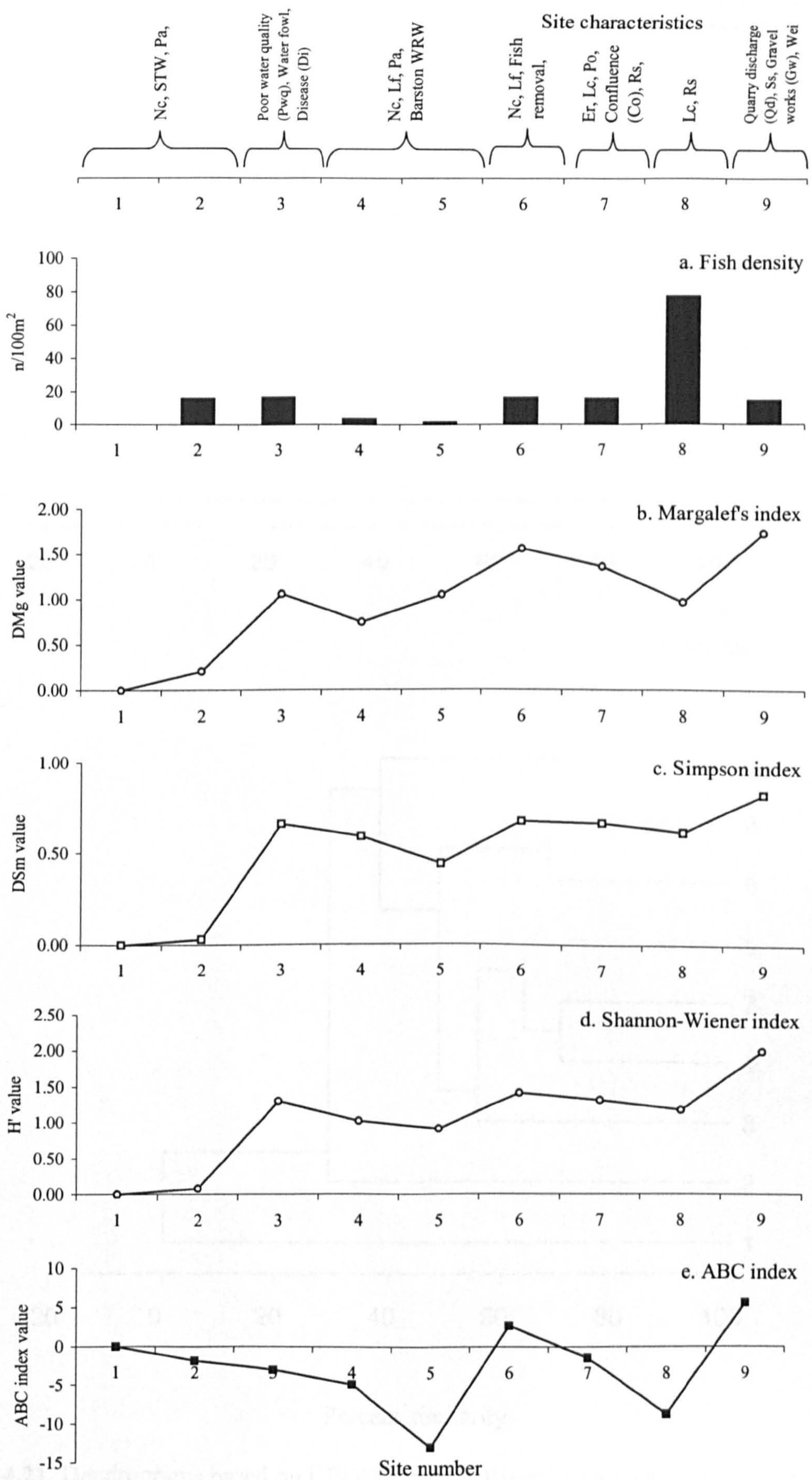
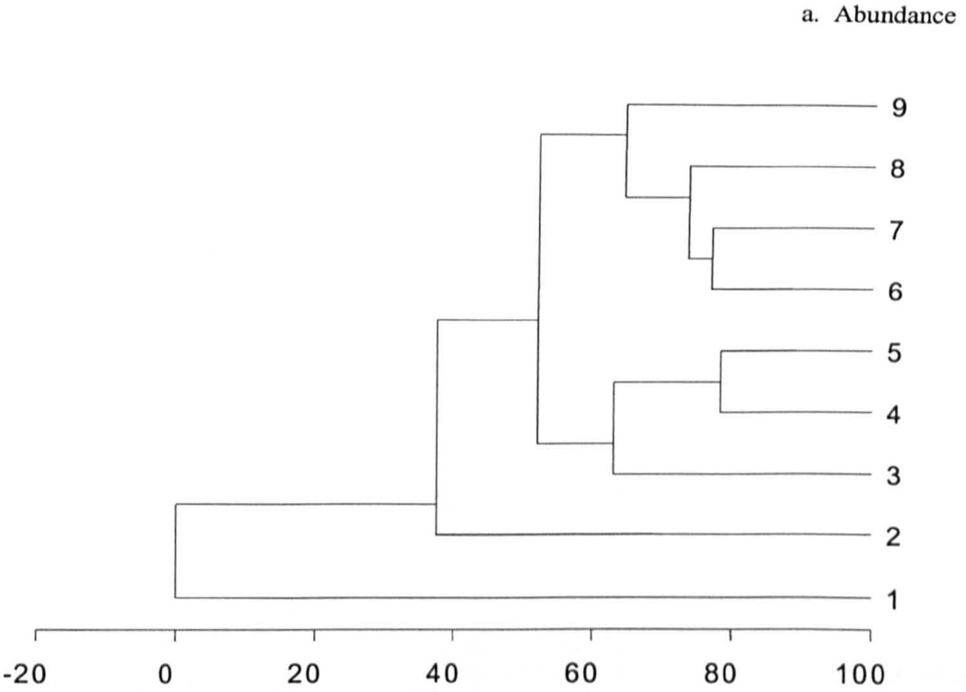


Fig. 4.20 Site characteristics and variations in different indices for the River Blythe



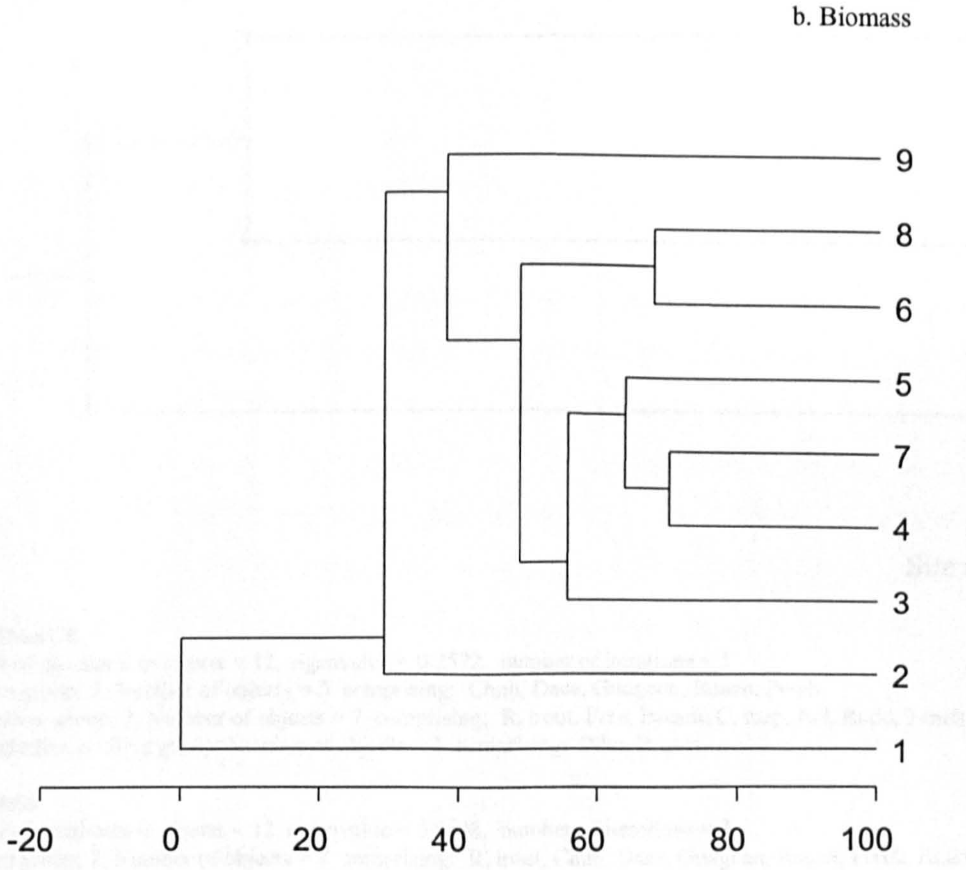
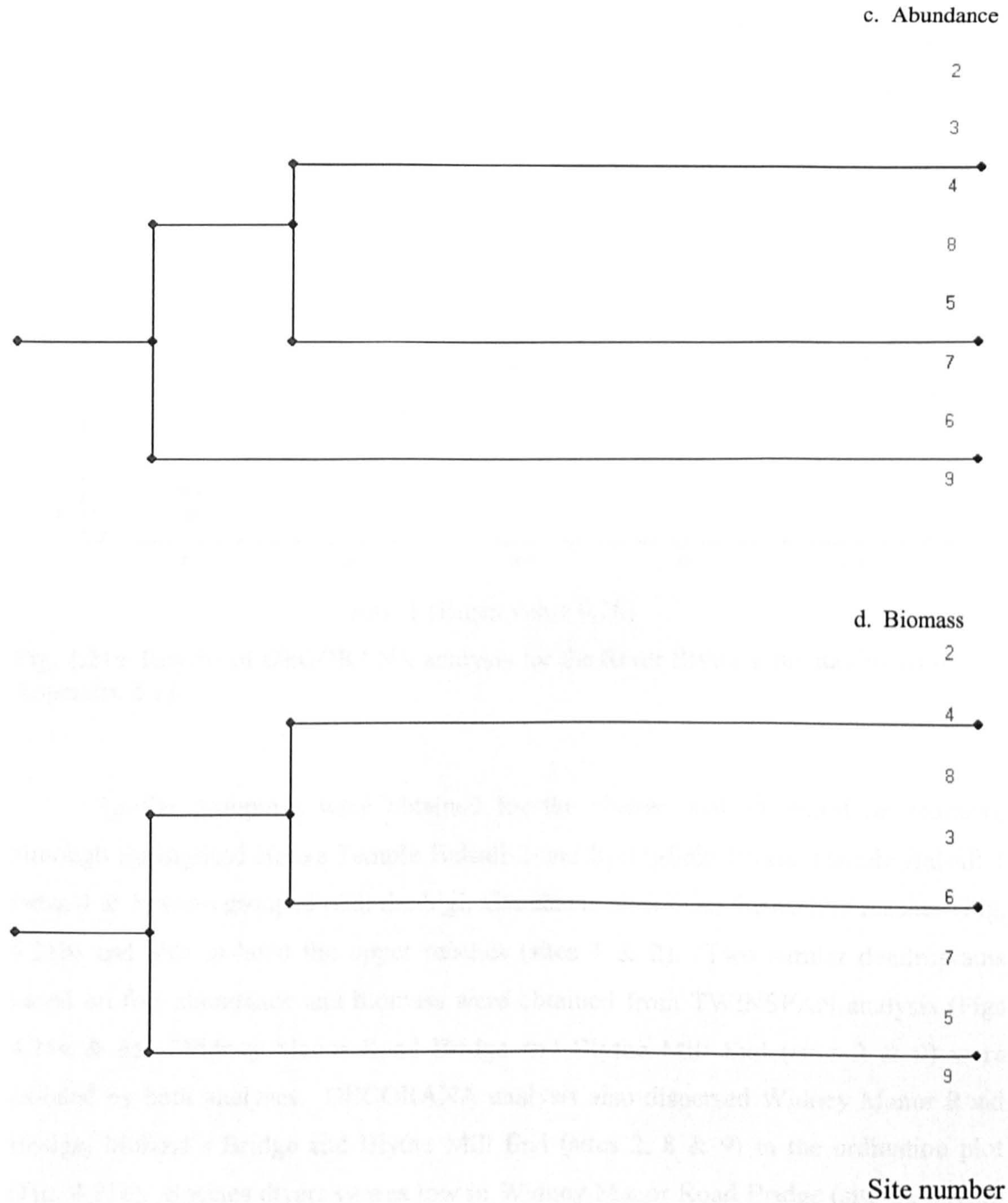


Fig. 4.21 Dendrograms based on UPGMA for the River Blythe (site number as in Appendix 2.1)



ABUNDANCE
Number of quadrates in cluster = 12, eigenvalue = 0.2572, number of iterations = 5
Negative group: 2, Number of objects = 5 comprising: Chub, Dace, Gudgeon, Roach, Perch
The positive group: 3, Number of objects = 7 comprising: R. trout, Pike, Bream, C. carp, Eel, Rudd, Tench
The borderline positive group: Number of objects = 2 comprising: Pike, Bream

BIOMASS
Number of quadrates in cluster = 12, eigenvalue = 0.3948, number of iterations = 3
Negative group: 2, Number of objects = 7 comprising: R. trout, Chub, Dace, Gudgeon, Roach, Perch, Rudd
The positive group: 3, Number of objects = 5 comprising: Pike, Bream, C. carp, Eel, Tench

Fig. 4.21 (Continued) Results of TWINPSAN for the River Blythe at one level of division (site number as in Appendix 2.1)

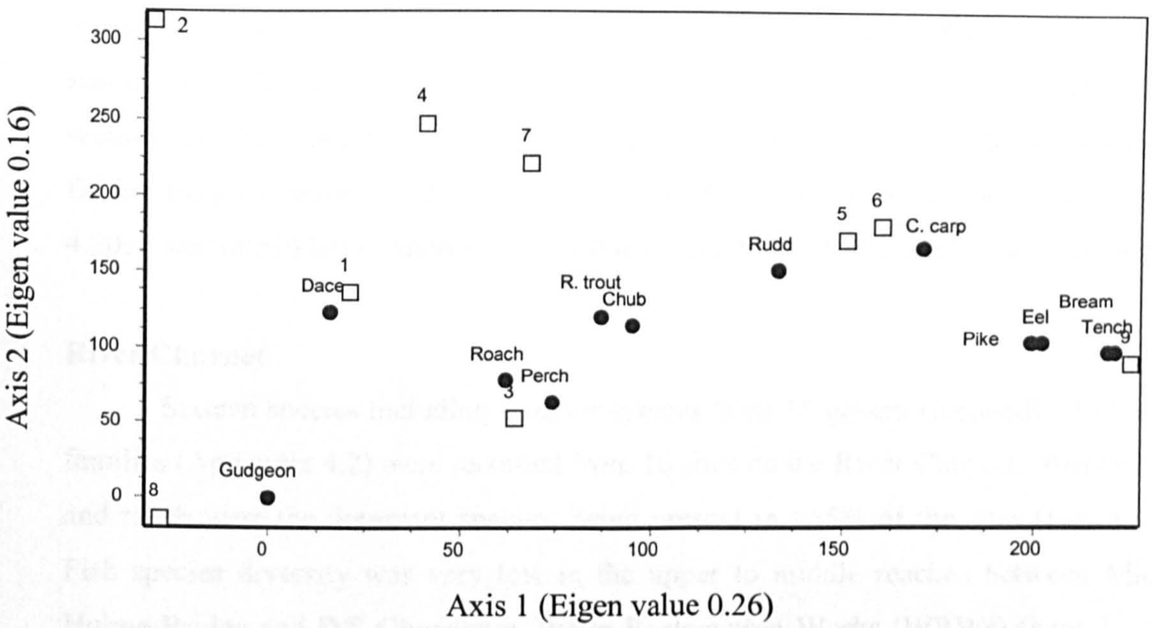


Fig. 4.21e Results of DECORANA analysis for the River Blythe (site number as in Appendix 2.1)

Similar groupings were obtained for the cluster analysis based on biomass, although Springfield House Temple Balsall-2 and Springfield House Temple Balsall-1 (sites 4 & 5) were grouped with the high abundance sites from the middle reaches (Fig. 4.21b) and also isolated the upper reaches (sites 1 & 2). Two similar dendrograms based on fish abundance and biomass were obtained from TWINSPAN analysis (Figs 4.21c & d). Widney Manor Road Bridge and Blythe Mill End (sites 2 & 9) were isolated by both analyses. DECORANA analysis also dispersed Widney Manor Road Bridge, Moland's Bridge and Blythe Mill End (sites 2, 8 & 9) in the ordination plot (Fig. 4.21e). Species diversity was low in Widney Manor Road Bridge (site 2), having high numbers of juvenile dace. Blythe Mill End (site 9) was dispersed due to presence of rare species (common bream & tench are rare for this river). Moland's Bridge (site 8) was found to have a high number of juvenile gudgeon, roach and dace, having low biomass.

The poor fish stocks in the upper reaches were due to poor water quality and habitat, as the reach receives domestic discharges from urban development (Fig. 4.20). In addition, a high number of waterfowl degrade water quality (Fig. 4.20) leading to outbreaks of fish disease (EA, LEAP 1998d). The middle reaches had poor fish stocks due to pollution from STWs and the reach around D/S Eastcote Brook (site 7) was subject to erosion (Fig. 4.20). High fish density was recorded at Moland's Bridge (site 8) (Fig. 4.20a), which was due to annual restocking of trout (EA, LEAP 1999c). Fish stocks show evidence of declining downstream of Moland's Bridge due to high suspended solids and habitat alteration by river engineering works (Fig. 4.20). This section also receives quarry discharges (EA, LEAP 1998d). The low value at Moland's Bridge (site 8), which contained high fish density but a negative ABC index (Fig. 4.20e), was due to large numbers of juvenile gudgeon, roach and dace (Appendix 4.4).

River Churnet

Sixteen species including 5 minor species from 15 genera (Appendix 4.1) and 9 families (Appendix 4.2) were recorded from 16 sites on the River Churnet. Brown trout and roach were the dominant species, being present in >55% of the sites (Fig. 4.15d). Fish species diversity was very low in the upper to middle reaches between Middle Hulme Bridge and D/S Cheddleton Water Reclamation Works (WRWs) (sites 1 - 10), but was moderate in the lower reaches (sites 11 - 16) (Appendix 4.5). A highly variable fish density ranging from 0 to 32 fish $100m^{-2}$ with a mean of 3.97 ± 8.00 fish $100m^{-2}$ (Fig. 4.22a) indicated a stressed fish community suffering from anthropogenic disturbances. Highest fish density was recorded at Middle Hulme Bridge (site 1), where only two species were found, but was very low in the reaches between D/S Tittesworth Reservoir and Thomas Boltons Ltd. (sites 2 - 11) and the site D/S Alton WRW (site 15) in the lower reaches.

These findings on the diversity, density and assemblage were supported by the various diversity indices (Figs 4.22b, c, d & e). The ABC index discriminated Middle Hulme Bridge, U/S Leekbrook WRW and Whiston Bridge (sites 1, 7 & 12) from other sites as poor (Fig. 4.22e). Cluster analysis based on fish biomass grouped the sites 3, 4 & 5 with poor biomass in the upper reaches (Fig. 4.23b). However, cluster analysis, based on fish abundance grouped South Hillswood Farm, Abby Green Road Site-1, Abby Green Farm, Westwood Golf Club, St. Edwards Hospital and D/S Cheddleton WRW (sites 3, 4, 5, 6, 8 & 10) with poor abundance sites and isolated the most upstream reach (site 1) (Fig. 4.23a). In TWINSPAN analysis, Middle Hulme Bridge (site 1) was included with high abundance sites (Fig. 4.23c) while it was grouped with low biomass sites in the biomass analysis (Fig. 4.23d). Middle Hulme Bridge, D/S Tittesworth Reservoir, Westwood Golf Club and D/S Cheddleton WRW (sites 1, 2, 6 & 10) were discriminated by the DECORANA analysis (Fig. 4.23e).

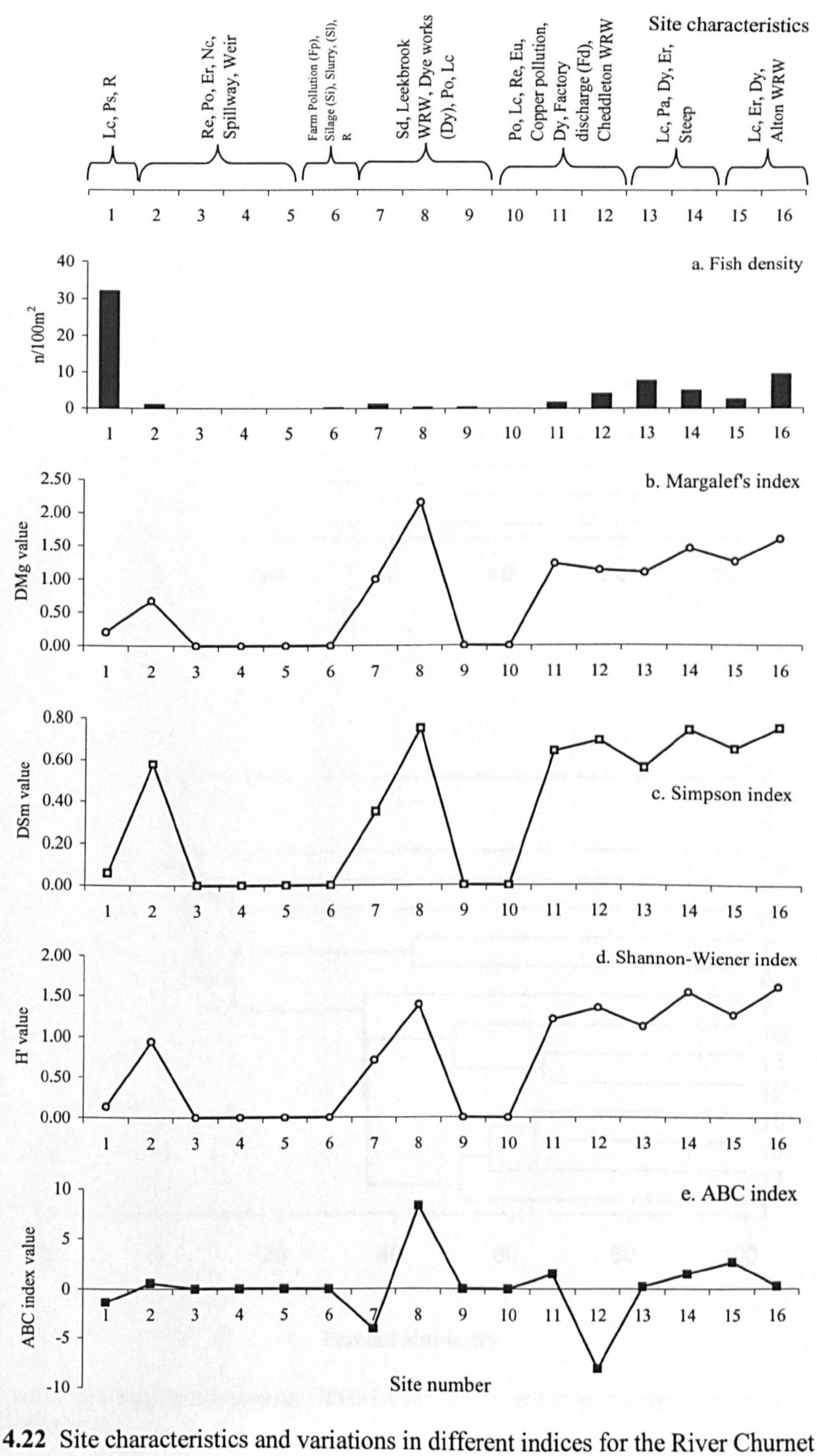


Fig. 4.22 Site characteristics and variations in different indices for the River Churnet

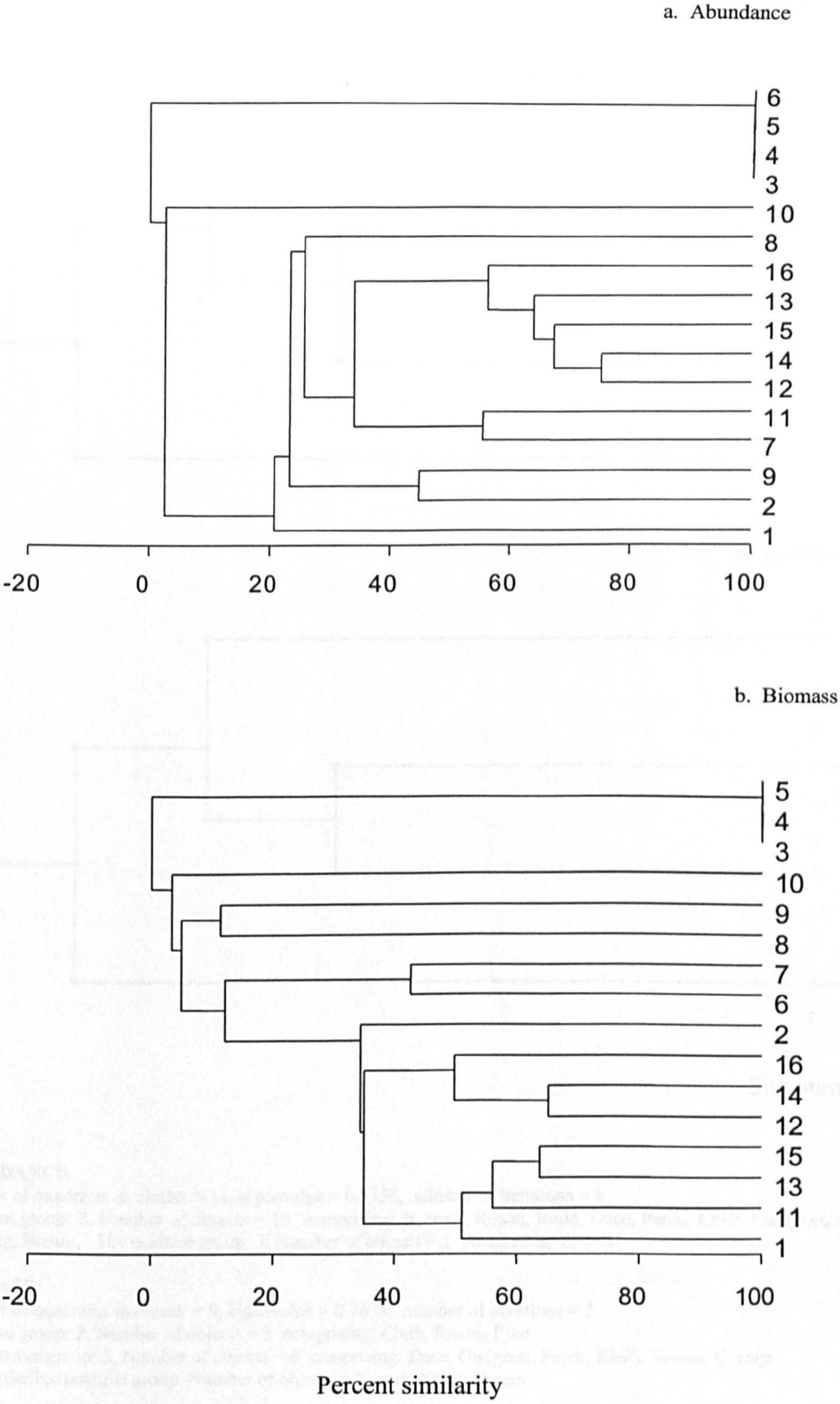
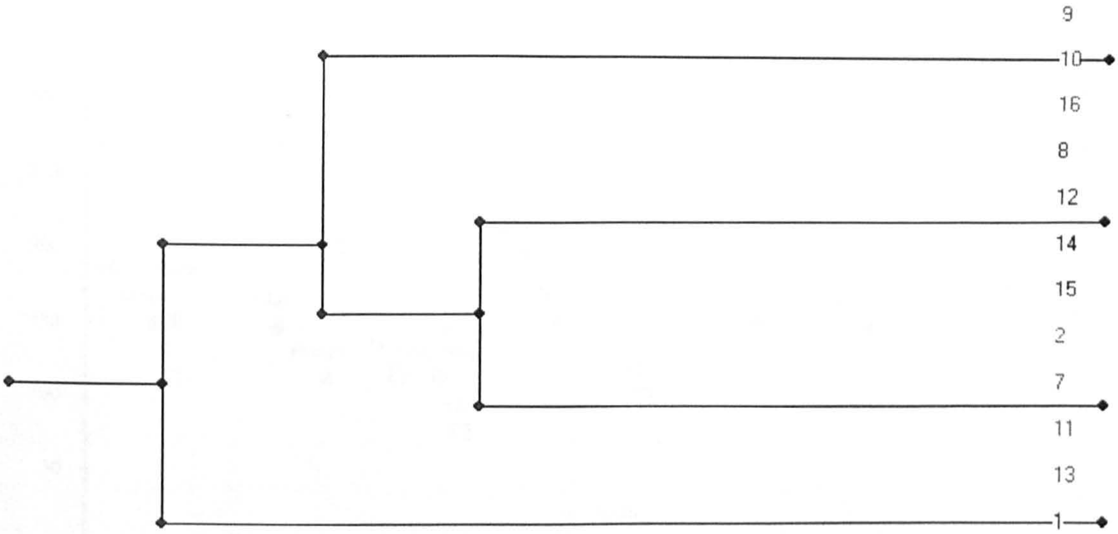
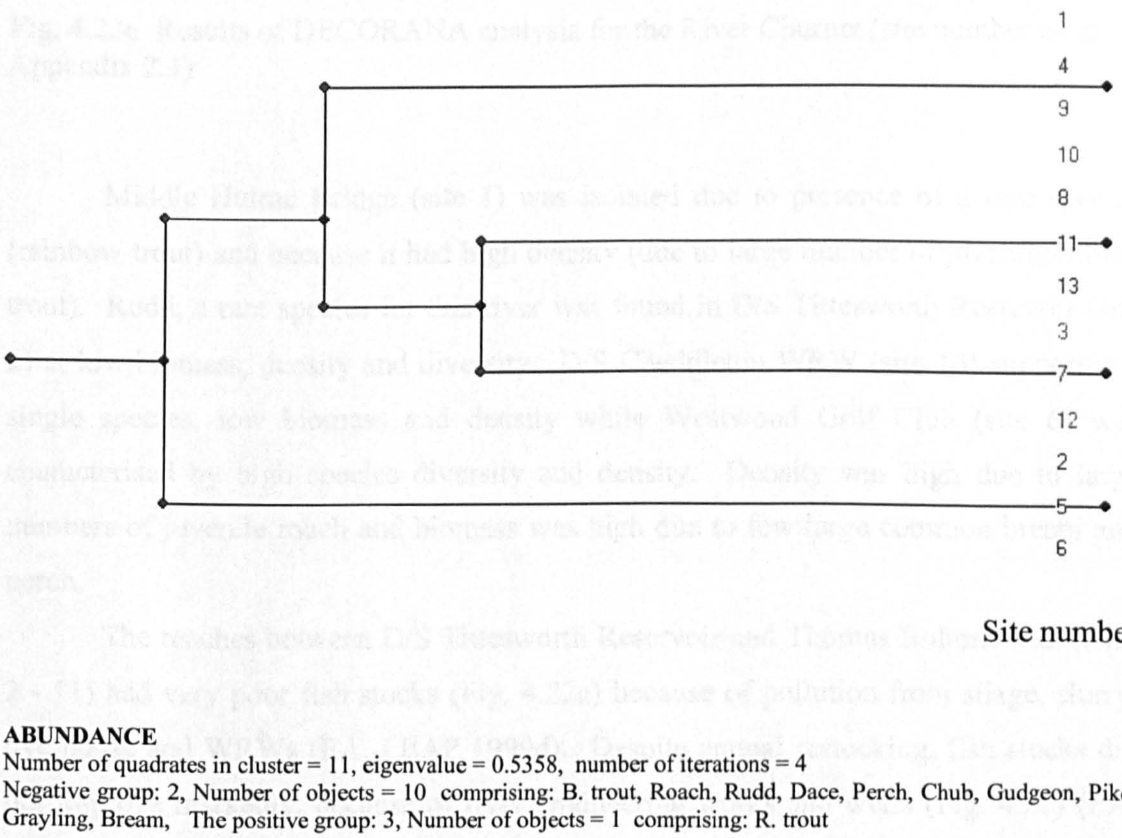


Fig. 4.23 Dendrograms based on UPGMA for the River Churnet (site number as in Appendix 2.1)

c. Abundance



d. Biomass



ABUNDANCE
Number of quadrates in cluster = 11, eigenvalue = 0.5358, number of iterations = 4
Negative group: 2, Number of objects = 10 comprising: B. trout, Roach, Rudd, Dace, Perch, Chub, Gudgeon, Pike, Grayling, Bream, The positive group: 3, Number of objects = 1 comprising: R. trout

BIOMASS
Number of quadrates in cluster = 9, eigenvalue = 0.2678, number of iterations = 5
Negative group: 2, Number of objects = 3 comprising: Chub, Roach, Pike
The positive group: 3, Number of objects = 6 comprising: Dace, Gudgeon, Perch, Bleak, Bream, C. carp
The borderline positive group: Number of objects = 1 comprising: Bream

Fig. 4.23 (Continued) Results of TWINSPAN for the River Churnet at one level of division (site number as in Appendix 2.1)

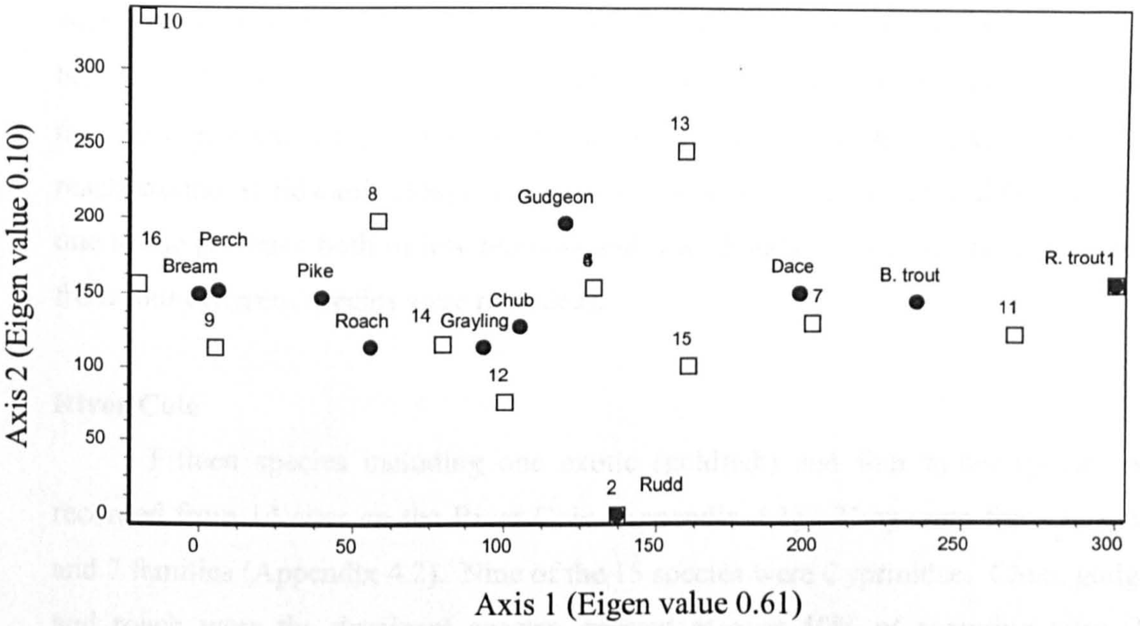


Fig. 4.23e Results of DECORANA analysis for the River Churnet (site number as in Appendix 2.1)

Middle Hulme Bridge (site 1) was isolated due to presence of a rare species (rainbow trout) and because it had high density (due to large number of juvenile brown trout). Rudd, a rare species for this river was found in D/S Tittesworth Reservoir (site 2) at low biomass, density and diversity. D/S Cheddleton WRW (site 10) supported a single species, low biomass and density while Westwood Golf Club (site 6) was characterised by high species diversity and density. Density was high due to large numbers of juvenile roach and biomass was high due to few large common bream and perch.

The reaches between D/S Tittesworth Reservoir and Thomas Boltons Ltd. (sites 2 - 11) had very poor fish stocks (Fig. 4.22a) because of pollution from silage, slurry, dye house and WRWs (EA, LEAP 1999d). Despite annual restocking, fish stocks did not improve markedly, because of river engineering works and weirs (Fig. 4.22) (EA, LEAP 2000c). In addition, the whole river was subject to erosion and the reach around D/S Cheddleton WRW (site 10) had been modified by constructing a flood embankment (Fig. 4.22). The lower reaches between Whiston Bridge and J.C.B. Rocester (sites 12 - 16) also had impoverished fish stocks due to pollution from a factory (Thomas Bolton Ltd.), dye house and WRWs (Fig. 4.22). This section was also prone to erosion, and had low instream cover (EA, LEAP 1999d).

Comparatively good fish stocks in the lower reaches were probably due to restocking and improvement in the effluent quality. High fish density and a negative ABC index at Middle Hulme Bridge (site 1) were due to high numbers of juvenile brown trout compared to total biomass and the presence of only two species, therefore, the abundance curve lay above the biomass curve in the analysis (Appendix 4.4). The reach around St Edwards Hospital (site 8) had low fish density but a high ABC index, due to the presence both of low biomass and low abundance (as four small individuals from four different species were recorded).

River Cole

Fifteen species including one exotic (goldfish) and four minor species were recorded from 14 sites on the River Cole (Appendix 4.1). They were from 13 genera and 7 families (Appendix 4.2). Nine of the 15 species were Cyprinidae. Chub, gudgeon and roach were the dominant species, present at over 50% of sampling sites (Fig. 4.15e). Distribution of carnivores was limited in the River Cole. Pike, the principal carnivore, was caught from only one site (Colehall, site 6) while perch were recorded from 4 sites (sites 7, 8, 9 & 10) (Appendix 4.5). Very poor species diversity was recorded in the upper reaches between Lowbrook Farm and Colehall (sites 1 - 6), but was moderate in the middle through lower reaches between Kingshurst-1 and Coleshill-2 (sites 7 - 14) (Appendix 4.5). Five seriously disturbed sites (sites 1 - 5) were identified in the upper reaches, where no fish were found. Fish density varied between 0 and 50.2 fish $100m^{-2}$ with a mean of 11.57 ± 15.80 fish $100m^{-2}$ (Fig. 4.24a). The highest fish density was at Coleshill-2 (site 14), where four species were recorded, but was very low at Colehall and D/S Cook's Lane Bridge (sites 6 & 9) in the middle reaches.

These findings on the diversity, density and assemblage were supported by the various diversity indices (Figs 4.24b, c, d & e). The ABC index discriminated D/S Cook's Lane Bridge (sites 9) as a good site and, Becons End and Coleshill Hospital-1 (sites 10 & 11) as poor (Fig. 4.24e). Two similar dendrograms were obtained on the basis of fish abundance and biomass, respectively (Figs 4.25a & b). In both dendrograms, the stretch between Lowbrook Farm and Colehall (1) in the upper reaches (sites 1 - 5) were grouped as poor abundance and biomass sites and isolated from site 9 in the middle reach. Colehall (site 6) was isolated as a poor abundance and biomass site by both TWINSPAN analyses (Figs 4.25c & d).

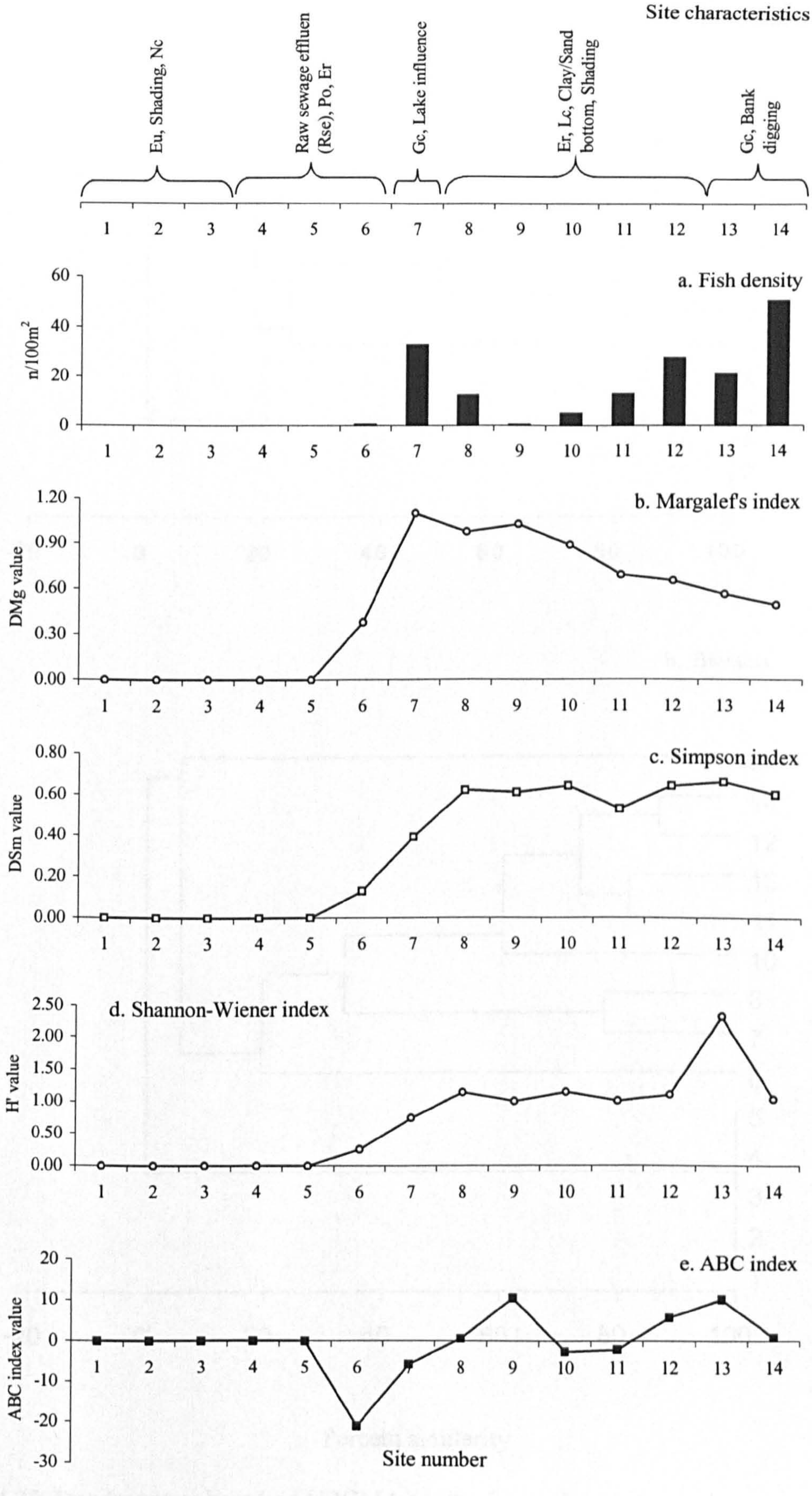


Fig. 4.24 Site characteristics and variations in different indices for the River Cole

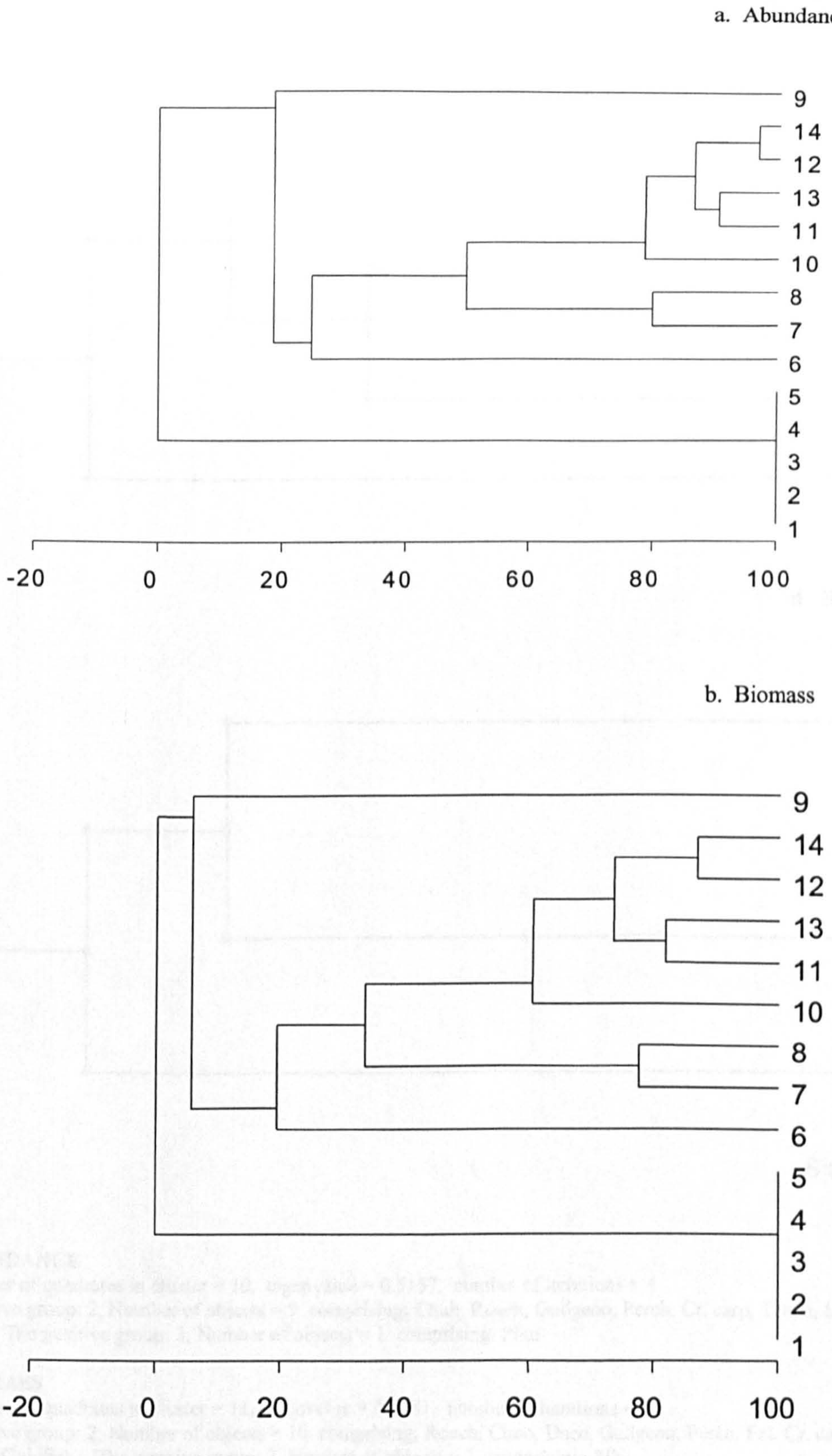
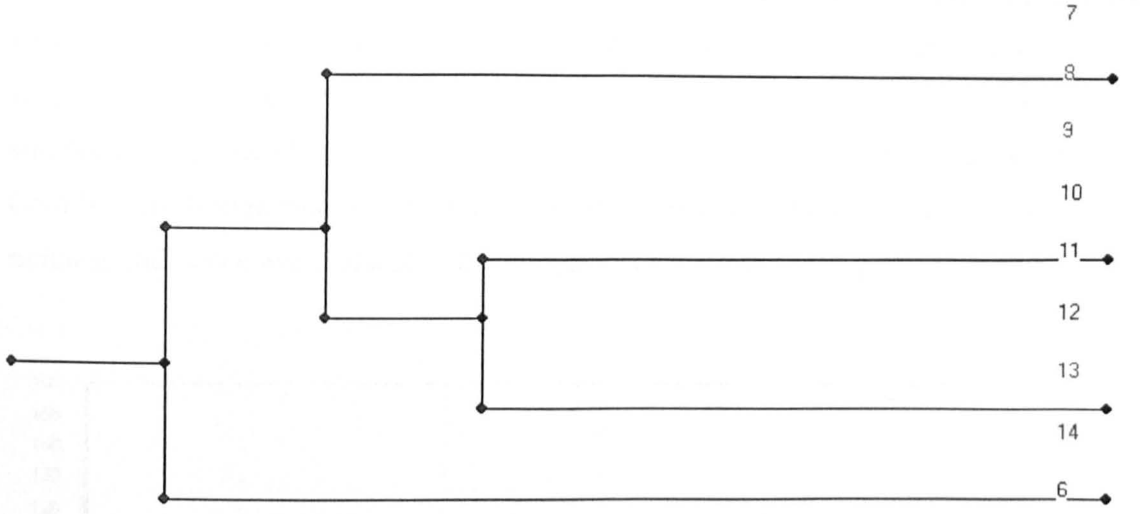
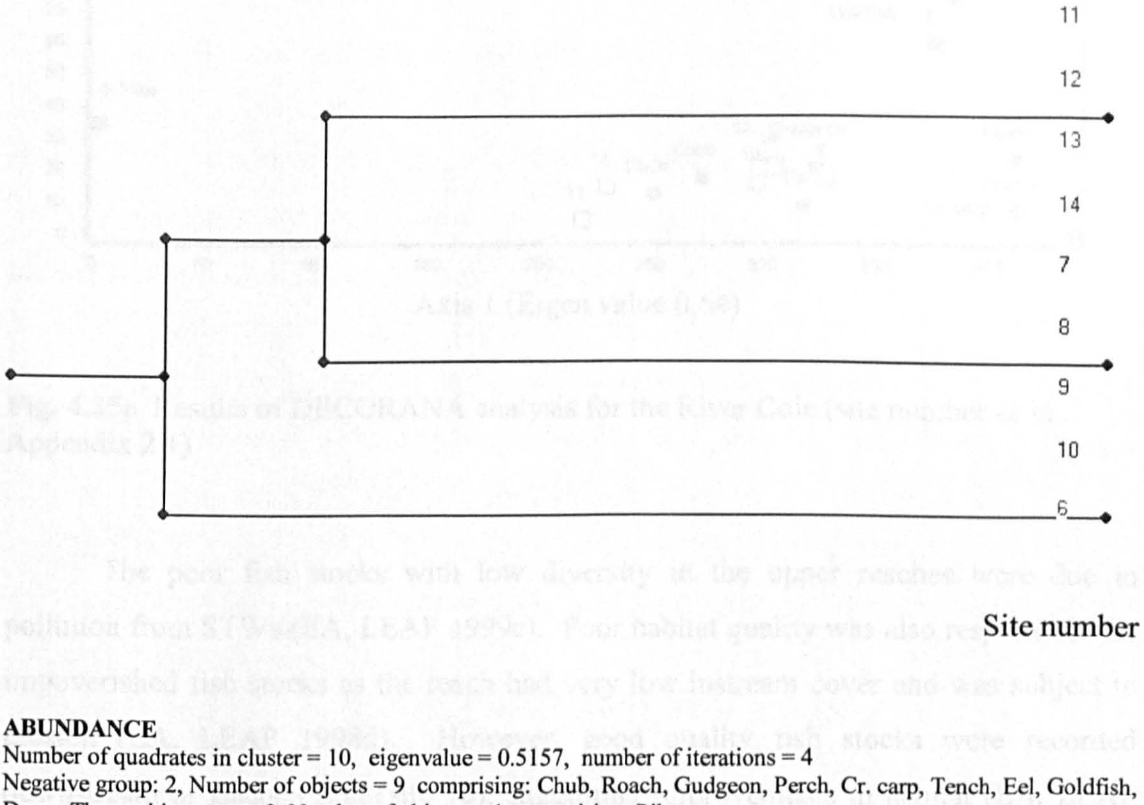


Fig. 4.25 Dendrograms based on UPGMA for the River Cole (site number as in Appendix 2.1)

c. Abundance



d. Biomass



Site number

ABUNDANCE
Number of quadrates in cluster = 10, eigenvalue = 0.5157, number of iterations = 4
Negative group: 2, Number of objects = 9 comprising: Chub, Roach, Gudgeon, Perch, Cr. carp, Tench, Eel, Goldfish, Dace. The positive group: 3, Number of objects = 1 comprising: Pike

BIOMASS
Number of quadrates in cluster = 11, eigenvalue = 0.5481, number of iterations = 5
Negative group: 2, Number of objects = 10 comprising: Roach, Chub, Dace, Gudgeon, Perch, Eel, Cr. carp, Tench, Rudd, Goldfish. The positive group: 3, Number of objects = 1 comprising: Pike

Fig. 4.25 (Continued) Results of TWINSPAN for the River Cole at one level of division (site number as in Appendix 2.1)

DECORANA analysis dispersed Colehall, Kingshurst-1 and D/S Cook's Lane Bridge (sites 6, 7 & 9) in the ordination plot (Fig. 4.25e). Colehall (site 6) had low species richness, density and biomass but a large portion of biomass was incorporated in large pike. Kingshurst-1 (site 7) had a rare species (crucian carp), high species richness and density but low biomass due to a large number of juvenile roach and gudgeon. D/S Cook's Lane Bridge (site 9) was characterised by very low biomass, density and species richness and hence was isolated in the periphery of the ordination plot.

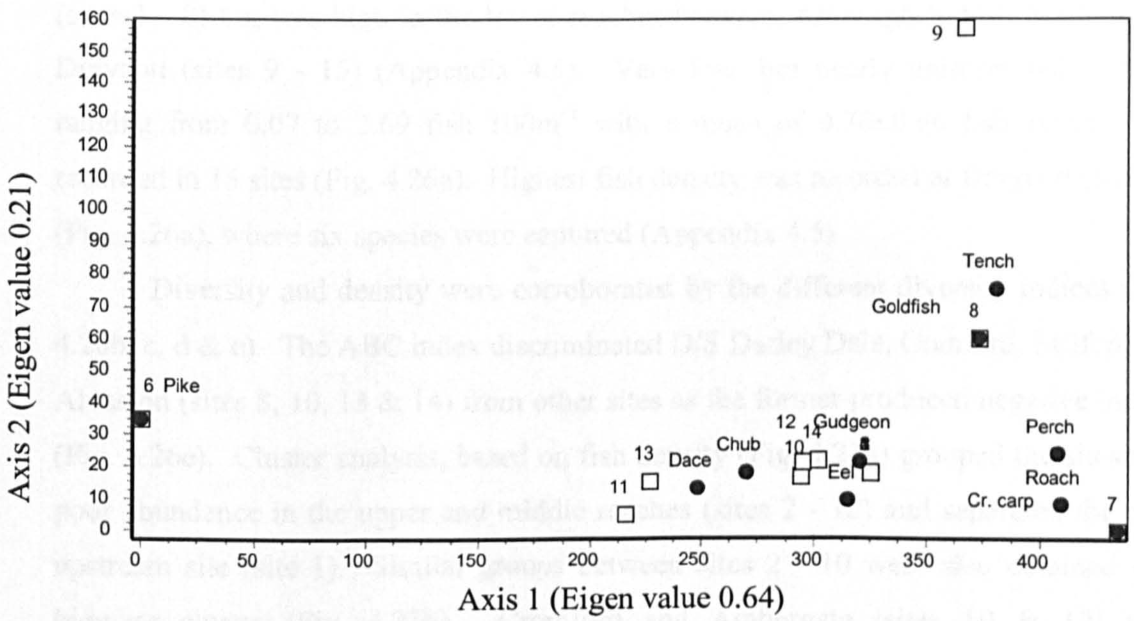


Fig. 4.25e Results of DECORANA analysis for the River Cole (site number as in Appendix 2.1)

The poor fish stocks with low diversity in the upper reaches were due to pollution from STWs (EA, LEAP 1999c). Poor habitat quality was also responsible for impoverished fish stocks as the reach had very low instream cover and was subject to erosion (EA, LEAP 1998d). However, good quality fish stocks were recorded downstream of Bacons End (site 10), suggesting improvements in habitat (EA, LEAP 1999c). Disturbances at D/S Cook's Lane Bridge, Becons End and Coleshill Hospital-1 (sites 9, 10 & 11) were discriminated by the ABC method (Fig. 4.24e). D/S Cook's Lane Bridge had low biomass but there were few small individuals from only a few species, consequently the biomass curve lay above the abundance curve in the analysis (Appendix 4.4). On the other hand, negative indices were found from sites 10 & 11, due to presence of large number of juveniles and low total biomass, consequently the abundance curve lay above the biomass curve in the analysis (Appendix 4.4).

River Derwent

The River Derwent supported 16 species (Appendix 4.1) belonging to 15 genera and 7 families (Appendix 4.2). Grayling and brown trout were present in >70% of sites in the River Derwent (Fig 4.15f). Poor species diversity was recorded in the upper and through the middle reaches between U/S Howden Gauging Weir and D/S Darley Dale (sites 1 - 8) but was high in the lower reaches between Arkwright's Mill Matlock and Draycott (sites 9 - 15) (Appendix 4.5). Very low, but nearly uniform fish density, ranging from 0.07 to 2.69 fish $100m^{-2}$ with a mean of 0.70±0.66 fish $100m^{-2}$, was recorded in 15 sites (Fig. 4.26a). Highest fish density was recorded at Draycott (site 15) (Fig. 4.26a), where six species were captured (Appendix 4.5).

Diversity and density were corroborated by the different diversity indices (Figs 4.26b, c, d & e). The ABC index discriminated D/S Darley Dale, Comford, Milford and Alvaston (sites 8, 10, 13 & 14) from other sites as the former produced negative indices (Fig. 4.26e). Cluster analysis, based on fish density (Fig. 4.27a) grouped the sites with poor abundance in the upper and middle reaches (sites 2 - 10) and separated the most upstream site (site 1). Similar groups between sites 2 - 10 were also obtained with biomass cluster (Fig. 4.27b). Cromford and Ambergate (sites 10 & 12) were discriminated by both TWINSPAN analyses based on abundance and biomass (Figs 4.27c & d). However, Whatstandwell (site 11) was isolated by the biomass analysis (Fig. 4.27d). The DECORANA analysis dispersed U/S Howden Gauging Weir, Alvaston and Draycott (sites 1, 14 & 15) in the periphery of the ordination plot (Fig. 4.27e). U/S Howden Gauging Weir (site 1) had a single species with low density and biomass while Alvaston (site 14) had high density but low biomass due to large numbers of juvenile dace, roach and perch. An abundant bleak population was found in Draycott (site 15) but supported a low biomass due to many juvenile chub and roach.

Due to lack of self-sustaining populations, fish stocks of the River Derwent are maintained by annual restocking. Poor, unsustainable fish stocks in the river were due to natural and human induced disturbances. The river is subject to erosion as the bank is steep and liable to erosion (EA, LEAP 1999a). Weirs hamper upstream migration of fishes. However, poor diversity in the upper reaches is also linked to zonation as the fishery was dominated by rheophilic species.

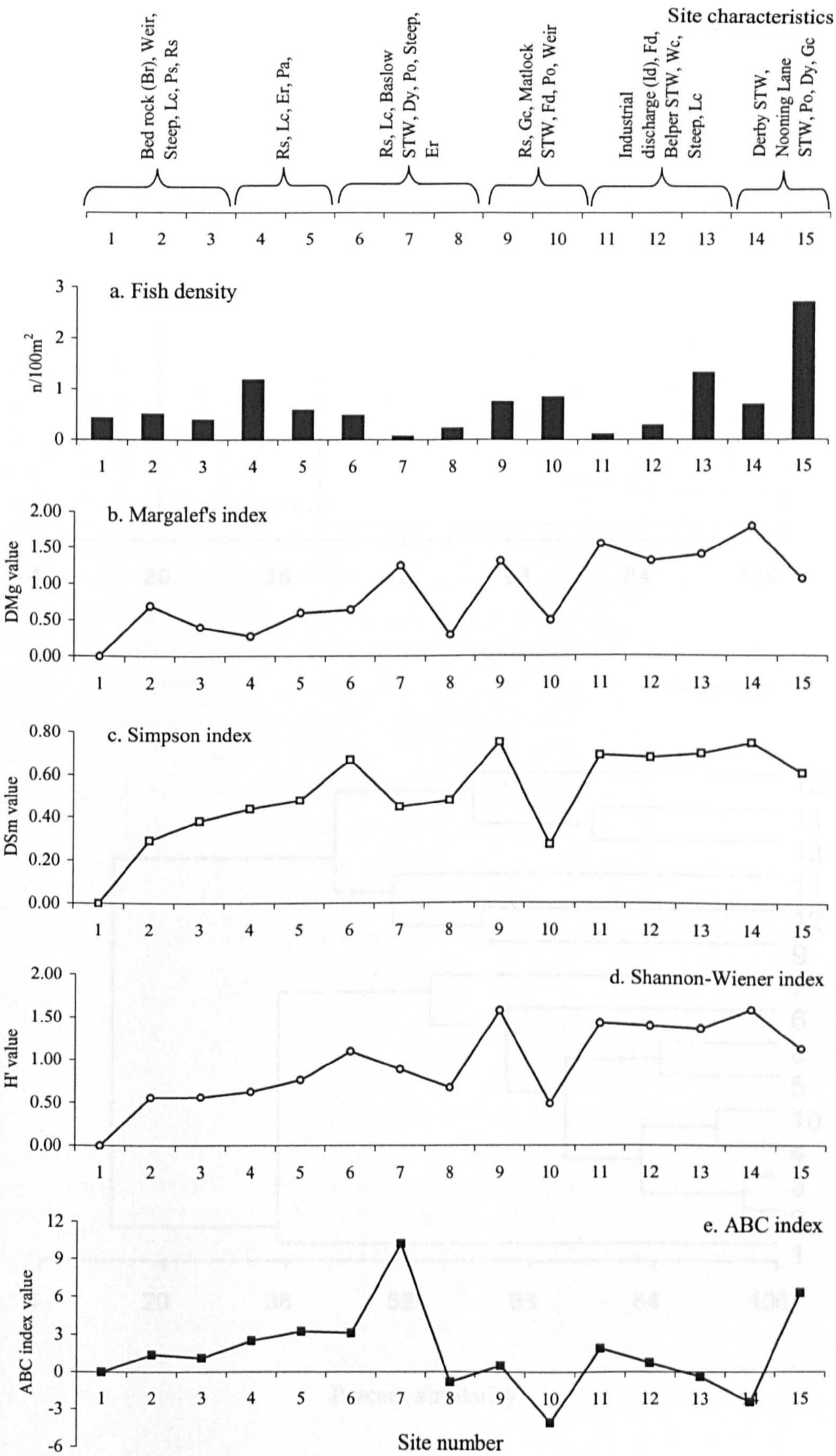


Fig. 4.26 Site characteristics and variations in different indices for the River Derwent

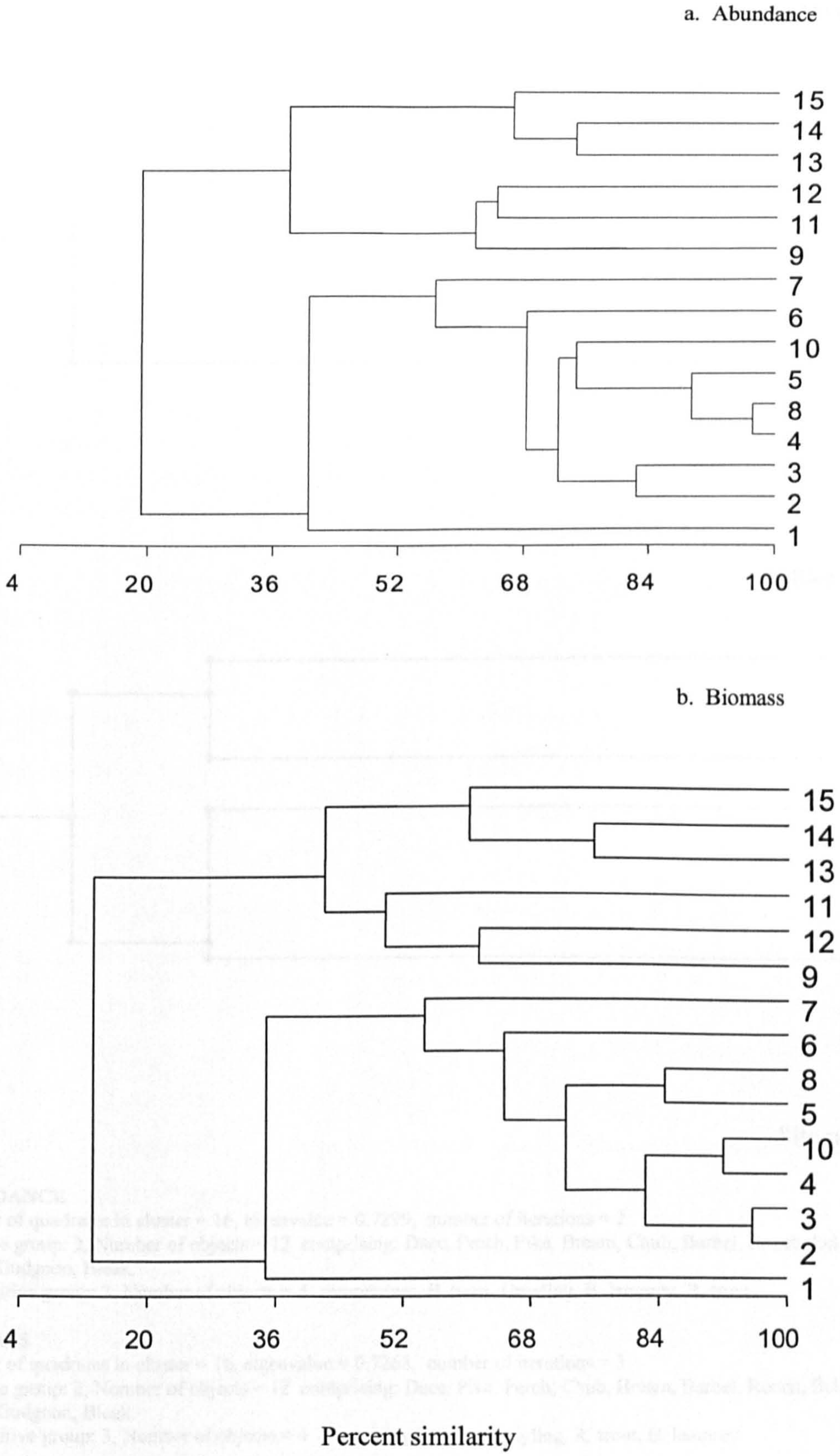
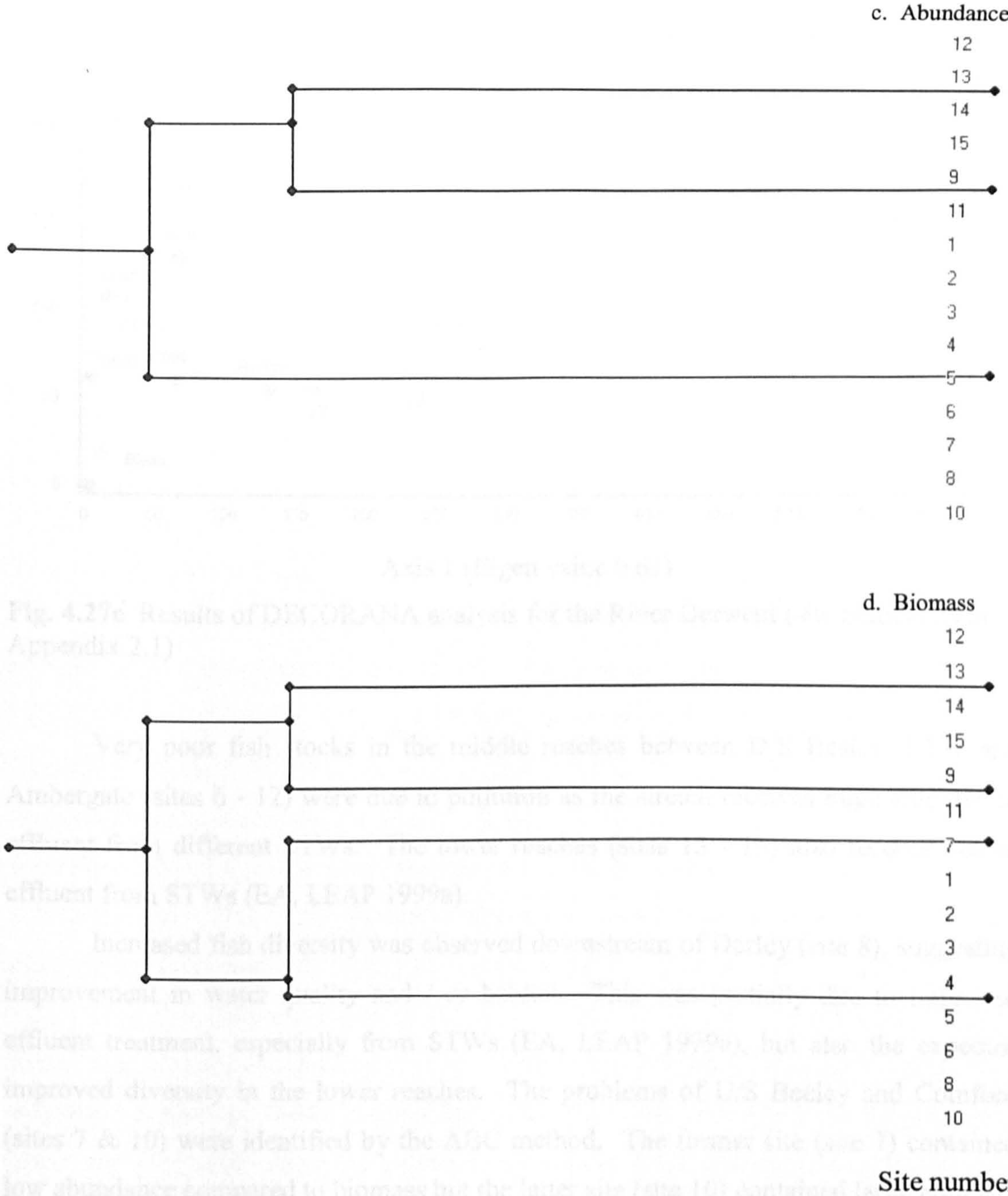


Fig. 4.27 Dendrograms based on UPGMA for the River Derwent (site number as in Appendix 2.1)



ABUNDANCE
Number of quadrates in cluster = 16, eigenvalue = 0.7299, number of iterations = 2
Negative group: 2, Number of objects = 12 comprising: Dace, Perch, Pike, Bream, Chub, Barbel, Roach, Eel, Ruffe, Tench, Gudgeon, Bleak.
The positive group: 3, Number of objects = 4 comprising: B. trout, Grayling, B. lamprey, R. trout

BIOMASS
Number of quadrates in cluster = 16, eigenvalue = 0.7268, number of iterations = 3
Negative group: 2, Number of objects = 12 comprising: Dace, Pike, Perch, Chub, Bream, Barbel, Roach, Eel, Ruffe, Tench, Gudgeon, Bleak.
The positive group: 3, Number of objects = 4 comprising:B. trout, Grayling, R. trout, B. lamprey.

Fig. 4.27 (Continued) Results of TWINSPAN for the River Derwent at one level of division (site number as in Appendix 2.1)

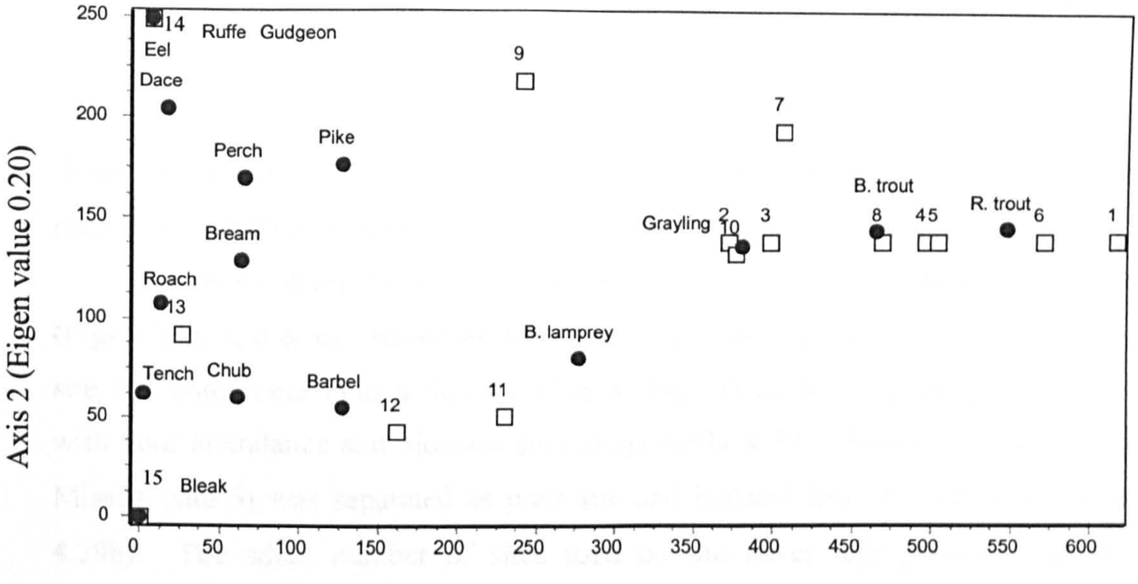


Fig. 4.27e Results of DECORANA analysis for the River Derwent (site number as in Appendix 2.1)

Very poor fish stocks in the middle reaches between D/S Baslow STW and Ambergate (sites 6 - 12) were due to pollution as the stretch receives huge amounts of effluent from different STWs. The lower reaches (sites 13 - 15) also receive treated effluent from STWs (EA, LEAP 1999a).

Increased fish diversity was observed downstream of Darley (site 8), suggesting improvement in water quality and / or habitat. This was partially due to improved effluent treatment, especially from STWs (EA, LEAP 1999a), but also the expected improved diversity in the lower reaches. The problems of U/S Beeley and Comford (sites 7 & 10) were identified by the ABC method. The former site (site 7) contained low abundance compared to biomass but the latter site (site 10) contained large numbers of juvenile grayling (Appendix 4.1).

River Idle

Fish species richness was poor in the River Idle as only 9 species (Appendix 4.1) from 8 genera and 3 families (Appendix 4.2) were caught from five sites. Chub and pike were the dominant species, present at 60% of sampling sites (Fig. 4.15g). Gudgeon, dace, roach, common bream, rudd, bleak and eel were captured from different sites (Appendix 4.5). Poor fish species diversity was observed throughout the river, although moderately high diversity was found at Bawtry (site 4), where six species were captured (Appendix 4.5). No fish were found at Misson (site 5) in the extreme lower reach (Appendix 4.5). Fish density was very low, ranging between 0 and 10.12 fish $100m^{-2}$ with a mean of 2.55 ± 4.27 fish $100m^{-2}$ (Fig. 4.28a), indicating a disturbed community. Highest density was recorded at Tiln (site 2), where only two species were found (Appendix 4.5), however, density was very low at Eaton (site 1) in the upper reaches and Mattersey Priory and Bawtry (sites 3 & 4) in the lower reaches.

Different diversity indices reflected the results on fish diversity and density (Figs 4.28b, c, d & e). However, the ABC index discriminated Tiln (site 2) as a poor site, but Eaton (site 1) as a rich site (Fig. 4.28e). Both dendrograms grouped all sites with poor abundance and biomass sites (Figs 4.29a & b). However, in both analyses Misson (site 5) was separated as poor site and isolated from the other reaches (Fig. 4.29b). The small number of sites used on the River Idle prevented the use of TWINSPAN and DECORNA analyses.

The poor fish stocks (Fig. 4.28a) with low diversity (Appendix 4.5) were due to human induced disturbances (Fig. 4.28). The upper reaches around Eaton and Tiln (sites 1 & 2) were subject to pollution from mine water discharge and sewage effluent from STWs (Fig. 4.28) situated in the Mansfield area (EA, LEAP 1999b). The lower reaches between Mattersey Priory and Misson (sites 3 - 5) were also affected by mine water and effluent discharges from the Worksop area (Fig. 4.28). In addition, the natural habitat of this section had been altered by constructing flood banks. Restocking programmes increased fish density at Tiln (site 2) (EA, LEAP 1999b).

The perturbations at Eaton and Tiln (sites 1 & 2) were identified by the ABC index (Fig. 4.28e). The former site (site 1) contained low abundance from few species compared to biomass. On the other hand, the abundance curve lay above the biomass curve for the latter site (site 2) which contained large numbers of juvenile chub (Appendix 4.4).

River Mease

A total of 14 fish species (Appendix 4.1) including 3 minor species, representing 13 genera and 6 families (Appendix 4.2) were recorded from 7 sites on the River Mease. Common carp, defined as an exotic species, was found in the river. Chub, dace, gudgeon, roach, perch and eel were caught in over 70% of sites, indicating their wide distribution in the river (Fig. 4.15h). Similar, but moderate fish diversity, was recorded throughout the river (Appendix 4.5).

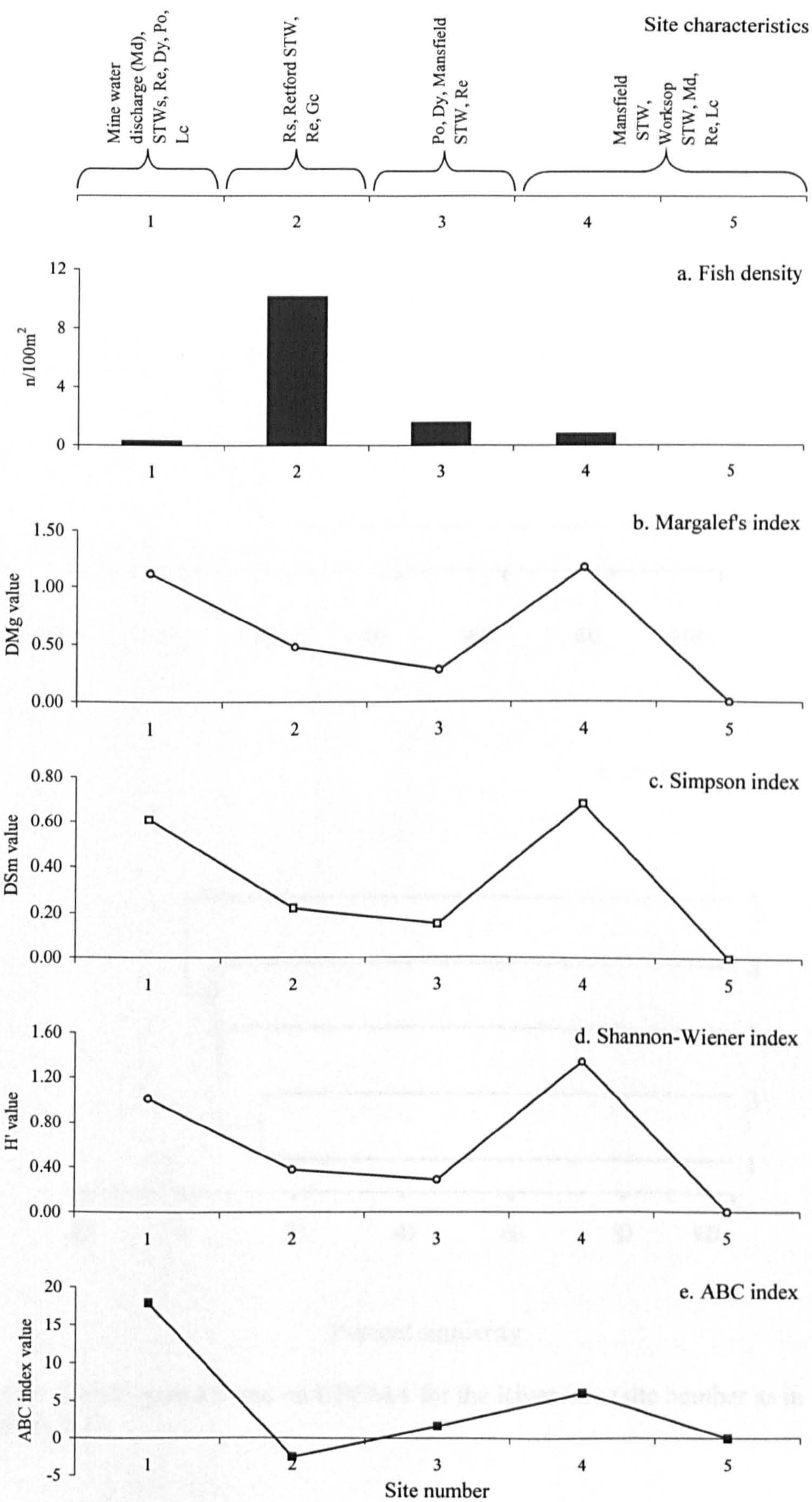


Fig. 4.28 Site characteristics and variations in different indices for the River Idle

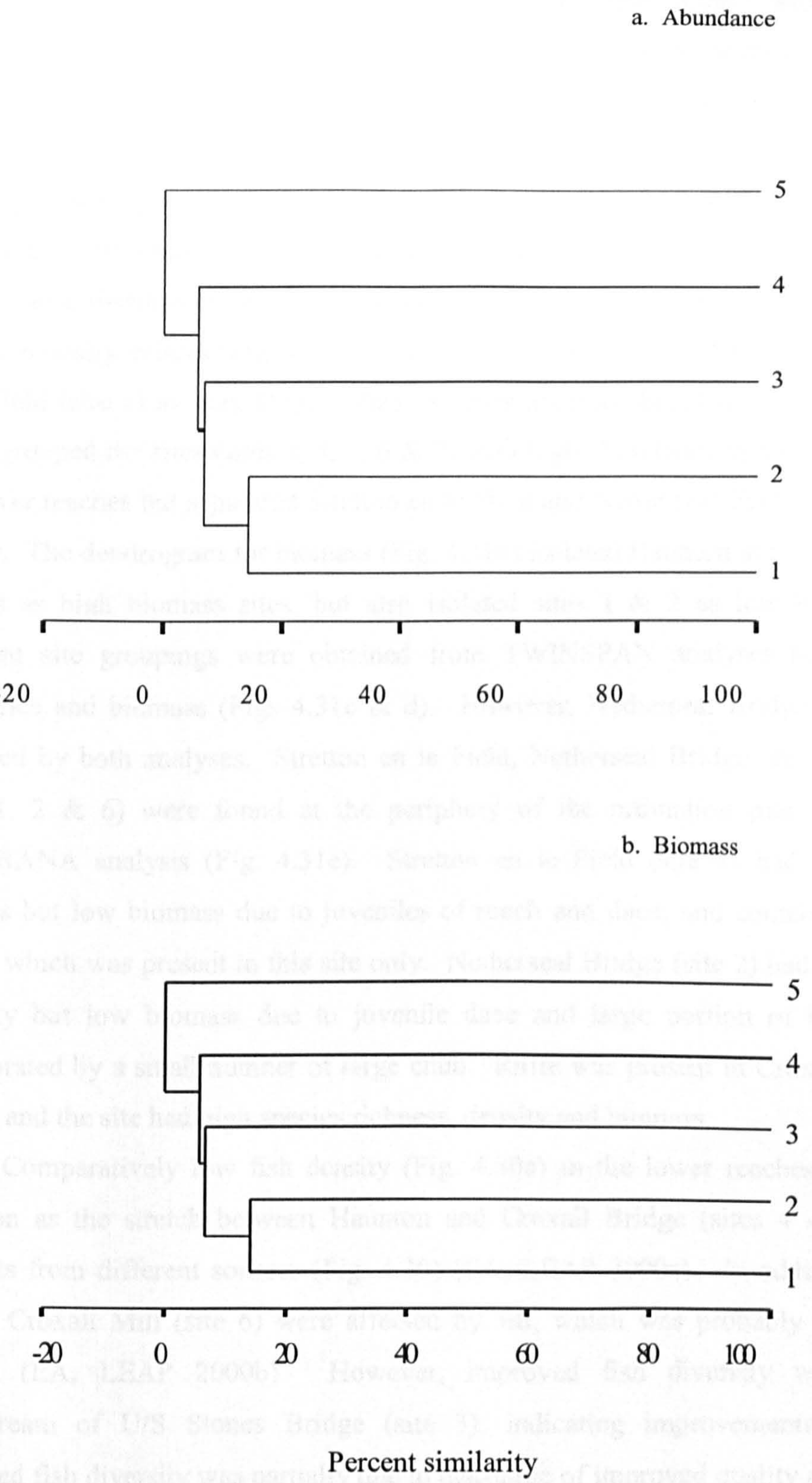


Fig. 4.29 Dendrograms based on UPGMA for the River Idle (site number as in Appendix 2.1)

Fish density ranged between 4.30 and 21.60 fish $100m^{-2}$ with a mean of 8.90 ± 5.90 fish $100m^{-2}$ (Fig. 4.30a) but was fairly stable, suggesting an undisturbed fish community with minimal disturbance from anthropogenic activities. Highest fish density was recorded at Stretton en le Field (site 1) (Fig. 4.30a), where seven species were captured (Appendix 4.5), however, fish density was low in the middle through lower reaches between Haunton and Croxall Bridge (sites 4 - 7).

These findings on the diversity, density and assemblage were supported by the various diversity indices (Figs 4.30b, c, d & e). The ABC index discriminated Stretton en le Field (site 1) as poor (Fig. 4.30e). Cluster analysis, based on fish density (Fig. 4.31a) grouped the sites (sites 3, 4, 5, 6 & 7) with high abundance in the upper, middle and lower reaches but separated Stretton en le Field and Netherseal Bridge (sites 1 & 2) as poor. The dendrogram for biomass (Fig. 4.31b) isolated Haunton and Edingale (sites 4 & 5) as high biomass sites, but also isolated sites 1 & 2 as low biomass sites. Different site groupings were obtained from TWINSPAN analyses based on fish abundance and biomass (Figs 4.31c & d). However, Netherseal Bridge (site 2) was separated by both analyses. Stretton en le Field, Netherseal Bridge and Croxall Mill (sites 1, 2 & 6) were found at the periphery of the ordination plot, obtained by DECORANA analysis (Fig. 4.31e). Stretton en le Field (site 1) had high species richness but low biomass due to juveniles of roach and dace, and contained common bream, which was present in this site only. Netherseal Bridge (site 2) had high species diversity but low biomass due to juvenile dace and large portion of biomass was incorporated by a small number of large chub. Ruffe was present in Croxall Mill (site 6) only and the site had high species richness, density and biomass.

Comparatively low fish density (Fig. 4.30a) in the lower reaches was due to pollution as the stretch between Haunton and Croxall Bridge (sites 4 - 7) receives effluents from different sources (Fig. 4.30) (EA, LEAP 2000a). In addition, reaches around Croxall Mill (site 6) were affected by silt, which was probably due to bank erosion (EA, LEAP 2000b). However, improved fish diversity was recorded downstream of U/S Stones Bridge (site 3), indicating improvements in habitat. Increased fish diversity was partially due to discharge of improved quality effluent from the STW situated upstream near Measham (EA, LEAP 2000b). From the ABC method, the negative index value at Stretton en le Field (site 1) (Fig. 4.30e) was due to large numbers of juvenile roach and dace and few large pike (Appendix 4.4).

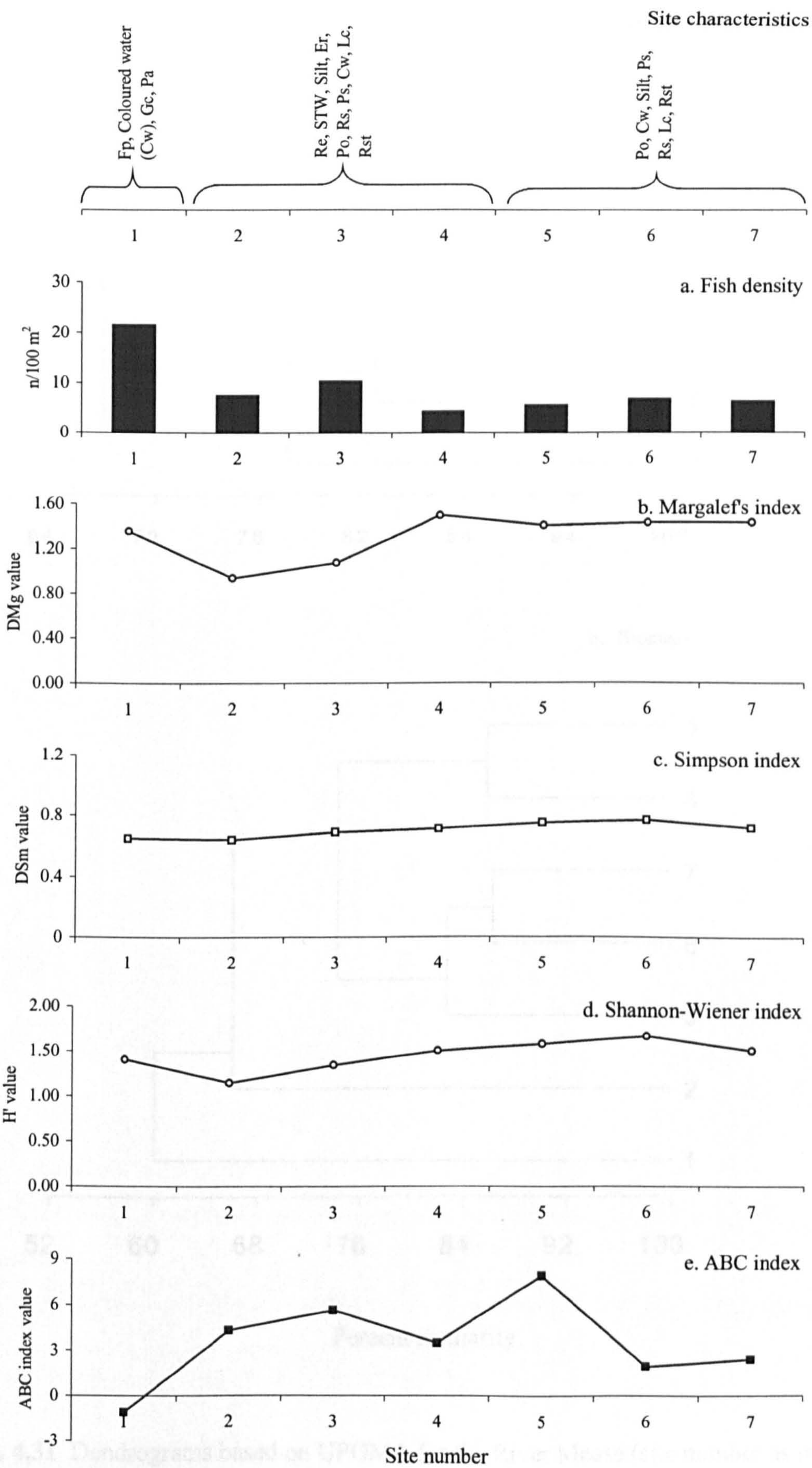


Fig. 4.30 Site characteristics and variations in different indices for the River Mease

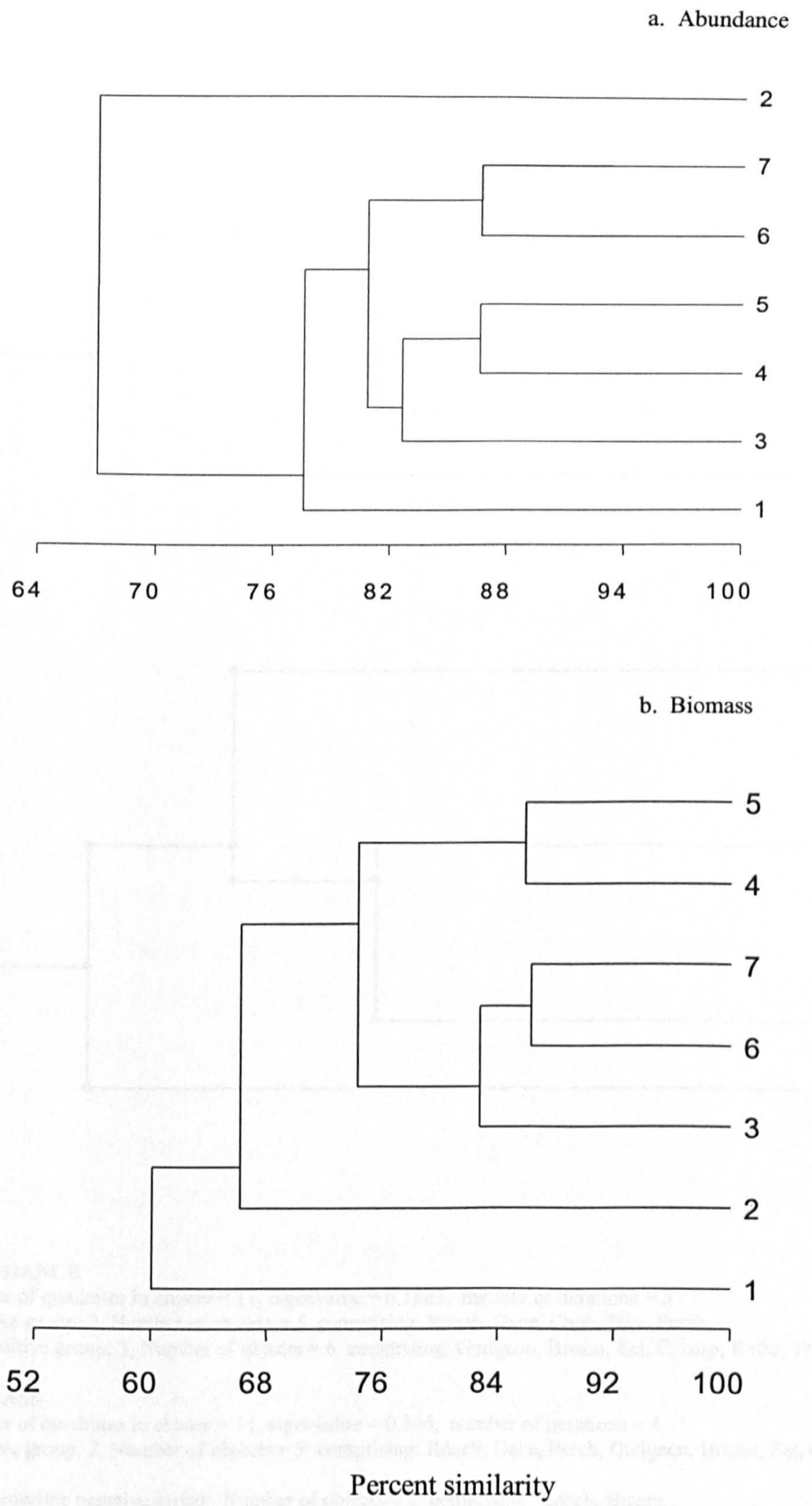
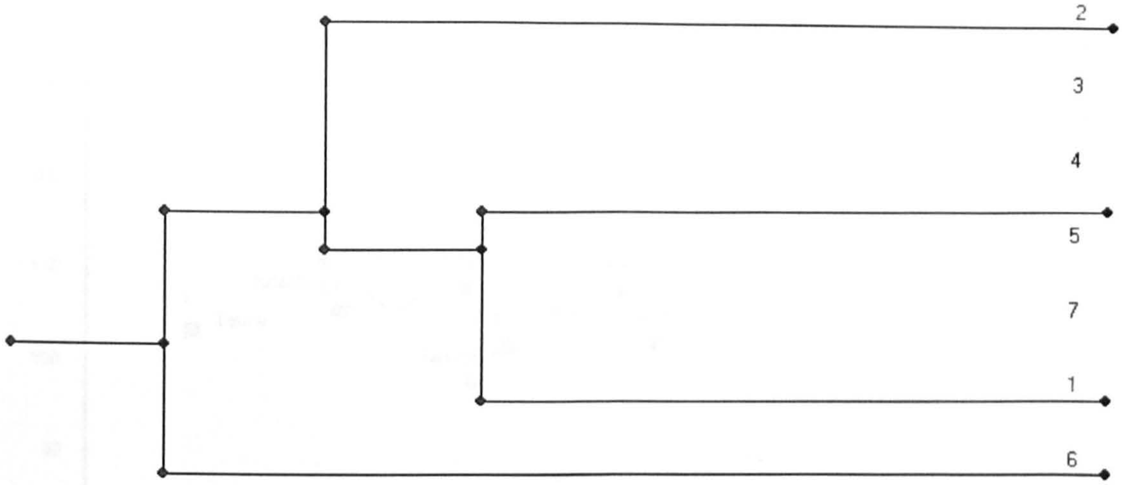
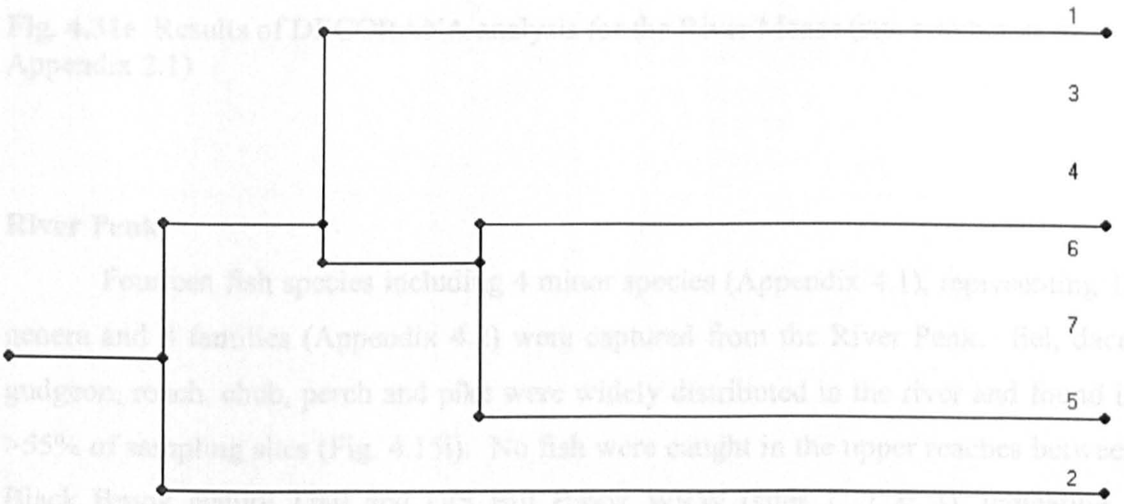


Fig. 4.31 Dendrograms based on UPGMA for the River Mease (site number as in Appendix 2.1)

c. Abundance



d. Biomass



Site number

ABUNDANCE
Number of quadrates in cluster = 11, eigenvalue = 0.1883, number of iterations = 5
Negative group: 2, Number of objects = 5 comprising: Roach, Dace, Chub, Pike, Perch.
The positive group: 3, Number of objects = 6 comprising: Gudgeon, Bream, Eel, C. carp, Ruffe, Tench.

BIOMASS
Number of quadrates in cluster = 11, eigenvalue = 0.306, number of iterations = 4
Negative group: 2, Number of objects = 9 comprising: Roach, Dace, Perch, Gudgeon, Bream, Eel, C. carp, Ruffe, Tench.
The borderline negative group: Number of objects = 2 comprising: Roach, Bream.
The positive group: 3, Number of objects = 2 comprising: Chub, Pike.

Fig. 4.31 (Continued) Results of TWINSPAN for the River Mease at one level of division (site number as in Appendix 2.1)

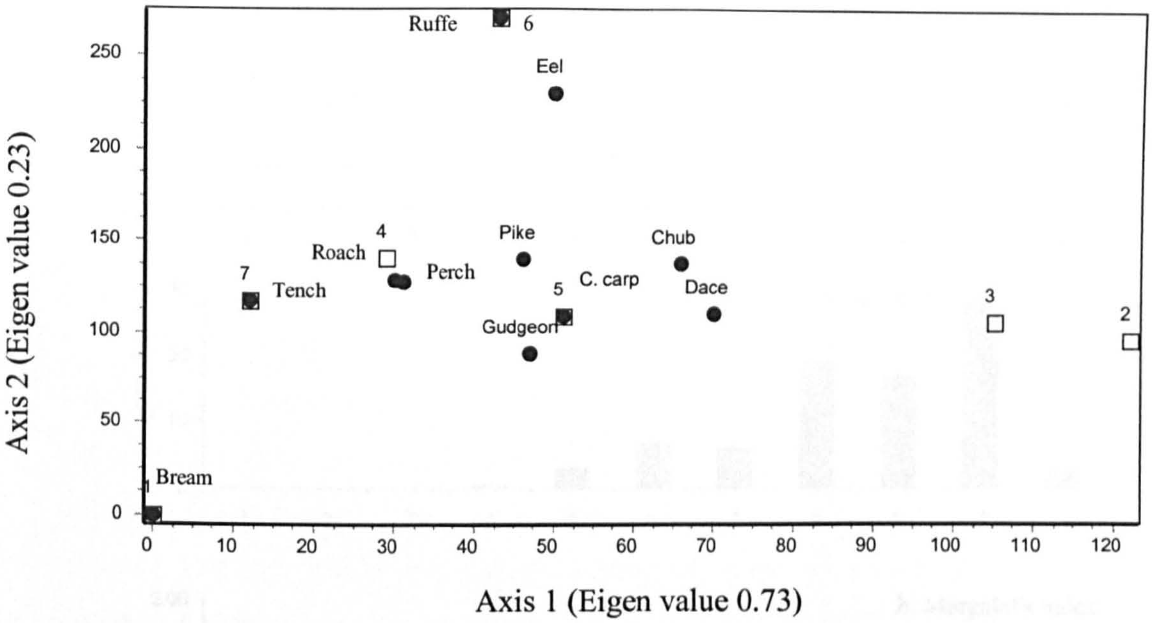


Fig. 4.31e Results of DECORANA analysis for the River Mease (site number as in Appendix 2.1)

River Penk

Fourteen fish species including 4 minor species (Appendix 4.1), representing 13 genera and 8 families (Appendix 4.2) were captured from the River Penk. Eel, dace, gudgeon, roach, chub, perch and pike were widely distributed in the river and found in >55% of sampling sites (Fig. 4.15i). No fish were caught in the upper reaches between Black Brook Nature Trail and U/S Bill Brook WRW (sites 1, 2 & 3), indicating a seriously stressed fish community in this section. Poor species diversity was recorded at D/S Bill Brook WRW and Pendeford Nature Reserve (sites 4 & 5), but was high in the middle through lower reaches between Brewood Park Farm and Radford Bridge (sites 6 & 11) (Appendix 4.5). Fish density was very poor to moderate and varied between 0 and 28.5 fish 100m^{-2} with a mean of 7.66±9.76 fish 100m^{-2} (Fig. 4.32a). Highest density was found at Acton Mill Bridge (site 10) (Fig. 4.32a), where seven species were recorded (Appendix 4.5). However, fish density was very low at D/S Bill Brook WRW and Pendeford Nature Reserve (sites 4 & 5) in the middle reaches.

These findings on the diversity, density and assemblage were corroborated by the various diversity indices (Figs 4.32b, c, d & e). The ABC index discriminated the stretch within Stretton Mill and Radford Bridge (sites 8 - 11) as poor.

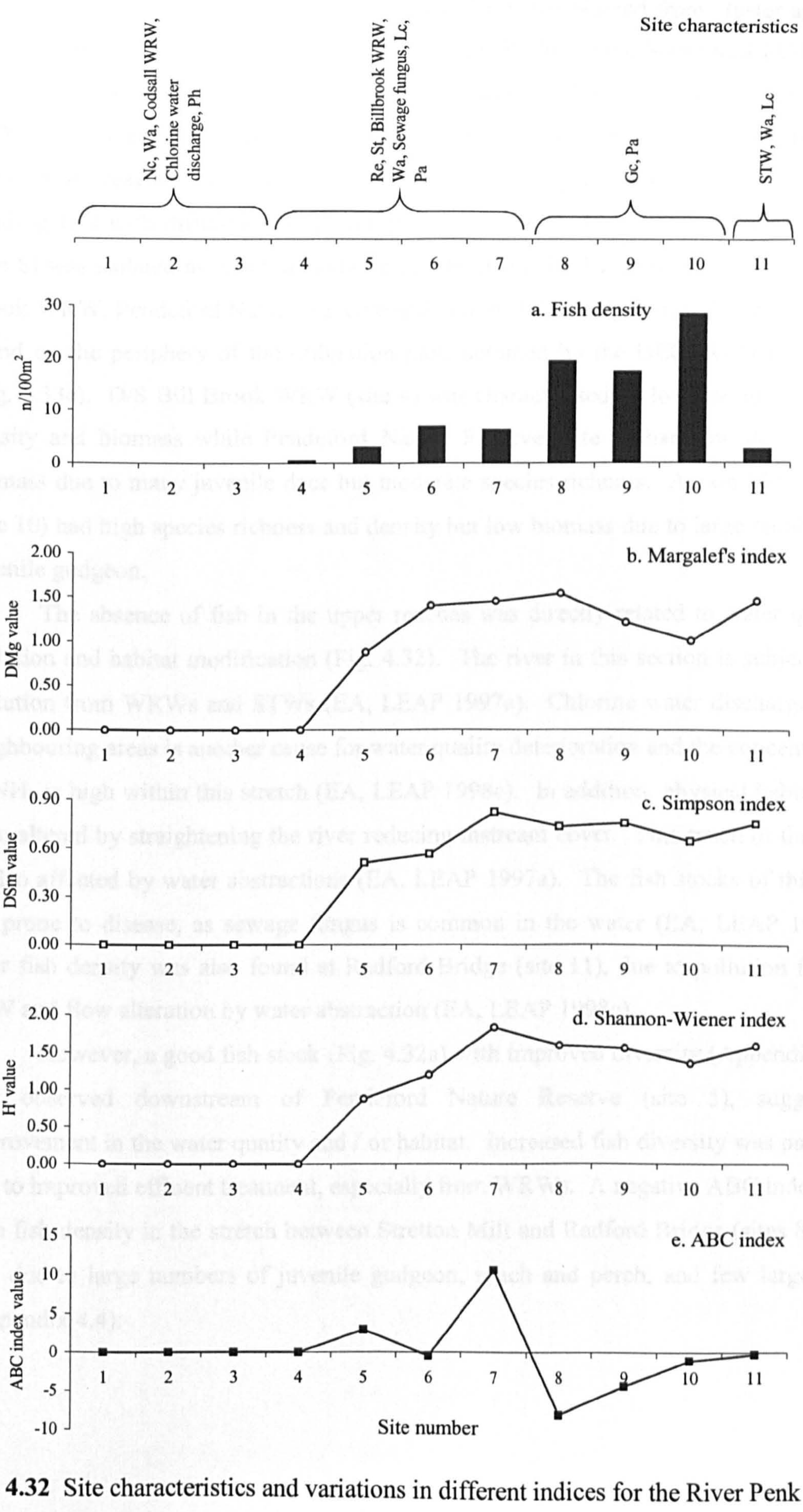


Fig. 4.32 Site characteristics and variations in different indices for the River Penk

Two similar dendrograms (Figs 4.33a & b) were obtained from cluster analyses on the basis of fish density and biomass. In both dendrograms, Somerford Mill Farm, Stretton Mill, Cuttlestone Bridge, Action Mill Bridge and Radford Bridge (sites 7, 8, 9, 10 & 11) were grouped with high abundance and biomass sites and separated from the upper most reaches (sites 1 - 3). TWINSPAN analyses produced two different dendrograms with different site groupings (Figs 4.33c & d). However, Stretton Mill (site 8) was isolated as a rich abundance and biomass site by both analyses. D/S Bill Brook WRW, Pendeford Nature Reserve and Action Mill Bridge (sites 4, 5 & 10) were found on the periphery of the ordination plot, obtained by the DECORANA analysis (Fig. 4.33e). D/S Bill Brook WRW (site 4) was characterised by low species richness, density and biomass while Pendeford Nature Reserve (site 5) had low density and biomass due to many juvenile dace but moderate species richness. Action Mill Bridge (site 10) had high species richness and density but low biomass due to large numbers of juvenile gudgeon.

The absence of fish in the upper reaches was directly related to water quality, pollution and habitat modification (Fig. 4.32). The river in this section is subjected to pollution from WRWs and STWs (EA, LEAP 1997a). Chlorine water discharge from neighbouring areas is another cause for water quality deterioration and the concentration of NH_3 is high within this stretch (EA, LEAP 1998c). In addition, physical habitat has been altered by straightening the river reducing instream cover. This reach of the river is also affected by water abstractions (EA, LEAP 1997a). The fish stocks of this area are prone to disease, as sewage fungus is common in the water (EA, LEAP 1997a). Poor fish density was also found at Radford Bridge (site 11), due to pollution from a STW and flow alteration by water abstraction (EA, LEAP 1998c).

However, a good fish stock (Fig. 4.32a) with improved diversity (Appendix 4.5) was observed downstream of Pendeford Nature Reserve (site 5), suggesting improvement in the water quality and / or habitat. Increased fish diversity was partially due to improved effluent treatment, especially from WRWs. A negative ABC index and high fish density in the stretch between Stretton Mill and Radford Bridge (sites 8 - 11) was due to large numbers of juvenile gudgeon, roach and perch, and few large pike (Appendix 4.4).

