MODELLING AND MANAGING THE EFFECTS OF TROUT FARMS ON CAPE RIVERS

by
Catherine Anne Brown

Submitted in fulfilment of the requirements for the Degree of Doctor of Philosophy
Department of Zoology, University of Cape Town

UNIVERSITY OF CAPE TOWN
JULY 1997

DST 590 BROW

98/9893

For my parents,
Anne and Gordon Brown

DECLARATION

This thesis reports the results of original research, which I did in the Freshwater Research Unit, Department of Zoology, University of Cape Town, between 1991 and 1996. None of it has been submitted in whole or in part for any other degree, and any technical assistance I received is fully acknowledged.

Signed by candidate

TABLE OF CONTENTS

ABSTRACT

The south-western Cape is currently responsible for 45% of the total annual trout production in South Africa and further expansion of this industry in the region is likely. A pilot study of seven trout farms situated on the upper reaches of rivers in the south-western Cape was undertaken to determine whether there was a common trend in their effect on the rivers. Results indicated that the impact of the farms on the benthic invertebrate communities of the rivers ranged from mild to severe, based on the degree of change in the structure of the communities form upstream to downstream of the effluent outlets. Those farms situated on mountain streams had the greatest impact and those on the downstream foothill had a lesser impact. The reduced impact in the foothill zone was probably because these reaches were already disturbed by other catchment activities. Of the three farms that were situated on mountain streams and source areas, two used plastic portapools and the third earth dams. There was a substantial increase in the number of oligochaetes downstream of both 'portapool' farms and yet, despite being situated in the same sensitive river zone, this did not occur downstream of the farm that used earth dams. The general impact of trout farm effluent on the mountain-stream and source zones was to eliminate or greatly reduce the number of Limnichidae, Helodidae, Plecoptera, Elmidae, Heptageniidae and Ephemerellidae, and, in the case of portapool farms, to replace these with Naididae, Lumbriculidae, Chironomidae and Planaria. Once-off chemical samples were also collected at each site and, acknowledging the limitations of the sampling strategy, results showed that the particulate fraction of the effluent was probably responsible for the recorded reaction of the biota.

In a follow-up study, macroinvertebrate community structure was investigated upstream and at various distances downstream of two trout farms situated alongside mountain streams in the south-western Cape, South Africa, in order to ascertain which of the constituents of trout-farm effluent were most responsible for changes in the macroinvertebrate communities downstream of the farms, and the concentrations at which the changes occurred. Of the farms, one used plastic-lined 'portapools' to house their fish, and the other, earthdams. Only the effluent discharged from the portapool trout farm resulted in notable changes in downstream benthic-macroinvertebrate community structure. The community downstream of the effluent outlets was dominated by non-insects, as opposed to the community upstream of the influence of the farm which was dominated by insect taxa. It was possible to recognise and identify differences in community structure upstream and downstream of the farms, and to relate these to organic pollution present in the systems. The macroinvertebrate community structure changed in response to relatively small changes in water physico-chemistry. Particulate organic material suspended in trout-farm effluent best correlated with the changes in the macroinvertebrate community structure recorded downstream of the portapool farm. Maintenance of suspended particulate organic matter concentrations to below 1.5 mg l^{-1} (dry weight) in the river, should protect the integrity of the community structure of macroinvertebrate fauna of mountain

streams in the south-western Cape, providing flows do not drop abnormally low relative to the historical condition.

Empirical data from the two farms that formed part of the main study were used to assess the effects of a reduction in taxonomic resolution using MDS and the Bray-Curtis measure of similarity. In addition, the performance of three community-structure (family-level meta-analysis, ABC curves, Shannon-Wiener Diversity) and two indicator-organism (SASS4 and EPT taxa richness) approaches was examined. Of the four taxonomic levels tested, the family-level abundance data seemed to provide the best indication of the effects of the trout-farm effluent on the macroinvertebrate community. Family-level meta-analysis proved extremely useful in assessing the severity of impacts in a regional context but is relatively sophisticated, requiring quantitative sampling, identification of the macroinvertebrate families and complex data analysis. The ABC curves did not follow predictions from marine studies. It may be that, in mountain streams the trends in ABC curves are the reverse of those expected from marine systems. Shannon-Wiener Diversity, SASS4 and EPT taxa richness showed similar responses to the changes in macroinvertebrate communities. Although Shannon-Wiener Diversity and SASS4 tracked the community changes fairly well, they both showed that the community at the most downstream site approximated that upstream of the effluent outfall. This is contrary to indications from the detailed study. A model of a mountain-stream macroinvertebrate community's response to increased organic suspended solids was generated using the data collected from the portapool farm during the main study. This model was used to assess the performance of six evenness/diversity indices (species richness, Shannon-Wiener, Pielou's Evenness, Simpson's D, Hurlbert's PIE, Pearson and Pinkham's B), one similarity index (Bray-Curtis Similarity) and one biotic index (SASS4). The performances of these indices are discussed in the light of prevailing stream theory. Indices which measure evenness within a community are not useful for assessing the effects of organic pollution on mountain stream ecosystems because evenly-distributed, stable communities occur at several levels of organic enrichment. This is in line with the River Continuum Concept, which states that stable communities will form based on the size and availability of organic matter. SASS4 and species richness avoided the "multiple response" pitfalls of the evenness indices but were sluggish in their response to potentially significant changes, mainly because they ignore abundances. The Bray-Curtis Similarity index was able to show subtle changes in community structure. Indices and other techniques that proved most valuable in assessing the effects of trout-farm effluents on upper rivers were those that had some biological basis, that sampled erosional and depositing habitats separately, that required familial-level identification of the macroinvertebrates and that incorporated some measure of abundance.

ACKNOWLEDGEMENTS

Part of this study was conducted under contract to the Department of Water Affairs and Forestry, and to Cape Nature Conservation. I gratefully acknowledge the financial support I received from those organisations and the assistance I received of members of the project Steering Committee. Thank you also to the University of Cape Town who awarded me a PhD bursary for one year of the study.

My sincere thanks go to Dr Jackie King, without whom none of this would have happened. Far, far more than just a supervisor, Jackie has been mentor and friend to me since first we met. I also thank Prof. John Field for his supervision of this project and for bringing new perspectives into the thesis.

Thank you also to Dr Jenny Day, Prof. Bryan Davies and all the other members of the Freshwater Research Unit for their friendship and support. I also extend a warm thank-you to my fellow Southern Waters' members, Jackie King, Geordie Ractliffe, Sean Eekhout, Rebecca Tharme, Helen Dallas and Liz Day, for their support and encouragement in this and other ventures. A special thank-you to Dr Jenny Day and to Liz Day for proof-reading various drafts of this document.

I would like to take this opportunity to thank Dr David Ward for his friendship and confidence in me over the years. Despite having been subjected to the first draft of the thesis, David encouraged me to keep plugging away. The fact that this thesis has reached completion is in no small way a result of his encouragement.

Thanks also to Mr D. Grobler and Mr G. Quibell of the Institute for Water Quality Studies (DWAF) for trace metal analyses; Mr M.Coxhill of the J.B.Visser Hatchery for information on the management of fish farms; Ms C.Coulson for laboratory assistance; and C. Coulson, L. Crossley, S. Eekhout, C. Fowler, D. Glassom, D. Gould, S. Harris, G. Ractliffe, C. Peckham, C. Smith, W. Steele, R. Townsend and A. van Aardt for field assistance.

This study could not have been completed without the co-operation and assistance of the trout farmers of the south-western Cape. Messers. D. Brink, M. Kruger, G. Stubbs, B. Smal, J. Visser and G. Watson allowed me access to their trout farms and rivers, provided valuable background information and generally made me feel very welcome.

Thanks also to all of my friends for their love and support, in particular, Carol Smith, Ros Townsend, Colette van Niftrik, Jenny Burkinshaw, Geordie Ractliffe, Sean Eekhout and Petra Muller. Also a very special thank-you to my partner, Dr Hildegarde Fast, for her love, patience and encouragement.

Finally, I wish to thank my parents, Anne and Gordon, for the love, the support, the education and the opportunities that were so much a part of being one of their children. This one's for you.

SECTION I

INTRODUCTION

CHAPTER 1
GENERAL INTRODUCTION

1.1 INTRODUCTION

Aldo Leopold encapsulated the sentiments of many ecologists of his and future generations with the statement:

> "*A thing is right when it tends to preserve the integrity, stability and beauty of the biotic community. It is wrong when it tends to do otherwise*" (Leopold 1949).

Unfortunately, all is not right in the world, and there are many 'things' that have done, and continue to do, 'otherwise'. Their injuriousness notwithstanding, detailed studies of these perturbations have afforded ecologists an understanding of the response of the natural environment to anthropogenic influences, leading to a greater overall understanding of the functioning of natural ecosystems.

Over the last few decades, Leopold's integrity concept has grown in influence, primarily as a result of a change in our awareness of our relationship to the physical, chemical and biological environment (Karr 1993). These changes have led to new policies for the protection of ecosystems, particularly with respect to the protection of water resources. For example, degradation of water resources resulting from land-based activities, which lead to an increased amount of human and industrial waste, and to complicated health hazards, was one of six fundamental water-related environmental challenges addressed by the United Nations Environmental Plan (UNEP). And, in South Africa, recent changes in the policy of the Department of Water Affairs and Forestry (DWAF) have reflected the recognition of riverine ecosystems as the resource on which a sustainable supply of water depends (DWAF 1997). As a result, the water quantity and quality requirements for maintenance of the riverine biota are now taken into consideration when allocating water from impoundments or setting standards for effluent discharges into rivers. This project arose directly from the changes in the policies and laws governing South Africa's freshwater systems.

1.2 POTENTIAL PROBLEMS ASSOCIATED WITH TROUT FARMING ON RIVERINE ECOSYSTEMS

Freshwater aquacultural activities increased dramatically across the globe during the 1970s and 1980s. The industry expanded in France (Oberdorff and Porcher 1994) and Australia (Purser and O'Sullivan 1992), and saw a resurgence in East Africa (Dadzie 1992). In South Africa, the trout-farming industry maintained a 30% growth rate between 1985 and 1991, despite a general downturn in the economy.

Fish farming, particularly trout farming, requires large quantities of good quality water to supply the needs of the cultured stock and to remove the wastes generated (Allcock and Buchanan 1994). These requirements tend to limit the establishment of trout-farming concerns to regions and river reaches, usually the mountain-stream or foothill zones, that can meet these needs. In many instances, the presence of trout farms in these areas resulted in the degradation of stream and lake ecosystems (Beveridge 1987, Nature Conservancy Council 1990). Scotland, for example, experienced an increase in trout production between 1970 and 1990 of two orders of magnitude (Nature Conservancy Council of Scotland 1990), with the result that the industry outstripped legislation designed to protect the water quality of Scottish streams for other uses (Smith and Haig 1991).

1.2.1 Potential polluting effects

The potential polluting effects of fish farms have been well documented (Jones 1990). The most obvious potential impacts of a land-based trout farm are those of over-abstraction of water from a river and the escape of farmed trout into surrounding waters (Carss 1990). Over-abstraction can lead to changes in channel shape and patterns of sedimentation, barriers to migration of fish, and alteration to the structure of biological communities (Jones 1990, Nature Conservancy Council 1990).

Potential pollutants in fish-farm effluent include faeces and uneaten food, which settle out on river beds and can result in increased rates of nutrient uptake into the sediments, leading, in extreme cases, to eutrophication. The quantity and quality of solid wastes in the effluents vary seasonally and diurnally depending on feeding time, stocking rate and other factors (Nature Conservancy Council 1990). This has important implications for the monitoring of these effluents.

Dissolved nutrients are also major potential pollutants. The amount of nitrogen in fish-farm effluent varies over time, with peaks following feeding and during tank cleaning. Phosphorus concentrations are dependent on feed quality, feed conversion ratios, fish size and fish-farm management (Nature Conservancy Council 1990). Nitrogen and phosphorus in the effluent can result in hypernutrification and increased primary production of macrophytes and algae in the river downstream of the farm.

The levels of dissolved oxygen in the river may be affected by localised reduction in oxygen levels at the effluent outlet. Factors likely to affect oxygen levels are consumption of oxygen during the breakdown of organic and other matter contained in the effluent (Biological Oxygen Demand, BOD, and Chemical Oxygen Demand, COD) and indirect downstream effects through changes in phytoplankton abundance. The impacts of changes in the level of dissolved oxygen will depend on the characteristics of the receiving waters and of the effluent but would affect the survival of the natural riverine fauna and flora (Nature Conservancy Council 1990).

Various chemicals are used in trout farms to supplement feed and to control diseases and ectoparasites. These chemicals may enter the riverine environment in the effluent. They range from fairly benign compounds (e.g. vitamins) to compounds that are extremely toxic to aquatic life (e.g. formaldehyde: toxic to algae at concentrations of 0.3 - 0.5 mg l^{-1}, and malachite green: sub-lethal effects on fish at concentrations as low as 0.03 - 0.05 mg l^{-1}). Little is known about the effects of these chemicals on the natural riverine biotas (Nature Conservancy Council 1990).

Cleaning of fish tanks and feeding can cause peaks in the concentration of pollutants in effluents, with peaks in 'cleaning' effluent being between 0.1 and 10 fold higher than concentrations of 'normal' effluent (Bergheim *et al.* 1984). These variations have important implications for both the monitoring of fish-farm effluents and for the natural environment.

Suspended solids have been identified, in other parts of the world, as the component in trout-farm effluents most responsible for the negative effects on the riverine biota downstream of trout farms, *viz.* Scotland (Nature Conservancy Council 1990), England (Jones 1990), Poland (Korzeniewski *et al.* 1982), Norway (e.g. Bergheim *et al.* 1984), France (Oberdorff and Porcher 1994) and the USA (e.g. Kendra 1991). A survey of four major fish farms in the Hampshire section of the Avon catchment in England showed that in all cases the macroinvertebrate fauna of the river bed downstream consisted of pollution-tolerant organisms, such as leeches (Hirudinea), flatworms (Planaria) and midge larvae (Chironomidae). These changes were attributed to increases in the amount of solid organic material, arising from faeces and uneaten food, deposited on the river bed (Jones 1990).

1.2.2 The influence of tank design

Factors other than the pollutants themselves contribute to the magnitude of the impact of effluent from land-based trout farms on rivers. These include the size and lay-out of the farm and the type of tanks used. Briefly, tanks can either be arranged in parallel or in series, with tanks arranged in parallel usually producing a more concentrated final effluent (Figure 1.1). There are two types of tanks used in land-based farms: unlined earth ponds (earthdams) and concrete- or plastic-lined tanks (portapools). Earthdams have a slower flow-through rate than do, for instance, portapools and thus some settlement of solids does occur. The solids in suspension may, therefore, be less concentrated in earthdams than in portapools or concrete-lined tanks. The flow-through of water in portapool or tank farms is too fast to allow waste food and faeces to decompose before they are discharged into the river. Earthdams also allow some interaction with the natural substratum and cannot be cleaned by scrubbing, so there is no flush of scoured material into the river.

1.2.3 The influence of food quality

There exists a close relationship between fish feed and the constitution of fish-farm effluents. The quality of fish feed plays a major role in determining the amount and characteristics of fish-farm effluent. The formulation and manufacture of the food

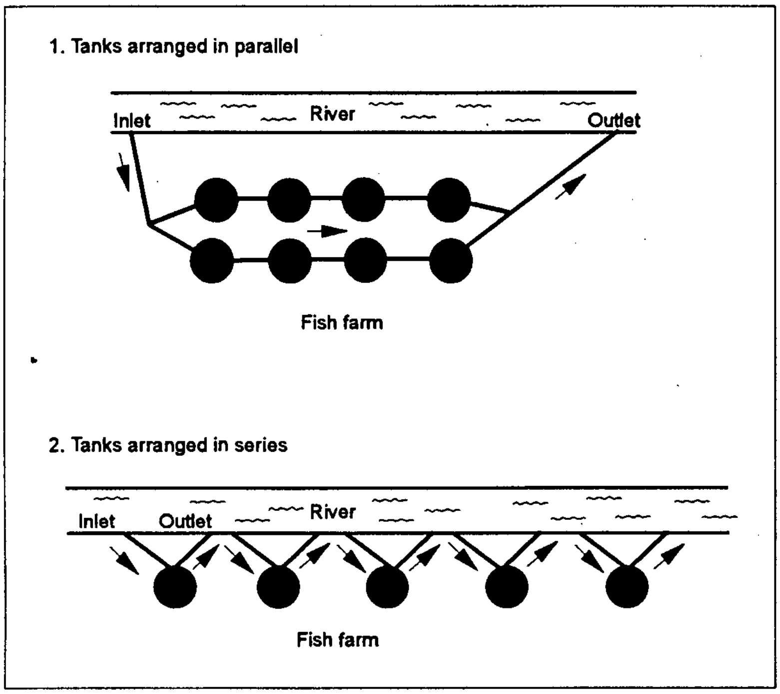

Figure 1.1 Arrangement of tanks on land-based farms

pellets, and the feeding methods employed on the farms, play important roles in this respect.

A proportion of feed given to fish is not eaten (Nature Conservancy Council 1990), either because too much is given or because the feed is unpalatable, and estimates of the amount of food that is left uneaten vary between 5 and 20 % (Beveridge *et al.* 1990). Of the food that is eaten, between 25 and 30 % is not digested and so is voided as faeces, together with small quantities of mucus, intestinal cells and gut microbes (Beveridge *et al.* 1990). Thus, depending on the quality of the food, between 24 and 44 % of the food used on the farm may be wasted, and this forms the bulk of the loading of solid matter from fish farms.

Various studies have shown that phosphorus is incorporated in excess in most trout diets (Wiesmann *et al.* 1988) with the result that phosphorus loads often form a significant proportion of the nutrient content of fish farm effluent. A major part of the total phosphorus in the effluent is associated with the particulate fraction (Nature Conservancy Council 1990).

Dietary control has been demonstrated as an efficient and cost-effective means of limiting ammonia and other soluble wastes (Henderson 1988) and there is considerable scope for reducing phosphorus loadings (Matty 1990) through using "low-pollution" feeds and environmentally sound management practices. Fish that were fed on so-called 'low-pollution' diets showed better food conversion ratios, faster growth rates and greater resistance to disease than those fed on 'normal' diets (Phillips *et al.* 1988; Roberts 1989 cited in Nature Conservancy Council 1990).

1.2.4 Background to the study

The South African freshwater-aquaculture industry expanded rapidly during the 1980s and by 1990 gross annual production was valued at approximately R72 million (Brink and Bekker 1991). Between 1985 and 1991 the industry maintained a 30% growth rate, with the result that, in 1988, 72% of aquacultural concerns in South Africa were between one and five years old. The trout-farming sector is probably the most significant contributor towards freshwater-aquaculture production figures, and in 1991 the commercial production of fresh trout in South Africa was approximately 1100 metric tonnes per annum (Brink and Bekker 1991, H. Bekker, Department of Agriculture, pers. comm.).

The south-western Cape was a focal area for growth in the South African trout-farming industry and, in 1991, was responsible for *c.* 45% of the total annual trout production in South Africa (550 tonnes in 1990). Furthermore, future expansion in the South African trout-farming industry is likely to concentrate on the south-western Cape (Brink and Bekker 1991).

Concern regarding the possible polluting effects of land-based trout farms on mountain streams and upper rivers in the south-western Cape resulted in the South African Department of Water Affairs and Forestry (DWAF), and Cape Nature Conservation (CNC) commissioning an investigation of the effects of these farms on riverine biotas in the region. The main aim of the investigation was to provide information on the reaction of the riverine biotas to different concentrations of trout-farm effluent (Bekker and Brown 1995). This would assist in the formulation of regulations controlling the concentration of trout-farming effluent entering the upper reaches of south-western Cape rivers.

When the investigation began, the south-western Cape supported nine commercial land-based, and two cage-culture, trout farms. Seven of the land-based farms (Figure 1.2) were selected for a pilot survey of the general impacts of trout-farm effluents on the rivers on which the farms were situated and, of these, two farms were later used as part of a year-long, detailed study.

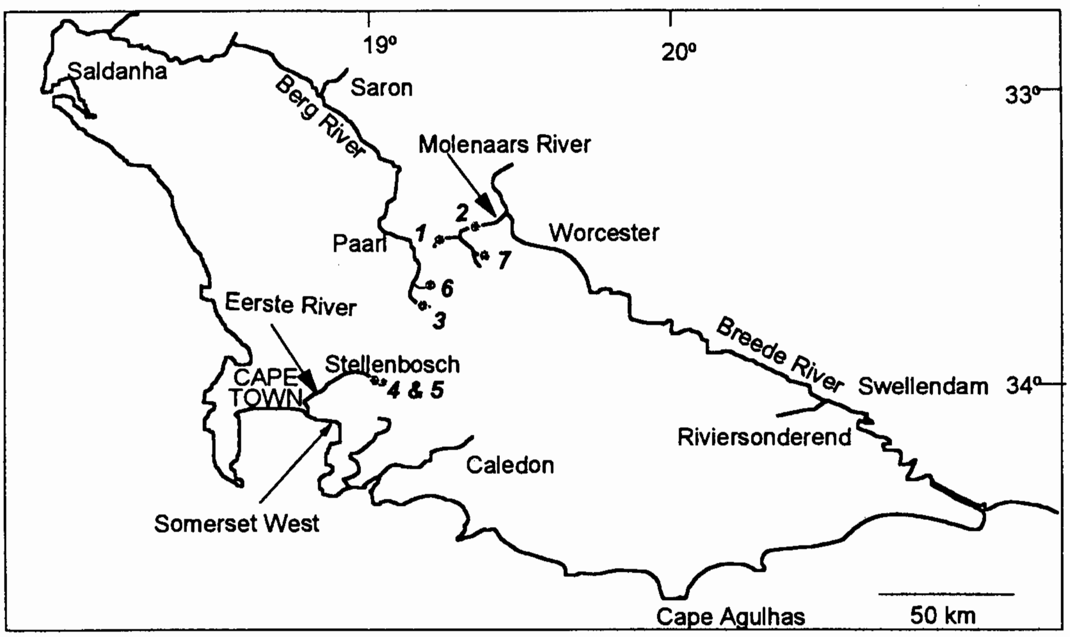

Figure 1.2 The locations of all of the land-based trout farms considered in the present study. The numbers represent the seven farms

1.3 STREAM ECOSYSTEM THEORY

Ecological theory is not some strange phenomenon best left within the confines of academia. It is the best available expression of the scientific communities' understanding of natural ecosystems. Problem-driven, applied studies and routine monitoring of riverine ecosystems must take cognisance of this theoretical basis, even if they are conducted outside of the academic environment. It is essential that the "short-cut" approaches, such as indices, adopted for monitoring impacts on freshwater ecosystems are rigorously tested, and that those that fail to indicate a sound biological basis are discarded. In this section, the current theories on stream, particularly mountain-stream, ecosystems are reviewed to provide a background against which to assess the performance of the different techniques used in this thesis to determine the impacts of trout farm effluent on mountain streams.

Streams and rivers change naturally along their lengths with respect to such properties as temperature, depth, current speed, substratum, turbidity and chemical composition (Hynes 1970). Since these factors are important determinants of the distribution of the riverine biota, the longitudinal physical and chemical changes are reflected in changes in species composition of the biotic communities. The result is a longitudinal biotic zonation that can be used to classify reaches of rivers. These zones are not discrete and attempts to define them in terms of a single variable have been unsatisfactory. Generally speaking, the rivers in the south-western Cape can be divided into six zones (Harrison and Elsworth 1958, Harrison 1965, Nobel and Hemens 1978, Brown *et al.* 1996), namely:

Mountain source and cliff waterfall

The source of a river, often consisting of boggy areas with sponge vegetation or humic turf and sometimes with waterfalls. Outside the sponges, the flow is usually fast and occasionally torrential. Turbidity is negligible and levels of oxygen saturation are high. Mean summer water temperatures may be below or about 20°C.

Mountain-stream zone

A narrow, defined channel with a very steep gradient, small waterfalls, rapids and little emergent vegetation. There may be occasional rock pools. The substratum consists of boulders, bed rock and cobbles, and flow is generally fast through riffle sections and slow in pools. The riparian trees may or may not form a closed canopy over the stream. Deposition of inorganic sediments is negligible and the surfaces of rocks and vegetation are virtually free of algal growth. Turbidity is negligible except during spates. Mean summer water temperatures are around 20°C.

Foothill zone

A zone of widening channel and decreasing bed gradient with lower flow velocities. The substratum consists of boulders and cobbles. Stony riffles and runs alternate with rock pools. Although there are still riparian trees, the river is wider and, because of this, the canopy is usually open. Turbidity is variable but usually low. Summer mean temperatures are above 20°C.

Transitional zone

A zone of single but occasionally anastamosing channel with alternating sections (or a mixture of) cobbles, sand and bedrock. Stony riffles, deep runs and deep backwaters dominate and there is usually a high incidence of marginal vegetation. The riparian belt is often narrow and the canopy is open. Turbidity is variable and mean summer water temperatures are above 20°C.

Lowland zone

A zone of very shallow gradient, with areas of deposition alternating with stony reaches. The riparian vegetation consists of reedbeds and few trees. The water is often turbid and mean summer water temperatures are usually well above 20°C.

Estuary

Flow is generally very slow and subject to tidal fluctuations. The riparian vegetation is specialised and tolerant of changes in salinity. Summer mean temperatures are generally above 20°C.

Several concepts concerning the longitudinal nature of rivers have been formulated in North America during the last few decades. Of these, the two best known are the River Continuum Concept (RCC, Vannote *et al.* 1980) and the Serial Discontinuity Concept (SDC, Ward and Stanford 1983a, Ward and Stanford 1983b) which view rivers as longitudinally-linked systems where ecosystem processes in downstream areas are linked to those in the upstream reaches by the unidirectional flow of water and materials (Naiman *et al.* 1988).

The RCC hypothesises that "the structural and functional characteristics of stream communities are adapted to conform to the most probable position or mean state of the physical system" (Vannote *et al.* 1980). The central tenet of the theory is that stable biological communities should form in a predictable fashion from a river's source to its mouth, based on size and availability of organic matter in the system. Thus, imported coarse organic material, the chief energy source of headwater streams, is processed into ever finer particles by successive downstream communities (Walker 1985). For instance, in the mountain-stream zone, allochthonous detritus is the primary source of food. Thus, the large number of whole leaves falling from riparian trees results in these zones being dominated by species known as "shredders", which break-up or shred leaves. Further downstream, as the river widens and sunlight can reach the bed, the emphasis switches from allochthonous to autochthonous food sources, such as algae. Shredders are proportionally less important here because proportionally less of the width of the river receives plant debris, and the macroinvertebrate community is dominated by species which scrape bacteria ("scrapers") off the surface of rocks or graze algae ("grazers"). In the lowland reaches of rivers, particulate matter suspended in the water column and deposited on the river bed results in a community dominated by detritivores or filter-feeders (see definitions in Chapter 5, Hynes 1970).

The dynamics of the macroinvertebrate populations inhabiting different river reaches are also determined by the hydraulic conditions in those reaches. In mountain streams, for instance, natural disturbance is high and unpredictable (Townsend 1989) and the landscape is erosional, producing communities with completely different life-history strategies from those communities inhabiting the lower reaches which have relatively low disturbance regimes and depositional landscapes.

The RCC has been modified by Minshall *et al.* (1985) to incorporate regional differences in climate, geology, lithology and geomorphology, and there is some evidence that biotic adjustments to energy input along the length of a river are biome-specific (e.g. Corkum 1991). For example, synchronous leaf-fall, such as, occurs in parts of the United States, does not occur in the south-western Cape, since most of the riparian trees in the south-western Cape are evergreen. Winterbourn *et al.* (1981) believe that river ecosystems are driven by stochastic events and, as such, are inhabited by opportunist and generalist biotas. Since climatic events are generally more unpredictable in southern Africa than in the temperate zones in the northern hemisphere (Alexander 1985), it is possible that the Winterbourn *et al.* (1981) view may be more appropriate for southern African river systems than the RCC (O'Keeffe *et al.* 1989a). In summary, although there is some doubt that the details described in the RCC apply to all river types (Gale 1992), the idea that the stream is a dynamic ecosystem that changes in a continuous fashion with respect to longitudinal environmental and resource gradients remains a useful framework on which to base stream ecosystem studies.

The SDC is based on the assumption that the RCC is conceptually sound, but states that streams are seldom, if ever, uninterrupted continua. The SDC predicts that a longitudinal change in a given parameter (physical or biological) will cause discontinuity in the stream continuum, after which a certain distance (the 'discontinuity distance') is required for the stream to return to its normal position in the continuum (Ward and Stanford 1983a, Minshall *et al.* 1985).

Within the river continuum (or continua), there is evidence that there are lateral linkages at play (Naiman *et al.* 1988). In mountain streams and foothill zones, for instance, marked spatial heterogeneity in conditions, such as flow velocities, substratum particle size (Gore 1989) and food resources (Townsend 1989) may result in spatial niche control. This may reduce interspecific competition and enhance species diversity. Vannote *et al.* (1980) suggest that increasing heterogeneity promotes faunal diversity, while increasing physical stability dampens faunal diversity. Once again there is some disagreement on this issue, and Winterbourn *et al.* (1981) suggest that increased physical stability combined with increased heterogeneity enhances faunal diversity. In his adaptation of patch dynamics to stream community ecology, Townsend (1989) states that the natural heterogeneity of mountain streams provides areas of reduced shear stress, that are physically stable, and can act as refugia during spate events, thereby enabling animals to maintain their position in the river at times of high physical disturbance. According to Townsend (1989), disturbance-driven temporal (climatic) niche control may further reduce the likelihood of competitive exclusions of species, thereby enhancing co-existence, and increasing diversity. In other words, animals are too busy dealing with the environmental pressures with which they are faced to worry about maintaining large territories (i.e., climatic rather than biological control). However, other evidence suggests that, although biotic interactions are suppressed during the actual disturbance events (spates), they can still influence stream communities significantly (Dudgeon 1993).

Studies conducted on the macroinvertebrate communities inhabiting rivers in the south-western Cape support the Winterbourn *et al.* (1981) notion (Harrison 1965, King 1981, Dallas *et al.* 1995). The upper reaches of these rivers have diverse arrays of hydraulic biotopes, highly unpredictable hydrological regimes and diverse macroinvertebrate faunas, relative to the lower reaches of the region's rivers. Furthermore, in the south-western Cape, the aquatic biotas most likely to be endemic and to be sensitive to changes in water quality and quantity are generally thought to be those that inhabit the upper reaches of rivers, that is, the source, mountain-stream and foothill zones (Harrison 1965, Davies and Day 1986).

1.4 OBJECTIVES AND ARRANGEMENT OF THE THESIS

1.4.1 Objectives of the thesis

This thesis had two main objectives. The first was to provide information on the reaction of the riverine macroinvertebrate communities to different kinds of trout farms and to different concentrations of trout-farm effluent. This first part of the thesis was conducted under contract to DWAF and CNC and the information was used in the formulation of regulations controlling effluents entering the upper reaches of south-western Cape rivers (King *et al.* 1991). The second objective was to use the data from the trout farm study to explore the applicability to freshwater systems of different biomonitoring approaches, with an emphasis on those developed for use in other fields of ecology. The results obtained using these approaches were then discussed in the light of prevailing stream theory, and related back to the results obtained in the first part of the thesis. Consequently, the first part of the thesis dealt with applied, question-orientated research, whereas the second part addressed more theoretical issues.

1.4.2 Methods of statistical analysis of the data

An understanding of the details of the statistical approaches used and their inherent assumptions is necessary for the results to be considered valid. Each section details the statistical analyses used to analyse the data collected during the study but there are some basic principles, which bear discussion here.

Rivers, by their nature, make the collection of statistically independent (replicate) samples from sites down their length implausible, since at any given time, the water quality at any given point on the river is likely to be dependent on the water quality upstream of that point. This is a result of the strong longitudinal linkages of stream patterns and, presumably, processes down the length of a river, and data collected in this way tend to violate the assumptions of many inferential analysis techniques, for example ANOVA (Zar 1984). Samples collected at an inappropriate scale, commonly referred to as pseudoreplicate samples *sensu* Hurlbert (1984), are also assumed to violate assumptions of many inferential analysis techniques. According to Hurlbert (1984), however, pseudoreplication results when the scale at which the samples are collected is smaller or more restricted than the inference space implicit in the hypothesis being tested. Pseudoreplication, therefore, concerns the interpretation, or rather misinterpretation of data, rather than describing a particular type of experimental design (Hargrove and Pickering 1992).

In ecology, a conflict exists between the need to collect sufficient replicate samples at any one place in order to account for the natural variability of the systems, and the need to study processes at appropriately large scales that hypotheses can be tested (Hargrove and Pickering 1992). To resolve this conflict, particularly in freshwater ecology, a more imaginative and innovative approach needs to be taken towards solving problems such as the effects of anthropogenic disturbance on ecosystems. Accordingly, the present study approached the problem of trout-farms, and their effect on riverine biota, from several perspectives.

The first perspective, provided by the initial survey, was designed to determine what impacts, if any, trout-farm effluents have on the river ecosystems on which they are situated; whether there were any common trends in the effects; and what components of the effluents might be responsible for the observed effects. In order to answer these questions, replicate samples were collected upstream and downstream of a number of different farms, thereby overcoming the problem of sample space versus inference space (pseudoreplication) by sampling at an appropriate scale. An inferential statistical technique, the paired sample T-test (Zar 1984), could therefore be applied to the data set and this was used in conjunction with multivariate techniques to interpret the results of the initial survey.

The second perspective was provided by the detailed study, which was designed to investigate seasonal and diurnal variations in the impacts of trout farms, to quantify the amount of organic material deposited downstream of the effluent outlets, and to determine the distances downstream of the outlets that the impact of the effluents persisted. The number of samples required for this study limited its scope to three farms. The detailed study concentrated on two main aspects of the effluent/river relationships, namely water-quality variables and composition of the macroinvertebrate community. It was necessary to determine whether or not sampling sites at any one farm differed from one another in terms of their water quality. This is best done using inferential statistical procedures. The assumption was made that, provided that the results of these analyses were used only to infer differences between sites at individual farms, and not downstream of all trout farms in general, then the problem of pseudoreplication would be avoided.

1.4.3 Arrangement of the thesis

The thesis is divided into five sections, each comprising one or more chapters. This section, SECTION 1, provides some background to the thesis, an indication of the directions taken, and places the work in its geographical and scientific context.

SECTION 2, the Pilot Study, is the first of two sections dealing specifically with the effects of trout farm effluent. It explores the general effects of land-based trout farms on south-western Cape mountain streams using data collected during a pilot study of seven farms.

SECTION 3, which is comprised of Chapters 3 and 4, looks in more detail at the influence of effluents from two of the original seven trout farms studied, on the riverine ecosystems with which they were associated. Chapter 3 concentrates on the physical and chemical changes and Chapter 4 looks at changes in macroinvertebrate community structure. These data were recorded over the course of a year, upstream and at various distances downstream of the farms in question.

SECTION 4 comprises Chapters 5 and 6, and focuses on rapid bio-assessment methodologies and their applicability for use in detecting the effects of organic pollution in mountain streams. Chapter 5 addresses some of the problems that are faced in the bid to minimise the time and financial costs of environmental monitoring, and assesses, in particular, the suitability of some the techniques used in the marine field. In Chapter 6, the performances of six diversity indices, one similarity index and one biotic index are investigated using a modelled response of a macroinvertebrate community to organic pollution. The results are discussed in the light of current stream theory.

The final section, SECTION 5, discusses the relevance of some of the issues explored in the thesis and considers some of the findings in the light of recent developments in aquatic-ecosystem monitoring in South Africa, with particular emphasis on the fledgling South African National Aquatic Ecosystem Biomonitoring Programme.

SECTION II

PILOT STUDY

CHAPTER 2

THE EFFECTS OF TROUT-FARM EFFLUENTS ON BENTHIC MACROINVERTEBRATE COMMUNITY STRUCTURE IN SOUTH-WESTERN CAPE RIVERS

2.1 INTRODUCTION

This chapter reports on the results of the first stage of the investigation into the reactions of riverine biotas to different concentrations and compositions of trout-farm effluent. The results were intended for use in setting site-specific or area-specific standards for trout-farm effluents entering rivers in the south-western Cape. The aim of the first stage was to visit several trout farms in order to ascertain: (a) what impacts, if any, trout-farm effluents have on the riverine ecosystems on which they are situated; (b) if there are any common trends in the downstream impacts; and (c) what components of the effluents might be responsible for the observed impacts. This is also the preliminary work which was used to decide which systems to use for more detailed analyses.

The potential polluting effects of land-based fish farms have been well documented (Jones 1990) and are dealt with in detail in SECTION 1. Considerable work has been undertaken in the northern hemisphere to evaluate the ecological impact of fish-farm wastes on the biota of rivers, although much of this has concentrated on fish assemblages (e.g. Karr 1991, Oberdorff and Porcher 1994), microbial communities (Nature Conservancy Council 1990) and algae (e.g. Beveridge 1984), and comparatively little on macroinvertebrates (e.g. Kendra 1991, Loch *et al.* 1996). In southern Africa, commercial fish farming is a relatively new activity, and there is very little in the published literature on the ecological impacts that it has on rivers in the sub-continent.

The effects of the trout-farm effluents on the rivers were assessed in terms of changes in the structure of macroinvertebrate communities upstream and downstream of the study farms. Aquatic macroinvertebrates, particularly benthic ones, are commonly used in biological assessments of rivers. These invertebrates include the aquatic larval stages of many terrestrial insects (e.g. Ephemeroptera and Trichoptera), aquatic adult insects (e.g. Coleoptera), and representatives of several non-insect groups such as crustaceans and worms (e.g. Amphipoda and Oligochaeta). Their usefulness in studies such as this one stems from the fact that they are relatively sedentary, widespread, easy to sample and, in general, display a rapid response to pollution (Hellawell 1977). Their use is sometimes limited by a scarcity of hard facts on cause-effect relationships between pollution and community structure (Dallas 1995), although the situation is changing.

2.1.2 Description of the study area

The south-western Cape is situated at the southern tip of the African continent (Figure 2.1). The region is characterised by a climate of cool, wet winters and dry summers. Its rivers therefore differ from those in other regions of southern Africa in that peak flow occurs during the winter months (Figure 2.2).

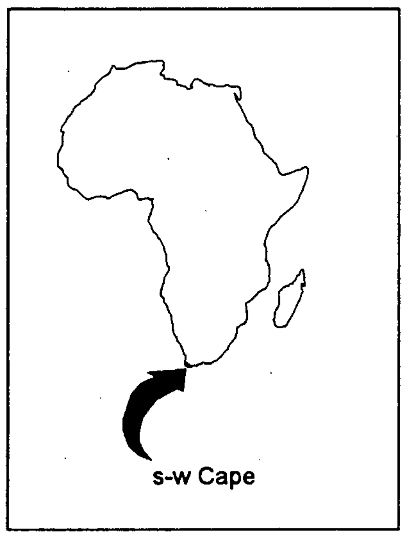

Figure 2.1 An outline of the African continent indicating the location of the south-western Cape, South Africa

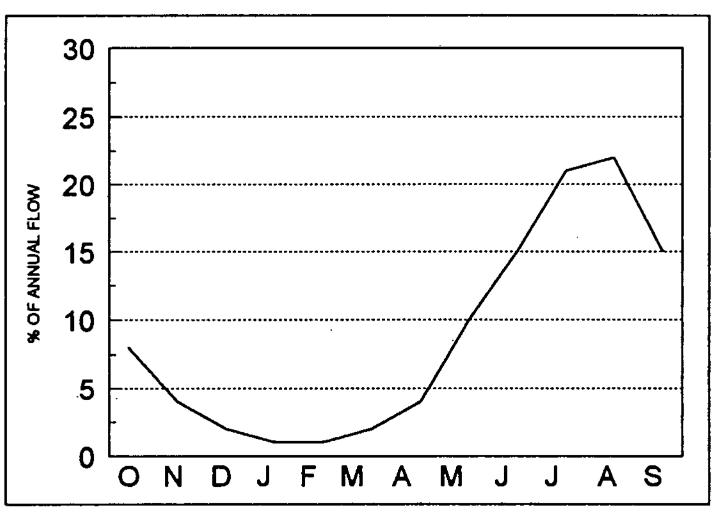

Figure 2.2 Mean monthly flow regime of south-western Cape rivers (Joubert and Hurly 1993)

South-western Cape rivers drain mountains comprised of quartzitic sandstones of the Table Mountain Series (Davies *et al.* 1993), which are vegetated with the heathlike, indigenous vegetation known as fynbos. Cape fynbos is a recognised floral kingdom distinguished by the presence of members of the plant families Proteaceae, Restionaceae, Ericaceae and Asteraceae (Taylor 1978, Campbell 1985; Cowling and Holmes 1992). The upper reaches of south-western Cape rivers are generally steep, fast-flowing streams that in some cases are shaded by a closed canopy of riparian trees, but in others have little or no canopy cover. Mostly because of the nature of the underlying geological formations and the vegetation, the streams are acid and poorly buffered (Dallas and Day 1993, Day and King 1995). Most nutrients and minerals are present in exceedingly low concentrations. Sediment loads are similarly low, with the exception of occasional peaks that are associated with natural small land slumps and flood events.

2.2 STUDY SITES

The survey was conducted by means of two sampling sessions in the rivers affected by each of seven trout farms (Figure 1.2, Chapter 1). The first session took place in October and November 1991, in spring, when the rivers were flowing strongly and water temperatures were low. Physical, chemical and benthic invertebrate samples were collected upstream, downstream and in the effluent of each of the farms during each visit. Since the dilution capacity of a river is proportional to its discharge, the chemical and physical samples collected at that time probably reflect the best water quality likely to be found at any time of the year. The second session took place in February and March 1992, at the end of summer, when discharge, and hence dilution, was low.

The rivers on which the trout farms were located were as follows:

- Farm 1 Molenaars River
- Farm 2 Molenaars River
- Farm 3 Berg River
- Farm 4 Eerste River
- Farm 5 Eerste River

- Farm 6 Franschhoek River
- Farm 7 Kraalstroom River

2.2.1 Sampling sites

Most of the farms abstracted water from the same river into which they discharged their effluent. In these cases, sampling sites were chosen upstream of the farm inlet (control), about 100 m downstream of the effluent outlet, and in the effluent itself (Figure 2.3). In the case of Farm 5, the water supply was not drawn from the river into which the effluent was discharged and the upstream site was chosen upstream of the outlet, that is, upstream of the influence of the farm.

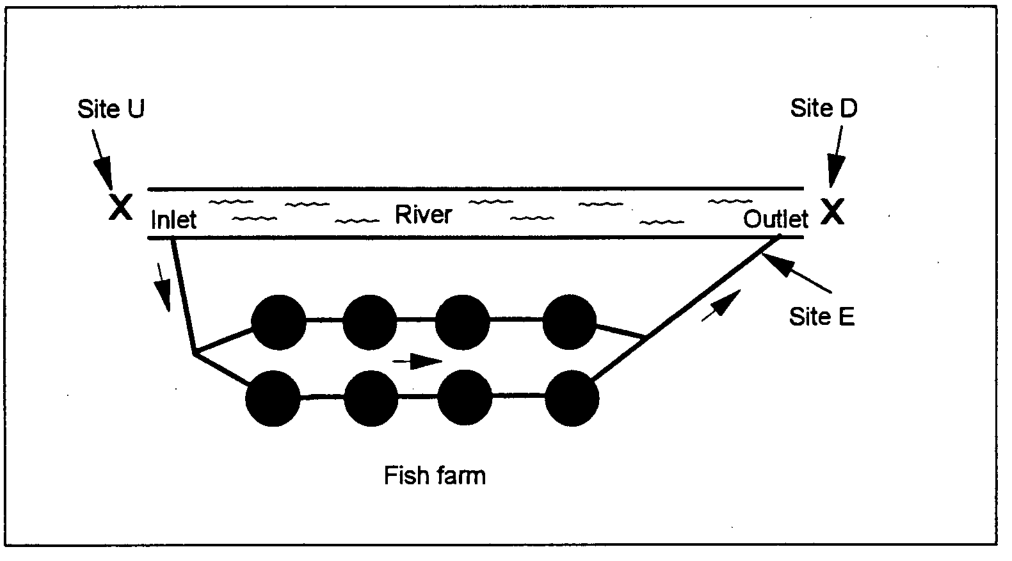

Figure 2.3 Positions of the sampling sites selected upstream (Site U), downstream (Site D) and in the effluent (Site E) at each of the trout farms visited during the pilot study

Farm 6 had characteristics that complicated the collection of data. The tanks were arranged in series along the banks of the Franschhoek River, near its source. The water from each tank was discharged directly into the river, and part of this subsequently flowed into the next tank. Thus, it was not possible to sample the 'total' effluent. In this instance, the effluent samples were collected from the outflow from the last tank in the series. Two further complications were (1) that a second stream joined the main river between the tanks, thus contributing to the dilution of the effluent and (2) that at least some of the effects on the river appeared to be a result of pollution from an adjacent stud farm, although the extent of this impact on the river could not be quantified.

2.3 MATERIALS AND METHODS

All samples were collected, and measurements made, in riffles, which are fast-flowing areas of shallow, broken water over cobble beds. Riffle biotopes were chosen in preference to other biotopes because they offer the most oxygenated and turbulent conditions in the stream and therefore the best possible environment for recovery of the biota to the status of the upstream community, following the input of effluent. The fauna of riffles is also generally rich both in number of species and in total biomass (Hynes 1970). Riffle invertebrates respond quickly and clearly to pollution (Harrison and Elsworth 1958), and most research on benthic macroinvertebrates in South African rivers has concentrated on the faunal communities of riffles (e.g. Harrison and Elsworth 1958; Chutter 1972; King 1981). There is thus more literature available on riffle communities than on those of other biotopes.

2.3.1 Analysis of chemical and physical variables

Measurements of conductivity (Crison CDTM-523 Conductivity Meter, standardised to 25°C and accurate to 0.1 mS cm^{-1}), pH (Crison 506 Portable pH Meter, accurate to 0.01 pH units), dissolved oxygen (Yellow Springs Institute Portable Oxygen Meter, compensated for altitude and temperature) and temperature (mercury thermometer, accurate to 0.5°C) were made in the field. The probes for each instrument were placed in the stream for a 30-minute equilibration period before the readings were taken.

Spot water samples, collected in the field, were filtered through Whatmann GF/F (0.45 µm pore-size) glass microfibre filters and cooled *in situ* to below 4°C. On return to the laboratory the filters and filtrate were frozen for later analysis. Contrad™-cleaned polyethylene bottles and vials were used for the collection of all water samples, with the exception of the samples for analysis of ammonia, which were collected in HCl-washed glass vials, and of trace metal samples, which were collected in specially prepared plastic bottles. All analyses were performed on these single, spot samples.

The size of the cobbles and/or boulders comprising the substratum was measured using a 0.25 m^2 metal grid, subdivided with string into 36 equal-sized squares (after Wright *et al.* 1984).

A known quantity of water (*c.* 1l) filtered through a pre-ashed, pre-weighed Whatmann GF/F filter was placed in a pre-weighed beaker and evaporated. The beaker was then re-weighed and the total dissolved solids (TDS) calculated. The filter papers were dried at 60°C for 48 hours and weighed to determine the mass of total suspended solids (TSS) in the sample. They were then placed in a muffle furnace for four hours at 450°C and re-weighed to determine the ratio of organic to inorganic suspended solids. The major cations Na^+, K^+ and Ca^{2+} were analysed using atomic flame absorption spectrophotometry with a VARIAN Spectra AA-30 and the major anions Cl^- and SO_4^{2-} were analysed using high-performance ion chromatography (Haddad and Jackson 1990). Soluble reactive phosphate (PO_4^{3-}-P), nitrite (NO_2^- -N) and nitrate (NO_3^- -N) and ammonium (NH_4^+-N) were analysed using a Technicon Autoanalyser at EMATEK, Council for Scientific and Industrial Research (CSIR, Mostert 1983). 500 ml water samples were analysed for dissolved trace metals and cold-water acid-extractable trace metal content (DWA 1985).

2.3.2 Aquatic benthic macroinvertebrates

A square-framed sampler (King 1981) with a 0.1 m^2 sample area was used to collect the benthic invertebrates. The downstream (collecting) side of the box was fitted with a funnel of 80 μm mesh netting and a detachable collecting jar. The frame was placed on the bed of the river and all the moveable stones inside the frame were lifted and gently brushed to remove the animals. The substratum was then agitated to a depth of *c.* 10 cm to disturb buried animals, which were carried downstream by the current into the collecting jar. The samples were fixed immediately in 4% formalin and were transferred to 70% alcohol within seven days of collection.

The faunal samples were sorted under a Nikon dissecting microscope. All animals were identified to family level, and in some cases to species level and counted. The following keys were used in the identification: McCafferty (1990), Pennak (1978), Scott (in prep), Wilmot (in prep.). Three replicate samples were collected at each site and processed individually, and the results averaged for each site in each season. Although an increased number of macroinvertebrate replicates would undoubtedly have increased the reliability of the estimates of abundance obtained, financial and time constraints prohibited their collection; Chutter and Nobel (1966) and Canton and Chadwick (1988) have shown that reliable estimates of abundance in a single habitat type can be obtained from three samples of 0.1 m^2.

2.3.3 Numerical analysis

Differences in water quality between sites were investigated using a paired-sample t-test (Zar 1984). Measurements that yielded data that had non-normal distributions were transformed using a log transformation: $y' = (\log_{10}y + 1)$.

Hierarchical clustering and multi-dimensional scaling (MDS) were used to detect similarities and differences in macroinvertebrate community composition among all the sites and among the seasons sampled during the surveys. The classification and ordination procedures contained in the software package PRIMER (Clarke and Warwick 1994), using the Bray-Curtis index of similarity (Bray and Curtis 1957), were applied to a matrix of 28 samples and 35 species (Tables 2.4 and 2.5).

Sensitive multivariate methods of the type used on this data set are only capable of detecting *differences* in the composition of collected samples. Multivariate methods alone do not indicate whether or not the change is deleterious (Clarke and Warwick 1994), although differences in species composition of the invertebrates can be correlated with measured levels of pollutants in the effluents in the rivers. Combined with a knowledge of the tolerances of benthic invertebrates to organic pollution, however, the combination of multivariate analysis of the benthic samples and statistically significant changes in measured chemical variables are a powerful technique for assessing the impact of pollutants on a system (e.g. Chutter 1994, Dallas 1995).

An analysis of the similarity between the macroinvertebrate communities recorded upstream and downstream of the trout farms was performed using Bray-Curtis SIMPER analysis (Clarke and Warwick 1994) in order to identify the species most responsible for the differences between the groups. In this instance, the farms were grouped according to the river zone on which they were situated, *viz*. source, mountain-stream or foothill zone. SIMPER compares the average abundances of species in two different groups, and identifies the average contribution of each species to the differences between the groups (Clarke and Warwick 1994). This allows the identification of species that are good *discriminating* species.

2.4 RESULTS

2.4.1 Substratum

No significant differences were recorded with respect to rock size and substratum type between the sites or between farms, indicating that the riffle areas chosen were fairly homogeneous in terms of substratum.

2.4.2 Water quality

The recorded concentrations and values for the chemical and physical parameters at each site, and the maximum values stipulated by Special Effluent Standards (Amendments to the Water Act 1984, Water Act No. 54 of 1956) are provided in Table 2.1. The results of the paired-sample t-tests are presented in Table 2.2 and discussed below. The test gives a good indication of the effect of a general trout-farm effluent because it combines each determinant from each of the farms into a set of paired (upstream/downstream) data. Thus, the upstream site of each farm is compared with its own downstream site and the general pattern of water quality changes determined. For any one variable there was no notable differences in the effect in the high- or low-flow survey, unless otherwise stated.

Values for pH and temperature upstream, downstream and in the effluent of any one farm were not significantly different. Conductivity was significantly higher in the effluents than at the upstream control sites. Although elevated at the downstream sites relative to the upstream sites, the differences were not statistically significant. There were also significantly higher concentrations of TSS both in the effluents and at the

Table 2.1 Physical and chemical variables recorded upstream, downstream and in the effluents of the seven trout farms visited during the high-flow and low-flow surveys. Major cations and anions were not sampled during the low-flow survey (n.a. = not available).

Farm	Site	Discharge	O_2 % sat	Temp. °C	pH	Cond. µS/cm	NO_3^{2-}-N mg mg l^{-1}	NO_2^--N mg l^{-1}	PO_4^{3-}-P mg l^{-1}	NH_4^+-N mg l^{-1}	Cl^- mg l^{-1}	SO_4^{2-} mg l^{-1}	Mg^{2+} mg l^{-1}	Na^+ mg l^{-1}	Ca^{2+} mg l^{-1}	TDS mg l^{-1}	TSS mg l^{-1}
Farm 1	Upstream	high-flow	71	16.0	5.5	18.2	0.01	0.003	0.01	0.04	4.36	0.14	0.3	4	0	8.38	0.5
		low-flow	83	17.5	n.a.	17.5	0.01	0.003	0.01	0.04						0.00	0.4
	Effluent	high-flow	73	17.7	6.1	43.0	0.40	0.009	0.14	0.36	7.19	0.85	0.5	6	0.1	29.25	5.1
		low-flow	68	18.7	n.a.	33.3	0.33	0.008	0.09	0.34						12.75	4.8
	Downstream	high-flow	94	16.0	5.9	22.9	0.04	0.003	0.02	0.14	4.99	0.48	0.3	5	0	8.25	1.6
		low-flow	100	17.3	n.a.	28.2	0.19	0.007	0.05	0.25						10.75	1.6
Farm 2	Upstream	high-flow	97	19.0	5.9	20.7	0.06	0.003	0.13	0.02	4.68	0.69	0.4	4	0.1	6.88	3.6
		low-flow	93	24.4	6.6	35.4	0.18	0.005	0.03	0.04						26.91	n.a.
	Effluent	high-flow	46	21.0	5.9	20.7	0.06	0.007	0.11	0.29	4.89	0.75	0.4	5	0.1	28.13	9.9
		low-flow	100	23.0	6.5	38.6	0.19	0.003	0.06	0.04						n/a	n.a.
	Downstream	high-flow	98	19.3	4.9	19.5	0.06	0.005	0.03	0.09	4.85	0.67	0.3	5	0.1	15.50	6.1
		low-flow	77	25.5	6.6	39.1	0.17	0.005	0.04	0.14						30.27	n.a.
Farm 3	Upstream	high-flow	95	15.3	6.5	34.3	0.07	0.003	0.01	0.07	8.90	1.63	1.0	6	0.1	36.75	3.1
		low-flow	n.a.	16.5	5.7	72.8	0.06	0.002	0.01	0.03						12.75	0.0
	Effluent	high-flow	91	17.0	6.2	33.0	0.09	0.005	0.03	0.28	10.5	1.58	0.7	6	0.1	22.50	5.2
		low-flow	n.a.	18.0	5.3	74.4	0.11	0.002	0.02	0.15						15.75	5.3
	Downstream	high-flow	91	17.0	6.4	33.8	0.07	0.008	0.01	0.13	8.88	1.68	1.0	6	0.1	21.00	3.9
		low-flow	n.a.	18.0	4.8	76.3	0.10	0.002	0.02	0.08						17.25	2.6
Farm 4	Upstream	high-flow	91	17.0	6.9	44.1	0.04	0.020	0.01	0.05	14.0	1.89	0.5	6	0	21.00	1.1
		low-flow	81	22.3	6.9	68.7	0.15	0.006	0.01	0.07						53.00	2.3
	Effluent	high-flow	73	17.0	6.4	47.3	0.03	0.010	0.10	0.31	14.0	2.96	1.1	7	0.1	28.88	3.5
		low-flow	76	21.9	6.2	64.9	0.16	0.009	0.04	0.31						50.28	3.3
	Downstream	high-flow	94	19.0	7.2	52.5	0.03	0.006	0.02	0.09	n.a.	n.a.	1.0	8	0.1	18.13	1.3
		low-flow	81	23.2	6.4	68.2	0.10	0.007	0.02	0.03						27.5	3.9
Farm 5	Upstream	high-flow	91	17.0	6.9	44.1	0.04	0.020	0.01	0.05	14.0	1.89	0.5	6	0	21.00	1.4
		low-flow	81	22.3	6.9	68.7	0.15	0.006	0.01	0.07						33.00	2.3
	Effluent	high-flow	82	16.0	5.6	62.4	0.14	0.020	0.04	0.21	16.6	1.69	0.9	9	0.1	41.50	4.8
		low-flow	87	21.1	6.6	80.3	0.18	0.010	0.01	0.08						45.25	11.4
	Downstream	high-flow	88	18.5	5.7	37.3	0.10	0.010	0.01	0.08	14.7	1.90	1.0	9	0.1	36.75	2.4
		low-flow	76	20.9	6.6	68.9	0.12	0.010	0.05	0.14						33.00	2.7
Farm 6	Upstream	high-flow	97	14.5	6.9	28.4	0.02	0.004	0.01	0.08	6.04	0.53	0.6	6	0.1	18.25	2.3
		low-flow	82	15.5	6.6	24.4	0.03	0.003	0.01	0.03						12.50	2.4
	Effluent	high-flow	82	15.5	6.5	31.3	0.08	0.009	0.06	0.26	5.99	0.61	0.6	6	0.1	24.75	5.1
		low-flow	69	16.0	6.5	43.5	0.35	0.350	0.25	0.41						16.50	4.7
	Downstream	high-flow	91	16.1	6.5	45.3	0.10	0.010	0.07	0.22	6.63	0.64	0.4	8	0.1	27.75	2.7
		low-flow	69	16.9	6.3	46.2	0.45	0.450	0.27	0.39						21.00	4.8
Farm 7	Upstream	high-flow	79	12.0	6.7	23.6	0.01	0.010	0.01	0.03	4.62	0.58	0.4	5	0	0.00	1.0
		low-flow	87	14.75	6.4	22.9	0.01	0.005	0.01	0.15						0.25	0
	Effluent	high-flow	58	12.5	6.9	40.0	0.06	0.005	0.10	0.21	5.48	0.74	0.5	5	0.1	14.88	11.6
		low-flow	70	16.0	6.5	332	0.09	0.007	0.14	0.35						7.75	2.5
	Downstream	high-flow	87	13.3	6.8	13.3	0.05	0.007	0.03	0.20	5.46	0.72	0.4	4	0	18.00	4.0
		low-flow	79	17.1	6.7	35.0	0.12	0.010	0.15	0.35						10.75	1.2
Special Stds max. allowable concentrations.			At least 75% saturation	Max. 25°C	5.5-7.5	< 250	1.50		1.00	1.00	0.0	0.05		<0.05 above the value at the inlet			10

downstream sites than at the upstream sites (Table 2.1).

No significant differences in the high-flow concentrations of any of the major anions or cations measured were recorded between sites or between farms (Table 2.2).

Table 2.2 Results of the paired-sample t-tests. If t>=2.447 then the difference between the two sets of samples is statistically significant (SS). The statistically significant variables are marked with an *, U/S = upstream and D/S = downstream of the effluent outlets.

Variable	Season	U/S vs. effluent $t_{0.05(2),7} = 2.447$		U/S vs. D/S $t_{0.05(2),7} = 2.447$	
Oxygen	High-flow	t=0.96		t=0.34	
	Low-flow	t=0.58		t=0.35	
Temperature	High-flow	t=1.11		t=1.60	
	Low-flow	t=0.72		t=0.57	
Conductivity	High-flow	t=6.18	*	t=0.07	
	Low-flow	t=0.91		t=1.08	
Total dissolved solids	High-flow	t=0.87		t=0.35	
	Low-flow	-		-	
Total suspended solids	High-flow	t=15.36	*	t=2.98	*
	Low-flow	t=3.27	*	t=4.58	*
Nitrite	High-flow	t=3.00	*	t=1.59	
	Low-flow	t=0.39		t=1.38	
Nitrate	High-flow	t=5.65	*	t=1.36	
	Low-flow	t=4.87	*	t=9.34	*
Phosphate	High-flow	t=16.67	*	t=0.00	
	Low-flow	t=4.62	*	t=9.36	*
Ammonia	High-flow	t=19.25	*	t=17.00	*
	Low-flow	t=3.81	*	t=17.41	*

The concentrations of nitrate, nitrite, ammonium and phosphate were significantly higher in the effluents than at the upstream control sites, with the exception of nitrite concentrations during the low-flow survey (Table 2.2). The concentrations of nitrate, phosphate and ammonium also were significantly higher in the river downstream of the effluent outlet than at the upstream sites, with the exception of nitrate and phosphate levels during the high flow survey.

By far the majority of the trace elements were below the level of detection of the methods used (Table 2.3). No significant trends between sites were evident, in either

Table 2.3 Results of the trace chemical analyses (dissolved metals, mg l^{-1}) of water samples collected during the high-flow survey, upstream, downstream, and in the effluents of the seven trout farms that formed part of the survey (analyses done by Hydrological Research Institute, Department of Water Affairs and Forestry).
< = below the detection limit.

