Some of the Ecological Effects of a Small Inter-Basin Water Transfer on the Receiving Reaches of the Upper Berg River, Western Cape

by

Catherine D. Snaddon

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Abstract

Inter-basin water transfer (IBT) is defined as the transfer of water from one geographically isolated river catchment, sub-catchment or river reach, to another. Thus, one river becomes a donor, and another the recipient. They are joined by a range of possible transfer routes. In many cases either or both the donor and recipient rivers are impounded. The volumes of water transferred and the operational criteria for IBT schemes vary considerably.

This study produced a review of the literature dealing with the ecological (physical, chemical and biological) effects of IBTs. The review highlighted the fact that much of the information available is conceptual, and provides few data collected from systems affected by transfer schemes. The main objective of this study was, therefore, to collect data from a donor and recipient river of a small IBT scheme in the Western Cape, the Riviersonderend-Berg-Eerste River Government Water Scheme (RBEGS).

The donor river, the Riviersonderend, is impounded by the Theewaterskloof Dam. Water from this reservoir is transferred through a tunnel to the upper reaches of the Berg River, and then on to the Cape Metropolitan Area (CMA). Approximately 38-45% of the water utilised in the CMA is transferred from Theewaterskloof. The release of water into the upper Berg is for irrigation in the upper catchment, and for use by a rainbow trout farm downstream of the outlet. The IBT tunnel supplies a maximum discharge of 5 $m^3 s^{-1}$ during summer months, while water transfer ceases during the winter months.

The Riviersonderend and Berg rivers were visited on a bi-monthly basis, from March 1994 through to April 1995. Two sampling sites on the Riviersonderend above the Theewaterskloof reservoirs were visited, while two above the IBT and one below it, were visited on the Berg River. Physical and chemical data were collected, and the benthic macroinvertebrate communities were sampled at each site. In addition to univariate analyses of all the data, multivariate techniques were used in the analysis of the benthic invertebrate data. The statistical package PRIMER (Plymouth Routines in Multivariate Ecological Research) was used to generate similarity dendrograms and multi-dimensional scaling ordination plots. The main objectives of data collection and analysis were to compare the

donor and recipient catchments, and to investigate the effects of transfer on the receiving reaches of the Berg River.

The water chemistry of the Riviersonderend differed significantly from that of the Berg River. Conductivity and the concentration of total dissolved solids (TDS) were higher in the Riviersonderend, as were the concentrations of sodium, magnesium and calcium ions. pH and temperature were lower in the Riviersonderend. Furthermore, a comparison of data between the Berg River sites above the IBT, and the site below it, showed that the release of water during the summer months led to an increase in pH, conductivity, TDS and total suspended solids (TSS) downstream of the IBT. These alterations in water chemistry accompanied the substantial increases in discharge and stream width due to the release of water into the Berg. In summer, the discharges below the IBT increased by 570-4400% over those measured above the tunnel.

In terms of benthic invertebrate communities, the two rivers differed in terms of community composition, while there were significantly fewer taxa and lower total abundances of invertebrates in the Riviersonderend than the unimpacted communities in the Berg River. During summer in the Berg River, the invertebrate communities below the IBT were markedly different to those collected above it. The invertebrate communities inhabiting the Berg River reaches downstream of the IBT were typical of outlet communities occurring below lakes and impoundments. These communities had significantly greater numbers of individuals, and lower overall richness and numbers of taxa. Furthermore, an overall loss of sensitive invertebrate taxa was recorded below the IBT, while several tolerant filter-feeding taxa appeared to benefit from the water transfer. The Hydropsychidae (Trichoptera) were recorded in particularly high numbers downstream of the IBT during summer. The success of this group was probably due to the transfer through the tunnel of zooplanktonic groups from Theewaterskloof.

During winter, when water transfer ceased, the Berg River invertebrate communities below the IBT recovered, to resemble the unimpacted communities found above the IBT. This temporal recovery reached its maximum towards the end of the winter, before the IBT tunnel discharged the first summer release.

The management and planning of IBTs were assessed, in addition to aspects of conflict resolution and legislation associated with IBTs. In most cases, the technical engineering components of a proposed IBT are given detailed attention, while the environmental and social impacts of such schemes are seldom assessed. A recommendation that was developed from this study was that assessment of the effects of IBTs, where the donor river is impounded before transfer, should take heed of the fact that an IBT represents a discontinuity along a river system, similar to an impoundment. Methodologies that have been developed to determine the flow requirements of river reaches downstream of impoundments should be applied to IBTs.

Of additional importance in ecological terms, and a feature not found where rivers are impounded, is the transfer and mixing of previously isolated biota between catchments, and thus the mixing of genetic material. In addition, the transfer of exotic and invasive fauna and flora is a likely consequence of IBTs. This requires great caution and extensive investigation during the assessment of the feasibility of such schemes.

Thesis Overview

Chapter 1 Introduction This general introduction provides a brief historical account of the use of inter-catchment water transfers for water supply around the world, and the research context of the thesis. Lastly, a definition of the term "inter-basin water transfer" is provided.

Chapter 2 Literature Review: the ecological effects of IBTs A review of the literature dealing with the ecological effects of inter-basin water transfer world-wide is given, with particular emphasis on their biological effects.

Chapter 3 Southern African IBTs A brief description of IBTs, extant and proposed, in southern Africa is presented, in order to provide a comprehensive picture of the water-use scenario and use of IBTs for water supply throughout the subcontinent.

Chapter 4 Description of the Study Area This chapter gives a description of the study areas: the upper Riviersonderend, Theewaterskloof Reservoir, upper Berg River and the transfer scheme chosen for this ecological study, that connects all of these systems. The Riviersonderend-Berg-Eerste River Government Water Scheme (RBEGS) is a small IBT in the Western Cape, serving farmers in the Upper Berg River. A description of the study sites selected for the fieldwork component of the study (below, Chapter 5) is also given here.

Chapter 5 The ecological effects of the Riviersonderend-Berg transfer scheme A description is provided of the field sampling programme, methodology and results of the fieldwork component of the project. Finally, a discussion of the results is presented.

Chapter 6 IBT Planning and Legislation This chapter examines current IBT planning in various countries around the world, and summarises some of the problems and conflicts that have arisen, in addition to legislation that has developed in connection with the planning or construction of IBTs. Lastly, recommendations are made for improved and more comprehensive planning and operation of IBTs in southern Africa.

References

Chapter 1 Introduction

"...And Elisha said: make this valley full of ditches. Ye shall not see wind; neither shall ye see rain, yet that valley shall be filled with water."
Kings II, 3:15-17

Manipulation of the rivers of the planet has always accompanied human civilisation. The historian Karl Wittfogel has suggested that modern society developed from the desire and need to control or regulate rivers (cited in Pearce, 1984). River regulation has evolved together with a steady increase in water requirements and the unchecked growth of human populations. Water storage techniques were developed in order to provide reliable water sources over drier periods, while the diversion of water away from natural water courses became necessary to transport water to the place of use.

In countries that lack adequate and well-placed water supplies, the diversion of water has occurred for thousands of years. At first, this was achieved through the use of canals and gravity. The history of ancient Mesopotamia, between the Tigris and Euphrates rivers, was determined almost entirely by the availability and quality of water diverted into the region (Warnick, 1969; Teclaff & Teclaff, 1973; Pearce, 1984). The first irrigation canals to transport water from the Tigris and Euphrates were constructed more than 6000 years ago (Pearce, 1984). The Nahrwan Canal, which ran alongside the Tigris for 400 km, the 600 km-long Pallacopas Canal that diverted the floodwaters of the Euphrates into the nearby Chaldean Marshes and the Shatt-el-Hai Canal joining the Tigris and Euphrates rivers approximately 4500 years ago (Warnick, 1969), are examples of early diversion schemes that illustrate the level of engineering skills possessed by this early civilisation.

The Egyptians and then the Romans were amongst the first civilisations to investigate new and more complex techniques (e.g. Biswas, 1978). In the case of Egyptian culture, labour-saving techniques such as the pivoted *shaduf* and the animal-driven Archimedes screw, were used to lift water from the river (or wells) to irrigation canals and, hence, onto the fields, while the Romans, with their knowledge of the equilibrium and pressure of water, built aqueducts that revolutionised municipal water supply, and transported water over large distances.

The current technological age has brought with it an expanded rate of use and manipulation of water resources. Globally, urban and agricultural expansion does not always occur alongside reliable water sources and, with developments in engineering skills, catchment boundaries are no longer barriers to water diversions. Inter-basin water transfers (IBTs) have become common solutions to the global problem of transporting water from where it is perceived to occur in excess to where it is required for storage and use.

Very few data exist on the effects of inter-basin water transfers, and thus their short- and long-term ecological consequences largely are unknown (e.g. Boon, 1988; Petitjean & Davies, 1988a,b; Davies *et al.*, 1992; Meador, 1992; Snaddon *et al.*, 1998). The main objective of this project was to address this research need, and to collect data on some of the ecological effects of IBTs in South Africa. The focus of the project was to describe the benthic macroinvertebrate communities that inhabit the donor and recipient river systems of a small IBT, and to explore the changes in community structure and functioning that occurred in the recipient river. The scheme chosen for this investigation was the Riviersonderend-Berg-Eerste River Government Water Scheme (RBEGS), in the Western Cape Province of South Africa. Data were collected from the donor system, the Riviersonderend, and one of the neighbouring recipients of the RBEGS, the Berg River.

The rationale behind the choice of benthic macroinvertebrates as indicators of the nature and degree of anthropogenic disturbance caused by IBTs, was based on evidence that riverine macroinvertebrate communities are adapted to fairly narrow ranges of hydraulic and chemical conditions, and thus will reflect conditions in a river (e.g. Hawkes, 1982; Dallas *et al.*, 1994). Alterations to physical and chemical conditions will lead to changes in the communities that are found at a particular river reach. For example, macroinvertebrate species that are particularly sensitive to flow conditions could be eliminated as a consequence of changes in the flow regime, while those that are more resistant to change will survive, and, possibly, flourish (e.g. O'Keeffe & de Moor, 1988; Snaddon & Davies, 1998).

This project was initiated as a result of the submission of a proposal to the South African Water Research Commission (WRC) in 1993, for funding for a research project that would investigate the ecological effects of inter-basin water transfers in dryland environments. The initial proposal was not successful. Nonetheless, the project commenced and provided the incentive for a re-submission of the original proposal to WRC, which was accepted in 1994. The main emphasis of the WRC research project was to develop ecological protocols for the informed planning, construction and management of IBTs. The project reported here presents an analysis of data collected during the first year of field sampling (March 1994 to April 1995), which then continued for a further two years (until July 1997) as part of the WRC contract. A report on the complete dataset will be presented as a final research product of the contract, in 1999. Additionally, a synthesis of the literature on the effects of IBTs on a global scale is currently being prepared for the WRC, and will be complete by July 1998.

1.1 Definition of inter-basin water transfer

International criteria used for defining inter-basin water transfers (IBTs) have commonly been adapted from Quinn (1981), who proposed the following definition:

1. the transferred water is not returned to the donor system within 20 km of the point of abstraction, and
2. the mean volume transferred is at least 0.5 $m^3 s^{-1}$ (equivalent to an annual volume of 16 million m^3).

Davies *et al.* (1992), however, attempted to broaden this definition and thus avoid the distance and volume components of Quinn's definition. They redefined IBTs as '*...the transfer of water from one geographically distinct river catchment, or basin to another, or from one river reach to another...*' This definition encompasses a broader variety of schemes and includes *intra*-basin transfers, or the transfer of water within catchment boundaries (such as, from headwaters to lower reaches), as well as *inter*-basin transfers, across catchment boundaries (Figure 1.1). This broader definition is realistic as the effects of intra-basin transfers, where water is not removed from the basin but is relocated within it, can be as ecologically deleterious as any major form of river regulation (e.g. Ward & Stanford, 1979; Lillehammer & Saltviet, 1984; Petts, 1984; Gore & Petts, 1989; Petts *et al.*, 1989), and small daily transfer volumes can amount to large annual volumes.

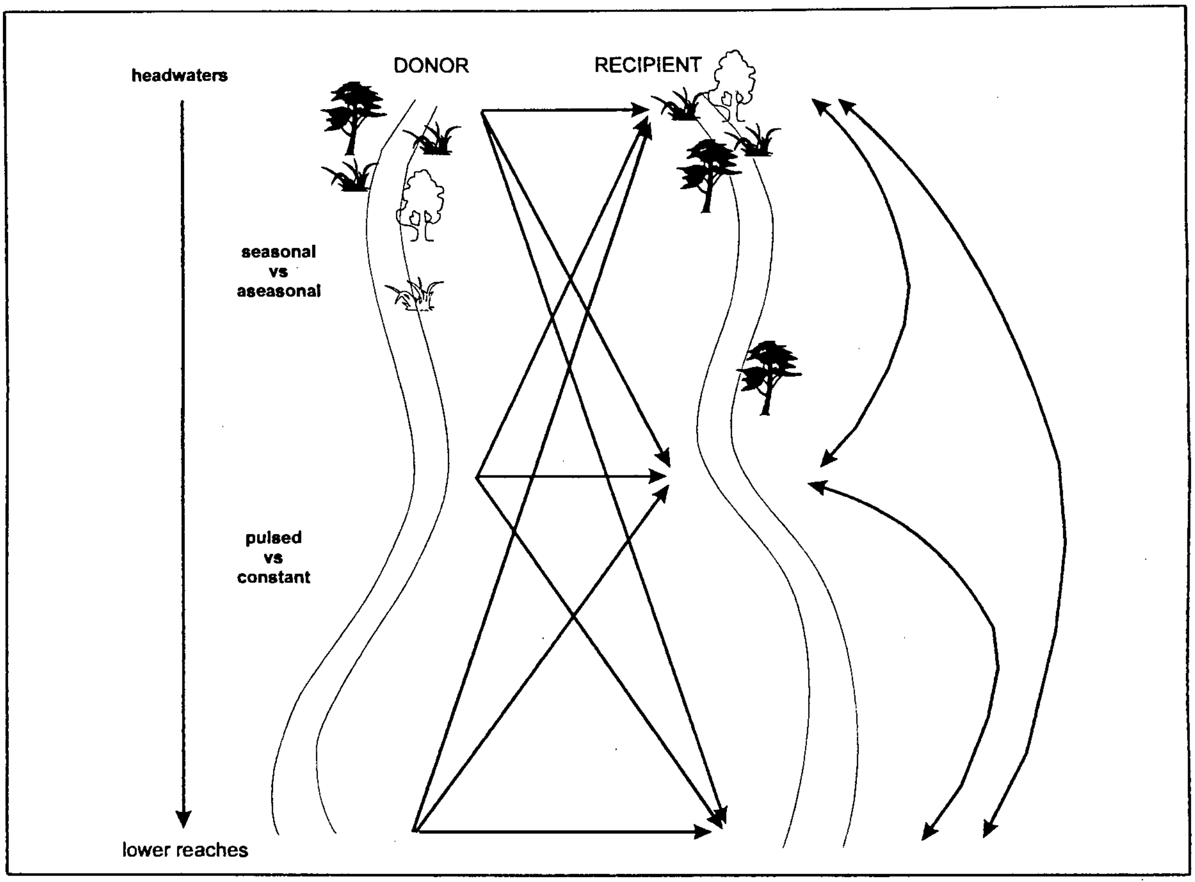


Figure 1.1 A schematic diagram showing the potential combinations of inter-basin water transfer, as defined by Davies *et al.* (1992). Transfers occur within or between rivers, and to and from different reaches; the duration of the transfer is either constant or pulsed, and occurs seasonally or aseasonally (irregularly throughout the year) (modified from Davies *et al.*, 1992).

For instance, the Jonglei Canal on the River Nile, and the Grootdraai Dam Emergency Augmentation Scheme on the Vaal River, which comprise manipulations of water *within* distinct basins and which are, therefore, included within the definition of Davies *et al.* (1992), have had, or will have, major ecological, sociological and hydrological effects both locally and over long distances (e.g. Critchfield, 1978; Charnock, 1983; Bailey & Cobb, 1984; Department of Water Affairs and Forestry, 1991a; Snaddon *et al.*, 1998).

Chapter 2 Literature Review: the Ecological Effects of IBTS

2.1 Introduction

In order to focus on the ecological effects of IBTs, it is important to identify the major differences between the effects of IBTs and the effects of river regulation by dams. Most of the literature which deals with the effects of river regulation on invertebrate communities, describes the effects of impoundments, and, in the UK, some workers define river regulation as impoundment (Boon, 1988). In many cases, water transfers require the impoundment of rivers and these reservoirs are often draw-off points from donor systems, as well as storages of transferred water on recipient systems. The consequences of transfer schemes, however, are more complex, and their effects require specific attention (Day, 1985). The ecological effects of IBTs in a donor catchment are largely similar to those that result from river regulation by dams, while effects that are unique to IBTs are manifested at the catchment scale, where differences in water quality and biotic assemblages between catchments assume importance, particularly in the recipient catchments and along transfer routes (Snaddon & Davies, 1997, 1998). For example, the rivers in South Africa can be divided into categories based on their water chemistry (Day & King, 1995; Day & Dallas, 1996), and thus the transfer of water between water quality regions could affect the water chemistry of the recipient systems. In the same vein, recent efforts in South Africa to determine biological regions according to distributions of riparian vegetation, fish and macroinvertebrates, have led to the description of 18 bioregions (Eekhout, 1996). Thus, the transfer of water across bioregional boundaries is likely to have effects on the biota of the receiving catchments.

Because of the wide variety of IBTs worldwide, they can only be categorised in very broad terms according to the following criteria:

Type of donor The donor is usually a river or stream that has been impounded, or which has a diversion weir downstream of the drawoff point. In the case of the impoundment, the transfer tunnel inlet is generally situated at variable depths within the reservoir. In some cases, however, water is transferred from groundwater systems.

Type of recipient In most cases, the recipient is a river or stream, but may also be an impoundment.

Type of transfer route There are several different types of transfer route that are employed for IBTs (Figure 2.1) - for example **pipes**, above or below ground, **canals**, open or closed, and **natural water courses.** An additional transfer route not illustrated in Figure 2.1 involves the use of **aquifers.**

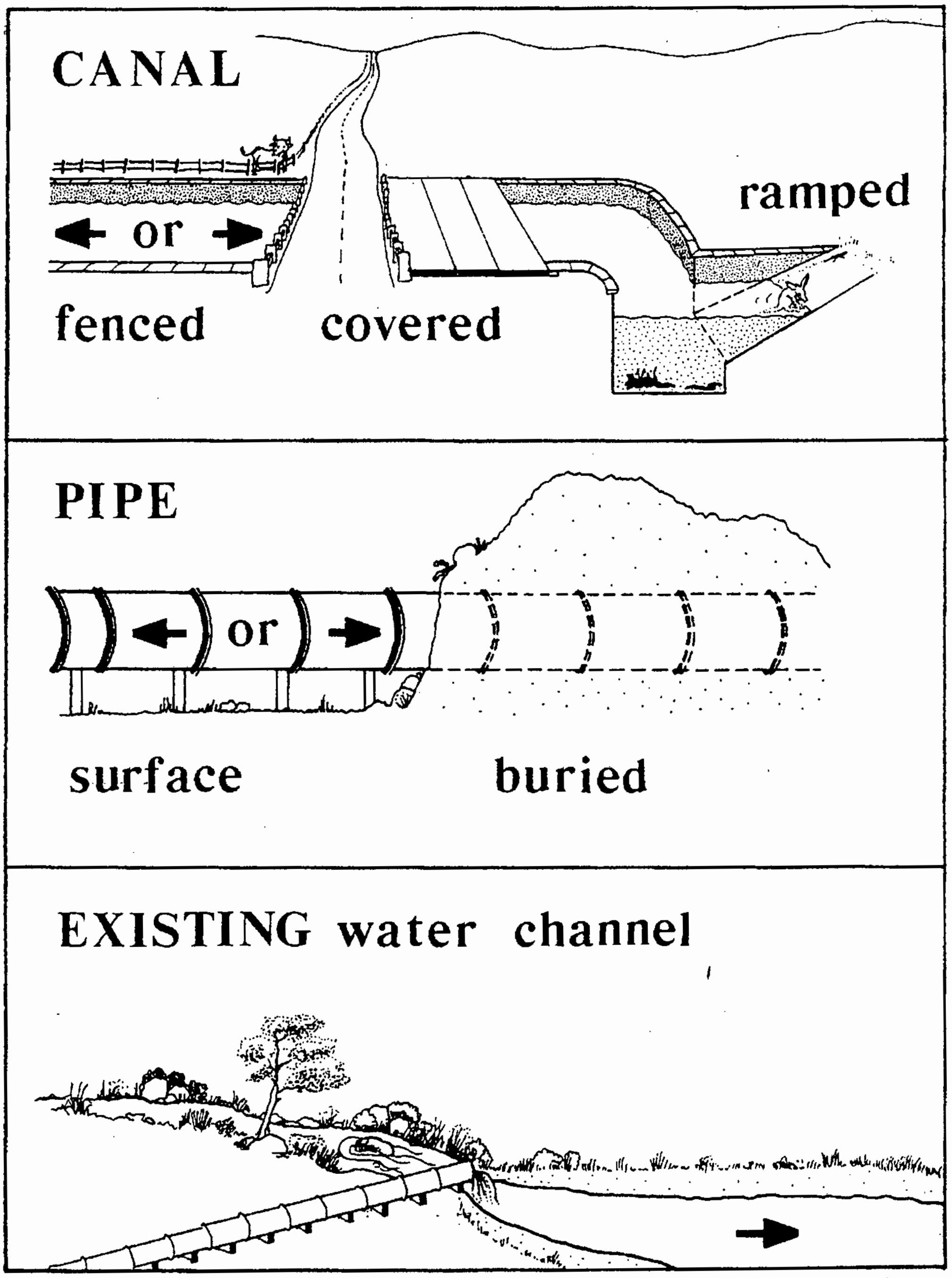


Figure 2.1 There are a variety of water transfer routes utilised in IBTs. These include canals, which can be open or covered, with fences and ramps, to provide an exit for trapped animals; pipes which are buried or laid on the surface, and existing water channels which are often used to transport the diverted water to and from storage reservoirs. The direction of the water through canals and pipes can, in some cases, be reversed.