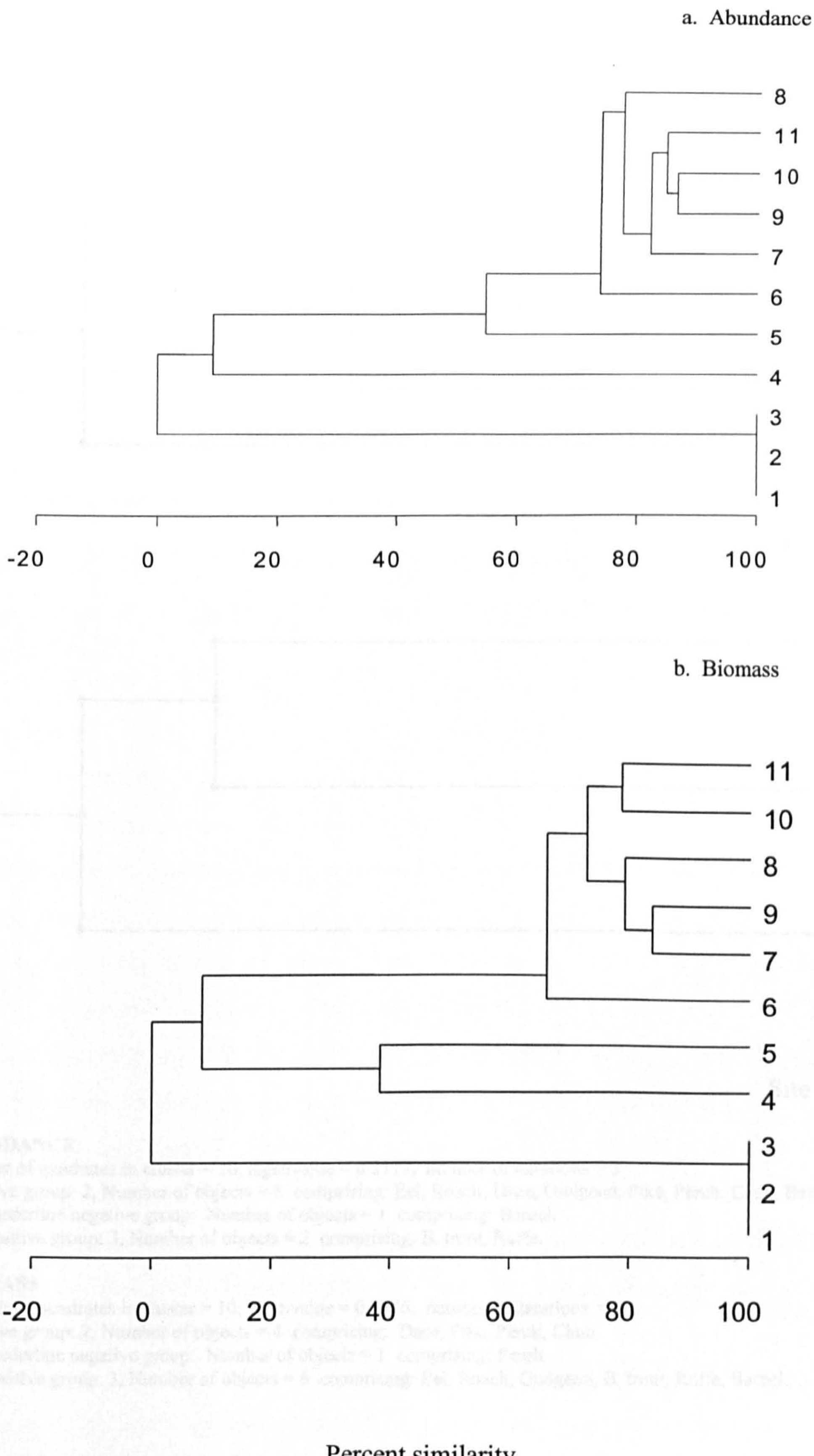
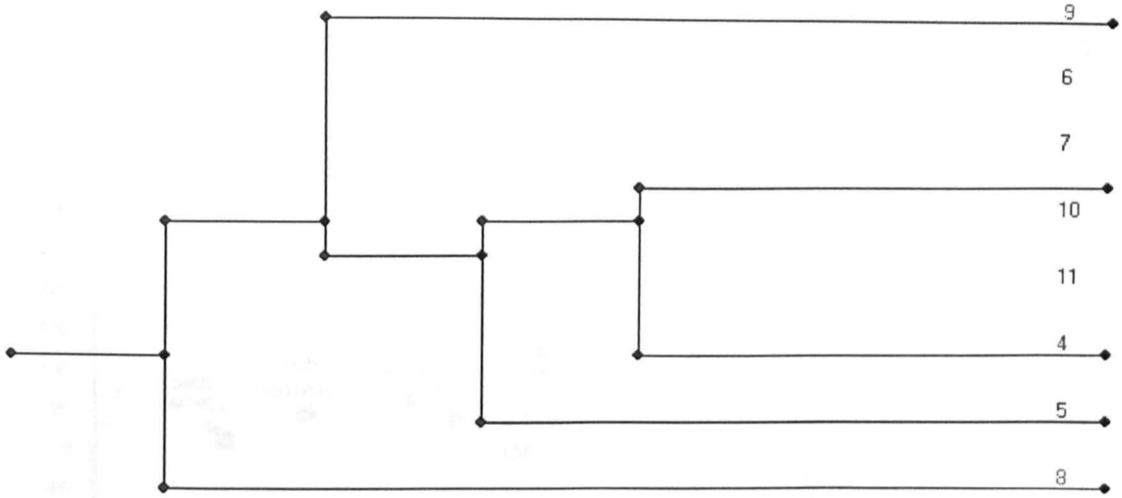
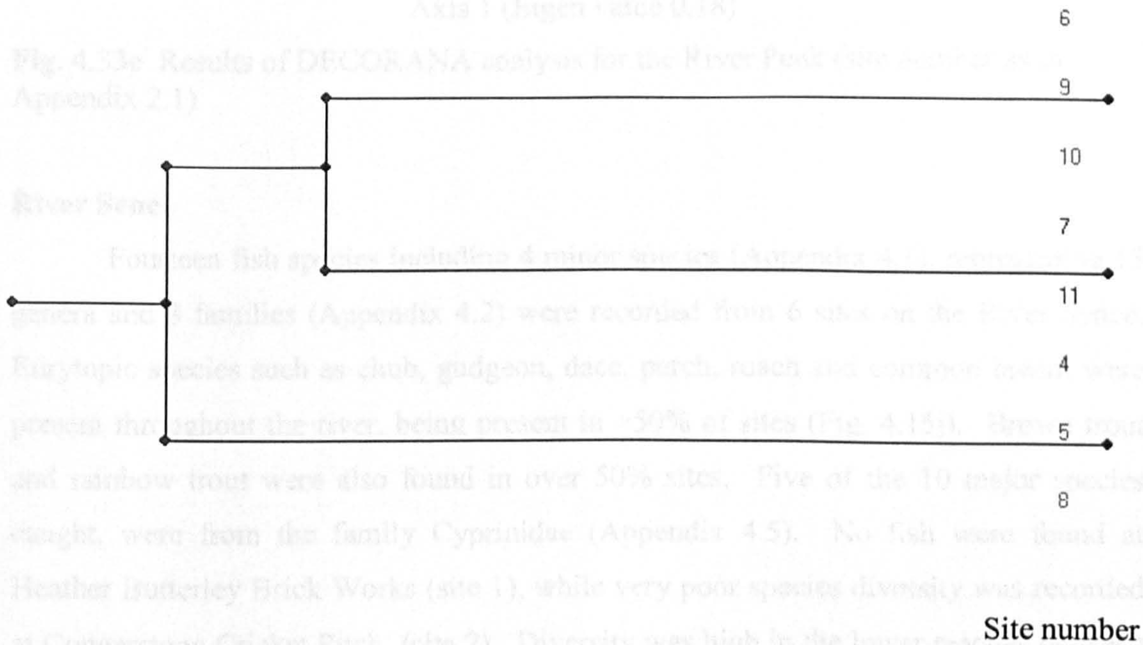


Fig. 4.33 Dendrograms based on UPGMA for the River Penk (site number as in Appendix 2.1)

c. Abundance



d. Biomass



ABUNDANCE
Number of quadrates in cluster = 10, eigenvalue = 0.2117, number of iterations = 3
Negative group: 2, Number of objects = 8 comprising: Eel, Roach, Dace, Gudgeon, Pike, Perch, Chub, Barbel.
The borderline negative group: Number of objects = 1 comprising: Barbel.
The positive group: 3, Number of objects = 2 comprising: B. trout, Ruffe.

BIOMASS
Number of quadrates in cluster = 10, eigenvalue = 0.3086, number of iterations = 3
Negative group: 2, Number of objects = 4 comprising: Dace, Pike, Perch, Chub.
The borderline negative group: Number of objects = 1 comprising: Perch.
The positive group: 3, Number of objects = 6 comprising: Eel, Roach, Gudgeon, B. trout, Ruffe, Barbel.

Fig. 4.33 (Continued) Results of TWINSPAN for the River Penk based on fish abundance and biomass (site number as in Appendix 2.1)

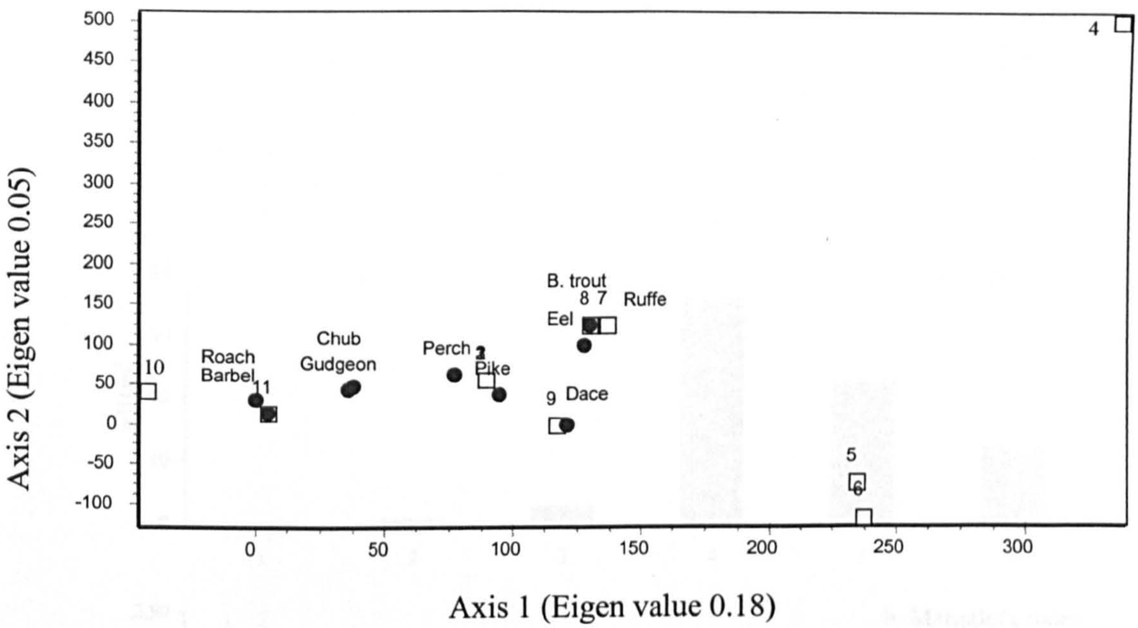


Fig. 4.33e Results of DECORANA analysis for the River Penk (site number as in Appendix 2.1)

River Sence

Fourteen fish species including 4 minor species (Appendix 4.1), representing 13 genera and 8 families (Appendix 4.2) were recorded from 6 sites on the River Sence. Eurytopic species such as chub, gudgeon, dace, perch, roach and common bream were present throughout the river, being present in >50% of sites (Fig. 4.15j). Brown trout and rainbow trout were also found in over 50% sites. Five of the 10 major species caught, were from the family Cyprinidae (Appendix 4.5). No fish were found at Heather Butterley Brick Works (site 1), while very poor species diversity was recorded at Congerstone Cricket Pitch (site 2). Diversity was high in the lower reaches between Congerstone and Ratcliffe Culey Bridge (sites 3 & 6) (Appendix 4.5). Fish densities, ranged from 0 to 36.20 fish $100m^{-2}$ with a mean of 12.53±14.52 fish $100m^{-2}$ (Fig. 4.34a). Highest density was found at Harris Bridge (site 4) (Fig. 4.34a), where nine species were recorded (Appendix 4.5), but was very low at Congerstone Cricket Pitch and Congerstone (sites 2 & 3) in the upper reaches.

These observations on the diversity, density and assemblage were supported by the various diversity indices (Figs 4.34b, c, d & e). The ABC index discriminated Harris Bridge (site 4) as poor (Fig. 4.34e). Based on fish density and biomass, two similar dendrograms were obtained from the cluster analysis (Figs 4.35a & b).

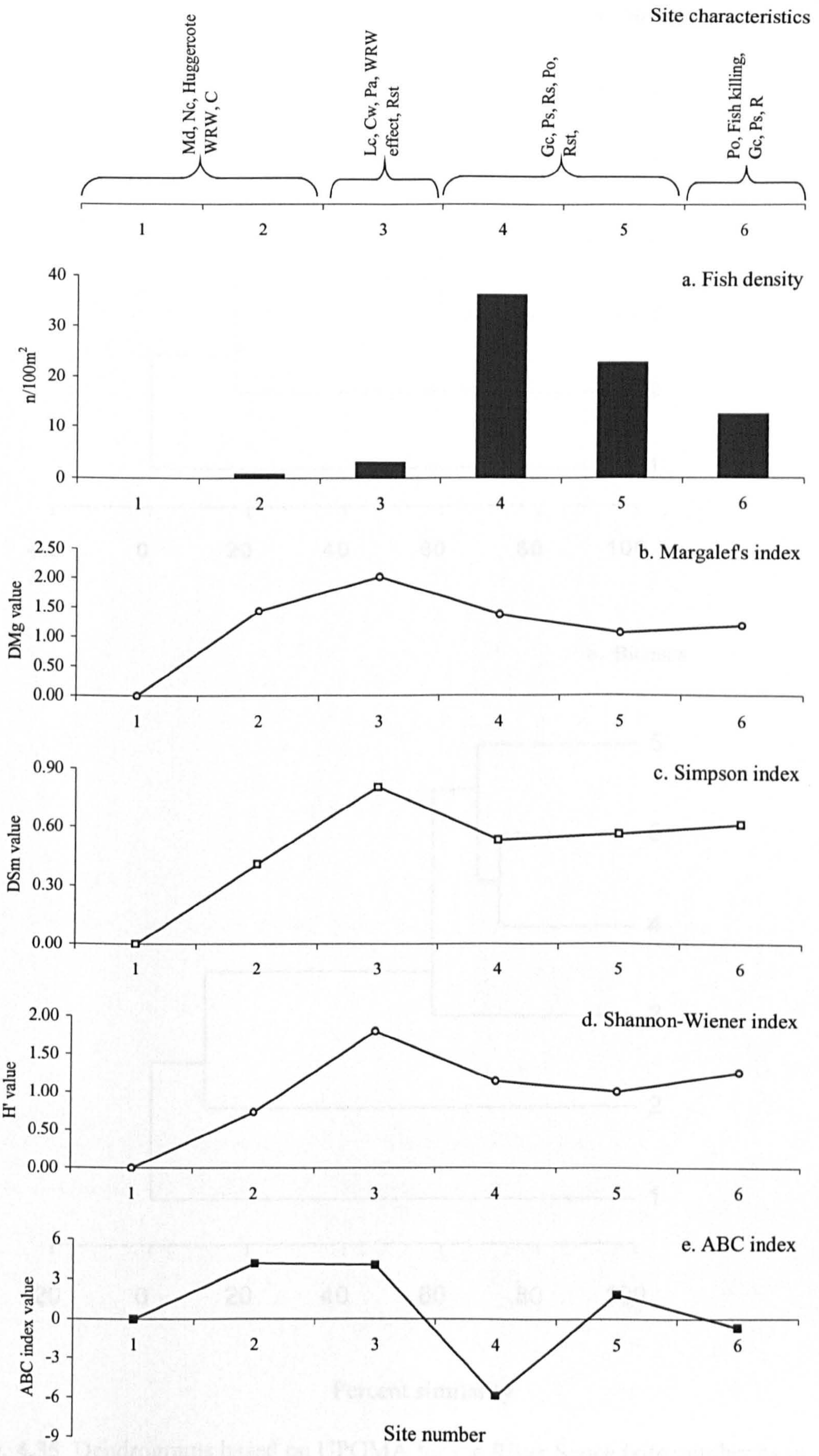


Fig. 4.34 Site characteristics and variations in different indices for the River Sence

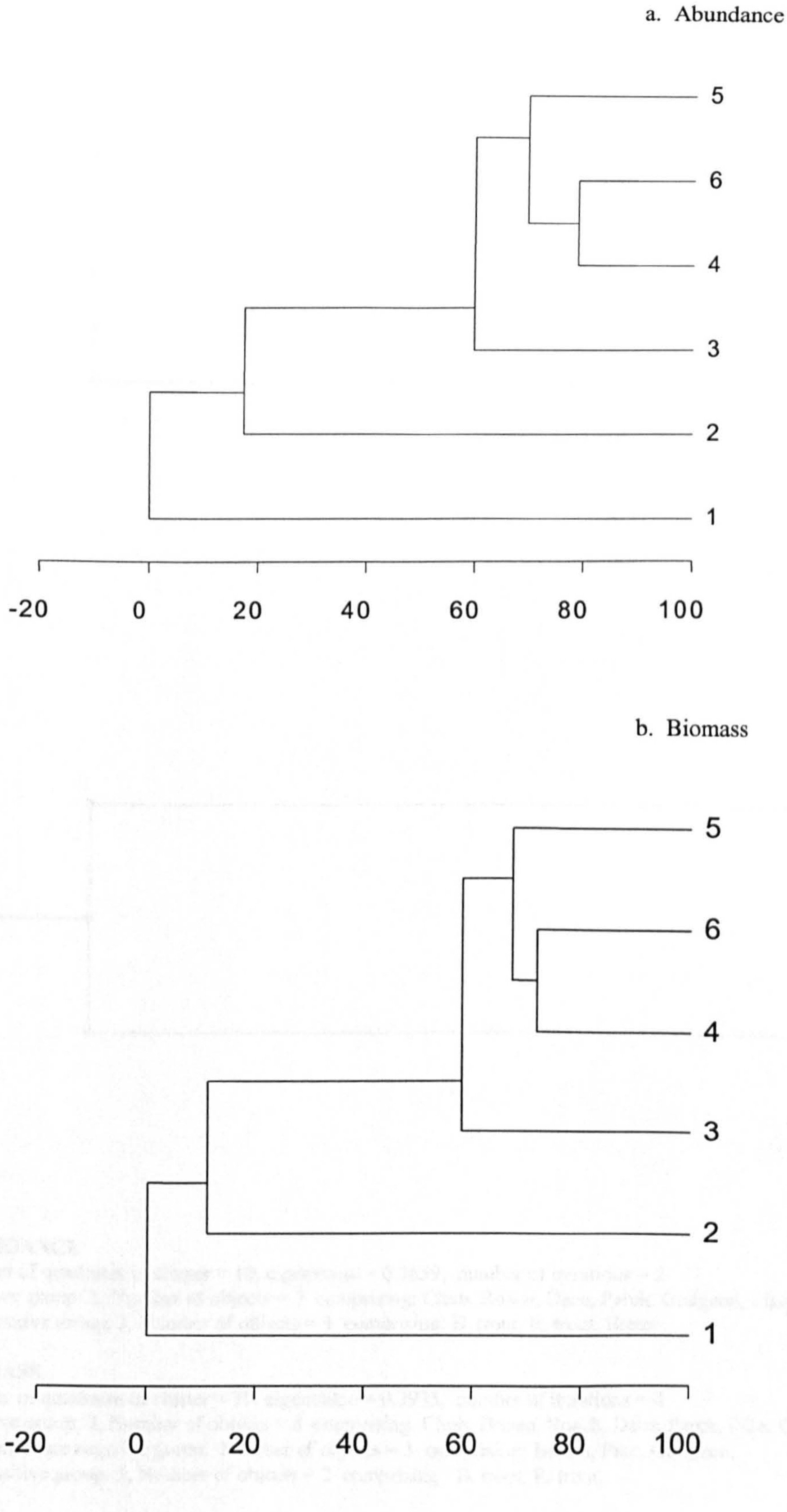
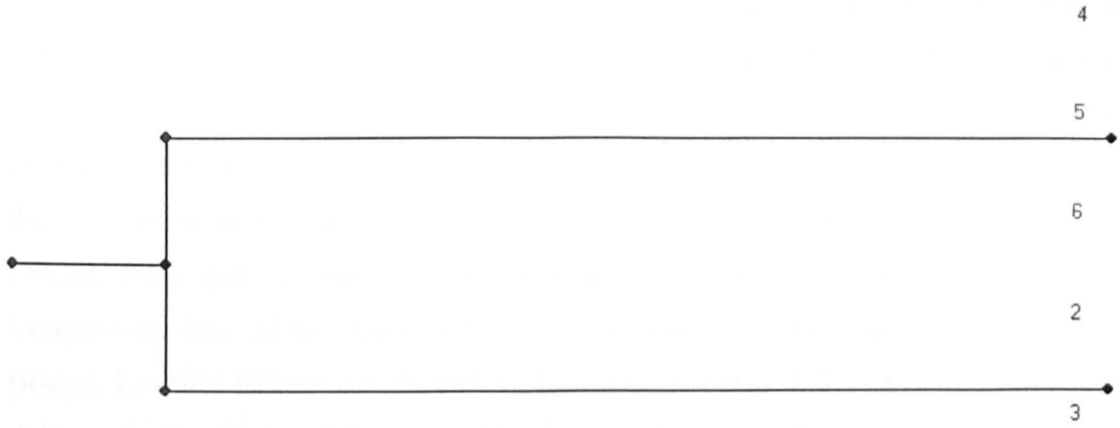
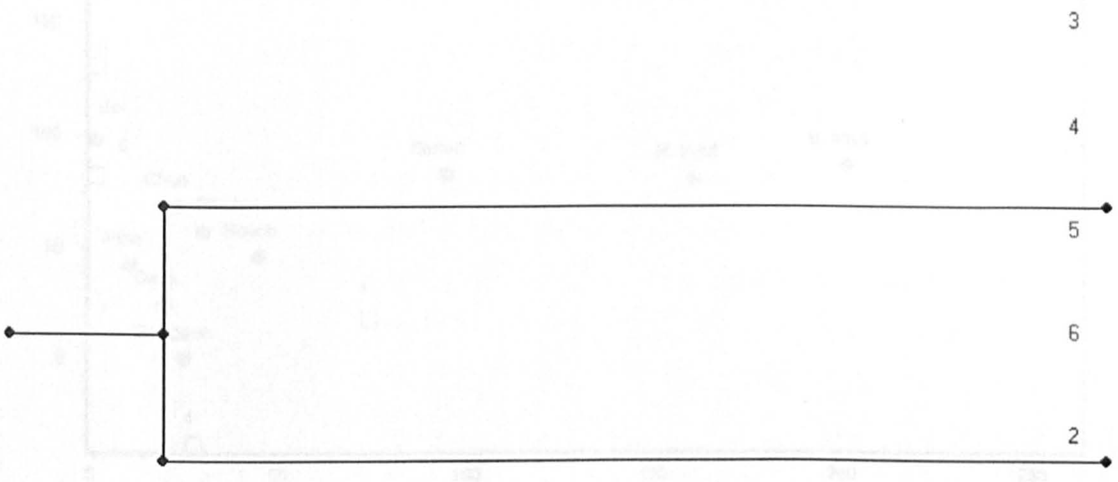


Fig. 4.35 Dendrograms based on UPGMA for the River Sence (site number as in Appendix 2.1)

c. Abundance



d. Biomass



Site number

ABUNDANCE
Number of quadrates in cluster = 10, eigenvalue = 0.3659, number of iterations = 2
Negative group: 2, Number of objects = 7 comprising: Chub, Roach, Dace, Perch, Gudgeon, Pike, Eel.
The positive group: 3, Number of objects = 3 comprising: B. trout, R. trout, Bream.

BIOMASS
Number of quadrates in cluster = 10, eigenvalue = 0.3933, number of iterations = 4
Negative group: 2, Number of objects = 8 comprising: Chub, Bream, Roach, Dace, Perch, Pike, Gudgeon, Eel.
The borderline negative group: Number of objects = 3 comprising: Bream, Pike, Gudgeon.
The positive group: 3, Number of objects = 2 comprising: B. trout, R. trout.

Fig. 4.35 (Continued) Results of TWINSPAN for the River Sence at one level of division (site number as in Appendix 2.1)

In both dendrograms, Heather Butterley Brick Works and Congerstone Cricket Pitch (sites 1 & 2) were grouped with poor abundance and biomass sites while sites 4, 5 & 6 were grouped with rich abundance and biomass sites. The TWINSPAN analysis produced two similar dendrograms and separated Congerstone (site 3) as rich abundance and biomass site (Figs 4.35c & d). All sites were found in the periphery of the ordination plot, obtained by DECORANA analysis (Fig. 4.35e). Congerstone Cricket Pitch (site 2) had only two species with low density and biomass, while Congerstone (site 3) had high species richness but low density and biomass. Harris Bridge, Lovett's Bridge and Ratcliffe Culey Bridge (sites 4, 5 & 6) had high species richness but low biomass due to juvenile dace, chub and roach.

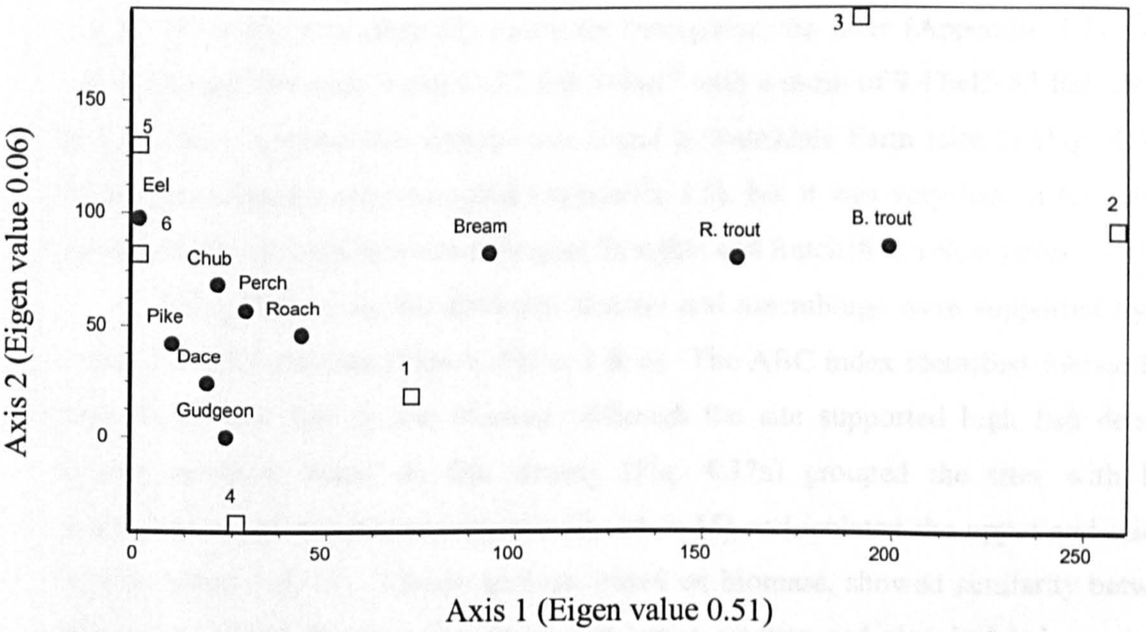


Fig. 4.35e Results of DECORANA analysis for the River Sence (site number as in Appendix 2.1)

Absence of fish in the extreme upper reaches (site 1) was due to pollution and lack of instream cover as the stretch receives mine water discharge and effluent from WRWs (EA, LEAP 2000a). Fish stocks in the upper reaches are maintained by restocking (EA, LEAP 2000b) and good quality fish stocks with improved diversity were recorded downstream of Congerstone (site 3), indicating improvement in the water quality and / or habitat. Increased fish diversity was due to improvement in the effluent quality discharged by the WRWs. High species richness but low biomass at Harris Bridge (site 4) predicted by the ABC method (Fig. 4.34e), was linked to large numbers of juvenile dace, gudgeon, chub and roach and a few large common bream and pike (Appendix 4.4).

River Soar

A total of 14 fish species (Appendix 4.1) belonging to 13 genera, representing 4 families (Appendix 4.2) were captured from the River Soar. Cyprinids, including one exotic species (common carp), contributed 79% of species diversity in the river. Eurytopic species such as perch, roach, dace, chub and pike were the most widespread species throughout the river, being present in >50% of the sites (Fig. 4.15k). No fish were found at Barrow-on-Soar (site 11), while poor species diversity was recorded at Ramsdale Farm (site 1) in the upper reaches and, Blue Bank Lock and D/S Wanlip STW Outfall (sites 6 & 9) in the middle reaches. However, with the above exceptions, species diversity was generally moderate throughout the river (Appendix 4.5). Fish density ranged between 0 and 47.27 fish $100m^{-2}$ with a mean of 9.47 ± 15.83 fish $100m^{-2}$ (Fig. 4.36a). Highest fish density was found at Ramsdale Farm (site 1) (Fig. 4.36a), where three species were recorded (Appendix 4.5), but it was very low in the middle through lower reaches between Leicester Straights and Ratcliffe-on-Soar (sites 7 - 15).

These studies on the diversity, density and assemblage were supported by the various diversity indices (Figs 4.36b, c, d & e). The ABC index identified Jubilee Park (site 5) as poor due to low biomass, although the site supported high fish density. Cluster analysis, based on fish density (Fig. 4.37a) grouped the sites with high abundance in the lower reaches (sites 13, 14 & 15) and isolated the upper and middle reaches (sites 1 & 11). Cluster analysis based on biomass, showed similarity between sites 2, 3, 12, 14 & 15 in the upper and lower reaches and also isolated upper and middle reaches (sites 1 & 11) (Fig. 4.37b). Different site groupings were obtained by the TWINSPAN analysis based on fish abundance and biomass (Figs 4.37c & d). However, both analysis separated Leicester Straights (site 7) as a poor site. The DECORANA analysis discriminated Ramsdale Farm, Jubilee Park and Leicester Straights (sites 1, 5 & 7) in the ordination plot (Fig. 4.37e). Ramsdale Farm (site 1) had high density but low biomass due to juvenile brown trout, while Jubilee Park (site 5) had high species richness and density but low biomass due to juvenile gudgeon and dace. Leicester Straights (site 7) had high species richness but low density and biomass, and crucian carp was found in this site only.

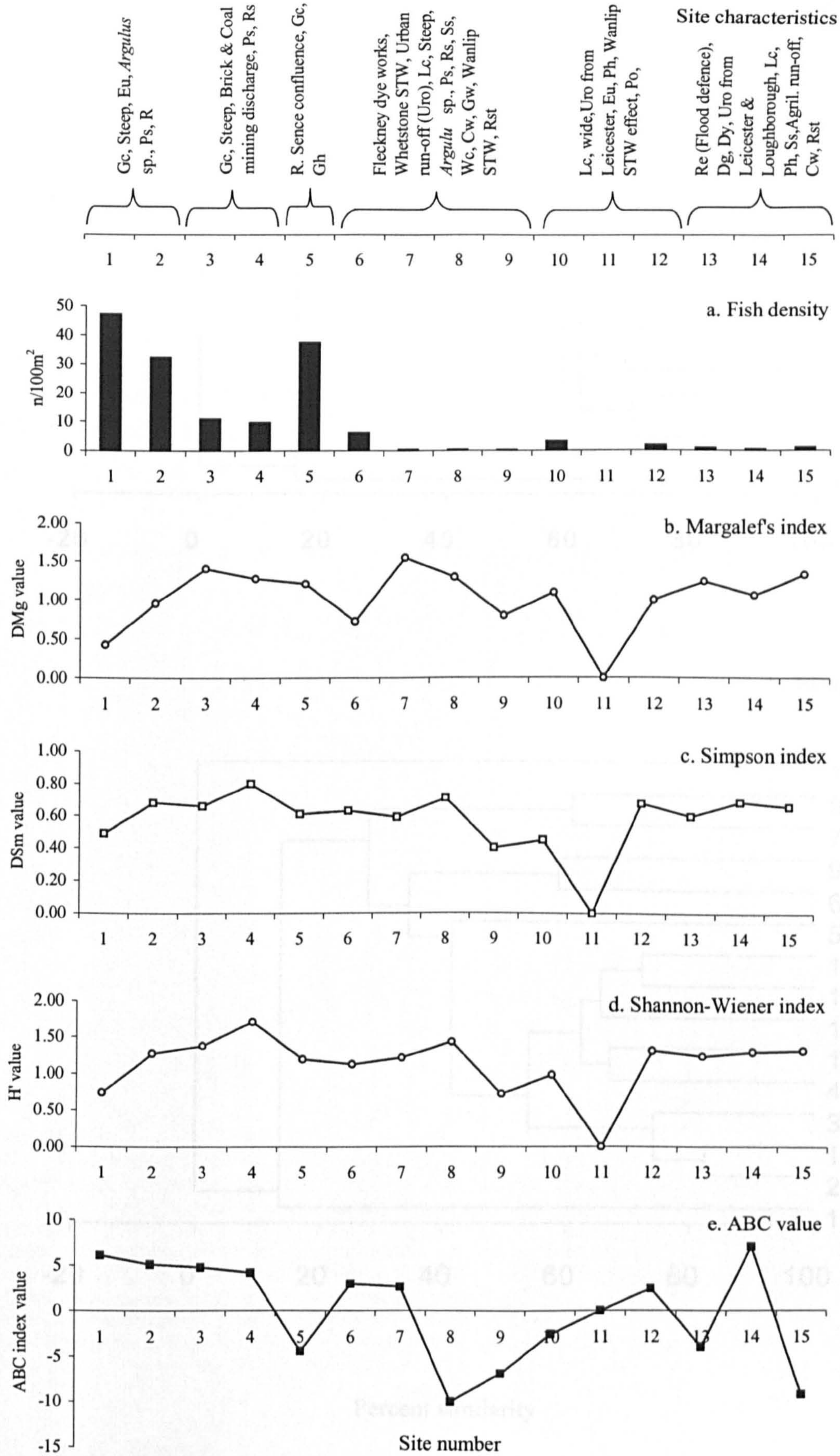
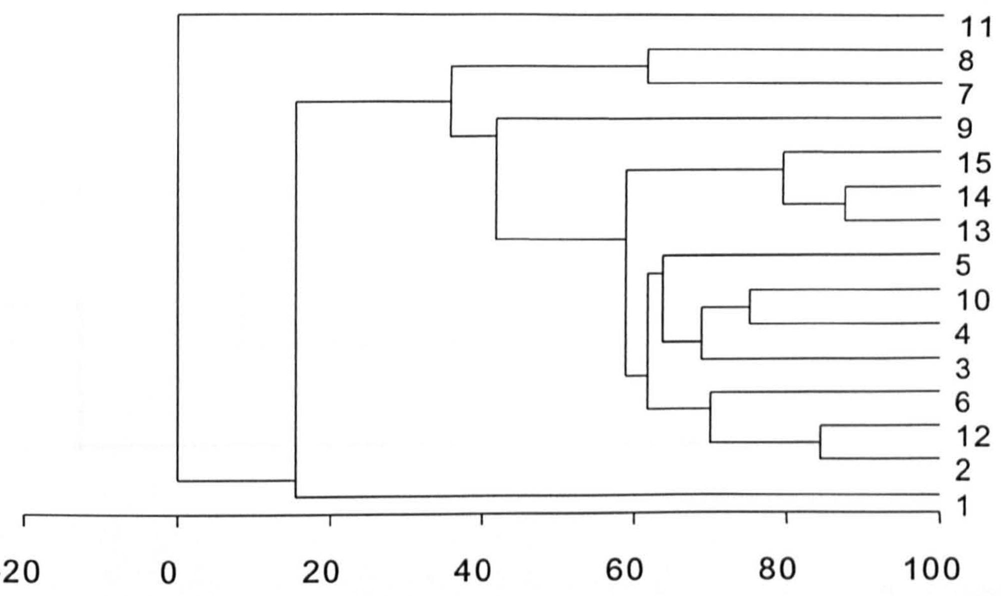
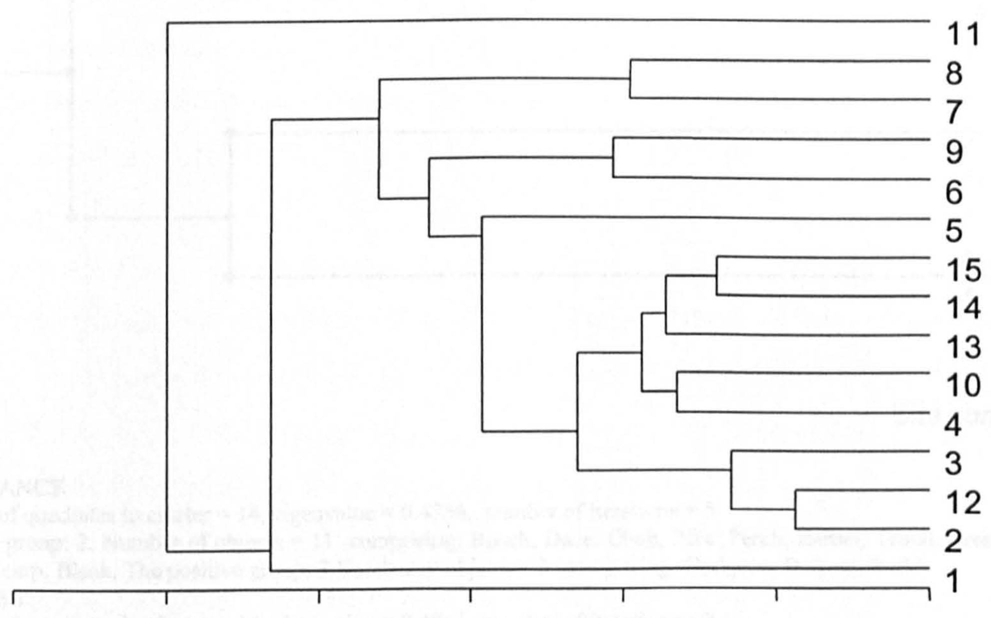


Fig. 4.36 Site characteristics and variations in different indices for the River Soar

a. Abundance



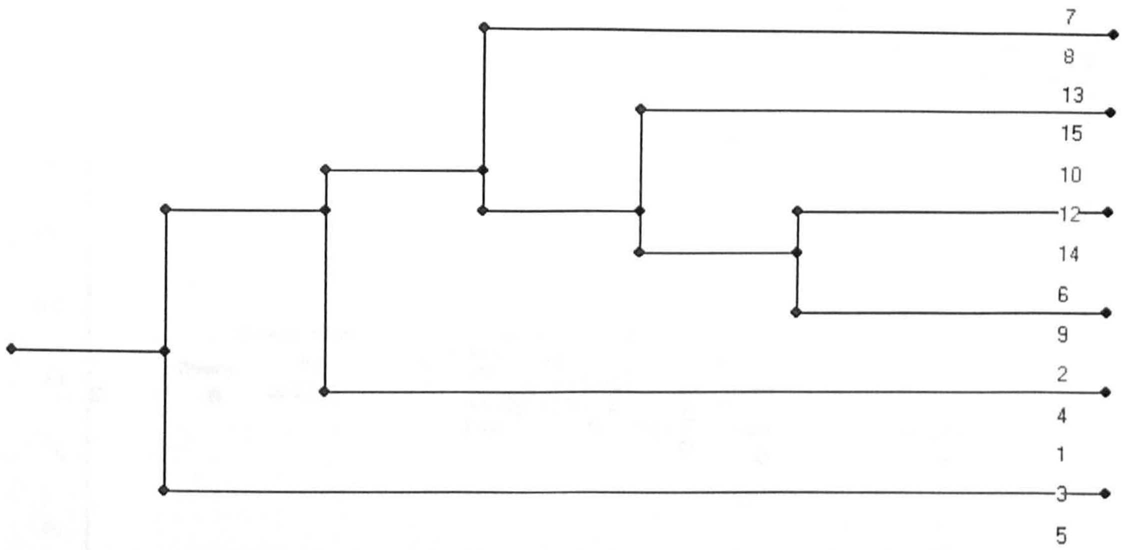
b. Biomass



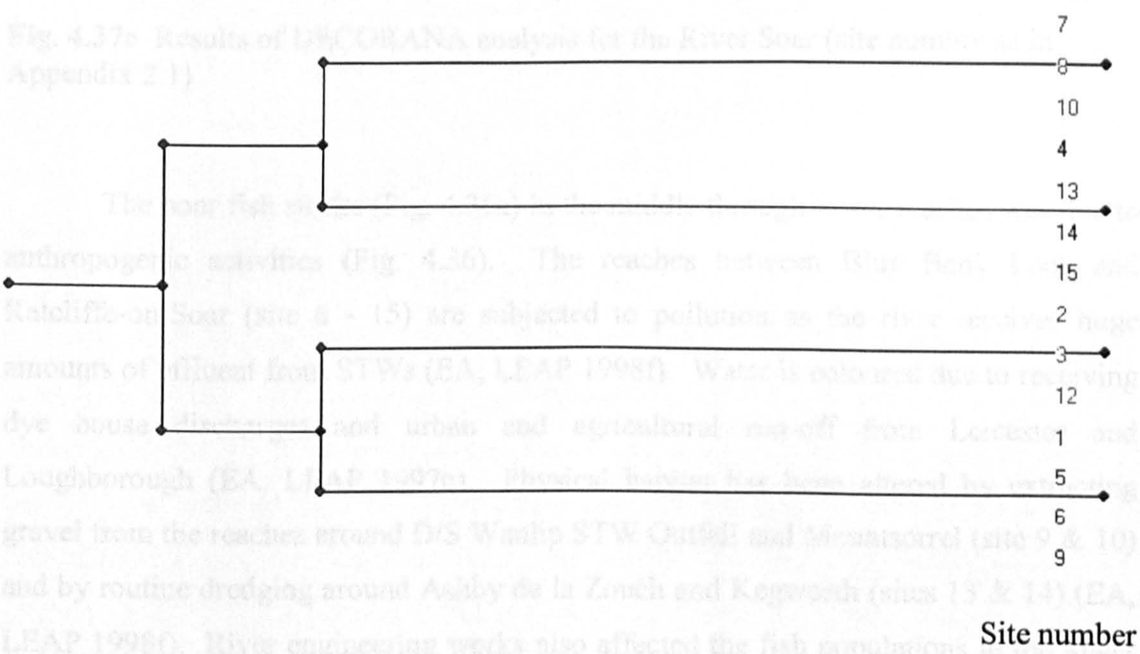
Percent similarity

Fig. 4.37 Dendrograms based on UPGMA for the River Soar (site number as in Appendix 2.1)

c. Abundance



d. Biomass



ABUNDANCE
Number of quadrates in cluster = 14, eigenvalue = 0.4756, number of iterations = 5
Negative group: 2, Number of objects = 11 comprising: Roach, Dace, Chub, Pike, Perch, Barbel, Tench, Bream, C. carp, Cr. carp, Bleak, The positive group: 3 Number of objects = 3 comprising: Gudgeon, B. trout, Rudd.
BIOMASS
Number of quadrates in cluster = 14, eigenvalue = 0.4863, number of iterations = 3
Negative group: 2, Number of objects = 7 comprising: Roach, Chub, Pike, Tench, Bream, C. carp, Cr. carp.
The borderline negative group: Number of objects = 1 comprising: Roach.
The misclassified negatives: Number of objects = 1 comprising: Chub.
The positive group: 3, Number of objects = 7 comprising: B. trout, Dace, Perch, Gudgeon, Barbel, Bleak, Rudd.
The borderline positive group: Number of objects = 1 comprising: Bleak.

Fig. 4.37 (Continued) Results of TWINSPAN for the River Soar at one level of division (site number as in Appendix 2.1)

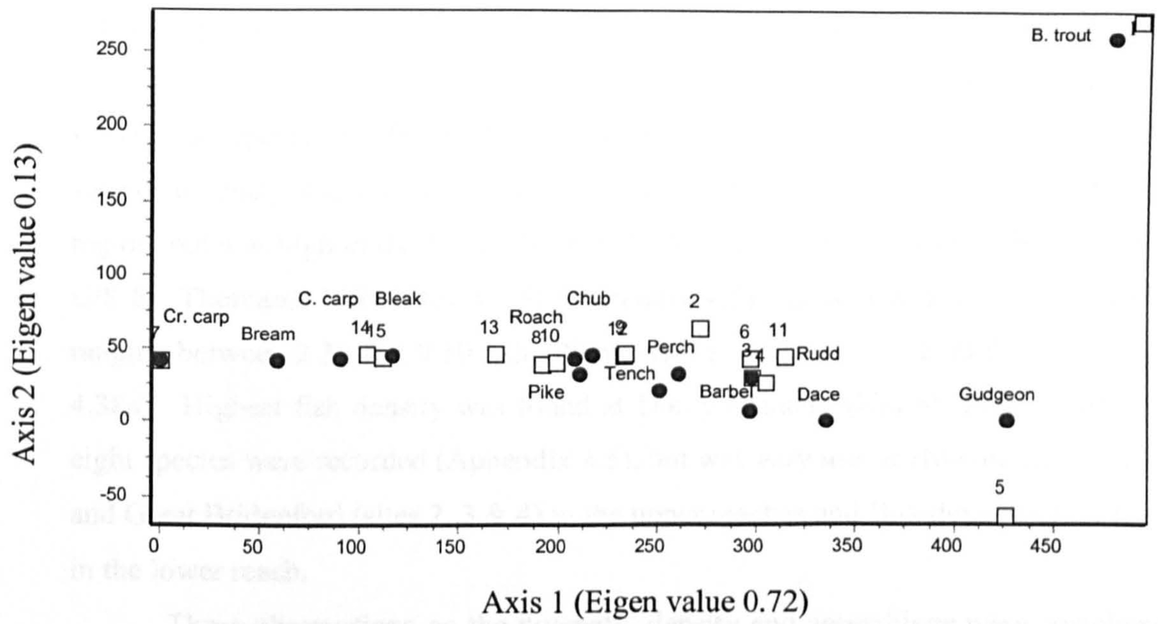


Fig. 4.37e Results of DECORANA analysis for the River Soar (site number as in Appendix 2.1)

The poor fish stocks (Fig. 4.36a) in the middle through lower reaches was due to anthropogenic activities (Fig. 4.36). The reaches between Blue Bank Lock and Ratcliffe-on-Soar (site 6 - 15) are subjected to pollution as the river receives huge amounts of effluent from STWs (EA, LEAP 1998f). Water is coloured due to receiving dye house discharges and urban and agricultural run-off from Leicester and Loughborough (EA, LEAP 1997c). Physical habitat has been altered by extracting gravel from the reaches around D/S Wanlip STW Outfall and Mountsorrel (site 9 & 10) and by routine dredging around Ashby de la Zouch and Kegworth (sites 13 & 14) (EA, LEAP 1998f). River engineering works also affected the fish populations in the lower reaches and because of degradation in the physical habitat and water quality, fish stocks in the lower reaches are maintained by restocking (EA, LEAP 1997c).

Poor water quality appears to have increased the fishes susceptibility to disease as *Argulus* sp. was recorded from cyprinids in the upper through middle reaches of the river (EA, LEAP 1997c). Negative ABC index at Jubilee Park (site 5) was due to large numbers of juvenile gudgeon and dace, and few large pike and tench (Appendix 4.4).

River Sow

Fourteen fish species, including 4 minor species (Appendix 4.1), belonging to 13 genera, representing 7 families (Appendix 4.2) were captured from nine sampling sites on the River Sow. Chub, pike, perch, dace, roach and gudgeon were the most widespread species in the river, being present in >65% of sites (Fig. 4.15l). Low species diversity was recorded at Hillcote Hall and Chebsey (sites 2 & 3) in the upper region, but was high in the middle through lower reaches between Great Bridgeford and U/S St. Thomases Mill (sites 4 - 9) (Appendix 4.5). Low fish density was recorded, ranging between 2.30 and 9.50 fish 100m^{-2} with a mean of 4.80±2.60 fish 100m^{-2} (Fig. 4.38a). Highest fish density was found at Dorey Marshes (site 6) (Fig. 4.38a), where eight species were recorded (Appendix 4.5), but was very low at Hillcote Hall, Chebsey and Great Bridgeford (sites 2, 3 & 4) in the upper reaches and Broadeye Stafford (site 7) in the lower reach.

These observations on the diversity, density and assemblage were corroborated by different diversity indices (Figs 4.38b, c, d & e). The ABC index discriminated Hillcote Hall, Chebsey, Dorey Marshes and Broadeye Stafford (sites 2, 3, 6 & 7) as poor (Fig. 4.38e). Cluster analysis, based on fish density (Fig. 4.39a) grouped Hillcote Hall and Chebsey (sites 2 & 3) with poor abundance sites but failed to include Great Bridgeford (site 4), containing similar fish density. Biomass cluster also grouped sites 2 & 3 with poor biomass sites in the upper reaches and isolated them from the remaining sites (Fig. 4.39b). Different dendrograms based on fish abundance and biomass were obtained by the TWINSPAN analysis (Figs 4.39c & d). However, Broadeye Stafford (site 7) was isolated as a rich abundance and biomass site by both analyses. The DECORANA analysis dispersed Eccleshall Castle, Chebsey and Broadeye Stafford (sites 1, 3 & 7) in the periphery of the ordination plot (Fig. 4.39e). Eccleshall Castle (site 1) had high species richness but low density and biomass while Chebsey (site 3) had low species richness, density and biomass. Broadeye Stafford (site 7) had high species richness but low biomass due to large numbers of juvenile roach.

In general, poor fish density (Fig. 4.38a) in the river was due to pollution, water abstraction and habitat alteration by river engineering works (Fig. 4.38). The river receives treated effluent from WRWs in the upper reaches around Eccleshall Castle (site 1) (EA, LEAP 1997a). Dense weed growth resulting from eutrophication also affected the fishery within the stretch between Chebsey and Great Bridgeford (site 3 & 4) (EA, LEAP 1998c).

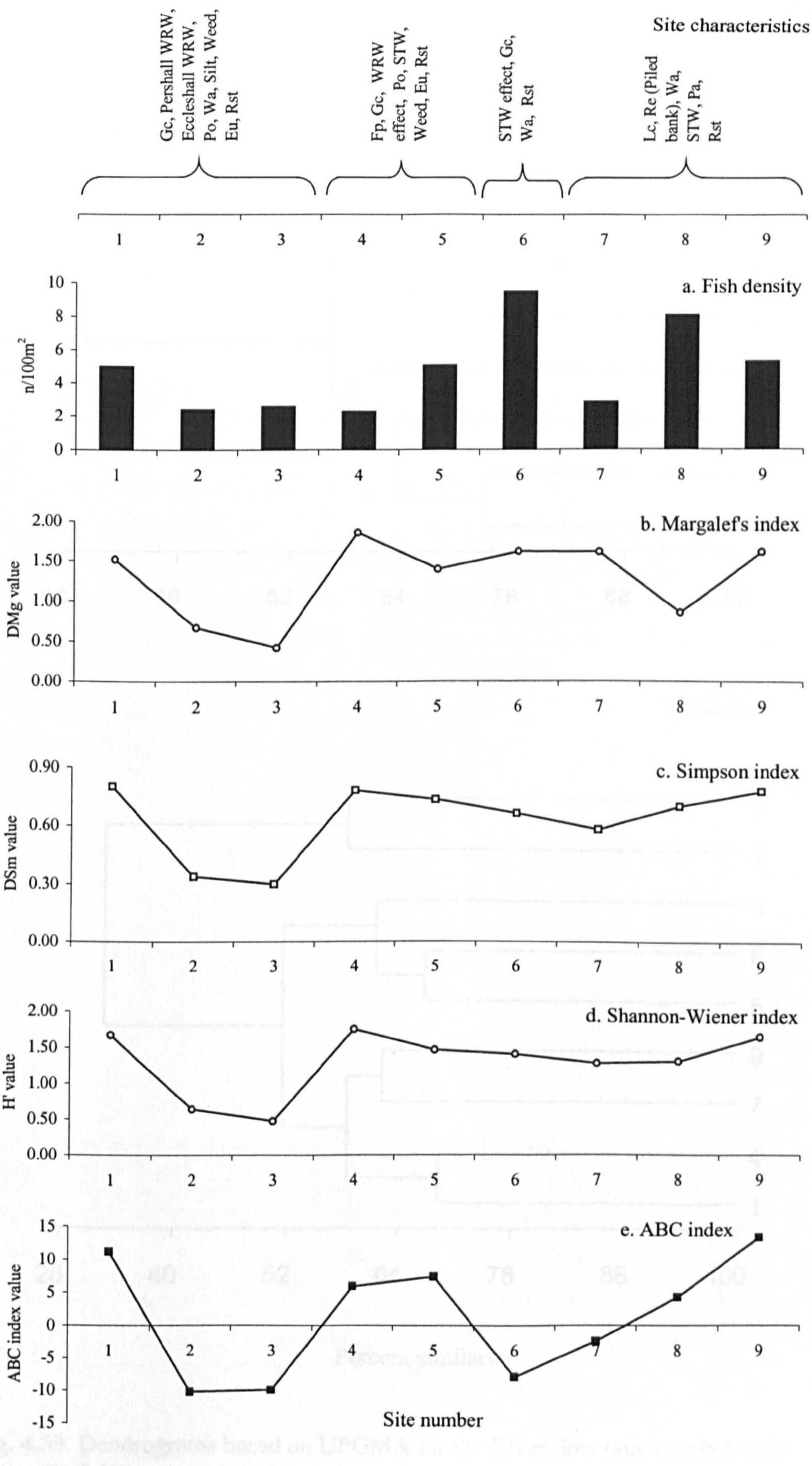


Fig. 4.38 Site characteristics and variations in different indices for the River Sow

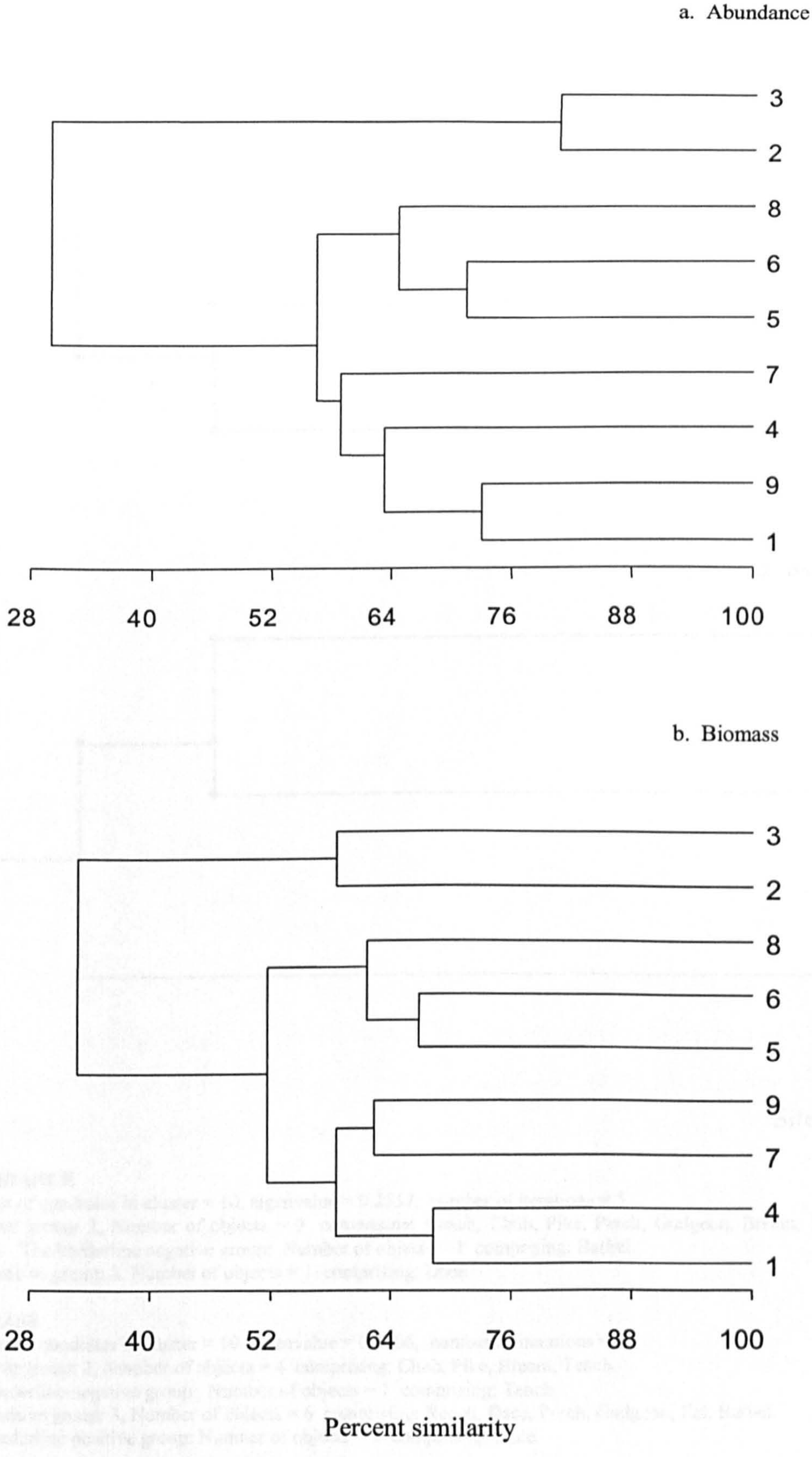
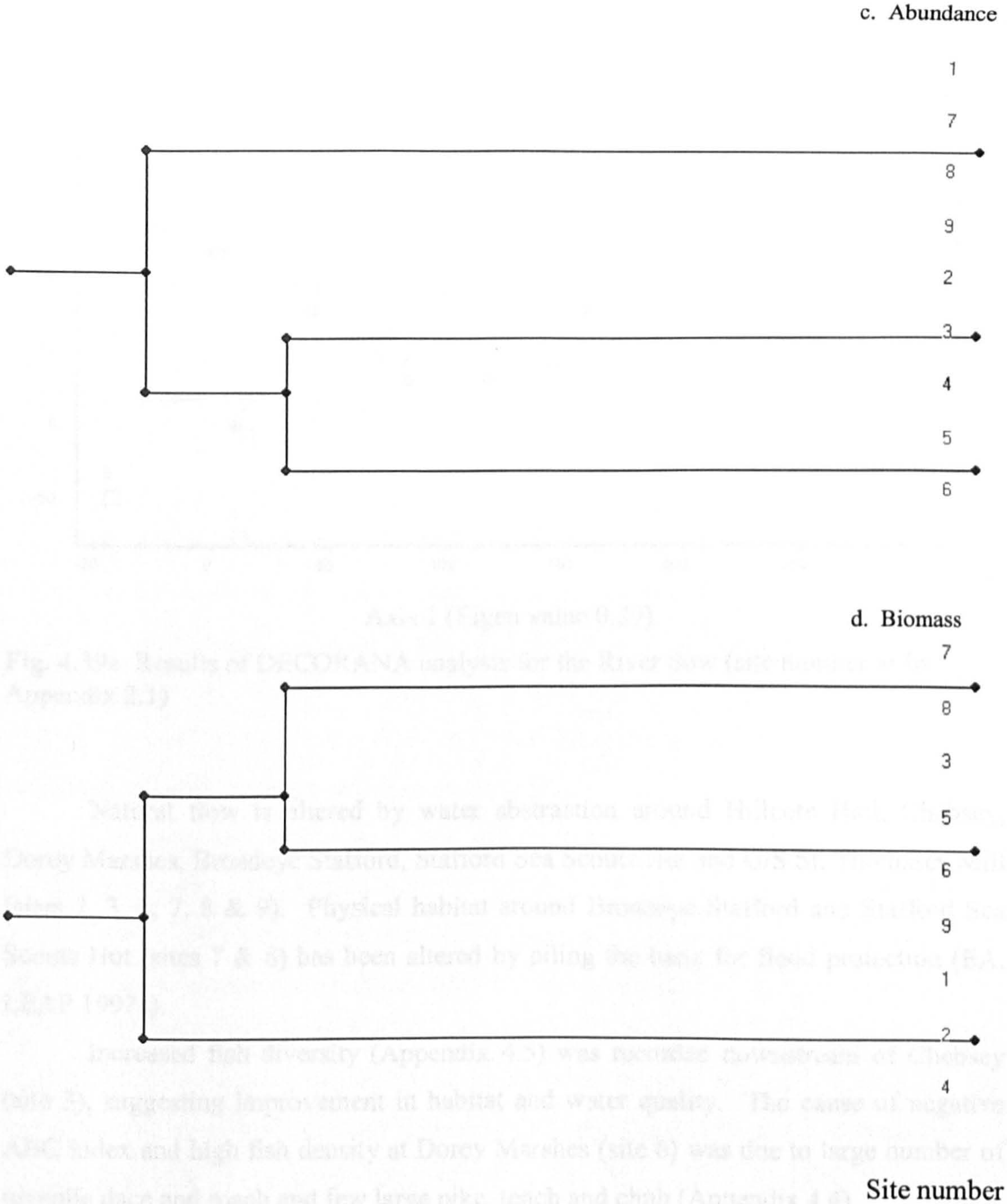


Fig. 4.39 Dendrograms based on UPGMA for the River Sow (site number as in Appendix 2.1)



ABUNDANCE
Number of quadrates in cluster = 10, eigenvalue = 0.2537, number of iterations = 5
Negative group: 2, Number of objects = 9 comprising: Roach, Chub, Pike, Perch, Gudgeon, Bream, Tench, Eel, Barbel. The borderline negative group: Number of objects = 1 comprising: Barbel.
The positive group: 3, Number of objects = 1 comprising: Dace.

BIOMASS
Number of quadrates in cluster = 10, eigenvalue = 0.3406, number of iterations = 4
Negative group: 2, Number of objects = 4 comprising: Chub, Pike, Bream, Tench.
The borderline negative group: Number of objects = 1 comprising: Tench.
The positive group: 3, Number of objects = 6 comprising: Roach, Dace, Perch, Gudgeon, Eel, Barbel.
The borderline positive group: Number of objects = 1 comprising: Dace.

Fig. 4.39 (Continued) Results of TWINSPAN for the River Sow at one level of division (site number as in Appendix 2.1)

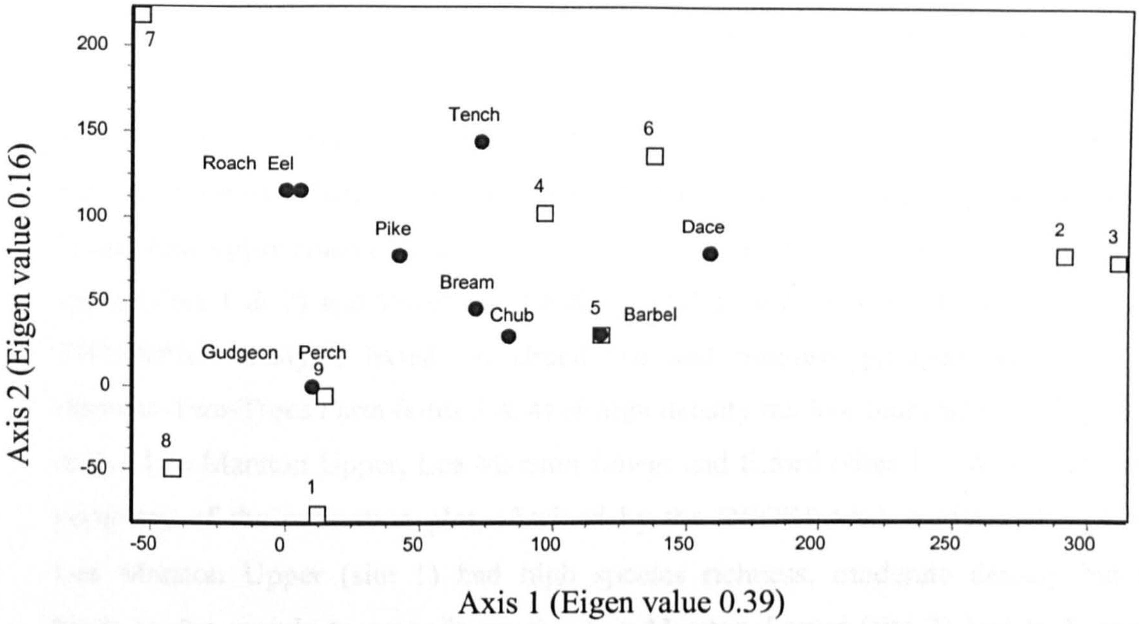


Fig. 4.39e Results of DECORANA analysis for the River Sow (site number as in Appendix 2.1)

Natural flow is altered by water abstraction around Hillcote Hall, Chebsey, Dorey Marshes, Broadeye Stafford, Stafford Sea Scouts Hut and U/S St. Thomases Mill (sites 2, 3, 6, 7, 8 & 9). Physical habitat around Broadeye Stafford and Stafford Sea Scouts Hut (sites 7 & 8) has been altered by piling the bank for flood protection (EA, LEAP 1997a).

Increased fish diversity (Appendix 4.5) was recorded downstream of Chebsey (site 3), suggesting improvement in habitat and water quality. The cause of negative ABC index and high fish density at Dorey Marshes (site 6) was due to large number of juvenile dace and roach and few large pike, tench and chub (Appendix 4.4).

River Tame

Fish species diversity was high in the River Tame with 18 species (Appendix 4.1) including 4 minor species, captured from six sampling sites. The fish species belonged to 17 genera, representing 8 families (Appendix 4.2). Roach, gudgeon, chub, dace, perch, pike, eel, bleak and tench were the most widespread species in the river, occupying >50% of sampling sites (Fig. 4.15m). Silver bream was recorded from Chetwynd Bridge (site 6) (Appendix 4.5). Fish diversity was high but density was very low within the sampling sites (sites 1 - 6). Fish density ranged from 0.6 to 6.0 fish

100m^{-2} with a mean of 2.47±2.03 fish 100m^{-2} (Fig. 4.40a). Highest density was at Lea Marston Lower (site 2) (Fig. 4.40a), where eight species were recorded (Appendix 4.5), but was very low at sites Hopwas-Two-Trees Farm and Elford (sites 4 & 5).

These observations on the diversity, density and assemblage were supported by the different diversity indices (Figs 4.40b, c, d & e). The ABC index depicted this section of the river (sites 1 - 6) as very poor. Cluster analysis based on fish abundance isolated the upper reach as poor (site 1) (Fig. 4.41a). However, biomass cluster, isolated upper (sites 1 & 2) and lower (sites 4 & 5) reaches with poor sites (Fig. 4.41b). The TWINSPAN analysis based on abundance and biomass grouped Middleton and Hopwas-Two-Trees Farm (sites 3 & 4) as high density but low biomass sites (Figs 4.41c & d). Lea Marston Upper, Lea Marston Lower and Elford (sites 1, 2 & 5) were in the periphery of the ordination plot, obtained by the DECORANA analysis (Fig. 4.41e). Lea Marston Upper (site 1) had high species richness, moderate density but low biomass due mainly to juvenile roach. Lea Marston Lower (site 2) had high species richness and density but low biomass due to juvenile roach. Very poor biomass, due to high numbers of juveniles, was the cause of isolation of Elford (site 5) in the ordination plot although species richness was high in the site.

The poor fish stocks with low density (Fig. 4.40a) in the river were due to pollution, erosion and eutrophication (Fig. 4.40). The reaches around Lea Marston Upper (site 1) receive effluent from STWs and mine water (EA, LEAP 1999c). In addition, the fish community was affected by dense weed resulting from eutrophication (EA, LEAP 2000a). The section between Middleton and Elford (sites 3 & 5) has also been affected by siltation as the stretch is subject to erosion (EA, LEAP 1998d). The lower section between Elford and Chetwynd Bridge (sites 5 & 6) had impoverished fish stocks, also due to pollution and eutrophication (EA, LEAP 1998d). The highly negative values for all the sites were due to serious structural imbalance in the fish community based on large numbers of juvenile roach and few large fish (Appendix 4.4).

River Tean

Eleven fish species including 4 minor species (Appendix 4.1), belonging to 10 genera and 7 families (Appendix 4.2) were captured from nine sites on the River Tean. Brown trout and perch dominated the river, being present in >50% of sites (Fig. 4.15n). Grayling were also abundant but no pike were found in this river (Appendix 4.5). No fish were found at Fole Hall and Fole D/S Creamery (sites 6 & 7).

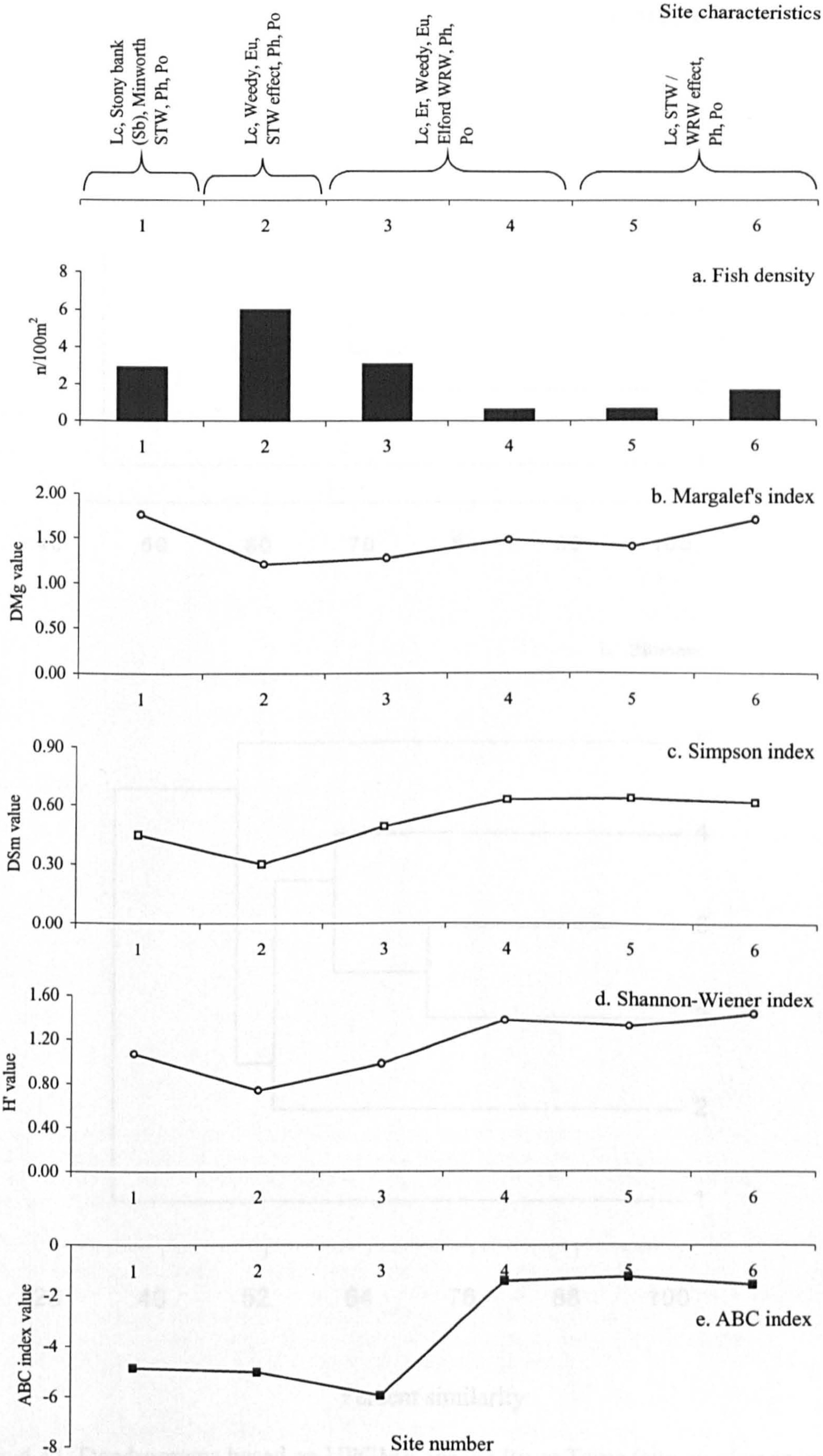


Fig. 4.40 Site characteristics and variations in different indices for the River Tame

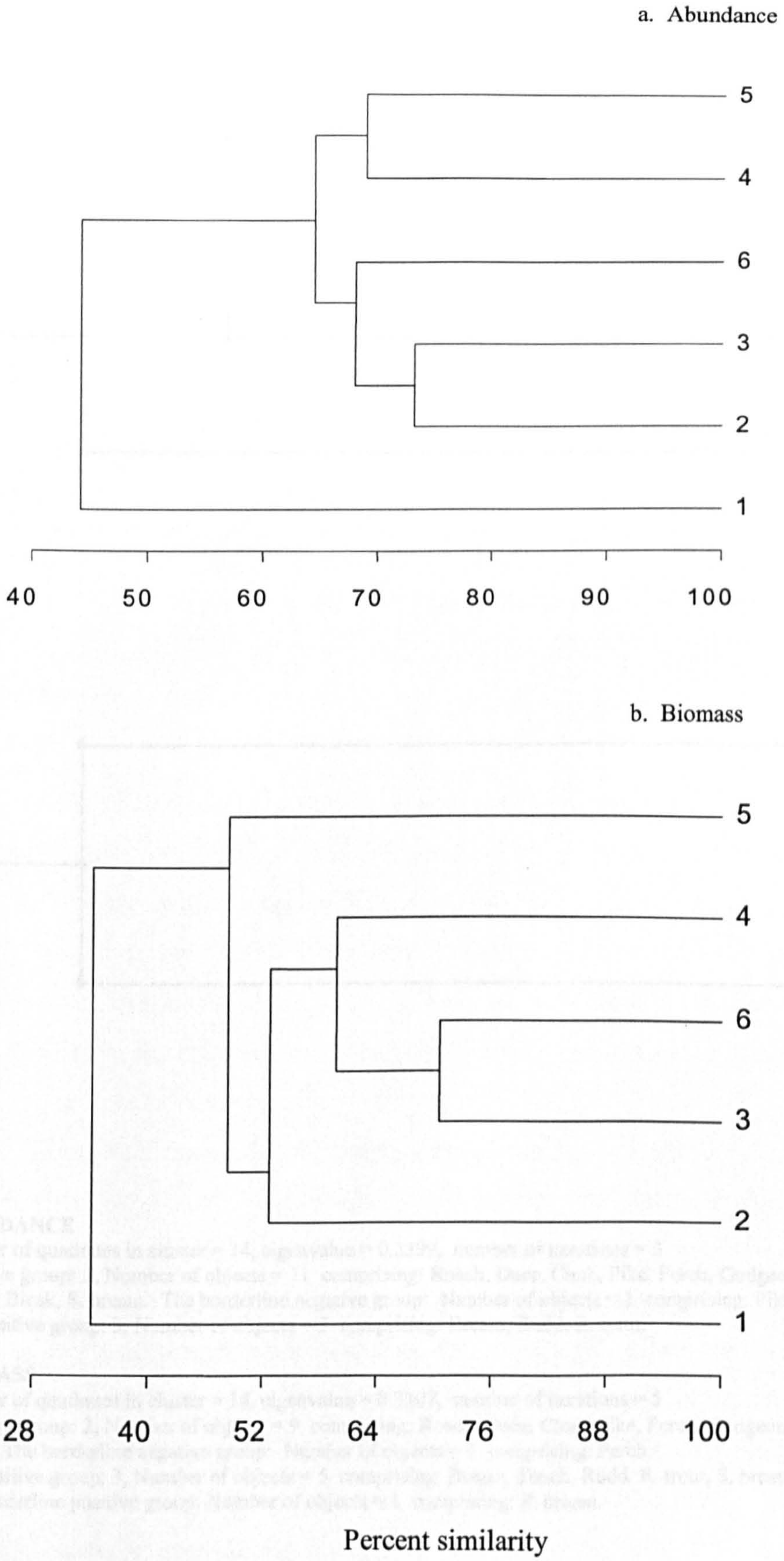
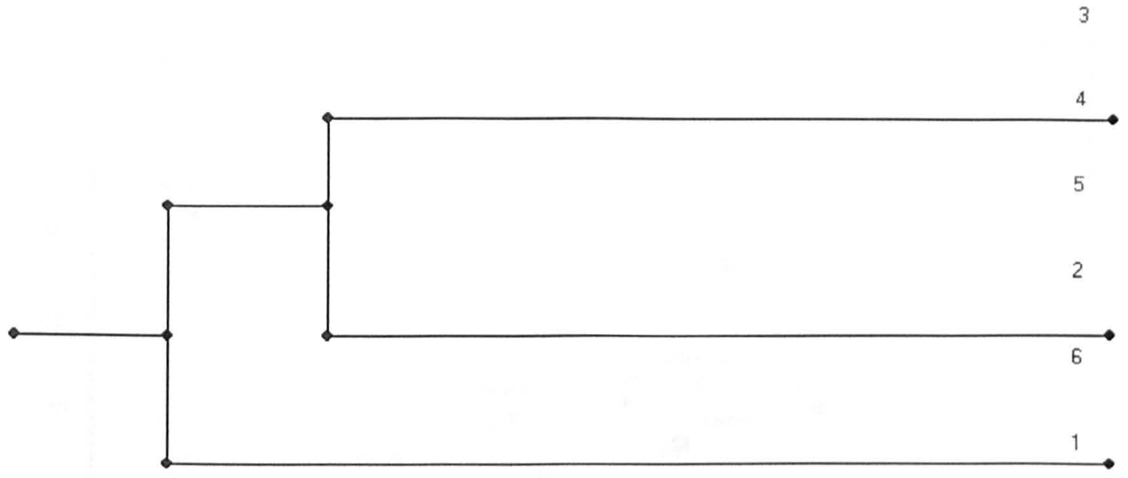
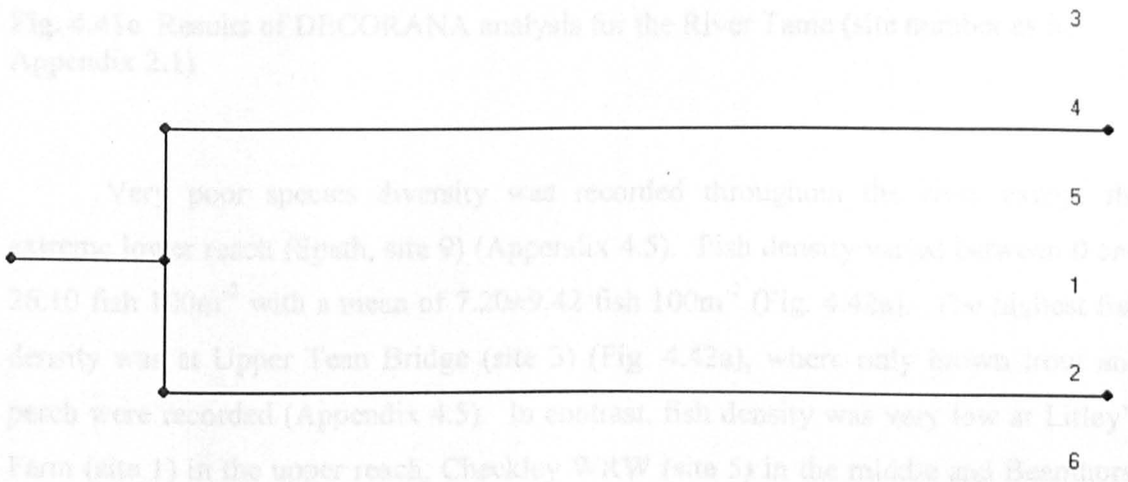


Fig. 4.41 Dendrograms based on UPGMA for the River Tame (site number as in Appendix 2.1)

c. Abundance



d. Biomass



Site number

ABUNDANCE
Number of quadrates in cluster = 14, eigenvalue = 0.3399, number of iterations = 3
Negative group: 2, Number of objects = 11 comprising: Roach, Dace, Chub, Pike, Perch, Gudgeon, Tench, Eel, Barbel, Bleak, S. bream. The borderline negative group: Number of objects = 1 comprising: Pike.
The positive group: 3, Number of objects = 3 comprising: Bream, Rudd, R. trout.

BIOMASS
Number of quadrates in cluster = 14, eigenvalue = 0.3307, number of iterations = 5
Negative group: 2, Number of objects = 9 comprising: Roach, Dace, Chub, Pike, Perch, Gudgeon, Eel, Barbel, Bleak. The borderline negative group: Number of objects = 1 comprising: Perch.
The positive group: 3, Number of objects = 5 comprising: Bream, Tench, Rudd, R. trout, S. bream.
The borderline positive group: Number of objects = 1 comprising: S. bream.

Fig. 4.41 (Continued) Results of TWINSPAN for the River Tame at one level of division (site number as in Appendix 2.1)

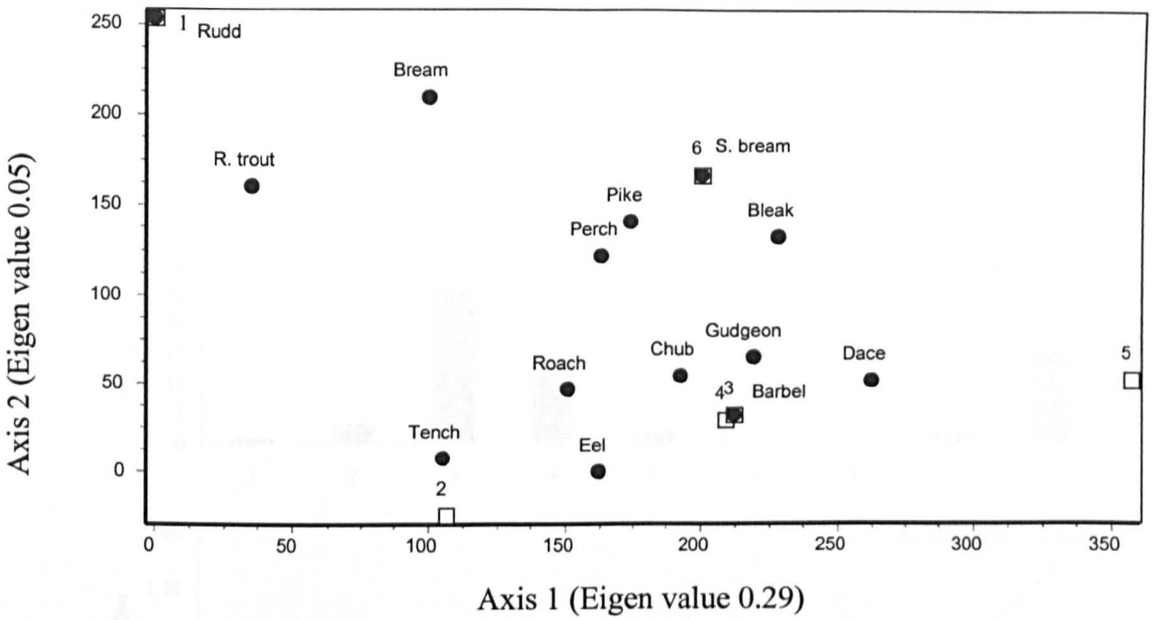


Fig. 4.41e Results of DECORANA analysis for the River Tame (site number as in Appendix 2.1)

Very poor species diversity was recorded throughout the river except the extreme lower reach (Spath, site 9) (Appendix 4.5). Fish density varied between 0 and 26.10 fish 100m^{-2} with a mean of 7.20±9.42 fish 100m^{-2} (Fig. 4.42a). The highest fish density was at Upper Tean Bridge (site 3) (Fig. 4.42a), where only brown trout and perch were recorded (Appendix 4.5). In contrast, fish density was very low at Litley's Farm (site 1) in the upper reach, Checkley WRW (site 5) in the middle and Beamhurst Bridge (site 8) in the lower reaches.

These findings on the diversity, density and assemblage were supported by the different diversity indices (Figs 4.42b, c, d & e). The ABC index discriminated Rectory Farm and Checkley WRW (sites 4 & 5) as poor. Two dendrograms with similar groupings were obtained from cluster analysis based on abundance and biomass (Figs 4.43a & b). In both analyses sites 4 & 8 were grouped together. In both dendrograms, Litley's Farm and Checkley WRW (sites 1 & 5) were grouped with poor abundance and biomass sites in the upper and middle reaches and isolated Fole Hall and Fole D/S Creamery (sites 6 & 7) from the remaining sites (Figs 4.43a & b). The TWINSPAN analysis produced two similar dendrograms with similar groupings (Figs 4.43c & d). However, DECORANA analysis discriminated Litleys Farm, Teanford Mill, Beamhurst Bridge and Spath (sites 1, 2, 8 & 9) in the ordination plot (Fig. 4.43e).

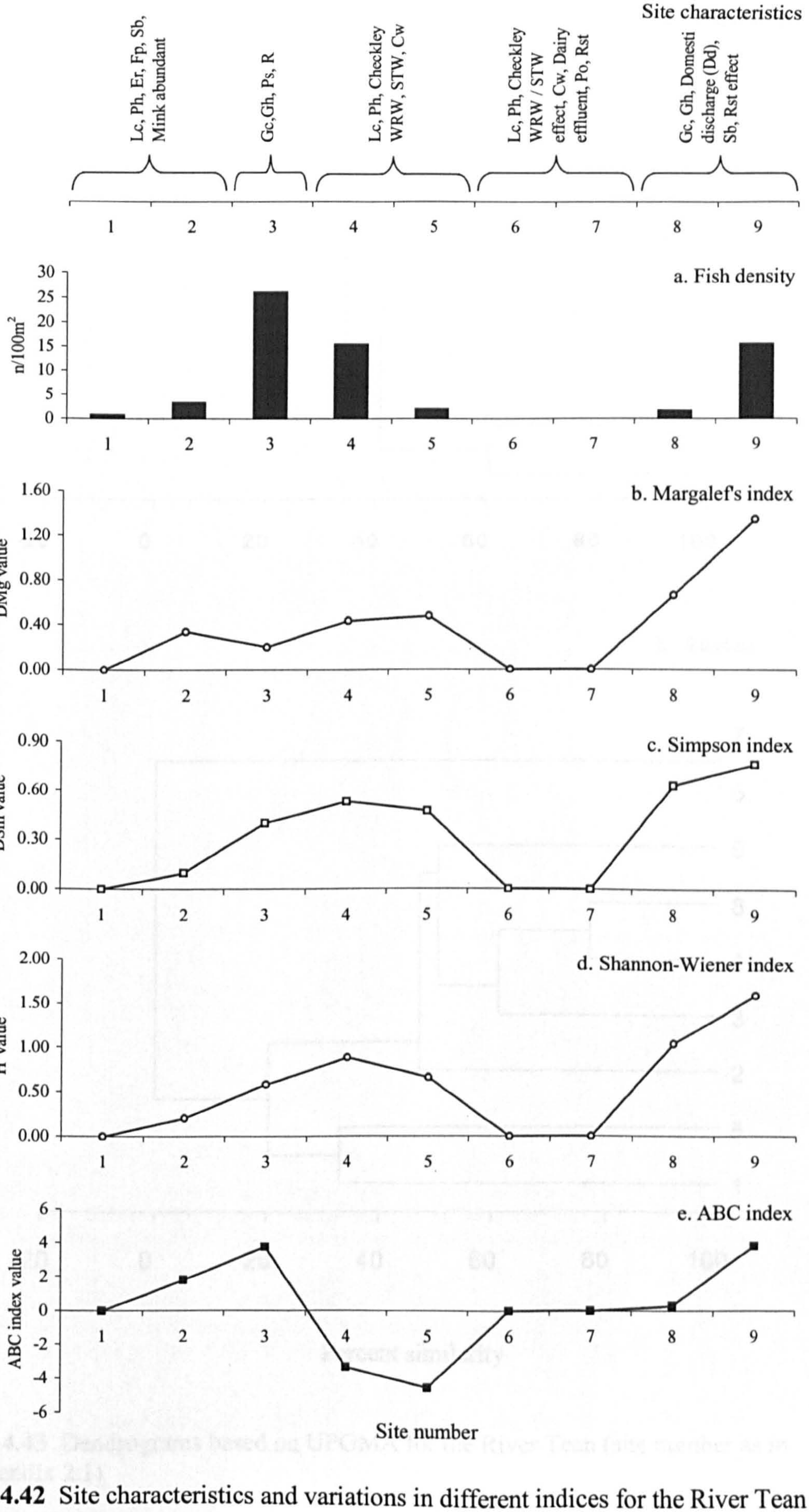


Fig. 4.42 Site characteristics and variations in different indices for the River Tean

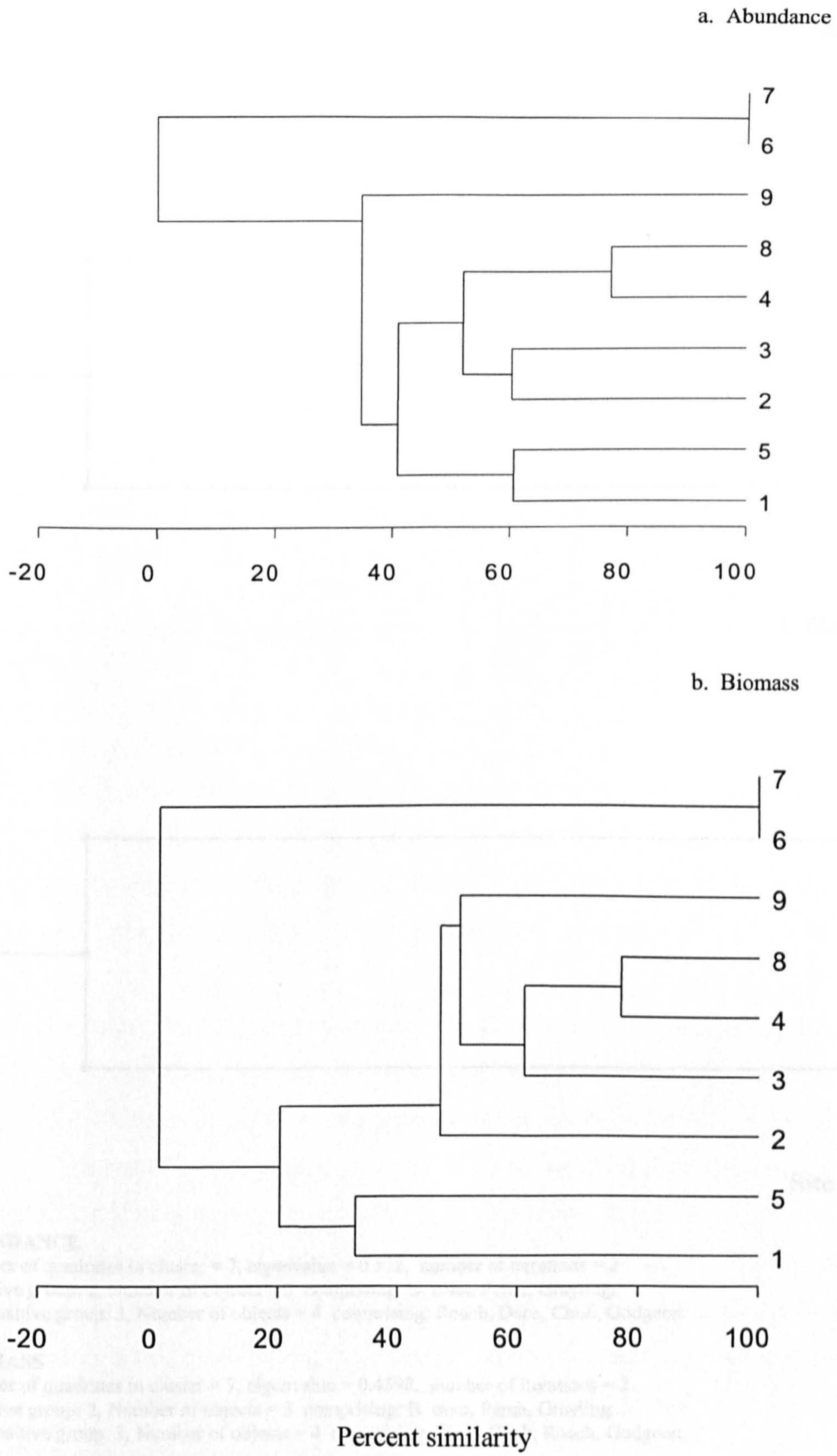
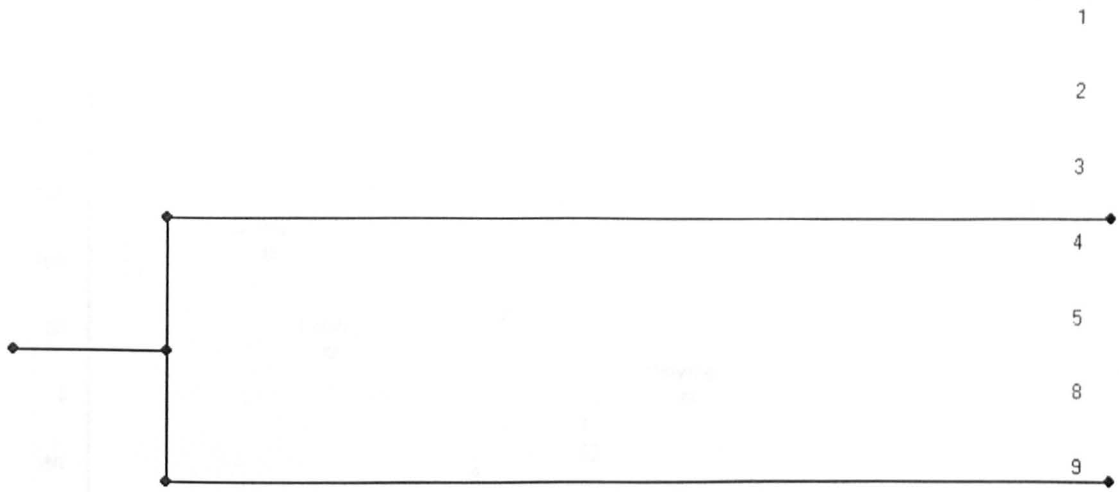
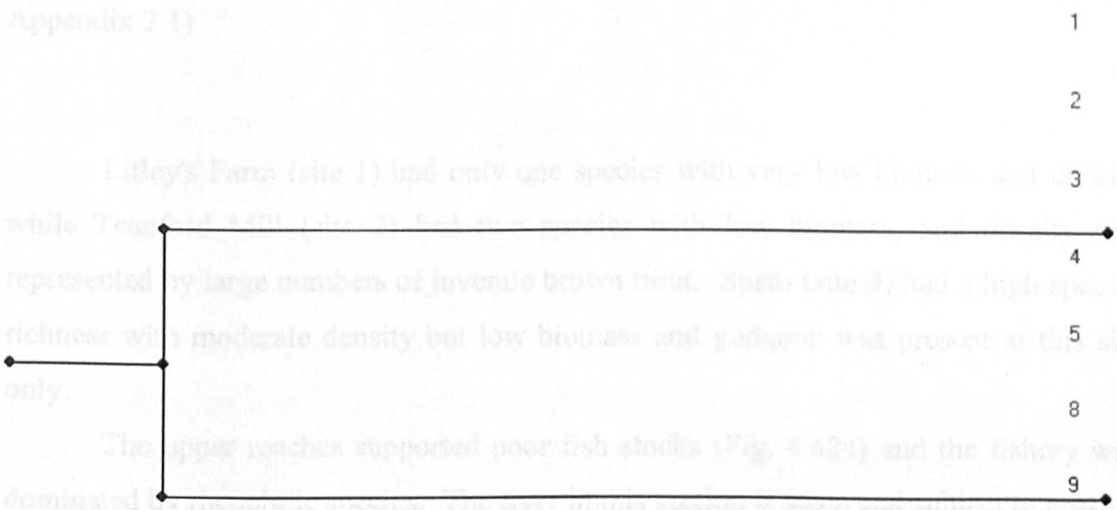


Fig. 4.43 Dendrograms based on UPGMA for the River Tean (site number as in Appendix 2.1)

c. Abundance



d. Biomass



Site number

ABUNDANCE
Number of quadrates in cluster = 7, eigenvalue = 0.522, number of iterations = 2
Negative group: 2, Number of objects = 3 comprising: B. trout, Perch, Grayling.
The positive group: 3, Number of objects = 4 comprising: Roach, Dace, Chub, Gudgeon.

BIOMASS
Number of quadrates in cluster = 7, eigenvalue = 0.4598, number of iterations = 2
Negative group: 2, Number of objects = 3 comprising: B. trout, Perch, Grayling.
The positive group: 3, Number of objects = 4 comprising: Dace, Chub, Roach, Gudgeon.

Fig. 4.43 (Continued) Results of TWINSPAN for the River Tean at one level of division (site number as in Appendix 2.1)

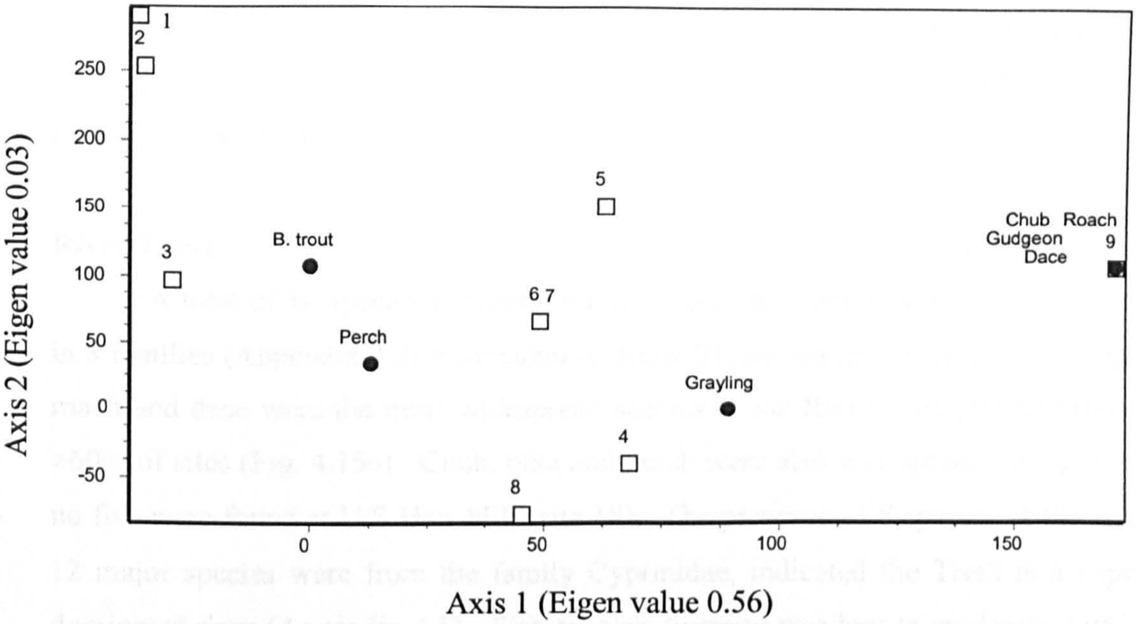


Fig. 4.43e Results of DECORANA analysis for the River Tean (site number as in Appendix 2.1)

Litley's Farm (site 1) had only one species with very low biomass and density while Teanford Mill (site 2) had two species with low biomass, and density was represented by large numbers of juvenile brown trout. Spath (site 9) had a high species richness with moderate density but low biomass and gudgeon was present at this site only.

The upper reaches supported poor fish stocks (Fig. 4.42a) and the fishery was dominated by rheophilic species. The river in this section is steep and subject to erosion (EA, LEAP 1999d). In addition, the reaches around Litley's Farm and Teanford Mill (site 1 & 2) were affected by farm pollution. Moreover, mink (*Mustela vison* Schreber) are abundant in this section (Fig. 4.42), and are suspected of having a detrimental impact on fish populations (EA, LEAP 2000c). Despite restocking, impoverished fish stocks occur in the middle reaches between Checkley WRW and Beamhurst Bridge (site 5 & 8) and were due to pollution, as the reach receives domestic and industrial effluents from STWs / WRWs and dairy farms (Fig. 4.42). Coloured water is a regular phenomenon in this section making it unsuitable for fish (EA, LEAP 1999d).

A quality fish stock (Fig. 4.42a) with increased diversity (Appendix 4.5) in the extreme lower reach (site 9, Spath), suggests improvement in the water quality and habitat. This was partially due to improvement in the effluent quality. An imbalance in the fish community structure at Rectory Farm and Checkley WRW (sites 4 & 5) was identified by the ABC method. Site 4 produced a negative index although it contained high fish density, due to large numbers of juvenile grayling and perch (Appendix 4.4). Site 5 produced a negative ABC index as the site contained only two species with low abundance and biomass (Appendix 4.4).

River Trent

A total of 16 species including 4 minor species (Appendix 4.1) from 15 genera in 8 families (Appendix 4.2) were captured from 20 sites on the River Trent. Gudgeon, roach and dace were the most widespread species in the River Trent, being present in >60% of sites (Fig. 4.15o). Chub, pike and perch were also widespread throughout but no fish were found at U/S Hoo Mill (site 18). The presence of 8 species of the total of 12 major species were from the family Cyprinidae, indicated the Trent is a cyprinid dominated river (Appendix 4.5). Fish species diversity was low to moderate with only 2 to 6 species being recorded per site (Appendix 4.5). Very poor diversity was recorded in the upper (sites 1 - 7) and middle reaches (sites 14 - 16) (Appendix 4.5). Generally low but highly variable fish density, ranging from 0 to 23.10 fish $100m^{-2}$ with a mean of 3.19 ± 5.64 fish $100m^{-2}$ was recorded from the river (Fig. 4.44a). Highest fish density was found at Finney Gardens (site 3), where five species were recorded, but was very low between Boothen End and Hissey's Scarp Yard (sites 6 - 8) and between D/S Park Brook Bridge and D/S Hoo Mill (sites 10 - 19).

These findings on the diversity, density and assemblage were supported by the various diversity indices (Figs 4.44b, c, d & e). The ABC index identified N. Staffs Polytech and Great Haywood Mill (sites 5 & 20) and, Boothen End and Walton Lane Stone (sites 6 & 14) as rich and poor sites, respectively (Fig. 4.44e). Two different dendrograms with different groupings were obtained from cluster analysis based on fish density and biomass (Figs 4.45a & b). Sites 2, 5, 9, 12, 13 & 15 showed similarity in the abundance dendrogram (Fig. 4.45a) and sites 10 & 12 showed similarity in the biomass (Fig. 4.45b). However, both dendrograms isolated Norton Green, Walton Lane Stone and U/S Hoo Mill (sites 1, 14 & 18) from the remaining sites and were grouped with poor abundance and biomass sites. TWINSPAN analysis based on fish abundance and biomass produced two different dendrograms (Figs 4.45c & d). However, Walton Lane Stone (site 14) was separated as a poor abundance and biomass site by both analyses. Except for Walton Lane Stone (site 14), all sites were found along the first Y-axis in the ordination plot, obtained by DECORANA analysis (Fig. 4.45e).