Trace metal	FARM 1			FARM 2			FARM 3			FARM 4			FARM 5			FARM 6			FARM 7		
	Up-stream	Effl.	Down-stream	Up-stream	Effl.	Down-stream	Up-stream	Effl.	Down-stream	Up-stream	Effl.	Down-stream	Up-stream	Effl.	Down-stream	Up-stream	Effl.	Down-stream	Up-stream	Effl.	Down-stream
Aluminium	< 0.100	< 0.100	< 0.100	< 0.100	< 0.100	< 0.100	< 0.100	< 0.100	< 0.100	< 0.100	< 0.100	< 0.100	< 0.100	< 0.100	< 0.100	< 0.100	< 0.100	< 0.100	< 0.100	< 0.100	< 0.100
Arsenic	< 0.100	< 0.100	< 0.100	< 0.100	< 0.100	< 0.100	< 0.100	< 0.100	< 0.100	< 0.100	< 0.100	< 0.100	< 0.100	< 0.100	< 0.100	< 0.100	< 0.100	< 0.100	< 0.100	< 0.100	< 0.100
Barium	< 0.004	0.009	< 0.004	0.010	< 0.004	0.004	0.004	< 0.004	0.004	0.008	< 0.004	0.007	0.008	< 0.004	0.006	< 0.004	0.017	0.005	< 0.004	0.006	0.009
Boron	< 0.002	< 0.002	< 0.002	< 0.002	< 0.002	< 0.002	< 0.002	< 0.002	< 0.002	< 0.002	< 0.002	< 0.002	< 0.002	< 0.002	< 0.002	< 0.002	< 0.002	< 0.002	< 0.002	< 0.002	< 0.002
Beryllium	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001
Cadmium	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005
Cobalt	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020
Chromium	< 0.005	< 0.005	< 0.005	0.005	< 0.005	0.005	0.005	< 0.005	< 0.005	0.007	< 0.005	0.008	0.007	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005
Copper	< 0.005	0.009	< 0.005	< 0.030	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005
Iron	0.144	0.031	< 0.020	0.251	0.168	0.097	0.008	0.128	0.127	< 0.020	0.126	0.013	< 0.020	0.097	0.206	0.082	0.072	0.568	0.043	0.042	< 0.020
Manganese	0.003	0.001	0.002	0.006	0.002	0.004	0.003	0.013	0.016	0.002	0.003	0.004	0.002	0.005	0.002	0.006	0.004	0.019	0.005	0.001	< 0.001
Molybdenum	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005
Nickel	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020
Lead	< 0.050	< 0.050	< 0.050	< 0.050	< 0.050	< 0.050	< 0.050	< 0.050	< 0.050	< 0.050	< 0.050	< 0.050	< 0.050	< 0.050	< 0.050	< 0.050	< 0.050	< 0.050	< 0.050	< 0.050	< 0.050
Strontium	0.009	0.013	0.012	0.010	0.008	0.018	0.015	0.002	0.010	0.006	0.010	0.006	0.006	0.007	0.010	0.004	0.013	0.013	0.006	0.001	0.006
Titanium	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001
Vanadium	< 0.002	0.003	< 0.002	0.006	0.006	0.005	< 0.002	< 0.002	< 0.002	0.006	0.006	0.006	0.006	< 0.002	< 0.002	< 0.002	< 0.002	0.004	< 0.002	< 0.002	< 0.002
Zinc	< 0.004	0.026	< 0.004	0.299	< 0.004	0.007	0.008	0.022	0.014	0.004	0.005	0.006	0.004	0.009	0.026	0.008	< 0.004	0.012	< 0.004	0.021	< 0.004
Zirconium	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	0.025	< 0.020	< 0.020	< 0.020	< 0.020	0.036	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020

dissolved or acid-extractable trace metals, although the data were obtained from spot-sampling which can miss one-off introductions of pollutants.

2.4.3 Macroinvertebrate communities

The macroinvertebrate species and their abundances recorded at each site during the winter and summer sampling sessions are presented in Table 2.4 and 2.5, respectively.

The results of the multivariate analysis of these data are presented in two ways: the cluster analysis is presented as a dendrogram (Figure 2.4) and the MDS as an ordination plot (Figure 2.5). The order in which the samples are presented in a dendrogram is optional (within defined limits) and, in Figure 2.4, they have been ordered to facilitate the explanation of the relationships between them. Cluster analysis attempts to group samples into discrete clusters, whereas MDS displays their inter-relationships on a continuous scale (Clarke and Warwick 1994). In the case of MDS, it is usually not possible to satisfy the relative similarity rankings between the samples in two dimensions. Hence, there will be some distortions, or stress, between the similarity rankings and the corresponding distances rankings in the ordination plot, which could render the final interpretation misleading (Clarke and Warwick 1994). However, when the same relationships between samples are clearly shown by both methods, then the patterns provide a good representation of the degree of similarity of invertebrate communities collected at different places and times.

In order to appraise the relative degree of impact at each site during each season, these results were combined with a knowledge of the tolerances to various water quality variables, and the feeding habits, of the benthic invertebrate groups present in each sample (Pennak 1978, Wiederholm 1984, Dallas and Day 1993).

Farms 1 and 7 are situated on mountain streams, Farm 6 on a source zone, and Farms 2 - 5 on foothill zones. In the dendrogram, the patterns of changes in faunal distribution due to the presence of trout farms were clearest for those farms situated on mountain-stream or source zones. In terms of the upstream sites the invertebrate community at Farm 6 was most similar to those at Farms 1 and 7, with the presence of

Table 2.4 Abridged list of taxa and average abundance (per m^2 of river bed) of benthic macroinvertebrates upstream and downstream of each trout farm sampled during the high-flow survey. Pollution-sensitive groups are indicated by means of shading and pollution-tolerant groups by a single thick border

Taxonomy Order	Family	Subfamily, Tribe or *genus*	Farm 1		Farm 2		Farm 3		Farm 4		Farm 5		Farm 6		Farm 7	
			Upstream	D-stream	Upstream	D-stream	Upstream	D-stream	Upstream	D-stream	Upstream	D-stream	Upstream	D-stream	Upstream	D-stream
Diptera	Simuliidae		210	40	70	35	30	230	870	1470	870	290	2480	2060	420	278
	Ceratopogonidae		0	0	0	0	0	0	0	0	0	0	0	0	10	0
	Rhagionidae		10	0	0	10	0	0	0	0	0	0	0	30	20	0
	Chironomidae	Orthocladinae	4070	4130	4060	6130	1310	4710	3690	303	3690	2430	290	4790	3890	2380
		Tanypodinae	30	0	60	20	30	50	120	30	120	90	70	60	60	0
		Chironomini	0	0	0	10	0	70	10	90	10	70	0	1480	0	10
		Tanytarsini	10	20	40	120	40	1330	620	250	620	1500	0	470	20	0
		Chaoborinae	0	0	0	0	10	70	0	0	0	0	0	0	0	0
Ephemeroptera	Leptophlebiidae	*Aprionyx* sp.	0	0	0	0	0	0	0	0	0	0	20	0	0	0
		Choroterpes sp.	0	0	0	40	0	0	0	0	0	0	0	0	20	0
		Adenophlebia sp.	0	0	0	0	0	0	0	0	0	0	0	10	0	0
		Castanophlebia sp.	10	0	10	20	0	0	0	10	0	10	600	310	60	0
	Baetidae		230	10	240	420	150	380	2480	2030	2480	3520	6800	1360	1460	50
		Baetis sp.	-	-	-	-	-	-	-	-	-	-	16%	61%	56%	100%
		Acentrella capensis	-	-	-	-	-	-	-	-	-	-	84%	38%	43%	0%
	Ephemerellidae	*Lestagella* sp.	10	10	0	20	0	0	0	0	0	0	60	0	0	10
		Ephemerellina sp.	10	0	0	10	0	10	10	0	10	0	10	0	30	10
Plecoptera			10	0	10	0	0	0	0	10	0	1	240	220	180	10
Trichoptera			10	10	0	0	0	0	20	0	20	0	290	50	10	0
Coleoptera	All Adults		0	0	0	0	0	0	0	0	0	0	0	0	270	0
Coleop. larva	Helodidae		10	10	0	10	10	0	0	0	0	0	170	10	580	10
	Elimidae	Species a	20	0	0	10	0	0	0	10	0	1	980	60	150	0
		Species b	10	0	0	0	0	0	0	10	0	0	40	10	0	0
		Species c	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Species d	0	0	0	0	0	0	0	0	0	0	190	0	340	0
Megaloptera	Corydalidae		0	10	0	0	0	0	0	0	0	0	0	0	20	0
Planaria			0	0	0	0	0	0	0	0	0	0	60	270	20	30
Oligochaeta	Lumbriculidae		10	10	10	10	0	0	0	10	0	0	0	1520	10	10
	Naididae	*Nais* sp.	10	30	40	230	0	0	0	0	0	60	180	11260	0	1630
Amphipoda			0	0	0	10	0	0	0	0	0	0	1580	0	0	0

Table 2.5 Abridged list of taxa and average abundances (per m^2 of river bed) of benthic macroinvertebrates upstream and downstream of each trout farm visited during the low-flow survey. Pollution-sensitive groups are indicated by means of shading and pollution-tolerant groups by a single thick border

Order	Family	Subfamily, Tribe or *Genus*	Farm 1		Farm 2		Farm 3		Farm 4		Farm 5		Farm 6		Farm 7	
			Upstream	D-stream	Upstream	D-stream	Upstream	D-stream	Upstream	D-stream	Upstream	D-stream	Upstream	D-stream	Upstream	D-stream
Diptera	Simuliidae		193	187	37	1093	167	687	3310	1483	3310	2240	217	2790	2690	1007
	Ceratopogonidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Rhagionidae		47	13	227	93	3	17	0	7	0	3	0	3	0	3
	Chironomidae	Orthocladinae	1497	1173	3180	4203	2980	5703	1900	1840	1900	1840	17	1353	2390	9460
		Tanypodinae	163	160	2397	383	77	73	0	20	0	7	27	0	393	7
		Chironomini	53	0	403	43	67	213	0	10	0	0	0	740	450	17
		Tanytarsini	950	14055	2410	960	827	583	130	227	130	360	30	770	87	213
		Chaoborinae	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ephemeroptera	Leptophlebiidae		557	17	1087	447	0	0	0	3	0	0	830	10	83	3
	Baetidae		1447	1540	2770	2030	827	393	2440	1387	2440	1317	1447	1540	983	273
		Baetis sp.	42%	55%	100%	100%	32%	100%	100%	84%	100%	96%	45%	94%	54%	87%
		Acentrella capensis	58%	45%	0%	0%	68%	0%	0%	16%	0%	4%	55%	6%	46%	13%
	Heptageniidae		463	143	0	0	3	13	0	0	0	3	36	0	440	3
Plecoptera			63	3	0	0	0	0	0	0	0	0	30	30	390	10
Trichoptera			147	183	90	303	4480	2660	1370	467	1370	607	30	13	93	0
Coleoptera	Adults		0	3	63	177	0	0	0	0	0	0	0	3	0	0
	Helodidae		540	20	20	40	0	0	0	3	0	3	383	0	70	3
	Elimidae	Species a	110	30	223	400	30	60	0	20	0	7	47	10	23	0
		Species b	0	0	0	0	0	0	0	0	0	0	707	33	117	0
		Species c	37	47	0	0	17	143	0	0	0	0	200	13	217	0
		Species d	40	0	0	0	0	0	0	0	0	0	0	0	83	0
Megaloptera	Corydalidae		7	23	3	7	7	3	10	3	10	0	0	0	10	0
Planaria			0	0	0	0	0	0	0	0	0	0	0	13	20	1443
Oligochaeta	Lumbriculidae		7	0	0	0	0	7	0	0	0	3	0	3	10	0
	Naididae	*Nais* sp.	47	7	7757	107	0	0	0	63	0	7	30	1340	103	2350
Amphipoda			0	0	0	0	0	0	0	0	0	0	1293	10	0	0

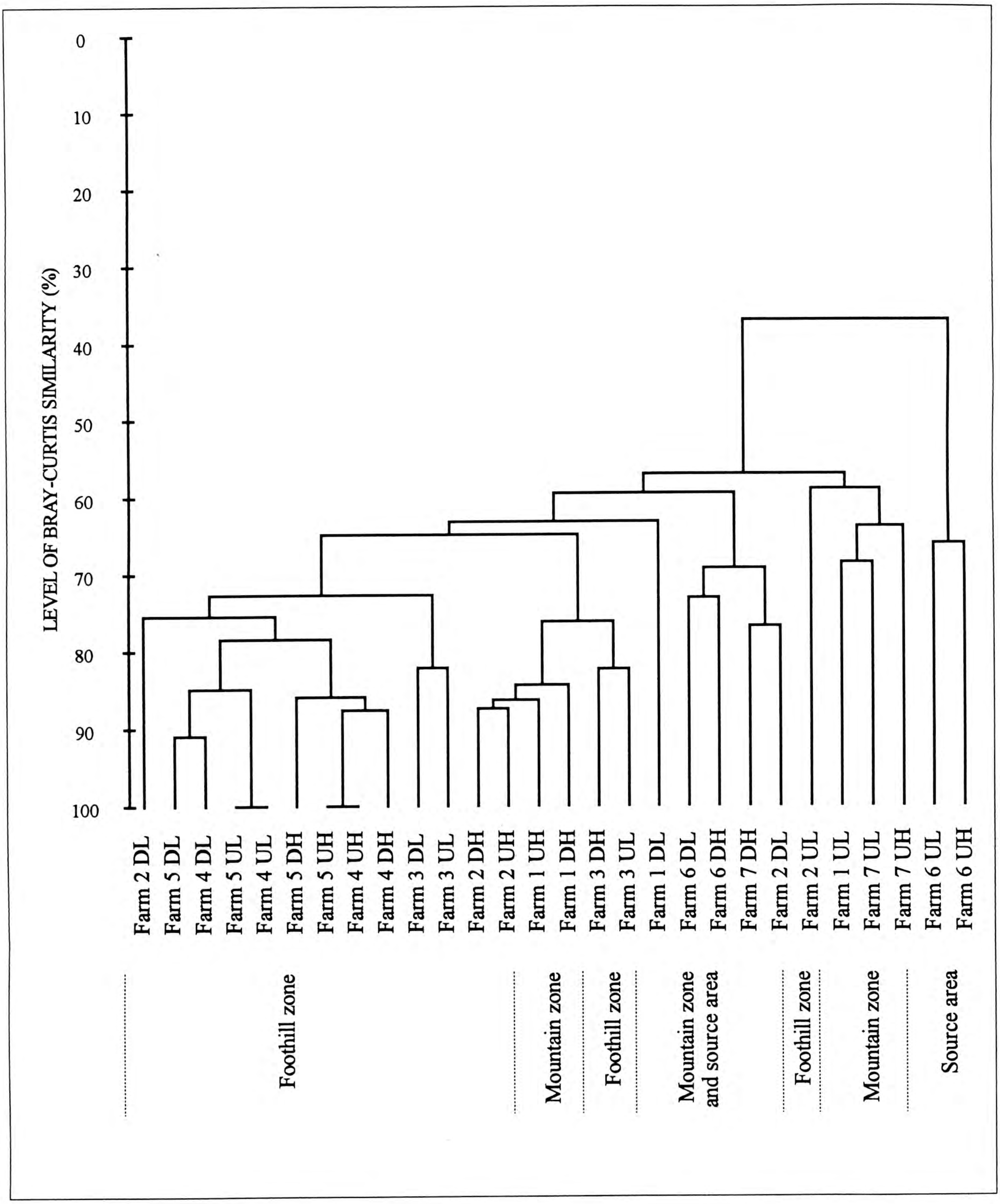

Figure 2.4 Dendrogram depicting the results of the Bray-Curtis hierarchical clustering, based on macroinvertebrate community structure of the river sites sampled during the surveys. U = upstream of the farm, D = downstream, H = high-flow survey and L = low-flow survey

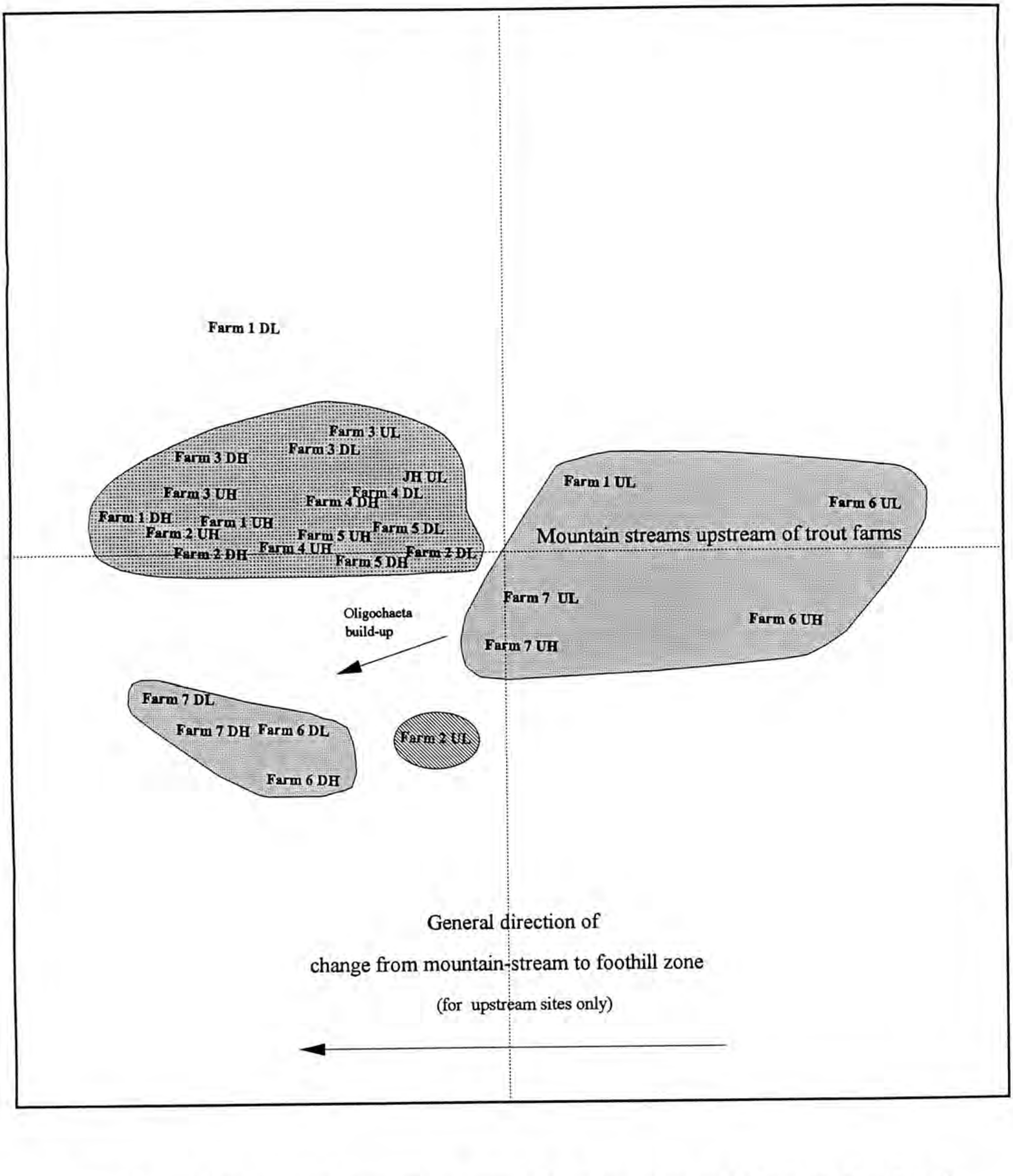

Figure 2.5 Ordination plot depicting the results of the Bray-Curtis multidimensional scaling of differences in community structure between sites sampled during the initial survey. U = upstream of the farm, D = downstream of the effluent outlet, H = high-flow survey and L = low-flow survey

Amphipoda at Farm 6 (Tables 2.4 and 2.5) being the main reason that the three farms did not cluster together in Figure 2.4. The communities upstream of these three farms (Farm 1 UL, Farm 6 UH, Farm 6 UL, Farm 7 UH, Farm 7 UL) were less than 60% similar to those of any of the other upstream sites (Figure 2.4) with the exception of Farm 1 in the high-flow survey (Farm 1 UH), the river upstream of which had been bulldozed shortly before the samples were collected (A. Coetzer, CNC, pers. comm.).

The communities downstream of these three farms were different from those at the upstream sites. Farm 1 DH and DL were similar to each other, and Farm 6 DH and DL and Farm 7 DH and DL were clustered with one another. Thus, the farms had similar effects on the rivers on which they were situated. Farm 6 had the greatest effect on the river on which it was situated in that its downstream sites were only 40% similar to the unimpacted upstream control sites, while Farm 1 and Farm 7 were 60% similar to their upstream sites. The remaining farms (those on foothill zones) showed between 75% and 90% similarity between their upstream and downstream sites in any one season.

The ordination plot (Figure 2.5) supported the findings of the cluster analysis. Concentrating again on those farms situated on mountain-stream and source zones, the samples collected in the undisturbed reaches upstream of the farms clustered together, with the exception of the site upstream of Farm 1 in the high-flow survey. The sites downstream of the same farms were in distinct clusters separate from the upstream sites, indicating the farms had a large impact on the community structure in their rivers. Sites downstream of farms in the foothill zone clustered close to their upstream control sites, suggesting the farms had less impact on their downstream rivers than did those farms situated on mountain streams or source zones.

The general trends with respect to the changes in abundance of key taxa, as identified by Bray-Curtis SIMPER analysis, are summarised in Table 2.6.

2.5 DISCUSSION

2.5.1 Physical and chemical changes

Of the variables measured upstream and downstream of the trout farms, only suspended particulate matter and nutrient concentrations were significantly higher in the river immediately downstream of the trout farms relative to their concentrations in the river upstream of influence of the farms.

Table 2.6 General trends with respect to the changes in abundance of key taxa, identified using SIMPER analysis (Clarke and Warwick 1994), upstream and downstream of trout farms grouped according to the hydroecological zone in which they were situated. Key: * = 0 – 10 animals m^{-2}, ** = 11 - 50, *** = 51 - 100, **** = 101 - 500, ***** = 501 - 1000, ****** = >1000.

Taxa	Source		Mountain-stream		Foothill zone	
	Upstream	Downstream	Upstream	Downstream	Upstream	Downstream
Amphipoda	******	*	-	-	-	-
Limnichidae	****	*	****	-	-	-
Helodidae	****	*	****	*	*	*
Plecoptera	****	****	****	*	*	*
Elmidae	*****	**	****	*	**	***
Trichoptera	****	**	***	**	*****	****
Heptageniidae	**	-	****	**	*	*
Ephemerellidae	**	-	**	*	***	*
Leptophlebiidae	*****	****	****	*	****	***
Tanytarsini	**	*****	****	******	*****	*****
Chironomini	-	******	****	*	***	***
Tanypodinae	**	**	****	**	****	***
Orthocladinae	***	******	******	******	******	******
Planaria	**	****	-	****	-	-
Simuliidae	******	******	****	******	******	******
Lumbriculidae	-	*****	-	*	-	*
Naididae	***	******	-	******	*****	***

The solids suspended in the effluents (TSS) appeared to consist mainly of uneaten fish food and faeces. Because the solids did not settle out completely in sediment ponds or were not removed by filtration, various amounts remained in suspension in the effluents and settled out in the rivers immediately downstream of the outlets, where they decomposed. The increases in the concentration of dissolved solids (TDS) in the effluent and downstream river, relative to those in the upstream reaches were not as marked as those in TSS. In some instances, however, the TDS concentration at the sampling site downstream of a farm was higher than that in the effluent. This could have been a result of leaching from the decomposing solids settled in the downstream reach.

Oxygen saturation levels in the mountain-stream and foothill zones of south-western Cape rivers are normally in excess of 80% (Dallas and Day 1993). The oxygen levels in the riffle sections of the rivers downstream of each effluent outlet were usually at about this level, and were not significantly different from those in the related upstream riffles. The backwaters upstream of the farms also yielded values around 80%. The levels of oxygen saturations measured in the slow-flowing areas downstream of the farms in this study, however, were seldom above 40%. This could have been because the organic material suspended in the effluents tended to settle out there. Where settlement had occurred, decomposition of the organic material could have resulted in an increase in chemical oxygen demand (COD) and biological oxygen demand (BOD), which would account for the reduction in available oxygen in the slow-flowing areas.

The average ammonium concentrations in the effluents were approximately eight times higher than at the upstream sites during the high-flow survey and approximately five times higher during the low-flow survey. The ammonium ion (NH_4^+) is non-toxic but exists in dynamic equilibrium with free ammonia (NH_3) which is highly toxic to aquatic life. The chemical species present depends on pH: NH_4^+ occurs exclusively at low pH (< 6; acid conditions) and NH_3 predominates at high pH (alkaline conditions). In natural waters in the south-western Cape the non-toxic ammonium ion predominates.

2.5.2 Changes in macroinvertebrate communities in the source and mountain-stream zones

There were three farms situated in the source and mountain-stream zones, namely Farms 1, 6 and 7. The species composition of the site upstream of Farm 6, situated on the source zone, differed from those in the other two zones in the large number of Amphipoda in the samples (>1000 m^{-2}). Apart from this difference, the mountain-stream and source zone sites upstream of the farms, with the exception of Farm 1 in the high-flow survey, had a similar species composition in terms of both numbers of individuals and taxa present in the sample. In the case of Farm 1, the paucity of representatives of the Coleoptera, Plecoptera, Trichoptera and the sensitive Ephemeroptera (e.g. Leptophlebiidae) at both upstream and downstream sites during the high-flow survey indicated that the river was, or had recently been, disturbed, masking the impact of the farm itself. Subsequent enquires revealed that the reaches upstream of the farm had been bulldozed shortly before the high-flow survey in order to create trout ponds.

The impact of the trout-farm effluent downstream of Farms 6 and 7 was marked. Downstream of Farm 1, using just the low-flow data (see above), the impact was considerably less dramatic, however. The primary difference between the farms was that Farm 1 used earth dams to house the fish while the other two farms used plastic portapools.

At Farm 7, the faunal composition of the samples collected downstream of the effluent outlet differed considerably from that of the upstream site. There was an increase in the numbers of simuliid larvae, suggesting increased particulate organic material in the water column. The pollution-sensitive, predatory tanypodine chironomids were absent from the downstream site. In addition, the ratio of the two tribes comprising the Chironominae swung in favour of the Chironomini *(Polypedilum* sp.), which benefit from organic pollution (Berhe *et al.* 1989). The second tribe, the Tanytarsini, which was present in moderate numbers in the river upstream of the influence of the farm, was absent from the downstream samples. The numbers of Coleoptera, Plecoptera and Ephemeroptera (including Baetidae) were all considerably lower downstream of the effluent outlet than upstream thereof. These were replaced by a large population (1630 m^{-2}) of worms of the genus *Nais* at the downstream site. The changes in species composition downstream of Farm 7 were almost identical to those recorded at Farm 6.

The river immediately upstream of the inlet to Farm 6 had a benthic faunal community characteristic of an undisturbed source zone as described by Harrison and Elsworth (1958) and King (1982). The Amphipoda are characteristic of source areas and large numbers are common in unpolluted, clear waters (Pennak 1978). Trichopteran, plecopteran, and coleopteran larva and the leptophlebiid and ephemerellid ephemeropterans were all present in the upstream samples. In contrast, all of these groups, with the exception of the plecopterans (*Aphanicerca* species complex), were either absent, or considerably reduced in number, in the samples from the downstream site. Under conditions of high flow, the numbers of planarians increased downstream of the farm, as did the numbers of lumbriculid worms (*Lumbriculus*: 0 m^{-2} upstream to 1520 m^{-2} downstream) and naidid worms (*Nais*: 180 m^{-2} upstream to 11 260 m^{-2} downstream). These aquatic worms obtain their food by ingesting quantities of the substratum and digesting the organic component, in much the same way as do earthworms. They are normally common in the organically rich mud and debris on the bottom of stagnant pools and ponds, and occur in large numbers in the presence of organic pollution (Pennak 1978). The situation under conditions of low flow was similar to that recorded under conditions of high flow. Many taxa present upstream of the farm were either absent or greatly reduced in numbers downstream of the effluent outfall. These taxa included Amphipoda and Coleoptera, as well as the ephemeropteran families, Leptophlebiidae and Heptageniidae. There was an accompanying increase in the number of Simuliidae (217 m^{-2} upstream; 2790 m^{-2} downstream) and Chironomidae (74 m^{-2} upstream; 2863 m^{-2} downstream). There was also an increase in the numbers of naidid worms (30 m^{-2} upstream; 1340 m^{-2} downstream) but, in contrast to the situation found in the high-flow survey, this was not accompanied by an increase in large Lumbriculidae.

At Farm 1, which had earthdams, the impact of the effluent appeared less marked than downstream of the other two farms. Nevertheless, some sensitive groups, such as ephemerellids of the *Ephemerellina* complex, and members of the coleopteran family Elmidae, present upstream, were missing from the downstream samples. This indicated some impact on the river by the trout farm in excess of the upstream impact(s). In the low-flow survey, the trichopteran family, the Glossosomatidae, a common component of summer mountain-stream communities in the south-western Cape, was absent downstream of the effluent outlet of Farm 1. Another trichopteran family, the Hydropsychidae, which is indicative of mild organic pollution (Wiederholm 1984), increased in number downstream of the outlet, however. The different responses of these two families can be attributed to differences in feeding habits. The Glossosomatidae have mouthparts that are specialised for scraping minute organic particles from rock surfaces, while the Hydropsychidae construct fine nets that strain particulate matter from the water (Pennak 1978). Hence the increase in hydropsychid larvae downstream of the farm was probably a response to increased suspended material in the water column downstream of the effluent outlet. Another component of the mountain-stream summer community, the Heptageniidae, *Afronurus* sp., was less abundant at the downstream site (143 m^{-2}) than at the upstream site (463 m^{-2}). Like the Glossosomatidae, the Heptageniidae feed on particles attached to rock surfaces (Pennak 1978), and the reduction of both these groups suggests that the farm adversely affected the species utilising this food source. An important aspect downstream of Farm 1 was that there were no marked increases in the numbers of naidid or lumbriculid worms compared to the upstream situation, under either high- or low-flow conditions. Such an increase occurred downstream of both the portapool farms situated in a mountain stream zone that were sampled during this survey.

In summary, the general impact of trout-farm effluent on the mountain-stream and source zones was to eliminate or greatly reduce the number of Limnichidae, Helodidae, Plecoptera, Elmidae, Heptageniidae and Ephemerellidae and, in the case of portapool farms, to replace these with Naididae, Lumbriculidae, Chironomidae and planarians.

2.5.3 Changes in macroinvertebrate communities in the foothill zone

There were four farms situated on the foothill zone, namely Farms 2, 3, 4 and 5. All of the rivers upstream of these farms had already been impacted by forestry, water abstraction or other disturbances, such as the presence of other trout farms. As a result the upstream sites either lacked or supported only small numbers of several of the more sensitive taxa that were present in the source and mountain-stream zone, such as Limnichidae, Helodidae, Plecoptera and Elmidae.

Farm 4 was situated downstream of a major forestry area and a small impoundment, and Farm 5 was situated *c.* 400 m downstream of Farm 4. This may have accounted for the relatively small differences recorded between the site upstream and downstream of the farms.

Farm 3 was situated on a stretch of river immediately downstream of an inter-basin transfer scheme. Although the impacts of the inter-basin transfer scheme on the river ecosystem are, as yet, poorly known, large numbers of copepods (K. Snaddon, Freshwater Research Unit, University of Cape Town, pers. comm.) characteristic of an impoundment, and, in the high-flow survey, Chaoborinae, which feed on the Crustacea, were present in both the upstream and downstream samples. Furthermore, the river had been bulldozed at the inlet to the farm to facilitate flow into the inlet channel and during periods of low flow, virtually no water flowed between the farm inlet and the outlet (A. von Felewski, Landowner, pers comm.). Since the upstream samples were collected slightly downstream of the inlet of the trout farm (high flow conditions prevented entry into the river at a higher point), it was not clear whether the low numbers and diversity recorded at the upstream site were a result of the transfer scheme, of bulldozing of the river bed at the inlet, or of insufficient flow between inlet and outlet during the summer months.

The river upstream of Farm 2 received water polluted with organic material and sediment from a small tributary. Interpretation of the results obtained at this farm was, however, further complicated by the fact that between the upstream and downstream sampling sites, several pristine mountain streams flowed into the main river. Downstream drift of benthic invertebrate species from these unimpacted streams and the input of unpolluted water could account for the increase in sensitive species downstream of the farm.

In summary, the impacts of the trout-farm effluents on the invertebrate communities in the foothill zones was considerably less noticeable than in the mountain-stream and source zones. This was probably because the sites upstream of the farms were already impacted and thus any additional impacts were difficult to detect (e.g. Storey et al. 1990). In general, there were only minor changes in community structure specifically among the chironomid sub-families, with a reduction in the number of Tanypodinae and an increase in Chironominae (particularly Chironomini), suggesting the presence of organic pollution (Berhe *et al.* 1989). In some cases, such as Farms 4 and 5, however, taxa such as ephemerellid *Ephemerellina* sp. and a trichopteran (Philopotamidae), which are sensitive to pollution, were present at the upstream sites in low numbers but were absent from the downstream sites.

2.5.4 Synthesis of faunal data

The impacts of the trout-farm effluents on the rivers, judged purely on their impacts on invertebrate community structure, were to eliminate some of the sensitive species and, in the worst cases, to provide abundant habitat for aquatic worms. Three main conclusions can be drawn from the data:

1. the responses of the benthic communities in the foothill zone to the release of trout-farm effluent were less marked than those of communities in the mountain-stream and source zones. This can be related to the condition of the river upstream of the inlets of the individual farms: the foothill zones were already impacted;
2. there was little difference between the impacts of most of the trout farms under conditions of low-flow and high-flow (see Figures 2.1 and 2.2), which may have been because reduced stocking rates in the summer counteracted the lower flows and higher temperatures;

2.5.5 Infrastructure: earth dams versus plastic portapools

Of the three farms that were situated on mountain streams and source areas, two used plastic portapools (Farms 6 and 7) and the third used earth dams (Farm 1). There was a substantial increase in the number of oligochaetes downstream of both 'portapool' farms and yet, despite being situated in the same sensitive river zone, this did not occur downstream of the farm that used earth dams. Oligochaetes derive most of their nutrition from bacteria and are found in stony streams when sufficient organic matter is introduced to maintain a thick bacterial slime on the substratum (Brinkhurst and Cook 1974). The oligochaetes downstream of Farms 6 and 7 occurred in organic deposits not evident downstream of the earth-dam farms.

Of the farms situated on foothill zones, only one used portapools (Farm 4) while all the others used earth dams. In contrast to farms in the mountain-stream or source zones using portapools, there was no significant build-up of oligochaetes in the site downstream of Farm 4, although *Nais* sp. were found at the downstream site (0 m^{-2} upstream; 63 m^{-2} downstream in the low-flow survey). The effluent from that farm, however, was not discharged directly into the river but into an earth-lined canal that flowed for approximately 60 m before entering the river. Organic deposits, similar to those downstream of the two mountain-stream farms, occurred in this canal but not in the downstream sampling site, 100 m downstream of the point where the canal flowed into the river. It thus appears that the canal was acting as a type of settlement facility.

Unlined earth dams provide for a certain amount of settlement of solids and thus the solids in suspension in the effluent are often less per corresponding weight of fish, than in the effluents from portapools (Drummond 1990). The interchange of water in portapool systems appears too fast to allow waste food and faeces to decompose before being discharged into the river. The total number of farms sampled during the surveys was small, however and, for this reason, only tentative conclusions have been reached concerning the different impacts of portapools and earth-dam facilities.

2.5.6 Probable causes of faunal change

The results indicate that, despite the fact that in most cases trout-farm effluents comply with Special Effluent Standards (Amendments to the Water Act 1984), they do have detrimental effects on downstream river ecosystems relative to their unimpacted upstream reaches. These downstream reaches showed signs of organic enrichment and the loss, to a greater or lesser extent, of pollution-sensitive species. This was accompanied by the appearance of pollution-tolerant species and, for some farms, the complete dominance of the downstream community by a pollution-tolerant species, such as naid worms. Enrichment of the river by the trout farms almost certainly provided the food for faunal components such as the Oligochaeta and Simuliidae that were abundant downstream of the effluent outlets.

Data collected during the surveys showed significant relationships between changes in benthic invertebrate community structure downstream of the trout farms and increases, in the effluent, of concentrations of total suspended solids, total dissolved solids, nitrite, nitrate, ammonia and phosphate. Studies in other parts of the world (e.g., Allcock and Buchanan 1994) have shown that the suspended solids in trout-farm effluent are responsible for the major impact on the downstream river ecosystem, and that a reduction in or removal of suspended organic solids will result in a marked reduction in nutrient levels in the effluents (Nature Conservancy Council 1990). The impressions gained during this preliminary investigation supported these research findings.

2.5.7 Conclusion

There is a need for further consideration of the potential impacts of effluents entering mountain streams and upper rivers in the south-western Cape. Clearly, the existing legislation (i.e. General and Special Effluent Standards) does not provide adequate protection for the endemic biota that would occur naturally in these upper river reaches. In the case of trout farms, the effluents had impacts on the benthic fauna that ranged from mild to severe depending on the location of the farm. However, more than the river reach, it was the condition of the river upstream of the farm which determined the extent of the impact of the effluent on the river ecosystem: those rivers that were already degraded by some other perturbation showed little change in invertebrate community structure following the input of trout-farm effluent. If those changes in DWAF policy designed to improve the protection given to riverine ecosystems are to be effective it is imperative that the response of the riverine biota to different types of pollution be quantified and appropriate site-specific standards set for industrial or agricultural enterprises releasing their effluents into the rivers. When such site-specific standards are set it will be necessary to consider whether or not they should be lower for rivers that are already degraded than for relatively undisturbed ones.

SECTION III

MAIN STUDY

CHAPTER 3

CHANGES IN THE PHYSICAL AND CHEMICAL PROPERTIES OF TWO SOUTH-WESTERN CAPE MOUNTAIN STREAMS AS A RESULT OF RECEIVING EFFLUENT FROM LAND-BASED TROUT FARMS

3.1 INTRODUCTION

The pilot study of seven farms in the south-western Cape (Chapter 2) indicated that effluents discharged from portapool farms were more detrimental to the downstream river than effluents from earthdam farms, because unlined earth dams provide for a certain amount of interaction with the sediments and settlement for solids. Thus, the solids in suspension in the effluent from earthdams are often lower for a corresponding weight of fish, than for portapools (Drummond 1990). The interchange of water in portapool systems appeared to be too fast to allow waste food and faeces to decompose before being discharged into the river. Furthermore, the plastic lining allowed the portapools to be scrubbed clean and flushed. The primary intention of this chapter is to compare the loadings originating from a farm that used plastic-lined "portapools" to house its fish with those from a farm that used unlined earthdams.

The fish at both of the study farms were fed on a diet of dry pellets. The relationship between fish diet and the constitution of fish-farm effluents is discussed in Chapter 1.

3.2 STUDY AREA

The two study farms were both situated in the upper catchment area of the Molenaars River, south-western Cape, South Africa (Figure 3.1). Both study rivers were steep, fast-flowing streams, shaded by a closed canopy of predominately indigenous riparian trees. The substratum consisted of cobbles and boulders, and a riffle-run-pool sequence of biotopes predominated.

"The portapool farm" was situated on the Kraalstroom River, a tributary of the Elands River, and used plastic-lined 'portapools' for housing the fish. The river upstream of the farm was undisturbed and, apart from the trout farm, no further impacts occurred on the Kraalstroom River before its confluence with the Elands River.

"The earthdam farm" was situated on the Molenaars River, upstream of its confluence with the Elands River, and used earthdams for housing the fish. It had minimal visible

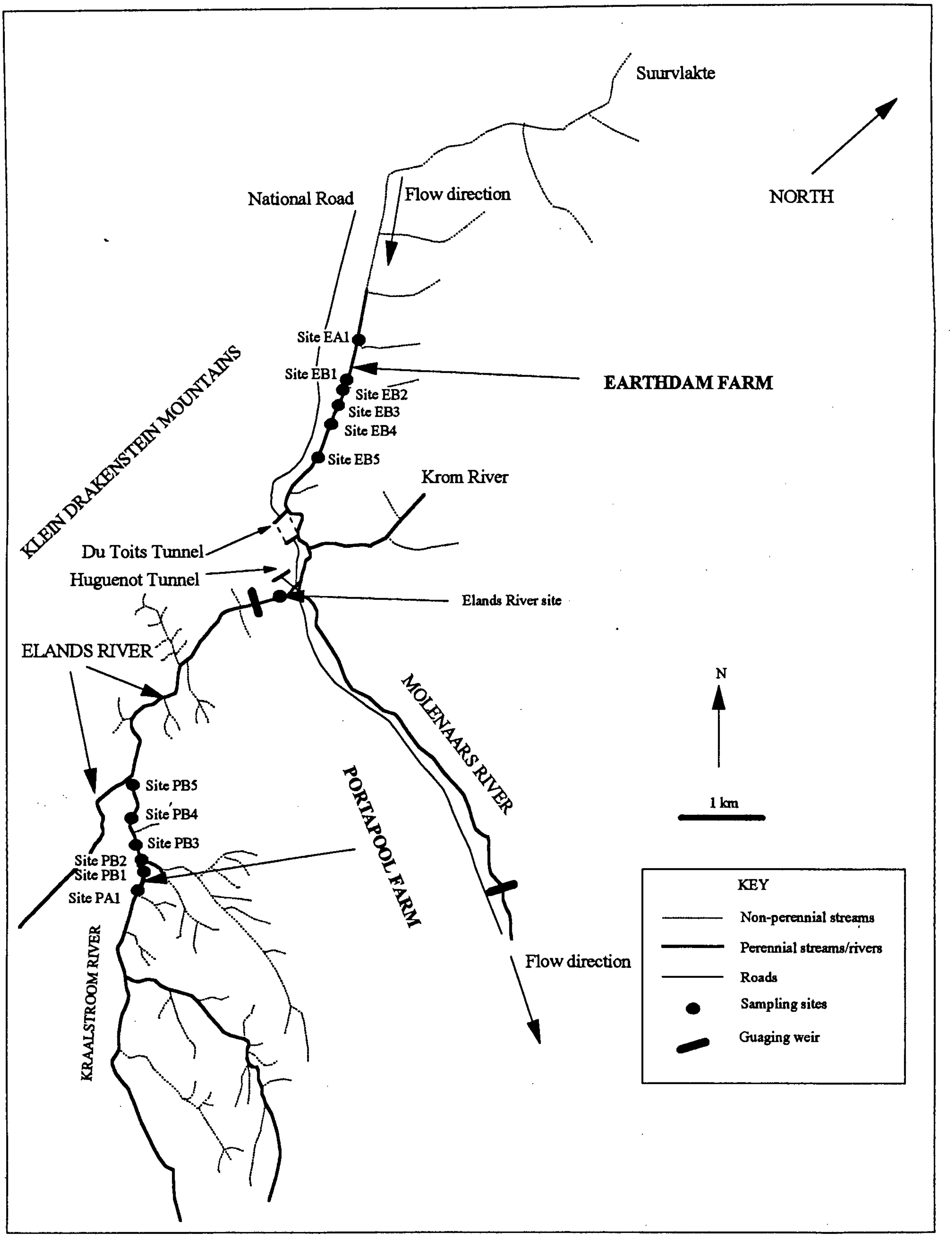

Figure 3.1 Locations of the earthdam farm and the portapool farm in the Molenaars River catchment, south-western Cape, South Africa

upstream disturbance, and was free of additional perturbations for the first kilometre downstream of the farm.

Six sites were chosen at each farm, namely:

• Site PA1, upstream of the portapool farm	Site EA1 - upstream of the earthdam farm
• Site PB1, at the effluent outlet	Site EB1 - at the effluent outlet
• Site PB2, 50 m downstream of the outlet	Site EB2 - 50 m downstream of the outlet
• Site PB3, 200 m downstream of the outlet	Site EB3 - 200 m downstream of the outlet
• Site PB4, 500 m downstream of the outlet	Site EB4 - 500 m downstream of the outlet
• Site PB5, 1000 m downstream of the outlet	Site EB5 - 1000 m downstream of the outlet.

It would have been ideal to include additional control sites in the programme, either a series of sites upstream of the farms with a similar spacing and habitat structure as the downstream sites or a series of control sites in a different river system, with similar habitat structure. However, once again financial and time constraints prohibited the collection of additional data at these sites. Where possible this short-coming has been addressed using data collected at similar sites during other studies. However it should be noted that the adopted design limited the ability to estimate variance or gradients in the control condition for direct or indirect comparison with the impact sites.

3.3 METHODS

3.3.1 Collection and laboratory analysis of physical and chemical variables

Based on the results of the pilot study (Chapter 2), the following physical and chemical variables were included in the main study: total dissolved solids (TDS), conductivity, total suspended solids (TSS), nitrate, ammonia, phosphate, dissolved oxygen and settled organic matter. Also included were nitrite, since the ratio of nitrite to nitrate is important in aquatic ecosystems, and temperature and pH, because many chemical reactions are dependent on those two variables.

Five replicate physical and chemical collections were made at each site on three separate occasions at the portapool farm, *viz*. August 1992, March 1993 and June 1993, and on two occasions at the earthdam farm, *viz*. March 1993 and June 1993.

During each sample trip, replicate samples were collected from each site on consecutive days (each replicate collection covered all the sites). This ensured that data were collected at a range of discharges. Originally, a different earthdam farm from the one presented here was selected for the detailed study, however, those sites were destroyed by bulldozing shortly after the start of the study, and a new farm had to be chosen. Hence, only two samples were collected at the earthdam farm. Additional data on oxygen levels and settled organic solids were also collected in November 1992 at the portapool farm.

The collection and analytical methods used for TDS, TSS, conductivity, nitrate, ammonia and phosphate and oxygen were the same as those detailed in Chapter 2. The methods for the collection and analysis of the settled organic material are presented in 3.3.4.

3.3.2 Numerical analysis of chemical and physical data

To recap, conductivity, temperature, pH, the concentrations of total dissolved solids (TDS), total suspended solids (TSS), nitrite, nitrate, ammonia, phosphate and dissolved oxygen and settled organic matter were recorded upstream and at various distances downstream of two trout farms. The data obtained, excluding the data for settled organic solids (see 3.3.4), were analysed using a variety of techniques. The questions that were asked were as follows:

1. how do the different sampling sites relate to one another with respect to their chemical and physical characteristics?
2. which variables correlated with one another?
3. which variables, if any, were significantly different in the river after, relative to before, the input of trout-farm effluent?
4. which variables had returned to their upstream concentrations by one kilometre downstream of the effluent outlet?

Hierarchical clustering and Principal Component Analysis (PCA) were used illustrate the relationships between the sampling sites associated with each farm with respect to their chemical and physical characteristics. The physical and chemical data obtained for the five replicates collected at each site during each sampling session were averaged and the multivariate techniques performed on a matrix of 10 variables by 18 samples and 12 samples for the portapool farm and earthdam farm, respectively (Table 3.1). The data were log-transformed, and analysed using Euclidean distance as a measure of dissimilarity (Clarke and Warwick 1994):

$$d_{jk} = \sqrt{\left[\sum_{i=1}^{p}\left(y_{ij} - y_{ik}\right)^2\right]}$$

where:
d_{jk} = direct distance between samples j and k.

Before calculating the Euclidean distance, the data for each variable were normalised. This was necessitated by the fact that the data were measured in a mix of units, for instance, temperature in °C, $[PO_4^+]$ in mg l^{-1} and conductivity in $\mu S\ m^{-1}$. Normalisation entails subtracting the mean count and dividing by the standard deviation of all the samples for that species (Clarke and Warwick 1994), and results in comparable (dimensionless) scales.

Table 3.1 The dates and sites of data collection at each of the study farms

	Portapool Farm						Earthdam Farm					
	PA1	PB1	PB2	PB3	PB4	PB5	EA1	EB1	EB2	EB3	EB4	EB5
Aug 92	*	*	*	*	*	*						
Feb/Mar 92	*	*	*	*	*	*	*	*	*	*	*	*
Jun 92	*	*	*	*	*	*	*	*	*	*	*	*

A correlation-based PCA was also performed (Clarke and Warwick 1994) on the normalised data, comparing the physical and chemical variables based on their concentrations in the samples to determine which of the variables correlated with one another (e.g. Clarke *et al.* 1996).

The physical and chemical data from each sampling session were combined are tested for differences upstream and downstream of each farm, and for downstream recovery to upstream conditions (e.g. Ward and Stanford 1983a, O'Keeffe *et al.* 1989a), using the sign test (Zar 1984), as follows:

- The immediate effects of the effluent were tested by comparing data from the site immediately upstream of the farm (Site A1) with those from immediately downstream of the effluent outlet (Site B1).
- Whether there were any significant downstream changes (i.e., if there was recovery) downstream of the farm was tested by comparing data from immediately downstream of the effluent outlet (Site B1) with those from the site 1 km downstream of the outlet (Site B5).
- Whether there was return to upstream conditions within the first kilometre was tested by comparing data from the site 1 km downstream of the outlet (Site B5) with those from the site immediately upstream of the farm (Site A1).

3.3.3. Calculation of discharge

Discharge (Q) was calculated from data obtained from the DWAF gauging weirs closest to each farm and corrected for the location of interest using differences in the size of the catchment areas (Gordon *et al.* 1992). For the portapool farm situated on the Kraalstroom River, the data were obtained from DWAF gauging weir H1H033, situated on the Elands River, except for the March discharge data which were not available for H1H033 and which were obtained from H1H018 on the Molenaars River. The discharge data for the earthdam farm were also obtained from H1H018, on the Molenaars River. The features of the gauging weirs are provided in Table 3.2.

Table 3.2 Location and catchment areas of the gauging weirs used during the detailed study (from DWAF 1990). S.F. = State Forest.

Gauging weir	River	Region	Catchment area	Latitude	Longitude
H1H033	Elands River	Hawequas S.F.	62.0 km^2	33° 44' 05"	19° 06' 54"
H1H018	Molenaars River	Hawequas S.F.	113.0 km^2	33° 43' 24"	19° 10' 13"

3.3.4. Analysis of settled organic material samples

In order to measure the amount of organic detritus that had settled on the river bed, three cross-channel transects, placed approximately 2 m apart, were established at each site. The material settled on the riverbed was collected at *c.* 1 m intervals along each transect. At each point, the area for collection was demarcated using a circular pipe of known diameter, which was placed on the river bed. The material which was settled on the river bed within the demarcated area was sucked up using a large syringe.

These samples were collected at the portapool farm in November 1992 and March 1993, and at the earthdam farm in March 1993. The collection of samples in the high flow season (June-August) was hampered by the fact that the river was subjected to unpredictable flood events, many of which flushed the riverbed of settled material. Consequently, samples were collected at the earthdam farm in June 1993, and at the portapool farm before and after a spate event in August 1992.

The samples were returned to the laboratory where they were dried at 60°C for 72 hours and weighed to determine the mass of solids in each sample. Each sample was then placed in a muffle furnace for eight hours at 450°C and re-weighed to determine the ratio of organic to inorganic solids.

The data used to answer the following questions:

1 is there settlement of organic material downstream of the effluent outlets and are there any differences in the amount of settled organic material downstream of the earthdam farm and the portapool farm?

2 if there is settlement, whereabouts in the channel does this occur and at what velocities?

3 is the settled organic material flushed from the system during winter spates, and if so, how often does this flushing occur?

3.3.5 Hydraulic conditions

Flow measurements were taken at each point on the cross-channel transects where settled material was collected. Flow velocities were recorded at 6/10 depth in the water column, which is recognised as mean column velocity (BS 3680 1980).

Froude numbers were used to represent the hydraulic conditions at each point on the transect. Froude numbers are dimensionless numbers that represent the ratio of inertial to gravitational forces, where gravitational forces encourage water to move downhill whereas the inertial forces reflect the water's compulsion to proceed or not (Gordon *et al.* 1992). They offer a more appropriate description of the interaction between flow and depth at a given point than do the better known Reynolds numbers (Reynolds 1883).

The Froude number (*Fr*) was calculated using the following equation (Gordon *et al.* 1992):

$$Fr = \frac{V}{\sqrt{gD}}$$

where: V = mean velocity
g = acceleration due to gravity
D = hydraulic depth.

Substratum composition at each site was measured according to the percentage cover of sand (*c.* 1-5 mm diameter particle size), gravel (*c.* 5-75 mm), cobble (*c.* 75-600 mm), boulder (>600 mm) and bedrock ('sheets' of rock), after Bovee (1982). A 0.25 m^2 metal grid, subdivided into 36 squares, was placed randomly on the river bed and used to estimate the relative proportions for each substratum type. Estimates were made for each square and then summed to produce an estimate of percentage cover. Three replicate sets of measurements were taken at each river site.

Differences in the hydraulic conditions (represented by Froude numbers) between sites, and between the two farms, were tested for using an analysis of variance (ANOVA). Differences in mean particle size of the substratum at each of the sites and between the farms were also tested using ANOVA.

3.4 RESULTS

3.4.1 Multivariate Analyses

The data matrices for the portapool farm and earthdam farm to which the multivariate techniques were applied are presented in Table 3.3 and Table 3.4, respectively, and the resultant plots in Figures 3.2 to 3.5.

Figure 3.2 depicts the results of hierarchical clustering, using normalised Euclidean distance, based on physical and chemical variables collected upstream and at various distances downstream of the portapool farm. The samples collected downstream of the farm, PB1-PB5, grouped according to the season in which they were collected. The samples collected upstream of the sites (PA1) grouped together. This pattern is most clearly seen in Figure 3.3, which shows the results of the PCA analysis. In Figure 3.3, the PA1 sites are scattered in the top right-hand corner of the plot, and the samples collected downstream of the farms grouped together, according to the month in which they were collected.

Figures 3.4 and 3.5 depict the results of the multivariate analyses of the data collected at the earthdam farm. In Figure 3.4, the sites grouped according to the season in which they were collected. In the PCA graph (Figure 3.5), the samples collected in June grouped closely together. However, the samples collected in March displayed a trend similar to those from the portapool farm (Figure 3.5); the sample collected upstream of the farm (EA1) did not group with those collected downstream of the farm.

The PCA graph based on samples of the physical and chemical variables collected at the portapool farm is presented in Figure 3.6.

Table 3.3 Physical and chemical data recorded in riffle biotopes at Sites A1, B1-B5 and in the effluents at the portapool farm during the high-flow (August 1992 and June 1993) and low-flow (November and February 1993) sampling sessions. The averages are of the five replicate collections done during each sampling session

Site	Temp. (°C)		O_2 % sat		pH		Cond. $\mu S\ cm^{-1}$		TDS $mg\ l^{-1}$		TSS $mg\ l^{-1}$		NO_3^--N $mg\ l^{-1}$		NO_2-N $mg\ l^{-1}$		NH_4^+-N $mg\ l^{-1}$		PO_4^{3-}-P $mg\ l^{-1}$	
August	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range
A1			89.7	79.6 - 100	5.9	5.6 - 6.4	19.6	18.8 - 20.4	0.016	0.009 - 0.021	0.82	0.56 - 1.60	0.009	0.004 - 0.013	0.001	0.000 - 0.002	0.029	0.012 - 0.066	0.010	0.008 - 0.014
Effluent			69.7	52.2 - 84.4	6.0		23.5	21.4 - 25.9	0.018	0.004 - 0.028	2.77	1.92 - 3.04	0.066	0.056 - 0.073	0.002	0.000 - 0.003	0.185	0.161 - 0.195	0.057	0.040 - 0.072
B1			81.4	72.5 - 89.9	5.8	5.3 - 6.2	23.7	20.2 - 28.8	0.027	0.014 - 0.028	3.23	1.52 - 5.84	0.061	0.040 - 0.082	0.002	0.000 - 0.003	0.171	0.140 - 0.240	0.047	0.032 - 0.066
B2			85.3	79.1 - 100	5.9	5.3 - 6.2	21.9	19.3 - 24.8	0.020	0.015 - 0.024	1.71	1.12 - 2.24	0.048	0.025 - 0.075	0.002	0.000 - 0.003	0.141	0.058 - 0.210	0.038	0.018 - 0.055
B3			88.7	73.4 - 100	5.6	5.2 - 6.1	21.2	19.1 - 25.5	0.021	0.016 - 0.027	2.06	1.44 - 2.56	0.069	0.040 - 0.103	0.002	0.000 - 0.003	0.107	0.058 - 0.155	0.036	0.016 - 0.066
B4			88.0	80.7 - 100	5.5	5.1 - 6.1	20.84	19.2 - 24.7	0.021	0.006 - 0.032	2.32	1.36 - 3.36	0.075	0.026 - 0.127	0.002	0.000 - 0.003	0.105	0.076 - 0.155	0.042	0.015 - 0.063
B5			83.9	69.6 - 98.3	5.6	5.0 - 6.1	21.4	18.9 - 24.2	0.022	0.006 - 0.040	1.74	0.96 - 2.56	0.073	0.043 - 0.109	0.003	0.001 - 0.005	0.123	0.073 - 0.197	0.034	0.015 - 0.061
June	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range
A1	8.7	8.0 - 9.0	97.3	95.0 - 100	6.0	6.0 - 6.1	18.4	17.3 - 18.9	0.016	0.007 - 0.024	1.12	0.6 - 1.3	0.023	0.009 - 0.068	0.002	0.001 - 0.003	0.020	0.013 - 0.032	0.003	0.002 - 0.006
Effluent	8.8	8.0 - 9.0	95.6	91.0 - 100	5.8	5.7 - 5.9	23.1	22.8 - 24.1	0.022	0.018 - 0.027	4.08	3.5 - 5.4	0.076	0.039 - 0.142	0.002	0.001 - 0.002	0.186	0.150 - 0.205	0.061	0.042 - 0.079
B1	8.8	8.0 - 9.0	96.6	94.0 - 100	6.0	6.0 - 6.0	21.7	21.0 - 22.9	0.018	0.012 - 0.029	2.98	2.1 - 4.7	0.051	0.036 - 0.060	0.002	0.001 - 0.002	0.157	0.112 - 0.231	0.037	0.023 - 0.052
B2	8.7	8.0 - 9.0	95.6	92.0 - 100	6.0	5.9 - 6.1	20.6	19.7 - 21.8	0.023	0.018 - 0.037	2.40	1.7 - 4.0	0.041	0.034 - 0.049	0.002	0.001 - 0.002	0.086	0.015 - 0.133	0.028	0.020 - 0.038
B3	8.4	8.0 - 9.0	95.0	92.0 - 100	6.2	6.1 - 6.3	19.4	15.9 - 20.6	0.023	0.014 - 0.032	2.02	1.5 - 2.6	0.057	0.047 - 0.057	0.002	0.001 - 0.002	0.106	0.065 - 0.192	0.027	0.016 - 0.044
B4	8.2	7.5 - 9.0	93.2	90.0 - 100	6.3	6.1 - 6.5	20.0	18.7 - 22.0	0.017	0.007 - 0.029	1.86	1.1 - 2.5	0.072	0.060 - 0.084	0.004	0.002 - 0.007	0.097	0.059 - 0.139	0.023	0.018 - 0.031
B5	7.7	6.8 - 9.0	90.6	84.0 - 100	6.2	5.9 - 6.5	18.9	14.7 - 21.3	0.021	0.015 - 0.028	1.88	1.6 - 2.2	0.062	0.024 - 0.084	0.002	0.001 - 0.002	0.082	0.048 - 0.101	0.027	0.012 - 0.055
November	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range
A1	13.6	12.5 - 15.0	81.9	73.5 - 87.8	6.2	5.8 - 6.6	20.9	16.5 - 24.1	0.018	0.013 - 0.020	0.85	0.25 - 1.73	0.006	0.004 - 0.006	0.002	0.001 - 0.003	0.032	0.013 - 0.038	0.007	0.001 - 0.020
Effluent	13.0	12.0 - 14.0	61.5	53.9 - 70.6	6.0	6.0 - 6.0	29.8	27.9 - 30.6	0.029	0.027 - 0.033	4.65	3.00 - 5.87	0.074	0.070 - 0.080	0.004	0.003 - 0.005	0.201	0.188 - 0.216	0.053	0.037 - 0.078
B1	12.9	12.0 - 14.0	74.9	72.1 - 76.5	6.1	6.0 - 6.2	28.2	24.7 - 30.4	0.038	0.038 - 0.039	5.33	2.17 - 7.83	0.070	0.063 - 0.075	0.003	0.002 - 0.004	0.186	0.179 - 0.190	0.078	0.031 - 0.120
B2	12.7	11.8 - 13.6	80.7	74.0 - 87.3	6.1	5.9 - 6.3	26.6	21.7 - 30.2	0.029	0.011 - 0.055	3.09	1.75 - 4.75	0.064	0.060 - 0.066	0.003	0.002 - 0.004	0.170	0.162 - 0.181	0.042	0.030 - 0.050
B3	12.5	12.0 - 13.1	83.5	79.4 - 86.3	6.2	6.1 - 6.3	25.2	22.3 - 29.3	0.036	0.020 - 0.072	3.33	1.08 - 7.20	0.076	0.073 - 0.078	0.004	0.003 - 0.005	0.141	0.113 - 0.159	0.046	0.031 - 0.069
B4	12.6	12.0 - 13.0	79.9	77.5 - 83.7	6.2	6.2 - 6.4	25.1	23.0 - 27.3	0.039	0.025 - 0.055	3.96	1.67 - 7.33	0.093	0.080 - 0.101	0.005	0.004 - 0.006	0.137	0.128 - 0.142	0.056	0.024 - 0.117
B5	12.4	12.0 - 13.0	84.2	76.5 - 88.5	6.3	6.1 - 6.3	24.4	20.0 - 27.9	0.028	0.019 - 0.038	2.08	1.08 - 3.07	0.092	0.058 - 0.113	0.005	0.004 - 0.006	0.128	0.110 - 0.149	0.046	0.043 - 0.058
February	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range
A1	17.3	17.0 - 18.0	91.6	16.7 - 17.5	6.4	6.2 - 6.5	33.8	27.0 - 42.2	0.020	0.016 - 0.026	0.80	0.00 - 1.10	0.032	0.008 - 0.114	0.003	0.001 - 0.008	0.116	0.044 - 0.240	0.001	0.001 - 0.003
Effluent	17.6	17.0 - 18.0	72.6	16.8 - 17.9	6.1	6.0 - 6.2	47.6	38.9 - 55.0	0.037	0.023 - 0.069	5.15	3.80 - 6.67	0.105	0.068 - 0.137	0.008	0.006 - 0.012	0.284	0.248 - 0.298	0.118	0.080 - 0.078
B1	17.6	17.0 - 18.0	89.0	16.7 - 17.9	6.2	6.1 - 6.3	46.1	38.9 - 53.6	0.091	0.020 - 0.047	4.83	2.80 - 6.80	0.114	0.077 - 0.134	0.012	0.010 - 0.015	0.285	0.278 - 0.298	0.136	0.107 - 0.187
B2	17.6	17.0 - 18.0	92.8	16.5 - 17.2	6.2	6.1 - 6.3	46.1	38.6 - 52.9	0.024	0.019 - 0.028	5.44	4.13 - 6.80	0.124	0.103 - 0.137	0.012	0.011 - 0.014	0.286	0.269 - 0.296	0.121	0.077 - 0.168
B3	17.3	17.0 - 18.0	90.8	86.0 - 96.0	6.3	6.2 - 6.5	43.3	35.0 - 50.0	0.027	0.020 - 0.031	3.72	2.40 - 4.60	0.168	0.164 - 0.171	0.020	0.018 - 0.022	0.275	0.252 - 0.292	0.112	0.103 - 0.164
B4	17.2	17.0 - 18.0	91.2	84.0 - 97.0	6.3	6.2 - 6.5	41.2	33.0 - 47.9	0.024	0.017 - 0.033	2.30	1.30 - 3.70	0.285	0.165 - 0.518	0.026	0.018 - 0.034	0.257	0.215 - 0.262	0.112	0.058 - 0.167
B5	17.2	17.0 - 18.0	88.8	79.0 - 97.0	6.3	6.1 - 6.5	40.3	35.1 - 44.3	0.028	0.025 - 0.031	1.42	1.00 - 1.90	0.465	0.360 - 0.571	0.028	0.027 - 0.034	0.220	0.188 - 0.254	0.087	0.001 - 0.139

Table 3.4 Physical and chemical data recorded in riffle biotopes at Sites A1, B1-B5 and in the effluents at the earthdam farm during the high-flow and low-flow sampling sessions (June 1993 and March 1993, respectively). The data for June 1993 are averages of five replicate collections done during that sampling session. The data for March 1993 are single spot-samples.