Operational criteria The operational criteria of IBTs refers to the type and timing of release, and the volume and rate of water transfer. These criteria have significant influences on the aquatic systems affected by an IBT (e.g. Boon, 1988; Davies *et al.*, 1992). The most common type of release is from the outlet of a tunnel, pipeline or canal. The release is commonly under pressure due to pumping or gravity. The timing of release is of great significance for the biota of the aquatic systems concerned, and can be seasonal or aseasonal, constant or pulsed, or combinations of these. Indeed, the possible combinations can be so variable world-wide, that well over 120 different types of IBT are possible (see Davies *et al.*, 1992).

The nature of the components of an IBT will influence the extent of its effects on the environment (Snaddon & Davies, 1998). Furthermore, IBTs may involve adjacent river basins, or basins with a very wide geographical separation, crossing biogeographical and even international boundaries. In all of these cases, the transfers of water will lead to alterations of the basin or basins involved, irrespective of their nature. A section of a river, or an entire river basin, becomes the donor of water, while the other becomes the recipient of that water, and a myriad attendant changes take place: water losses or additions, water quality changes, alterations in physical properties of channels, temperature regimes and so on, as well as the transfer and mixing of organisms, to name a few.

An increasing scientific and public awareness of the ecological and environmental effects of large-scale engineering developments has resulted in IBTs being brought to the attention of the scientific community, particularly where biogeographical barriers are broken down or breached, water quality is affected, reduced flow conditions occur and high ecological costs are incurred. The following sections summarise the work that has been done on the ecological effects of IBTs.

2.2 Physical and geomorphological effects

2.2.1 Water quantity and flow patterns

The main aim of water transfers is often to supply a continuous and reliable source of water, either throughout the year, or for a season. Thus, a recipient river will almost invariably lose its natural flow patterns, changing from possibly an intermittent or seasonal system, to one that is perennial (e.g. Morel, 1978). Richards & Wood (1977) looked at the hydrological effects of IBTs in the United Kingdom (UK), where the uses of transferred water are mostly non-consumptive, (i.e. not used for irrigation or lost to the atmosphere), but generally are for use in sewage treatment processes. The transferred water, if it is not directly transferred to a river before use, reaches the recipient basin in the form of return flows, primarily from sewage works. The transfer of water thus raises the baseflow and leads to heightened flood peaks, resulting in the more frequent occurrence of flashy floods, especially in urban areas. Low flows in these recipient catchments, particularly in the midlands and south-east England, comprise mostly imported water; the River Tame at Lea Marston, for example, is 90% effluent (Richards & Wood, 1977) .

The Orange River Project (ORP) in South Africa involves a constant transfer of water from the Orange River southwards into the Great Fish River, and then the Sundays River (Figure 3.1, Chapter 3). Since 1977, this scheme has transferred an annual volume of $350 \times 10^6 m^3 yr^{-1}$ of water for irrigation in the receiving catchments. The primary influence of the Orange River Project has been a staggering 500 to 800% increase in flow in the upper section of the Great Fish River, and although the flow in the lower river remains similar to pre-transfer values, the river is now perennial as opposed to seasonal (O'Keeffe & de Moor, 1988). One of the largest IBTs on the Australian continent is the Snowy Mountains Scheme, which involves a transfer of $1.1 \times 10^9 m^3 yr^{-1}$ from the Snowy River to the Murray River system. The scheme comprises fifteen large dams, seven power stations, two pumping stations and 135 and 36km of tunnels and aqueducts respectively, ultimately connecting six rivers. Both donor and recipient rivers of the Snowy Mountains Scheme have been affected by the transfer. Flows in donor rivers have been reduced to almost negligible volumes, while substantial flow fluctuations have been experienced in recipient river channels, as a result of hydroelectric power demand. At one of the power stations, for example, releases of up to

1130 m^3 s^{-1} can be made on demand and, as a result, the construction of regulation structures on the recipient Murray River has become necessary, in order to reduce diurnal fluctuations in water level, and to minimise river bank erosion (Johnson & Millner, 1978).

Similarly massive transfer schemes were proposed in the former USSR. These were the European Transfer Project (ETP), which involves the redistribution of water resources of northern rivers and lakes to the Caspian Sea Basin in the south, and the Siberia-Central Asia Project (SCAP), comprising the diversion of Siberian rivers southwards to the Aral Sea Basin (Micklin, 1985). If completed, these diversions would have transferred 120km^3 of water, and although this volume is not the largest ever proposed, the potential impacts of these transfers would have surpassed those of all existing transfer schemes. After fairly extensive research on the effects of both of these proposals, however, SCAP was shelved in November 1985, while the ETP was abandoned in March 1986 (Micklin, 1986). The ETP would have resulted in a 53% reduction in the flow of the Onega River at its outflow from Lake Lacha (Voropaev & Velikanov, 1985; Micklin, 1986). Furthermore, a reduction of 43% of the flow of the Upper Sukhona River would have led to sedimentation and tributary downcutting, resulting in difficulties for navigation and for timber rafting on the river. Furthermore, a second stage diversion of water from Lake Onega (3.5km^3) would have reduced flow in the Svir and Neva rivers. A further third stage diversion from the Pechora River of 9.8km^3 *via* the Kolva and Kama to the Volga would have had a significant impact on the Pechora, thereby reducing flows below the Mitrofanovskoye hydro-electric dam by between 40 and 76%, with consequent effects on bank stability, navigation and on floodplain ecology (Voropaev & Velikanov, 1985; Micklin, 1986).

In the proposed recipient area of the ETP, the Caspian Sea, the Sea of Azov, the Volga-Akhtuba floodplain and the Volga estuary are all areas that have suffered greatly from over-exploitation of their water resources through abstraction and IBTs. In 1978, the Caspian Sea dropped 29m to its lowest level ever, and the Sea of Azov suffered from high salinities. The deterioration of these systems could have been checked by the ETP.

The Siberia-Central Asia Project (SCAP) was a larger transfer proposal, covering a greater distance than the ETP schemes. It was designed respectively to transfer 27.2 and 60 km^3 of water per year in Phases I and II. The donor rivers, the Ob and Irtysch, would have been substantially reduced in volume below points of diversion, resulting in a wet-year flow reduction in the Ob at Salekhard of between 9 (Phase I) and 15% (Phase II), and between 19 and 31% for dry years. These reductions in flow, combined with unnatural flow fluctuations, would have reduced inundation of important floodplains in the Ob and Irtysch river catchments. One of the major reasons for the SCAP proposal was to improve conditions in the recipient Aral Sea basin (Section 2.3.1). The Sea has been shrinking rapidly for the last 30 years, and the cause is directly attributable to the diversion of 100 x 10^6 m^3 yr^{-1} from the Amudar'ya and Syrdar'ya rivers, which feed into the Aral, for irrigation of the desert regions of Kazakhstan and Uzbekistan (Micklin, 1985). Micklin (1988) has described how the Aral Sea, which lies in one of the recipient catchments of the SCAP, has decreased in area since 1960 from 68 000 to 41 000km^2, halving its volume and dropping its level by more than 12m (see also Williams & Aladin, 1991).

The Churchill River Diversion in Canada was commissioned in 1976, for hydro-electric power generation (Day, 1985). Southern Indian Lake on the Churchill River is dammed at its northern outlet and water is diverted from the Lake's southern basin into the Rat River, from where it flows into the Burntwood and Nelson rivers, eventually passing *via* the Kettle and Long Spruce generating stations to Hudson Bay. The Rat River receives an average annual transferred volume of 775 m^3 s^{-1}, some 35 times its normal flow, and seven times the normal flow of the next recipient, Burntwood River. On the lower Nelson, a 34% increase in the mean annual flow was expected. Concurrently, the scheme has reduced the flow of the Churchill River at the outflow of Southern Indian Lake by 75%, from 1011 to 251 m^3 s^{-1}. The James Bay Project is another hydro-electric development (Day, 1985) in Canada, which involves the transfer of 87% of flow in the Eastmain River (810 m^3 s^{-1}) and 43% of water flowing in the Caniapiscau River (776 m^3 s^{-1}), into the La Grande Rivière. An additional transfer from the Sakami River in a neighbouring catchment brings a further 30 m^3 s^{-1} into the La Grande, *via* a hydro-power generation unit. The total amount transferred to the La Grande is 1586 m^3 s^{-1}, almost double its natural flow volume.

One of the largest transfer schemes ever conceived is the ambitious North American Water and Power Alliance (NAWAPA). Proposals were first suggested in the mid-1960s, and these were to transfer water from north-western North America to other areas of Canada, the United States and Mexico (e.g. Micklin, 1977). The original transfer volume of $136km^3$ of water was to be divided between the United States (61%), Canada (20%) and Mexico (19%), with a potential increase to $308km^3$. Purposes of the scheme were multiple: expanded irrigation (enough to irrigate a total of 22×10^6 hectares), increased hydroelectric power production (100 000 MW, of which 70% would be available for industrial and domestic use), the development of a navigable network of canals of which the Canadian Great Lakes Waterway is the most substantial, allowing ocean-going vessels access to the interior of the country, flood control, municipal water supply, recreation amenities and pollution control. NAWAPA has since been shelved due to opposition from numerous vocal and politically active public organisations (Micklin, 1977).

Remaining at this massive scale of water transfer, the largest dam on the African continent is the Katse Dam, which is the major impoundment of Phase IA of the Lesotho Highlands Water Project (LHWP, Figure 3.1, Chapter 3), in southern Africa. If the first phase (IA and IB) of this scheme is completed as planned, it will involve the transfer of 30 $m^3 s^{-1}$ from the headwaters of the Orange River in Lesotho, into a tributary of the Vaal River, in South Africa (Department of Water Affairs, 1985). A major consequence of Phase IA of the LHWP will be a substantial reduction in flow and changes to the natural flow regime of the donor river, the Malibamatso. The 182m-high Katse Dam will store 3.3 times the mean annual runoff (MAR) of the Malibamatso catchment, which is an unusually large storage (Chutter, 1993). The compensation flow has been calculated as 2.5% of the MAR, giving a discharge of only 0.5 $m^3 s^{-1}$, which will be the only flow in the Malibamatso for long periods. The river channel is expected to decrease in size due to the diversion of water, with bankfull discharges only occurring when the Katse Dam spills during heavy summer rains. The final transfer volume for the LHWP was initially calculated at 70 $m^3 s^{-1}$; this volume approaches the present 20-year-flood discharge in the recipient Ash River, in the neighbouring Vaal River catchment. Lower down the system, after the Ash has joined the Liebenbergsvlei and then the Wilge River, the combined 'natural' flow of these latter two rivers is predicted to exceed the flow due to the transfer scheme in only 11% of the months in a 55-year synthetic flow record (Chutter, 1993). In places, the augmented flow in the Liebenbergsvlei River will exceed the capacity of the channel, and some inundation of land will occur. Currently, the costs of transporting the water to its point of use, through an underground pipeline, are being evaluated, in order to avoid the consequences of increased flow in the Ash and Liebenbergsvlei rivers (E. van den Berg, Department of Water Affairs and Forestry, personal communication, October, 1997).

2.2.2 Groundwater Resources

In many cases, water is drawn from groundwater sources, to feed transfer schemes directly, or even to replenish supplies that have been over-utilised. In southern Africa, for example, Namibian farmers have reported severe water-table depression, which will be exacerbated by the Karstveld Borehole Scheme, that will supply water to the Eastern National Water Carrier (ENWC, Figure 3.1, Chapter 3). The ENWC involves the transfer of water from a number of dams on perennial rivers in Namibia, and the abovementioned borehole scheme, to the Okahanja/Windhoek area (Petitjean & Davies, 1988a,b; Davies *et al.*, 1992, Davies *et al.*, 1993).

Another example of groundwater as an IBT source can be drawn from Israel where, in the 1970s, rainfall was inadequate to provide sufficient runoff to the rivers flowing into Lake Kinneret (the Sea of Galilee), which is the source of water for the National Water Carrier that feeds Tel Aviv and the Negev region (Overman, 1976; Gavaghan, 1986). To exacerbate this water crisis, the water-table in the north and south of the country is sinking to very low levels due both to over-abstraction and inadequate recharge. Israeli engineers have attempted to solve this problem by pumping water out of Lake Kinneret and into the depleted wells in the north. This water flowed underground and along an aquifer that acted as a natural conduit extending down to the drier southern region, and soon the level of water in the southern wells rose.

Excessive groundwater abstraction in the vicinity of Mexico City has resulted in more than 9m of land subsidence, with consequent poor quality water and drainage problems necessitating a new deep drainage system (Garduño *et al.*, 1978). In order to replenish supplies, groundwater is abstracted from the Lerma basin for transfer to Mexico City. Consequently, excessive abstraction from the Lerma Basin has resulted in land subsidence and the drying up of lakes. The reliance of Mexico City on the Lerma basin has resulted in social, environmental and political problems.

A further problem linked with IBTs is the seepage of water from unlined canals. This is a likely consequence of two water transfer proposals, the East Route in China (Changming *et al.*, 1985), which will transfer water from the lower reaches of the Yangtze at Jiangdu to the Huang-Huai-Hai North China Plain, and SCAP in Asia (Gorodetskaya, 1982). In Russia, the 2000 km unlined transfer canal from Tobol'sk in Western Siberia to the Amu Dar'ya in Central Asia (the transfer route for SCAP) would have raised groundwater levels in areas adjacent to the canal.

2.2.3 Erosion

The increased flows due to water transfers and the increased frequency of high flows capable of transporting sediment often lead to bank and bed erosion in recipient systems. This has been found on a fairly small scale in the UK (Richards & Wood, 1977), while rapid erosion has been recorded and continues to be a problem in diversion channels used by the Long Lake and Ogoki Diversions in Canada (Day *et al.*, 1982; Day, 1985). These two schemes came into operation in 1943, with the dual functions of inter-basin pulpwood transport, and hydropower generation. The Ogoki and Long Lake systems drain into Lake Superior and the Albany River system, and water is also diverted to Niagara and to the St Mary's and St Lawrence rivers. Flows in one of the diversion channels, the Little Jackfish River, increased from 4 to 118 $m^3 s^{-1}$ with mean peak flows of 311 $m^3 s^{-1}$ as a result of the Ogoki Diversion (Day *et al.*, 1982).

In South Carolina, USA, the Santee-Cooper Project was constructed in 1942, to divert Santee River water into the Cooper River for hydroelectric power generation (Snaddon *et al.*, in prep.). However, environmental impacts such as an increased sediment load in the Cooper River, necessitated the Santee-Cooper Re-diversion Project in 1985 to redivert flow from the Cooper River to the Santee River.

The Mpofana River, a tributary of the Mgeni River in Kwazulu/Natal in South Africa, is the recipient of a water transfer from the Mooi River, the Mooi-Mgeni Transfer Scheme (Figure 3.1, Chapter 3). The capacity of this scheme is due to be increased, from a maximum of $6\,m^3\,s^{-1}$ to upwards of $10\,m^3\,s^{-1}$ (Henderson, 1995). Sections of the Mpofana are characterised as unconfined channel flowing through wide or restricted floodplains (Heritage, 1995), and it is predicted (albeit based on limited information) that these reaches are likely to be susceptible to erosion. The substratum is largely fine sand and silt and the banks are composed of alluvium, often forming vertical cliffs on meanders (Heritage, 1995), and elevated flows will invariably cause geomorphological changes in this system.

The downstream reaches of the Malibamatso River, recipient of the LHWP, will receive sediment-free water from Katse Dam, due to the settling out of finer sediments, which will in turn and in all likelihood, result in erosion, stream-bed armouring, and the transport of sediment down the river (Chutter, 1993). Similarly, and in addition to changes in flow volumes in the recipient Ash River, the river will receive water that is clear and which will carry little sediment (Chutter, 1993). Accordingly, geomorphological consequences for the Ash River may be considerable, with increased erosion and armouring of the streambed.

2.2.4 Seismic Activity

A recent concern in southern Africa is reservoir-induced seismic activity, which has occurred as the result of filling of the Katse reservoir in Lesotho, donor impoundment to the LHWP. The reservoir commenced filling towards the end of 1995, and a series of seismic events was triggered almost immediately. At 29% capacity, at least three events greater than magnitude 2.5 (2.7, 3.0 and 3.5) were recorded, the last of which caused a 1.3-kilometre-long crack in the Mapeleng region in April 1996, less than six months after Katse commenced filling (Hendron & Gibson, 1996). Current predictions estimate that seismic events could continue and may reach magnitude 4 to 4.5 on the Richter Scale as the reservoir continues to fill. Katse (built) and Mohale (presently on track for construction) reservoirs, both large donor impoundments of the LHWP, are in an area of seismic activity (e.g. Hartnady, 1985). Katse Dam is designed to withstand events up to 6.5 on the Richter Scale (LHDA Press Release, 19 February, 1996), however, the seismic strengths of the transfer tunnels northwards to

Gauteng, South Africa, appear not to have been determined. Earthquake damage to transfer tunnels and other LHWP infrastructure will have severe effects on water supply, and would present a potential threat to human life and damage to property, not only in Lesotho, but also in the provinces of KwaZulu/Natal, Free State and Gauteng (Davies, 1996).

2.3 Water quality

Due to the fairly intensive use and re-use of water resources in the UK, many of the problems and benefits linked to IBTs are associated with water quality. Johnson (1988) has stated that IBTs can be used to dilute point-source pollution, and thus restrict ecological damage caused by this pollution. One of the unplanned benefits of the Ely Ouse to Essex Scheme, which is able to transfer almost 6 $m^3\ s^{-1}$ from the Ouse to the River Stour in Essex in order to increase potable water supplies in this area, was the control of a liquid ammonia spill in the upper reaches of the River Stour. The spill originated from a sewage works unable to cope with a sudden surge of ammonia (Guiver, 1976). Water transfer from the Ely Ouse to the Stour enabled dilution of the spill to acceptable levels. Furthermore, the cost of the pumping was recovered from the sewage works responsible for the ammonia spill.

Other water quality problems are associated with the transfer tunnels and canals, especially when schemes are inactive for long periods. The Kielder Water Scheme in the UK transfers water from the River Tyne to the rivers Wear and Tees. The quality of water remaining in the transfer tunnel during times when the scheme is not operational is monitored as part of the management strategy for this scheme (Hall, 1977). This is an attempt to prevent slugs of poor quality water being washed into the Wear each time the transfer is re-activated. Still in the UK, Huntingdon & Armstrong (1974) have stated that knowledge of the time required for water to flow from abstraction point to user along the Ely Ouse to Essex Scheme, is necessary for the efficient operation of the scheme. As with the Kielder Water Scheme, water quality in the tunnel is monitored, since transfer of water is intermittent and water may be left to stagnate for considerable periods. The ENWC in Namibia is also likely to suffer from water quality deterioration in its open sections due to decomposing filamentous algae mats, for which chemical control is being considered (Petitjean & Davies, 1988a,b). Water quality in lined canals has been investigated by several authors, in particular by Oksiyuk *et al.* (1979)

and Oksiyuk & Stolberg (1981), who worked on the Severskiy Donets-Donbas Canal in Russia. They identified filamentous algae as the main cause of water quality deterioration in this canal.

A further concern associated with IBTs is the transfer of water quality problems experienced in the donor catchments, to the recipient catchments. One of the anticipated areas of great concern in the proposed Chinese East Route is the transfer of water polluted by phenols, cyanides and mercury, to recipient catchments (Jinghua & Yongke, 1983). Returning to the UK, the transfer of *Stephanodiscus* blooms from donor to recipient in the Ely Ouse to Essex Scheme gave rise to complaints about potable water quality, and transfer was temporarily halted (Guiver, 1976; Jollans & Zabel, 1982). Furthermore, in 1973 in the Essex region, local fruit farmers developed problems with tomato fruit formation, which rendered the crop unmarketable. An analysis of water supplies revealed the presence of the pesticide 2.3.6 trichlorobenzoic acid (TBA), originating from a factory discharging into the River Cam, a tributary of the Ely Ouse (Guiver, 1976). The problem had not occurred prior to the transfer, since Ely Ouse water had never been used for crop irrigation in its own catchment (see also Huntingdon & Armstrong, 1974).