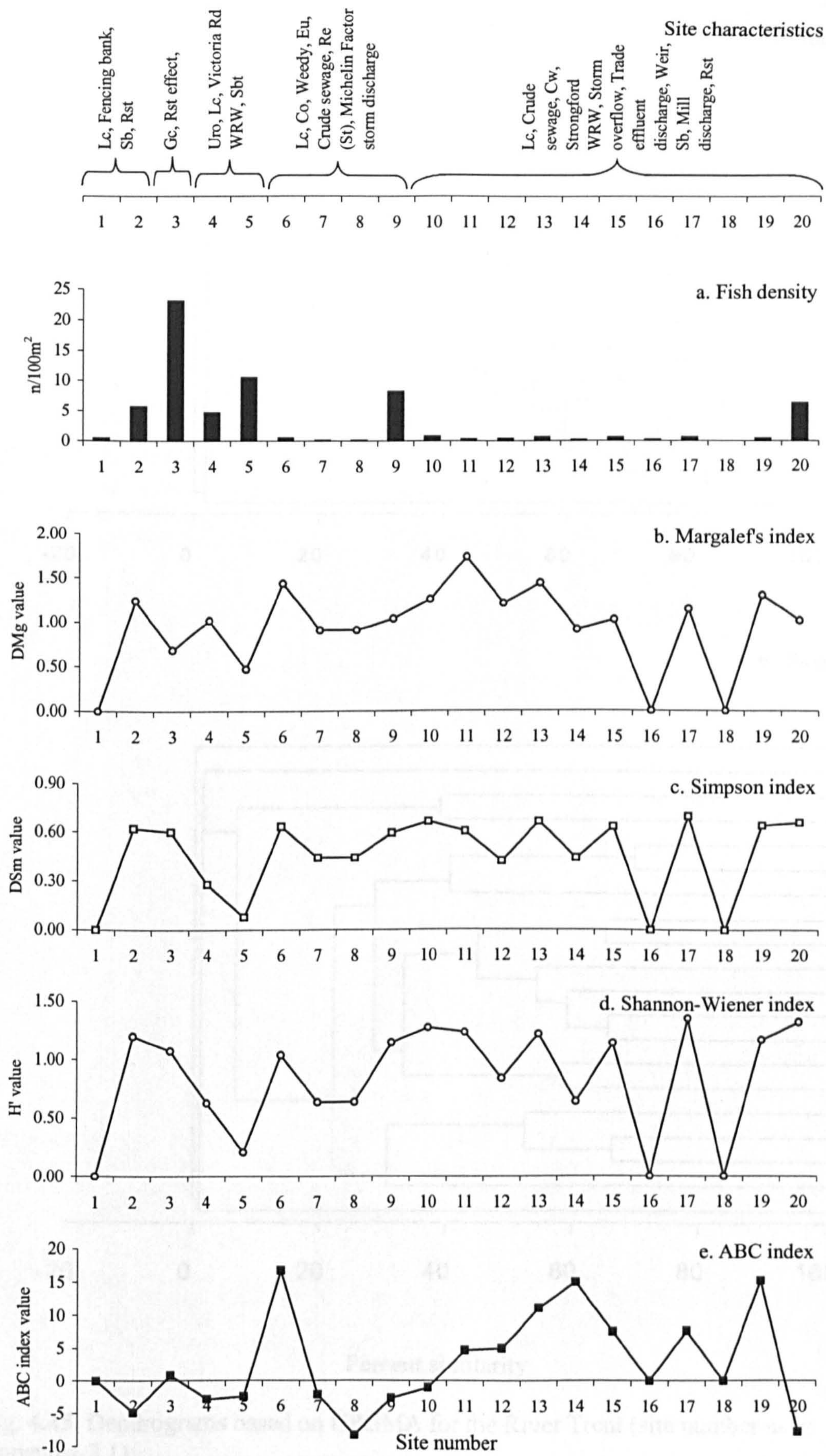
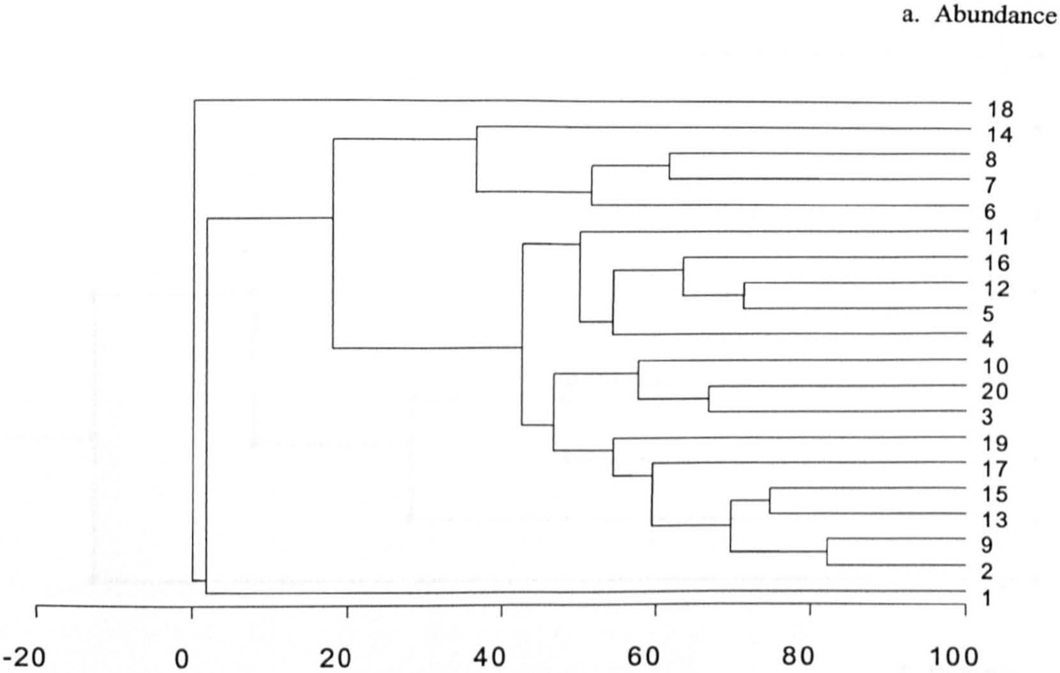


Fig. 4.44 Site characteristics and variations in different indices for the River Trent



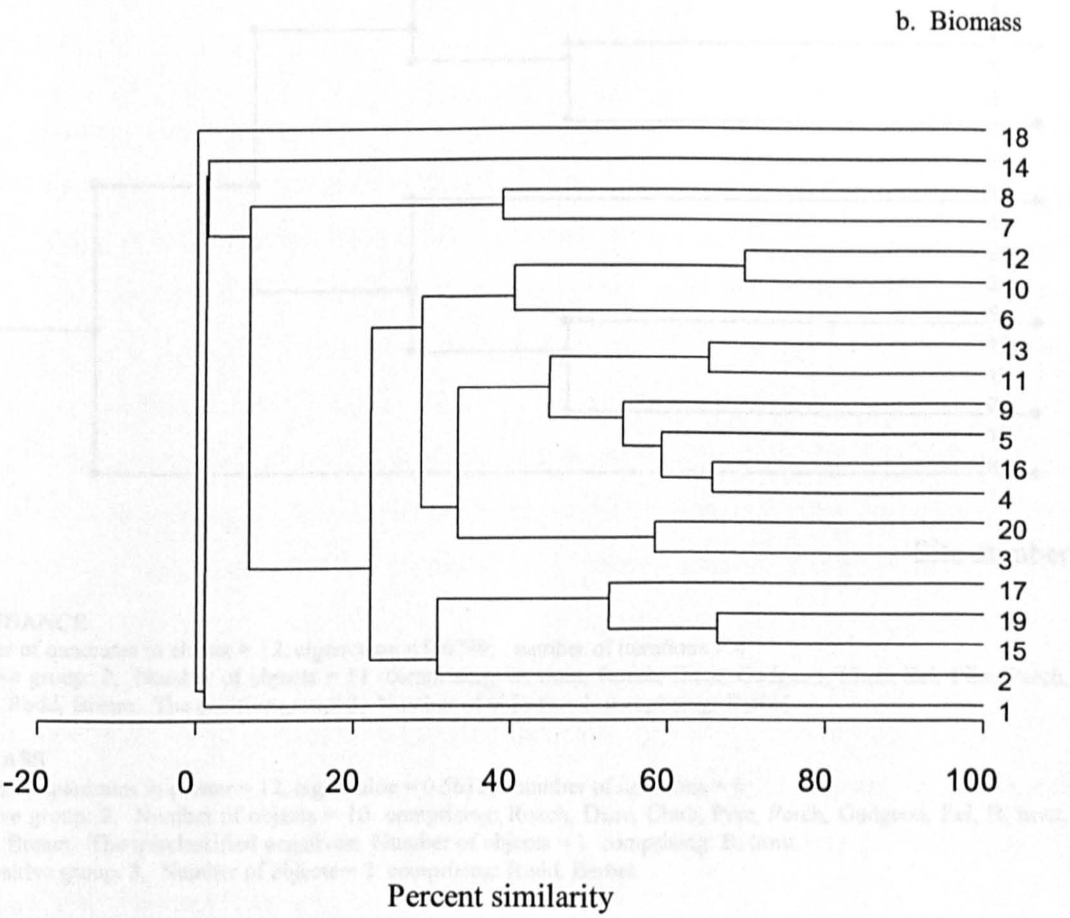
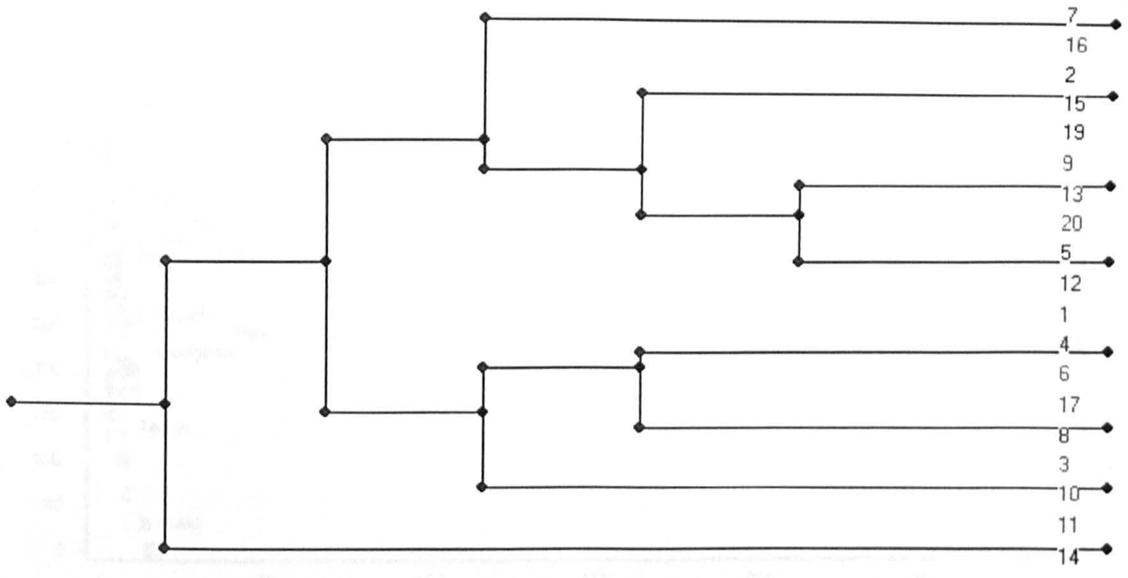
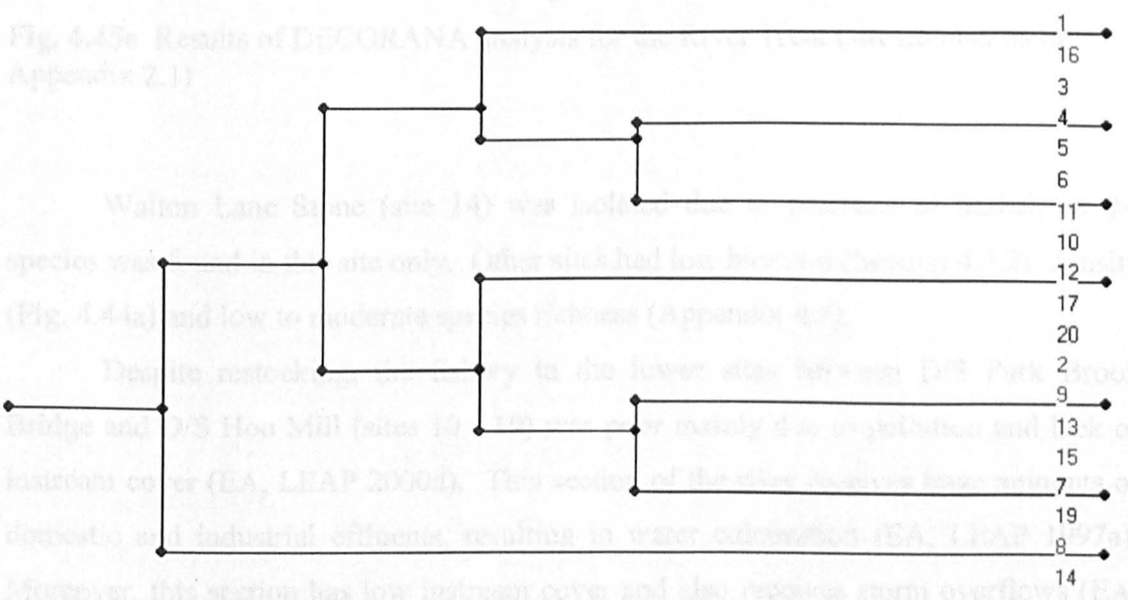


Fig. 4.45 Dendrograms based on UPGMA for the River Trent (site number as in Appendix 2.1)

c. Abundance



d. Biomass



ABUNDANCE
Number of quadrates in cluster = 12, eigenvalue = 0.6749, number of iterations = 4
Negative group: 2, Number of objects = 11 comprising: B. trout, Roach, Dace, Gudgeon, Chub, Eel, Pike, Perch, Tench, Rudd, Bream. The positive group: 3, Number of objects = 1 comprising: Barbel.

BIOMASS
Number of quadrates in cluster = 12, eigenvalue = 0.5612, number of iterations = 6
Negative group: 2, Number of objects = 10 comprising: Roach, Dace, Chub, Pike, Perch, Gudgeon, Eel, B. trout, Tench, Bream. The misclassified negatives: Number of objects = 1 comprising: B. trout.
The positive group: 3, Number of objects = 2 comprising: Rudd, Barbel.

Fig. 4.45 (Continued) Results of TWINSPAN for the River Trent at one level of division (site number as in Appendix 2.1)

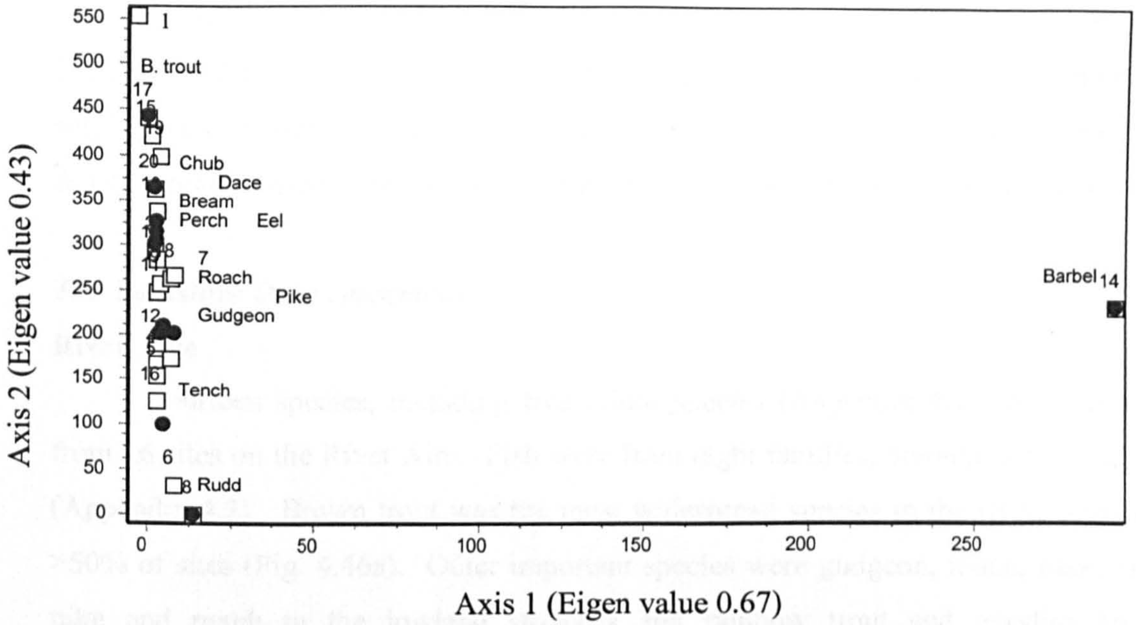


Fig. 4.45e Results of DECORANA analysis for the River Trent (site number as in Appendix 2.1)

Walton Lane Stone (site 14) was isolated due to presence of barbel, as the species was found in this site only. Other sites had low biomass (Section 4.3.2), density (Fig. 4.44a) and low to moderate species richness (Appendix 4.5).

Despite restocking, the fishery in the lower sites between D/S Park Brook Bridge and D/S Hoo Mill (sites 10 - 19) was poor mainly due to pollution and lack of instream cover (EA, LEAP 2000d). This section of the river receives huge amounts of domestic and industrial effluents, resulting in water colouration (EA, LEAP 1997a). Moreover, this section has low instream cover and also receives storm overflows (EA, LEAP 1998c). Impoverished fish stocks were a feature between Seven Arches, Stoke-upon-Trent and Hissey's Scarp Yard (sites 4 - 8), which were also due to pollution from STWs / WRWs. In addition reaches between Boothen End and Hanford U/S Lyme Brook (sites 6 & 7) have been straightened as a result of river engineering works (EA, LEAP 1997a). Moreover, this section (sites 6 & 7) receives crude sewage effluents and urban run off (Fig. 4.44). Comparatively high fish density at Finney Gardens (site 3) was probably due to restocking the stretch between Norton Green and Abbey Farm (site 1 & 2) with chub and dace (EA, LEAP 2000d).

Comparatively improved fish stocks (Fig. 4.44a) with moderate diversity (Appendix 4.5) in the downstream (site 20, Great Haywood Mill), indicated improvement in water quality and / or habitat, partially due to improvement in the effluent quality, especially from STWs and WRWs. The perturbations at N. Staffs Polytech, Boothen End, Walton Lane Stone and Great Haywood Mill (sites 5, 6, 14 & 20) predicted by the ABC index were due to numerical dominance of juveniles of a single species (roach) (Appendix 4.4). At Boothen End and Walton Lane Stone (sites 6 & 14), a high positive ABC index was due to low biomass against low abundance.

The Yorkshire Ouse catchment

River Aire

Fourteen species, including five minor species (Appendix 4.1), were recorded from 26 sites on the River Aire. Fish were from eight families, belonging to 13 genera (Appendix 4.2). Brown trout was the most widespread species in the river, present in >50% of sites (Fig. 4.46a). Other important species were gudgeon, roach, dace, chub, pike and perch in the lowland stretches and rainbow trout and grayling in the headwaters. Brown trout were widespread in the upper reaches, from Malham Beck, below Malham Cove to D/S Snaygill STW, above Cononley (sites 1 - 15), covering 58% of sampling sites (Appendix 4.6). No cyprinids were found upstream of Near Gargrave STW (site 13). No fish were recorded from Malham Beck below Malham Cove and Calverley below A6120 d/s Rawdon STW (sites 1 & 18), while single species were caught from 13 sites (sites 1- 8, 10 - 13, 22 & 24). Fish species diversity was very low in the upper reaches between Malham Beck below Malham Cove and Near Gargrave STW (site 1 - 13) and moderate in the middle through lower reaches (sites 14 - 26) (Appendix 4.6).

Highly fluctuating fish density, ranging from 0 to 69.0 fish $100m^{-2}$ with a mean of 18.02±17.68 fish $100m^{-2}$ (Fig. 4.47a) was recorded from the river. Highest fish density was found at River Aire Hanlith Bridge (site 9), where only two species were recorded. However, fish density was very low at Crossflatts and Esholt U/S STW (sites 16 & 17) in the middle reaches and Kirkstall, Swillington Bridge and Castleford alongside Hicksons Ltd (sites 19, 22 & 24) in the lower reaches. The above results revealed that the River Aire supported fish species richness with poor assemblage structure.

These results on the diversity, density and assemblage were corroborated by the different diversity indices (Figs 4.47b, c & d). Cluster analysis, based on fish abundance grouped the sites (sites 17, 22 & 24) with poor abundance in the lower reaches and isolated the most upstream (site 1) and middle (site 18) reaches (Fig. 4.48a).

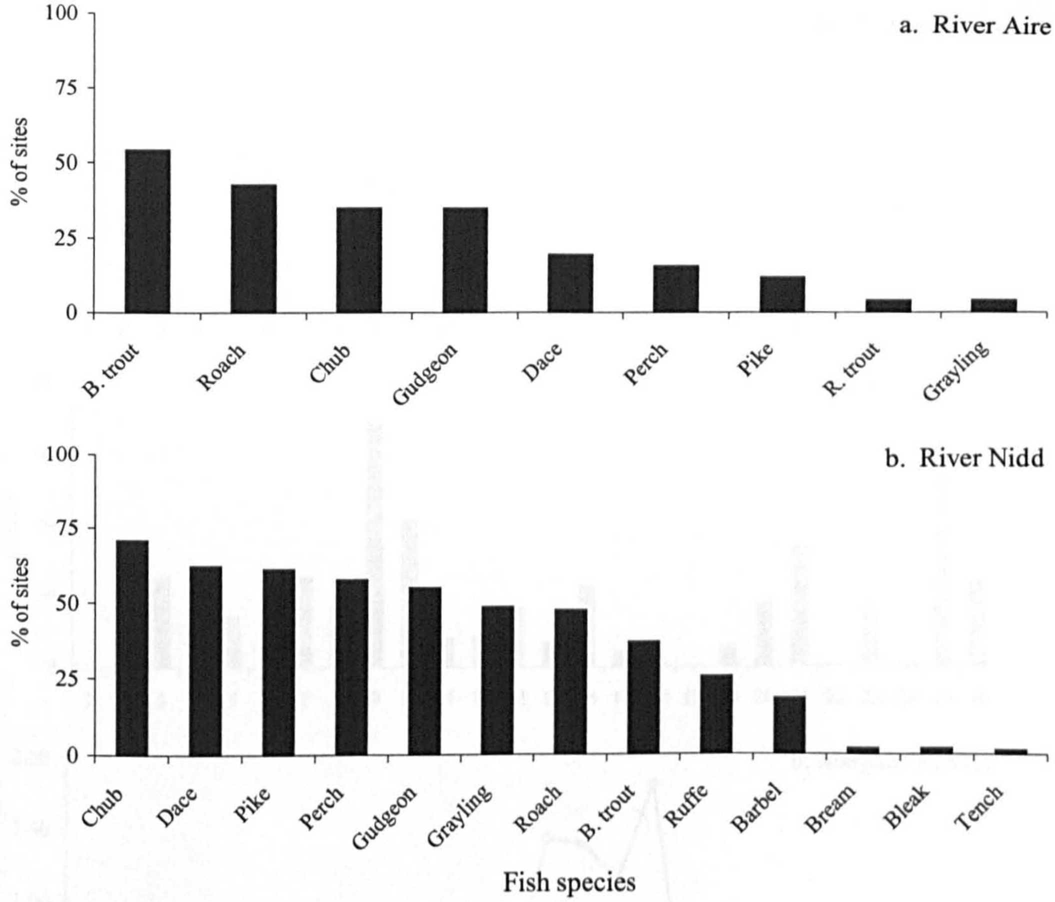


Fig. 4.46 Percentage of sites in the Yorkshire Ouse catchment containing major fish species

The ABC, UPGMA for biomass, TWINSPAN and DECORANA analyses were not performed for the River Aire due to lack of biomass data.

In general, poor fish stocks (Fig. 4.47a) with low to moderate fish diversity (Appendix 4.6) were due to pollution, construction of weirs and lack of instream cover (Fig. 4.47). The river receives huge volumes of effluent from 21 STWs whilst 30 weirs in the main channel (EA, LEAP 1998a) hamper upstream fish migration. Sites (especially sites 14 - 19) have been seriously affected by effluent discharges (EA, LEAP 1998a). Very poor fish stocks in the reach around Swillington Bridge and Castleford alongside Hicksons Ltd (sites 22 & 24) were probably due to discharges from the nearby power station and Hickson and Welch Chemical companies, respectively. In addition, these sections had low instream cover (EA, LEAP 1998a). Physical habitat has been altered in the middle reaches between "above Gargrave below Bridge" and Kirkstall (site 11 - 19) for flood defence purposes. Moreover, these sections, especially Crossflatts and Kirkstall, were subject to organic pollution from organo-phosphorous pesticides through Marley Sewage Treatment Works (EA, LEAP 1998a).

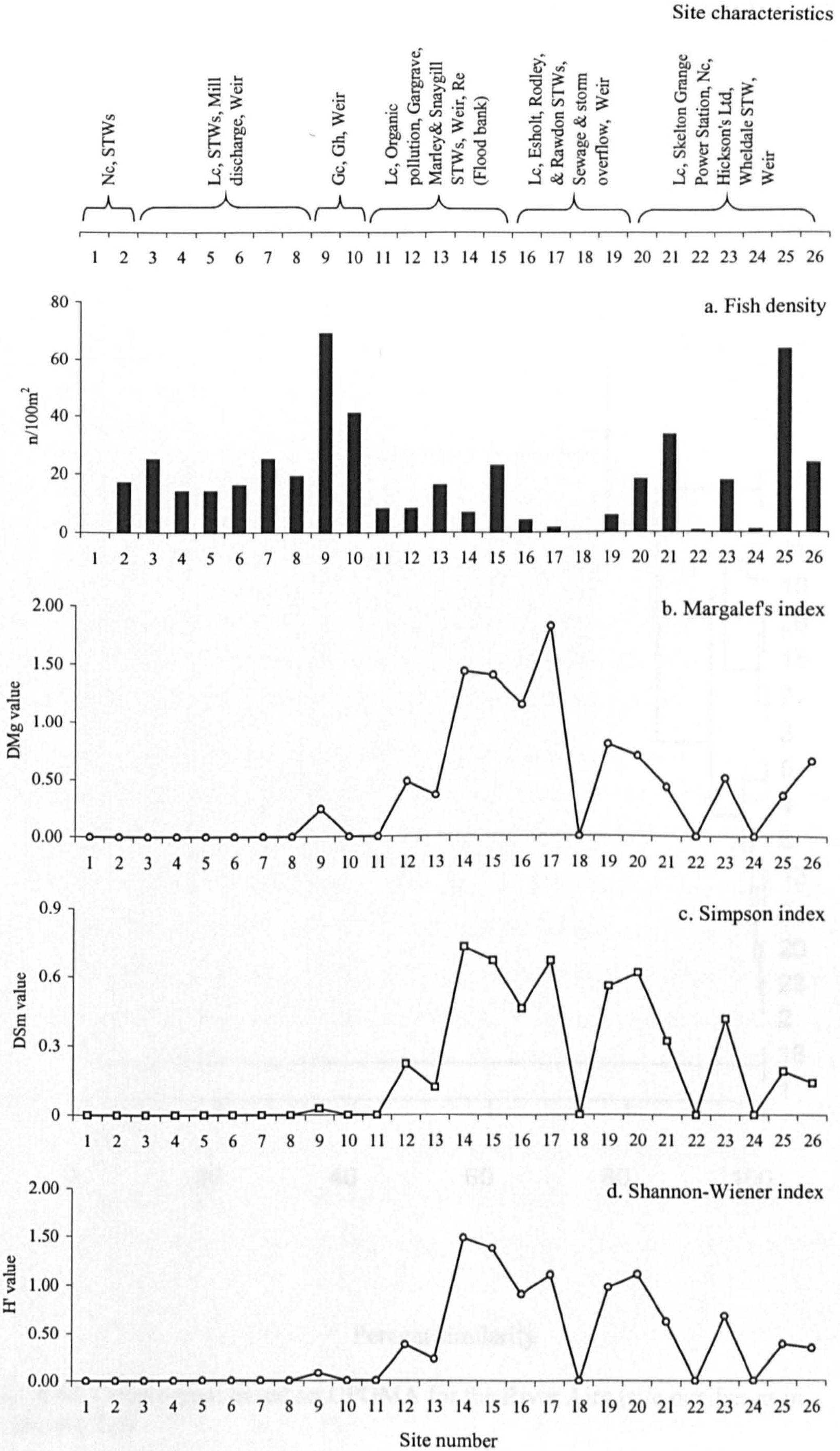


Fig. 4.47 Site characteristics and variations in different indices for the River Aire

a. Abundance

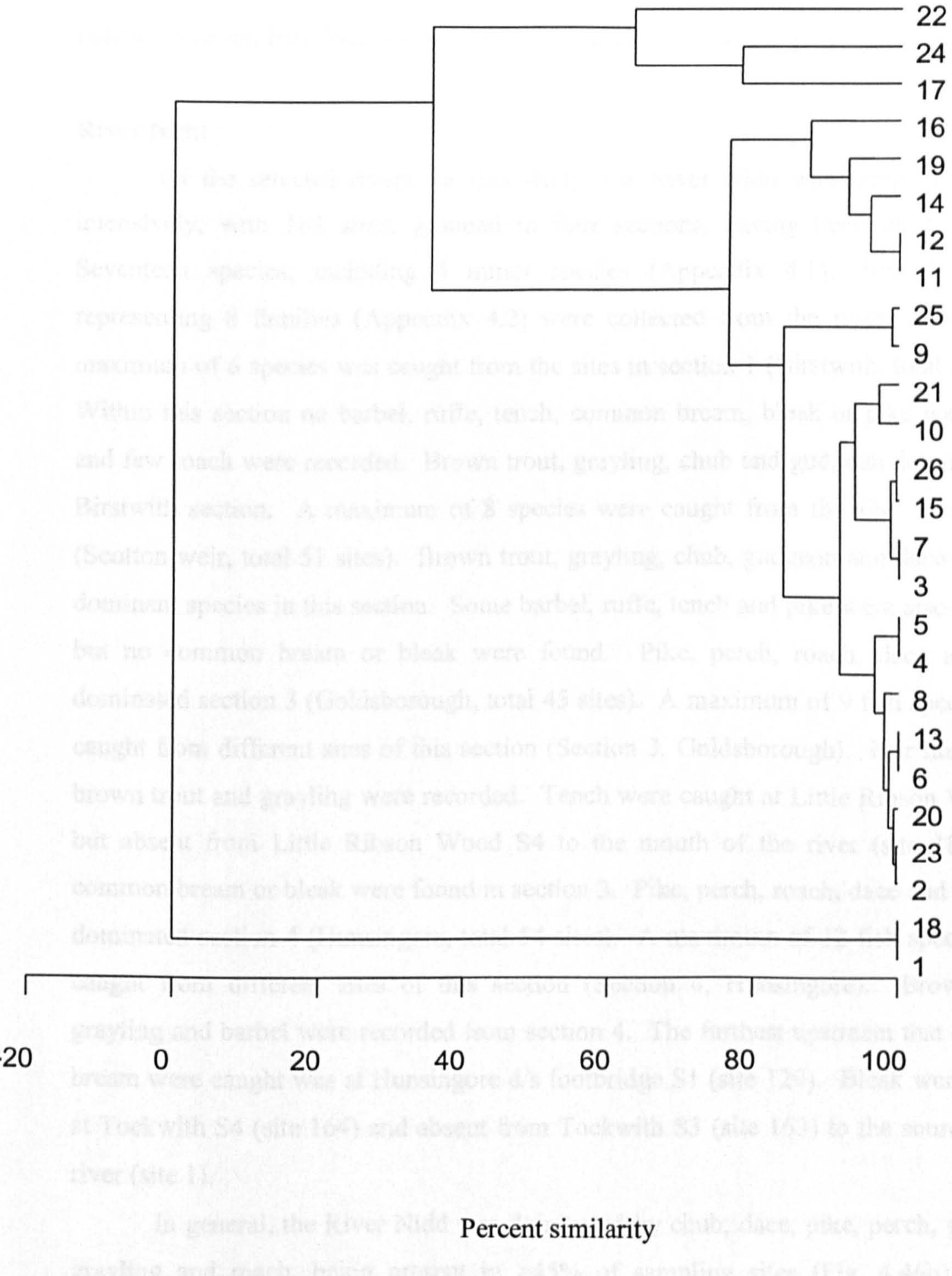


Fig. 4.48 Dendrogram based on UPGMA for the River Aire (site number as in Appendix 2.1)

The upper reaches between Malham Beck below the waterfall and River Aire Airton Bridge (site 2 & 10) had high to moderate fish density (Fig. 4.47a), which may be due to good instream cover and low pollution (EA, LEAP 1998a). The fishery in the upper reaches (sites 1 - 8) was totally dominated by brown trout (Appendix 4.6). Comparatively moderate diversity but high fish density were recorded in the lower reaches between Beal Weirpool and Chapel Haddlesey U/S A19 (sites 25 & 26).

River Nidd

Of the selected rivers for this study, the River Nidd was sampled the most intensively, with 182 sites, grouped in four sections, having been electric fished. Seventeen species, including 4 minor species (Appendix 4.1), from 16 genera, representing 8 families (Appendix 4.2) were collected from the river. However, a maximum of 6 species was caught from the sites in section 1 (Birstwith, total 32 sites). Within this section no barbel, ruffe, tench, common bream, bleak or pike were found and few roach were recorded. Brown trout, grayling, chub and gudgeon dominated the Birstwith section. A maximum of 8 species were caught from the sites in section 2 (Scotton weir, total 51 sites). Brown trout, grayling, chub, gudgeon and dace were the dominant species in this section. Some barbel, ruffe, tench and pike were also recorded but no common bream or bleak were found. Pike, perch, roach, dace and chub dominated section 3 (Goldsborough, total 45 sites). A maximum of 9 fish species were caught from different sites of this section (Section 3, Goldsborough). Fair numbers of brown trout and grayling were recorded. Tench were caught at Little Ribson Wood S3 but absent from Little Ribson Wood S4 to the mouth of the river (site 182). No common bream or bleak were found in section 3. Pike, perch, roach, dace and gudgeon dominated section 4 (Hunsingore, total 54 sites). A maximum of 12 fish species were caught from different sites of this section (Section 4, Hunsingore). Brown trout, grayling and barbel were recorded from section 4. The furthest upstream that common bream were caught was at Hunsingore d/s footbridge S1 (site 129). Bleak were caught at Tockwith S4 (site 164) and absent from Tockwith S3 (site 163) to the source of the river (site 1).

In general, the River Nidd was dominated by chub, dace, pike, perch, gudgeon, grayling and roach, being present in >45% of sampling sites (Fig. 4.46b). Other important species were brown trout, ruffe and barbel (Appendix 4.6). No fish were found at U/S Scotton Weir S1 and U/S Scotton Weir S2 (sites 33 & 34) and only one species with few individuals was captured from Holme bottom Farm S3, U/S Scotton

Weir S3, U/S Scotton Weir S4, Scotton Weir S1 and D/S A1 Bridge S1 (sites 31, 35, 36, 37 &117). Highest fish diversity was found at Little Ribston Wood S3, Ornamental Bridge S2, Hunsingore D/S Footbridge S1, Hunsingore D/S Footbridge S4, Cowthore Gauging Hut S2, Cowthorpe dog kennels S4, Cattal D/S Bridge S4 and Cattal D/S Old Thornville S1 (sites 95, 106, 129, 132, 134, 140, 152 & 157) in the lower reaches, where nine species were recorded.

Fish density was very low, ranging from 0 to 8.75 fish $100m^{-2}$ with a mean of 1.81 ± 1.82 fish $100m^{-2}$ (Fig. 4.49a). Highest fish density was recorded at Cattal U/S Bridge S3 (site 147) and was very low in the stretch between D/S Killinghall Bridge S2 and Scotton Weir S1 (sites 22 - 37), and U/S High Bridge S1 and U/S High Bridge S4 (sites 56 - 59) in the upper reaches. Very low fish density was also found at Lido Bottom S2 and Lido Bottom S4 (sites 69 - 71), D/S A59 Bridge S1 (site 76), U/S Goldsborough Mill S3 (site 82) and Little Ribston Wood S1 (site 93) in the middle reaches. In the lower reaches, very low fish density was recorded at Ribston Park Bottom S1 (site 109), Crimple mouth S1 (site 113), U/S Broad Wath Beck S1 and U/S Broad Wath Beck S3 (sites 121 - 123), Cattal U/S Bridge S4 (site 148), Cattal U/S Old Thornville S2 (site 153), Cattal D/S Old Thornville S2 (site 158), Tockwith S4 (site 164), Opposite Skewkirk S1 and Opposite Skewkirk S1 (sites 171 & 172) and, U/S Skip Bridge S1 and U/S Skip Bridge S3 (sites 179 - 181).

These findings on the diversity, density and assemblage were supported by the various diversity indices (Figs 4.49b, c, d & e). The ABC index discriminated 14% of sites (sites 1, 2, 5, 6, 7, 11, 16, 17, 25, 28, 29, 30, 44, 46, 55, 56, 60, 64, 87, 102, 103, 122, 123, 133 & 134) as poor. Two dendrograms based on fish density and biomass (Figs 4.50a & b) grouped the sites with poor abundance and biomass sites and isolated U/S Scotton Weir S1 and U/S Scotton Weir S2 (sites 33 & 34) in the upper and A1 Bridge S1 and U/S Skip Bridge S2 (sites 117 & 180) in the middle and lower reaches, respectively. The TWINSPAN analysis produced two different dendrograms with different site groupings (Figs 4.50c & d) indicating variable fish density in the river. The DECORANA analysis was not presented for this river due to an unknown error in the data analysis.

The fishery in the upper reaches was poor and dominated by rheophilic species. The upper reaches receives mine water discharge and effluent from STWs (EA, LEAP 1998b) and the stretch between D/S Birstwith S1 and Scotton weir S1 (sites 1 - 37), were subject to farm pollution (EA, LEAP 1998b).

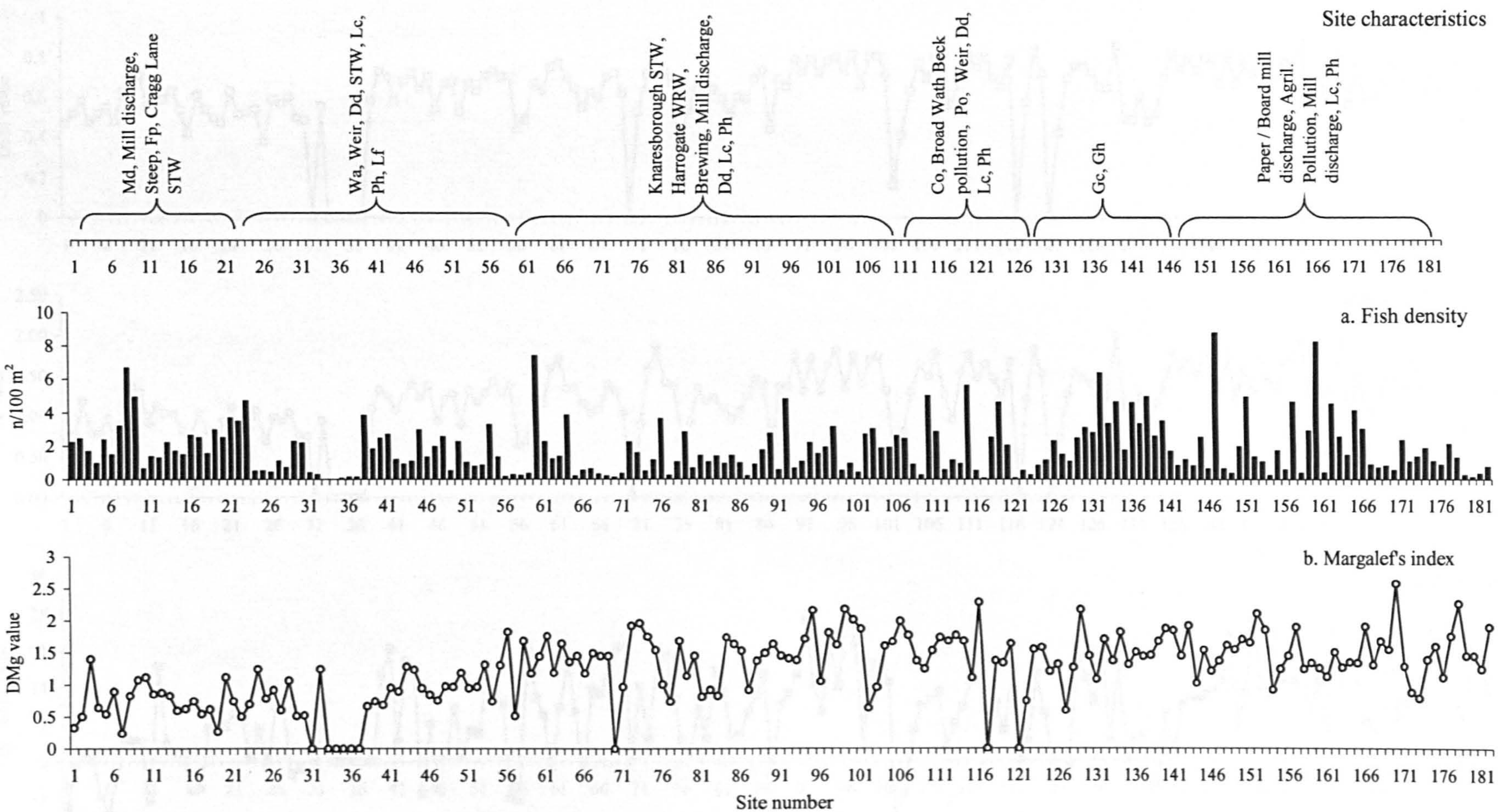


Fig. 4.49 Site characteristics and variations in different indices for the River Nidd

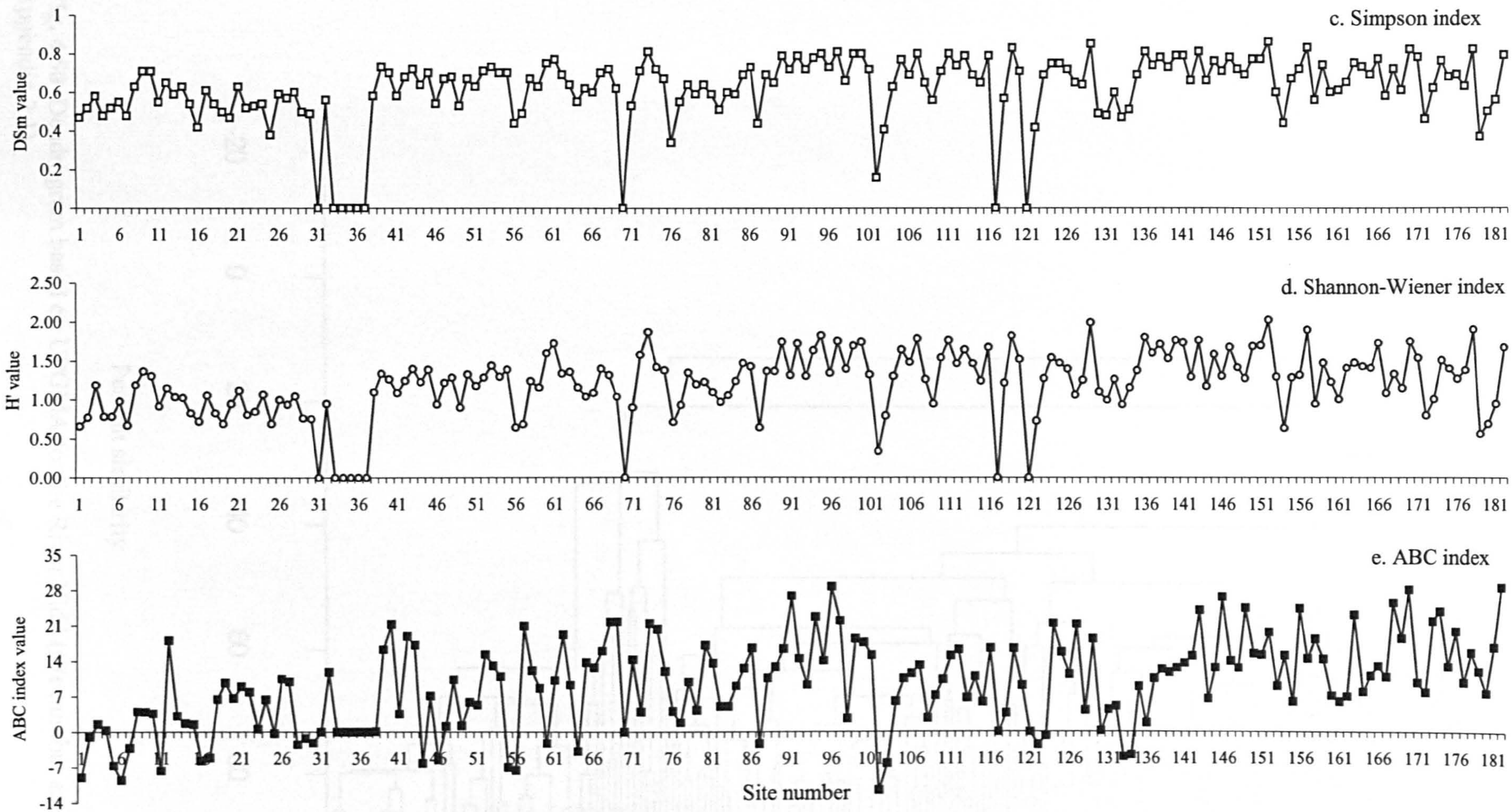


Fig. 4.49 (Continued) Site characteristics and variations in different indices for the River Nidd

a. Abundance

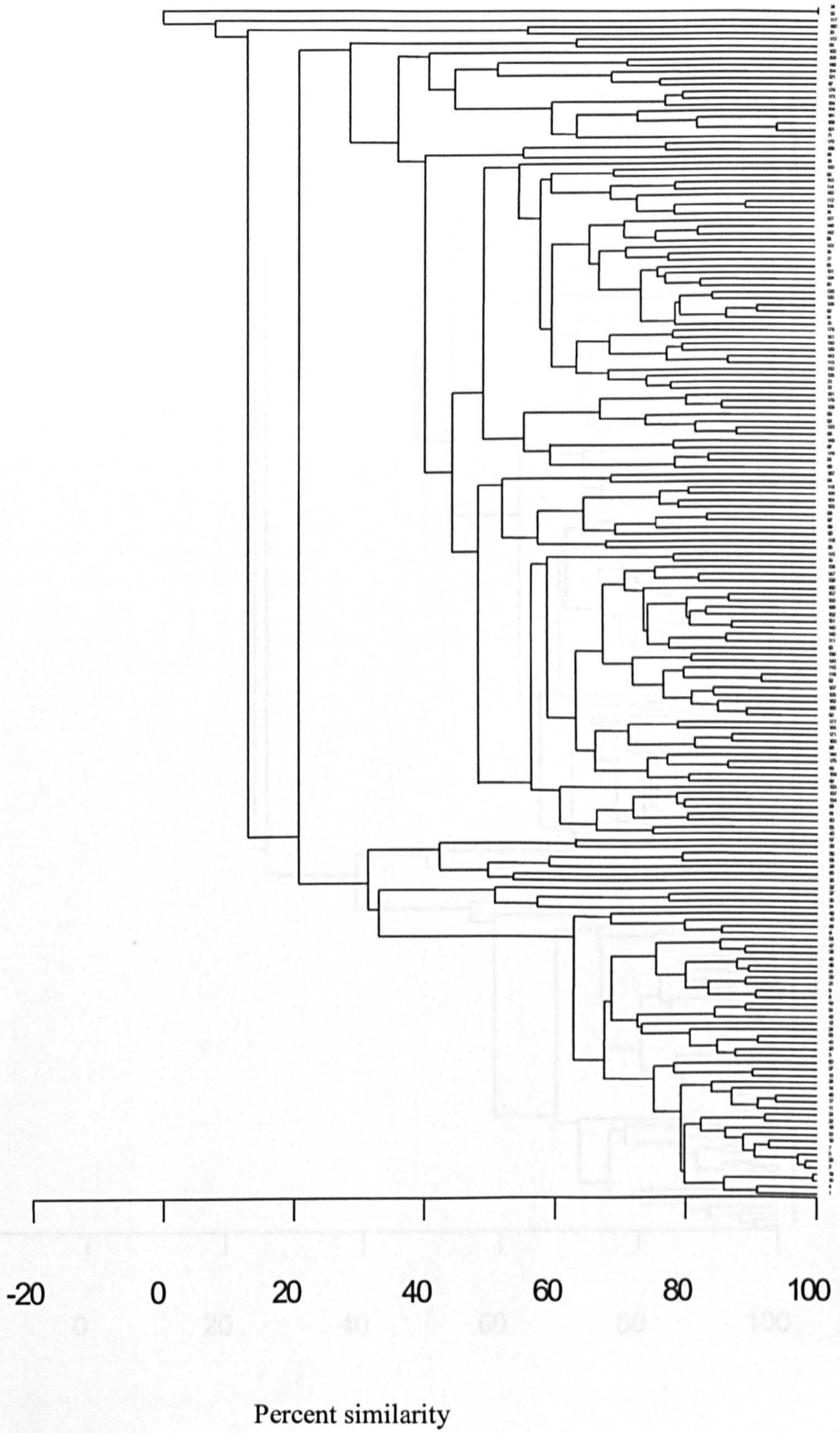


Fig. 4.50 Dendrogram based on UPGMA for the River Nidd (site number as in Appendix 2.1)

b. Biomass

Fig. 4.50 (Continued) Dendrogram based on UPGMA for the River Nidd (site number as in Appendix 2.1)

c. Abundance

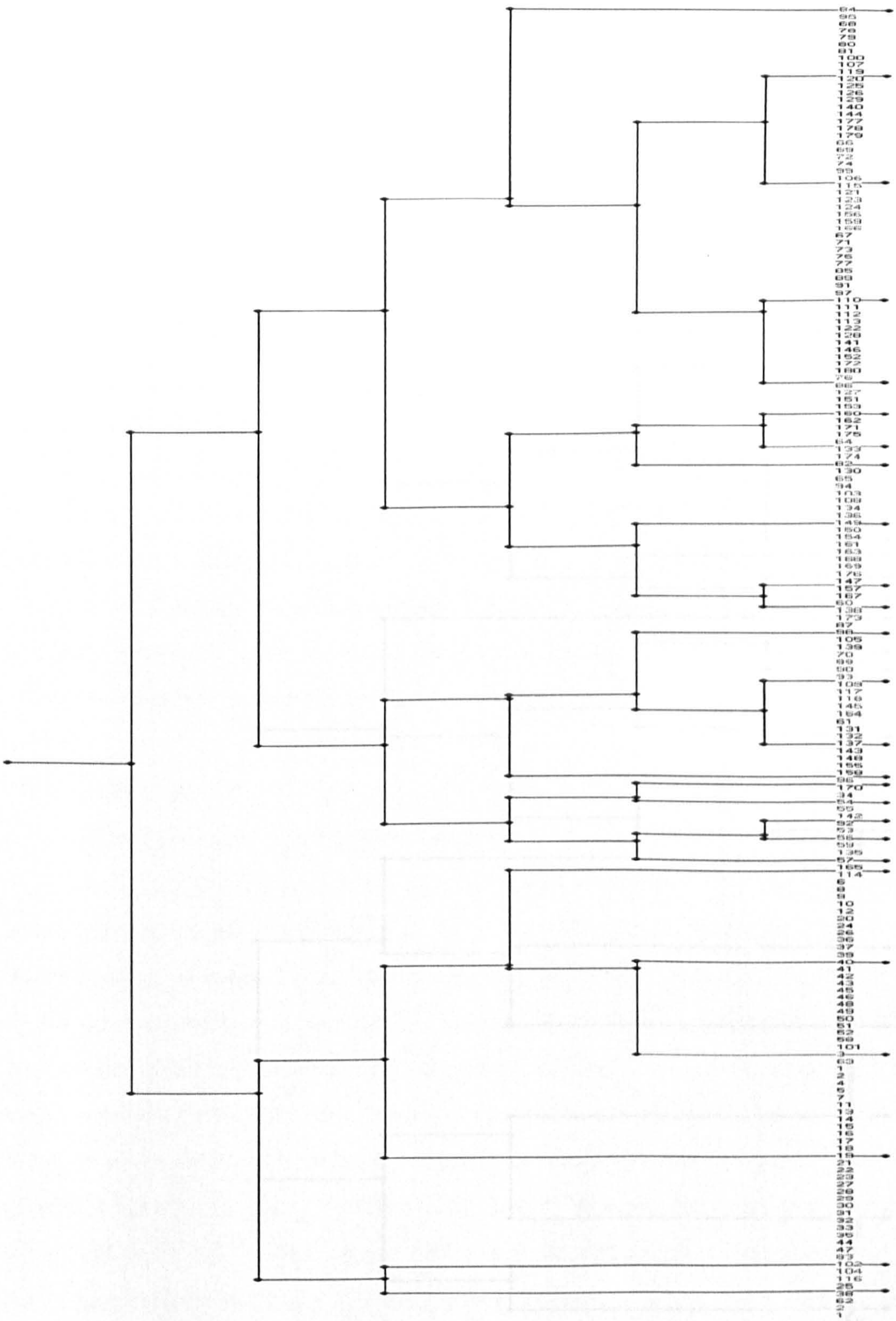


Fig. 4.50 (Continued) Results of the TWINSPAN for the River Nidd (site number as in Appendix 2.1)

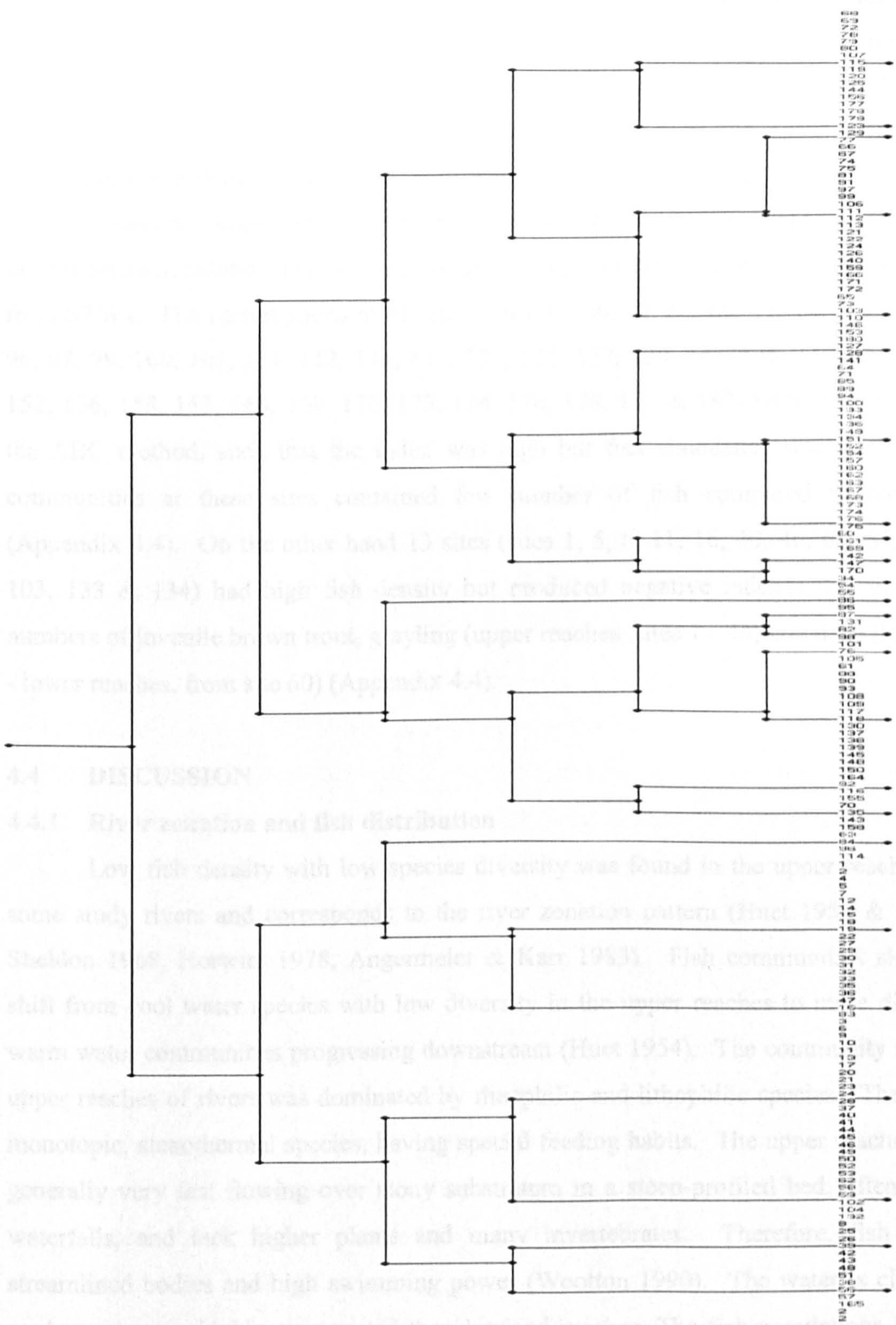


Fig. 4.50 (Continued) Results of the TWINSPAN for the River Nidd (site number as in Appendix 2.1)

The upper middle reaches between Conningham Hall S2 and U/S Little Ribson S2 (sites 54 - 90) had very poor stocks, due to pollution from domestic and industrial effluents. Natural habitat has been altered by constructing flood defence structures (EA, LEAP 1998b). With two exceptions, the lower section between Cowthorpe dog kennels S3 and Upstream Skipbridge S4 (sites 139 - 182) has very poor fish stocks because of farm pollution, mill discharges and lack of instream cover (EA, LEAP 1998b).

However, increased fish diversity was recorded in the lower reaches, suggesting improvement in habitat. This was due to improvement in the effluent quality, especially from STWs. The perturbations at 41 sites (sites 39, 40, 62, 68, 69, 73, 74, 80, 91, 94, 96, 97, 99, 100, 101, 111, 112, 116, 119, 124, 125, 127, 129, 143, 146, 149, 150, 151, 152, 156, 158, 163, 168, 169, 170, 173, 174, 176, 178, 181 & 182) were identified by the ABC method, such that the index was high but fish abundance was low. Fish communities at these sites contained few number of fish compared to biomass (Appendix 4.4). On the other hand 13 sites (sites 1, 5, 6, 11, 16, 30, 46, 60, 64, 102, 103, 133 & 134) had high fish density but produced negative indexes due to large numbers of juvenile brown trout, grayling (upper reaches, sites 1 - 46) and dace (middle - lower reaches, from site 60) (Appendix 4.4).

4.4 DISCUSSION

4.4.1 River zonation and fish distribution

Low fish density with low species diversity was found in the upper reaches of some study rivers and corresponds to the river zonation pattern (Huet 1954 & 1959, Sheldon 1968, Horwitz 1978, Angermeier & Karr 1983). Fish communities show a shift from cool water species with low diversity in the upper reaches to more diverse warm water communities progressing downstream (Huet 1954). The community in the upper reaches of rivers was dominated by rheophilic and lithophilic species. They are monotopic, stenothermal species, having special feeding habits. The upper reaches are generally very fast flowing over stony substratum in a steep-profiled bed, often with waterfalls, and lack higher plants and many invertebrates. Therefore, fish have streamlined bodies and high swimming power (Wootton 1990). The water is clearer, cooler and more highly oxygenated than lowland reaches. The fish populations in the upstream reaches are controlled principally by the abiotic characteristics of the environment (Mann 1995).

Distribution patterns of fish species were affected by habitat degradation in the rivers Churnet, Derwent, Idle, Sow, Tame, Tean, Trent, Aire and Nidd (Appendices 4.3, 4.5 & 4.6). However, to confirm river zonation patterns, it is necessary to survey all sections of these rivers. The sampled sections of these rivers had poor fish stocks due to poor water quality and / or habitat degradation (Section 4.3.2). All these rivers had common problems of poor habitat and low instream cover and pollution due to discharges from STWs / WRWs (Section 4.3.2). Pollution is a widespread cause of degradation in fisheries (Axford 1994, Firth 1997), not only because of the STWs and WRWs discharges but also because of non-point source inputs such as nutrient runoff, pesticides, and acid rain. River engineering is one of the major problems of the rivers Churnet, Idle, Trent and Aire (Section 4.3.2).

Fish populations were severely depleted by river channel works, such as dredging in the rivers Stort, Thame, Windrush, Blithe and Soar, and weed cutting in the rivers Anker, Stort and Derwent (Section 4.3.2). Eutrophication, caused by organic effluent, affected the River Anker, Sow, and Tame and the fisheries were dominated both in terms of biomass and numbers by species tolerant of eutrophic conditions, typically roach (Section 4.3.2). Fish populations in the upper reaches of the rivers Churnet, Derwent and Tean were affected by erosion resulting in a change of fish assemblage from monotopic to eurytopic species (Section 4.3.2). Dye works caused coloured water and affected fish populations in the rivers Churnet, Idle and Soar (Section 4.3.2). The rivers Idle and Nidd received mine water discharges, while fish populations in the River Tean were affected by mink, a piscivorous mammal (Section 4.3.2). The River Soar had impoverished fish stocks in the middle through lower reaches also due to anthropogenic disturbances including dye works, urban run-off, high suspended solids, gravel works, discharge from STWs and eutrophication (Section 4.3.2).

Fish distribution patterns were also affected by river impoundment and water abstractions and did not follow zonation theory in the rivers Blithe, Churnet, Derwent, Soar, Tame, Aire and Nidd. This is similar to the work of Hodgson (1993) in the River Dee, where zonation pattern did not fit because of the same reasons. Generally, where river zonation theory shows inconsistencies with observed data, man has impacted the river (Cowx 2001).

River management, maintenance practices and use of rivers are similar for many English rivers. Therefore, similar types of perturbations, mainly effluent discharge, water abstractions and river engineering were observed throughout the three catchments. The EA always try to minimise perturbations by imposing regulations on river use by different user groups, but it is necessary to pay special attention to particular problems, such as mink in the River Tean and mine water discharge in the rivers Idle and Nidd. In addition to minimising perturbations, the EA also carry out programmes to rehabilitate degraded sites / rivers to improve habitat and fish stocks. It is practically impossible to stop all disturbances in the river but it is possible to reduce disturbance rate to a minimal level and that will help natural distribution of fishes in English rivers.

4.4.2 River Continuum Concept (RCC) and fish species richness

Rich fish stocks with high fish species diversity were observed in the lower reaches of most study rivers. This agrees with the river continuum concept that predicted higher species richness in the lower reaches of rivers (Vannote *et al.* 1980, Barmuta & Lake 1982, Li *et al.* 1987, Naiman *et al.* 1988, Cowx & Welcomme 1988). The RCC explicitly predicts changes in fish community structure from headwater streams (1st to 3rd order) to large rivers (≥7th order). Fish stocks and species diversity in the lower reaches are usually controlled by complex interactions of biotic and abiotic factors and the fish community is dominated by cyprinids and predators. Most are eurytopic and eurythermal species, having capacity to thrive on wide variety of foods. Some species are euryhaline, having migratory habits. However, poor fish stocks with low species diversity were found in the lower reaches of the rivers, Thame, Windrush, Anker, Blithe, Blythe, Idle, Penk, Sence, Soar, Sow, Tame, Aire and Nidd, due to human-induced disturbances (Section 4.3.2). This agrees with the work of Horowitz (1978) and Cowx (2001). The RCC concept has been developed for natural, unperturbed stream ecosystem and usually deviates from the general pattern due to perturbations (Statzner & Higler 1985) as was found in this study.

Comparatively rich fish stocks and high species diversity was found in the middle reach of the rivers Cherwell, Stort, Thame, Windrush, Blithe, Cole, Sow, Aire and Nidd (Section 4.3.2). This also agrees with the RCC as midreaches show highest variability in the abiotic and biotic factors (Statzner & Higler 1985). Therefore, it may be said that with some exceptions, study rivers follow the RCC that describes the structure and function of fish communities along a river system.

4.4.3 Indices, multivariate analyses and fish assemblage

A number of indices were used to evaluate fish assemblage in the study rivers with each index having advantages and limitations. Washington's (1984) review of diversity indices found that there was no general consensus on which index is the most effective. Cairns (1977) stated that a diversity index is the best single means of assessing biological integrity in freshwater streams and is less effective or possibly inappropriate for lakes and seas. The Shannon-Weiner index H' has been criticised by several authors, e.g. Hulburt (1971) and Goodman (1975), for lacking in any biological significance but Heip & Engels (1974) recommended it. Eloronta and Eloronta (1977) considered H' a useful parameter for describing diversity of fish populations. However, May (1975) stated that H' is an insensitive measure of the character of species distribution and recommended the Simpson index. Peet (1974), Alatalo & Alatalo (1977) and Routledge (1979) also recommended the Simpson index to evaluate species diversity.

Magurann (1991) showed that of the indices used, H' has moderate discriminant abilities, moderate sensitivity to sample size and is richness biased. Margalef's index is a good discriminator, has high sensitivity and is richness biased. Simpson's index has moderate discriminant ability, low sensitivity and is dominance biased. Graphical techniques such as K-dominance plots (Shaw *et al.* 1983) and ABC curves (Warwick 1986) are useful for visual inspection of the structure of the fish communities in terms of abundance, species richness and biomass.

A wide variety of multivariate techniques based on software package are in common use. However, no one package includes all methods, and as such the techniques used often depend on the software available with little regard to the properties of the data or desired outcome. As yet there is no general consensus on which method should be used although it is generally agreed that a combination of differing techniques should be used when possible (Allen 1999).

Both simple and complex indices and multivariate techniques were used to examine fish communities in the study rivers. Margalef, Simpson and Shannon-Wiener indices were found suitable to evaluate fish communities in different study rivers of England. High Margalef's indexes were found where the number of species was high (e.g. rivers Derwent, Evenlode, Soar, Tame, Thame, Anker & Nidd) while the Simpson index decreased with the increase of species diversity (Section 4.3.2). Both Margalef and Simpson indices are simple to use to evaluate species diversity rapidly. Margalef's index is sensitive to sample size and sampling effort (Kempton 1979) while Simpson's index is based on random sampling and is sensitive to sample size and dominance. Margalef's index evaluated the conditions of the River Nidd efficiently, having high sample size but failed to explain the conditions of the river Idle, Mease, Sence and Tame with low sample size.

Simpson's index is heavily weighted towards the most abundant species in the sample while being less sensitive to species richness. Consequently, if a single species dominates a community (so that the community's diversity is low) then Simpson's index value will be high, while numerous species that all are fairly evenly present, will produce low index value (Magurann (1991). Both indices are based on structural properties of the fish community and failed to address the situation of juvenile-dominated sites in the rivers Anker, Blithe, Blythe, Cole, Penk, Sence, Soar, Tame, Tean and Trent (Section 4.3.2). These indices are ineffective in the upper reaches where the community is dominated by single species or a few species. The indices give an over estimate of the conditions in the lower reaches where species diversity is generally high but tend to exhibit high numbers of juveniles.

The H' assumes that individuals are randomly sampled from an infinitely large population (Pielou 1975). The index also assumes that all species are represented in the sample. The index has an advantage that H' estimates the diversity of the unsampled as well as sampled portion of the community (Magurran 1988). The H' is simple in computation and calculation but difficult to interpret. The H' is moderately sensitive to sample size and shows bias on species richness (Washington 1984). The index discriminated fish communities of the River Nidd effectively as the river was sampled randomly. However, H' was insensitive to discriminate fish communities of the rivers Idle, Mease, Sence and Tame where sample size was low. The H' generally increases (to an asymptote) with sample size (Allen 1999). The index also failed to explain the conditions associated with juvenile fish of a single or a few species in the rivers Anker, Blithe, Blythe, Cole, Penk, Sence, Soar, Tame, Tean and Trent.

Status and integrity of a fish community are based on structural and functional components. All diversity indices used in this study were based on structural component of a fish community and did not include functional attributes. All the indices successfully explained the species diversity but failed to address status and integrity of fish communities in the study rivers.

The ABC index appeared to provide a better assessment of fish communities in the study rivers than other indices. Fish communities in most of the study rivers were structurally imbalanced due to the presence of high numbers of juveniles. The ABC index addressed this situation in the rivers Tame, Stort, Windrush, Anker, Blythe, Cole, Soar and Trent. The results of ABC method were supported by other diversity indices as the ABC method included 'numerical diversity' (fish abundance), 'biomass diversity' (biomass) and total number of species from a community. Distribution of the numbers of individuals by species differed from the distribution of biomass between species in disturbed sites of all study rivers. The results agreed with Warwick (1986), Warwick *et al.* (1987) and Warwick & Ruswahyuni (1987). They concluded that the ABC index is a sensitive indicator of natural, physical and biological disturbance as well as pollution-induced disturbance over space and time. The index is applicable to the assessment of disturbance in fish communities before and after river channel works or natural and human induced river regulations (Coeck *et al.* 1993). All the study rivers impacted by man resulted in poor fish stocks with low species diversity and the ABC index successfully identified the perturbations. However, this technique did not include functional components of the fish community, which has great influence on the stability of a fish population. The ABC index was partially successful in explaining the status of the fish communities but failed to address the integrity.

Multivariate analysis was found to be effective in evaluating sampling sites on the basis of fish abundance and biomass. The dendrograms produced by UPGMA and TWINSPAN analysis successfully grouped and isolated river reaches with rich or poor fish stocks. Isolation of river reaches was also supported by different diversity indices. The UPGMA based on abundance and biomass produced similar groupings (rich and poor sites) in the rivers Cherwell, Blithe, Churnet, Cole, Derwent, Idle, Mease, Sence and Tean and different groupings in the rivers Evenlode, Stort, Windrush, Anker, Blythe, Penk, Soar, Sow, Tame, Trent and Nidd (Section 4.3.2). These groupings indicated structural imbalance in the fish communities related to fish abundance and biomass, which have been addressed by the ABC index. The UPGMA linked the average distance between samples with recalculation of linkage distance after each successive linkage and maximised the correlation between the original (dis) similarity matrix and the (dis) similarities between the samples in the dendrogram (Krebs 1999).

The TWINSPAN technique not only classified samples and species but also rearranges the species and samples into a two-way table that shows how the species distribution changes across the samples and lists the abundance class according to the ranges (Gauch 1982). The TWINSPAN based on fish abundance and biomass classified sites as poor or rich and produced dendrograms with similar groupings for the rivers Blythe, Derwent, Mease, Sence and Tean, but different groupings for the rivers Cherwell, Evenlode, Stort, Windrush, Anker, Blithe, Churnet, Cole, Penk, Soar, Sow, Trent and Nidd. Different groupings for abundance and biomass were due to high number of juveniles and low biomass as indicated by the ABC method.

However, neither analysis included the functional attributes of the fish communities, e.g. trophic structure and reproductive guilds of fishes, and were unable to address the integrity of the fish community. Fish density may be the same in the upper and lower reaches and hence two reaches may fall in the same group even though fish species diversity is different. This problem is not addressed in this study but it should be in future studies. Similarly a highly disturbed site having high density of a tolerant species may be grouped with a good site having similar density. The former can be explained by the river zonation theory and the latter by the Simpson index. These analysis are therefore, not enough to detect the integrity of a river system.

The DECORANA analysis discriminated sampling sites in an ordination plot on the basis of fish abundance and was supported by other diversity indices. Sites with uncommon or rare species were isolated successfully for the rivers Cherwell (common bream & common carp), Stort (brown trout & rainbow trout), Thame (barbel), Windrush (brown trout, rainbow trout & grayling), Anker (barbel), Blithe (grayling), Blythe (tench & common bream), Churnet (rainbow trout & rudd), Cole (crucian carp), Derwent (gudgeon & common bream), Mease (ruffe & common bream), Sence (eel), Soar (crucian carp & brown trout), Tame (rudd), Tean (gudgeon) and Trent (barbel) (Section 4.3.2).

The DECORANA analysis also isolated sites and species with high fish abundance, such as gudgeon in site 19 on the Cherwell, bleak in site 14 on the Stort, barbel in site 18 on the Thame, common carp in site 9 on the Anker, common bream and perch in site 9 on the Churnet, pike in site 8 on the Cole, roach in site 13 on the Derwent, common bream in site 1 on the Mease, eel in sites 5 & 6 on the Sence, crucian carp in site 7 on the Soar, roach and gudgeon in site 3 on the Trent (Section 4.3.2). The former conditions are supported by the Simpson's index while the later by the ABC index. The DECORANA analysis is sensitive to uncommon or rare species, which occur in a few sites (Minchin 1987) and is able to segregate fish community by fish abundance. This analysis also did not include functional components of fish communities. Consequently, output of DECORANA analysis is not sufficient to explain the status of a fishery and needs support from other indices.

4.5 SUMMARY

Before applying an index, method or technique for measuring ecological health, it is necessary to know the behaviour of the index, method or technique under normal environmental conditions and the underlying limitations of the data set on which the index is applied. In summary, a comparison of diversity indices, ABC method and multivariate techniques used in this study (Section 4.3.2) are presented in Table 4.2 and detailed the advantages and disadvantages of each method. In the study rivers, the diversity indices, ABC method and multivariate techniques appeared inappropriate to measure ecological health (Section 4.3.2) as the indices, method and multivariate techniques were only based on the structural component of fish communities (Section 4.2). Therefore, they were unable to evaluate the overall condition of the study rivers even when used in combination. Rather they tend to be affected by certain changes in the structural composition due to particular perturbation (Section 4.3.2). Therefore, a technique comprising both structural and functional attributes of fish community is needed to evaluate the over all condition or ecological health of a river. The next Chapter presents the assessment of the ecological health of the study rivers using a method that included both structural and functional attributes of fish communities.

Table 4.2 Comparison of diversity indices, ABC method and multivariate techniques used for study rivers

Indices / Method / Techniques	**Advantages**	**Disadvantages**	**Comments**
D_{Mg}	Easy to calculate. Deals with number of species and total number of individuals.	Sensitive to sample size. No information on abundance patterns. Does not consider the size of fish i.e. juvenile or adult.	Does not consider functional attributes of fish community. Gives partial information on fish community structure. Inaccurate assessment of ecological health of rivers.
D_{Sm}	Deals with relative abundance of each species and total number of individuals and hence provides more informative results than D_{Mg}.	Requires random sampling. Sensitive to sample size. Biased to dominant species. Low sensitivity to species richness and rare species. No information on abundance patterns.	As above.
H'	In addition to the relative abundance of each species and total number of individual individuals, the H' also uses natural log to provide more accurate results than the D_{Mg} and D_{Sm}. Uses evenness of abundance of species.	Requires random sampling. Requires infinitely large population for sampling. Requires presence of all species in any sample. Sensitive to sample size. Less sensitive to species richness and rare species. No information on abundance patterns. Difficult to interpret the output.	As above.

Table 4.2 (Continued). Comparison of diversity indices, ABC method and multivariate techniques used for study rivers

Indices / Method / Techniques	**Advantages**	**Disadvantages**	**Comments**
Ra	Based on total number of individuals and hence simple to calculate and easy to interpret the output.	Requires random sampling. Sensitive to sample size and sampling effort. Does not consider the size of fish i.e. juvenile or adult. No information on abundance patterns or species richness.	As above.
ABC	Based on difference between cumulative biomass and abundance of all species, and total number of species. Therefore, the index provides more accurate assessments than the D_{Mg}, D_{Sm} and H'.	Requires random samples of fish community. Sensitive to species richness and rare species. Based on macrobenthic communities and does not test thoroughly on fish communities. Difficult to interpret the output.	As above. More sensitive to pollution-induced disturbances than physical and biological disturbances.
UPGMA	Able to handle large data sets. Provides percent similarity of sites on the basis of fish abundance and biomass through cluster analysis. Diversity indices and the ABC method do not give such results.	More complex than the D_{Mg}, D_{Sm} H', Ra and ABC method. Requires random samples of fish community. Based on macrobenthic communities and does not test thoroughly on fish communities. Loss of information during output production. Difficult to interpret the output.	Does not consider functional attributes of fish community. Gives partial information on fish assemblages. Incomplete assessment of ecological health of rivers.

Table 4.2 (Continued). Comparison of diversity indices, ABC method and multivariate techniques used for study rivers

Indices / Method / Techniques	Advantages	Disadvantages	Comments
TWINSPAN	Able to handle large data sets. In addition to abundance and biomass, the TWINSPAN uses fish species richness, and classifies the sites and constructs an ordered two-way table from a sites-by-species matrix. Therefore provides more accurate measure of ecological health of rivers than diversity indices, ABC method and UPGMA.	Requires random samples of fish community. Based on macrobenthic communities and does not test thoroughly on fish communities. Loss of information during output production. Difficult to interpret the output.	Does not consider functional attributes of fish community. Gives partial information on fish community structure. Incomplete assessment of ecological health of rivers.
DECORANA	Able to handle large data sets. In addition to abundance, biomass and species richness, the DECORANA uses rare fish species to classify sites. Gives a low "weight" to a species that occurs in a few sites and minimises its influence in the assemblage, hence provides more accurate assessments of ecological health of rivers. Diversity indices, ABC method, UPGMA and TWINSPAN do not give such results.	Requires random samples of fish community. Based on macrobenthic communities and does not test thoroughly on fish communities. Loss of information during output production. Sensitive to species richness and rare species. Difficult to interpret the output.	As above

CHAPTER FIVE

5. INDEX OF BIOTIC INTEGRITY FOR ENGLISH RIVERS

5.1 INTRODUCTION TO IBI

Fish community characteristics have been used for many years to measure ecological health of waterbodies. Ecological health may be quantified by integrative ecological indices, which directly relate fish communities to other biotic and abiotic components of the ecosystem (Simon 1999). A variety of quantitative indices are used to define specific biocriteria. Indices include: indicator species or guilds; species richness, diversity, and similarity indices; the Index of Well-Being; multivariate ordination and classification; and the Index of Biotic Integrity (IBI) (Karr 1981). Of these, the most commonly used and it is suggested the most effective (Simon 1999), has been the Index of Biotic Integrity.

The index of biotic integrity (IBI) was developed to measure ecological health of a water body on the basis of biological criteria. Karr & Dudley (1981) defined biotic (or biological) integrity as " the ability to support and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organisation comparable to that of natural habitat of the region". The IBI was originally developed for use in Midwestern US streams characterised by undisturbed environments and relatively rich fish faunas (Karr *et al.* 1986). Many investigators have modified the IBI to assess degradation in a variety of ecoregions throughout the world (Hocutt 1981, Fausch *et al.* 1984 & 1990, Leonard & Orth 1986, Saylor & Scott 1987, Angermeier & Schlosser 1987, Miller *et al.* 1988, Ohio EPA 1988, Steedman 1988, Allen 1989, Schrader 1989, Fisher 1989, Crumby *et al.* 1990, Bramblett & Fausch 1991, Oberdorff & Hughes 1992, Hughes & Noss 1992, Gatz & Harig 1993, Angermeier & Karr 1994 & 1986, Gutierrez 1994, Oberdorff & Porcher 1994, Kerans & Karr 1994, Ribeiro *et al.* 1995, Shields *et al.* 1995, Simon & Emery 1995, Lyons *et al.* 1995 &1996, Didier & Kestemont 1996, Hugueny *et al.* 1996, Hay *et al.* 1996, Wallace *et al.* 1996, Chun *et al.* 1996, Koizumi & Matsumiya 1997, Liang & Menzel 1997, Ganasan & Hughes 1998 and Hughes & Oberdorff 1998).

The IBI has also been applied to various habitats such as estuaries (Thompson & Fitzhugh 1986, Deegan *et al.* 1997 and Weisberg *et al.* 1997) and lakes (Dionne & Karr 1992, Minns *et al.* 1994). They modified a variety of metrics for different ecological regions, waterbodies and specific applications but retained the basic ecological foundation proposed by Karr (1981).

However, scientists have found difficulty applying the index in a system with naturally depauperate fish faunas (Leonard & Orth 1986, Hughes & Gammon 1987, Miller *et al.* 1988, Schrader 1989). Problems arise because the ecological framework of the IBI relies on nominal levels of taxonomic diversity, as well as on diversity in trophic guilds and levels of tolerance to environmental degradation (Hay *et al.* 1996). Moreover, ecology of fish communities varies with the geographical location of the system (Cowx & Welcomme 1998), which may hamper suitable modification of the IBI to detect degradation level. The main objectives covered in this Chapter are to select IBI metrics, to develop rating and scoring systems for metrics and to calculate IBIs for a number of English lowland rivers.

5.2 CONCEPT OF IBI

The IBI (Karr 1981) is an ecologically based index used to assess degradation of aquatic ecosystems. The method integrates attributes of communities, populations, and individual organisms to assess biological integrity on the basis of accurate measures of relative abundance. The main advantages are that IBI is sensitive to different sources of degradation and that it produces biologically meaningful and reproducible results when applied by competent fish biologists (Fausch *et al.* 1990). However, the IBI does not replace chemical and toxicological methods but it does increase the probability that an assessment programme will detect degradation due to anthropogenic influences (Angermeir & Karr 1994).

The logical foundations of the IBI are easily adaptable to fish assemblages in the rivers of tropical and temperate regions and are applicable to a waterbody with no ichthyofauna (Fausch *et al.* 1990). The method is also effectively applicable to a fish community having only one or two families (Ganasan & Hughes 1998). The IBI is suitable for use in other aquatic environments and with taxa other than fish (Fausch *et al.* 1990) such as aquatic macrophytes (Canfield & Jones 1984), amphibians (Moyle *et al.* 1986) and macroinvertebrates (Schaeffer *et al.* 1985). The IBI is a flexible index, which needs modification, reduction, or addition of the metrics to reflect regional differences in fish distribution and assemblage structure (Hughes & Oberdorff 1999). A useful feature of the IBI is that direct observations on many aspects of fish communities are used. This limits the possibility of errors (Steedman 1988).

5.3 ESTABLISHING IBI METRICS

The original IBI (Karr 1981, Karr *et al.* 1986) consisted of 12 community attributes, termed metrics, grouped into three broad categories: species richness and composition, trophic composition, and fish abundance and condition. Geographical differences in fish communities require modification of the IBI metrics for basins or ecoregions (Fausch *et al.* 1990). Ecoregions are defined as areas of homogeneous ecological systems or areas that have the potential (if undisturbed) for similar biological communities (Omernik 1987). Basic ecological information needed to modify the IBI for a new ecoregion includes knowledge of which fish species are native or introduced, their trophic, reproductive and habitat guilds, and their relative tolerance to environmental degradation (Simon 1999). The original version of IBI has been modified to apply in new ecoregions according to local biodiversity by various authors. Worldwide modifications of IBI metrics are tabulated in Table 5.1.

Table 5.1 Metric changes for IBIs developed in countries other than the USA and Canada (Simon 1999)

Original metric (Karr *et al.* 1986)	Substitute	Country	Reference
1. Total number of fish species	No change	France	Oberdorff & Hughes (1992)
		France	Oberdorff (1996)
		Guinea	Hugueny *et al.* (1996)
		India	Ganasan & Hughes (1998)
		Australia	Harris (1995)
		Venezuela	Gutierrez (1994)
	Native fish species	France	Oberdorff & Porcher (1994)
		Belgium	Didier *et al.* (1996)
		Namibia	Hocutt *et al.* (1994)
		Mexico	Lyons *et al.* (1995)
	Number of families	India	Ganasan & Hughes (1998)
2. Number and identity of darter species	Benthic species	France	Oberdorff & Hughes (1992)
		France	Oberdorff & Porcher (1994)
		Belgium	Didier *et al.* (1996)
		India	Ganasan & Hughes (1998)
	Benthic specialists	France	Oberdorff (1996)
		Namibia	Hocutt *et al.* (1994)
	Riffle benthic species	Australia	Harris (1995)
	% Native benthic individuals	Mexico	Lyons *et al.* (1995)
	Mormyrid species	Guinea	Hugueny *et al.* (1996)
	Characidiin / parodontin species	Venezuela	Gutierrez (1994)
3. Number and identity of sunfish species	Water column species	France	Oberdorff & Hughes (1992)
		India	Ganasan & Hughes (1998)
		Mexico	Lyons *et al.* (1995)
	Cichlid species	Guinea	Hugueny *et al.* (1996)
		Namibia	Hocutt *et al.* (1994)
		Venezuela	Gutierrez (1994)
	Pelagic pool species	Australia	Harris (1995)
	Deleted	France	Oberdorff & Porcher (1994)
		Belgium	Didier *et al.* (1996)

Table 5.1 Continued

Original metric (Karr *et al.* 1986)	**Substitute**	**Country**	**Reference**
4. Number and identity of sucker species	Trout / pike age classes	France	Oberdorff & Hughes (1992)
	Trout/pike/perch ages	France	Oberdorff (1996)
	Trout age classes	France	Oberdorff & Porcher (1994)
	Dominant / intolerant age classes	Belgium	Didier *et al.* (1994)
	Large siluriform species	Guinea	Hugueny *et al.* (1996)
	Loricariid species	Venezuela	Gutierrez (1994)
	Pool benthic species	Australia	Harris (1995)
	Pelagic / rheophilic species	Namibia	Hocutt *et al.* (1994)
	Benthic species metric	India	Ganasan & Hughes (1998)
	% benthic individuals metric	Mexico	Lyons *et al.* (1995)
	Deleted	India	Ganasan & Hughes (1998)
		Mexico	Lyons *et al.* (1995)
5. Number and identity of intolerant species	No change	France	Oberdorff & Hughes (1992)
		France	Oberdorff (1996)
		Belgium	Didier *et al.* (1996)
		India	Ganasan & Hughes (1998)
		Australia	Harris (1995)
		Venezuela	Gutierrez (1994)
	% intolerant individuals	Belgium	Didier *et al.* (1996)
	Sensitive species	Mexico	Lyons *et al.* (1995)
	% Sculpin individuals	France	Oberdorff & Porcher (1994)
	Deleted	Guinea	Hugueny *et al.* (1996)
		Namibia	Hocutt *et al.* (1994)
6. Proportion of individuals as green sunfish	% Tolerant individuals	India	Ganasan & Hughes (1998)
		Mexico	Lyons *et al.* (1995)
	% Roach	France	Oberdorff (1996)
		France	Oberdorff & Hughes (1992)
	% Eel and roach	France	Oberdorff & Porcher (1994)
	% Alien / invasive individuals	Belgium	Didier *et al.* (1996)
	% Native individuals	Australia	Harris (1995)
	% Dominants	Venezuela	Gutierrez (1994)
	Deleted	Namibia	Hocutt *et al.* (1994)
		Guinea	Hugueny *et al.* (1996)
7. Proportion of individuals as omnivores	No change	France	Oberdorff & Hughes (1992)
		France	Oberdorff & Porcher (1994)
		France	Oberdorff (1996)
		Belgium	Didier *et al.* (1996)
		Guinea	Hugueny *et al.* (1996)
		India	Ganasan & Hughes (1998)
		Mexico	Lyons *et al.* (1995)
	% Omnivorous species	Venezuela	Gutierrez (1994)
	% species as herbivores	Venezuela	Gutierrez (1994)
	% species as detritivores	Venezuela	Gutierrez (1994)
	% species as parasites	Venezuela	Gutierrez (1994)
	% Scavenger individuals	Namibia	Hocutt *et al.* (1994)
	% herbivore / detritivore individuals	Namibia	Hocutt *et al.* (1994)
	% Microphagic omnivorous individuals	Australia	Harris (1995)
8. Proportion of individuals as insectivorous cyprinids	% Invertivorous individuals	France	Oberdorff & Hughes (1992)
		France	Oberdorff & Porcher (1994)
		France	Oberdorff (1996)
		Guinea	Hugueny *et al.* (1996)
		Namibia	Hocutt *et al.* (1994)

	% Microphagic carnivorous individuals	Australia	Harris (1995)
	% Herbivore individuals	India	Ganasan & Hughes (1998)
	% Insectivore species	Venezuela	Gutierrez (1994)
	Deleted	Belgium	Didier *et al.* (1996)
		Mexico	Lyons *et al.* (1995)
9. Proportion of individuals as piscivores (top carnivores)	No change	France	Oberdorff & Hughes (1992)
		India	Ganasan & Hughes (1998)
		Guinea	Hugueny *et al.* (1996)
		Namibia	Hocutt *et al.* (1994)
	% Piscivorous / invertivorous individuals	Belgium	Didier *et al.* (1996)
	% Macrophagic carnivorous individuals	Australia	Harris (1995)
	% Piscivorous species	Venezuela	Gutierrez (1994)
	Deleted	France	Oberdorff & Porcher (1994)
		France	Oberdorff (1996)
		Mexico	Lyons *et al.* (1995)
10. Number of individuals in sample	No change	India	Ganasan & Hughes (1998)
		Guinea	Hugueny *et al.* (1996)
		Namibia	Hocutt *et al.* (1994)
		Australia	Harris (1995)
	Catch / effort	France	Oberdorff & Hughes (1992)
		Mexico	Lyons *et al.* (1995)
	Individuals / 100 m^2	France	Oberdorff & Porcher (1994)
		France	Oberdorff (1996)
	Total biomass	France	Oberdorff & Porcher (1994)
	Biomass	Belgium	Didier *et al.* (1996)
	Deleted	Venezuela	Gutierrez (1994)
11. Proportion of individuals as hybrids	No change	Guinea	Hugueny *et al.* (1996)
	% Gravel spawning individuals	France	Oberdorff & Hughes (1992)
	% Generalist spawning individuals	Belgium	Didier *et al.* (1996)
	% Alien / invasive species	Belgium	Didier *et al.* (1996)
	% Specialist spawner individuals	Belgium	Didier *et al.* (1996)
	% Native livebearing individuals	Mexico	Lyons *et al.* (1995)
	% Alien species	Mexico	Lyons *et al.* (1995)
	% Alien / invasive individuals	Namibia	Hocutt *et al.* (1994)
	% Alien individuals	India	Ganasan & Hughes (1998)
	% Introduced species	Venezuela	Gutierrez (1994)
	% Native species	Australia	Harris (1995)
	Deleted	France	Oberdorff & Porcher (1994)
		France	Oberdorff (1996)
12. Proportion of individuals with disease, fin damage, and skeletal anomalies	No change	France	Oberdorff & Hughes (1992)
		France	Oberdorff & Porcher (1994)
		France	Oberdorff (1996)
		Guinea	Hugueny *et al.* (1996)
		Namibia	Hocutt *et al.* (1994)
		India	Ganasan & Hughes (1998)
		Mexico	Lyons *et al.* (1995)
		Australia	Harris (1995)
		Venezuela	Gutierrez (1994)
	Deleted	Belgium	Didier *et al.* (1996)

5.3.1 Selection of IBI metrics for English rivers

The number of metrics in the different versions of IBI vary between 6 in Belgium, (Didier 1997) and 22 in USA (Lyons *et al.* 1996), according to ecoregion, biodiversity and watershed. The second highest number of metrics (19) was adopted for the rivers of Namibia, (Hay *et al.* 1996) (Appendix 5.1). Although, the number and identity of metrics differ among different versions of the IBI, all versions have metrics that measure both structural and functional characteristics of fish communities. Asian versions of IBI are based on 10 (Japan) to 12 (India) metrics (Koizumi & Matsumiya 1997, Koizumi *et al.* 1997, Ganasan & Hughes 1998) while Australian and South American (Venezuela) versions were developed on 12 and 13 metrics, respectively (Gutierrez 1994, Harris 1995). The literature details thirty-two versions of IBI that have been developed for streams and rivers around the world to date (Appendix 5.1).

Retaining the basic ideas, a number of modifications and additions were made to structure a new version of the IBI for English rivers. Initially 19 candidate metrics were tentatively selected to calculate IBIs for English lowland rivers. The metrics "percentage of individuals with deformities, eroded fins, lesions, and tumours", "percentage of hybrids", "percentage of standard growth of fishes" and "percentage of juvenile fishes" were not tested due to lack of existing data. English rivers have low species and habitat diversities (Sections 3.3.2 & 3.3.3) and generally suffer similar types of perturbations (Section 4.3.2). With these factors in mind, 15 metrics were selected to calculate the IBI. These metrics are aimed at English rivers. The metrics were classified into 4 broad categories: species richness and composition, habitat composition, trophic composition and, fish abundance and biomass (Table 5.2). The justification for the choice of each metric is given below.

Species richness and composition

Four metrics were considered within the "Species richness and composition" category. "Total number of native species" was preferred to the "total number of species" initially proposed by Karr *et al.* (1986), which also included alien (introduced) species. Numbers of native species are low in UK rivers and some have become extinct (e.g. burbot). Moreover, some species are at risk of extinction as they already are extinct from a specific area (Table 3.1). It is extremely important to use this metric for English lowland rivers. This metric was used in 16 versions of IBI around the world (Appendix 5.1). Some alien species such as common carp, goldfish, bitterling, pikeperch, ide and rainbow trout have been present in English rivers for a long time and some West European authors assimilated them as native species (Spillman 1961). However, no naturalised species was considered as a native species for English rivers. It is expected that naturalised fish species will not significantly affect the results of this study as diversity and density of such species are generally low in English rivers (Tables 3.3 & 3.4).

Table 5.2 Details of modified IBI metrics adopted for English rivers

Category	Metric	Expected trend in fish community structure after degradation
Species richness	1. Total number of native fish species	Declining
	2. Percentage of individuals as non-natives	Increasing
	3. Number of intolerant species	Declining
	4. Percentage of individuals as tolerant species	Increasing
Habitat composition	5. Number of water-column species	Declining
	6. Number of benthic species	Declining
	7. Percentage of individuals as rheophilic species	Declining
	8. Percentage of individuals preferring vegetated areas	Declining
	9. Percentage of individuals as gravel spawners	Declining
Trophic composition	10. Percentage of individuals as omnivores	Increasing
	11. Percentage of individuals as invertivores	Declining
	12. Percentage of individuals as piscivores	Declining
Fish abundance and biomass	13. Number individuals of long-lived species (No. 100 m^{-2})	Declining
	14. Number of individuals in a sample (No. 100 m^{-2})	Declining
	15. Total biomass (g m^{-2})	Declining

"Total number of native fish species" (Table 5.2) is a strong overall indicator of ecosystem health. The metric measures the species richness component of diversity (Boet *et al.* 1999). This metric is based on the hypothesis that a disturbed environment will have fewer native species than an undisturbed one, as species that are intolerant of the disturbance will be absent.

Table 5.3 Summary of the reference condition for English lowland rivers (This table is replicated here from Chapter 3)

Criteria	Reference number	Table / Species
Maximum expected number of common native species in lowland rivers	18	See Table 3.4 & Section 3.3.2
Maximum expected exotic species in lowland rivers	10	Common carp, pikeperch, bitterling, rainbow trout, black bullhead, goldfish, sunbleak, ide, wels and asp
Tolerance		
Intolerant species	5	Barbel, minnow, chub, dace and bleak
Tolerant species	13	Crucian carp, tench, roach, common bream, silver bream, 3-spined stickleback, 10-spined stickleback, rudd, gudgeon, pike, perch, ruffe and eel
Habitat guild		
Limnophilic (Typically vegetation loving species)	12	Crucian carp, tench, roach, common bream, silver bream, 3-spined stickleback, 10-spined stickleback, rudd, pike, perch, ruffe and eel
Rheophilic species	6	Barbel, minnow, chub, dace, bleak and gudgeon
Water-column species	10	Minnow, chub, dace, bleak, 3-spined stickleback, 10-spined stickleback, rudd, pike, perch and ruffe
Benthic species	8	Barbel, crucian carp, roach, tench, common bream, silver bream, gudgeon and eel
Trophic guild		
Omnivores	9	Crucian carp, tench, roach, common bream, silver bream, rudd, chub, 3-spined stickleback and 10-spined stickleback
Invertivores	5	Barbel, minnow, dace, bleak and gudgeon
Piscivores	4	Pike, perch, ruffe and eel
Reproductive guild		
Phytophilic species	5	Crucian carp, tench, rudd, perch, ruffe
Phytolithophilic species	5	Roach, common bream, silver bream, pike and bleak
Lithophilic species	4	Barbel, minnow, chub and dace
Total gravel spawners	9	Barbel, minnow, chub, dace, roach, common bream, silver bream, pike and bleak
Psammophils	2	Gudgeon
Nest builders	2	3-spined stickleback and 10 spined stickleback
Abundance		
Number of individuals of long-lived species used in this study (No 100 m^{-2})	≥25	Chub and common bream (Section 3.3.7 & Table 3.9)
Number of individuals in sample (Fish 100 m^{-2})	≥200	Section 3.3.4
Biomass for English lowland rivers (g m^{-2})	≥35	Section 3.3.4

Therefore, total number of native species decreases with increased degradation. This metric was applied by Oberdorff & Porcher (1994), Didier *et al.* (1996) and Kestemont *et al.* (2000) for European rivers. Most English rivers are affected by different anthropogenic activities and thus the metric "total number of native species" was considered useful for evaluating fish stocks. Minor and migratory species were not included in "total number of native fish species" metric. Most minor species live in upper reaches and lowland areas are not their preferred habitat. As the objective of this study is to calculate IBIs for lowland rivers based on freshwater species, minor and migratory species were therefore, not included in "total number of native fish species". If minor and migratory species are included in "total number of native fish species", total IBI score will be different for the lowland reaches, leading to an unreliable assessment of the ecological health.

Non-native or introduced species create problems for the native species by occupying their niche and trophic level. It is therefore, important to separate non-natives from native species. If non-natives are included in the "total number of native fish species" metric, then the scoring criteria would be raised so that a site with no introduced species could receive a lower score than a site with non-natives (Ganasan & Hughes 1998). Therefore, a new metric was added as "percentage of individuals as introduced / non-native species" to measure the degree of fish community degradation. The percent non-natives reflects biological pollution, which is usually more difficult to reverse than chemical and physical disturbance. This metric is based on the hypothesis that a biologically undisturbed water body will have no non-natives and a disturbed site will have a higher density therein. This metric includes the species introduced intentionally or accidentally and gives a measure of the degree that invasive alien species dominate the assemblage. Non-native species are generally more successful where native species are depauperate or in anthropogenically altered systems (Ross 1991).

British rivers are apparently susceptible to invasion by non-natives, as 41 exotic species have been introduced into the country (Table 3.2). This is probably due to the comparatively low number of native species and the presence of vacant niches. Although most alien species have not been successful in terms of reproductive potential in the wild a few (e.g. pikeperch) are exerting a potential threat to the native riverine fish populations (Hickley & North 1983, Hickley 1986). Common carp, goldfish, ide, rainbow trout, pikeperch and sunbleak are found in a number of English rivers (Wheeler 1977). Therefore, use of the metric will be helpful to evaluate the biotic integrity of these rivers. The use of non-natives / introduced species as a metric is considered appropriate because these species generally disrupt biotic integrity (Karr & Dudley 1981). The percentage of non-natives was also used by Hughes & Gammon (1987), Schrader (1989), Crumby *et al.* (1990) and Bramblett & Fausch (1991). This metric was adopted in 10 versions of IBI in different parts of the world (Appendix 5.1). Ganasan and Hughes (1998) successfully applied the metric to evaluate the ecological health of Indian rivers. However, the metric has not been used to evaluate ecological health of African or European rivers (Oberdorff & Hughes 1992, Hay *et al.* 1996, Hugueny *et al.* 1996, Kestemont *et al.* 2000).

The metric "number of intolerant species" was considered as one of the main indicators of water quality (Table 5.2). The metric was retained for English rivers as the rivers harbour a number of intolerant species (Cowx 2001) and are subjected to anthropogenic disturbances. "Number of intolerant species" is considered to be a powerful metric, which is able to detect minimal disturbances created by anthropogenic activities (Karr *et al.* 1986). The metric is based on the hypothesis that the number of intolerant species will be low in a disturbed site and will be zero in highly disturbed areas. The number of species intolerant of various physical, chemical and biological habitat perturbations will distinguish high and moderate quality sites (Karr *et al.* 1986). Intolerant species are usually the first species to disappear after a disturbance and are the last to reappear after restoration (Oberdorff & Hughes 1992). Endangered or threatened species, however, are not considered intolerants because their low numbers may be due to factors other than perturbation. They might, for example, be glacial relics. If a high number of intolerant species is included in this metric, the usefulness is reduced because intolerants are most often found only when stream conditions are good to excellent. However, Hughes & Oberdorff (1999) suggested the retention of this metric as it detects initial signs of ecosystem perturbation. This metric was retained in 24 versions of IBI around the world (Appendix 5.1) and was also used in seven of ten modifications of the IBI for use outside the USA and Canada (Hughes & Oberdorff 1998).

Although most authors (Karr *et al.* 1986, Hughes & Gammon 1987, Oberdorff & Hughes 1992, Richard 1994, Lyons *et al.* 1995 & 1996, Shields *et al.* 1995, Didier & Kestemont 1996, Ganasan & Hughes 1998, Kestemont *et al.* 2000) used a classification scale (tolerant, intolerant, intermediate etc.), the metric used in this study integrated most degrees of tolerance to water quality degradation. However, for West African rivers, this metric was deleted due to limited knowledge of species responses to general disturbance (Hugueny *et al.* 1996), which is not a problem for English species.

The percentage of tolerant species increases with increased physical and chemical habitat degradation. Tolerant species are the last to disappear following a disturbance and the first to return as the system begins to recover (Lyons *et al.* 1996); several have accessory respiratory organs that allow them to move on land areas with little water and exist in waters lacking dissolved oxygen or in isolated pools. Usually they are generalists having ability to thrive on a wide range of foods (Goldstein &

Simon 1999). The European eel is able to survive with low dissolved oxygen in turbid and muddy waters and may move from one water body to another through the wet grasslands while the bullhead is able to utilise atmospheric air for intestinal respiration (Wootton 1990).

Roach, common bream and gudgeon (Table 5.3) were selected to replace "green sunfish" as the tolerant species in the English rivers because the latter is absent from the UK and the former are widespread in medium to low gradient rivers and appear tolerant of many pollutants (Cowx 2001). Therefore, the metric "percentage of individuals as green sunfish" was replaced with "percentage of individuals as tolerant species". In France, "green sunfish" was replaced by "percentage of individuals as roach" (Oberdorff & Hughes 1992), while the metric was adopted as "percentage of individuals intolerant to dissolved oxygen and ammonia" in Belgium (Kestemont *et al.* 2000). The metric " percentage of individuals as tolerant species" is used in 18 versions of IBI (Appendix 5.1) and is directly related to the assessment of water quality degradation and will be able to distinguish low and moderate quality waters. English rivers have several tolerant fish species (Cowx 2001), their tolerance limit is thoroughly studied and well documented (Varley 1967) and hence it was considered suitable to use this metric to evaluate biotic integrity of these rivers.

Habitat composition

Three metrics specific to North America ("number of darter species", "number of sunfish species" and "number of sucker species") were replaced with "number of water column species" and "number of benthic species". All were suggested by Karr *et al.* (1986) and these metrics are common substitutes when the IBI is modified for use outside the USA (Hughes & Oberdorff 1999). These two modified metrics are strongly responsive to changes in water quality and habitat structure. Increased diversity, size and overall quality of benthic and water column habitats are associated with increased richness of benthic and water column species (Karr *et al.* 1986, Hughes *et al.* 1998).

Water-column species (Table 5.3) are active swimmers that typically feed on drifting and surface invertebrates or other fishes. These fishes are sensitive to the degradation of pool habitats and instream cover (Karr 1981). Use of the water-column metric was considered as such species occur frequently in English rivers (Wheeler 1983), and erosion and siltation affect habitats of these species. Hughes & Oberdorff (1999) suggested this metric be used unless these species are absent, unresponsive to the disturbance, or have highly variable abundance. This metric was used in 7 versions of

IBI, developed for streams and rivers of North America, Asia, Europe and South America (Appendix 5.1).

A number of benthic species are found in English rivers (Table 3.10). Benthic species, are impacted by siltation, turbidity, toxic chemicals and benthic oxygen depletion because they feed and reproduce in benthic habitats. In English rivers, dredging and erosion affect the habitats of benthic species. Any change in benthic habitat will adversely affect growth, reproduction, density and assemblage of benthic species (Cowx 2001). Therefore, "number of benthic species" was selected as a metric to evaluate ecological health of English rivers (Table 5.2). The metric was retained in 12 versions of IBI around the world (Appendix 5.1).

Each fish species has preferred habitat requirements, which result in changes in community structure along the upstream-downstream gradient of a river. Moving from the headwaters downstream Huet (1959) described a trout, grayling, barbel and bream zonation pattern (Fig. 3.1). Anthropogenic disturbance results in the change of fish distribution in these zones. Therefore, two new metrics, "percentage of individuals as rheophilic species" and "percentage of individuals preferring vegetated areas", were added to reflect the habitat conditions of the river system (Table 5.2).

Rheophils are commonly found in riffles and rapids, and are generally hidden in rock crevices, under pebbles or gravel (Table 5.3). They prefer well-oxygenated high quality water for their survival, breeding, growth and recruitment (Cowx & Welcomme 1998). Their distribution and abundance are affected by the alteration of flow regime due to dams, weirs and other structures. The metric "percentage of individuals as rheophilic species" might prove especially useful as many English rivers are characterised by the presence of dams and weirs (Petts 1984). Species such as trout, grayling, minnow, barbel and dace live in the upper reaches of the river with fast flow conditions while common bream, common carp, tench, and roach are found in the more lowland sections with low to moderate flow conditions (Cowx 2001). In their recent synthesis, Cowx & Welcomme (1998) summarised the preferred water velocities for reproduction of coarse fishes in European waters (Table 5.4). Any change in habitat will adversely affect these species, which will limit their distribution, abundance, assemblage, growth and recruitment. Therefore, rheophils will give a measure of the flowing water habitat. This metric was used in 5 versions of IBI developed for the streams and rivers of Africa, Australia and Europe (Appendix 5.1).

Vegetation is important for feeding, spawning, shelter and cover for individuals from adverse flow conditions and predation (Cowx 2001). Without vegetation cover, populations of species such as pike and perch may become dominated by single age groups that may lead to an improvement in recruitment success of cyprinids through a reduction in predation pressure (Cowx *et al.* 1995) or vice versa. Abundance of species preferring vegetated habitats such as tench, common carp, crucian carp and common bream, will reflect whether the aquatic vegetation or floodplains have been damaged. The most likely causes of damage are agricultural activities, overgrazing, deforestation and soil erosion (Hay *et al.* 1996, Hughes & Oberdorff 1999). Soil erosion will result in siltation of the system and is likely to be reflected by the abundance and composition of benthic species. The abundance and composition of aquatic vegetation in English rivers is affected by river maintenance works and erosion. For example, the removal of midstream and riparian vegetation by dredging reduced the populations of chub and roach by up to 64% in some sections of the River Thames (Armstrong 1983). Therefore, the metric, "percentage of individuals preferring vegetated areas" was included to evaluate biological integrity of English rivers (Table 5.2). This metric was adopted in 3 versions of IBI (Appendix 5.1).

Table 5.4 Range velocities for spawning of coarse fishes in European waters (Cowx & Welcomme 1998)

Species	Water velocity (cm s^{-1})
Roach	> 20
Dace	20 - 50
Chub	20 - 50
Common bream	< 20
Silver bream	< 20
Barbel	35 - 49
Tench	< 20
Gudgeon	10 - 80
Minnow	> 20
Common carp	< 5

Gravel spawners or lithophilic species such as minnow, barbel, chub and dace require suitable, clean gravel for spawning success (Cowx & Welcomme 1998). Gravel spawners are able to illustrate the degree that environmental degradation alters reproductive isolation. These species are early-warning indicators of anthropogenic disturbance and they rarely occur in highly turbid, warm, chemically polluted, or heavily silted rivers (Lyons *et al.* 1996). A lack of suitable substrate for spawning will adversely affect reproductive success, which will lead to the reduction of population density (Cowx 2001). For example, barbel prefer gravel of 20 - 50 mm diameter, minnow prefer 20 - 100 mm, whereas dace prefer 30 - 250 mm as spawning substrate (Cowx & Welcomme 1998). The percentage of lithophils is reduced with loss of interstitial pore space (Berkman & Rabeni 1987). Therefore, the metric "percentage of individuals as gravel spawners" was selected to measure the degree of degradation due to siltation over the gravel or displacement of gravel due to strong currents. Use of lithophils as metric is also suggested by Ohio EPA (1988) to evaluate the habitat(s) of gravel spawners when calculating IBI scores. Oberdorff & Hughes (1992) excluded the metric "number of hybrids" and included "percent gravel spawners". Karr (1981) suggested the use of reproductive guilds (Balon 1975) as a metric to develop IBI, although he did not use it in his initial work. The metric "percentage of individuals as gravel spawners" was retained in 10 versions of IBI around the world (Appendix 5.1). The majority of English freshwater species are lithophilic (Mann 1996, Cowx 2001). Hence, it was considered appropriate to use "percentage of individuals as gravel spawners" as a metric for English rivers (Table 5.2).

Trophic composition

Trophic composition metrics reflect the trophic dynamics of a fish assemblage based on the feeding patterns of adults (Goldstein & Simon 1999). They measure the divergence from expected production and consumption patterns resulting from alterations in river quality that, in turn, modify the food base of the fish assemblage. Trophic composition metrics are thus used to assess changes in ecological processes or functions, thereby broadening the IBI to include both structural and functional components (Miller *et al.* 1988).

All original metrics of Karr (1981) were retained including "percentage of individuals as omnivores", "percentage of individuals as invertivores", and "percentage of individuals as piscivores (top carnivores)" (Table 5.2). The term "generalist" and "specialist" are often used to designate the fish as "omnivores" and "invertivores", respectively (Table 5.3). The omnivore metric is designed to measure increasing levels of environmental degradation due to a disruption of the food base. Omnivores are defined as species that consistently feed on substantial proportions of plant (≥ 25%) and animal (at least 25%) materials but the category does not include filter feeding species or herbivores (Miller *et al.* 1988, Goldstein & Simon 1999). Omnivores are multitrophic, having the ability to change their feeding habits when the food chain is under pressure. The number of omnivores usually increases in disturbed environments, where specialised sources of food are rare or absent (Angermeier & Karr 1986). An increase in the percentage of omnivores is caused by decreases in biotic integrity (Leonard & Orth 1986). The metric "percentage of individuals as omnivores" was retained in the highest number of IBIs (26) developed around the world, due to the global dominance of omnivores (Appendix 5.1). In British rivers, a considerable number of fish are omnivorous (Cowx 2001) and thus the metric was considered appropriate to calculate IBI for English rivers (Table 5.2).

Invertivores are the dominant trophic guild in most streams and rivers of the UK (Cowx 2001). Thus, it is appropriate to use the metric "percentage of individuals as invertivores" to develop IBI for those rivers (Table 5.2). Therefore, this metric is chosen to be sensitive over the middle range of biotic integrity. In this study, instead of "insectivore", a general term "invertivore", is used which includes fish feeding on crustaceans, oligochaetes, snails and molluscs, as well as on insects. The invertivores are a measure of the secondary productivity of the system. The metric reflects increases in the proportion of invertivores with increasing biological integrity. A low abundance of invertivores will typically reflect a degradation of the invertebrate food base of a stream (Karr *et al.* 1986). Hughes & Oberdorff (1999) recommended that wherever the fauna is rich enough, invertivores or some substitute group of small organism or specialised feeder be evaluated as a metric. This metric was used in 21 versions of IBI developed for streams and rivers of North America, South America, Europe and Africa (Appendix 5.1).

A consistently high number of piscivores (top carnivores) usually indicates a healthy trophic composition of a river (Goldstein & Simon 1999). Top carnivores are species that as adults feed primarily on fish, other vertebrates (birds, amphibians and mammals), or large invertebrates such as crayfish. This metric will help discriminate between high and moderate quality systems. Pike and perch are the top carnivores in many English rivers and play an important role in maintaining the balance in fish communities (Table 5.3). Therefore, use of the metric "percentage of individuals as piscivores (top carnivores)" was considered appropriate for the study rivers. Hughes & Oberdorff (1998) suggested the use of the "top carnivore metric" for assessing fish assemblages outside the United States and Ganasan & Hughes (1998) applied this metric successfully in Indian rivers to develop an IBI. This metric describes the trends in decline of trophic composition with disturbance (Miller *et al.* 1988). This metric was also retained in 21 versions of IBI covering North America, South America, Europe, Africa, Asia and Australia (Appendix 5.1).

Fish abundance and biomass

Another new metric, "number of individuals of long-lived species (No. $100m^{-2}$)" was included in this study (Table 5.2). Chub and common bream were selected as long-lived species for English lowland rivers. Chub are used as an indicator of moderate to high water quality while common bream usually indicate low to moderate water quality (Cowx 2001). Both species are representative of long-lived individuals (life span 20 - 25 years, Table 3.9) and are likely be a substitute for Karr's "number of sucker species metric" as sucker species are also considered as long-lived species. These species offer a means to integrate disturbances over many years because of their long life span. Oberdorff & Hughes (1992) proposed other long-lived species as a replacement for suckers, and Steedman (1988) classified suckers and catfishes into the same metric. The metric adopted for this study, is thus a compromise between these two suggestions while in the case of common bream retaining the association with the river bed displayed by suckers. Chub and common bream are sensitive to physical, chemical and biological habitat degradation. The chub is an omnivore while the common bream is a benthivore, both feeding on invertebrates (Cowx 2001). Chub live in clear and well-oxygenated areas where current is moderate. They are sensitive to pollution and increased water temperatures. On the other hand, younger common bream live in vegetated waters, usually close to the bank. The common bream is sensitive to removal or reduction in vegetation as they use aquatic vegetation for shelter and breeding purposes (Cowx 2001). Chub and common bream are widespread in UK rivers and hence, it was considered appropriate to use this metric for these rivers. This metric was used in 3 versions of IBI developed for the streams and rivers of North America and Africa (Appendix 5.1).

Fish abundance is a common surrogate for system productivity and highly disturbed sites are expected to support fewer individuals than high-quality sites (Karr 1981). The number of fish captured at a site is indicative of the biotic integrity of that site (Hughes & Oberdorff 1999). "Total number of individuals" is used to evaluate population density and fish abundance. The "number of individuals in a sample" is an important parameter because disturbed areas often have reduced fish abundance with poor physical conditions (Simon & Emery 1995). However, there is a lack of historical data on what numbers to expect in an undisturbed river. "Number of individuals in a sample" is defined as the number per unit area sampled. A high number of species in a sample is often associated with warm, enriched agricultural streams while numbers are very low with toxic influences and degraded urban streams (Hughes & Oberdorff 1999).

"Number of individuals in a sample" varies in UK rivers due to several reasons. For example, this metric shows a large amount of variation as a result of point and non-point source effluents (Boet *et al.* 1999). "Number of individuals in a sample" is commonly used, as most fishery samples (no matter what the purpose of the study) provide an abundance statistic. Therefore, it was considered appropriate to use this metric for UK rivers (Table 5.2). This metric was retained in 22 versions of IBIs developed around the world, indicating its wide application (Appendix 5.1).