Site	Temp. (°C)		O_3 % sat		pH		Cond. µS cm^{-1}		TDS mg l^{-1}		TSS mg l^{-1}		NO_3^--N mg l^{-1}		NO_2-N mg l^{-1}		NH_4^+-N mg l^{-1}		PO_4^{3-}-P mg l^{-1}	
June	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range
A1	7.8	7.0 - 9.0	98.0	94.0 - 100	4.9	4.8 - 5.1	15.8	15.3 - 16.8	0.018	0.017 - 0.019	2.75	2.5 - 3.0	0.003	0.001 - 0.007	0.002	0.001 - 0.004	0.05	0.027 - 0.081	0.000	0.000-0.000
Effluent	9.4	none	68.0	38.0 - 85.0	5.5	5.1 - 5.7	27.0	23.4 - 29.5	0.033	0.029 - 0.041	6.23	4.3 - 6.4	0.175	0.135 - 0.240	0.003	0.002 - 0.003	0.214	0.198 - 0.232	0.064	0.047 - 0.078
B1	7.7	7.0 - 9.0	96.7	94.0 - 100	5.1	4.9 - 5.5	15.7	14.2 - 18.0	0.021	0.019 - 0.022	3.30	2.9 - 3.7	0.028	0.018 - 0.043	0.003	0.002 - 0.005	0.049	0.035 - 0.070	0.007	0.001 - 0.012
B2	7.6	6.8 - 9.0	96.7	94.0 - 100	5.3	5.1 - 5.5	15.3	12.6 - 17.6	0.025	0.021 - 0.029	2.93	1.6 - 3.7	0.031	0.020 - 0.044	0.003	0.002 - 0.004	0.050	0.043 - 0.057	0.010	0.003 - 0.023
B3	7.7	7.0 - 9.0	97.0	94.0 - 100	5.3	5.1 - 5.6	16.1	14.6 - 17.9	0.022	0.021 - 0.024	3.43	3.4 - 3.5	0.029	0.026 - 0.034	0.003	0.002 - 0.005	0.065	0.051 - 0.072	0.005	0.003 - 0.008
B4	8.3	8.0 - 9.0	93.3	91.0 - 95.0	5.3	5.1 - 5.7	16.4	14.9 - 17.3	0.017	0.016 - 0.019	3.07	2.7 - 3.3	0.037	0.029 - 0.044	0.002	0.002 - 0.003	0.068	0.051 - 0.087	0.005	0.003 - 0.007
B5	8.3	8.0 - 8.5	96.0	91.0 - 100	5.4	5.3 - 5.7	16.0	10.0 - 20.7	0.020	0.018 - 0.025	3.27	2.6 - 3.7	0.030	0.015 - 0.046	0.003	0.002 - 0.006	0.082	0.043 - 0.143	0.006	0.001 - 0.013
March	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range
A1	14.0	none	61.0	none	5.6	none	18.5	none	0.020	none	0.80	none	0.028	none	0.004	none	0.171	none	0.005	none
Effluent	none	none	none	none	none	none	none	none	none	none		none	none	none	none	none	none	none	none	none
B1	15.5	none	57.0	none	6.5	none	24.3	none	0.027	none	1.40	none	0.101	none	0.004	none	0.064	none	0.001	none
B2	15.9	none	99.3	none	6.1	none	31.9	none	0.032	none	1.35	none	0.124	none	0.004	none	0.043	none	0.001	none
B3	15.0	none	100.0	none	6.2	none	31.4	none	0.026	none	3.45	none	0.129	none	0.003	none	0.065	none	0.001	none
B4	13.5	none	97.3	none	6.3	none	26.9	none	0.029	none	2.88	none	0.129	none	0.004	none	0.096	none	0.006	none
B5	18.0	none	94.0	none	6.3	none	37.9	none	0.021	none	1.652	none	0.135	none	0.004	none	0.123	none	none	none

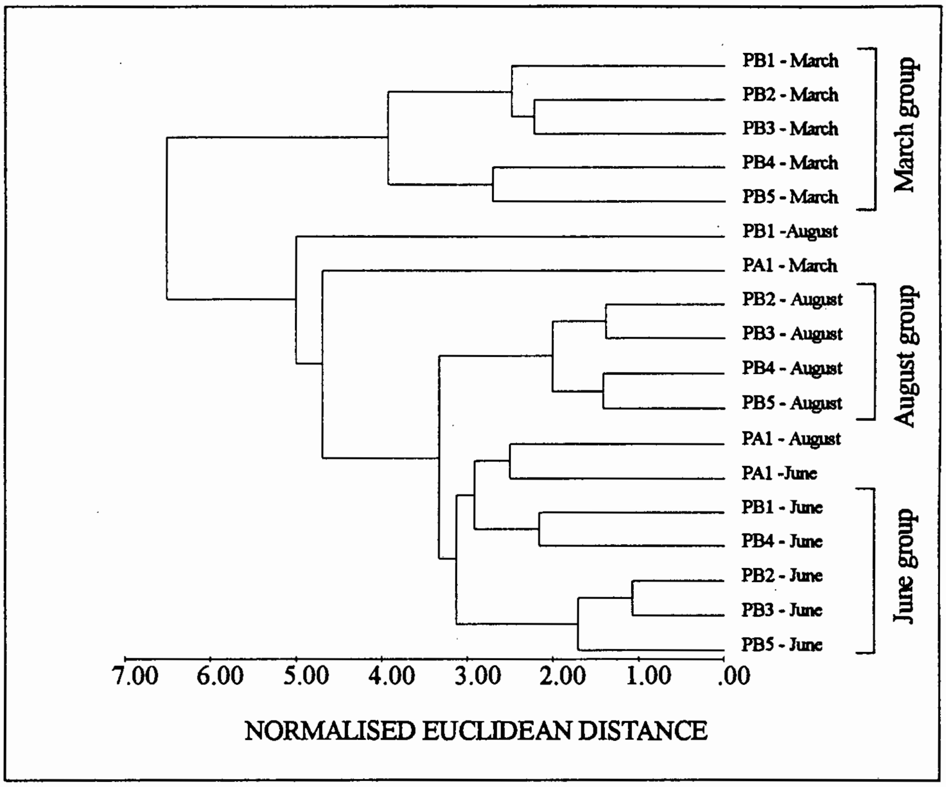

Figure 3.2 Dendrogram depicting the results of the hierarchical clustering, using normalised Euclidean distance, based on the physical and chemical measurements obtained from the portapool farm sampling sites PA1 - PB5 during March, June and August 1993

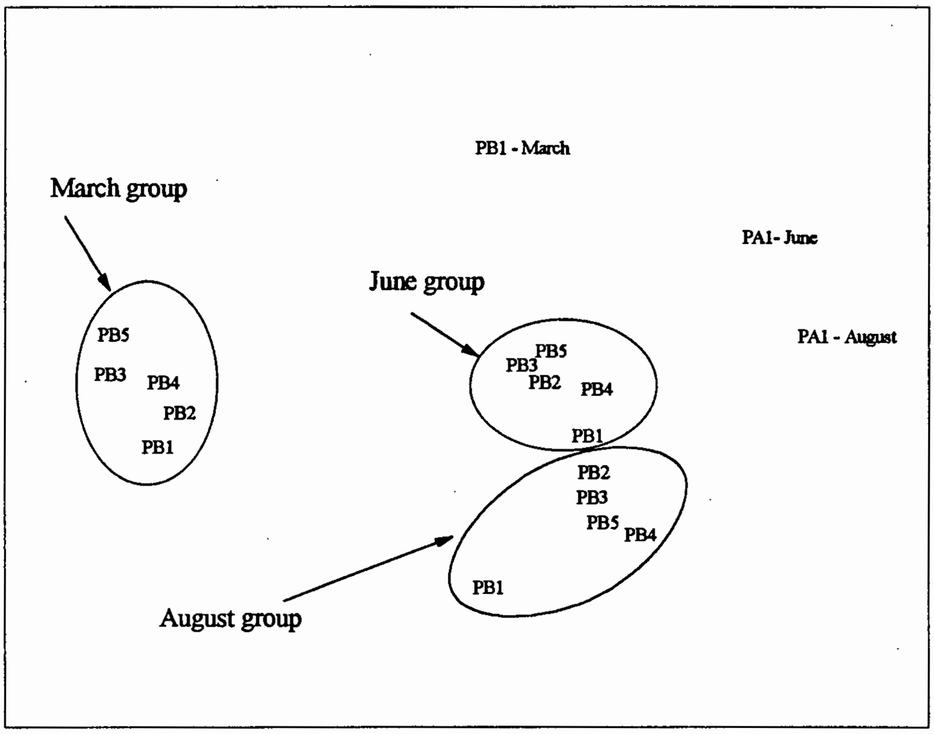

Figure 3.3 Two-dimensional graph depicting the results of the Principal Components Analysis (PCA), using Normalised Euclidean Distance conducted on the physical and chemical measurements made at the portapool farm sampling sites PA1 - PB5 during March, June and August 1993

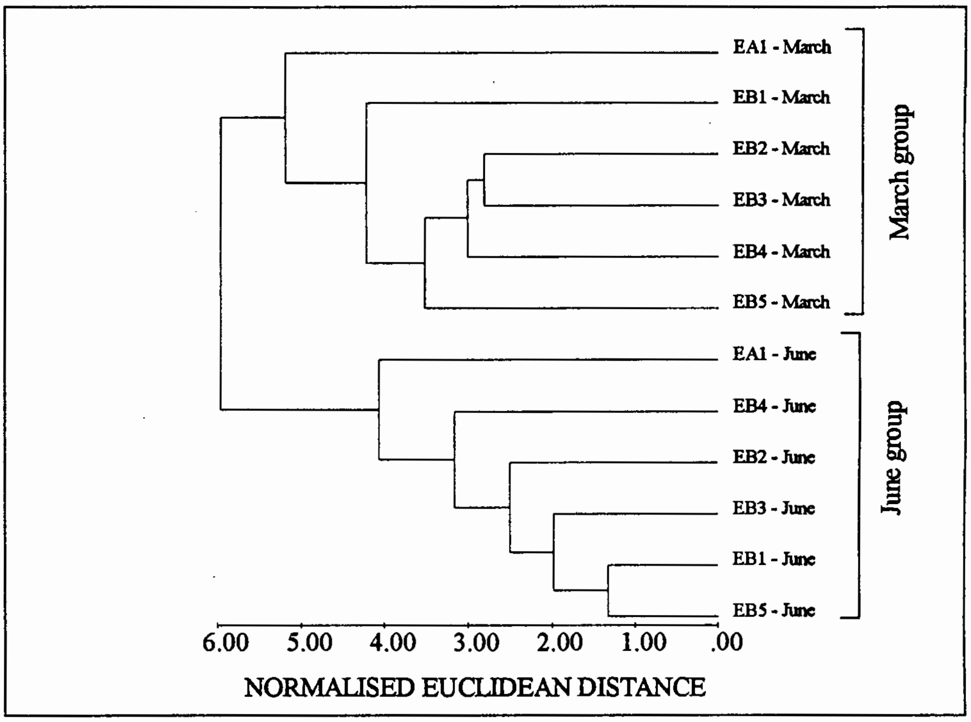

Figure 3.4 Dendrogram depicting the results of the hierarchical clustering, using normalised Euclidean distance, based on the physical and chemical measurements obtained from the earthdam farm sampling sites EA1 - EB5 during March and June

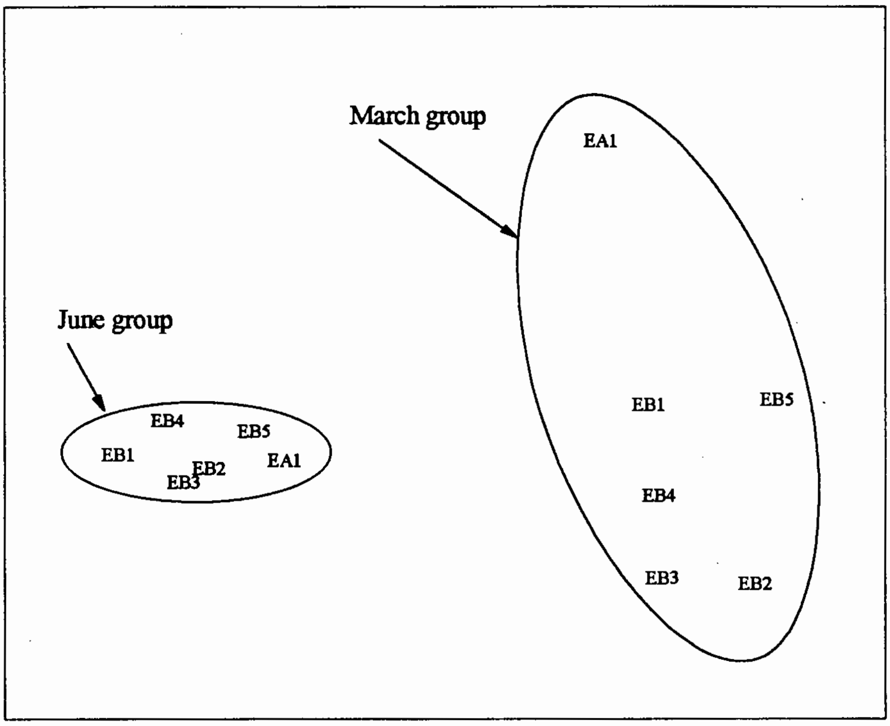

Figure 3.5 Two-dimensional graph depicting the results of the Principal Components Analysis (PCA), using Normalised Euclidean Distance conducted on the physical and chemical measurements made at the earthdam farm sampling sites EA1 - EB5 during March and June 1993

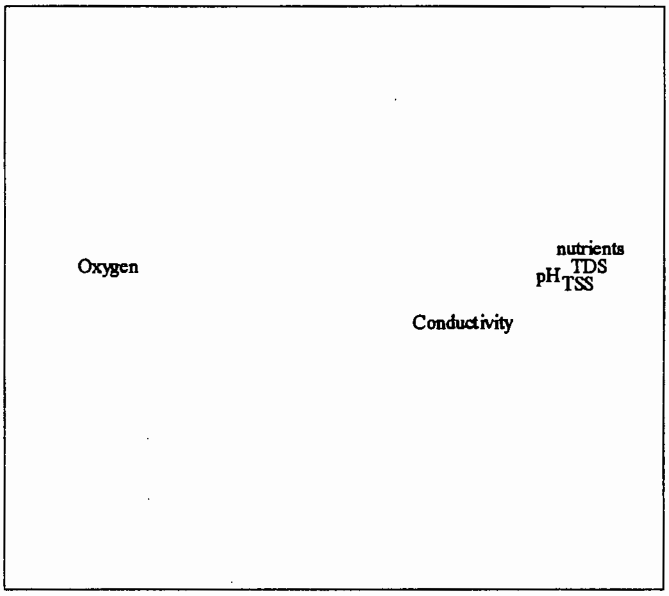

Figure 3.6 PCA graph based on samples of the physical and chemical variables collected at the portapool farm

From Figure 3.6 it is clear that there is a considerable amount of correlation between the chemical variables. This is chiefly a result of the fact that the levels of almost all the variables increased downstream of the trout farm relative to their upstream levels.

3.4.2 Univariate analyses

Temperature, pH, conductivity levels and the concentrations of total suspended solids recorded upstream and downstream of the two study farms in the high-flow (June-August) and low-flow (November-April) seasons are presented in Figure 3.7a-d. Similarly, the concentrations of nitrite (NO_2^--N), nitrate (NO_3^--N), ammonium (NH_4^+-N) and phosphate (PO_4^{3-}-P) are depicted in Figure 3.7a-d.

Table 3.5 presents the results of the paired-sample sign tests. For any one variable, a significant difference between Site A1 and Site B1 indicates that the effluent had a significant effect on that variable, and a significant difference between Site B1 and B5 indicates that some downstream 'recovery' to upstream conditions had taken place. A significant difference between Site A1 and Site B5 indicates that the levels of the variable 1 km below the effluent had not returned to those recorded upstream of the farm.

Neither farm had any significant effect on temperatures (Figure 3.7a) in the downstream river and, although pH increased downstream of the earthdam farm, the increases were small: between 0.25 and 0.5 pH units (Figure 3.7b), and not significant. Increases in conductivity, at both farms were more noticeable during the summer, when the flow in their receiving rivers was low (Figure 3.7c) but were only significant at the portapool farm. Total suspended solids (Figure 3.7d) increased significantly downstream of both farms and, in the case of the portapool farm, did not return to upstream conditions by Site B5, 1000 m downstream of the effluent outlets.

[NO_2^--N] (Figure 3.8a) was elevated downstream of the earthdam farm and, to a lesser extent, the portapool farm. The concentrations measured at the site 1000 m (Site PB5) downstream of the portapool farm approximated those recorded upstream of the farm (Site PA1). [NO_3^--N] (Figure 3.8b) increased in the river immediately downstream of the effluent outlet and then continued to increase steadily, but not significantly, from PB1 to PB5. [NH_4^+-N] and [PO_4^{3-}-P] increased in the river immediately downstream of the portapool effluent outlet, and then decreased steadily with increasing distance down the river (Figures 3.8 c&d, respectively).

3.4.3 Accumulated organic debris

Settlement of organic material downstream of the study farms

The measured amounts of organic solids accumulated on the river bed downstream of the portapool and earthdam farms are depicted in Figure 3.9 and 3.10, respectively. The accumulation of organic material on the river bed downstream of the portapool farm (Figure 3.9) was far greater than that downstream of the earthdam farm (Figure 3.10; $t_{0.05(1),4} = 1.935$). Deposition upstream of both farms, at Site A1, was negligible, even in the slow-flowing areas. Downstream of the portapool farm (Figure 3.9), deposition was greatest at Sites PB1 and PB2, but some deposition was evident as far downstream as Site PB5, 1 km downstream of the effluent outlet. Site PB2, in particular, had a deep, slow-flowing area where a considerable amount of organic material was trapped.

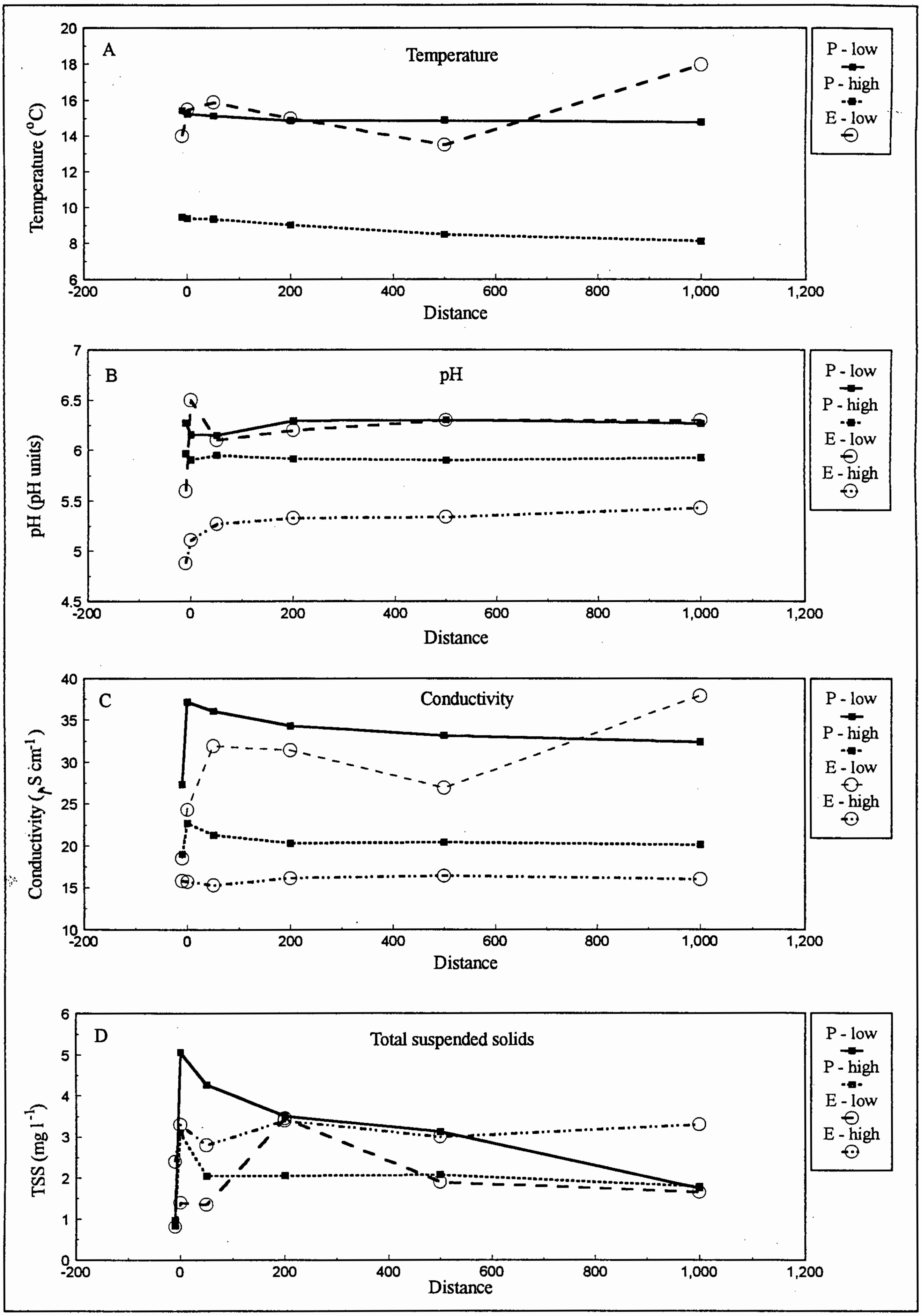

Figure 3.7 (a) Water temperature, (b) pH, (c) Conductivity and (d) Total suspended solid measurements recorded upstream (-10 m) and various distances (0 m, 50 m, 200 m, 500 m and 1000 m) downstream of the portapool and earthdam farms in the low-flow (summer) and high-flow (winter) seasons

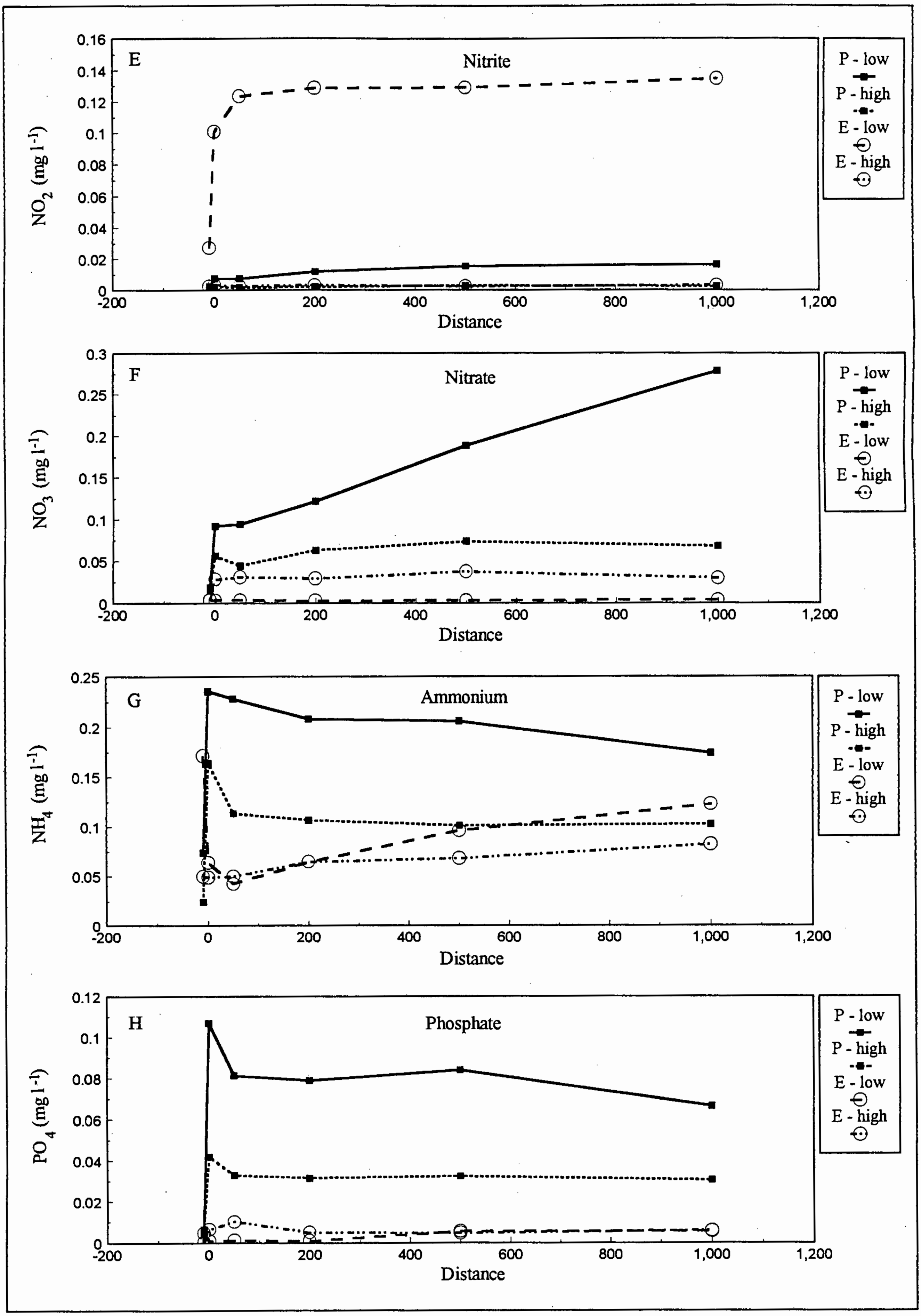

Figure 3.8 (a) Nitrite, (b) Nitrate, (c) Ammonium and (d) Phosphate concentrations recorded upstream (-10 m) and various distances (0 m, 50 m, 200 m, 500 m and 1000 m) downstream of the portapool and earthdam farms in the low-flow (summer) and high-flow (winter) seasons

Table 3.5 Results of the Sign paired-sample tests. Because a number of tests of the same hypothesis are being performed, there is a high comparison-wise error. Hence a Bonferroni adjustment of α was performed, *viz.* α/no. variables tested (10) = 0.005 = p. For Sites A1 vs. B1, and Sites A1 vs. B5, P $\leq$ 0.005 indicates that the measurements taken at Site A1 were statistically significantly ***lower*** than those at Sites B1 or B5. For Sites B1 vs. B5, P $\leq$ 0.005 indicates that the measurements taken at Site B1 were statistically significantly ***higher*** than those at Site B5. The cells containing significant values are shaded.

Variable	A1 vs. B1 immediate effect if significant		B1 vs. B5 Downstream recovery if significant		A1 vs. B5 recovery to upstream conditions if not significant	
	Portapool PA1vs. PB1 (n = *c.* 18)	Earthdam EA1 vs. EB1 (n = *c.* 4)	Portapool PB1 vs. PB5 (n = *c.* 18)	Earthdam EB1 vs. EB5 (n = *c.* 4)	Portapool PA1 vs.PB5 (n = *c.* 18)	Earthdam EA1 vs. EB5 (n = *c.* 4)
Oxygen (in riffles)	P = 0.2851	P = 0.6875	P = 0.0309	P = 0.6875	P = 0.4544	P = 0.6875
Temperature	P = 0.9999	P = 1.8750	P = 0.9525	P = 0.6250	P = 0.8816	P = 0.6250
pH	P = 0.0108	P = 0.2187	P = 0.1670	P = 0.2187	P = 1.0000	P = 0.0312
Conductivity	P = 0.0001	P = 0.3437	P = 0.0001	P = 0.2187	P = 0.0044	P = 0.2187
TDS	P = 0.03	P = 1.0000	P = 1.0000	P = 1.0000	P = 0.03	P = 0.0000
TSS	P = 0.0000	P = 0.0004	P = 0.0000	P = 1.0000	P = 0.0000	P = 0.0000
Nitrite	P = 0.0256	P = 0.6250	P = 0.9999	P = 1.8750	P = 0.6476	P = 0.6250
Nitrate	P = 0.0000	P = 0.1250	P = 0.0023	P = 0.6250	P = 0.0007	P = 0.1250
Phosphate	P = 0.0000	P = 0.6250	P = 0.0308	P = 1.8750	P = 0.0001	P = 0.2500
Ammonium	P = 0.0004	P = 0.6250	P = 0.0212	P = 0.1250	P = 0.0007	P = 0.6250

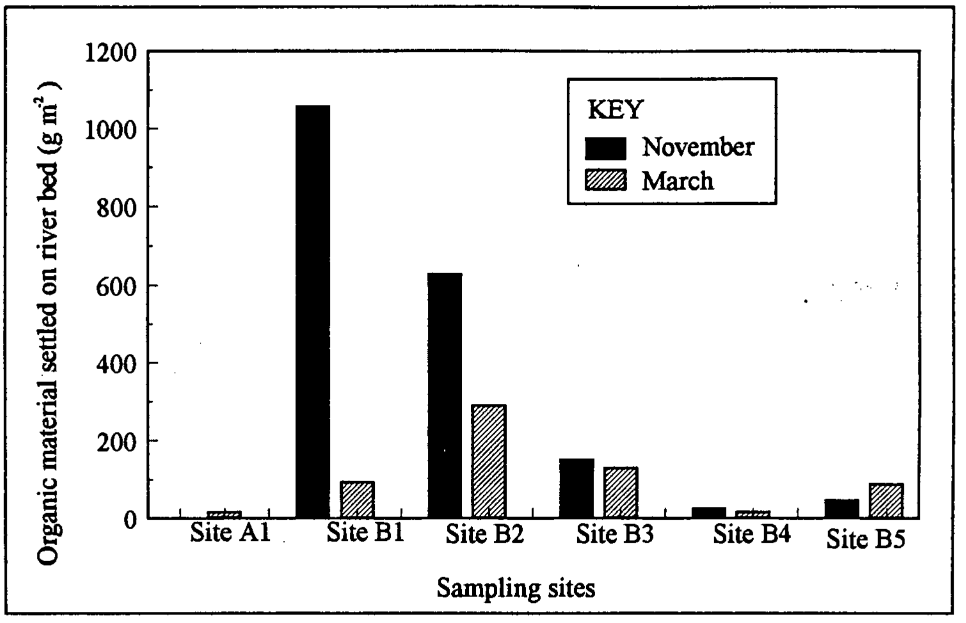

Figure 3.9 Average dry weight of organic material settled on the surface of the river bed upstream and downstream of the portapool farm in November 1992 and March 1993

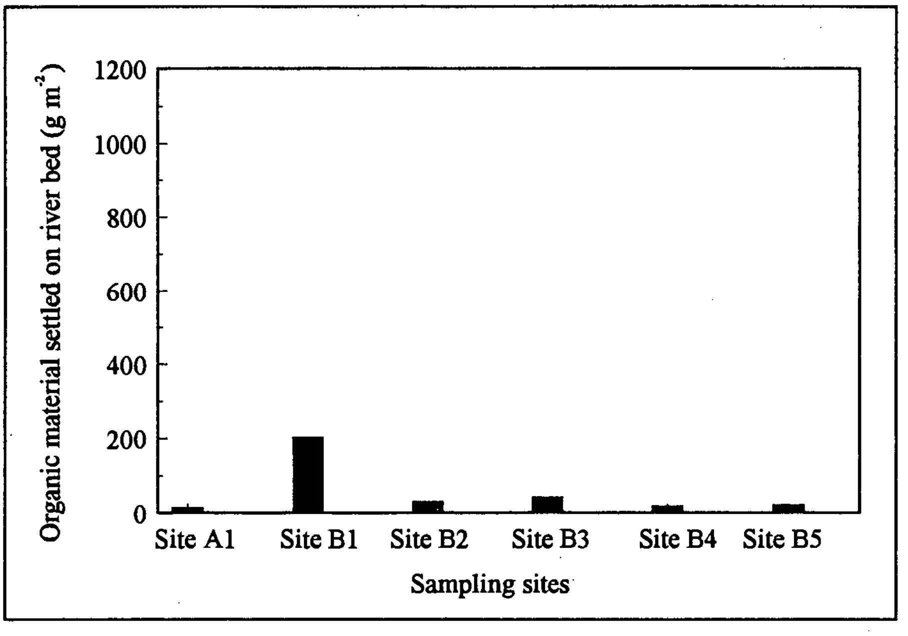

Figure 3.10 Average dry weight of organic material settled on the surface of the river bed upstream and downstream of the earthdam farm in November 1993. There were no measurable amounts of organic material settled on the riverbed upstream or downstream of the farm in June 1993

Spatial distribution of settled material

The spatial distributions of the settled material in relation to the velocities at the sampling sites at the portapool farm in the low flow season (summer) are shown in Figure 3.11 (March) and Figure 3.12 (November). The organic material tended to settle in the slow-flowing areas, particularly in the deep backwaters where the settled organic material was up to a metre deep.

Figure 3.13 depicts relationship between velocities of the water and the dry weight of organic material settled on the river bed at the portapool sites, PB1 and PB2, under base-flow conditions. Downstream of the portapool farm, over 40% of the river bed was covered with some organic debris that originated from the farm. Twenty to twenty-two percent of the riverbed had heavy deposits (mean = > 200 g m^{-2} dry weight) of organic material, mostly in the slow-flowing areas. The greatest settlement occurred at velocities less than 0.04 m s^{-1}, although a fair amount of deposition (up to 5 g m^{-2} dry weight of organic material) was recorded at velocities between 0.04 m s^{-1} and 0.08 m s^{-1}. There were still noticeable deposits measured at velocities up to 0.14 m s^{-1}, and occasionally up to 0.28 m s^{-1}. As the transects were evenly spaced across the width of the river, the results obtained are probably a fair approximation of what was occurring in the river channel as a whole.

Flushing of settled organic material

Measurements of the amount of organic matter that settled on the surface of the river bed at Site PB1 and Site PB5 before and after the spate which occurred in the study river in August 1992 are provided in Figure 3.14. The pre-spate data were collected 37 days after the last spate in the river, which meant that the material had been accumulating on the river bed since then.

Figure 3.14 shows that the organic material settled on the river bed was completely flushed during the spate. The magnitude of the spate was measured as between 10.4 - 24.2 m^3 s^{-1} at H1H033. Corrected for the study river, the Kraalstroom, these measurements yielded discharges of between 2.0 and 4.1 m^3 s^{-1}. Thus, a spate with a magnitude between 2.0 and 4.1 m^3 s^{-1} is sufficiently strong to flush the settled organic

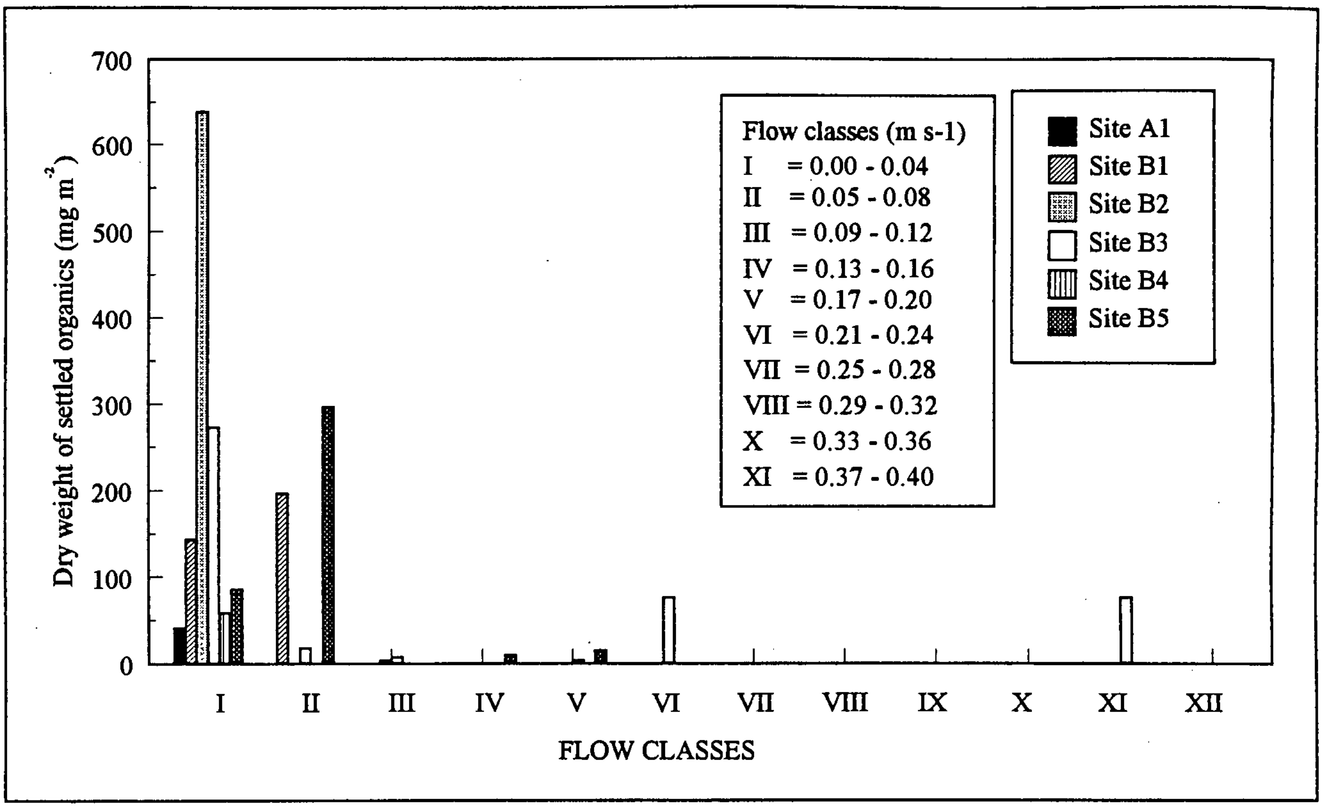

Figure 3.11 Dry weight of organic solids collected from the riverbed upstream (Site A1) and downstream (Sites B1 - B5) of the portapool farm during the March sampling session

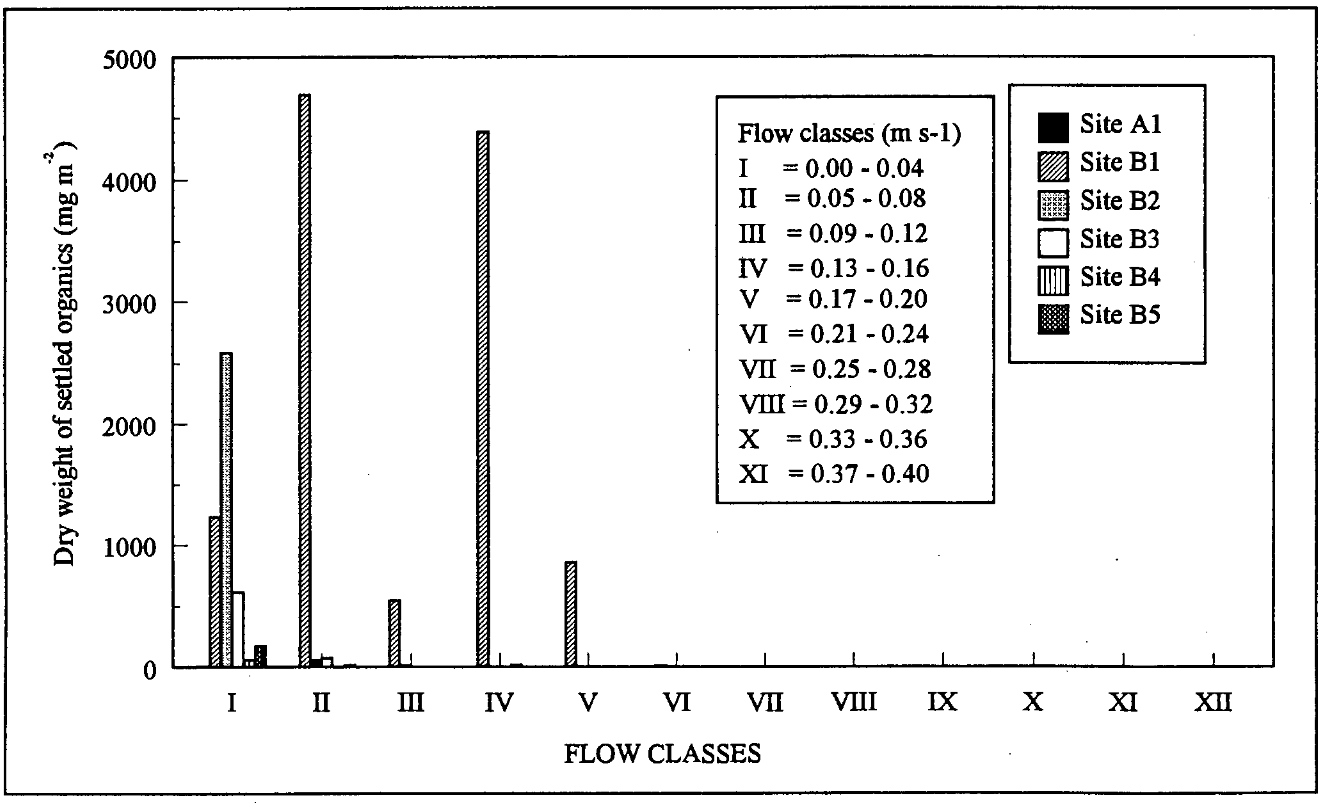

Figure 3.12 Dry weight of organic solids collected from the riverbed upstream (Site A1) and downstream (Sites B1- B5) of the portapool farm during the November sampling session

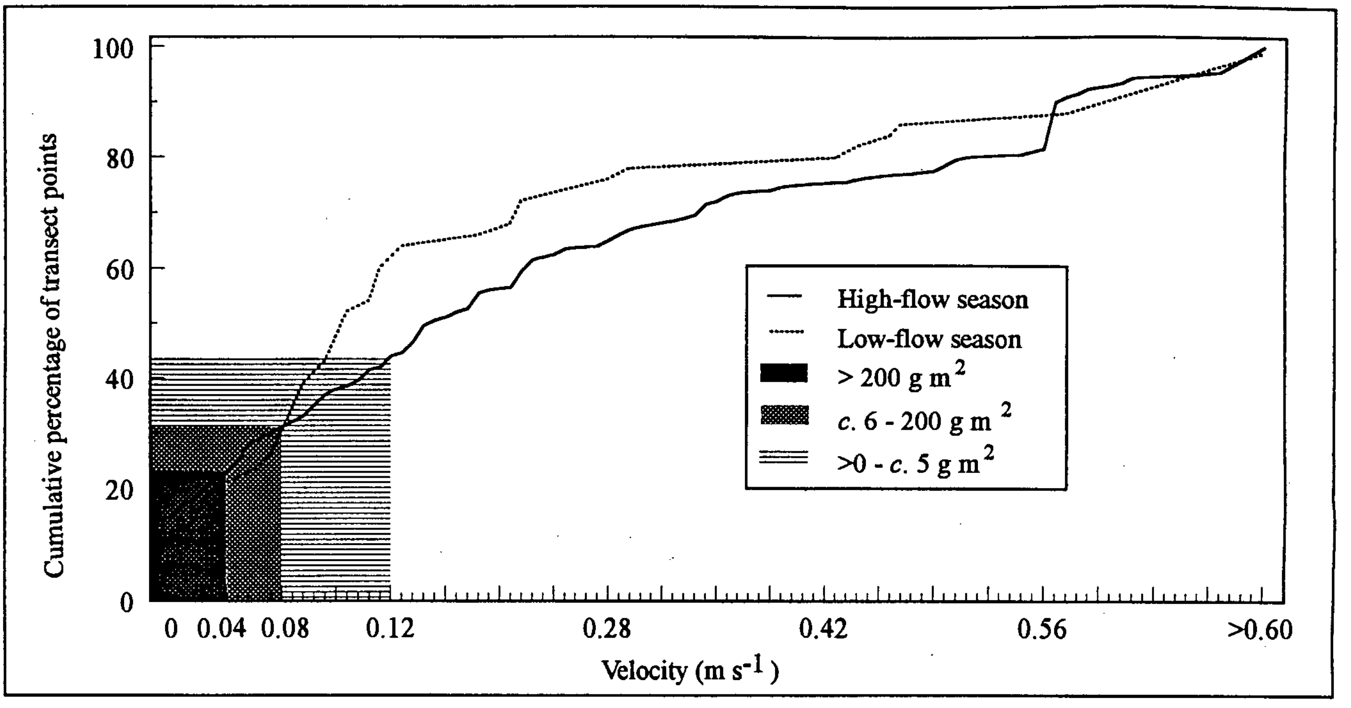

Figure 3.13 Range of velocities recorded in the river at PB1 and PB2 downstream of the portapool farm during base-flow conditions in the high-flow and low-flow season. The shaded areas indicate the velocities at which the majority of settlement of organic solids occurred

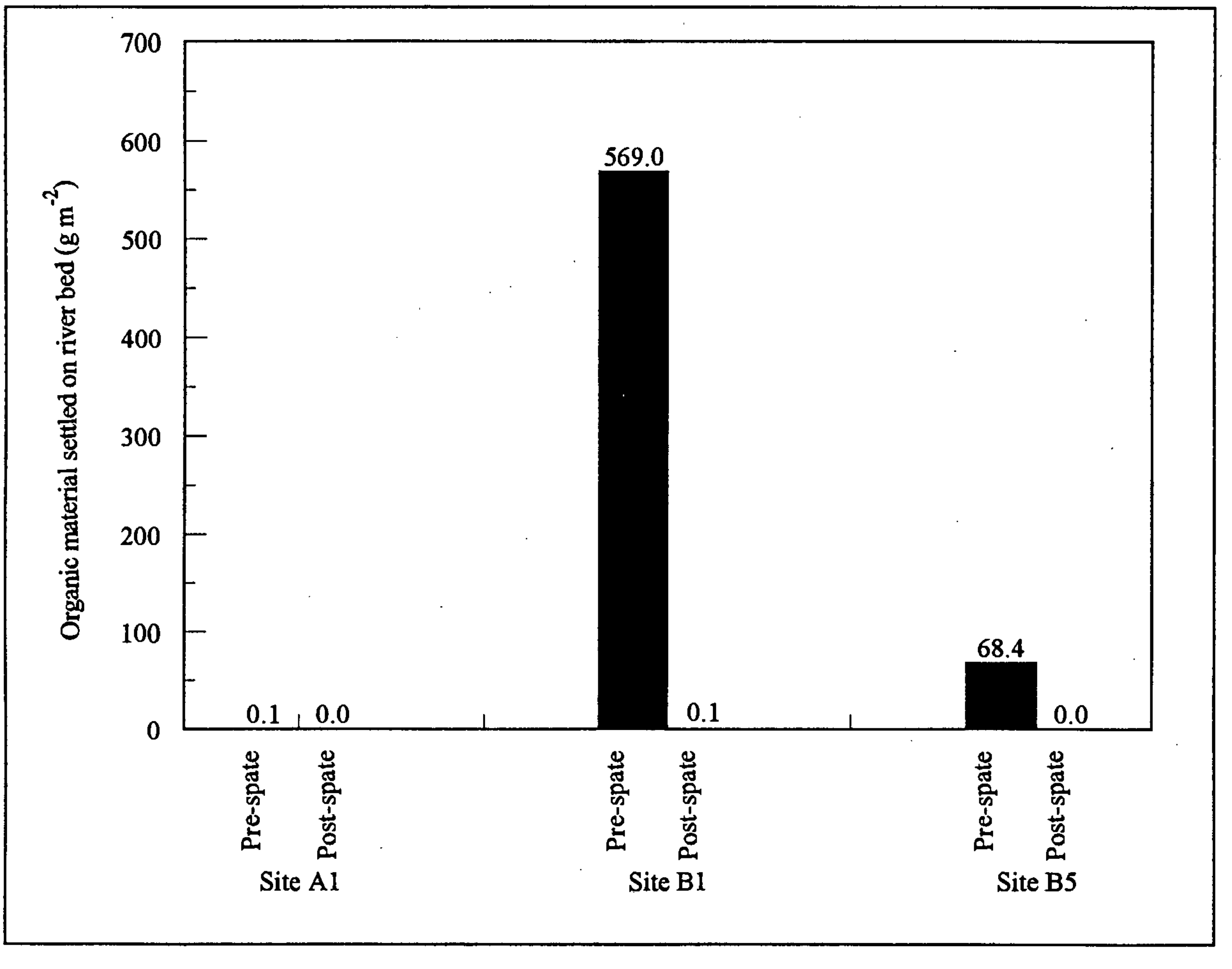

Figure 3.14 Measurements of the organic material settled on the riverbed downstream of the portapool farm collected before and after a flushing spate in August 1992

material. However, it is not known whether a smaller spate would also flush the material.

The flow duration curve for the Elands River at H1H033, compiled from data for ten randomly selected years is provided in Figure 3.15, and a hydrograph for the Elands River, constructed using data from H1H033 for a single representative year, is presented in Figure 3.16. A discharge of 10 m^3s^{-1} at H1H033 is equivalent to a discharge of *c.* 2 m^3s^{-1} at PB1 and PB2. Between them, Figures 3.15 and 3.16 indicate that discharges of sufficient magnitude to flush the organic material accumulated on the surface of the river bed downstream of the effluent outlet occur on average between 5 and 11 times a year, during the high-flow (winter) months.

Resettlement of organic material

The June sampling session was immediately preceded by a spate which flushed the organic material from the river bed. The peak discharge of this flushing spate was in the region of 4.0 m^3s^{-1}. Resettlement of organic material in the backwater areas only began on the penultimate day of the trip, *c.* 10 days after the peak in the hydrograph. The discharge in the Kraalstroom River at the time that resettlement was first observed was 0.4 m^3 s^{-1} and can be used to provide an indication of the discharge at which resettlement of organic solids recommences following a flushing spate.

Differences in the composition of the total suspended solids at the two study farms

Table 3.6 provides the percentage of the organic component of the total suspended solids measured in the effluents and in the rivers downstream of the two study farms. The organic content of the total suspended solids was considerably higher at the portapool farm than at the earthdam farm.

Effects of settled organic material on oxygen levels

Table 3.7 gives the oxygen concentrations recorded in riffles and in the backwaters in the river downstream of the portapool farm. The concentration of dissolved oxygen

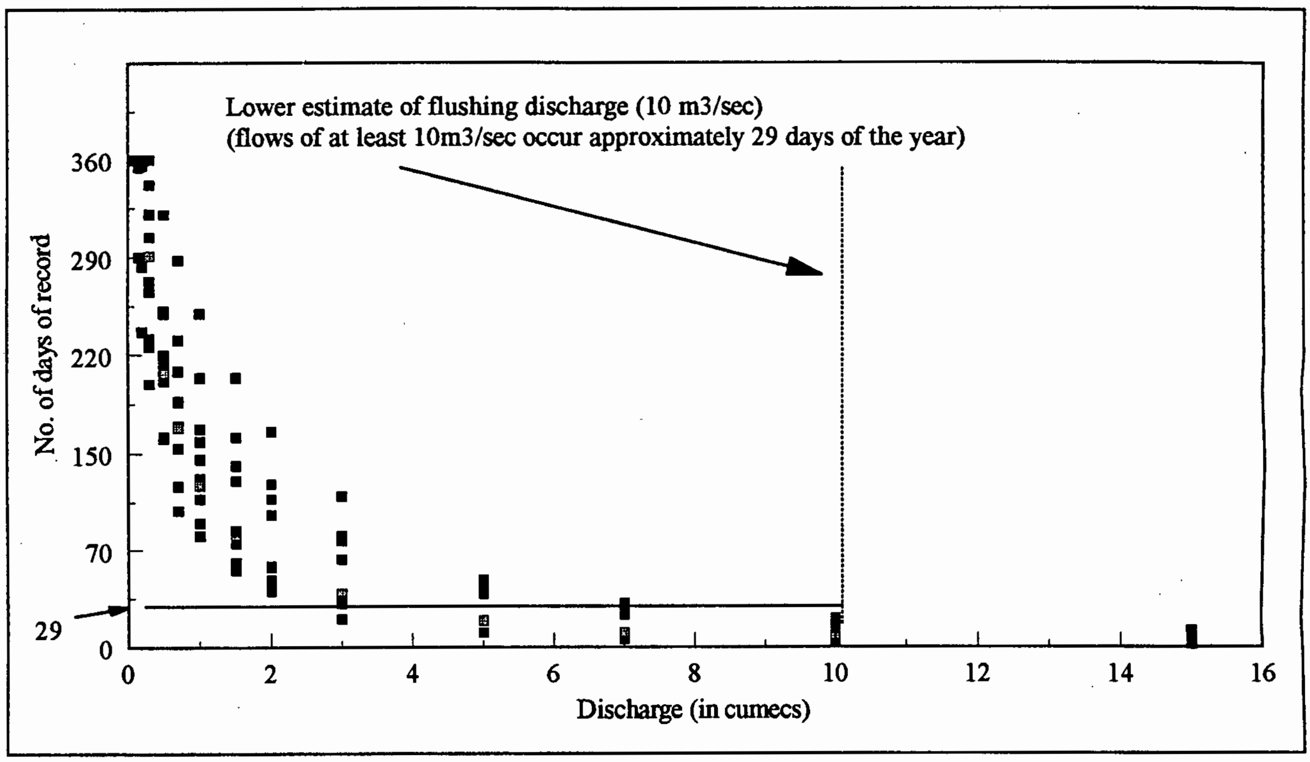

Figure 3.15 Flow duration curve for the Elands River (Data obtained from DWAF gauging weir no. H1H003)

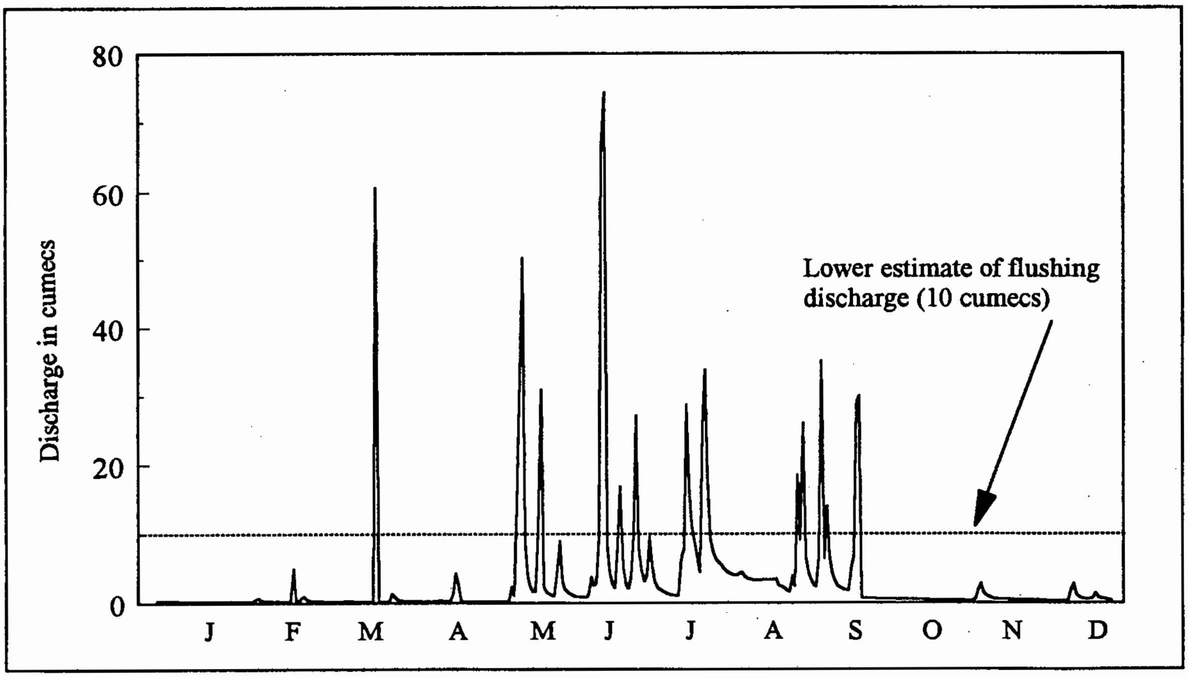

Figure 3.16 Hydrograph for the Elands River (Data obtained from guaging weir no. H1H017)

Table 3.6 Percentage of the organic component of the total suspended solids measured in the effluents (E) and downstream (B1-B5) of the two farms. n = the number of replicates

Farm and Sampling session	E %org (n)	B1 %org (n)	B2 %org (n)	B3 %org (n)	B4 %org (n)	B5 %org (n)
Portapool						
August 1992	60 (4)	59 (5)	61 (5)	53 (5)	47 (5)	46 (5)
November 1992		75 (4)	72 (4)	86 (4)	63 (4)	75 (4)
March 1993		76 (13)	66 (4)	70 (4)	66 (4)	99 (4)
June 1993		72 (5)	48 (5)	63 (5)	65 (5)	43 (5)
Earthdam						
March 1993	n/a	29 (1)	60 (1)	28 (1)	40 (1)	52 (1)
June 1993	31 (4)	43 (4)	49 (4)	30 (4)	34 (4)	35 (4)

Table 3.7 Oxygen concentrations in riffle and backwater biotopes measured at the portapool farm. Oxygen measurements were taken between 07h00 and 16h00, and adjusted for temperature and altitude

High-flow season					Low-flow season			
Session	Site	Days since last flush	Riffle O_2 - % saturation	Backwater O_2 - % saturation		Days since last flush	Riffle O_2 - % saturation	Backwater O_2 - % saturation
August Pre-spate (n = 3)	A1	37	82.9	68.7	November (n = 4)	23	81.9	59.7
	B1	37	76.5	65.0		23	74.9	43.6
	B2	37	76.0	61.8		23	80.7	49.5
	B3	37	81.2	61.0		23	83.5	40.6
	B4	37	80.0	68.9		23	79.9	66.8
	B5	37	79.1	65.4		23	84.2	56.6
August Post-spate (n = 2)	A1	1	100	80.15	March (n = 5)	131	91.6	74.6
	B1	1	88.1	81.1		131	89.0	69.4
	B2	1	100	96.8		131	92.8	61.0
	B3	1	100	83.5		131	90.8	75.2
	B4	1	100	95.5		131	91.2	76.2
	B5	1	98.3	93.2		131	88.8	69.0
June Post-spate (n = 3)	A1	1	97.3	97.3				
	B1	1	96.6	97.6				
	B2	1	95.6	98.2				
	B3	1	95.0	99.0				
	B4	1	93.2	99.8				
	B5	1	90.6	100				

recorded in the backwater areas downstream of the portapool farm in the low flow season and before the August spate were noticeably lower than those recorded in either the riffle areas, or in the backwater areas after the spate. This suggests that the presence of the settled organic material had a dampening affect on the levels of oxygen in the water column, particularly in the backwater areas.

3.4.4. Hydraulic conditions

There were no significant between-site differences in the Froude numbers at either of the two trout farms (ANOVA; $p>0.05$ for all sites) nor was there any significant difference in Froude numbers between the rivers at each of the farms (ANOVA; $F = 1.069$, $p = 0.4751$, d.f. = 70).

The proportions of different size particles making up the substratum at the sites associated with the two farms is given in Figure 3.17. Although the overall proportions did not appear to differ greatly, the mean particle size at the portapool farm was significantly greater (ANOVA; $F = 6.123$, $p = 0.000$, d.f. = 496) than that at the earthdam farm.

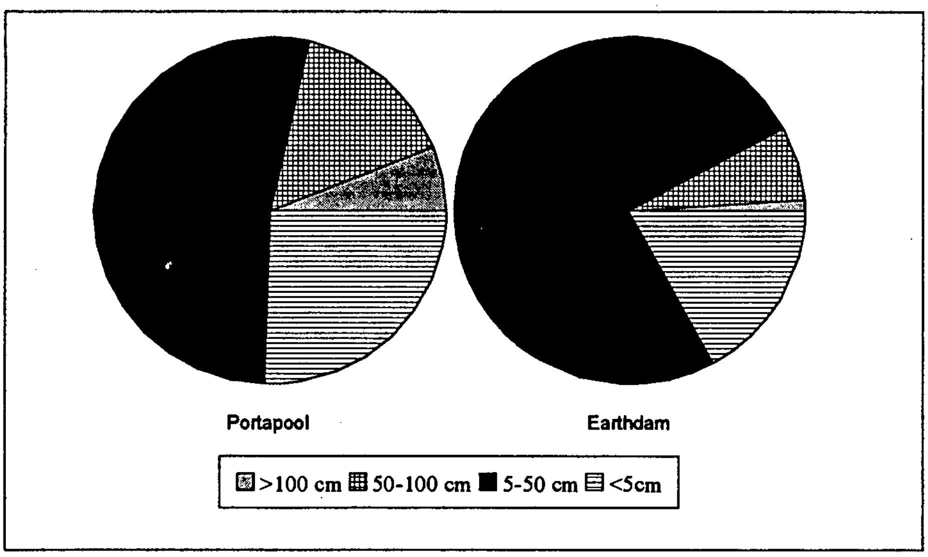

Figure 3.17 Pie diagrams indicating the proportion of different-sized substrata recorded in the rivers associated with the two study farms

3.5 DISCUSSION

Overall trends

The concentrations of the variables measured downstream of the two farms were similar to one another, as were the trends towards recovery to upstream conditions downstream of the effluent outlets. The grouping of the sites in the ordination plots for both study farms suggested a gradient, running from top right to bottom left diagonally across the plot from least to most impacted, although this trend was less marked in the case of the earthdam farm. This gradient provides evidence in support of the theory that there is a gradual recovery in overall water quality in the river with increasing distance downstream of the effluent outlets. In addition, in the case of the portapool farm, the samples collected in the wet season tended to group to the bottom-right of the plot and those collected in the dry season, to the top-left, indicating that a seasonal or discharge-related gradient may also exist.