Collie & Lund (1979) have reported on preliminary work on the water quality of the rivers of the proposed Severn-Thames Transfer in the UK. This transfer would have a maximum transfer capacity of approximately 4.5 $m^3 s^{-1}$ of water from the Severn into the upper reaches of the Thames (National Rivers Authority, 1994). Algal bioassays revealed that the waters of both systems were highly fertile. This led them to the conclusion that water transfer from the Severn to the Thames would not significantly alter either the algal or the nutrient status of the Thames, even if water was drawn only from the Clywedog Reservoir, the main upland reservoir on the Severn.

Reduced flows in the Onega River, donor to the ETP, would probably have led to a higher percentage of pollutants in this river (Voropaev & Velikanov, 1985; Micklin, 1986). Flow in another donor, the upper Sukhona River, would have been reversed below the towns of Sokol and Vologda, which are sources of pulp- and paper-mill effluents, transferring these back to Lake Kubenskoye. This would have resulted in pollution and increased eutrophication in the lake, with a consequent threat to the Kubenskaya Whitefish. The construction of the transfer would have been delayed in order to allow construction of water treatment works for the pulp and paper mill effluents. The diversion of water from the Neva River, Phase IB of the ETP, would have resulted in a 5% reduction in flow, increasing pollution concentrations in the Gulf of Finland due to reduced dilution of effluents from Leningrad (Voropaev & Velikanov, 1985; Micklin, 1986).

A further water quality aspect of water transfer involves temperature changes in recipient waters. One of the unpredicted impacts of the Churchill River Diversion in Canada, for example, was a decrease in water temperature in Southern Indian Lake, a receiving water body (Day, 1985). Similarly, thermal effects as a result of the ambitious NAWAPA proposals are a real possibility since the water in donor reservoirs in Canada and northern North America would be several degrees colder than receiving waterbodies in the south (Micklin, 1977).

Johnson & Millner (1978) commented on the Snowy Mountains Scheme in Australia where water that is 6°C cooler than the average temperatures of the Murray is released into the river. Katse Reservoir of the LHWP in southern Africa will be a deep, sinuous impoundment, storing just under 2 billion tonnes of water which are predicted to stratify thermally in summer. Although there is provision made for the withdrawal of water at varying depths, the temperature range of the transferred water will be smaller than that of water in the recipient Ash River (Chutter, 1993). Furthermore, temperatures in the reservoir are expected to be lower in winter and higher in summer than those of the Ash (Chutter, 1993).

The dilution or concentration of salinity levels of soils and waterbodies in recipient catchments is often associated with IBTs. The diversion of $100 \times 10^6\ m^3\ yr^{-1}$ from the Amudar'ya and Syrdar'ya rivers for irrigation of the desert regions of Kazakhstan and Uzbekistan, has resulted in salinity values in excess of $27\ g\ l^{-1}$ in the Aral Sea, which is fed by these two rivers (Micklin, 1988). On the other hand, the Snowy Mountains Scheme diverts high quality water to the Murray system, which has led to a decrease in the high salinities that were becoming problematic in the middle and lower reaches of the Murray River. Similarly, the ORP in South Africa has also enabled dilution of the salinised Great

Fish River (recipient), particularly with respect to sodium, chloride, magnesium and sulphate, but in the lower river there appears to be no significant reduction in any of the chemical variables measured (O'Keeffe & de Moor, 1988). Salinisation has been, and still is of serious concern to agriculture in the areas to which water will be transferred by the East Route in China, as well as along the transfer route (Huanting *et al.*, 1983).

2.4 Biological effects

2.4.1 Invertebrates

In the United States, Golladay & Hax (1995) investigated the effects of a transfer of water from Lake Texoma on the Oklahoma-Texas border (Red River basin), through a 16-km pipeline, to the headwaters of Sister Grove Creek and eventually into Lake Lavon in Texas (Trinity River basin) (see also Schorr *et al.*, 1993), on the invertebrates of the recipient river. The transferred water flows along 50km of Sister Grove Creek before reaching Lake Lavon. Sister Grove Creek is usually an intermittent stream, with seasonal extremes in flow (Matthews *et al.*, 1996). Golladay & Hax (1995) investigated the effects of experimentally increased flows on the meiofauna of Sister Grove Creek. Meiofauna (45-500μm) are particularly susceptible to higher discharges: they are poor swimmers and are prone to drift, rather than able to resist flow. The meiofauna of Sister Grove Creek was sampled from two different habitat types, namely sediment and small woody debris (<3mm diameter), at two disturbed sites and one undisturbed site, before and after a 2-week water diversion from Lake Texoma. Before diversion, meiofaunal populations in Sister Grove Creek were stable or increasing in numbers in both habitat types at all sites. Nematodes, copepods and rotifers accounted for 85-99% of the meiofauna in the sediment; while chironomids, nematodes and rotifers accounted for 80-92% of the meiofauna on the woody debris. After the diversion, meiofaunal densities in the sediment habitat type were reduced to 1-2% of pre-diversion levels, and all taxa were affected, while on the woody debris, meiofaunal densities decreased to 10-17% of pre-diversion levels. The wood meiofauna were thus more resistant to flow increases than those in the sediments. Furthermore, one month after diversion, greater recovery was observed on the woody debris than in the sediments. The authors concluded that woody debris probably provides important refuges during floods, and recolonisation sources after floods.

In Norway, Brittain *et al.* (1984) investigated the effects of a diversion of 80% of the flow of the Glåma to the Rena River on the benthic macroinvertebrates of the Glåma, while Borgstrøm & Løkensgard (1984) undertook a similar study on the fish fauna. The purpose of the transfer is to provide water for hydro-electric power generation in the Rena catchment. Only the donor river was sampled, and thus the results of the study are not unique to water transfer, but rather are as a result of water abstraction. The effects on the benthic invertebrates were greatest during winter, when the percentage reduction in water flow was greatest. Brittain *et al.* (1984) found that densities of Trichoptera, Ephemeroptera and Plecoptera were considerably lower below the transfer site. The benthic species which grew most during winter were most affected by the diversion. For example, the winter-growing Capniidae Plecoptera were eliminated below the transfer, while the winter generations of the baetid Ephemeroptera, *Baetis rhodani*, were severely reduced. In addition, species that are characteristic of the slower-flowing sections of the Glåma tended to colonise the river below the diversion. Densities of filter-feeding trichopterans, for example of the genus *Hydropsyche*, increased below the transfer site, while a species of plecopteran, *Diura nanseni*, increased in dominance. Recovery was fairly rapid below the transfer site during summer, with benthic densities by August resembling those at the undisturbed sites above the diversion. Spring spates played an important role in recolonisation of depauperate sections of the river below the transfer, especially from tributaries that were unaffected by the transfer. Working on the same system, Brittain & Bildeng (1995) found that the life-cycles of a species of Trichoptera, *Arctopsyche ladogensis*, occurring in the regulated reaches of the Glåma, were fairly flexible. Individuals of this species had both one- and two-year life-cycles, which they suggested was advantageous in regulated rivers, where temperatures and flows fluctuate. Such flexibility is not common amongst riverine invertebrates.

One of the few studies of the ecological effects of IBTs in South Africa was undertaken by O'Keeffe & de Moor (1988). A comparison of the pre- and post-transfer benthic macro-invertebrate communities of the Great Fish River, one of the recipients of the ORP, was developed, using data collected by researchers in the 1970s (e.g. Scott *et al.*, 1972). O'Keeffe

& de Moor (1988) reported that the species richness of the benthic invertebrate fauna of riffles had changed little as a result of flow increases in the river. The numbers of taxa increased from 41 taxa before to 47 taxa after the transfer, but the species composition had changed considerably, with only 33% of taxa common both to pre- and to post-transfer samples. In particular, they recorded significant shifts in the dominant dipteran Chironomidae (ghost midges) and Simuliidae (blackfly) species, and trichopteran Hydropsychidae (caddisflies). Most pronounced was the shift to dominance by the pest blackfly species *Simulium chutteri*, to the detriment of the original benign populations of *Simulium adersi* and *S. nigritarse*, which were previously co-dominant. *Simulium chutteri* now causes severe damage to livestock in the lower reaches of the river: the feeding activities of swarms of adult females cause stock damage and disturbance during spring (see also Scott *et al.,* 1972; O'Keeffe, 1982, 1985). All of these shifts in the invertebrate fauna can directly be attributed to the changes in flow regime caused by the transfer, particularly the loss of flow variability, and the shift from a seasonal to a perennial river. This has led to an increase in the total area of available erosional habitats, which are favoured, in particular, by simuliid larvae.

The benthic macroinvertebrate communities downstream of Katse Dam of the LHWP are also predicted to be primarily affected by physical alterations of habitat - such as smothering by benthic algae, and loss of riffle biotope for the establishment of early instars - by changes in water quality and, by a reduction of inputs of coarse organic matter from riparian vegetation (Chutter, 1993). In addition, lentic faunal groups, such as zooplankton developing in the reservoir, can be expected to be released from Katse Dam, thereby providing a ready and greatly altered food source for the downstream biota (e.g. Snaddon & Davies, 1998). Some invertebrate taxa will benefit from these conditions, such as filter-feeding hydropsychid caddisfly larvae and simuliid larvae, and predators such as *Hydra* sp., while others will not be able to tolerate the new environmental conditions (Snaddon & Davies, 1998). The major pest species of Simuliidae of the Orange River, *Simulium chutteri*, has been recorded in low numbers in both the Malibamatso and Liebenbergsvlei rivers, and of concern is the fact that a rapid increase in numbers of this species to economically serious proportions has already been well-documented for river reaches below dams further down the Orange/Vaal River system, so much so, that major eradication campaigns are in progress using biological control methods (e.g. Car & de Moor, 1984; de Moor, 1982; de Moor *et al.*, 1986). Similar increases in numbers could well occur in the Malibamatso, Nqoe and Ash rivers, where manipulation of flow conditions coupled with temperature changes and alterations in type and quantity of food material could encourage the establishment of this species.

Lastly, in Britain, screening and chlorination of transfer water in the Ely Ouse to Essex Scheme is primarily done in order to prevent the growth of the zebra mussel, *Dreissena polymorpha*, and to reduce diatom blooms. It appears that this has been effective (Boon, 1988).

2.4.2 Fish

In many cases, water transfers result in the alteration of habitat necessary for the survival or establishment of fish fauna. Matthews *et al.* (1996) conducted a study on the Lake Levon-Lake Texoma transfer which is described above (Section 2.4.1). They investigated the effects of experimentally increased flows on the fish fauna of the recipient stream, Sister Grove Creek. The releases were made into Sister Grove Creek during periods of lowest flow, resulting in marked increases in flow and conductivity. Three trial flows were released, each lasting for 2 weeks or less. Matthews *et al.* (1996) collected data before, during and after the experimental releases. Analyses of the data showed moderate changes in the fish fauna of Sister Grove Creek after the trial flows. Small changes in abundance of individual fish species were recorded, while, at some sampling stations, quantitative and qualitative changes in species composition of the fish assemblage were substantial.

Diversion of water from Lake Texoma resulted in increased discharges that were well above the average natural flows for the season, leading to a doubling of the conductivity of Sister Grove Creek (the salts that increased were largely sodium and chloride). The increased flows were sustained for periods that exceeded the natural period of high flow after a storm, and, in addition, these artificial flows were not accompanied by other events associated with storms, such as increased allochthonous input, meteorological changes, and increased surface runoff. All of these events are significant in terms of fish distribution and behaviour.

Analysis of the data showed an overall decrease in variation in the fish fauna after water transfers began, while no lake species were introduced into Sister Grove Creek upstream from Lake Lavon, or from the Red River basin. Most species showed non-significant changes in abundance after the transfer, with a few exceptions: the abundance of two minnows increased significantly, while the abundance of one centrarchid species in Sister Grove Creek decreased after the transfer. The species showing increases in abundance were those adapted to harsh physical conditions.

The authors suggested that conclusive results realistically can only be gained after the transfer has been in operation for a longer period. For example, prolonged high discharges may have different effects on the river, such as modified habitat due to erosion of the channel and stream banks and the removal of woody debris. The fish of the Sister Grove Creek system are adapted to seasonal flooding. During floods, juveniles and adults find hydraulic refugia in low-flow areas, where they remain until water levels decrease. Larval fish, however, are washed out of stream segments. The authors stated that probably the most important condition for the success of the fish species is the availability of suitable habitat. If the prolonged increases in flow due to the transfer lead to increased scouring of the channel, input of mud and silt, and the export of debris and its associated biota, fish that rely on these habitats may decrease.

Borgstrøm & Løkensgard (1984) investigated the effects of a diversion of flow from the Glåma to the Rena River on the fish fauna of the Glåma. The authors reported a reduction in grayling (*Thymallus thymallus*) recruitment and numbers, while brown trout (*Salmo trutta*) appeared unaffected by the reductions in flow and wetted area that resulted from the transfer. They attributed these observations to the reduction in the density and diversity of benthic invertebrates below the transfer, as recorded by Brittain *et al.* (1984; see above), which are the favoured prey of grayling; trout feed on small fish. In addition, grayling appeared to be more dependent on flow and the availability of habitat, especially as refuges for immature individuals. Reduced recruitment of grayling has also led to increased fishing pressure on this species.

As a result of the Long Lake and Ogoki Diversions in Canada, changes in the magnitude of artificially high or low water levels have resulted in erosion and thus in increased fish-egg mortality (Day, 1985). In the UK, Mann (1988) noted that treated Wye and Severn water which had been used by the Birmingham Industrial Complex and which was released to the River Tame, had enabled the establishment of fish populations in the river as a result of increased baseflow. However, reduction in flow as a result of abstraction from the River Severn has resulted in the creation of barriers to migratory fish. Hancock (1977) has dealt theoretically with the effects on fish and trout populations (in particular grayling), of the Kielder Water Scheme in the UK, and he predicted that flow changes would lead to a shift of spawning grounds.

The effects of flow alterations on fish stocks were learnt the hard way in Canada, where the likely environmental consequences of the Kemano Diversion were not evaluated before construction, and which led to the decimation of Nechako Chinook Salmon stocks (Day, 1985). In an attempt to encourage industrialisation in its mid-coastal region, the provincial government of British Columbia provided substantial financial incentives to the Canadian Aluminium Company, Alcan, to develop a hydroelectric power plant and smelter at Kemano. The result was the storage and diversion of up to 269 $m^3 s^{-1}$ of water from the Nechako and Nanika rivers and Skins Lake, into the Kemano River. The Nechako Chinook Salmon were adversely affected by flow disruptions from the closure of dams, warm-water releases and siltation (Day, 1985). Subsequently, flows to the Nechako River had to be ensured by a British Columbia Supreme Court injunction, in order to protect Sockeye Salmon migration routes. Increased discharges in the Kemano River have, however, resulted in Pink and Chum Salmon runs in this river. In 1980, Alcan refused to allow the renewal of the court injunction permitting the Minister of Fisheries and Oceans to set flows on the Nechako River that are necessary to maintain fish populations. This coincided with plans to further develop the Kemano Diversion, and resulted in a trial in August 1987. An out-of-court settlement was reached in September 1987, whereby Alcan renounced all rights to the Nanika River and the nearby Murray-Cheslatta system in return for a variety of benefits, among them the freedom from the responsibility to provide more water to the Nechako should compensation flows prove inadequate to support the salmon and sports fisheries, and the assurance that no further mitigation and compensation would be demanded from Alcan. In addition, compensation and cold-water flows that were set in the final agreement are substantially lower than those previously determined by the Minister. Most significant in the settlement, a review of which is provided by the Rivers Defence Coalition (1988), is that the Minister of Fisheries lost the right to regulate flows on the Nechako for fish protection. Additional effects of the final agreement, such as reduced irrigation water available from the Nechako River, and damage to the aesthetics and recreational potential of the area, were ignored in the final, out-of-court settlement.

Remaining in North America, Rozengurt *et al.* (1985), in a review of the ecological impacts of the California State Water Project (CSWP) on the River-Delta-Estuary Sea Ecosystems of San Francisco Bay, have outlined a variety of serious effects of this scheme on fish populations. The CSWP was constructed in 1972 to transfer water from northern California's Feather River to southern California, involving 21 dams and reservoirs, 22 pumping plants, and 1100km of canals, tunnels, and pipelines (Snaddon *et al.*, in prep.). Substantial reductions in freshwater flow to San Franscisco Bay of up to 63% of annual runoff, have resulted in massive reductions of fish populations. Chinook Salmon (*Oncorhyncus tshawytscha*) decreased by 30%, while striped bass (*Morone saxatilis*) decreased by 80%. The economic impact of these losses was estimated to be about 1.3 billion US dollars (Rozengurt *et al.*, 1985).

Apart from fish habitat changes brought about by IBTs, the actual transfer of species has been recorded in a number of cases. For instance, it appears that a minimum of four species are likely to have been introduced to the Great Fish River from the Orange River *via* the Orange/Fish tunnel of the ORP (e.g. Cambray & Jubb, 1977; Laurenson & Hocutt, 1984, 1986). These include the smallmouth yellowfish, *Barbus aeneus*, the Orange River mudfish *Labeo capensis*, the sharptooth catfish, *Clarias gariepinus* and the rock barbel, *Gephyroglanis sclateri*. Transfer of these species may threaten the endangered natural and endemic populations in the Great Fish. Furthermore, it is likely that individuals of species already present in the recipient system have also been transferred from the Orange River, thereby mixing previously isolated gene pools of the same species in the Great Fish River (Jubb, 1976; Cambray & Jubb, 1977; Cambray & Hahndiek, 1979; Laurenson & Hocutt, 1984, 1986; Bruton & van As, 1986).

Additionally, there are a few indigenous fish species in the Malibamatso River that make upstream spawning runs: these runs will be obstructed by Katse Dam, thereby reducing habitat availability to these fish species, restricting spawning to downstream reaches of the river (Chutter, 1993), and generally impinging upon the health of the river.

Mann (1988) and Solomon (1975) have reported that in Wales, roach and dace were able to enter Llandegfedd Reservoir (River Severn) despite the installation of 0.38 mm-mesh screens over the intake of the Severn-Thames Transfer tunnel. The transfer of the predatory fish *Stizostedion lucioperca* appears to have occurred from the Ely Ouse system to the Stour, part of the Ely Ouse to Essex Transfer. It was originally thought that the pumps and high water pressures associated with the transfer system would prevent the transfer of live fish. Live fry have, however, successfully completed the journey, with as yet unassessed impacts on the indigenous fish fauna. Although electric screens have been considered in order to prevent the further transfer of fish, no action has as yet been taken. Since it is most likely that both fry and eggs are transferred, the installation of electric screens will most likely be of minimal efficiency (Guiver, 1976).

García De Jalón (1987) has reported that one of the ecological impacts of the Tajo-Segura transfer, Spain, has been the introduction, and colonisation of the Segura by the gudgeon (*Gobio gobio* L.). This IBT diverts between 100 and 350 x 10^6 m^3 yr^{-1} of water over 286 km from the Atlantic River basin of the Tajo to the Mediterranean river basin of the Segura. Operational since 1979, it has permitted a 150% increase in irrigated land (220 000ha). However, the effects of the introduction of the gudgeon on the indigenous fish populations of the Segura, and other rivers in Spain, is at present unknown. These populations are adapted to high temperatures and low summer flows, and river regulation in the Segura has resulted in a complete reversal of flow and temperature regime (i.e. cooler summer maximum flows). In Namibia, transfers of water from the Kavango River to the Omatako Dam and Swakop River *via* the ENWC are likely by the end of the century. These transfers will probably facilitate the introduction of *Oreochromis andersonii* (three spot bream), *Clarias gariepinus*, and *Tilapia sparmanii* (banded bream) to the recipient systems, which could lead to direct intraspecific competition with existing populations of the same species (Skelton & Merron, 1984).

Hubbs *et al.* (1943) reported on the transfer of the Owens sucker, *Catostomus fumeiventris*, to the Los Angeles Basin *via* the Los Angeles Aqueduct. This is the oldest IBT in California, which was completed in 1913 in order to transfer water from Owens Valley on the eastern slopes of the Sierra Nevada Mountains to the city of Los Angeles. Further north, Lindsey (1957) reviewed the distribution of freshwater fish in drainages of the British Columbia mainland, with reference to proposed water diversions for hydroelectric power development. The proposed Liard-Stikine diversion, which would link the Arctic and Pacific drainages, would almost certainly introduce trout to the upper Liard River, and would probably transfer northern pike (*Esox lucius*), along with its tape-worm (*Triaenophorus crassus*), from the Stikine to the Liard rivers. The introduction of the tapeworm to the Liard River would have a significant impact on the economic value of Pacific salmon in this river. With respect to the Long Lake and Ogoki Diversions in Canada, the prevention of transfer of sea lamprey from Lake Superior to the Albany River system (into which both the Ogoki and Long Lake systems drain) was fortunately avoided by the presence of hydro-power dams, which, along with some natural barriers, have prevented the spread of these aliens (Day *et al.,* 1982; Day, 1985). However, it must be added that the severe erosion as a result of the diversion has caused siltation of fish-spawning beds as well as changes in species composition of the recipient systems.