In the fish abundance and biomass category, another new metric, "total biomass (g m^{-2})" was added (Table 5.2). "Total biomass (g m^{-2})" gives a measure of standing crop at a site (Kestemont *et al.* 2000). The logic behind this is that degraded sites may have lower or higher total biomass than an undisturbed site, as biomass is influenced by the fish community structure and function. In a degraded site, total biomass may be higher as the tolerant fish dominate the site, while biomass may be less due to poor survival, growth and recruitment of the relatively intolerant species (Didier & Kestemont 1996). Fish biomass has greater impact on the trophic equilibrium and resources than the number of species, which is highly influenced by the abundance of small species (Oberdorff & Porcher 1994). A range of total fish biomass is found in English rivers due to a variety of reasons (Table 3.8) and hence, use of this metric was considered appropriate (Table 5.2). This metric was also adopted in 3 versions of IBIs developed for the streams and rivers of Europe (Appendix 5.1).

5.4 RATING AND SCORING OF IBI METRICS

Basic ecological information on fish and habitats is required for rating and scoring IBI metrics for a new ecoregion (Steedman 1988). Ecological information includes the knowledge of which fish species are native and which are introduced, their trophic class, reproductive guild and their tolerance to environmental degradation. The number of fish species expected at undisturbed sites varies with stream size and ecoregion (Fausch *et al.* 1984). Expectation criteria for fish species richness, density, diversity, abundance and biomass vary according to the watershed and ecoregion. An expectation criterion was developed for English rivers (Table 3.3) and according to the expectation criterion, IBI metrics were rated.

5.4.1 Existing methods for rating and scoring

Karr (1981) developed an IBI with a discontinuous rating system such as 5 (best), 3 (fair) and 1 (worst) to score each of 12 metrics according to whether its value approximates, deviates somewhat from, or deviates strongly from the value expected at the minimally disturbed sites. Values for all 12 metrics were then summed and resulting total IBI scores ranged from 12 to 60. Karr (1981) initially suggested 9 continuous integrity classes with the following boundaries: Excellent (57 - 60), Excellent to Good (53 - 56), Good (48 - 52), Good to Fair (39 - 44), Fair to Poor (36 - 38), Poor (28 - 35), Poor to Very Poor (24 - 27) and Very Poor (≤ 23). However, in his later work, Karr *et al.* (1986) assigned total score (12 x 5 = 60) to 6 integrity classes according to the following arbitrary scale: Excellent (58 - 60), Good (48 - 52), Fair (40 - 44), Poor (28 - 34), Very Poor (≤ 24) and No Fish (0). Most IBIs were developed by adopting these 6 integrity classes and class boundaries (Crumby *et al.* 1990; Oberdorff & Hughes 1992; Fore *et al.* 1994). Moyle *et al.* (1986) used 5 integrity classes with class boundary from ≤13 to 40 (Very Poor to Excellent) to evaluate biotic integrity of the rivers of California. Moyle *et al.* (1986) scored 8 IBI metrics with traditional rating system (5-3-1) and did not classify a site with "No Fish". Lyons *et al.* (1996) adopted a discontinuous, traditional type rating system, 20 (good), 10 (fair) and 0 (poor) with five integrity classes to evaluate coldwater streams in Wisconsin, USA. Ganasan & Hughes (1998) compared traditional scoring systems (5-3-1) with continuous systems (0 to 10) having three integrity classes such as acceptable, moderately impaired and impaired and found a similar pattern of overall IBI scores for Indian rivers. Kestemont *et al.* (2000) used a continuous rating scale ranging from 1 (low) to 5 (high) and found the scale suitable for wadable streams and rivers of the Meuse basin, Belgium.

Minns *et al.* (1994) and Hughes *et al.* (1998) introduced a continuous decimal scoring system, ranging from 0.0 - 10.0 to develop IBI for Great Lakes and Willamette Valley, Oregon, USA, respectively. Their IBI scores were the sums of the metric scores multiplied by 10 and divided by the number of metrics used, producing a maximum IBI score of 100 regardless of the number of metrics selected. Bramblett & Fausch (1991) introduced a proportionate scoring system, while developing an IBI containing 9 metrics for Western Great Plains rivers, USA. The authors multiplied the total score by 1.33 to obtain 60 instead of 45 (9 x 5), although they used traditional (5-3-1) rating scale to score the IBI metrics. Lyons (1992) used a discontinuous scoring criterion such as 10, 7, 5, 2 and 0 to develop an IBI for the warmwater streams of Wisconsin, USA. Although he retained the 6 integrity classes of Karr *et al.* (1986), he adopted different class boundaries with a continuous scale (Excellent: 65 - 100, Good: 50 - 64, Fair: 30 - 49, Poor: 20 - 29 and Very Poor: 1 - 19 and No Fish: 0). However, in their later experiments, Lyons *et al.* (1995) used a traditional type of scoring system (0 - 5 - 10, poor-fair-good) to calculate 10 metrics adopted for the streams and rivers of West-Central Mexico. The authors divided the total IBI score (10 x 10 = 100) into four integrity classes, Good (70 - 100), Fair (45 - 65), Poor (1 - 40) and Very Poor (No Score). Retaining the basic traditional scoring system (5 - 3 - 1), Steedman (1988) and Oberdorff & Porcher (1994) divided total IBI scores (10 x 5 = 50) of 10 metrics into five integrity classes, Excellent (48 - 50), Good (38 - 42), Fair (30 - 34), Poor (18 - 24) and Very Poor (≤ 14). However, these authors also used a "No Fish" class, when repeated sampling found no fish. Didier & Kestemont (1996) used a traditional system to score 13 metrics and a total score (13 x 5 = 65) divided into five integrity classes to evaluate the River Meuse, Belgium with following class boundaries: Excellent (63 - 65), Good (52 - 56), Fair (43 - 48), Poor (30 - 37) and very Poor (≤ 24).

The upper limit of the total score varies and depends on the total number of metrics used and rating scale. In summary, choice of rating scale, number of integrity classes and their boundaries depend on the choice of the experimenting scientist. Some authors also use arbitrary class ranges on the basis of professional experience and judgement. Therefore, an attempt was made to adopt a suitable rating scale, calculation and scoring procedures of metrics, identifying appropriate integrity classes and fixing their boundaries to define the IBI for English lowland rivers.

5.4.2 Scoring scale and integrity classes for English rivers

A continuous rating scale was adopted to score IBI metrics for English lowland rivers. Use of a continuous scale has several advantages. A continuous scale includes all values, yielded from a sum of metrics calculated, producing a continuous range of score values. Therefore, it is easy to transfer the value to an appropriate integrity class thus explaining simply the condition of a site on the basis of that value. On the other hand, a discontinuous scale produces discontinuous total scores, which may be difficult to transfer to an appropriate integrity class. Furthermore, the score values may fall between the upper limit of a class and lower limit of the next class and are unable to explain the condition of the site.

Ratings of 5, 4, 3, 2, 1 and 0 were chosen to assign each metric according to whether its value approximates to (best), deviates somewhat from (good), deviates more from (intermediate), deviates considerably from (bad), deviates strongly (worst) or "No Fish" from the value expected at the minimally disturbed sites (Table 5.5).

Table 5.5 Scoring criteria of IBI metrics used for the study rivers (according to Table 5.3).

Metric description (reference number)	Score 5	4	3	2	1
1. Total number of native species (18)	14-18 (80-100%)	11-13 (60-79%)	7 - 10 (40-59%)	4 - 6 (20-39%)	1 - 3 (1-19%)
2. Percentage of individuals as non-natives (Max. expected no.=10)	1-2 (1-16%)	3-4 (17-27%)	5-6 (28-38%)	7-9 (39-55%)	10 (56%)
3. Number of intolerant species (5)	5	4	3	2	1
4. Percentage of individuals as tolerant species (13)	1-16%	17-21%	22-38%	39-71%	72%
5. Number of water - column species (10)	10	8-9	6-7	4-5	1-3
6. Number of benthic species (8)	8	6 -7	5	3-4	1-2
7. Percentage of individuals as rheophilic species (6)	33%	28-32%	22-27%	11-21%	1-10%
8. Percentage of individuals preferring vegetated areas (12)	67%	56-66%	39-55%	28-38%	1-27%
9. Percentage of individuals as gravel spawner (9)	50%	39-49%	28-38%	22-27%	1-21%
10. Percentage of individuals as omnivores (9)	1-21%	22-27%	28-38%	39-49%	50%
11. Percentage of individuals as invertivores (5)	28%	22-27%	17-21%	11-16%	1-10%
12. Percentage of individuals as piscivores (4)	22%	17-21%	11-16%	1-10%	-
13. Number of individuals of long-lived species (No. 100 m^{-2}) (≥ 25)	≥25	20-24	15-19	10-14	1-9
14. Number of individuals in sample (No. 100 m^{-2}) (≥ 200 fish)	≥200	100-199	50-99	20-49	1-19
15. Total biomass (≥35 g m^{-2})	≥35	28-34	17-27	6-16	1-5

Metric ratings were then summed to yield a numerical score, ranging from 0 (15 x 0), indicating a "No Fish" site to 75 (15 x 5), indicating an excellent site. The score range (0 - 75) was used to assign sites to qualitative classes of biotic integrity.

A total of six integrity classes on a continuous scale were chosen to define biotic integrity of English lowland rivers with following class boundaries: Excellent (56 - 75), Good (42 - 55), Fair (28 - 41), Poor (16 - 27), Very Poor (1 - 15) and "No Fish" (0) (Table 5.6). When repeated sampling failed to produce any fish, sites were assigned to the "No Fish" category. The ranges of class boundaries were fixed arbitrarily on the basis of overall biodiversity and ecology of British freshwater fishes, and supported with reference to other IBIs for temperate regions. Moreover, class boundary ranges were adjusted with the increase in number of metrics compared to Karr (1981).

The selected number of integrity classes is considered by the author to be optimum for English lowland rivers. Selecting / grouping into too many classes, has a chance of overlapping the conditions, while too few classes are unable to identify and segregate the real condition of the site. Boundaries of the integrity classes were fixed to reflect the real condition of the site. Score ranges with respective integrity classes and the attributes of those classes are given in Table 5.6.

The River Ecosystem Classification (RE) and General Quality Assessment (GQA) (EA, LEAP 1998d) based on chemical and biological parameters of river water, respectively, also contained six quality classes / grades (Appendix 5.2). The chemical classification does include physical and biological stress on a fish community while the other (e.g. Habitat index) fails to include chemical aspects of stress on fish community. However, there is a similarity in the quality categories / grades regarding explanation of habitat quality as RE1 to Unclassified category of the Chemical Classification and A to F grades of the Biological GQA system were comparable to the "Excellent" to "No Fish" category of the IBI system.

5.4.3 Calculations and scoring of IBI metrics for English rivers

The rating system of 7 metrics (metric 1, 3, 5, 6, 13 14 and 15, (Table 5.5)) was based on a numerical scale and the remainder (metric 2, 4, 7, 8, 9, 10, 11 and 12, (Table 5.5)) were on a percentile scale. Three metrics, "percentage of individuals as non-natives", "percentage of individuals as tolerant species" and "percentage of individuals as omnivores", received the highest score 5, when these species were absent.

Table 5.6 Details of score ranges, integrity classes and the attributes of those classes on the basis of the 15 selected metrics

Total IBI score	Integrity class of site	Attributes
56 - 75	**Excellent**	Comparable to the best situations without human disturbance; all regionally expected species for the habitat and stream size, including the most intolerant forms, are present with a full array of age (size) classes; balanced trophic structure.
42 - 55	**Good**	Species richness somewhat below expectation, especially due to the loss of the most intolerant forms; some species are present with less than optimal abundance or size distributions; trophic structure shows some signs of stress.
28 - 41	**Fair**	Signs of additional deterioration including loss of intolerant forms, fewer species, highly skewed trophic structure (e.g., increasing frequency of omnivores and other tolerant species); older age classes of top predators (carnivore) may be rare.
16 - 27	**Poor**	Dominated by omnivores, tolerant forms, and habitat generalists; few top carnivores; growth rates and condition factors commonly depressed; hybrids and diseased fish often present.
1 - 15	**Very Poor**	Few fish present, mostly introduced or tolerant forms; hybrids common; disease, parasites, fin damage, and other anomalies regular.
0	**No Fish**	Repeated sampling finds no fish.

However, these scores were not used for the "No Fish" categories, as "No Fish" category indicates heavy degradation of a site. A summary of scoring criteria and calculations are given in Table 5.5 and details are provided below.

Total number of native fish species: A total of 18 common native species were included to establish the reference condition for English lowland rivers (Section 3.3.2 & Table 5.3). These are the most common species. This number (18) was taken as standard reference for an unimpacted site. Rating of this metric was done according to the number of species found at a site (Table 5.5). For example, eighteen species indicated 100% presence whilst presence of 80% of total reference species is 14 ($18/100 \times 80 = 14.4 \approx 14$ species).

Percentage of individuals as non-natives: At the time of writing a total of 41 species had been introduced into the UK (Table 3.2). However, most exotic species are confined to isolated areas and only 10 species are normally found in the open waters of the UK (Section 3.3.2 & Table 5.3). Therefore, the lowest score (1) was allocated for the presence of 10 alien species. The number of non-natives was calculated proportionately from the highest expected number (10) for this group, according to ascending order of score number i.e. 1, 2, 3, 4 & 5 (e.g. score 1 for 10 non-natives, score 2 for 8 species) (Table 5.7).

Table 5.7 Scoring of "percentage of individuals as non-natives"

Number of non-native species	% of reference number	Score
1 - 2	1 - 16	5
3 - 4	17 - 27	4
5 - 6	28 - 38	3
7 - 9	39 - 55	2
10	56	1

Number of intolerant species: The number of intolerant species was 5, amounting to approximately 28% (=5/18 x 100) of total species identified in the study area (Table 5.5). This value (5) was taken as reference for English lowland rivers (Table 5.8).

Table 5.8 Scoring of "number of intolerant species"

Number of intolerant species	% of reference number	Score
5	28	5
4	22 - 27	4
3	17 - 21	3
2	11 - 16	2
1	1 - 10	1

Percent of individuals as tolerant species: The number of tolerant species to ecological degradation is 13, which is taken as the standard reference for the network of English lowland rivers (Table 5.3). Scoring was in reverse order, i.e. lowest score for presence of highest number of tolerant species. The number of tolerant forms such as 13, 7, 4, 3 and 2 were calculated proportionately from the reference number (i.e. score 1 for the presence of 13/1 = 13 tolerant species, score 2 for 13/2 = 6.5 ≈ 7, score 3 for 13/3 = 4.3 ≈ 4, score 4 for 13/4 = 3.25 ≈ 3 and score 5 for 13/5 = 2.6 ≈ 2 tolerant species) (Table 5.9).

Table 5.9 Scoring of "percentage of individuals as tolerant species"

Number of tolerant species	% of reference number	Score
1 - 2	1 - 16	5
3	17 - 21	4
4 - 6	22 - 38	3
7 - 12	39 - 71	2
13	72	1

Number of water-column species: The diversity of water-column species was 10 and this was taken as the standard reference (Table 5.3). Descending order of score, i.e. 5, 4, 3, 2, & 1 was used to rate the metric (highest score for the presence of highest water-column species) (Table 5.10).

Table 5.10 Scoring of "number of water-column species"

Number of water-column species	% of reference number	Score
10	56	5
8 - 9	44 - 55	4
6 - 7	33 - 43	3
4 - 5	22 - 32	2
1 - 3	1 - 21	1

Number of benthic species: The reference condition for benthic species was based on eight for English lowland rivers (Table 5.3). The same procedures were followed as for number of water-column species. Scores of 5, 4, 3, 2 and 1 were given for the presence of 8, 6 - 7, 5, 3 - 4 and 1 - 2 of benthic species, respectively (score 5 for the presence of 8 benthic species, so score 4 for $8/5 \times 4 = 6.4 \approx 6$ species) (Table 5.11).

Table 5.11 Scoring of "number of benthic species"

Number of benthic species	% of reference number	Score
8	44	5
6 - 7	33 - 43	4
5	28 - 32	3
3 - 4	17 - 27	2
1 - 2	1 - 16	1

Percent of individuals as rheophilic species: A total of six species were found to be rheophilic and this was taken as reference for English lowland rivers (Table 5.3). According to the scores, 6 species followed by, 5, 4, 3 - 2, and 1 were calculated proportionately from the reference number (score 5 for the presence of 6 rheophils, so score 4 for $6/5 \times 4 = 4.8 \approx 5$ species). Percentages were calculated from the reference number for total species (18) (Table 5.12).

Table 5.12 Scoring of "percentage of individuals as rheophilic species"

Number of rheophilic species	% of reference number	Score
6	33	5
5	28 - 32	4
4 - 3	22 - 27	3
2	11 - 21	2
1	1 - 10	1

Percentage of individuals preferring vegetated areas: Twelve species prefer to live in vegetated areas (Table 5.3). This was taken as standard for English lowland rivers. Twelve, 10, 7, 5 and 2 vegetated area preferring species were calculated proportionately from the reference number (score 5 for the presence of 12 species, so, score 4 for $12/5 \times 4 = 9.6 \approx 10$ species). Percentages were then calculated from the reference number for total species (Table 5.13).

Table 5.13 Scoring of "percentage of individuals preferring vegetated areas"

Number of limnophilic species	% of reference number	Score
12	67	5
10 - 11	56 - 66	4
7 - 9	39 - 55	3
5 - 6	28 - 38	2
1 - 4	1 - 27	1

Percentage of individuals as gravel spawners: Nine species found were gravel spawners (Table 5.3). This was taken as reference for English lowland rivers (Table 5.14).

Table 5.14 Scoring of "percentage of individuals as gravel spawners"

Number of gravel spawners	% of reference number	Score
9	50	5
7 - 8	39 - 49	4
5 - 6	28 - 38	3
4	22 - 27	2
1 - 3	1 - 21	1

Percentage of individuals as omnivores: A total of nine species were found as omnivores (Section 3.3.5 & Table 5.3). The metric was given the lowest score for the presence of highest number of omnivores (Table 5.15).

Table 5.15 Scoring of "percentage of individuals as omnivores"

Number of omnivores	% of reference number	Score
1 - 3	1 - 21	5
4	22 - 27	4
5 - 6	28 - 38	3
7 - 8	39 - 49	2
9	50	1

Percentage of individuals as invertivores: Total number of invertivores is 5 and was considered as the standard reference for that group (Section 3.3.5, Table 5.3 & Table 5.16).

Table 5.16 Scoring of "percentage of individuals as invertivores"

Number of invertivores	% of reference number	Score
5	28	5
4	22 - 27	4
3	17 - 21	3
2	11 - 16	2
1	1 - 10	1

Percentage of individuals as piscivores: Four was designated as the reference number of species of piscivores (Section 3.3.5, Tables 5.3 & 5.17).

Table 5.17 Scoring of "percentage of individuals as piscivores"

Number of invertivores	% of reference number	Score
4	22	5
3	17 - 21	4
2	11 - 16	3
1	1 - 10	2

Number of individuals of long-lived species (No. 100 m^{-2}): Chub and common bream were identified as long-lived species in English rivers (Section 3.3.7). Frequency distribution for chub and common bream was analysed for 188 sites from 16 English rivers, to establish a standard reference. A total number of 0 - 25, 0 - 55 and 5 - 55 individuals (No. 100 m^{-2}) of long-lived species were present at 78, 92 and 59% of total sites, respectively. Therefore, presence of 0 - 25 individuals of long-lived species (Table 5.3) covering 78% of total sites were taken as standard for English lowland rivers. The metric was rated as 5, 4, 3, 2 and 1 for the presence of ≥25, 24 - 20, 19 - 15, 14 - 10, 9 - 1 (No. 100 m^{-2}) of long-lived species a site, respectively (score 5 for the presence of 25 individuals of long-lived species, so, score 4 for 25/5 x 4 = 20 species) (Table 5.18). A score of zero was allocated in the case of absence of long-lived species.

Table 5.18 Scoring of "number of individuals of long-lived species (No. $100m^{-2}$)"

Number of individuals of long-lived species (No. 100 m^{-2})	Score
≥25	5
20 - 24	4
15 - 19	3
10 - 14	2
1 - 9	1

Number of individuals in a sample (No. 100 m^{-2}): Number of individuals in a sample depends on several factors such as sampling gear, efficiency and type of gear, time and season of sampling, type and nature of the habitat, water temperature and the actual number of fish present at a site. Frequency distribution of total catch for 188 sites of 16 rivers was analysed to establish reference condition for total number in a sample. All sites were sampled by electric fishing. The total catch of fish ranged between 0 and 1256 fish $100m^{-2}$. However, it was found that a range of 0 - 200 fish 100 m^{-2} were caught from 151 sites, covering 84% of total sites (Section 3.3.4 & Table 5.3). Therefore, this catch range was taken as the standard reference for a site of an English lowland river. Arbitrary ranges for total number in a sample, ≥200, 100 - 199, 50 - 99, 20 - 49 and 1 - 19, fish 100 m^{-2} were fixed to score the metric as 5, 4, 3, 2 and 1, respectively (Table 5.19). A score of zero was allocated for the catch of no fish.

Table 5.19 Scoring of "number of individuals in a sample (No. 100 m^{-2})"

Number of fish in sample (No. 100 m^{-2})	Score
≥200	5
100 - 199	4
50 - 99	3
20 - 49	2
1 - 19	1

Total biomass (g m^{-2}): Total biomass of a site was calculated as g m^{-2}. Historical data for fish biomass of UK rivers were evaluated (Table 3.8). Frequency distribution of total fish biomass from 188 sites of 16 different English rivers was analysed to establish a reference condition. Approximately 80% of total fish biomass values were distributed within the class 5 to 35 g m^{-2} (Section 3.3.4 & Table 5.3). Therefore, this range of fish biomass was taken as reference for English lowland rivers. Arbitrary ranges, ≥35, 34 - 28, 27 - 17, 16 - 6 and 5 - 1 g m^{-2} of total fish biomass were fixed on the basis of judgement by the author to score the metric as 5, 4, 3, 2 and 1, respectively (Table 5.20). Scores of all metrics were tabulated in a sheet to calculate the IBI for a site (Table 5.21).

Table 5.20 Scoring of "total biomass (g m^{-2})"

Total biomass (g m^{-2})	Score
≥35	5
34 - 28	4
27 - 17	3
16 - 6	2
5 - 1	1

Table 5.21 Example of IBI score calculation sheet

River: Cherwell **Site No.: 4, Spiceball Park**

Metric	Number	% reference number	Score
1. Total number of native species	7	39	2
2. Percentage of individuals as non-natives	0	0	5
3. Number of intolerant species	2	11	2
4. Percentage of individuals as tolerant species	5	28	2
5. Number of water-column species	5	28	3
6. Number of benthic species	2	11	1
7. Percentage of individuals as rheophilic species	3	17	3
8. Percentage of individuals preferring vegetated areas	4	22	2
9. Percentage of individuals as gravel spawners	5	28	3
10. Percentage of individuals as omnivores	3	17	4
11. Percentage of individuals as invertivores	2	11	2
12. Percentage of individuals as piscivores	2	11	2
13. Number of individuals of long-lived species (Chub & common bream) (No. 100 m^{-2})	13	-	3
14. Number of individuals in a sample (No. 100 m^{-2})	52	-	3
15. Total biomass (g m^{-2})	23.93	-	4
	N = 15	Total IBI Score	**41**
	Integrity Class: Fair		

5.5 CASE STUDIES : GENERAL DESCRIPTION OF IBI OUTPUT

5.5.1 The Thames catchment

Eighty-six sites from five rivers in the Thames catchment were used to develop IBI models. The IBI value for the individual sites ranged from 18 to 55 with an average of 40.00±8.87 (n = 86), indicating a "Fair" class catchment (Appendix 5.3 & Table 5.6). A "Fair" category catchment is characterised by few intolerant species having few total numbers of species. Trophic structure is highly skewed as the number of omnivores and other tolerant forms are high. Abundance of older classes of top predators is low in "Fair" class rivers (Table 5.6).

The range of values indicates that the catchment contained extremely degraded sites as well as sites with conditions close to the natural state. However, no "Excellent" sites were found in the Thames catchment. The average value for the individual rivers varied between 35 and 45 with a mean of 39.40±3.88 (n = 5) (Table 5.22). The River Thame had the highest average IBI followed by the rivers Evenlode and Windrush.

Table 5.22 IBI scores on the basis of 15 metrics for 5 rivers in the Thames catchment

	River	Total number of sites	Average IBI score	SD±	Range	Integrity Class
1	Cherwell	13	35	8.53	22 - 47	Fair
2	Evenlode	20	41	7.39	18 - 55	Fair
3	Stort	16	35	7.80	23 - 50	Fair
4	Thame	18	45	7.85	26 - 55	Good
5	Windrush	19	41	8.82	23 - 52	Fair
Total / Average		**86**	**39**	**3.88**	**35 - 45**	**Fair**

River Cherwell

The IBI was calculated for 13 sites on the River Cherwell (Fig. 5.1a). The score ranged from 22 to 47 with a mean of 35.00±8.53 (n = 13), indicating a "Fair" category river (Table 5.22 & Table 5.6). Except for the Sor Brook Confluence (site 9), all sites were free from alien species. West Farndon Mill (site 1), was the only site not supporting tolerant species. The number of intolerant species ranged between 0 (site 5, Tramroad Industrial Estate) and 3. These features of the river indicated a disrupted fish community, having imbalanced structural and disfunctional fish assemblages.

The integrity class of the sites ranged between "Poor" and "Good" (Fig. 5.2b). The mean IBI for the River Cherwell was close to the lower limit of the "Good" category boundary. The three "Poor" sites (sites 1, 5 & 6) were in the upper reaches of the river, containing 1 - 2 fish species only. Spiceball Park (site 4) had an IBI index close to the "Good" category boundary. The majority of "Fair" sites (sites 7, 8, 9, 10 & 13) were in the lower reaches of the river. However, one "Good" site (site 3) was found in the upper reaches and two (sites 11 & 12) in the lower reaches (Appendix 5.3).

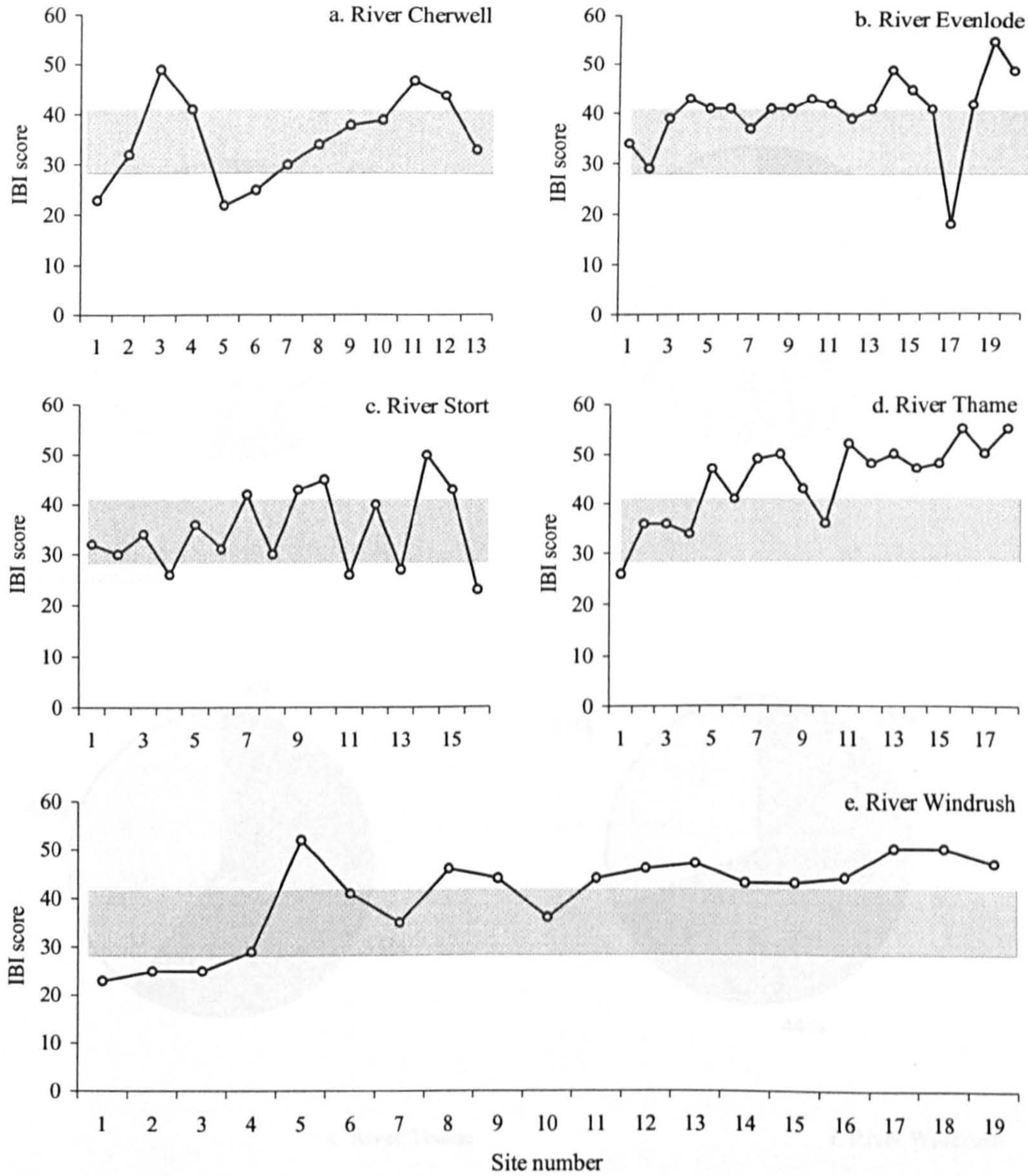


Fig. 5.1 Trends of IBI score for the Thames catchment (shaded area represents "Fair" range)

River Evenlode

Except for the site 17, an increasing trend in the IBI score was found within study sites on the River Evenlode (Fig. 5.1b). The IBI score ranged between 18 and 55 with a mean of 41.00±7.39 (n = 20), indicating a "Fair" class river (Table 5.22 & Table 5.6). The mean IBI score for the river was close to the "Good" category boundary. No non-natives were found in the river, indicating low pressure from human intervention. Goose Eye Farm (site 17) was the only site not to support intolerant forms. With one exception (site 17), fish diversity (Appendix 4.3) and density (Fig. 4.7a) were relatively stable throughout the river.

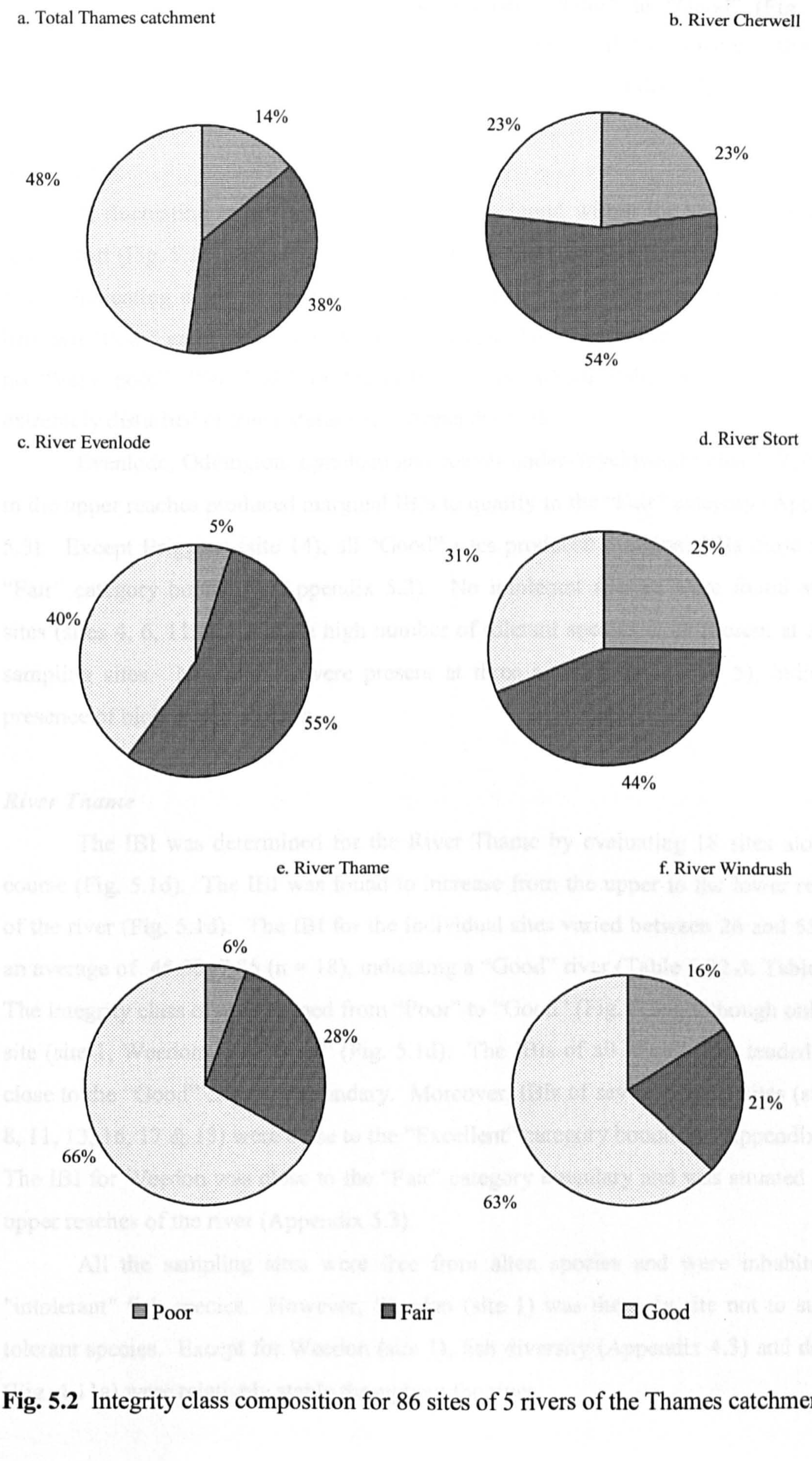


Fig. 5.2 Integrity class composition for 86 sites of 5 rivers of the Thames catchment

The integrity class of the sites varied from "Poor" to "Good" (Fig. 5.2c), although only one was "Poor" (site 17) (Fig. 5.1b). Most "Fair" sites were in the upper reaches and most "Good" sites were in the lower reaches (Appendix 5.3).

River Stort

A fluctuating trend in the IBI scores was found within the study sites on the River Stort (Fig. 5.1c). The IBI varied between 23 and 50 with a mean of 35.00±7.80 (n = 16), indicating a "Fair" river (Table 5.22 & Table 5.6). The integrity class ranged between "Poor" and "Good" within 16 sites on the River Stort (Fig. 5.2d). There were no "Very poor", "No Fish" or "Excellent" sites, which indicated the absence of extremely disturbed or true natural sites (Appendix 5.3).

Evenlode, Oddington, Lyneham and Ascott-under-Wychwood (sites 1, 2, 6 & 8) in the upper reaches produced marginal IBIs to qualify in the "Fair" category (Appendix 5.3). Except Briggens (site 14), all "Good" sites produced marginal IBIs close to the "Fair" category boundary (Appendix 5.3). No intolerant species were found at four sites (sites 4, 6, 11 & 16) but a high number of tolerant species were present at all the sampling sites. Non-natives were present at three sites (sites 1, 3 & 5), indicating presence of biological pollution.

River Thame

The IBI was determined for the River Thame by evaluating 18 sites along its course (Fig. 5.1d). The IBI was found to increase from the upper to the lower reaches of the river (Fig. 5.1d). The IBI for the individual sites varied between 26 and 55 with an average of 45.00±7.85 (n = 18), indicating a "Good" river (Table 5.22 & Table 5.6). The integrity class of sites ranged from "Poor" to "Good" (Fig. 5.2e), although only one site (site 1, Weedon) was "Poor" (Fig. 5.1d). The IBIs of all "Fair" sites tended to be close to the "Good" category boundary. Moreover, IBIs of seven "Good" sites (sites 7, 8, 11, 13, 16, 17 & 18) were close to the "Excellent" category boundary (Appendix 5.3). The IBI for Weedon was close to the "Fair" category boundary and was situated in the upper reaches of the river (Appendix 5.3).

All the sampling sites were free from alien species and were inhabited by "intolerant" fish species. However, Weedon (site 1) was the only site not to support tolerant species. Except for Weedon (site 1), fish diversity (Appendix 4.3) and density (Fig. 4.11a) were relatively stable throughout the river.

River Windrush

The IBI was found to increase from the upper to the lower reaches of the River Windrush (Fig. 5.1e). The IBI score varied between 23 and 52 with a mean of 41.00±8.82 (n = 19), indicating a "Fair" river (Table 5.22 & Table 5.6). For the 19 sites on the River Windrush, integrity class varied between "Poor" and "Good" (Fig. 5.2f). The average IBI was very close to the "Good" category boundary. The IBIs of two of the four "Fair" sites (sites 6 & 10) were also close to the "Good" category boundary. On the other hand, the IBIs of 3 of the 12 "Good" sites (sites 5, 17 & 18) were close to the "Excellent" category boundary. Three "Poor" sites (sites 1, 2 & 3) had IBIs close to the "Fair" category boundary (Appendix 5.3).

The three "Poor" sites (sites 1, 2 & 3) were in the upper reaches (Fig. 5.1e) and except for D/S of Dikler Confluence (site 5), all the "Good" sites were in the downstream reaches (Appendix 5.3). No exotic species were recorded from the river and all sites were inhabited by "intolerant" fish species. Four upstream sites (sites 1, 2, 3, & 4) did not support tolerant species. Fish species diversity (Appendix 4.3) and density (Fig. 4.13a) were relatively stable throughout the river except for four sites (sites 1, 2, 3 & 4).

5.5.2 The Trent catchment

The IBI was determined for the Trent catchment by evaluating 163 sites from 15 rivers (Table 2.1). The IBI for the individual sites ranged from 0 to 54 with a mean of 32.96±14.44 (n = 163), indicating a "Fair" catchment (Appendix 5.3, & Table 5.6). The range of values indicated that the Trent catchment contained sites showing extreme degradation as well as sites with fish populations very close to the natural condition. The average IBI for the individual rivers varied from 23 to 44 with a mean of 33.87±6.96 (n = 15), also indicating a "Fair" catchment (Table 5.23). The integrity class ranged from "No Fish" to "Good", but with no "Excellent" or "Very Poor" sites in the catchment.

River Anker

The IBI was calculated for 10 sites on the River Anker (Fig. 5.3a). The IBI ranged between 31 and 50 with a mean of 42.80±5.81 (n = 10), indicating a "Good" river overall (Table 5.23 & Table 5.6). Integrity class ranged between "Fair" and "Good" (Fig. 5.4b) with no "Excellent", "Poor", "Very Poor" or "No Fish" sites. Although, the mean IBI marginally qualified for "Good" category, two of the six

"Good" sites (sites 2 & 10) had IBIs close to the "Excellent" category boundary. Moreover, two of the four "Fair" sites (sites 1 & 7) had IBIs very close to the "Good" category boundary (Appendix 5.3).

Polesworth-2 (site 8) was the only site supporting exotic species. Although, Polesworth-2 contained alien species and also had a single intolerant species however, it had an IBI of 43, indicating a "Good" site. All sites contained intolerant and tolerant fish species.

Table 5.23 IBI scores on the basis of 15 metrics for 15 rivers in the Trent catchment

	River	Total number of sites	Average IBI score	SD±	Range	Integrity class
1	Anker	10	43	5.81	31 - 50	Good
2	Blithe	11	39	9.04	23 - 48	Fair
3	Blythe	9	37	14.54	0 - 50	Fair
4	Churnet	16	27	15.00	0 - 48	Poor
5	Cole	14	23	17.98	0 - 45	Poor
6	Derwent	15	37	8.50	23 - 52	Fair
7	Idle	5	25	14.18	0 - 44	Poor
8	Mease	7	43	3.21	39 - 48	Good
9	Penk	11	29	19.09	0 - 48	Fair
10	Sence	6	36	17.51	0 - 54	Fair
11	Soar	15	37	12.30	0 - 49	Fair
12	Sow	9	38	6.88	26 - 50	Fair
13	Tame	6	44	4.49	37 - 49	Good
14	Tean	9	24	14.00	0 - 45	Poor
15	Trent	20	29	8.94	0 - 43	Fair
Total / Average		**163**	**34**	**6.96**	**23 - 44**	**Fair**

River Blithe

Eleven sites were used to calculate IBI for the River Blithe (Fig. 5.3b). The IBI ranged between 23 and 48 with a mean of 39.27±9.04 ($n = 11$), indicating a "Fair" river (Table 5.23 & Table 5.6). The River Blithe contained "Poor" to "Good" sites (Appendix 5.3). No "Excellent", "Very Poor" or "No Fish" sites were found in the river, indicating absence of absolute natural or extremely degraded sites (Fig. 5.4c). The mean IBI was close to the "Good" category boundary (Appendix 5.3). The three "Poor" sites (sites 1, 2 & 3) were in the upper reaches and all the "Good" sites were in the lower reaches.

No alien species were found in the river and all sites contained intolerant fish species. Conversely, four sites (sites 1, 2, 3 & 4) did not support tolerant species. With few exceptions, fish species diversity (Appendix 4.5) was relatively stable throughout the river.

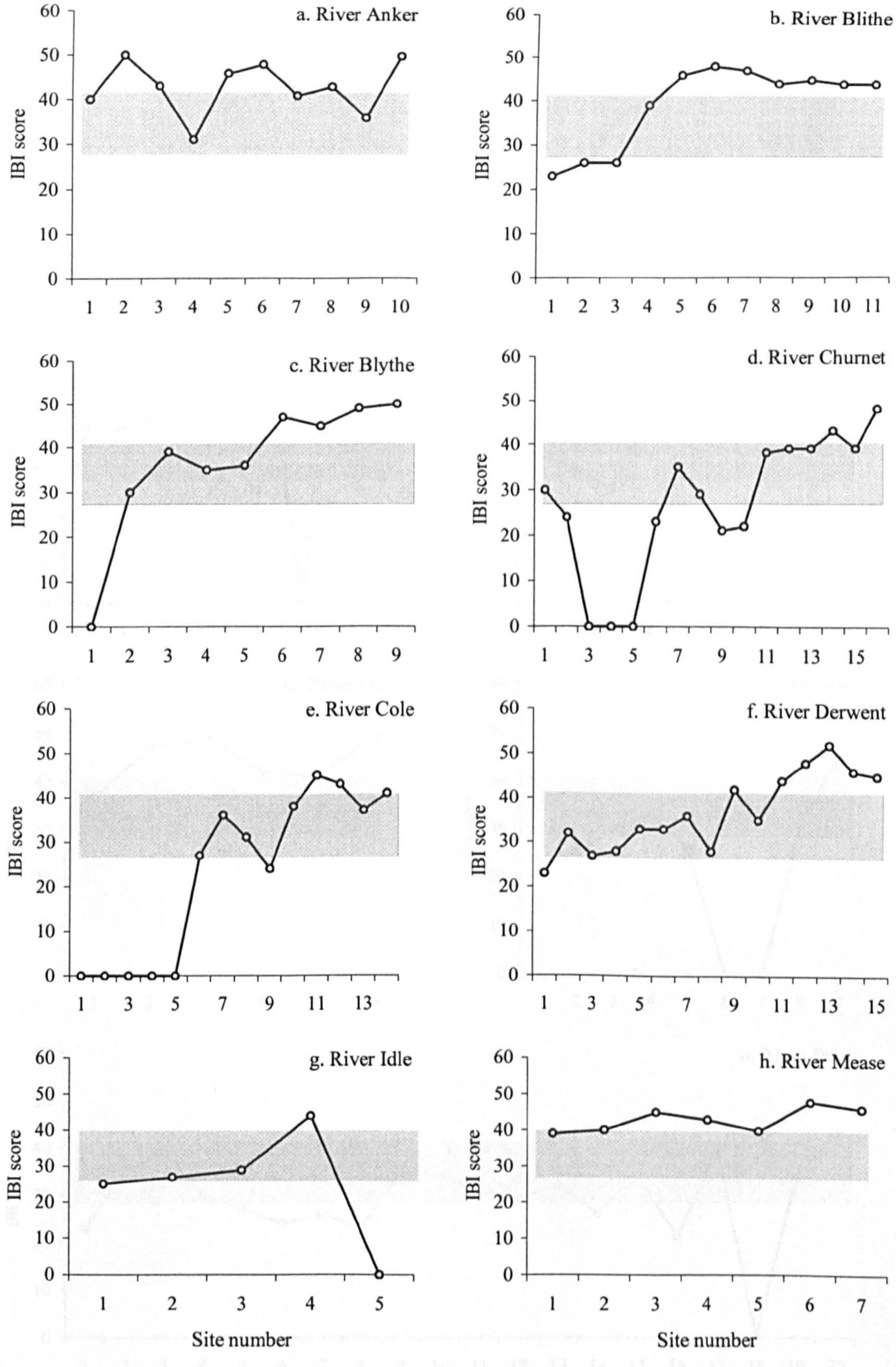


Fig. 5.3 Trends of IBI score for the Trent catchment (shaded area represents "Fair" range)

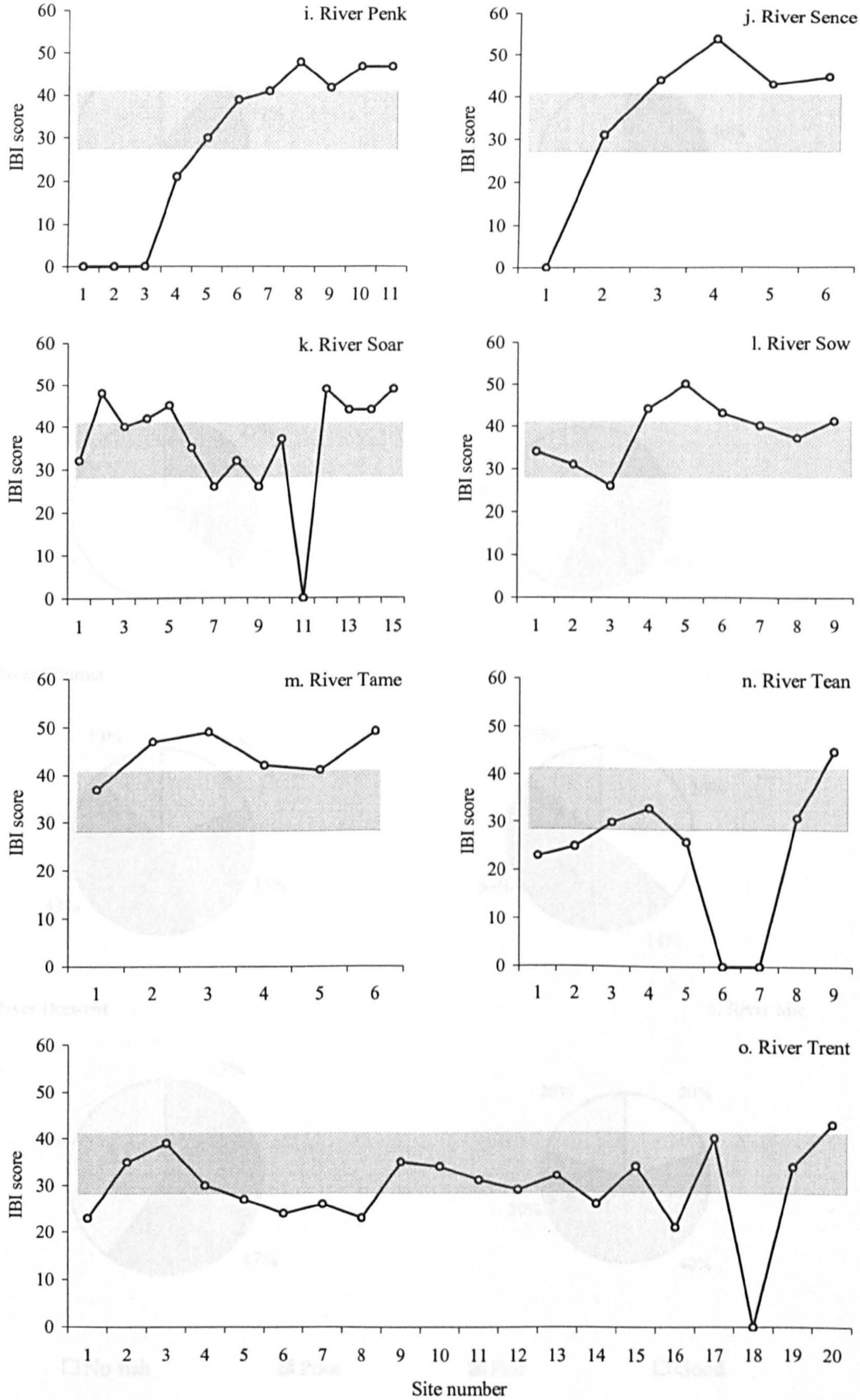


Fig. 5.3 (Continued) Trends of IBI score for the Trent catchment (shaded area represents "Fair" range)

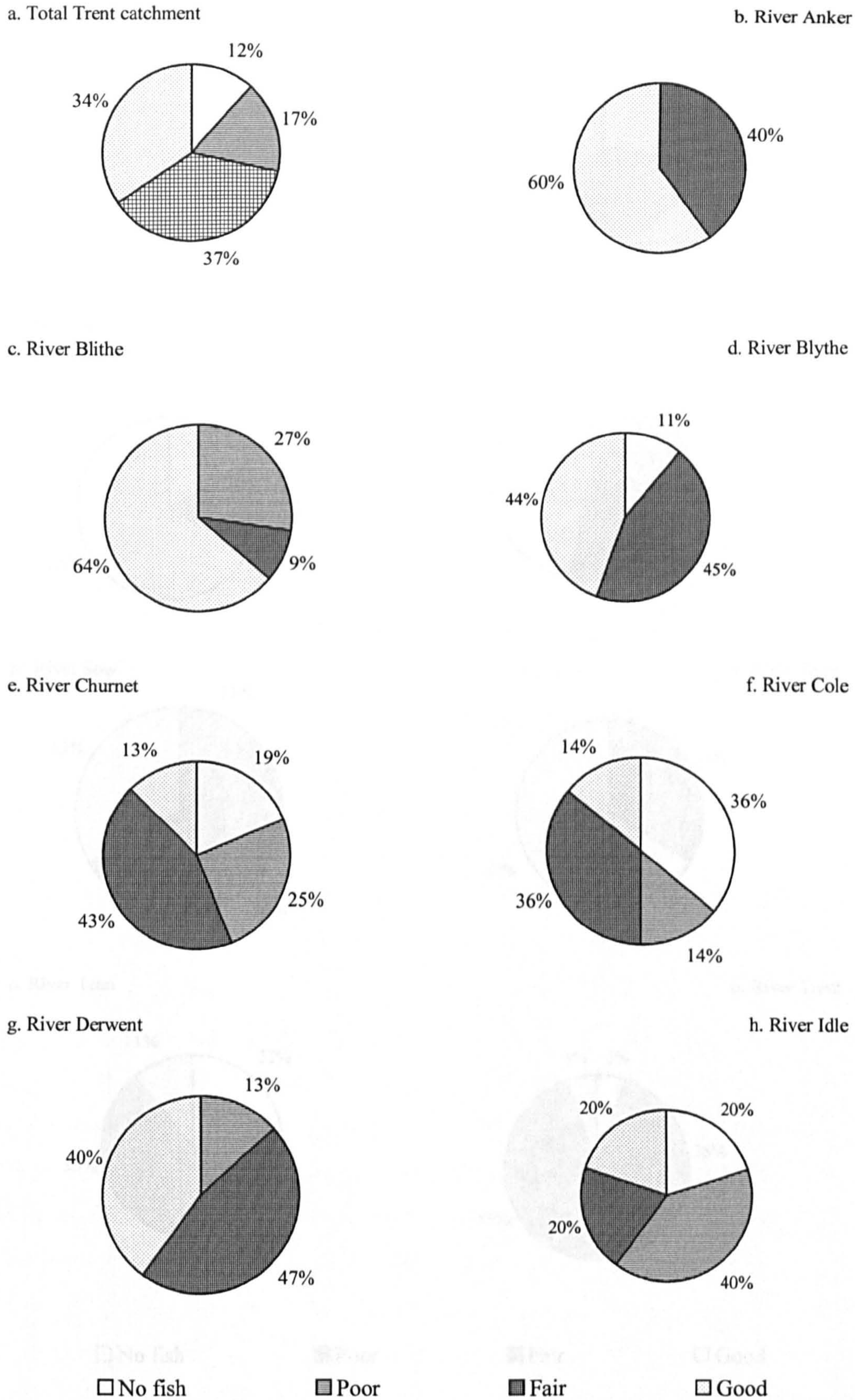


Fig. 5.4 Integrity class composition for 163 sites of 15 rivers of the Trent catchment

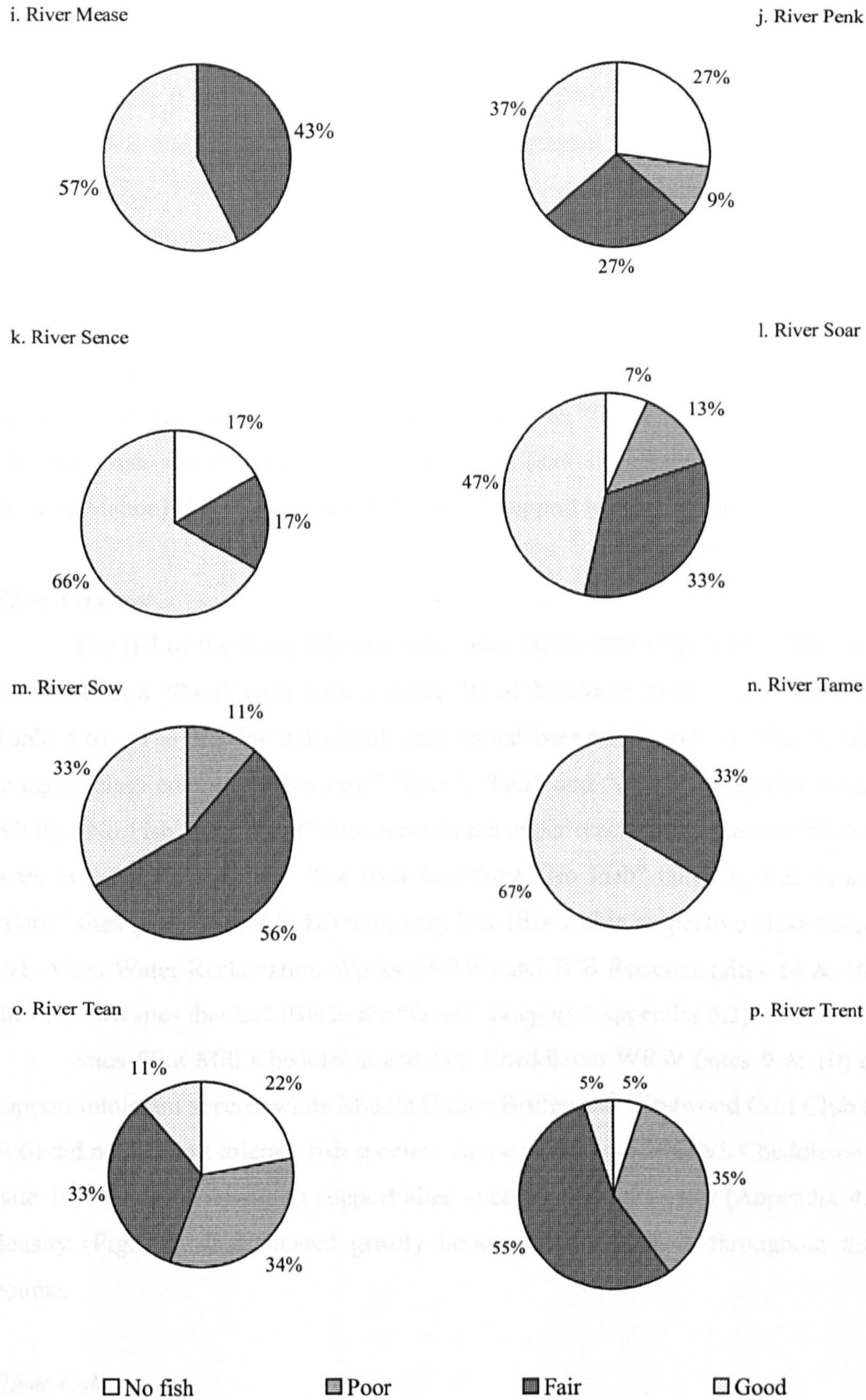


Fig. 5.4 (Continued) Integrity class composition for 163 sites of 15 rivers of the Trent catchment

River Blythe

The IBI was calculated for 9 sites on the River Blythe (Fig. 5.3c). The IBI score ranged between 0 and 50, with a mean of 36.78±14.54 (n = 9), indicating a "Fair" river (Table 5.23 & Table 5.6). The integrity class contained "No Fish", "Fair" and "Good" categories (Fig. 5.4d). The "No Fish" site was in the upper reaches and all the "Good" sites were in the lower reaches (Appendix 5.3). The river had low mean IBI due to presence of one "No Fish" site (site 1, Cheswick Green). However, Blythe Mill End (site 9) produced an IBI close to the "Excellent" category boundary (Appendix 5.3).

Exotic species were recorded from U/S of Eastcote Brook and Blythe Mill End (sites 6 & 9), although these sites were classified as "Good". All the sites supported intolerant fish species except Cheswick Green (site 1). Sites Cheswick Green and Widney Manor Rd Bridge (sites 1 & 2) did not support tolerant species.

River Churnet

The IBI of the River Churnet was based on 16 sites (Fig. 5.3d). The river was identified as a "Poor" river with a mean IBI of 27.00±15.00 (n = 16) (Table 5.23 & Table 5.6). The IBI for individual sites varied between 0 and 48 (Fig. 5.3d). The integrity class contained "No Fish" "Poor", "Fair" and "Good" categories (Fig. 5.4e). All the "No Fish" and "Poor" sites were in the upper reaches and the two "Good" sites were in the lower reaches. The river had three "No Fish" (sites 3, 4 & 5) and four "Poor" sites (sites 2, 6, 9 & 10) with very low IBIs within respective class boundaries. U/S Alton Water Reclamation Works (WRW) and JCB Rocester (sites 14 & 16) were the only two sites that had IBIs in the "Good" category (Appendix 5.3).

Sites Flint Mill Cheddleton and D/S Cheddleton WRW (sites 9 & 10) did not support intolerant species while Middle Hulme Bridge and Westwood Golf Club (sites 1 & 6) did not support tolerant fish species. In the middle reaches D/S Cheddleton WRW (site 10) was the only site to support alien species. Fish diversity (Appendix 4.5) and density (Fig. 4.22a) fluctuated greatly between different sites throughout the river course.

River Cole

The IBI was determined for the River Cole by evaluating 14 sites along its course (Fig. 5.3e). The IBI was found to increase from site 5 to the lower reaches of the river (Fig. 5.3e). The IBI for individual sites ranged between 0 and 45, with a mean of 23.00±17.98 (n = 14), indicating a "Poor" river (Table 5.23 & Table 5.6). Sites with four integrity classes, i.e. "No Fish", "Poor", "Fair" and "Good", were found in the river (Fig. 5.4f). Five (sites 1 to 5) of the 14 sites in the upper reaches of the River Cole were determined in the "No Fish" category (Appendix 5.3). One (site 6) of the two "Poor" sites had IBI close to the "Fair" category boundary (Appendix 5.3). Both "Good" sites had IBIs marginal for that category while two (sites 10 & 14) of the five "Fair" sites had IBIs close to the "Good" category boundary (Appendix 5.3).

Sites D/S Cook's Lane Bridge and Coleshill-1 (sites 9 & 13) did not support intolerant and tolerant species, respectively. Exotic species were recorded from Kingshurst-1 and Kingshurst-2 (sites 7 & 8), indicating presence of biological pollutants.

River Derwent

Fifteen sites were used to determine the IBI for the River Derwent (Fig. 5.3f). The IBI varied from 23 to 52 with a mean of 36.80±8.50 (n = 15), indicating a "Fair" class river (Table 5.23 & Table 5.6). The integrity class varied between "Poor" and "Good" (Fig. 5.4g), having no "Excellent", "Very Poor" or "No Fish" sites within its length. All "Poor" sites and all "Fair" sites except for site 10, were in the upper reaches and all "Good" sites were in the lower reaches (Appendix 4.5). The river had six "Good" sites (sites 9, 11, 12, 13, 14 & 15) and one of those (site 13) had an IBI close to the "Excellent" category boundary (Appendix 5.3). Most "Fair" sites had marginal IBIs within the class boundary, which decreased the mean IBI for the river.

All sites support intolerant fish species but only five sites (sites 9, 11, 12, 13 & 14) had tolerant species. The River Derwent was free from alien species and had high fish species diversity (Appendix 4.5).

River Idle

Five sites were used to develop an IBI model for the River Idle (Fig. 5.3g). The IBI for individual sites varied between 0 and 44, with a mean of 25.00±14.18 (n = 5), indicating a "Poor" river (Table 5.23 & Table 5.6). Sites with four integrity classes, i.e. "No Fish", "Poor", "Fair" and "Good", were found in the river (Fig. 5.4h). The range of values indicated that the river contained both extremely degraded sites and sites with conditions close to the natural state. Both "Fair" (Mattersey Priory, site 3) and "Good" (Bawtry, site 4) sites had marginal IBIs within respective class boundaries (Appendix 5.3 & Table 5.6).

Site Eaton (site 1) did not support intolerant fish species and four sites (sites 1 - 4) contained tolerant species. Sites at Mattersey Priory and Bawtry (sites 3 & 4) had low fish density (Fig. 4.28a) and species diversity (Appendix 4.5), respectively. No exotic species were recorded in the river.

River Mease

The IBI of the River Mease was based on seven sites (Fig. 5.3h). The IBI for individual sites ranged between 39 and 48, with a mean of 43.00±3.21 (n = 7), indicating a "Good" river (Table 5.23 & Table 5.6). Sites fell into only two integrity classes, "Fair" and "Good" on the River Mease (Fig. 5.4i). Three "Fair" sites (sites 1, 2 & 5) had IBIs close to the "Good" category boundary. Except for Croxall Mill (site 6), all "Good" sites had marginal IBIs to qualify for the "Good" category (Appendix 5.3).

All the sites were free from alien species. Two intolerant fish species were present at each site, while the number of tolerant species ranged between 3 and 6 (Appendix 4.5). Except for Stretton en le Field (site 1), fish diversity (Appendix 4.5) and density (Fig. 4.30a) were relatively stable throughout the river.

River Penk

The IBI was calculated for the River Penk by evaluating 11 sites in its course (Fig. 5.3i). The IBI for the individual sites varied between 0 and 48 with a mean of 28.64±19.09 (n = 11), indicating a "Fair" river (Table 5.23 & Table 5.6). The integrity class consisted of "No Fish", "Poor", "Fair" and "Good" categories (Fig. 5.4j). D/S Bill Brook WRW (site 4) was the only "Poor" site, having marginal IBI for the "Poor" category. Sites Brewood Park Farm and Somerford (sites 6 & 7) had IBIs close to the "Good" category boundary (Appendix 5.3).

The sites at Black Brook Nature Trail, Allotment Site Codsall and U/S Bill Brook WRW (sites 1, 2 & 3) had no fish. The site D/S Bill Brook WRW (site 4) did not support intolerant species and only one alien species was found there. Tolerant species were present at all sites.

River Sence

The IBI for the River Sence was based on six sites (Fig. 5.3j). The IBI ranged from 0 to 54, with a mean of 36.17±17.51 (n = 6), indicating a "Fair" river (Table 5.23 & Table 5.6). Sites with three integrity classes, i.e. "No Fish", "Fair" and "Good", were found in the river (Fig. 5.4k). All "Good" sites were in the lower reaches (Appendix

5.3). One (site 4, Harris Bridge) of the four "Good" sites (sites 3, 4, 5 & 6) had an IBI close to the "Excellent" category boundary (Appendix 5.3). The IBI was zero for Heather Butterley Brick Works (site 1) as no fish were found.

No exotic species were recorded from the river and all sites supported intolerant species, except Heather Butterley Brick Works (site 1). Congerstone Cricket Pitch site (site 2) contained no tolerant species. Fish density was variable throughout the river (Fig. 4.34a), although fish diversity (Appendix 4.5) was relatively stable.

River Soar

The IBI score was calculated for 15 sites on the River Soar (Fig. 5.3k). The IBI for the individual sites ranged between 0 and 49 with a mean of 36.60±12.30 (n = 15), indicating a "Fair" river (Table 5.23 & Table 5.6). The integrity class consisted of "No Fish", "Poor", "Fair" and "Good" categories (Fig. 5.4l), indicating presence of extremely degraded sites as well as sites with habitat close to the natural state. Four (sites 12, 13, 14 & 15) of the seven "Good" sites were in the lower reaches, although a "No Fish" site (site 11, Barrow-on-Soar) was found in this reach. Four of the seven "Good" sites (sites 4, 5, 13 & 14) marginally qualified for the "Good" category. Leicester Straights and D/S Wanlip STW Outfall (sites 7 & 9) were two "Poor" category sites with IBIs close to "Fair" category boundary (Appendix 5.3).

Sites Leicester Straights and Abbey Meadows (sites 7 & 8) did not support intolerant species. All sites contained tolerant species except site 11. Exotic species were recorded from Abbey Meadows and Zouch (sites 8 & 13), indicating presence of biological pollutants. Variable fish density was determined throughout the river (Fig. 4.36a). Except for a number of sites, fish species diversity was relatively stable (Appendix 4.5).

River Sow

Nine sites were examined to determine an IBI for the River Sow (Fig. 5.3l). The IBI for the individual sites varied between 26 and 50, with a mean of 38.44±6.88 (n = 9), indicating a "Fair" river (Table 5.23 & Table 5.6). Sites on the River Sow fell into the "Poor", "Fair" and "Good" integrity classes (Fig. 5.4m). The mean IBI was comparatively low due to presence of a "Poor" site (site 3), although the score was close to the "Good" category boundary. Cresswell Farm (site 5) had an IBI close to the "Excellent" category and four (sites 1, 7, 8 & 9) of the five "Fair" sites had IBIs close to the "Good" category boundary (Appendix 5.3).

No exotic fish were recorded for the River Sow and all sites supported intolerant species. Chebsey (site 3) was the only site not to support tolerant species. Variable fish density was found throughout the river (Fig. 4.38a), although fish species diversity was relatively stable (Appendix 4.5).

River Tame

The IBI was calculated for six sites on the River Tame (Fig. 5.3m). The IBI for the individual sites ranged between 37 and 49, with a mean of 44.17±4.49 (n = 6), indicating a "Good" river (Table 5.23 & Table 5.6). The range of values indicated that the river contained sites with fish populations close to the natural state. Although, the average IBI only marginally qualified for the "Good" category. Among the sampling sites, only two integrity classes, "Fair" and "Good", were found (Fig. 5.4n). The IBIs of the two "Fair" sites (sites 1 & 5) were close to the "Good" category boundary (Appendix 5.3).

The River Tame was free from exotic species and all sites supported intolerant and tolerant fish species. Fish species diversity was relatively stable (Appendix 4.5) than fish density (Fig. 4.40a).

River Tean

The IBI for the River Tean was based on nine sites (Fig. 5.3n). The River Tean was identified as a "Poor" river with a mean IBI score of 23.67±14.00 (n = 9) (Table 5.23 & Table 5.6). The IBI range for the individual sites was 0 to 45, indicating presence of extremely degraded sites, and sites with fish populations close to the natural state (Table 5.24). The integrity class consisted of "No Fish", "Poor", "Fair" and "Good" categories (Fig. 5.4o), although only one "Good" site was found in the river. The mean IBI was low due to presence of two "No Fish" sites (sites 6 & 7) and three "Poor" sites (sites 1, 2 & 5) in the rivers course (Fig. 5.3n). The IBIs of different sites were marginal within respective integrity class ranges (Appendix 5.3).

No alien species were found in the river and all the sites supported intolerant species. Tolerant fishes were absent from four sites (sites 1, 5, 6 & 7). Fish diversity (Appendix 4.5) and density (Fig. 4.42a) were variable between different sites.

River Trent

The IBI was determined for 20 sites on the River Trent (Fig. 5.3o). The IBI for the individual sites ranged between 0 and 43, with a mean of 29.30±8.94 (n = 20), indicating a "Fair" river (Table 5.23 & Table 5.6). The values indicated that both extremely degraded sites and sites with fish populations close to the natural state were prevailing in the river. The integrity class consisted of "No Fish", "Poor", Fair" and "Good" categories, although only one "Good" site (site 20, Great Haywood Mill) was recognised (Fig. 5.4p). Except for Finney Gardens and Weston U/S Gayton Brook (sites 3 & 17), all other "Fair" sites marginally qualified for the "Fair" category. Great Haywood Mill (site 20), the only "Good" site also qualified marginally for "Good" category. Most "Poor" sites had IBIs close to the "Fair" category boundary (Appendix 5.3).

No exotic species were recorded and four sites (sites 6, 8, 16 & 18) did not support intolerant species. All sites contained tolerant species except for Norton Green and U/S Hoo Mill (sites 1 & 18). No fish were found at U/S Hoo Mill. Highly fluctuating but generally low fish density was determined throughout the river (Fig. 4.44a). Fish species diversity was also variable (Appendix 4.5).

5.5.3 The Yorkshire Ouse catchment

The IBI was calculated for 208 sites from two rivers of the Yorkshire Ouse catchment (Figs 5.5a & b). The IBI of individual sites ranged from 0 to 54 with a mean of 33.64±8.99 (n = 208), indicating a "Fair" catchment (Appendix 5.3 & Table 5.6). The range of values also indicated that the catchment contained both extremely degraded sites and sites with ecology very close to the natural state. The mean IBI for individual rivers varied between 25 and 35 with a mean of 30.00±5.00 (n = 2), also indicating a "Fair" catchment (Table 5.24 & Table 5.6). Both mean values marginally qualified for the "Fair" category.

The integrity class ranged from "No Fish" to "Good" (Fig. 5.6a) and no "Excellent" and "Very Poor" category sites were found within the catchment. No exotic species were recorded from the catchment. Fish species diversity fluctuated greatly, and density was very low throughout the catchment.

Table 5.24 IBI scores on the basis of 15 metrics for 2 rivers in the Yorkshire Ouse catchment

	River	Total number of sites	Average IBI score	SD±	Range	Integrity class
1	Aire	26	25	9.05	0 - 41	Poor
2	Nidd	182	35	8.18	0 - 54	Fair
Total / Average		**208**	**30**	**5.00**	**25 - 35**	**Fair**

River Aire

Twenty-six sites were sampled to calculate the IBIs for the River Aire (Fig. 5.5a). The IBI calculations were based on 14 metrics, as no data on fish biomass (metric 15) was available for this river. Although, the IBI for each site was evaluated with the same scale of integrity class, adopted for other rivers, based on 15 metrics. It was assumed that for this river, missing of a metric (biomass) would not influence the overall IBI as low numbers of fish were caught in the river. However, the IBI of a specific site (especially sites 9, 15, 20, 21, 25 or 26) could be increased a little, but would not cross the limit of the present integrity class boundary. The sites 9, 15, 20, 21, 25 and 26 contained 69 to 253 individuals in the sample.

The IBI for individual sites ranged from 0 to 41 with a mean of 24.50±9.05, indicating a "Poor" river (Table 5.24 & Table 5.6). The range value also indicated that the River Aire contained some extremely degraded and some moderately degraded sites in its course. The integrity classes varied between "No Fish" and "Fair" (Fig. 5.6b). No "Excellent", "Good" or "Very Poor" sites were found within the river length. Most of "Poor" sites had IBIs, close to the "Very Poor" category boundary (Appendix 5.3). Except for U/S Snaygill STW and D/S Snaygill STW above Cononley (sites 14 & 15), all other "Fair" sites had IBIs close to the "Poor" category boundary (Appendix 5.3). Malham Beck below Malham Cove and Calverley D/S Rawdon STW (sites 1 & 18) had scores of zero as no fish were found (Appendix 5.3).

The River Aire was free from alien species. Tolerant species were absent from 54% sites, most were in the upper reaches (sites 1 - 13), although, these sites contained 0 to 2 species only. Twenty three percent of total sites did not support intolerant species. All these sites were in the downstream from Calverley D/S Rawdon STW (site 18) to the mouth of the river. With two exceptions (sites 14 & 15), very low fish species diversity (Appendix 4.6) and density (Fig. 4.47a) were found throughout the river.

River Nidd

An intensive survey was conducted (Table 2.1) to assess the ecological health of the River Nidd through the IBI method and 182 sites within the river length were examined (Fig. 5.5b). The IBI of the individual sites ranged from 0 to 54 with a mean of 34.95±8.18 (n = 182), indicating a "Fair" river (Table 5.24 & Table 5.6). The range of values also indicated that the river contains sites with extreme degradation as well as sites with ecology very close to the natural state.

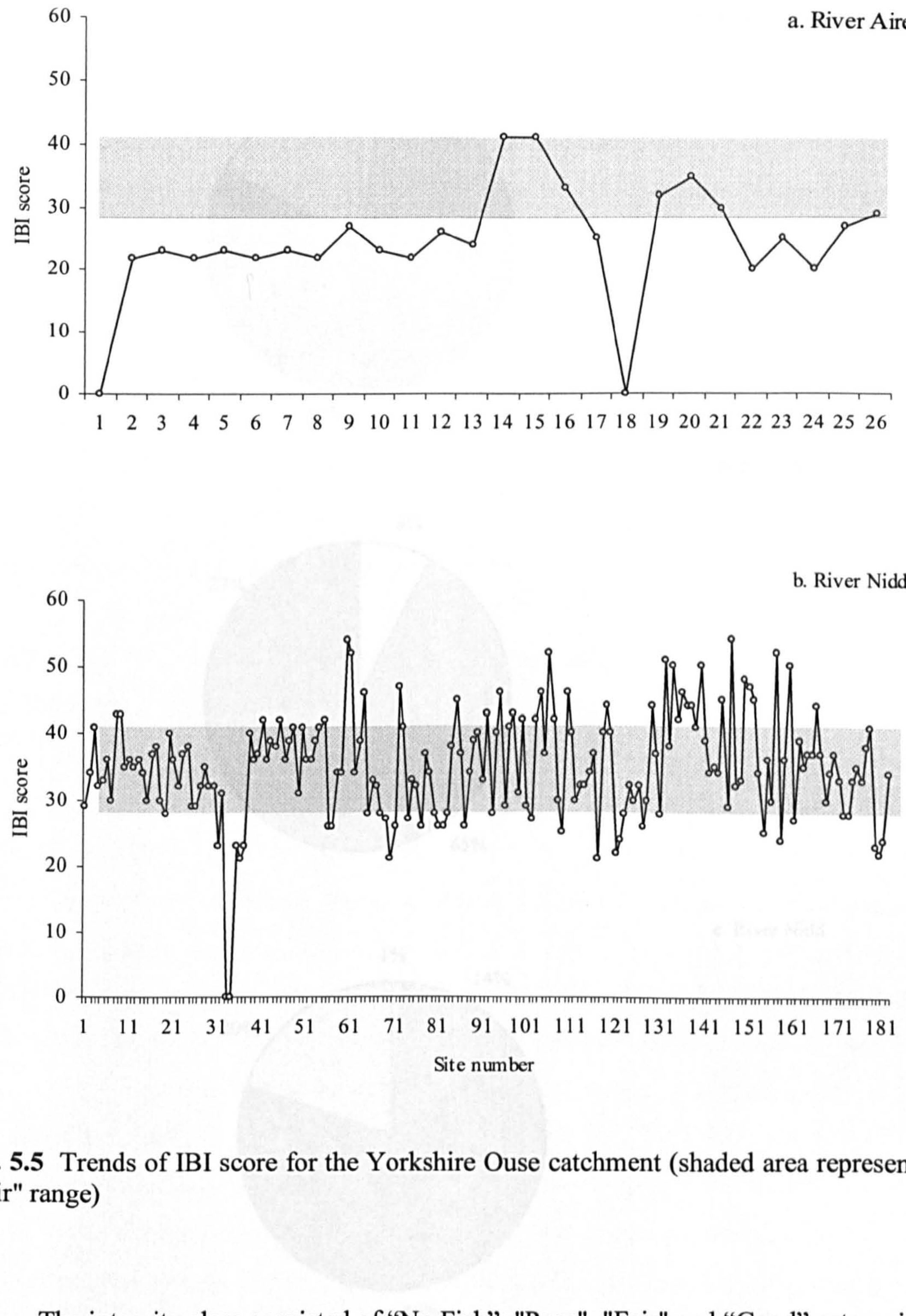


Fig. 5.5 Trends of IBI score for the Yorkshire Ouse catchment (shaded area represents "Fair" range)

The integrity class consisted of "No Fish", "Poor", "Fair" and "Good" categories (Fig. 5.6c), although only two "No Fish" sites (sites 33 & 34) were recorded in the river and these two sites were in the upper reaches (Appendix 5.3). The "Good" sites were distributed throughout the river although most "Good" sites were in the middle reaches and most "Poor" sites in the lower reaches (Appendix 5.3).

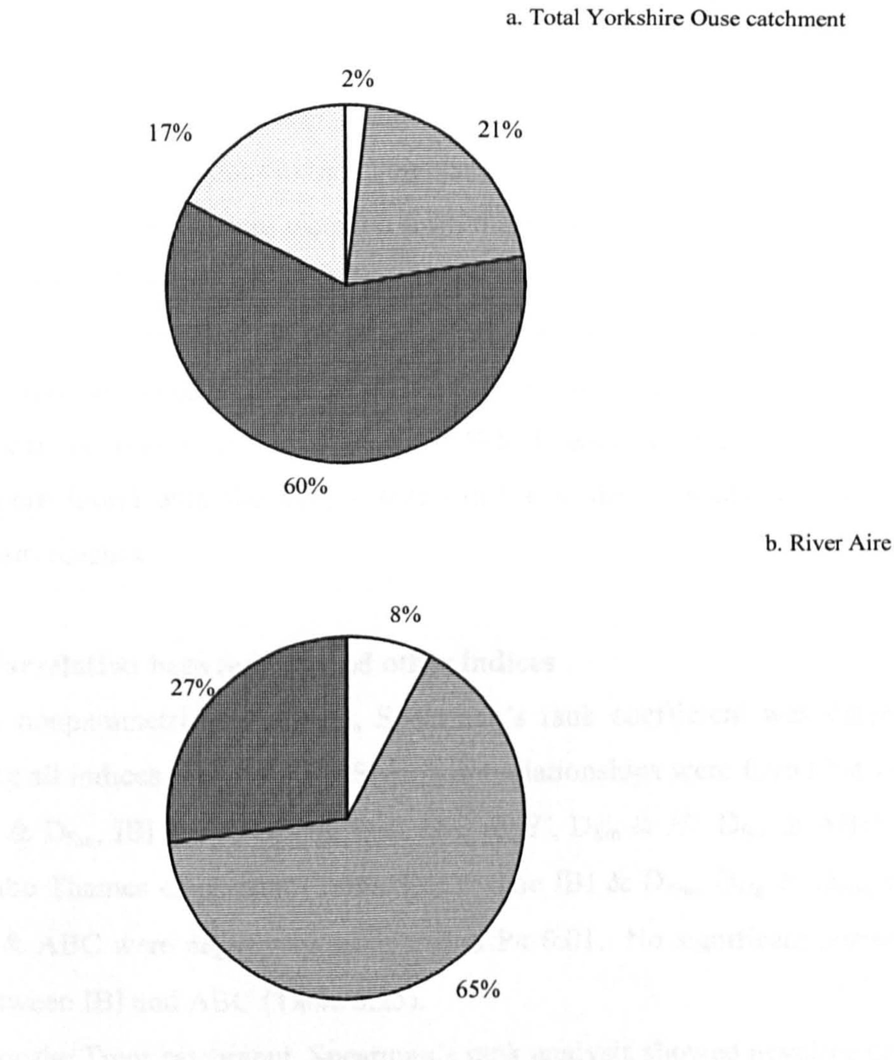


Fig. 5.6 Integrity class composition for 208 sites of 2 rivers of the Yorkshire Ouse catchment

The IBIs of nine (sites 60, 61, 106, 132, 134, 140, 147, 157 & 160) of the 36 "Good" sites were close to the "Excellent" category boundary (Appendix 5.3). Twenty-seven sites, representing 23% of all "Fair" sites had IBI values very close to the "Good" category boundary (Appendix 5.3). Fourteen sites, representing 54% of all "Poor" sites, also had IBI values very close to the "Fair" category boundary (Appendix 5.3).

No exotic species were recorded from the river and 20 sites, representing 11% of the total sites, did not support intolerant species. All these sites were in the middle through the lower reaches. On the other hand, 29 sites, representing 16% of the total sites were free from tolerant species. The majority of these sites were in the upstream reaches near the source of the river. Low fish diversity (Appendix 4.6) and density (4.49a) were found with the sites situated in the upstream areas compared with the downstream reaches.

5.5.4 Correlation between IBI and other indices

A nonparametric correlation, Spearman's rank coefficient was determined by comparing all indices (Table 5.25). Significant relationships were found between IBI & D_{Mg}, IBI & D_{Sm}, IBI & H', D_{Mg} & D_{Sm}, D_{Mg} & H', D_{Sm} & H', D_{Sm} & ABC and H' & ABC in the Thames catchment (Table 5.25). The IBI & D_{Sm}, D_{Mg} & D_{Sm}, D_{Sm} & H' and D_{Sm} & ABC were negatively correlated at $P< 0.01$. No significant correlation was found between IBI and ABC (Table 5.25).

For the Trent catchment, Spearman's rank analysis showed negative correlations between D_{Sm} & H' and D_{Sm} & ABC at 0.05 and 0.01 levels respectively, while the correlations were positive between IBI & D_{Mg}, IBI & H', D_{Mg} & H', D_{Mg} & ABC and H' & ABC (Table 5.25). No significant correlation was found between IBI and ABC.

In the Yorkshire Ouse catchment, significant positive correlations were found between IBI & ABC, IBI & H' and IBI & D_{Mg} (Table 5.25). The ABC showed positive correlations with D_{Mg} and H', while a negative correlation was found with D_{Sm} ($P< 0.01$). Negative correlations also exist between IBI & D_{Sm} and D_{Mg} & D_{Sm}.

Scatter diagrams for the Thames catchment, IBI showed positive relationships with D_{Mg} and H' (Fig. 5.7a & c) and a negative relationship with D_{Sm} (Fig. 5.7b). These results are supported by the Spearman's Rank correlation as the IBI showed similar relationships with D_{Mg}, H' and D_{Sm} (Table 5.25). No significant correlation exists between IBI and ABC (Fig. 5.7d).

Scatter diagrams for the Trent catchment suggested positive relationships between IBI and other indices (Figs 5.8a, b & c) except ABC (Fig. 5.8d). No significant relationship was found between IBI and ABC (Fig. 5.8d). Spearman's Rank coefficient supports the correlation between IBI and D_{Mg}, and IBI and H', while there is no correlation between IBI and D_{Sm} (Table 5.25).

Table 5.25 Spearman's Rank Correlation Coefficient (1 tailed) for all indices

Thames catchment (total sites, n = 86)

	D_{Mg}	D_{Sm}	H'	ABC	IBI
IBI	0.638**	-0.575**	0.683**	0.071	1.000
D_{Mg}	1.000	-0.472**	0.594**	0.063	
D_{Sm}		1.000	-0.963	-0.500**	
H'			1.000	0.400**	
ABC				1.000	

** Significant at 0.01 level

Trent catchment (n = 163)

	D_{Mg}	D_{Sm}	H'	ABC	IBI
IBI	0.675**	0.093	0.790**	-.002	1.000
D_{Mg}	1.000	0.066	0.838**	0.138*	
D_{Sm}		1.000	-0.135*	-0.241**	
H'			1.000	0.222**	
ABC				1.000	

* Significant at 0.05 level, ** Significant at 0.01 level

Yorkshire Ouse catchment (n = 208)

	D_{Mg}	D_{Sm}	H'	ABC	IBI
IBI	0.582**	-0.351**	0.734**	0.213**	1.000
D_{Mg}	1.000	-0.483**	0.834**	0.253**	
D_{Sm}		1.000	-0.684**	-0.321**	
H'			1.000	0.266**	
ABC				1.000	

** Significant at 0.01 level

Scatter diagrams for the Yorkshire Ouse catchment, the IBI showed positive relationships with D_{Mg}, H' and ABC (Figs 5.9a, c & d), and a negative relationship with D_{Sm} (Fig. 5.9b). These findings are also supported by the Spearman's Rank correlation as the IBI showed positive relationships with D_{Mg}, H' and ABC, and a negative relationship with D_{Sm} (Table 5.25).

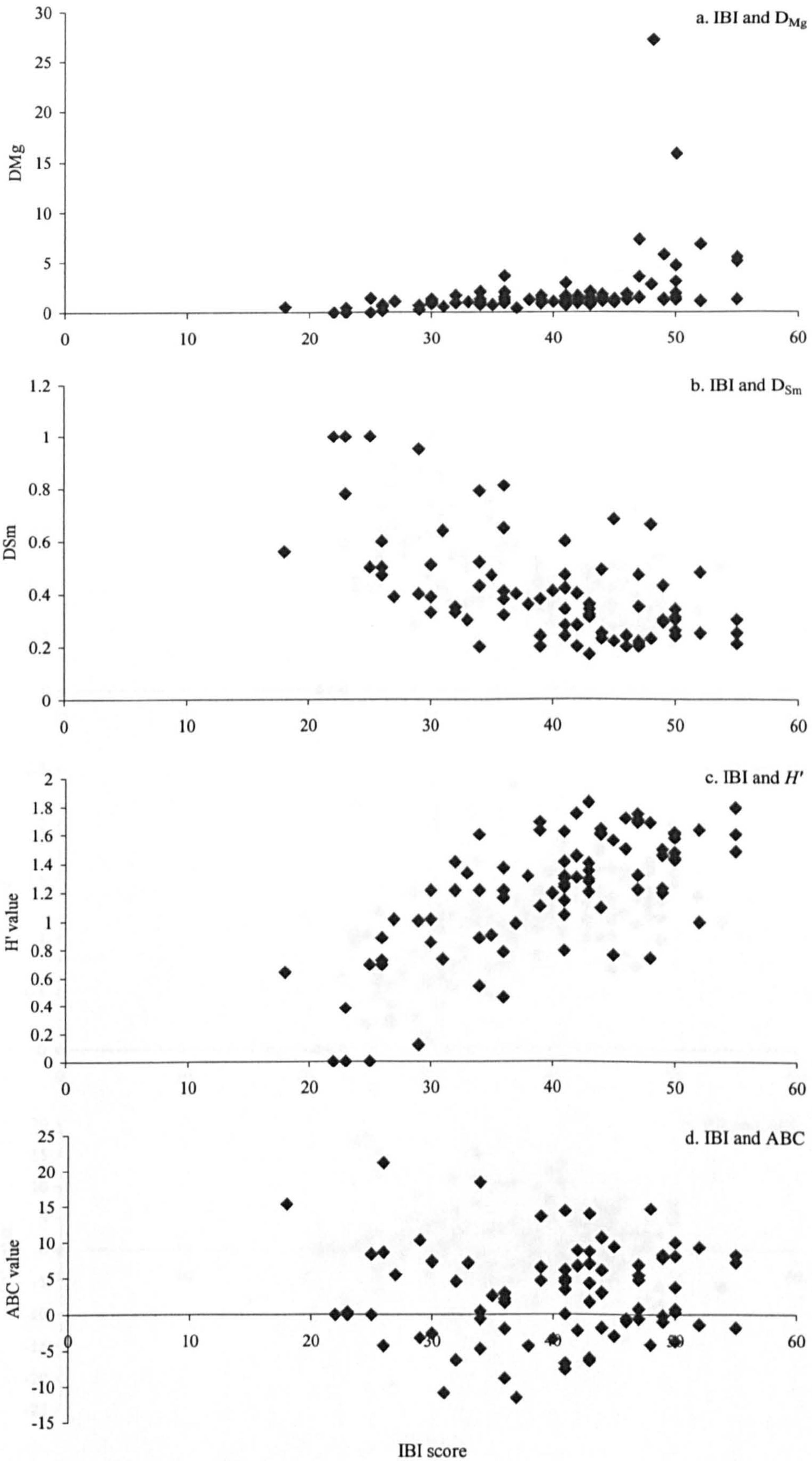


Fig. 5.7 Scatter diagram between IBI and other indices for the Thames catchment

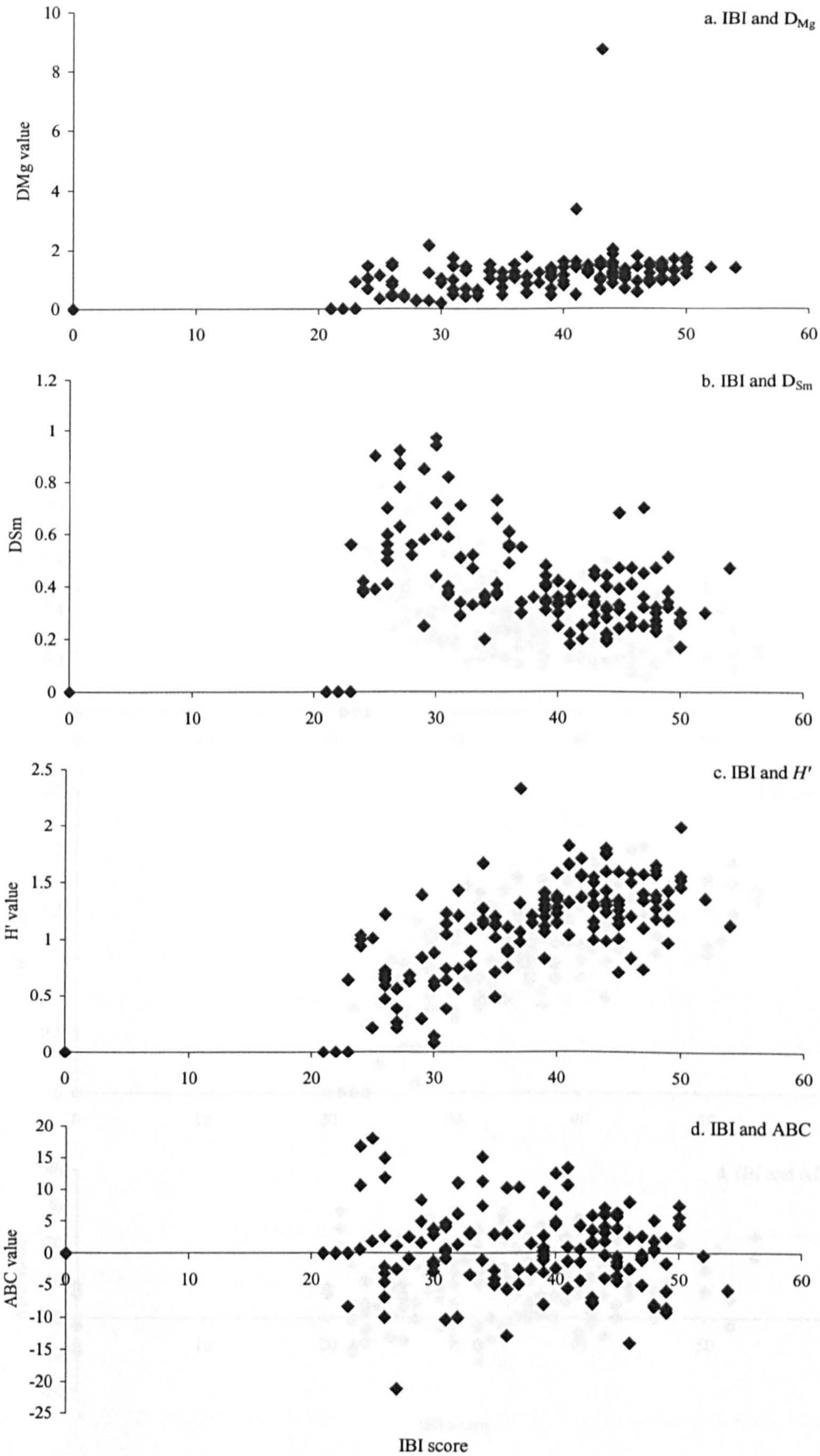


Fig. 5.8 Scatter diagram between IBI and other indices for the Trent catchment

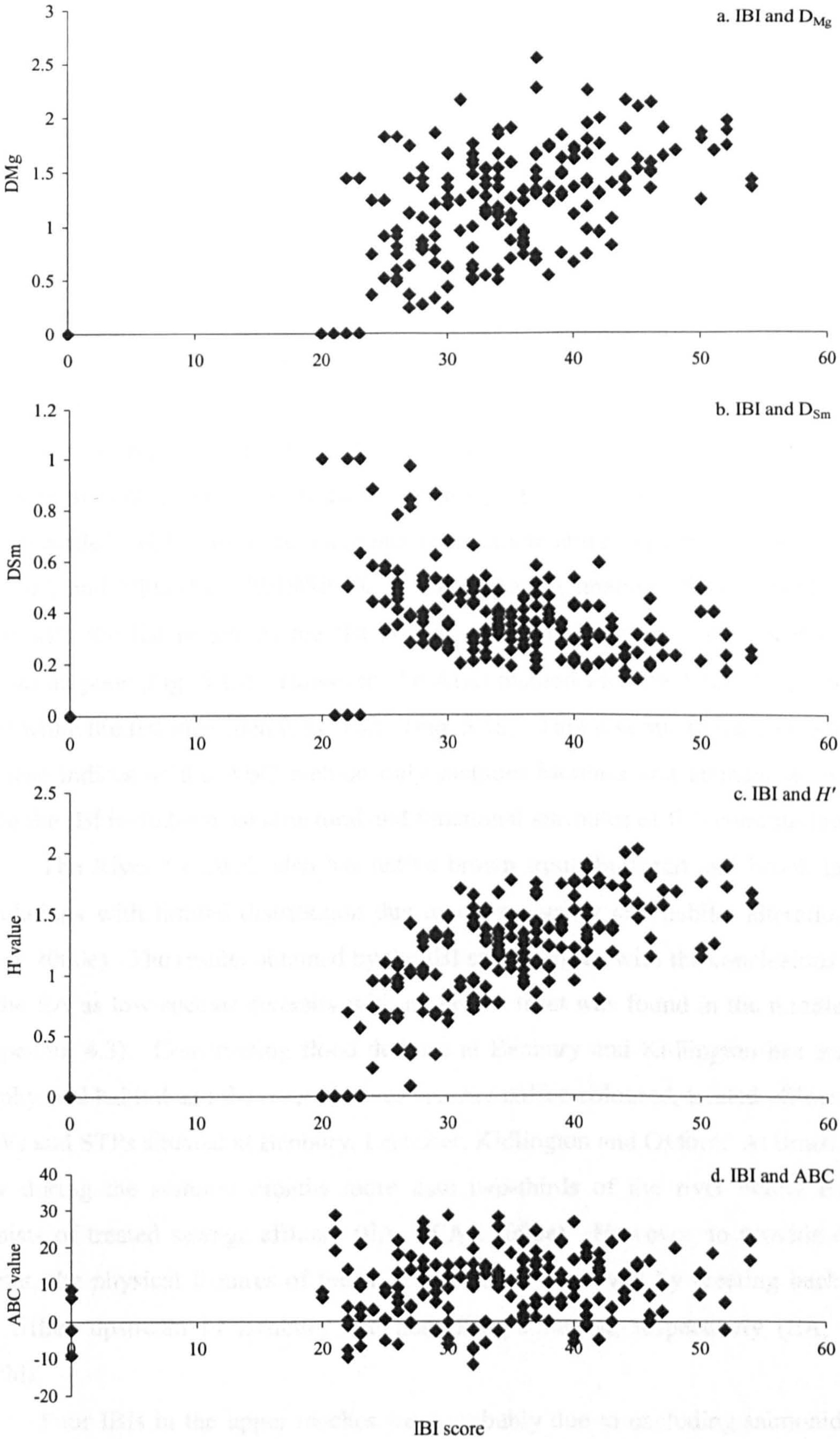


Fig. 5.9 Scatter diagram between IBI and other indices for the Yorkshire Ouse catchment

5.6 DISCUSSION

5.6.1 The Thames catchment

River Cherwell

The River Cherwell supports a high quality coarse fishery dominated by roach, dace and chub (EA, LEAP 2000e). The water quality class of the river varies between RE2 and RE4 (River Ecosystem Classification-2 & 4), indicating good to fair water quality (EA, LEAP 1999d). These recent findings of the EA do not support the results obtained under the IBI system which may be due to assessment criteria as the former used only fish density while the IBI included a range of fish community attributes to classify the river.

According to the IBI, the River Cherwell was generally "Fair" with higher IBI scores in the lower reaches than upper reaches (Fig. 5.1a). Ten species of coarse fish were recorded, with pike, dace, chub and roach dominating (Appendix 4.3). The D_{Mg}, D_{Sm}, H', and UPGMA, TWINSPAN & DECORANA analyses (Figs 4.5a-d, 4.6a-e) agree with the IBI results as the IBI also identified the sites 5, 6 & 7 in the middle reaches as poor (Fig. 5.1a). However, the ABC method identified site 9 as poor (Fig. 4.5e) while the IBI identified it as "Fair" (Fig. 5.1a). This was due to the characteristics of these indices as the ABC method only includes biomass and abundance of fishes while the IBI includes most structural and functional attributes of fish communities.

The River Cherwell also has native brown trout, bullhead and brook lamprey populations with limited distribution due to water quality and habitat alteration (EA, LEAP 2000e). The results obtained by the IBI system agree with the conclusions drawn by the EA as low species diversity with no brown trout was found in the middle reach (Appendix 4.3). Constructing flood defence at Banbury and Kidlington has modified the physical habitat and the river receives brown / coffee-coloured, treated effluent from STWs and STPs situated at Banbury, Leicester, Kidlington and Oxford. At times of low flow during the summer months more than two-thirds of the river below Banbury consists of treated sewage effluent (EA, LEAP 2000e). However, to provide desired habitat, the physical features of the river have been improved by creating backwaters and riffles upstream of Banbury and near King's Sutton, respectively (EA, LEAP 1999d).

Poor IBIs in the upper reaches were probably due to excluding salmonids (e.g. salmon & trout), grayling and, minor and headwater species (e.g. minnow, stone loach, spined loach & bullhead) as metrics (Fig. 5.1a). The reference condition used (Table 3.11) and the metrics chosen (Section 5.3.1) for this study were specific to the middle and lower reaches of English rivers, which were not appropriate for the upper reaches of a river.

The lower reaches also suffer from high suspended solids from the Oxford canal and water abstraction at Grimsbury and Cropredy Mill (EA, LEAP 1999d). As a result, the lower reaches contain impoverished or patchy fish populations (Fig. 4.5a). Punts and canoes also disturb the fish populations in the lower reaches. All these activities result in classification of sites on the river as "Poor" and "Fair", hence the IBI system was found to be an appropriate method to assess the ecological health of the middle and lower reaches of this river.

River Evenlode

On the basis of the IBI, the River Evenlode was classified as "Fair", although its mean IBI was very close to the lower limit of the "Good" category boundary (Table 5.22 & Table 5.6). Being an EC salmonid river (EA, LEAP 1996), the integrity class should be high but the IBI classified the river as "Fair" which was probably due to exclusion of salmonids in a metric. It is assumed that if the presence of brown trout was evaluated through a metric then the integrity class of the river may be changed to the "Good" category. Brown trout is the dominant fish species upstream of Oddington but recruitment is generally poor due to the "flashy" flow regime, as the River Evenlode responds rapidly to rainfall events (EA, LEAP 1997b). Fish diversity was high as 14 species, including brown trout, were recorded from the river (Appendix 4.3; EA, LEAP (1997b). According to the EA, LEAP (1997b), the River Evenlode harbours good mixed fish populations throughout most of its length with quality coarse fish dominating.

Water quality in the upper reaches is high and the river was classified as RE2 (EA, LEAP 1996). Habitat in the river has been improved by creating Off River Supplementation Units (ORSUs), narrowing the channel and creating marginal shelves at Ashford Mill, Combe and Cassington (EA, LEAP 1997b). These improvements in habitat exerted a positive impact on the fish populations at sites 12, 14 and 19 as reflected in the high IBI score (Fig. 5.1b). The average IBI was close to the boundary of the next integrity class.

River Stort

The IBI classified the River Stort as "Fair" while the EA classified water quality as good to fair (EA, LEAP 2001). The lower reaches of the river has a good quality coarse fishery, dominated by roach and perch followed by chub, dace, pike, bleak, tench, common bream and eel (EA, LEAP 2001), which were reflected in the IBI as most "Good" sites were found in the lower reaches (Fig. 5.1c). Twelve species of coarse fish were recorded from the river and 6 to 8 were found at each site in the lower reaches (Appendix 4.3). Fish species density was comparatively high in this section (Fig. 4.9a). All other indices, i.e. D_{Mg}, D_{Sm} and H', and UPGMA, TWINSPAN and DECORANA analyses (Figs 4.9a-d, 4.10a-e), which isolated site 16 as poor, agreed with the IBI (Fig. 5.1c). However, the IBI included sites 1, 2 & 3 as "Fair" (Fig. 5.1c) but the ABC method separated them as poor (Fig. 4.9e).

The upper Stort is a rural river with intensive agriculture. This section has limited physical and ecological diversity with a tendency to dry up in the summer months (EA, LEAP 1999e). Eutrophication affects the river due to agricultural run-off and discharge from sewage treatment works in the upstream areas (EA, LEAP 2001). These features of the river also reflect the IBI output as most "Fair" sites and one "Poor" site were in the upper reaches (Fig. 5.1c).

The fish communities are under pressure as the river receives treated sewage effluent from STWs situated at Clavering, Bishop's Stortford and Stansted Mountfitchet (EA, LEAP 1999e). The fish community also suffers stress from recreational activities such as angling, boating and canoeing (EA, LEAP 2001). Fish have lost their preferred habitat due to construction of flood defences (EA, LEAP 1999e) and aquatic plants are declining downstream of Harlow (site 13), limiting habitat for phytophils (EA, LEAP 1999f). All these characteristics indicated the variability of the habitat quality, which was reflected in the distribution pattern of the IBI (Fig. 5.1c).

River Thame

The River Thame supports a good to excellent coarse fishery from Nether Winchendon (site 12) to the confluence with the River Thames (site 18), with chub and dace the dominant species (EA, LEAP 1998e). The IBI classified the river as "Good" (Table 5.22 & Table 6.5) and would have been better except for the presence of one "Poor" sites" (site 1) in the upper reaches. This "Poor" site reduced the mean IBI score. Fish biomass was greater than 20 g m^{-2} downstream from the confluence of the Scotsgrove Brook (site 9) to the River Thames (site 18) (EA, LEAP 1997a). The biological quality of water varies between "Very good" and "Good" throughout the river. Physical habitat has been enhanced by re-instatement of gravel riffles at Chearsley and Nether Winchendon (EA, LEAP 1998e). These conclusions made by the

EA support the results obtained under the IBI system. In the downstream reach, only one "Poor" site and a high number of "Good" sites were found (Fig. 5.1d), indicating its generally high ecological integrity. Other indices (Figs 4.11a-d & 4.12a-c), which identified site 1 as poor, agree with the IBI (Fig. 5.1d). However, the ABC method isolated sites 12, 13 & 14 as poor (Fig. 4.11e) but the IBI identified them as "Good" (Fig. 5.1d) as the former method fails to include all population characteristics and is therefore, considered less reliable.

Fish populations in the River Thame between Aylesbury (site 4) and Nether Winchendon (site 12) are affected by a combination of water quality and habitat characteristics (EA, LEAP 1997a). The water quality problem is most pronounced upstream of Eythrope weir / sluice structures, where the impoundment exacerbates problems caused by high nutrient loading. This section also has poor quality habitat resulting from the wide dredged channel. Roach, with small chub and dace (EA, LEAP 1997a) dominate the fish community in this section, with fish standing crop ranging from 10 to 20 g m^{-2} (EA, LEAP 1997a). The fish community suffers pressure from sport fishing, as angling is one of the main recreational activities in the river. Other activities such as boating and canoeing also disturb the fish community (EA, LEAP 1997a). The IBI scores also agree with the above statement of the EA as three sites (sites 2, 3 & 4) were "Fair" within this section (Fig. 5.1d). Fish species diversity (Appendix 4.3) and density (Fig. 4.11a) were also low, indicating a disrupted fish community suffering from anthropogenic disturbances.

The upper reaches of the River Thame, especially from Stonebridge to Holman's Bridge, have poor fish communities as a result of low flows during summer months and chronic organic pollution (EA, LEAP 1998e). This section also receives treated effluent from STWs situated at and Weedon, and Lower and Nether Winchendon (EA, LEAP 1997a). Only minor species are present in the upper reaches (EA, LEAP 1997a). These problems in the upper reaches were also identified by the IBI system. Site Weedon (site 1) was classified as "Poor" while Nether Winchendon (site 6) had an IBI in the "Fair" category (Fig. 5.1d). The former site contained low fish diversity; density and biomass while the later contained low biomass but high diversity with high number of tolerant species (Fig. 4.11a).

River Windrush

The River Windrush is designated as an EC salmonid fishery from Harford Bridge to its confluence with the River Thames. Brown trout is the dominant fish species above Bourton-on-the-Water (EA, LEAP 1997b). However, the IBI classified the river as "Fair" with a mean IBI score very close to the lower limit of the "Good" category boundary (Table 5.23 & Table 5.6). This "Fair" classification was probably due to presence of three "Poor" sites (sites 1, 2 & 3) in the upper reaches (Fig. 5.1e), which reduced the mean IBI score. The three "Poor" sites in the upper reaches were probably due to omission of salmonids (e. g. brown trout) and other headwater species as metrics (Table 5.2). As mentioned before, specific reference condition and metrics are needed to assess ecological health of the upper reaches of a river. In addition the headwaters always harbour few fish species (Carpenter 1928, Huet 1959) and the upper reaches of the River Windrush also suffer erosion and pollution. However, it is assumed that if the presence of brown trout was evaluated through a metric then the integrity class of the river may change to the "Good" category. This again shows that the IBI as developed in this study is unsuitable in its present form for headwater sections. The D_{Mg}, D_{Sm} and H', and UPGMA, TWINSPAN and DECORANA analyses (Figs 4.13a-d & 4.14a-e) produced similar results to the IBI which isolated sites 1, 2 & 3 as poor in the upper reaches (Fig. 5.1e). Although, the ABC method classified site 5 as poor (Fig. 4.13e), the IBI identified site 5 as "Good" (Fig. 5.1e).

The middle reaches of the river have an impoverished habitat caused by dredging and impoundment for milling, with intermittent pollution from Witney STW. The river also receives domestic and industrial discharges, and agricultural and surface run-off. Water abstraction above Witney at Minster Lovell causes low flows (EA, LEAP 1997b). Angling is an important recreational activity that disturbs the fish populations of the lower Windrush. Other forms of physical disturbance include windsurfing, powerboating, waterskiing and sailing. These disturbances were reflected by the IBI system as all the "Fair" and only two "Good" sites were found in the middle reaches (Fig. 5.1e).

Fish communities are particularly good downstream of Witney due to the quality of the habitat (EA, LEAP 1996). Physical habitat has been improved by narrowing the channel, creating spawning substrate, shelves, riffles and re-instatement of gravel and groynes (EA, LEAP 1996). The quality class varied between RE1 and RE2 under the Rivers Ecosystem Classification (EA, LEAP 1996). Coarse fish dominate downstream from Bourton-on-the-Water to the Thames, although the river supports a mixed fishery (EA, LEAP 1997b). Theses characteristics of the river are reflected by the IBI as all "Good" sites were found in the downstream areas (Fig. 5.1e). The IBIs of sites 16 & 17 were close to the lower limit of "Excellent" although the ABC index identified site 17 as poor (Fig. 4.13e).

Spearman's Rank Correlation

Spearman's Rank correlations on 86 sites of the Thames catchment were positive for the IBI & D_{Mg}, IBI & D_{Sm}, and IBI & *H'* (Table 5.25 & Fig. 5.7a-c). This was due to the nature of data or attributes of the fish community used for these indices (e.g. fish abundance and biomass). The value of D_{Mg} is influenced by the sample size and a large numbers of juveniles are caught from the catchment (Section 4.3.2). The value of D_{Sm} is influenced by the abundance and dominant fish species. The catchment has high abundance of juvenile fishes and is dominated by few species (Section 4.3.2). The *H'* is influenced by the proportional abundance of species and by the sampling strategy. The IBI also incorporates all the attributed used in the D_{Mg}, D_{Sm} and *H'*, therefore, correlation was expected. The ABC index value was greatly influenced by juveniles (numerical abundance) and few large fishes (biomass abundance), which were the common features of the catchment (Section 4.3.2). In the IBI calculations, juveniles were not separated from the sample or not included as a separate "metric" for the catchment (Table 5.2). The biomass of a few large individuals did not influence the total IBI for a river or a catchment as many other attributes were involved in the IBI calculations.