Downstream recovery

The results of the paired-sample Signs tests showed significant differences between the levels of conductivity, total suspended solids, nitrate, phosphate and ammonium upstream (PA1) and immediately downstream (PB1), whereas, in the case of the earthdam farm, only total suspended solids differed significantly. Of the five variables that differed significantly between PA1 and PB1, three: conductivity, total suspended solids and nitrate, were significantly lower 1000 m downstream of the effluent outlet (PB5) than immediately below the outlet (PB1). However, only the conductivity levels approximated those recorded upstream of the farm (PA1). Thus, even though the levels of some of the variables showed a recovery towards upstream conditions with distance downstream of the portapool effluent outlet, by 1000 m downstream most had not yet recovered to their upstream levels. Furthermore, little or no downstream reduction in the concentrations of phosphate or ammonium was recorded.

Accumulated debris

The quantities of organic material settled on the river bed downstream of the two study farms differed markedly. Excepting for immediately after a flushing spate, the river bed downstream of the portapool farm was coated with a thick layer of organic material consisting of uneaten food and fish faeces. Differences in the composition of the suspended solids in the effluents from these two farms may explain the differences in the quantity of organic material settled on the river bed downstream of their effluent outlets. The effluent from the portapool farm contained a greater percentage of organic solids than did that from the earthdam farm which tended to be richer in inorganic suspended solids.

Naturally-occurring benthic organic matter, which originates from riparian trees, instream algae or macrophyte production (Maridet *et al.* 1995), is widely recognised as the primary source for benthic fauna in the heterotrophic headwaters of many streams (e.g. Minshall 1967, Vannote *et al.* 1980). Numerous studies have been conducted which illustrate the importance of the accumulation, retention and entrainment of leaf-litter and other woody debris for the macroinvertebrate communities inhabiting streams (e.g. Campbell *et al.* 1992, Prochazka *et al.* 1991, Lamberth *et al.* 1995, Malmqvist *et al.* 1978; Ractliffe *et al.* 1995, Maridet *et al.* 1995). Organic matter from effluent outfalls from, *inter alia*, aquaculture (Kelly *et al.* 1997) and intensive-feeding schemes for pigs or cows (Quinn and McFarlane 1989), however, does not offer the same benefits to the faunal or floral communities in these rivers. It differs from the naturally occurring particulate organic matter in rivers in terms of its size (in the case of mountain streams it is much smaller), quantity and texture (Dallas and Day 1993). Where it settles on the river bed, this organic material results in marked changes in the physical and chemical nature of the streams as well as changes in the micro- and macro-organisms living there (e.g. Hynes 1960, Chutter 1969, Seager and Abrahams 1990). Presumably, these deposits also alter the contribution of the hyporheic metabolism to the overall riverine ecosystem (Findlay 1995).

In the riffle biotopes downstream of the portapool farm, at base-flow conditions, almost half the total available habitat was affected by surface deposition. Presumably, since the organic matter settles preferentially in areas of slow flow, the pool and backwater biotopes would experience a greater loss of habitat area than the riffle areas. The deposition mosaic as a result of different hydraulic conditions and the effects of flushing have important implications for monitoring impacts of this nature on the benthic invertebrate fauna in the rivers, since different parts of the river experience different degrees of impact. A sample collected in a riffle or immediately after a flushing spate is likely to produce widely different results from one collected in a backwater some time after the last spate.

Flushing of the organic material

The temporal occurrence of the organic material is at least as important as its spatial distribution. The seasonal differences in the concentration of organic suspended solids in the effluent from the portapool farm were small. Although stocking rates dropped during the summer, the reduction was offset by a reduction in the flow through the farm, with the result that the concentration of suspended organic solids remained fairly constant. Thus, any seasonal differences in the quantity of organic material settled on the river bed were likely to be related to discharge in the river.

Several spate events that occurred during or immediately before sampling trips to the portapool farm provided valuable first estimates of the settlement and flushing dynamics of organic material in mountain streams. During the August 1992 sampling session at the portapool farm, a spate of between 2.0 and 4.1 $m^3 s^{-1}$ was recorded in the study river. Measurements of the amount of organic matter settled on the surface of the river bed at Site PB1 and Site PB5 before and after the event show that organic material was completely flushed during the spate. Combined with the hydrological data from the gauging weir on the Elands River, these data suggest that the river downstream of the portapool farm was flushed between 5 and 11 times a year, during the high-flow (winter) months. Observed changes in the consistency of the settled material appeared to be a complicating factor with respect to it being flushed from the system, although these changes were not quantified during the study. Being organic, the material on the river bed tended to coagulate with time and form a gelatinous mass, with the particles sticking to one another. This meant that the flushing spates probably had to be sufficiently strong to move all the material settled at the same time. Thus, once the material had lain on the river bed for a few days, many of the smaller flushing spates were unable to dislodge the settled material. Furthermore, since most of the spates occurred during the high-flow (winter) season, organic material which settled on the river bed after the last winter spate would in all likelihood remain there until the onset of the rains in the following winter.

It should also be noted that the flushing spates only cleared the organic material settled on the surface of the river bed. Their effects on the organic material trapped in the interstitial spaces between the cobbles was not recorded. Preliminary observations also suggest that resettlement of organic material occurs when discharge in the river drops below 0.4 $m^3\ s^{-1}$.

Importantly, not all areas of the channel are equally affected by the depositions. This may result in refuge areas being available for macroinvertebrates inhabiting the river. Presumably animals that are able to withstand velocities in excess of 0.12 $m^3 s^{-1}$ are likely to be less affected by the organic deposits than are those that prefer backwater areas.

Total load of organic material

Total suspended solid (TSS) values recorded in the effluent at the portapool farm during the detailed study yielded values of between 1.9 and 6.8 mg l^{-1}. Using a conservative estimate of *c.* 0.1 $m^3\ s^{-1}$ for the annual average discharge of the effluent, these translate to between 6 and 21.4 tonnes dry weight of suspended solids being deposited into the river each year. An average of 60% of the TSS consists of organic material, representing between 3.6 and 12.8 tonnes dry weight of organic material entering the river from the farm each year.

The effect on oxygen levels

Oxygen concentrations measured in the water column in the riffles did not change significantly downstream of the effluent outlets. However, oxygen levels recorded in the backwater biotopes downstream of the portapool effluent outfall before the flushing spate in August, and in November and March were considerably lower than those recorded in the riffles. This was undoubtedly a result of the decomposition of the organic matter that had settled in these areas. Oxygen levels recorded in the backwater areas in June and post-spate in August did not differ from those recorded in riffle areas. Both of these sampling sessions were immediately preceded by a spate, which flushed the backwater areas. Thus, apart from the possible physical changes which resulted from the presence of the organic material, there were also chemical changes associated with its presence.

Hydraulic conditions

It is possible that differences in the hydraulic conditions in the two study rivers could have explained some of the differences in the quantity of organic material on the riverbeds downstream of the two farms. The hydraulic conditions, as reflected by Froude numbers, in the two study rivers did not differ significantly from one another. The substrata in the river at the portapool farm were significantly coarser than those in the river at the earthdam farm. However, generally speaking, larger particles (gravel, cobble and boulders) such as those recorded in both rivers, are associated with faster currents, and smaller particles (sand, silt and clay) with slower ones (Gordon *et al.* 1992). Hence of the two, greater deposition would be expected at the earthdam farm. Both study reaches were considered to be representative of the erosional zone of the rivers, however, since silt and clay did not occur in appreciable amounts at any of the sample sites. Hence, differences in the hydraulic conditions or the composition of the substrata are unlikely to be sufficient to account for the marked differences in the amount of settlement occurring in the two rivers.

Thus, assuming the rivers on which the farms were situated did not differ considerably from one another in terms of physical attributes of the water, the amount of organic material settled downstream of each effluent outlet must be related to the nature of the effluent. Although there were no striking differences in the concentration of total suspended solids leaving the farms, there were differences in the composition thereof, with the effluent from the portapool farm containing a far higher percentage of organic solids than did that from the earthdam farm. This difference appears to be a major factor contributing to the differences in their effects on the river.

Hence, although the general physical and chemical character of the study rivers changed downstream of the effluent outlets of both farms, by far the most noticeable effect was related to the deposition of organic matter downstream of the portapool farm. Furthermore, since the hydraulic conditions in the rivers did not differ significantly from one another, the greater organic deposition downstream of the portapool farm was probably related to the greater organic content of the suspended solids in the effluent. The organic content of the effluent from the earthdam farm was considerably lower than that from the portapool farm. Since the fish at both farms were fed the same or similar diets, the increased organic content of the effluent is most likely a consequence of operational differences between the two farms. The slow flow-through rate of earthdam farms, coupled with the fact that they cannot be flushed in the same way as portapool farms, means that they act as settlement tanks. This effectively prevents the particulate organic component of fish wastes from entering the river. In fact the greatest effect on the river from the earthdam was recorded during the low-flow summer months when very few fish inhabited the farm and the ponds were being cleaned. This may be related to an increase in inorganic suspended solids during this time, since inorganic sediments have been shown to interfere with the processing of leaf litter (Bunn 1988), and hence the presence of many species of Trichoptera and Plecoptera in mountain streams.

CHAPTER 4

MACROINVERTEBRATE COMMUNITY PATTERNS RECORDED AT TWO LAND-BASED TROUT FARMS AFFECTING STREAMS IN THE SOUTH-WESTERN CAPE, SOUTH AFRICA

4.1 INTRODUCTION

The concept of using stream and lake benthos to detect pollution is not new, and the subject has recently been reviewed by Cairns and Pratt (1993). Organic pollution is probably the most common, and best documented, type of pollution occurring in rivers. The main distinction between naturally occurring dissolved and particulate organic matter in rivers, and organic matter that pollutes, is related to the size, quantity and texture of the organic matter (Dallas and Day 1993). Polluting organic matter that enters a river system can have detrimental effects on many species of the aquatic biota (Hynes 1960, Wiederholm 1984) and often results in changes in biotic community structure, and in species richness and diversity (Chutter 1972, Biggs and Price 1987, Whitehurst and Lindsay 1990, Sheath and Cole 1992). For instance, sewage overflows in the Pendle River in the United Kingdom led to a reduction in macroinvertebrate densities, almost complete elimination of certain species (e.g., the isopod, *Asellus aquaticus*, and the mayfly, *Baetis rhodani*), and a concomitant increase in oligochaetes, particularly tubificids (Seager and Abrahams 1990). Berhe *et al.* (1989) reported similar changes in macroinvertebrate community structure in a stream flowing through Addis Ababa in Ethiopia. Certain macroinvertebrates, such as larvae of the fly *Eristalis* (rat-tailed maggots), *Psychoda* (sewage flies) and *Chironomus* (bloodworms), and oligochaete genera such as *Tubifex* and *Limnodrilus*, are indicators of severe organic pollution (Dallas and Day 1993). Subtle changes in community structure are, however, often evident at much lower pollution levels (Armitage *et al.* 1983, Dallas 1995).

This chapter details the changes in macroinvertebrate community structure in the river upstream and downstream of the portapool and earthdam trout farms introduced in Chapter 3. It relates these changes to the changes in physical and chemical character of the water as presented in Chapter 3, and discusses them in terms of current ecological theory.

4.2 METHODS

The location and characteristics of the two farms are described in Chapter 3, as are the locations of the study sites. The methods of collection and analysis are given in Chapter 2.

Samples of the benthic macroinvertebrates were collected from both backwater (three samples taken at each of the six sites, n = 3) and riffle (n = 3) biotopes at each site. Collections were made once during the summer (March) and twice in the winter (June and August). These data were collected at the same time and at the same sites as the physical and chemical data analysed in Chapter 3.

4.2.1 Numerical analysis

Insect:non-insect ratios

The data collected at each of the sites were initially compared at the taxonomic level of Class, *viz.* insect to non-insect. The ratios of insect to non-insect individuals in the samples were used to assess whether the effect of the effluent being discharged into the river was to shift the structure of the macroinvertebrate community from an upper river, dominated by insects, to a lower-river type of community, dominated by non-insect taxa (e.g. King 1981).

Analysis of similarity (ANOSIM)

The benthic macroinvertebrate samples collected from the different sites were grouped on the basis of *a priori* decisions as to their relationships with one another, on the basis of the following hypotheses:

1. Effluents from trout farms, discharged into an upper river or mountain stream, will result in a predictable change in benthic-invertebrate community structure. The community will tend to change from one dominated by insects, to one dominated by non-insects. The degree of change will be dependent on the pollutant levels in the stream.
2. The effects of trout-farm effluent on benthic community structure will be greatest at the point where the pollution enters the river (assuming similar hydraulic conditions). Gradual recovery of the river ecosystem to the upstream condition (or

an approximation thereof) will occur with increasing distance downstream of that point.

3. The effects of the trout farm effluent will be greatest in the slow-flowing biotopes, such as backwaters.

The samples collected upstream and those collected downstream of the farms were subjected to a one-way Analysis of Similarity (ANOSIM, PRIMER Version 3.1a; Clarke and Warwick 1994), on the assumption that the former represent unaffected communities, and the latter communities that are affected by trout-farm effluent.

ANOSIM is analogous to the well-known Analysis of Variance procedure, ANOVA. Unlike ANOVA, however, it does not depend on the assumption that the data are normally distributed. ANOSIM is a simple, non-parametric permutation procedure that is applied to the (rank) Bray-Curtis similarity matrix underlying the ordination or classification of samples (Clarke and Warwick 1994), which allows the testing of the null hypothesis that there is no difference between the groups. The test is possible because, unlike many other multivariate procedures, MDS plots and hierarchical clustering dendrograms are constructed from pairwise similarities among samples and are not based on any assumed statistical distribution. Differences between samples, such as those from impacted and unimpacted sites, can therefore be tested. The main assumption of the ANOSIM test is that the groups that are tested must be specified prior to analysis, based on hypotheses. Cluster analysis or MDS of the biota cannot be used to define the groups that are tested with ANOSIM using the same data, because the argument would then be a circular one and would violate one of the few basic assumptions of ANOSIM.

A one-way ANOSIM was also used to test whether or not there were significant differences in community structure with distance downstream of the effluent outlet. The community data from downstream of the farms were divided into sites close to (e.g. Sites PB1 and PB2) and those distant from (Sites PB3, PB4 and PB5) the effluent outlet. The hypothesis was that, if downstream recovery to upstream conditions (e.g. O'Keeffe *et al.* 1989b) was occurring, then the macroinvertebrate community in the river immediately downstream of the effluent outlet would be expected to be significantly different from that at the more downstream sites.

Finally, a one-way ANOSIM was used to test whether or not there were significant differences in community structure between backwater biotopes and riffle biotopes.

The ANOSIM tests were performed on root-root transformed data, analysed using Bray-Curtis measurements of similarity (Bray and Curtis 1957). The raw data are presented in Appendix A.

Classification and Ordination

Following the ANOSIM, the data were analysed using the multivariate techniques of classification (hierarchical clustering) and ordination (multidimensional scaling) which form part of the software package PRIMER Version 3.1a (Clarke and Warwick 1994). The results obtained from the three replicates collected in each biotope at each site were combined and averaged, and the multivariate techniques performed on a matrix of abundance data for *c.* 80 species from 48 samples and 36 samples for the portapool farm and earthdam farm respectively. The data were root-root transformed to reduce the influence of numerically dominant taxa and analysed using Bray-Curtis measurements of similarity (Bray and Curtis 1957). The raw data are presented in Appendix A.

SIMPER analysis

The macroinvertebrate data collected in the riffle habitats were combined into two groups on the basis of the results from ANOSIM analysis, and the species composition of the macroinvertebrate communities in the two groups was analysed using SIMPER (PRIMER Version 3.1a; Clarke and Warwick 1994). SIMPER measures the degree of separation between the groups, compares the average abundances of species in pre-selected groups with one another, and identifies the species that are most responsible for the Bray-Curtis inter-group differences (Clarke and Warwick 1994).

BIO-ENV analysis

The BIO-ENV analysis, which also forms part of the software package PRIMER (Clarke and Ainsworth 1993), was used to examine the extent to which the water quality data collected in conjunction with the biotic samples (Chapter 3) explains the observed biological pattern.

The premise for this procedure is that, if a set of environmental variables responsible for structuring the community were known, then samples having similar combinations of these variables would be expected to have similar species composition (Clarke and Warwick 1994).

The BIO-ENV procedure matches the biotic and environmental patterns between the samples and identifies the abiotic variable(s) that best group(s) the sampling sites in a manner consistent with the faunal patterns.

4.3 RESULTS

4.3.1 Ratio of insect to non-insect taxa

The combined ratios of insect to non-insect taxa making up the samples at each of the farms (Figure 4.1) revealed that:

- at the portapool farm, there was a shift from a community dominated by insects upstream of the effluent outlet to one dominated by non-insects downstream of the effluent outlet, and there was no discernible recovery to the upstream condition within the first kilometre downstream of the effluent outlet.
- at the earthdam farm, the expected shift from an insect-dominated macroinvertebrate community upstream of the farm to a non-insect-dominated macroinvertebrate community downstream of the effluent outlet did not occur. Indeed, this farm appeared to have little effect on the species composition of the macroinvertebrate community.
- there were no discernible differences in the patterns recorded during the high-flow and low-flow sampling sessions at the two farms.

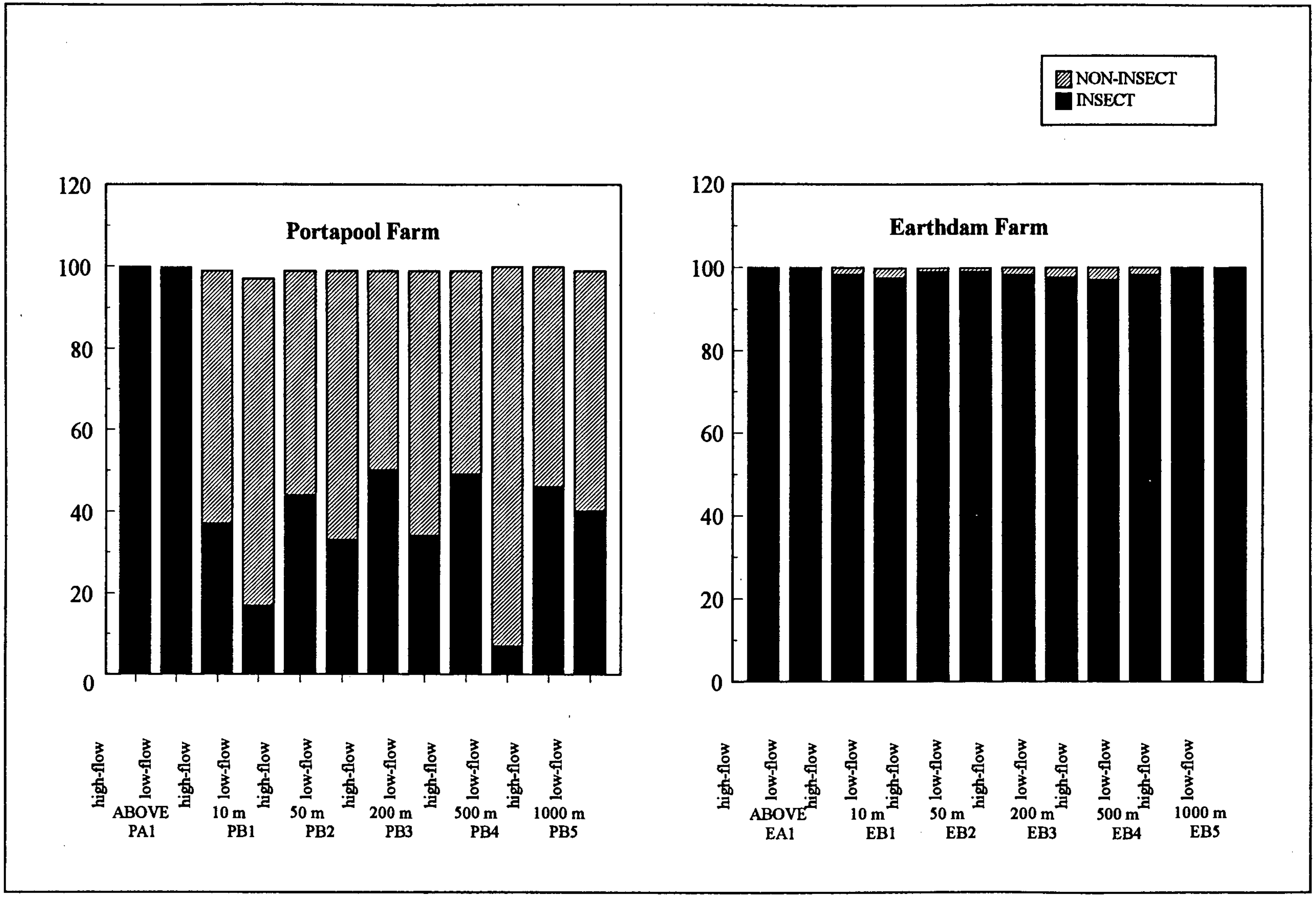

Figure 4.1 Ratios of insect to non-insect taxa making up the samples collected upstream and downstream of the two trout farms

4.3.2 ANOSIM

In ANOSIM, a Global R value of approximately zero indicates that the null hypothesis is true and that similarities between and within sites are roughly the same (Clarke and Warwick 1994). The one-way ANOSIM analyses yielded the following results:

- There was a statistically significant difference between the community recorded upstream of the portapool trout farm and that recorded downstream of the farm (Global R = 0.636, p = 0.000).
- At the portapool trout farm there were no significant differences between the communities collected at the sites ≤ 50 m downstream of the outlet and those collected at sites > 500 m downstream of the outlet (Global R = -0.062, p = 0.709), suggesting that 500 m was insufficient distance to allow statistically detectable differences between the macroinvertebrate communities to emerge.

With the exception of the august sample, there were no significant differences between the macroinvertebrate communities recorded in the backwaters and riffles upstream of the portapool farm (Table 4.1). The results of the ANOSIM test between the macroinvertebrate communities recorded in the backwater biotopes versus those recorded in the riffle biotopes downstream of the potapool farms are also presented in Table 4.1.

Table 4.1 The results of the one-way ANOSIM test for the backwater versus the riffle macroinvertebrate communities downstream of the two study farms

Farm	Month	Global R	Significance level
Portapool farm (upstream)	March	-0.185	p = 0.90
	June	0.000	p = 0.60
	August	0.926	p = 0.10
Portapool Farm (downstream)	March	0.173	p = 0.012
	June	0.252	p = 0.000
	August	0.336	p = 0.001
	June	0.312	p = 0.024

Significant differences were detected between the backwater and the riffle communities recorded during any of the sampling sessions. Hence, the hypothesis that the effects of the trout farm effluent will be greater in the slow-flowing biotopes, such as backwaters, than in the fast-flowing biotopes, such as riffles, was at least partially supported by that data collected during this study.

4.3.3 Hierarchical clustering and MDS

Hierarchical clustering of the macroinvertebrate community data collected in the stream at the portapool farm during winter (high-flow conditions; Figure 4.2) and summer (low-flow conditions; Figure 4.3) revealed that the macroinvertebrate communities at the upstream control site were less than 45% similar to those recorded at any of the downstream sites. This illustrates the differences in macroinvertebrate community structure upstream and downstream of the effluent that resulted in a significant result in the ANOSIM test. The separation between the samples collected at the sites upstream of the effluent outlet and those collected downstream thereof is shown clearly in both Figures 4.2 and 4.3. In Figure 4.3, the samples collected in the backwater biotopes at the site immediately downstream of the effluent outlet, *viz.* Site B1 (0 m downstream) and Site B2 (50 m downstream) grouped together, which suggests that perhaps the impact in the backwater areas is greater nearer the effluent outlet.

The cluster and MDS analyses of macroinvertebrate communities recorded upstream and downstream of the earthdam farm (Figures 4.4 and 4.5) revealed a different picture from those for the river affected by the portapool farm. The communities recorded at all the former sites were >70% similar to each other; there was no logical sequence in the relationships among the sites; and the clearest divisions occurred in the high-flow samples, between the backwater and riffle biotopes. This suggests that the overriding differences between the communities sampled were a result of natural variation associated with different biotopes (e.g. Palmer *et al.* 1991). Hence, the effluent from the earthdam farm had a smaller effect than did the portapool farm on macroinvertebrate communities inhabiting the downstream receiving river.

The MDS plot in Figure 4.6a enables the relative differences in macroinvertebrate community structure downstream of the portapool and earthdam farms to be assessed. Two macroinvertebrate samples collected from undisturbed mountain streams elsewhere in the south-western Cape have been included in the analysis as reference points (FRU, unpublished data). The two new sites cluster together with the site upstream of the portapool farm and with the sites upstream and downstream of the earthdam farm. Hence, from Figure 4.6a, two broad groups are identifiable, namely:

Group A: represented by the riffle samples from the upstream site at the portapool farm and the riffle samples from all the sites at the earthdam farm

Group B: represented by the riffle samples from Sites P0, P50, P200, P500 and P1000, downstream of the portapool farm.

Figure 4.6b shows a PCA of the environmental values with the same two groups delineated.

4.3.4 SIMPER analysis

SIMPER analysis performed on two groups shown in Figure 4.6a (Table 4.2) indicated that Group A was characterised by families, such as the Helodidae and Elmidae, that are considered to be indicative of undisturbed south-western Cape mountain streams (Dallas 1995). On the other hand, Group B was characterised by the abundance of taxa such as the Oligochaeta and Chironomidae (specifically *Chironomus* sp.), that are considered to be tolerant of organic pollution (Hynes 1960).

The average abundances of the macroinvertebrates that inhabit the surfaces of boulders and cobbles relative to those that burrow into the substratum for each of the groups are illustrated in Figure 4.7. Group B had larger numbers of burrowing individuals than did Group A.

4.3.5 BIO-ENV - Linking the macroinvertebrate community data with environmental variables

The physical and chemical conditions in the water associated with the macroinvertebrate communities represented by Group A and those associated with the communities represented by Group B are summarised in Table 4.3. Sites with different community structure might be expected to differ with respect to the concentrations of

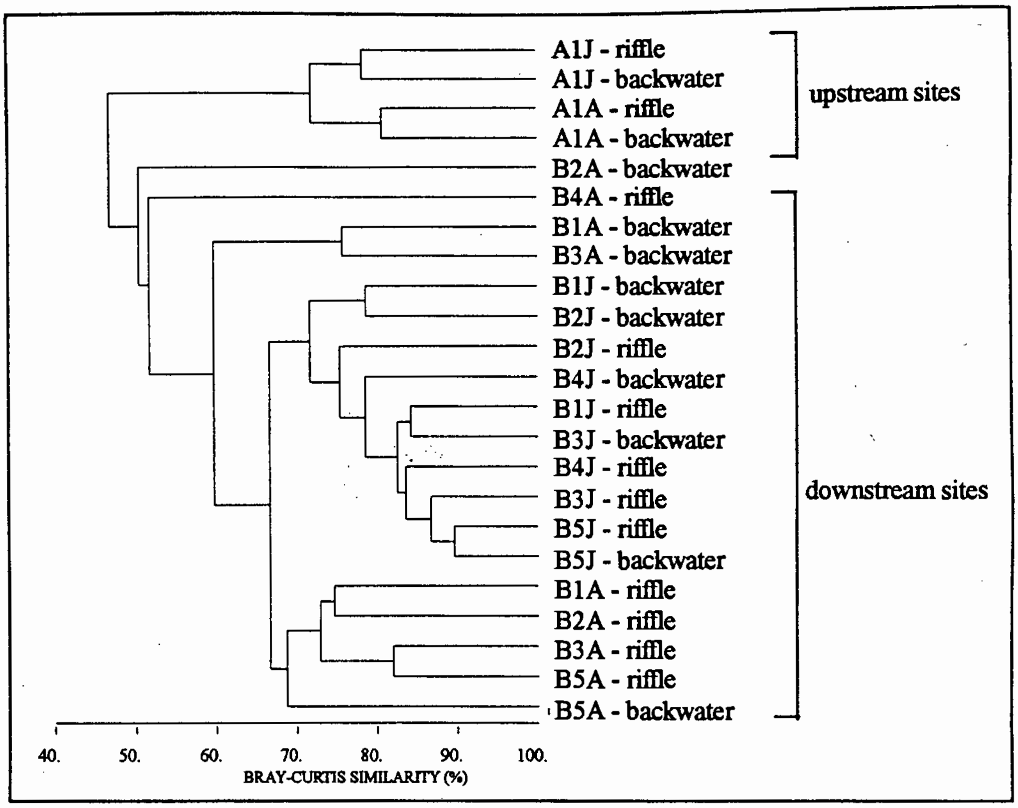

Figure 4.2a Dendrogram showing the results of the hierarchical classification of macro-invertebrate samples collected in riffles and backwaters during the high-flow sampling sessions (June and August) upstream (Site A1) and downstream (Sites B1 - B5) of the portapool farm. Codes: A = upstream, B = downstream, 1 - 5 = site codes, J = June and A = August. B4A - backwater was not collected

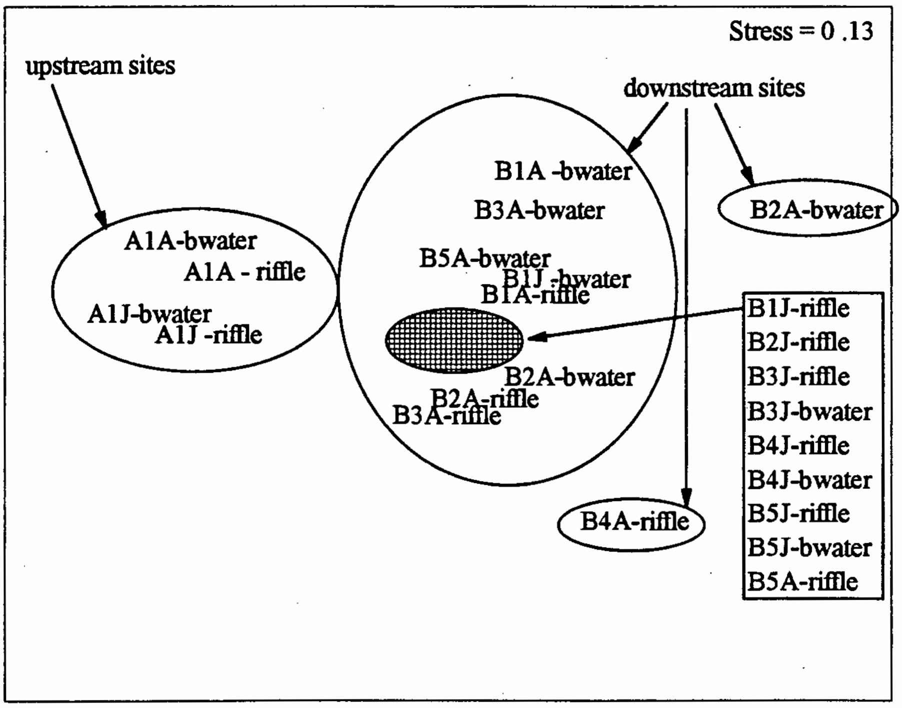

Figure 4.2b Ordination plot of the macroinvertebrate samples collected during the high-flow sampling sessions (June and August) upstream (Site A1) and downstream (Sites B1 - B5) of the portapool farm, using the same similarity matrix used in Figure 4.2a. Codes: A = upstream, B = downstream, 1 - 5 = site codes, J = June and A = August. B4A - backwater was not collected. Axes have arbitary scales which are therefore not shown

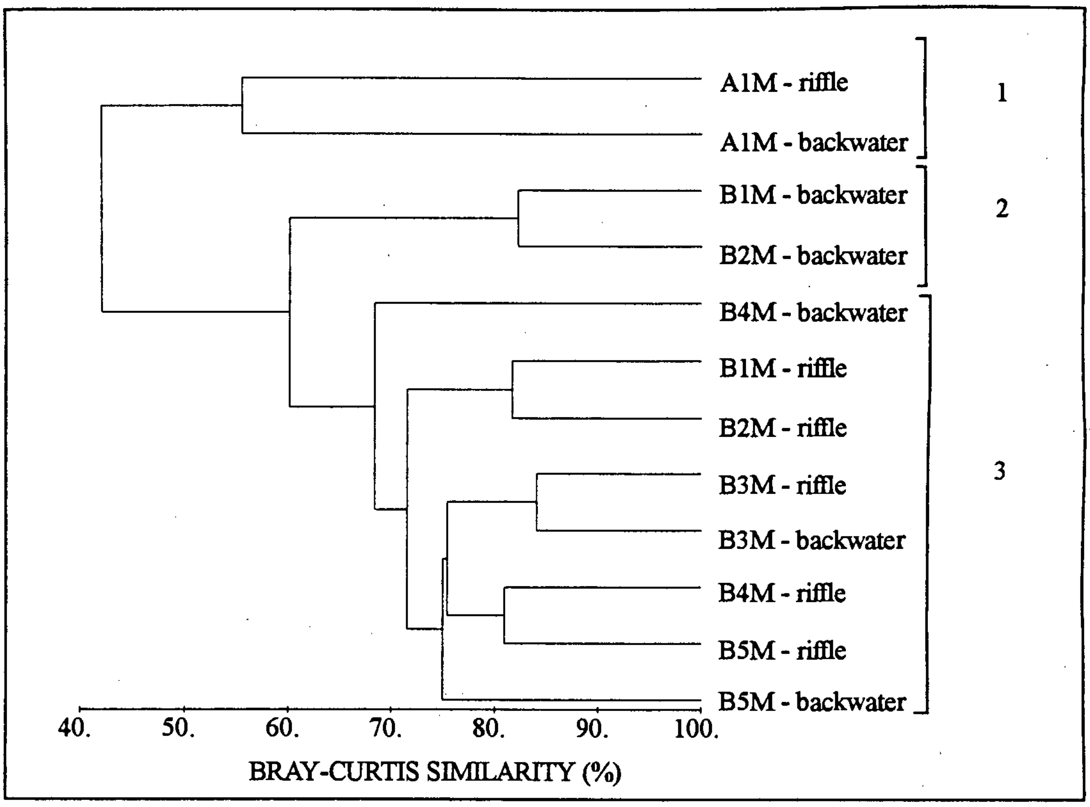

Figure 4.3a Dendrogram showing the results of the hierarchical classification of macroinvertebrate samples collected during the low-flow sampling session (March) upstream (Site A1) and downstream (Sites B1 - B5) of the portapool farm. Codes: A = upstream, B = downstream, 1 - 5 = site codes, M = March

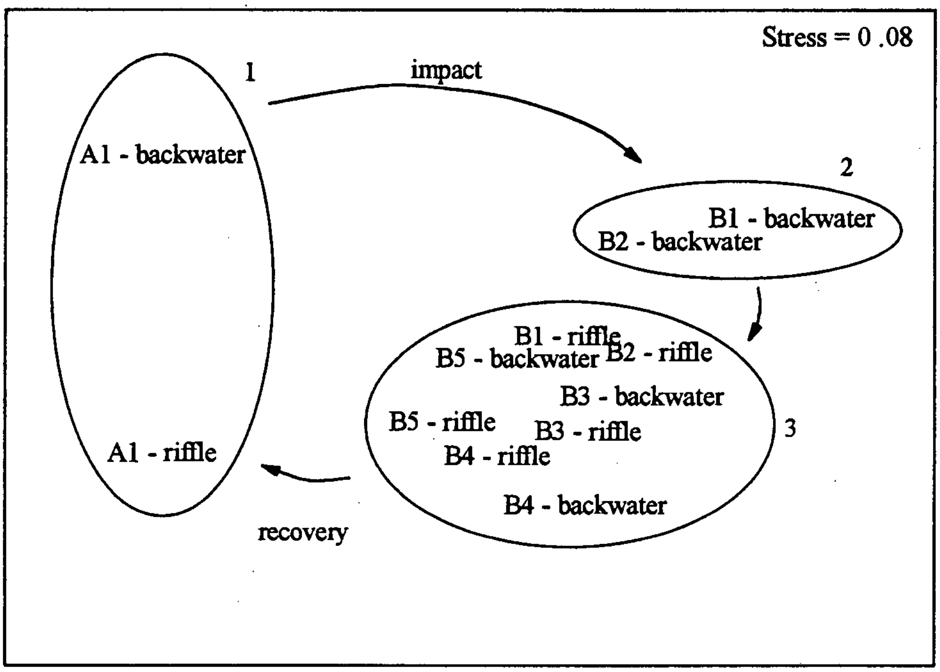

Figure 4.3b Ordination plot of the macroinvertebrate samples collected during the low-flow sampling session (March) upstream (Site A1) and downstream (Sites B1 - B5) of the portapool farm, using the same similarity matrix used in Figure 4.3a. Codes: A = upstream, B = downstream, 1 - 5 = site codes. Axes have arbitary scales which are therefore not shown. A suggested trend of impact and recovery is illustrated by the arrows

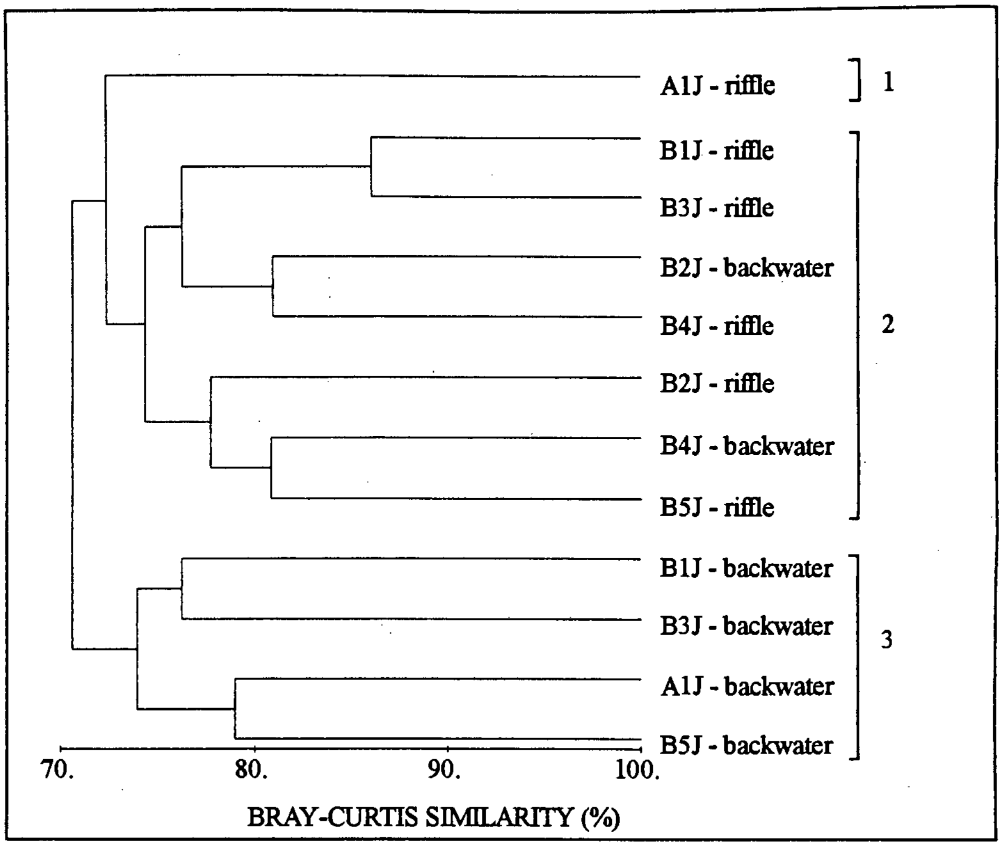

Figure 4.4a Dendrogram showing the results of the hierarchical classification of macroinvertebrate samples collected during the high-flow sampling session (June) upstream (Site A1) and downstream (Sites B1 - B5) of the earthdam farm. Codes: A = upstream, B = downstream, 1 - 5 = site codes, J = June

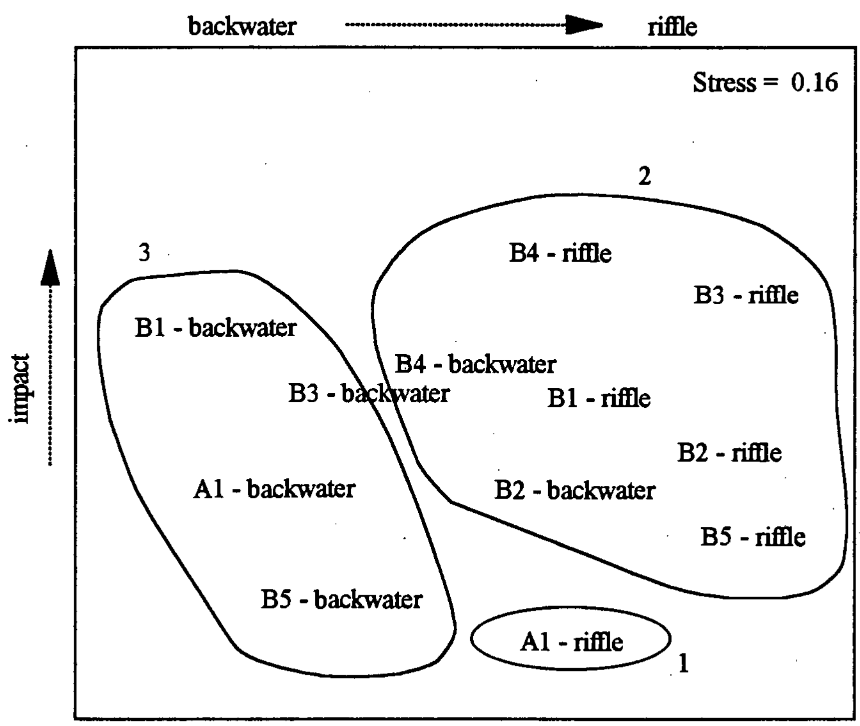

Figure 4.4b Ordination plot of the macroinvertebrate samples collected during the high-flow sampling session (June) upstream (Site A1) and downstream (Sites B1 - B5) of the earthdam farm, using the same similarity matrix used in Figure 4.4a. Codes: A = upstream, B = downstream, 1 - 5 = site codes. Axes have arbitary scales which are therefore not shown. The arrows indicate possible trends

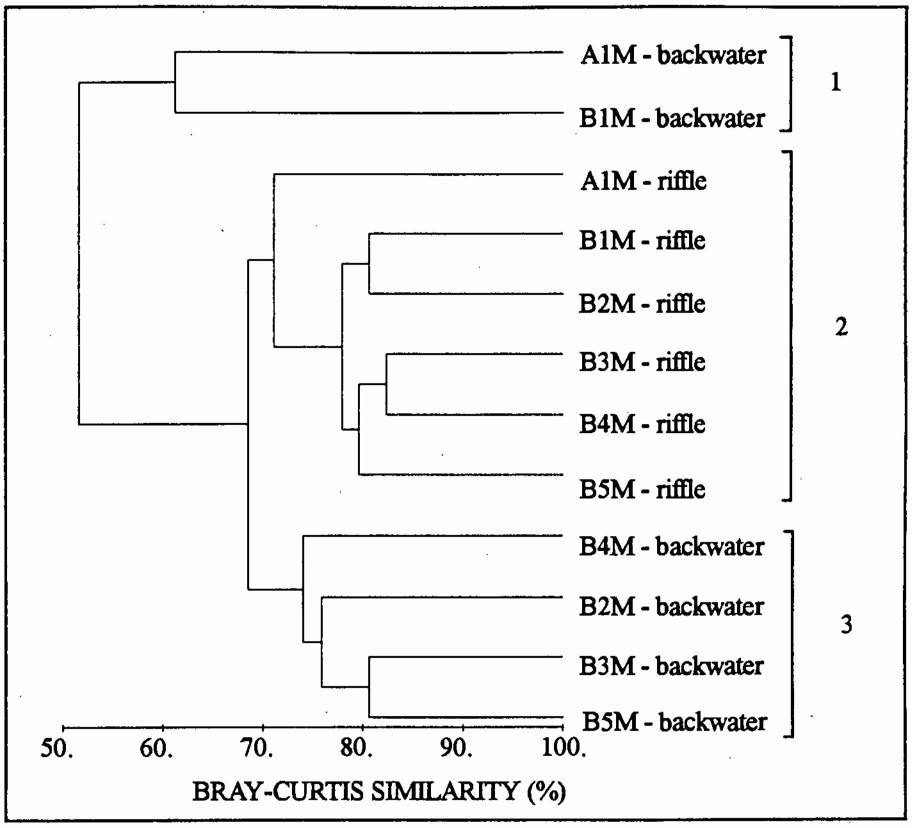

Figure 4.5a Dendrogram showing the results of the hierarchical classification of macroinvertebrate samples collected during the low-flow sampling session (March) upstream (Site A1) and downstream (Sites B1 - B5) of the earthdam farm. Codes: A = upstream, B = downstream, 1 - 5 = site codes, M = March

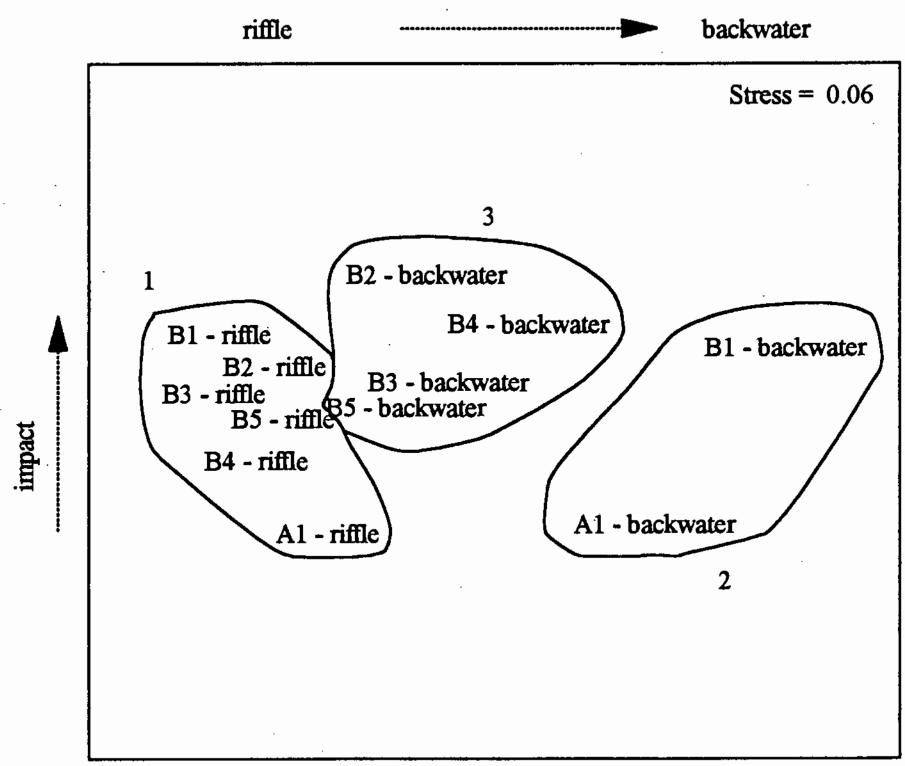

Figure 4.5b Ordination plot of the macroinvertebrate samples collected during the low-flow sampling sessions (March) upstream (Site A1) and downstream (Sites B1 - B5) of the earthdam farm, using the same similarity matrix used in Figure 4.5a. Codes: A = upstream, B = downstream, 1 - 5 = site codes. Axes have arbitary scales which are therefore not shown. The arrows indicate possible trends. The greater distance left-right suggests riffle-backwater community effects are greater than impact effects (top-bottom)

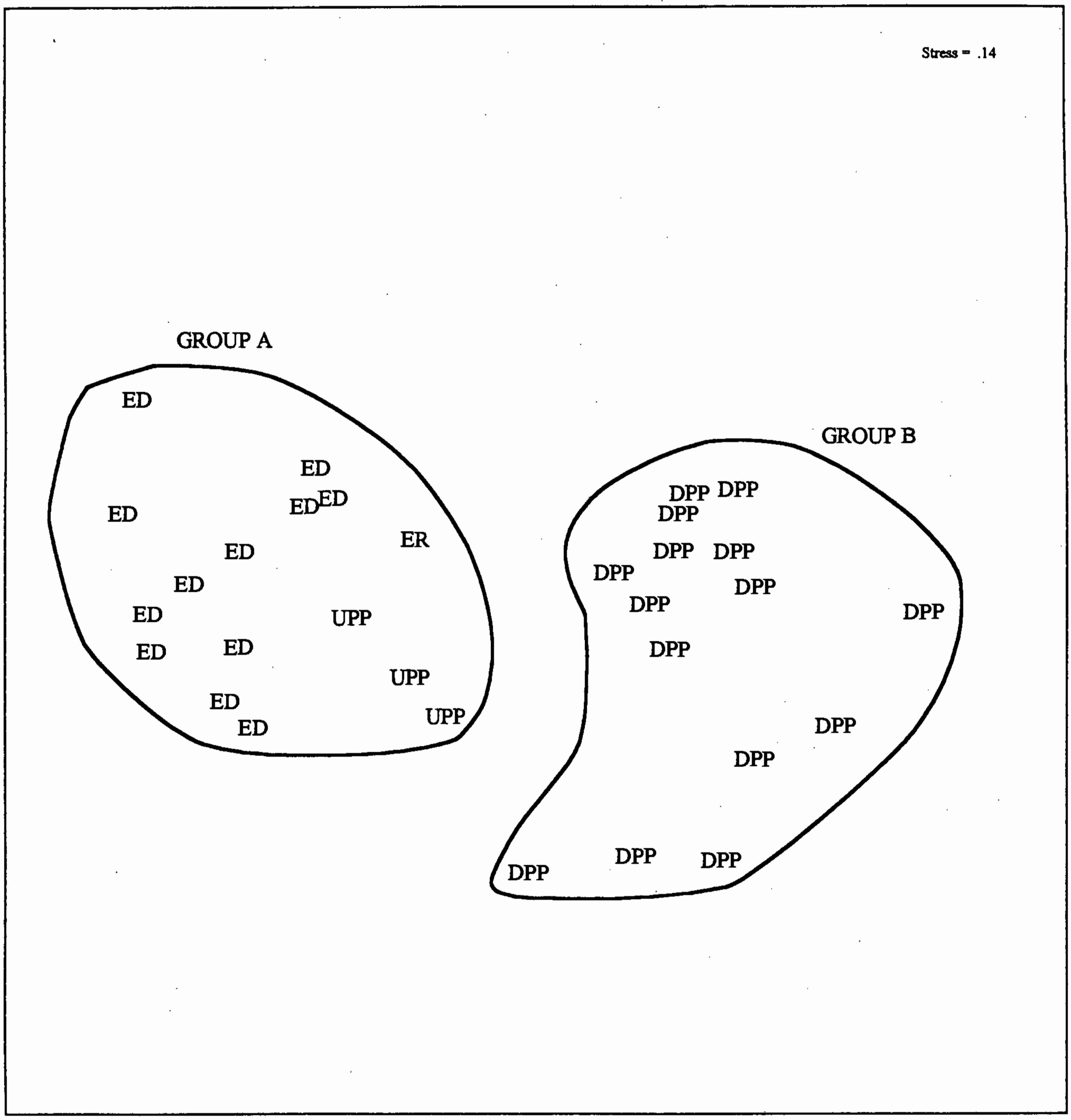

Figure 4.6a MDS ordination plot of all the macroinvertebrate samples collected during the detailed study and allocated into two groups: Group A, upstream of portapool farm (UPP), Elands River (ER) and all sites (upstream and downstream) at earthdam farm (ED). Group B , downstream of the portapool farm (DPP). The data were root-root transformed and analysed using the Bray Curtis Measure of Similarity

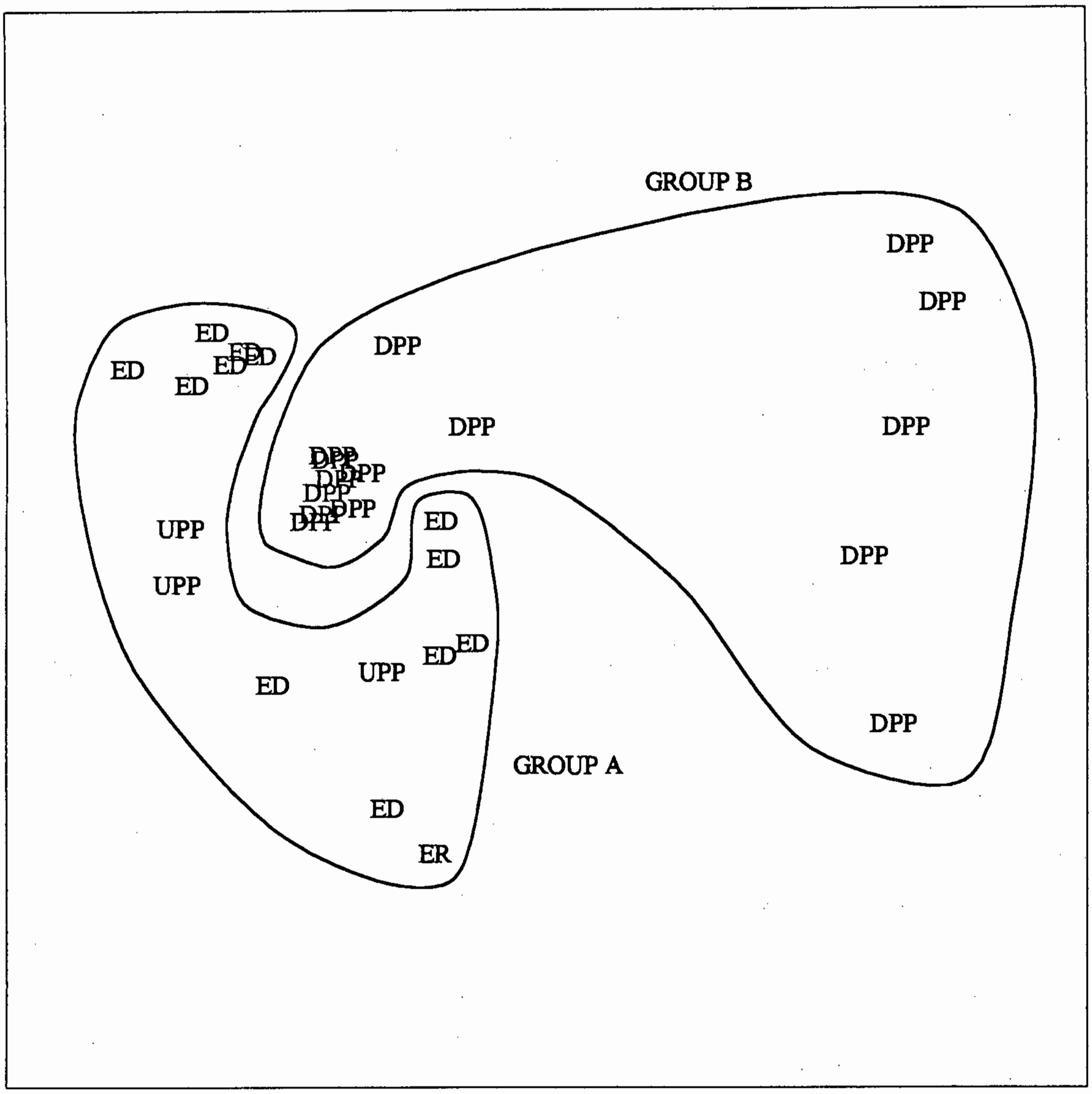

Figure 4.6b PCA of the environmental variables measured during the detailed study and allocated into two groups: Group A, upstream of portapool farm, Elands River and all sites at earthdam farm. Group B, downstream of the portapool farm

Table 4.2 Average abundances of macroinvertebrate taxa in Group A and Group B communities (see Figure 4.6), and the cumulative percentage difference between the groups that each accounts for, using SIMPER (Clarke and Warwick 1994). For each taxa the group with the highest abundance is in bold. Group A and B relate to the groups in Figure 4.6. The data were root-root transformed and analysed using the Bray-Curtis Measure of Similarity

Group A vs. Group B community groupings	TAXON	Average abundance per m^2		Cumulative percentage of difference accounted For
		Group A	Group B	
Average dissimilarity = 50%	Oligochaeta	270	**5050**	15
	Leptophlebiidae	**516**	5	24
	Helodidae	**579**	13	32
	Philopotamidae	**381**	45	39
	Planaria	156	**533**	45
	Elmidae	**170**	16	50
	Baetidae	**2976**	529	56
	Glossosomatidae	**129**	0	62
	Simuliidae	**1019**	265	67
	Chironomidae	1899	**4002**	73
	Hydropsychidae	**176**	1	78
	Ephemerellidae	**713**	53	82
	Notonemouridae	71	**114**	87
	Dryopidae	**57**	0	90
	Leptoceridae	**31**	8	93
	Corydalidae	**6**	0	96
	Heptageniidae	**11**	0	97
	Polycentropodidae	**4**	1	98
	Ceratopogonidae	1	0	99
	Blephariceridae	1	0	99
	Caenidae	2	0	100

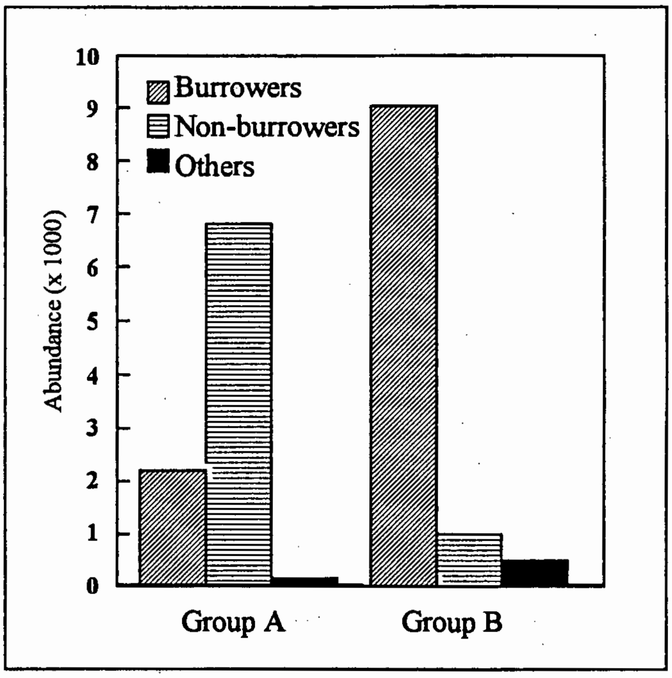

Figure 4.7 The average abundances of the macroinvertebrates in Group A and Group B that inhabit the surfaces of boulders and cobbles relative to those that burrow into the substratum

Table 4.3 Nitrate, phosphate, ammonium, suspended organic material and total suspended solid concentrations, and conductivity associated with the Group A and Group B macroinvertebrate communities. T-Test values were calculated using MS Excel. The percentage significance is provided in brackets after the p-values

Parameter	Group	Mean	Std. Dev.	n	95% confidence limits	T-test p-values
Suspended Organics	Group A	0.79	0.39	8	0.58 - 0.99	
($mg\ l^{-1}$)	Group B	1.65	0.82	6	1.29 -2.00	P = 0.0079 (1%)
Total suspended solids	Group A	2.10	1.17	8	1.44 - 2.69	
($mg\ l^{-1}$)	Group B	3.41	2.22	6	2.46 - 4.38	P = 0.0383 (5%)
Nitrate	Group A	5.30	5.91	8	2.17 - 8.47	
($mg\ l^{-1}$)	Group B	5.70	7.57	6	2.42 - 8.97	P = 0.2027 (ns)
Ammonium	Group A	4.84	3.01	8	8.28 - 12.50	
($mg\ l^{-1}$)	Group B	10.37	4.87	6	3.23 - 6.45	P = 0.0000 (0.1%)
Phosphate	Group A	0.27	0.53	8	0 - 0.56	
($mg\ l^{-1}$)	Group B	1.54	1.20	6	1.02 - 2.06	P = 0.0000 (0.1%)
Conductivity	Group A	22.75	7.61	8	18.69 - 26.80	
($\mu S\ m^{-1}$)	Group B	26.26	9.41	6	22.19 - 30.33	P = 0.0002 (0.5%)

the chemical and physical variables responsible for structuring of the communities at those sites (Clarke and Warwick 1994). In Table 4.3, the physical and chemical conditions in the water at the sites with Group B communities and those with Group A communities differed with respect to the concentration of suspended organic material, phosphate and ammonium. Hence, these three variables may be important in determining macroinvertebrate structure.

The BIO-ENV procedure yielded a maximum positive correlation (weighted Spearman rank correlation (ρ_w) = 0.569) between suspended organic material and macroinvertebrate community structure, indicating that the changes in the concentration of suspended organic material best explained the patterns observed in the macroinvertebrate data. The BIOENV data are presented in Table 4.4.