An interesting and possibly severe effect of IBTs relates to the rélease of population-specific pheromones by juvenile, anadromous salmonids and their homeward migration as adults. The hypothesis, which was put forward by Nordeng (1977) as a result of work on the Salangen River in Norway, suggests that homeward navigation by these commercially important fish species is an inherited response to population-specific pheromone trails released by descending smolt. The juvenile fish release the pheromones from their skin mucus as they move downstream during their migration to the sea during spring and summer. The hypothesis has serious implications for IBTs where the rivers involved support anadromous fish species that use this adaptation for home recognition. Water transferred between basins would have the consequence of leading homebound adults into an incorrect river for spawning due to deceptive pheromone trails, which would ultimately result in the depletion of fish stocks. It was on the strength of this hypothesis that Tøndevold (1984) recommended that transfers of water from Lake Suldalsvatnet to Hylsfjorden, also in Norway, cease during June and July during salmon runs, in order to reduce the possibility of returning salmon entering Hylsfjorden rather than Suldalslågen. This recommendation was, in fact, carried out.

Hesthagen & Fjellheim (1987) have studied the effects of glacier-fed water transferred from the River Veo on the production and food source of brown trout (*Salmo trutta*) of the receiving Smådøla River, in Norway. Although not an indigenous fish in the Smådøla, transfers of water to the river significantly reduced fish production from 271.5 to 103.1 g $100m^{-2}$ a^{-1}. This was due primarily to reduced recruitment and availability of important food organisms.

2.4.3 Fish Parasites

Arai & Mudry (1983) presented a study on the possibility of protozoan and metazoan fish parasite transfers as a result of a proposed transfer in north-eastern British Columbia, the McGregor Diversion. The transfer was proposed to divert water from the McGregor River (Pacific drainage) to the Parsnip River (Arctic drainage). Their results showed that 26 parasites had disjunct distributions in the study area. Three forms were identified as posing a significant threat to the fisheries resources of the immediate area and also to downstream areas. Based in part on these studies, the British Columbia Hydro and Power Authority, in a public announcement, suspended engineering studies of the proposed diversion.

Arthur *et al.* (1976) undertook a similar study of the potential consequences of the transfer of fish parasites from Stevens Lake to Aishihik Lake (Yukon Territory, Canada) as a result of an interlake diversion for hydro-electric power generation, the Stevens-Aishihik Inter-Lake Transfer. Several species of parasite, of potential economic and pathogenic importance, were identified as likely candidates for transfer.

2.4.4 Human Diseases and Disease Vectors

An environmental problem associated with the proposed Texas Water Plan (USA) was the transfer of water to certain areas of West Texas, where the encephalitis-carrying mosquito is found (Greer, 1983). The delivery of large quantities of water could lead to their proliferation. The Texas Water Plan was designed to transfer water from the Mississippi River, and rivers in the eastern half of Texas, to rivers and reservoirs in the west and south-west of the state. These plans have been shelved, due to the discovery that abstraction of water from the Mississippi at the intended scale, was not economically feasible (Greer, 1983). In Russia, the potential exists for the transfer of water-borne diseases and parasites along the 2000-km unlined transfer canal from Tobol'sk in Western Siberia to the AmuDar'ya in Central Asia, which is the transfer route for SCAP (Gorodetskaya, 1982).

Still in Russia, an aspect of the ETP investigated by Voronov *et al.* (1983), is the possible effect that the transfer would have on the aggravation, or attenuation, of a variety of epidemiological conditions within the recipient area. The main effect was likely to have been the spread of opisthorchiasis (liver-fluke disease) and diphyllobothriasis (Broad Fish tapeworm) along the entire length of the proposed canal (see also Kuperman, 1978). This would be due to the possible transfer of the respective intermediate hosts *Bithynia leachii* (a freshwater snail) and *Cyclops* sp. (a crustacean) to the southern regions where, at present, these invertebrates are almost absent. The increased supply of water to the midland regions was also likely to encourage the establishment of water fever in these areas. The authors warn that strict adherence to sanitary and preventative measures along the entire canal route will be required to prevent the spread of diseases.

The ENWC, in Namibia, could transfer schistosomiasis from the Kavango River to recipient areas. Control of the host snails is being studied, but it is possible that the three dams in the scheme could become reservoirs of infection (Department of Water Affairs & the Department of Agriculture and Nature Conservation SWA/Namibia, 1987).

The only ecological investigations that accompanied the construction of the Tugela-Vaal Scheme in South Africa, were related to the potential transfer of the schistosomiasis (bilharzia) host snails *Biomphalaria* sp. and *Bulinus* (*Physopsis*) sp. from the donor Tugela

River to the recipient Vaal system: the disease does not occur in the Vaal catchment (Pitchford & Visser, 1975; Pretorius *et al.*, 1976). The problem was of concern due to the fact that Pitchford (1953) had shown that snail ova are hardy and could pass through the mechanical components used in IBTs. However, there has fortunately been no evidence of the establishment of snail populations in the Vaal due possibly to prevailing winter temperatures in the system (Professor J. de Kock, Snail Research Unit, Potchefstroom University, personal communication, 1996). The Tugela-Vaal Scheme was one of the first designed in South Africa, involving the pumping of water from the Tugela, in KwaZulu/Natal Province in the east of the country, over the intervening mountain range to the interior, in order to augment water supply in the Vaal River catchment (Department of Water Affairs and Forestry, 1991b).

2.4.5 Vertebrates other than fish

It is believed that the migration routes of great herds of Caribou would be blocked by the physical barriers that the canals and reservoirs of NAWAPA would have created, and this would also have affected those human communities reliant on the Caribou as a source of food (Nace, 1966).

The Namibian ENWC has been severely criticised for its design due to the fact that it consists in part of a 203-km long open concrete canal. Estimates of the number of animals that fall into the canal are as high as 17500 per annum (these include large mammals and smaller reptiles) although crossing points are provided and sloping ramps are being installed. The ideal solution would have been for the canal to be covered. However, the cost was deemed prohibitive at the time (Comrie-Grieg, 1986; Jones, 1987; Petitjean & Davies, 1988a,b). Of course, attention has focused on large-animal deaths, with no cognisance of the myriad smaller organisms (vertebrates and invertebrates) that fall into the canal, with obvious consequences for water quality.

2.4.6 Plants

Most of the literature relating to IBTs and their effects on plant groups, deals with the transfer of diatoms and macrophytes through transfer schemes in the United Kingdom. Holmes & Whitton (1977) undertook a survey of the River Tyne (Kielder Water Scheme) in an attempt to predict the floristic and algal changes that would occur with the transfer of water to the Wear and Tees rivers. They suggested that the Wear would be subject to recurrent inoculation by several species presently absent but that it was unlikely that any of these would become pests. Later, Belcher & Swale (1979) made an interesting observation while studying the distribution of the coastal diatom, *Actinocyclus normanii,* in freshwater systems in England. With particular reference to reservoirs, it was noted that these diatom populations appeared to be non-viable, disappearing after only a few generations. The diatom owed its presence in these particular reservoirs to the water transfers which feed them, and which periodically re-inoculate the waters. The diatom was found only in reservoirs filled with water pumped from donor rivers. Belcher & Swale (1979) postulated that if the transfer of water were to cease, these populations would disappear.

Due to water transfer by the Ely Ouse to Essex Scheme in the United Kingdom, transfers of *Stephanodiscus* blooms from the point of abstraction have given rise to public complaints about potable water quality, and transfers were halted (Guiver, 1976). However, on a subsequent occasion, the transfers could not be halted since they were required to maintain water levels in the Essex reservoirs (see also Jollans & Zabel, 1982). Furthermore, prior to water transfers the dominant diatom in the River Stour, one of the recipients of the Ely-Ouse to Essex Scheme, was the alga *Melosira* sp. The inoculation provided by the substantial transfer of *Stephanodiscus* appears to have caused a shift in dominance to the latter algal species (Guiver, 1976). In an analysis of the algal characteristics of Silverwood Lake in the USA, Kubomoto *et al.* (1974) noted that some of the species occurring in the lake appear to have been transported from the north through the California Aqueduct, part of the California State Water Project.

Turning to macrophytes, Holmes & Whitton (1977) conducted a macrophyte survey of the River Tees, recipient of the Kielder Water Scheme in the UK, in order to describe the changes in the river since 1965. They attributed the upstream spread of four submerged angiosperms to river regulation, at Cow Green Reservoir. They also stated that transfer of water from the Tyne to the Tees is likely to cause further changes, since four Tyne species (found upstream of the abstraction point of the transfer scheme) are absent in the Tees, and another 22 are found only in the lower reaches of the Tees.

In southern Africa, the clarity of the water released from Katse Dam in Lesotho (LHWP) could lead to increased light penetration, and this, combined with the more constant flow below the dam, will encourage the growth of various macrophytes, such as filamentous algae (especially in riffle areas), *Potamogeton* sp. (pond weed) and *Typha* sp. (bulrush) (Chutter, 1993).

2.5 Estuarine, coastal and oceanic implications

Yuexian & Jialian (1983), in their review of the probable effects of China's East Route water transfer, anticipated that the use of the Chang Jiang as China's main navigation channel would not be affected by this transfer, even when all three proposed routes are operational. However, siltation in the lower reaches of the Yangtze River was expected as a result of the decreased flow in the river due to the East Route diversion, especially during low-flow months (Changming & Dakang, 1987). Altered flows, sediment transport and tidal gradients in the Yangtze estuary will adversely affect navigation in these parts of the river and estuary. Estuarine and brackish water fish will also negatively be affected (Xuefang, 1983).

In Iraq, the coastal marshes of the Tigris and Euphrates rivers are threatened by the almost total diversion of the flow of the Euphrates and many of its tributaries that carried flood waters to these marshes (Section 2.1.1). In addition, abstractions from the Euphrates by Turkey in order to fill the relatively recently completed Ataturk Dam have demonstrably reduced the flow of the river to the marshes by 10% between 1985 and 1990 (Pearce, 1993). The marshes are regarded as the most important wetland bird habitat in the whole of Eurasia with millions of waterfowl feeding on the fish and invertebrates of the marshes every year. The area is not declared under the Ramsar Convention and its loss will be measured on an inter-continental, if not a global scale.

Chapter 3 Southern African IBTs

3.1 Introduction

In 1990, 300 million people were living in water scarce regions throughout the world; in 2025, this figure will increase to 3 billion (Falkenmark, 1993). Water scarcity is defined by Falkenmark (1993) as the case where there are more than 600 people for every million cubic metres of available water per year. Most of southern Africa falls within this category (Basson, 1996; van Niekerk *et al.*, 1996), although water availability and usage varies substantially throughout the subcontinent. South Africa accounts for the bulk of consumption, while only 10% of the water on the subcontinent is located here (Table 3.1).

Table 3.1 Statistics on water availability and consumption by country, in southern Africa (adapted from Conley, 1995).

Country	Total Water Available (10^6 m^3)	% Water Available	Total Water Use (10^6 m^3 yr^{-1})	% Water Use
Angola	158 000	32	480	2
Zambia	96 000	19	360	2
Tanzania	76 000	15	480	2
Moçambique	58 000	12	760	3
South Africa	50 000	10	19 040	84
Zimbabwe	23 000	5	1 220	5.5
Namibia	9 000	2	n/av	n/av*
Botswana	9 000	2	90	0.4
Malawi	9 000	2	160	0.7
Swaziland	7 000	1.5	n/av'	n/av*
Lesotho	4 000	1	50	0.2

*n/av: not available

Climatic stochasticity is a feature of the southern African subcontinent, and South African rivers have been shown by Alexander (1985) (see also Braune, 1985) to be highly variable in terms of their hydrology. Davies *et al.* (1995) have described the rivers of southern Africa as 'predictably unpredictable', and the prediction of droughts on a year-by-year basis is extremely difficult. Added to this is the substantially low conversion of rainfall to runoff on the southern African subcontinent (a ratio of 8.9%; Alexander, 1985). Long-term water-resources planning in South Africa is a necessary but complex task (Conley, 1995).

Coupled with the uneven distribution of water in South Africa, anthropogenic pressures such as over-abstraction, salinisation and pollution, are exacerbating problems of water supply (Snaddon *et al.*, 1998). Furthermore, the distribution of current water supply infrastructure amongst the South African population is inequitable. For example, between 12 and 14 million South Africans, of a population of approximately 45 million, have no access to clean and safe potable water, while some 20 million people have no water-borne sanitation (e.g. Palmer Development Group, 1995). Thus, in order to meet the water requirements of previously disadvantaged communities, water demands inevitably will increase, and are predicted to do so at a rate of approximately 2% per annum until the year 2020 (Heyns *et al.*, 1994; Pitman & Hudson, 1994).

In southern Africa, many of the larger rivers have been impounded, and, in many cases, more than once (Davies *et al.*, 1993). According to figures given by the Department of Water Affairs and Forestry (Department of Water Affairs & Forestry, 1986a), *ca* 50% of the mean annual runoff (MAR) of the region is stored in approximately 520 'major' regulating structures. In order to ensure a continual supply of water, the high degree of hydrological variability in such arid areas demands larger reservoir storage capacities than are required in more humid parts of the globe (e.g. Alexander, 1985). Furthermore, many of the larger impoundments, such as Gariep Dam on the Orange River, are situated in parts of the region which are geomorphologically appropriate for dam construction, but where water demand is low. Thus it has become necessary to transport this stored water to areas where demand is perceived to be higher, and of a greater priority, than use within its catchment of origin.

This has led to the creation of water redistribution networks involving complex IBTs, and the extent of these networks is increasing (Conley, 1995; van Niekerk *et al.*, 1996; Basson, 1997). There is growing pressure to look beyond both provincial and national boundaries for water. The Congo and Zambezi rivers are the only truly large perennial systems in southern Africa (Table 3.2), and both of these rivers are being investigated as potential sources of water, for countries such as Zimbabwe, Botswana and South Africa (e.g. Alexander, 1996). In hyper-arid regions such as Namibia, where there are no assured permanent surface resources (e.g. Jacobson *et al.*, 1995), major groundwater resources are over-exploited and, in this instance, it is inevitable that permanent water supplies from perennial waters must come from neighbouring regions.

Table 3.2 The mean annual precipitation (MAP) and mean annual runoff (MAR) of some major rivers of southern Africa, listed in order of decreasing catchment area. (After Conley (1995): extracted from Pitman & Hudson (1994)).

River	Area (km^2)	MAP (mm)	MAR ($10^6 m^3 yr^{-1}$)
Democratic Republic of Congo	3 981 000	1500*	1 250 000
Zambezi	1 234 000	860	110 000
Orange	973 000	330	11 860
Okavango	586 000	580	11 650
Limpopo	413 000	520	7 330
Rovuma	155 000	1100	28 000
Cunene	117 000	830	5 550
Save	104 000	680	6 200

*Estimated MAP.

3.2 IBTs in South Africa

Petitjean & Davies (1988a,b) reviewed the extant and planned IBTs for southern Africa, and Figure 3.1 has been adapted from their work, to illustrate the location of all current and proposed IBT schemes, while Table 3.3 provides a list of their data for both existing and potential IBTs of the subcontinent. The list has been updated using information from Pitman & Hudson (1994), van Niekerk *et al.* (1996) and Basson (1997). The following section does not attempt to provide technical details on IBTs in South Africa, as this has been done by the above authors. In addition, a synthesis of information on South African IBTs is currently in preparation by this author, for the Water Research Commission. For the purposes of this study, a brief overview of South African transfers is given in order to provide a regional context for the scheme which was studied here.

In 1988, Petitjean & Davies (1988a,b) calculated that the total volume involved in IBTs exceeded $1.63 \times 10^9 m^3 yr^{-1}$, and this volume was predicted to rise to $4.82 \times 10^9 m^3 yr^{-1}$: this is almost equivalent to the total MAR for the Western Cape Province. This total has not been re-evaluated or updated here, as many of the transfer schemes are pulsed throughout the year, and thus a total volume would be an overestimate.

Table 3.3 Some attributes of (a) existing inter-basin transfer schemes in South Africa, and (b) those proposed or under construction. (Modified from Petitjean & Davies, 1988b). Note that the capital costs provided in this table are not comparable, as they have been distilled from a variety of sources, that provide estimates from a range of years.

(a) Existing Schemes

Scheme	Phase	Capital Cost (R million)	Volume of transfer ($x10^6 m^3 yr^{-1}$)	Donor river(s)	Recipient river(s)
Usutu Vaal River Government Water Scheme	I	-	500	Vaal	Trichardtspruit (Olifants)
		-		Vaal	Steenkoolspruit (Olifants)
	II	84	78	Assegaai (Usutu)	Little Vaal/Vaal
	Emergency Scheme	-	-	Assegaai (Usutu)	Ngwempisi Spruit (Usutu)
Grootdraai Dam Emergency Augmentation Scheme (1983)	-	-	-	Vaal	Vaal
Usutu River Government Water Scheme	5 phases	-	103	Usutu	Mpamaspruit (Usutu)
				Mpamaspruit (Usutu)	(Hydro-electric power station)
				Ngwempisi Spruit (Usutu)	Mpamaspruit(Usutu)
				Bonnie Brook (Usutu)	Usutu
Komati-Usutu River link system Government Water Scheme	-	-	227	Komati	Olifants
Slang River Government Water Scheme	-	-	-	Slang (Buffels)	Perdewaterspruit (Vaal)
Tugela-Vaal Scheme	I	-	160	Tugela	Nuwejaarsspruit (Vaal)
	IIa	566	347	Tugela	Nuwejaarsspruit (Vaal
	IIb	-	611	Tugela	Nuwejaarsspruit (Vaal)
Lesotho Highlands Water Project	Ia	-	533	Malibamatso	Nqoe (Caledon)/Ash (Vaal)
Mhlatuze Government Water Scheme	-			Mhlatuze	Mhlatuze (Lake Nseze)
Orange River Project	-	-	354	Orange	Teebusspruit (Great Fish)
	-	-	26	Great Fish to Little Fish	Sundays
Riviersonderend-Berg-Eerste River Government Water Scheme	I & II	-	0.5-5 $m^3 s^{-1}$	Riviersonderend (Breede)	Berg
	III	-	130	Riviersonderend (Breede)	Eerste
Mooi-Mgeni Scheme	Mearns Emergency Pumping Scheme	-	3-30	Mooi (Tugela)	Mpofana (Mgeni)
Amatole Scheme	-	225	36	Toise & Kubusi	Yellowwoods & Nahoon
Palmiet River Scheme	I	676	29	Palmiet	Steenbras

Table 3.3 continued...

(b) Proposed Schemes					
Scheme	**Phase**	**Capital Cost (R million)**	**Volume of transfer ($x10^6 m^3 yr^{-1}$)**	**Donor river(s)**	**Recipient river(s)**
Lesotho Highlands Water Project	Ib	-	851	Matsoku (Malibamatso)/ Senqunyane	Malibamatso/Ash (Vaal)
	Further Phases	25 000	2207	Senqunyane (Orange)	Ash (Vaal)
Palmiet River Scheme	II	-	127	Palmiet	Steenbras
Mooi-Mgeni Scheme	Ia	-	$3 m^3 s^{-1}$	Mooi (Tugela)	Mpofana (Mgeni)
	Ib	-	$4\text{-}10 m^3 s^{-1}$	Mooi (Tugela)	Mpofana (Mgeni)
	II	-	$6 m^3 s^{-1}$	Mooi (Tugela)	Mpofana (Mgeni)
Mzimkulu-Mkomaas-Illovo Scheme	I	680	375	Mkomaas	Mgeni & Illovo
	II	-	-	Mzimkulu	Mkomaas
Sabie River Government Water Scheme	I	270	25	Marite	Sand
	II	-	-	Sand	Sand/Klein Sand
The Zambezi Aqueduct	-	6000	2500 to 4000	Zambezi	Botswana & Vaal
Tugela-Mhlatuze Transfer	-	260	-	Tugela	Mhlatuze
Skuifraam Supplementary Scheme	-	82	19	Berg	Upper Berg
Misverstand Dam Raising	-	436	70	Lower Berg	Voëlvlei Dam (Berg River catchment)
Keerom Diversion	-	776	90	Olifants	Berg
Michell's Pass Diversion	-	20	15	Breede	Boontjies (Klein Berg River catchment)
Molenaars Diversion	-	89	35	Molenaars	Berg
Brandvlei to Theewaterskloof	-	317	100	Breede River	Elands (Riviersonderend catchment)

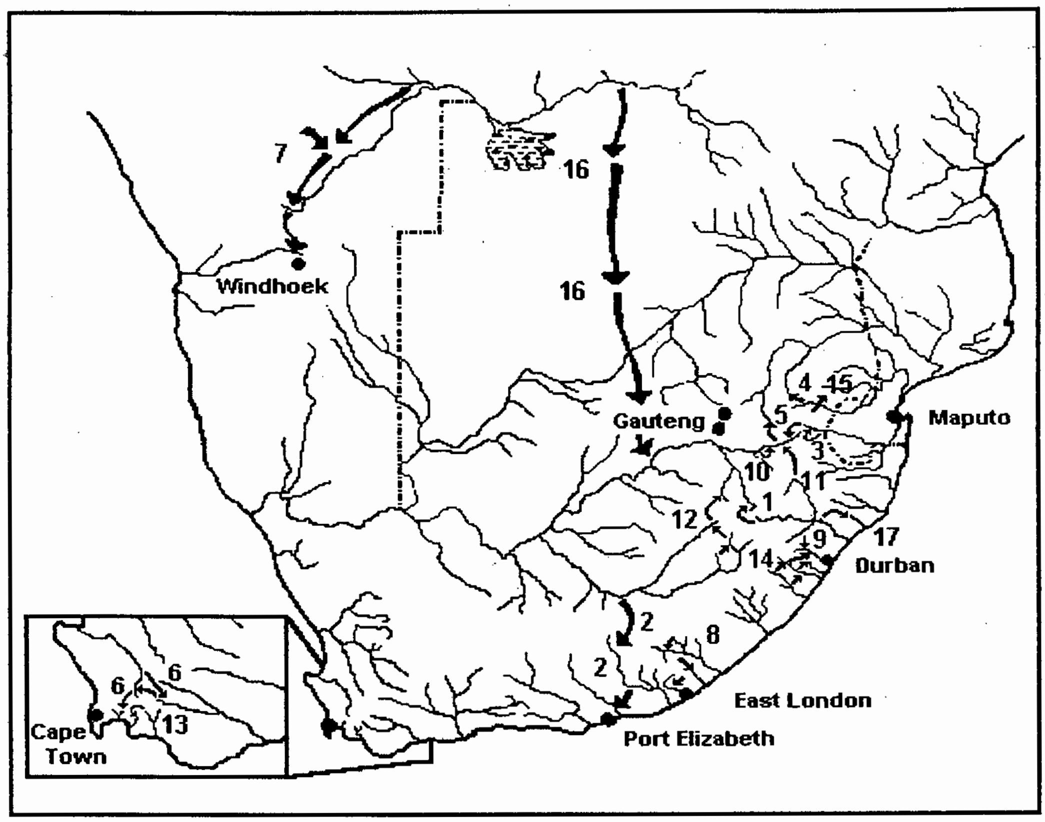


Figure 3.1 A map of some of the larger extant and proposed IBT schemes in southern Africa. 1, Tugela-Vaal Scheme; 2, Orange River Project; 3, Usutu Scheme; 4, Komati Scheme; 5, Usutu-Vaal Scheme; 6, Riviersonderend-Berg-Eerste River Government Water Scheme; 7, Eastern National Water Carrier; 8, Amatole Scheme; 9, Mooi-Mgeni Scheme; 10, Grootdraai Emergency Scheme; 11, Slang River Scheme; 12, Lesotho Highlands Water Project; 13, Palmiet River Scheme; 14; Mzimkulu-Mkomaas-Illovo Scheme; 15, Sabie River Government Water Scheme; 16, Zambezi Aqueduct; 17, Tugela-Mhlatuze Transfer Scheme.