5.6.2 The Trent catchment

River Anker

The water quality of the River Anker is variable along its length from fairly good to good in its upper reaches, but deteriorates to fair water quality below Nuneaton STW, before returning to fairly good quality at Tamworth (EA, LEAP 2000a). The quality class varied from RE2 to RE4 (EA, LEAP 2000a). The IBI scores for the River Anker agree with these assessments as some "Fair" and "Good" sites were in the upper reaches, while most "Fair" sites were confined to the middle reaches and again "Fair" and "Good" sites were in the lower reaches (Fig. 5.3a). These variable characteristics of the river were probably due to input of treated effluent from STWs. Apart from the ABC method, other indices (Figs 4.16a-d & 4.17a-e), which also identified site 4 as "Fair", agreed with the IBI (Fig. 5.3a). However, the ABC method identified sites 3, 4, 5, 6, 7 & 8 as poor (Fig. 4.16e) but the IBI classified them as "Fair" to "Good" (Fig. 5.3a) and the latter is considered to be a more accurate assessment.

The River Anker receives acidified water from Mancetter quarries (site 4) (EA, LEAP 2000b). Moreover, in the rural areas, numerous smaller sewage treatment plants (STPs) discharge to the river. Water quality is fair from Nuneaton STW to Mancetter Bridge (site 4) and was classified as RE4 (EA, LEAP 2000a). The IBI supports this classification as the Mancetter (site 4) was found as "Fair" with marginal IBI score (Fig. 5.3a). The river also experiences agricultural pollution from silage, slurry stores and other farm wastes. The river has eutrophication problems, resulting in excessive weed growth, which limits the habitat of different species (EA, LEAP 2000a). Reshaping of banks, altering natural flows and water levels, and constructing flood defences such as flood relief by-pass channels has also altered natural habitats. All these activities results in the unstable nature of fish communities reflected in the IBI (Fig. 5.3a).

River Blithe

The IBI classified the River Blithe as "Fair", although the mean IBI was close to the "Good" category boundary (Table 5.23). Three sites (sites 1, 2 & 3) in the upper reaches had a low IBI and were classified as "Poor" (Fig. 5.3b). These findings agree with the assessments undertaken by the EA, as the upper reaches have limited fish stocks due to low river flows and organic pollution from combined sewer overflows (EA, LEAP 1997a). The D_{Mg}, D_{Sm} and H', and UPGMA, TWINSPAN and DECORANA analyses (Figs 4.18a-d & 4.19a-e), which identified sites 1, 2 & 3 as poor in the upper reaches, agree with the IBI (Fig. 5.3b). However, the ABC method grouped site 3 as good (Fig. 4.18e).

From Leigh (site 4), the river supports good coarse fisheries especially populations of dace and chub (EA, LEAP 1997a) and the lower section has good brown trout fisheries (EA, LEAP 1998c). Downstream of Field (site 5), the fishery status is consistently high and grayling are present. The IBI was also supported by the EA conclusions as all the "Good" sites were in the downstream sections (Fig. 5.3b), especially below Field where grayling were present. If the presence of brown trout and grayling was evaluated through an appropriate metric then the integrity class of the river is likely to improve.

Water of the River Blithe is of good quality and the quality class varied between RE2 and RE3 based on chemical indicators (EA, LEAP 1997). Although there have been occasional pollution incidents in the vicinity of Blythe Bridge (site 1) and some problems with urban storm run-off. There are no major sewage works discharging to the river and only limited combined sewer overflows around its headwaters. This good quality river is impounded in Blithfield Reservoir for public water supplies (EA, LEAP 1997a). Water is also abstracted below Blithfield Reservoir at Nethertown close to the confluence with the River Trent. Coarse fish dominate reaches below the reservoir and densities are generally high (EA, LEAP 1997a). The IBI adequately reflects the situation as high fish diversity (Appendix 4.5) and density (Fig. 4.18a) were found below the reservoir.

Blithfield Reservoir effectively divides the River Blithe and two gauging weirs at Hamstall Ridware (site 11) and Nethertown (EA, LEAP 1997a) limit downstream migrations of fish. The river bed was dredged to increase depth (Fig. 4.18). The above features of the river adversely impact on the fish community causing a degraded condition. Due to the combined effects of all these stresses, the River Blithe is characterised by a low IBI and classified as "Fair" (Table 5.23).

River Blythe

The River Blythe is a high quality rural river, and has an excellent coarse fishery for much of its length with the middle reaches managed for trout (EA, LEAP 1999c). Dace, chub, rainbow trout and brown trout are the dominant species in the river. Biologically the water of the river is generally of good to fair quality while the quality class ranges between RE2 and RE3 under chemical classification, also indicating a fair to good class river (EA, LEAP 1998d).

These findings of the EA do not support the results obtained under the IBI system as the latter classified the river as "Fair" (Table 5.23). This contradiction may be due to assessment criteria as the EA used only fish density while the IBI included a range of fish community attributes to classify the river. Moreover, as mentioned before, if brown trout and rainbow trout were evaluated through an appropriate metric then the integrity class of the river is likely to improve. Four sites (sites 6, 7, 8 & 9) from U/S Eastcote to Blythe Mill End in the lower reaches had IBIs of "Good" category (Fig. 5.3c). Dace and chub dominated throughout the river and rainbow trout dominated in the middle reaches of the river (Appendix 4.5). Fish species diversity (Appendix 4.5) and density (Fig. 4.20a) were also high. However, the Blythe has serious water quality and low flow problems in the headwaters as no fish were found at Cheswick Green (site 1). This was probably due to urbanisation in the headwaters that caused deterioration in the water quality and reduced flows. With the exception of the ABC method, other indices and multivariate analyses (Figs 4.20a-d & 4.21a-e), which identified site 1 in the upper reaches as poor, agree with the IBI (Fig. 5.3c). However, the ABC method identified site 8 as poor (Fig. 4.20e) while the IBI grouped it as "Good" (Fig. 5.3c).

During prolonged dry periods almost the whole of the flow in the River Blythe is abstracted for public water supply. The river receives treated effluent from 10 large STWs above the abstraction point and also from a large number of smaller STPs. Downstream of the Eastcote Brook (site 7), the treated effluent constitutes about 50% of the flow under dry weather conditions (EA, LEAP 1998d). However, water quality is good due to improvement in the sewage treatment system. The IBI also reflects these improvements made by the EA, as "Good" sites were found only in the downstream reaches (Fig. 5.3c).

The River Blythe suffers from eutrophication problems and from pesticide pollution. It also receives discharges from limestone and sandstone quarries (Fig. 4.20). The natural flow regime has been altered by installation of a surface water balancing system downstream of Solihull (site 6). High levels of angling and boat traffic (EA, LEAP 1998d) have also degraded the physical habitat. Combined effects of all these anthropogenic activities have been reflected in the variation in the IBIs along the water course (Fig. 5.3c).

River Churnet

According to the IBI, the River Churnet was "Poor", although an improvement in the IBI score was observed in the lower reaches (Fig. 5.3d). The upper and middle reaches produced low IBIs (Fig. 5.3d). The variations in IBI in the river reflect the findings of the EA. The fish populations in the middle reaches have been detrimentally affected in the past by the impact of sewage and industrial effluents (EA, LEAP 1999d). However, due to improvements in the quality of the effluents, the fish population is showing signs of recovery in the lower reaches (Fig. 4.22a). The D_{Mg}, D_{Sm} and H', and UPGMA, TWINSPAN and DECORANA analyses (Figs 4.22a-d & 4.23a-e), which separated the upper reaches as poor, agree with the IBI (Fig. 5.3d). Although the ABC method identified site 12 as poor (Fig. 4.22e), the IBI classified it as "Fair" (Fig. 5.3d).

The lower reaches have good mixed and coarse fisheries (EA, LEAP 2000c). The recent report of the EA also supports the IBI as the "Fair" and "Good" sites were found in the lower reaches (Fig. 5.3d). No fish were found at three sites (sites 3, 4 & 5) downstream of Tittesworth Reservoir (Appendix 4.5). This may be due to low flows associated with other habitat deterioration (Fig. 4.22) as huge amounts of water are removed for drinking purposes by 20 abstraction licence holders (EA, LEAP 1999d).

The River Churnet receives surface run-off from the catchment and treated effluent from Cheddleton and Groghall STWs (sites 9, 10 & 11). According to the Rivers Ecosystem Classification, quality class ranged from RE1 to RE4, indicating very good to fair quality, respectively (EA, LEAP 1999d). Anthropogenic disturbance (e.g. water abstraction, habitat degradation) and fluctuating water quality resulted in variable fish species diversity (Appendix 4.5) and density (Fig. 4.22a) resulting in variable IBIs (Fig. 5.3d). Corroboration of these observations by the IBI suggests it is a suitable measure of ecological health of the river.

River Cole

Due to variable water quality as a result of sewage and industrial discharges, and surface run-off (Fig. 4.23), the upper reaches of the River Cole support only sparse coarse fish populations (EA, LEAP 1999c). The quality class varied between RE3 and RE4 indicating fair to moderate water quality (EA, LEAP 1998d). The assessments of the EA support the IBI as the River Cole was classified as "Poor". Fish diversity (Appendix 4.5) and density (Fig. 4.24a) were poor in the upper reaches, because of anthropogenic disturbances (Fig. 4.24). The D_{Mg}, D_{Sm} and H', and UPGMA, TWINSPAN and DECORANA analyses (Figs 4.24a-d & 4.25a-e), that identified sites 1 - 6 as "Poor" in the upper reaches, agree with the IBI (Fig. 5.3e). However, the ABC method identified site 11 in the "Poor" category (Fig. 4.24e) as the method did not include the functional attributes of fish community but the IBI, which takes account of most community attributes, categorised the site as "Good" (Fig. 5.3e).

The River Cole shows poor biological quality, particularly in the Haybarnes Bridge to Stechford section (sites 1 - 5). This section of the river suffers from the effects of combined sewer overflows and urban run-off (EA, LEAP 1998d). In this study, no fish were found in the sites around Haybarnes Bridge (sites 1 - 5), reflecting the problem (Fig. 4.24).

The physical habitat of the river has been improved by removing all sheet piling and concrete bank re-enforcements and by creating pools and wetlands across the River Cole (EA, LEAP 1998d). Sustainable populations of dace, chub and roach have been present for several decades in the lower reaches at Coleshill (sites 10 - 13) (EA, LEAP 1998d). This has been reflected in the IBI scores for the aforesaid sites (Appendix 4.5).

The River Cole suffers from low flow conditions, and pollution from limestone and sandstone quarries (Fig. 4.24). Upstream migration is disrupted due to two weirs at Cook's Lane and Moorend Avenue (sites 8 & 9) and as a result the IBI was poor for

D/S Cook's Lane Bridge (site 9) (Fig. 5.3e), although this has been partially addressed by construction of a "rock chute" pass for the Cook's Lane weir (EA, LEAP 1998d). It was considered that the IBI developed in this study, provided an appropriate measure of the ecological health of the middle and lower reaches of the River Cole.

River Derwent

The IBI classified the River Derwent as "Fair" (Table 5.23), although the EA reported that the upper reaches (sites 1 - 6) harbour a high quality trout fishery, and downstream of Derby (sites 14 & 15) supports a good quality coarse fishery (EA, LEAP 1999a). This contradiction between the IBI and EA assessments was probably due to the exclusion of salmonids (e.g. brown trout) in the metrics. As mentioned previously, the reference condition and the metrics chosen for this study were aimed at the middle and lower reaches of English rivers and have resulted in low IBI scores in the upper reaches (Fig. 5.3f). Clearly the IBI needs to be modified before it can be used on upper reach sites. It is assumed that if the presence of brown trout was evaluated through an appropriate metric then the integrity class of the river would probably change to the "Good" category and this could be tested in future studies. Angling is an important activity in the River Derwent, which exerts considerable pressure on trout and coarse fisheries. Other recreational activities that disturb the fish communities include navigation, boating and canoeing (EA, LEAP 1999a). The D_{Mg}, D_{Sm} and H', and multivariate analyses (Figs 4.26a-d & 4.27a-e), that separated site 1 in the upper reaches as "Poor", agree with the IBI (Fig. 5.3f). However, the ABC method grouped sites 8, 10, 13 & 14 with poor sites (Fig. 4.26e) while the IBI classified them as "Fair" to "Good" (Fig. 5.3f).

The River Derwent receives consented discharges from 19 and 8 sewage treatment and industrial effluent treatment works, respectively. There are 42 STWs in the catchment and the majority of these discharge directly to the river or its tributaries. The river also receives a number of direct discharges from small privately-owned STPs (EA, LEAP 1999a). The water quality immediately downstream of such discharges may be significantly affected, resulting in diurnal variations in the dissolved oxygen level, pH and un-ionised ammonia, which may lead to fish kills. During warm summer nights in June and July, the DO concentration declines to low levels due to the discharge of huge amounts of treated effluents from different sources (EA, LEAP 1999a). Moreover, constructing 171.2 km of flood defence structure has modified the natural habitat of the river. Removing surface water through 95 licensed abstractions

(EA, LEAP 1999a) has also altered water flow and level. Impacts of all these activities were reflected in the IBI as seven sites (sites 2, 4, 5, 6, 7, 8 & 10) in the upper through middle reaches which were classified as "Fair" (Fig. 5.3f). Therefore, it was considered that the IBI provided an appropriate measure of the ecological health of the middle and lower reaches of the River Derwent.

River Idle

Very low fish species diversity (Appendix 4.5), density (Fig. 4.28a) and biomass (Section 4.3.2) was observed throughout the River Idle, which were reflected in the IBI scores at each site (Fig. 5.3g). These were due to fluctuating water levels and river flows (Fig. 4.28) affected by periodic pumping (EA, LEAP 1999b). Water is pumped from Gringley Carr (an area of low-lying land) into the River Idle and a pumping station at West Stockwith then pumps water from the Idle into the River Trent. This reduces flow velocity in the River Idle and gives rise to eutrophic conditions. Moreover, the river is characterised by poor habitat features due to heavy engineering works (EA, LEAP 1999b) as the river has been straightened and deepened for different purposes such as navigation (Fig. 4.28). Downstream of Gamston, the river has been highly modified for flood relief purposes and embankments constructed in some areas. This area is intensively farmed and has sparse tree cover, and the river is affected by sewage effluent and urban runoff, causing eutrophication (Fig. 4.28). The river also receives floodwater from mines in the catchment (Fig. 4.28), which affects fish spawning success and fry survival (EA, LEAP 1999b). With the exception of the ABC method, other indices and multivariate analyses (Figs 4.28a-d & 4.29a-b), which identified site 4 as "Good", agree with the IBI (Fig. 5.3g). The ABC index identified site 1 as "Good" (Fig. 4.28e) due to the index's simple character but the IBI classified the site as "Poor" by considering more fish community attributes (Fig. 5.3g).

The IBI was zero, i.e. "No Fish", at Misson (site 5). This section of the river is highly disturbed by different activities such as sailing, windsurfing, boating, canoeing taking place in this area (EA, LEAP 1999b). Moreover, the pumping station at West Stockwith also influences the section by altering the flow regime. In addition, water in this section was characterised by high conductivity and low visibility. Although no fish were found at site 5, angling pressure was high in this section of the river, suggesting that survey was inadequate to catch fish. Therefore, the IBI was unable to assess this site (site 5). However, all these features indicated that the River Idle, in particular, Misson section, was suffering from serious anthropogenic disturbances.

River Mease

The River Mease is of fairly good quality along its entire length and was classified as RE3 on the basis of chemical indicators (EA, LEAP 2000a). The IBI adequately reflects the findings of the EA. Under the IBI system, the River Mease was classified as "Good" (Table 5.23) and no "Poor", "Very Poor" or "No Fish" sites were found on the river (Appendix 5.3). Fish species diversity (Appendix 4.5), density (Fig. 4.30) and biomass (Section 4.3.2) were also high. However, a number of "Fair" sites were identified in the river (Appendix 5.3). This was probably due to discharges from a number of coal mines around Moira (sites 1 & 2) and treated effluent from STWs (sites 3 & 4). Moreover, a large amount of surface water is abstracted for irrigation purposes through surface water licences that alters the natural flow regime (EA, LEAP 2000b).

Other indices, i.e. D_{Mg}, D_{Sm} and H', and multivariate analyses, UPGMA, TWINSPAN and DECORANA analyses (Figs 4.30a-d & 4.31a-e) agree with the IBI (Fig. 5.3h). However, the ABC method separated site 1 in the upper reaches as "Poor" (Fig. 4.30e) but the IBI grouped it in the "Fair" category (Fig. 5.3h) and the latter was considered to be a more accurate assessment.

River Penk

The IBI classified the River Penk as "Fair" (Table 5.23) although the river supports both coarse and trout fisheries (EA, LEAP 1998c). This contradiction between the IBI and EA assessments was probably due to the exclusion of salmonids from the IBI (e.g. brown trout), resulting in low IBI scores in the upper reaches (Fig. 5.3i). Chub, dace and brown trout are the dominant species in the river (Appendix 4.5) (EA, LEAP 1997a).

The River Penk in its upper reaches (sites 1 - 7) receives treated sewage effluent from Cannock, Codsall and parts of Wolverhampton and contaminated urban storm run-off (Fig. 4.32). This effluent and urban run-off gives a poor to fair biological quality of the river while water quality based on chemical parameters varies between RE2 and RE3 (EA, LEAP 1997a). The IBI adequately reflects the findings of the EA as three sites (sites 1 - 3) in the upper reaches had no fish (Fig. 5.3i). Other indices and multivariate analyses (Figs 4.32a-d & 4.33a-e), which separated the upper region as "Poor" (Fig. 5.3i), agree with the IBI. However, the ABC method grouped sites 8, 9, 10 & 11 as poor sites (Fig. 4.32e), while the IBI classified them as "Good" (Fig. 5.3i).

The water quality improves from poor to fairly good from Brewood (site 6), downstream to its confluence with the River Sow at Stafford (site 11) (EA, LEAP

1997a). This assessment of the EA is also reflected in the IBI as all the "Fair" and "Good" sites were found in Brewood area. Due to three "No Fish" sites in the upper reaches (sites 1 - 3, Fig. 4.32a), the average IBI was low and the river was classified as "Fair".

The fish community is also under pressure from anglers as sport fishermen exploit the River Penk. Construction of extensive flood defences (EA, LEAP 1997a) has altered natural habitat of the river. In some sections, narrowing the channel through the installation of deflectors and creation of artificial weed beds has altered habitat (Fig. 4.32). In other sections, the river has been deepened by dredging (Fig. 4.32). Moreover, a low weir upstream of Stretton Mill (site 7) restricts upstream movement of fish in dry weather (EA, LEAP 2000d). All these activities also reflect the "Fair" ecological health of the river and therefore, it is considered the IBI provided an accurate assessment of the ecological health of the River Penk.

River Sence

The IBI analysis classified the River Sence as "Fair" (Table 5.24). However, fish diversity (Appendix 4.5), density (Fig. 4.34a) and biomass (Section 4.3.2) were high in the lower reaches especially from Congerstone (site 3) to the confluence with the River Anker. This contradiction was due to the existence of one "No Fish" and one "Fair" site in the upper reaches, which reduced the mean IBI and did not qualify for the "Good" category (Table 5.23). As mentioned before, upper reaches of a river usually have low fish diversity and density (Huet 1949). The IBI scores are in line with the findings of the EA who classified the river as RE3 on the basis of chemical indicators, which is able to support high-class coarse fish populations (EA, LEAP 2000a). The D_{Mg}, D_{Sm} and H', and UPGMA, TWINSPAN and DECORANA analyses (Figs 4.34a-d & 4.35a-e) were also similar to that of the IBI (Fig. 5.3j). However, the ABC method separated site 4 as poor (Fig. 4.34e) but the IBI recognised the site as "Good" (Fig. 5.3j).

The River Sence receives treated effluent from STWs and numerous smaller STPs (Fig. 4.34). The river also receives agricultural run-off from the catchment containing fertilisers, suspended solids, pesticides and herbicides (Fig. 4.34). Natural habitat of the river has been altered by constructing flood defences (Fig. 4.34). Fish populations are also exploited by anglers, as angling is the major water based recreational activity in the river (EA, LEAP 2000a). All these activities result in "Fair" and "No Fish" category sites in the river. No fish were found at Heather Butterley

Brick Works (site 1) in the upper reaches and this may be due to discharges from the brick works (Fig. 4.34), or to the sampling methods being unable to catch fish. Results of this study (Appendix 5.3) indicated that with the exception of the IBI, other indices partially assessed the ecological health of the river.

River Soar

The IBI identified the River Soar as "Fair" (Table 5.23). However, the EA stated that the water quality is of fair to good, having good fisheries throughout its length, containing 5 to 12 species with a biomass range of 2 to 56 g m^{-2} (EA, LEAP 1997b). This contradiction between the IBI and EA assessments was due to low IBI scores (Appendix 5.3) in the middle reaches (sites 6 - 10) of the river. This section of the river was affected by different anthropogenic activities (Fig. 4.36) and at Barrow-on-Soar (site 11) an IBI score of zero was recorded as no fish were found (Fig. 5.3k). This extreme situation was the result of discharge of mine water from a Gypsum mine at Barrow-on-Soar (Fig. 4.36). Except for the ABC method, other diversity indices and multivariate analyses (Figs 4.36a-d & 4.37e) supported the IBI (Fig. 5.3k). The ABC method grouped sites 5, 8, 10, 13 & 15 with "Poor" category (Fig. 4.36e) but the IBI identified them as "Fair" to "Good" (Fig. 5.3k).

Other anthropogenic activities such as flood defences, flow regulation structures, dredging and widening the channel have impacted on the habitat, leading to poor fish stocks in the lower reaches (sites 6 - 15) of the river (Fig. 4.36a). In addition, the river receives urban drainage, causing pollution. The water often becomes pink, purple or inky black in colour due to receiving discharges from dyehouses, associated with the textile industry, located in Wigston (sites 3 - 5), Leicester (sites 6 - 8) and Loughborough (sites 10 - 12) (Fig. 4.36).

At Ashby-de-la-Zouch (site 13), where there are many septic tanks, caused localised water quality deterioration in May 1992. Weirs and locks were constructed to regulate water level for navigation that disrupted migration routes of many fish species. About 110 Mld^{-1} of water is abstracted for different purposes (EA, LEAP 1997b). Fish populations are also under pressure from pleasure boating, rowing and canoeing. Angling and match fishing also disturb the fish community. The combined effects of all these activities decreased the biotic integrity of the River Soar and resulted in IBIs varying from "Poor" to "Good".

River Sow

The River Sow is a good quality rural river, which is noted for its coarse fish populations, in particular chub (EA, LEAP 1997a). Biologically, the water quality is very good within the stretches between Eccleshall and Hilcote (sites 1 -5) (EA, LEAP 1997a). Water quality class varies from RE2 to RE5, based on chemical indicators (EA, LEAP 1997a). The IBI classified the River Sow as "Fair" (Table 5.23). Fish species diversity (Appendix 4.5), density (Fig. 4.38a) and biomass (Section 4.3.2) were average throughout the river. This contradiction between the findings of the EA and the IBI may be due to assessment criteria as the former used only fish density while the IBI included a range of fish community attributes to classify the river. Three "Fair" (sites 7, 8 & 9) and three "Good" (sites 4, 5 & 6) sites were found from Great Bridgeford to its confluence with the River Trent. The D_{Mg}, D_{Sm} and H', and UPGMA, TWINSPAN and DECORANA analyses (Figs 4.38a-d, 4.39a-e), which also separated site 3 in the upper reaches as poor, also agree with the IBI (Fig. 5.31). However, the ABC method isolated sites 6 & 7 as poor (Fig. 4.38e), while the IBI identified them as "Good" and "Fair", respectively (Fig. 5.31).

The upper reaches of the river suffer from high densities of algae due to eutrophication at the Mere (site 1) and downstream to Chebsey (site 3) (Fig. 4.38). Weed growth is high in summer and seriously blocks the channel, restricting the available habitat for fish populations (Fig. 4.38). Habitat diversity is also poor in some sections due to land drainage works (Fig. 4.38). One "Poor" (Chebsey, site 3) and two "Fair" (Eccleshall Castle and Hilcote Hall, sites 1 & 2) sites were found in the upper reaches (Fig. 5.31), which may be due to such disturbances (EA, LEAP 1998c).

Angling pressure is high from Eccleshall (site 1) to its confluence with the River Trent (EA, LEAP 1998c). Fish populations also suffer from pollution (Fig. 4.38). In addition, constructing of flood defences, narrowing the channel and dredging the bed of the river has altered the physical habitat (Fig. 4.38). Great Bridgeford (site 4) and Milford (site 6) gauging stations alter the natural flow. There are low weirs at Great Bridgeford (site 4) and Stafford town centre (site 8), which have disrupted the migration patterns (EA, LEAP 1997a). The river also receives sewage and industrial effluents from the Bramcote STW (sites 4 & 5). All these activities have exerted pressure on the fish communities and resulted in low IBI scores.

River Tame

Water quality in the lower reaches of the River Tame has improved significantly due to construction of Lea Marston Purification Lakes in the early 1980s. Moreover, a number of improvements to the sewerage system including combined sewer overflows have been completed. The physical habitat has also been improved along the banks, and fish refuges have been created. All these activities have helped to improve the water quality and habitat of the river. As a result, fish are recolonising the River Tame as far upstream as the Lea Marston lakes (site 1) (EA, LEAP 1998d).

These observations support the results obtained from the River Tame using the IBI system (Appendix 5.3). The IBI classified the lower section of the river as "Good" (Table 5.23). Fish species diversity (Appendix 4.5), density (Fig. 4.40a) and biomass (Section 4.3.2) were high in this section indicating improvements in the water quality and habitat. The fish community comprised mixed species from different trophic and reproductive guilds (Appendix 4.5). This section not only supports tolerant species but also intolerant species like rainbow trout, barbel and dace (Appendix 4.5). The high fish species diversity in this section of the river suggests that the Lea Marston Purification Lakes are acting as good refuge for the fish community of the lower reaches of the river or may be due to improvements in the water quality. Except for the ABC method, other indices, D_{Mg}, D_{Sm} and H', and multivariate analyses, UPGMA, TWINSPAN and DECORANA (Figs 4.40a-d & 4.41a-e) agree with the IBI (Fig. 5.3m). The ABC method grouped all the sites (site 1 - 6) as poor category (Fig. 4.40e), while the IBI classified them as "Fair and "Good" (Fig. 5.3m).

To assess the ecological integrity of the entire river, it is necessary to sample the reaches above the Lea Marston Lake, as fish populations in the upper Tame remain extremely sparse. This is especially the case above Lea Marston Purification Lakes, where there are no sustainable fish populations. Only minor tolerant species, particularly 3-spined stickleback, exist locally where refuges are available. Small populations of roach, derived from off line balancing lakes, survive temporarily (EA, LEAP 1998d). Under the Rivers Ecosystem Classification based on chemical parameters, the River Tame was classified as RE4 category, however, some stretches were classified as RE5. The water quality is bad to fair and may be suitable for some coarse fish species but not for salmonids or other intolerant species.

In the upper reaches, some improvements in the sewerage system have already been carried out and some treatment works have been closed (e.g. Oldbury Treatment Works). These closures of sewage treatment works relate to site-specific improvements in the water quality, which follow relocation of the treated sewage outfall. The River Tame is still extremely vulnerable to periodic water quality deterioration due to urban run-off and discharges from combined sewer overflows (EA, LEAP 1998d). Summer rainfall reduces dissolved oxygen level and urban run-off discolours the receiving watercourse giving it, a cloudy or grey appearance due to the level of suspended solids (EA, LEAP 2000a).

Apart from the water quality problems, the physical habitat of the river has also been altered by the construction of extensive flood banks at Witton, Hamstead, Bescot and Oldbury (sites 1 - 5) (EA, LEAP 1999c). Moreover, five flood balancing areas have been constructed at Ocker Hill, Sheepwash, Bescot, Sandwell, and Perry Hall Playing Fields (EA, LEAP 1998d). The River Tame and its tributaries also suffer from litter, unauthorised tipping and other forms of aesthetic pollution (EA, LEAP 1998d). At Lea Marston the average flow in the river consists of 55% treated sewage effluent and industrial waste, and under dry conditions up to 90% of the surface water draining from the West Midlands - Tame catchment is made up of treated effluent (EA, LEAP 1999c).

River Tean

The middle reach of the River Tean supports poor fish populations due to the effect of sewage and trade effluents (Fig. 4.42). The river receives discharges from a milk processing plant at Fole (site 6) (EA, LEAP 1999d). The river receives major discharges from the Checkley STW (site 5) and also receives discharges from sand and gravel quarries (Fig. 4.42). The treated sewage effluent discharge constitutes over half the flow in dry season (EA, LEAP 1999d). These characteristics of the river were adequately included in the IBI as the river was classified as "Poor" (Table 5.23). "Poor" and "No Fish" sites were found in the upper through middle reaches especially downstream from the Checkley STW (site 5) (Appendix 5.3). Other indices, i.e. D_{Mg}, D_{Sm} and H', and UPGMA, TWINSPAN and DECORANA analyses (Figs 4.42a-d & 4.43a-e), which also identified sites 1 & 2 as "Poor", agree with the IBI (Fig. 5.3n). The ABC method identified site 4 as poor (Fig. 4.42e), while the IBI identified the site as "Fair" (Fig. 5.3n).

The fish populations are, however, recovering as improvements have been carried out in the effluent treatment works at Checkley (site 5) (EA, LEAP 1999d). The quality class ranged between RE2 and RE4, indicating good to fair quality water (EA, LEAP 1999d). This assessment of the EA is also reflected in the IBI analysis as "Fair" and "Good" sites were found in the downstream of the Checkley STW (site 5). It is therefore, considered that the IBI was a better index than other diversity indices, the ABC method or multivariate analyses to measure ecological health of the river.

River Trent

The section of the River Trent included in the study supports low coarse fish populations, although the status of the fishery varies quite widely within the catchment (EA, LEAP 1998c). These observations made by the EA agree with the determination by the IBI system. The IBIs were low and classified the river as "Fair" (Table 5.23). The mean IBI (Table 5.24) only marginally qualified for the "Fair" category (Table 5.6). A large number of the sites were suffering from poor water quality due to habitat deterioration (Fig. 4.44). The River Trent receives effluents from 138 sewage and storm overflows, 94 private sewage treatment plants, 99 industrial discharges and 38 sewage treatment works (EA, LEAP 1998c). The D_{Mg}, D_{Sm} and H', and multivariate analyses, UPGMA, TWINSPAN and DECORANA (Figs 4.44a-d & 4.45a-e), which identified sites 1, 5, 8 & 16 as "Poor", agree with the IBI (Fig. 5.3o). However, the ABC method grouped sites 2, 4, 9, 10 & 20 in the poor category (Fig. 4.44e), while the IBI identified them as "Fair" (Fig. 5.3o).

The stretch from Abbey Hulton to Hanley (sites 7 - 9) suffers pollution from a number of sources (Fig. 4.44). Water quality of the River Trent is also affected by discharges from combined sewer overflows including Mill Farm (Fig. 4.44). Both water quality and fishery are poor on the River Trent between Tittensor (site 12) and Hoo Mill (site 19) (Fig. 4.44a). These findings were also reflected in the IBI scores as no fish were found at U/S Hoo Mill (site 18) (Fig. 5.3o).

The water flow is altered by removing 284 Mld^{-1} of surface water through 184 abstractions. In addition, physical habitat has been modified by constructing flood defences for a length of about 218 km (EA, LEAP 1998c). Fish migration is affected by weirs and other channel structures (Fig. 4.44). Fish populations were imbalanced both in terms of quality (Appendix 4.5) and quantity (Fig. 4.44a) due to all these activities. This was reflected in the IBIs at each site (Fig. 5.3o).

Spearman's Rank Correlation

Nonparametric Spearman's rank correlation analysis and scatter diagrams showed significant relationships exists between IBI & D_{Mg}, and IBI & H' for 163 sites in the Trent catchment (Table 5.25 & Fig. 5.8). As mentioned previously (Section

5.6.1) this significant relationships was due to the nature of the data or attributes of the fish community used for these indices (e.g. fish abundance and biomass). The IBI takes account of most structural and functional attributes of a fish community (Table 5.2) but other indices, ABC method and multivariate analyses are based only on structural attributes (Table 4.2). No significant correlation exists between IBI & ABC as the latter is markedly influenced by the number of juvenile fishes. Therefore, it is considered that the IBI is a better index to measure the ecological health of the river than other diversity indices, ABC method and multivariate analyses.

5.6.3 The Yorkshire Ouse catchment

River Aire

On the basis of the IBI, the River Aire was assessed as "Poor" (Table 5.24). The IBI was zero at Malham Beck below Malham Cove and Calverley D/S Rawdon STW (sites 1 & 18) as no fish were found because of pollution resulting from the disposal of sheep dip pesticides. Moreover, Otterburn Beck (site 3) and Crosber Beck (site 12) in the upper catchment (northwest of Gargrave) were adversely affected by organic pollution (Fig. 4.47). The site at Calverley D/S Rawdon STW (site 18) was affected by the discharges from Esholt and Rawdon STWs (Fig. 4.47). The IBI scores for individual sites were low in the River Aire throughout its length (Appendix 5.3). This was due for example, to poor water quality, low water level and flow, water abstraction and lack of instream cover. The river and many of its tributaries carry effluents from the industrial conurbations of West Yorkshire (EA, LEAP 1998a). The lower reaches also receive sewage effluent from domestic properties through village drains (Sewer Dykes) via small watercourses (EA, LEAP 1998a). Other indices, i.e. D_{Mg}, D_{Sm}, H', multivariate analysis and UPGMA (Figs. 4.47a-d & 4.48a) also identified sites 14, 15, 16, 19, 20, 21 & 26 as "Fair", which agrees with the IBI (Fig. 5.6a). However, the D_{Mg}, D_{Sm} and H' indices grouped sites 12 & 13 with good sites while the IBI identified them as "Poor" (Fig. 5.6a).

Fish populations are under stressed conditions as the river is extensively used for boating, cruising, canoeing and angling. Moreover, from Hunslet to Goole (sites 24 - 26) commercial vessels use the river (EA, LEAP 1998a). From Skipton to Bingley the area has been modified for flood defence purpose and a large volume of water is removed downstream of Leeds and at Ferrybridge (site 21). The river between Knottingley and its confluence with the River Ouse lacks bankside vegetation. There are 30 weirs in the river, which act as barriers to the free passage of both trout and coarse fish species, preventing natural migration patterns (Fig. 4.47). All these activities suggest that the fish community of the River Aire was under serious pressure from antropogenic disturbances. The effects of these activities seem to be reflected in the distribution pattern of the IBI scores (Fig. 5.6a). Fish species diversity (Appendix 4.6), density (Fig. 4.47a) and biomass (Section 4.3.2) were low from source to the mouth of the river. As a result, low IBIs were found and the river was classified as "Poor". This result (Appendix 5.3) agrees with the EA, which classified the River Aire as a poor quality river (EA, LEAP 1998a).

River Nidd

The River Nidd was classified as "Fair" on the basis of IBI scores (Table 5.24). This finding is different from the assessments of the EA as the water quality of the river is generally high that supports a high quality fishery containing salmonids and cyprinids (EA, LEAP 1998b). As mentioned previously, if the presence of brown trout and grayling were evaluated through an appropriate metric then the integrity class of the river is likely to improve. Data used in this study revealed that brown trout, grayling, perch, chub, dace, gudgeon and roach were distributed throughout the river (Appendix 4.6). Despite barbel, bleak, tench, common bream, ruffe and pike being absent from Holme bottom farm (site 29) to the source of the river, no "Poor" site was found within this section (Appendix 5.3). Barbel were abundant from D/S Scotton Hospice (site 49) to the mouth of the river. Fish species diversity was high from Knaresborough STW (site 72) to the mouth of the river (Section 4.3.2). The D_{Mg}, D_{Sm} and H', and multivariate analyses, UPGMA and TWINSPAN (Figs 4.49a-d & 4.50a-d) also agree with the IBI (Fig. 5.6b). However, the IBI identified most sites as "Fair" throughout the river (Fig. 5.6b) but the ABC method identified more sites as poor in the upper reaches (Fig. 4.49e) even though they often weren't. As stated before this was due to the organisational basis of the indices as the IBI includes all community characters while the ABC method relies only on abundance and biomass.

In some stretches, a number of STWs and storm overflows affect the water quality. Pollution from the storage of silage, slurry and agricultural fuel oil sometimes affects water quality (Fig. 4.49). In addition, a large volume of water is abstracted to supply the West Yorkshire conurbations and these activities alter the natural flow regime (EA, LEAP 1998b). These disturbances were reflected in this study as two sites had IBI's of zero and 26 sites were classified as "Poor" (Fig. 5.6b). Most "Poor" sites were in the downstream section below the site upstream of Scotton weir (site 36), indicating the presence of anthropogenic disturbances. Therefore, it is considered that the IBI provided a more accurate measure of ecological health of the river than other indices.

Spearman's Rank Correlation

In the Yorkshire Ouse catchment, Spearman's Rank correlation analysis and scatter diagrams for 208 sites showed significant correlation between the IBI and other indices (Table 5.25 & Fig. 5.9).

5.7 SUMMARY

The IBI developed in this study was modified from Karr (1981), which was based on 12 metrics. Instead of 12 metrics, the IBI developed in this study for England is based on 15 metrics (Table 5.2). The three new metrics included were "percent of individuals as rheophilic species", "percent of individuals preferring vegetated areas" and "total biomass (g m^{-2}) (Table 5.2). The reference condition developed (Table 5.3) and the metrics chosen (Table 5.2) for this study were specific to the middle and lowland reaches of English rivers. Therefore, salmonids, minor and headwater species were not included in the reference condition (Table 5.3) and no metric were chosen specific to these fishes (Table 5.2). Not surprisingly therefore, low IBIs were found when the IBI was tested in the upper reaches of some rivers (e.g. River Evenlode, Windrush, Churnet and Derwent) (Appendix 5.3). Therefore, for a headwater IBI, it will be necessary to develop a separate reference condition, and metrics should be chosen specific to headwater fish communities. No fish were caught from a small number of sites on the rivers Blythe, Churnet, Cole, Idle, Penk, Sence Soar, Tean, Trent, Aire and Nidd, which reduced the mean IBI scores of these rivers (Appendix 5.3).

A continuous rating scale (i.e. 5, 4, 3, 2, 1 & 0) is suggested to be appropriate to score IBI metrics as it included all values, yielded from a sum of metrics calculated (Section 5.4.2). A total of six integrity classes (e.g. Excellent, Good, Fair, Poor, Very Poor & No Fish) on a continuous scale were chosen to define biotic integrity of English lowland rivers and the number of integrity classes was considered optimum for the study rivers. The value of class boundaries was fixed arbitrarily on the basis of overall biodiversity and ecology of British freshwater fishes, and supported with reference to other IBIs for temperate regions but ranges were adjusted to take account of the number of metrics compared to Karr (1981) (Section 5.4.2). Calculations of IBI metrics are simple and include numerical and percentile scales with no complicated mathematical formulae needed (Section 5.4.3).

To summarise, the IBI developed was considered appropriate to evaluate the ecological health of the middle and lower reaches of the study rivers as the index generally agrees with the assessments provided by EA based on water quality criteria etc. The IBI, however, failed to predict the quality of the fisheries in headwater streams because of the exclusion of salmonid species and general poor species diversity found in these zones. If metrics which accounted for salmonid population characteristics are included this scenario would probably change and the IBI may become a better predictor of ecosystem health in these zones.

Although the diversity indices and multivariate analyses appeared to predict the quality of the environment reasonably well, these indices were shown to be less sensitive to change. The IBI includes more relevant information, so has less chance of underestimating problems. Under certain circumstances such as high abundance of juveniles, other indices and especially the ABC, are unreliable. The IBI is, however, able to identify such problems and is able to produce an appropriate index for a particular river. Throughout the study the ABC index produced results which conflicted with the IBI and other indices. This arose because the ABC uses only species abundance and biomass and does note account for the structural and functional aspects of the fish communities. Consequently, the ABC is not considered a good indicator of ecosystem health based on fish. In contrast the IBI seems to be a good indicator of fishery quality because it uses more attributes related to community structure and function. Overall it was thought to be the best method tested.

CHAPTER SIX

6. DISCUSSION AND CONCLUSIONS

6.1 INTRODUCTION

The Index of Biotic Integrity is a widely adopted and apparently effective tool which uses fish assemblage data to assess the environmental quality of aquatic habitats. The original version of the IBI has been modified in numerous ways for application in many different countries and habitat types (Table 5.1). Retaining the basic principles of Karr (1981), the IBI was modified and tested on a number of English lowland rivers using data from the EA archives. A reference condition was developed (Table 3.11) and metrics appropriate for English lowland rivers were selected (Section 5.3.1). A continuous metric rating scale was selected for six integrity classes with continuous class boundaries (Section 5.4.2). This chapter describes merits and weakness of the components used to develop an IBI for English lowland rivers and compares the output with a variety of commonly used indices (Section 4.3.2). The possibility of reducing the number of metrics to minimise the data requirements is examined. Modifications (e.g. inclusion of other metrics) and the possible application of the English version of IBI to headwaters are discussed. On the basis of the discussion, a number of recommendations are made for further development and application of the IBI to situations in the UK. It is anticipated that this work will form the basis to develop indices to meet UK obligations under the Water Framework Directive (Section 1.2).

6.2 IBI, DIVERSITY INDICES AND MULTIVARIATE ANALYSIS

A number of indices were used in this study (Section 4.3.2) and compared with the newly developed IBI (Section 5.5). No single diversity index appeared to be effective in evaluating the ecological health of a river (Section 4.3.2). All diversity indices tested showed specificity to certain perturbations.

6.2.1 Shannon-Wiener index (*H'*)

The Shannon-Wiener index (*H'*) was more discriminatory at evaluating fish community change than other indices but was biased towards high species richness and requires random sampling (Maguraan 1988) (Table 6.1). The *H'* incorporates species richness and evenness, both of which are implicitly assumed to be positively correlated with ecosystem well-being or ecological integrity (Washington 1984). However, due to independent responses of species richness and evenness to different forms of ecosystem degradation, analyses based on H' may yield results that lead to ambiguous interpretations of ecosystem status. The H' analysis also needs large randomly sampled data (Maguraan 1988), a condition which is difficult to satisfy with sampling on rivers. Due to access problems and resource constraints (e.g. manpower, money) collection of large random sampled data from a river may be problematic. On the other hand, non-random sampled data usually under or over estimates the status of a fish community (Peet 1974). Therefore, caution should be taken when using non-random sampled data to calculate H' values of a river. Maguraan (1988) came to a similar conclusion when calculating H' for polychaete populations in temperate intertidal habitats.

Table 6.1 Comparison between indices (mean values) applied in this study
DNA = Data not available as no biomass information

River	IBI	ABC	H'	D_{Sm}	D_{Mg}
Cherwell	35	6.01	1.17	0.59	1.10
Evenlode	41	4.96	1.25	0.65	1.06
Stort	35	-0.67	1.03	0.49	1.10
Thame	45	3.16	0.55	0.60	5.68
Windrush	41	1.57	1.12	0.54	1.09
Anker	43	-1.86	0.52	0.58	2.10
Blithe	39	3.08	0.44	0.51	0.88
Blythe	37	-2.64	0.41	0.50	0.96
Churnet	27	0.12	0.31	0.36	0.74
Cole	23	-0.21	0.30	0.34	0.48
Derwent	37	1.55	0.41	0.51	0.87
Idle	25	4.66	0.26	0.33	0.61
Mease	43	3.52	0.63	0.70	1.31
Penk	29	-0.04	0.39	0.43	0.81
Sence	36	0.60	0.43	0.49	1.18
Soar	37	-0.42	0.45	0.53	0.93
Sow	38	1.30	0.56	0.63	1.28
Tame	44	-3.34	0.50	0.52	1.17
Tean	24	0.20	0.24	0.32	0.38
Trent	29	2.57	0.36	0.45	0.94
Aire	25	DNA	0.37	0.28	0.40
Nidd	35	9.23	1.20	0.66	1.22

May (1975) described H' as an insensitive measure of the character of species distribution, and unsatisfactory due to lack of exploration of its biological relevance (Kovalak 1981, Washington 1984). Rees *et al.* (1990) stated that H' has no biological significance and the same value of H' may arise from different communities with varying species evenness and diversity.

Due to potential bias to high species richness, the H' was considered inappropriate to address adequately the biotic integrity of the study rivers (Section 4.3.2). For example, the rivers Windrush and Cole had the same total species richness (10 species, Appendices 4.3 & 4.5) but the former had a higher average H' value than the latter (Table 6.1). This was because number of fish species in the Windrush at individual sites was higher (6 - 9 species) and the catch more evenly spread (Appendix 4.3) than the River Cole, where number of species fluctuated between zero and 5 and the community was thus dominated by few species (Appendix 4.5). Variable H' values in the rivers Windrush and Cole were also probably due to the presence of different numbers of dominant species, as the fish community of the River Windrush was dominated by six fish species (Appendix 4.3) while the River Cole was dominated by only three species (Appendix 4.5). The H' values of the study rivers were also linked to river zonation (Section 4.4.1) as the upper reaches (usually because of lower species richness) tend to have lower H' values than lowland reaches (Figs 4.13d & 4.24d). Conclusions from the H' analyses (Section 4.3.2) were supported by Angermeier & Schlosser (1987), who also found H' an inappropriate index to assess the biotic integrity of small Illinois streams. This is partly because H' is sensitive to the presence of large numbers of juveniles and consequently Angermeier & Schlosser (1987) suggested that H' is more sensitive than the IBI to short-term population fluctuations such as seasonal abundance. The high abundance of juveniles may thus explain the high H' obtained at some sites (Table 6.1, e.g. River Anker) in the present study.

6.2.2 Diversity indices

Pielou (1975) stated that a diversity index is a single descriptive statistic and in itself not very informative and may lack any biological significance. Generically, diversity indices suffer from the following limitations:

a. Diversity indices incorporate relatively little biological information, which severely hampers their use in detailed analyses of ecological systems (Peet 1974);

b. Diversity indices address community structure but ignore the function of species in communities (Fausch *et al.* 1990);

c. Diversity indices do not consider species ecology (e.g. monotopic, eurytopic, herbivore, carnivore) or absolute abundance (Fausch *et al.* 1990);

d. Under pristine conditions, fish species diversity may vary substantially by season (Murphy 1978), among years (Angermeier & Schlosser 1987), and longitudinally in streams (Angermeier & Karr 1983);

e. Calculation of diversity indices is easy but interpretation is difficult. After calculating the value, one must then determine how it can be used to assess environmental degradation (Karr *et al.* 1986);

f. Although diversity indices generally decline with severe degradation, species richness and diversity may actually increase with minor or moderate degradation (Leidy & Fielder 1985);

g. Although community structure is influenced by both numbers and biomass of species it is unclear which data are best to use (Fausch *et al.* 1990).

In this study, the use and interpretation of diversity indices also suffered from these broad-spectrum limitations. The indices used were based on species presence/absence and relative abundance in one sample per location. Attempts were made to relate outputs to perturbations (Section 4.3.2) but this did not always prove easy because the degraded sites which maintained diversity at lower abundance still produced a higher diversity index value (Table 6.1, e.g. River Tame). Therefore, it is concluded that diversity indices are not an appropriate tool to measure the ecological health of a river. Bowen *et al.* (1996) found diversity indices were poor indicators of biotic integrity and supported this conclusion. Thus, understanding what the various diversity indices are indicating is problematic and they may not be identifying degradation but merely natural, temporal and spatial variations in species distribution and abundance.

6.2.3 ABC method

In comparison, the ABC method evaluated fish community structure more clearly and meaningfully than diversity indices, as the abundance/biomass comparison reflects the result of a combination of input parameters (Section 4.3.2). However, this method also fails to integrate functional components of a fish community. The method is data intensive and needs a large number of random samples (Coeck *et al.* 1993), and is heavily biased by high abundance of small individuals and few samples (e.g. five sites in the River Idle). In this study, negative ABC index values were recorded for a large number of sites due to prevalence of many juvenile fishes at these sites (Section 4.3.2). The presence or absence of few large individuals (e.g. presence and absence of large pike, at sites 9 and 10 on the River Cherwell, respectively, Fig. 4.5e) can significantly change the results. At site 9, a few large pike gave a high biomass but low abundance and consequently produced a negative ABC value. Coeck *et al.* (1993) found similar problems with presence or absence of large fishes when calculating ABC index for Belgian lowland rivers. Furthermore, the ABC index is more sensitive to water pollution than physical disturbances such as habitat modification and dredging. For example, site 7 on the River Churnet had a negative ABC value probably due to effects of water pollution (Fig. 4.22e) while site 14 on the River Soar had a high positive ABC value, despite site 14 was affected by river engineering works (Fig. 4.36e). Therefore, the ABC index has limitations when used for discriminating the biotic integrity of rivers. Despite these criticisms, Coeck *et al.* (1993) considered the ABC method to be a useful tool for assessing disturbance in rivers and stated that the method is able to give information both about pollution and physical disturbance before and after river channel works or natural and human induced river restorations. Meire & Dereu (1990) supported this conclusion regarding the sensitivity of the ABC index to pollution and physical disturbances. Part of the problem (i.e. identifying pollution and physical disturbances) may arise because the ABC index was developed mainly for macrobenthic invertebrate communities (Warwick 1986, Warwick *et al.* 1987, Warwick & Ruswahyuni 1987, Meire & Dereu 1990) and has not been thoroughly tested on fish communities. The ABC method also lacks a comparative classification scale, in contrast to the IBI method (Karr 1981), although reference to pristine habitats (reference condition) does allow some comparison to be made. However, it is difficult to compare ABC results with natural habitats as very few pristine sites exits in English rivers.

6.2.4 Multivariate techniques

Multivariate techniques, unlike univariate methods (e.g. diversity indices and ABC method) do not lose data during the simplification of complex data sets (Allen 1999). They include all variables and, it is suggested, give informative results (Krzanowski 1972). Many workers have used multivariate analyses of species data to assess the biological condition of aquatic ecosystems (e.g. Reynoldson & Metcalfe-Smith 1992, Norris 1995, Pan *et al.* 1996). Multivariate techniques allocate species variability to functional interactions but rarely consider the influence of environmental factors (ter Braak & Verdonschot 1995, Pires *et al.* 1999). Consequently, Rexstad *et al.* (1988) and Karr & Chu (1997b) questioned this approach. The UPGMA, TWINSPAN and DECORANA analyses used in this study classified and subsequently grouped sites into poor or rich assemblage categories (Section 4.3.2). The UPGMA and TWINSPAN showed similarity or dissimilarity between sites based on fish abundance and biomass (Section 4.3.2). The DECORANA analysis isolated sites according to rare species and high or low fish abundance (Section 4.3.2). However, it is questionable whether the output adequately reflects perturbations or can be used for assessment of ecological health. Indeed the same arguments raised for diversity indices are relevant for multivariate techniques. Thus conclusions drawn from multivariate methods can be artefacts of the procedures (Rexstad *et al.* 1988). All multivariate analyses are data intensive and these analyses challenge statistical theory and depend on available computer software (Allen 1999). The latter tends to be complex to operate (Fausch *et al.* 1990). In cluster analysis, clusters are formed sometimes regardless of meaningful biological or geographical associations (van Groenewoud 1992). Notwithstanding these objections, these analyses may be helpful to support other diversity indices. Alone they are often less than satisfactory for measuring the biotic integrity of a river.

6.2.5 Index of Biotic Integrity

Having discussed the limitations of the various indices and multivariate methods, it is considered pertinent to compare the outputs with those of the IBI. The IBI incorporates many fish-assemblage attributes that reflect predominant anthropogenic effects on rivers. Each metric has been chosen to describe a particular taxonomic, trophic, reproductive, or tolerance feature of the assemblage (Table 5.2). Other indices and multivariate techniques do not consider these aspects of a fish community and hence those indices and methods are unlikely to provide complete assessments of ecological health of a waterbody. The IBI is very simple to calculate and no complicated statistical analysis or formulae are involved, as required by other indices and multivariate analyses (Sections 4.2.1, 4.2.3 & 4.2.4). Data requirements for the IBI are not too demanding. For example, the method only requires taxonomic information of fish species and total number and weight of individual fish species. Thus the IBI is not data intensive and can potentially be based on semi-quantitative data, provided there are no obvious biases in the collection procedure. Halliwell *et al.* (1999) used semi-quantitative data to develop an IBI, and came to a similar conclusion. The IBI is also useful for interpreting large amounts of data from complex fish communities (Hay *et al.* 1996). In the case of other indices and methods, interpretation of results needs expertise and thus is of lower value to the layman.

From a negative perspective, extensive background information on the fish species (Chapter 3), waterbody (Section 2.3) and data collection processes (Section 2.2) are required to develop an IBI. Therefore, Rankin & Yoder (1999) warned not to apply the IBI "blindly" and without consideration of the ecological principles underlying the metrics and the rich information each contains.

Spearman's rank correlation indicated the IBI outputs were more similar to diversity indices than other measures, as significant relationships were found between IBI & D_{Mg}, IBI & D_{Sm} and IBI & H' at $\alpha = 0.01$ level (Table 5.25). Significant relationships were probably due to the use of fish density and abundance in the models. This significant relationship is probably a numerical similarity and necessarily does not mean that all diversity indices and the IBI are similar in measuring ecological health of a river. Statistical analyses include numerical values, not the attributes of a fish community and hence, significant relationships may exist between variables (e.g. biomass, density). As mentioned previously, the IBI includes all fish community characteristics whereas the diversity indices only handle species richness. No significant relationship was found between the IBI and ABC index, as the ABC index is a ratio of abundance and biomass while the IBI uses absolute values of biomass and abundance separately. The IBI also includes functional aspects of the community, which is not accounted for in the ABC index. In summary, results of this study indicate that the IBI has fewer limitations and makes better use of fisheries survey data. However, the following issues should be addressed when developing and applying a new version of IBI.

- Establishing a reference condition for a new ecoregion;
- Sampling technique, period and time;
- Site selection for data collection;
- Metric selection for IBI development;

- Scoring criteria of selected metrics.

These aspects will be discussed in the following sections.

6.3 ESTABLISHMENT OF THE REFERENCE CONDITION

Establishing reference criteria is vital for developing an IBI for a new ecoregion. Data from pristine or near pristine sites are the preferred option for establishing reference conditions. Where this is not available both historical and the best available data on fish fauna and habitat may be used. For the middle and lowland reaches of rivers of England used in this study, no pristine sites were considered available. Consequently, best available data were used. The major disadvantage of best available data is inappropriateness, when data are collected from sites that have been strongly affected by anthropogenic activities. Using severely affected sites to establish reference conditions is clearly inappropriate but the data from these sites, in conjunction with data from "good" habitats, are important to elucidate the range of values for scoring a metric. Historical data, on the other hand, may allow identification of pristine conditions or near pristine conditions, but using these data for reconstructing reference conditions rarely allows estimation of natural variability. Moreover, historical data are subject to unknown sources of bias, for example, when methodologies, gears and sampling techniques are poorly documented or methods potentially inefficient.

Notwithstanding the arguments about reference conditions being based on pristine sites, a reference condition may be defined as a known state against which change can be measured (Caddy and Mahon 1995). Consequently, the use of best available condition becomes acceptable. By referencing the best available conditions for all metrics it is possible to measure deviation caused by ecological and environmental changes. The degree of deviation is thus a measure of degradation and any change in deviation is a measure of deterioration or improvement.

The process adopted in this study to establish the reference condition (Table 3.11) follows Fausch *et al.* (1984), Leonard & Orth (1986), Fausch (1987), Steedman (1988), Oberdorff & Hughes (1992), Goldstein *et al.* (1994), Hughes (1995), Hugueny *et al.* (1996), Koizumi & Matsumiya (1997), Didier (1997), Ganasan & Hughes (1998), Hughes *et al.* (1998), Mundahl & Simon (1999), Niemela *et al.* (1999), Boet *et al.* (1999), Schleiger (2000), Lyons *et al.* (2000) and Kestemont *et al.* (2000), who developed reference conditions based on local stream size, region and fish fauna, and information gathered from consultation with local resource managers, historical fish collections by various departments and by on-site reconnaissance.

Reference conditions need to be related to the zoogeography of a region or country as species diversity varies with regions. For example, in the UK, rivers draining to the south coast have higher species diversity than the rivers draining to the west coast (Varley 1967). In addition rivers of the south coast are dominated by cyprinids, while salmonids are dominant in the rivers of the west coast (Varley 1967). Although all study rivers are draining to the East coast of England little variation in the fish species diversity was found between the different regions (i.e. the Thames, Midlands and Northeast) (Table 6.2). Hence, it is desirable to develop regional reference conditions, especially for salmonid-dominated rivers (west coast), which are likely to differ from the study rivers.

Table 6.2. Variations in fish species richness and IBI scores with zoogeography of study rivers. Parentheses indicate number of species measured and used for IBI calculations

River	Region	Total number of Fish species	Range of number of fish species used in IBI in each region	Mean IBI score	Range of IBI score
Cherwell	Thames	14 (10)		35	
Evenlode	Thames	19 (14)		41	
Stort	Thames	16 (12)	10 - 14	35	35 - 45
Thame	Thames	18 (13)		45	
Windrush	Thames	16 (11)		41	
Anker	Midlands	16 (13)		43	
Blithe	Midlands	13 (9)		39	
Blythe	Midlands	16 (12)		37	
Churnet	Midlands	16 (11)		27	
Cole	Midlands	15 (11)		23	
Derwent	Midlands	16 (16)		37	
Idle	Midlands	9 (9)	7 - 16	25	23 - 44
Mease	Midlands	14 (11)		43	
Penk	Midlands	14 (10)		29	
Sence	Midlands	14 (10)		36	
Soar	Midlands	14 (10)		37	
Sow	Midlands	14 (10)		38	
Tame	Midlands	18 (14)		44	
Tean	Midlands	11 (7)		24	
Trent	Midlands	16 (12)		29	
Aire	Northeast	14 (9)	9 - 13	25	25 – 35
Nidd	Northeast	17 (13)		35	

Fish species diversity also varies with different zones of a river (Cowx 2001) and usually "trout" and "grayling" zones have lower species diversity than "barbel" and "bream" zones (Varley 1967). Such variations were found in the study rivers (Section 4.3.2). Therefore, reference condition should be related to Huet's (1949) zonation patterns. As the current IBI is aimed specifically at middle and lowland reaches, headwater species (salmon, brown trout and brook lamprey) and minor species (stone loach, spined loach, bullhead, minnow, 3-spined stickleback and 10-spined stickleback) were excluded (Section 3.3.2). However, the IBI developed in this study was applied to headwaters of some study rivers with the reference conditions developed for lowland reaches and produced low IBIs for upper reaches (e.g. Appendix 5.3, Rivers Churnet, Cole, Aire). Consequently, it was confirmed that IBI developed for the middle and lowland reaches was not appropriate for headwaters. However, with modification, the IBI can be adapted to headwaters, but this must start with the development of a new reference condition. For a headwaters IBI, possible metrics such as "density of salmonids", "percentage of fry and parr of salmon and trout", "number of headwater species" and "number of minor species" should be included but again these must be linked to zoogeographical characteristics. Most English minor species live in upper reaches of rivers, are intolerant of poor water quality (e.g. minnow, bullhead) and habitat degradation (e.g. stone loach) (Hawkes 1975). Therefore, presence of minor species should identify moderate to high quality sites. In this study, a number of minor species were recorded but were not included in the IBI calculations due to the metrics used. This is probably one of the main reasons for the low IBI scores in upper reaches of some rivers (Figs 5.1a, d & e). Furthermore, data on minor species were inappropriate for IBI calculations as the abundance of minor species were assessed subjectively (Section 2.2.3). Schlosser (1985) and Angermeier & Karr (1986) recommended excluding minor species from IBI calculations to reduce sampling costs and time. For a headwater IBI, it is likely that several metrics, e.g. "percentage of individuals preferring vegetated areas" should be deleted, as they are not relevant to the zone.

Otherwise, in principle the procedure adopted in this study for developing an IBI for middle and lowland reaches of river should be followed.

6.4 SAMPLING TECHNIQUE, PERIOD, TIME AND FREQUENCY

6.4.1 Sampling technique

A representative and appropriate sampling technique is crucial to obtain the data required for any biological assessment. A number of survey methodologies are available for sampling rivers (Cowx 1995) and most methodologies have distinct advantages and disadvantages (Harvey 1996). Data used in this study were collected exclusively by electric fishing. The electric fishing technique provides detailed information on community structure and population characteristics (Cowx 1990a), which are the main attributes of IBI metrics. However, the technique tends to be selective against fish <80 mm (Junge & Libosvarsky 1965, Cowx 1990a). Size selectivity did not appear to affect the IBI developed in this study as fish <80 mm were not specifically included as a metric (Table 5.2). Generally, efficiency of electric fishing gear gets progressively poorer with depth, especially depths >3 m (Harvey 1996). Depth was not considered a significant contributory factor in the present study because most sites were <3 m deep (Appendix 2.1).

Site characteristics (e.g. length, depth & width) are important factors for consideration when applying a sampling technique to collect representative fish samples. The length of the sample reach should be long enough to include all major habitat types, for example, riffles, pools and backwater areas (USEPA 1991, Lyons 1992, Niemela *et al.* 1999, Yoder & Smith 1999 and Smogor & Angermeier 1999 & 2001). The majority of selected sites for this study included major habitat types available in the river reach and this was reflected in the variable length, width and depth of sites. Fish diversity and abundance vary with zone of the river (Huet 1949); region (e.g. East Coast or West Coast) (Varley 1967); geographical location (e.g. temperate & tropical) (Fausch *et al.* 1984) and intensity of anthropogenic disturbances (Cowx 2001). Hence, minimum length of river to obtain a representative sample needs adjustment accordingly. Sampling 150 m reaches was found suitable for coldwater streams in the Upper Midwestern United States (Mundahl & Simon 1999), while 0.5 km long reaches were chosen for Indian rivers (Ganasan & Hughes 1998). Karr (1981) suggested sampling 100 m reaches in small streams / rivers and 1 km reaches in large rivers where electric fishing is employed. Angermeier & Karr (1986), Angermeier & Smogor (1995), Simoson & Lyons (1995) and Yoder & Smith (1999) recommended sampling a length, which is at least 20 times greater than the width. For the UK, Cowx (1995) and CEN (2001) described the absolute minimum length of sampling sites to obtain specific data (e.g. fish species composition, abundance and age structure of a given site) under certain conditions (e.g. sampling by electric fishing gear in daylight hours, small rivers by wading, large rivers by boat). These dimension were 20 m in small streams rising to 20 times the river width in large rivers. From the discussion, it is appeared that length of sampling sites used in this study (range: 50 - 1045 m) was appropriate to collect representative fisheries survey data, which is in line with the description of length of sampling sites described by Cowx (1995) and CEN (2001). Therefore, for a minimum length of a sampling site, it is recommended that the criteria prescribed by Cowx (1995) and CEN (2001) (Table 6.3) for various waters in Europe be followed. This will at least standardise the effort and reduce biases associated with sampling too small a length of river. However, consideration should also be given to the problems of access and resources, which may influence the length of river that can be sampled.

Table 6.3 Minimum dimension of a site to be sampled to collect representative fish data in European waterbodies (after Cowx 1995 and CEN 2001)

Dimension of waterbody	Minimum length to be sampled
Small stream, width <5 m	20 m
Small river, width 5-15 m, depth <1 m Pool - pool/riffle topography	50 m
Large river and canal, width >15 m, depth >1 m	>50 m of river margin either on one side or on both sides
Large shallow water, depth <70 cm	Area: 200 m^2
Large water bodies (e.g. lakes)	>50 m of littoral zone

Usually fish sampling by electric fishing is effective in small rivers and streams (Hickley & Starkie 1985). However, in medium sized rivers and tributaries, electric fishing may not be effective due to strong water currents (e.g. electro-shocked fish may be swept away), increased depth & width or excessive weed growth (e.g. fish may hide during electric fishing). Hence, Nielsen & Johnson (1983) came to the conclusion that use of one sampling technique may cause bias in the assessment. Therefore, when appropriate, a combination of techniques should be employed according to particular conditions of a river to ensure the best possible assessment is made. Having said this, however, a combination of sampling techniques may still not be effective in small rivers and streams, which have strong current or thick vegetation growth. For example, seine netting may also not be effective in strong currents as the net cannot be set while in a thick vegetation growth area setting and pulling of a seine net may be difficult.

Sampling fishes in large rivers (e.g. River Severn, lower Trent, Thames and Great Ouse) for IBI data acquisition may be problematic as large rivers have more complex habitats than small rivers and a single sampling method (e.g. electric fishing) may not adequately characterise the entire fish assemblage (Lyons *et al.* 2001). For example, surveys at a single site should include all habitats, which rarely occurs as efforts are usually concentrated in the main river channel, and it should be recognised that backwaters, floodplain lakes etc. are all part of the river (Lyons *et al.* 2001). Different habitat types therefore, may require different sampling methods. Due to these problems, data collected from the large rivers in this study which chose to use only electric fishing survey results may not be truly representative of the fish communities and accordingly the IBI scores may be lower than expected (Table 5.23). To overcome these problems multiple sampling methods should be used to assess the overall assemblage, as multigear techniques produce less biased samples of the fish community and reduce selectivity (CEN 2001). However, use of multiple gears is often not possible. Use of multiple gears is time consuming, labour intensive and costly. Some gears (e.g. seine net) may not be suitable for large lowland rivers, especially in fast-flow conditions. Moreover, data collected by different gears have different selectivities and it can be difficult to aggregate data (Simon & Lyons 1995). Consequently, Jennings *et al.* (1999) recommended developing method-specific metric values. For example, electric fishing gear is more effective for sampling mid-water and surface-water dwelling species (e.g. for tolerant, intolerant, rheophilic species metrics) than fyke or seine nets, which tend to be more efficient for benthic species (e.g. for benthic species metric) (Jennings *et al.* 1999). The Young of the Year (YOY) are often best sampled by micro-meshed seine nets (e.g. for percentage of YOY metric if developed to assess recruitment success (Section 6.6.2)), which are more efficient than using electric fishing. Species richness (e.g. for number of native and percentage of non-native species metrics) and relative abundance (e.g. for numbers in a sample and total biomass metrics) are typically less for gillnets than electric fishing as gillnets are selective with respect to fish size (Simon & Sanders 1999). In essence the method-specific metric values will allow incorporation of the advantages of a multiple-method sampling regime without adding the complication of combining results from different methods. Against these arguments, Ganasan & Hughes (1998) collected fish assemblage data using multiple gears to develop an IBI for Indian rivers. Riverine fisheries of India are multispecies and multigear in nature and thus they were obliged to use multigear techniques to obtain representative samples. The IBI developed by Ganasan & Hughes (1998) was considered appropriate for assessing the fish communities and thus combining data from multigear techniques should be possible. Therefore, it is recommended to test the multigear approach in UK rivers.

The sampling methodology used in this study was considered acceptable, since precise population estimates were not required for IBI calculations with the metrics chosen. Indeed, Simon & Sanders (1999) considered estimation of biotic integrity does not require intensive sampling, as this would increase resource expenditure to collect every last species that may occur at a site. Since rare species generally add very little to the total IBI score, the failure to collect a few rare species at a site will not detract from the assessment of biotic integrity (Yoder and Rankin 1995). In summary, a multigear approach may be preferable but, as mentioned above, many considerations make this difficult. The IBI developed in this study indicated a single gear approach was acceptable and in principle this approach may be adopted. However, it is recommended that single gear approach for a river should be compared with a miltigear approach in future studies when developing IBIs for other rivers.

6.4.2 Sampling period

Sampling efficiency for fishery data collection also depends on selecting the most appropriate period of the year. Fish species diversity varies with season (i.e. summer, winter) and time (i.e. day, night) (Cowx 2001). Therefore, sampling should be linked to the life-history strategies (e.g. spawning migration in spring and summer, feeding migrations, effects of lunar cycle) when most fish species are easier to capture. Species richness tends to appear higher later in summer and early autumn due to increased chance of catching YOY of rare species. These may not necessarily be captured when sampling in early summer due to size selectivity of the gear as the fish are too small in size. Unfortunately it is difficult to define the optimum time for sampling because of environmental variables and conflicting requirements which often override the time of sampling. However, periods of low to moderate stream flow are recommended and relatively variable flow conditions of winter should be avoided (Karr *et al.* 1986). Karr (1981) suggested collecting data several times each year but this increased cost may not be justified.

In this study, sites were sampled both in summer and winter periods (Table 2.2). No obvious seasonal effect was found. This may be due to reasonably efficient sampling in the rivers or lack of discrete seasonal variation in community structure in the rivers studied. Therefore, it is recommended that fish populations in English rivers are sampled in the summer and early autumn, when water temperatures are higher, fish are active and flows do not prevent efficient sampling.

6.4.3 Diurnal sampling time

Data used in this study were collected in daylight hours and seem appropriate for calculating IBIs for the respective rivers. Lyons *et al.* (2001) also collected fish assemblage data in the daytime and developed an IBI for Wisconsin's large warmwater rivers but stated that night electric fishing yields more fish species and greater biomass than day electric fishing. Sanders (1991) found night catches contained significantly more species, higher numbers and weights of fish, and were compositionally more evenly distributed than day catches ($P > 0.05$). Simon & Sanders (1999) also emphasised night sampling, as they believe that night electric fishing is the best way to collect representatives of all species. Harvey (1996) also found a markedly different fish community structure, mainly in terms of the size of fish caught and species composition when electric fishing at dusk on the Yorkshire Ouse compared with daylight hours fishing. Based on these data it is possible the IBI would change if night-time sampling was adopted. However, there are many logistical and safety concerns associated with working at night. Management agencies are unlikely to adopt night time sampling for routine monitoring because of the dangers of working on rivers at night and the higher cost that will possibly be incurred. Therefore, it is recommended that the daytime sampling adopted by the EA is continued for collection of fishery survey data to calculate the IBI for English rivers. This will also allow comparison with historical data sets and elucidate any trends in the status of the fisheries. However, to identify possible variation in the IBI, data collected during the night time may be tested on English lowland rivers in future studies.

6.4.4 Sampling frequency

In this study, "single or one-time" samples, i.e. one sample was collected from a site, were used to calculate the IBI. This process of sample collection was supported by Fausch *et al.* (1984), Angermeier & Karr (1986) and Lyons *et al.* (1996) who used "one-time" samples to calculate IBIs. Combining several samples collected by different gears at different times can potentially produce substantial errors in calculating species diversity, abundance and produce an inappropriate IBI. Multiple samples collected over an extended time frame should be treated independently and used to depict change in the status of the fishery. Results of this study suggest that single-sweep electric fishing is acceptable if carried out with rigour (Section 6.4), although quantitative sampling, if the time and resources are available, is preferable.

6.5 SITE SELECTION

An appropriate choice of sampling sites is critical to any survey and for successful application of an IBI. The sites chosen for sampling should be representative of the overall habitat of the river (Lyons 1992).

6.5.1 Historical and new sites

In this study, fisheries survey data were collected from sites historically selected for routine monitoring. The data were found appropriate to calculate IBIs for respective rivers as they contained basic information such as number of fish species and total number and biomass of individual fish species. The EA, and its predecessor the NRA, selected the sites for various purposes, such as water quality monitoring, habitat degradation or impact assessment. Consequently, all the different habitats of a river may not be included in the samples. Results of this study (Appendices 2.1 & 5.3) suggested that impacts on the site affect on the IBI score. As expected, sites around STWs / WRWs or immediately below a STWs / WRWs, produced low, or even zero IBI scores (Fig. 5.1a, River Cherwell, site 5, IBI score 22) while above and for a short distance below the STWs, IBI scores were higher (Fig. 5.1a, River Cherwell, sites 3 & 11, IBI score 49 & 47, respectively). The former situation was probably due to immediate effects of effluent discharges, which were detrimental to the fish assemblage while the latter was due to dilution of effluents and ecosystem recovery processes. However, exceptions were found, e.g. in the River Windrush where a high IBI was found between STWs (Fig. 5.1e, site 19, IBI score 47). The latter was probably due to higher quality discharge from the STWs in this reach. This variation shows the efficacy of the IBI to detect change in ecosystem health under different degrees of intensity of degradation.

Variations in the IBI score were also found at sites around physical barriers such as weirs, gauging stations, pumping stations, power stations, locks and bridges. Physical barriers can disrupt migration patterns of fishes resulting in low species diversity and abundance at a site, and ultimately produce a low IBI (e.g. site 6, River Cherwell; site 4, River Stort; site 2, River Blithe; site 3 River Tean, site 11, River Aire, site 33, River Nidd). Variations in the IBI values around physical barriers indicate the need to include such sites in IBI assessment. The IBIs also fluctuated at sites around river confluences and immediately below and above reservoirs (e.g. River Cherwell, sites 9 & 10, IBI scores 38 and 39, respectively) (Appendices 2.1 & 5.3). River confluences usually have higher species diversity and abundance mainly due to greater heterogeneity of habitat and higher quality water than other sections of a river thus producing higher IBI scores. Low IBIs were found at sites below waterfalls (e.g. River Aire sites 2 & 3, IBI scores 22 and 23 respectively) (Appendices 2.1 & 5.3), while the highest IBI scores for the River Aire were found at sites above Snaygill STW (sites 14 & 15, IBI scores 41 & 41, respectively). Low IBI scores at sites below waterfalls may be due to low species diversity and abundance associated with the harsh environmental conditions found at these locations (e.g. turbulence, strong current, deep pools, erosion and turbidity) or problems with sampling such different habitats. These findings were supported by Hugueny *et al.* (1996), Koizumi & Matsmiya (1997) and Ganasan & Hughes (1998), who found variable IBI values in sites with such barriers.

As the weirs, gauging stations, pumping stations, power stations, locks and bridges are an integral part of management of English rivers, and it is not possible to remove such barriers, it is logical to include these sites to assess the overall integrity of a river. It is recommended that when setting up a new monitoring regime all habitat types are represented and all anthropogenic disturbances are included. However, for the warmwater streams of Wisconsin, USA, Lyons (1992) recommended that sampling areas should not normally include bridges, dams, mouths of the tributaries, or other atypical habitat features, since fish assemblages in the vicinity of such features are often not representative of the overall fish community of a stream reach.

6.5.2 Number of sampling sites

Although the IBI works at a site level, as each site produces an independent IBI score (Appendix 5.3) which reflects the anthropogenic disturbance (Section 4.3.2), the question that needs to be answered is what is the minimum number of sites that must be surveyed on a river or reach to give an indicative score for the river or reach, i.e. mean IBI, for the river or reach as a whole. The number of sampling sites on a river varied between 5 (e.g. River Idle) and 182 (e.g. River Nidd) (Table 2.1). For the number of sites needed to give a true reflection of the river or reach, the IBI must account for the variability between sampling sites brought about by natural geomorphological and hydrological conditions, and the effects of human disturbance. Sufficient number of sites must be sampled so the mean IBI score stays at least within the class boundary which is indicative of the river or reach, and deviates little when sites are added. Because each river has its own inherent variability the analysis should be based on a number of rivers to account for this change. To determine the number of sites needed to stabilise the IBI score in a river, figures were produced by plotting cumulative average IBI scores against randomly selected increasing numbers of sites for a number of rivers (Figs 6.1a-e, 6.2a-o & 6.3a-b). Using a number of rivers rather than repeating the random selection several times for one river was deemed the best approach because the latter analysis would be river specific and the former a more generalised output.

In most study rivers, the IBI score stabilised within a certain integrity class boundary with 10 and 20 sites (Figs 6.1a-e, 6.2a-o & 6.3a-b). For example, a stable IBI score within the "Good" integrity class was obtained after 9 sites on the River Thame (Fig. 6.1d) while a comparatively stable IBI score within the "Poor" integrity class was found after 16 sites on the River Aire (Fig. 6.3a). In many rivers (e.g. Rivers Evenlode, Stort, Derwent, Penk) the mean IBI score did not stabilise until the total number of sites sampled on that river had been included. This shows that natural variability is high and a large number of sites may be necessary to give a true reflection of the biotic integrity of these rivers. This was exemplified for the River Nidd (Fig. 6.3b) where stability of the IBI score was not achieved until about 81 sites had been included. Part of this problem arises because the River Nidd was sampled entirely and large scale natural variability between downstream and middle reaches are inherent within the data, coupled with the extensive habitat degradation that exists throughout this catchment (Section 2.3.3). A better strategy for setting IBIs for large rivers is to divide them into reaches, probably based around Huet's zonation patterns (Fig. 3.1) to address some of the natural variability.

In view of the above arguments, the number of sampling sites on a river should be related to the length, width and depth of the river, habitat types, and intensity of perturbations. Notwithstanding the problems relating to predicting the number of sampling sites to give an accurate representation of the river, it appears that a minimum of between 10 and 20 sites are needed to stabilise the IBI within the class boundary (Figs 6.1a-e, 6.2a-o & 6.3a-b). As the key issue is accurate representation of the status of the river this should be adequate to satisfy the needs of the Water Framework Directive. However, 10 to 20 sampling sites may not be representative of long, wide and deep rivers such as the Severn, Trent, Thames or Great Ouse.

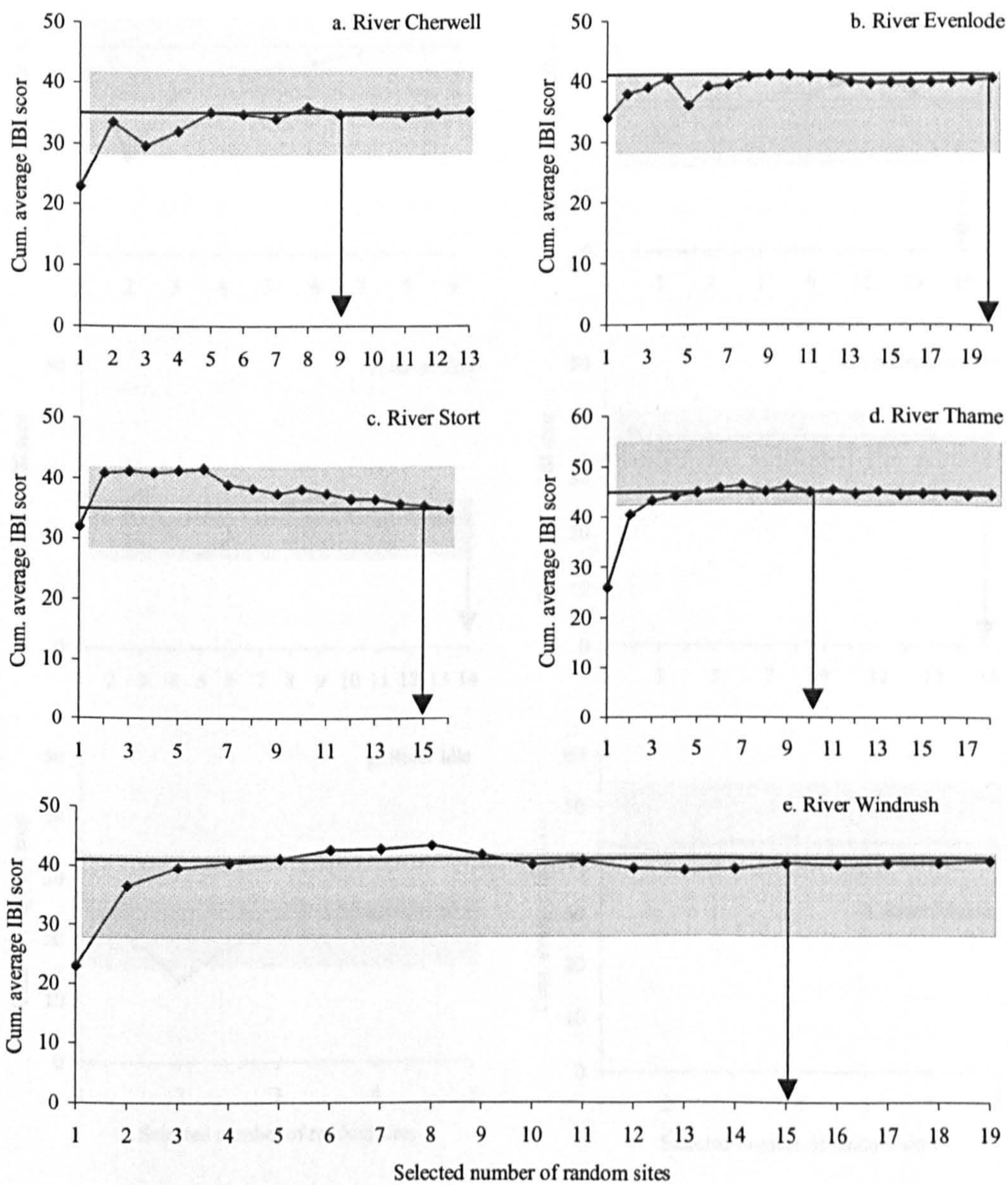


Fig. 6.1 (a-e) Number of selected random sites for stabilising IBI score in the Thames catchment (shaded area indicates the range value of integrity class while the arrow indicates stabilising point)

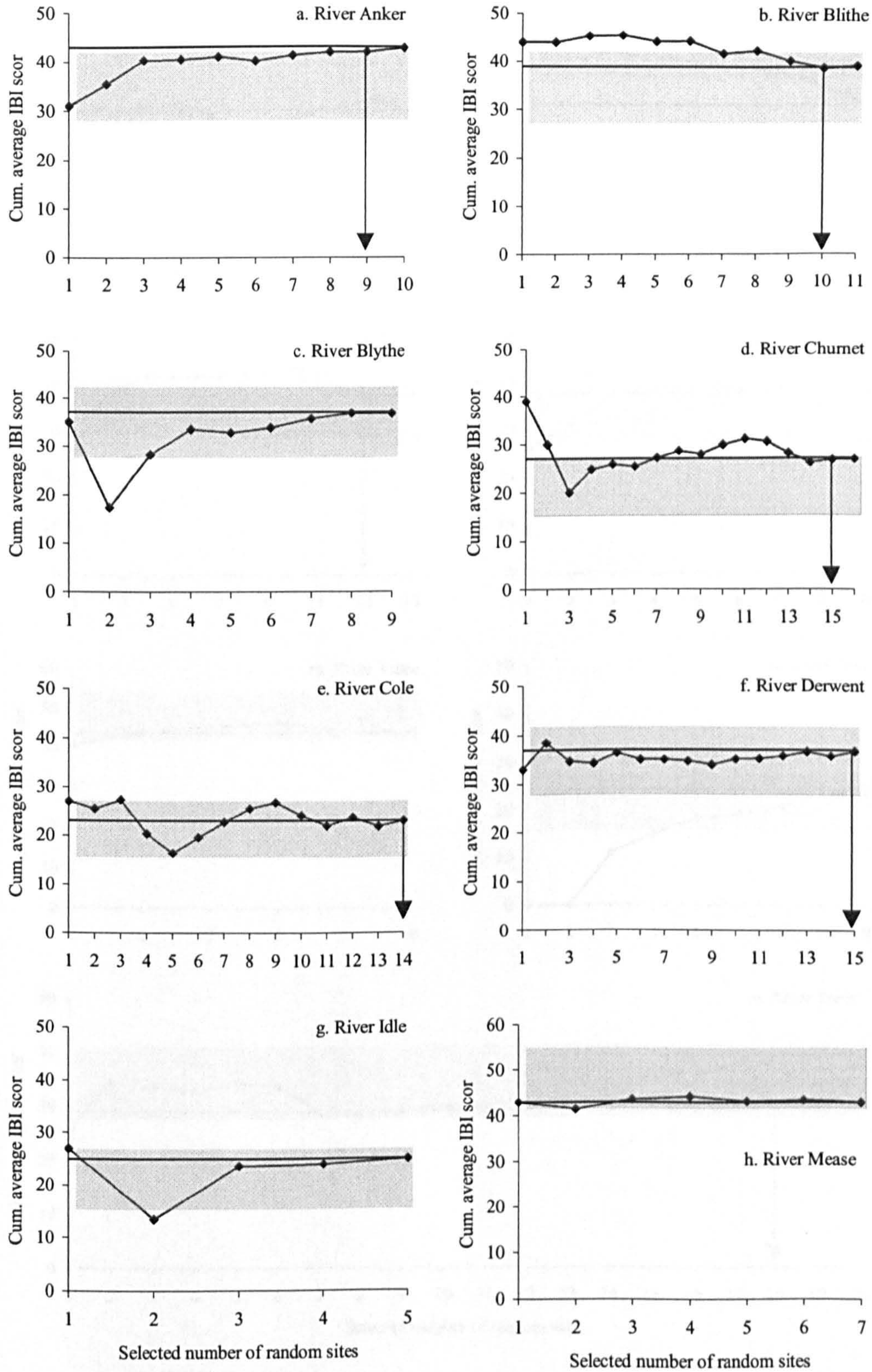


Fig. 6.2 (a-o) Number of selected random sites for stabilising IBI score in the Trent catchment (shaded area indicates the range value of integrity class while the arrow indicates stabilising point)

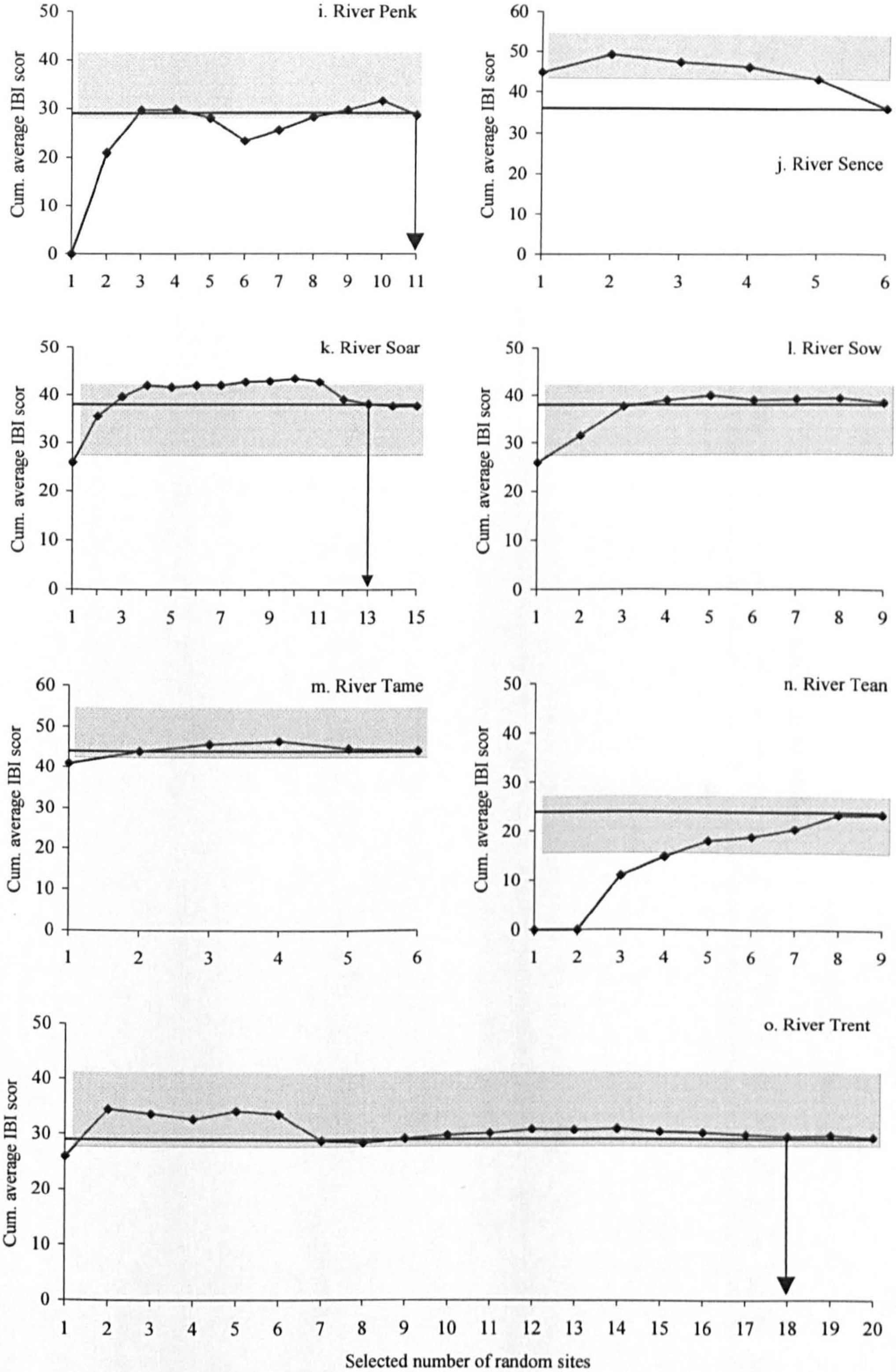


Fig. 6.2 (a-o)(Continued) Number of randomly selected sites for stabilising IBI score in the Trent catchment (shaded area indicates the range value of integrity class while the arrow indicates stabilising point)

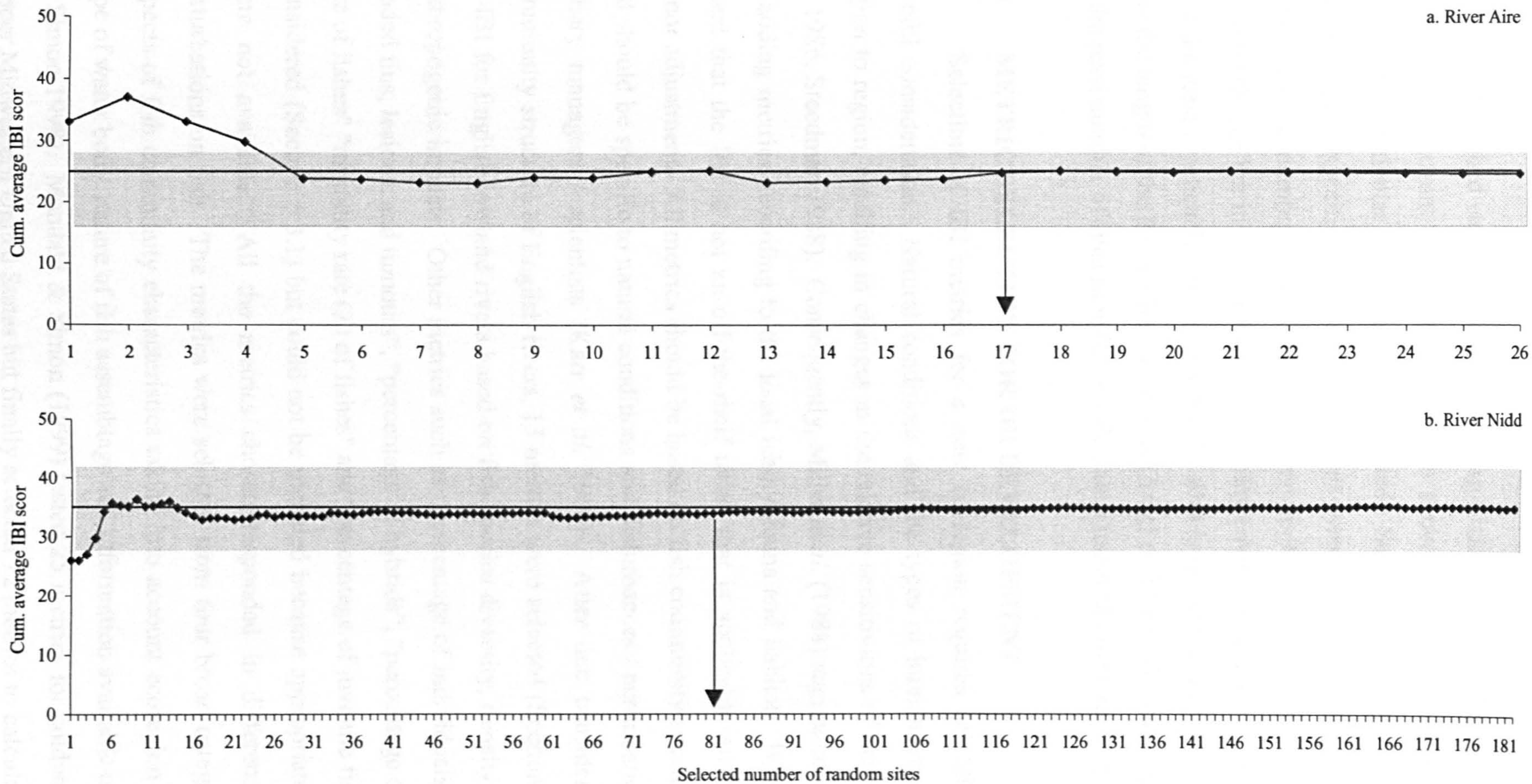


Fig. 6.3 (a-b) Number of randomly selected sites for stabilising IBI score in the Yorkshire Ouse catchment (shaded area indicates the range value of integrity class while the arrow indicates stabilising point)

As described earlier (Section 6.3), long rivers should be divided into several zones or reaches to account for zonation and then a number of representative sites (10 - 20) specific to a particular zone should be selected. Samples collected from river zones will lead to a zone-specific IBI. One important issue that needs to be considered when determining the number of sites is the resource implications. In an ideal world many sites would be chosen but in a financially constraining environment the number of sites will have to be the least practicable. It appears that 10 sites is the minimum acceptable number for a river the length of the River Cherwell (96 km long) or Thame (77 km long), and this should be the least number of sites sampled to calculate a mean IBI for a river or reach.

6.6 METRIC SELECTION FOR IBI DEVELOPMENT

Selection of IBI metrics for a new ecoregion requires critical observation and careful consideration. Natural conditions and the types of human impacts change from region to region, resulting in changes in the relative sensitivities of many metrics (Karr *et al.* 1986, Steedman 1988). Consequently, Miller *et al.* (1988) suggested replacing, deleting or adding metrics according to the local ichthyofauna and habitat. Whittier *et al.* (2001) stated that the IBI is not an off-the-shelf index that is applicable everywhere with only minor adjustment. All metrics should be based on fish community specific to an ecoregion and should be specific to natural conditions and disturbances / perturbations noticed by the fishery managers / scientists (Karr *et al.* 1986). After due consideration of the fish community structure of English rivers, 15 metrics were selected (Section 5.3.1) to develop an IBI for English lowland rivers based on fish species diversity, density and biomass, and anthropogenic impacts. Other metrics such as "percentage of individuals with deformities, eroded fins, lesions, and tumours", "percentage of hybrids", "percentage of standard growth rate of fishes" "mortality rate (Z) of fishes" and "percentage of juvenile fishes (YOY)" were considered (Section 5.3.1) but could not be included because appropriate or adequate data were not available. All the metrics chosen responded in different ways to human perturbations on fish. The metrics were selected from four broad categories, covering all aspects of fish community characteristics taking into account ecoregion faunal variability, type of water body, nature of fish assemblage and information available on fishes (Mundahl & Simon 1999). Mundahl & Simon (1999) tested 25 metrics for coldwater streams in the Upper Midwestern United States but finally selected 12 metrics to calculate IBI. Hughes *et al.* (1998) tested 16 metrics and selected 13 for IBI calculations. Biggs *et al.* (1998)

proposed 20 potential fish metrics divided into four broad classes, to evaluate still water integrity in the UK. Among them, "trout or pike year classes", "number of minor species", "number of threatened species", "total biomass of coarse fish", "abundance/biomass of eels", "biomass of tolerant/intolerant species", "density of salmonids", "growth rate" and "condition index" were new additions as IBI metrics. Whittier *et al.* (2001) considered as many as 100 candidate metrics to develop an IBI for Florida lakes. From this study (Chapter 5) it appears development of an IBI should be based on choosing a number of metrics according to local/regional habitat and fish fauna, which are sufficient to give the desired predictive response of the fishery status.

6.6.1 Validation of selected metrics

All metrics finally selected for the IBI were considered sensitive to particular perturbations and were used to calculate an IBI for English lowland rivers (Appendix 5.3). This was based on determining the relationships between individual metrics and total IBI score (Fig 6.4a-o). Seventy randomly selected sites from a total of 457 sites (Table 2.1) were used to establish the relationship. The "total number of native fish species" was positively correlated (r^2 = 0.85) with the IBI score and was therefore a key metric (Fig. 6.4a). Fore *et al.* (1994), Oberdorff & Porcher (1994), Didier *et al.* (1996) and Kestemont *et al.* (2000) came to a similar conclusion when determining statistical properties of IBIs for Ohio streams and European rivers, respectively. The metric followed the general consequences of habitat degradation (Karr *et al.* 1986) as the number of fish species was less in highly degraded sites and consequently produced a lower IBI score (Figs 4.5 & 5.1a, River Cherwell, sites 1 & 8). It therefore, appears that this metric can be applied universally to all types of habitat irrespective of geographical location. Therefore, use of this metric is recommended for the development of IBI for other English rivers.

"Percentage of individuals as non-natives" showed weak correlation with the IBI (Fig. 6.4b). This is because the number of sites with "introduced" species was low (5 sites). Although not a powerful metric in the current study it is considered of particular importance because it infers "biological" pollution or fish community degradation. Furthermore, non-native species tend to be more resistant to human impacts (Courtenay & Moyle 1992) and they tend to dominate in degraded systems where the ecological function has been disrupted allowing them to proliferate (Cowx 2002).

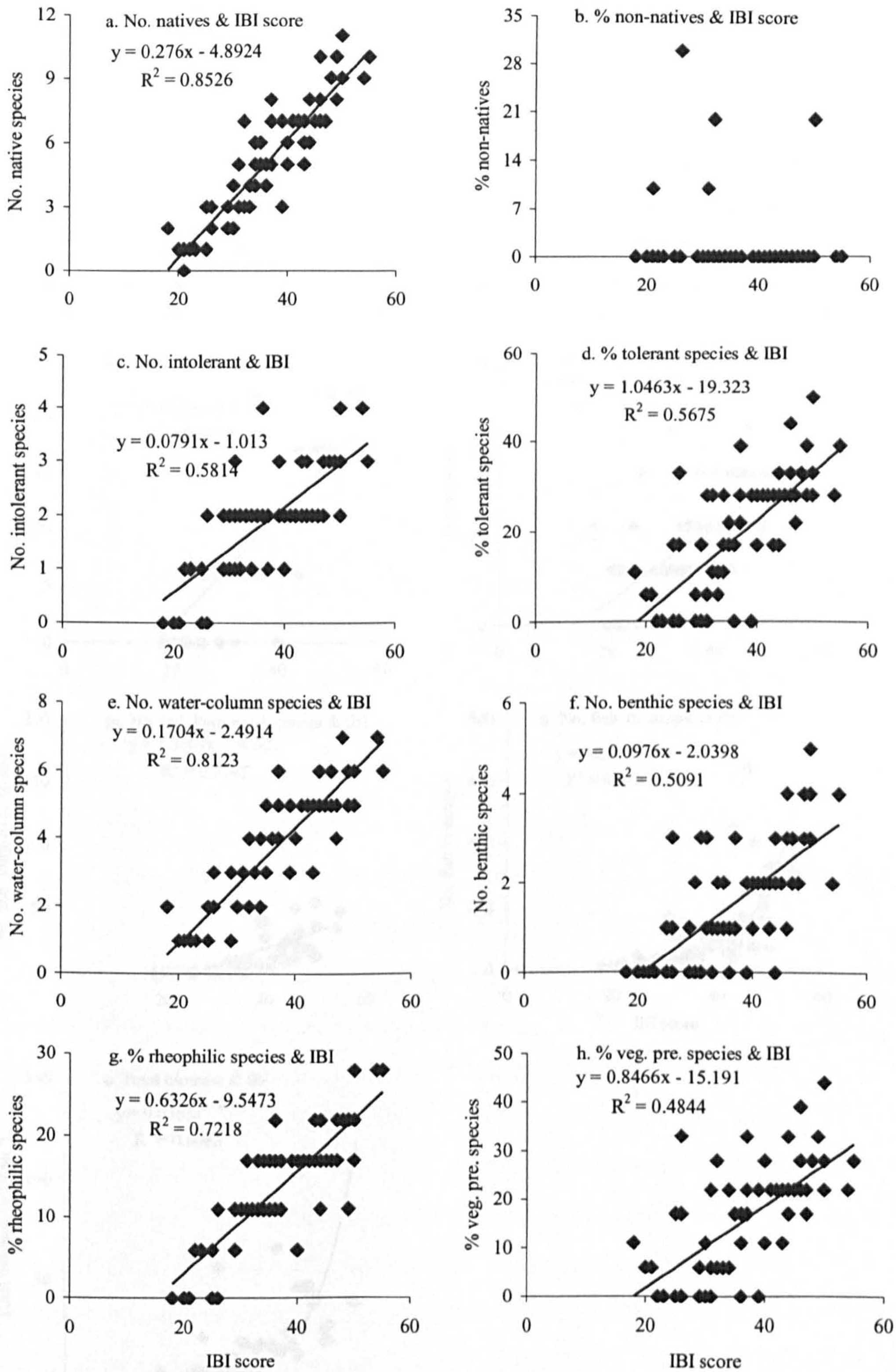


Fig. 6.4(a-o) Correlation between metrics and IBI scores. (% veg. pre. species = % vegetation preferring species)

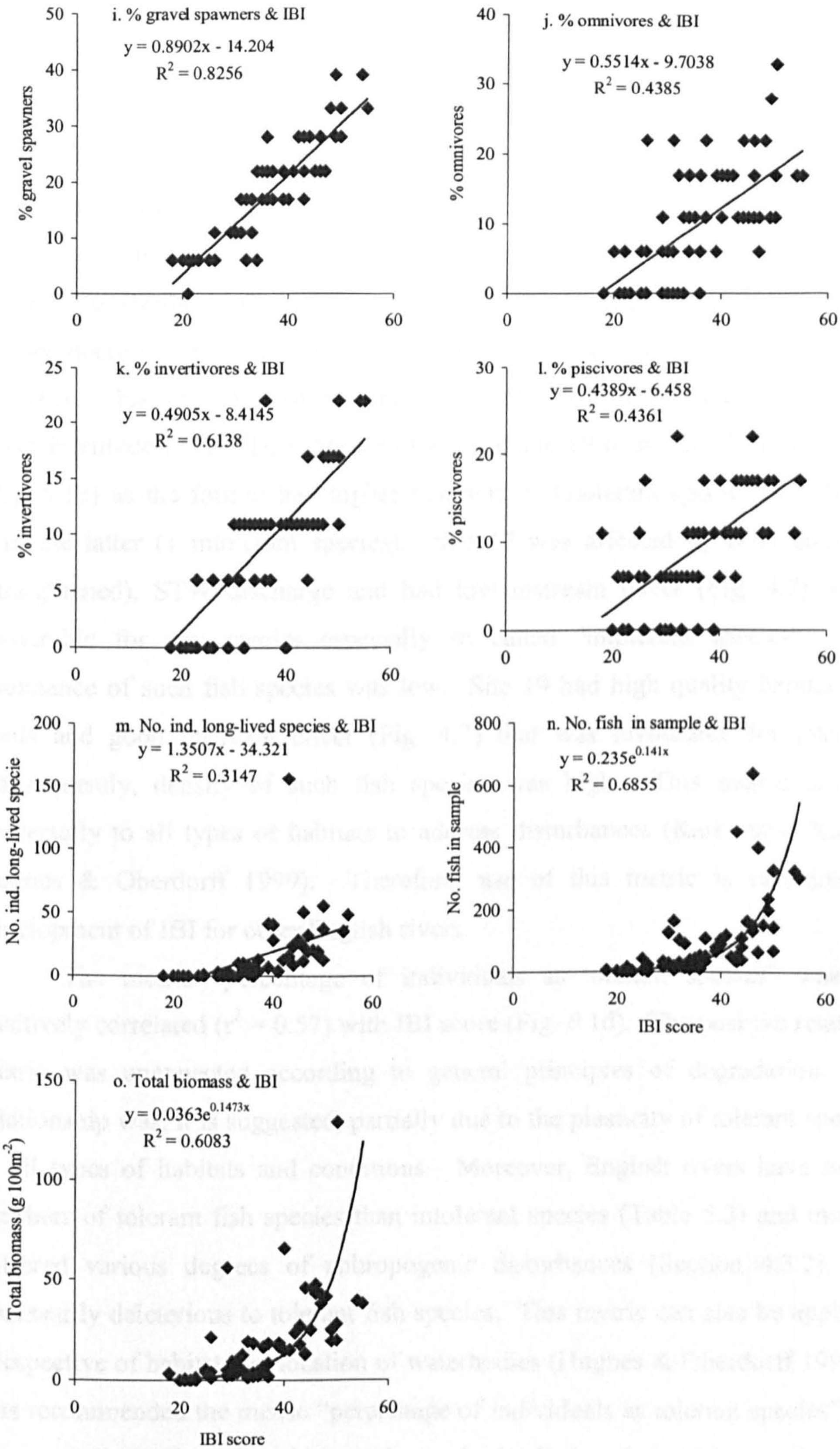


Fig. 6.4(a-o)(Continued) Correlation between metrics and IBI scores. (No. ind. long-lived species = Number of individuals of long-lived species)

These species generally disrupt biotic integrity (Karr & Dudley 1981). A considerable number of exotic fish species have been introduced in the UK (Table 3.2) (Cowx 1997), therefore, it is recommended the metric "percentage of individuals as non-natives" be retained in future studies.

"Number of intolerant species" was positively correlated ($r^2 = 0.58$) with the IBI score (Fig. 6.4c). The correlation indicates that the metric influenced the IBI score. This finding was similar to that of Fore *et al.* (1994). Generally the number of intolerant fish species decreases with habitat degradation (Karr *et al.* 1986) and degraded sites produce low IBIs. This was reflected in some of the IBI scores of this study (e.g. Figs 4.7 & 5.1b, River Evenlode). The IBI score was higher at site 19 than site 17 on the River Evenlode (Fig. 5.1b) as the former had higher numbers of intolerant species (4 intolerant species) than the latter (1 intolerant species). Site 17 was affected by river engineering works (straightened), STW discharge and had low instream cover (Fig. 4.7), which was not favourable for any species especially so called "intolerant species". Consequently, abundance of such fish species was low. Site 19 had high quality habitat having riffles, pools and good instream cover (Fig. 4.7) that was favourable for intolerant species. Consequently, density of such fish species was high. This metric could be applied universally to all types of habitats to address disturbances (Karr 1981, Karr *et al.* 1986, Hughes & Oberdorff 1999). Therefore, use of this metric is recommended for the development of IBI for other English rivers.

The metric "percentage of individuals as tolerant species" was found to be positively correlated ($r^2 = 0.57$) with IBI score (Fig. 6.1d). The positive relationship of this metric was unexpected according to general principles of degradation. The positive relationship was, it is suggested, partially due to the plasticity of tolerant species to survive in all types of habitats and conditions. Moreover, English rivers have naturally higher numbers of tolerant fish species than intolerant species (Table 5.3) and most study rivers suffered various degrees of anhropogenic disturbances (Section 4.3.2) that were not necessarily deleterious to tolerant fish species. This metric can also be applied universally irrespective of habitat and location of waterbodies (Hughes & Oberdorff 1999). Therefore, it is recommended the metric "percentage of individuals as tolerant species" be retained in future studies. However, this metric needs detailed and careful investigation to classify tolerant species accurately and what perturbations (e.g. water quality or habitat degradation) they can tolerate.

"Number of water-column species" was positively correlated ($r^2 = 0.81$) with the IBI score (Fig. 6.4e). Positive correlation indicates that the metric influenced the IBI score. This finding is in line with those of Oberdorff & Hughes (1992) and Fore *et al.* (1994), who found similar results with this metric when developing IBIs for American and French rivers, respectively. Generally the number of water-column species decreases with habitat degradation and produces a low IBI score (Karr *et al.* 1986). For example, a high IBI score was obtained from site 9 on the River Blythe (Fig. 5.3c) with a high number of water-column species and high quality habitat (Fig. 4.20). A low IBI score was found with a low number of water-column species at site 9 on the River Cole (Fig. 5.3e) which may be due to poor quality habitat (Fig. 4.24). Oberdorff & Hughes (1992) recommended using this metric unless these species are absent because of biogeographic reasons. River engineering works (e.g. dredging) affected most of the study rivers and decreased water level, which is critical for all fish species including water-column species. In Belgian rivers, however, the water-column species (similar type of species but number of species is higher than English rivers) showed low responsiveness to perturbations (Kestemont *et al.* 2000), which may be due to their ability to feed on a variety of food resources and adapt to a variety of habitats (Cowx 2001).

The metric "number of benthic species" was also positively correlated ($r^2 = 0.51$) with IBI score (Fig. 6.4f) and thus influenced the IBI. The metric tended to predict the general consequences of habitat degradation as low numbers of benthic species were recorded from degraded sites (Fig. 4.47, River Aire, site 12), consequently producing low IBIs (Fig. 5.6a). A high IBI score was found with a high number of benthic species at site 6 on the River Tame (Fig. 5.3m), which may be due to high quality habitat and / or water quality (Fig. 4.40). The output agrees with Mundahl & Simon (1999) and Thoma (1999), who found a high IBI score with a high number of benthic fish species. Oberdorff & Hughes (1992), Oberdorff & Porcher (1994), Didier *et al.* (1996) and Kestemont *et al.* (2000) applied this metric to evaluate biotic integrity of European rivers, and Lyons *et al.* (1996) showed that the metric is also applicable where few benthic species are found. As engineering works are performed on a periodic basis in English rivers for drainage and flood control purposes, it is important to use "number of benthic species" in future studies. It should be noted that water-column species is complementary to benthic species (if total number of species is used) and Simon (1999) suggested excluding one of the metrics, depending on the representation of such species in the fish community. Equal numbers of benthic and water-column species were found in the study rivers (Section 3.3.3) and both the metrics were tested (Table 5.2). Similar types of results were obtained with these metrics (Figs 6.4e & f). However, it is considered that benthic species are more sensitive to perturbations than water-column species. Hence the metric "number of water-column species" has removed when testing the possibility of reduction of metrics (Section 6.6.3).