Table 4.4 The results of the BIOENV procedure, using concentrations of organic suspended solids (OSS) and phosphate PO_4, and measurements of conductivity (COND.). ρ_w = weighted Spearman rank correlation and n = number of variables combined. * denotes which components we included in each procedure.

n	ρ_w	OSS	PO_4	COND.
1	0.569	*		
1	0.411		*	
1	0.208			*
2	0.564		*	*
2	0.432	*	*	
2	0.408	*		*
3	0.506	*	*	*

Table 4.5 gives the percentages of the organic component of the total suspended solids in the effluents of the portapool farm relative to the earthdam farm (from Chapter 3). The percentage of the organic component of the suspended solids was noticeably higher in the effluent of the portapool farm.

Table 4.5 Percentage of the organic component of the total suspended solids in the samples collected from the effluents and downstream sites (B1-B5) of the two trout farms

Farm	Effluent % org (n)	B1 % org (n)	B2 %org (n)	B3 %org (n)	B4 %org (n)	B5 %org (n)
Portapool	60 (4)	70.5 (27)	62 (18)	68 (18)	60 (18)	46 (5)
Earthdam	31(4)	36(6)	54 (5)	29 (5)	37 (5)	52 (1)

4.4 DISCUSSION AND CONCLUSIONS

Distinct changes in macroinvertebrate community structure were recorded downstream of the portapool farm, with the communities downstream of the farm being dominated by non-insect taxa, compared to an upstream community dominated by insect taxa. No significant differences in macroinvertebrate community structure were recorded downstream of the earthdam farm, relative to the upstream communities.

It appears that the changes in the macroinvertebrate community structure downstream of the portapool farm occurred in response to relatively small changes in selected chemical and physical variables, primarily in the amount of particulate organic material suspended in the water. It is worth noting, however, that although the concentration of suspended organic material entering the river from the portapool trout farm appears relatively small, it can result in a substantial increase immediately downstream of the outlet (Chapter 3). Suspended organic concentrations recorded downstream of the portapool farm 2 - 40 times higher than those which would naturally occur in the river.

It has been widely recognised and documented that when solids settle out on a river bed they reduce habitat availability for the benthic macroinvertebrate fauna normally found in the upper reaches of rivers (e.g. Harrison and Farina 1965; Chutter 1969; Reid and Wood 1976; Hellawell 1986), because they coat the stones and impair the attachment mechanisms and normal feeding activities of the stony-bed fauna (Wiederholm 1984). The results of this study support these findings.

sufficiently strong to move all the settled material at the same time. Hence, animals living in the organic deposits were protected from the smaller spates and experienced relatively long periods of homogenous conditions. The implications of this for the structure of the macroinvertebrate community and for biological monitoring using macroinvertebrate fauna is discussed further in Chapters 5 and 6.

In terms of protection of the rivers downstream of trout farms, the results obtained in this study showed clearly that not all trout farms affected the river equally. In fact, although the portapool farm had a significant affect on the structure of the macroinvertebrate community downstream of its effluent outlet, the recorded differences between the structure of the macroinvertebrate communities upstream and downstream of the earthdam farm were negligible. Since both farms fed their fish on a similar diet of pellets, these differences are most likely related to differences in the rates of flow-through of water, where the flow of water through the portapool farms was too fast to allow waste food and faeces to decompose before they were discharged into the river. Differences in the impacts of farms on the macroinvertebrate communities downstream of their effluent outlets were directly linked with the concentration of organic suspended solids in their effluent – the concentration of suspended organics being far higher in the effluent from the portapool farm. Hence, regulation of the amount of organic solids entering the river should mitigate against the sorts of conditions recorded downstream of this farm. Furthermore, the use of settling ponds or some other form of effluent treatment is common practice in trout farms internationally and should be made a compulsory feature of trout farms in the south-western Cape.

No significant impact on the macroinvertebrate community structure was recorded downstream of the earthdam farm. Suspended organic solid levels measured upstream of the farms, and downstream of the earthdam farm, were in the range of 0.58 - 0.99 mg l^{-1}. Similar concentration of particulate organic matter have also been measured in other relatively undisturbed south-western Cape mountain streams, *viz.* mean = 0.8 mg l^{-1} (FRU unpublished data). Levels greater than 1.5 mg l^{-1} have never been recorded in undisturbed south-western Cape mountain streams.

Significant changes in macroinvertebrate community structure were, however, recorded downstream of the portapool farm, where the level recorded was 1.9 mg l^{-1} and the highest level was 2.9 mg l^{-1}.

On the basis of these results, it is suggested that, maintenance of concentrations of particulate organic matter *in the river* below 1.5 mg l^{-1} should protect the integrity of the structure of the macroinvertebrate communities, provided that flows do not drop abnormally low relative to the historical condition. This value is in agreement with those recorded in other studies on the impacts of trout farms (e.g. Allcock and Buchanan 1994).

SECTION IV

METHODOLOGY

CHAPTER 5

DETECTING THE EFFECTS OF ORGANIC POLLUTION IN MOUNTAIN STREAMS: A COMPARISON BETWEEN FIVE METHODS OF ASSESSING CHANGES IN BENTHIC MACROINVERTEBRATE COMMUNITIES

5.1 INTRODUCTION

Much use is being made of monitoring programmes incorporating biological data, in conjunction with physical and chemical variables, to assess the impacts of perturbations on rivers (e.g. Hawkes 1982; Armitage *et al.* 1991, Whitton *et al.* 1991, Lenat and Crawford 1994, Winterbourn and Ryan 1994). Chemical assessment of water quality provides useful information about the nature of effluents entering a system, but chemical surveillance sampling is usually discontinuous. This can result in underestimates (or overestimates) of daily pollutant loads because of temporal fluctuations in effluent quality, and once-off introductions of harmful wastes may be missed. In addition, the number of criteria used to monitor water quality and the number of samples analysed are usually dictated by financial constraints. Some pollutants may thus simply not be analysed. Furthermore, the term *water quality* can only be defined relative to a user: for example, different species of animals and plants have different water quality requirements (Washington 1984). Because of the difficulty of measuring every pollutant likely to be found in a sample of water, and of interpreting results in terms of the severity of impact, it makes sense to turn to the aquatic biota for assistance. The main advantage of a biological approach is that it examines organisms whose exposure to the water (and any pollutants therein) is continuous. Species present reflect the present and the past history of the water quality at a particular point in the river, allowing detection of disturbances that might otherwise be missed. Changes in the composition of the benthic invertebrate community in space and time can often be related to changes in the concentrations of pollutants in the water.

In recent years, the use of community data (i.e., the numerical abundance of each taxon in a community: Washington 1984) for environmental impact studies has gained more widespread acceptance in both marine (Warwick 1993) and freshwater systems (Rosenberg and Resh 1993, Zamora-Muñoz and Alba-Tercedor 1996). Changes in community structure may result from behavioural responses or from shifts in the competitive ability or fecundity of organisms making up that community and are not necessarily dependent on the deaths of organisms. Thus, the use of community data makes intuitive sense, as anthropogenic effects are often subtle, affecting growth and reproduction (Marcucella and Abramson 1978). The acceptance of community-based environmental monitoring has come about partly as a result of the emergence of protocols (Warwick 1993), such as the BACI sampling designs of Green (1979) and Underwood (1993), and various multivariate methods of data analysis (e.g. Field *et al.* 1982, Norris *et al.* 1982, Clarke and Warwick 1994, Dolédec and Chessel 1994) for sampling and analysing communities. In addition, developments in the field of computer hardware and software have contributed considerably to the use of community data (Warwick 1993), since many of the complicated data analyses, once incredibly time-consuming, can now be done at the push of a button. Conversely, these advances have led to analysis for the sake thereof. The remaining bottleneck in the use of biological community data in environmental monitoring is the time-consuming and expensive task of collecting samples and identifying specimens.

Attempts to address the expensive and time-consuming aspects of data collection and identification have resulted in the development of a number of methods for the qualitative rapid assessment of biotic communities (Resh and Jackson 1993). In fresh waters, these approaches typically use macroinvertebrate (e.g. Armitage *et al.* 1983, Lenat 1988, Chutter 1994), algal (e.g. Merschhemke 1991) or fish (e.g. Karr 1981) assemblages as a baseline for assessing problems associated with point and non-point source pollution into rivers and lakes, although some use has also been made of the microbiota (Tuchman and Peterson 1995). Rapid assessments are designed to fulfil two primary criteria: (i) a reduction in the cost of assessing environmental conditions at a site relative to that using conventional quantitative methods, and (ii) summarisation of the results in a manner that is understandable by non-specialists, such as managers and decision-makers (Resh and Jackson 1993). The priority is to collect only the most important information and not collect that which is peripheral to the task at hand. Most often this is done by (i) sampling using a coarser mesh (thereby avoiding excess silt and debris, and very small animals) or only identifying specimens to familial or higher taxonomic levels, (ii) dispensing with abundance data and (iii) presenting the results in the form of a single summary statistic (e.g. Clarke 1990, Chutter 1994).

While there are advantages to using cheaper, quicker methods of assessment to monitor freshwater systems, it is essential that results obtained using those methods accurately reflect changes in the condition of the systems in question, and that the limitations of the method(s) employed are fully explored and understood. This chapter attempts to address some of the problems that are faced in the bid to minimise the time and financial costs of environmental monitoring. At the same time, it aims to assess the usefulness to freshwater studies of some of the advances being made in the marine sciences. To make the chapter more focussed, only the methods believed to be the most appropriate for illustrating each aspect were chosen.

This chapter concentrates on the results obtained from the samples collected at the portapool farm presented in Chapter 4. The conclusions from those data were that the levels of some, but not all, of the physical and chemical variables measured increased downstream of the effluent outlet relative to upstream; that there was a difference in macroinvertebrate community structure immediately downstream of the effluent outlet than that upstream; that despite a change in community structure with distance downstream from the outlet, full recovery to upstream conditions did not occur; and that there was no significant improvement in water quality with distance downstream of the outlet within the total study distance. In this chapter, these data were used to assess the following aspects:

1. The effects of decreasing taxonomic resolution (and therefore decreasing requirements of time and expertise). Hence, the efficacy with which different levels of taxonomic resolution could detect change were assessed using the following criteria:
   - the degree of difference between the upstream site and the site immediately downstream of the effluent outlet
   - the degree of 'recovery' to upstream conditions with distance downstream of the effluent outlet
   - the extent of congruence between the pattern for the grouping of sites and that obtained for environmental variables *unlikely* to be affected by the presence of the trout farm. This was to assess whether the differences in community structure were related to the presence of the trout farm or if they were a response to differences in environmental variables that were unrelated to the farm.

2. The relative performance of five measures of community stress calculated from the data in Chapter 4. As with taxonomic resolution, the performance of the techniques was assessed using the following criteria:
   - the degree of difference between the upstream site and the site immediately downstream of the effluent outlet
   - the degree of 'recovery' to upstream conditions with distance downstream of the effluent outlet.

   In addition, the trends between the sites, obtained from calculating each measure of community stress for each site, were compared to the trends obtained for the physical and chemical variables which were shown in Chapter 3 to differ significantly upstream and downstream of the portapool farm. Those measures of community stress which match the physical and chemical data most closely were considered to have performed best.

   Since the data used in this chapter are from a single farm, they lack generality and as such are not an ideal basis for rigorous evaluation and comparison of methods. Nonetheless they illustrate some of the potential problems encountered with the use of measures of community stress.

5.2 METHODS

5.2.1 The data set

The physical, chemical and macroinvertebrate data used are the same as presented for the summer sampling at the portapool farm in Chapters 3 and 4, respectively. These data were chosen because they illustrated most clearly the differences between samples collected upstream and downstream of the trout farm, and between samples collected at different distances downstream of the trout farm. The methods of collection and analysis are explained in Chapter 2. The location and characteristics of the farm is described in Chapter 3, as are the locations of the study sites.

In this chapter the sites were numbered as follows:

Site P1	Upstream of the influence of the farm
Site P2	Immediately downstream of the effluent outlet

Site P3	50 m downstream of the effluent outlet
Site P4	200 m downstream of the effluent outlet
Site P5	500 m downstream of the effluent outlet
Site P6	1000 m downstream of the effluent outlet.

5.2.2 Taxonomic resolution

The effects of decreasing taxonomic resolution were assessed using non-metric multidimensional scaling ordination (MDS), using a Bray-Curtis measure of similarity (Bray and Curtis 1957). The data sets consist of the same abundance data, grouped at three different taxonomic levels, *viz.* order, family and species. Before ordination, the data were root-root transformed to downplay the influence of dominant taxa on the final ordination (Clarke and Warwick 1994). Dispensing with abundances saves time and therefore money, and so the same analyses were also performed using the binary (presence/absence) data for each taxonomic level.

The MDS plot obtained for each taxonomic level was tested against environmental variables that were unlikely to change as a result of the presence of the trout farm. The two plots were compared to see whether they grouped the sites in a similar fashion (Clarke and Warwick 1994). The variables chosen were depth, flow velocities, substratum composition, number of upstream tributaries and temperature. For the environmental variables, the MDS ordination was performed using Euclidean Distance as a measure of dissimilarity between the samples collected at each site.

5.2.3 Measures of community stress

Five measures of community stress were examined. They ranged from those requiring quantitative sampling and species-level identification to those requiring only qualitative sampling and an ability to distinguish between taxa. The techniques examined are listed in Table 5.1, roughly in order of decreasing time, effort and expertise required.

Meta-analysis refers to the combined analysis of a range of individual case studies which in themselves are of limited value but which in combination provide a more global insight into the problem under investigation (Warwick and Clarke 1993)

Warwick and Clarke (1993) presented an example of phylum-level meta-analysis using macroinvertebrate community data collected at unimpacted and impacted sites on the north-east Atlantic shelf. They suggested that the relative severity of a perturbation in that area could be assessed by comparing data collected when investigating the "new" impact with existing data collected when investigating previous impacts. The existing data represent communities where the pollution/disturbance status is known and are the 'training data set', and can be used as a template against which community data from sites of unknown status can be assessed.

Table 5.1 A list of the five techniques examined in this chapter. The techniques are listed in roughly decreasing costs in terms of time, effort and expertise.

Rank	Technique	Requirements
1	Family-level meta-analysis of macroinvertebrate communities	Quantitative sampling; a large dataset from similar systems; family-level identification; sophisticated analytical techniques.
2	Abundance/biomass plots	Quantitative sampling; measurements of biomass.
3	Shannon-Wiener Diversity	Quantitative sampling, distinction between taxa (but no identification).
4	South African Scoring System	Qualitative sampling; family-level identification.
5	EPT Taxa Richness	Qualitative sampling, order-level identification thereafter distinction between different taxa (but no identification).

Using this concept, a "training data set" was created for mountain streams in the south-western Cape. This was done by performing a family-level meta-analysis of macroinvertebrate communities recorded during this and other studies (Dallas 1995, Tharme, unpublished data). The following data were used:

- summer data presented for the earthdam farm in Chapter 4. These sites were as follows:

Site E1	upstream of the influence of the farm
Site E2	immediately downstream of the effluent outlet
Site E3	50 m downstream of the effluent outlet
Site E4	200 m downstream of the effluent outlet
Site E5	500 m downstream of the effluent outlet
Site E6	1000 m downstream of the effluent outlet.

- data collected from the Berg River, south-western Cape (labelled HD: Dallas 1995)
- data from the Elands River. These data were collected at a relatively undisturbed site situated approximately 3 kilometres downstream of its confluence with the Kraalstroom River on which the portapool farm is situated (labelled RT: Rebecca Tharme, Freshwater Research Unit, University of Cape Town, unpublished data).

The training data set was used as a template to compare the extent to which communities recorded at sites situated downstream of the portapool trout farm differed from those in undisturbed rivers.

Abundance/biomass plots (ABC curves)

Ranked abundance and biomass curves (ABC curves) are based on the ranking of taxa (in this case families), in increasing order of total abundance and in increasing order of total biomass per taxon. The ranked abundances and biomasses, expressed as a percentage of total abundance or biomass of all taxa, are plotted against the relevant taxon rank. Log transformations of one or both axes can be used to downweight or emphasise different sections of the curves. Logging (log_{10}) the x-axis allows the distribution of the more common taxa to be displayed more clearly (Warwick 1986; Clarke and Warwick 1994).

ABC plots have proved useful as measures of the pollution status of marine macrobenthic communities (Warwick *et al.* 1987), particularly in instances where polychaete species are dominant (Warwick and Clarke 1994). ABC plots are potentially extremely useful in freshwater studies because they provide some idea of the functional response of macroinvertebrates to organic pollution, something that most indices fail to do. The plots show the comparison between the abundance and biomass curves of species ranked in order of importance on the x-axis and percentage dominance on the y-axis.

In marine systems, under stable conditions of infrequent disturbance, K-selected species (large bodied, long lived) often dominate the macrobenthic fauna in terms of biomass, while r-selected (small-bodied, short lived) species are numerically dominant. Hence, the dominance curve for biomass will lie above the curve for abundance for its entire length. Under heavily-polluted conditions, however, small opportunistic species are favoured and the macrobenthic community present is often dominated both numerically and in terms of biomass by r-selected species. Hence, the k-dominance curve for abundance will lie above the curve for biomass. Conditions of intermediate pollution would result in a plot in which the biomass and abundance curves crossed one another (Clarke and Warwick 1994).

Biomass values were calculated from triplicate samples containing a known number of individuals from single taxa. The samples were air-dried and weighed, and the total mass divided by the number of individuals in the sample to obtain a mean individual biomass. To obtain the biomass for a particular site, the mean number of individuals of that taxon recorded at a site was multiplied by the individual biomass calculated for that taxon. These values, together with the mean number of individuals recorded at a site (Appendix A), were used to produce the biomass ABC curves.

Species diversity

The species diversity for each site was calculated using the Shannon-Wiener diversity index (H'), where:

$$H' = -\sum_{i=1}^{k} p_i \log p_i$$

H' has been criticised as an index for many years and Washington (1984) dubbed it a "dubious index" as far as its biological relevance is concerned. Nevertheless, it has been, and continues to be, used extensively throughout the biological world, especially amongst ecologists (Washington 1984) because it is easy and quick to use and requires no taxonomic identification.

The South African Scoring System (SASS)

In recent years in South Africa, the chosen method for rapid bioassessment of rivers has most often been the South African Scoring System (Chutter 1994), currently referred to as SASS (Dallas 1995). It is has also been suggested that SASS be used as the basis of a new South African National Biomonitoring Programme (Uys *et al.* 1996). SASS is a field-based, rapid bioassessment method that uses family-level information on benthic macroinvertebrates to assess the degree of impairment of water quality in rivers (Dallas 1995). It is based loosely on the British Monitoring Working Party (BMWP) scoring system (Armitage *et al.* 1983). Tolerance/sensitivity scores allocated to different taxa in the BWMP formed the foundation for SASS, which was then modified to take account of those families that are found in South Africa but not Britain. The taxon "scores" that are currently used in SASS are subjective, and are based on the opinion of several South African freshwater ecologists (Dallas *et al.* 1994). SASS requires that all the available biotopes at a given site are sampled using a kick-net with a 900 μm mesh size. The animals are collected in the net, identified on site to the taxonomic level of family, and then returned to the river. SASS results are reported either as a Total Score, calculated by summing the individual taxon scores, or as an Average Score per Taxon (ASPT), calculated by dividing the Total Score by the number of taxa. The higher the Total Score or ASPT obtained for a particular river, the better the water quality is deemed to be. A low Total Score may indicate impairment of habitat, or water quality.

EPT taxa richness

Ephemeroptera+Plecoptera+Trichoptera taxa richness is widely used as a measure of the physical and chemical properties of water (e.g. Quinn & Hickey 1993, Lenat and Crawford 1994). EPT taxa richness is calculated by counting the number of species belonging to Ephemeroptera, Plecoptera and Trichoptera.

5.2.4 Judging performance of the techniques

The performance of each method was assessed relative to the first principal component of a *correlation-based* Principal Component Analysis (PCA; Clarke and Warwick 1994) on a set log-transformed physical and chemical data. This is essentially the ordering of the samples/sites according to the overall levels of the different physical and chemical variables recorded for each one. It serves as a useful summary of the inverse of the pollution levels at each of the sites.

In this instance, the variables chosen were total suspended solids, organic suspended solids, phosphate, nitrate and ammonium concentrations, conductivity and pH. These variables were shown to increase downstream of a trout-farm effluent outlet (Chapter 3). Temperature was added to the data set because it influences so many other variables.

The feeding habits of the taxa inhabiting each site on the river were also taken into account when assessing the performance of the different techniques. Most aquatic insects can be grouped into general categories or functional feeding groups (after Cummins 1973, Cummins and Klug 1979). In this chapter, the following functional groups were identified (after Palmer 1991):

predators	feeding on other consumers
scrapers	eating periphyton, including algae
shredders	striping, boring or chewing CPOM (coarse particulate organic material)
collectors	feeding on deposits of UFPOM and FPOM (ultra-fine and fine particulate organic material)
detritivores	living buried in and feeding on deposits of UFPOM and FPOM (note: the term detritivores as it is used here excludes animals living buried in and feeding on CPOM, e.g. collectors and shredders)
filterers	passive filtering of suspended UFPOM and FPOM
brushers	active filtering of UFPOM and FPOM by 'brushing' it off submerged surfaces (McShaffrey and McCafferty 1988).

Although there is evidence to suggest that there is some overlap between the groups, i.e. some filter feeders also graze (e.g. Chessman 1986), the distribution of animals belonging to each of these feeding groups provides insight into the food-sources and abundance in a stream. In very general terms, detritivores would be expected to be dominant in the lower depositional zones of a stream, and predators, scapers and shredders in the upper zones (*sensu* Vannote *et al.* 1980).

5.3 RESULTS

5.3.1 The data set

The raw data are presented in Appendix A. At the site immediately downstream of the effluent outlet, there was an elimination or great reduction in the number of Limnichidae (Sp 1), Helodidae (*Prionocyphon* spp.), Plecoptera (*Aphanicercus* sp. and *Aphanicercella* sp.), Elmidae, Heptageniidae (*Afronurus capensis*) and Emphemerellidae (*Lestagella* sp., *Lithogloea* sp. and *Ephemerellina* sp.), relative to the upstream situation. These taxa were replaced with Naididae (*Nais* sp.), Lumbriculidae, Chironomidae (*Chironomus* sp.) and planarians (*Dugesia* sp.). One kilometre downstream of the effluent, the community consisted of primarily *Nais* sp. and *Rheotanytarsus* sp. There were also large numbers (relative to the upstream condition) of the ephemeropteran species *Baetis harrisoni* and *Acentrella capensis*, and the trichopteran, *Cheumatopsyche* sp.

Table 5.2 gives the functional feeding group to which families were allocated. The community upstream of the trout farm was dominated by scrapers, grazers and shredders. Immediately downstream of the effluent outfall, the community was dominated by detritivores. The community recorded one kilometre downstream of the effluent outlet (Site P6) differed from both the upstream community (Site P1) and that immediately downstream of the effluent (Sites P2 and P3). It consisted of detritivores (*Nais* sp.), predators (*Rheotanytarsis* sp.), filter-feeders (*Cheumatopsyche* sp.) and opportunistic brusher species (*Baetis* spp. and *Acentrella capensis*).

Table 5.2 Functional feeding groups to which families recorded at the portapool trout farm were allocated

Order	Family	Genus	Functional feeding group	Reference
Diptera	Simuliidae		Filter feeder	Hildrew *et al.* (1984)
	Rhagionidae		Predators	Pennak (1978)
	Chironomidae	*Rheotanytarsis* sp.	Predator	Hildrew *et al.* (1984)
		Chironomus sp.	Detritivore	Hynes (1960)
	Blephariceridae		Grazers	Pennak (1978)
Plecoptera		*Aphanicercella* sp.	Shredders	Hynes (1984)
Ephemeroptera	Leptophlebiidae	Choroterpes sp.	Brusher	Palmer (1991)
		Castanophlebia sp.	Brusher	-
		Aprionyx	Brusher	-
		Adenophlebia sp.	Brusher	Palmer (1991)
	Caenidae		Gatherer	Palmer (1991)
	Baetidae	Beatis harrisoni	Brusher	Palmer (1991)
		Acentrella capensis	Brusher	Pers. obs.
	Ephemereliidae		Brusher (?)	McShaffery & McCafferty (1988)
	Heptageniidae		Scraper	Pennak (1978)
Trichoptera	Hydropsychidae	*Chematopsyche* sp.	Filter feeder	Palmer (1991)
	Hydroptilidae		Scraper	Pennak (1978)
	Phyolopotamidae	*Chimarra* sp.	Predator	Pennak (1978)
	Phylopotamidae	Excl. *Chimarra* sp.	Filter feeders	Pennak (1978)
	Glossosomatidae		Scraper	Pennak (1978)
	Polycentropodidae		Filter feeders	Pennak (1978)
	Leptoceridae		Grazers	Pennak (1978)
	Others		Shredders	Hynes (1984)
Coleoptera	Helodidae	*Prionocyphon* sp.	Grazers	Pennak (1978)
	Limnichidae		Detritivores	Pennak (1978)
	Dytiscidae		Predator	Hildrew *et al.* (1984)
	Dryopidae		Scrapers	Pennak (1978)
	Elmidae		Grazers	Pennak (1978)
	Helodidae		Grazers	Pennak (1978)
Odenata	Aeshnidae	*Aeschna* spp.	Predators	Pennak (1978)
Hydracarina			Predators	Pennak (1978)
Turbellaria		*Dugesia* sp.	Detritivores	Pennak (1978)
Megaloptera	Corydalidae		Predators	Hynes (1984)
Annelida	Hirudinidae		Detritivore	Pennak (1978)
	Naididae		Detritivore	Pennak (1978)
	Lumbriculidae	*Lumbriculus* sp.	Detritivores	Hynes (1960)

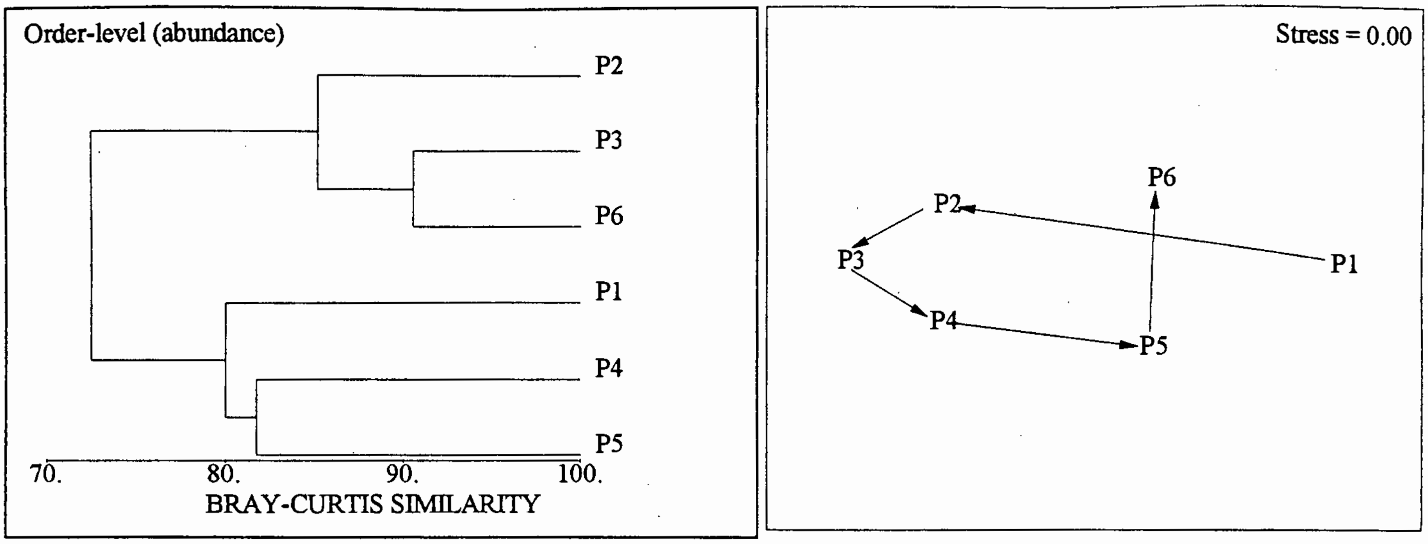

Figure 5.1 Non-metric multidimensional scaling ordination of ordinal abundance data. The arrows indicate the direction of flow

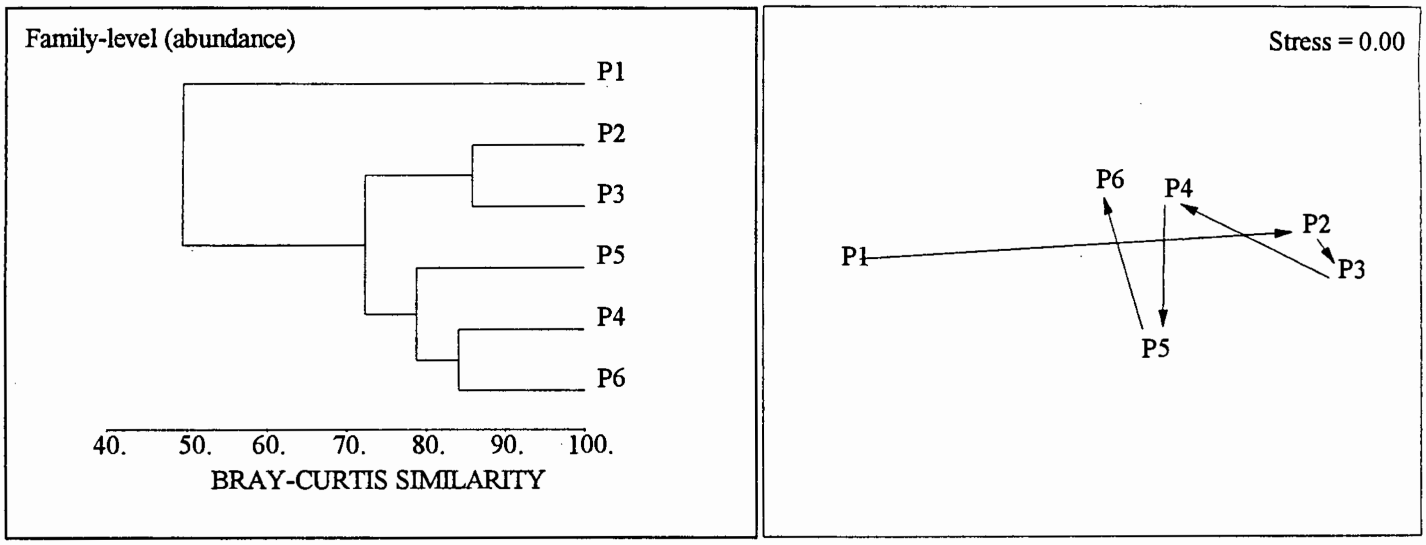

Figure 5.2 Non-metric multidimensional scaling ordination of familial abundance data. The arrows indicate the direction of flow

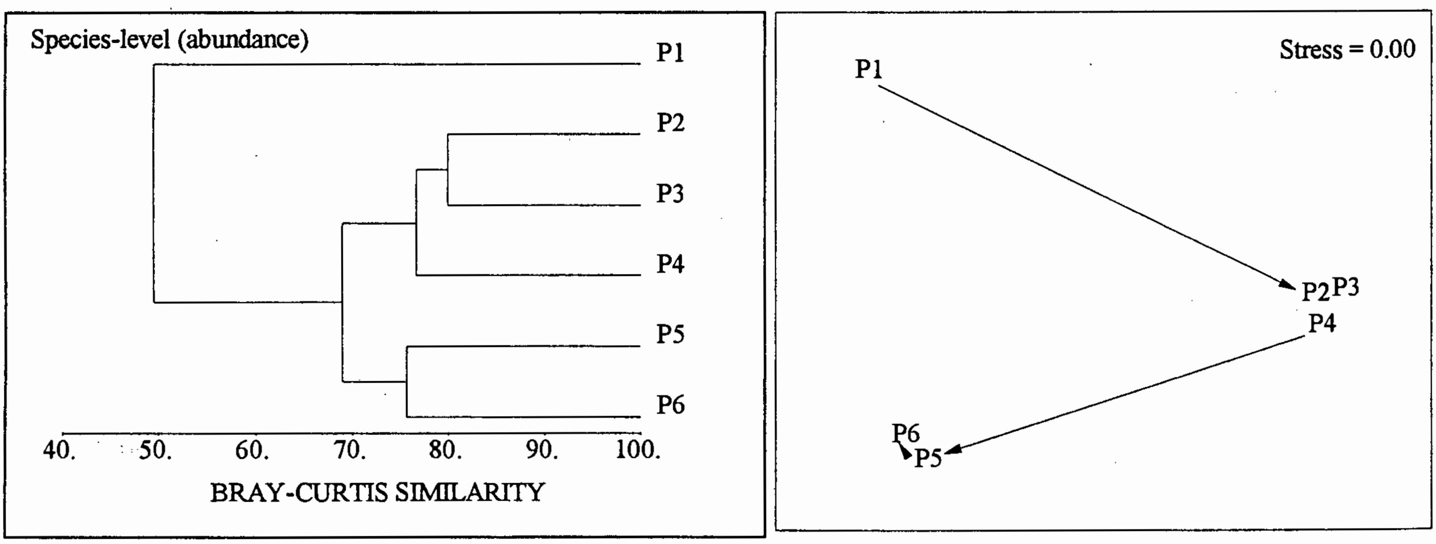

Figure 5.3 Non-metric multidimensional scaling ordination of species-level abundance data. The arrows indicate the direction of flow

5.3.2 Taxonomic level

Abundance data

The MDS ordinations and dendrograms using abundance data at the taxonomic level of order, family and species are depicted in Figures 5.1 - 5.3, respectively. All the plots showed a difference in community composition immediately downstream of the effluent outfall relative to that upstream. At the taxonomic level of order, however, the upstream site P1 was 70-80 % similar to the downstream sites. In Figures 5.2 and 5.3, the similarities between the sites were far lower (*c.* 50%).

The MDS plot of the environmental variables likely to be unaffected by the trout farm is presented in Figure 5.4a, and that for those likely to be affected by the farm in Figure 5.4b. The pattern obtained using these four "unaffected" environmental variables was most similar to that obtained using the species-level data (Figure 5.3). This suggests that these variables were at least partly responsible for the pattern obtained in the species plot, i.e. changes at the species level are sensitive to natural variation in addition to anthropogenic variation.

Binary data

The MDS ordinations using binary data at the taxonomic level of order, family and species are depicted in Figures 5.5 - 5.7, respectively. The binary data for the ordinal identification indicated little difference between Sites P1 and P6, i.e., 85 % similarity between the sites. The plot using familial binary data was similar to that obtained using familial abundance data but the species-level binary plot discriminated less between the sites (especially Sites P1 and P6) than did the species-level abundance plot.

Figure 5.8 is an MDS plot using the family-level data and 'ranked abundance' data. The abundances of individuals in each family were ranked as follows:

1 = 1 - 10

2 = 10 - 100

3 = 100 - 1000, and

4 = >1000.

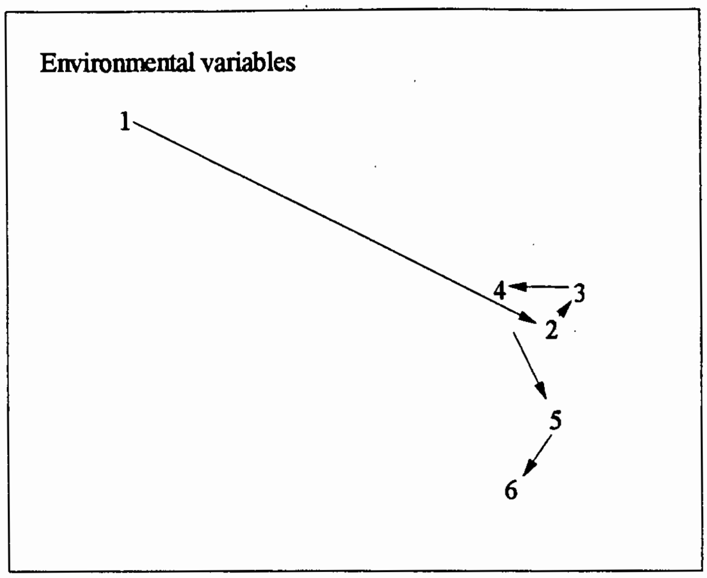

Figure 5.4a Non-metric multidimensional scaling ordination of four environmental variables likely to be unaffected by the presence of the trout farm. The arrows indicate the direction of flow.

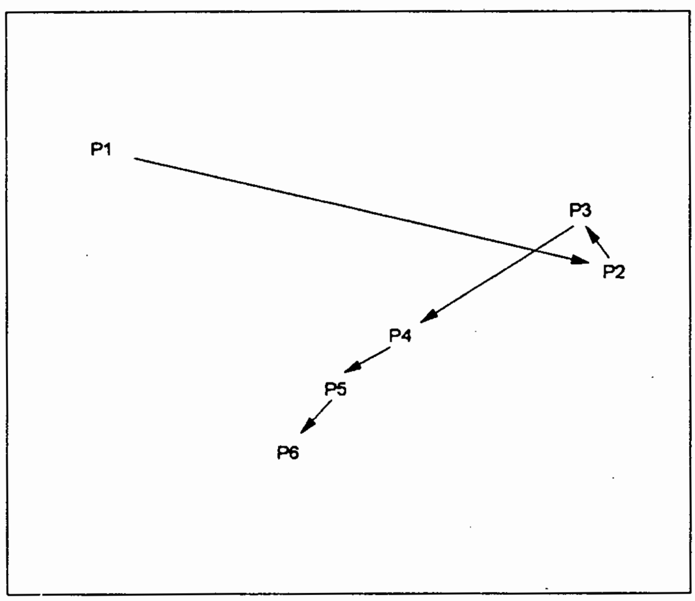

Figure 5.4b PCA for environmental values measured upstream and downstream of the portapool farm

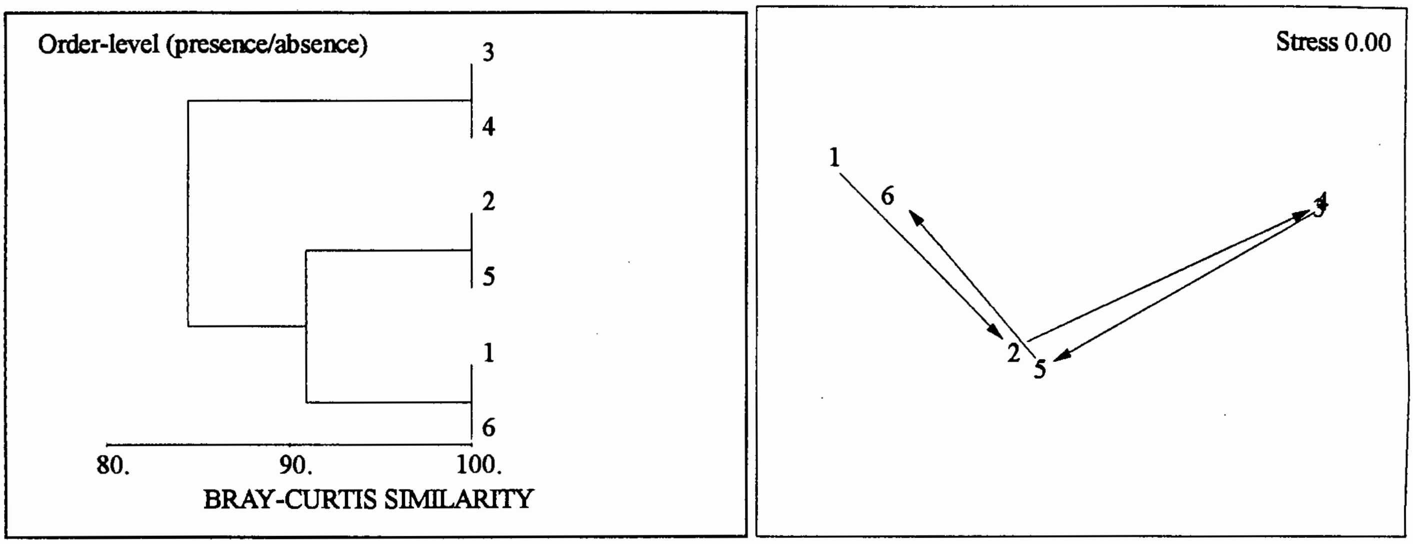

Figure 5.5 Non-metric multidimensional scaling ordination of the ordinal presence/absence data. The arrows indicate the direction of flow.

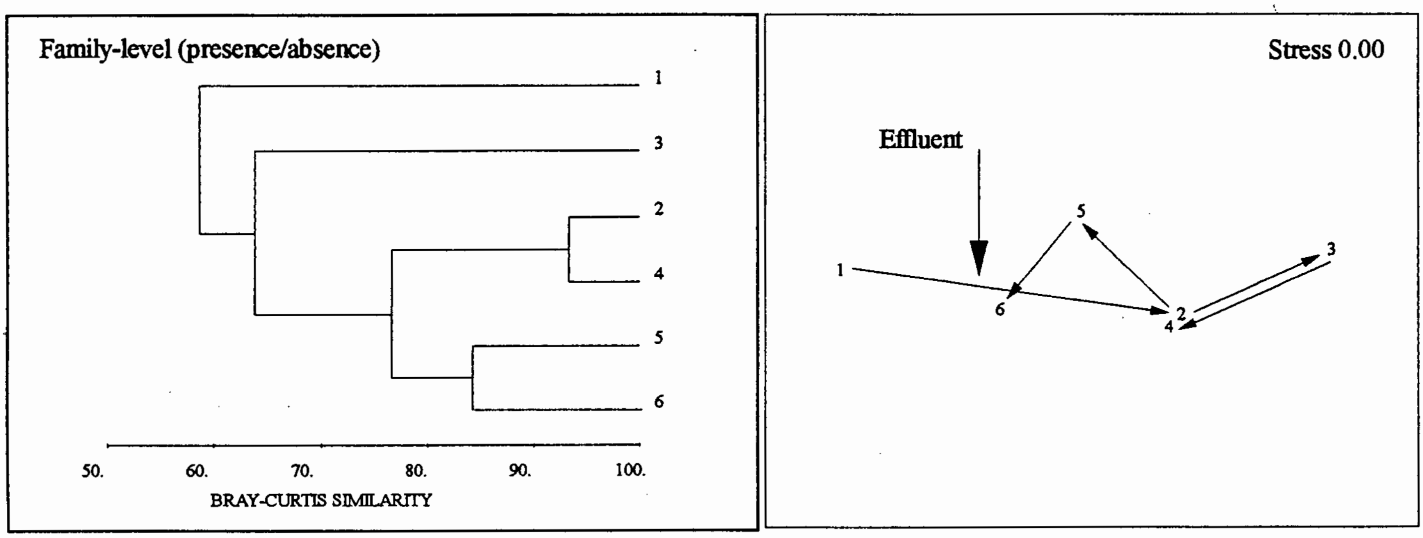

Figure 5.6 Non-metric multidimensional scaling ordination of the familial presence/absence data. The arrows indicate the direction of flow.

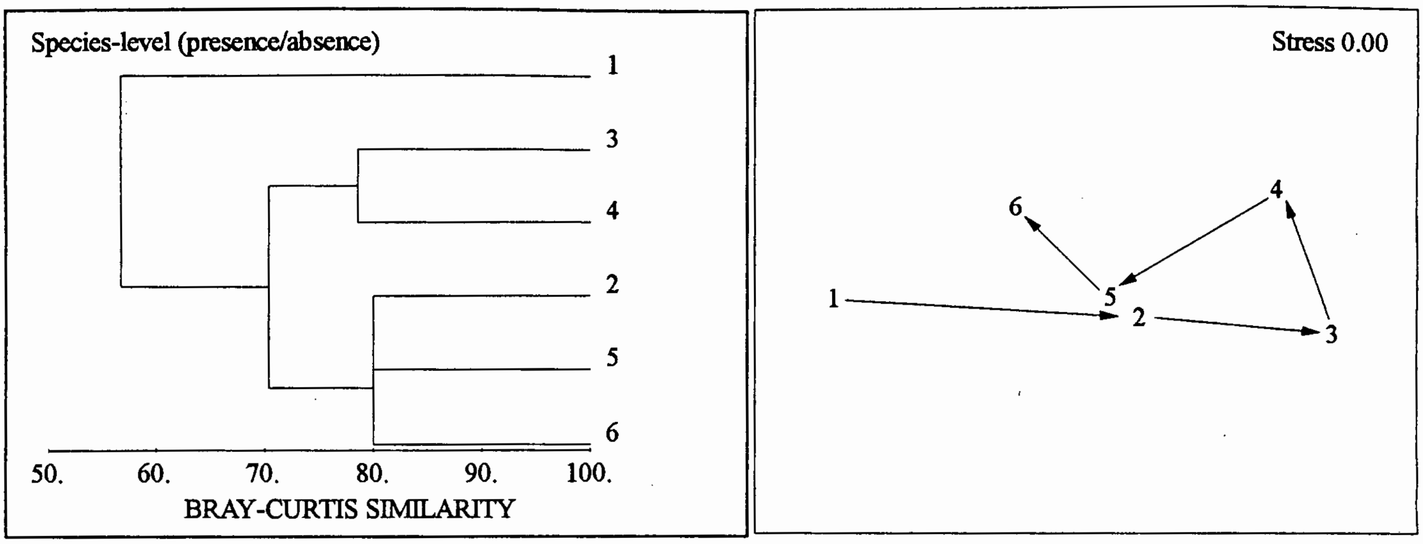

Figure 5.7 Non-metric multidimensional scaling ordination of the species-level binary data. The arrows indicated the direction of flow.

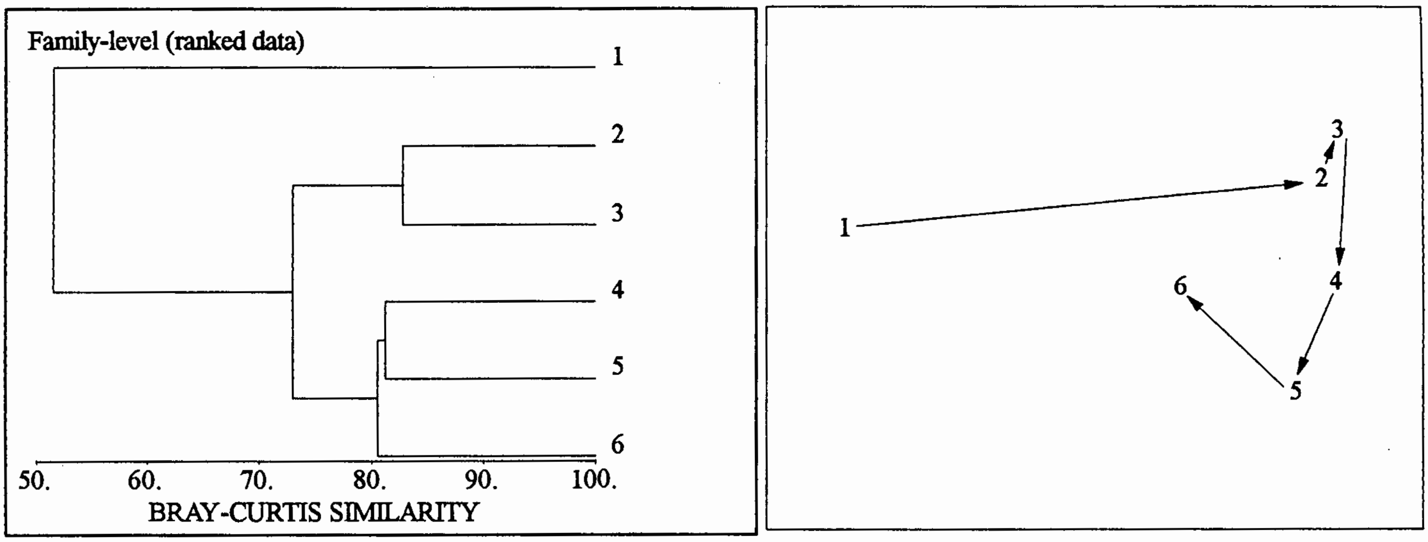

Figure 5.8 Non-metric multidimensional scaling ordination of family-level ranked data. The arrows indicated the direction of flow.

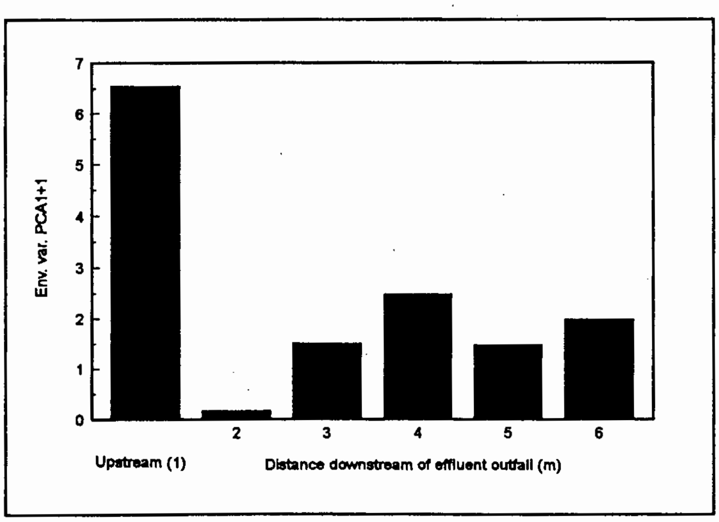

Figure 5.9 Pollution loadings for the sites upstream and downstream of the portapool farm represented by the first principal component of a correlation-based PCA on a set of log-transformed environmental data

It is interesting to note that the differences between the sites are as more marked using the family-level ranked data than they are using either the family-level abundance data.

5.3.3 Monitoring techniques

The differences in pollution loadings depicted in Figure 5.9 give the first principal component of a *correlation-based* PCA (Clarke and Warwick 1994) on the set log-transformed environmental data.

Family-level meta-analysis of macroinvertebrate communities

The combined MDS analysis for the "training set data" plus the data obtained downstream of the portapool farm is presented in Figure 5.10. The connected sequence of samples on the right-hand side of the plot were collected upstream (E1) and at increasing distances downstream (E2 to E6) of the earthdam trout farm. Chapters 3 and 4 have shown that the effect of this farm on its downstream river was minimal. The other connected sequences are the data collected upstream (P1) and downstream (P2-P6) of the portapool farm. The points marked RT and HD represent the data obtained from Tharme (unpublished data) and Dallas (1995), respectively. In Figure 5.10, the log axis represents a scale of disturbance, with the samples collected from the most disturbed reaches of river to the left.

Abundance/biomass plots (ABC curves)

Table 5.3 provides a list of mean individual biomass values used for the invertebrate taxa recorded in the river affected by the portapool farm.

The ABC curves for the species-level macroinvertebrate data collected at the portapool farm are presented in Figure 5.11. The relationship between the biomass and abundance curves is the opposite to that found in marine studies. In undisturbed marine systems the curve for abundance would be expected to lie below the curve for biomass. However, in this study the sample collected at the undisturbed upstream site, showed a dominance curve for abundance that lay above the biomass curve. In the samples collected from the grossly polluted sites (0, 50 and 200 m downstream of the

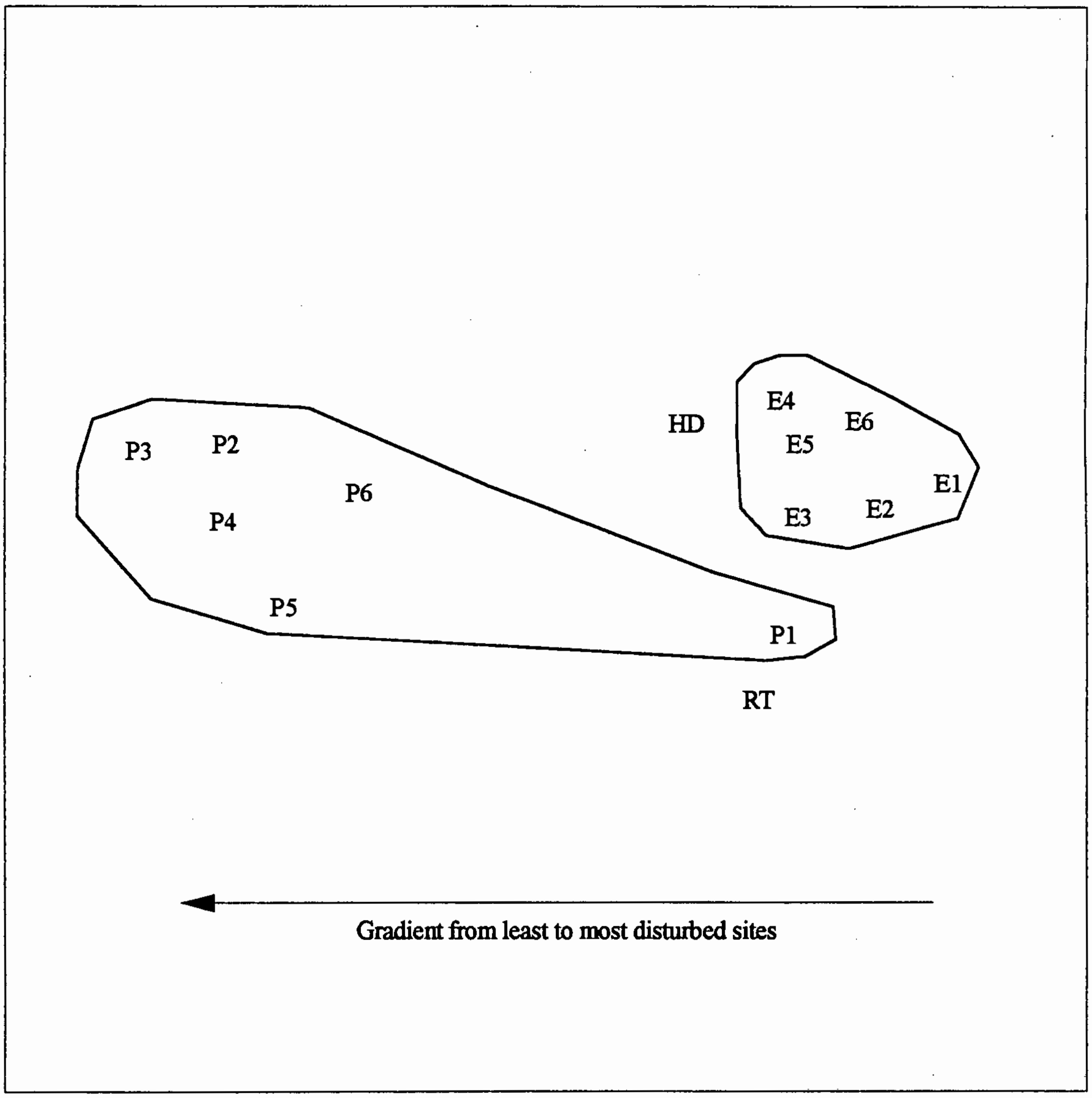

Figure 5.10 A family-level meta-analysis of macroinvertebrate communities recorded at 14 sites situated on mountain streams in the south-western Cape. Sites P1 to P6 are situated on the river affected by the portapool trout farm, Sites E1 to E6 are situated on the river affected by the earthdam farm, Site HD is situated on a relatively undisurbed section of the Berg River and Site RT is situated on relatively undisturbed section of the Elands River.

Table 5.3 An abbreviated list of macroinvertebrate species recorded at the trout farm and their individual biomass (n = 3 weightings of bunches of individuals). Those taxa not recorded upstream of the farm are marked with a cross

Name	Average biomass (g individual^{-1})	Recorded upstream?
Castanophlebia sp.	0.001	✓
Aprionyx sp.	0.001	✓
***Lumbriculus* spp.**	**0.0007**	×
Adenophlebia sp.	0.0006	✓
Aphanicerca spp.	0.0005	✓
***Chironomus* sp.**	**0.0004**	×
Afronurus sp.	0.0004	✓
Cheumatopsyche sp.	0.0004	✓
Athericidae	0.0002	✓
Dryopidae	0.0002	✓
Prionocyphon spp.	0.0002	✓
Aphanicercella sp.	0.0002	✓
Aeschna spp.	0.0002	✓
Chimarra sp.	0.0002	✓
Dugesia spp.	0.0002	×
Lestagella sp.	0.0001	✓
Lithogloea sp.	0.0001	✓
Ephemerellina sp.	0.0001	✓
Baetis spp.	0.00008	✓
Acentrella sp.	0.00005	✓
Elmidae	0.00005	✓
Chironomidae (excl. *Chironomus* sp.)	0.00009-0.00002	✓
Simulium sp.	0.00003	✓
Nais sp.	0.000007	×

effluent outlet), the curve for biomass shifted position to above the curve for abundance.

A similar reversal of the expected pattern was obtained using data collected on macroinvertebrates in the Eerste River, an undisturbed mountain stream near Stellenbosch (King 1982, Figure 5.12 a&b). Once again, for the mountain-stream community, the k-dominance curve for abundance lay above the curve for biomass.

Shannon-Wiener Diversity

The species diversities for each site were calculated using the Shannon-Wiener diversity index (H'; Figure 5.13). There was good agreement between the trend obtained for the Shannon-Wiener diversity (H') and that obtained for the pollution loadings. However, H' did not distinguish clearly between the undisturbed upstream and the disturbed downstream sites (from Figure 5.10). This was particularly true for the site furthest downstream. This is contrary to the results obtained in Chapter 4, which indicated that recovery to upstream conditions did not occur.

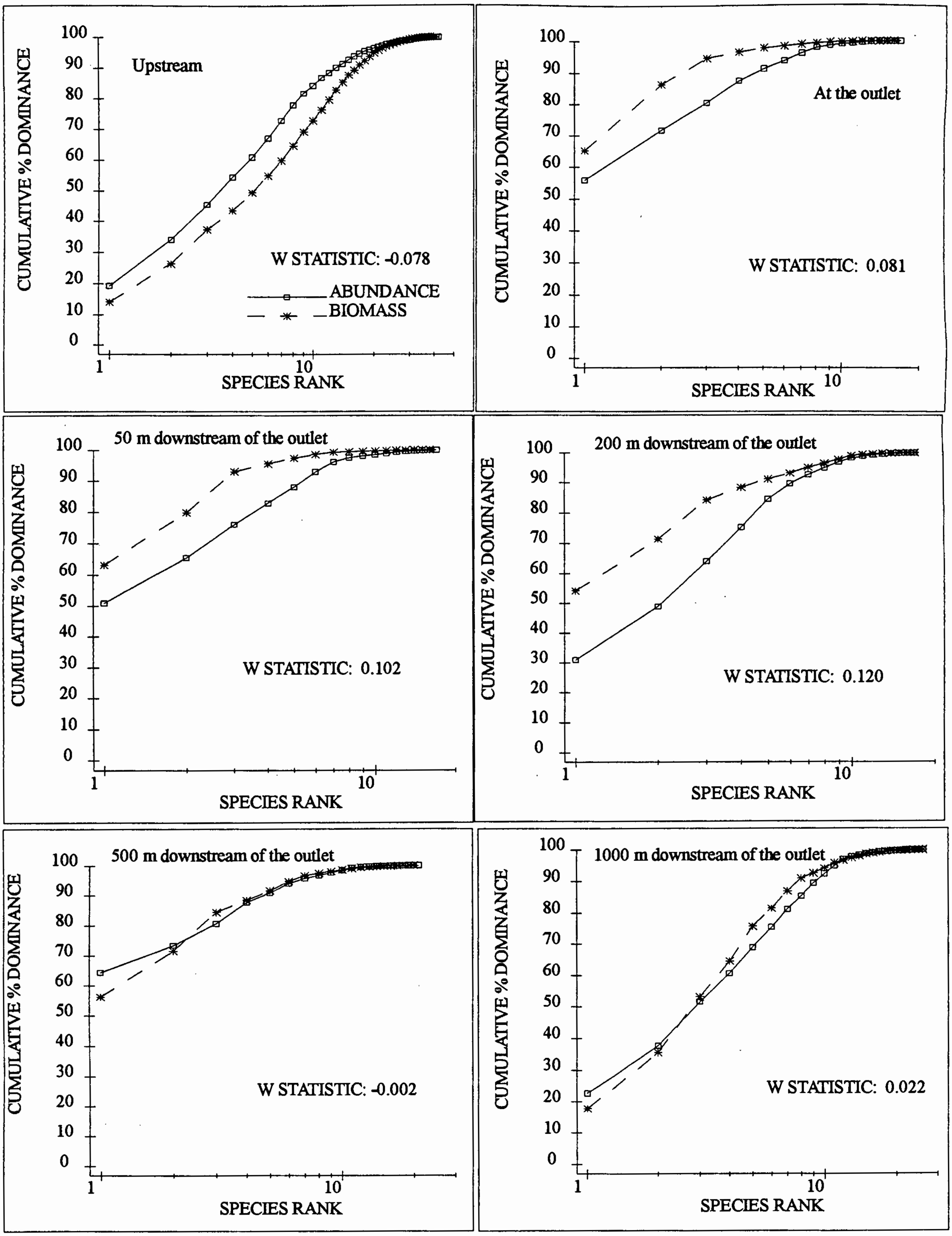

Figure 5.11 ABC curves for species-level data collected upstream and downstream of the portapool at the trout farm.