The significance of IBTs to water resources development in South Africa is clear from Figure 3.1, while Figure 3.2 illustrates the importance of these schemes to the economy of the country, in terms of the percentage of the Gross Geographic Product (GGP) that is at least partially dependent on IBTs. Since the 1970s, major IBT schemes have been an important part of water resources development in South Africa, with the first of these being the Tugela-Vaal Transfer Scheme, constructed in 1975 (e.g. Department of Water Affairs and Forestry, 1991b; Asmal, 1996; Snaddon *et al.*, 1998) (Scheme 1, Figure 3.1). This scheme was soon followed by the Orange River Project (ORP; Scheme 2, Figure 3.1) and the Komati and Usutu schemes (Schemes 3 and 4, Figure 3.1).

The major water demand in South Africa is located in the province of Gauteng (Pretoria, Johannesburg, Vereeniging and a few other industrial cities), in the catchments of the Vaal (south), the Crocodile (north), and the Olifants rivers (east). Thus, as can be seen in Figure 3.1 and Table 3.3, most of the current and proposed schemes in southern Africa are received by the catchments of this province. The Vaal River is presently augmented by several river basins, including the Tugela, Orange, Usutu, Komati, Olifants and Buffels rivers, and, currently, almost 50% of water supply originates beyond the catchment boundaries (Pitman & Hudson, 1994; van Niekerk *et al.*, 1996). Water demands are steadily increasing in the province, such that additional storages and diversions are being planned and constructed. Construction of the first international transfer scheme in southern Africa, the Lesotho Highland Water Project (LHWP; Scheme 12, Figure 3.1) has recently entered its second subphase, Phase IB. As described in Chapter 2, the completion of this phase of the project will result in approximately 30 $m^3 s^{-1}$ being transferred from the headwaters of the Orange River in Lesotho, into a small tributary of the Vaal River in South Africa. Further phases of this project remain at the planning stage.

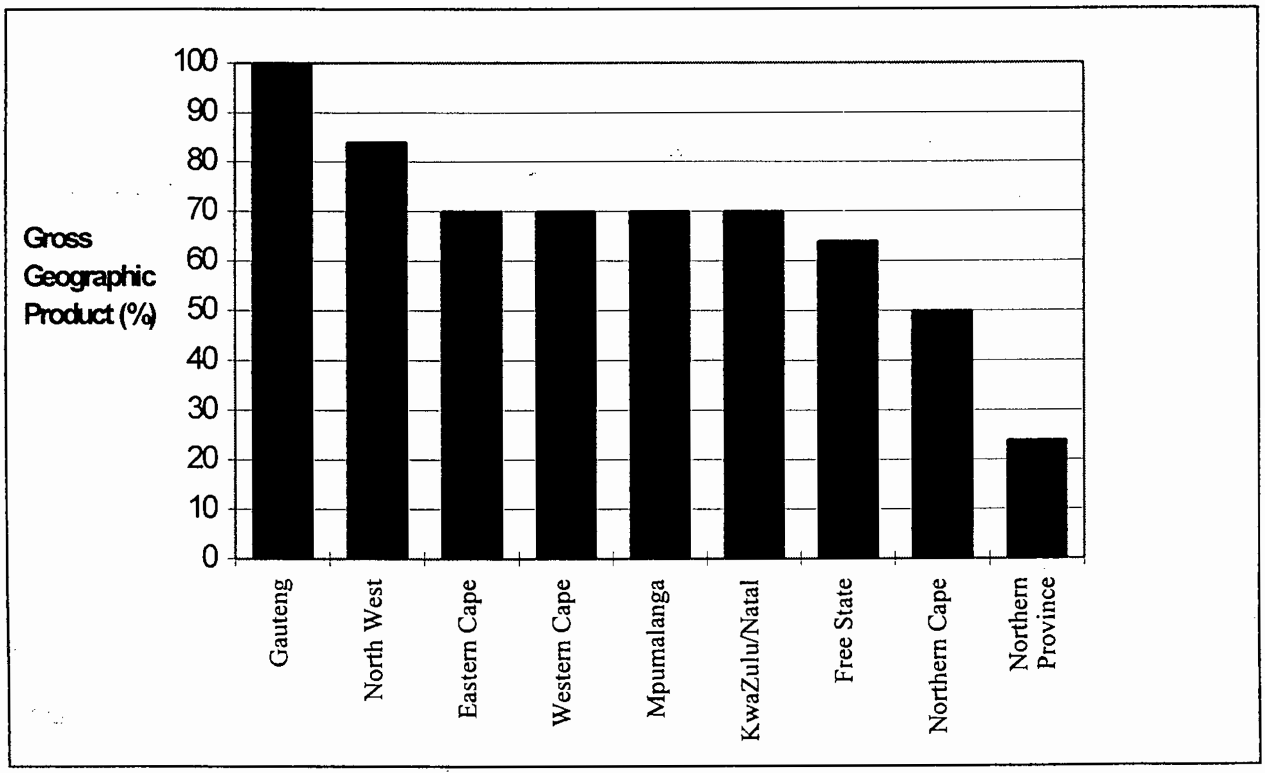


Figure 3.2 A graphic presentation of the percentage of Gross Geographic Product (GGP), as an indicator of productivity per province, that is at least partly reliant on inter-basin water transfer schemes (from van Niekerk *et al.* (1996)).

Moving to the east of the country, the Durban-Pietermaritzburg metropolitan area in KwaZulu/Natal Province, is one of the fastest growing urban areas in the world (van Niekerk *et al.*, 1996). The Mooi-Mgeni Transfer Scheme brings water to this area; plans to increase transfer capacity from the Mooi are complete and construction will begin when the need arises (Henderson, 1995). Further transfers are planned, to augment flows in the Mgeni River, bringing water from the Mkomaas and Mzimkulu rivers. These coastal rivers are relatively short, however, and their yields are limited. The Richards Bay/Empangeni area is another growth node within the province. The Mhlatuze River Scheme is an intra-basin transfer scheme which has supplied water to this area since 1984 (van Niekerk *et al.*, 1996; Wepener & Cyrus, 1997). However, the Mhlatuze no longer meets the demands of the area, and augmentation of this source is planned from the Tugela River. Thus, although KwaZulu/Natal is situated on the wetter eastern coast of South Africa, 70% of the province's production is dependent on IBTs (Figure 3.2).

The East London/Bisho and Port Elizabeth area, in the Eastern Cape Province, is supplied by a number of small diversions for urban and industrial water use, and the larger ORP (Scheme 2, Figure 3.1), which began transferring an annual volume of $350 \times 10^6 \, m^3 \, yr^{-1}$ of water in 1977. The purpose of the ORP was to provide irrigation water to the receiving catchments. Further west, the Western Cape Province is a winter rainfall area, with a mean annual precipitation (MAP) of *ca* 1250mm yr^{-1}. Intensive irrigation agriculture predominates, with viniculture and fruit and wheat farming forming the major activities. Recently there has been a rapid human population growth in the Greater Cape Town Metropolitan Area, swelling the population from less than a million people in the early 1980s to over three million currently. By 2020, the population is projected to swell to 4.5 million (Shand, 1992). Because of its historical development, the Cape Town area has traditionally been short of water. As such, IBTs have assumed increasing significance in the water supply of the region, with 98% of Cape Town's current water demand of 360 million $m^3 \, yr^{-1}$ coming from outside its boundaries (Table 3.4): demand is increasing at a rate of 4% yr^{-1} (Clayton, 1994, 1996).

Table 3.4 Supply of water to Cape Town, giving a breakdown of the supply reservoirs, and water demand, for July 1993 to June 1994, inclusive (from Clayton, 1994, 1996).

Reservoir	River	Percentage of Supply		
		1993/4	**1994/5**	**1995/6**
Cape Town catchments:				
-	Liesbeek (Albion Springs)	-	0.1	0.3
*Hely Hutchinson, Woodhead, De Villiers, Victoria, Alexandra	Table Mountain rivers	1.3	1.2	1.3
Catchments beyond city:				
Steenbras	Steenbras	13.7	10.4	11.5
Wemmershoek	Berg	21.5	21	20.2
Voelvlei	Berg	25.6	25.6	22.2
Theewaterskloof	Riviersonderend	37.9	41.7	44.5

*The total volume from these sources comprises 3.7 x 10^6 m^3 yr^{-1}, compared to 185.9 x 10^6 m^3 yr^{-1} from catchments beyond the boundary of the City.

The rivers now supplying this water include the Riviersonderend, Du Toits, Elands, Berg, Eerste and Steenbras systems with an additional IBT which will shortly abstract water from the Palmiet River (Palmiet Phase 1: Department of Water Affairs and Forestry, 1992) (Scheme 13, Figure 3.1; Table 3.3). Importation of water on this scale places major stresses on the donor rivers in so far as they also supply irrigation water to farms and to human communities within their own watersheds. One of the Western Cape schemes, the Riviersonderend-Berg-Eerste River Government Water Scheme (RBEGS; Scheme 6, Figure 3.1), was the subject of this study, and is described in further detail in Chapter 4.

The ecological effects of IBTs have been discussed in the previous chapter, but the political consequences are of equal importance. In many instances, the rivers that are regulated by IBTs form international boundaries or traverse more than one country. For example, Moçambique is downstream of a variety of IBT schemes on rivers in South Africa, such as the Sabie, Komati (Inkomati) and Letaba/Olifants rivers (Table 3.3). Thus the major sources of water for Moçambique often run dry, and this, combined with poor water supply infrastructure, leaves many Moçambicans without water. Similarly, virtually all rivers in Swaziland rise beyond the borders of this country. All but two of its major rivers originate in South Africa and flow either through Moçambique, or back into South Africa. One such example, the Usutu River, rises in South Africa, and flows eastwards through Swaziland, but is part of a South African IBT that diverts 260 x 10^6 m^3 yr^{-1} to the Vaal River, which flows westwards (Department of Water Affairs and Forestry, 1991a; Table 3.3). Thus, the Usutu ultimately enters the Orange River which in turn, in its lower reaches, forms the border with Namibia and is a source of water for both South Africa and Namibia.

Namibia presently uses $20 \times 10^6 m^3 y^{-1}$ from the Orange River and irrigates 2000ha of land adjacent to the river (Martin, 1993), but South Africa is committed to a further allocation of water to Namibia from the Orange River. This allocation will not be determined until Namibia has completed a comprehensive assessment of its national water requirements. However, the final transfer volume of the Lesotho Highlands Water Project (LHWP) was estimated without taking this allocation into account. Furthermore, the high evaporative losses from the middle and lower Orange River catchment were not considered in the process of calculating the yield of the LHWP (Benade, 1993), and thus the Northern Cape Province would also suffer from upstream over-utilisation.

3.3 Concluding Remarks

After fairly comprehensive assessments of the water resources of South Africa, most water managers and engineers have concluded that the full development of current resources will meet demands for water only for the next two or three decades (van Niekerk *et al.*, 1996; Basson, 1997). The development of water resources in South Africa appears to be taking the form of increasingly larger and more complex water supply projects, such as the LHWP and the Zambezi Aqueduct. It appears to be only a matter of time before the southern African subcontinent is supplied by a water network (Snaddon *et al.*, 1998), similar to the energy grid currently in use. In some cases, flow is already being augmented in rivers that have been diverted across other catchment boundaries. It is essential that IBTs are considered a form of river regulation associated with an array of ecological consequences, that require comprehensive assessment at both the planning and operational stages.

Chapter 4 Description of the Study Area

4.1 Berg River

The Berg River rises at approximately 1500m AMSL, in the Groot Drakenstein Mountains, Western Cape (Figure 4.1). These mountains are characteristic of the Cape Fold Mountains, comprising sandstones and quartzites of the Table Mountain Group, part of the Cape Supergroup (Lambrechts, 1979). The river flows north through agricultural lands and several towns, before reaching the Atlantic Ocean at Velddrif (Figure 4.1). Mean annual precipitation (MAP) within the basin ranges from 2600mm in the mountain region, to 400mm on the coastal plain, while mean annual evaporation increases from 1500mm near the source, to 2400mm near the coast (Berg, 1993).

The vegetation of the upper catchment of the Berg River is dominated by mesic mountain fynbos. Mountain fynbos is a vegetation type which is characterised by ericoid shrubs and restioid herbs, with the frequent occurrence of proteoid shrubs (Kruger, 1979). This vegetation occurs on the foothills, slopes and summits of mountains of the Cape Fold Belt, on soils that are well-leached and often sandy. The expected rainfall in areas vegetated by mountain fynbos ranges from 250 to 3300 mm yr^{-1}. Lower down its course, the river flows through hilly valleys of Malmesbury Shale deposits, which largely support renosterveld (Lambrechts, 1979). Coastal renosterveld is characteristic of the lower altitudes of the Western Cape, growing on undulating lowlands, at a rainfall of 300-600 mm yr^{-1}. This vegetation occurs on clay-loams or clay, in shallow, slightly acid to neutral soils, and is characterised by low narrow-leafed shrubs, deciduous grasses and other succulent shrubs (Kruger, 1979). Closer to the sea, the vegetation changes to strandveld, growing on sandy soils and surface limestones, at a rainfall of between 200 and 300 mm yr^{-1} (Harrison & Elsworth, 1958). Strandveld vegetation is characterised by a diverse range of growth forms, including broad-leafed shrubs, succulents, spiny shrubs and spring annuals (Kruger, 1979).

The outlet of the IBT tunnel is located in the upper reaches of the Berg River, where it flows through the La Motte Forest Reserve, near the town of Franschhoek (Figure 4.1; Plates 1 and 2). The riparian vegetation of the study area is dominated by exotic pines and acacias

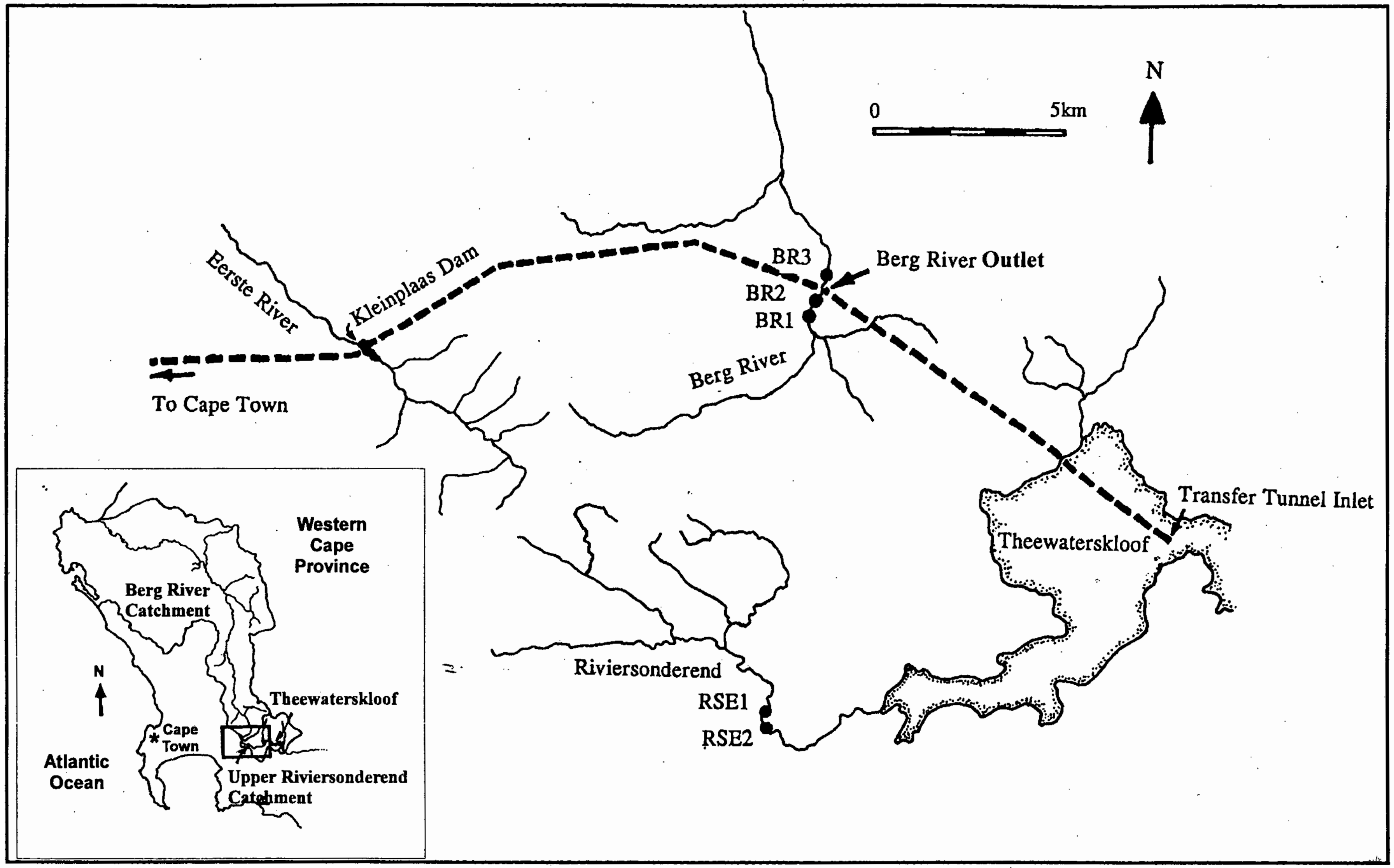


Figure 4.1 A map of the rivers involved in the RBEGS IBT, showing the location of the five study sites described in Section 4.4.

Plate 1: The Riviersonderend-Berg water transfer tunnel outlet, releasing water into the upper Berg River. The photograph was taken in summer 1994.

Plate 2: This photograph was taken on the same day as Plate 1, and gives a view of the tunnel outlet from the downstream sampling site, BR3 (see Figure 4.1).