The metric “percentage of rheophilic species” was positively correlated ($r^2 = 0.72$) with IBI score (Fig. 6.4g) and influenced the IBI. This finding is similar to that of Fore *et al.* (1994). Generally the number of rheophilic species decreases with habitat degradation (Karr *et al.* 1986) and produces low IBIs (Figs 4.13 & 5.1e, River Windrush, site 3). A high IBI score was found with a high percentage of rheophilic species at site 18 on the River Windrush (Fig. 5.1e), together with high quality habitat (Fig. 4.13). The rheophilic species are usually affected by water flow alteration, and therefore, the metric is particularly important for regulated rivers. In several study rivers, river engineering works (e.g. dredging and removal of dams & weirs) facilitated the speeding-up of water current, which favours rheophilic species (Fig. 4.7, River Evenlode, sites 1, 9 & 14). This metric is related to “percentage of individuals as gravel spawners” as the majority of gravel spawners are rheophilic (Section 6.6.3) and one or other metric may be sufficient.

The metric “percentage of individuals preferring vegetated areas” showed a weak but positive correlation ($r^2 = 0.48$) with IBI score (Fig. 6.4h), which was supported by Hay *et al.* (1996). The number of such fish species usually decreases with loss of aquatic vegetation (Hay *et al.* 1996). For example, a high IBI score was obtained from site 16 on the River Thame (Figs 4.11 & 5.1d) with a high percentage of vegetation-preferring species and a low IBI score was found with a low percentage of such species at site 154 on the River Nidd (Figs 4.49 & 5.6b). Site 154 on the River Nidd was affected by removal of aquatic vegetation while site 16 on the River Thame had adequate vegetation to support vegetation-preferring species. Hay *et al.* (1996) and Hughes & Oberdorff (1999) found similar results with vegetation-preferring species when developing IBIs for African and European rivers, respectively. The weak correlation ($r^2 = 0.48$) indicates the metric had limited impact on the IBI scores in this study and was not therefore, the most important factor. Moreover, this metric is related to other metrics in the trophic composition category and considered inappropriate as a metric to calculate IBI for the study rivers. However, this metric may be vital detecting change in rivers affected by cutting of over-hanging trees and aquatic vegetation.

The metric "percentage of individuals as gravel spawners" was positively correlated (r^2 = 0.83) with the IBI score (Fig. 6.4i), as found by Fore *et al.* (1994). The strong correlation indicates that the metric influenced the IBI score. The number of gravel spawners also declines with habitat degradation (Karr *et al.* 1986), which was reflected in this study as a low IBI score was frequently associated with a low percentage of gravel spawners, for example at site 6 on the River Cherwell (Figs 4.5 & 5.1a). Conversely, a high percentage of gravel spawners gave a high IBI score at site 14 on the River Evenlode (Fig. 5.1b), together with high quality habitat (Fig. 4.7). Berkman & Rabeni (1987) and Oberdorff & Hughes (1992) found similar results for gravel spawners. The metric "percentage of individuals as gravel spawners" appears to be an important metric in this study as the majority of English freshwater fish species are gravel spawners (Mann 1996, Cowx 2001). It is therefore, recommended that the metric "percentage of individuals as gravel spawners" be used in future studies when developing IBIs for other English rivers.

The metric "percentage of individuals as omnivores" showed a weak but positive correlation (r^2 = 0.44) with the IBI score (Fig. 6.4j). Positive relationship of this metric is unusual as percentage of onmivores usually increases with degradation (Karr *et al.* 1986). The positive relationship was probably due to plasticity of omnivores to survive on various food resources and most omnivores are eurytopic in habitat use. Moreover, English rivers normally have more omnivores than fishes from other trophic guilds (Table 5.3) (Cowx 2001). "Percentage of individuals as omnivores" can also be applied universally as most types of waterbody contain some omnivores. It is recommended that this metric should be used in future studies when developing IBIs for other English rivers.

The metric "percentage of individuals as invertivores" was positively correlated (r^2 = 0.61) with the IBI (Fig. 6.4k) and influenced the IBI score. This finding is similar to that of Fore *et al.* (1994). Generally number of invertivores decreases with perturbations related to food base alterations (Karr *et al.* 1986). Diversity of invertivores reflected general habitat degradation. For example, a low percentage of invertivores gave a low IBI score for site 1 on the River Tean (Fig. 5.3n) reflecting habitat degradation at this site (Fig. 4.42). Conversely, a high IBI score was obtained with a high percentage of invertivores at site 4 on the River Sence (Figs 4.34 & 5.3j). Site 4 on the River Sence had good instream cover with pools and riffles (Fig. 4.34). It is assumed that the site was rich with food for invertivores, and consequently the abundance of invertivores was high. Results for the "percentage of invertivores" are in line with those of Oberdorff & Hughes (1992), Oberdorff & Porcher (1994) and Oberdorff (1996) who found similar results for European rivers. Hughes & Oberdorff (1999) suggested that wherever the fauna is sufficiently rich, invertivores or some substitute group of small organism or specialised feeders should be evaluated as a metric. Again, it is recommended this metric be used in future studies when developing IBIs for other English rivers.

The metric "percentage of individuals as piscivores" was also positively correlated with the IBI (Fig. 6.4l) although the relationship was weak (r^2 = 0.44). This weak relationship was probably due to low diversity of piscivores in the study rivers (Table 5.3). Moreover, piscivore abundance was generally low in the lowland rivers studied, partly because of the removal of vegetation and woody debris habitat which they prefer (Cowx 2002). Additionally the data used related to a time when the perch stocks had not fully recovered from the perch ulcer disease which decimated the stocks in the 1970s and 1980s (Dr. Ian G. Cowx, personal communication). Generally the number of piscivores decreases with habitat degradation (Karr *et al.* 1986). For example, a low percentage of piscivores gave a low IBI score for site 9 on the River Soar (Fig. 5.3k), which had poor quality habitat (Fig. 4.36). Oberdorff & Hughes (1992) supported the use of the piscivore metric in their work and it is recommended this metric be used in future studies when developing IBIs for other English rivers.

The metric "number of individuals of long-lived species (No. 100m^{-2})" had limited impact on the IBI score, as the metric showed weak correlation (r^2 = 0.31) (Fig. 6.4m). The weak correlation with the IBI was probably due to low abundance of individuals of such species (chub and common bream) at most sampling sites. Moreover, these species have the ability to survive with variable food resources and habitats. Generally "number of individuals of long-lived species (No 100m^{-2})" are high at high quality habitats. A high number of individuals of long-lived species was contributing to a high IBI score for the site 6 on the River Mease (Figs 4.30 & 5.3h), while the opposite was true at site 3 on the River Sow (Figs 4.38 & 5.3l). The poor performance of the metric in this study is in line with that of Bramblett & Fausch (1991) who found similar results with long-lived species while developing an IBI for Western Great Plains River. As diversity and abundance of long-lived fish species are low in the study rivers, this metric is not considered effective in measuring ecological health of rivers. However, the presence of long-lived species indicates existence of good quality habitat condition over an extended period of time.

Therefore, in other rivers where long-lived species are abundant, this metric may more useful.

"Number of individuals in a sample (No. $100m^{-2}$)" was positively correlated (r^2 = 0.69) with the IBI (Fig. 6.4n) and influenced the IBI score. This finding is similar to that of Fore *et al.* (1994). For example the "number of individuals in a sample (No $100m^{-2}$)" tended to be lower at degraded sites (Fig. 4.44, River Trent) and contributed to lower IBI (Fig. 5.3o, River Trent, site 14). Conversely, a high IBI score was obtained with a high number of individuals in a sample at site 4 on the River Sow (Fig. 5.3l) where habitat quality was high (Fig. 4.38). The metric can be used universally for all types of waterbodies, as it is a common surrogate for system productivity and it is recommended this metric be used in future studies when developing IBIs for other English rivers. A problem exists in meso-trophic rivers where the increased productivity can lead to increased fish abundance even though the water quality is deteriorating.

The metric "total biomass (g m^{-2})" showed a positive correlation (r^2 = 0.61) with the IBI and influenced the IBI score (Fig. 6.4o). A large biomass was associated with a high IBI score for site 10 on the River Penk (Figs 4.32 & 5.3i), while a low biomass gave a low IBI score at site 5 on the River Tean (Figs 4.42 & 5.3n). IBIs linked with this metric are in line with those of Oberdorff & Hughes (1992), Oberdorff & Porcher (1994), Didier *et al.* (1996) and Kestemont *et al.* (2000) who found high IBI scores associated with high biomass for European rivers. "Total biomass (g m^{-2})" may vary for different reasons and total biomass may be high in disturbed sites due to abundance of tolerance species (e.g. site 5, River Mease). Therefore, this metric was removed while testing the possibility of reduction of metrics for English rivers (Section 6.6.3).

6.6.2 Consideration of additional metrics

"Percentage of individuals with deformities, eroded fins, lesions, and tumours" was not included as a metric for this study (Section 5.3.1). This was one of Karr's (1981) original metrics, used to examine gross external anomalies. Hughes & Oberdorff (1999) said that this metric should be retained where diseased and deformed fish may be prevalent. This metric may be useful for English rivers as fishes infected with ectoparasites were recorded from a few study rivers (e.g. Fig. 4.32, River Penk; Fig. 4.36, River Soar) but existing data are weak and preclude the inclusion of the metric. In future studies, this metric or a synonym thereof may be included if appropriate data, currently lacking, are collected.

Concomitantly, there is a growing concern about damage of fish by anglers and predators in many rivers and stillwaters of the UK. To account for this problem, a new metric "percentage of fish damaged by anglers and predators" may be included, where severity of such disturbance is high. However, introduction of such a metric will require close scrutiny of individual fish, which potentially could be a large undertaking in prolific fisheries. It also assumed that the survey operators are able to discriminate damage caused by anglers or predators. Notwithstanding the above, damage caused by birds, e.g. cormorants (*Phalacrocorax carbo carbo* (L.) & *Phalacrocorax carbo sinensis* (Blumenbach)) and herons (*Ardea cinerea* L.) could be deemed a natural phenomenon and as such is not strictly indicative of degradation.

Hybridisation between species of the family Cyprinidae is a common phenomenon found in many UK lowland rivers (Wheeler 1969, Cowx 1983, Pitts 1994). Cowx (1983) considered the proportion of hybrids in lowland fish communities was considerably greater than in the past and argued this was due to degradation and loss of spawning habitat. Consequently species that would normally be reproductively isolated by geographic or physical barriers are drawn together and the proportion of hybrids increases. Therefore, this metric could be a useful measure of degradation. One of the problems with introducing this metric is the ability of researchers or technicians to identify hybrids in the field. Whilst it is relatively straightforward for experienced workers to identify F1 generation hybrids between species such as roach and common bream in the field, the high prevalence of introgressive back-crossing (Verspoor & Hammer 1991) produces offspring that are difficult to discern from their pure parental form and need specialist laboratory analysis. Also some hybrids are not easy to discriminate, e.g. common bream x silver bream or bleak x chub. Moreover, Hughes & Oberdorff (1999) stated that this metric has limited usefulness and evaluation of reproductive guilds is favoured where this knowledge is available.

Initially, "percentage of standard growth of fishes" (Hickley & Dexter 1979, Hickley & Sutton 1984) was considered as a metric in this study (Section 5.3.1) but was excluded due to lack of data. However, this metric could potentially be important in identifying reaches where the primary and secondary productivity has been disrupted and is manifest in the growth rates of the fish. Fish growth varies with quality and quantity of food and other biotic (e.g. competition) and abiotic (e.g. water temperature) factors. Abundance of fish food varies with physical (loss of benthic food: dredging), chemical (loss of all kinds of food: pollution) and biological (loss of predator's food: presence of voracious non-natives, *S. lucioperca* and *S. glanis*) disturbances and hence growth of fish also varies accordingly. This highlights the importance of this metric and the value of its inclusion if appropriate data are available. The EA collects data on growth of fishes but need to be standardised and made a strategic element in survey monitoring before the metric "percentage of standard growth of fishes" can be included.

Another metric, "mortality rate (Z) of fishes" was also considered but was not tested because of lack of data. In future studies, this metric may be included if appropriate data, currently lacking, are collected. Whilst mortality can be assessed from a single large data set it requires that the sampling is not selective for larger individuals. Unfortunately this is often not the case. Bias due to poor representation of small younger age classes can be accommodated in the estimation but problems with natural variability in year class strength can have a marked influence on the estimate.

Initially the metric, "percentage of juvenile fishes (YOY)" was considered (Section 5.3.1) but again was not tested due to paucity of adequate data. YOY are indicators of spawning and recruitment success of a fishery. Low recruitment due to spawning failure is usually associated with degradation of water quality, loss of spawning habitat, blocking of migration routes and presence of a high percentage of piscivores. However, introducing this metric may need adjustment of the timing of sampling and use of more appropriate sampling methods. For the study rivers, the best sampling period for YOY is mid summer (July) to early autumn (September) when YOY of most English fishes are found. Micro-mesh seine netting is probably the best method to sample YOY but other methods such as Point Abundance Sampling (PAS) with electric fishing gear may be used. In the surveys, large numbers of YOY were caught from many study rivers using electric fishing gear but the abundance is assessed subjectively and may need quantification by species. There are also possible problems (e.g. equipment and expertise) with identifying YOY fish, which may restrict usage of this metric. After solving these problems only then can this metric be included to evaluate biotic integrity of English rivers.

When selecting metrics for an IBI, it is sensible to avoid complementary metrics such as "number of water-column species" & "number of benthic species" (Table 5.2), "percentage of omnivores" & "percentage of microphagic omnivores", and "percentage of specialist spawners" & " percentage of non-specialist spawners". Complementary metrics may give misleading IBI scores, as the groups of fishes under complementary metrics use feeding and breeding resources opposite to each other (Section 3.2.2) and thus double account for the particular metric, leading to biases that may give a false picture of the status of ecological health of a river. The metrics chosen should therefore be scrutinised to remove the potential for complementarity.

6.6.3 Possible reduction of metrics

Most metrics used in this study were considered of same value as correlation of metrics (r^2 value) with the IBI varied between 0.31 and 0.85 (Fig. 6.4a-o). However, 5 metrics were subsequently considered unnecessary due either (1) to complementary characteristics of metrics (e.g. "number of water-column species" is complementary to "number of benthic species" and "percentage of individuals as rheophilic species" is related to "percentage of individuals as gravel spawners"), or (2) to weak correlation with IBI (e.g. "percentage of individuals preferring vegetated areas" and "number of individuals of long-lived species (No. $100m^{-2}$)") and (3) ability to give variable IBI scores under similar types of degradation (e.g. "total biomass ($g\ m^{-2}$)"). Both gravel spawners and rheophilic species showed a strong correlation with the IBI, possibly for the same reasons as most rheophilic species are gravel spawners. Moreover, gravel spawners are probably more sensitive to loss of spawning habitat than rheophilic species are to a reduction in water current. It therefore, appears that there is some scope to remove metrics such as "number of water-column species", "percentage of individuals as rheophilic species", "percentage of individuals preferring vegetated areas", "number of individuals of long-lived species (No. $100m^{-2}$)" and "total biomass ($g\ m^{-2}$)".

The possible removal of these 5 metrics was tested on the River Mease (Table 6.4). The River Mease was classified using the IBI as "Good", following the integrity class boundaries based on 15 metrics (Table 5.6). The same rating scale (5, 4, 3, 2, 1 and 0) was used to score 10 metrics after removing the above-mentioned metrics (Table 5.5) (5 metrics were removed at a time). As the number of metrics was reduced from 15 to 10, the total IBI score ranged between 50 (10 x 5) and 0 (10 x 0). Six integrity classes (Table 5.6) were defined with the following class boundaries; Excellent: 38 - 50, Good: 27 - 37, Fair: 19 - 26, Poor: 11 - 18, Very Poor: 1 - 10 and No Fish: 0. The new class boundaries for 10 metrics were calculated proportionately from the class boundaries developed for 15 metrics.

Adjustment of class boundaries for 10 metrics is needed as the total score is 50 whilst for 15 metrics, the total score was 75. This test for the possible reduction of the number of metrics showed no differences in the integrity class for individual site or for the mean class quality (i.e. "Good") of the River Mease (Table 6.4). Results showed that up to 5 metrics may be removed from the metric list (Table 5.2) for future application of IBI in other rivers. However, it is concluded that reduction, addition or modification of metrics should be tested nationally first.

Table 6.4 Comparison of IBI scores between 15 and 10 metrics for the River Mease

Site no.	IBI score for 15 metrics	IBI score for 10 metrics	Integrity class based on 15 metrics	Integrity class based on 10 metrics	Comments
1	39	26	Fair	Fair	No change
2	40	23	Fair	Fair	No change
3	45	27	Good	Good	No change
4	43	28	Good	Good	No change
5	40	25	Fair	Fair	No change
6	48	30	Good	Good	No change
7	46	30	Good	Good	No change
Mean	**43±3.21**	**27±2.39**	**Good**	**Good**	**No change**

Although all the study rivers are from the East coast they covered nearly two-third of the UK catchment area (rivers of the Thames, Midlands and Northeast regions [Section 2.1]). Moreover, many UK rivers have similar habitats and fish species diversity, and suffer similar types of perturbations (Section 2.3). Therefore, it is suggested that the metrics used in this study can be applied to middle and lower reaches of other UK rivers. However, to address specific issues such as translocated species (e.g. roach in Scottish rivers, barbel in the Severn catchment), it may be necessary to include new metrics, such as "number of translocated species". Translocated species may compete for food and spawning substrate with the resident species of the recipient waterbody (Cowx 1998b). In this study, no naturalised fish species (e.g. *C. carpio*) was considered as a native species. It was expected that naturalised fish species will not significantly affect the results of this study as diversity and density of such species are generally low in English rivers (Tables 3.3 & 3.4). However, naturalised species could be included in the native fish species category in future studies as they have adapted themselves to the local habitat.

6.7 SCORING CRITERIA OF SELECTED METRICS

The choice of scoring criteria, integrity class ranges and boundaries depends on the scientist developing the index. A traditional but continuous scoring scale, having a continuous class range, was used to score IBI metrics and to assign integrity classes for English lowland rivers (Section 5.4.2). This type of scoring system was also used by Minns *et al.* (1994) and Ganasan & Hughes (1998). Rankin & Yoder (1999) suggested the adjustment of scoring scales and integrity class boundaries with reference criteria from a new ecoregion (i.e. study ecoregion) that will improve the ability to characterise and quantify the severity of impairment in a particular stream or river segment. Mundahl & Simon (1999) used a discontinuous scoring system (0 - 5 - 10) with discontinuous class range (105 - 120, 70 - 100, 35 - 65, 10 - 30, 1 - 5 & "No Fish") to develop an IBI for coldwater streams with 12 metrics. However, in the present study a continuous scoring system was considered appropriate as it helped to maintain continuity of the scoring system (Table 5.5). The scoring scale was adjusted for the increased number of metrics (15 metrics) chosen for English lowland rivers. In a discontinuous scale, it is more difficult to explain the IBI scores between upper boundary of a class (e.g. "Poor") and lower boundary of the next class (e.g. "Fair"). A continuous scale makes it possible to transfer all scores within a boundary range to a specific integrity class. Therefore, it is recommended that a continuous rating scale be selected when developing an IBI for other English lowland rivers.

6.8 LIMITATIONS OF IBI

Any tool can be misused and, if the limitations of IBI are not recognised, it can be misapplied or misinterpreted. The IBI is designed for use when the objective is to monitor biotic integrity (at specific sites). When other objectives are pursued, for example, the management of a single species, the index is of little value. Some limitations of IBI are as follows:

a. Management decisions based on IBI are best made with the guidance of a fish biologist familiar with IBI and with knowledge of the local fish fauna and watershed conditions. The use of IBI by individuals without biological training is likened to the use of econometric or engineering tools by those without specialised training. Non-biologists may give importance to the face value of the data and results and ignore the underlying causes of variations in the IBI scores.

b. A potentially dangerous practice is to turn interpretation of IBI over to a computer software package (Karr *et al.* 1986). A major advantage of IBI is its ability to integrate and summarise the collective wisdom of biologists. Computer programs, on the other hand, overemphasise numerical data and minimise evaluation and interpretation. Because fish community usually vary with stream size (watershed area) and regional zoogeography and a considerable investment of time is required to define expectation criteria and to collect, collate and interpret data from sampling sites. Computer software can be used to carry out the simple mathematics but the result must be interpreted by biologists.

c. Management at the watershed level is essential if the problems indicated by low IBI scores are to be solved (Karr & Schlosser 1978). Some management practices could merely improve metric scores temporarily but not improve biotic integrity. The stocking of piscivores / top carnivores, for example, may increase a local IBI value temporarily, but if these fishes have little chance of long-term survival, the measure is pointless (Lyons 1992).

d. Representative samples are essential for IBI calculations. Among the most common problems associated with any sampling are reliance on river reaches that are too short and gear that is ineffective for certain species or habitats.

e. The importance of professional judgement during the sampling, the development of expectation criteria, the assignment of metric scores, and the interpretation of those scores is critical.

f. The IBI, of course, is not the last word in river management. Instead, it is a tool that aids in the interpretation of complex biological data and a method that integrates physical and chemical data. The IBI score for given sites are always relative to one another and have no absolute meaning.

g. Finally, for a variety of reasons, caution must be exercised when comparing streams/rivers from different geographic regions. Qualitative labels ("excellent" to "no fish") may be used in making comparative statements but quantitative IBI scores cannot.

6.9 CONCLUSIONS AND RECOMMENDATIONS

6.9.1 Conclusions

Measuring biotic integrity of a stream / river is, in a sense, analogous to measuring human health. "Good health" is not a simple function of the attributes described in section 6.6. Rather, a biological system – whether it is a human health system or a river ecosystem

– can be considered healthy when its inherent potential is realised, its condition is stable, its capacity for self-repair when perturbed is preserved, and minimal external management support is needed. The IBI is an index that is useful in a variety of situations and incorporates information from many biotic variables. No single index (Chapter 4) or set of metrics (Chapter 5) can be expected to detect all water resource problems. However, the IBI developed in this study was considered very successful as a broad-based approach in assessing the ecological health of the middle and lower reaches of study rivers. Based on the previous discussions, the following conclusions are made:

a. The diversity indices, ABC method and multivariate techniques, appeared inappropriate to measure ecological health of study rivers even when used in combination (Section 4.3.2) as the indices, method and multivariate techniques are only based on the structural component of fish communities (Section 4.2). Rather they tend to be affected by certain changes in the structural composition due to particular perturbation (Section 4.3.2).

b. The IBI is very flexible with respect to data collection. Statistical design for data collection is not obligatory and it allows the use of subjectively selected sites. This is because the IBI is site specific. Similarly, data collected during *ad hoc* surveys can be used from sites of different lengths (Section 6.4).

c. The IBI was relatively robust with regard to sampling requirements (Section 6.4). Electric fishing appears to be an appropriate technique for sampling fish stocks in English rivers for IBI calculations. Single-sweep electric fishing is acceptable if carried out with rigour (Section 6.4), although quantitative sampling is preferable.

d. Although considerable expertise is necessary for the metric identification, scoring system development and integrity class determination, over all calculations are very simple. No complicated statistical analysis or formulae are required (Section 6.6).

e. The IBI allows simple interpretation of the score (e.g. "Excellent", "Good", "Fair", "Poor") and easy presentation and communication to the layman.

f. The success of the IBI is partly due to the ability of users to adapt and calibrate the index to reflect regional conditions and expectations (Section 6.3).

g. The IBI was an appropriate tool to measure biotic integrity of the middle and lower reaches of both small (e.g. River Anker) and large rivers (e.g. River Trent) (Section 5.6). The IBI developed in this study was inappropriate for headwaters and there is a need to develop a separate IBI based on separate reference conditions and metrics for this zone of the river.

h. The number of metrics may be removed depending on the local fish fauna and habitat. For English lowland rivers, 10 metrics (Table 6.4) appear to be adequate for calculating the IBI.

i. The EA has historical data on fish stocks of various rivers. These data can be used to develop IBIs for specific river types and may take account of regional variations.

j. The IBI was a rapid assessment method with great flexibility and can be easily modified for different applications (Section 5.6).

k. The IBI provides a straightforward method for assessing different rivers, so that those systems or reaches most in need of protection or restoration can be identified.

l. As the IBI measures ecological health, it can be used in conservation management of endangered and threatened species, as these species require a healthy ecosystem for their survival.

m. The IBI can be used as a monitoring and evaluation tool to identify streams and rivers where restoration activities are needed and to monitor biodiversity change over time.

n. The EA could use the IBI in Local Environment Agency Plan (LEAP) for integrated management of watersheds.

o. In short, IBI satisfies three basic conditions named by Schindler (1987) for useful monitoring programmes: inexpensive, simple to use and highly sensitive to changes in ecosystems. However, data collection is expensive but the high cost of data collection is inherent in all biological assessment methods.

6.6.2 Recommendations

The IBI developed in the present study is based on a number of assumptions and has a number of limitations, which need further investigation and development, if and when the appropriate data become available. The IBI developed in this study was also based on reference conditions and metrics that are specific to middle and lowland reaches. Consequently, the IBI should be further tested on a wide range of lowland rivers in the UK to assess whether it is appropriate for assessing ecological health of lowland rivers in all regions of the UK, but most particularly in England where cyprinids are a major component of the fish fauna. If the IBI proves ineffective, appropriate modifications should be made in relation to regional differences in fish communities.

The IBI developed was based on three main river catchments (Thames, Trent and Yorkshire Ouse). These catchments all drain to the east coast and probably have a much more diverse fish fauna than those draining to the west coast (Varley 1967) or those catchments in Wales and Scotland. Indeed, rivers in Wales and Scotland generally have a very depauperate fish fauna and the IBI will probably not be appropriate to these rivers. Furthermore, the IBI developed for this study was shown to be inappropriate for headwaters and it is likely that rivers dominated by salmonids throughout their watercourse, as is commonly found in Scotland and Wales, will not fit to this IBI. Therefore, a separate IBI should be developed and tested on headwater streams where the fauna is dominated by salmonids and minor species, e.g. stone loach and bullheads. Any IBI developed for headwaters of rivers could prove problematical because of the low diversity of the fish fauna. It is anticipated that the metrics used will be more orientated towards individual species, population structure and dynamics rather than fish community structure as used in the current study.

Inclusion of the IBI in wider aquatic resource monitoring programmes (e.g. Water Framework Directive) is an important issue for consideration. The WFD requires that the European Union member states establish monitoring and ecological quality classification systems for the purpose of constantly assessing the ecological status of surface waters and defining the level of human impact on ecosystems. The WFD is based on four "quality elements", phytoplankton, macrophytes and phytobenthos, benthic invertebrate fauna and fish fauna (Section 1.2). The IBI developed in this study is possibly the first step towards meeting the obligations of the UK under the WFD for assessing quality of rivers using fish fauna.

Whilst the IBI method obviously needs further development as outlined above, its general applicability for assessing ecological health has been vindicated in this study. It is recommended that the IBI approach should be adopted for assessing ecological health in all other water bodies as required under the WFD, i.e. stillwaters and estuaries. These types of waters have very different fish faunas, so new indices will have to be developed based on reference conditions applicable to the characteristic fish faunas. The actual metric structure of the IBIs may also be very different because the fish community and ecosystem dynamics are very different. Difficulties are envisaged in developing IBIs for estuaries in particular because of the transient nature of many of the fish species, many of which only use estuaries for specific life stages. However, metrics that reflect the role of estuaries as nursery areas for marine species and reflect unhindered migration of anadromous species may be a starting point for detecting ecological health. It is therefore, recommended that the methods developed by Jennings *et al.* (1999) for lakes and by Deegan *et al.* (1997) for estuaries in USA be adjusted for use in UK stillwaters and estuaries, respectively.

Acquisition of standardised data is an important issue in the calculation of IBIs (Section 6.4.1), although the IBI method is not considered data hungry. Due to variations in gear type and specifications, data quality may vary between different regions or zones of rivers resulting in inaccurate assessment of fish populations. Consequently, it is necessary to establish survey monitoring programmes, which provide a standard suite of data outputs to meet the IBI requirements. It must be accepted that different river types require different sampling methodologies but the gear type plus intensity and frequency of sampling must be appropriate to provide an adequate picture of the fish community structure and dynamics. Several initiatives are in hand to meet these requirements including those of the Council of Europe, Committee for Standardisation (CEN) and revision of the EA monitoring programme. In the case of CEN, standard procedures for sampling using electric fishing and gill netting are proposed and, if adopted by member states, should meet the requirements of the IBI.

Fish sampling and data collection is a costly activity (Hickley & Starkie 1985). As already indicated, the data requirements for the IBI are not necessarily intensive and can be met by traditional methods such as electric fishing. However, consideration should be given to using more cost effective sources of data such as creel census, match fishing or angler's log books (Cowx 1990b, Hickley 1996). These methods, however, may provide a biased picture of the fish community because anglers generally target specific species and size groups of fish. However, if the intensity of angling is high and the recording process accurate, e.g. the Nottingham Federation of Anglers stretch of the River Trent in Nottingham (Cowx 1991), it may be possible to adapt the IBI specifically for this type of data. Criteria such as quality of angling, catch rates, and size distribution of fish caught may have to be incorporated into the modified IBI through appropriate metrics but the development of a simplified IBI should be investigated.

The IBI was developed as a tool to categorise ecosystem degradation. However, there is no reason why the method cannot be used as a tool to monitor change. The present philosophy in Europe, driven by various EU Directives (e.g. Habitats Directives or WFD) is to improve the ecological status of water courses. To achieve this, many types of management practices are being undertaken, including reduction of pollution discharges, rehabilitation of rivers, construction of instream habitat features and fish stock enhancement. Although the tendency is not to carry out pre and post project monitoring of the impact of these potentially positive and negative activities, this is seen as a short coming. If the IBI is sufficiently robust, it should be possible to use it to measure ecosystem change and assess whether such activities are appropriate and achieve their desired output. Where the IBI may fail is that it is not sensitive enough to detect subtle changes in the ecosystems, brought about by small scale interventions, i.e. the species abundance and standing crop may increase but the diversity may not, thus change (improvement) may not be detected. Notwithstanding the above arguments, the applicability of using IBIs for this type of pre and post project monitoring should be investigated.

One problem that needs to be overcome is correlation of the IBI scores to those of other indices. Even if the IBI is adopted as the standard measure of ecosystem health, there is a need to correlate the output against other classifications such as habitat index, diatom index, microinvertebrate index, chemical index for water quality and GQA index. Direct correlation between indices (e.g. IBI & H' or IBI & ABC) or classifications (e.g. "Good" & RE1 or "Poor" & RE5) is not expected, because fish respond differently to environmental perturbation than other animal groups. Moreover, different fish community attributes (e.g. fish density and biomass, trophic [i.e. omnivore, carnivore] and reproductive [i.e. phytophils, lithophils] guilds, habitat utilisation [i.e. benthic or rheophilic]) are used in different indices to assess ecological health of waterbodies. However, it is critical to know how the various indices behave in response to different types of degradation. Once the IBI has been established for English rivers, and many sites have been classified, a type of matrix analysis, perhaps using non-parametric correlation methods, should be undertaken and a series of comparative tables produced. This will allow a better integration of the methods for more accurate assessment of the status of the rivers based on a multiple criteria approach.

For more effective application and understanding, the IBI should be incorporated into a GIS (Geographical Information System) environment. Presentation of the IBI through GIS will help to inform the general public, especially anglers and other river users. Coloured and annotated maps depicting the state of the river are a powerful way of providing information to the general public.

Although there has been criticism by different workers to turn the interpretation of IBI over to a computer software package (Karr *et al.* 1986, Lyons 1992), it is suggested that a suitable computer package is developed to reduce repetitive and time consuming calculations on various aspects of IBI metrics. As biologists develop such a package, they will be able to integrate most components of the fish community to calculate an IBI. However, interpretation of an IBI score in relation to the status of the ecosystem requires considerable expertise and should include judgement of those with local knowledge.

REFERENCES

Alabaster J. S. and R. Lloyd (1982). *Water quality criteria for freshwater fish.* Second Edition. Food and Agriculture Organisation of the United Nations, London: Butterworth Scientific, 361 pp.

Alatalo R. and R. Alatalo (1977). Components of diversity: multivariate analysis with interaction. *Ecology,* **56:** 1040 - 1053.

Allen J. H. (1999). The analysis and prediction of the shallow subtidal benthic communities along the East coast of England. Unpublished *PhD Thesis.* University of Hull, UK. 309 pp.

Allen S. M. (1989). The adaptation of an index of biotic integrity to streams of the Coastal Plain Drainage Basins in North Carolina. Unpublished *M.Sc. Thesis,* School of Forestry and Environmental Studies, Duke University, NC, USA. 124 pp.

Angermeier P. L. and J. R. Karr (1983). Fish communities along environmental gradients in a system of tropical streams. *Environmental Biology of Fishes,* **9(2):** 117 - 135.

Angermeier P. L. and J. R. Karr (1986). Applying an index of biotic integrity based on stream-fish communities: considerations in sampling and interpretation. *North American Journal of Fisheries Management,* **6:** 418 - 429.

Angermeier P. L. and I. J. Schlosser (1987). Assessing biotic integrity of the fish community in a small Illinois stream. *North American Journal of Fisheries Management,* **7(3):** 331 - 338.

Angermeier P. L. and J. R. Karr (1994). Biological integrity versus biological diversity as policy directives: Protecting biotic resources. *BioScience,* **44(10):** 690 - 697.

Angermeier P. L. and R. A. Smogor (1995). Estimating number of species and relative abundance in stream-fish communities: effects of sampling effort and discontinuous spatial distributions. *Canadian Journal of Fisheries and Aquatic Sciences,* **52:** 936 - 949.

Angermeier P. L., R. A. Smogor and J. R. Stauffer (2000). Regional frameworks, and candidate metrics for assessing biotic integrity in mid-Atlantic highland streams. *Transactions of the American Fisheries Society,* **129:** 962 - 981.

Appelberg M., B. C. Bergquist and E. Degerman (2000). Using fish to assess environmental disturbance of Swedish lakes and streams – a preliminary approach. *Verhandlungen Internationale Vereinigung fur Theorestiche und angewandte Limnologie.,* **27:** 311 - 315.

Armstrong G. S. 1983. Some effects of maintenance dredging on the fish populations in a lowland river. *Proceedings of 3rd British Freshwater Fisheries Conference,* University of Liverpool, UK. pp. 203 - 215.

Atkinson M. J. (1994). The factors influencing the fisheries in the River Aire. Unpublished *MSc Thesis.* University of Hull, UK. 114 pp.

Austen D. J., P. B. Bayley and B. W. Menzel (1994). Importance of the Guild Concept to Fisheries Research and Management. *Fisheries,* **19(6):** 12 - 20.

Axford S. (1994). Rehabilitation of inland fisheries: water quality aspects. In: I. G. Cowx (ed.) *Rehabilitation of Freshwater Fisheries.* Fishing News Books, Blackwell Scientific Publications, Oxford, UK. pp 361 - 365.

Bagenal T. B. and F. W. Tesch (1978). Age and growth. In: T. B. Begenal (ed.) *Methods for Assessment of Fish Production in Fresh Waters.* IBP Handbook No. 3, Blackwell Scientific Publications, Oxford, UK. pp 101 - 136.

Bain M. B., J. T. Finn and H. E. Booke (1988). Streamflow regulation and fish community structure. *Ecology*, **69:** 382 - 392.

Balon E. K. (1975). Reproductive guilds of fishes- a proposal and definition. *Journal of the Fisheries Research Board of Canada*, **32(6):** 821 - 864.

Balon E. K., W. T. Momot and H. A. Regier (1977). Reproductive guilds of percids: results of the paleogeographical history and ecological succession. *Journal of the Fisheries Research Board of Canada*, **34:** 1910 - 1921.

Balon E. K (ed.) (1980). "Charrs: Salmonid Fishes of the Genus *Salvelinus*". Junk Publications, The Hague, The Netherlands, 115 pp.

Balon E. K. (1981a). Reproductive guilds and the ultimate structure of fish taxocenes: amended contribution to the discussion presented at the mini-symposium. *Environmental Biology of Fishes*, **3:** 149 - 152.

Balon E. K. (1981b). Additions and amendments to the classification of reproductive styles in fishes. *Environmental Biology of Fishes*, **6(3-4):** 377 - 389.

Baras E. (1995). Thermal related variations of seasonal and daily spawning periodicity in *Barbus barbus*. *Journal of Fish Biology*, **46:** 915 - 917.

Barmuta L. A. and P. S. Lake (1982). On the value of the river continuum concept. *New Zealand Journal of Marine and Freshwater Research*, **16:** 227 - 231.

Bayley P. B. (1988). Factors affecting growth rates of young tropical floodplain fishes: seasonality and density dependence. *Environmental Biology of Fishes*, **21:** 127 - 142.

Bayley P. B. , R. W. Larimore and D. C. Dowling (1989). Electric seine as a fish-sampling gear in streams. *Transactions of the American Fisheries Society*, **118:** 447 - 453.

Belliard J., P. Boet and E. Tales (1997). Regional and longitudinal patterns of fish community structure in the Seine River basin, France. *Environmental Biology of Fishes*, **50(2):** 133 - 147.

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

Biggs J., M. Whitfield, P. Williams, A. Corfield, G. Fox and K. Adare (1998). *Biological Techniques of Still Water Quality Assessment*: Phase 2: Method Development. R & D Technical Draft Report to Environment Agency, Bristol, UK. pp 41 - 49.

Boet P., D. Chessel, B. Hugueny, T. Oberdorff, D. Pont and J. P. Porcher (1999). *Rapport final de la Phase II du programme national (Indice Poisson).* Conseil Superieur de la Peche, Direction Generale, 134, Avenue de Malakoff, 75116 Paris, France. 60 pp.

Bone Q. and N. B. Marshall (1982). *Biology of Fishes*. Blackie, Glasgow, UK. 344 pp.

Boon P. J. (1992). Essential elements in the case for river conservation. In: P. J. Boon, P. Calow and G. E. Petts (eds) *River Conservation and Management,* John Wiley & Sons, Chichester, UK. pp. 11 - 34.

Bowen Z. H.; M. C. Freeman and D. L. Watson (1996). Index of biotic integrity: Applied to a flow-regulated river system. *Proceedings of the Annual Conference Southeast Association Fish. and Wildlife Agencies*, **50:** 26 - 37. [Online]-*http://www.mesc.usgs.gov/pubs/online/ifim-chron/ind_biotic_integ.html,* Accessed on 02 December 1998.

Braddock A. (1977). The River Trent. *Coarse Angler*, **1:** 38 - 40.

Bramblett R. G. and K. D. Fausch (1991). Variable fish communities and the index of biotic integrity in a Western Great Plains River. *Transactions of the American Fisheries Society*, **120(6):** 752 - 769.

Brewin D. J. and J. R. Martin (1988). Water quality management: A regional perspective - The Severn - Trent area. *Regulated Rivers: Research & Management*, **2(3):** 257 - 275.

Britton J. R. (1999). The impact of cormorants (*Phalacrocorax carbo carbo* and *Phalacrocorax carbo sinensis*) on inland fisheries in the UK. *Unpublished PhD thesis*, University of Hull, UK. 377 pp.

Bruton M. N. and G. S. Merron (1990). The proportion of different eco-ethological sections of reproductive guilds of fishes in some African inland waters. *Environmental Biology of Fishes*, **28:** 179 - 187.

Burnett J., M. Clegg, M. Shannon, S. Warburton, S. Dallow and G. Hobbs (1978). The Yorkshire Derwent: A Case for Conservation. England. 73 pp.

Butterworth A. J. (1990). Fish populations of the lower River Kennet and Foudry Brook. Thames Water Report, UK. 68 pp.

Caddy J. F. and R. Mahon (1995). Reference points for fisheries management. *FAO Fisheries Technical Paper* **347:** 83 pp.

Cairns J. Jr. (1971). A simple method for the biological assessment of the effects of waste discharges on aquatic bottom dwelling organisms. *Journal of Water Pollution Control Federation*, **40:** 755 - 782.

Cairns J. Jr. (1977). Quantification of biological integrity. In: R. K. Ballentine and L. J. Guarraia (eds) *The integrity of water: a symposium.* US Environmental Protection Agency, Washington DC, USA. pp 171 - 185.

Calow P. and G. E. Petts (eds) (1994). *The Rivers Handbook: Hydrological and Ecological Principles*; Vol. II. Blackwell Scientific Publications, Oxford, UK. 523 pp.

Canfield D. E. and J. R. Jones (1984). Assessing the trophic status of lakes with aquatic macrophytes. *Lake and Reservoir Management*, **1:** 446 - 450.

Carpenter K. E. (1928). *Life in inland waters.* Sidgwick & Jackson Ltd., London, UK. 267 pp.

CEN documents (2001). Water analysis - Sampling of fish with electricity. CEN/TC230/WG2/TG4NXX, 12 pp.

Chadwick E. M. P. (1976). Ecological fish production in a small Precambrian shield lake. *Environmental Biology of Fishes*, **1:** 13 - 60.

Chandler G. L, T. R. Maret and D. W. Zaroban (1993). Protocols for assessment of biotic integrity (fish) in Idaho streams. *Idaho Department of Health and Welfare*, Division of Environmental Quality, Water Quality Monitoring Protocols, Report 6, Boise, USA, 112 pp.

Chun K., M. J. Weaver and L. A Deegan (1996). Assessment of fish communities in New-England Embayments:- Application of the estuarine biotic integrity index. *Journal of Biological Bulletin*, **191(2):** 320 - 321.

Coeck J., A. Vandelannoote, R. Yseboodt and R. F. Verheyen (1993). Use of the Abundance / Biomass method for comparison of fish communities in regulated and unregulated lowland rivers in Belgium. *Regulated Rivers: Research & Management*, **8:** 73 - 82.

Coles T. F., J. S. Wortley and P. Noble (1985). Survey methodology for fish population assessment with Anglian Water. *Journal of Fish Biology*, **27 (Suppl. A):** 175 - 186.

Cooper M. J. and G. A. Wheatley (1981). An examination of the fish populations in the River Trent, Nottinghamshire using angler catches. *Journal of Fish Biology*, **19:** 539 - 556.

Courtenay W. R. Jr. and P. B. Moyle (1992). Crimes against biodiversity: the lasting legacy of fish introductions. In: J. E. Williams and R. J. Neves (eds) *Biological Diversity in Aquatic Management*. Transactions 57th North American Wildlife & Natural Resources Conference. Wildlife Management institute, Washington DC., USA. pp. 365 - 372.

Cowx I. G. (1980). Population ecology and management of coarse fish in the River Exe and River Culm. Unpublished *PhD Thesis*, University of Exeter, Exeter, UK. 413 pp.

Cowx I. G. (1983). The biology of bream, *Abramis brama* (L.) and its natural hybrid with roach, *Rutilus rutilus* (L.) in the River Exe. *Journal of Fish Biology*, **22:** 631 - 646.

Cowx I. G. (1988). Distribution and variation in the growth of roach *Rutilus rutilus* (L.) and dace, *Leuciscus leuciscus* (L.), in a river catchment in southwest England. *Journal of Fish Biology*, **33:** 59 - 72.

Cowx I. G., G. A. Wheatley and P. Hickley (1988). Developments of boom electric fishing equipment for use in large rivers and canals in the United Kingdom. *Aquaculture and Fisheries Management*, **19:** 205 - 212.

Cowx I. G. (1990a). *Developments in Electric Fishing*. Fishing News Books, Blackwell Scientific Publications Ltd., Oxford, UK. 358 pp.

Cowx I. G. (1990b). Application of creel census data for the management of fish stocks in large rivers in the United Kingdom. In: W. L. T. van Densen, B. Steinmetz & R. H. Hughes (eds) 1990. *Management of freshwater fisheries*. Proceedings of a symposium organized by the European Inland Fisheries Advisory Commission, Goteborg, Sweden, 31 May-3 June 1988. *Pudoc. Wageningen*. pp. 526 - 534.

Cowx I. G. and P. Lamarque (1990). *Fishing with Electricity: Applications in Freshwater Fisheries Management*. Fishing News Books, Oxford, UK. 248 pp.

Cowx I. G., G. A. Wheatley, P. Hickley and A. S. Starkie (1990). Evaluation of electric fishing equipment for stock assessment in large rivers and canals in the United Kingdom. In: I. G. Cowx (ed.) *Developments in Electric Fishing*. Fishing News Books, Blackwell Scientific Publications Ltd., Oxford, UK. pp 34 - 40.

Cowx I. G. (1991). The use of angler catch data to examine potential fishery management problems in the lower reaches of the River Trent, England. In: I. G. Cowx (ed.) *Catch Effort Sampling Strategies and their Application in Freshwater Fisheries Management*. Oxford: Fishing News Books, Blackwell Scientific Publications. pp 154 - 165.

Cowx I. G. (1995). Fish Stock Assessment - A Biological Basis for Sound Ecological Management. In: D. M. Harper & A. J. D. Ferguson (eds) *The Ecological Basis for River Management*. John Wiley & Sons Ltd, England. pp 375 - 388.

Cowx I. G., C. S Pitts, K. L. Smith, P. J. Hayward and S. W. F. van Breukelen (1995). *Factors influencing coarse fish populations in lowland rivers*. R & D Report-0429/6/N &Y, National Rivers Authority, Bristol, UK. 125 pp.

Cowx I. G. (1997). Introduction of fish species into European freshwaters: economic success or ecological disasters ? *Bulletin Francais de la Peche et de la Pisciculture*, **344-345:** 57 - 78.

Cowx I. G. and R. L. Welcomme (eds) (1998). *Rehabilitation of Rivers for Fish*. Fishing News Books, Blackwell Science Ltd., Oxford, UK. 260 pp.

Cowx I. G. (1998a). *Factors influencing coarse fish populations in lowland rivers: Literature Review*. Environment Agency, R & D Report., Bristol, UK. 185 pp.

Cowx I. G (ed.). (1998b). *Stocking and Introduction of Fish.* Fishing News Books, Oxford, UK. 456 pp.

Cowx I. G. (1999). The importance of planning in the rehabilitation of freshwater fisheries. *Rybactwo Jeziorowe, IV Krajowa Konferencja Rybackich Uzythowników Jezior* June 1999, Olsztyn, Poland, pp. 43 - 68.

Cowx I. G. and P. A. Godkin (2000). *Analysis of the environmental and economic impact of operations to reinforce the aquatic fauna of fresh waters for fishery purposes.* Report to Director-General for Fisheries, European Commission. HIFI, University of Hull, UK. 141 pp + Appendix.

Cowx I. G. (ed.) (2000). *Management and Ecology of River Fisheries.* Fishing News Books, Oxford, UK. 444 pp.

Cowx I. G. (2001). *Factors influencing coarse fish populations in rivers: A literature review.* Environment Agency, Rio House, Waterside Drive, Bristol BS32 4UD, UK. R & D Publication 18, 146 pp.

Cowx I. G. (2002). Analysis of threats to freshwater fish conservation: past and present challenges. In: M. J. Collares-Pereira, I. G. Cowx and M. M. Coelho (eds) *Conservation of Freshwater Fishes: Options for the Future.* Fishing News Books, Blackwell Scientific Publications, Oxford, UK. pp 201 - 220.

Cragg-Hine D. and J. W. Jones (1969). The growth of Dace, *Leuciscus leuciscus* (L.), Roach, *Rutilus rutilus* (L.) and Chub, *Squalis cephalus* (L.) in Willow Brook, Northamptonshire. *Journal of Fish Biology,* **1(1):** 59 - 82.

Crumby W. D, M. A Webb, F. J. Bulow and H. J. Cathey (1990). Changes in biotic integrity of a river in North-Central Tennessee. *Transactions of the American Fisheries Society,* **119(5):** 885 - 893.

Cummins K. W. (1974). Structure and function of stream ecosystems. *BioScience,* **24:** 631 - 641.

Cuvier G and A. Valenciennes (1839). *Histoire Naturelle des Poissons.* Paris: Pitois-Levrault. **Vol. 13,** 505 pp.

Davies A. W. (1977). What water quality is necessary to maintain a satisfactory coarse fishery ? *Proceedings of 8th British Coarse Fish Conference,* University of Liverpool, UK. pp. 169 - 194.

Day F. (1880-84). *The Fishes of Great Britain and Ireland.* London: Williams & Norgate. **Vol. 1,** 336 pp.; **Vol. 2,** 368 pp.

de Vlaming V. L. (1972). The effects of diurnal photoperiod treatments on reproductive function in the estuarine gobiid fish, *Gillichthys mirabilis* Cooper. *Journal of Experimental Marine Biology and Ecology,* **9:** 5 - 163.

De Vries A. L. (1980). Freezing resistance in Antarctica fishes. In: M. W. Holdgate (ed) *Antarctic Ecology I.* Academic Press, New York, USA. pp. 320 - 328.

Dearsley A. F. and J. Reeves (1977). *Investigation into the fish populations of the Stort Navigation in the Sawbridgeworth area.* Thames Water Authority, Scientific Department, Internal Report, UK, 52 pp.

Deegan L. A., J. T. Finn, S. G. Ayvazian, C.A. Ryder-Kieffer and J. Buonaccorsi (1997). Development and validation of an estuarine biotic integrity index. *Estuaries,* **20(3):** 601 - 617.

Didier J. (1997). Indice Biotique d'Integrite Piscicole pour evaluer la qualite ecologique des ecosystemes aquatiques. *These de Doctorat,* Universite Notre-Dame de la Paix, Namur, Belgique, 313 pp.

Didier J. and P. Kestemont (1996). Relationships between mesohabitats, ichthyological communities and IBI metrics adapted to a European river basin (The Meuse, Belgium). *Hydrobiologia,* **341(2):** 133 - 144.

Dill L. M. (1983). Adaptive flexibility in the foraging behaviour of fishes. *Canadian Journal of Fisheries and Aquatic Sciences*, **40:** 398 - 408.

Dionne M. and J. R. Karr (1992). Ecological monitoring of fish assemblages in Tennessee River reservoirs. In: D. H. McKenzie and V. J. McDonald (eds) *Ecological Indicators*, **Vol. 1.** Elsevier, NY, USA. pp. 259 - 281.

EA (Environment Agency, UK), LEAP (Local Environment Agency Plan) (1996). *River Thames (Buscot to Eynsham): Windrush and Evenlode: Consultation Report September 1996.* EA, Thames Region, West Area, Wallingford, Oxon, OX10 8BD, UK. Appendices A to G + 138 pp.

EA, LEAP (1997a). *Thame Valley: Consultation Report October 1997.* EA, Thames Region, West Area, Wallingford, Oxon, OX10 8BD, UK. 120 pp.

EA, LEAP (1997b). *River Thames (Buscot to Eynsham): Windrush and Evenlode: Action Plan September 1997.* EA, Thames Region, West Area, Wallingford, Oxon, OX10 8BD, UK. 53 pp.

EA, LEAP (1997c). *Soar: Consultation Report April 1997.* EA, Midlands Region, Lower Trent, Trentside, Nottingham, NG2 5FA, UK. 162 pp.

EA, LEAP (1997d). *Staffordshire Trent Valley: Consultation Report June 1997.* EA, Upper Trent, Lichfield, WS13 8RR, UK. 165 pp.

EA (1998). River Habitat Quality: the physical character of rivers and streams in the UK and Isle of Man. Report No. 2. 86 pp.

EA, LEAP (1998a). *Aire: Consultation draft June 1998.* EA, Ridings Area, Northeast Region, Leeds, LS1 2QG, UK. 43 pp.

EA, LEAP (1998b). *Nidd and Wharfe: Action plan September 1998.* Environment Agency, Northeast Region, Dales Area Office, York, YO30 4GZ, UK. 32 pp.

EA, LEAP (1998c). *Staffordshire Trent Valley: Action plan February 1998.* EA, Upper Trent, Lichfield, WS13 8RR, UK. 61 pp.

EA, LEAP (1998d). *West Midlands - Tame: Consultation Report March 1998.* EA, Upper Trent, Lichfield, WS13 8RR, UK. 171 pp.

EA, LEAP (1998e). *Thame Valley: Action Plan August 1998.* EA, Thames Region, West Area, Wallingford, Oxon, OX10 8BD, UK. 50 pp.

EA, LEAP (1998f). *Soar: Action Plan June 1998.* EA, Midlands Region, Lower Trent, Trentside, Nottingham, NG2 5FA, UK. 59 pp.

EA, LEAP (1999a). *Derbyshire Derwent: Action Plan January 1999.* EA, Lower Trent, Midlands Region, Trentside, Nottingham, NG2 5FA, UK. 68 pp.

EA, LEAP (1999b). *Idle and Torne: Consultation draft Novmber 1999.* EA, Lower Trent, Midlands Region, Trentside, Nottingham, NG2 5FA, UK. 94 pp.

EA, LEAP (1999c). *West Midlands - Tame: Action Plan March 1999.* EA, Upper Trent, Lichfield, WS13 8RR, UK. 64 pp.

EA, LEAP (1999d). *Cherwell Valley: Consultation Draft November 1999.* EA, Thames Region, West Area, Wallingford, Oxon, OX10 8BD, UK. 60 pp.

EA, LEAP (1999e). *Upper Lee: Environmental Overview May 1999.* EA, Thames Region, North East Area, Hatfield, Herts AL10 9EX, UK. 90 pp.

EA, LEAP (1999f). *Upper Lee: Consultation Draft May 1999.* EA, Thames Region, North East Area, Hatfield, Herts AL10 9EX, UK. 65 pp.

EA, LEAP (2000a). *Burton, Nuneaton and Tamworth: Environmental Overview March 2000.* EA, Upper Trent, Lichfield, WS13 8RR, UK. 74 pp.

EA, LEAP (2000b). *Burton, Nuneaton and Tamworth: Action Plan August 2000.* EA, Upper Trent, Lichfield, WS13 8RR, UK. 65 pp.

EA, LEAP (2000c). *Dove: Action Plan March 2000.* EA, Upper Trent, Lichfield, WS13 8RR, UK. 59 pp.

EA, LEAP (2000d). *Staffordshire Trent Valley: Second Annual Review June 2000.* EA, Upper Trent, Lichfield, WS13 8RR, UK. 32 pp.

EA, LEAP (2000e). *Cherwell Valley LEAP June 2000.* EA, Thames Region, West Area, Wallingford, Oxon, OX10 8BD, UK. 68 pp.

EA, LEAP (2001). *Upper Lee January 2001.* EA, Thames Region, Northeast Area, Hatfield, Herts AL10 9EX, UK. 71 pp.

EA (2002). *Decade of clean-up brings best ever river and estuary quality results.* EA Doc. No. 156/1. http://www.environment-agency.gov.uk, Accessed on 12 March 2002.

Edevane J. (1994). Impacts on Fisheries in the Bedford Ouse. Unpublished *MSc Thesis*, University of Hull, UK, 81 pp.

EIFAC (European Inland Fisheries Advisory Commission) (1984) Documents presented at the symposium on stock enhancement in the management of freshwater fish. *EIFAC Technical Paper,* **42 (Suppl 1)**: 281 pp.

EIFAC. (1993). Water quality and fish health. FAO, Rome. 67 pp.

Eloronta P. and A. Eloronta (1977). Properties of water, plankton and fish in Lake Keurusselka. In: J. S. Alabaster (ed.) *Biological Monitoring of Inland Fisheries.* Applied Science, New York, USA. pp. 175 - 185.

English Nature. (1996). The impact of water abstraction on wetland SSSIs. *English Nature, Freshwater Series No. 4. August 1996.* English Nature, Peterborough, UK. 48 pp.

EU (European Union) (2000). *Establishing a Framework for Community Action in the Field of Water Policy.* Directive of the European Parliament and of the Council 2000/60/EC, **PE-CONS 3639/1/00 REV 1,** 62 pp.

FAO (Food and Agriculture Organization of the United Nations) (2000). [On-line]:- http://www.fao.org/fi, Accessed on 12 November 2000.

Fausch K. D., J. R. Karr and P. R Yant (1984). Regional application of an index of biotic integrity based on stream fish communities. *Transactions of the American Fisheries Society,* **113(1):** 39 - 55.

Fausch K. D. (1987). Development and use of the index of biotic integrity to monitor fish communities in the St. Croix National Science Riverway. Final Report (Contract PX 6000 - 6 -0380) to National Park Service, Midwest Regional Office, Omaha, Nebraska, USA.

Fausch K. D., J. Lyons, J. R. Karr and P. L. Angermeier (1990). Fish communities as indicators of environmental degradation. *American Fisheries Society Symposium,* **8:** 123 - 144.

Fauth J. E., J. Bernardo, M. Camara, W. J. Jr. Resetarits, J. Van Buskirk and S. A. McCollum (1996). Simplifying the jargon of community ecology: a conceptual approach. *American Naturalist,* **147:** 282 - 286.

Feltham M. J., J. M. Davies, B. R. Wilson, T. Holden, I. G. Cowx, J. P. Harvey and J. R. Britton (1999). Case studies of the impact of fish-eating birds on inland fisheries in England and Wales. Project Report, MAFF Project VC0106, Ministry of Agriculture, Fisheries and Food, UK. 146 pp.

Firth C. J. (1997). *900 years of the Don Fishery.* Environment Agency Report. Environment Agency, Ridings Area, Leeds LS12 6DD, UK.

Fisher T. R. (1989). Application and testing of indices of biotic integrity in Northern and Central Idaho headwater streams. Unpublished *MSc Thesis.* University of Idaho, USA.

Fore L. S, J. R. Karr and L. L Conquest (1994). Statistical properties of an index of biological integrity used to evaluate water resources. *Canadian Journal of Fisheries and Aquatic Sciences,* **51(5):** 1077 - 1087.

Ganasan V. and R. M. Hughes (1998). Application of an index of biological integrity (IBI) to fish assemblages of the rivers Khan and Kshipra (Madhya Pradesh), India. *Freshwater Biology*, **40(2):** 367 - 383.

Gatz A. J. and A. L. Harig (1993). Decline in the index of biotic integrity of Delaware Run, Ohio, over 50 years. *Ohio Journal of Science*, **93(4):** 95 - 100.

Gauch H. G. (1982). *Multivariate Analysis in Community Ecology.* Cambridge University Press, Cambridge. 298 pp.

Gee A. S., N. J. Milner and R. J. Hemsworth (1978). The production of juvenile Atlantic Salmon, *Salmo salar*, in the upper Wye, Wales. *Journal of Fish Biology*, **13:** 439 - 451.

Ghetti P. F. and G. Bonazzi (1977). A comparison between various criteria for the interpretation of biological data in the analysis of water. *Water Research,* **11:** 819 - 831.

Giles N., M. Street and R. M. Wright (1990). Diet composition and prey preference of tench, *Tinca tinca* (L.), common bream, *Abramis brama* (L.), perch, *Perca fluviatilis* L and roach, *Rutilus rutilus* (L.), in two contrasting gravel pit lakes: potential trophic overlap with wildfowl. *Journal of Fish Biology*, **37:** 945 - 957.

Giles N. (1994). *British Freshwater Fish.* Kingfisher Books, London. 194 pp.

Goldstein R. M., T. P. Simon, P.A. Bailey, M. Ell, E. Pearson, K. Schmidt and J. W. Enblom (1994). Concepts for an index of biotic integrity for the streams of the Red River of the North basin. In: *Proceedings of the North Dakota Water Quality Symposium,* March 30 - 31, 1994, Fargo, ND, USA. [On-line]:- http://wwwmn.cr.usgs.gov/redn/abs/awra.html Accessed on 03 December 1998. pp 169 - 180.

Goldstein R. M. and T. P. Simon (1999). Toward a United Definition of Guild Structure for Feeding Ecology of North American Freshwater Fishes. In: T. P. Simon (ed) *Assessing the Sustainability and Biological Integrity of Water Resources Using Fish Communities.* CRC Press LLC, Boca Raton, USA. pp. 123 - 220.

Goodman J. (1975). The theory of diversity-stability relationships in ecology. *Quarterly Review of Biology*, **50:** 237 - 266.

Gore J. A. and G. E. Petts (eds) (1989). *Alternatives in Regulated River Management.* CRC Press LLC, Boca Raton, Florida, USA.

Gorman O. T and J. R. Karr (1978). Habitat structure and stream fish communities. *Ecology*, **59(3):** 507 - 515.

Goulding M. (1980). *The Fishes and The Forest.* University of California Press, Berkeley, 225 pp.

Grossman G. D., P. B. Moyle and J. O. Whitaker Jr. (1982). Stochasticity in structural and functional characteristics of an Indiana stream fish assemblage: a test of community theory. *American Naturalist*, **120:** 423 - 454.

Gunther A. (1862). *Catalogue of the Fishes in the British Museum.* London: British Museum. **Vol. 4**, 534 pp.

Gutierrez M. A. R. (1994). Utilizacion de la Ictiofauna como Indicadora de la Integridad Biotica de los Rios Guache y Guanare, Estado Portuguesa, Venezuela. *MSc Thesis*, Universidad Nacional Experimental de los Llanos Occidentales 'Ezequiel Zamora', Guanare, Venezuela. 174 pp.

Halliwell D. B., R. W. Langdon, R. A. Daniels, J. P. Kurtenbach and R. A. Jacobson (1999). Classification of Freshwater Fish Species of the Northeastern United States for Use in the Development of indices of Biological Integrity, with Regional Applications. In: T. P. Simon (ed) *Assessing the Sustainability and Biological Integrity of Water Resources Using Fish Communities.* CRC Press LLC, Washington DC, USA. pp 301 - 337.

Harper D., C. Smith, P. Barham and R. Howell (1995). The Ecological Basis for the Management of the Natural River Environment. In: D. M. Harper & A. J. D. Ferguson (eds). *The Ecological Basis for River Management.* pp. 219 - 238.

Harris J. H. (1995). The use of fish in ecological assessments. *Australian Journal of Ecology,* **20(1):** 65 - 80.

Hartley P. H. T. (1947). The natural history of some British Freshwater Fishes. *Proceedings of the Zoological Society of London,* **117:** 129 - 206.

Harvey J. P. (1996). The evaluation and development of electric fishing gears and sampling strategies for the assessment of fish stocks in large water bodies. Unpublished *PhD Thesis.* University of Hull, UK. 291 pp.

Harvey J. P. and I. G. Cowx (1996). Electric fishing for the assessment of fish stocks in large rivers. In: I. G. Cowx (ed.) *Stock Assessment in Inland Fisheries.* Fishing News Books, Oxford, UK. pp 11 - 26.

Hawkes H. A. (1975). River zonation and classification. In: B. A Whitton (ed.) *River Ecology,* Blackwell Scientific Publications, Oxford, UK. pp. 312 - 374.

Hawksworth D. L (ed) (1974). *The Changing Flora and Fauna of Britain.* The Systematics Association Special **Vol. 6**. Academic Press, London, UK. 461 pp.

Hay C. J, B. J. Van Zyl and G. J. Steyn (1996). A quantitative assessment of the biotic integrity of the Okavango River, Namibia, based on fish. *Water South Africa,* **22(3):** 263 - 284.

Heip C. and P. Engels (1974). Comparing species diversity and evenness indices. *Journal of the Marine Biological Association of the United Kingdom,* **54:** 559 - 563.

Hellawell J. M. (1971). The autecology of the chub, Squalis cephalus (L.), at the River Lugg and Afon Llynfi. I. Age determination, population structure and growth. *Freshwater Biology,* **1:** 29 - 60.

Hellawell J. M. (1977). Change in natural and managed ecosystems: detection, measurement and assessment. *Proceedings of the Royal Society of London B.,* **197:** 31 - 57.

Hellawell J. M. (1988). River regulation and nature conservation. *Regulated Rivers: Research & Management,* **2(3):** 425 - 443.

Hendricks M. L., C. H. Hocutt and J. R. Stauffer (1980). Monitoring of fish in lotic habitats. In: C. H. Hocutt and J. R. Stauffer (eds) *Biological Monitoring of Fish.* Heath, Lexington, MA, USA. pp. 205 - 231.

Hickley, P. and K. F. Dexter (1979). A comparative index for quantifying growth in length of fish. *Fisheries Management.,* **10(4):** 147 - 151.

Hickley P. and R. G. Bailey (1982). Observations on the growth and production of chub *Leuciscus cephalus* and dace *Leuciscus leuciscus* in a small lowland river in Southeast England. *Freshwater Biology,* **12:** 167 - 178.

Hickley P. and E. North (1983). Zander threaten the Severn: true or false ? Proceedings of the 3rd British Freshwater Fisheries Conference, University of Liverpool, UK. pp 106 - 114.

Hickley P. and A. Sutton (1984). A standard growth curve for pike. *Fisheries Management,* **15(1):** 29 - 30.

Hickley P. and A. Starkie (1985). Cost effective sampling of fish populations in large waterbodies. *Journal of Fish Biology*, **27(Suppl.- A):** 151 - 161.

Hickley P. 1986. Invasion by Zander and the management of fish stocks. *Philosophical Transactions of the Royal Society of London*, **314:** 571 - 582.

Hickley P. (1996). Fish population survey methods: a synthesis. In: I. G. Cowx (ed.) *Stock Assessment in Inland Fisheries*. Fishing News Books, Oxford, UK. pp 3 - 10.

Higgs G. and G. Petts (1988). Hydrological changes and river regulation in the UK. *Regulated Rivers: Research & Management*, **2(3):** 349 - 368.

Hill M. O. (1979a). TWINSPAN – A FORTRAN *Program for Arranging Multivariate Data in an Ordered Two-way Table by Classification of the Individuals and Attributes*. Cornell University, Ithaca, New York, 90 pp.

Hill M. O. (1979b). DECORANA – A FORTRAN *Program for Detrended Correspondence Analysis and Reciprocal Averaging*. Cornell University, Ithaca, New York, USA, 52 pp.

Hillary J. (1999). A scoping study to address issues on coarse fish and fisheries in England and Wales. *R & D Technical Report* W 137, Environment Agency, Bristol, UK, 117 pp.

Hilsenhoff W. L. (1977). Use of arthropods to evaluate water quality of streams. Department of Natural Resources, Madison, Wisconsin, USA. *Technical Bulletin Number* **100:** 15 pp.

Hilsenhoff W. L. (1987). An improved biotic index of organic stream pollution. *Great Lakes Entomologist*, **20:** 31 - 39.

Hocutt C. H. (1981). Fish as indicators of biological integrity. *Fisheries*, **6(6):** 28 - 31.

Hocutt C. H., P. N. Johnson, C. Hay and B. J. van Zyl (1994). Biological basis of water quality assessment: the Kavango River, Namibia. *Reviews Hydrobiologie Tropicale*, **27(4):** 361 - 384.

Hodgson N. B. (1945). *Freshwater Fishes of the British Isles*. John Crowther Ltd., Reading, UK. 59 pp.

Hodgson B. P. (1993). The ecology and management of the coarse fish populations of the Lower Welsh Dee. Unpublished *PhD Thesis*, University of Liverpool, UK. 192 pp.

Holcik J. (1984). Some considerations on the role of introduced species of fish in the management on inland fisheries. *EIFAC Technical Paper*,. **42 (Suppl. 2):** 488 - 495.

Holcik J. (1991). Fish introductions in Europe with particular reference to its central and eastern part. *Canadian Journal of Fisheries and Aquatic Sciences*, **48(Suppl. 1):** 13 - 23.

Homer M. (1976). Seasonal abundance, biomass, diversity and trophic structure of fish in a salt-marsh tidal creek affected by a coastal power plant. In: G. W. Esch and R. W. McFarlane (eds) *Thermal Ecology II*, US Energy Research, Development and Administration, Washington DC, USA. pp. 259 - 267.

Horoszewicz L. (1973). Lethal and disturbing temperature in some fish species from lakes with normal and artificially elevated temperature. *Journal of Fish Biology*, **5:** 165 - 181.

Horwitz R. J. (1978). Temporal variability patterns and the distributional patterns of stream fishes. *Ecological Monographs*, **48:** 307 - 321.

Huet M. (1949). Apercu dés relations entre la pente et les populations des eaux courantes. *Schweizerische Zeitschrift fur Hydrologie*, **11:** 333 - 351.

Huet M. (1954). Biologie, profils en long et en travers des eaux courantes. *Bulletin Francais de Pisciculture*, **175:** 41 - 53.

Huet M. (1959). Profiles and biology of Western European streams as related to fish management. *Transactions of the American Fisheries Society*, **88(3):** 155 - 163.

Hughes S. and S. Morley (2000). Aspects of fisheries and water resources management in England and Wales. *Fisheries Management and Ecology*, **7(1-2)**: 75 - 84.

Hughes R. M., D. P Larsen and J. M. Omernik (1986). Regional reference sites: a method for assessing stream potentials. *Environmental Management*, **10:** 629 - 635.

Hughes R. M. and J. R. Gammon (1987). Longitudinal changes in fish assemblages and water quality in the Willamette River, Oregon. *Transactions of the American Fisheries Society*, **116:** 196 - 209.

Hughes R. M. (1995). Defining acceptable biological status by comparing with reference conditions. In: W. S Davis and T. P. Simon (eds) *Biological Assessment and Criteria: Tools for Water Resource Planning and Decision Making*. Lewis Publishers. Boca Raton, FL, USA. pp. 31 - 47.

Hughes R. M., P. R. Kaufmann, A. T. Herlihy, T. M. Kincaid, L. Reynolds and D. P Larsen (1998). A process for developing and evaluating indices of fish assemblage integrity. *Canadian Journal of Fisheries and Aquatic Sciences*, **55(7):** 1618 - 1631.

Hughes R. M. and T. Oberdorff (1999). Applications of IBI concepts and metrics to waters outside the United States and Canada. In: T. P. Simon (ed.) *Assessing the sustainability and Biological Integrity of Water Resources Using Fish Communities*. CRC press, Lewis Publishers, Boca Raton, FL, USA. pp. 79 - 93.

Hugueny B., S. Camara, B. Samoura and M. Magassouba (1996). Applying an index of biotic integrity of fish assemblages in a West African river. *Hydrobiologia*, **331(1-3):** 71 - 78.

Hurlbert S. H. (1971). The nonconcept of species diversity: a critique and alternative parameters. *Ecology*, **52:** 577 - 586.

Illies J. and L. Botosaneanu (1963). Problémes et methodes de la classification et de la zonation ecologique des eaux courantes considerees surtout du point de rue faunistique. *Mitteilungen der Internationalen Vereinigung fur theoretische und angewandte Limnologie,* **12**: 1 - 57.

Jacklin T. E. (1996). The River Trent: Is it too clean ? In: *Proceedings of the Inland Fisheries Management conference 1996*, **26:** 129 - 142.

Jenkins J. T. (1925). *The Fishes of the British Isles.* Frederick Warne & Co. Ltd., London, UK. 408 pp.

Jennings M. J., J. Lyons, E. E. Emmons, G. R. Hatzenbeler, M. A. Bozek, T. D. Simonson, T. D. Beard Jr. and D. Fago (1999). Towards the Development of an Index of Biotic Integrity for Inland Lakes in Wisconsin. In: T. P. Simon (ed.) *Assessing the Sustainability and Biological Integrity of Water Resources Using Fish Communities*. CRC Press LLC, Washington DC, USA. pp 541 - 562.

Jobling M. 1995. *Environmental Biology of Fishes.* Fish and Fisheries Series 16. Chapman & Hall, London. 455 pp.

Junge C. O. and J. Libosvarsky (1965). Effect of size selectivity on population estimates based on successive removals with electrofishing gear. *Zoologicke Listy*, **14 (2):** 171 - 178.

Kaesler R. L. and E. E. Herricks (1979). Hierarchical diversity of communities of aquatic insects and fishes. *Water Resources Bulletin*, **15:** 1117 - 1125.

Karr J. R. and I. J. Schlosser (1978). Water resources and the land -water interface. *Science*, **201:** 229 - 234.

Karr J. R. (1981). Assessment of biotic integrity using fish communities. *Fisheries (Bethesda),* **6(6):** 21 - 27.

Karr J. R. and D. R Dudley (1981). Ecological perspective on water quality goals. *Environmental Management,* **5(1):** 55 - 68.

Karr J. R., K. D Fausch, P. L. Angermeier, P. R Yant and I. J. Schlosser (1986). Assessing biological integrity in running waters: a method and its rationale. *Illinois Natural History Survey, Special Publication* **5:** 28 pp.

Karr J. R. (1991). Biological integrity: a long-neglected aspect of water resource management. *Ecological Applications,* **1(1):** 66 - 84.

Karr J. R. and E. W. Chu (1997). *Biological Monitoring and Assessment: Using Multimetric Indexes Effectively.* EPA 235-R97-001. US Environmental Protection Agency, Washington DC, USA. 32 pp.

Keast A. (1985). The piscivore feeding guild of fishes in small freshwater ecosystems. *Environmental Biology of Fishes,* **12 :** 119 - 129.

Keenleyside M. H. A. (1979). *Diversity and Adaptation in Fish Behaviour.* Springer-Verlag. Berlin, Germany.

Kell L. (1991). A comparison of methods used for coarse fish population estimation. In: I. G. Cowx (ed.) *Catch Effort Sampling Strategies.* Fishing News Books, Oxford, UK. pp. 184 - 200.

Kempton R. A. (1979). Structure of species abundance and measurement of diversity. *Biometrics,* **35:** 307 - 322.

Kerans B. L and J. R. Karr (1994). Development and testing of a benthic index of biotic integrity (B-IBI) for rivers of the Tennessee Valley. *Ecological Applications,* **4(4):** 768 - 785.

Kestemont P., J. Didier, E. Depiereux and J.C. Micha (2000). Selecting ichthyological metrics to assess river basin ecological quality. *Archiv fur Hydrobiologie,* **121 (3-4) Suppl:** 321 - 348.

Kestemont P., J. Didier, F. Dewandel and J.-Cl. Micha (1998). Preliminary results on the adaptation of a biotic index of fish integrity to a European river basin (The Meuse, Belgium). *Verhandlungen Internationale Vereinigung fur Theorestiche und angewandte Limnologie,* **26:** 1243 - 1244.

Killingbeck A., E. Hopkins, J. Perkins, E. Goddard, S. Carter, V. Lewis, S. Hughes and R. Preston (1993). *River Evenlode Fisheries Survey 1992/1993.* NRA Report, Thames Region, Thames West Area, UK, 126 pp.

Killingbeck A., A. Irvine, E. Hopkins, J. Perkins and S. Simonds (1994). *River Windrush Fisheries Survey 1993/1994.* NRA Report, Thames Region, Thames West Area, UK, 127 pp.

Killingbeck A., R. Preston, E. Hopkins, J. Perkins, V. Shackle, F. Williams, T. Dapling and S. Hughes (1996). *River Cherwell Fisheries Survey 1994/1995.* NRA Report, Thames Region, Thames West Area, UK, 112 pp.

Koizumi N. and Y. Matsumiya (1997). Assessment of Stream Fish Habitat Based on Index of Biotic Integrity. *Japan Soc. Fish. Oceanogr.,* **61:** 144 - 156 (In Japanese with English abstract).

Kolavak W. P. (1981). Assessment and predation of impacts of effluents on communities of benthic stream macroinvertebrates. In: J. W. Bates and C. I. Webber (eds) *Ecological Assessment of effluent Impacts on Communities of indigenous Aquatic Organisms.* American Society For Testing and Materials, Philadelphia. pp. 255 - 263.

Kovach W. L. (1998). *MVSP - A MultiVariate Statistical Package for Windows, ver. 3.0.* Kovach Computing Services, Pentreath, Wales, UK. 127 pp.

Krebs C. J. (1999). *Ecological Methodology*. Second Edition. Addison Welsey Longman, Inc., Menlo Park, Canada. 620 pp.

Kreutzer R. (1984). Fish in religion and myths of Mesopotamia and Egypt. In: B. Gunda (ed.) *The Fishing Culture of the World. Studies in Ethnology: Cultural Ecology and Folklore*. **Vol. 2**. Akademia Kiado, Budapest. pp 593 - 618.

Krzanowski W. J. (1972). Techniques in multivariate analysis. In: F. G. H. Lupton, G. Jenkins and R. Johnson (eds) *The Way Ahead in Plant Breeding*. Plant Breeding Institute, Thrumpington. pp 147 - 155.

Kushlan J. A. (1976). Environmental stability and fish community diversity. *Ecology*, **57:** 821 - 825.

Lambshead P. J. D., H. M. Platt and K. M. Shaw. (1983). The detection of differences among assemblages of marine benthic species based on an assessment of dominance and diversity. *Journal of Natural History*, **17:** 859 - 874.

Langford T. F. (1979). Observations on sonic-tagged coarse fish in rivers. *Proceedings of the 1st British Freshwater Fisheries Conference*. University of Liverpool, UK. pp. 106 - 114.

Larimore R. W. (1961). Fish population and electrofishing success in a warm-water stream. *Journal of Wildlife Management*, **25:** 1 - 12.

Leeming J. B. (1967). The biology of some coarse fish of the River Welland. Unpublished *PhD Thesis*. Department of Zoology, University of Liverpool, UK. 210 pp.

Leidy R. A and P. L. Fielder (1985). Human disturbance and patterns of fish species diversity in the San Francisco Bay drainage, California. *Biological Conservation*, **33(3):** 247 - 267.

Lelek A. (1987). *The Freshwater Fishes of Europe*. Vol. 9: Threatened Fishes of Europe. AULA-Verlag: Wiesbaden, Germany. 343 pp.

Leonard P.M. and D. J. Orth (1986). Application and testing of an index of biotic integrity in small, coolwater streams. *Transactions of the American Fisheries Society*, **115(3):** 401 - 414.

Lester W. F. (1975). Polluted river: River Trent, England. In: B. A. Whitton (ed.) *River Ecology*, Blackwell Scientific Publications, Oxford, UK. pp. 489 - 513.

Lewin J (ed.) (1981). *British Rivers*. George Allen & Unwin Ltd., London. 216 pp.

Lewis V. (1991). *River Cherwell Fisheries Survey 1988/89*. NRA, Thames region, Thames West area, UK.

Lewis V., A. Killingbeck, D. Willis, E. Hopkins and W. Teale (1992). *River Thame Fisheries Survey 1990*. National Rivers Authority, Thames Region, Upper Thames Area, UK. 106 pp.

Li H. W., C. B. Schreck, C. E. Bond and E. Rexstad (1987). Factors influencing changes in fish assemblages of Pacific Northwest streams. In: Matthews W. J. & D. C. Heins (eds) *Community and Evolutionary Ecology of North American Stream Fishes*. University of Oklahoma Press, Oklahoma. pp. 193 - 202.

Liang S. H. and B. W. Menzel (1997). A new method to establish scoring criteria of the index of biotic integrity. *Zoological Studies*, **36(3):** 240 - 250.

Lobb M. D. and D. J. Orth (1991). Habitat use by an assemblage of fish in a large warmwater stream. *Transactions of the American Fisheries Society*, **120:**65 - 78.

Lyons J. (1992). Using the index of biotic integrity (IBI) to measure environmental quality in warmwater streams of Wisconsin. Forest Service, United States Department of Agriculture, St. Paul, Minnesota, USA. *General Technical Report*, NC-149, 54 pp.

Lyons J., S. Navarro-Perez, P.A. Cochran, E. Santana C. and M. Guzman-Arroyo (1995). Index of biotic integrity based on fish assemblages for the conservation of streams and rivers in West-Central Mexico. *Conservation Biology*, **9(3)**: 569 - 584.

Lyons J., L. Wang and T. D. Simonson (1996). Development and validation of an index of biotic integrity for coldwater streams in Wisconsin. *North American Journal of Fisheries Management*, **16(2)**: 241 - 256.

Lyons J., A. Gutierrez-Hernandez, E. Diaz-Pardo, E. Soto-Galera, M. Medina-Nava and R. Pineda-Lopez (2000). Development of a preliminary index of biotic integrity (IBI) based on fish assemblages to assess ecosystem condition in the lakes of central Mexico. *Hydrobiologia*, **418**: 57 - 72.

Lyons J., R. R. Piette and K. W. Niermeyer (2001). Development, Validation, and Application of a Fish-Based Index of Biotic Integrity for Wisconsin's Large Warmwater Rivers. *Transactions of the American Fisheries Society*, **130(6)**: 1077 - 1094.

Magurran A. E. (1988). *Ecological Diversity and Its Measurement.* Croom Helm, London, UK. 179 pp.

Magurran A. E. (1991). *Ecological Diversity and Its Measurement.* 3rd Edition. Croom Helm, London, UK. 192 pp.

Mahon R. (1980). Accuracy of catch-effort methods for estimating fish density and biomass in streams. *Environmental Biology of Fishes*, **5**: 343 - 360.

Mahon R. (1984). Divergent structure in fish taxocenes of North Temperate streams. *Canadian Journal of Fisheries and Aquatic Sciences*, **41**: 330 - 350.

Maitland P. S. (1972). *A key to the Freshwater Fishes of the British Isles with notes on their distribution and ecology.* Scientific Publication of the Freshwater Biological Association, **No. 27**, 153 pp.

Maitland P. S. (1979). The status and conservation of rare freshwater fishes in the British Isles. *Proceedings of the 1st British Freshwater Fisheries Conference*, University of Liverpool, UK. pp. 237 - 248.

Maitland P. S. and R. N. Campbell (1992). *Freshwater Fishes of the British Isles.* Harper Collins Publishers, London, UK. 368 pp.

Mann R. H. K. (1974). Observations on the age, growth, reproduction and food of the dace *Leuciscus leuciscus* (L.) in two rivers in southern England. *Journal of Fish Biology*, **6**: 237 - 253.

Mann R. H. K. (1976). Observations on the age, growth, reproduction and food of the pike, *Esox lucius* (L.) in two rivers in southern England. *Journal of Fish Biology*, **8**: 179 - 197.

Mann R. H. K. and T. Penczak (1986). Fish production in rivers: a review. *Polskie Archwum Hydrobiologia*, **33**: 233 - 247.

Mann R. H. K. (1991). Growth and production. In: I. J. Winfield and J. S. Nelson (eds) *Cyprinid Fishes: Systematics, biology and exploitation*, Fish and Fisheries Series 3. Chapman & Hall, London. pp. 457 - 482.

Mann R. H. K. (1995). Natural Factors Influencing Recruitment Success in Coarse Fish Populations. In: D. M. Harper and A. J. D. Ferguson (eds) *The Ecological Basis for River Management*, John Wiley & Sons Ltd., Chichester, England. pp. 339 - 348.

Mann R. H. K. (1996). Environmental requirements of European non-salmonid fish in rivers. *Hydrobiologia*, **323**: 223 - 235.

Maret T. R. (1999). Characteristics of Fish Assemblages and Environmental Conditions in Streams of the Upper Snake River Basin in Eastern Idaho and Western Wyoming. In: T. P. Simon (ed.) *Assessing the Sustainability and Biological Integrity of Water Resources Using Fish Communities*. CRC Press LLC, Washington, D. C., USA. pp 273 - 299.

Margalef R. (1951). Diversidad de especies en las comunidades naturales. *Publnes. inst. Biol. apl., Barcelona*, **6:** 59 - 72.

Margalef R. (1972). Homage to Evelyn Hutchinson, or why is there an upper limit to diversity. *Trans. Connect. Acad. Arts Sci.*, **44:** 211 - 235.

Marlborough D. (1970). The status of the burbot *Lota lota* (L.) Gadidae in Britain. *Journal of Fish Biology*, **2:** 217 - 222.

Mason W. T. Jr. (ed.) (1978). *Methods for the assessment and prediction of mineral mining impacts on aquatic communities: a review and analysis*. Workshop Proceedings. FWS/OBS-28/30. U.S. Fish and Wildlife Service, USDI, Harpers Ferry, West Virginia. 157 pp.

Mason W. T. Jr., P. A. Lewis and C. I. Webber (1985). An Evaluation of benthic macroinvertebrates biomass methodology. *Environmental Monitoring and Assessment*, **5:** 399 - 422.

Mathews C. P. (1971). Contribution of young fish to total production of fish in the River Thames near Reading. *Journal of Fish Biology*, **3:** 157 - 180.

May R. M. (1975). Patterns of species abundance and diversity. In: M. L. Cody & J. M. Diamond (eds) *Ecology and Evolution of Communities*. Harvard University, Cambridge, MA. pp. 81 - 120.

Meire, P. M. and J. Dereu (1990). Use of the Abundance / Biomass Comparison method for detecting environmental stress: some considerations based on intertidal macrozoobenthos and bird communities. *Journal of Applied Ecology*, **27:** 210 - 223.

Metcalfe-Smith J. L. (1994). Biological Water Quality Assessment of Rivers: Use of macroinvertebrate communities. In: P. Calow and G. E. Petts (eds) *The Rivers Handbook: Hydrological and Ecological Principles*; Vol. II. Blackwell Scientific Publications, Oxford, UK. pp 144 - 170.

Miller P. J. and M. J. Loates (1997). *Fish of Britain & Europe*. HarperCollins Publishers, London, UK. 288 pp.

Miller D. L., P.M. Leonard, R. M. Hughes, J. R. Karr, P. B. Moyle, L. H. Schrader, B. A. Thompson, R. A. Daniels, K. D. Fausch, G. A. Fitzhugh, J. R. Gammon, D. B Halliwell, P. L. Angermeier and D. J. Orth (1988). Regional applications of an index of biotic integrity for use in water resource management. *Fisheries (Bethesda)*, **13(5):** 12 - 20.

Minns C. K., V. W. Cairns, R. G. Randall and J. E. Moore (1994). An index of biotic integrity (IBI) for fish assemblages in the littoral zone of Great Lake's area of concern. *Canadian Journal of Fisheries and Aquatic Sciences*, **51(8):** 1804 - 1822.

Moon N. and G. Souter (1994). Socio-economic review of angling 1994. *Fisheries Technical Report* No. **5**, National Rivers Authority (NRA), Bristol, UK. 31 pp.

Moss B. (1988). *Ecology of Freshwaters : Man and Medium*. Blackwell Scientific Publications, Oxford, UK. 417 pp.

Moss B. (1998). *Ecology of Freshwaters*. 3rd edition. Blackwell Scientific Publications, Oxford, UK. 560 pp.

Mount D. I. (1985). Scientific problems in using multispecies toxicity tests for regulatory purpose. In: J. Cairns, Jr. (ed.) *Multispecies toxicity testing*. Pergamon, NY, USA. pp 13 - 18.

Moyle P. B. and F. R. Senanayake (1984). Resource partitioning among the fishes of rainforest streams in Sri Lanka. *Journal of Zoology, London*, **202:** 195 - 223.

Moyle P. B., L. R. Brown and B. Herbold (1986). Final Report on "*Development and Preliminary tests of indices of biotic integrity for California*". Report to US Environmental Protection Agency, Corvallis Environmental Research Laboratory, Corvallis, Oregon, USA. 22 pp.

Mundahl N. D. and T. P. Simon (1999). Development and Application of an Index of Biotic Integrity for Coldwater Streams of the Upper Midwestern United States. In: T. P. Simon (ed.) *Assessing the Sustainability and Biological Integrity of Water Resources Using Fish Communities*. CRC Press LLC, Washington DC, USA. pp 383 - 415.

Murphy P. M. (1978). The temporal variability in biotic indices. *Environmental Pollution*, **17:** 227 - 236.

Muus B. J. (1971). *Freshwater Fish of Britain and Europe*. Wm Collins Sons and Co. Ltd., London, UK. 222 pp.

Naiman R. J., H. Decamps, J. Pastor and C. A. Johnston (1988). The potential importance of boundaries to fluvial ecosystems. *Journal of the North American Benthological Society*, **7(4):** 289 - 306.

Naismith I. A. and B. Knights (1994). Fyke-netting and coarse fisheries in lowland Britain: practical advice for fishery owners and managers. *Fisheries Management and Ecology*, **1:** 107 - 116.

Nelson J. S. (1994). *Fishes of the World*. 3rd Edition. John Wiley & Sons, New York, USA, 628 pp.

NERC (Natural Environmental Research Council) (1996). *Hydrological data, United Kingdom, 1995 Year Book*. Institute of Hydrology, Oxon, UK. 176 pp.

Nielsen L. A. and D. L. Johnson (eds). 1983. *Fisheries Techniques*. American Fisheries Society, Bethesda, MD, USA.

Niemela S., E. Pearson, T. P. Simon, R. M. Goldstein and P. A. Baily (1999). Development of an Index of Biotic Integrity for the Species-Depauperate Lake Agassiz Plain Ecoregion, North Dakota and Minnesota. In: T. P. Simon (ed.) *Assessing the Sustainability and Biological Integrity of Water Resources Using Fish Communities*. CRC Press LLC, Washington DC, USA. pp 339 - 366.

Norman J. R. (1943). *Fishes of Britain's Rivers and Lakes*. Penguin Books Ltd., London, UK. 48 pp.

Norris R. H. (1995). Biological monitoring: the dilemma of data analysis. *Journal of the North American Benthological Society*, **14:** 440 - 450.

NRA (National Rivers Authority) (1986). *River Windrush Survey 1986*. Thames Water Report, Thames Water Authority, UK, 112 pp.

NRA (1994a). National Angling Survey 1994. *Fisheries Technical Report* **5**, NRA, Bristol, UK, 31 pp.

NRA (1994b). *The NRA National Fisheries Classification Scheme: A Guide for Users*. NRA, Newcastle upon Tyne, NE85 4ET, UK. R & D Note 206, 102 pp.

Oberdorff T. and R. M. Hughes (1992). Modification of an index of biotic integrity based on fish assemblages to characterize rivers of the Seine Basin, France. *Hydrobiologia*, **228(2):** 117 - 130.

Oberdorff T. and J. P Porcher (1994). An index of biotic integrity to assess biological impacts of salmonid farm effluents on receiving waters. *Aquaculture*, **119(2-3):** 219 - 235.

Oberdorff T., J. F. Guegan and B. Hugeueny (1995). Global scale patterns of fish species richness in rivers. *Ecography*, **18:** 345 - 352.

Oberdorff T. (1996). Reseau Hydrobiologique et Piscicole. Synthese des Donnees 1995 sur le Bassin Seine Normandie. Final report to the Agence de I'Eau Seine-Normandie, Conseil Superieur de la Peche, Compiegne, France, 212 pp.

O'Hara K. and T. R. Williams (1991). Analysis of catches from the British National Angling Championships.. In: I. G. Cowx (ed.) *Catch Effort Sampling Strategies: Their application in Freshwater Fisheries Management.* Fishing News Books, Oxford, UK. pp. 214 - 222.

Ohio Environmental Protection Agency. (1988). *Biological criteria for the protection of aquatic life.* Ohio Environmental Protection Agency, Division of Water Quality Monitoring and Assessment, Surface Water Section, Columbus, USA. **Vols. 1-3** : 343 pp.

Omernik J. M. (1987). Ecoregions of the United States. *Annales of the Association of American Geographers,* **77:** 118 - 125.

OU (Open University, UK) (1972). *Rivers and Lakes.* The Open University Press, Bucks, UK. 72 pp.

OVB (Organisate ter veretering van de Binnenvisserij) (1988). *Manual: Fish species:* Organization for Improvement of the Inland Fisheries. Nieuwegein The Netherlands. (in Dutch).

Paloumpis A. A. (1958). Responses of some minnows to flood and drought conditions in an intermittent stream. *Iowa State Journal of Science,* **32:** 547 - 561.

Pan Y., R. J. Stevenson, B. H. Hill, A. T. Herlihy and G. B. Collins (1996). Using diatoms as indicators of ecological conditions in lotic systems: a regional assessment. *Journal of the North American Benthological Society,* **15:** 481 - 494.

Patric R. (1975). Stream communities. In: M. L. Cody and J. M. Diamond (eds) *Ecology and Evolution of Communities.* Harvard University Press, Cambridge, Massachusetts. pp. 445 - 459.

Patric R. (1973). Use of algae, especially diatoms, in the assessment of water quality. *ASTM Special Technical Publication* No. **528.**

Peet R. K. (1974). The measurement of species diversity. *Annual Review of Ecology and Systematics,* **5:** 285 - 307.

Penczak T. and H. Jakubowski (1990). Drawbacks of electric fishing in rivers. *In*: I. G. Cowx (ed.) *Developments in Electric Fishing.* Fishing News Books, Blackwell Scientific Publications, Oxford, UK. pp. 115 - 122.

Pennycuik L. (1971). Quantitative effects of three species of parasites on a population of Three-Spined Stickleback, *Gasterosteus aculeatus* L. *Journal of Zoology London,* **165:** 143 - 162.

Persson L. (1994). Natural shifts in the structure of fish communities: mechanisms and constraints on perturbation sustenance. In: I. G. Cowx (ed.) *Rehabilitation of Freshwater Fisheries.* Fishing News Books, Blackwell Scientific Publications, Oxford, UK. pp. 421 - 434.

Petts G. E. (1978). The Adjustment of Channel Capacity Downstream from Reservoirs in Great Britain. Unpublished *PhD Thesis.* University of Southampton, UK. 224 pp.

Petts G. E. (1980). Long term consequences of upstream impoundment. *Environmental Conservation,* **7(4):** 325 - 332.

Petts G. E. (1984). *Impounded Rivers: Perspectives for Ecological Management.* John Wiley & Sons, Chichester, UK. 326 pp.

Petts G. E. (1988). Regulated rivers in the United Kingdom. *Regulated Rivers: Research & Management,* **2(3):** 201 - 220.

Philippart J. -C. (1987). Demographie, conservation et restoration du barbeau fluviatile, *Barbus barbus* (L.) (Teleostei: Cyprinidae) dans la Meuse et ses affluents. Quinze annees de recherches. *Annales de Societe Royale Zoologie Belgique*, **117:** 9 - 62.

Pianka E. R. (1970). 'On r - and K - selection'. *American Naturalist*, **104:** 592 - 597.

Pielou E. C. (1969). *An Introduction to Mathematical Ecology.* Wiley-Interscience. New York, USA.

Pielou E. C. (1975). *Ecological Diversity.* John Wiley and Sons Inc., New York, USA. 165 pp.

Pilcher M. W., R. Tyner, N. Buck and N. Sampson (1991). *River Stort Fisheries Survey 1990/1991.* National Rivers Authority Report, Thames Region, Thames East area, UK, 119 pp.

Pincher C. (1947). *A Study of Fishes.* Herbert Jenkins Ltd., London, 304 pp.

Pires A. M., I. G. Cowx and M. M. Coelho (1999). Seasonal changes in the fish community structure of intermittent streams in the middle reaches of the Guadiana basin (Portugal). *Journal of Fish Biology*, **54:** 235 - 247.

Pitcher T. J. and P. J. Hart (1982). *Fisheries Ecology.* Chapman & Hall, London, UK. 414 pp.

Pitts C. S. (1994). Inter-specific hybridization in the fish family Cyprinidae. Unpublished *PhD Thesis.* University of Hull, UK. 103 pp.

Preston R. L., A. Killingbeck, E. Hopkins and J. Perkins (1996). *Kennet tributaries fisheries survey 1994-1995.* NRA Report, Thames Region, Thames West area, UK. 127 pp.

Punchard N. T., M. R. Perrow and A. J. D. Jowitt (2000). Fish habitat associations, community structure, density and biomass in natural and channelised lowland streams in the catchment of the River Wensum, UK. In: I. G. Cowx (ed.) *Management and Ecology of River Fisheries.* Fishing News Books, Oxford, UK. pp. 143 - 157.

Pygott J. R. , K. O'Hara and J. W. Eaton (1990). Fish community structure and management in navigated British canals. In: W. L. T. van Densen, B. Steinmetz & R. H. Hughes (eds) *Management of freshwater fisheries.* Proceedings of a symposium organised by the European Inland Fisheries Advisory Commission, Goteborg, Sweden, 31 May - 3 June 1988. *Pudoc. Wageningen.* pp. 547 - 557.

Rahman A. K. A. (1989). *Freshwater Fishes of Bangladesh.* Published by Zoological Society of Bangladesh, Dhaka. 364 pp.

Rankin E. T and C. O. Yoder (1999). Adjustments to the Index of Biotic Integrity: A Summary of Ohio Experiences and Some Suggested Modifications. In: T. P. Simon (ed.) *Assessing the Sustainability and Biological Integrity of Water Resources Using Fish Communities.* CRC Press LLC, Washington DC, USA. pp 625 - 637.

Rees H. L., D. C. Moore, T. H. Pearson, M. Elliott. M. Service, J. Pomfret and D. Johnson (1990). Procedures for the Monitoring of Marine Benthic Communities at UK Sludge Sites. Department of Agriculture and Fisheries for Scotland, Aberdeen, UK.

Regan C. T. (1917). *Guide to the British Fresh-Water Fishes.* Department of Zoology, British Museum (Natural History), London SW, 39 pp.

Reice S. T., R. C. Wissmar and R. J. Naiman (1990). Disturbance regimes, resilience, and recovery of animal communities and habitats in lotic ecosystems. *Environmental Management*, **14:** 647 - 660.

Resh V. and J. D. Unzicker (1975). Water quality monitoring and aquatic organisms: the importance of species identifications. *Journal of the Water Pollution Control Federation*, **47:** 9 - 19.

Rexstad E. A., D. D. Miller, C. H. Flather, E. M. Anderson, J. H. Hupp and D. R. Anderson (1988). Questionable multivariate statistical inference in wildlife habitat and community studies. *Journal of Wildlife Management*, **52:** 794 - 798.

Reynoldson T. B. and J. L. Metcalfe-Smith (1992). An overview of the assessment of aquatic ecosystem health using benthic invertebrates. *Journal of Aquatic Ecosystem Health*, **1:** 295 - 308.

Ribeiro M. C. L. B., M. Petrere Jr. and A. A Juras (1995). Ecological integrity and fisheries ecology of the Araguaia-Tocantins River Basin, Brazil. *Regulated Rivers: Research & Management*, **11(3&4):** 325 - 350.

Richard Y. (1994). Les communautes ichtyologiques du bassin de la riviere l'Assomption et "Integrite Biotique des Ecosystemes Fluviaux" - Envirodoq n° EN940235, rapp. n° QE-89. Direction des Ecosystemes Aquatiques, Ministere de l'Environnement et de la Faune du Quebec. Sainte-Foy, Quebec, Canada, 153 pp.

Root R. B. (1967). The niche exploitation pattern of the blue-gray gnatcatcher. *Ecological Monographs*, **37:** 317 - 350.

Rosenberg D. M. and R. J. Reash (1993). The utility of benthic invertebrates in water quality monitoring. *Water Quality Bulletin*, **10:** 21 - 28.

Ross S. T. (1991). Mechanisms structuring stream fish assemblages: are there lessons from introduced species ? *Environmental Biology of Fishes*, **30:** 359 - 368.

Routledge R. D. (1979). Diversity indices: which ones are admissible. *Journal of Theoretical Biology*, **76:** 503 - 515.

Sanders R. E. (1991). Day versus night electrofishing catches from near-shore waters of the Ohio and Muskingum Rivers. *Ohio Journal of Science*, **92:** 51 - 59.

Saxby S. and V. Lewis (1982). *River Evenlode Fisheries Survey 1982.* (Reprinted in 1988). NRA, Thames region, Thames West area, UK.

Saylor C. and E. M. Scott Jr. (1987). Application of the index of biotic integrity to existing TVA data. Tennessee Valley Authority, Chattanooga, Tennessee, USA. TVA/ONRED/AWR 87/32. 31 pp.

Schaeffer D. J., W. H. Ettinger, W. J. Tucker and H. W. Kerster (1985). Evaluation of a community-based index using benthic indicator organisms for classifying stream quality. *Journal of Water Pollution Control Federation*, **57:** 167 - 171.

Schiemer F., T. Spindler, H. Wintersberger, A. Schneider and A. Chovance (1991). Fish fry associations: Important indicators for the ecological status of large rivers. *Verhandlungen Internationale Vereinigung fur Theorestiche und angewandte Limnologie*, **24:** 2497 - 2500.

Schindler D. W. (1987). Detecting ecosystem responses to anthropogenic stress. *Canadian Journal of Fisheries and Aquatic Sciences*, **44(Suppl.-1):** 6 - 25.

Schleiger S. L. (2000). Use of an Index of Biotic Integrity to Direct Effects of Land Uses on Stream Fish Communities in West-Central Georgia. *Transactions of the American Fisheries Society*, **129:** 1118 - 1133.

Schlosser I. J. (1982a). Trophic structure, reproductive success, and growth rate of fishes in a natural and modified headwater stream. *Canadian Journal of Fisheries and Aquatic Sciences*, **39:** 968 - 978.

Schlosser I. J. (1982b). Fish community structure and function along two habitat gradients in a headwater stream. *Ecological Monographs*, **52(4):** 395 - 414.

Schlosser I. J. (1985). Flow regime, juvenile abundance and the assemblage structure of stream fishes. *Ecology*, **66:** 1484 - 1490.

Schrader L. H. (1989). Use of the index of biotic integrity to evaluate the effects of habitat, flow and water quality on fish communities in three Colorado Front Range rivers. Unpublished *MSc. Thesis*. Colorado State University, USA, 120 pp.

Scott J. M., B. Csuti, J. D. Jacobi and J. E. Estes (1987). Species richness: a geographic approach to protecting future biological diversity. *BioScience*, **37:** 782 - 788.

Severn-Trent Water Authority (1983). Review of Fisheries surveys undertaken in the Trent area of Severn-Trent Water Authority during the period 1975 - 1981. Directorate of Technical Services, Fisheries Section, UK. 122 pp.

Shannon C. E. (1948). A mathematical theory of communication. *Bell Systems Tech. J.*, **27**: 379 - 423.

Shaw K. M., P. J. D. Lambshead and H. M. Platt (1983). Detection of pollution induced disturbance in marine benthic assemblages with special reference to nematodes. *Marine Ecology Progress Series*, **12:** 195 - 202.

Sheldon A.L. (1968). Species diversity and longitudinal succession in stream fishes. *Ecology*, **49(2):** 193 - 198.

Shields, F. D. Jr., S. S Knight and C. M Cooper (1995). Use of the index of biotic integrity to assess physical habitat degradation in warmwater streams. *Hydrobiologia*, **312(3):** 191 - 208.

Simon T. P. and E. B. Emery (1995). Modification and assessment of an index of biotic integrity to quantify water resource quality in Great Rivers. *Regulated Rivers: Research and Management*, **11 (3-4):** 283 - 298.

Simon T. P. and J. Lyons (1995). Application of the index of biotic integrity to evaluate water resources integrity in freshwater ecosystems. In: W. S. Davis and T. P. Simon (eds.) *Biological assessment and criteria: tools for water resources planning and decision making.* Lewis Publishers, Boca Raton, Florida, USA. pp. 243 - 262.

Simon T. P. (ed.) (1999). *Assessing the Sustainability and Biological Integrity of Water Resources Using Fish Communities.* CRC Press LLC, Washington DC, USA. 671 pp.

Simon T. P. and R. E. Sanders (1999). Applying an Index of Biotic Integrity Based on Great-River Fish Communities: Considerations in Sampling and Interpretation. In: T. P. Simon (ed.) *Assessing the Sustainability and Biological Integrity of Water Resources Using Fish Communities.* CRC Press LLC, Washington DC, USA. pp 475 - 505.

Simonson T. and J. Lyons (1995). Comparisons of catch per effort and removal procedures for sampling stream fish assemblages. *North American Journal of Fisheries Management*, **15:** 419 - 427.

Simpson E. H. (1949). Measurement of diversity. *Nature (London)*, **163:** 688.

Smith R. P (1989). Distribution and habitat requirements of chub, *Leuciscus cephalus* (L.), in several lowland rivers in eastern England. Unpublished *PhD Thesis*, University of East Anglia, UK. 202 pp.

Smogor R. A. and P. L. Angermeier (1999). Relations Between Fish Metrics and Measures of Anthropogenic Disturbance in Three IBI Regions in Virginia. In: T. P. Simon (ed.) *Assessing the Sustainability and Biological Integrity of Water Resources Using Fish Communities.* CRC Press LLC, Washington DC, USA. pp 585 - 610.

Smogor R. A. and P. L. Angermeier (2001). Determining a Regional Framework for Assessing Biotic Integrity of Virginia Streams. *Transactions of the American Fisheries Society*, **130 (1):** 18 - 35.

Sokal R. R. and C. D. Michener (1958). A statistical method for evaluating systematic relationships. *University of Kansas Science bulletin*, **38:** 1409 - 1439.

Solomon D. J. (1992). *Diversion of entrapment of fish at water intakes and outfalls.* National Rivers Authority, R& D Report 1, 51 pp.

Spellerberg I. F. (1991). *Monitoring Ecological Change.* Cambridge University Press, UK. 334 pp.

Spillman C. J. (1961). Faune de France: 65 poissons d'eau douce - Le Chevalier, Paris, 303 pp.

Sprague J. B. (1973). The ABC's of pollutant using fish. In: J. Cairns Jr. and K. L. Dickson (eds) *Biological Methods for the Assessment of Water Quality.* American Society for Testing and Materials. STP 528. Philadelphia, Pennsylvania, USA. pp 6 - 30.

Statzner B. and B. Higler. (1985). Questions and comments on the River Continuum Concept. *Canadian Journal of Fisheries and Aquatic Sciences,* **42:** 1038 - 1044.

Steedman R. J. (1988). Modification and assessment of an index of biotic integrity to quantify stream quality in Southern Ontario. *Canadian Journal of Fisheries and Aquatic Sciences,* **45(3):** 492 - 501.

Steinmetz B. (1990). Electric fishing: some remarks on its use. In: I. G. Cowx (ed.) *Developments in Electric Fishing.* Fishing News Books, Blackwell Scientific Publications, Oxford, UK. pp. 1 - 4.

Svardson G. and G. Molin (1968). Growth, Weight, and Year-class fluctuations in the pike-perch (*Lucioperca lucioperca* L.) of Lakes Hjalmaren and Malaren. *Report of the Institute for Freshwater Research. Drottningholm.* **48:** 17 - 35.

Swales S. (1988). Fish populations of a small lowland channelized river in England subject to long-term river maintenance and management works. *Regulated Rivers: Research & Management,* **2:** 493 - 506.

Templeton R. G. and A. Churchward (1990). Fisheries management practices of the Severn Trent Water Authority in England 1976 - 1986. In: W. L. T. van Densen, B. Steinmetz & R. H. Hughes (eds) *Management of freshwater fisheries.* Proceedings of a symposium organized by the European Inland Fisheries Advisory Commission, Goteborg, Sweden, 31 May - 3 June 1988. *Pudoc. Wageningen.* pp. 558 - 568.

ter Braak C. J. F. and P. F. M. Verdonschot (1995). Canonical correspondence analysis and related multivariate methods in aquatic ecology. *Aquatic Sciences,* **57:** 255 - 289.

Thames Water (1985). *River Thame Fisheries Survey 1985.* Internal report, Thames Water Authority, UK.

Thames Water (1986). *River Thame Fisheries Survey 1986.* Internal report, Thames Water Authority, UK.

Thoma R. F. (1999). Biological Monitoring and Index of Biotic Integrity for Lake Erie's Nearshore Waters. In: T. P. Simon (ed.) *Assessing the Sustainability and Biological Integrity of Water Resources Using Fish Communities.* CRC Press LLC, Washington DC, USA. pp 417 - 461.

Thompson B. A. and G. R. Fitzhugh (1986). *A Use Attainability Study: an Evaluation of fish and Macroinvertebrate Assemblages of the Lower Calcasieu River, Louisiana.* Center for Wetland Resources, Louisiana State University, USA, 143 pp.

Toham A. K. and G. G. Teugels (1999). First data on an Index of Biotic Integrity (IBI) based on fish assemblages for the assessment of the impact of deforestation in a tropical West African river system. *Hydrobiologia,* **397:** 29 - 38.

Tonn W. M. (1990). Cimate change and fish communities – a conceptual framework. *Transactions of the American Fisheries Society*, **119:** 337 - 352.

Townsend C. R. (1989). The patch dynamics concept of stream ecology. *Journal of the North American Benthological Society*, **8:** 36 - 50.

USEPA (United States Environmental Protection Agency) (1977). Interagency 316a technical guidance manual and guide for thermal effects sections of nuclear facilities environmental impact statements. U.S. Environmental Protection Agency Publication, Washington D.C. 79 pp.

USEPA (1991). *Environmental Monitoring and Assessment Program.* EMAP–Surface Waters Monitoring and Research Strategy. Fiscal Year 1991. EPA/600/3-91/022. Office of the Research and Development, Environmental Research Laboratory, Corvallis, OR, USA.

Usher M. B. (1983). Species diversity: a comment on a paper by W. B. Yapp. *Field Studies*, **5:** 825 - 832.