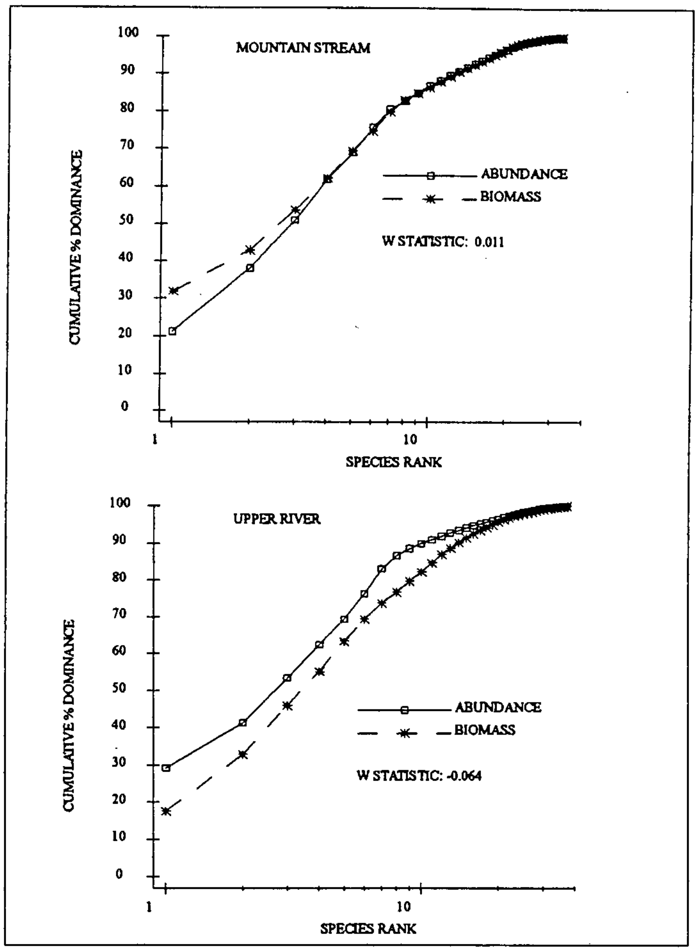

Figure 5.12 ABC curves for macroinvertebrate data collected in the relatively undisturbed upper reaches of the Eerste River, near Stellenbosch, south-western Cape, South Africa (from King 1982)

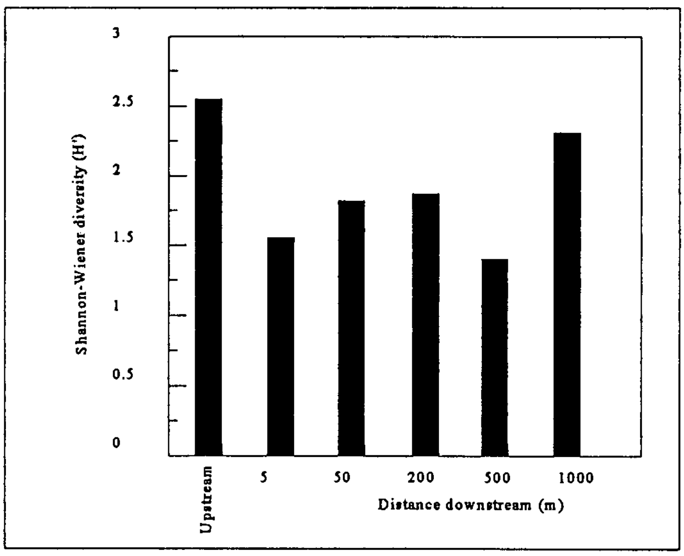

Figure 5.13 Shannon-Wiener diversity for the sites upstream and downstream of the portapool trout farm (Chapters 3 and 4)

The South African Scoring System (SASS)

SASS Total Scores and ASPTs are presented in Figures 5.14 a & b, respectively. Both of these showed good agreement with the changes in pollution loadings (Figure 5.9). SASS Total Scores and ASPTs were also positively correlated with distance downstream of the effluent outlet (r^2 = 0.82; d.f. = 16 and r^2 = 0.60; d.f. = 12, respectively). This indicates recovery to a less disturbed community.

The SASS method, which requires that all available habitats be sampled and combined in a single score, resulted in about 30% more families per sample being recorded than the number recorded using the box-sampling technique. The additional families included Ancylidae, Empidae, Corixidae, Gerridae, Veliidae, Ecnomidae, Culicidae, Athericidae, Muscidae and Brachyura. The greatest discrepancies occurred at the sites 500 and 1000 m from the effluent outlet (# SASS = 4 and # box sampler = 5.7). At the site immediately below the effluent outlet, the box sampler recorded more families than SASS (# SASS - # box sampler = -1.3).

Ephemeroptera/Plecoptera/Trichoptera (EPT) taxa richness

EPT scores for the sampling sites at the trout farm are provided in Figure 5.15. The EPT scores distinguish clearly between the undisturbed upstream and the disturbed downstream sites (from Figure 5.9). They also tracked the changes in pollution loadings well.

5.4 DISCUSSION

A biological monitoring technique should indicate whether an anthropogenic disturbance is taking place and provide some indication of how severe that disturbance is. It should also indicate the extent of the disturbance. To do that, it should be able to detect subtle changes in response with increasing spatial or temporal distance from the disturbance or the introduction of mitigatory measures.

In this chapter, the "biological reality" against which the performance of the techniques was judged is the degree of the difference between the macroinvertebrate community

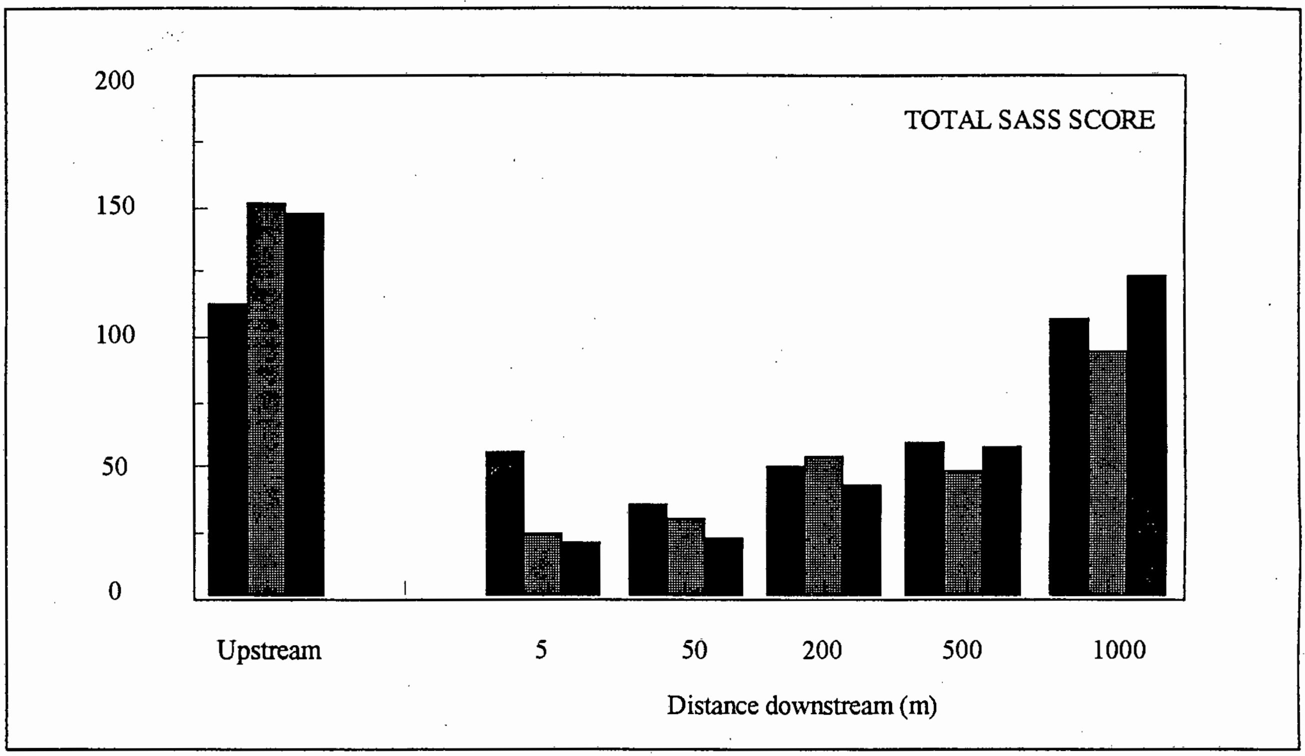

Figure 5.14a SASS scores (three replicate collections) for the sites upstream and downstream of the portapool farm

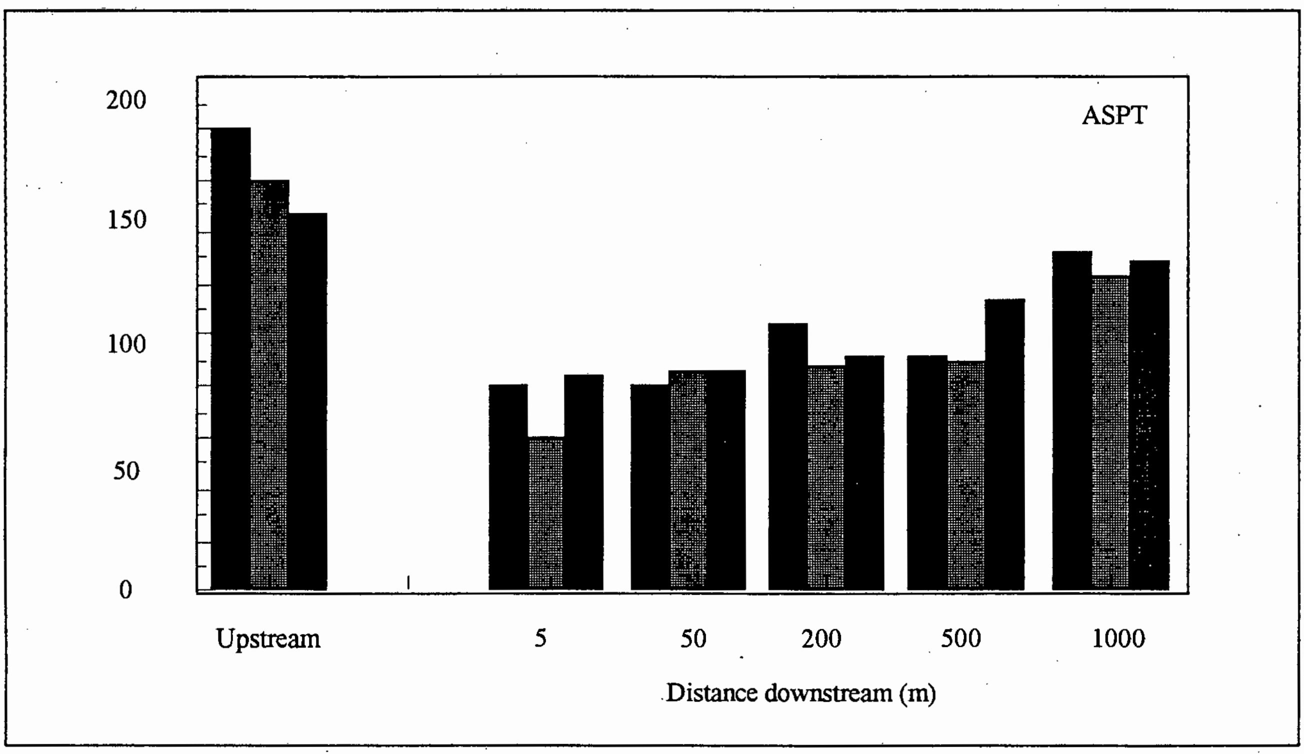

Figure 5.14b ASPT scores (three replicate collections) for the sites upstream and downstream of the portapool farm

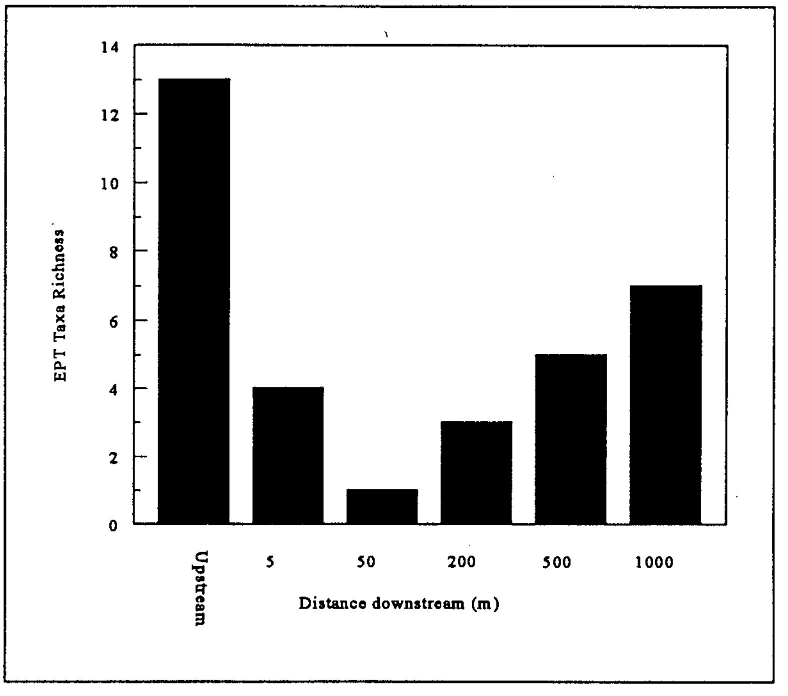

Figure 5.15 EPT taxa richness for the sites upstream and downstream of the portapool trout farm (Chapters 3 and 4)

upstream and downstream of the portapool trout farm. It should be borne in mind therefore that the conclusions drawn from the results refer a specific farm on a specific river in the south-western Cape.

The clean-water species present upstream of the farm were eliminated from the river by the farm effluent for a downstream distance of at least one kilometre. Immediately downstream of the effluent outfall there was abundant habitat for aquatic worms. One kilometre downstream of the effluent outfall, the community was still dominated by worms (albeit a different genus, Appendix A), although some of the more cosmopolitan, clean-water species were also present. The data presented in Chapter 4 show that the most downstream site sampled was dominated by *Nais* sp. (2803 m^{-2}) and *Dugesia* sp. (1743 m^{-2}) and lacked many of the families, such as the Leptophlebidae, Ephemerellidae, Heptogeniidae as well as various colepteran families, all of which are characteristic of south-western Cape mountain streams.

The Bray-Curtis measure of similarity was used in this chapter to illustrate the effects of different levels of taxonomic resolution on the perceived relationships between sampling sites. Of the four taxonomic levels tested, the family-level data seemed to provide the best indication of the effects of the trout-farm effluent on the macroinvertebrate community. The species data, on the other hand, seemed to agree most closely with the pattern obtained for the environmental variables thought to be unaffected by the presence of the trout farm. This suggests that the more exaggerated patterns obtained using the species-level data may have been influenced by environmental factors other than the ones of immediate concern. There is evidence from marine studies to suggest that change in community structure following a perturbation is *more* obvious at higher taxonomic levels, because the community response is more evident above the natural environmental noise (Warwick 1993). There is some evidence to suggest the same may be true in fresh waters. For instance, Bournaud *et al.* (1996) reported similar patterns at the familial and species level in their study of longitudinal changes along the River Rhône in France, except in the undisturbed upper reaches where familial and species-level data displayed different patterns. The argument that familial data help reduce natural 'noise' levels, thereby simplifying the interpretation of data collected to assess impacts, is supported by the results shown in this chapter, and merits further investigation.

The results presented in this chapter also suggest, however, that identification to a resolution lower than familial may result in important information being missed, and increase the chances of the effects of a perturbation going unnoticed. Quinn & Hickey (1993), working in New Zealand streams, failed to show any significant changes in macroinvertebrate density downstream of sewage farms using either ordinal data or by using functional-feeding groups, and concluded that enrichment stress is more easily seen at the species level. In this study, however, the ordinal data indicated that the upstream sites and downstream sites were 80-90% similar to one another, yet a cursory glance at the functional-feeding groups comprising the two communities suggests that there were major functional differences between them. Furthermore, these differences were clearly indicated by the familial data. Recent studies on other rivers have also indicated that, for macroinvertebrates, the effects of perturbation are detectable at the level of family. For instance, the Australian Rapid Biological

Assessment Method (Chessman 1994) requires that animals be classified only to family level and there has been success in the use of familial identifications in Great Britain using the Biological Monitoring Working Party (BMWP) score system (Chesters 1980; Wright *et al.* 1988, Wright *et al.* 1995). It would appear therefore that family-level identification offers the most reliable data and best value for money for monitoring of rivers and streams using macroinvertebrates.

The presence-absence data also showed the expected trends reasonably well, especially for familial data. However, the differences in community composition between the site 1000 m downstream of the effluent outlet and the upstream site were not illustrated as clearly as those using abundance data. Family-level, ranked-abundance data, on the other hand, highlighted the differences between the sites. Ranking or scaling the abundances would have a very similar effect to a log transformation, but has the advantage of being a technique that can be conducted in the field. A simple ranking of the abundances of each family on a 4- or 5-point scale is easily and quickly accomplished, and the benefits of including these sorts of data into an analysis should be fully explored.

The family-level meta-analysis yielded trends similar to those obtained by Warwick (1993). The technique has several disadvantages from a monitoring point of view, in that it requires quantitative sampling and sophisticated analytical techniques. Its advantage, however, is that it places a disturbance within a regional context, with respect to both natural variations and other disturbances. Hence, the relative severity of the disturbance is highlighted.

The ABC curves did not follow the predictions made by marine studies. Part of the reason for this may be that macroinvertebrate communities in "flashy" discharge regimes are controlled by natural physical disturbance (Townsend 1989). Thus, in the absence of anthropogenic disturbance, most patches in such streams are occupied by r-selected individuals. Downstream of some of the trout farms, however, there are large depositions of organic matter in slow-flowing areas. The data presented in Chapter 3 indicated that these deposits are flushed on average seven times a year, during the winter months (see Chapter 4). Thus, organic material that settles on the river bed after the last winter spate will, in all likelihood, remain there throughout the summer (pers. obs.). This results in the formation of relatively stable, hydraulically controlled habitat patches of glutinous waste organic matter. Conditions in these patches could be compared to those in large, stable lower rivers. Downstream of the trout farm, these patches were inhabited by large numbers of the detritivores *Lumbriculus* sp. and *Chironomus* sp. These species live in the substratum and are indicative of a high degree of organic pollution. They are also relatively large-bodied (Table 5.3), out-weighing many of the so-called "clean water" species. Bourassa and Morin (1995) found that overall macroinvertebrate abundance was higher in eutrophic streams than in oligotrophic streams but that only animals > 1 mm were more abundant. Similar trends have been reported from studies on lentic communities and it appears that, even in the absence of the organic deposits found in this study, larger organisms are better able to take advantage of changes in food availability than can small animals (Sprules and Munawar 1986). It is possible that, in systems controlled by physical disturbance, the trends in ABC curves are the reverse of those in marine systems. Whether this is so would be well worth investigating, since ABC curves could potentially provide a theoretical underpinning for the results obtained using other biomonitoring techniques, in much the same way as they do for marine systems.

Shannon-Wiener Diversity, SASS and EPT taxa richness showed similar responses to the changes in the invertebrate communities. A disquieting aspect, however, is that both SASS Total Scores and ASPTs obtained at the most downstream site approximated those recorded upstream of the trout farm. Because SASS does not incorporate any measure of abundance, a site dominated by oligochaetes but with some representatives of "clean water" taxa present (1 km downstream of the outlet) may have the same or similar scores as a site with an even distribution of "clean water" species (upstream). Over the last year or so, practitioners of SASS have been ranking the abundances of each of the families recorded when doing a SASS survey. The results in this thesis suggest that incorporation of such a measurement would greatly enhance the sensitivity of the index. Without it there is a real chance that moderate and slight impacts would be missed. This could lead to the effects of perturbations affecting water quality going undetected. The same is true for EPT taxa richness, which also does not incorporate a measure of abundance. Both indices are also susceptible to being influenced by "chance" records resulting from downstream drift.

The SASS sampling technique requires that all the habitats present at a site be sampled and the scores combined into a single sample. In this instance, this resulted in SASS recording about 30% more families per sample than the number recorded using the box-sampling technique. Results presented in this and earlier chapters have suggested that perturbations can affect different biotopes differently. Thus, it is advisable to incorporate a number of different biotopes into bioassessments of the impact of perturbations. However, since the communities found in different biotopes are likely to differ naturally from one another, data collected from a combination of biotopes are likely to include natural 'noise', which complicates the interpretation of data in much the same way as that illustrated by the species-level data. The downstream 'recovery' recorded by SASS could be partly as a result of a slight increase in marginal vegetation (and therefore the addition of species that inhabit marginal vegetation) at the sites furthest away from the effluent outlet (pers. obs.), although this was never quantified. The shape of the species/family accumulation curve is dependent to some extent on the amount of habitat sampled by an individual replicate (Marchant 1990). Fontoura and De Pauw (1994) have suggested that bioassessment samples taken in different microhabitats present at a site should be combined to provide an accurate assessment of the influence of changes in water quality. Dallas (1995) found that for SASS, the number of biotopes sampled at a sites with intermediate or poor water quality did not affect the scores obtained but that at sites with good water quality, the SASS Total Scores increased with an increased number of biotopes. This study suggests that combining the results from different microhabitats can artificially inflate the scores obtained at a site and supports Chessman's (1994) contention that they be sampled separately. Differences in habitat introduce variation that can mask other differences, such as water quality, between sites since taxa expected to occur at a site may be as much a product of habitat as of water quality tolerances (Parsons and Norris 1996). Furthermore, sampling a variable number of habitats confounds the detection of biological impairment because of unequal sampling effort (Parsons and Norris 1996).

In summary, in the analyses presented here it would appear that familial data are best to ensure the detection of moderate impacts as a result of effluents from trout-farm or other intensive feeding concerns, such as piggeries. In other words, quantitative or semi-quantitative family-level data probably represented the best compromise between the costs of collecting and processing the samples, and the information content of the resultant data. Quantification need not entail the laborious counting of every individual but some assessment of numerical dominance should be made. The use of several different techniques would enhance any conclusions drawn from data collected as part of a biological monitoring programme. In this regard, the inclusion of some form of sensitivity rating is a useful addition (*sensu* Lenat 1993, Chutter 1994). This provides a built-in 'expert' interpretation of the data, which obviates the need for the technician who is responsible for data collection, to interpret them from scratch. Finally, techniques with a theoretical basis (such as the ABC plots) should be developed and implemented since they can provide a theoretical underpinning for results obtained.

CHAPTER 6

AN ASSESSMENT OF THE RESULTS OBTAINED FOR COMMONLY USED INDICES USING MODELLED COMMUNITIES

6.1 INTRODUCTION

Indices summarising community structure are used to evaluate species interactions, biogeographical factors and environmental stress (Boyle *et al.* 1990). They have been used widely and have been the subject of numerous scientific papers (e.g. Jansen and Vegelius 1981, Wolda 1981). Community structural analysis is presently a diverse, well-developed field (Washington 1984, Boyle *et al.* 1990). There can be little doubt that there is a place for indices in monitoring the effects of anthropogenic disturbances on natural ecosystems. They can provide answers considerably more quickly, and therefore more cheaply, than can conventional methods of assessment. They are also amenable to simple statistical analysis (Clarke and Warwick 1994). Most importantly, however, they summarise potentially confusing data into a single number that can be readily understood by the layperson. Their popularity is unsurprising, but their potential for misuse is enormous.

This chapter investigates how the values calculated from seven commonly-used indices changed with subtle changes in community structure. The indices chosen for reporting in this chapter were the following:

- Species richness: a simple measure of the number of species.
- Pielou's evenness (Pielou 1975) and Simpson's D (Simpson 1949): indices that measure the evenness of distribution of individuals across the species in a community (how evenly the individuals in a sample are distributed among the different species). For instance, Krebs (1985) defines Simpson's D as "the probability of picking two organisms at random that are different species".
- Shannon Wiener Diversity (Good 1953, Krebs 1985) and Hurlbert's PIE (Probability of Interspecific Encounter; Hurlbert 1971): diversity indices that combine measures of evenness and species richness.
- Bray-Curtis Similarity (Bray and Curtis 1957): a similarity index which take account of the abundance of all the species in a sample but which requires comparison among two (or more) samples.
- South African Scoring System Version 4 (SASS4, Chutter 1994): a biotic index which was developed specifically for South African rivers, and which combines the presence of a macroinvertebrate taxon with an *a priori* weighting related to that

taxon's response to a particular type of pollution. SASS4 was developed specifically to assess the effect of organic pollution (and more recently other perturbations) on riverine ecosystems. The SASS4 methodology is explained in Chapter 5. Briefly, each taxon is pre-allocated a 'sensitivity' score (1 - 15), based on expert opinion, according to the water-quality conditions the taxon normally inhabits (Dallas 1995).

Species richness was included in this chapter because it is the simplest expression of the composition of a community. Simpson's D, Hurlbert's PIE and Bray Curtis were chosen because they were identified by Washington (1984) in his review of diversity, biotic and similarity indices with special relevance to aquatic ecosystems, as having a sound theoretical basis and as having more apparent biological relevance than the other indices he examined. Shannon Wiener Diversity (H') was included in this chapter because, despite criticism that H' is insensitive to rare species (Sager and Hasler 1969) and that it lacks a clear ecological basis (Goodman 1975, Hilsenhoff 1977), it continues to be widely used in studies of communities in aquatic ecosystems. Pielou's evenness (J) is derived from H'. The reason for its selection was that it is the most common expression of evenness (Clarke and Warwick 1994). Finally, the biotic index, SASS4, was chosen because it is widely used in South Africa for monitoring water quality in rivers (e.g. Dallas 1995) and is likely to become the cornerstone of the new National Biomonitoring Programme (Uys *et al.* 1996).

Each index was calculated for each macroinvertebrate community predicted to occur at each, progressively higher, concentration of organic pollution. This was done in order to examine how the indices respond to changes in community structure and why they respond the way that they do. The 'communities' for which the indices were calculated were produced using modelled responses of macroinvertebrate taxa to suspended organic pollution, based on data collected upstream and at various distances downstream of the portapool farm described in detail in Chapters 3 and 4 of this thesis.

This chapter has three main aims: (1) to examine the results obtained from analyses for a number of indices, in an attempt to illustrate the potential problems associated with their use in assessing water quality changes in riverine ecosystems, (2) to explain the possible reasons for these problems in the light of the basic principles embodied in the River Continuum Concept (Vannote *et al.* 1980) and the Serial Discontinuity Concept (Ward and Stanford 1983a) of riverine ecosystems, and (3) to highlight the role that modelling can play in developing an understanding of some of the problems that are encountered in applied ecology.

6.2 METHODS

6.2.1 The macroinvertebrate "community"

Data collected during the summer in the river at the portapool farm described in Chapter 4 were used to generate a hypothetical community response to increasing concentrations of suspended organic solids. The methods of data collection and identification are described in Chapter 2, and the raw data are provided in Appendix A. Details of the chemical analyses are given in Chapter 3.

The results of the BIO-ENV procedure presented in Chapter 4 showed that changes at the level of overall community structure were correlated with changes in the concentration of organic suspended solids (which does not necessarily imply a causal relationship between organic suspended solids and community structure). A statistically-significant relationship between the abundance of individual species and the concentration of organic suspended solids proved more difficult to establish. This was probably because of a combination of insufficient samples and the influence of antecedent events (such as spates) on animal numbers.

For the model, the average abundance of a species in the riffle biotopes (from three replicates) at each sample site was related to the average concentration of suspended solids measured (from five replicates) at the time that the faunal samples were collected. Samples collected upstream of the portapool farm had low concentrations of organic suspended solids, and the conditions measured there were taken to represent the "undisturbed" condition. Downstream of the farm, there was a gradual reduction in organic suspended solids with distance downstream of the farm (Chapter 3). Thus, after the upstream site, the site furthest downstream (1 km) had the next lowest concentration of organic suspended solids, the site 500 m downstream the next lowest after that, and so on.

A subset of the 22 most abundant species recorded in the river at the portapool trout farm during the summer was used to represent the macroinvertebrate community (Table 6.1). This was necessitated by the fact that the less-abundant species were only recorded once or twice and, therefore, the relationship between concentration of

Table 6.1 Macroinvertebrate species chosen to represent the 'community'

Order	Family	Sub-Family/Tribe	Genus/Species
Diptera	Simuliidae		*Simulium* sp.
	Chironomidae	Orthocladinae	*Tvetenia* sp. orthoclad sp AA orthoclad sp AB orthoclad sp J orthoclad sp X *Chironomus* sp.
		Tanypodinae	tanypod sp.
		Tanytarsini	*Rheotanytarsus* sp.
		Chironomini	*Polypedilum* sp.
Ephemeroptera	Baetidae		*Baetis harrisoni* *Acentrella capensis*
	Leptophlebidae		*Castanophlebia* sp.
	Ephemerellidae		*Lestagella penicillata*
Coleoptera	Elmidae		elmid sp. A
	Dryopidae		dryopid sp. A
	Helodidae		*Prionocyphon* sp.
Plecoptera			*Aphanicerca* sp.
Trichoptera			*Cheumatopsyche* sp.
Oligochaeta	Lumbriculidae		*Lumbriculus* sp.
	Naididae		*Nais* sp.
Tricladida			*Dugesia* sp.

organic solids and abundance of those species could not be quantified. The use of a subset of the data has a dampening effect on diversity: the modelled community is already less diverse than the natural community from which it was generated, and it is possible that any differences in diversity would also be dampened. However, the modelled community was intended only to represent a hypothetical community in a mountain stream, and changes in that community in response to an increase in organic pollution. The most important requirement was thus that the changes in community structure should represent ecologically-realistic responses of particular species to an initial increase, and subsequent downstream reduction, in suspended organic solids.

An ecologically-realistic response can be defined as changes in the abundance and proportions of a species that can be explained in terms of their life histories, feeding strategies or other aspects of their biology.

It would have been preferable to use data from samples collected in the winter as well as those collected in summer (Chapter 4), but seasonal differences in community structure and water quality precluded this. For instance, the ephemerellid, *Lestagella penicillata*, occurs in the rivers in greatest numbers during the summer, but is often also recorded during the winter months. Hence, differences in abundance between the two seasons are as likely to be a product of the life cycle of the species as of its response to the concentrations of suspended organic solids in the river.

For each species, a data set consisting of mean abundance data (number per m^2, $n = 3$) and corresponding mean concentrations of suspended organic solids ($n = 5$) was constructed. The data for each species were subjected to a series of statistical tests to find the best fit to any one of four regression equations, *viz.*

- linear ($y=a+bx$)
- logarithmic ($y=a+b(\log_{10}x)$)
- exponential ($y=ae^{bx}$), and
- power ($y=ax^b$).

For those species for which the concentration of organic suspended solids in the water column accounted for, an arbitarily chosen, > 60 % of the variation, the relevant equation was used to generate a model of the changes in abundance of that species with changes in the concentration of organic suspended solids.

For those species where the concentration of organic suspended solids accounted for < 60 % of the variation, the raw, unsmoothed data were used to describe the species' response (after Bovee 1982).

The individual curves are referred to in this chapter as the "species response curves". The 22 species response curves were combined to describe the overall response of the modelled community to increasing levels of suspended organic solids.

6.2.2 Diversity and similarity indices

The indices calculated were:

1. Species richness S
2. Shannon Wiener $H' = -\sum_{i=1}^{k} p_i \log p_i$
3. Pielou's Evenness $J' = H'(observed) / H'_{\max}$
4. Simpson's D = $\dfrac{\sum n_j[n_j - 1]}{I(I-1)}$
5. Hurlbert's PIE = $\dfrac{I}{I-1}\left[1 - \sum p_j^2\right]$
6. Bray-Curtis = $0.5 * \sum |p_{oj} - p_j| = 1 - PSC/100$

where:

ln =	natural logarithm
I =	total number of individuals of all species in the sample from the community
S =	total number of species in a community
H_{max} =	the maximum possible diversity which would be achieved if all species were equally abundant (= log S)
$p_j = n_j/I$ =	proportion of the perturbed community belonging to species j
$p_{oj} = n_{oj}/I$ =	proportion of the original community belonging to species j
PSC =	Percentage Similarity = $100 * \left[1.0 - 0.5 * \sum |p_{oj} - p_i|\right]$, and
	ratio (j) $\dfrac{\min[n_j, n_{oj}]}{\max[n_j, n_{oj}]}$

7. The South African Scoring System Version 4 (SASS4, Chutter 1994).

Following Boyle *et al.* (1990) who tested the response of nine indices using modelled communities, in this chapter each index was calculated for the original community and for the communities generated by the model at 0.05 mg l^{-1} incremental increases in the concentration of suspended organic solids in the water column. Similarity indices were calculated between the original community and the community at each 0.05 mg l^{-1} increase in suspended organic solid concentration. The results obtained for each of the indices were compared graphically.

6.3 RESULTS

6.3.1 Species response curves

The individual response curves for each of the species are presented in Figures 6.1 - 6.4 and the regression equations used to construct each curve are provided in Table 6.2. Species for which a dotted line is shown are those species whose abundances were well explained by suspended organic solid concentration. The line depicted in Figure 6.2 is the line of the equation of the curve. Those with a solid line are species where the raw data were used and the line links individual data points. The modelled abundances are presented in Appendix B. The species response curves are combined in Figure 6.5 to give a hypothetical community response to increasing concentrations of suspended organic solids in a mountain-stream ecosystem.

6.3.2 The indices

Using the community curves presented above, the various indices were calculated at each 0.05 mg l^{-1} increase in suspended organic solids. The results are depicted in Figures 6.6 and 6.7. In each instance, the particular index value for the modelled community is plotted against the concentration of suspended organic solids at which that community occurred. That is, the community is progressively more disturbed moving from left to right across the x-axis.

The Bray Curtis similarity index (Figure 6.7) was plotted as an ordination plot. Ordination is a multivariate technique that arranges samples along axes, using multidimensional scaling (MDS). MDS creates a 'map' or ordination plot of the samples in a specified number of dimensions, in this case three, which attempts to satisfy all the conditions imposed by a ranked similarity matrix (Clark & Warwick 1990). The placing, and the *relative* distance apart, on a two-dimensional plot (depicting only the x-axis and the y-axis, not the z-axis), of the samples gives an idea

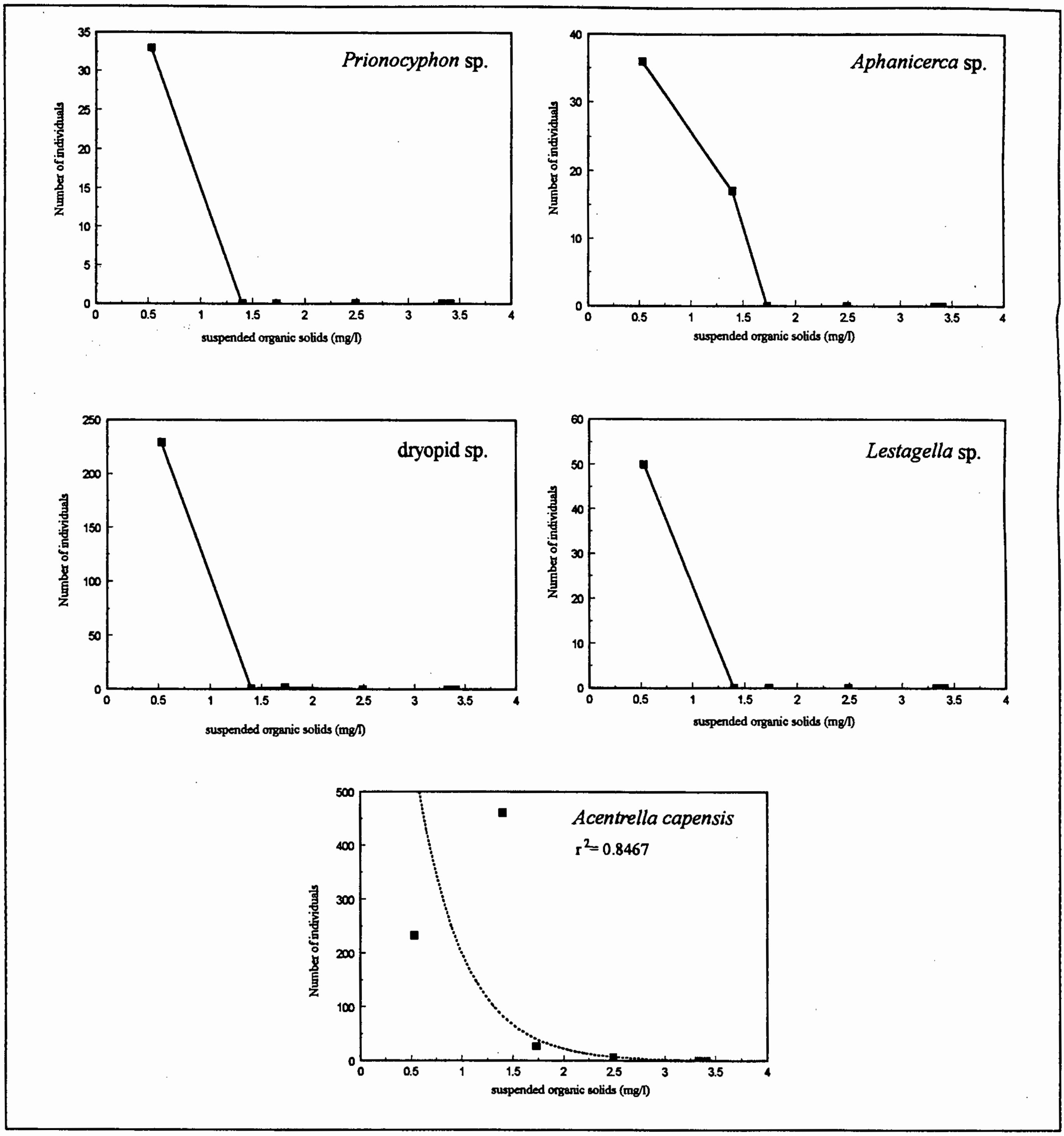

Figure 6.1 Curves of the measured response of *Prionocyphon* sp., *Aphanicerca* sp., dryopid Sp. A, *Lestagella penicellata* and *Acentrella capensis* to increased concentrations of suspended organic solids. n = 6. Dotted lines represent those species where the concentration of suspended organic solids accounted for > 60% of the variation in abundance. In these cases, the r-squared value is shown in the top right hand corner of the graph. Solid lines represent those species where the raw unsmoothed data were used to calculate the species response curves

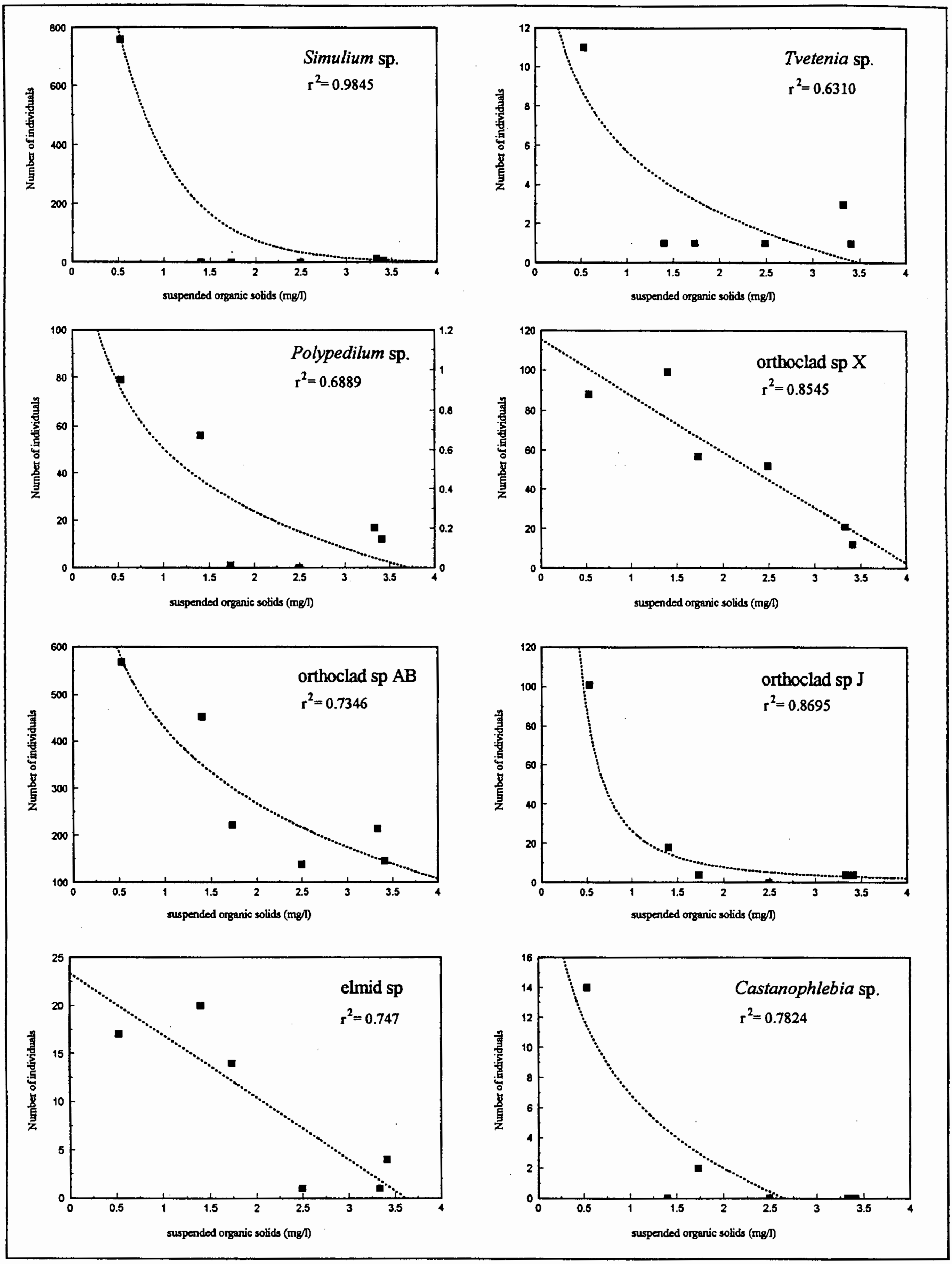

Figure 6.2 Curves of the measured response of *Simulium* sp., *Tvetenia* sp., orthoclad Sp X, orthoclad Sp AB, orthoclad Sp J, elmid sp., and *Castanophlebia* sp. to increased concentrations of suspended organic solids. n = 6. Dotted lines represent those species where the concentration of suspended organic solids accounted for > 60% of the variation in abundance. The r-squared values are shown in the top right hand corner of the graphs

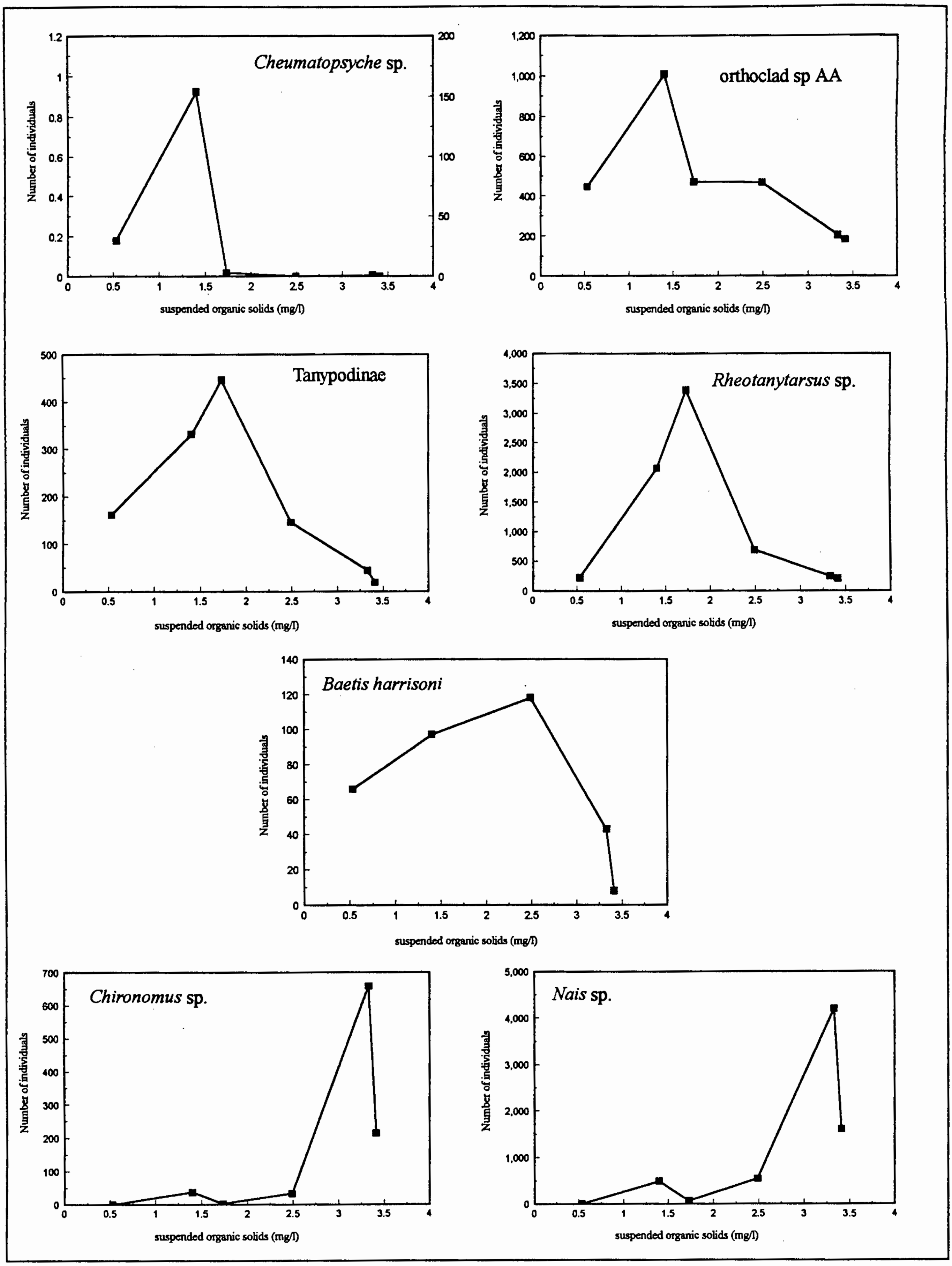

Figure 6.3 Curves of the measured response of *Cheumatopsyche* sp., orthoclad Sp AA, Tanypodinae sp., *Rheotanytarsus* sp., *Baetis harrisoni*, *Chironomus* sp. and *Nais* sp. to increased concentrations of suspended organic solids. Solid lines represent those species where the raw unsmoothed data were used to calculate the species response curves

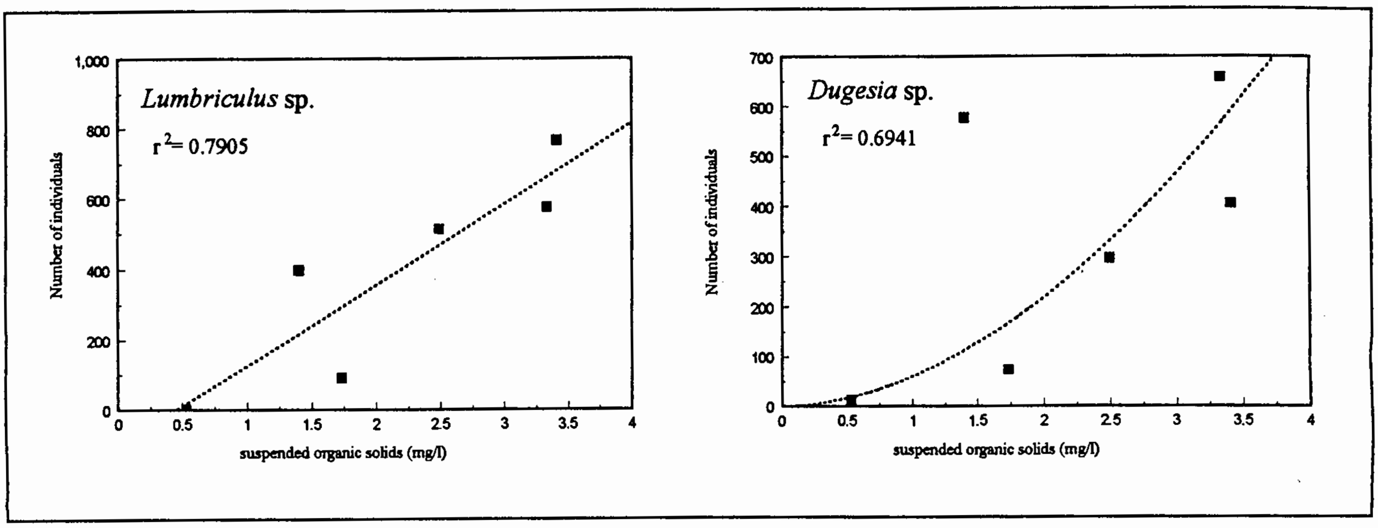

Figure 6.4 Curves of the measured response of *Lumbricus* sp. and Dugesia sp. to increased concentrations of suspended organic solids. n = 6. Dotted lines represent those species where the concentration of suspended organic solids accounted for > 60% of the variation in abundance. The r-squared values are shown in the top right hand corner of the graphs

Table 6.2 Methods used to construct the individual species response curves

Species	Method
Simulium sp.	$y=ae^{bx}$
Tvetenia sp.	$y=a+b(\log_{10}x)$
orthoclad sp AA	raw unsmoothed
orthoclad sp AB	$y=a+bx$
orthoclad sp J	$y=ax^b$
orthoclad sp X	$y=a+bx$
Chironomus sp.	raw unsmoothed
tanypod sp.	raw unsmoothed
Rheotanytarsus sp.	raw unsmoothed
Polypedilum sp.	$y=a+b(\log_{10}x)$
Baetis harrisoni	raw unsmoothed
Acentrella capensis	$y=a+b(\log_{10}x)$
Castanophlebia sp.	$y=a+b(\log_{10}x)$
Lestagella penicillata	raw unsmoothed
elmid sp. A	$y=a+bx$
dryopid sp. A	raw unsmoothed
Prionocyphon sp.	raw unsmoothed
Aphanicerca sp.	raw unsmoothed
Cheumatopsyche sp.	raw unsmoothed
Lumbriculus sp.	$y=a+bx$
Nais sp.	$y=ax^b$
Dugesia sp.	$y=ax^b$

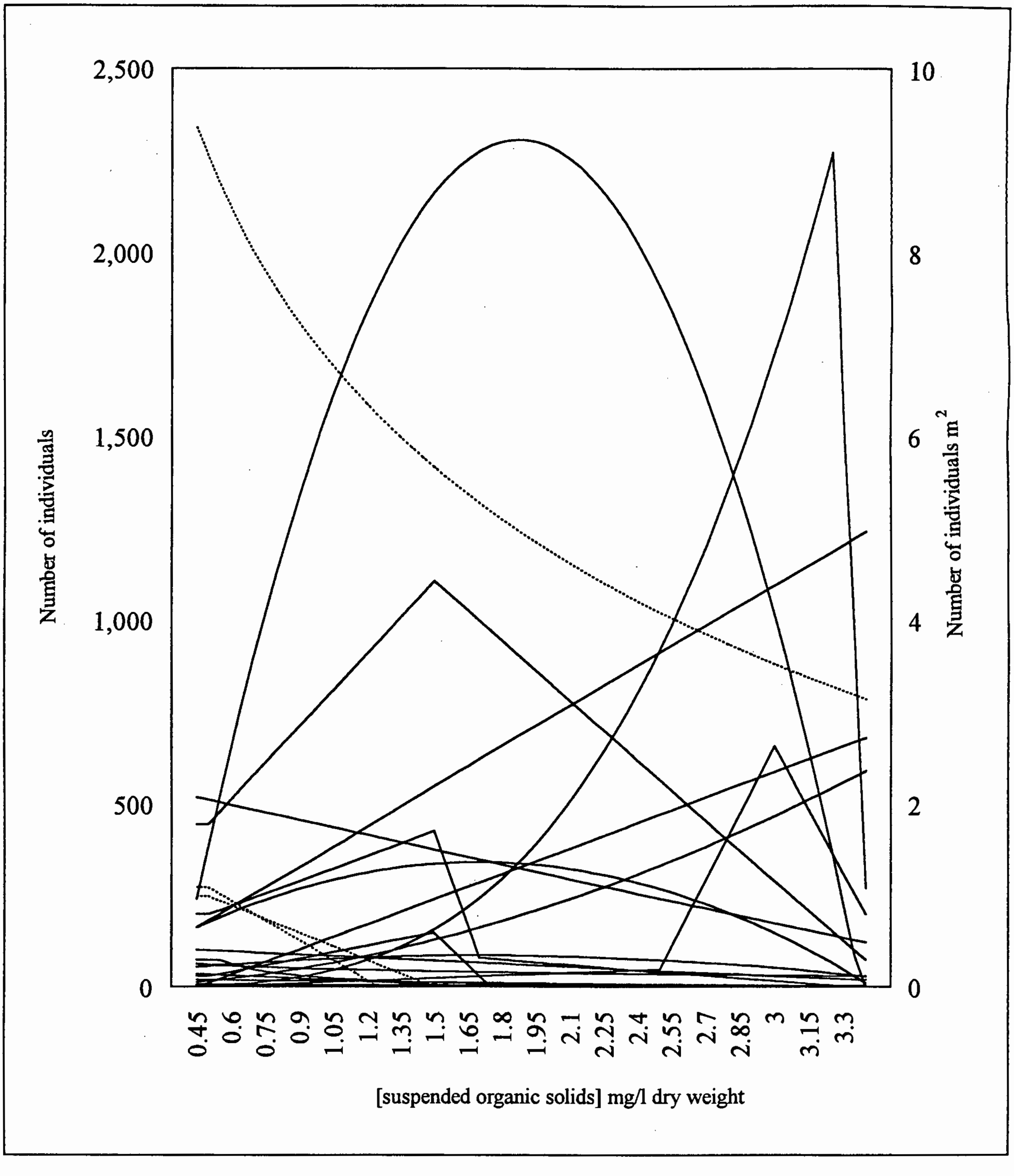

Figure 6.5 The response curves for all 22 species combined on a single graph. Those species depicted with a solid line have their abundances on the left-hand y-axis and those depict with a dotted line, have their abundances on the right-hand y-axis

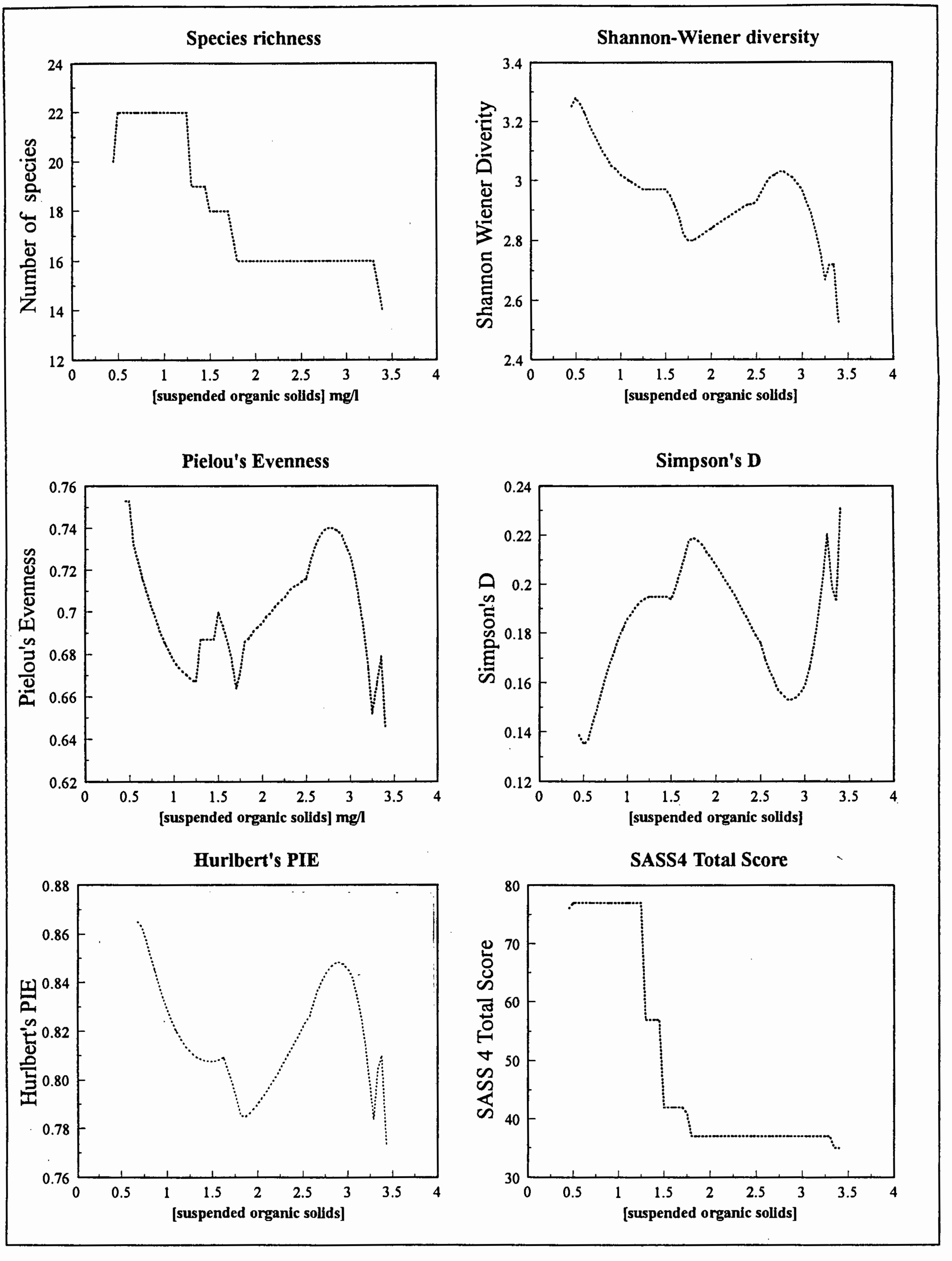

Figure 6.6 The results for Species Richness, Shannon Wiener Diversity, Pielou's Evenness, Simpson's D, Hurlbert's PIE and SASS4 Total Score calculated using the modelled macroinvertebrate communities at 0.05 mg/l increments in the concentration of suspended organic solids.

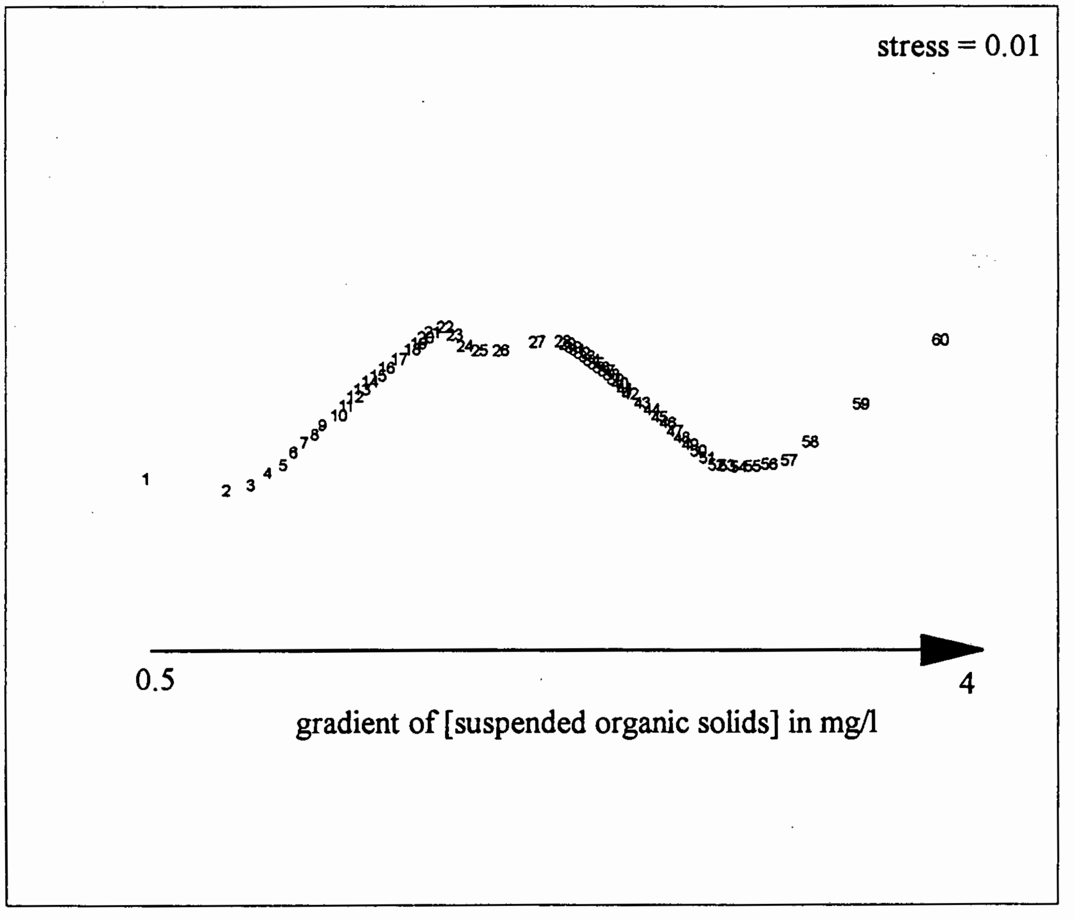

Figure 6.7 The 2-dimensional MDS results obtained for the Bray-Curtis Similarity Index using root-root transformed modelled data of changes in community structure at 0.05 mg/l increases in the concentration of suspended organic solids. The numbers 1-60 represent 60 hypothetical stations at each 0.05 increment in suspended organic solid concentrations. The axes have arbitary scales which are therefore not shown. The arrow indicating the direction of the [suspended organic solids] gradient has been superimposed on the MDS plot to aid in interpretation

of the relationships between them: that is, the closest are most similar. The resultant plot (Figure 6.7) shows a clear gradient, running from right to left across the page from the least to most impacted community. In Figure 6.7, the communities generated by the model for each 0.05 mg l^{-1} increment in the concentration of suspended organic solids are numbered from 1 (0.05 mg l^{-1}) to 60 (4 mg l^{-1}).

6.4 DISCUSSION

6.4.1 The modelled communities

Much of what is drawn from the results of this chapter depends on whether or not the hypothetical community responses on which the calculations were based are accepted as being likely to occur in nature. The 'responses' generated for each species was based on empirical data; they reflect the responses observed downstream of the outlet of the portapool farms discussed in Chapters 3 and 4, but in reverse sequence from low to high concentrations of suspended organic solids.