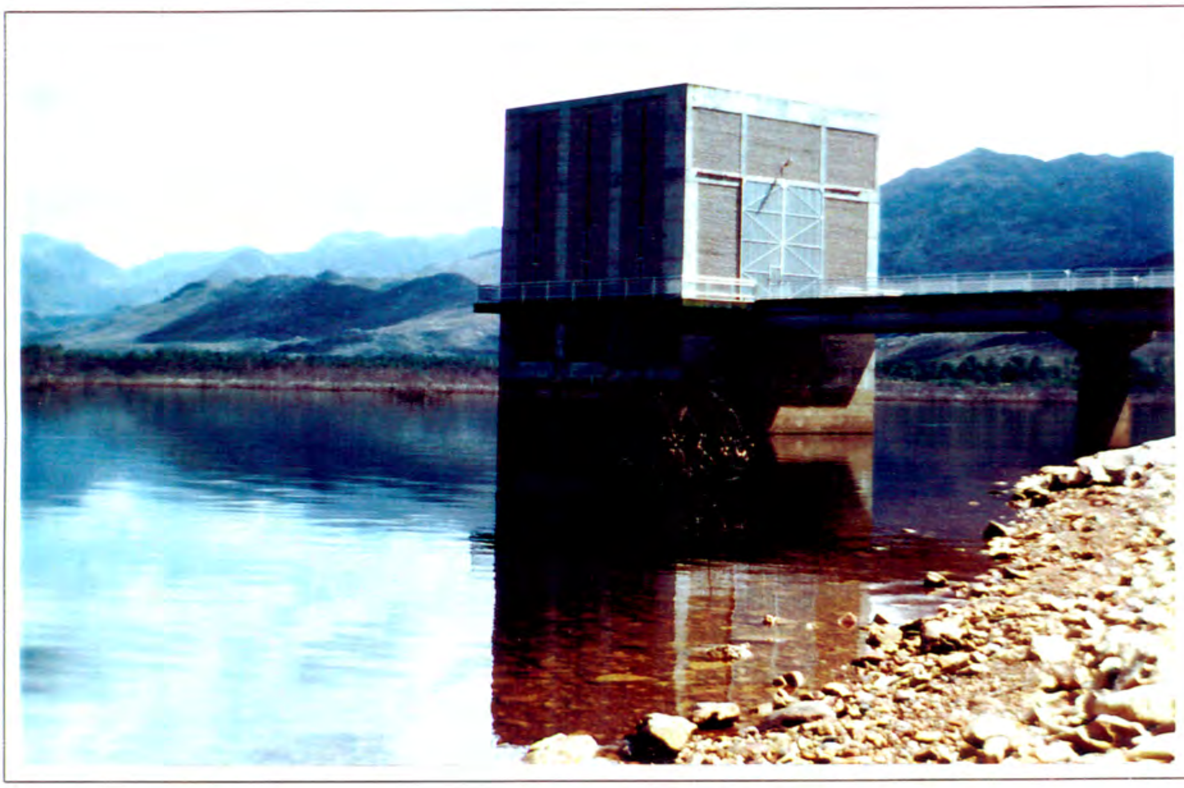


Plate 3: The Riviersonderend-Berg-Eerste River Government Water Scheme inlet tower, in the Theewaterskloof reservoir.

Plate 4: This photograph was taken in summer at BR2, one of the unimpacted sampling sites above the Berg River IBT outlet (see Figure 4.1). Three replicate benthic macroinvertebrate samples were taken on each sampling occasion, in the riffle in the centre of the photograph. The first unimpacted site, BR1, is situated approximately 300 metres upstream of this riffle.

(respectively, *Pinus radiata* D. Don and *Acacia longifolia* (Andr.) Willd.), with a few indigenous species such as wild almond (*Brabejum stellatifolium* L.) and the lance-leafed myrtle (*Metrosideros angustifolia* (L.)). In the upper Berg River, submerged vegetation is fairly limited due to the regular occurrence of scouring flows during winter months (Harrison & Elsworth, 1958). A commonly occurring species, however, is the sedge *Scirpus digitatus*, which grows on hard substrata of the upper river, in a range of flow types. The palmiet reed, *Prionium serratum* (L.), also occurs in patches on the streambed of the upper river. The streambed comprises small to large cobble, with patches of white, quartzitic sand and gravel. The water of the upper reaches of the Berg River is 'white', and normally clear. White waters are characteristic of rivers rising on the north-facing slopes of inland mountains in the Cape (Midgley & Schafer, 1992). The observed mean annual runoff (MAR) from the sub-catchment above the study sites (Figure 4.1; Section 4.4) is 152.3 x 10^6 m^3 yr^{-1}.

Over the last few decades the Berg River has been well-studied. Thus, like the Great Fish River, the Berg River fortunately also has a detailed historical data base (a rarity in southern Africa). Harrison & Elsworth (1958) reported on the chemical and biological status of the Berg River as it was in 1951, while Scott (1958) has provided information on the dipteran Chironomidae. Further, Dallas (1992; 1995) has worked on the Berg River over the period 1992-1997, looking at the riverine macroinvertebrate communities, and the link between macroinvertebrates and the water quality of the system.

The upper reaches of the Berg River have been described as being of good water quality, with a low pH, conductivity, Total Dissolved Solids (TDS) and Total Suspended Solids (TSS) (Bath, 1993). The faunal communities inhabiting these reaches of the river have been described by Harrison & Elsworth (1958) as the most diverse of the entire river. The main anthropogenic disturbances in the upper catchment are afforestation, agricultural runoff and trout farm effluent, and the transfer of water from Theewaterskloof (Dallas, 1992).

A comparison of invertebrate and water quality data collected in 1951 by Harrison & Elsworth (1958), and again in 1991 by Dallas (1992), led to the conclusion that the greatest changes have occurred in the Upper Foothill Stony-Run zone, which is where the IBT tunnel outlet is located (Figure 4.1). Upstream of the IBT, groups such as the dipteran

Chironomidae and the Oligochaeta were more common in 1991 than in 1951, while the Plecoptera, Simuliidae and a species of the ephemeropteran family, Baetidae, were less common during the more recent study (Dallas, 1992). It was surmised that this was due to marked increases in silt deposits and reductions in the occurrence of marginal vegetation, such as the palmiet reed. At a site lower down the river, below the IBT, Dallas (1992) found that the faunal community was dominated by the Chironomidae, followed by the Oligochaeta and Simuliidae. It is likely that this community was indicative of organic and nutrient enrichment from the Franschhoek tributary, which flows through the town of Franschhoek, or from the effluent from the trout farm further upstream. It was also suggested that the water transferred from Theewaterskloof could have had a negative impact on the faunal communities of the Berg.

4.2 Riviersonderend

The Riviersonderend is a tributary of the Breede River in the Western Cape (Figure 4.1). The river flows eastwards from its source at an altitude of approximately 1590mAMSL in the Groot Drakenstein Mountains. For the first 15km it flows through the Riviersonderend Gorge within the Hottentots Holland Nature Reserve and State Forest (Figure 4.1). The river flows out of the mountains into the Theewaterskloof Reservoir at approximately 300mAMSL. The Riviersonderend catchment area above the reservoir is 497 km^2, with a virgin MAR of 291 x 10^6 m^3 yr^{-1}, measured 2 km downstream of the current dam (Ractliffe *et al.*, 1996). Further east, the Riviersonderend flows past the town of the same name before joining the Breede River near Swellendam, approximately 100km below Theewaterskloof Dam. The Riviersonderend Mountains run parallel and north of the river.

Like the Groot Drakenstein Mountains, the Riviersonderend Mountains are of the Cape Fold series, while Bokkeveld Group shales, also of the Cape Supergroup, predominate along the length of the river once it leaves Theewaterskloof (Lambrechts, 1979). The vegetation reflects these changes in the geology of the catchment. Along the upper reaches of the river, mesic and dry fynbos predominate, while the wide valleys east of Theewaterskloof support patches of renosterveld, which is characteristic of the Bokkeveld shales. The valleys have been extensively cleared for agriculture, however, and farmland encroaches on the river channel in many areas. The riparian zone is heavily infested by non-indigenous tree species

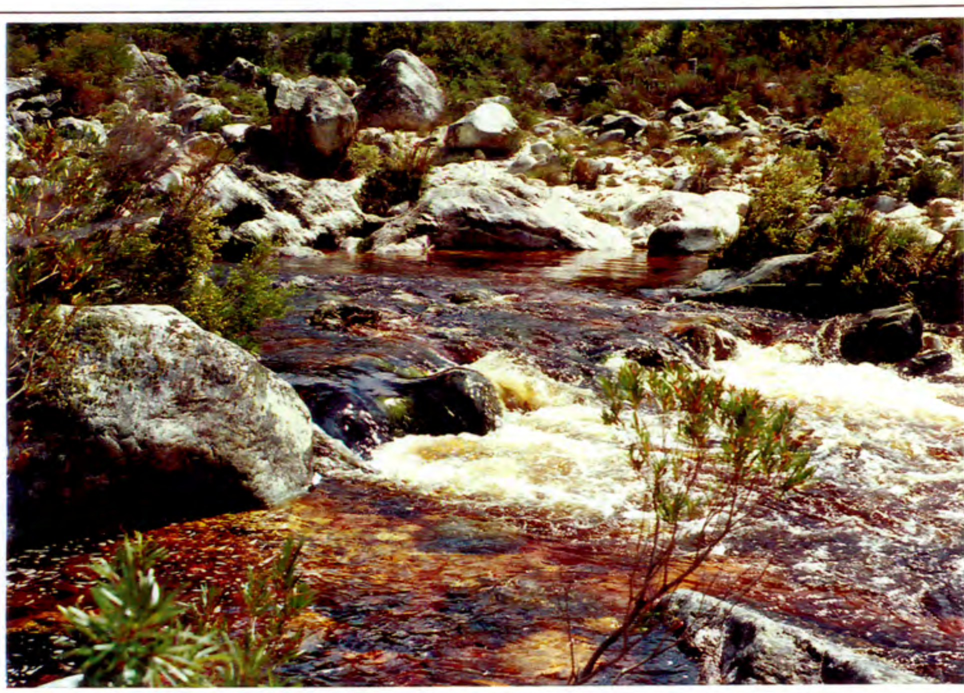


Plate 5: The photograph was taken in early winter at the lower Riviersonderend sampling site, RSE2. The riffle in the photograph was sampled on each occasion, and water quality samples were taken immediately upstream of the riffle.

such as *Acacia mearnsii*, and, in many places, is choked by the indigenous palmiet reed, *P. serratum*. Winter high-flows are not released from Theewaterskloof and, thus, scouring flows cannot keep the growth of the reed in check (Ractliffe *et al.*, 1996).

The water quality of the upper reaches of the Riviersonderend is very pure, and the waters are darkly stained by humic acids, which is typical of rivers which drain south-facing, fynbos-covered slopes. Humic acids originate as polyphenols, which are secondary plant compounds found in fynbos plants (Midgley & Schafer, 1992; Davies & Day, 1998). The polyphenols are leached into the soils of the catchment, as a result of the death and decay of the vegetation. These compounds are transformed into humic acids, which are then washed into the river. The Riviersonderend was especially darkly stained in winter, when the increased runoff carried large quantities of humic acids into the river.

4.3 Riviersonderend-Berg-Eerste River Government Water Scheme (RBEGS)

The Riviersonderend-Berg-Eerste River Government Water Scheme (RBEGS) forms part of the water supply network to the Cape Metropolitan Area (CMA), providing 38-45% of the total water supply (Clayton, 1994, 1996; see also Table 3.4, Chapter 3). The floodwaters of the upper reaches of the Riviersonderend are regulated and stored in Theewaterskloof Reservoir, which has a capacity of $482 \times 10^6\,m^3$ (Bath, 1993; Figure 4.1; Plate 3). Three other rivers feed Theewaterskloof, namely the Du Toits, Elands and Waterkloof rivers. From Theewaterskloof, water is transferred (maximum $30\,m^3\,s^{-1}$) in a 12km-long tunnel through the Franschhoek Mountains to the upper Berg River (Figure 4.1; Plates 1 and 2). At this point, a siphon with a maximum discharge capacity of approximately $5\,m^3\,s^{-1}$ releases water almost continuously (and occasionally very variably) from November through to the end of May each year (Figure 4.2), with an average discharge of *ca* $3.5m^3\,s^{-1}$ in mid-summer (January through March) for irrigation purposes and for exotic rainbow trout (*Oncorhyncus mykiss*) farming in the Berg catchment. As mentioned above, the summer season is the natural period of low-flow in rivers in the Western Cape. The transfer tunnel continues a further 13km westwards to a balancing dam (Kleinplaas Dam: capacity $3.77 \times 10^5\,m^3$) on the Eerste River, and then 5km through the Stellenbosch Mountain to Blackheath Purification

Works (Figure 4.1), from where it is piped to Cape Town and environs. There are outlets from the RBEGS tunnel to supply water along this route to the Eerste River catchment. Two additional small abstractions are made to the IBT from rivers between the Berg River outlet and the Kleinplaas Dam (Wolwekloof and Banhoek rivers), while a much larger abstraction from Voëlvlei Reservoir, lower down on the Berg River, to Theewaterskloof regularly occurs such that Voëlvlei, an off-channel reservoir, is drawn down in order to keep the levels in Theewaterskloof relatively high.

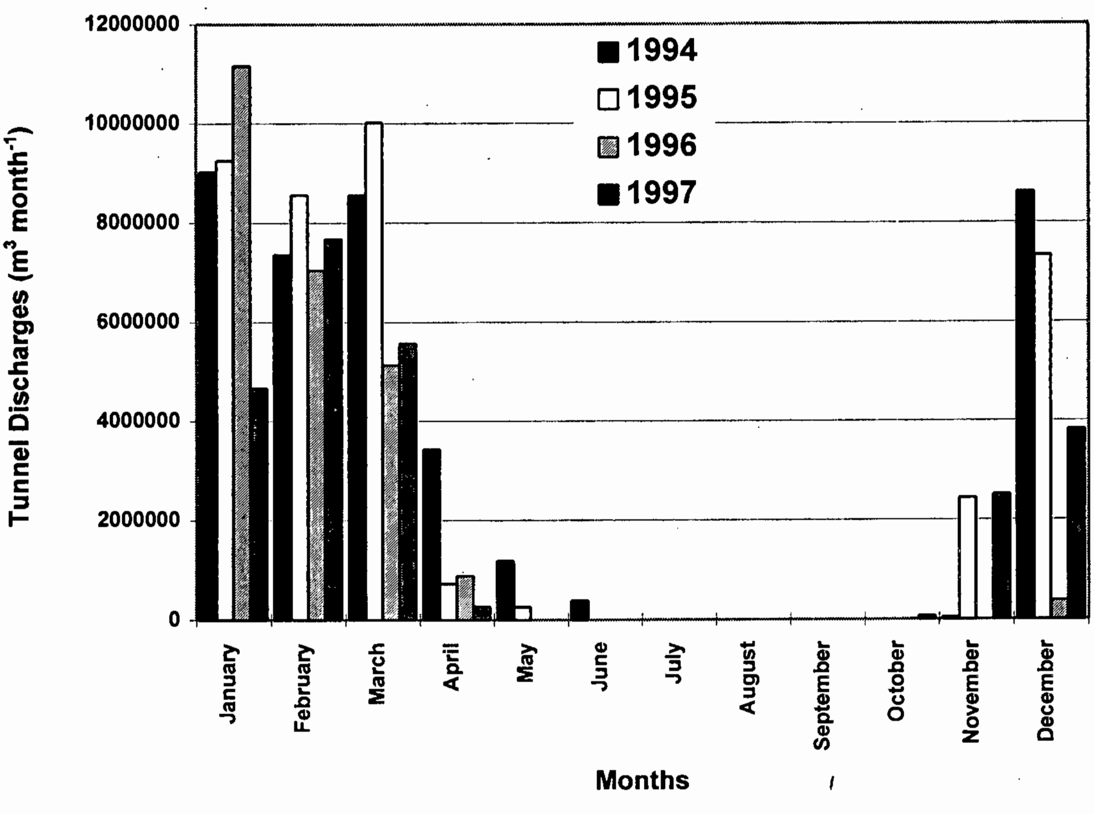


Figure 4.2 Water transfers through the RBEGS tunnel into the upper Berg River are seasonal, with releases reaching a maximum in mid-summer - January to March - and ceasing during winter.

Construction work on the RBEGS started in 1970, and water was first transferred to the upper Berg River in November 1980 (Department of Water Affairs, 1986b). Originally a dam, the Assegaaibos Dam, was to have been built above the IBT outlet on the upper Berg River to store winter floodwaters of the Berg for later diversion to Theewaterskloof: the direction of water between Theewaterskloof and the Berg River could then have become reversible between seasons. This dam has, however, not been built, and water has never been transferred back to Theewaterskloof. However, the Skuifraam Dam is due to be built on the upper Berg River in the La Motte Forest Reserve in 1998, in order to supply water to the Cape Metropolitan Area (Ninham Shand Consulting Engineers, 1994). The new impoundment will drown the IBT outlet, but reversal of water transfer is envisaged through the IBT tunnel, from Skuifraam to Theewaterskloof. Farmers in the upper Berg River catchment will thus be provided with water from Skuifraam Dam, rather than from Theewaterskloof.

4.4 Site Selection

In order to investigate the overall ecological effects of the RBEGS IBT, five sampling sites were selected, three on the upper Berg River (recipient system) and two on the Riviersonderend (donor) (Figure 4.1).

Two of the upper Berg River sites (BR1 and BR2) were situated respectively 500m and 200m upstream of the siphon (Figure 4.1, Plate 4), both above a small weir, while the third site (BR3) was selected approximately 100m below the siphon (Figure 4.1, Plate 2). This asymmetrical design is based on the advice of Underwood (1993), who suggested that the choice of several unimpacted control sites and a single impacted site would solve the problem of a lack of spatial replication in riverine systems (e.g. Hurlbert, 1984; Stewart-Oaten *et al.*, 1986). This is an extension of the Before-After, Control-Impact (BACI) design of Stewart-Oaten *et al.* (1986), which is confounded by the fact that natural spatial and temporal variation in biotic assemblages will affect the interpretation of data. In order to isolate and assess the effects of the IBT on the invertebrate assemblages of the Berg, variations in abundance between the control sites (BR1 and BR2) could be compared with variations between the control and impacted (BR3) sites.

All three sites lie within the Upper-Foothill-Stony-Run-Zone of Harrison & Elsworth (1958), which is characterised by a gentle gradient and a bed that comprises long runs and riffles, interspersed with pools (Plate 4). Samples at each site were consistently taken from the riffle or stones-in-current biotope (see Chutter, 1968, 1972). The invertebrate assemblages from this biotope have been well-documented for the Berg River (e.g. Harrison & Elsworth, 1958;

Scott, 1958), and are easily sampled. Furthermore, the riffle biotope, with its characteristically complex cobble substratum, generally supports the greatest diversity of lotic macroinvertebrates (e.g. Dallas, 1992; Downes *et al.*, 1995; Wohl *et al.*, 1995; Angradi, 1996).

Although sites BR1 and BR2 are relatively undisturbed, their riparian zones have been invaded by exotic acacias, while the upper catchment is extensively afforested with exotic pines. Nonetheless, the stream bed and banks at these sites are stable and the water is of exceptional quality. Below the siphon (BR3), both the bed and banks have been affected by the construction works for the tunnel, and by summer releases of water, which occur under high pressure, displacing cobble, and resulting in bed instability. The northern bank of the river at this site comprises granite rubble, which supports pine and acacia trees, while the southern bank comprises bedrock, vegetated by acacia trees and some indigenous riparian species.

Two sites were selected on the Riviersonderend (RSE1 and RSE2), above Theewaterskloof, in the foothill zone (Figure 4.1; Plate 5). Like the Berg River, the river is characterised within this zone by riffle-run sequences with pools, and mixed substratum dominated by cobble and boulders of various sizes (Ractliffe *et al.*, 1996). The sites were situated approximately 100 and 200m above a small DWAF weir, which no longer functions as a gauging weir. Both sites fall within the Hottentots Holland State Forest, but are immediately upstream of the extent of the pine forest plantations. The riparian vegetation is dominated by fynbos shrubs with a few tree species, such as *Meterosideros stellatifolium*. The upper Riviersonderend is relatively pristine, with the only human disturbances being forestry and hiking activities. Very little work has been done on this river system.

Chapter 5 The ecological effects of the Riviersonderend-Berg transfer scheme

5.1 Introduction

The review of the literature on IBTs presented in Chapter 2 revealed a paucity of data on the ecological effects of these water schemes. Many authors have warned of the consequences of this information gap, at a time when the planning and construction of IBTs appear to be on the increase (e.g. Boon, 1988; Petitjean & Davies, 1988; Davies *et al.*, 1992; Meador, 1992; Snaddon *et al.*, 1998). The following chapter describes the collection and analysis of data from sites on the Riviersonderend and Berg rivers, described in Chapter 4 (also see Figure 4.1). The main objective of the study was to explore the effects of the Riviersonderend-Berg IBT, a part of the Riviersonderend-Berg-Eerste River Government Water Scheme (RBEGS), on the physico-chemical attributes and macroinvertebrate community structure of the receiving reaches of the upper Berg River. The physical, chemical and biotic components of the neighbouring Riviersonderend and Berg River catchments were also compared, in order to assess whether there were marked differences between the two catchments.

5.2 Methods

5.2.1 Data Collection

Sampling Frequency

All sites (Figure 4.1; Section 4.4) were visited on a bi-monthly basis, and the sampling was completed within two days for all sites on all occasions. The sampling period extended from March 1994 to April 1995, inclusive. During the months of March, May and November 1994, and January and April 1995, the IBT released water into the Berg River, and these months were termed "summer" months, while July and October 1994 were "winter" months, during which the transfer ceased (see also Figure 4.2) .