Vallipuram S. and J. S. Wortley (1990). Acquisition and computation of routine fisheries management data by the Anglian Water Authority, England. In: W. L. T. van Densen, B. Steinmetz & R. H. Hughes (eds) *Management of freshwater fisheries.* Proceedings of a symposium organized by the European Inland Fisheries Advisory Commission, Goteborg, Sweden, 31 May - 3 June 1988. *Pudoc. Wageningen.* pp. 569 - 581.

van Groenewoud H. (1992). The robustness of correspondence, detrended correspondence, and TWINSPAN analysis. *Journal of Vegetation Science*, **3:** 239 - 246.

Vannote R. L., G. W. Minshall, K. W Cummins, J. R. Sedell and C. E Cushing (1980). The river continuum concept. *Canadian Journal of Fisheries and Aquatic Sciences,* **37(1):** 130 - 137.

Varley M. E. (1967). *British Freshwater Fishes.* Fishing News (Books) Ltd., London, UK. 148 pp.

Verspoor E. and J. Hammer (1991). Introgressive hybridization in fishes: the biochemical evidence. *Journal of Fish Biology*, **39 (Suppl.-A):** 309 - 334.

Vibert R. (ed.) (1967). *Fishing with Electricity – Its application to Biology and Management.* London: Fishing News (Books) Ltd. 276 pp.

Walker P. A. (1994). Development of pike and perch populations after biomanipulation of fish stocks. In: I. G. Cowx (ed.) *Rehabilitation of Freshwater Fisheries.* Fishing News Books; Blackwell Scientific Publications, Oxford, UK. pp. 376 - 389.

Wallace J. B. J. W. Grubaugh and M. R. Whiles (1996). Biotic indexes and stream ecosystem processes - Results from an experimental study. *Ecological Applications*, **6(1):** 140 - 151.

Ward D., N. Holmes and P. Jose (1993). *The New Rivers and Wildlife Handbook.* Royal Society for the Protection of Birds, National Rivers Authority, & Royal Society for Nature Conservation, UK. pp. 137 - 143.

Ward R. C., J. C. Loftis and G. B. McBridge (1986). The "data-rich and information-poor syndrome" in water quality monitoring. *Environmental Management*, **10:** 291 - 297.

Warwick R. M. (1986). A new method for detecting pollution effects on marine macrobenthic communities. *Marine Biology*, **92:** 557 - 562.

Warwick R. M. and Ruswahyuni (1987). Comparative study of the structure of some tropical and temperate marine soft-bottom macrobenthic communities. *Marine Biology*, **95:** 641 - 649.

Warwick R. M., T. H. Pearson and Ruswahyuni (1987). Detection of pollution effects on marine macrobenthos: further evaluation of the species abundance / biomass method. *Marine Biology*, **95:** 193 - 200.

Washington H. G. (1984). Diversity, biotic and similarity indices: a review with special relevance to aquatic ecosystems. *Water Research*, **18:** 653 - 694.

Weisberg S. B., J. A. Ranasinghe, L.C. Schaffner, R. J. Diaz, D. M Dauer and J. B. Frithsen (1997). An estuarine benthic index of biotic integrity (B-IBI) for Chesapeake Bay. *Estuaries*, **20(1):** 149 - 158.

Welcomme R. L. (1979). *The fisheries ecology of floodplain rivers.* Longman, London, UK. 317 pp.

Welcomme R. L. (1984). International transfers of inland fish species. In: W. R. Courtenay, Jr. and J. R. Stauffer Jr. (eds) *Distribution, biology and management of exotic fishes*. Baltimore, John Hopkins University Press. pp. 22 - 40.

Welcomme R. L. (1985). *River Fisheries.* Food and Agriculture Organization of the United Nations, Rome, Italy. *FAO Fisheries Technical Report*, **262:** 330 pp.

Welcomme R. L. (1988). International introductions of inland aquatic species. *FAO Fisheries Technical Paper*, **294**. 318 pp.

Welcomme R. L. (1995). Relationship between fisheries and the integrity of river system. *Regulated Rivers : Research & Management*, **11:** 121 - 136.

Welcomme R. L. (2000). Principles and approaches for river fisheries management. In: I. G. Cowx (ed.). *Management and Ecology of River Fisheries.* Fishing News Books, Oxford, UK. pp. 331 - 345.

Welcomme R. L. (2001). *Inland Fisheries : Ecology and Management.* Fishing News Books, Blackwell Science, Oxford OX2 0EL, UK. 358 pp.

Wheeler A. (1969). *Fishes of the British Isles and North-West Europe.* London, Macmillan, 613 pp.

Wheeler A. and P. S. Maitland (1973). The scarcer freshwater fishes of the British Isles. I. Introduced species. *Journal of Fish Biology*, **5:** 49 - 68.

Wheeler A. (1974). Changes in the freshwater fish fauna of Britain. In: D. L. Hawksworth (ed.) *The changing flora and fauna of Britain.* Systematics Association (**Special Volume: 6**), Academic Press, London. pp. 157 - 178.

Wheeler A. (1977). The origin and distribution of the freshwater fishes of the British Isles. *Journal of Biogeography*, **4:** 1 - 24.

Wheeler A. (1983). *Freshwater Fishes of Britain and Europe.* Kingfisher Books Limited, London. 124 pp.

Wheeler A. (1992). A list of the Common and Scientific Names of Fishes of the British Isles. *Journal of Fish Biology*, **41 (Suppl.-A):** 37 pp.

Whittaker R. H. (1965). Dominance and diversity in land plant communities. *Science*, **147:** 250 - 260.

Whittier T. R., R. M. Hughes and D. V. Peck (2001). Comment : Test of an Index of Biotic Integrity. *Transactions of the American Fisheries Society*, **130 (1):** 169 - 172.

Williams W. P. (1967). The growth and mortality of four species of fish in the river Thames at Reading. *Journal of Animal Ecology*, **36:** 695 - 720.

Wilson J. and C. Turnbull (1993). *The Angler's Guide to Freshwater Fish : Habitat and characteristics of over 50 British fish.* Boxtree Ltd. London SE1 9PL, UK. 128 pp.

Winfield I. J., J. M. Fletcher and D. Cragg-Hine (1994). Status of Rare Fish. Vol. 2, R & D Project Record 249/8/NW. National Rivers Authority, UK. 108 pp.

Wolda H. (1981). Similarity indices, sample size and diversity. *Oecologia*, **50:** 296 - 302.

Woolland J. V., K. O'Hara and D. R Wilkinso (1977). Fish populations in the Welsh Dee, past and present. *Proceedings of the 8th British Coarse Fish Conference*, University of Liverpool, Liverpool, UK. pp. 13 - 17.

Wootton R. J. (1990). *Ecology of Teleost Fishes*. Fish and Fisheries Series 1. Chapman and Hall Ltd., London, UK. 455 pp.

WRc (Water Research Council) (1990). Fish stock management in the UK - A review. WRc, Buckinghamshire, SL7 2HD, UK, Report No. PRS 2352-M, 95 pp.

Yarrel W. (1836). *A History of British Fishes*. London, United Kingdom: John van Voorst, 408 pp.

Yoder C. O. and E. T. Rankin (1995). Biological response signatures and the area of degradation value: new tools for interpreting multimetric data. In: W. S. Davis and T. P. Simon (eds) *Biological Assessment and Criteria: Tools for Water Resource Planning and Decision Making*. Lewis, Boca Raton, FL, USA. pp. 263 - 286.

Yoder C. O. and M. A. Smith (1999). Using Fish Assemblages in a State Biological Assessment and Criteria Program: Essential Concepts and Considerations. In: T. P. Simon (ed.) *Assessing the Sustainability and Biological Integrity of Water Resources Using Fish Communities*. CRC Press LLC, Washington DC, USA. pp. 17 - 56.

Zalewski M. and T. Penczak (1981). Characterisation of the fish community of the Utrata river drainage basin, and evaluation of the efficiency of the catching methods. *Polskie Archiwum Hydrobiologii, Warszawa*, **28**: 385 - 396.

Zalewski M. and I. G. Cowx (1990). Factors affecting the efficiency of electric fishing. In: I. G. Cowx and P. Lamarque (eds) *Fishing with electricity: applications in freshwater fisheries management*. Fishing News Books, Oxford. pp. 89 - 111.

APPENDICES

Appendix 2.1 Sampling sites on the River Cherwell

Flow and site direction: North to South (S - M)

Site No.	Location/Name	NGR	Width (m)	Depth (m)	Length (m)	Area (m^2)
1	West Farndon Mill	SP532 518	2.2	0.2	140	308
2	Trafford Bridge	SP518 479	5.5	1.4	111	611
3	Slat Mill	SP472 444	5.5	0.75	124	682
4	Spiceball Park	SP459 409	5	0.8	120	600
5	Tramroad Industrial Estate	SP461 404	6.3	0.3	134	844
6	Footbridge near M40	SP476 390	7.3	0.4	107	781
7	Twyford Mill	SP486 371	7	0.5	142	994
8	Millhouse Farm	SP490 352	5.3	1	90	477
9	Sor Brook Confluence	SP493 337	7.5	1.5	123	923
10	Somerton	SP495 290	8.1	1	100	810
11	Lower Heyford	SP487 250	10	1	120	1200
12	Bunkers Hill	SP476 184	17.3	1.8	111	1920
13	Angel & Greyhound Meadows	SP523 063	14.3	1.7	128	1830

Appendix 2.1 (Continued) River Evenlode

Flow and site: North to Southeast (S - M)

Site No.	Location	NGR	Width (m)	Depth (m)	Length (m)	Area (m^2)
1	Evenlode	SP222 282	4.2	0.6	108	454
2	Oddington	SP234 265	4.3	0.8	119	512
3	Kingham	SP246 243	3.8	0.5	150	570
4	Bledington	SP253 224	5.4	0.6	166	896
5	Bruern Abbey	SP262 208	7.1	0.8	116	824
6	Lyneham	SP272 198	7.4	1	108	799
7	Shipton-under-Wychwood	SP281 183	7.4	0.6	192	1421
8	Ascott-under-Wychwood	SP297 188	9.1	1.3	102	928
9	Chadlington	SP331 206	7.2	1.4	106	763
10	Charlbury	SP354 195	8.1	0.7	149	1199
11	Finstock Station	SP368 174	7.9	0.8	105	830
12	Ashford Mill	SP386 156	6.9	0.8	102	704
13	Lower Riding Farm	SP397 159	9	0.5	86	774
14	Combe	SP407 152	10.8	0.8	133	1436
15	D/S Blenheim Sawmill	SP421 149	10.6	0.5	113	1198
16	Bladon	SP442 146	13	1.5	78	1014
17	Goose Eye Farm	SP438 121	11.4	1.4	77	878
18	Upstream of A40	SP104 446	9.2	1.2	113	1040
19	Canal Stream (Cassington)	SP454 095	8.7	0.7	200	1740
20	Mill Stream (Cassington)	SP454 095	3.8	0.8	129	490

Appendix 2.1 (Continued) River Stort

Flow and site: North to Southwest (S - M)

Site No.	Location	NGR	Width (m)	Depth (m)	Length (m)	Area (m^2)
1	Hazel End	TL501 243	6	0.7	116	696
2	Grange Paddocks	TL490 223	7.3	0.7	110	803
3	Bishops Stortford	TL490 208	12.5	1.2	100	1250
4	Spellbrook Lock	TL490 176	13.4	0.9	100	1340
5	Thorley Marsh	TL489 176	9.3	1.5	65	605
6	Tednambury Lock	TL494 168	14.6	1	100	1460
7	Tednambury Mill Overflow	TL495 168	8	0.7	122	976
8	Sawbridgeworth Lock	TL487 153	11.7	1.1	100	1170
9	Sawbridgeworth mead Ditch	TL493 158	5	0.5	180	900
10	Pishiobury Meander	TL482 139	4.5	0.9	160	720
11	Harcamlow Way	TL463 122	13.7	1.5	100	1370
12	Eastwick Lodge Farm	TL439 116	7.5	0.7	90	675
13	A414, Harlow Road	TL431 114	12	1.5	100	1200
14	Briggens	TL413 108	7.6	1.1	111	844
15	St. Albans Sand & Gravel	TL398 104	7	0.5	107	749
16	Brick Lock	TL393 096	15.5	1.2	100	1550

Appendix 2.1 (Continued) River Thame

Flow and site: North to Southwest (S - M)

Site No.	Location	NGR	Width (m)	Depth (m)	Length (m)	Area (m^2)
1	Weedon Lodge Farm	SP827 175	4.1	1	114	467
2	Stone Bridge	SP794 151	5.9	1.3	165	974
3	Lower Hartwell	SP785 145	7.8	1.4	100	780
4	Eythrope	SP777 135	9.9	1.3	95	941
5	Ridge Barn Farm	SP748 123	5.1	0.6	152	775
6	Nether Winchendon	SP732 118	8	1.6	125	1000
7	U/S Notley Abbey	SP720 093	7.7	1.4	150	1155
8	Scotgrove Brook confluence	SP704 070	10.1	1.7	134	1353
9	Shabbington West Arm	SP668 064	8.2	1.2	150	1250
10	Shabbington East Arm	SP669 064	7.6	1.6	120	912
11	Ickford	SP652 064	10.8	0.6	164	1771
12	Waterstock	SP633 056	11.7	0.3	62	725
13	Cuddesdon	SP611 033	7.1	0.6	125	888
14	Cuddesdon Mill Channel	SP610 033	7.3	1.8	122	891
15	Chippinghurst Manor	SP602 014	9.7	1	110	1067
16	Chiselhampton	SU592 987	12	0.4	137	1644
17	Drayton St. Leonard	SU598 960	13.7	0.7	101	1384
18	Dorchester	SU580 937	13.9	1	94	1307

Appendix 2.1 (Continued) River Windrush

Flow and site : North to Southeast (S - M)

Site No.	Location	NGR	Width (m)	Depth (m)	Length (m)	Area (m^2)
1	Kineton	SP097 267	2.5	0.3	60	150
2	Guiting Power	SP099 243	2.9	0.3	126	365
3	Harford Bridge	SP128 228	2.8	0.5	120	336
4	Upstream of A429	SP158 209	7.1	0.6	126	895
5	D/S Dikler Confluence	SP179 184	6.3	0.9	166	1046
6	Great Rissington	SP184 168	7.1	0.5	99	703
7	Sherborne Common	SP192 147	7.2	0.7	136	979
8	Barrington Park	SP196 135	8.2	0.8	97	795
9	Little Barrington	SP214 131	8.8	1	126	1109
10	Upton	SP243 127	10	1.4	85	850
11	Widford	SP266 115	11	0.9	107	1177
12	Asthall	SP289 115	5.8	1.1	100	580
13	Worsham	SP303 108	7.1	0.9	92	653
14	Minster Lovell	SP319 111	12.8	0.9	97	1242
15	New Mill	SP342 109	8	1.2	120	960
16	Ducklington (West Arm)	SP362 074	5.4	0.9	111	599
17	Ducklington (East Arm)	SP365 079	7.3	0.7	120	876
18	Beard Mill	SP398 063	4.3	0.7	105	452
19	Standlake STW	SP403 023	10.7	1	134	1434

Appendix 2.1 (Continued) River Anker

Flow and site: South to Northeast (S - M)

Site No.	Location	NGR	Width (m)	Depth (m)	Length (m)	Area (m^2)
1	Weddington	SP361 934	6.5	0.7	190	1235
2	Leather Mill	SP340 955	6.5	0.7	173	1125
3	Woodford Bridge	SP333 962	9	0.6	200	1800
4	Mancetter Mill	SK323 968	9	1.4	120	1080
5	Ratcliffe Bridge	SP318 986	9	0.5	210	1890
6	Fieldon Bridge	SP308 994	13	1	355	4615
7	Polesworth 1	SK265 023	17	1.4	180	3060
8	Polesworth 2	SK263 023	15	2	236	3540
9	U/S Tamworth Cowells Farm	SK217 052	13	1.5	240	3120
10	Tamworth Station Field	SK216 044	12	1	180	2160

Appendix 2.1 (Continued) River Blithe — Flow and site: North to Southeast (S - M)

Site No.	Location	NGR	Width (m)	Depth (m)	Length (m)	Area (m^2)
1	Blythe Bridge	SJ960 404	4	0.5	137	548
2	Cresswell U/S Blithe Colours	SJ975 394	4	0.4	137	548
3	Newton Crossing	SK989 384	4.5	0.4	140	630
4	Lower leigh	SK014 358	5	0.2	112	560
5	Field	SK024 333	6	0.4	169	1014
6	Burnthurst Mill	SK044 308	8	0.5	130	1040
7	Booth Bridge	SK043 280	3.9	0.4	290	1131
8	Lower Booth Farm	SK047 266	6.5	1.2	110	715
9	U/S Newton Bridge	SK048 265	6.9	0.9	125	863
10	Priory Farm	SK095 207	5	0.6	160	800
11	Hamstall Ridware	SK110 185	6.5	0.6	165	1073

Appendix 2.1 (Continued) River Blythe — Flow and site: South to Northeast (S - M)

Site No.	Location	NGR	Width (m)	Depth (m)	Length (m)	Area (m^2)
1	Cheswick Green	SP129 755	4	0.4	82	328
2	Widney Manor Rd. Bridge	SP157 775	6	0.6	130	780
3	Sandall's Bridge	SP165 791	5.5	0.7	120	660
4	Springfield House Temple Balsall (2)	SP200 765	10.4	0.7	150	1560
5	Springfield House Temple Balsall (1)	SP200 763	12	0.6	225	2700
6	U/S Eastcote Brook	SP213 801	7	1	275	1925
7	D/S Eastcote Brook	SP215 803	8	0.8	135	1080
8	Moland's Bridge	SP221 823	9.7	0.7	200	1940
9	Blythe Mill End	SP212 911	12.5	0.6	180	2250

Appendix 2.1 (Continued) River Churnet — Flow and site direction: North to South (S - M)

Site No.	Location	NGR	Width (m)	Depth (m)	Length (m)	Area (m^2)
1	Middle Hulme Bridge	SK001 603	4	0.6	100	400
2	Tittesworth Reservoir D/S	SJ994 583	7	0.4	300	2100
3	South Hillswood Farm	SJ988 579	6	0.4	80	480
4	Abbey Green Road S1	SJ979 573	12	0.6	88	1056
5	Abbey Green Farm	SJ979 572	6	0.5	112	672
6	Westwood Golf Club	SJ973 554	9	0.8	385	3465
7	U/S Leekbrook WRW	SJ979 543	5	1	400	2000
8	St. Edwards Hospital	SJ970 537	6	0.8	200	1200
9	Flint Mill Cheddleton	SJ971 527	7	1.2	260	1820
10	D/S Cheddleton WRW	SJ983 511	7.3	1	345	2519
11	Thomas Boltons Ltd.	SK025 473	9.5	1.5	195	1853
12	Whiston Bridge	SK026 465	11	1	190	2090
13	Eastwall Farm	SK037 452	9	0.8	150	1350
14	U/S Alton WRW	SK072 425	9	1	290	2610
15	D/S Alton WRW	SK082 427	10	0.8	200	2000
16	JCB Rocester	SK106 393	10	1	110	1100

Appendix 2.1 (Continued) River Cole — Flow and site: South to Northeast (S - M)

Site No.	Location	NGR	Width (m)	Depth (m)	Length (m)	Area (m^2)
1	Lowbrook Farm	SP095 758	2	0.3	115	230
2	Mill lodge	SP103 786	1.5	0.3	100	150
3	Haybarn Recreation Ground	SP117 858	6	0.4	146	876
4	Glebe Farm Recreation Ground	SP140 886	6	0.5	115	690
5	Colehall (1)	SP152 881	6.5	0.3	225	1463
6	Colehall	SP153 880	6.5	0.4	490	3185
7	Kingshurst 1	SP168 878	8.5	0.3	214	1819
8	Kingshurst 2	SP169 879	7	0.7	190	1330
9	Cook's Lane Bridge D/S	SP175 874	7	0.3	330	2310
10	Bacons end	SP184 880	9.5	0.3	200	1900
11	Coleshill Hospital 1	SP187 889	12	0.4	220	2640
12	Coleshill Hospital 2	SP188 889	7.5	0.3	220	1650
13	Coleshill 1	SP201 906	8	0.4	120	960

14	Coleshill 2	SP202 907	8	0.2	110	880

Appendix 2.1 (Continued) River Derwent — Site and flow direction: North to South (S - M)

Site No.	Location	NGR	Width (m)	Depth (m)	Length (m)	Area (m^2)
1	U/S Howden Gauging Weir	SK169 952	6.5	0.2	110	715
2	Bamford Gauging Station	SK208 823	18	0.4	208	3744
3	Bamford	SK211 822	18	0.4	170	3060
4	Grindleford	SK241 794	19	0.8	179	3401
5	Baslow	SK254 720	22	1.1	243	4853
6	D/S Baslow STW	SK255 714	23	1.2	211	4853
7	Beeley U/S site	SK252 678	26	1.4	624	16224
8	Darley dale D/S site	SK259 645	40	1.6	370	14800
9	Arkwright's Mill, Matlock	SK294 572	36	1.3	173	6228
10	Cromford	SK298 572	38	1.3	178	6764
11	Whatstandwell	SK338 530	29	2.3	1000	29000
12	Ambergate	SK346 517	25	1.2	650	16250
13	Milford	Sk352 453	28	2.5	400	11200
14	Alvaston	SK381 343	36	1.5	600	21600
15	Draycott	SK445 237	28	1.4	150	4200

Appendix 2.1 (Continued) River Idle — Flow and site: South to Northeast (S - M)

Site No.	Location	NGR	Width (m)	Depth (m)	Length (m)	Area (m^2)
1	Eaton	SK709 779	10	0.7	200	2000
2	Tiln	SK703 843	7	0.4	102	714
3	Mattersey priory	SK704 895	9	0.8	258	2322
4	Bawtry	SK655 927	15	1.5	600	9000
5	Misson	SK693 948	15	1.5	400	6000

Appendix 2.1 (Continued) River Mease — Flow and site: East to West (Source to Mouth)

Site No.	Location	NGR	Width (m)	Depth (m)	Length (m)	Area (m^2)
1	Stretton en le Field	SK307 124	4.5	1	166	747
2	Netherseal Bridge	SK286 126	4.4	0.7	210	924
3	U/S Stone Bridge	SK263 114	5.8	0.5	170	986
4	Haunton	SK235 113	8.5	1	150	1275
5	Edingale	SK214 116	8.5	1	150	1275
6	Croxall mill	SK197 129	8.5	1	228	1938
7	Croxall Bridge	SK193 139	8	1	270	2160

Appendix 2.1 (Continued) River Penk — Flow and site: South to North (S - M)

Site No.	Location	NGR	Width (m)	Depth (m)	Length (m)	Area (m^2)
1	Black Brook Nature Trail	SJ888 027	2.4	0.3	115	276
2	Allotment Site Codsall	SJ874 043	1.8	1.5	100	180
3	U/S Bill Brook WRW	SJ885 037	2.5	0.3	115	288
4	D/S Bill Brook WRW	SJ886 037	2.5	0.4	110	275
5	Pendeford Nature Reserve	SJ893 043	4	0.4	260	1040
6	Brewood Park Farm	SJ904 073	3.5	0.4	305	1068
7	Somerford Mill Farm	SJ895 093	6	0.5	170	1020
8	Stretton Mill	SJ897 108	6.2	0.7	150	930
9	Cuttlestone Bridge	SJ916 138	6.5	0.4	125	813
10	Action Mill Bridge	SJ932 189	8.5	0.7	165	1403
11	Radford Bridge	SJ948 217	10.4	1	480	4992

Appendix 2.1 (Continued) River Sence — Flow and site: North to Southwest (S - M)

Site No.	Location	NGR	Width (m)	Depth (m)	Length (m)	Area (m^2)
1	Heather Butterley Brick Works	SK393 103	4	0.5	110	440
2	Congerstone/ Cricket Pitch	SK374 060	5	0.6	220	1100
3	Congerstone	SK366 056	5	0.6	215	1075
4	Harris Bridge	SK352 032	6	0.6	150	900
5	Lovett's Bridge	SK335 023	5.5	0.4	90	495
6	Ratcliffe Culey Bridge	SP320 996	6	0.8	220	1320

Appendix 2.1 (Continued) River Soar — Flow and site: South to North (S - M)

Site No.	Location	NGR	Width (m)	Depth (m)	Length (m)	Area (m^2)
1	Ramsdale Farm	SP497 924	2.2	0.2	100	220
2	Sutton Hill	SK512 944	5	0.4	112	560
3	Croft	SK505 952	6	0.3	105	630
4	Littlethorpe	SK541 974	7.4	0.6	150	1110
5	Jubilee Park	SK551 985	8.5	0.3	100	850
6	Blue Bank Lock	SK555 992	8	0.8	124	992
7	Leicester Straights	SK581 034	40	0.8	200	8000
8	Abbey Meadows	SK588 062	40	1.8	400	16000
9	D/S Wanlip STW Outfall	SK598 119	20	2	747	14940
10	Mountsorrel	SK588 155	16	1	186	2976
11	Barrow on Soar	SK571 175	30	2.5	400	12000
12	Cotes	SK551 208	17.5	0.6	450	7875
13	Ashby-de-la-Zouch	SK123 456	20	2.4	737	14740
14	Kegworth	SK495 271	27	1.8	1045	28215
15	Ratcliffe on Soar	SK496 294	25	1	750	18750

Appendix 2.1 (Continued) River Sow — Flow and site: North to Southeast (S - M)

Site No.	Location	NGR	Width (m)	Depth (m)	Length (m)	Area (m^2)
1	Eccleshall Castle	SJ831 296	2.7	1	200	540
2	Hillcote Hall	SJ843 296	4.8	1.2	175	840
3	Chebsey	SJ861 285	4	1	105	420
4	Great Bridgeford	SJ887 277	6	1.1	310	1860
5	Cresswell Farm	SJ892 262	7	1	410	2870
6	Doxey Marshes	SJ909 241	4	0.6	200	800
7	Broadeye Stafford	SJ918 231	8.5	1.3	316	2686
8	Stafford Sea Scout Hut	SJ929 229	7.5	0.5	180	1350
9	U/S St.Thomases Mill	SJ945 229	4.3	1	185	796

Appendix 2.1 (Continued) River Tame — Flow and site direction: South to North (S - M)

Site No.	Location	NGR	Width (m)	Depth (m)	Length (m)	Area (m^2)
1	Lea Marston Upper	SP214 943	17	1	105	1785
2	Lea Marston Lower	SP214 945	20	1.5	270	5400
3	Middleton	SP203 988	22	1	350	7700
4	Hopwas Two Trees Farm	SK181 051	30	1.5	315	9450
5	Elford	SK189 104	30	2	400	12000
6	Chetwynd Bridge	SK188 138	38	1.5	328	12464

Appendix 2.1 (Continued) River Tean — Flow and site: North to Southeast (S - M)

Site No.	Location	NGR	Width (m)	Depth (m)	Length (m)	Area (m^2)
1	Litleys Farm	SK001 424	4	0.3	100	400
2	Teanford Mill	SK006 406	4	0.6	145	580
3	Upper Tean Bridge	SK008 396	5	0.6	115	575
4	Rectory Farm	SK031 375	5.5	0.6	110	605
5	Checkley WRW	SK035 374	3.5	0.4	115	403
6	Fole Hall	SK047 369	5	0.4	140	700
7	Fole D/S Creamery	SK049 368	5	0.5	190	950
8	Beamhurst Bridge	SK065 359	8	0.7	153	1224
9	Spath	SK087 348	5	0.5	115	575

Appendix 2.1 (Continued) River Trent — Flow and site : North to Southeast (S - M)

Site No.	Site	NGR	Width (m)	Depth (m)	Length (m)	Area (m^2)
1	Norton green	SJ 901 520	1.7	0.9	140	238
2	Abbey Farm	SJ 903 492	3.5	0.8	125	438
3	Finney Gardens	SJ 899 474	5.5	0.8	250	1375
4	Seven Arches, StokeUponTrent	SJ 886 457	6.3	0.6	172	1084
5	N. Staffs Polytech.	SJ 885 456	5.7	0.4	115	656
6	Boothen End	SJ 878 444	8.5	0.2	170	1445
7	Hanford U/S Lyme Brook	SJ 867 427	8	0.4	360	2880
8	Hissey's Scrap Yard	SJ 863 417	11	0.4	190	2090
9	U/S Park Brook Bridge	SJ 866 410	8	1	70	560
10	D/S Park Brook Bridge	SJ 867 407	11.5	0.3	260	2990
11	Trentham U/S Strongford WRW	SJ 873 393	8	1	450	3600
12	Tittensor D/S Strongford WRW	SJ 876 380	12	1	240	2880
13	Meaford Power Station	SK 885 368	12	1	215	2580
14	Walton Lane Stone	SJ 894 339	8	1	245	1960
15	Aston Lock	SJ 916 318	11.4	0.8	270	3078
16	Sandon	SJ 936 294	10	0.9	530	5300
17	Weston U/SGayton Brook	SJ 966 273	11	1	493	5423
18	U/S Hoo Mill	SJ 995 240	12.3	1	1000	12300
19	D/S Hoo Mill	SJ 996 237	15	0.6	175	2625
20	Great Haywood Mill	SJ 995 230	11	1	205	2255

Appendix 2.1 (Continued) River Aire — Flow and site direction: North to Southeast (Source to Mouth)

Site No.	Location	NGR	Width (m)	Depth (m)	Length (m)	Area (m^2)
1	Malham Beck, below Malham cove	SD 897637	4		100	400
2	Malham Beck, below waterfalls	SD 899632	4.5		100	450
3	Malham Beck, Malham village	SD 901627	6		100	600
4	Malham Beck, above STW	SD 902623	3		100	300
5	Malham Beck, below STW	SD 903622	3.5		50	175
6	Gordale Beck, Gordale bridge	SD 913635	3		100	300
7	Malham Beck, below Gordale beck	SD 903621	5		100	500
8	River Aire above Skelgill mill	SD 899617	6		100	600
9	River Aire Hanlith bridge	SD 900612	8		100	800
10	River Aire Airton bridge	SD 903593	7		100	700
11	Above Gargrave (below bridge)	SD 921538	8		100	800
12	Gargrave (above stepping stone)	SD 929541	12		100	1200
13	Near Gargrave STW	SD 944538	15		100	1500
14	U/S Snaygill STW	DS 984501-984496	15		500	7500
15	D/S Snaygill STW (above Cononley)	SD 992485-993676	15		320	4800
16	Crossflatts	SE 095404-098403	15		350	5250
17	Esholt U/S STW	SE 174397	20		200	4000
18	Calverley (below A6120) D/S Rawdon STW	SE 224369	20		200	4000
19	Kirkstall	SE 264350-271347	20		800	16000
20	Thwaite Weir	SE 327313-324312	25		400	10000
21	Below Skelton Grange Power Station	SE 333308-336306	22		300	6600
22	Swillington Bridge	SE 368295-373293	25		500	12500
23	Castleford below weir, above Hicksons Ltd.	SE 429260	25		300	7500
24	Castleford alongside Hicksons Ltd.	SE 434267	25		300	7500
25	Beal Weirpool	SE 535255	35		400	14000
26	Chapel Haddlesey U/S A19	SE 576263-572263	35		400	14000

Appendix 2.1 (Continued) River Nidd   Flow and site: North to Southeast (S - M)

Site No.	Location	NGR	Width (m)	Depth (m)	Length (m)	Area (m^2)
1	D/S Birstwith, Section 1		16		60	960
2	D/S Birstwith, S2		19		120	2280
3	D/S Birstwith, S3		20		110	2200
4	D/S Birstwith, S4		22		110	2420
5	U/S Hampsthwaite bridge, S1		18		100	1800
6	U/S Hampsthwaite bridge, S2		20		100	2000
7	U/S Hampsthwaite bridge, S3		22		100	2200
8	U/S Hampsthwaite bridge, S4		20		100	2000
9	Cragg Lane STW, S1		22		100	2200
10	Cragg Lane STW, S2		25		100	2500
11	Cragg Lane STW, S3		25		100	2500
12	Cragg Lane STW, S4		25		100	2500
13	Upper Cragghill farm, S1		17.5		100	1750
14	Upper Cragghill farm, S2		17.5		100	1750
15	Upper Cragghill farm, S3		17.5		100	1750
16	Upper Cragghill farm, S4		22		100	2200
17	Cragghill farm ford, S1		17		100	1700
18	Cragghill farm ford, S2		17		100	1700
19	Cragghill farm ford, S3		14		100	1400
20	Cragghill farm ford, S4		15		100	1500
21	D/S Killinghall bridge, S1		20		80	1600
22	D/S Killinghall bridge, S2		18		90	1620
23	D/S Killinghall bridge, S3		15		110	1650
24	D/S Killinghall bridge, S4		20		100	2000
25	Roch farm, S1		28		100	2800
26	Roch farm, S2		23		100	2300
27	Roch farm, S3		26		100	2600
28	Roch farm, S4		25		100	2500
29	Holme bottom farm, S1		22		100	2200
30	Holme bottom farm, S2		22		100	2200
31	Holme bottom farm, S3		22		100	2200
32	Holme bottom farm, S4		22		100	2200
33	U/S Scotton weir S1					3000
34	U/S Scotton weir S2					3000
35	U/S Scotton weir S3					3000
36	U/S Scotton weir S4					3000
37	Scotton weir S1					2250
38	Scotton weir S2					2500
39	Scotton weir S3					3200
40	Scotton weir S4					3300
41	D/S Scotton weir S1					2500
42	D/S Scotton weir S2					2500
43	D/S Scotton weir S3					2600
44	D/S Scotton weir S4					2400
45	Scotton Hospice S1					2440
46	Scotton Hospice S2					2675
47	Scotton Hospice S3					2800
48	Scotton Hospice S4					2500
49	D/S Scotton Hospice S1					1700
50	D/S Scotton Hospice S2					1350
51	D/S Scotton Hospice S3					2420
52	D/S Scotton Hospice S4					3000
53	Conningham Hall S1					2500
54	Conningham Hall S2					1800
55	Conningham Hall S3					3600
56	U/S High Bridge S1					2380

57	U/S High Bridge S2	3100
58	U/S High Bridge S3	3100
59	U/S High Bridge S4	4160
60	Mother Shiptons S1	1800
61	Mother Shiptons S2	2500
62	Mother Shiptons S3	2500
63	Mother Shiptons S4	3000
64	Lido Top S1	2200
65	Lido Top S2	2500
66	Lido Top S3	2500
67	Lido Top S4	2500
68	Lido Bottom S1	2700
69	Lido Bottom S2	2730
70	Lido Bottom S3	2470
71	Lido Bottom S4	2500
72	Knaresborough STW S1	1800
73	Knaresborough STW S2	2300
74	Knaresborough STW S3	2200
75	Knaresborough STW S4	2300
76	D/S A59 Bridge S1	1530
77	D/S A59 Bridge S2	1650
78	D/S A59 Bridge S3	2000
79	D/S A59 Bridge S4	3000
80	U/S Goldsborough Mill S1	2430
81	U/S Goldsborough Mill S2	2970
82	U/S Goldsborough Mill S3	2700
83	U/S Goldsborough Mill S4	2800
84	D/S Goldsborough mill S1	1980
85	D/S Goldsborough mill S2	1600
86	D/S Goldsborough mill S3	1500
87	D/S Goldsborough mill S4	1900
88	Pylons D/S Goldsborough mill	2200
89	U/S Little Ribston S1	1700
90	U/S Little Ribston S2	1500
91	U/S Little Ribston S3	1500
92	U/S Little Ribston S4	1500
93	Little Ribston wood S1	1440
94	Little Ribston wood S2	1870
95	Little Ribston wood S3	1920
96	Little Ribston wood S4	1200
97	Above Ribston park S1	1500
98	Above Ribston park S2	1350
99	Above Ribston park S3	2000
100	Above Ribston park S4	2200
101	Ribston park S1	1240
102	Ribston park S2	900
103	Ribston park S3	2260
104	Ribston park S4	2400
105	Ornamental bridge S1	2000
106	Ornamental bridge S2	2200
107	Ornamental bridge S3	2200
108	Ornamental bridge S4	2200
109	Ribston park-bottom S1	2200
110	Ribston park-bottom S2	2000
111	Ribston park-bottom S3	2000
112	Ribston park-bottom S4	2000
113	Crimple mouth S1	1500
114	Crimple mouth S2	2200

115	Crimple mouth S3		1700
116	Crimple mouth S4		1800
117	D/S A1 Bridge S1		2000
118	D/S A1 Bridge S2		1600
119	D/S A1 Bridge S3		2000
120	D/S A1 Bridge S4		2000
121	U/S Broad Wath beck S1		3000
122	U/S Broad Wath beck S2		3000
123	U/S Broad Wath beck S3		3000
124	U/S Broad Wath beck S4		3000
125	Cowthorpe hall farm S1		2500
126	Cowthorpe hall farm S2		2000
127	Cowthorpe hall farm S3		2090
128	Cowthorpe hall farm S4		2310
129	Hunsingore D/S footbridge S1	SE 429 531	1600
130	Hunsingore D/S footbridge S2		2080
131	Hunsingore D/S footbridge S3		1440
132	Hunsingore D/S footbridge S4		1760
133	Cowthore - Gauging hut S1	SE 435 552	1200
134	Cowthore - Gauging hut S2		1800
135	Cowthore - Gauging hut S3		2660
136	Cowthore - Gauging hut S4		2340
137	Cowthorpe dog kennels S1	SE 442 528	2000
138	Cowthorpe dog kennels S2		2520
139	Cowthorpe dog kennels S3		1400
140	Cowthorpe dog kennels S4		2080
141	Cowthorpe (bottom limit) S1	SE 443 532	1600
142	Cowthorpe (bottom limit) S2		2000
143	Cowthorpe (bottom limit) S3		2000
144	Cowthorpe (bottom limit) S4		2400
145	Cattal (upstream bridge) S1	SE 448 536	2000
146	Cattal (upstream bridge) S2		2000
147	Cattal (upstream bridge) S3		2000
148	Cattal (upstream bridge) S4		2000
149	Cattal (downstream bridge) S1		2000
150	Cattal (downstream bridge) S2		2760
151	Cattal (downstream bridge) S3		1400
152	Cattal (downstream bridge) S4		3400
153	Cattal (U/S Old thornville) S1		1500
154	Cattal (U/S Old thornville) S2		1500
155	Cattal (U/S Old thornville) S3		1500
156	Cattal (U/S Old thornville) S4		1500
157	Cattal (D/S Old thornville) S1		1500
158	Cattal (D/S Old thornville) S2		1500
159	Cattal (D/S Old thornville) S3		1500
160	Cattal (D/S Old thornville) S4		1500
161	Tockwith S1		1680
162	Tockwith S2		1200
163	Tockwith S3		960
164	Tockwith S4		1440
165	Hammerton mill S1		1050
166	Hammerton mill S2		1350
167	Hammerton mill S3		1200
168	Hammerton mill S4		900
169	Hammerton mill S5		900
170	Hammerton mill S6		1350
171	Opposite Skewkirk S1		1000
172	Opposite Skewkirk S2		1000

173	Opposite Skewkirk S3	1000
174	Opposite Skewkirk S4	1000
175	Wilstrop S1	2250
176	Wilstrop S2	1845
177	Wilstrop S3	1500
178	Wilstrop S4	1800
179	Upstream Skipbridge S1	2000
180	Upstream Skipbridge S2	2000
181	Upstream Skipbridge S3	2000
182	Upstream Skipbridge S4	2000

Appendix 4.1 Distribution of fish species in different English rivers
(P = Present, A = Absent, 3-STB = 3-spined stickleback, 10-STB= 10-spined stickleback)

Species	Cherwell	Evenlode	Stort	Thame	Windrush	Anker	Blithe	Blythe	Churnet	Cole	Derwent	Idle	Mease	Penk	Sence	Soar	Sow	Tame	Tean	Trent	Aire	Nidd	Number	% rivers
Chub	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	22	100
Dace	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	22	100
Roach	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	22	100
Gudgeon	P	P	A	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	21	95
Pike	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	A	P	P	P	21	95
Perch	P	P	P	P	P	P	P	P	P	P	P	A	P	P	P	P	P	P	P	P	P	P	21	95
Minnow	P	P	P	P	P	P	P	P	P	P	A	A	P	P	P	A	P	P	P	P	P	P	19	86
Stone loach	P	P	P	P	P	P	P	P	P	P	A	A	P	P	P	A	P	P	P	P	P	P	19	86
Bullhead	P	P	P	P	P	A	P	P	P	P	A	A	P	P	P	A	P	P	P	P	P	P	18	82
3-STB	P	P	P	P	P	P	P	P	P	P	A	A	A	P	P	A	P	P	P	P	P	P	18	82
Common bream	P	P	P	P	A	P	A	P	P	A	P	P	P	A	P	P	P	P	A	P	P	P	17	77
Eel	A	A	P	P	P	P	P	P	A	P	P	P	P	P	P	A	P	P	A	P	A	A	15	68
Tench	P	P	A	P	A	P	A	P	A	P	P	A	P	A	A	P	P	P	A	P	A	P	14	64
Brown trout	A	P	P	P	P	A	P	A	P	A	P	A	A	P	P	P	A	A	P	P	P	P	14	64
Barbel	A	P	A	P	P	P	A	A	A	A	P	A	A	P	A	P	P	P	A	P	A	P	11	50
Bleak	P	P	P	P	A	P	A	A	A	A	P	P	A	A	A	P	A	P	A	A	A	P	10	45
Ruffe	A	P	A	P	A	P	A	A	A	A	P	A	P	P	A	A	A	A	A	A	A	P	10	45
Rainbow trout	A	A	P	A	P	A	A	P	P	A	P	A	A	A	P	A	A	P	A	A	P	A	9	41
Grayling	A	P	A	A	P	A	P	A	P	A	P	A	A	A	A	A	A	A	P	A	P	P	8	36
Rudd	A	P	A	A	A	A	A	P	P	P	A	P	A	A	A	P	A	P	A	P	A	A	8	36
Common carp	P	A	P	A	A	P	A	P	A	A	A	A	P	A	A	P	A	A	A	A	A	A	7	32
River lamprey	A	P	A	P	P	A	A	A	P	A	A	A	A	A	A	A	A	A	A	A	A	A	4	18
Crucian carp	A	A	P	A	A	A	A	A	A	P	A	A	A	A	A	P	A	A	A	A	A	A	3	14
Brook lamprey	A	A	A	A	A	A	A	A	A	A	P	A	A	A	A	A	A	A	A	A	A	A	2	9
Goldfish	A	A	A	A	A	A	A	A	A	P	A	A	A	A	A	A	A	A	A	A	A	A	1	5
Silver bream	A	A	A	A	A	A	A	A	A	A	A	A	A	A	A	A	A	P	A	A	A	A	1	5
Pikeperch	A	A	A	A	A	A	A	A	A	A	A	A	A	A	A	A	A	A	A	A	A	A	0	0
Spined loach	A	A	A	A	A	A	A	A	A	A	A	A	A	A	A	A	A	A	A	A	A	A	0	0
10-STB	A	A	A	A	A	A	A	A	A	A	A	A	A	A	A	A	A	A	A	A	A	A	0	0
Total species	**14**	**19**	**16**	**18**	**16**	**16**	**13**	**16**	**16**	**15**	**16**	**9**	**14**	**14**	**14**	**14**	**14**	**18**	**11**	**16**	**14**	**17**		

Appendix 4.2 Distribution of fish families in different English rivers

P = Present, A = Absent

	The Thames catchment					**The Trent catchment**															**The Ouse catchment**		Distribution	
Species	Cherwell	Evenlode	Stort	Thame	Windrush	Anker	Blithe	Blythe	Churnet	Cole	Derwent	Idle	Mease	Penk	Sence	Soar	Sow	Tame	Tean	Trent	Aire	Nidd	Number	% rivers
Cyprinidae	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	**22**	**100**
Percidae	P	P	P	P	P	P	P	P	P	P	P	A	P	P	P	P	P	P	P	P	P	P	**21**	**95**
Esocidae	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	A	P	P	P	**21**	**95**
Cobitidae	P	P	P	P	P	P	P	P	P	P	A	A	P	P	P	A	P	P	P	P	P	P	**19**	**86**
Gasterosteidae	P	P	P	P	P	P	P	P	P	P	A	A	A	P	P	A	P	P	P	P	P	P	**18**	**82**
Cottidae	P	P	P	P	P	A	P	P	P	P	A	A	P	P	P	A	P	P	P	P	P	P	**18**	**82**
Salmonidae	A	P	P	P	P	A	P	P	P	A	P	A	A	P	P	P	A	P	P	P	P	P	**16**	**73**
Anguillidae	A	A	P	P	P	P	P	P	A	P	P	P	P	P	P	A	P	P	A	P	A	A	**15**	**68**
Thymallidae	A	P	A	A	P	A	P	A	P	A	P	A	A	A	A	A	A	A	P	A	P	P	**8**	**36**
Petromyzonidae	A	P	A	P	P	A	A	A	P	A	P	A	A	A	A	A	A	A	A	A	A	A	**5**	**23**
Total family	**6**	**9**	**8**	**9**	**10**	**6**	**9**	**8**	**9**	**7**	**7**	**3**	**6**	**8**	**8**	**4**	**7**	**8**	**7**	**8**	**8**	**8**		

Appendix 4.3 Site-wise distribution of fish species in the Thames catchment

River Cherwell **Total sites = 13**

	Site number													Distribution	
Species	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**Total**	**% sites**
Pike	P	P	P	P	P	P	P	P	P	P	P	P	P	13	100
Chub		P	P	P	P	P	P	P	P	P	P	P	P	12	92
Dace	P		P	P		P	P	P	P	P	P	P	P	11	85
Roach		P	P	P				P	P	P	P	P	P	9	69
Perch			P	P					P	P	P	P	P	7	54
Gudgeon		P	P	P				P		P	P		P	7	54
Bleak			P							P	P	P		4	31
Common bream				P							P			2	15
Tench									P			P		2	15
Common carp									P					1	8
Diversity = 10	2	4	7	7	2	3	3	5	7	7	8	7	6		

River Evenlode **Total sites = 20**

	Site number																				Distribution	
Species	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**	**Total**	**% sites**
Chub	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P		P	P	P	19	95
Dace	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P		P	P	P	19	95
Pike								P	P	P	P	P	P	P	P	P	P	P	P	P	13	65
Perch		P	P	P		P		P			P		P			P	P	P	P	P	12	60
Roach	P		P	P	P			P	P	P	P					P		P	P	P	12	60
Brown trout	P		P	P	P	P	P		P	P		P		P	P						11	55
Gudgeon			P		P					P	P		P	P		P		P	P	P	10	50
Grayling													P	P	P						3	15
Barbel																		P	P	P	3	15
Bleak																			P		1	5
Common bream																P					1	5
Tench																P					1	5
Rudd																P					1	5
Ruffe																			P		1	5
Diversity = 14	4	3	6	5	5	4	3	5	5	6	6	4	6	6	5	9	2	7	9	7		

Appendix 4.3 (continued) Site-wise distribution of fish species in the Thames catchment

River Stort	Site number							Total sites = 16									Distribution	
Species	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**Total**	**% sites**
Perch	P		P	P	P	P	P	P	P	P	P	P	P	P	P	P	15	94
Pike			P	P	P	P	P	P	P	P	P	P	P	P	P	P	14	88
Eel		P		P	P	P	P	P	P	P	P	P		P			11	69
Dace			P				P		P	P	P	P	P	P	P		9	56
Chub					P		P	P	P	P		P	P	P	P		9	56
Roach			P		P	P	P		P	P		P		P			8	50
Common bream			P		P					P		P		P			5	31
Common carp	P		P		P												3	19
Bleak										P				P	P		3	19
Crucian carp	P		P														2	13
Brown trout	P	P															2	13
Rainbow trout	P	P															2	13
Diversity = 12	5	3	7	3	7	4	6	4	6	8	4	7	4	8	5	3		

River Thame	Site number							Total sites = 18											Distribution	
Species	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**Total**	**% sites**
Dace	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	18	100
Roach		P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	17	94
Pike		P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	17	94
Chub	P	P			P	P	P	P	P	P	P	P	P	P	P	P	P	P	16	89
Perch		P	P	P	P	P		P			P	P	P	P	P	P	P	P	14	78
Gudgeon				P	P	P	P	P	P		P	P	P	P	P	P	P	P	14	78
Common bream				P	P		P	P		P	P	P	P	P			P		10	56
Tench		P	P	P		P	P		P				P			P			8	44
Bleak							P	P			P		P	P			P	P	7	39
Barbel																P		P	2	11
Brown trout																P			1	6
Ruffe																		P	1	6
Eel																P			1	6
Diversity = 13	2	6	5	7	7	7	8	8	6	5	8	7	9	8	6	10	8	9		

Appendix 4.3 (continued) Site-wise distribution of fish species in the Thames catchment

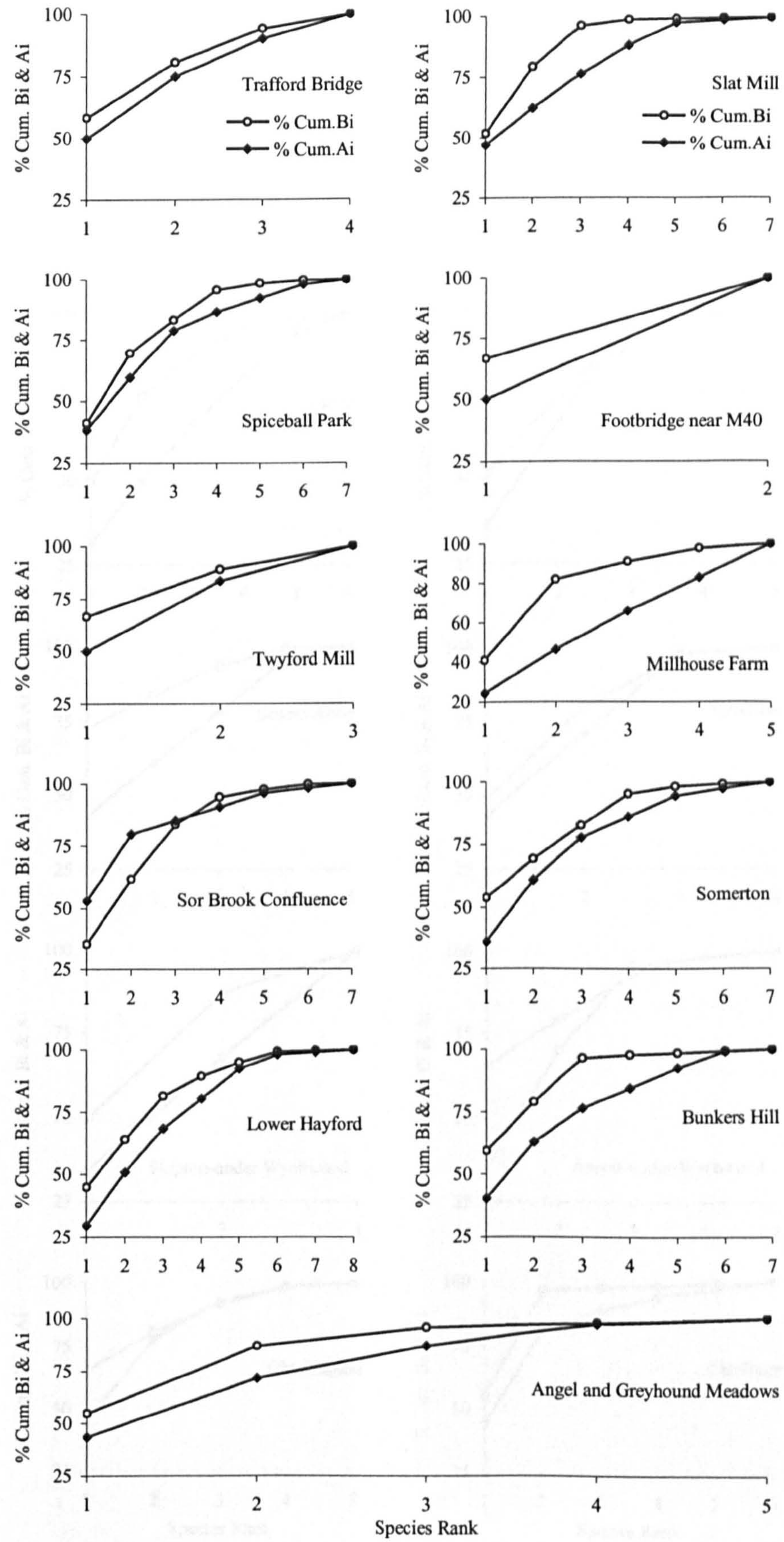
River Windrush **Total sites = 19**

Species	Site number 1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	Distribution Total	% sites
Brown trout	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P		18	95
Dace					P	P	P	P	P	P	P	P	P	P	P	P	P	P	P	15	79
Perch					P	P	P	P	P	P	P	P	P	P	P	P		P	P	14	74
Chub					P			P	P	P	P	P	P	P	P	P	P	P	P	13	68
Gudgeon								P	P	P	P	P	P	P	P	P	P	P	P	12	63
Pike					P			P			P		P	P	P	P	P	P	P	10	53
Roach										P	P	P	P	P	P	P	P	P		9	47
Eel								P			P	P	P	P			P	P	P	8	42
Grayling				P	P	P	P	P	P											6	32
Barbel																	P	P	P	3	16
Rainbow trout					P	P														2	11
Diversity = 11	1	1	1	2	7	5	4	8	6	6	8	7	8	8	7	7	8	9	7		

Appendix 4.4 k-dominance curves for the Thames catchment

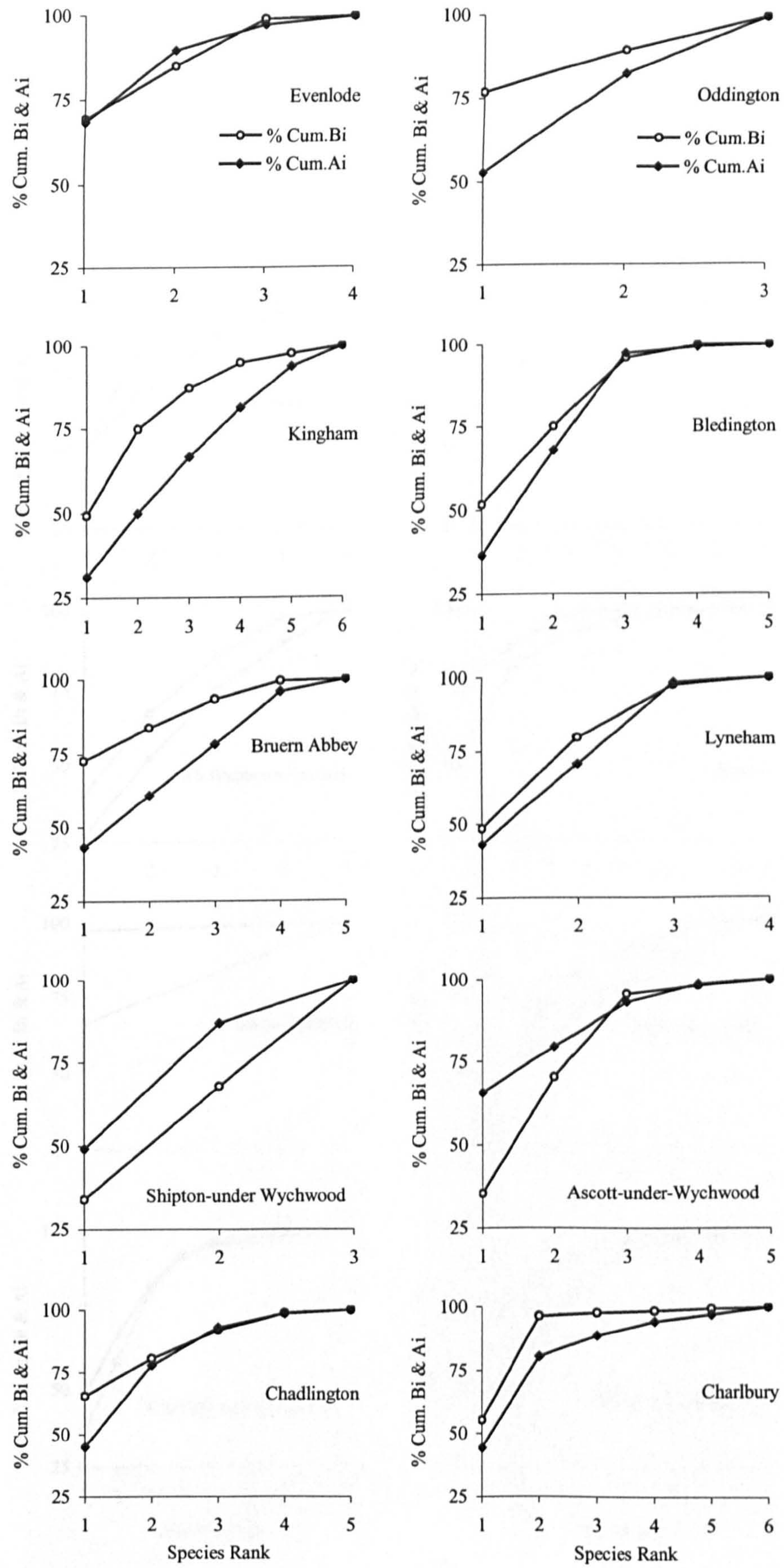
River Cherwell Total site = 13

Single species was caught from West Farndon Mill & Tramroad Industrial Estate (sites 1 & 5)



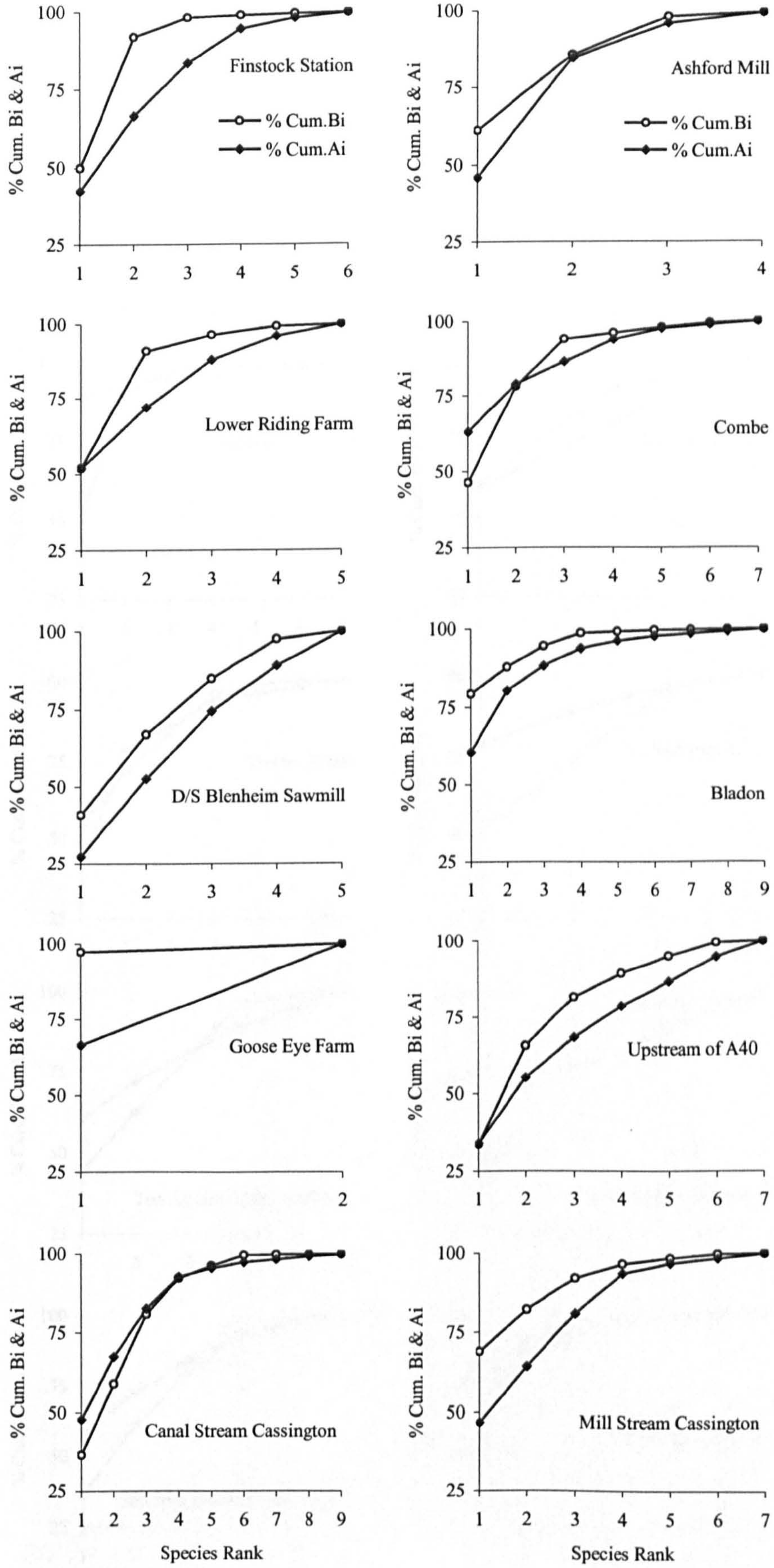
Appendix 4.4 (Continued) k-dominance curves for the Thames catchment

River Evenlode Site = 20 Flow & site direction: North to Southeast (S - M)



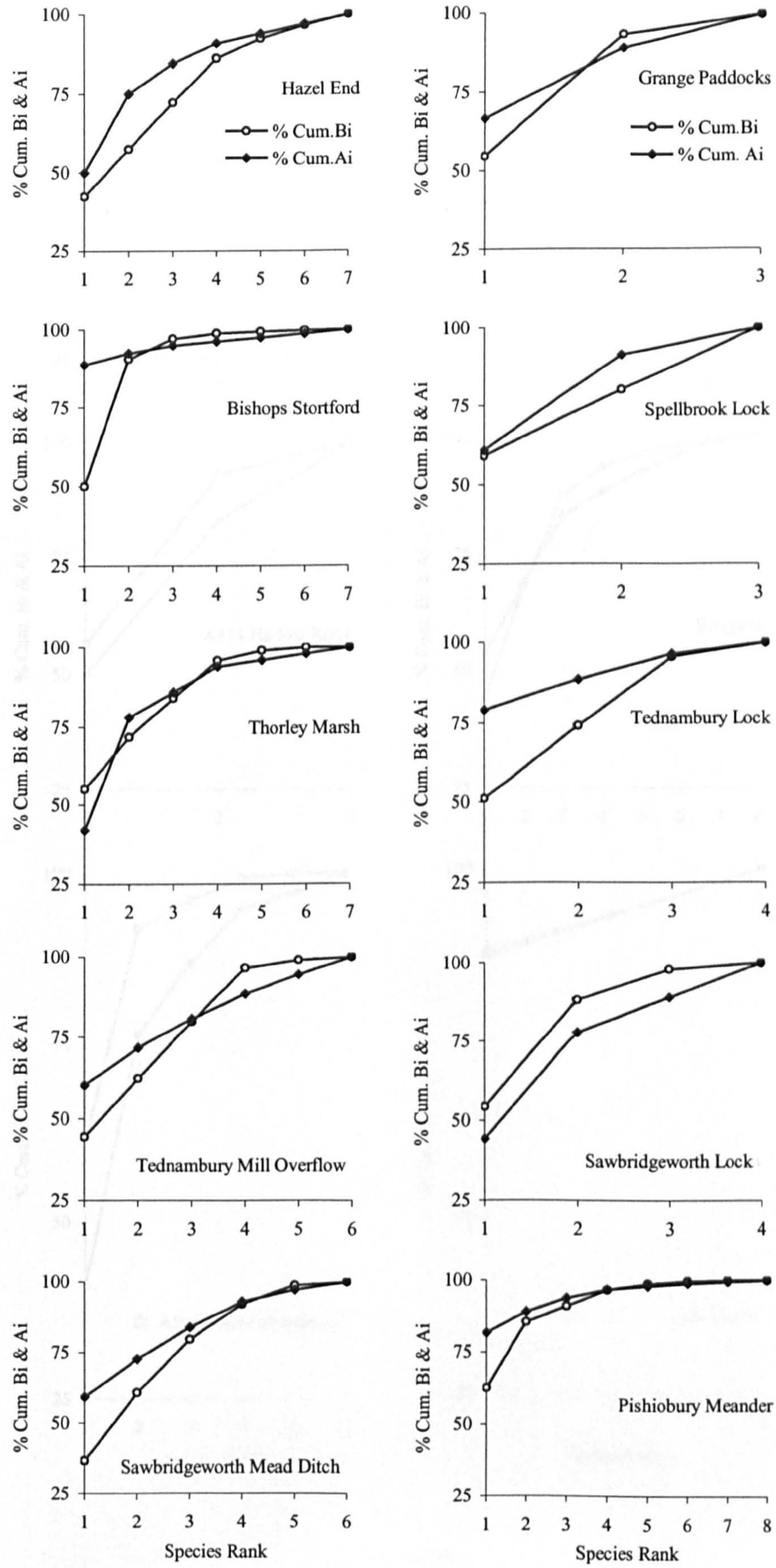
Appendix 4.4 (Continued) k-dominance curves for the Thames catchment

River Evenlode Site = 20 Flow & site direction: North to Boutheast (S - M)



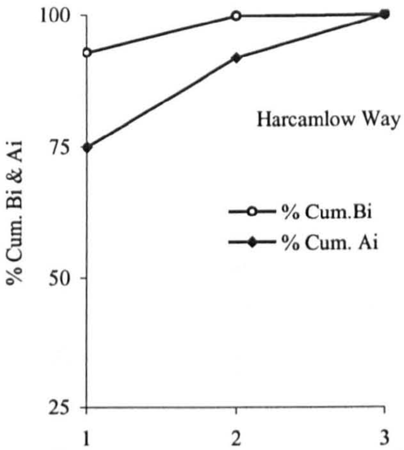
Appendix 4.4 (Continued) k-dominance curves for the Thames catchment

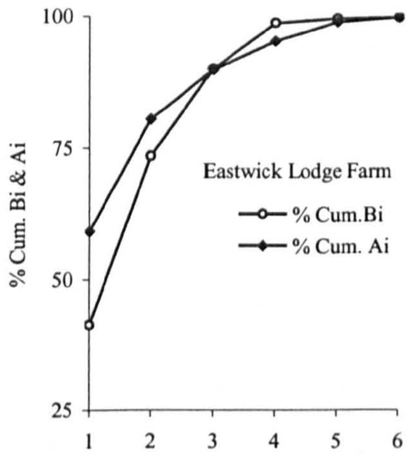
River Stort Site: 16 Flow and site direction: North to Southwest (Source to Mouth)

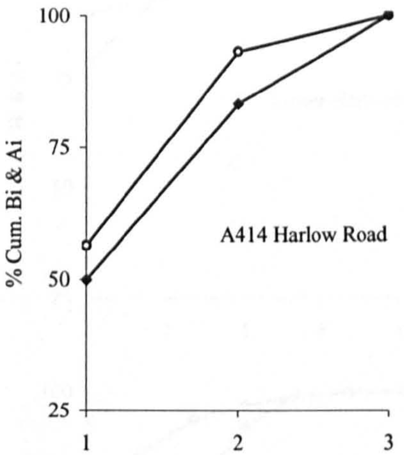


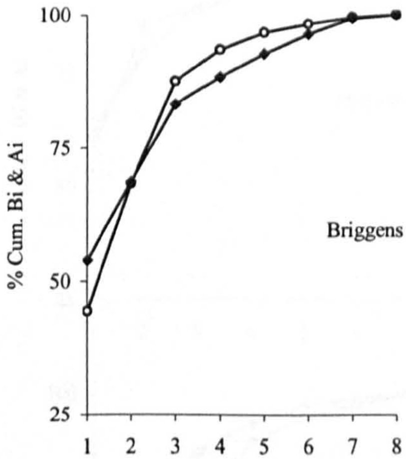
Appendix 4.4 (Continued) k-dominance curves for the Thames catchment

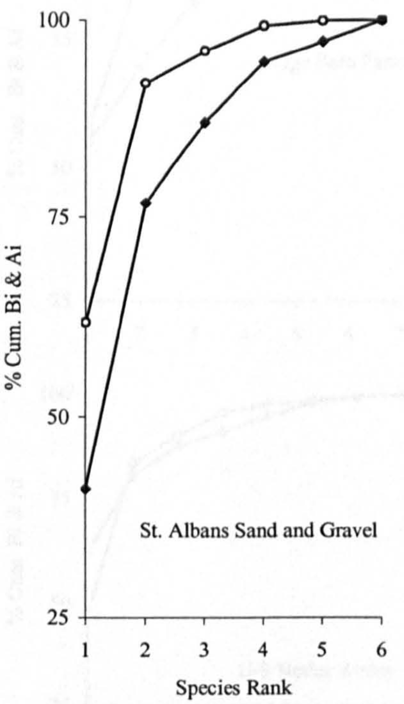
River Stort Site: 16 Flow and site direction: North to Southwest (Source to Mouth)

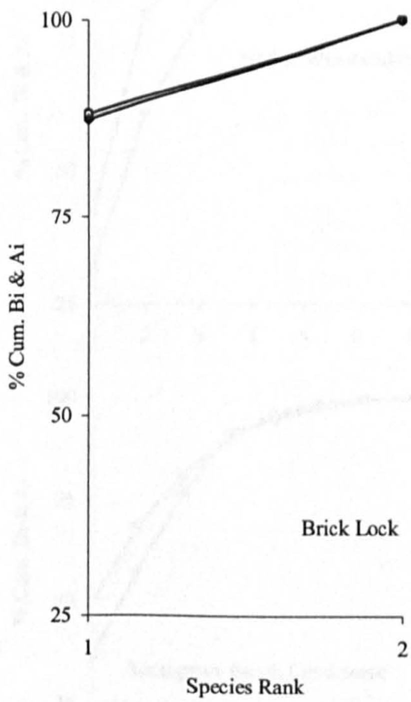








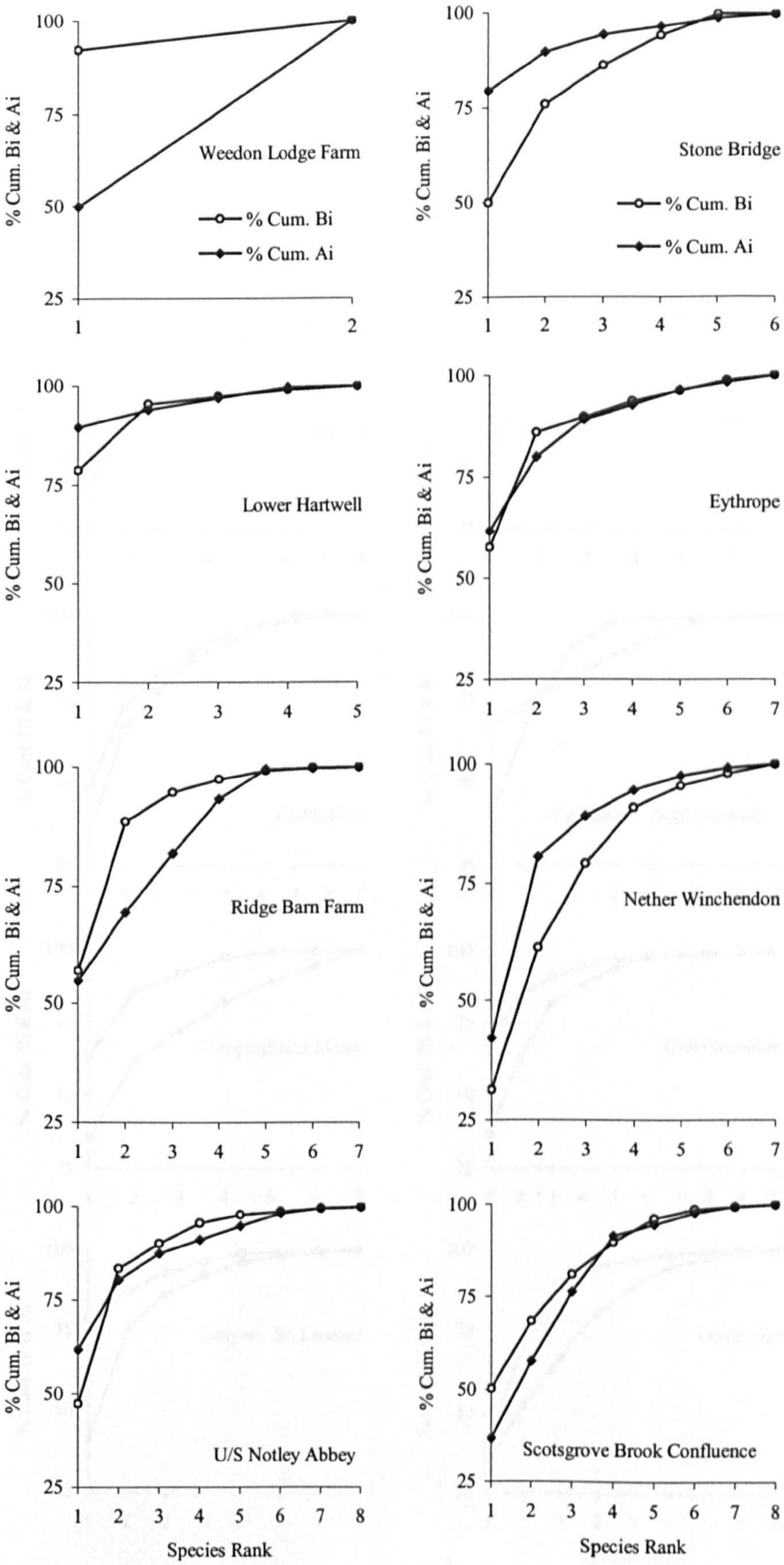




Appendix 4.4 (Continued) k-dominance curves for the Thames catchment

River Thame Site = 18

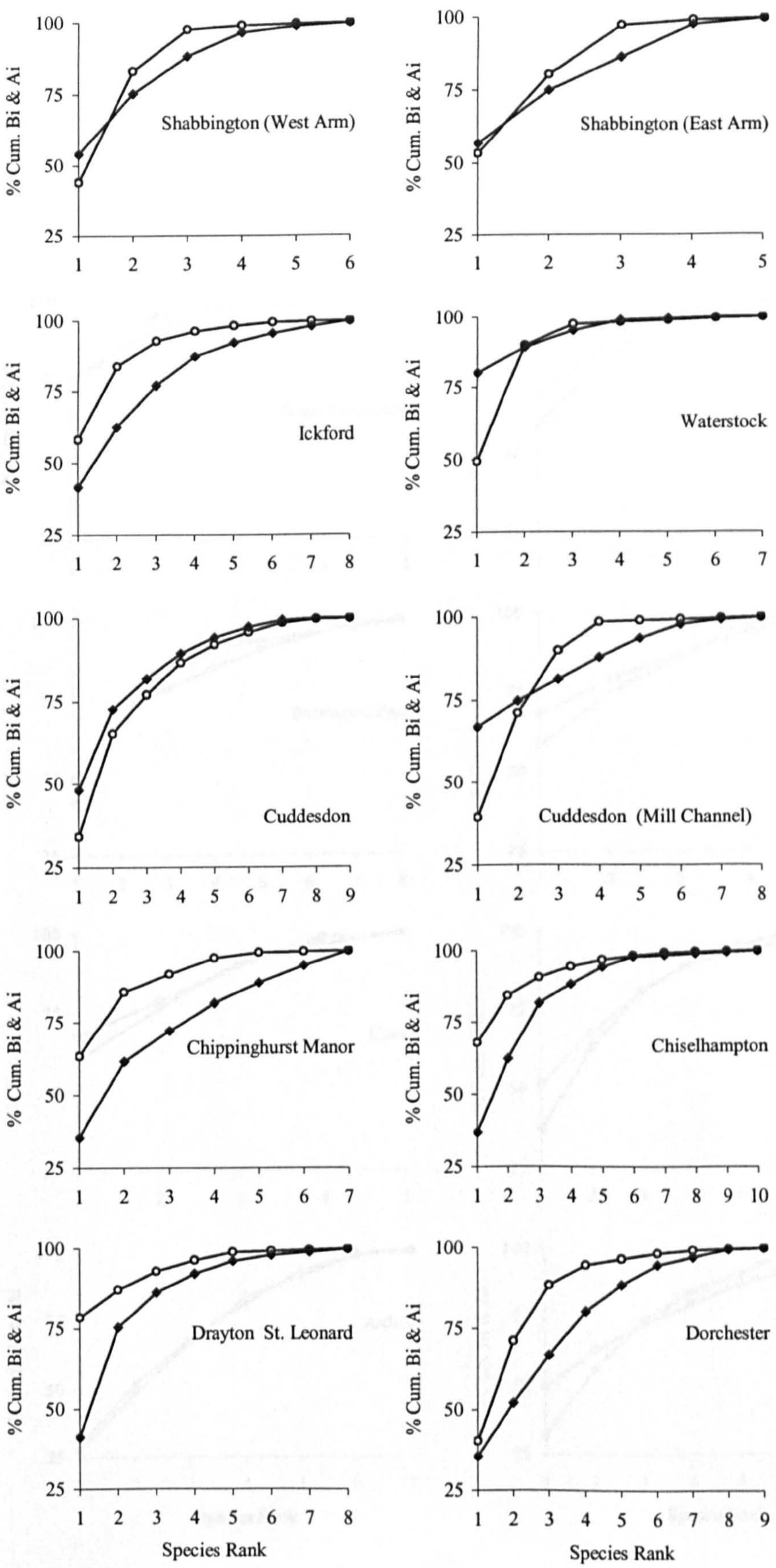
Flow & site direction: North to Southwest (Source to Mouth)



Appendix 4.4 (Continued) k-dominance curves for the Thames catchment

River Thame Total site = 18

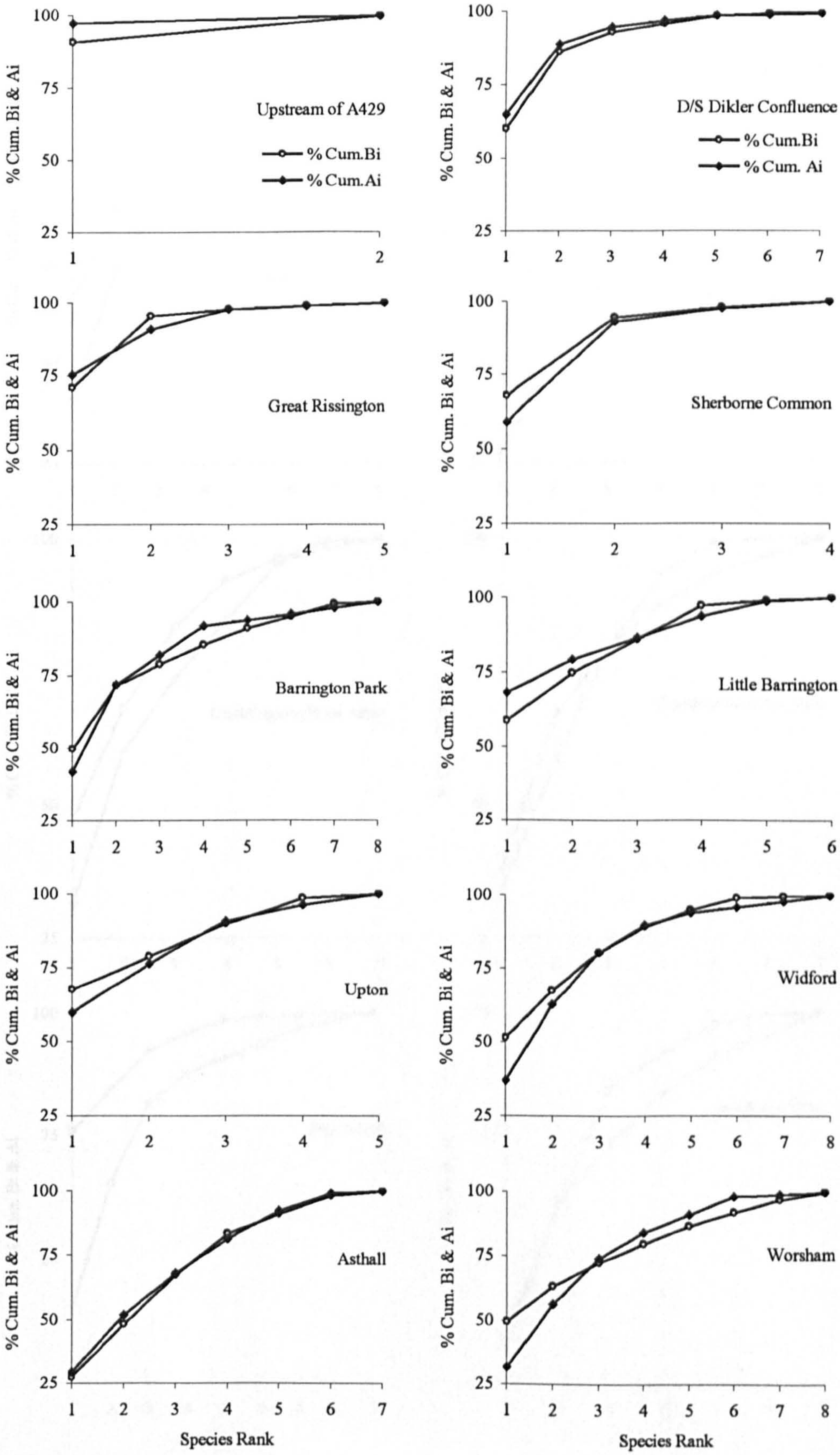
Flow & site direction: North to Southwest (Source to Mouth)



Appendix 4.4 (Continued) k-dominance curves for the Thames catchment

River Windrush Site: 19 Flow and site direction: North to Southeast (Source to Mouth)

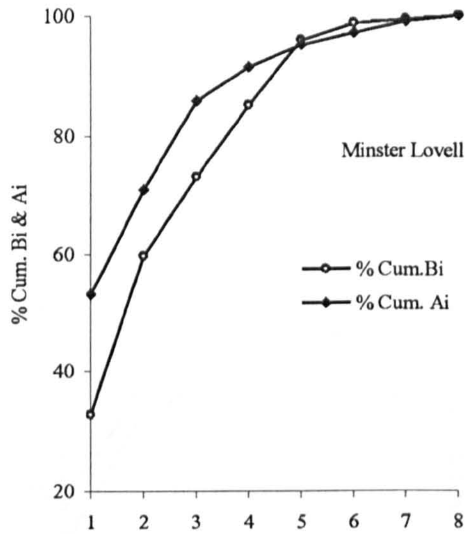
Single species was caught from Kineton, Guiting Power and Harford Bridge (sites 1, 2 & 3)

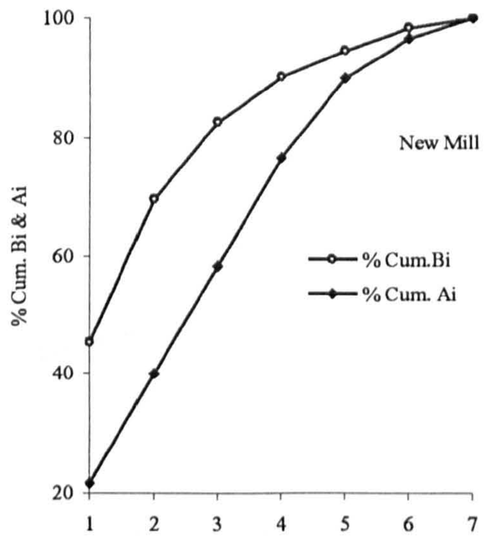


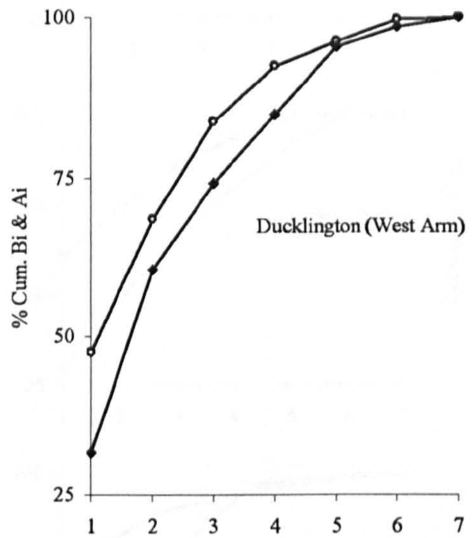
Appendix 4.4 (Continued) k-dominance curves for the Thames catchment

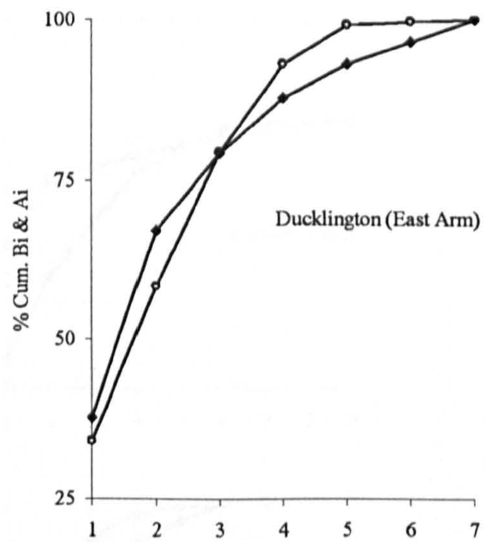
River Windrush Site: 19 Flow and site direction: North to Southeast (Source to Mouth)

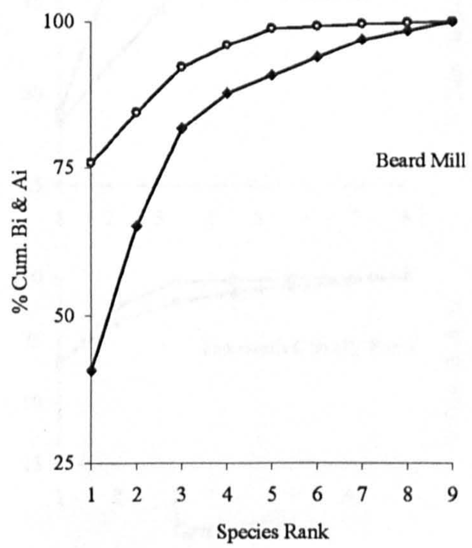
Single species was caught from Kineton, Guiting Power and Harford Bridge (sites 1, 2 & 3)

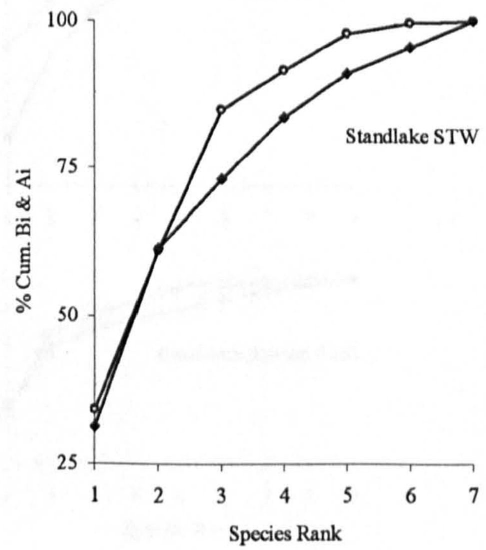








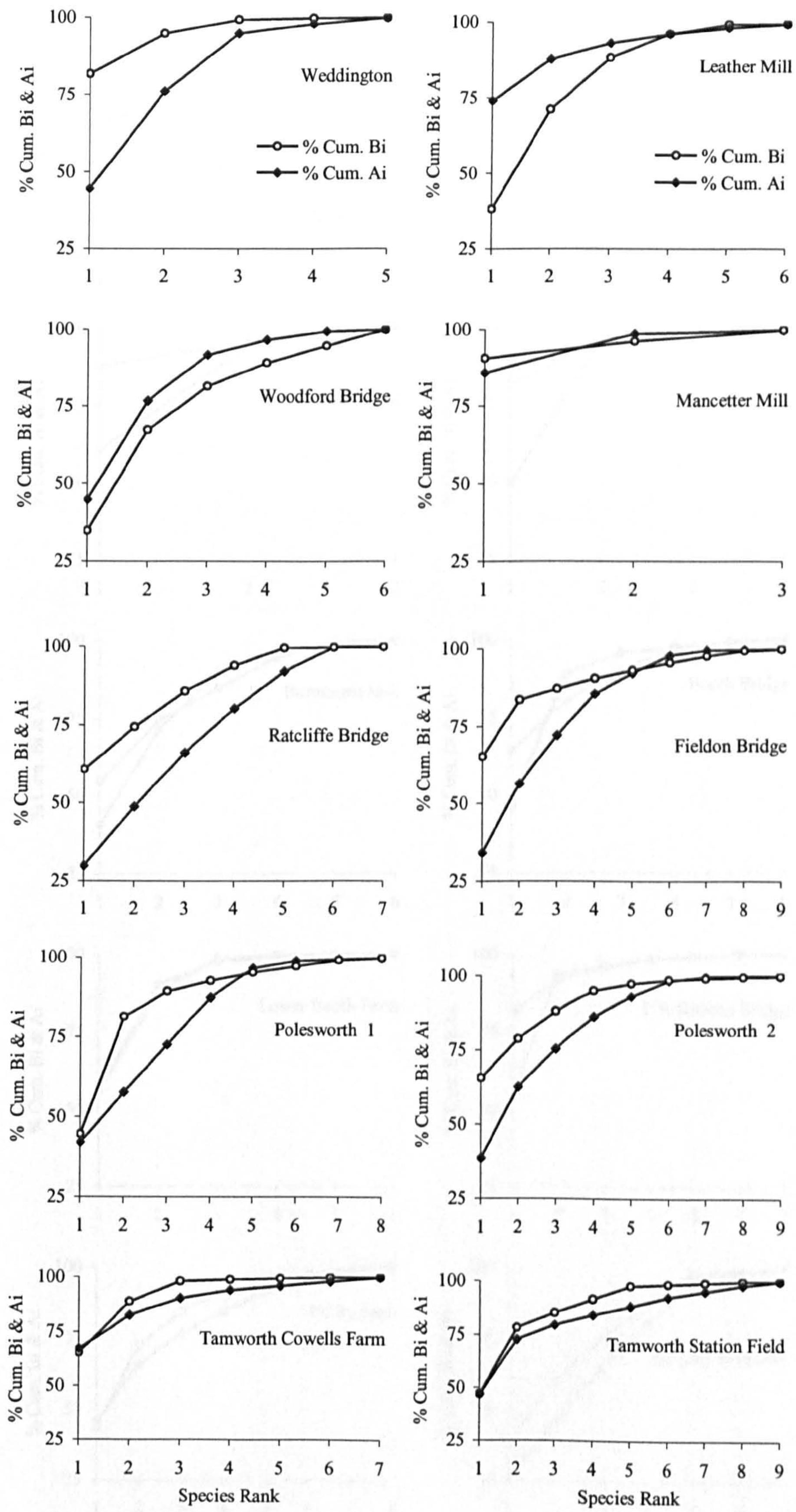




Appendix 4.4 (Continued) k-dominance curves for the Trent catchment

River Anker Total sites = 10

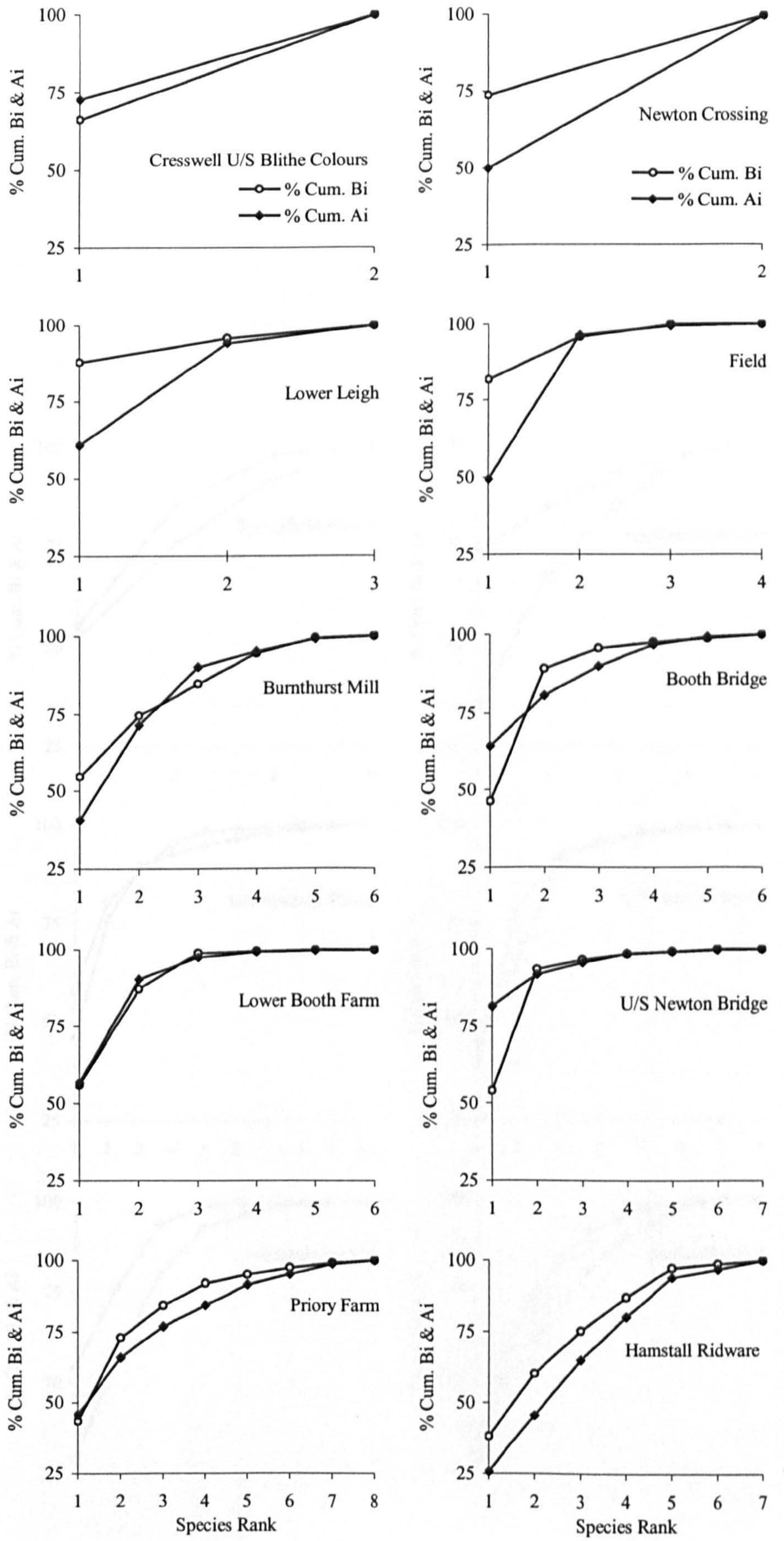
Flow & site direction: South to Northwest (Source to Mouth)



Appendix 4.4 (Continued) k-dominance curves for the Trent catchment

River Blithe Site = 11 Flow & site direction: North to Southeast (Source to Mouth)

Single species was caught from Blythe Bridge (site 1).

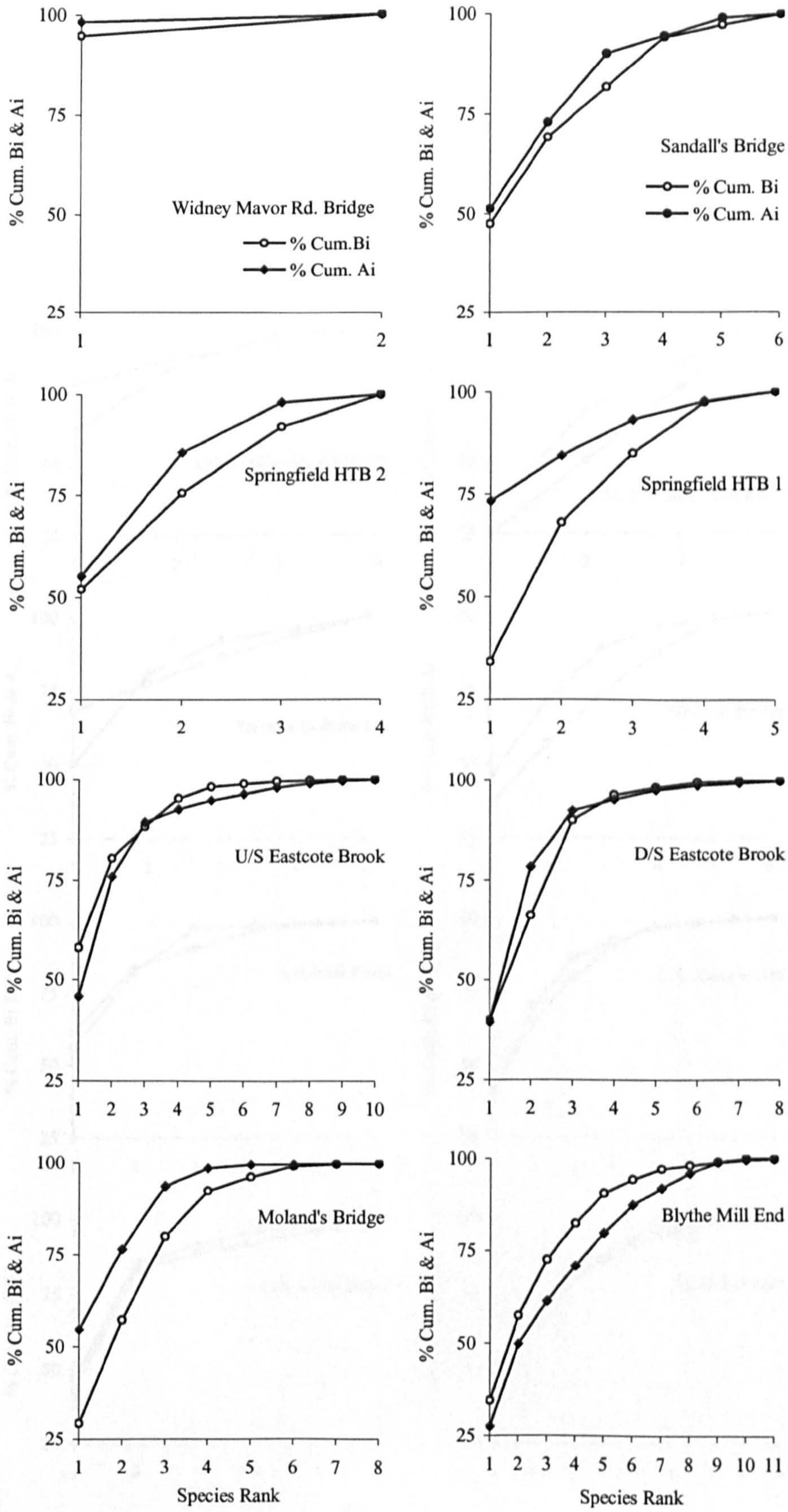


Appendix 4.4 (Continued) k-dominance curves for the Trent catchment

River Blythe Total site = 9

No fish were caught from Cheswick Green (site 1).