There were essentially four types of species response to an increase in suspended organic material in the water column, namely:

1. Species considered to be highly intolerant of organic pollution (Figure 6.1). These species included *Prionocyphon* sp., *Aphanicerca* sp., Dryopidae Sp 1 and *Lestagella penicillata*. These four species are indicative of undisturbed mountain streams in the south-western Cape (King 1981, Dallas 1995) and were rarely recorded downstream of any of the trout farms that formed part of this study.
2. Species considered to be intolerant of organic pollution (Figure 6.2). These were: *Simulium* sp., *Tvetenia* sp., *Polypedilum* sp., Orthoclad Sp. X, Orthoclad Sp. AB, Orthoclad Sp. J, Elmid Sp 1, *Castanophlebia* sp. and *Acentrella capensis*. These species were recorded upstream of the farm, and were absent immediately below the effluent outlet but were recorded at the more distant sites downstream.
3. Species whose numbers increased with a slight increase in suspended organic solids but decreased at higher concentrations. These species, such the filter-feeders *Cheumatopsyche* sp. and many of the Chironomidae; *Rheotanytarsus* sp., Tanypod Sp 1. and Orthoclad Sp. AA, were considered to benefit from slight organic pollution (Figure 6.3). They are naturally present in low numbers in south-western

Cape mountain streams (King 1981). Also included in this group are *Chironomus* sp. and *Nais* sp., which inhabit the settled organic material.

4. *Lumbriculus* sp. and *Dugesia* sp. (Figure 6.4), which increased dramatically in abundance with increasing concentrations of suspended organic solids.

Groups 4 and 5 are referred to as pollution-tolerant fauna, although it must be stressed that they are not confined to polluted waters. They are normal inhabitants of the depositional areas of rivers, such as backwaters and ponds, and they merely happen to benefit from organic pollution (Hynes 1960).

In terms of functional feeding groups, increasing organic material suspended in the water column results in a decrease in the numbers of grazers and shredders, and an increase in the abundance of filter-feeders. However, if the concentration of the suspended organic solids continues to increase, the numbers of filter-feeders decreases, and the community becomes numerically dominated by detritivorous species (see Chapter 4). In the case of the trout farm, the reach immediately downstream of the effluent outlet supported a community of detritivores, but the amount of suspended solids decreased with increasing distance downstream. This resulted in the reverse pattern to that just described, and reflects the so-called recovery or discontinuity distance referred to by the Serial Discontinuity Concept (Ward and Stanford 1983a). The resultant community response curves (Figure 6.5) closely resemble the classic curves downstream of a point-source of organic pollution (Hynes 1960). The community on the left-hand side of the graph (low concentration of suspended organic solids) represents a relatively undisturbed mountain-stream community. Such a community tends to have a fairly large number of species, each with a fairly low abundance. The right-hand side depicts a community severely disturbed by suspended organic pollution (Figure 6.5). The disturbed community would tend to have relatively few species and be numerically dominated by one or two species.

It may be tempting to dismiss the results obtained for some of the indices as anomalies of the data set used to create the modelled community; of necessity, models of community structure are a simplification of the actual situation. However, it is this characteristic which makes them so useful. Through simplification they enable detailed examination of the results obtained from summary statistics, such as indices. Models of changes in community structure in response to changes in a particular variable permit detailed scrutiny of each stage of a change from one community to another. Used in combination with accepted theories of community structure, they can considerably improve our understanding of the potential difficulties that can arise in the interpretation of such results.

6.4.2 Assessing the modelled community changes using indices

In general terms, increasing levels of environmental impact are considered to *decrease* diversity, *decrease* species richness and evenness, *increase* dominance and *decrease* stability (Clarke and Warwick 1994). The results of richness, diversity and evenness indices are generally evaluated using this interpretation. Thus, the higher the value obtained for an index, the less impacted the system is assumed to be. This interpretation may, however, be an oversimplification. Certainly from the results obtained in this chapter using a modelled community, it appears that the situation is far more complicated.

The simple measure of species richness (number of species) appeared to follow the form of a negative exponential (i.e., increased impact = decreased richness). This was only after an initial slight increase in species richness, however. An increase in species richness under conditions of mild organic pollution, particularly in shallow, reasonably well-oxygenated waters, is a well-documented phenomenon. Generally, under these conditions, although there is an increase in oligochaete and chironomid species, many of the 'clean water' species are able to persist, albeit in lower numbers (Hynes 1960). Hence, the overall species richness is increased.

The results obtained for Pielou's evenness, Simpson's D, Hurlbert's PIE and Shannon-Wiener were more complicated. These indices all incorporate some measure of the evenness of distribution of individuals among the species present in a community. In each case, plots of the results of the indices against increasing concentrations of suspended organic solids produced a wavelike pattern, which resulted in identical values being achieved for different concentrations of suspended organic solids. Hence, changes in community structure were, at times, undetected by the index, which returned the same value for very different communities found at much higher, or lower, concentrations of suspended organic solids. This suggests that macroinvertebrate communities pass through multiple stable states in response to increasingly severe suspended organic pollution. As it happens, this is in agreement with the predictions of both the River Continuum (Vannote *et al.* 1980) and the Serial Discontinuity concepts (Ward and Stanford 1983a&b). The central notion of both concepts is that ***stable*** (i.e. higher evenness) biological communities should form in a predictable fashion based on size and availability of organic matter in the system. Thus, it is plausible that a stable community formed in the clear waters of a Cape mountain stream could be replaced by an equally stable but very different community, following the addition of large quantities of fine organic matter. The second community is more likely to be reminiscent of that usually found in the lower reaches of these rivers.

A problem arises, however, if indices that measure evenness or diversity are used to assess the affects of suspended organic pollution on mountain-stream ecosystems. If one of these indices were being used as the basis of a monitoring programme to assess the impacts of pollution resulting from suspended organic solids, the implications from a management point of view could be enormous. For instance, Pielou's evenness returned the same result for a community consisting of individuals of nine species characteristic of an undisturbed Cape mountain stream and for a community consisting of five species that are normally found only in the lower depositional reaches of these river systems (Hynes 1960, Harrison 1965). This is because univariate indices are not species specific, and two communities with completely different taxonomic composition have the same univariate configuration (Warwick and Clarke 1991). Whereas a simple examination of the actual species inhabiting the stream would give a clear indication of the changes taking place, there is a chance that an index such as these could fail to indicate a fundamental change in the structure of the macroinvertebrate community.

Multivariate analyses on the other hand are species (or taxon) specific. Using the Bray-Curtis Similarity Index, two samples are considered to be the same only if they contain the same taxa in exactly the same abundance. Hence, the Bray-Curtis Index was able to show the subtle differences in community structure that resulted with each incremental increase in the concentration of suspended organic solids. Also, because a similarity 'score' is a measure of one site relative to the next, on a continuum, the results obtained using Bray-Curtis provide an indication of how different an impacted site is from an upstream, unimpacted (or control) site. What similarity indices cannot do in this instance, however, is give an indication of how impacted the control site itself may or may not be without first determining the natural or reference condition for that river.

SASS4 combines some of the characteristics of both univariate and multivariate indices, in that it is a taxon-specific, univariate index. SASS4 combines the presence of macroinvertebrate families with an *a priori* weighting related to that family's known response to impaired water quality; the higher the SASS4 score, the healthier a community is presumed to be. The SASS4 scores calculated for the modelled communities dropped dramatically as the concentration of suspended organic solids increased. However, because SASS4 does not take into account abundances, it operates on the 'spilt milk' principle. That is, the researcher is only alerted to change once one or more species have been lost from the system. This leads to the sort of erroneous "recovery" signals reported in Chapter 5. In their study on the effects of organic pollution on the Adur River in England, Whitehurst and Lindsey (1990) concluded that for this reason simple indices (in their case the *Gammarus:Asellus* Ratio) were useful only over a limited pollution range. The sensitivity of the SASS4 to changes in the macroinvertebrate community also varied widely, however. For example, SASS4 showed a clear negative response to the elimination of the 'sensitive' macroinvertebrate species (*Prionocyphon* sp., *Lestagella* sp., *Dryopidae* sp 1, *Aphanicerci* sp.), to which the method allocates high scores. However, because it does not take cognisance of abundances, SASS4 showed no response to an increase from 310 m^{-2} to 2271 m^{-2} in lumbriculid worms and the concomitant reduction of *Acentrella capensis* from 75 individuals m^{-2} to 2 individuals m^{-2} (SASS4 = 37 throughout). Furthermore, SASS4 appeared no more sensitive to the modelled community changes than did species richness. This begs the question: "why use a biotic index, which is dependant on family-level identification and on subjective weightings assigned to each family, when a species richness curve, which is free of both, provides the same resolution?" As discussed in Chapter 5, the inclusion of some form of sensitivity rating (*sensu* Lenat 1993) can provide a built-in 'expert' interpretation of the data, which means that it is not necessary for a technician to interpret the data themselves. However, the results of this chapter and Chapter 5 both identify the lack of abundance estimates and biotope specificity as serious weaknesses of the SASS4 methodology.

In summary, the potential usefulness of indices to summarise complicated environmental data cannot be denied. If they serve no purpose other than to convey information in an easily understandable manner to managers, funders and the general public, their purpose is an important one. However, it is imperative that scientists do not lose sight of the fact that the environment and its responses to anthropogenic disturbance **are** complicated, and are dependent on the life-histories of the organisms involved. Indices and other summary statistics are always an oversimplification of the actual situation. The responses of riverine macroinvertebrate communities to organic pollution cannot be depicted simply as a reduction in the lumped abundance of some or all of the species making up that community. Nor is the distribution of individuals among taxa a particularly reliable measure of the community response to organic pollution, unless something is known about which taxa are present. Furthermore, unless the measure used has a sound biological basis or is grounded in accepted ecological theory, it should not be used to summarise ecological data. This is simply because the result it provides, albeit "sellable", may be misleading.

SECTION V

DISCUSSION

CHAPTER 7
DISCUSSION

In turbulent, steeply-graded mountain streams, the patterns of faunal community structure induced by organic pollution are not as clear cut as those recorded in the lower, more geomorphologically uniform reaches of rivers (Harrison 1965, Hynes 1960) mainly because of spatial heterogeneity and sampling across microhabitat boundaries. The result is a flow-dependent mosaic of different communities with different degrees of pollution tolerance (*sensu* Rabe and Gibson 1984, Townsend 1989, Fontoura and De Pauw 1994). Whereas in unpolluted streams the natural heterogeneity of streambeds provides areas of reduced shear stress that can act as refugia during spate events (Townsend 1989, Lancaster and Hildrew 1993), in streams that receive particulate organic pollution, the converse may be true. Areas of higher shear stress may provide refugia for the pollution-sensitive benthic macroinvertebrates from the effects of settled organic matter. In times of strong flows, however, these animals are not able to shelter in the backwater areas where the flows are slower, nor are they able to shelter in the interstitial spaces between the rocks (e.g. Poole and Stewart 1976, Marmonier and Creuzé des Châtelliers 1991, Dole-Olivier *et al.* 1997) because, when the flows first start to increase, these areas are clogged with settled organic material. Consequently, the chance of animals being swept downstream in floods increases markedly. This has interesting implications for biological monitoring using macroinvertebrate communities.

The results presented in this thesis suggest that, for the purposes of assessing the impact of particulate organic pollution in mountain streams or foothill zones of rivers, three major habitat types should be defined: erosional, transporting and depositing habitats (Palmer *et al.* 1991, Fontoura and De Pauw 1994). The macroinvertebrate communities in these habitat types should be sampled separately in order to determine whether they are being affected by organic pollution. Furthermore, in order to understand the effects of any one kind of anthropogenic disturbance within each of these habitat types need to be considered in the light of overall stream functioning. For instance, the direct impacts of the portapool farm were greatest in the backwater areas where the most deposition occurred but elimination of backwater refuge areas may have meant that animals that would ordinarily inhabit the riffle areas were also negatively affected, despite settlement in riffles being negligible. Interestingly, Parsons and Norris (1996) recently found that these habitats (edge and riffle) produced the best results in a biological assessment of water quality using a predictive model.

The influence of antecedent events, particularly spates, was highlighted in Chapters 3 and 4. The levels of physical and chemical variables in the water, and macroinvertebrate community structures differed considerably depending on whether they were measured before or after a spate. This supports the results obtained by Boulton and Lake (1992) which showed that antecedent events affected macroinvertebrate community structure at least as much as did site-specific disturbances. In the case of biomonitoring, where different rivers are sampled, the effects of natural antecedent events could make interpretation of the results extremely difficult. This is particularly true where no record is available of the occurrence of such events prior to sampling.

The organic material that settles in the slow-flowing areas coagulates with time, which reduces its mobility. Since flushing spates have to be sufficiently strong to move all the settled material at once, this considerably increases the magnitude of the spates required to flush the settled material, and many of the smaller spates will move over the settled material without being able to dislodge it. Animals living in these deposits are sheltered from the hydraulic disturbance to which animals living on and under the cobble surfaces are subjected, and thus experience relatively stable hydraulic conditions. Furthermore, food is not a limiting factor, and so the macroinvertebrates inhabiting the organic deposits tend to be larger-bodied, less-mobile species than their clean-water counterparts. This could explain why the ABC curves did not follow the predictions from the marine studies (Clarke and Warwick 1994, Chapter 5) which are that, under conditions of anthropogenic disturbance, small opportunistic species are favoured and the macrobenthic community present is often dominated both numerically and in terms of biomass by r-selected species. Hence, the k-dominance curve for abundance will lie above the curve for biomass. This clearly was not the case in the mountain stream, and the results presented in this thesis suggest that, in mountain-stream ecosystems, 'reverse' ABC plots maybe more useful for indicating the effect of organic pollution, *viz.* biomass overriding abundance in organically-polluted streams and rivers. Ascertaining if this is the case would, however, require considerably more investigation than was possible here. Since ABC plots can potentially provide a graphic display of functional relationships, it is recommended that their applicability to upper river systems be further explored.

One method of assessing the potential usefulness of ABC plots in riverine ecosystems would be through the use of models. Chapter 6 of this thesis illustrated how, by simplifying the natural situation, models of changes in community structure enable detailed scrutiny of each stage of a change from one community to another. Used in combination with accepted theories of community structure, these can considerably improve our understanding of the results obtained from different analytical techniques or summary statistics. In this thesis, the community model was used to interpret values of the different indices used to summarise community data. However, the application of models is by no means limited to that illustrated in Chapter 6.

Measurements of Pielou's evenness, Simpson's D, Hurlbert's PIE and Shannon-Weiner diversity indicated the possibility of multiple stable states in response to increasing organic pollution. Macroinvertebrate communities with a roughly even distribution of individuals between species existed upstream of the farm and at various distances downstream, and represented points at which species belonging to a particular feeding group were able to coexist (Chapters 4 and 6). Both the River Continuum Concept (Vannote *et al.* 1980, Minshall *et al.* 1985) and the Serial Discontinuity Concept (Ward and Stanford 1983a) recognise this phenomenon. Both of these concepts are based on the notion that stable biological communities will form in a river ecosystem, based on size and availability of organic matter (Chapter 6). However, the presence of an evenly-distributed, stable community is not necessarily synonymous with a naturally-occurring community. For instance, the macroinvertebrate community recorded 50 m downstream of the outlet from the portapool trout farm was comprised of a fairly even numerical distribution of individuals of *Chironomus* sp., *Lumbriculus* sp. and *Nais* sp. It is, however, highly unlikely that this community would have existed in a mountain stream in the absence of organic pollution for the farm.

Since biomonitoring is aimed primarily at assessing the deviation from the natural condition, the implications, from a management point of view, of equating a community with an even distribution of species with a natural community could be calamitous. Because univariate indices are not taxon specific, an univariate analysis of two communities made up of completely different taxa could return identical index values. For instance, in Chapter 6, Pielou's evenness returned the same result for a community characteristic of an undisturbed Cape mountain stream and for a community normally found only in the lower depositional reaches of these river systems (Hynes 1960, Harrison 1965). Hence, if that index was the sole basis of a biomonitoring programme to assess the effects of trout farm effluent on Cape mountain streams, the results would fail to reveal any effects.

At the other end of the spectrum, the Bray-Curtis Similarity Index responded well to subtle changes in macroinvertebrate community structure. Multivariate analyses are taxon specific and two samples are considered to be the same only if they contain the same taxa in exactly the same abundance. There is some concern, however, that the index is sensitive to natural environmental variations in species composition and that this may make the interpretation of results difficult were it to be used in biomonitoring aimed at detecting anthropogenic disturbances. The results obtained in this study indicate that this may be the case but that the problem can be overcome by only identifying the animals to the family level.

That is not to say that rapid bioassessment cannot produce ecologically-meaningful results. Indeed its usefulness would be short-lived were this the case. Faced with the dual pressures of time and money, detailed species-level studies to answer management questions are simply not feasible, particularly in a developing country such as South Africa. Indeed, the financial and other limitations of being developing country are the chief reasons why the biomonitoring system, River Invertebrate Prediction and Classification (RIVPACS), which is being adopted elsewhere in world was not adopted in South Africa.

RIVPACS development began in Britain in the late 1970s (Wright *et al.* 1984) and although the original database has since been expanded (Wright *et al.* 1993), the basic approach to selecting sites has remained unchanged. The RIVPACS approach can be summarised as follows:

- Selection, based on water quality data and expert opinion, of comparatively unstressed sites (438 sites across England and Wales were selected for RIVPACS II).
- Collection of macroinvertebrate data from the major habitats (combined in a single sample), together with water quality and environmental data, at each site during spring, summer and autumn.
- The macroinvertebrate communities present at each site were assessed using a rapid sampling procedure and identified to the taxonomic level of species.
- Multivariate analysis of macroinvertebrate data to establish groups of sites with similar macroinvertebrate communities.
- Description of the environmental characteristics of each group of sites.
- Comparison of the macroinvertebrate communities found at monitoring sites with those recorded at unimpacted sites with the most similar environmental characteristics (reference sites), in order to assess the degree of difference between them.

The chief disadvantage of this approach in a South African situation was that it requires an extremely large database of the biotic community structure and environmental conditions at many sites in order to create a predictive model. Only then can monitoring sites be assessed by comparing the biotic community actually recorded with that predicted for the site.

In this investigation, the preferred technique for rapid bioassessment in South Africa, SASS4 (Chutter 1994, Dallas 1995, Uys *et al.* 1996), did not fare particularly well using empirical data. However, its potential usefulness as an index of organic pollution (for which it was originally designed) was illustrated by its response to the hypothetical community changes explored in Chapter 6. This advantage lies mainly in the fact that it is an univariate, taxon-specific index and, as such, combines some of the characteristics of both univariate and multivariate indices. Its biggest downfall is that it does not incorporate abundance data, with the result that it is insensitive to subtle changes in community structure which could potentially act as early-warning signals of perturbation. The example given in Chapter 6, for instance, was that SASS4 Total Score did not change when the modelled macroinvertebrate community changed from one which included 310 m^{-2} lumbriculid worms and 75 m^{-2} *Acentrella capensis*, to one which included 2271 m^{-2} in lumbriculid worms and only 2 m^{-2} *Acentrella capensis*. It is likely that the incorporation of abundance into the SASS4 Total Score would greatly increase the sensitivity of the index to subtle, but important, changes in macroinvertebrate community structure. This, in turn, would increase its sensitivity to subtle changes in water quality.

Until the mid-1980s biological river research in South Africa was poorly-funded and, apart from some notable exceptions, the consequence has been a paucity of biological information on the nation's rivers. If, as seems likely, biomonitoring of rivers and other freshwater ecosystems becomes standard management practice in South Africa (e.g. Hohls 1996, Eekhout *et al.* 1996, Uys *et al.* 1996), it will generate large amounts of much needed data on the country's rivers. Depending on the sorts of data that are collected, this could contribute to a major boost in our understanding of the structure and of the function of our freshwater ecosystems. It is understandable that, in terms of cost and management, the methods chosen for use in such a biological monitoring programme should be easy, quick and cheap. However, a little extra effort, such as the inclusion of abundance data and separate sampling of biotopes, can often make the difference between a meaningful and a meaningless result.

This study has shown that the criteria for successful biomonitoring of fish farm effluents include:

1. taxon-specific biomonitoring techniques
2. family-level identification
3. separate sampling of erosional and depositing habitats
4. some measure of abundance.

If met, these criteria should ensure the best compromise between the costs of collecting and processing of the samples, and the information content of the resultant data. It is however acknowledged that while the results of this study have implications for biomonitoring programmes in general, similar studies focussing on other kinds of impacts should be undertaken before the full implications for a general biomonitoring programme that covers a diverse array of impacts can be understood.

Finally, the results obtained in this study clearly illustrated the potentially damaging effects that trout farms can have on their downstream rivers. Equally clear is that removal of solids from the trout farm effluent, before it enters the river, considerably reduces these impacts. The use of settling ponds or some other form of effluent treatment is common practice in trout farms internationally and should be made a compulsory feature of trout farms in the south-western Cape.

REFERENCES

LITERATURE CITED IN ALL SECTIONS

Alexander, W.J.R. (1985) Hydrology of low latitude Southern Hemisphere landmasses. In: Davies, B.R. and Walmsley, R.D. (eds.) Perspectives in Southern Hemisphere Limnology. Junk, Dordrecht: Chapter 6.

Allcock, R. and Buchanan, D. (1994) Agriculture and Fish Farming. In: Maitland,P.S., Boon, P.J. and McLusky, D.S. (eds.) The Fresh Waters of Scotland: A National Resource of International Significance. John Wiley & Sons Ltd.

Amendments to the Water Act (1984) Special and General Effluent Standards. Section 15 of Act No. 96 of 1984. South African Government Gazette, Pretoria.

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

Armitage, P.D., Furse, M.T. and Wright, J.F. (1991) Environmental quality and biological assessment in Britich rivers - past and future perspectives. Journadas sobre le Gestion Integral de Ecosistemas Acuaticos. Direccion de Investigacion y Formacion Agropesqueras: Gobierno Vasco.

Bekker, H. and Brown, C.A. (1995) Developing environmental guidelines for freshwater aquacultural operations: a case study. Water SA 21(4):379-383.

Bergheim, A., Hustvleit, H., Kittelsen, A. and Selmer-Olsen, A.R. (1984) Estimated pollution loadings from Norwegian fish farms II. Investigations 1980-1981. Aquaculture 36:157-168.

Berhe, T., Harrison, A.D. and Hynes, H.B.N. (1989) The degradation of a stream crossing the city of Addis Ababa, Ethiopia. Tropical Freshwater Biology 2:112-120.

Beveridge, M.C.M. (1984) Cage and pen farming: carrying capacity models and environmental impact. FAO Fisheries Technical Paper 255: FAO Rome.

Beveridge, M.C.M. (1987) Cage aquaculture. Fishing News Books Ltd., Farnham, Surrey. 352pp.

Beveridge, M.C.M., Lindsay, G.R. and Kelly, L.A. (1990) Aquaculture and Biodiveristy. Ambio 23(8):497-502.

Biggs, B.J.F. and Price, J.M. (1987) A survey of filamentous algae proliferations in New Zealand rivers. New Zealand Journal of Marine and Freshwater Research 21:175-191.

Boulton, A.J. and Lake, P.S. (1992) The ecology of two intermittent streams in Victoria, Australia: II. Comparisons of faunal composition between habitats, rivers and years. Freshwater Biology 27:99-121.

Bourassa, N. and Morin, A. (1995) Relationships between size structure of invertebrate assemblages and trophy and substrate composition in streams. Journal of the North American Benthological Society 14(3):393-403.

Bournaud, M., Cellot, B., Richoux, P. and Berrahou, A. (1996) Macroinvertebrate community structure and environmental characteristics along a large river: congruity of patterns for identification to species or family. Journal of the North American Benthological Society 15(2):232-253.

Boyle, T.P, Smillie, G.M., Anderson, J.C. and Beeson, D.R. (1990) A sensitivity analysis of nine diversity and seven similarity indices. Journal of the Water Pollution Control Federation 62:749-762.

Bray, J.R. and Curtis, J.T. (1957) An ordination of the upland forest communities of southern Wisconsin. Ecological Monographs 27:325-349.

Brink, D. and Bekker, H. (1991) 'n Verslag oor die potential van akwakultuur in die Winterreënstreek. Departement Landbou-ontwikkeling. Winterreënstreek, Elsenburg. 22pp.

Brinkhurst, R.O. and Cook, D.G. (1974) Aquatic earthworms (Annelida: Oligochaeta). In: Hart, C.W.Jr., Fuller, S.L.H. (Eds.). Pollution ecology of freshwater invertebrates. Academic Press, Inc. London.

Brown, C.A., Eekhout, S. and King, J.M. (eds.) (1996) South African National Biomonitoring Programme: Proceedings of spatial framework workshop. Unpublished Report to Institute for Water .Quality Studies, Department of Water Affairs and Forestry, Pretoria: *c.* 200pp.

Bovee, K.D. (1982) A guide to stream habitat analysis using the Instream Flow Incremental Methodology. Instream Flow Information Paper 12, FWS/OBS-82/26. Co-operative Instream Flow Group, US Fish and Wildlife Service, Office of Biological Services.

BS 3680 (1980) British Standard. Methods of measurement of liquid flow in open channels. Part 3A. Velocity-area methods.

Bunn, S.E. (1988) Processing of leaf litter in two northern jarrah forest streams, Western Australia: II. The role of macroinvertebrates and the influence of soluble polyphenols and inorganic sediments. Hydrobiologia 162:211-233.

Cairns, J Jr. and Pratt, J.R. (1993) A history of biological monitoring using benthic macroinvertebrates. In: Rosenberg, D.M. and Resh, V.H. (eds.): Freshwater biomonitoring and benthic macroinvertebrates, 10-29. Chapman and Hall, New York.

Canton, S.P. and Chadwick, J.W. (1988) Variability in benthic invertebrate densities from small streams. Freshwater Ecology 4:291-297.

Campbell, B.M. (1985) A classification of the mountain vegetation of the fynbos biome. Memoirs of the Botanical Society of South Africa 50:1-115.

Campbell, I.C., James, K.R., Hart, B.T. and Devereaux, A. (1992) Allochthonous coarse particulate organic material in forest and pasture reaches of two south-eastern Australian streams. I. Litter accession. Freshwater Biology 27:341-352.

Carss, D.N. (1990) Concentrations of wild and escaped fishes immediately adjacent to fish farm cages. Aquaculture 90:29-40.

Chesters, R.K. (1980) Biological Monitoring Working Group Party.The 1978 national testing exercise. Department of the environment, Water Data Unit. Technical Memorandum 19:1-37.

Chessman, B.C. (1986) Dietary studies of aquatic insects from two Victorian rivers. Australian Journal of marine and Freshwater Research 37:129-146.

Chessman, B.C. (1994) Rapid assessment of rivers using macroinvertebrates: A procedure based on habitat-specific sampling, family-level identification and a biotic index. Australian Journal of Ecology 20:122-129.

Chutter, F.M. (1969) The effects of silt and sand on the invertebrate fauna of streams and rivers. Hydrobiologia 34:57-76.

Chutter, F.M. (1972) An empirical biotic index of the quality of water in South African streams and rivers. Water Research 6:19-30.

Chutter, F.M. (1994) Research on the rapid biological assessment of water quality impacts in streams and rivers. Progress Report to the Water Research Commission, Pretoria.

Chutter, F.M. and Nobel, R.G. (1966) The reliability of a method of sampling stream invertebrates. Archive für Hydrobiologie 62:95-103.

Clarke, K.R. and Ainsworth, M. (1993) A method of linking multivariate community structure to environmental values. Marine Ecological Progress Series 92:205-219.

Clarke, B.M., Bennett, B.A. & Lamberth, S.J. (1996) Factors affecting spatial variability in seine net catches of fish in the surf zone of False Bay, South Africa. Marine Ecology Progress Series 131:17-34.

Clarke, E.R., Harman, J.P. and Forster, R.M. (1985) Production of metabolic waste products by intensively farmed rainbow trout, *Salmo gairdneri* Richardson. Journal of Fish Biology 27:381-393.

Clarke, K.R. (1990) Comparison of dominance curves. Journal of Experimental Marine Biology and Ecology 138:143-157.

Clarke, K.R. and Warwick, R.M. (1990) Statistical analysis and interpretation of marine community data. IOC Manuals and Guides No.22. Unpublished draft Sept (1990). Intergovernment Oceanographic Commission, GEEP. Plymouth Marine Laboratory, UK.

Clarke, K.R. and Warwick, R.M. (1994) Change in marine communities: An approach to statistical analysis and interpretation. Natural Environment Research Council, UK. 144pp.

Corkum, l.D. (1991) Spatial patterns of macroinvertebrate distributions along rivers in eastern deciduous forest and grassland biomes. Journal of the North American Benthological Society 10(4):358-371.

Cowling, R.M. and Holmes, P.M. (1992) Flora and vegetation. In: Cowling, R.M. (ed.): The ecology of fynbos: Nutrients, fire and diversity. Oxford University Press, Cape Town:23-61.

Cummins, K.W. (1973) Structure and function of aquatic ecosystems. BioScience 24:631-641.

Cummins, K.W. and Klug, M.J. (1979) Feeding ecology of stream invertebrates. Annual Review of Ecology and Systematics 10:627-672.

Dadzie, S. (1992) An overview of aquaculture in eastern Africa. Hydrobiologia 232:99-110.

Dallas, H.F. (1995) An evaluation of SASS (South African Scoring System) as a tool for the rapid bioassessment of water quality. MSc Thesis, University of Cape Town. 169pp.

Dallas, H.F. and Day, J.A. (1993) The effect of water quality variables on riverine ecosystems: a review. Water Research Commission Report No. TT 61/93. 240pp.

Dallas, H.F., Day, J.A. and Reynolds, E.G. (1994) The effects of water quality variables on riverine biotas. Water Research Commission Report No. 351/1/94. 230pp.

Davies, B.R. and Day, J.A. (1986) The biology and conservation of South Africa's vanishing waters. Centre for Extra-Mural Studies, University of Cape Town, South Africa. 186pp.

Day, J.A. and King, J.M. (1995) Geographical patterns, and their origins, in the dominance of major ions in South African Rivers. South African Journal of Science 91:299-306.

Davies, B.R., O'Keeffe, J.H. and Snaddon, C.D. (1993) A synthesis of ecological functioning, conservation and management of South African rivers. Water

Reserach Commission Report No. TT 62/93. 232pp.

Dolédec, S. and Chessel, D. (1994) Co-inertia analysis: an alternative method for studying species-environment relationships. Freshwater Biology 31:277-294.

Dole-Olivier, M.J., Marmonier, P. and Betty, J-L. (1997) Response of invertebrates to lotic disturbance: is the hyporheic zone a patchy refugium? Freshwater Biology 37:257-276.

Drummond, S. (1990) Trout farming handbook. Fishing News Books, Oxford.

Dudgeon, D. (1993) The effects of spate-induced disturbance, predation and environmental complexity on macroinvertebrates in a tropical stream. Freshwater Biology 30:189-197.

DWA (1985) 16th Edition of Standard Methods for the examination of water and wastewater. Department of Water Affairs, Pretoria.

DWAF (1990) List of hydrological gauging stations. Hydrological Information Publication No. 15, July 1990. Volume 1. Gauging stations in flowing water. Directorate of Hydrology, Pretoria.

DWAF (1997) White paper on a National Water Policy for South Africa. Department of Water Affairs and Forestry, April 1997: 37 pp.

Eekhout, S., Brown, C.A. and King, J.M. (1996) National Biomonitoring Programme for Riverine Ecosystems: NBP Report Series No. 3. Institute for Water Quality Studies, Department of Water Affairs and Forestry, Pretoria, South Africa.

Field, J.G., Clarke, K.R. and Warwick, R.M. (1982) A practical strategy for analysing multispecies distribution patterns. Marine Ecology Progress Series 8:37-52.

Findlay, S. (1995) Importance of surface-subsurface exchange in stream ecosystems: The hyporheic zone. Limnological Oceanography 40(1):159-164.

Fontoura, A.P. and De Pauw, N. (1994) Microhabitat preferences of stream macrobenthos and its significance in water quality assessment. Verh. Internat. Verein. Limnol. 25:1936-1940.

Gale, B.A. (1992) The effect of regulation by two impoundments on an acid, blackwater, Cape mountain stream. Unpublished PhD Thesis. Zoology Department. University of Cape Town. 272 pp.

Good, I.J. (1953) The population frequencies of species and the estimation of population parameters. Biometrika 40:237-264.

Goodman, D. (1975) The theory of diversity-stability relationships in ecology. Quarterly Review of Biology 50:237-266.

Gore, J.A. (1989) Models for predicting benthic macroinvertebrate habitat suitability under regulated flows. In: Gore, J.A. and Petts, G.E. (eds.) Alternatives in regulated river management. CRC Press, Inc. Florida:253-265.

Gordon, N.D., McMahon, T.A. and Finlayson, B.L. (1992) Stream hydrology. An introduction for ecologists. John Wiley and Sons, Queensland, Australia.

Green, R.H. (1979) Sampling design and statistical methods for environmental biologists. Wiley, New York.

Haddad, P.R. and Jackson, P.E. (1990) Ion chromatography. Principles and applications. Journal of Chromatography Library 46. Elsevier, Amsterdam. 751pp.

Hargrove, W.W. and Pickering, J. (1992) Pseuadoreplication: *a sine qua non* for regional ecology. Landscape Ecology 6:251-258.

Harrison, A.D. (1965) Geographic distribution of riverine invertebrates in Southern Africa. Archive für Hydrobiologie 61:387-394.

Harrison, A.D. and Elsworth, J.F. (1958) Hydrobiological studies on the Great Berg River, Western Cape Province. Part IV. The effect of organic pollution on the fauna of parts of the Great Berg River system and of the Krom Stream, Stellenbosch. Transations of the Royal Society of South Africa 25:125-226.

Harrison, A.R. and Farina, T.D.W. (1965) A naturally turbid water with deleterious effects on the egg capsules of planorbid snails. Ann. Trop. Med. Para. 59:327-330.

Hawkes, H.A. (1982) Biological surveillance of rivers. Water pollution Control 1982:329-342.

Hellawell, J.M. (1977) Change in natural and managed ecosystems: detection, measurement and assessment. Proceedings of the Royal Society of London 197:31-57.

Hellawell, J.M. (1986) Biological indicators of freshwater pollution and environmental management. Elsevier Applied Science, London. 546pp.

Henderson, J.P. (1988) Fish farm effluent control and the development of an expert system. Unpublished PhD Thesis, University of Aston.

Hildrew, A.G., Townsend, C.R. and Francis, J. (1984) Community structure in some southern English streams: the influence of species interactions. Freshwater

Biology 14:297-310.

Hilsenhoff, W.L. (1977) Use of arthropods to evaluate water quality of streams. Technical Bulletin No. 100. US department of Nature Research: 16pp.

Hohls, D. (1996) National Biomonitoring Programme for Riverine Ecosystems: Framework document for the programme. NBP Report Series No. 1. Institute for Water Quality Studies, Department of Water Affairs and Forestry, Pretoria, South Africa: 39pp.

Hurlbert, S.H. (1971) The nonconcept of species diversity: a critique and alternative parameters. Ecology 52:577-586.

Hulbert,S.H. (1984) Pseudoreplication and the design of ecological field experiments. Ecological Monographs 54(2):187-211.

Hynes, H.B.N. (1960) The biology of polluted waters. Liverpool University Press, Liverpool.

Hynes, H.B.N. (1970) The ecology of running waters. Liverpool University Press, Liverpool.

Jansen, S. and Vegelius, J. (1981) Measures of ecological association. Oecologia 49:371-376.

Jones, J.G. (1990) Pollution from fish farms. Journal of IWEM 4:14-18.

Joubert, A. and Hurly, P.R. (1993) The use of daily flow data to classify South African rivers. In: King,J.M. and Tharme, R.E. Assessment of the instream flow incremental methodology and initial development of alternative instream flow methodologies in South Africa. WRC Report No. 295/1/94:286-368.

Karr, J.R. (1981) Assessment of biotic integrity using fish communities. Fisheries 6:21-27.

Karr, J.R. (1991) Biological integrity: A long-neglected aspect of water resource management. Ecological Applications 1:66-84.

Karr, J.R. (1993) Measuring biological integrity: Lessons from streams. In. Woodley, S., Kay, J. and Francis, G. (eds.) Ecological Integrity and the management of ecosystems. St Lucie Press, Canada:83-103.

Kelly, L.A., Bergheim, A. and Stellwagen, J. (1997) Particle size distribution of wastes from freshwater fish farms. Aquaculture International 5:65-78.

Kendra, W. (1991) Quality of salmonid hatchery effluents during a summer low-flow season. Trans. American Fisheries Society 120:43-51.

King, J.M. (1981) The distribution of invertebrate communities in a small South African river. Hydrobiologia 83:43-65.

King, J.M. (1982) An ecological study of the macroinvertebrate fauna of the Eerste River, Western Cape Province, South Africa. Unpublished PhD Thesis, Zoology Department, University of Cape Town.

King, J.M, Day, J.A. and Brown, C.A. (1991) Proposal to investigate the effects of trout farms on riverine biotas. Unpublished Research Proposal to Department of Water Affairs and Forestry. 10pp.

Korzeniewski, K., Banat, Z. and Moczulska, A. (1982) Changes in water of the Uniesc and Skotawa Rivers caused by intensive trout culture. Polskie Archiwum Hydrobiologii 29:683-691.

Krebs, C.J. (1985) Ecology: The experimental analysis of distribution and abundance. Harper, New York.

Lamberth, S.J., Stewart, B.A. and Davies, B.R. (1995) The processing of unconfined, naturally-entrained leaves in two low-order southern African mountain streams. South African Journal of Aquatic Sciences.

Lancaster, J. and Hildrew, A.C. (1993) Flow refugia and the microdistribution of lotic macroinvertebrates. Journal of the North American Benthological Society 12(4):385-393.

Lenat, D.R. (1988) Water quality assessment of streams using a qualitative collection method for macroinvertebrates. Journal of the North American Benthological Society 7:222-233.

Lenat, D.R. (1993) A biotic index for southeastern United States: derivation of tolerance values with criteria for assigning water-quality ratings. Journal of the North American Benthological Society 12:279-290.

Lenat, D.R. and Crawford, J.K. (1994) Effects of land use on water quality and aquatic biota of three North Carolina Piedmont streams. Hydrobiologia 294:185-199.

Leopold, A. (1949) A Sand County Almanac: and Sketches Here and There. Oxford University Press, New York.

Loch, D.D., West, J.L. and Perlmutter, D.G. (1996) The effects of trout farm effluent on the taxa richness of benthic macroinvertebrates. Aquaculture 147:37-55.

Malmqvist, B., Nilsson, L. and Svensson, B.S. (1978) Dynamics of detritus in a small stream in southern Sweden and its influence on the distribution of the bottom animal communities. Oikos 31:3-16.

Marchant, R. (1990) Robustness of classification and ordination techniques applied to macroinvertebrate communities for the La Trobe River, Victoria. Australian Journal of Marine and Freshwater Research 41:493-504.

Marcucella, H. and Abramson, C.I. (1978) Behavioural toxicity and teleost fish. In: Mofstofsky, D.I. (ed.) The behaviour of fish and other aquatic animals. Academic Press New York:33-77.

Maridet, L., Wasson, J.G., Philippe, M. and Amoros, C. (1995) Benthic organic matter dynamics in three streams: Riparian vegetation or bed morphology control? Archive für Hydrobiologie 132(4):415-425.

Marmonier, P. and Creuzé des Châtelliers, M. (1991) Effects of spates on interstitial assemblages of the Rhone River: Importance of heterogeneity. Hydrobiologia 210:243-251.

Matty, A.J. (1990) Feeds in the fight against pollution. Fish Farming International.

McCafferty, W.P. (1990) A preliminary guide to the Ephemeroptera of southern Africa. In: Aquatic Entomology; McCafferty,W.P., Jones & Barlett Publ., Inc. Boston, U.S.A.

McShaffrey, D. and McCafferty, W.P. (1988) Feeding behaviour of *Rithrogena pellucida* (Ephemeroptera: Ephemerellidae). Journal of Insect Behaviour 3:673-688.

Merschhemke, C. (1991) Development, testing and installation of automatic bioassays along the River Rhine. In: Whitton, B.A., Rott, E. and Friedrich, G. (eds.). Use of algae for monitoring rivers. Institut für Botanik, Universität Innsbruck:119-122.

Mostert, S.A. (1983) Procedures used in South Africa for the automatic photometric determination of micronutrients in seawater. South African Journal of Marine Science 1:189-198.

Minshall, G.W. (1967) The role of allochthonous detritus in the trophic structure of a woodland springbrook community. Ecology 48:139-149.

Minshall, G.W., Cummins, K.W., Petersen, R.C., Cushing, C. E., Bruns, D.A., Sedell, J.R. and Vannote, R.L. (1985) Developments in stream ecosystem theory. Canadian Journal of Fisheries and Aquatic Sciences 42:1045-1055.

Naiman, R.J., Johnston, C.A. and Kelley, J.C. (1988) Alteration of North American streams by beaver. Bioscience 38:753-762.

Nature Conservancy Council (of Scotland) (1990) Fish farming and the Scottish freshwater environment. NCC. Contract No. HF3-03-450.

Nobel, R.G. and Hemens, J. (1978) Inland water ecosystems in South Africa – A review of research needs. South African National Scientific Programmes Report No. 34. Council for Scientific and Indutrial Reseach, Pretoria, 150pp.

Norris, R.H., Lake, P.S. and Swain, R. (1982) Ecological effects of mine effluents on the South Esk River, north-eastern Tasmania. III. Benthic macroinvertebrates. Australian Journal of Marine and Freshwater Research 33:789-809.

O'Keeffe, J.H., Davies, B.R., King, J.M. and Skelton, P.H. 1989a. The conservation status of southern African rivers. In: Huntley, B.J (ed.) Biotic diversity in southern Africa: Concepts and conservation. Oxford University Press, Cape Town. 266-289.

O'Keeffe, J,H., King, J.M., Day, J.A., Roussouw, D., van der Zel, D.W. and Skorozewski, R. (1989b): General concepts and approaches to in-stream flow assessment. South African National Scientific Programmes Report 162. Council for Scientific and Industrial Research, Pretoria.

Oberdorff, T. and Porcher, J.P. (1994) An index of biotic integrity to assess biological impacts of salmonid farm effluents on receiving waters. Aquaculture 119:219-235.

Palmer, C.G. (1991) Benthic assemblage structure and the feeding biology of sixteen macroinvertebrate taxa from the Buffalo River, Eastern Cape, South Africa. Unpublished PhD Thesis, Rhodes University:256pp.

Palmer, C.G, O'Keeffe, J.H. and Palmer, A.R. (1991) Are macroinvertebrate assemblages in the Buffalo River, southern Africa, associated with particular biotopes? Journal of the North American Benthological Society 10(4):349-357.

Parsons, M. and Norris, R.H. (1996) The effect of habitat-specific sampling in the biological assessment of water quality using a predictive model. Freshwater Biology 36:419-434.

Pennak, R.W. (1978) Freshwater invertebrates of the United States. Second Edition. John Wiley & Sons, New York.

Phillips, M.J, Mowatt, A. and Clarke, R. (1988) Phosphorus wastage from a new low phosphorus and traditional salmon diet. Unpublished report to Fulmar Feeds.

Pielou, E.C. (1975) Ecological diversity. Wiley, New York:165pp.

Poole, W.C. and Stewart, K.W. (1976) The vertical distribution of macrozoobenthos within the substratum of the Brazos River, Texas. Hydrobiologia 50:151-160.

Prochazka, K., Stewart, B.A. and Davies, B.R. (1991) Leaf litter retention and its implications for shredder distribution in two headwater streams. Archive für Hydrobiologie 120:315-325.

Purser, O. and O'Sullivan, D. (1992) Effluents from freshwater aquaculture - reducing conflicts through suitable management. Waterkwalitiet by forelproduksie. Elsenburg Agricultural Development Institute. 58pp.

Quinn, J.M. and McFarlane, P.N. (1989) Effects of slaughterhouse and dairy factory wastewaters on epilithon: A comparison in laboratory streams. Water Research 23(10):1267-1273.

Quinn, J.M. and Hickey, C.W. (1993) Effects of sewage waste stabilization lagoon effluent on stream invertebrates. Journal of Aquatic Ecosystem Health 2:205-219.

Rabe, F.W. and Gibson, F. (1984) The effect of macrophyte removal on the distribution of selected invertebrates in a littoral environment. Journal of Freshwater Ecology 2(4):359-371.

Ractliffe, S.G., Davies, B.R., Stewart, B.A. and Snaddon, C.D. (1995) The influence of discharge on entrainment of bank litter in a headwater stream. Archive für Hydrobiologie 134(1):103-117.

Reid, G.K. and Wood, R.D. (1976) Ecology of inland waters and estuaries. Second Edition. D. van Nostrand Company, New York.

Resh, V.H. and Jackson, J.K. (1993) Rapid assessment approaches to biomonitoring using benthic macroinvertebrates. In: Rosenberg, D.M. and Resh, V.H. (eds.) Freshwater biomonitoring and benthic macroinvertebrates. Chapman & Hall, New York.

Reynolds, O. (1883) An experimental investigation of the circumstances which determine whether the motion of water shall be direct or sinuous, and the law of resistance in parallel channels. Transactions of the Royal Society of London:932-982.

Roberts, J.K. (1989) Low pollution feeds. Trout News 8 June 1989:22-27.

Rosenberg, D.M. and Resh, V.H. (eds.) (1993) Freshwater biomonitoring and benthic macroinvertebrates. Chapman and Hall, New York.

Sager, P.E. and Hasler, A.D. (1969) Species diversity in lacustrine phytoplankton. I. The components of the index of diversity from Shannon's formula. American Naturalist 103:51-59.

Scott, K.M.F. (In prep) A provisional key to genera of hydroptilid trichoptera from the afrotropical region.

Seager, J. and Abrahams, R.G. (1990) The impact of storm sewage discharges on the ecology of a small urban river. Water Scientific Research 22:163-171.

Sheath, R.G. and Cole, K.M. (1992) Biogeography of stream macroalgae in North America. Journal of Phycology 28:448-460.

Simpson, E.H. (1949) Measurement of diversity. Nature 163:688.

Smith, H. and Haig, A.J.N. (1991) The development of a control policy for fish farming. Journal of IWEM 5:641-648.

Sprules, W.G. and Munawar, M. (1986) Plankton size spectra in relation to ecosystem productivity, size and perturbation. Canadian Journal of Fisheries and Aquatic Sciences 43:1789-1794.

Storey, A.W., Bunn, S.E., Davies, P.M. and Edward, D.H. (1990) Classification of the macroinvertebrate fauna of two river systems in southwestern Australia in relation to physical and chemical parameters. Regulated Rivers: Research and Management 5:217-232.

Taylor, H.C. (1978) Capensis. In: Werger, M.J.A. Biogeography and ecology of southern Africa. W. Junk, Amsterdam:171-229.

Townsend, C.L. (1989) The patch dynamics concept of stream community ecology. Journal of the North American Benthological Society 8(1):36-50.

Tuchman, N.C. and Petersen, C.G. (1995) Substratum-associated microbiota. Water Environment Research 67(4):702-713.

Underwood, A.J. (1993) The mechanics of spatially replicated sampling programmes to detect environmental impacts in a variable world. Australian Journal of Ecology 18:99-116.

Uys, M.C., Goetsch, P-A. and O'Keeffe, J.H. (1996) National Biomonitoring Programme for Riverine Ecosystems: Ecological indicators, a review and recommendations. NBP Report Series No. 4. Institute for Water Quality Studies, Department of Water Affairs and Forestry, Pretoria, South Africa. 93 pp.

Vannote, R.L., Minshall, G.W., Cummins, G.W., Sedell, J.R. and Cushing, C.E. (1980) The river continuum concept. Canadian Journal of Fisheries and Aquatic Sciences 37:130-137.

Walker, K.F. (1985) A review of the ecological effects of river regulation in Australia. Hydrobiologia 125:111-112.

Ward, J.V. and Stanford, J.A. (eds.) (1979) The Ecology of Regulated Streams. Plenum Press, New York. 398pp.

Ward, J.V. and Stanford, J.A. (1983a) The serial discontinuity concept of lotic ecosystems. In: Fontaine, T.D. and Bartell (eds.) Dynamic of lotic ecosystems. Ann Arbor Science, Ann Arbor, Michigan:29-42.

Ward, J.V. and Stanford, J.A. (1983b) The intermediate disturbance hypothesis: an explanation of biotic diversity patterns in lotic ecosystems. In: Fontaine, T.D. and Bartell (eds.) Dynamic of lotic ecosystems. Ann Arbor Science, Ann Arbor, Michigan:347-356.

Warwick, R.M. (1993) Environmental impact studies on marine communities: pragmatic considerations. Australian Journal of Ecology 18:63-80.

Warwick, R.M. and Clarke, K.R. (1991) A comparison of some methods for analysing changes in benthic community structure. Journal of the Marine Biology Association of the United Kingdom 71:225-244.

Warwick, R.M. and Clarke, K.R. (1993) Comapring the severity of disturbance: a meta-analysis of marine macrobenthic community data. Marine Ecology Progress Series 92: 221-231.

Warwick, R.M., Pearson, T.H. and Ruswahyuni. (1987) Detection of pollution 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 reference to aquatic ecosystems. Water Research 18:653-694.

Water Act No. 54 (1956) Statutes of the Republic of South Africa - Water.

Whitehurst, I.T. and Lindsey, B.I. (1990) The impact of organic enrichment on benthic macroinvertebrate communities of a low land river. Water Research 24:625-630.

Whitton, B.A., Rott, E. and Friedrich, G. (1991). Use of algae for monitoring rivers. Institut für Botanik, Universität Innsbruck:119-122.

Wiederholm, T. (1984) Responses of aquatic insects to environmental pollution. In: Resh, V. and Rosenberg, D.M. (eds.) The Ecology of Aquatic Insects. Praeger Publications, Connecticut.

Wiesmann, D, Scheid, H. and Pfeffer, E. (1988) Water pollution with phosphorus of dietary origin by intensively fed rainbow trout (*Salmo gairdneri* Rich.). Auqaculture 69:263-270.

Wilmot, B.C. (In prep) A preliminary key to the identification othe the aquatic stages of the odenata of southern Africa.

Winterbourn, M.J., Rounick, J.S. and Cowie, B. (1981) Are New Zealand streams really different? New Zealand Journal of Marine and Freshwater Research 15:321-328.

Winterbourn, M.J. and Ryan, P.A. (1994) Mountain streams in Westland, New Zealand: benthic ecology and management issues. Freshwater Biology 32:359-373.

Wolda, H. (1981) Similarity indices, sample size and diversity. Oecologia 50:296-302.

Wright, J.F., Armitage, P.D., Furse, M.T. and Moss, D. (1988) A new approach to the biological surveillance of river quality using macroinvertebrates. Verh. Internat. Verien. Limnol. 23:1548-1552.

Wright, J.F., Moss, P.D., Armitage, P.D. and Furse, M.T. (1984) A preliminary classification of running-water sites in Great Britain based on macroinvertebrate species and the prediction of community type using environmental data. Freshwater Biology 14:221-256.

Wright, I.A., Chessman, B.C., Fairweather, P.G. and Benson, L.J. (1995) Measuring the impact of sewage effluent on the macroinvertebrate community of an upland stream: The effect of different levels of taxonomic resolution and quantification. Australian Journal of Ecology 20:142-149.

Zamora-Muñoz, C. and Alba-Tercedor, J. (1996) Bioassessment of organically polluted Spanish rivers, using a biotic index and multivariate methods. Journal of the North American Benthological Society 15(3):332-352.

Zar, J.H. (1984) Biostatistical analysis. Second Edition. Prentice-Hall, Inc., New Jersey.

APPENDIX A
MACROINVERTEBRATE DATA

Table A1 AUGUST 1992: Abundances (x10 m^2) for the macroinvertebrate taxa recorded at the sampling sites in **riffle biotopes**, upstream (PA1) and downstream (PB1-PB5) of the portapool trout farm

		Site code	PA1	PA1	PA1	PB1	PB1	PB1	PB2	PB2	PB2	PB3	PB3	PB4	PB4	PB4	PB5	PB5	PB5
		Biotope	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle
		Replicate number	1	2	3	1	2	3	1	3	1	2	3	1	2	3	1	2	3
Order	**Family**	**Genus/species**																	
DIPTERA	Simuliidae	*Simulium* sp.	3	0	0	92	19	1	236	1	9	67	5	5	9	4	28	29	34
	Chironomidae	*Tvetenia* sp	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		SpA	27	9	1	2	0	0	0	0	0	9	0	4	0	0	0	1	5
		SpAA	0	0	0	0	7	2	24	0	3	2	5	0	0	0	0	3	4
		SpX(AB)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		SpAB	71	49	93	124	186	41	258	106	29	76	35	8	4	0	46	62	70
		SpBB	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		SpD	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		SpF	4	1	0	0	0	0	0	0	0	0	0	0	0	0	1	0	4
		SpJ	5	0	0	0	0	0	8	4	0	0	4	0	0	0	4	0	0
		SpJJ	17	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		SpK	8	4	20	4	48	4	0	4	0	0	0	0	0	0	4	0	0
		SpP	0	0	0	0	0	1	1	4	0	0	0	0	0	4	0	0	0
		Sp Q	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		SpS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Tanypodinae	1	0	8	0	1	1	0	0	1	0	0	0	0	0	0	0	5
		Rheotanytarsus sp.	0	1	4	0	1	0	0	0	0	0	0	0	0	0	0	0	0
		Polypedilum sp.	0	3	6	0	1	3	0	0	0	0	0	0	0	0	0	0	0
		Chironomus sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Typanidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		'Cone worms'	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0
	Rhagionidae		0	3	1	0	4	0	0	0	0	0	0	0	0	0	0	0	0
	Ceratopogodinae		0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0

Table A1 cont AUGUST 1992: Abundances (x10 m^2) for the macroinvertebrate taxa recorded at the sampling sites in **riffle biotopes**, upstream (PA1) and downstream (PB1-PB5) of the portapool trout farm

		Site code	PA1	PA1	PA1	PB1	PB1	PB1	PB2	PB2	PB2	PB3	PB3	PB4	PB4	PB4	PB5	PB5	PB5
		Biotope	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle
		Replicate number	1	2	3	1	2	3	1	3	1	2	3	1	2	3	1	2	3
Order	**Family**	**Genus/species**																	
EMPHEMEROPTERA	Baetidae	*Baetis* sp.	59	49	69	0	24	5	3	8	21	4	3	0	0	3	16	19	9
		Acentrella sp.	77	10	60	28	0	0	56	9	58	8	13	14	1	1	72	174	152
	Leptophlebidae	*Castenophlebia* sp.	3	2	2	0	0	0	0	0	0	0	0	0	0	0	2	1	0
		Adenophlebia sp.	0	0	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Aprionyx sp.	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1	0
		Choroterpes sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
	Ephemerellidae	*Lestagella* sp.	11	5	3	0	0	1	0	0	0	0	0	1	0	0	1	5	4
		Lithogloea sp.	15	10	21	0	4	0	0	15	17	0	0	0	0	0	1	8	0
		Ephemerellina sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Heptageniidae	*Afronurus* sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Caenidae	*Austrocaenis* sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
COLEOPTERA	Elmidae	Elmid sp 1	0	2	0	0	0	8	0	0	0	0	0	0	0	0	0	0	0
		Elmid sp 2	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
		Elmid sp 3	10	1	9	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Dryopidae	Dryopid sp 1	63	0	3	0	0	0	0	0	0	0	0	0	0	0	1	0	0
		Dryopid sp 2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Helodidae	*Prionocyphon* sp. 1	36	12	52	0	8	0	0	0	4	1	4	0	0	0	12	0	0
		Prionocyphon sp. 2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PLECOPTERA	Notonemouridae	*Aphanicerca* sp.	23	9	7	0	5	2	9	0	13	0	9	1	0	0	2	9	20
		Aphanicercella sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Table A1$_{cont.}$ AUGUST 1992: Abundances (x10 m^2) for the macroinvertebrate taxa recorded at the sampling sites in **riffle biotopes**, upstream (PA1) and downstream (PB1-PB5) of the portapool trout farm

		Site code	PA1	PA1	PA1	PB1	PB1	PB1	PB2	PB2	PB2	PB3	PB3	PB4	PB4	PB4	PB5	PB5	PB5
		Biotope	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle
		Replicate number	1	2	3	1	2	3	1	3	1	2	3	1	2	3	1	2	3
Order	**Family**	**Genus/species**																	
TRICHOPTERA	Leptoceridae	*Athripsoddes* sp.	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Glossosomatidae	*Agapetus* sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Hydropsychidae	*Cheumatopsyche* sp.	0	0	0	1	0	0	0	0	0	0	0	0	1	0	0	0	0
	Philopotamidae	Chimarra	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Polycentropodidae		4	0	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		TRICOP X	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
MEGALOPTERA	Coryalidae	Chloriniella sp	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Chauliodinae spp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
HEMIPTERA			0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
COLLEMBOLA			0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
HYDRACARINA			0	0	0	0	0	0	0	0	4	0	0	0	4	0	0	4	0
ANNELIDA	Lumbriculidae	Gen1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Gen4	0	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Gen5	0	0	4	0	0	0	0	0	400	0	0	0	0	0	0	0	0
		Gen6	8	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Lumbriculus sp.	0	0	0	0	17	42	0	16	1	0	4	0	0	0	0	4	0
		Thin, curly	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0
		Earthy	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	0
	Naididae	*Nais* sp.	0	0	0	451	1516	1002	444	1044	50	272	407	298	112	401	356	582	238
TURBELLARIA		*Dugesia* sp.	0	2	4	7	19	25	4	3	6	17	0	0	1	2	4	0	0

Table A2 AUGUST 1992: Abundances (x10 m^2) for the macroinvertebrate taxa recorded at the sampling sites in **backwater biotopes**, upstream (PA1) and downstream (PB1-PB5) of the portapool trout farm

		Site code	PA1	PA1	PA1	PB1	PB1	PB1	PB3	PB3	PB3	PB5	PB5	PB5
		Biotope	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water
		Replicate number	1	2	3	1	2	3	1	2	3	1	2	3
Order	**Family**	**Genus/species**												
DIPTERA	Simuliidae	*Simulium* sp.	4	0	1	0	0	0	0	1	0	0	0	0
	Chironomidae	*Tvetenia* sp	0	0	0	0	0	0	0	0	0	0	0	0
		SpA	5	0	0	0	0	0	0	0	0	1	0	0
		SpAA	0	0	0	0	0	1	0	0	0	0	0	0
		SpX(AB)	0	0	0	0	0	0	0	0	0	0	0	0
		SpAB	21	12	1	12	11	12	84	8	26	8	82	57
		SpBB	0	0	0	0	0	0	0	0	0	0	0	0
		SpD	4	0	0	0	0	0	0	0	0	0	0	0
		SpF	0	0	0	0	0	0	0	0	0	0	0	0
		SpJ	0	0	4	0	0	0	8	0	0	4	4	0
		SpJJ	17	0	1	0	0	0	0	0	0	0	0	0
		SpK	0	12	1	0	0	16	8	0	0	0	0	5
		SpP	0	0	0	0	0	0	0	1	13	0	0	0
		Sp Q	1	0	0	0	0	0	0	0	0	0	0	0
		SpS	0	0	0	0	0	0	0	0	0	0	0	0
		Tanypodinae	0	4	1	0	1	1	0	1	0	0	0	1
		Rheotanytarsus sp.	0	0	0	1	32	0	0	0	0	0	4	0
		Polypedilum sp.	3	1	3	0	5	0	6	1	4	0	0	0
		Chironomus sp.	0	0	0	0	0	0	0	0	0	0	0	0
	Typanidae		0	0	0	0	0	0	0	0	0	0	0	0
		'Cone worms'	0	0	5	0	0	4	0	0	0	0	0	0
	Rhagionidae		0	0	0	0	0	0	0	0	1	0	0	0
	Ceratopogodinae		0	0	5	0	0	4	0	0	0	0	0	0

Table A2$_{cont.}$ AUGUST 1992: Abundances (x10 m^2) for the macroinvertebrate taxa recorded at the sampling sites in **backwater biotopes**, upstream (PA1) and downstream (PB1-PB5) of the portapool trout farm

		Site code	PA1	PA1	PA1	PB1	PB1	PB1	PB3	PB3	PB3	PB5	PB5	PB5
		Biotope	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water
		Replicate number	1	2	3	1	2	3	1	2	3	1	2	3
Order	**Family**	**Genus/species**												
EMPHEMEROPTERA	Baetidae	*Baetis* sp.	72	11	109	0	1	2	4	3	2	7	58	1
		Acentrella sp.	72	13	6	0	0	0	0	0	0	1	11	0
	Leptophlebidae	*Castenophlebia* sp.	12	2	1	0	0	0	0	1	0	0	0	0
		Adenophlebia sp.	4	6	1	0	1	0	1	3	3	0	0	0
		Aprionyx sp.	0	0	0	0	0	0	0	0	0	0	0	0
		Choroterpes sp.	1	0	0	0	0	0	0	0	0	0	1	0
	Ephemerellidae	*Lestagella* sp.	9	0	79	0	2	0	0	1	0	1	8	2
		Lithogloea sp.	15	2	37	6	13	0	34	1	6	0	0	1
		Ephemerellina sp.	0	0	0	0	0	0	0	0	0	0	0	0
	Heptageniidae	*Afronurus* sp.	0	0	0	0	0	0	0	0	0	0	0	0
	Caenidae	*Austrocaenis* sp.	0	0	0	0	0	0	0	0	0	0	0	0
COLEOPTERA	Elmidae	Elmid sp 1	25	0	0	0	0	0	0	0	0	0	0	0
		Elmid sp 2	32	0	0	0	4	2	0	0	0	0	0	0
		Elmid sp 3	0	20	4	0	0	0	0	0	0	0	0	0
	Dryopidae	Dryopid sp 1	25	18	2	0	0	0	0	0	0	0	0	0
		Dryopid sp 2	0	0	0	0	0	0	0	0	0	0	0	0
	Helodidae	*Prionocyphon* sp. 1	84	21	15	1	5	0	4	0	0	0	1	0
		Prionocyphon sp. 2	0	0	0	0	0	0	0	0	0	0	0	0
PLECOPTERA	Notonemouridae	*Aphanicerca* sp.	26	4	1	0	0	0	0	0	0	1	15	0
		Aphanicercella sp.	0	0	0	0	0	0	0	0	0	0	0	0