Physical Attributes and Water Chemistry

Physical variables measured at each site were discharge, depth profile, stream width and water temperature. Flow was recorded using an OTT Flow Meter during March and May, 1994, while for the remainder of the sampling period, a more sophisticated Price AA Current Meter equipped with a graded wading rod was used. Discharge was calculated using specified propellerconversion equations to convert multiple flow/depth measurements along channel profiles of known width at each site. Stream width was measured using a tape measure. Discharge, depth and stream width were measured at only one site above the IBT (BR1), and once at every other site in both rivers. Temperature was measured at all sites using an alcohol thermometer, accurate to within 0.5°C.

Water chemistry variables measured at each site were pH, conductivity, Total Dissolved and Suspended Solids, and a variety of anions and cations. pH and conductivity were measured using Crison meters, model 506 for pH, and model CDTM 523 for conductivity, and three measurements were taken at each site. Total Dissolved Solids (TDS) was measured through the evaporation at approximately 70°C of 800ml of pre-filtered river water in pre-weighed pyrex glass beakers. Beakers were weighed on a Sartorius precision laboratory balance, accurate to 1mg, and TDS was recorded in mg l^{-1}. Total Suspended Solids (TSS) was determined by filtering a known volume of river water *in situ*, at each sample site, through a pre-weighed Whatman GF/F filter, which was pre-combusted at 500°C for 3 hours. The filter papers were then dried at approximately 60°C for at least 48 hours and weighed, then combusted at 500°C for 3 hours and re-weighed in order to determine the organic TSS. Filter papers were weighed on a Mettler AE100 laboratory balance, accurate to 0.1mg, and measurements were recorded in mg l^{-1}. Filtered water was stored in polythene vials that were pre-washed in 5% Contrad (phosphate-free surface active agent) solution, and returned to the laboratory where they were frozen until analysis for various cations and anions. Anions (chloride, sulphate, phosphate and nitrate) and cations (calcium, magnesium, sodium and potassium) were measured on an HPIC ion exchange column, with an accuracy up to 0.005mg l^{-1}. Analyses were performed by the Department of Geology, UCT.

Invertebrates of the stones-in-current biotope

Three randomly-selected replicate benthos samples were taken from the stones-in-current biotope (Chutter, 1968) at each site on each visit, using a box sampler with an area of 0.1m^2. The box sampler was placed on the stream bed and all moveable cobble and pebbles were gently scrubbed to remove the invertebrates, while gravel and sand was agitated, down to an approximate depth of 15cm. Animals were retained on a detachable 80µm-mesh sieve and were preserved in the field with 4% buffered formaldehyde solution which was later replaced by 70% alcohol in the laboratory, within 7 days after collection. Laboratory treatment included fractionation into four size categories: >950µm fraction, between 500µm and 950µm, between 250µm and 500µm, and a fraction between 80µm and 250µm, which was curated but not analysed. Some fractions were subsampled using a standard plankton-splitter. Invertebrates were counted and identified to family using Usinger (1956), Smith (1969), McCafferty & Provonsha (1981), Merritt & Cummins (1984), Scott (1985) and Davies & Day (1998).

The decision to identify invertebrates only to the level of family was based primarily on the temporal and spatial scales of the project. The objectives of this project were to compare the invertebrate communities in the donor and recipient rivers of the RBEGS, and further, to investigate the responses of the invertebrate communities of the Berg River to an IBT. The effects on individual invertebrates or species were not considered and thus it was deemed unnecessary, at this stage, to concentrate on the taxonomy of the individuals that were collected from the two rivers, and rather to gather more information at the community level. The current state of aquatic invertebrate taxonomic knowledge in South Africa is such that the time and skills required for identification of individuals to species were not available for this project. More detailed taxonomic results are envisaged within the scope of the WRC research contract period, as it is accepted that investigation of the effects of anthropogenic disturbances at the invertebrate species level provides more refined assessment of such effects than family-level analyses (e.g. Ractliffe, 1992; de Moor, 1997; de Moor *et al.*, 1997).

5.2.2 Data Analyses

Physical Attributes and Water Chemistry

The physical attributes and water chemistry of the Riviersonderend and Berg River were subjected to standard statistical analyses, to test for significant differences between measurements made at different sites and on different occasions. For each group of data, a paired t-test was performed on the assumption that all measurements of variables at different sites on the same sampling occasion will not be independent (Zar, 1984; Kranzler & Moursund, 1995). Furthermore, due to the asymmetrical design of the sampling programme in the Berg River, where two unimpacted sites and one impacted site were sampled (Section 4.4), t-tests were performed between unimpacted sites, and again between one unimpacted site (BR1) and the impacted site (BR3). This was done in order to determine whether significant differences between the impacted and unimpacted sites were due to the IBT, and not simply due to natural fluctuations in measured variables. Furthermore, the chemical data from the two rivers were compared, in order to assess whether or not the two catchments differed significantly in terms of water quality.

Invertebrates of the stones-in-current biotope

Most of the analyses performed on the invertebrate data made use of multivariate techniques. This was due to the fact that univariate methods require that the data are collapsed into single measures, such as average abundances, taxon richness, or diversity indices. Hence, much of the complexity of community structure is lost for such analyses, while multivariate statistics have been developed to assess and take account of this complexity (Field *et al.*, 1982; Clarke & Warwick, 1994). Methodologies for analyses were taken largely from the manual prepared by Clarke & Warwick (1994) of the Plymouth Marine Laboratory, in which all of the techniques used here are described in greater detail.

Univariate Analyses Standard statistical techniques were utilised to generate measures of total abundances and taxon richness for all invertebrate samples from both rivers. The proportional abundances of some Berg River taxa were determined, in order to demonstrate changes in dominance of some invertebrate groups, as a result of the transfer of water into the catchment.

Furthermore, a variety of diversity indices was generated that commonly are used in ecological studies to represent various attributes of community structure. The value of diversity indices lies in their ability to measure the way in which the species assemblage is divided up between taxa. There are two components to this measure of diversity: taxon richness and equitability. An index which combines these components, and which was used here, is the Shannon-Wiener diversity index (H'):

$$H' = -\sum_i p_i (\log p_i)$$

where p_i is the proportion of the total abundance contributed by the ith taxon. Log_e was used for this study. Lastly, taxon richness was calculated, as a measure of the total number of taxa in the communities. Taxon richness be expressed as the total number of taxa (as above), or as an index, which incorporates the total number of individuals. The index that was used for this study was Margalef's index (d):

$$d = (S - 1) / \log N$$

where S is the total number of taxa, and N is the total number of individuals, using $\log_e$. All of the above indices were calculated using the statistical package PRIMER (Plymouth Routines in Multivariate Ecological Research: Clarke & Warwick, 1994). Statistical differences between all of the above indices were tested between various groups of samples, by using a simple t-test for differences between sample means.

Multivariate Analyses The statistical package PRIMER was used to perform a sequence of multivariate statistical routines on the data. The approach upon which PRIMER is based is one where the biotic data are analysed first, in order to search for patterns in distribution and abundance, and, secondly, the biotic patterns are interpreted in terms of the environmental data (such as physico-chemical variables) (Field *et al.*, 1982). In this way, biotic and environmental data analyses are kept separate, in an attempt to avoid the influences of previously assumed relationships between the data. PRIMER is of particular use for biological studies at the community level (Clarke & Warwick, 1994), where taxa-by-samples matrices are fairly large, and where communities comprise a few taxa in large numbers accompanied by a diversity of taxa represented only by a few individuals. In this study, for example, the riverine invertebrate communities often comprised a few dominant families, such as chironomid and simuliid dipterans, and baetid ephemeropterans, while most of the other families were represented by small numbers of individuals. The equations used by PRIMER have specifically been chosen to deal with these situations.

Testing for Significance Firstly, it was necessary to test for statistically significant differences between various groups of samples. The nature of the dataset of this study presented a difficulty for significance testing, in that the data were clearly not univariate. If they were univariate, simple analysis of variance (ANOVA) would have sufficed. This would have been the case if average taxon richness values across all taxa were compared between sites. As mentioned above, however, the calculation of average richness across all taxa would have led to the loss of many of the patterns contributing to community differences between groups of samples. Hence there was a need to test for significant differences in the multivariate structure between groups.

For these reasons, this study made use of the ANOSIM routine of PRIMER, to test for significant differences between groups of samples. ANOSIM (analysis of similarities) is a test based on the ranked similarities between samples, which itself underlies the ordinations produced by MDS. The null hypothesis was that there were no differences between sites. The test statistic, R, was calculated from the rank similarities of the original similarity matrix, and is defined as:

$$R = (\bar{r}_B - \bar{r}_W) / (M / 2)$$

where r_B is the average of rank similarities arising from all pairs of all pairs of replicate samples between sites, r_W is the average of all rank similarities among replicate samples within sites, and where $M = n(n\text{-}1)/2$ and n is the total number of samples in the analysis. The R-statistic can range from -1 to 1, and will equal 1 if all replicate samples within sites are more similar to each other than any replicates from different sites, and will approximate zero when there are, in fact, no differences between samples, and the null hypothesis is true.

For this study, differences between various combinations of groups of samples, based on sampling sequence (i.e. *a priori*), not on groups determined through cluster analysis (*a posteriori*), were tested for significance. The two-way nested ANOSIM was chosen for this dataset, as the sampling design was considered to be hierarchical, with three sites, two unimpacted and one impacted, which were sampled on seven different occasions, and which contained replicate macroinvertebrate samples.

ANOSIM was used to perform a global test, testing for significant differences across all groups, and also to generate statistics for paired, two-way comparisons. In this way, two null hypotheses were tested: H_1, no differences between months, and H_2 no differences between sites. Subsequently, pairs of samples which contributed significantly to the separation of groups of samples, were identified. The ANOSIM routine was performed on four subsets of the invertebrate community data: (1) summer samples taken at BR1, BR2 and BR3, (2) winter samples taken at BR1, BR2 and BR3, (3) winter and summer samples at RSE1 and RSE2, and (4) winter and summer samples from BR1, BR2, RSE1 and RSE2. In all cases, the critical significance level was taken as 5%. The Berg River subsets for tests (1) and (2) were divided into summer and winter samples, in order to test specifically for the effects of the IBT.

Multivariate representation of the data Two multivariate methods were used in this study in order to develop graphic representations of the invertebrate community data. These were (1) cluster analysis, and (2) ordination.

1. Cluster Analysis: The component routine of PRIMER that generates a hierarchically clustered representation of biological community data, is CLUSTER. This routine computed similarity coefficients between each pair of samples, and generated triangular matrices containing the coefficients. The similarity coefficients measured the differences between the abundances of taxa between each sample, which were then averaged over all taxa. In this way, abundance values were compared between control sites and between control and impacted sites (see Section 4.4). The coefficient used here was the Bray-Curtis Similarity Index (Bray & Curtis, 1957) which has been reported to be appropriate for complex ecological data (Clarke *et al.*, 1993), and has widely been used in marine ecosystem analyses (e.g. Field *et al.*, 1982; Warwick *et al.*, 1990). The Bray-Curtis similarity between the abundances of taxa j and k across all s=1,...,n columns (samples) is calculated as:

$$S_{jk} = 1 - \delta_{jk}$$

where

$$\delta jk = \frac{\sum_{i=1}^{S} | Y_{ij} - Y_{ik} |}{\sum_{i=1}^{S} (Y_{ij} + Y_{ik})}$$

and Y_{ij} is the abundance of taxon i in the jth sample, Y_{ik} is the abundance of taxon i in the kth sample, and where δ_{jk} can vary from 0 (identical abundances for all taxa) to 1 (no species in common). The data were log(x+1)-transformed in order to ensure that dominant families alone did not determine the multivariate ordinations (e.g. Clarke *et al.*, 1993), while avoiding problems with zero entries.

The similarity matrix described above was used to find clusters of samples where samples were grouped together based on their calculated similarities. The cluster analysis method used by PRIMER for this study was the group-average linking method, where samples were grouped together based on the average similarities between successively larger groups. The graphic product of the hierarchical clustering routine was a dendrogram, which displayed groups of samples, but not the inter-relationships between them. The latter was done through ordination.

2. Ordination: The starting point for multi-dimensional scaling was the similarity matrices generated above, but using rank (relative) similarities, rather than absolute values. The final ordination map was in two dimensions, where the placement of samples reflected similarities between the biological communities. There are several ordination techniques, and the one used here was non-metric Multi-dimensional Scaling, performed by the PRIMER routines, MDS, and CONPLOT, which plotted the final ordination map. The MDS algorithm is an iterative process, whereby the placement of samples on the ordination is progressively refined, in order to satisfy the original similarity matrix. The principle of the MDS algorithm is to minimise the "stress" or distortion between the calculated similarity rankings and the final ordination. Stress increases with decreasing dimensionality, and with increasing quantity of data, and can be interpreted as follows:

- Stress < 0.05: indicates an excellent representation of the data
- 0.05 < Stress < 0.1: indicates a good ordination
- 0.1 < Stress < 0.2: indicates a potentially useful 2-dimensional plot, and
- 0.2 < Stress < 0.3: the points are close to an arbitrary placement.

Determining Discriminating Taxa A further useful component offered by the PRIMER package and used here, was the SIMPER routine. This technique allowed the determination of which taxa influenced the spread of samples on the dendrograms and ordinations. This was achieved by computing the average dissimilarity (δ) between all pairs of inter-group samples (i.e. group 1 samples were all paired with samples in group 2). This average was then broken down into separate contributions from each taxon.

Using the Bray-Curtis dissimilarity coefficient δ_{jk} described above, between two samples *j*, in group 1 and *k*, in group 2, the contribution from the *i*th taxon, $\delta_{jk}(i)$ is defined as the *i*th term in the following equation:

$$\delta_{jk}(i) = 100.\left|y_{ij} - y_{ik}\right| / \sum_{i=1}^{p}\left(y_{ij} + y_{ik}\right)$$

$\delta_{jk}(i)$ is consequently averaged over all pairs of groups, to determine the average contribution from each taxon to the dissimilarity between groups 1 and 2. Furthermore, the standard deviation of taxon i, SD(δ_i), of the $\delta_{jk}(i)$ values, gives an indication of how consistently taxon i contributes to the dissimilarities. A low SD(δ_i) indicates consistent contribution from taxon i, which would thus be a good discriminating taxon. The SIMPER routine was performed on **all** summer (release) data from the Berg River, and again on the same dataset **without** the transferred taxa (i.e. all zooplanktonic groups), in order to investigate which invertebrate taxa in the Berg River were responding to the IBT.

5.3 Results

5.3.1 Physical Attributes and Water Chemistry

Figures 5.1 to 5.10 illustrate the variations in physical attributes and water chemistry in the Berg River and Riviersonderend over the sampling period (see also Appendix 2). Looking first at the Berg River, the average depth of the river was greatest at the site above the IBT release tunnel (BR1) in July (Figure 5.1a). On either side of this peak, the depth decreased towards summer. None of the sites were significantly deeper than others (*summer*: $t=-1.057$, $p>0.1$; *winter*: $t=1.842$, $p>0.1$; two-tailed tests). In terms of stream width, BR1 was of a significantly narrower width than BR3 on all sampling occasions when the IBT was releasing water ($t=-5.572$;, $p<0.001$; one-tailed test), but was slightly wider during the winter, no-release months ($t=3.341$, $p>0.05$; one-tailed test) (Figure 5.1b). The upper Riviersonderend site (RSE1) was significantly deeper than the lower site (RSE2) ($t=3.133$, $p<0.05$; one-tailed test), while, in most months, the lower site was wider ($t=-2.26$, $p<0.05$; one-tailed test) (Figure 5.2).

The timing of releases of water from the IBT into the Berg River was reflected in the discharges measured in the river (Figure 5.3a). Discharges at BR3, below the IBT, were consistently higher than those measured above the IBT (BR1), but this difference was significant ($t=-3.787$, $p<0.01$; one-tailed test) when the IBT was releasing water into the river, and minimal during the winter months ($t=-1.504$, $p>0.05$; two-tailed test). Discharges measured in the Riviersonderend were not significantly different between sites ($t=1.34$, $p>0.05$; two-tailed test), while discharges tended to be slightly higher in the Riviersonderend than at BR1 in the Berg, with the exception of July 1994 (Figure 5.3b).

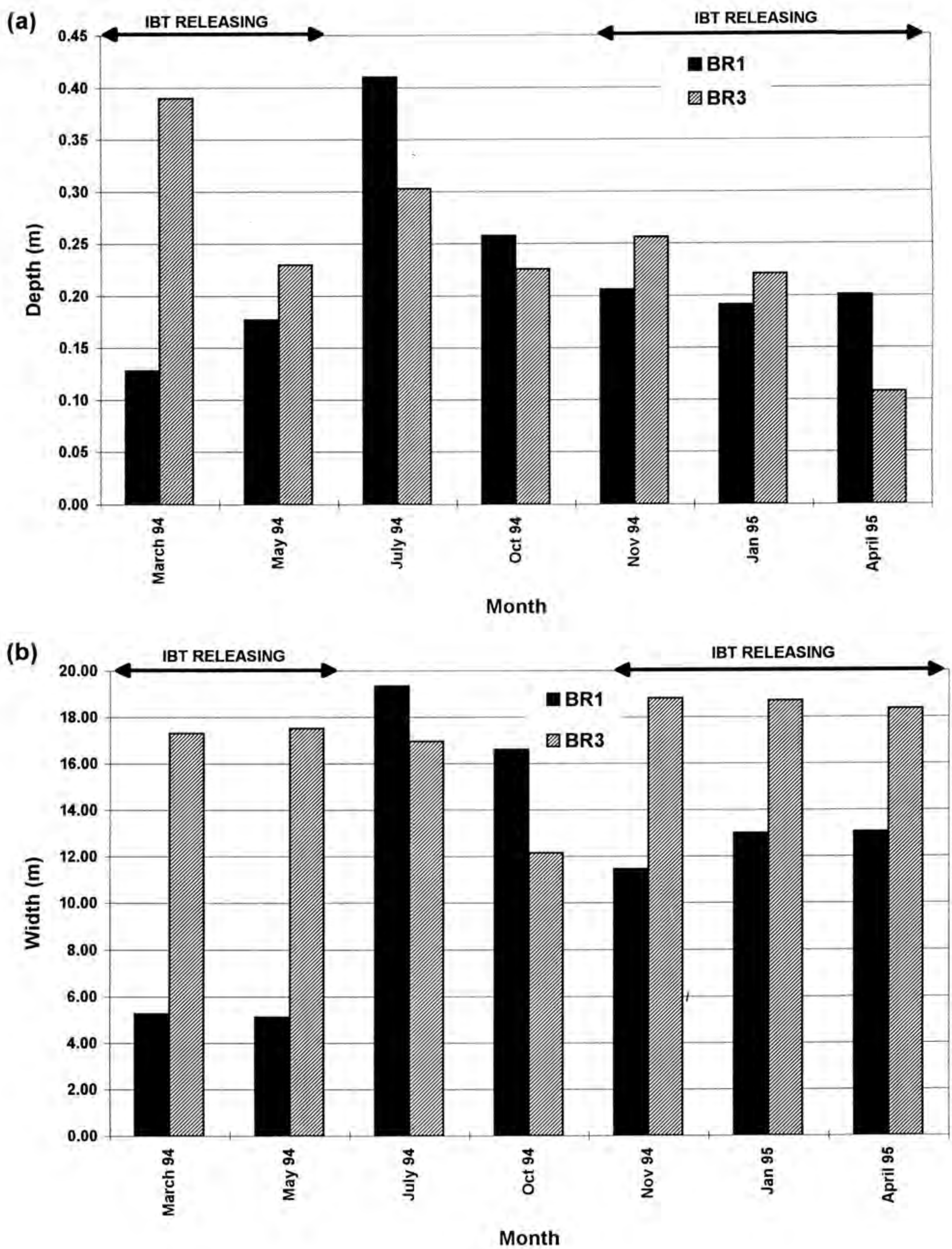


Figure 5.1 (a), Depth, and (b), width of two sampling sites on the Berg River, from March 1994 to April 1995. The arrows at the tops of the graphs indicate the period during which the IBT was releasing water into the river.