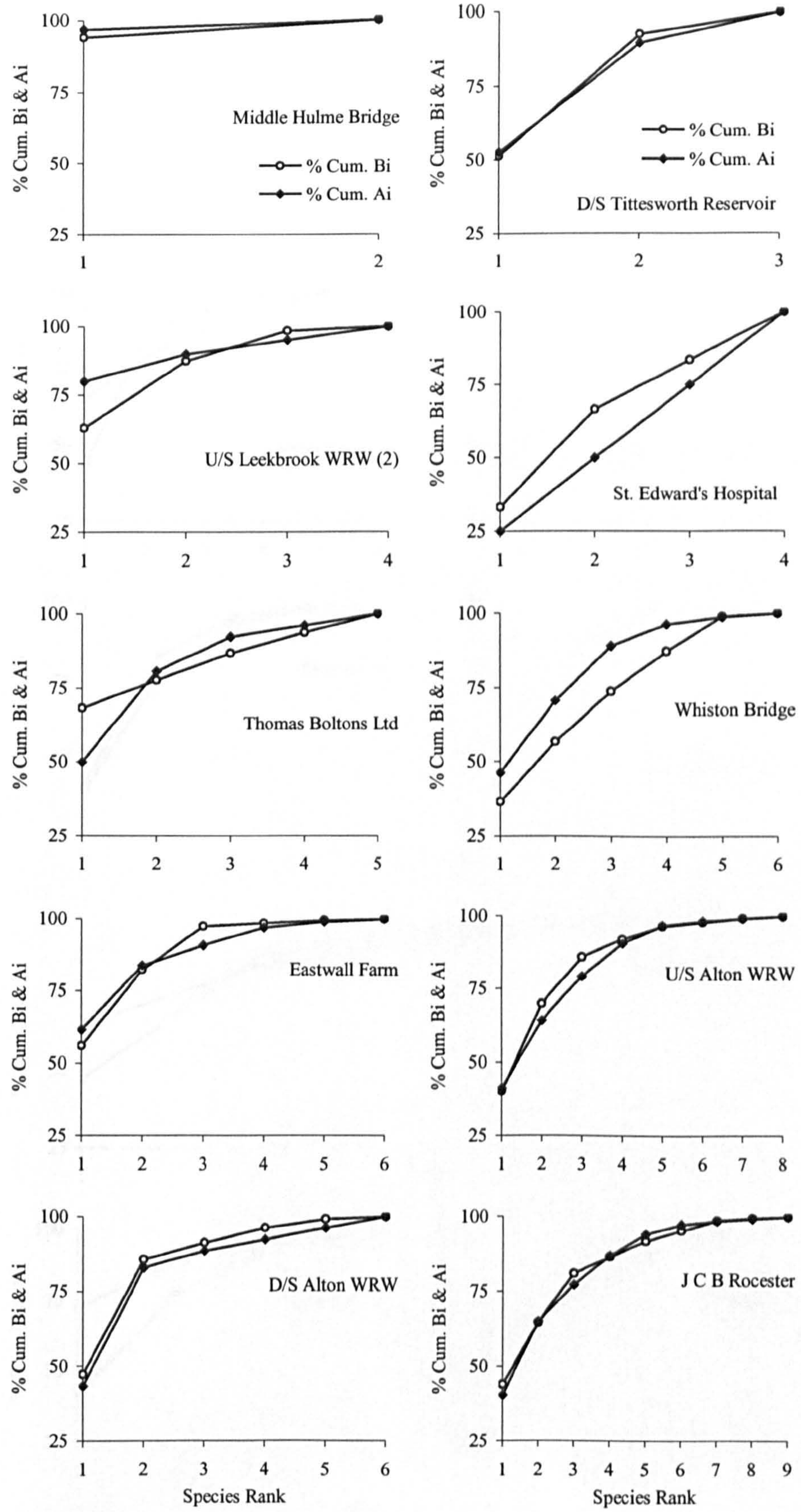
Flow & site direction: South to Northeast (Source to Mouth)



Appendix 4.4 (Continued) k-dominance curves for the Trent catchment

River Churnet Total site = 16 Flow & site direction: N-S (S - M)

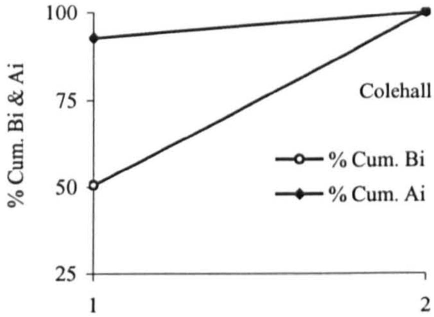
No fish were caught from South Hillswood Farm, Abbey Green Road S1,
Abbey Green Farm (sites 3, 4 & 5) while single species was caught from Westwood Golf Club,
Flint Mill Cheddleton and D/S Cheddleton WRW (sites 6, 9 & 10)

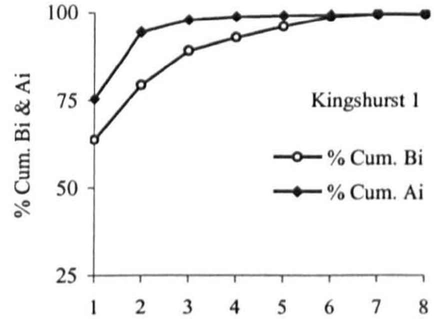


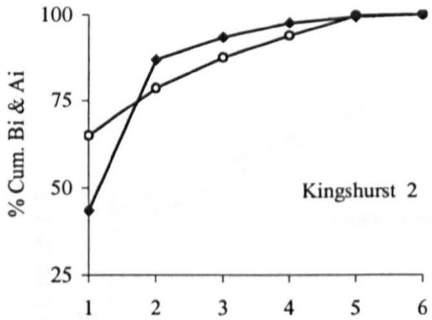
Appendix 4.4 (Continued) k-dominance curves for the Trent catchment

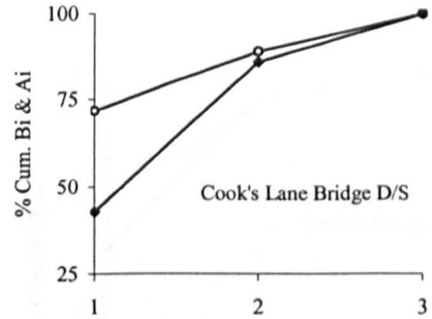
River Cole Total site = 14 Flow & site direction: South to Northeast (S - M)

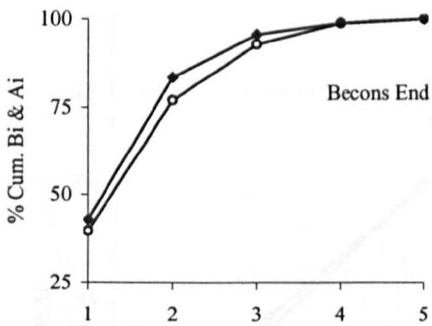
No fish were found at Lowbrook Farm, Mill Lodge, Haybarn Recreation Ground, Glebe Farm Recreation Ground and Colehall (1) (sites 1, 2, 3, 4 & 5)

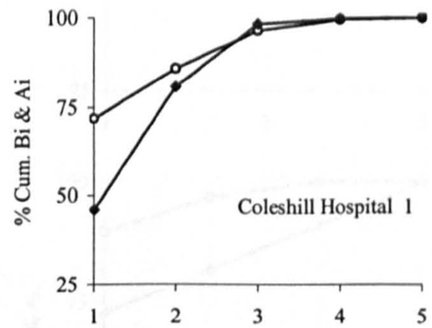


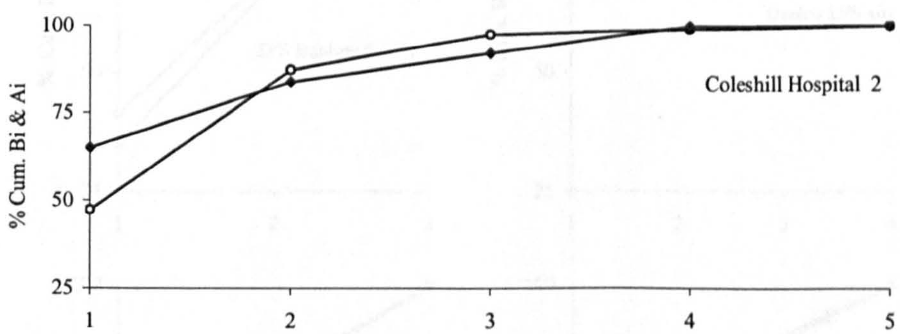


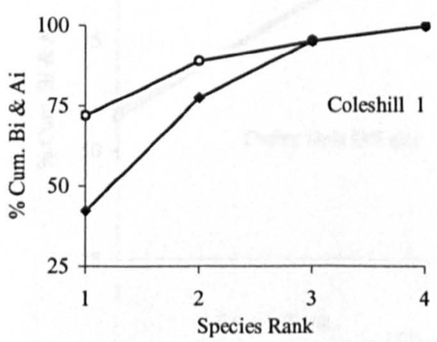


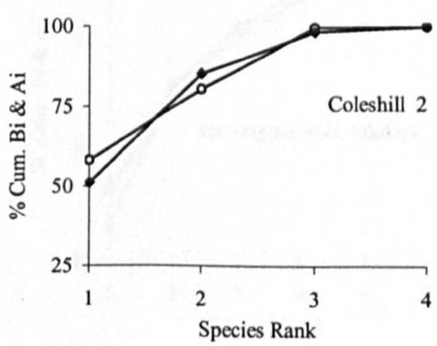








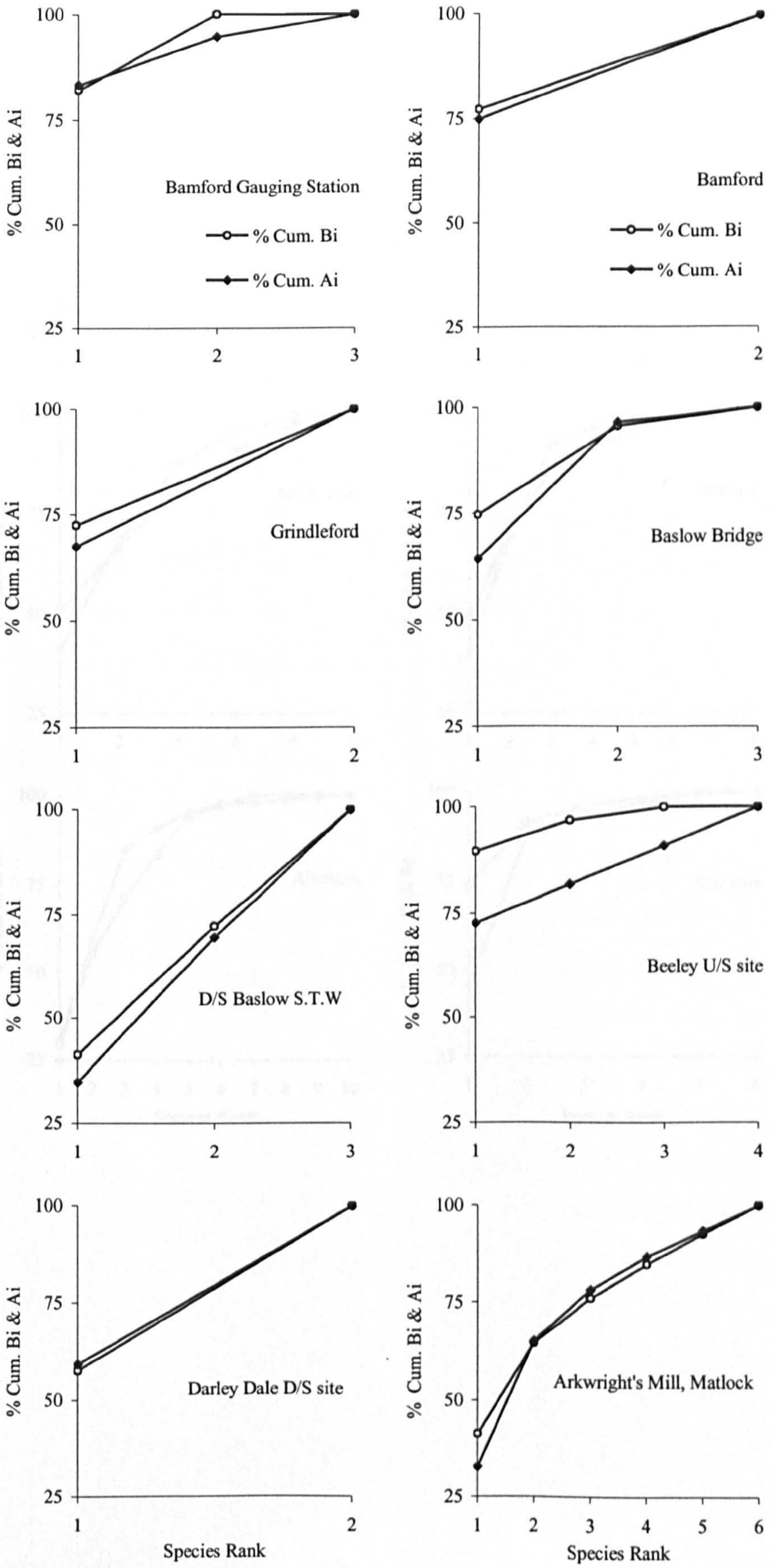




Appendix 4.4 (Continued) k-dominance curves for the Trent catchment

River Derwent Site = 15 Flow & site direction: North to South (S - M)

Single species caught from U/S Howden Gauging Weir (site 1)

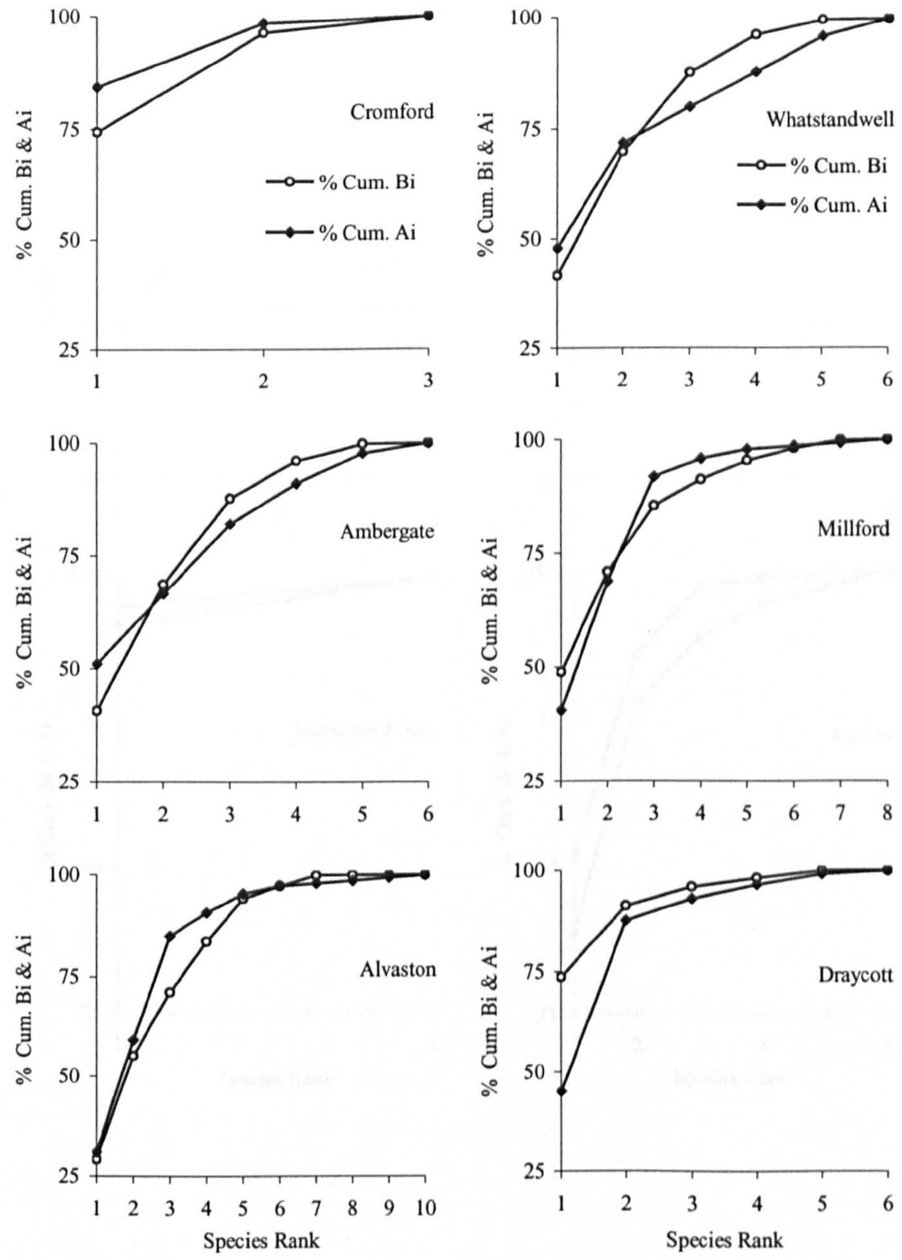


Appendix 4.4 (Continued) k-dominance curves for the Trent catchment

River Derwent Total site = 15

Flow & site direction: North to South (Source to Mouth)

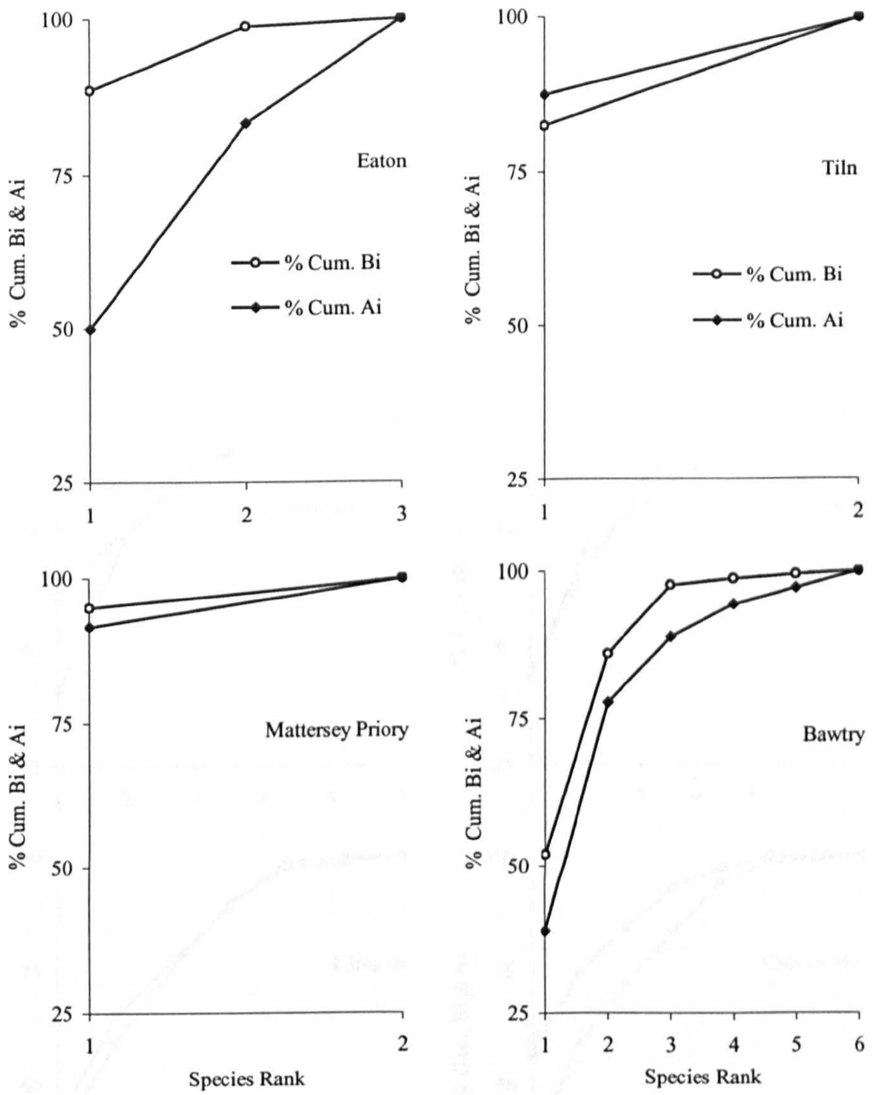
Single species caught from U/S Howden Gauging Weir (site 1)



Appendix 4.4 (Continued) k-dominance curves for the Trent catchment

River Idle Site = 5 Flow & site direction: South to Northeast (Source to Mouth)

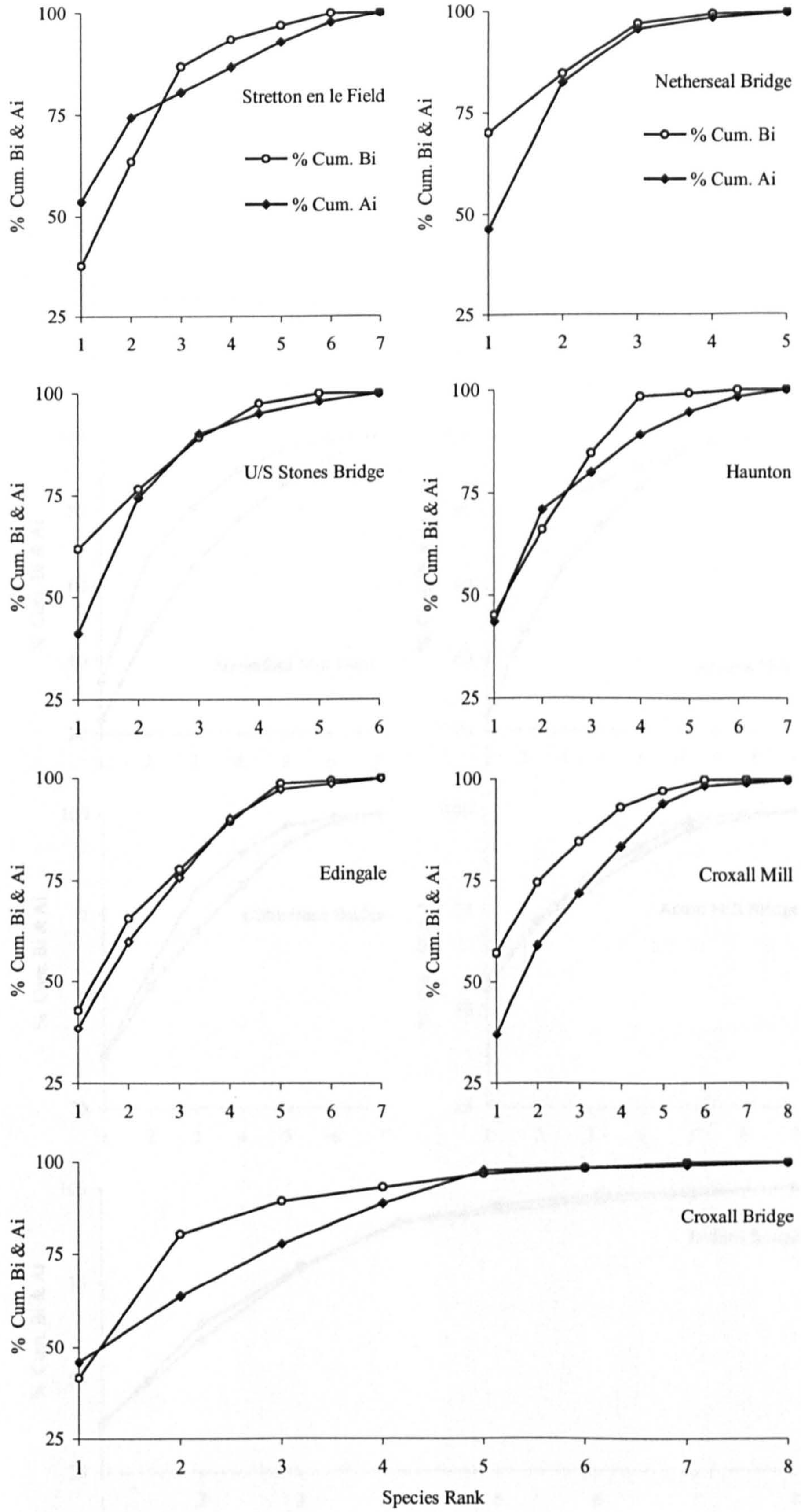
No fish were found at Misson (site 5)



Appendix 4.4 (Continued) k-dominance curves for the Trent catchment

River Mease Total site = 7

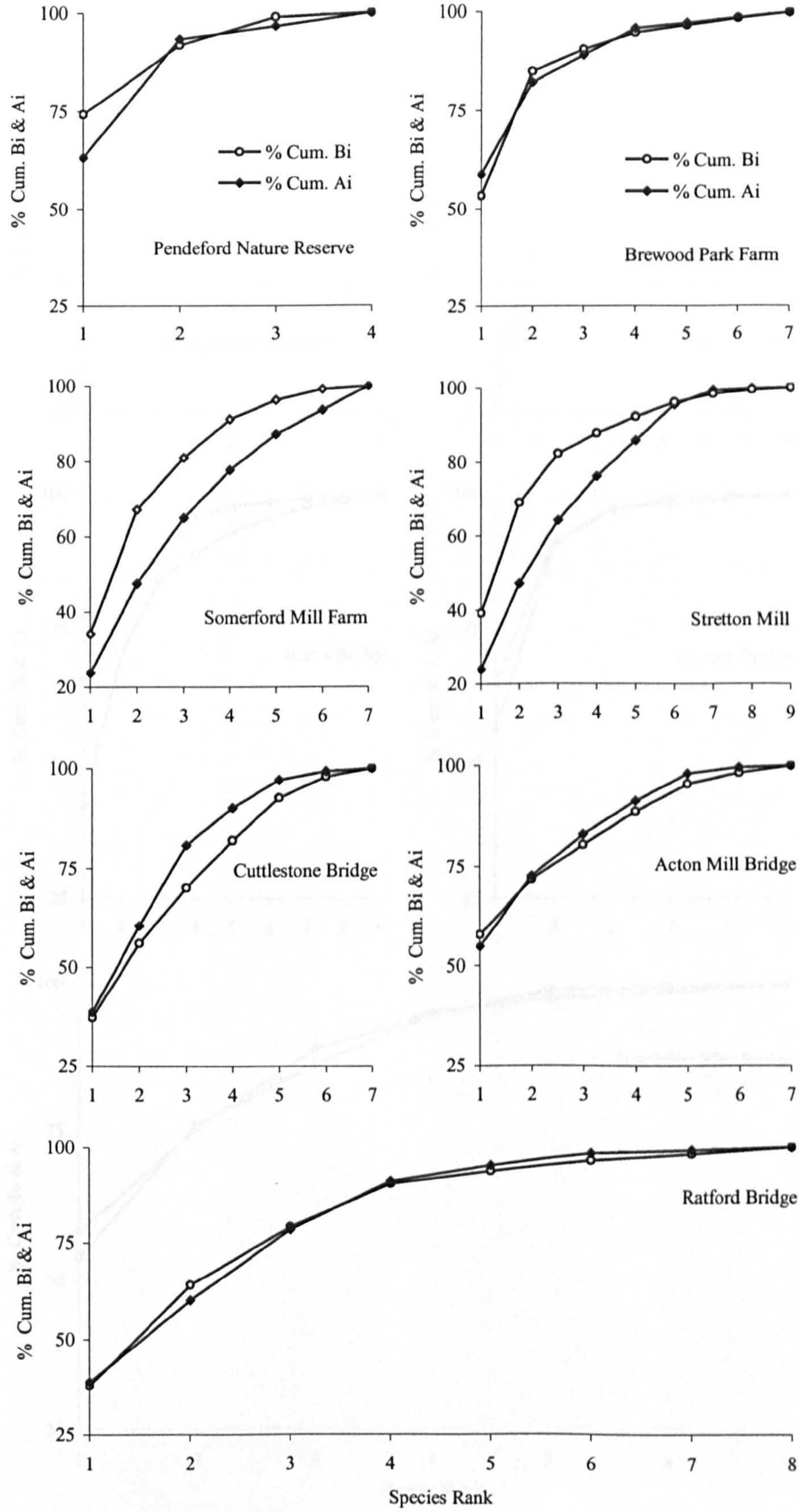
Flow & site direction: East to West (Source to Mouth)



Appendix 4.4 (Continued) k-dominance curves for the Trent catchment

River Penk Site = 11 Flow & site direction: South to North (Source to Mouth)

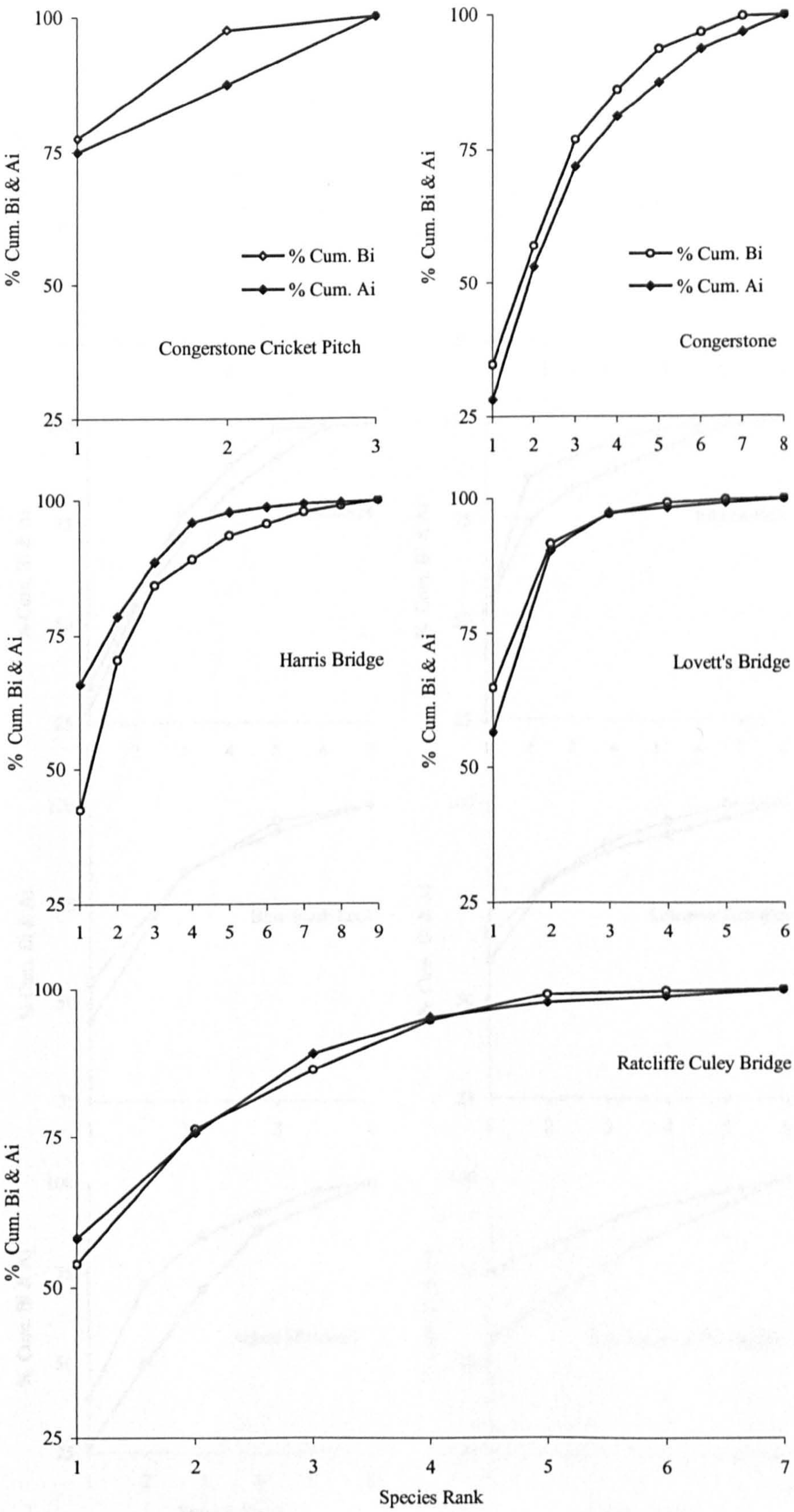
No fish were found at Black Brook Nature Trail, Allotment Site Codsall and U/S Bill Brook WRW (sites 1, 2 & 3) and single species was caught from D/S Bill Brook WRW (site 4)



Appendix 4.4 (Continued) k-dominance curves for the Trent catchment

River Sence Site = 6 Flow & site direction: North to Southwest (Source to Mouth)

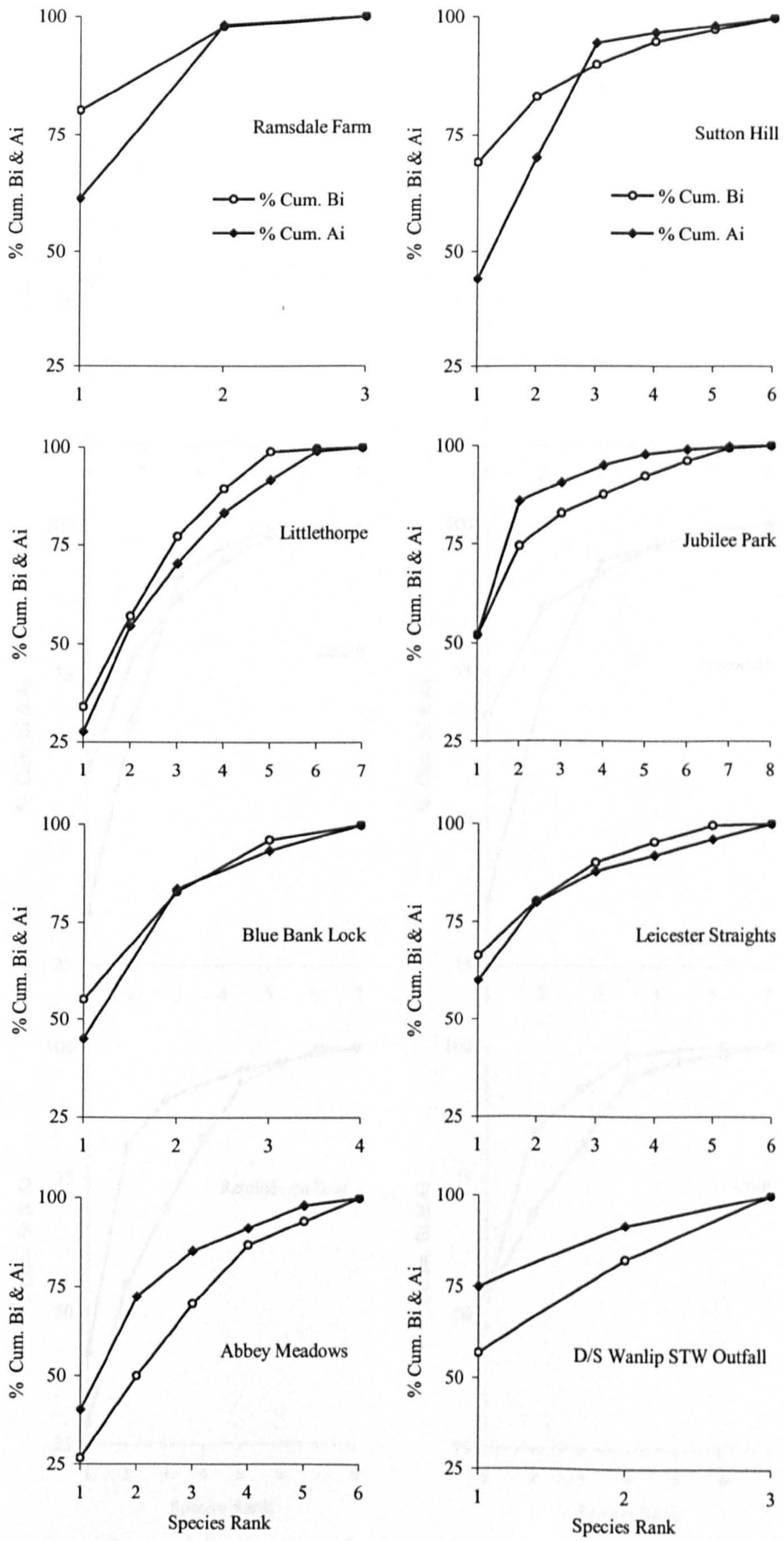
No fish were found at Heather Butterley Brick Works (site 1)



Appendix 4.4 (Continued) k-dominance curves for the Trent catchment

River Soar Site = 15 Flow & site direction: South to North (Source to Mouth)

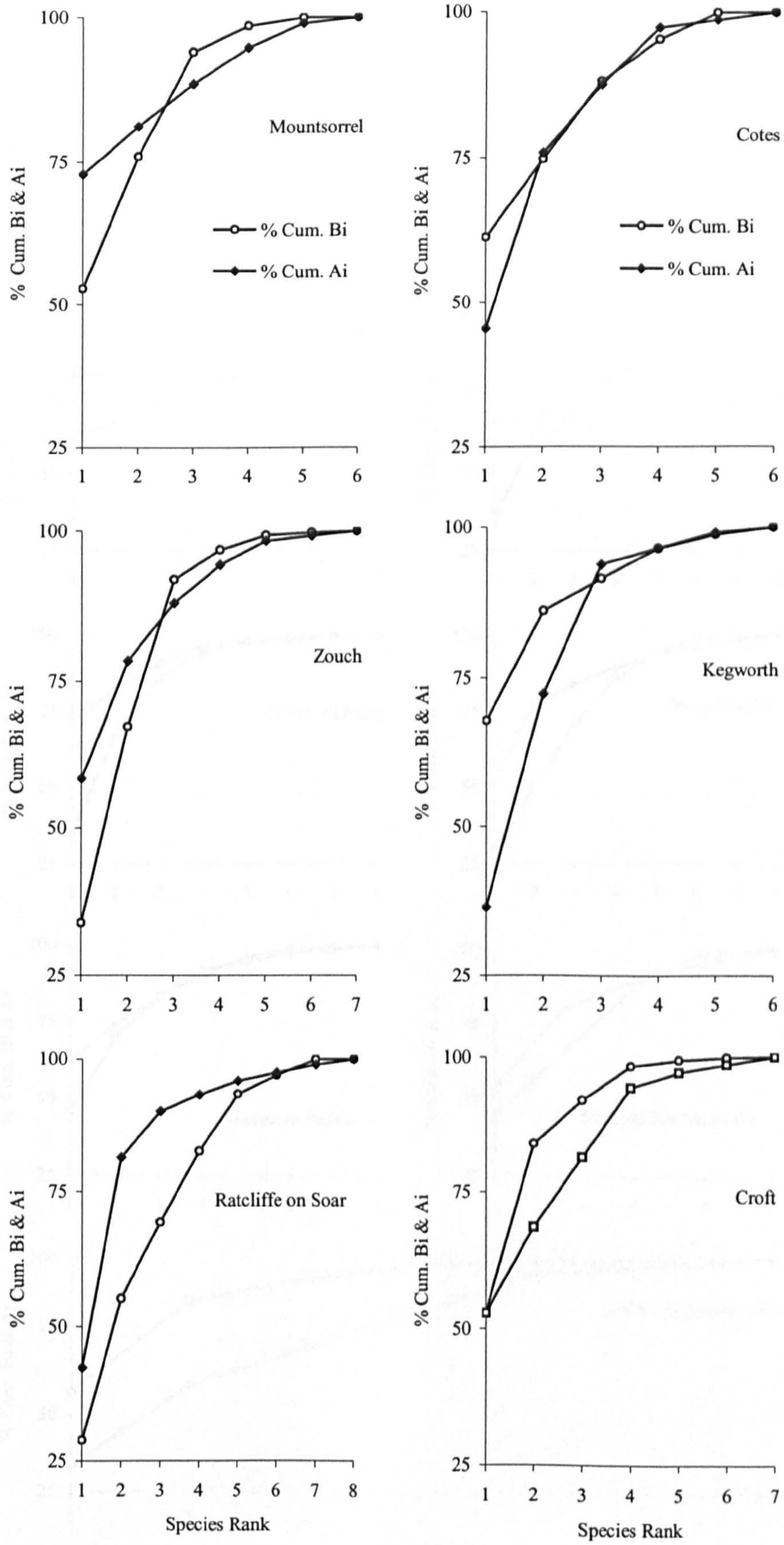
No fish were found at Barrow on Soar (site 11)



Appendix 4.4 (Continued) k-dominance curves for the Trent catchment

River Soar Site = 15 Flow & site direction: South to North (Source to Mouth)

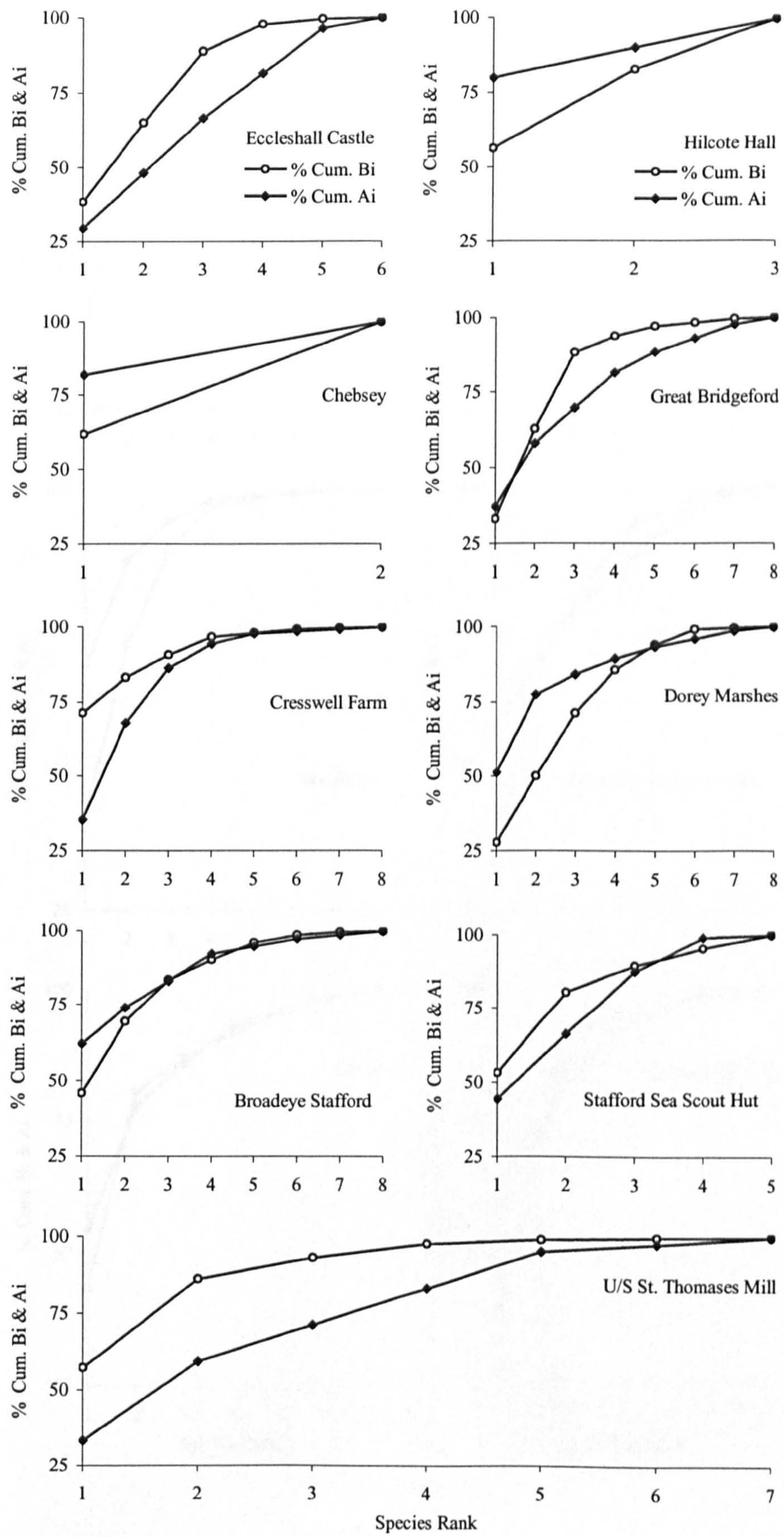
No fish were found at (Barrow on Soar (site 11)



Appendix 4.4 (Continued) k-dominance curves for the Trent catchment

River Sow Total site = 9

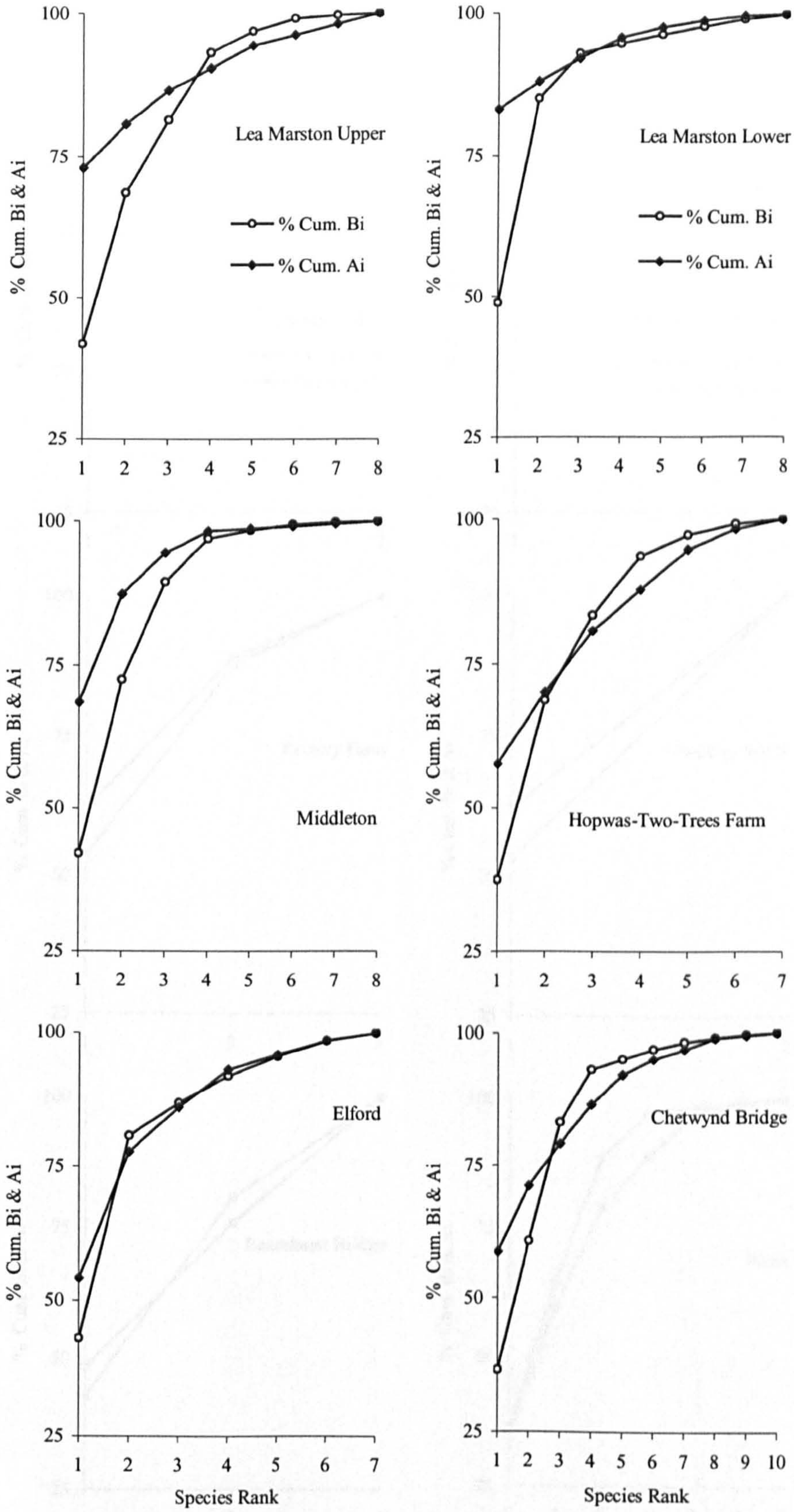
Flow & site direction: North to Southeast (Source to Mouth)



Appendix 4.4 (Continued) k-dominance curves for the Trent catchment

River Tame Total site = 6

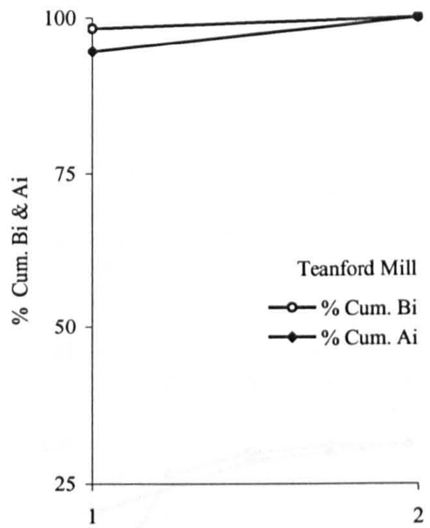
Flow & site direction: South to North (Source to Mouth)

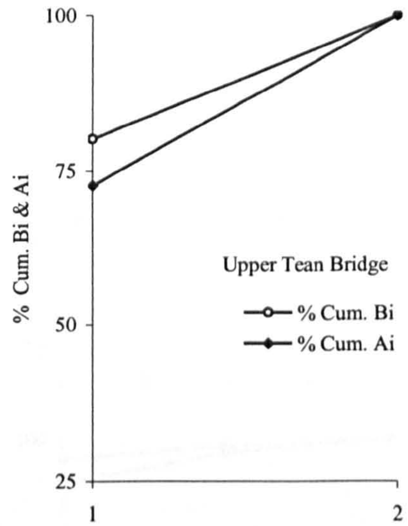


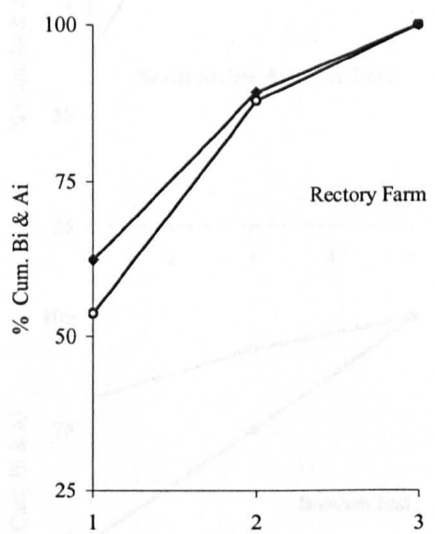
Appendix 4.4 (Continued) k-dominance curves for the Trent catchment

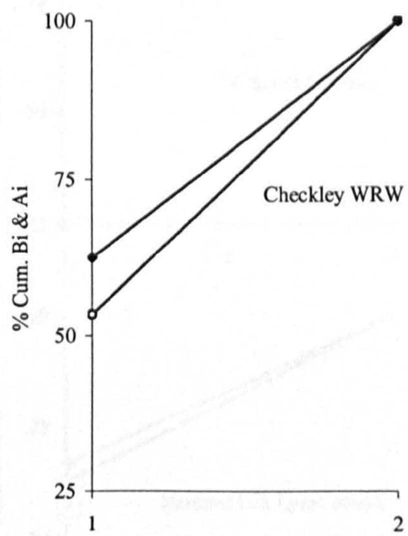
River Tean Site = 9 Flow & site direction: North to South (Source to Mouth)

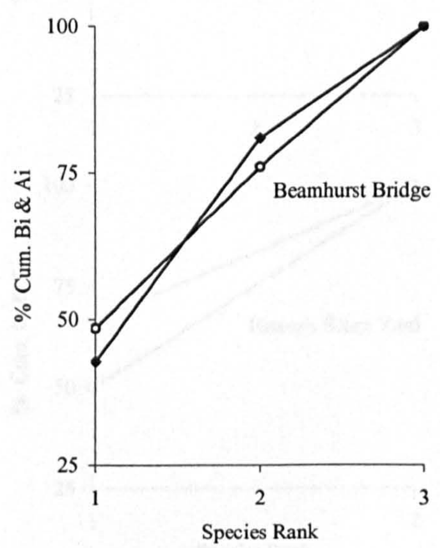
No fish were found at Fole Hall & Fole D/S Creamery (sites 6 & 7) and single species caught from Litley's Farm (site 1)

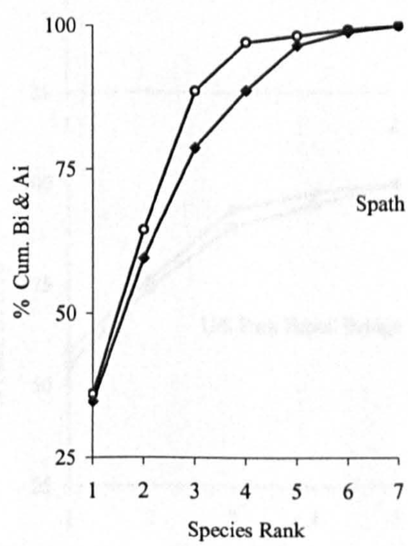








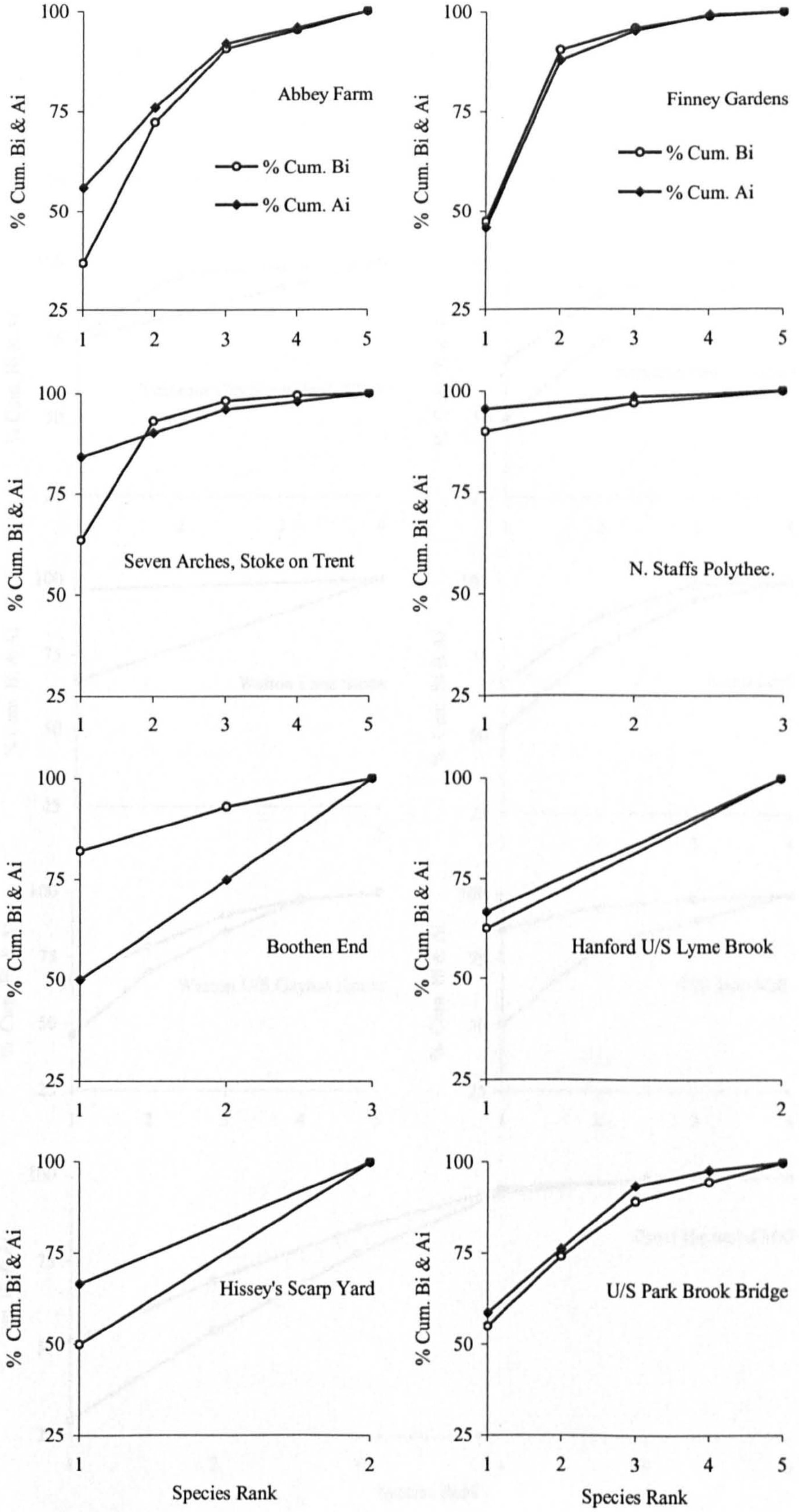




Appendix 4.4 (Continued) k-dominance curves for the Trent catchment

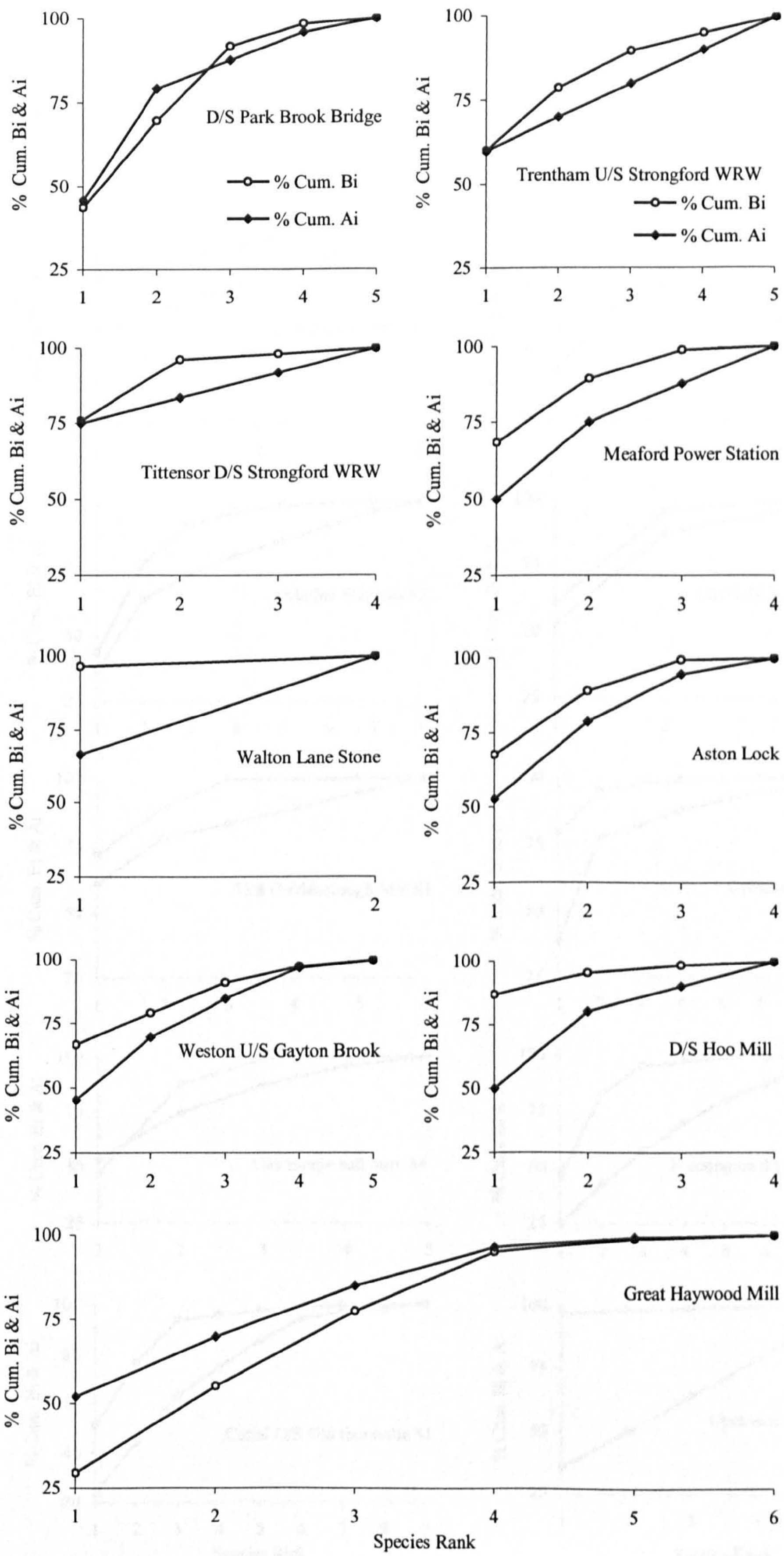
River Trent Site = 20 Flow & site direction: North to Southeast (Source to Mouth)

No fish were found at U/S Hoo Mill (site 18) and single species caught from Norton Green and Sandon (sites 1 & 16)



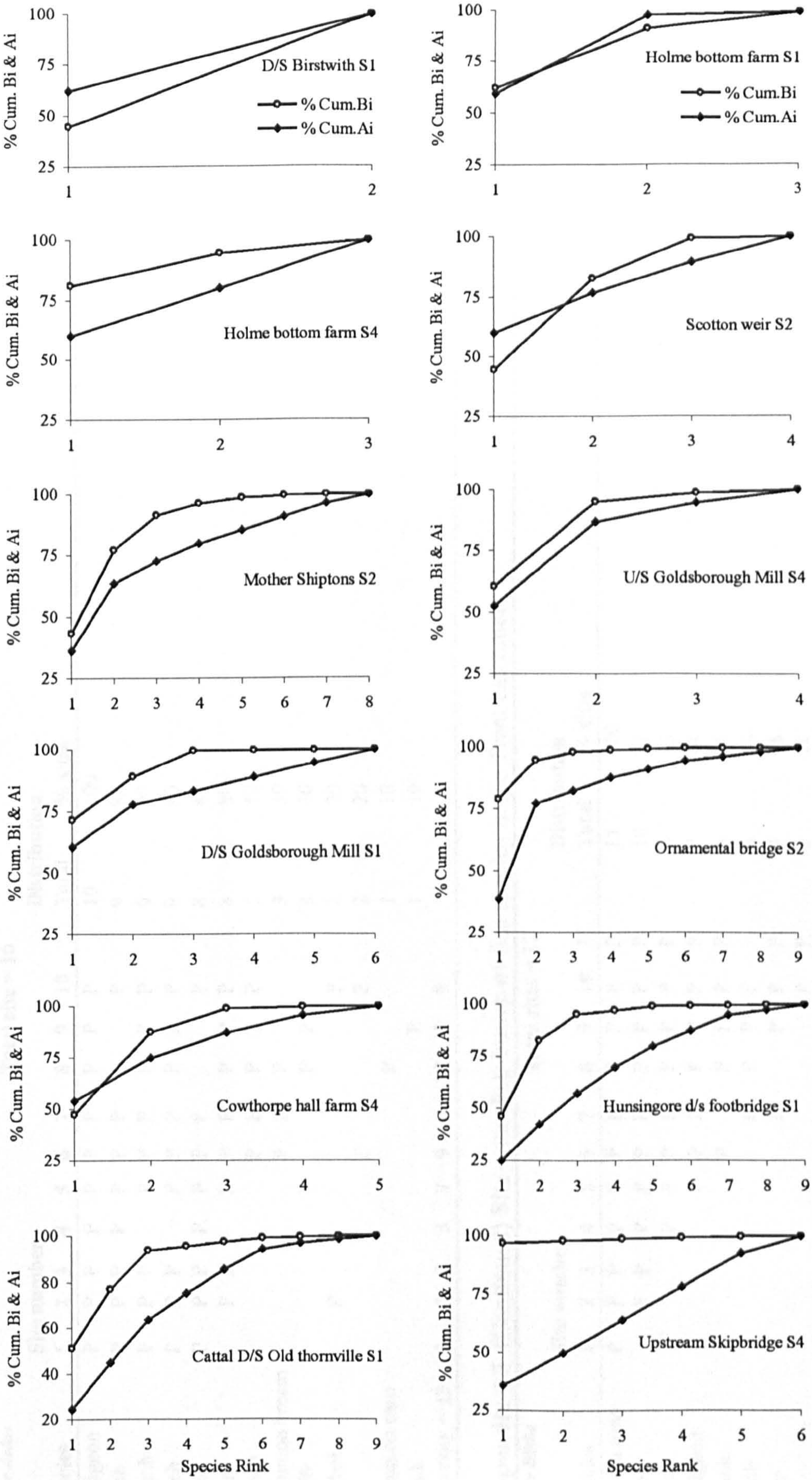
Appendix 4.4 (Continued) k-dominance curves for the Trent catchment

River Trent Site = 20 Flow & site direction: North to Southeast (Source to Mouth)



Appendix 4.4 (Continued) k-dominance curves for the Yorkshire Ouse catchment

River Nidd Total site: 182 (Twelve sites were chosen to represent the river)



Appendix 4.5 Site-wise distribution of fish species in the Trent catchment

River Anker						Total site = 10						
	Site number										**Distribution**	
Species	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**Total**	**% sites**
Gudgeon	P	P	P	P	P	P	P	P	P	P	10	100
Chub	P	P	P	P	P	P	P	P		P	9	90
Roach	P	P	P		P	P	P	P	P	P	9	90
Perch	P	P	P		P	P	P	P	P	P	9	90
Dace	P	P	P	P	P	P	P			P	8	80
Pike		P	P		P	P	P	P	P	P	8	80
Tench						P	P	P	P	P	5	50
Common bream						P	P	P			3	30
Ruffe					P			P	P		3	30
Barbel		P								P	2	20
Eel						P				P	2	20
Common carp								P			1	10
Bleak									P		1	10
Diversity = 13	5	7	6	3	7	9	8	9	7	9		

Appendix 4.5 (Continued) Site-wise distribution of fish species in the Trent catchment

River Blithe						Total sites = 11							
	Site number											**Distribution**	
Species	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**Total**	**% sites**
Brown trout	P	P	P	P	P	P	P	P	P	P	P	11	100
Chub		P	P	P	P	P	P	P	P	P	P	10	91
Dace				P	P	P	P	P	P	P	P	8	73
Gudgeon					P	P	P	P	P	P	P	7	64
Roach						P		P	P	P	P	5	45
Perch							P	P	P	P		4	36
Pike							P		P	P	P	4	36
Eel										P	P	2	18
Grayling						P						1	9
Diversity = 9	1	2	2	3	4	6	6	6	7	8	7		

Appendix 4.5 (Continued) Site-wise distribution of fish species in the Trent catchment

River Blythe **Total site = 9**

Species	Site Number 1	2	3	4	5	6	7	8	9	Distribution Total	% sites
Chub		P	P	P	P	P	P	P	P	8	89
Dace		P	P	P	P	P	P	P	P	8	89
Roach			P	P	P	P	P	P	P	7	78
Perch			P		P	P	P	P	P	6	67
Gudgeon			P			P	P	P	P	5	56
Pike			P	P	P				P	4	44
Rudd						P	P	P	P	4	44
Eel						P	P	P	P	4	44
Rainbow trout						P	P	P	P	4	44
Common bream					P				P	2	22
Common carp						P			P	2	22
Tench						P			P	2	22
Diversity = 12	0	2	6	4	6	10	8	8	12		

River Churnet **Total sites = 16**

Species	Site number 1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	Distribution Total	% sites
Brown trout	P	P					P				P	P	P	P	P	P	9	56
Roach		P						P	P		P	P	P	P	P	P	9	56
Dace						P	P	P			P		P	P		P	7	44
Grayling											P	P	P	P	P	P	6	38
Perch							P	P					P	P	P	P	6	38
Pike										P		P		P	P	P	5	31
Gudgeon												P	P	P	P	P	5	31
Chub							P				P	P				P	4	25
Common bream														P	P	P	3	19
Rainbow trout	P																1	6
Rudd		P															1	6
Diversity = 11	2	3	0	0	0	1	4	3	1	1	5	6	6	8	7	9		

Appendix 4.5 (Continued) Site-wise distribution of fish species in the Trent catchment

River Cole								Total sites = 14							
	Site number													**Distribution**	
Species	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**Total**	**% sites**
Chub						P	P	P		P	P	P	P	P	8	57
Gudgeon							P	P	P	P	P	P	P	P	8	57
Roach							P	P		P	P	P	P	P	7	50
Dace										P	P	P	P	P	5	36
Perch							P	P	P	P					4	29
Tench							P	P	P						3	21
Eel							P				P	P			3	21
Pike	P														1	7
Crucian carp							P								1	7
Goldfish								P							1	7
Rudd							P								1	7
Diversity = 11	1	0	0	0	0	1	8	6	3	5	5	5	4	4		

River Derwent																	
	Site number							**Total sites = 15**								**Distribution**	
Species	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**Total**	**% sites**
Grayling		P	P	P	P	P	P	P	P	P	P	P	P			12	80
Brown trout	P	P	P	P	P	P	P	P	P	P	P					11	73
Chub									P	P	P	P	P	P	P	7	47
Dace							P					P	P	P	P	5	33
Perch									P		P	P	P	P		5	33
Pike									P		P	P	P	P		5	33
Common bream									P				P	P	P	4	27
Roach													P	P	P	3	20
Rainbow trout					P	P	P									3	20
Barbel												P	P			2	13
Tench														P	P	2	13
Brook lamprey		P									P					2	13
Gudgeon														P		1	7
Bleak															P	1	7

Ruffe														P		1	7
Eel														P		1	7
Diversity = 16	1	3	2	2	3	3	4	2	6	3	6	6	8	10	6		

Appendix 4.5 (Continued) Site-wise distribution of fish species in the Trent catchment

River Idle — **Total sites = 5**

	Site number					Distribution	
Species	**1**	**2**	**3**	**4**	**5**	**Total**	**% sites**
Chub		P	P	P		3	60
Pike	P	P		P		3	60
Gudgeon				P		1	20
Dace				P		1	20
Roach				P		1	20
Common bream			P			1	20
Rudd	P					1	20
Bleak				P		1	20
Eel	P					1	20
Diversity = 9	3	2	2	6	0		

River Mease — **Total sites = 7**

	Site number							Distribution	
Species	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**Total**	**% sites**
Chub	P	P	P	P	P	P	P	7	100
Dace	P	P	P	P	P	P	P	7	100
Pike	P	P	P	P	P	P	P	7	100
Gudgeon	P	P	P	P		P	P	6	86
Roach	P		P	P	P	P	P	6	86
Perch	P		P	P	P	P	P	6	86
Eel		P		P	P	P	P	5	71
Tench							P	1	14
Common carp					P			1	14
Common bream	P							1	14
Ruffe						P		1	14
Diversity = 11	7	5	6	7	7	8	8		

Appendix 4.5 (Continued) Site-wise distribution of fish species in the Trent catchment

River Penk	**Site number**							**Total sites = 11**				**Distribution**	
Species	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**Total**	**% sites**
Eel				P	P	P	P	P	P	P	P	8	73
Dace					P	P	P	P	P	P	P	7	64
Gudgeon					P	P	P	P	P	P	P	7	64
Roach					P	P	P	P	P	P	P	7	64
Chub						P	P	P	P	P	P	6	55
Perch						P	P	P	P	P	P	6	55
Pike						P	P	P	P	P	P	6	55
Barbel											P	1	9
Ruffe								P				1	9
Brown trout								P				1	9
Diversity = 10	0	0	0	1	4	7	7	9	7	7	8		

River Sence	**Site number**				**Total sites = 6**		**Distribution**	
Species	**1**	**2**	**3**	**4**	**5**	**6**	**Total**	**% sites**
Chub		P	P	P	P	P	5	83
Gudgeon			P	P	P	P	4	67
Dace			P	P	P	P	4	67
Perch			P	P	P	P	4	67
Roach			P	P		P	3	50
Brown trout		P	P	P			3	50
Rainbow trout		P	P	P			3	50
Common bream			P	P	P		3	50
Eel					P	P	2	33
Pike				P		P	2	33
Diversity = 10	0	3	8	9	6	7		

Appendix 4.5 (Continued) Site-wise distribution of fish species in the Trent catchment

River Soar	Site number							Total sites = 15							Distribution	
Species	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**Total**	**% sites**
Perch	P	P	P	P	P	P	P	P	P	P		P	P	P	P	14	93
Roach		P	P	P	P	P	P	P	P	P		P	P	P	P	13	87
Dace		P	P	P	P	P			P	P		P	P	P	P	11	73
Chub		P	P	P	P					P		P	P	P	P	9	60
Pike				P	P		P	P		P			P	P	P	8	53
Barbel		P			P	P						P			P	5	33
Bleak												P	P	P	P	4	27
Tench				P	P			P		P						4	27
Gudgeon	P		P	P	P											4	27
Common bream							P	P							P	3	20
Common carp							P	P					P			3	20
Brown trout	P	P	P													3	20
Crucian carp							P									1	7
Rudd			P													1	7
Diversity = 14	3	6	7	7	8	4	6	6	3	6	0	6	7	6	8		

River Sow	Site number							Total site = 9		Distribution	
Species	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**Total**	**% sites**
Chub	P	P	P	P	P	P	P	P	P	9	100
Pike	P	P		P	P	P	P	P	P	8	89
Perch	P			P	P	P	P	P	P	7	78
Dace		P	P	P	P	P	P		P	7	78
Roach	P			P	P	P	P	P	P	7	78
Gudgeon	P				P	P	P	P	P	6	67
Eel				P	P		P		P	4	44
Tench				P		P	P			3	33
Common bream	P			P		P				3	33
Barbel					P					1	11
Diversity = 10	6	3	2	8	8	8	8	5	7		

Appendix 4.5 (Continued) Site-wise distribution of fish species in the Trent catchment

River Tame **Total sites = 6**

Species	**1**	**2**	**3**	**4**	**5**	**6**	**Distribution Total**	**% sites**
Roach	P	P	P	P	P	P	6	100
Gudgeon	P	P	P	P	P	P	6	100
Chub		P	P	P	P	P	5	83
Dace		P	P	P	P	P	5	83
Perch	P	P	P		P	P	5	83
Pike	P		P	P		P	4	67
Eel		P	P	P	P		4	67
Bleak				P	P	P	3	50
Tench	P	P				P	3	50
Rainbow trout	P	P					2	33
Common bream	P					P	2	33
Barbel			P				1	17
Rudd	P						1	17
Silver bream						P	1	17
Diversity = 14	8	8	8	7	7	10		

River Tean **Total site = 9**

Species	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**Distribution Total**	**% sites**
Brown trout	P	P	P	P	P			P	P	7	78
Perch		P	P	P				P	P	5	56
Grayling				P	P			P	P	4	44
Chub									P	1	11
Dace									P	1	11
Gudgeon									P	1	11
Roach									P	1	11
Diversity = 7	1	2	2	3	2	0	0	3	7		

Appendix 4.5 (Continued) Site-wise distribution of fish species in the Trent catchment

River Trent	**Total sites = 20**																				
	Site number																			**Distribution**	
Species	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**	**Total**	**% sites**
Gudgeon	P	P				P	P	P	P	P	P	P	P	P	P				P	P	14	70
Roach		P	P	P	P				P	P	P	P	P		P	P	P			P	13	65
Dace		P	P	P			P		P	P	P		P		P		P		P	P	12	60
Chub		P			P				P			P	P		P		P		P	P	9	45
Pike			P	P	P	P				P		P								P	7	35
Perch			P	P							P						P				4	20
Brown trout	P									P							P				3	15
Eel		P							P										P		3	15
Tench				P		P															2	10
Common bream											P									P	2	10
Barbel														P							1	5
Rudd								P													1	5
Diversity = 12	2	5	4	5	3	3	2	2	5	5	5	4	4	2	3	1	4	0	4	6		

Appendix 4.6 Site-wise distribution of fish species in the Yorkshire Ouse catchment

River Aire	Site number							Total site - 26																			Distribution	
Species	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**	**21**	**22**	**23**	**24**	**25**	**26**	**Total**	**% sites**
Brown trout		P	P	P	P	P	P	P	P	P	P	P	P	P	P												14	54
Roach															P	P	P		P	P	P	P	P	P	P	P	11	42
Chub													P	P	P	P	P		P	P	P					P	9	35
Gudgeon														P	P	P			P	P	P		P		P	P	9	35
Dace														P	P	P			P	P							5	19
Perch															P								P		P	P	4	15
Pike														P	P		P										3	12
Rainbow trout									P																		1	4
Grayling														P													1	4
Diversity = 9	0	1	1	1	1	1	1	1	2	1	1	1	2	6	7	4	3	0	4	4	3	1	3	1	3	4		

River Nidd — Total sites = 182

Species	Distribution Total site	% site
Chub	128	70
Dace	113	62
Pike	111	61
Perch	105	58
Gudgeon	100	55
Grayling	88	48
Roach	86	47
Brown trout	67	37
Ruffe	46	25
Barbel	33	18
Bream	3	2
Bleak	3	2
Tench	2	1
Diversity = 13 species		

Appendix 5.1 Metrics adopted for this study, used by other authors in different parts of the World (Total IBI versions = 32, m = metric)

Metrics used in this study	Reference (Number of metrics, Waterbody / Country)
1. Total number of native fish species (16 versions, 50%)	Appelberg *et al.* 2000 (7 m, Stream, Sweden, as abundance / 100 m^2 of native species), Kestemont *et al.* 2000 (12 m, River, Belgium), Boet *et al.* 1999 (15 m, River, France), Ganasan & Hughes 1998 (12 m, River, India), Hughes *et al.* 1998 (16 m, Stream, Oregon, USA), Koizumi & Matsumiya 1997 (10 m, Stream, Japan), Hay *et al.* 1996 (19 m, River, Namibia), Lyons *et al.* 1996 (22 m, Cold water River, Wisconsin, USA), Lyons *et al.* 1995 (10 m, Streams & Rivers, West-Central Mexico), Minns *et al.* 1994 (12 m, Great Lakes, USA), Lyons 1992 (12 m, Warmwater stream, Wisconsin, USA), Karr 1991 (12 m, River, Midwest, USA), Crumby *et al.* 1990 (12 m, River, Tennessee, USA), Steedman 1988 (10 m, River, Canada), Moyle *et al.* 1986 (8 m, River, California, USA, as % native species)
2. Percentage of individuals as non-natives / introduced species (10, 3%)	Appelberg *et al.* 2000, Ganasan & Hughes 1998, Hughes *et al.* 1998 (as % alien species), Koizumi & Matsumiya 1997 (as % of immigrant species), Lyons *et al.* 1995, Minns *et al.* 1994, Bramblett & Fausch 1991, Karr 1991 (% as hybrids), Crumby *et al.* 1990,
3. Number of intolerant species (24, 75%)	Kestemont *et al.* 2000, Boet *et al.* 1999, Toham & Teugels 1999 (12 m, River, Cameroon, West Africa), Ganasan & Hughes 1998, Hughes *et al.* 1998 (as Number of sensitive species), Koizumi & Matsumiya 1997, Lyons *et al.* 1996, Hugueny *et al.* 1996 (12 m, River, Guinea, West Africa), Bowen *et al.* 1996 (9 m, River, Alabama, USA), Didier & Kestemont 1996 (13 m, River, Belgium), Lyons *et al.* 1995 (as Number of sensitive species), Simon & Emery 1995 (12 m, Great Rivers, Ohio, USA, as Number of sensitive species), Oberdorff & Porcher 1994 (10 m, Stream, France, as % of sculpin, intolerant species), Minns *et al.* 1994, Goldstein *et al.* 1994 (12 m, Red River basin, North & South Dakota, Minnesota, USA), Oberdorff & Hughes 1992 (12 m, River, France), Karr 1991, Crumby *et al.* 1990, Miller *et al.* 1988 (12 m, River, Midwest, USA), Angermeier & Schlosser 1987 (12 m, River, Illinois, USA), Angermeier & Karr 1986 (12 m, River, Illinois, Ohio & West Virginia, USA), Moyle *et al.* 1986 (as Sculpin abundance) Fausch *et al.* 1984 (12 m, Stream of Illinois, Kentucky, Michigan, Nebraska, & South & North Dakota, USA)

4. Percentage of individuals as tolerant species (18, 56%)	Boet *et al.* 1999, Toham & Teugels 1999, Ganasan & Hughes 1998, Hughes *et al.* 1998, Koizumi & Matsumiya 1997, Didier & Kestemont 1996, Lyons *et al.* 1996, Lyons *et al.* 1995, Simon & Emery 1995, Oberdorff & Porcher 1994 (as % eel and roach), Goldstein *et al.* 1994 (as green sunfish), Oberdorff & Hughes 1992 (as % of roach), Lyons 1992, Bramblett & Fausch 1991 (9 m, River, Colorado, USA, Karr 1991 (% as green sunfish), Crumby *et al.* 1990, Fausch *et al.* 1984 (as green sunfish)
5. Number of water-column species (7, 22%)	Ganasan & Hughes 1998, Hughes *et al.* 1998 (as native), Koizumi & Matsumiya 1997, Didier & Kestemont 1996, Lyons *et al.* 1995, Oberdorff & Hughes 1992, Karr 1991 (Number of sunfish species)
6. Number of benthic species (12, 38%)	Kestemont *et al.* 2000, Boet *et al.* 1999, Ganasan & Hughes 1998, Hughes *et al.* 1998 (as native), Koizumi & Matsumiya 1997 (as Number of sub- and benthic species of cyprinid), Hay *et al.* 1996, Bowen *et al.* 1996 (as % of individuals as benthic fluvial specialists), Didier & Kestemont 1996, Lyons *et al.* 1995, Oberdorff & Hughes 1992, Karr 1991 (Number of darter species)
7. Percentage of individuals as rheophilic species (5, 16%)	Boet *et al.* 1999, Toham & Teugels 1999, Hay *et al.* 1996, Hocutt *et al.* 1995 (10 m, River, Namibia, as Number of pelagic / rheophilic species), Harris 1995 (12 m, River, Australia, as Number of riffle benthic species)
8. Percentage of individuals preferring vegetated areas (3, 9%)	Hughes & Oberdorff 1999, Hay *et al.* 1996, Koizumi & Matsumiya 1997,
9. Percentage of individuals as gravel spawners (10, 31%)	Appelberg *et al.* 2000 (as reproduction of salmonids), Kestemont *et al.* 2000 (as specialised spawners), Boet *et al.* 1999 (as lithophils), Hughes *et al.* 1998 (as Number of nonguarding lithophils), Koizumi & Matsumiya 1997, Didier & Kestemont 1996 (as % of individuals lithophil or phytophil), Lyons *et al.* 1996 (as simple lithophilic spawners), Simon & Emery 1995 (as % simple lithophils), Lyons 1992 (as simple lithophilic spawners), Oberdorff & Hughes 1992

10. Percentage of individuals as omnivores (26, 81%)	Kestemont *et al.* 2000, Boet *et al.* 1999, Toham & Teugels 1999, Ganasan & Hughes 1998, Hughes *et al.* 1998, Koizumi & Matsumiya 1997, Lyons *et al.* 1996, Didier & Kestemont 1996, Lyons *et al.* 1995, Hay *et al.* 1996, Hugueny *et al.* 1996, Simon & Emery 1995, Goldstein *et al.* 1994, Minns *et al.* 1994 (as % generalist biomass), Oberdorff & Porcher 1994, Lyons 1992, Oberdorff & Hughes 1992, Bramblett & Fausch 1991, Karr 1991, Crumby *et al.* 1990, Miller *et al.* 1988, Steedman 1988, Angermeier & Schlosser 1987, Leonard & Orth 1986 (7 m, Coolwater streams, West Virginia, USA), Angermeier & Karr 1986, Fausch *et al.* 1984,
11. Percentage of individuals as invertivores (21, 66%)	Boet *et al.* 1999, Toham & Teugels 1999, Bowen *et al.* 1996 (as % of individuals as insectivorous cyprinids), Lyons *et al.* 1996, Didier & Kestemont 1996, Hay *et al.* 1996, Hugueny *et al.* 1996, Simon & Emery 1995 (% insectivores), Minns *et al.* 1994 (% specialist biomass), Goldstein *et al.* 1994 (as % of individuals as insectivorous cyprinids), Oberdorff & Porcher 1994, Oberdorff & Hughes 1992, Lyons 1992 (as insectivores), Bramblett & Fausch 1991, Karr 1991 (as insectivores), Crumby *et al.* 1990 (as insectivores), Miller *et al.* 1988 (as % of individuals as insectivorous cyprinids), Angermeier & Schlosser 1987 (as % of individuals as insectivorous cyprinids), Leonard & Orth 1986 (as % of individuals as insectivorous cyprinids), Angermeier & Karr 1986 (as % of individuals as insectivorous cyprinids), Fausch *et al.* 1984 (as % of individuals as insectivorous cyprinids)

12. Percentage of individuals as piscivores (21, 66%)	Kestemont *et al.* 2000, Toham & Teugels 1999 (as carnivores), Ganasan & Hughes 1998 (as top carnivores), Hughes *et al.* 1998 (as native), Koizumi & Matsumiya 1997, Hay *et al.* 1996, Hugueny *et al.* 1996, Lyons *et al.* 1996 (as top carnivores), Didier & Kestemont 1996 (as top carnivores), Simon & Emery 1995 (as % carnivores), Minns *et al.* 1994 (as % piscivore biomass), Goldstein *et al.* 1994 (as top carnivores), Lyons 1992 (as top carnivores), Oberdorff & Hughes 1992 (as top carnivores), Karr 1991, Crumby *et al.* 1990, Miller *et al.* 1988 (as top carnivores), Steedman 1988 (as large piscivores), Angermeier & Schlosser 1987, Angermeier & Karr 1986, Fausch *et al.* 1984 (as top carnivores),
13. Number of individuals of long-lived species (No. 100 m^{-2}) (Chub & common bream) (3, 9%)	Toham & Teugels 1999 (as Number of benthic siluriform species), Hugueny *et al.* 1996 (as Number of large benthic siluriform species), Karr 1991 (as Number of sucker species)
14. Number of individuals in a sample (No. 100 m^{-2}) (22, 69%)	Boet *et al.* 1999 (as density, number individuals/100m^2), Toham & Teugels 1999, Ganasan & Hughes 1998 (as total number of individuals), Hughes *et al.* 1998 (as total number of individuals), Bowen *et al.* 1996 (as density, mean number per PAE sample), Hugueny *et al.* 1996 (as Number of individuals), Lyons *et al.* 1996 (as CPUE), Lyons *et al.* 1995 (as number per half-hour sampling), Simon & Emery 1995 (as CPUA), Goldstein *et al.* 1994, Oberdorff & Porcher 1994 (as catch per 100 m^2 of sampling), Lyons 1992 (No./ 300 m sampled, excluding tolerants), Oberdorff & Hughes 1992 (as catch per minute of sampling), Bramblett & Fausch 1991, Karr 1991, Miller *et al.* 1988, Steedman 1988, Angermeier & Schlosser 1987 (as total number of individuals), Leonard & Orth 1986, Angermeier & Karr 1986, Moyle *et al.* 1986 (as total fish abundance), Fausch *et al.* 1984,
15. Total biomass (g m^{-2}) (3, 9%)	Kestemont *et al.* 2000 (as estimated biomass, kg/ha), Didier & Kestemont 1996 (as estimated biomass, kg/ha), Oberdorff & Porcher 1994

Appendix 5.2 Rivers Ecosystem Classification based on chemical characteristics of water (after EA, LEAP 1998d)

Quality class	DO % saturation 10 percentile	BOD (ATU) mg/l 90 percentile	Total Ammonia mg N/l 90 percentile	Un-ionised Ammonia mg N/l 95 percentile	pH lower limit as 5 percentile; upper limit as 95 percentile	Hardness mg/l $CaCO_3$	Dissolved Copper µg/l 95 percentile	Total Zinc µg/l 95 percentile
RE1	80	2.5	0.25	0.021	6.0-9.0	≤10	5	30
						>10 & ≤50	22	200
						>50 & ≤100	40	300
						≤100	112	500
RE2	70	4.0	0.6	0.021	6.0-9.0	≤10	5	30
						>10 & ≤50	22	200
						>50 & ≤100	40	300
						≤100	112	500
RE3	60	6.0	1.3	0.021	6.0-9.0	≤10	5	300
						>10 & ≤50	22	700
						>50 & ≤100	40	1000
						≤100	112	2000
RE4	50	8.0	2.5	-	6.0-9.0	≤10	5	300
						>10 & ≤50	22	700
						>50 & ≤100	40	1000
						≤100	112	2000
RE5	20	15.0	9.0	-	-	-	-	-
Unclassified	-	-	-	-	-	-	-	-

Appendix 5.2 (Continued) River Ecosystem Classification (after EA, LEAP 1998d)

Quality class	Characteristics
RE1	Water of very good quality suitable for all fish species
RE2	Water of good quality suitable for all fish species
RE3	Water of fair quality suitable for high class coarse fish populations
RE4	Water of fair quality suitable for coarse fish populations
RE5	Water of poor quality which is likely to limit coarse fish populations
Unclassified	Water of bad quality in which fish are unlikely to be present, or insufficient data available by which to classify water quality (e.g. small streams not regularly sampled)

Appendix 5.2 (Continued) General Quality Assessment (GQA) for biological study (after EA, LEAP 1998d)

Grade / Quality class	Outline description
A – Very good	Biology similar (or better) than expected. High diversity of taxa, usually with several species in each. Dominance of one taxon rare.
B - Good	Biology falls a little short of that expected. Small reduction in pollution sensitive taxa. Moderate increase in individual species in pollution tolerant taxa.
C – Fairly good	Biology worse than expected. Many sensitive taxa absent, or number of individual species reduced. Marked rise in individual species in pollution tolerant taxa present, some with high numbers of individual species.
D - Fair	Biology worse than expected. Sensitive taxa scarce. Pollution tolerant taxa present, some with high numbers of individual species.
E - Poor	Biology restricted to pollution tolerant species with some taxa dominant in terms of the numbers of individual species. Sensitive taxa will be rare or absent.
F - Bad	Biology limited to small number of very pollution tolerant taxa, often only worms, midge larvae, leeches and the water hoglouse. They may be present in very high numbers. In the worst case, no life presents.

Appendix 5.3 IBI scores on the basis of 15 selected metrics for English rivers

The Thames catchment

River Cherwell

Site No.	Location	IBI Score	Integrity Class	Comment
1	West Farndon Mill	23	Poor	
2	Trafford Bridge	32	Fair	n = 13
3	Slat Mill	49	Good	Score range= 22 - 47
4	Spiceball	42	Good	Average = 35
5	Tramroad Industrial Estate	22	Poor	SD = 8.53
6	Footbridge near M40	25	Poor	
7	Twyford Mill	30	Fair	***Fair Class River***
8	Millhouse Farm	34	Fair	
9	Sor Brook Confluence	38	Fair	
10	Somerton	39	Fair	
11	Lower Hayford	47	Good	
12	Bunkers Hill	44	Good	
13	Angel and Greyhound Meadows	33	Fair	
	Average	**35**	**Fair**	

Appendix 5.3 IBI scores on the basis of 15 selected metrics for English rivers

River Evenlode		IBI Score	Integrity Class	Comment
1	Evenlode	34	Fair	
2	Oddington	29	Fair	n = 20
3	Kinhham	39	Fair	Score range= 18 - 55
4	Bledington	43	Good	Average =41
5	Bruern Abbey	41	Fair	SD = 7.39
6	Lyneham	41	Fair	
7	Shipton-under-Wychwood	37	Fair	***Fair Class River***
8	Ascott-under-Wychwood	41	Fair	
9	Chadlington	41	Fair	
10	Charlbury	43	Good	
11	Finstock Station	42	Good	
12	Ashford Mill	39	Fair	
13	Lower Riding Farm	41	Fair	
14	Combe	49	Good	
15	D/S Blenheim Saw Mill	45	Good	
16	Bladon	41	Fair	
17	Goose Eye Farm	18	Poor	
18	U/S A40	42	Good	
19	Canal Stream (Cassington)	55	Good	
20	Mill Stream (Cassington)	49	Good	
	Average	**41**	**Fair**	

Appendix 5.3 (Continued) IBI scores on the basis of 15 selected metrics for English rivers

The Thames catchment

River Stort

Site No.	Location	IBI Score	Integrity Class	Comment
1	Hazel End	32	Fair	
2	Grange Paddocks	30	Fair	n = 16
3	Bishops Stortford	34	Fair	Score range= 23 - 50
4	Spellbrook	26	Poor	Average = 35
5	Thorley Marsh	36	Fair	SD = 7.80
6	Tednambury	31	Fair	
7	Tednambury Mill Overflow	42	Good	***Fair Class River***
8	Sawbridgeworth Lock	30	Fair	
9	Sawbridgeworth Mead Ditch	43	Good	
10	Pishiobury Meander	45	Good	
11	Harcamlow Way	26	Poor	
12	Eastwick Lodge Farm	40	Fair	
13	A414, Harlow Road	27	Poor	
14	Briggens	50	Good	
15	St. Alban's Sand and Gravel	43	Good	
16	Brick Lock	23	Poor	
	Average	**35**	**Fair**	

Appendix 5.3 (Continued) IBI scores on the basis of 15 selected metrics for English rivers

River Thame

Site No.	Location	IBI Score	Integrity Class	Comment
1	Weedon	26	Poor	n = 18
2	Stone Bridge	36	Fair	Score range= 26 - 55
3	Lower Hartwell	36	Fair	Average = 45
4	Eythorpe	34	Fair	SD = 7.85
5	Ridge Barn Farm	47	Good	
6	Nether Winchendon	41	Fair	***Good Class River***
7	U/S Notley Abbey	49	Good	
8	Scotsgrove Brook Confluence	50	Good	
9	Shabbington (West arm)	43	Good	
10	Shabbington (East Arm)	36	Fair	
11	Ickford	52	Good	
12	Waterstock	48	Good	
13	Cuddesdon	50	Good	
14	Cuddesdon Mill Channel	47	Good	
15	Chippinghurst Manor	48	Good	
16	Chiselhampton	55	Good	
17	Drayton St. Leonard	50	Good	
18	Dorchester	55	Good	
	Average	**45**	**Good**	

Appendix 5.3 (Continued) IBI scores on the basis of 15 selected metrics for English rivers

The Thames catchment

River Windrush

Site no.	**Location**	**IBI Score**	**Integrity Class**	**Comment**
1	Kineton	23	Poor	
2	Guiting Power	25	Poor	n = 19
3	Harford Bridge	25	Poor	Score range= 23 - 52
4	Up Stream of A429	29	Fair	Average = 41
5	D/S Dikler Confluence	52	Good	SD = 8.82
6	Great Rissington	41	Fair	
7	Sherborne Common	35	Fair	***Fair Class River***
8	Barrington Park	46	Good	
9	Little Barrington	44	Good	
10	Upton	36	Fair	
11	Widford	44	Good	
12	Asthall	46	Good	
13	Worsham	47	Good	
14	Minster Lovell	43	Good	
15	New Mill	43	Good	
16	Ducklington (West Arm)	44	Good	
17	Ducklington (East Arm)	50	Good	
18	Beared Mill	50	Good	
19	Standlake STW	47	Good	
	Average	**41**	**Fair**	

Appendix 5.3 (Continued) IBI scores on the basis of 15 selected metrics for English rivers

The Trent catchment

River Anker

Site No. Location	IBI score	Integrity Class	Comment
1 Weddington	40	Fair	
2 Leather Mill	50	Good	n = 10
3 Woodford Bridge	43	Good	Score range= 31 - 50
4 Mancetter Mill	31	Fair	Average = 43
5 Ratcliffe Bridge	46	Good	SD = 5.81
6 Fieldon Bridge	48	Good	
7 Polesworth 1	41	Fair	***Good Class River***
8 Polesworth 2	43	Good	
9 U/S Tamworth Cowells Farm	36	Fair	
10 Tamworth Station Field	50	Good	
Average	**43**	**Good**	

Appendix 5.3 (Continued) IBI scores on the basis of 15 metrics for English rivers

River Blithe	IBI score	Integrity Class	Comment
1 Blythe Bridge	23	Poor	n = 11
2 Cresswell U/S Blithe colours	26	Poor	Score range= 23 - 48
3 Newton Crossing	26	Poor	Average = 39
4 Lower Leigh	39	Fair	SD = 9.04
5 Field	46	Good	
6 Burnthurst Mill	48	Good	***Fair Class River***
7 Booth Bridge	47	Good	
8 Lower Booth Farm	44	Good	
9 U/S Newton Bridge	45	Good	
10 Piory Farm	44	Good	
11 Hamstall Ridware	44	Good	
Average	**39**	**Fair**	

Appendix 5.3 (Continued) IBI scores on the basis of 15 selected metrics for English rivers

River Blythe	IBI Score	Integrity Class	Comment
1 Cheswick Green	0	No Fish	
2 Widney Manor Rd. Bridge	30	Fair	n = 9
3 Sandall's Bridge	39	Fair	Score range= 0 - 50
4 Springfield House Temple Balsall (2)	35	Fair	Average = 37
5 Springfield House Temple Balsall (1)	36	Fair	SD = 14.54
6 U/S Eastcote Brook	47	Good	
7 D/S EastCote Brook	45	Good	***Fair Class River***
8 Moland's Bridge	49	Good	
9 Blythe Mill End	50	Good	
Average	**37**	**Fair**	

Appendix 5.3 (Continued) IBI scores on the basis of 15 selected metrics for English rivers

The Trent catchment

River Churnet

Site No.	Location	IBI Score	Integrity Class	Comment
1	Middle Hulme Bridge	30	Fair	n = 16
2	Tittesworth Reservoir D/S	24	Poor	Score range= 0 - 48
3	South Hillswood Farm	0	No Fish	Average = 27
4	Abby Green Road Site 1	0	No Fish	SD = 15.00
5	Abby Green Farm	0	No Fish	
6	Westwood Golf Club	23	Poor	***Poor Class River***
7	U/S Leek Brook WRW (2)	35	Fair	
8	St. Edwards Hospital	29	Fair	
9	Flint Mill Cheddleton	21	Poor	
10	D/S Cheddleton WRW	22	Poor	
11	Thomas Boltons Ltd.	38	Fair	
12	Whiston Bridge	39	Fair	
13	Eastwall Farm	39	Fair	
14	U/S Alton WRW	43	Good	
15	D/S Alton WRW	39	Fair	
16	JCB Rocester	48	Good	
	Average	**27**	**Poor**	

Appendix 5.3 (Continued) IBI scores on the basis of 15 selected metrics for English rivers

River Cole		IBI Score	Integrity Class	Comment
1	Lowbrook Farm	0	No Fish	
2	Mill Lodge	0	No Fish	n = 14
3	Haybarn Recreation Ground	0	No Fish	Score range= 0 - 45
4	Glebe Farm Recreation Ground	0	No Fish	Average = 23
5	Colehall (1)	0	No Fish	SD = 17.98
6	Colehall	27	Poor	
7	Kingshurst 1	36	Fair	***Poor Class River***
8	Kingshurst 2	31	Fair	
9	Cook's Lane Bridge D/S	24	Poor	
10	Becons End	38	Fair	
11	Coleshill Hospital 1	45	Good	
12	Coleshill Hospital 2	43	Good	
13	Coleshill 1	37	Fair	
14	Coleshill 2	41	Fair	
	Average	**23**	**Poor**	

Appendix 5.3 (Continued) IBI scores on the basis of 15 selected metrics for English rivers

River Derwent		IBI Score	Integrity Class	Comment
1	U/S Howden Gauging Weir	23	Poor	n = 15
2	Bamford Gauging Station	32	Fair	Score range= 23 - 52
3	Bamford	27	Poor	Average = 37
4	Grindleford	28	Fair	SD = 8.50
5	Baslow Bridge	33	Fair	
6	D/S Baslow S.T.W.	33	Fair	***Fair Class River***
7	Beeley U/S site	36	Fair	
8	Darley Dale D/S Site	28	Fair	
9	Arkwright's Mill, Matlock	42	Good	
10	Cromford	35	Fair	
11	Whatstandwell	44	Good	
12	Ambergate	48	Good	
13	Milford	52	Good	
14	Alvaston	46	Good	
15	Draycott	45	Good	
	Average	**37**	**Fair**	

Appendix 5.3 (Continued) IBI scores on the basis of 15 selected metrics for English rivers

The Trent catchment

River Idle

Site No.	Location	IBI Score	Integrity Class	Comment
1	Eaton	25	Poor	n = 5
2	Tiln	27	Poor	Score range= 0 - 44
3	Mattersey Priory	29	Fair	Average = 25
4	Bawtry	44	Good	SD = 14.18
5	Misson	0	No Fish	
	Average	**25**	**Poor**	***Poor Class River***

Appendix 5.3 (Continued) IBI scores on the basis of 15 selected metrics for English rivers

River Mease

Site No.	Location	IBI Score	Integrity Class	Comment
1	Stretton en le Field	39	Fair	
2	Netherseal Bridge	40	Fair	n = 7
3	U/S Stones Bridge	45	Good	Score range= 39 - 48
4	Haunton	43	Good	Average = 43
5	Edingle	40	Fair	SD = 3.21
6	Croxall Mill	48	Good	
7	Croxall Bridge	46	Good	***Good Class River***
	Average	**43**	**Good**	

Appendix 5.3 (Continued) IBI scores on the basis of 15 selected metrics for English rivers

River Penk

Site No.	Location	IBI Score	Integrity Class	Comment
1	Black Brook Nature Trail	0	No Fish	
2	Allotment Site Codsall	0	No Fish	n = 11
3	U/S Bill Brook WRW	0	No Fish	Score range= 0 - 48
4	D/S Bill Brook WRW	21	Poor	Average = 29
5	Pendeford Nature Reserve	30	Fair	SD = 19.09
6	Brewood Park Farm	39	Fair	
7	Somerford Mill Farm	41	Fair	***Fair Class River***
8	Stretton Mill	48	Good	
9	Cuttlestone Bridge	42	Good	
10	Acton Mill Bridge	47	Good	
11	Radford Bridge	47	Good	
	Average	**29**	**Fair**	

Appendix 5.3 (Continued) IBI scores on the basis of 15 selected metrics for English rivers

River Sence		IBI Score	Integrity Class	Comment
1	Heather Butterley Brick Works	0	No Fish	n = 6
2	Congerstone / Cricket Pitch	31	Fair	Score range= 0 - 54
3	Congerstone	44	Good	Average = 36
4	Harris Bridge	54	Good	SD = 17.51
5	Lovett's Bridge	43	Good	
6	Ratcliffe Culey Bridge	45	Good	***Fair Class River***
	Average	**36**	**Fair**	

Appendix 5.3 (Continued) IBI scores on the basis of 15 selected metrics for English rivers

The Trent catchment

River Soar	**IBI Score**	**Integrity Class**	**Comment**
1 Ramsdale Farm	32	Fair	
2 Sutton Hill	48	Good	n = 15
3 Croft	40	Fair	Score range= 0 - 49
4 Littlethorpe	42	Good	Average = 37
5 Jubilee Park	45	Good	SD = 12.30
6 Blue Bank Lock	35	Fair	
7 Leicester Straights	26	Poor	***Fair Class River***
8 Abbey Meadows	32	Fair	
9 W/S Wanlip S.T.W. Outfall	26	Poor	
10 Mountsorrel	37	Fair	
11 Barrow on Soar	0	No Fish	
12 Cotes	49	Good	
13 Zouch	44	Good	
14 Kegworth	44	Good	
15 Ratcliffe on Soar	49	Good	
Average	**37**	**Fair**	

Appendix 5.3 (Continued) IBI scores on the basis of 15 selected metrics for English rivers

River Sow	**IBI Score**	**Integrity Class**	**Comment**
1 Eccleshall Castle	34	Fair	n = 9
2 Hillcote Hall	31	Fair	Score range= 26 - 50
3 Chebsey	26	Poor	Average = 38
4 Great Bridgeford	44	Good	SD = 6.88
5 Cresswell Farm	50	Good	
6 Dorey Marshes	43	Good	***Fair Class River***
7 Broadeye Stafford	40	Fair	
8 Stafford Sea Scout Hut	37	Fair	
9 U/S St. Thomases Mill	41	Fair	
Average	**38**	**Fair**	

Appendix 5.3 (Continued) IBI scores on the basis of 15 selected metrics for English rivers

River Tame

Site No. Location	**IBI Score**	**Integrity Class**	**Comment**
1 Lea Marston Upper	37	Fair	
2 Lea Marston Lower	47	Good	n = 6
3 Middleton	49	Good	Score range= 37 - 49
4 Hopwas-Two-Trees Farm	42	Good	Average = 44
5 Elford	41	Fair	SD = 4.49
6 Chetwynd Bridge	49	Good	
Average	**44**	**Good**	***Good Class River***

Appendix 5.3 (Continued) IBI scores on the basis of 15 selected metrics for English rivers

River Tean

Site No. Location	**IBI Score**	**Integrity Class**	**Comment**
1 Litley's Farm	23	Poor	n = 9
2 Teanford Mill	25	Poor	Score range= 0 - 45
3 Upper Tean Bridge	30	Fair	Average = 24
4 Rectory Farm	33	Fair	SD = 14.00
5 Checkley WRW	26	Poor	
6 Fole Hall	0	No Fish	***Poor Class River***
7 Fole D/S Creamery	0	No Fish	
8 Beamhurst Bridge	31	Fair	
9 Spath	45	Good	
Average	**24**	**Poor**	

Appendix 5.3 (Continued) IBI scores on the basis of 15 selected metrics for English rivers

River Trent

Site no.	Location	IBI Score	Integrity Class	Comment
1	Norton Green	23	Poor	n = 20
2	Abbey Farm	35	Fair	Score range= 0 - 43
3	Finney Gardens	39	Fair	Average = 29
4	Seven Arches, Stoke upon Trent	30	Fair	SD = 8.94
5	N. Staffs Polytech.	27	Poor	
6	Boothen End	24	Poor	***Fair Class River***
7	Hanford U/S Lyme Brook	26	Poor	
8	Hissey's Scarp Yard	23	Poor	
9	U/S Park Brook Bridge	35	Fair	
10	D/S Park Brook Bridge	34	Fair	
11	Trentham U/S Strongford WRW	31	Fair	
12	Tittensor D/S Strongford WRW	29	Fair	
13	Meaford Power Station	32	Fair	
14	Walton Lane Stone	26	Poor	
15	Aston Lock	34	Fair	
16	Sandon	21	Poor	
17	Weston U/S Gayton Brook	40	Fair	
18	U/S Hoo Mill	0	No Fish	
19	D/S Hoo Mill	34	Fair	
20	Great Haywood Mill	43	Good	
	Average	**29**	**Fair**	

Appendix 5.3 (Continued) IBI scores on the basis of 15 selected metrics for English rivers

The Yorkshire Ouse catchment

River Aire (IBI scores based on 14 metrics)

Site No.	Location	IBI score	Integrity Class	Comment
1	Malham Beck, below Malham cove	0	No Fish	n = 26
2	Malham Beck, below waterfalls	22	Poor	Score range= 0 - 41
3	Malham Beck, Malham village	23	Poor	Average = 25
4	Malham Beck, above STW	22	Poor	SD = 9.05
5	Malham Beck, below STW	23	Poor	
6	Gordale Beck, Gordale bridge	22	Poor	***Poor Class River***
7	Malham Beck, below Gordale beck	23	Poor	
8	River Aire above Skelgill mill	22	Poor	
9	River Aire Hanlith bridge	27	Poor	
10	River Aire Airton bridge	23	Poor	
11	Above Gargrave (below bridge)	22	Poor	
12	Gargrave (above stepping stone)	26	Poor	
13	Near Gargrave STW	24	Poor	
14	U/S Snaygill STW	41	Fair	
15	D/S Snaygill STW (above Cononley)	41	Fair	
16	Crossflatts	33	Fair	
17	Esholt U/S STW	25	Poor	
18	Calverley (below A6120) D/S Rawdon STW	0	No Fish	
19	Kirkstall	32	Fair	
20	Thwaite Weir	35	Fair	
21	Below Skelton Grange Power Station	30	Fair	
22	Swillington Bridge	20	Poor	
23	Castleford below weir, above Hicksons Ltd.	25	Poor	
24	Castleford alongside Hicksons Ltd.	20	Poor	
25	Beal Weirpool	27	Poor	
26	Chapel Haddlesey U/S A19	29	Fair	
	Average	**25**	**Poor**	

Appendix 5.3 (Continued) IBI scores on the basis of 15 selected metrics for English rivers

The Yorkshire Ouse catchment

River Nidd

Site No.	Location	IBI Score	Integrity Class	Comment
1	D/S Birstwith, Section 1	29	Fair	
2	D/S Birstwith, S2	34	Fair	n = 182
3	D/S Birstwith, S3	41	Fair	Score range= 0 - 54
4	D/S Birstwith, S4	32	Fair	Average = 35
5	U/S Hampsthwaite bridge, S1	33	Fair	SD = 8.18
6	U/S Hampsthwaite bridge, S2	36	Fair	
7	U/S Hampsthwaite bridge, S3	30	Fair	***Fair Class River***
8	U/S Hampsthwaite bridge, S4	43	Good	
9	Cragg Lane STW, S1	43	Good	
10	Cragg Lane STW, S2	35	Fair	
11	Cragg Lane STW, S3	36	Fair	
12	Cragg Lane STW, S4	35	Fair	
13	Upper Cragghill farm, S1	36	Fair	
14	Upper Cragghill farm, S2	34	Fair	
15	Upper Cragghill farm, S3	30	Fair	
16	Upper Cragghill farm, S4	37	Fair	
17	Cragghill farm ford, S1	38	Fair	
18	Cragghill farm ford, S2	30	Fair	
19	Cragghill farm ford, S3	28	Fair	
20	Cragghill farm ford, S4	40	Fair	
21	D/S Killinghall bridge, S1	36	Fair	
22	D/S Killinghall bridge, S2	32	Fair	
23	D/S Killinghall bridge, S3	37	Fair	
24	D/S Killinghall bridge, S4	38	Fair	
25	Roch farm, S1	29	Fair	
26	Roch farm, S2	29	Fair	
27	Roch farm, S3	32	Fair	
28	Roch farm, S4	35	Fair	
29	Holme bottom farm, S1	32	Fair	
30	Holme bottom farm, S2	32	Fair	
31	Holme bottom farm, S3	23	Poor	
32	Holme bottom farm, S4	31	Fair	
33	U/S Scotton weir S1	0	No Fish	
34	U/S Scotton weir S2	0	No Fish	
35	U/S Scotton weir S3	23	Poor	
36	U/S Scotton weir S4	21	Poor	
37	Scotton weir S1	23	Poor	
38	Scotton weir S2	40	Fair	
39	Scotton weir S3	36	Fair	
40	Scotton weir S4	37	Fair	
41	D/S Scotton weir S1	42	Good	
42	D/S Scotton weir S2	36	Fair	
43	D/S Scotton weir S3	39	Fair	
44	D/S Scotton weir S4	38	Fair	
45	Scotton Hospice S1	42	Good	
46	Scotton Hospice S2	36	Fair	
47	Scotton Hospice S3	39	Fair	
48	Scotton Hospice S4	41	Fair	
49	D/S Scotton Hospice S1	31	Fair	
50	D/S Scotton Hospice S2	41	Fair	

Appendix 5.3 (Continued) IBI scores on the basis of 15 selected metrics for English rivers

River Nidd (Continued)

Site No.	Location	IBI Score	Integrity Class	Comment
51	D/S Scotton Hospice S3	36	Fair	
52	D/S Scotton Hospice S4	36	Fair	
53	Conningham Hall S1	39	Fair	
54	Conningham Hall S2	41	Fair	
55	Conningham Hall S3	42	Good	
56	U/S High Bridge S1	26	Poor	
57	U/S High Bridge S2	26	Poor	
58	U/S High Bridge S3	34	Fair	
59	U/S High Bridge S4	34	Fair	
60	Mother Shiptons S1	54	Good	
61	Mother Shiptons S2	52	Good	
62	Mother Shiptons S3	34	Fair	
63	Mother Shiptons S4	39	Fair	
64	Lido Top S1	46	Good	
65	Lido Top S2	28	Fair	
66	Lido Top S3	33	Fair	
67	Lido Top S4	32	Fair	
68	Lido Bottom S1	28	Fair	
69	Lido Bottom S2	27	Poor	
70	Lido Bottom S3	21	Poor	
71	Lido Bottom S4	26	Poor	
72	Knaresborough STW S1	47	Good	
73	Knaresborough STW S2	41	Fair	
74	Knaresborough STW S3	27	Poor	
75	Knaresborough STW S4	33	Fair	
76	D/S A59 Bridge S1	32	Fair	
77	D/S A59 Bridge S2	26	Poor	
78	D/S A59 Bridge S3	37	Fair	
79	D/S A59 Bridge S4	34	Fair	
80	U/S Goldsborough Mill S1	28	Fair	
81	U/S Goldsborough Mill S2	26	Poor	
82	U/S Goldsborough Mill S3	26	Poor	
83	U/S Goldsborough Mill S4	28	Fair	
84	D/S Goldsborough mill S1	38	Fair	
85	D/S Goldsborough mill S2	45	Good	
86	D/S Goldsborough mill S3	37	Fair	
87	D/S Goldsborough mill S4	26	Poor	
88	Pylons D/S Goldsborough mill	34	Fair	
89	U/S Little Ribston S1	39	Fair	
90	U/S Little Ribston S2	40	Fair	
91	U/S Little Ribston S3	33	Fair	
92	U/S Little Ribston S4	43	Good	
93	Little Ribston wood S1	28	Fair	
94	Little Ribston wood S2	40	Fair	
95	Little Ribston wood S3	46	Good	
96	Little Ribston wood S4	29	Fair	
97	Above Ribston park S1	41	Fair	
98	Above Ribston park S2	43	Good	
99	Above Ribston park S3	31	Fair	
100	Above Ribston park S4	42	Good	

Appendix 5.3 (Continued) IBI scores on the basis of 15 selected metrics for English rivers

River Nidd (Continued)

Site No.	Location	IBI Score	Integrity Class	Comment
101	Ribston park S1	29	Fair	
102	Ribston park S2	27	Poor	
103	Ribston park S3	42	Good	
104	Ribston park S4	46	Good	
105	Ornamental bridge S1	37	Fair	
106	Ornamental bridge S2	52	Good	
107	Ornamental bridge S3	42	Good	
108	Ornamental bridge S4	30	Fair	
109	Ribston park-bottom S1	25	Poor	
110	Ribston park-bottom S2	46	Good	
111	Ribston park-bottom S3	40	Fair	
112	Ribston park-bottom S4	30	Fair	
113	Crimple mouth S1	32	Fair	
114	Crimple mouth S2	32	Fair	
115	Crimple mouth S3	34	Fair	
116	Crimple mouth S4	37	Fair	
117	D/S A1 Bridge S1	21	Poor	
118	D/S A1 Bridge S2	40	Fair	
119	D/S A1 Bridge S3	44	Good	
120	D/S A1 Bridge S4	40	Fair	
121	U/S Broad Wath beck S1	22	Poor	
122	U/S Broad Wath beck S2	24	Poor	
123	U/S Broad Wath beck S3	28	Fair	
124	U/S Broad Wath beck S4	32	Fair	
125	Cowthorpe hall farm S1	30	Fair	
126	Cowthorpe hall farm S2	32	Fair	
127	Cowthorpe hall farm S3	26	Poor	
128	Cowthorpe hall farm S4	30	Fair	
129	Hunsingore D/S footbridge S1	44	Good	
130	Hunsingore D/S footbridge S2	37	Fair	
131	Hunsingore D/S footbridge S3	28	Fair	
132	Hunsingore D/S footbridge S4	51	Good	
133	Cowthore - Gauging hut S1	38	Fair	
134	Cowthore - Gauging hut S2	50	Good	
135	Cowthore - Gauging hut S3	42	Good	
136	Cowthore - Gauging hut S4	46	Good	
137	Cowthorpe dog kennels S1	44	Good	
138	Cowthorpe dog kennels S2	44	Good	
139	Cowthorpe dog kennels S3	41	Fair	
140	Cowthorpe dog kennels S4	50	Good	
141	Cowthorpe (bottom limit) S1	39	Fair	
142	Cowthorpe (bottom limit) S2	34	Fair	
143	Cowthorpe (bottom limit) S3	35	Fair	
144	Cowthorpe (bottom limit) S4	34	Fair	
145	Cattal (upstream bridge) S1	45	Good	
146	Cattal (upstream bridge) S2	29	Fair	
147	Cattal (upstream bridge) S3	54	Good	
148	Cattal (upstream bridge) S4	32	Fair	
149	Cattal (downstream bridge) S1	33	Fair	
150	Cattal (downstream bridge) S2	48	Good	

Appendix 5.3 (Continued) IBI scores on the basis of 15 selected metrics for English rivers

River Nidd (Continued)

Site No.	Location	IBI Score	Integrity Class	Comment
151	Cattal (downstream bridge) S3	47	Good	
152	Cattal (downstream bridge) S4	45	Good	
153	Cattal (U/S Old thornville) S1	34	Fair	
154	Cattal (U/S Old thornville) S2	25	Poor	
155	Cattal (U/S Old thornville) S3	36	Fair	
156	Cattal (U/S Old thornville) S4	30	Fair	
157	Cattal (D/S Old thornville) S1	52	Good	
158	Cattal (D/S Old thornville) S2	24	Poor	
159	Cattal (D/S Old thornville) S3	36	Fair	
160	Cattal (D/S Old thornville) S4	50	Good	
161	Tockwith S1	27	Poor	
162	Tockwith S2	39	Fair	
163	Tockwith S3	35	Fair	
164	Tockwith S4	37	Fair	
165	Hammerton mill S1	37	Fair	
166	Hammerton mill S2	44	Good	
167	Hammerton mill S3	37	Fair	
168	Hammerton mill S4	30	Fair	
169	Hammerton mill S5	34	Fair	
170	Hammerton mill S6	37	Fair	
171	Opposite Skewkirk S1	33	Fair	
172	Opposite Skewkirk S2	28	Fair	
173	Opposite Skewkirk S3	28	Fair	
174	Opposite Skewkirk S4	33	Fair	
175	Wilstrop S1	35	Fair	
176	Wilstrop S2	33	Fair	
177	Wilstrop S3	38	Fair	
178	Wilstrop S4	41	Fair	
179	Upstream Skipbridge S1	23	Poor	
180	Upstream Skipbridge S2	22	Poor	
181	Upstream Skipbridge S3	24	Poor	
182	Upstream Skipbridge S4	34	Fair	
	Average	**35**	**Fair**	

GLOSSARY

Br	= Bed rock
Co	= River confluence
Cw	= Coloured water
Dd	= Domestic discharge
Dg	= Dredged
Dw	= Dense weed
Dy	= Dye works
Er	= Erosion
Eu	= Eutrophication
Fd	= Factory discharge
Fp	= Farm pollution
Fr	= Fish removed
Ft	= Fast flow
Gc	= Good cover
Gh	= Good habitat
Gw	= Gravel works
He	= Habitat enhanced
Id	= Industrial discharge
Ip	= Industrial pollution
Lc	= Low cover
Lf	= Low flow
Lw	= Land works
Md	= Mine water discharge
Nc	= No cover
Op	= Oil pollution
Pa	= Pasture
Ph	= Poor habitat
Po	= Pollution
Ps	= Pools
Pwq	= Poor water quality
Qd	= Quarry discharge
Re	= River engineering
Rs	= Riffles
Rse	= Raw sewage effluent
Rst	= Restocking
Sb	= Stony bank
Sh	= Shallow
Si	= Silage
Sl	= Slurry
Ss	= High suspended solids
St	= Straightened
Wa	= Water abstraction
Wc	= Weed cutting
Wd	= Widened
Uro	= Urban run-off