Table A2cont. AUGUST 1992: Abundances (x10 m^2) for the macroinvertebrate taxa recorded at the sampling sites in **backwater biotopes**, upstream (PA1) and downstream (PB1-PB5) of the portapool trout farm

		Site code	PA1	PA1	PA1	PB1	PB1	PB1	PB3	PB3	PB3	PB5	PB5	PB5
		Biotope	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water
		Replicate number	1	2	3	1	2	3	1	2	3	1	2	3
Order	**Family**	**Genus/species**												
TRICHOPTERA	Leptoceridae	*Athripsoddes* sp.	0	0	0	0	0	0	0	0	4	0	0	0
	Glossosomatidae	*Agapetus* sp.	4	0	0	0	0	0	0	0	0	0	0	0
	Hydropsychidae	*Cheumatopsyche* sp.	0	0	0	0	0	0	0	0	0	0	0	0
	Philopotamidae	*Chimarra* sp.	1	0	0	0	0	0	0	0	0	0	0	0
	Polycentropodidae		0	0	2	0	0	0	0	0	0	0	0	0
		TRICOP X	0	0	0	0	0	0	0	0	0	0	0	0
MEGALOPTERA	Coryalidae	*Chloriniella* sp	0	0	0	0	0	0	0	0	0	0	0	0
		Chauliodinae spp.	0	0	0	0	0	0	0	0	0	0	0	0
HEMIPTERA			0	0	0	0	0	0	0	0	0	0	0	0
COLLEMBOLA			0	0	0	0	0	0	0	0	0	0	0	0
HYDRACARINA			0	0	0	0	0	0	0	0	0	0	0	0
ANNELIDA	Lumbriculidae	Gen4	0	0	0	0	0	0	0	0	0	0	0	0
		Gen5	0	0	0	0	0	0	0	0	0	0	0	0
		Gen6	0	0	0	0	0	0	0	0	0	0	0	0
		Lumbriculus sp.	0	0	0	38	39	49	15	11	76	0	0	0
		Thin, curly	4	4	0	0	0	0	0	0	0	0	0	0
		Earthy	4	0	0	0	0	0	0	0	0	0	0	0
	Naididae	*Nais* sp.	0	0	0	1489	882	888	464	576	944	555	608	530
TURBELLARIA		*Dugesia* sp.	16	16	0	36	15	8	45	22	6	8	8	0

Table A3 MARCH 1993: Abundances (x10 m^2) for the macroinvertebrate taxa recorded at the sampling sites in **riffle biotopes**, upstream (PA1) and downstream (PB1-PB5) of the portapool trout farm

		Site	PA1	PA1	PA1	PB1	PB1	PB1	PB2	PB2	PB3	PB3	PB4	PB4	PB4	PB5	PB5	PB5
		Biotope	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle
		Replicate	1	2	3	1	2	3	2	3	2	3	1	2	3	1	2	3
Order	**Family**	**Genus/species**																
DIPTERA	Simuliidae	*Simulium* sp.	54	54	54	0	13	0	0	6	0	0	0	0	0	0	0	0
	Chironomidae	*Tvetenia* sp	5	0	6	0	4	0	0	0	1	0	0	0	0	0	1	0
		SpA	0	0	0	0	0	0	45	0	0	0	0	0	0	0	0	0
		SpAA	203	80	159	18	127	62	0	171	123	274	104	128	114	217	169	472
		SpX(AB)	0	22	62	24	0	21	0	12	35	17	20	5	20	0	56	43
		SpAB	152	176	198	0	159	37	20	39	0	71	66	45	50	39	131	130
		SpBB	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		SpD	0	0	0	0	1	0	0	0	0	0	0	0	0	0	1	0
		SpF	0	0	4	0	0	0	0	0	0	0	0	0	0	0	20	0
		SpJ	44	36	21	0	0	4	0	0	0	0	0	4	0	0	0	13
		SpJJ	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		SpK	4	1	5	0	0	0	0	0	0	0	0	0	0	0	0	0
		SpP	2	6	9	22	100	49	2	0	36	61	23	17	236	89	17	147
		Sp Q	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		SpS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Tanypodinae	22	12	61	8	27	6	0	13	8	7	62	43	125	37	17	81
		Rheotanytarsus sp.	50	79	54	84	129	28	32	127	127	10	313	563	916	685	202	584
		Polypedilum sp.	14	5	32	6	9	0	0	4	0	0	0	1	0	0	8	47
		Chironomus sp.	0	0	0	9	7	24	34	5	1	0	0	1	0	2	0	8
	Typanidae		0	4	0	0	0	0	0	0	0	0	0	0	0	0	1	0
		'Cone worms'	0	4	2	0	0	0	0	0	0	0	0	0	0	0	0	0
	Rhagionidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Ceratopogodinae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Table A3cont. MARCH 1993: Abundances (x10 m^2) for the macroinvertebrate taxa recorded at the sampling sites in **riffle biotopes**, upstream (PA1) and downstream (PB1-PB5) of the portapool trout farm

		Site	PA1	PA1	PA1	PB1	PB1	PB1	PB2	PB2	PB3	PB3	PB4	PB4	PB4	PB5	PB5	PB5
		Biotope	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle
		Replicate	1	2	3	1	2	3	2	3	2	3	1	2	3	1	2	3
Order	**Family**	**Genus/species**																
EMPHEMEROPTERA	Baetidae	*Baetis* sp.	10	41	15	2	25	15	8	0	18	20	16	0	21	45	31	9
		Acentrella sp.	158	14	61	0	1	0	0	0	0	6	0	16	0	1	242	217
	Leptophlebidae	*Castenophlebia* sp.	1	1	8	0	0	0	0	0	0	0	0	2	0	0	0	0
		Adenophlebia sp.	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
		Aprionyx sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Choroterpes sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Ephemerellidae	*Lestagella* sp.	13	13	20	0	0	0	0	0	0	0	0	0	0	0	0	0
		Lithogloea sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Ephemerellina sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Heptageniidae	*Afronurus* sp.	0	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0
	Caenidae	*Austrocaenis* sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
COLEOPTERA	Elmidae	Elmid sp 1	0	0	13	0	1	0	0	0	0	0	4	6	4	0	8	4
		Elmid sp 2	0	0	24	0	0	0	0	0	0	0	0	0	4	4	4	8
		Elmid sp 3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Dryopidae	Dryopid sp 1	63	27	87	0	0	0	0	0	0	0	1	0	0	0	1	0
		Dryopid sp 2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Helodidae	*Prionocyphon* sp. 1	28	11	14	0	0	0	0	0	0	0	0	0	0	0	0	0
		Prionocyphon sp. 2	1	1	7	0	0	0	0	0	0	0	0	0	0	0	0	0
PLECOPTERA	Notonemouridae	*Aphanicerca* sp.	15	1	20	0	0	0	0	0	0	0	0	0	0	5	3	5
		Aphanicercella sp.	0	1	0	0	0	0	0	0	0	0	0	0	0	5	1	0

Table A3$_{cont.}$ MARCH 1993: Abundances (x10 m^2) for the macroinvertebrate taxa recorded at the sampling sites in **riffle biotopes**, upstream (PA1) and downstream (PB1-PB5) of the portapool trout farm

		Site	PA1	PA1	PA1	PB1	PB1	PB1	PB2	PB2	PB3	PB3	PB4	PB4	PB4	PB5	PB5	PB5
		Biotope	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle
		Replicate	1	2	3	1	2	3	2	3	2	3	1	2	3	1	2	3
Order	**Family**	**Genus/species**																
TRICHOPTERA	Leptoceridae	*Athripsoddes* sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Glossosomatidae	*Agapetus* sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Hydropsychidae	*Cheumatopsyche* sp.	12	1	17	0	1	0	0	0	0	0	0	1	2	4	47	103
	Philopotamidae	*Chimarra* sp.	0	0	6	0	0	0	0	0	0	0	0	0	0	0	0	4
	Polycentropodidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		TRICOP X	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
MEGALOPTERA	Coryalidae	*Chloriniella* sp	7	0	6	0	0	0	0	0	0	0	0	0	0	0	0	0
		Chauliodinae spp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
HEMIPTERA			0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
COLLEMBOLA			0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
HYDRACARINA			0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0
ANNELIDA	Lumbriculidae	Gen4	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
		Gen5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Gen6	0	0	325	0	0	0	0	0	0	0	0	0	0	0	0	0
		Lumbriculus sp.	0	0	0	55	40	171	283	81	24	26	16	16	32	270	21	25
		Thin, curly	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Earthy	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Naididae	*Nais* sp.	0	0	0	517	1535	836	304	320	266	58	0	0	44	132	44	200
TURBELLARIA		*Dugesia* sp.	0	0	10	49	402	115	58	245	72	92	8	4	60	251	135	169

Table A4 MARCH 1993: Abundances (x10 m^2) for the macroinvertebrate taxa recorded at the sampling sites in **backwater biotopes**, upstream (PA1) and downstream (PB1-PB5) of the portapool trout farm

		Site	PA1	PA1	PA1	PB1	PB1	PB1	PB2	PB2	PB2	PB3	PB3	PB3	PB4	PB4	PB4	PB5	PB5	PB5
		Biotope	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water
		Replicate	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3
Order	**Family**	**Genus/species**																		
DIPTERA	Simuliidae	*Simulium* sp.	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Chironomidae	*Tvetenia* sp	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		SpA	0	0	0	0	0	0	0	45	0	0	0	0	0	0	0	0	0	0
		SpAA	0	0	4	0	0	0	4	0	8	104	0	104	0	20	0	57	37	57
		SpX(AB)	4	0	0	0	0	0	0	0	0	0	0	0	0	0	12	0	0	0
		SpAB	16	23	3	9	0	10	31	20	36	15	36	16	0	1	44	66	19	24
		SpBB	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		SpD	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		SpF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		SpJ	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	5	0
		SpJJ	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		SpK	4	12	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		SpP	0	0	0	4	0	0	0	2	32	0	20	0	1	119	44	24	4	0
		Sp Q	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		SpS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Tanypodinae	8	26	33	1	0	2	6	0	0	67	0	63	27	128	61	71	28	98
		Rheotanytarsus sp.	20	8	12	0	0	0	0	32	12	525	8	17	33	1398	169	252	100	244
		Polypedilum sp.	12	14	2	0	0	2	0	0	8	0	0	0	0	0	0	0	0	1
		Chironomus sp.	0	0	0	121	194	304	108	34	67	16	0	16	1	0	0	0	0	27
	Typanidae		0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		'Cone worms'	4	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Rhagionidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0
	Ceratopogodinae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Table A4 cont. MARCH 1993: Abundances (x10 m^2) for the macroinvertebrate taxa recorded at the sampling sites in **backwater biotopes**, upstream (PA1) and downstream (PB1-PB5) of the portapool trout farm

		Site	PA1	PA1	PA1	PB1	PB1	PB1	PB2	PB2	PB2	PB3	PB3	PB3	PB4	PB4	PB4	PB5	PB5	PB5
		Biotope	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water
		Replicate	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3
Order	**Family**	**Genus/species**																		
EMPHEMEROPTERA	Baetidae	*Baetis* sp.	0	0	0	1	0	0	0	8	0	40	40	0	0	0	1	3	5	4
		Acentrella sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	2	9	1	0	0
	Leptophlebidae	*Castenophlebia* sp.	2	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Adenophlebia sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Aprionyx sp.	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Choroterpes sp.	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Ephemerellidae	*Lestagella* sp.	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Lithogloea sp.	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4
		Ephemerellina sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Heptageniidae	*Afronurus* sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Caenidae	*Austrocaenis* sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
COLEOPTERA	Elmidae	Elmid sp 1	4	0	0	0	0	0	0	0	4	0	0	1	0	0	0	8	0	0
		Elmid sp 2	0	4	0	0	0	0	0	0	0	4	0	0	0	0	0	16	0	0
		Elmid sp 3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Dryopidae	Dryopid sp 1	9	10	33	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
		Dryopid sp 2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Helodidae	*Prionocyphon* sp. 1	0	0	8	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Prionocyphon sp. 2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PLECOPTERA	Notonemouridae	*Aphanicerca* sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4
		Aphanicercella sp.	0	0	0	0	0	0	0	0	0	32	0	0	0	1	0	4	0	0

Table A4 cont. MARCH 1993: Abundances (x10 m^2) for the macroinvertebrate taxa recorded at the sampling sites in **backwater biotopes**, upstream (PA1) and downstream (PB1-PB5) of the portapool trout farm

		Site	PA1	PA1	PA1	PB1	PB1	PB1	PB2	PB2	PB2	PB3	PB3	PB3	PB4	PB4	PB4	PB5	PB5	PB5
		Biotope	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water
		Replicate	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3
Order	**Family**	**Genus/species**																		
TRICHOPTERA	Leptoceridae	*Athripsoddes* sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Glossosomatidae	*Agapetus* sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Hydropsychidae	*Cheumatopsyche* sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Philopotamidae	*Chimarra* sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Polycentropodidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		TRICOP X	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
MEGALOPTERA	Coryalidae	*Chloriniella* sp	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Chauliodinae spp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
HEMIPTERA			0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
COLLEMBOLA			0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
HYDRACARINA			0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ANNELIDA	Lumbriculidae	Gen4	0	0	0	0	0	0	0	0	0	44	0	48	0	0	0	0	0	0
		Gen5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Gen6	0	0	24	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Lumbriculus sp.	0	0	0	174	25	111	25	294	84	152	152	160	9	8	10	12	22	48
		Thin, curly	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Earthy	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Naididae	*Nais* sp.	0	4	0	324	124	806	362	524	94	76	72	76	28	0	0	0	12	105
TURBELLARIA		*Dugesia* sp.	2	0	0	9	0	84	14	58	32	64	5	64	0	2	0	0	0	23

Table A4 JUNE 1993: Abundances (x10 m^2) for the macroinvertebrate taxa recorded at the sampling sites in **riffle biotopes**, upstream (PA1) and downstream (PB1-PB5) of the portapool trout farm

		Site	PA1	PA1	PA1	PB1	PB1	PB1	PB2	PB2	PB3	PB3	PB3	PB4	PB4	PB4	PB5	PB5	PB5
		Biotope	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle
		Replicate	1	2	3	1	2	3	1	2	1	2	3	1	2	3	1	2	3
Order	**Family**	**Genus/species**																	
DIPTERA	Simuliidae	*Simulium* sp.	0	24	1	25	0	0	19	124	21	66	47	60	6	1	36	16	11
	Chironomidae	*Tvetenia* sp	0	1	0	32	0	0	16	12	23	6	56	137	4	10	32	20	34
		SpA	0	0	0	0	24	4	71	57	37	64	68	0	25	13	76	36	61
		SpAA	0	0	0	8	0	0	4	0	0	0	0	4	1	0	4	4	0
		SpX(AB)	8	19	26	92	40	88	294	254	145	238	161	114	57	70	132	120	252
		SpAB	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		SpBB	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		SpD	4	4	0	9	0	0	4	12	4	13	8	12	8	4	4	4	4
		SpF	4	28	49	18	12	41	28	8	3	8	21	28	8	1	56	4	16
		SpJ	0	0	0	0	0	0	0	0	0	0	0	0	0	0	8	0	0
		SpJJ	4	0	4	0	8	32	0	0	8	0	0	12	0	0	8	0	0
		SpK	0	0	9	0	4	4	0	0	5	8	0	8	8	1	4	0	0
		SpP	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Sp Q	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		SpS	0	8	0	0	0	0	0	0	3	4	0	0	0	1	8	0	8
		Tanypodinae	0	0	8	0	4	8	8	0	12	6	4	12	8	4	8	0	22
		Rheotanytarsus sp.	0	4	8	0	0	4	0	0	1	0	1	0	1	0	0	4	0
		Polypedilum sp.	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	4
		Chironomus sp.	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Typanidae		0	4	0	17	32	53	25	30	12	4	16	12	24	0	0	8	7
		'Cone worms'	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Rhagionidae		0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Ceratopogodinae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Table A4 cont. JUNE 1993: Abundances (x10 m^2) for the macroinvertebrate taxa recorded at the sampling sites in **riffle biotopes**, upstream (PA1) and downstream (PB1-PB5) of the portapool trout farm

		Site	PA1	PA1	PA1	PB1	PB1	PB1	PB2	PB2	PB3	PB3	PB3	PB4	PB4	PB4	PB5	PB5	PB5
		Biotope	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle
		Replicate	1	2	3	1	2	3	1	2	1	2	3	1	2	3	1	2	3
Order	**Family**	**Genus/species**																	
EMPHEMEROPTERA	Baetidae	*Baetis* sp.	0	3	39	4	33	1	33	23	49	26	30	20	1	16	10	1	25
		Acentrella sp.	51	71	16	0	0	8	0	26	0	46	27	20	7	4	128	27	21
	Leptophlebidae	*Castenophlebia* sp.	0	4	2	0	1	0	0	0	0	0	0	0	1	0	0	0	0
		Adenophlebia sp.	0	0	0	0	0	3	0	0	0	0	0	0	0	0	4	1	0
		Aprionyx sp.	0	7	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Choroterpes sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Ephemerellidae	*Lestagella* sp.	18	2	14	16	9	15	8	0	8	12	8	0	12	5	4	5	10
		Lithogloea sp.	4	4	25	4	0	8	0	0	0	8	1	0	0	0	0	0	4
		Ephemerellina sp.	0	0	0	1	0	12	0	0	0	0	0	4	0	0	0	0	0
	Heptageniidae	*Afronurus* sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Caenidae	*Austrocaenis* sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
COLEOPTERA	Elmidae	Elmid sp 1	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Elmid sp 2	0	8	8	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Elmid sp 3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Dryopidae	Dryopid sp 1	0	0	19	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Dryopid sp 2	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Helodidae	*Prionocyphon* sp. 1	22	56	171	0	0	8	1	0	4	0	4	0	0	0	0	0	4
		Prionocyphon sp. 2	0	4	0	0	0	0	0	0	0	0	4	0	0	0	0	0	0
PLECOPTERA	Notonemouridae	*Aphanicerca* sp.	12	48	22	24	16	107	30	10	11	21	18	56	21	0	108	9	0
		Aphanicercella sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Table A4 cont. JUNE 1993: Abundances (x10 m^2) for the macroinvertebrate taxa recorded at the sampling sites in **riffle biotopes**, upstream (PA1) and downstream (PB1-PB5) of the portapool trout farm

		Site	PA1	PA1	PA1	PB1	PB1	PB1	PB2	PB2	PB3	PB3	PB3	PB4	PB4	PB4	PB5	PB5	PB5
		Biotope	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle
		Replicate	1	2	3	1	2	3	1	2	1	2	3	1	2	3	1	2	3
Order	**Family**	**Genus/species**																	
TRICHOPTERA	Leptoceridae	*Athripsoddes* sp.	0	1	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0
	Glossosomatidae	*Agapetus* sp.	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Hydropsychidae	*Cheumatopsyche* sp.	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0
	Philopotamidae	*Chimarra* sp.	0	4	1	0	0	0	0	0	0	4	0	0	0	0	0	0	0
	Polycentropodidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		TRICOP X	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0
MEGALOPTERA	Coryalidae	*Chloriniella* sp	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Chauliodinae spp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
HEMIPTERA			0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
COLLEMBOLA			0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
HYDRACARINA			0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0
ANNELIDA	Lumbriculidae	Gen4	0	0	8	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Gen5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Gen6	0	20	16	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Lumbriculus sp.	0	0	0	29	6	124	205	8	27	6	51	34	10	0	10	9	6
		Thin, curly	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Earthy	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Naididae	*Nais* sp.	0	0	0	580	144	324	666	399	604	192	400	584	113	49	548	44	914
TURBELLARIA		*Dugesia* sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Table A5 JUNE 1993: Abundances (x10 m^2) for the macroinvertebrate taxa recorded at the sampling sites in **backwater biotopes**, upstream (PA1) and downstream (PB1-PB5) of the portapool trout farm

		Site	PA1	PA1	PA1	PB1	PB1	PB1	PB2	PB2	PB2	PB3	PB3	PB3	PB4	PB4	PB4	PB5	PB5	PB5
		Biotope	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water
		Replicate	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3
Order	**Family**	**Genus/species**																		
DIPTERA	Simuliidae	*Simulium* sp.	4	0	0	0	0	0	0	0	8	0	22	0	8	4	0	0	16	0
	Chironomidae	*Tvetenia* sp	0	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0
		SpA	0	0	4	0	0	0	0	0	8	1	20	1	0	4	0	0	32	28
		SpAA	0	0	0	0	2	0	0	1	8	0	28	3	28	16	4	9	48	28
		SpX(AB)	0	0	0	0	7	0	0	0	0	0	0	0	0	0	0	0	0	0
		SpAB	0	2	0	34	48	3	53	91	124	89	47	61	77	78	20	212	120	84
		SpBB	0	0	0	0	0	8	0	0	0	0	0	0	0	0	0	0	4	0
		SpD	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		SpF	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	8	0
		SpJ	16	36	0	0	8	0	1	20	10	0	9	9	8	19	16	16	8	24
		SpJJ	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		SpK	4	0	4	0	12	8	0	0	4	12	8	0	0	16	8	0	12	4
		SpP	0	0	0	0	0	8	12	1	0	8	4	1	32	1	0	4	0	12
		Sp Q	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		SpS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Tanypodinae	4	29	0	0	0	0	4	0	0	0	0	4	8	1	20	0	0	16
		Rheotanytarsus sp.	0	0	0	0	0	4	4	0	4	0	8	0	59	28	0	25	0	32
		Polypedilum sp.	4	6	0	0	4	0	0	0	0	0	1	1	1	0	0	4	0	4
		Chironomus sp.	0	0	0	10	0	0	3	13	5	0	0	0	5	0	1	1	0	0
	Typanidae		5	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		'Cone worms'	0	8	24	71	76	25	0	16	31	11	40	17	8	16	4	28	20	3
	Rhagionidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4	0	0
	Ceratopogodinae		0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Table A5 cont. JUNE 1993: Abundances (x10 m^2) for the macroinvertebrate taxa recorded at the sampling sites in **backwater biotopes**, upstream (PA1) and downstream (PB1-PB5) of the portapool trout farm

		Site	PA1	PA1	PA1	PB1	PB1	PB1	PB2	PB2	PB2	PB3	PB3	PB3	PB4	PB4	PB4	PB5	PB5	PB5
		Biotope	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water
		Replicate	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3
Order	**Family**	**Genus/species**																		
EMPHEMEROPTERA	Baetidae	*Baetis* sp.	27	0	4	0	4	5	0	0	16	3	26	9	20	12	4	2	36	0
		Acentrella sp.	8	16	32	0	4	0	0	0	0	0	30	4	0	4	0	0	51	8
	Leptophlebidae	*Castenophlebia* sp.	4	12	8	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
		Adenophlebia sp.	4	0	1	0	0	1	0	0	0	0	1	0	0	0	0	0	1	1
		Aprionyx sp.	0	0	2	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
		Choroterpes sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Ephemerellidae	*Lestagella* sp.	32	17	26	0	4	4	0	12	32	4	9	10	0	12	0	2	12	0
		Lithogloea sp.	25	33	17	0	0	4	0	4	16	8	2	0	5	0	0	4	0	0
		Ephemerellina sp.	0	0	0	0	0	0	0	0	0	0	0	0	8	0	0	0	0	0
	Heptageniidae	*Afronurus* sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Caenidae	*Austrocaenis* sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
COLEOPTERA	Elmidae	Elmid sp 1	0	0	12	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Elmid sp 2	40	32	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Elmid sp 3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Dryopidae	Dryopid sp 1	54	38	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Dryopid sp 2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Helodidae	*Prionocyphon* sp. 1	29	40	29	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Prionocyphon sp. 2	6	10	2	0	0	0	0	0	0	0	4	0	0	4	0	0	0	0
PLECOPTERA	Notonemouridae	*Aphanicerca* sp.	23	76	11	0	6	8	2	0	4	7	40	6	1	15	0	1	23	10
		Aphanicercella sp.	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Table A5 cont. JUNE 1993: Abundances (x10 m^2) for the macroinvertebrate taxa recorded at the sampling sites in **backwater biotopes**, upstream (PA1) and downstream (PB1-PB5) of the portapool trout farm

		Site	PA1	PA1	PA1	PB1	PB1	PB1	PB2	PB2	PB2	PB3	PB3	PB3	PB4	PB4	PB4	PB5	PB5	PB5
		Biotope	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water
		Replicate	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3
Order	**Family**	**Genus/species**																		
TRICHOPTERA	Leptoceridae	*Athripsoddes* sp.	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Glossosomatidae	*Agapetus* sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Hydropsychidae	*Cheumatopsyche* sp.	0	0	0	0	0	0	1	0	0	0	0	4	0	0	0	0	0	0
	Philopotamidae	*Chimarra* sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Polycentropodidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		TRICOP X	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
MEGALOPTERA		*Chloriniella* sp	0	0	0	0	0	0	0	0	0	0	0	0	0	4	0	0	0	0
		Chauliodinae spp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
HEMIPTERA			0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
COLLEMBOLA			0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	4	0
HYDRACARINA			0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ANNELIDA	Lumbriculidae	Gen4	8	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Gen5	0	8	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Gen6	24	226	4	0	0	0	0	0	40	0	0	0	0	0	0	0	0	0
		Lumbriculus sp.	0	0	0	626	210	51	866	98	160	17	16	31	167	30	8	8	13	78
		Thin, curly	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Earthy	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Naididae	*Nais* sp.	0	0	0	1084	366	435	308	632	444	10	1476	450	492	484	120	154	564	564
TURBELLARIA		*Dugesia* sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Table A6 MARCH 1993: Abundances (x10 m^2) for the macroinvertebrate taxa recorded at the sampling sites in **riffle biotopes**, upstream (PA1) and downstream (PB1-PB5) of the earthdam trout farm

		Site	EA1	EA1	EA1	EB1	EB1	EB2	EB2	EB2	EB3	EB3	EB3	EB4	EB4	EB4	EB5	EB5	EB5
		Biotope	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle
		Replicate	1	2	3	2	3	1	2	3	1	2	3	1	2	3	1	2	3
Order	**Family**	**Genus/species**																	
DIPTERA	Simuliidae	*Simulium* sp.	68	12	29	216	1747	16	73	131	850	255	80	48	88	48	32	8	4
	Chironomidae	SpA	0	0	0	0	361	0	0	0	0	0	0	0	0	0	0	0	0
		SpAA	0	0	0	24	0	36	28	10	32	8	8	0	0	0	0	0	4
		SpAB	0	0	0	20	32	100	68	60	15	20	44	18	0	16	8	8	0
		SpF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		SpJ	12	0	4	0	4	4	0	0	4	0	4	4	0	0	0	0	0
		SpJJ	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		SpK	0	0	4	8	0	0	0	0	0	4	4	0	0	4	0	0	8
		SpP	8	0	0	12	20	16	36	32	44	28	44	20	0	8	8	0	0
		Tanypodinae sp.	8	0	28	24	12	112	81	61	32	98	104	150	80	140	84	68	60
		Rheotanytarsus sp.	32	4	4	4	0	0	8	0	0	0	0	28	0	16	8	0	4
		Polypedilum sp.	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Chironomus sp.	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0
EMPHEMEROPTERA	Baetidae	*Baetis* sp.	361	32	51	272	330	135	265	208	56	237	117	762	52	37	114	242	185
		Acentrella sp.	29	226	269	425	462	187	143	355	731	620	569	174	690	488	1077	107	112
	Leptophlebiidae	*Castenophlebia* sp.	183	76	114	40	20	20	46	34	24	32	64	92	50	96	68	116	54
		Adenophlebia sp.	2	1	1	12	0	12	2	0	0	4	4	11	0	6	4	3	3
		Aprionyx sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Choroterpes sp.	8	0	0	44	4	32	88	6	28	36	28	0	98	20	37	56	111
	Ephemerellidae	*Lestagella* sp.	133	185	148	184	41	419	184	136	48	84	124	377	108	17	120	299	309
		Lithogloea sp.	8	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4	0
		Ephemerellina sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Heptageniidae	*Afronurus* sp.	2	4	2	0	0	0	0	0	0	0	0	4	0	0	0	0	0
	Caenidae	*Austrocaenis* sp.	0	0	0	0	0	0	4	0	4	0	0	0	0	0	0	0	0

Table A6cont. MARCH 1993: Abundances (x10 m^2) for the macroinvertebrate taxa recorded at the sampling sites in **riffle biotopes**, upstream (PA1) and downstream (PB1-PB5) of the earthdam trout farm

		Site	EA1	EA1	EA1	EB1	EB1	EB2	EB2	EB2	EB3	EB3	EB3	EB4	EB4	EB4	EB5	EB5	EB5
		Biotope	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle
		Replicate	1	2	3	2	3	1	2	3	1	2	3	1	2	3	1	2	3
Order	**Family**	**Genus/species**																	
COLEOPTERA	Elmidae	Elmid sp 1	0	4	0	8	4	0	4	12	12	32	20	0	4	2	0	4	4
		Elmid sp 2	0	0	4	8	0	16	16	28	0	25	16	14	24	9	28	60	36
		Elmid sp 3	0	0	4	0	0	0	0	0	0	0	0	64	0	0	0	0	0
	Dryopidae	Dryopid sp 1	4	1	0	0	0	12	8	0	0	1	0	0	0	4	0	0	0
		Dryopid sp 2	4	2	2	8	0	4	0	0	4	0	0	0	4	0	0	0	0
	Helodidae	*Prionocyphon* sp. 1	77	44	15	36	32	0	20	48	20	172	104	33	260	160	216	296	148
		Prionocyphon sp. 2	28	5	1	12	0	80	8	0	0	0	4	0	8	4	16	32	0
PLECOPTERA	Notonemouridae	*Aphanicerca* sp.	235	2	151	0	8	0	0	5	32	16	4	9	64	20	39	25	4
		Aphanicercella sp.	0	0	0	20	8	16	24	10	16	0	16	2	0	4	4	1	32
TRICHOPTERA	Leptoceridae	Athripsoddes sp.	0	0	0	0	0	12	0	4	0	0	0	0	8	4	0	48	8
	Glossosomatidae	Agapetus sp.	0	21	25	1	0	0	0	0	0	0	4	16	4	0	20	40	32
	Hydropsycinae	*Cheumatopsyche* sp.	4	3	0	1	8	8	4	20	107	63	121	44	84	92	34	20	0
	Philopotamidae	*Chimarra* sp.	12	0	0	0	15	8	0	4	160	299	115	88	230	160	120	36	0
MEGALOPTERA	Coryalidae	*Chloriniella* sp.	2	0	1	3	1	0	0	4	3	1	0	0	1	0	1	2	0
ANNELIDA		*Lumbriculus* sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Naididae	*Nais* sp.	0	0	0	0	0	0	0	0	84	0	0	164	0	0	0	0	0
TURBELLARIA		*Dugesia* sp.	0	0	0	57	20	9	12	16	4	24	0	0	0	0	4	12	0

Table A7 MARCH 1993: Abundances (x10 m^2) for the macroinvertebrate taxa recorded at the sampling sites in **backwater biotopes**, upstream (PA1) and downstream (PB1-PB5) of the earthdam trout farm

		Site	EA1	EA1	EB1	EB1	EB1	EB2	EB2	EB2	EB3	EB3	EB3	EB4	EB4	EB4	EB5	EB5	EB5
		Biotope	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water
		Replicate	1	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3
Order	**Family**	**Genus/species**																	
DIPTERA	Simuliidae	*Simulium* sp.	0	0	0	0	4	0	12	0	20	0	0	8	0	0	0	12	0
	Chironomidae	SpA	0	0	0	0	0	0	0	0	0	0	0	0	0	4	0	0	0
		SpAA	0	0	0	0	0	0	8	0	0	0	0	0	0	0	0	0	0
		SpAB	0	0	0	0	0	0	4	0	4	0	0	4	4	4	0	4	12
		SpF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		SpJ	0	0	0	0	0	0	8	0	0	0	0	0	0	0	0	0	0
		SpJJ	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		SpK	4	0	0	4	0	0	0	4	4	0	28	0	8	12	4	8	28
		SpP	0	0	0	0	0	0	0	0	8	4	36	0	0	16	0	4	0
		Tanypodinae sp.	6	1	8	12	0	36	40	60	64	72	64	8	64	280	0	136	172
		Rheotanytarsus sp.	1	0	16	0	0	0	4	4	0	4	4	0	0	0	0	0	12
		Polypedilum sp.	0	5	0	0	0	0	0	0	0	0	0	0	0	8	0	4	4
		Chironomus sp.	0	0	0	0	3	0	0	0	0	0	0	0	0	0	0	0	0
EMPHEMEROPTERA	Baetidae	*Baetis* sp.	79	134	158	40	23	321	300	340	426	19	484	273	260	140	204	144	242
		Acentrella sp.	12	0	0	0	1	4	4	8	49	40	44	4	0	10	0	12	38
	Leptophlebiidae	*Castenophlebia* sp.	4	15	0	1	0	0	49	8	0	7	3	8	4	12	4	32	84
		Adenophlebia sp.	4	15	3	12	0	1	6	11	2	0	12	8	5	6	36	34	29
		Aprionyx sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Choroterpes sp.	0	8	1	0	0	0	0	0	0	22	5	0	0	0	0	8	60
	Ephemerellidae	*Lestagella* sp.	36	46	8	16	0	8	149	139	100	227	172	32	328	702	20	116	320
		Lithogloea sp.	4	5	0	4	0	0	8	0	0	0	12	0	4	0	0	0	0
		Ephemerellina sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Heptageniidae	*Afronurus* sp.	1	4	0	0	0	0	0	0	0	10	4	0	0	5	0	0	0

Table A7cont. MARCH 1993: Abundances (x10 m^2) for the macroinvertebrate taxa recorded at the sampling sites in **backwater biotopes**, upstream (PA1) and downstream (PB1-PB5) of the earthdam trout farm

		Site	EA1	EA1	EB1	EB1	EB1	EB2	EB2	EB2	EB3	EB3	EB3	EB4	EB4	EB4	EB5	EB5	EB5
		Biotope	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water
		Replicate	1	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3
Order	**Family**	**Genus/species**																	
EMPHEMEROPTERA	Caenidae	*Austrocaenis* sp.	5	0	0	4	0	12	0	48	76	4	8	0	0	64	4	0	4
COLEOPTERA	Elmidae	Elmid sp 1	0	0	0	0	0	0	20	20	8	4	8	0	0	0	0	0	4
		Elmid sp 2	6	0	0	0	0	8	12	4	8	32	4	20	4	52	0	4	80
		Elmid sp 3	0	0	0	0	5	0	0	0	0	0	0	0	12	0	0	0	0
	Dryopidae	Dryopid sp 1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	4	4
		Dryopid sp 2	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Helodidae	*Prionocyphon* sp. 1	0	0	0	0	0	0	12	8	12	0	8	0	4	36	0	84	80
		Prionocyphon sp. 2	0	0	0	0	0	0	12	0	0	8	0	0	0	0	0	0	8
PLECOPTERA	Notonemouridae	*Aphanicerca* sp.	5	0	0	8	0	0	0	0	0	0	12	8	0	4	0	13	0
		Aphanicercella sp.	0	16	4	0	0	20	28	0	8	4	4	0	0	0	8	0	0
TRICHOPTERA	Leptoceridae	Athripsoddes sp.	0	0	0	12	0	0	0	0	0	4	0	0	0	0	0	0	0
	Glossosomatidae	Agapetus sp.	0	0	0	0	0	0	12	0	0	0	0	0	0	4	0	0	4
	Hydropsycinae	*Cheumatopsyche* sp.	0	0	0	0	0	0	0	0	0	0	4	0	0	0	0	4	8
	Philopotamidae	*Chimarra* sp.	0	0	0	0	0	0	12	0	0	4	16	4	0	8	0	28	0
MEGALOPTERA	Coryalidae	*Chloriniella* sp.	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	1
ANNELIDA		*Lumbriculus* sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Naididae	*Nais* sp.	0	0	0	0	38	0	0	0	0	0	0	0	0	0	0	0	0
TURBELLARIA		*Dugesia* sp.	0	0	15	76	0	4	11	28	12	0	8	4	25	4	0	4	24

Table A8 JUNE 1993: Abundances (x10 m^2) for the macroinvertebrate taxa recorded at the sampling sites in **riffle biotopes**, upstream (PA1) and downstream (PB1-PB5) of the earthdam trout farm

		Site	EA1	EA1	EA1	EB1	EB1	EB1	EB2	EB2	EB2	EB3	EB3	EB3	EB4	EB4	EB4	EB5	EB5	EB5
		Biotope	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle
		Replicate	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3
Order	**Family**	**Genus/species**																		
DIPTERA	Simuliidae	*Simulium* sp.	0	0	0	13	8	8	52	24	9	4	76	20	0	13	0	36	4	4
	Chironomidae	SpA	8	0	0	8	86	24	509	154	126	80	49	144	5	179	16	24	12	0
		SpAA	0	0	0	0	0	0	0	0	3	0	0	0	0	0	8	4	0	0
		SpAB	0	8	8	20	0	16	56	17	72	16	36	0	0	20	44	100	4	36
		SpF	4	0	0	0	5	4	4	4	0	12	4	16	0	0	0	40	0	0
		SpJ	4	0	8	12	0	4	12	0	0	0	4	16	0	0	4	0	8	0
		SpJJ	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		SpK	0	4	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0	4
		SpP	0	0	0	0	0	0	0	0	0	0	4	4	0	0	0	0	0	0
		Tanypodinae sp.	4	0	0	8	0	0	0	0	8	0	0	4	0	0	0	0	12	24
		Rheotanytarsus sp.	0	0	12	0	0	0	0	0	4	0	0	0	0	0	8	4	4	0
		Polypedilum sp.	0	0	0	0	0	4	0	0	0	0	1	0	0	0	0	0	0	8
		Chironomus sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
EMPHEMEROPTERA	Baetidae	*Baetis* sp.	0	12	0	12	5	7	2	12	9	0	9	32	0	11	44	27	22	2
		Acentrella sp.	28	12	20	32	15	35	71	35	42	59	11	97	50	240	21	461	182	55
	Leptophlebiidae	*Castenophlebia* sp.	72	32	55	19	0	37	8	40	16	8	2	28	12	13	12	17	20	25
		Adenophlebia sp.	0	0	0	1	0	0	0	0	1	5	0	0	0	0	0	0	0	16
		Aprionyx sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Choroterpes sp.	0	0	8	0	0	4	4	0	0	12	0	0	4	0	0	36	56	69
	Ephemerellidae	*Lestagella* sp.	221	232	285	41	16	162	53	78	69	44	15	127	28	53	139	434	252	130
		Lithogloea sp.	4	0	24	0	4	29	0	0	0	0	0	9	0	0	28	0	1	0
		Ephemerellina sp.	8	0	0	0	0	0	0	0	0	12	5	0	0	0	0	0	0	0
	Heptageniidae	*Afronurus* sp.	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	5	0	0
	Caenidae	*Austrocaenis* sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Table A8$_{cont.}$ JUNE 1993: Abundances (x10 m^2) for the macroinvertebrate taxa recorded at the sampling sites in **riffle biotopes**, upstream (PA1) and downstream (PB1-PB5) of the earthdam trout farm

		Site	EA1	EA1	EA1	EB1	EB1	EB1	EB2	EB2	EB2	EB3	EB3	EB3	EB4	EB4	EB4	EB5	EB5	EB5
		Biotope	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle	Riffle
		Replicate	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3
Order	**Family**	**Genus/species**																		
COLEOPTERA	Elmidae	Elmid sp 1	0	8	4	0	0	0	8	0	0	0	0	0	0	0	0	4	0	12
		Elmid sp 2	8	0	0	0	0	8	0	4	4	0	4	0	0	4	4	0	0	12
		Elmid sp 3	0	0	4	0	0	0	0	0	0	0	8	4	0	0	0	4	0	0
	Dryopidae	Dryopid sp 1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Dryopid sp 2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Helodidae	*Prionocyphon* sp. 1	30	12	52	93	24	28	49	20	116	44	8	64	100	16	28	13	120	84
		Prionocyphon sp. 2	0	0	0	8	0	0	0	0	0	0	0	8	0	0	8	0	0	0
PLECOPTERA	Notonemouridae	*Aphanicerca* sp.	55	19	57	93	2	76	60	39	13	43	27	87	31	46	4	74	65	0
		Aphanicercella sp.	0	2	0	0	0	4	0	0	1	0	0	0	0	0	0	0	0	32
TRICHOPTERA	Leptoceridae	Athripsoddes sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	16	4	0	0
	Glossosomatidae	Agapetus sp.	12	69	156	0	8	24	0	16	40	0	0	40	0	4	0	84	20	16
	Hydropsycinae	*Cheumatopsyche* sp.	0	5	0	13	2	4	0	4	0	6	22	5	4	0	0	15	5	4
	Philopotamidae	*Chimarra* sp.	0	13	4	17	15	12	0	8	4	33	47	21	8	24	0	27	17	149
MEGALOPTERA	Coryalidae	*Chloriniella* sp.	1	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	3
ANNELIDA		*Lumbriculus* sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Naididae	*Nais* sp.	0	0	0	0	28	0	0	0	0	0	36	0	0	0	0	0	0	0
TURBELLARIA		*Dugesia* sp.	0	0	0	0	0	0	4	8	8	0	0	0	16	0	8	4	4	4

Table A9 JUNE 1993: Abundances (x10 m^2) for the macroinvertebrate taxa recorded at the sampling sites in **backwater biotopes**, upstream (PA1) and downstream (PB1-PB5) of the earthdam trout farm

		Site	EA1	EA1	EA1	EB1	EB1	EB2	EB2	EB2	EB3	EB3	EB3	EB4	EB4	EB4	EB5	EB5	EB5
		Biotope	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water
		Replicate	1	2	3	1	2	1	2	3	1	2	3	1	2	3	1	2	3
Order	**Family**	**Genus/species**																	
DIPTERA	Simuliidae	*Simulium* sp.	0	4	0	0	4	12	0	4	0	0	0	4	4	0	0	4	0
	Chironomidae	SpA	0	0	0	0	0	32	0	32	0	24	4	8	0	32	0	0	0
		SpAA	0	0	0	0	0	0	0	0	0	0	0	20	0	0	0	0	0
		SpAB	12	12	0	4	12	12	0	24	8	8	8	56	0	12	0	4	0
		SpF	0	0	0	0	0	0	0	0	0	0	0	0	4	0	0	0	0
		SpJ	4	0	0	0	4	20	0	4	0	12	0	0	0	8	0	12	0
		SpJJ	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		SpK	0	0	0	8	4	0	0	0	0	0	0	4	0	0	0	4	0
		SpP	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Tanypodinae sp.	0	0	0	0	4	0	0	0	0	4	2	4	0	4	0	0	0
		Rheotanytarsus sp.	0	0	0	0	0	8	0	0	0	0	0	0	0	0	0	4	0
		Polypedilum sp.	4	0	0	0	0	0	0	0	0	4	0	4	0	0	0	0	0
		Chironomus sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
EMPHEMEROPTERA	Baetidae	*Baetis* sp.	0	5	20	454	28	31	51	0	24	17	105	103	17	64	64	241	178
		Acentrella sp.	21	18	0	8	48	32	16	24	24	44	0	39	7	85	4	8	2
	Leptophlebiidae	*Castenophlebia* sp.	45	40	18	4	24	24	48	20	0	13	20	33	0	16	12	28	18
		Adenophlebia sp.	1	4	2	2	3	0	16	0	1	0	0	9	0	3	3	4	1
		Aprionyx sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Choroterpes sp.	28	8	1	0	0	0	0	16	0	0	0	0	4	8	0	0	16
	Ephemerellidae	*Lestagella* sp.	201	68	12	42	79	107	13	78	43	187	82	148	69	188	31	206	40
		Lithogloea sp.	4	31	8	9	44	20	12	0	24	40	20	21	4	21	20	32	1
		Ephemerellina sp.	4	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0
	Heptageniidae	*Afronurus* sp.	1	0	4	0	0	0	0	0	0	0	0	0	0	5	0	0	4
	Caenidae	*Austrocaenis* sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Table A9cont. JUNE 1993: Abundances (x10 m^2) for the macroinvertebrate taxa recorded at the sampling sites in **backwater biotopes**, upstream (PA1) and downstream (PB1-PB5) of the earthdam trout farm

		Site	EA1	EA1	EA1	EB1	EB1	EB2	EB2	EB2	EB3	EB3	EB3	EB4	EB4	EB4	EB5	EB5	EB5
		Biotope	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water	B-water
		Replicate	1	2	3	1	2	1	2	3	1	2	3	1	2	3	1	2	3
Order	**Family**	**Genus/species**																	
COLEOPTERA	Elmidae	Elmid sp 1	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	4	0
		Elmid sp 2	0	0	0	0	4	4	0	0	0	8	0	0	4	0	0	8	0
		Elmid sp 3	0	0	0	0	0	0	0	0	0	0	0	0	0	4	0	0	0
	Dryopidae	Dryopid sp 1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Dryopid sp 2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Helodidae	*Prionocyphon* sp. 1	34	52	20	8	192	33	76	44	20	60	4	185	144	28	16	56	28
		Prionocyphon sp. 2	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0
PLECOPTERA	Notonemouridae	*Aphanicerca* sp.	28	54	1	16	40	61	32	40	18	106	4	16	0	45	0	8	1
		Aphanicercella sp.	4	0	0	0	0	0	49	0	0	4	0	5	0	0	6	9	2
TRICHOPTERA	Leptoceridae	Athripsoddes sp.	4	0	0	0	12	0	0	0	0	4	4	0	0	4	0	0	0
	Glossosomatidae	Agapetus sp.	8	4	0	0	148	0	0	4	0	8	0	0	0	0	0	12	16
	Hydropsycinae	*Cheumatopsyche* sp.	3	0	0	0	0	4	0	0	0	8	0	0	0	2	0	0	4
	Philopotamidae	*Chimarra* sp.	5	0	4	0	16	4	0	0	0	20	0	21	0	17	0	0	8
MEGALOPTERA	Coryalidae	*Chloriniella* sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ANNELIDA		*Lumbriculus* sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Naididae	*Nais* sp.	0	0	0	0	0	0	0	16	0	0	0	0	0	0	0	0	0
TURBELLARIA		*Dugesia* sp.	0	0	0	10	0	0	8	0	0	0	0	0	4	0	0	0	0

APPENDIX B
MODELLED DATA

Appendix B Abundances generated using the community model developed in Chapter 6

[Suspended organic solid]	*Simulium* sp.	*Tvetenia* sp.	Orthoclad AA	Orthoclad X	Orthoclad AB	Orthoclad J	Tanypod sp.	*Rheotanytarsus* sp.	*Polypedilum* sp.	*Chironomus* sp.	*Baetis harrisoni*	*Acentralla capensis*
0.45	164.47	7.29	446.00	102.85	517.96	75.00	163.81	241.32	63.90	0.00	55.17	200.00
0.50	182.75	7.08	446.00	101.44	511.26	75.00	177.90	383.80	62.14	1.20	57.68	200.00
0.55	201.02	6.90	479.12	100.03	504.55	74.86	191.42	521.18	60.54	2.40	60.07	211.40
0.60	219.30	6.72	512.23	98.61	497.85	64.30	204.36	653.48	59.09	3.60	62.37	222.80
0.65	237.57	6.57	545.35	97.20	491.15	55.90	216.73	780.68	57.75	4.80	64.57	234.20
0.70	255.85	6.42	578.47	95.78	484.45	49.10	228.52	902.78	56.51	6.00	66.66	245.60
0.75	274.12	6.28	611.59	94.37	477.75	43.52	239.74	1,019.80	55.36	7.20	68.66	257.00
0.80	292.40	6.16	644.70	92.96	471.05	38.88	250.38	1,131.72	54.28	8.40	70.55	268.40
0.85	310.67	6.04	677.82	91.54	464.35	34.96	260.45	1,238.56	53.27	9.60	72.34	279.80
0.90	328.95	5.92	710.94	90.13	457.64	31.64	269.94	1,340.30	52.31	10.80	74.03	291.20
0.95	347.22	5.82	744.05	88.72	450.94	28.78	278.86	1,436.94	51.41	12.00	75.61	302.60
1.00	365.50	5.72	777.17	87.30	444.24	26.31	287.20	1,528.50	50.55	13.20	77.10	314.00
1.05	383.77	5.62	810.29	85.89	437.54	24.16	294.97	1,614.96	49.73	14.40	78.48	325.40
1.10	402.05	5.53	843.40	84.47	430.84	22.27	302.16	1,696.34	48.95	15.60	79.77	336.80
1.15	420.32	5.44	876.52	83.06	424.14	20.61	308.78	1,772.62	48.21	16.80	80.95	348.20
1.20	438.60	5.36	909.64	81.65	417.44	19.13	314.82	1,843.80	47.50	18.00	82.03	359.60
1.25	456.87	5.28	942.76	80.23	410.73	17.81	320.29	1,909.90	46.82	19.20	83.01	371.00
1.30	475.15	5.20	975.87	78.82	404.03	16.63	325.18	1,970.90	46.16	20.40	83.88	382.40
1.35	493.42	5.13	1,008.99	77.41	397.33	15.57	329.50	2,026.82	45.53	21.60	84.66	393.80
1.40	511.70	5.05	1,042.11	75.99	390.63	14.61	333.24	2,077.64	44.92	22.80	85.33	405.20
1.45	529.97	4.98	1,075.22	74.58	383.93	13.74	336.41	2,123.36	44.33	24.00	85.90	416.60
1.50	548.24	4.92	1,108.34	73.16	377.23	12.95	339.00	2,164.00	43.77	25.20	86.38	428.00
1.55	566.52	4.85	1,081.14	71.75	370.53	12.23	341.02	2,199.54	43.22	26.40	86.74	341.20
1.60	584.79	4.79	1,053.94	70.34	363.83	11.57	342.46	2,230.00	42.69	27.60	87.01	254.40
1.65	603.07	4.73	1,026.74	68.92	357.12	10.96	343.33	2,255.36	42.17	28.80	87.18	167.60
1.70	621.34	4.67	999.54	67.51	350.42	10.40	343.62	2,275.62	41.68	30.00	87.24	80.80
1.75	639.62	4.61	972.34	66.10	343.72	9.89	343.34	2,290.80	41.19	31.20	87.21	78.35
1.80	657.89	4.56	945.14	64.68	337.02	9.41	342.48	2,300.88	40.72	32.40	87.07	75.90
1.85	676.17	4.50	917.94	63.27	330.32	8.97	341.05	2,305.88	40.26	33.60	86.83	73.45
1.90	694.44	4.45	890.74	61.85	323.62	8.56	339.04	2,305.78	39.82	34.80	86.49	71.00
1.95	712.72	4.40	863.54	60.44	316.92	8.18	336.46	2,300.58	39.38	36.00	86.04	68.55
2.00	730.99	4.35	836.34	59.03	310.21	7.83	333.30	2,290.30	38.96	37.20	85.50	66.10
2.05	749.27	4.30	809.14	57.61	303.51	7.50	329.57	2,274.92	38.54	38.40	84.85	63.65
2.10	767.54	4.25	781.94	56.20	296.81	7.19	325.26	2,254.46	38.14	39.60	84.11	61.20
2.15	785.82	4.21	754.74	54.79	290.11	6.90	320.38	2,228.90	37.75	40.80	83.26	58.75
2.20	804.09	4.16	727.54	53.37	283.41	6.63	314.92	2,198.24	37.36	42.00	82.31	56.30
2.25	822.37	4.12	700.34	51.96	276.71	6.37	308.89	2,162.50	36.99	43.20	81.26	53.85
2.30	840.64	4.07	673.14	50.55	270.01	6.13	302.28	2,121.66	36.62	44.40	80.10	51.40
2.35	858.92	4.03	645.94	49.13	263.31	5.91	295.10	2,075.74	36.26	45.60	78.85	48.95
2.40	877.19	3.99	618.74	47.72	256.60	5.69	287.34	2,024.72	35.91	46.80	77.49	46.50

Appendix B cont. Abundances generated using the community model developed in Chapter 6

[Suspended organic solid]	*Simulium* sp.	*Tvetenia* sp.	Orthoclad AA	Orthoclad X	Orthoclad AB	Orthoclad J	Tanypod sp.	*Rheotanytarsus* sp.	*Polypedilum* sp.	*Chironomus* sp.	*Baetis harrisoni*	*Acentralla capensis*
2.45	895.47	3.95	591.54	46.30	249.90	5.49	279.01	1,968.60	35.56	48.00	76.03	44.05
2.50	913.74	3.91	564.34	44.89	243.20	5.30	270.10	1,907.40	35.23	49.20	74.48	41.60
2.55	932.02	3.87	537.14	43.48	236.50	5.12	260.62	1,841.10	34.90	110.00	72.81	39.15
2.60	950.29	3.83	509.94	42.06	229.80	4.95	250.56	1,769.72	34.57	170.80	71.05	36.70
2.65	968.57	3.80	482.74	40.65	223.10	4.79	239.93	1,693.24	34.25	231.60	69.19	34.25
2.70	986.84	3.76	455.54	39.24	216.40	4.63	228.72	1,611.66	33.94	292.40	67.22	31.80
2.75	1,005.12	3.72	428.34	37.82	209.69	4.49	216.94	1,525.00	33.63	353.20	65.16	29.35
2.80	1,023.39	3.69	401.14	36.41	202.99	4.35	204.58	1,433.24	33.33	414.00	62.99	26.90
2.85	1,041.67	3.65	373.94	34.99	196.29	4.21	191.65	1,336.40	33.04	474.80	60.72	24.45
2.90	1,059.94	3.62	346.74	33.58	189.59	4.09	178.14	1,234.46	32.74	535.60	58.35	22.00
2.95	1,078.22	3.58	319.54	32.17	182.89	3.97	164.06	1,127.42	32.46	596.40	55.87	19.55
3.00	1,096.49	3.55	292.34	30.75	176.19	3.85	149.40	1,015.30	32.18	657.20	53.30	17.10
3.05	1,114.76	3.52	265.14	29.34	169.49	3.74	134.17	898.08	31.90	600.07	50.62	14.65
3.10	1,133.04	3.49	237.94	27.93	162.78	3.64	118.36	775.78	31.63	542.94	47.85	12.20
3.15	1,151.31	3.45	210.74	26.51	156.08	3.54	101.98	648.38	31.36	485.81	44.97	9.75
3.20	1,169.59	3.42	183.54	25.10	149.38	3.44	85.02	515.88	31.10	428.68	41.99	7.30
3.25	1,187.86	3.39	156.34	23.68	142.68	3.35	67.49	378.30	30.84	371.55	38.91	4.85
3.30	1,206.14	3.36	129.14	22.27	135.98	3.26	49.38	235.62	30.58	314.42	35.72	2.40
3.35	1,224.41	3.33	101.94	20.86	129.28	3.18	30.70	87.86	30.33	257.29	32.44	0.00
3.40	1,242.69	3.30	74.74	19.44	122.58	3.10	11.44	0.00	30.08	200.16	29.05	0.00

Appendix B cont. Abundances generated using the community model developed in Chapter 6

[Suspended organic solid]	*Castanophlebia* sp.	*Lestagella* sp.	Elmid sp.	Dryopid sp.	*Prionocyphon* sp.	*Aphanicerca* sp.	*Cheumatopsyche* sp.	*Lumbriculus* sp.	*Nais* sp.	*Dugesia* sp.
0.45	9.37	1.00	20.48	1.10	1.10	36.00	30.00	0.00	2.54	14.01
0.50	9.04	1.00	20.16	1.10	1.10	36.00	30.00	11.24	3.64	17.02
0.55	8.75	0.95	19.84	1.03	1.03	34.56	36.00	22.76	5.06	20.30
0.60	8.48	0.90	19.51	0.95	0.95	33.12	42.00	34.28	6.82	23.85
0.65	8.23	0.85	19.19	0.88	0.88	31.68	48.00	45.80	8.98	27.65
0.70	8.01	0.80	18.87	0.81	0.81	30.24	54.00	57.32	11.59	31.71
0.75	7.79	0.75	18.54	0.74	0.74	28.80	60.00	68.84	14.69	36.03
0.80	7.60	0.70	18.22	0.66	0.66	27.36	66.00	80.36	18.34	40.60
0.85	7.41	0.65	17.90	0.59	0.59	25.92	72.00	91.88	22.59	45.41
0.90	7.23	0.60	17.57	0.52	0.52	24.48	78.00	103.40	27.49	50.48
0.95	7.07	0.55	17.25	0.44	0.44	23.04	84.00	114.92	33.11	55.79
1.00	6.91	0.50	16.93	0.37	0.37	21.60	90.00	126.44	39.50	61.34
1.05	6.76	0.45	16.60	0.30	0.30	20.16	96.00	137.96	46.71	67.13
1.10	6.62	0.40	16.28	0.22	0.22	18.72	102.00	149.48	54.81	73.16
1.15	6.48	0.35	15.96	0.15	0.15	17.28	108.00	161.00	63.86	79.43
1.20	6.35	0.30	15.63	0.08	0.08	15.84	114.00	172.52	73.92	85.94
1.25	6.22	0.25	15.31	0.01	0.01	14.40	120.00	184.04	85.06	92.68
1.30	6.10	0.20	14.99	0.00	0.00	12.96	126.00	195.55	97.34	99.65
1.35	5.99	0.15	14.66	0.00	0.00	11.52	132.00	207.07	110.82	106.86
1.40	5.87	0.10	14.34	0.00	0.00	10.08	138.00	218.59	125.58	114.29
1.45	5.77	0.05	14.02	0.00	0.00	8.64	144.00	230.11	141.69	121.96
1.50	5.66	0.00	13.69	0.00	0.00	7.20	150.00	241.63	159.20	129.85
1.55	5.56	0.00	13.37	0.00	0.00	5.76	120.00	253.15	178.20	137.97
1.60	5.46	0.00	13.05	0.00	0.00	4.32	90.00	264.67	198.75	146.31
1.65	5.37	0.00	12.72	0.00	0.00	2.88	60.00	276.19	220.93	154.88
1.70	5.28	0.00	12.40	0.00	0.00	1.44	30.00	287.71	244.81	163.67
1.75	5.19	0.00	12.08	0.00	0.00	0.00	3.00	299.23	270.46	172.69
1.80	5.10	0.00	11.75	0.00	0.00	0.00	0.00	310.75	297.97	181.93
1.85	5.02	0.00	11.43	0.00	0.00	0.00	0.00	322.27	327.40	191.38
1.90	4.93	0.00	11.11	0.00	0.00	0.00	0.00	333.79	358.83	201.06
1.95	4.85	0.00	10.78	0.00	0.00	0.00	0.00	345.31	392.35	210.96
2.00	4.78	0.00	10.46	0.00	0.00	0.00	0.00	356.83	428.04	221.07
2.05	4.70	0.00	10.14	0.00	0.00	0.00	0.00	368.35	465.96	231.40
2.10	4.63	0.00	9.81	0.00	0.00	0.00	0.00	379.87	506.21	241.95
2.15	4.55	0.00	9.49	0.00	0.00	0.00	0.00	391.38	548.86	252.71
2.20	4.48	0.00	9.17	0.00	0.00	0.00	0.00	402.90	594.00	263.69
2.25	4.41	0.00	8.84	0.00	0.00	0.00	0.00	414.42	641.71	274.88
2.30	4.35	0.00	8.52	0.00	0.00	0.00	0.00	425.94	692.08	286.28
2.35	4.28	0.00	8.20	0.00	0.00	0.00	0.00	437.46	745.19	297.90
2.40	4.21	0.00	7.87	0.00	0.00	0.00	0.00	448.98	801.13	309.73

Appendix B cont. Abundances generated using the community model developed in Chapter 6

[Suspended organic solid]	*Castanophlebia* sp.	*Lestagella* sp.	Elmid sp.	Dryopid sp.	*Prionocyphon* sp.	*Aphanicerca* sp.	*Cheumatopsyche* sp.	*Lumbriculus* sp.	*Nais* sp.	*Dugesia* sp.
2.45	4.15	0.00	7.55	0.00	0.00	0.00	0.00	460.50	859.98	321.77
2.50	4.09	0.00	7.23	0.00	0.00	0.00	0.00	472.02	921.84	334.02
2.55	4.03	0.00	6.90	0.00	0.00	0.00	0.00	483.54	986.78	346.48
2.60	3.97	0.00	6.58	0.00	0.00	0.00	0.00	495.06	1,054.91	359.15
2.65	3.91	0.00	6.26	0.00	0.00	0.00	0.00	506.58	1,126.30	372.03
2.70	3.85	0.00	5.93	0.00	0.00	0.00	0.00	518.10	1,201.06	385.12
2.75	3.80	0.00	5.61	0.00	0.00	0.00	0.00	529.62	1,279.27	398.41
2.80	3.74	0.00	5.29	0.00	0.00	0.00	0.00	541.14	1,361.02	411.91
2.85	3.69	0.00	4.96	0.00	0.00	0.00	0.00	552.66	1,446.41	425.62
2.90	3.63	0.00	4.64	0.00	0.00	0.00	0.00	564.18	1,535.54	439.54
2.95	3.58	0.00	4.32	0.00	0.00	0.00	0.00	575.69	1,628.48	453.65
3.00	3.53	0.00	3.99	0.00	0.00	0.00	0.00	587.21	1,725.35	467.98
3.05	3.48	0.00	3.67	0.00	0.00	0.00	0.00	598.73	1,826.24	482.51
3.10	3.43	0.00	3.35	0.00	0.00	0.00	0.00	610.25	1,931.24	497.24
3.15	3.38	0.00	3.02	0.00	0.00	0.00	0.00	621.77	2,040.45	512.17
3.20	3.33	0.00	2.70	0.00	0.00	0.00	0.00	633.29	2,153.97	527.31
3.25	3.28	0.00	2.38	0.00	0.00	0.00	0.00	644.81	2,271.90	542.65
3.30	3.23	0.00	2.05	0.00	0.00	0.00	0.00	656.33	1,605.24	558.20
3.35	3.19	0.00	1.73	0.00	0.00	0.00	0.00	667.85	938.58	573.94
3.40	3.14	0.00	1.41	0.00	0.00	0.00	0.00	679.37	271.92	589.88