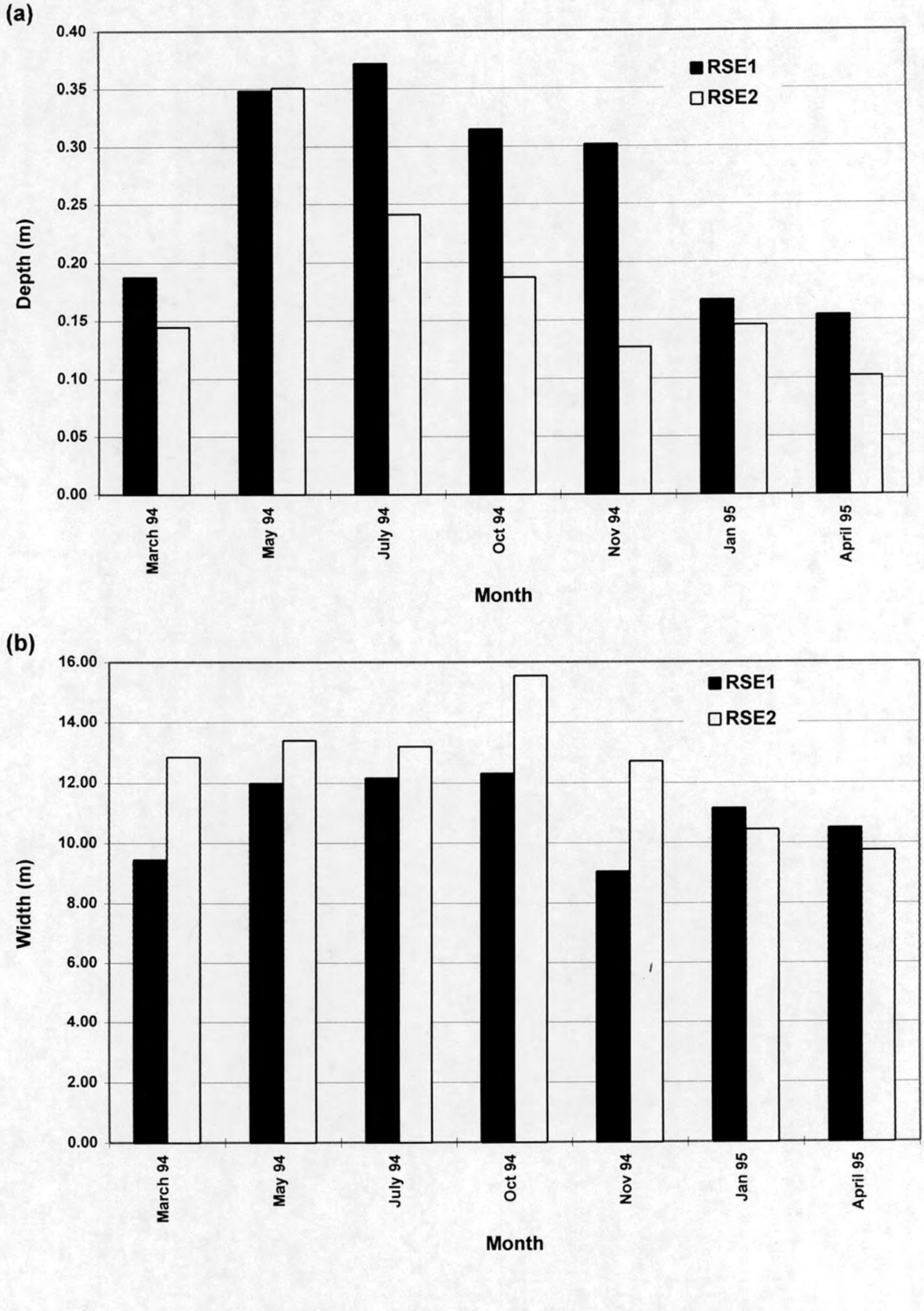


Figure 5.2 Bar graphs showing (a) depth, and (b) width of the two sampling sites on the Riviersonderend, from March 1994 to April 1995.

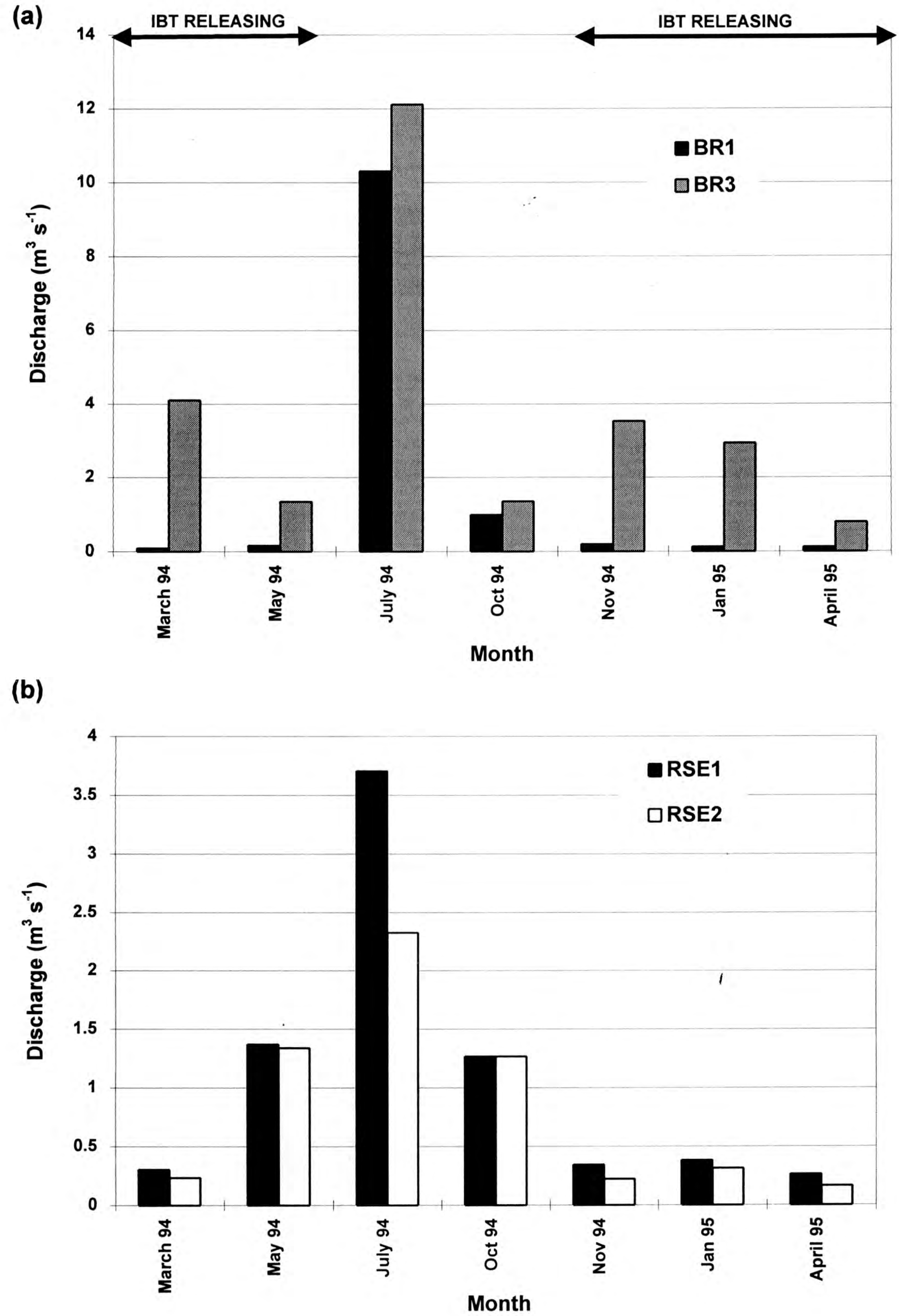


Figure 5.3 Discharges measured at two sampling sites on (a) the Berg and (b) the Riviersonderend rivers, from March 1994 to April 1995.

Sites BR1 and BR2 in the Berg, and RSE1 and RSE2 in the Riviersonderend were not significantly different to each other (0.05 significance level), in terms of temperature, pH, conductivity, TDS and TSS. Temperatures measured in the Berg River and Riviersonderend followed a distinctly seasonal pattern - highest in November/January and lowest in July (Figure 5.4). However, there were no significant differences between temperatures taken in summer at BR3, compared to BR1 ($t=0.862$, $p>0.05$; two-tailed test). Temperatures were significantly lower in the Riviersonderend than in the Berg ($t=-3.408$, $p<0.01$; one-tailed test). Both pH and conductivity were significantly higher in the Berg River at BR3 than above the IBT at BR1, during months when water was released (*pH*: $t=-6.785$, $p<0.01$; one-tailed test; *conductivity*: $t=-3.941$, $p<0.05$; one-tailed test) (Figures 5.5a and 5.6a). The Riviersonderend sites had a consistently lower pH and higher conductivity than the Berg River site above the IBT (*pH*: $t=-5.505$, $p<0.01$; *conductivity*: $t=11.302$, $p<0.01$; one-tailed tests), while there were no significant differences in these variables between RSE1 and RSE2 (Figures 5.5b and 5.6b). TDS and TSS were significantly higher in the Berg River below the IBT during summer (*TDS*: $t=-4.316$, $p<0.01$; *TSS*: $t=-8.329$, $p<0.001$; one-tailed tests), with no significant differences in winter (Figures 5.7a and 5.8a). Again, there were no differences in TDS and TSS between RSE1 and RSE2 (Figures 5.7b and 5.8b;). TDS was significantly higher in the Riviersonderend than in the Berg ($t=3.822$, $p<0.01$; one-tailed test), while TSS was similar between the two rivers.

In summer in the Berg River, the concentrations of all cations measured at BR3 were significantly greater than those at BR1 (*Na*: $t=-3.925$, $p<0.05$; *K*: $t=-4.761$, $p<0.05$; *Mg*: -6.443, $p<0.05$; *Ca*: -6.571, $p<0.05$; one-tailed tests) (Figure 5.9a). This was also the case for chloride and sulphate anions (*Cl*: $t=-3.033$, $p<0.05$; SO_4: -3.998, $p<0.05$; one-tailed tests) (Figure 5.10a). On the other hand, ionic concentrations at BR1 and BR2 were similar, with the exception of calcium, which was significantly higher at BR2 (t=-2-215, $p<0.05$; one-tailed test). There were no significant differences between measurements made at any of the Berg River sites in winter, although sodium and potassium concentrations were consistently higher at BR3 than at the two sites above the IBT. Similarly, there were no significant differences between ionic concentrations measured at RSE1 and RSE2, with the exception of potassium ions which were significantly higher at RSE2 ($t=2.884$, $p<0.05$; one-tailed test) (Figures 5.9b and 5.10b). In terms of cation concentrations, the Riviersonderend had significantly higher concentrations of sodium, magnesium and calcium ions than the Berg River (*Na*: $t=-4.182$, $p<0.01$; *Mg*: $t=-5.479$, p<0.01; *Ca*: t=-4.977, p<0.01; one-tailed tests), while measurements of the remaining ions (potassium, chloride, phosphate and sulphate) and were similar between the two rivers.

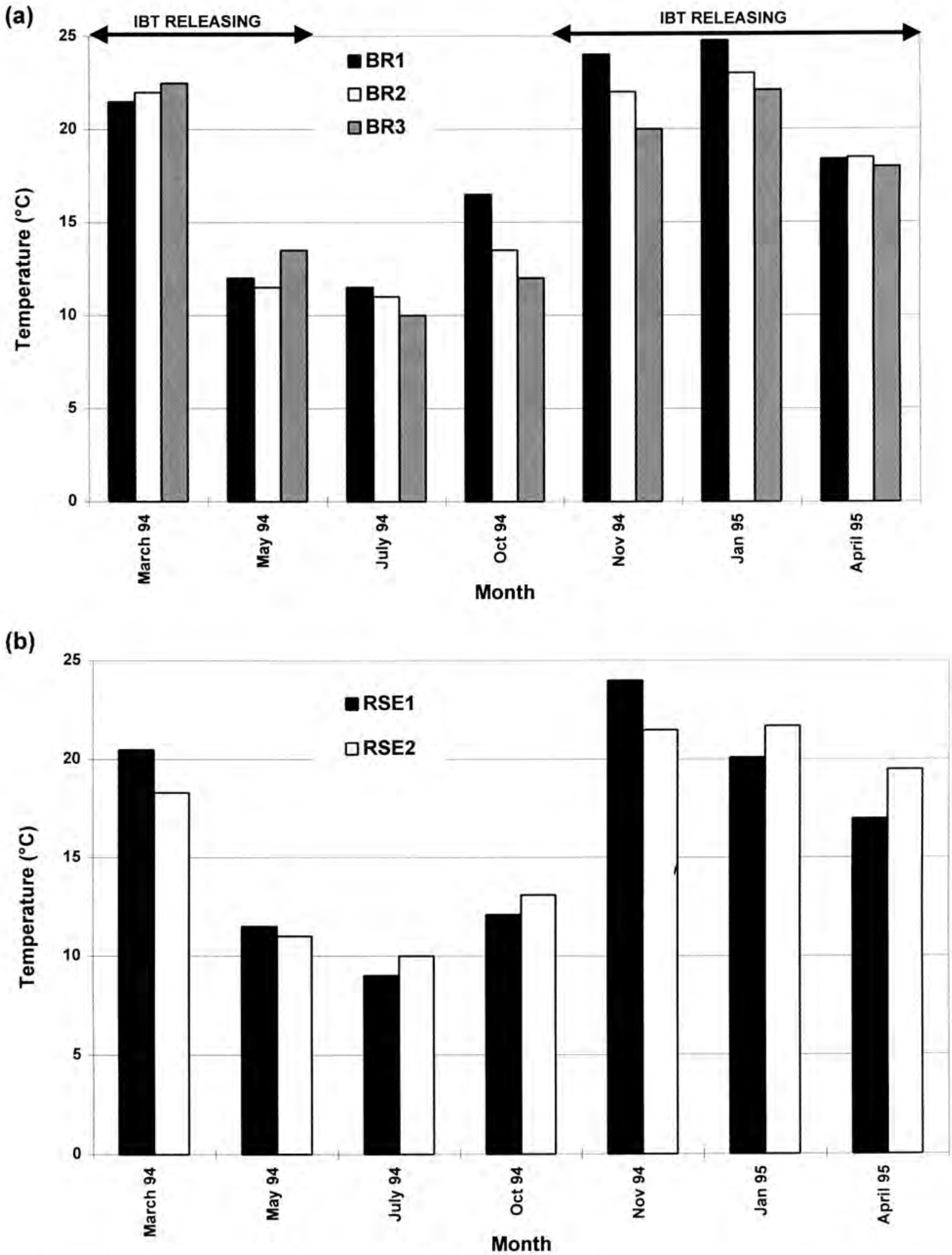


Figure 5.4 Temperatures measured at all the sampling sites on (a) the Berg and (b) the Riviersonderend rivers, from March 1994 to April 1995.

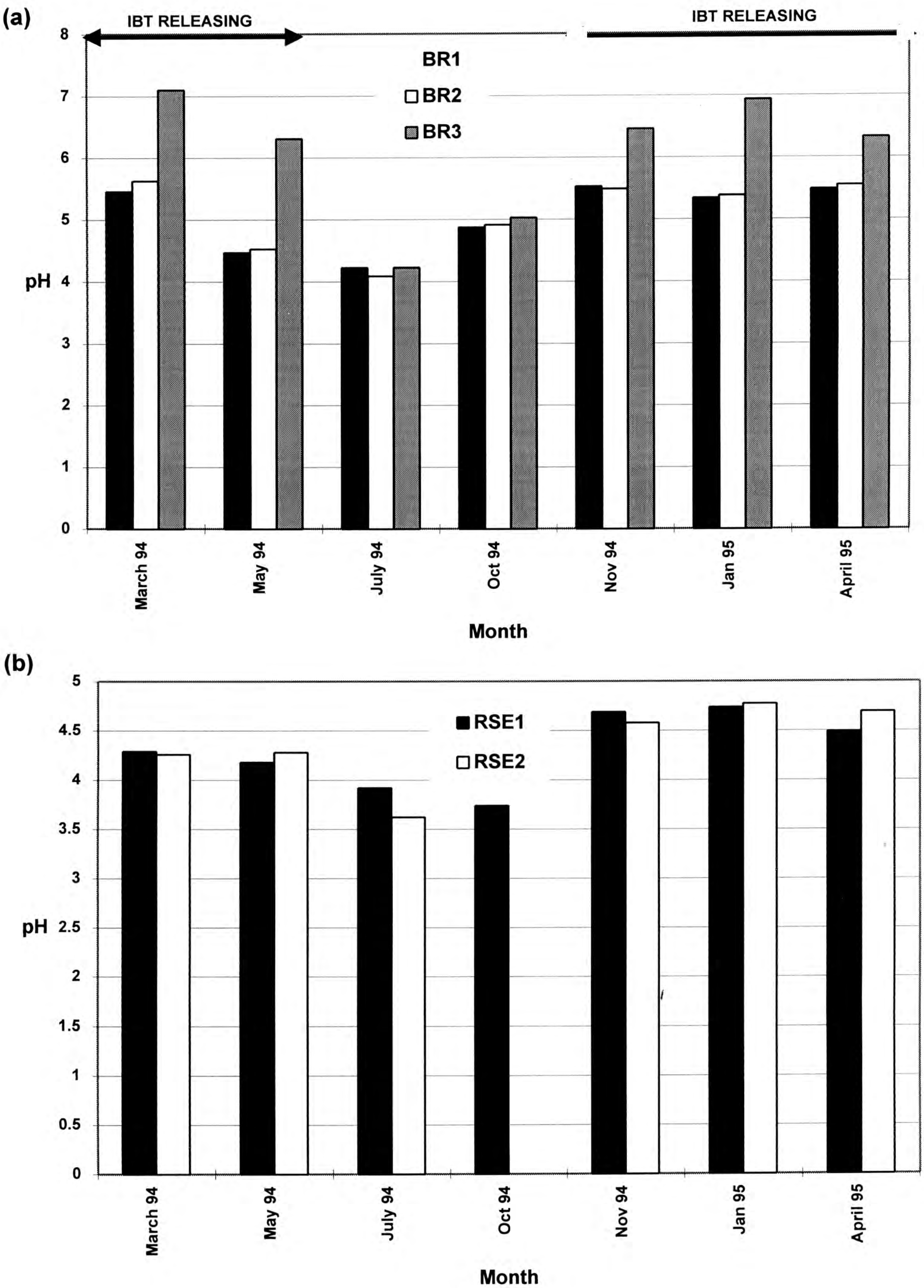


Figure 5.5 pH measurements at all the sampling sites on (a) the Berg and (b) the Riviersonderend rivers, from March 1994 to April 1995.

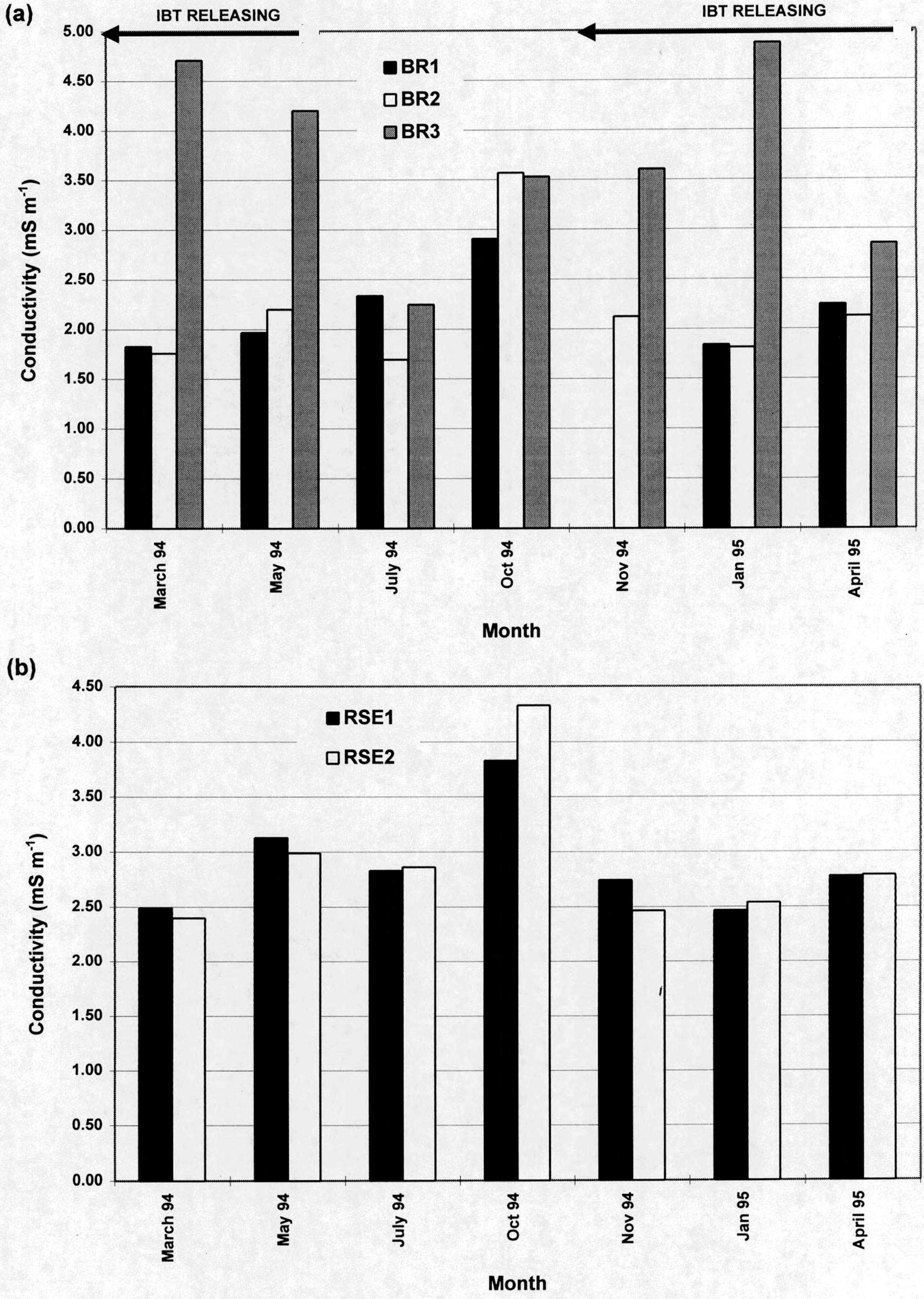
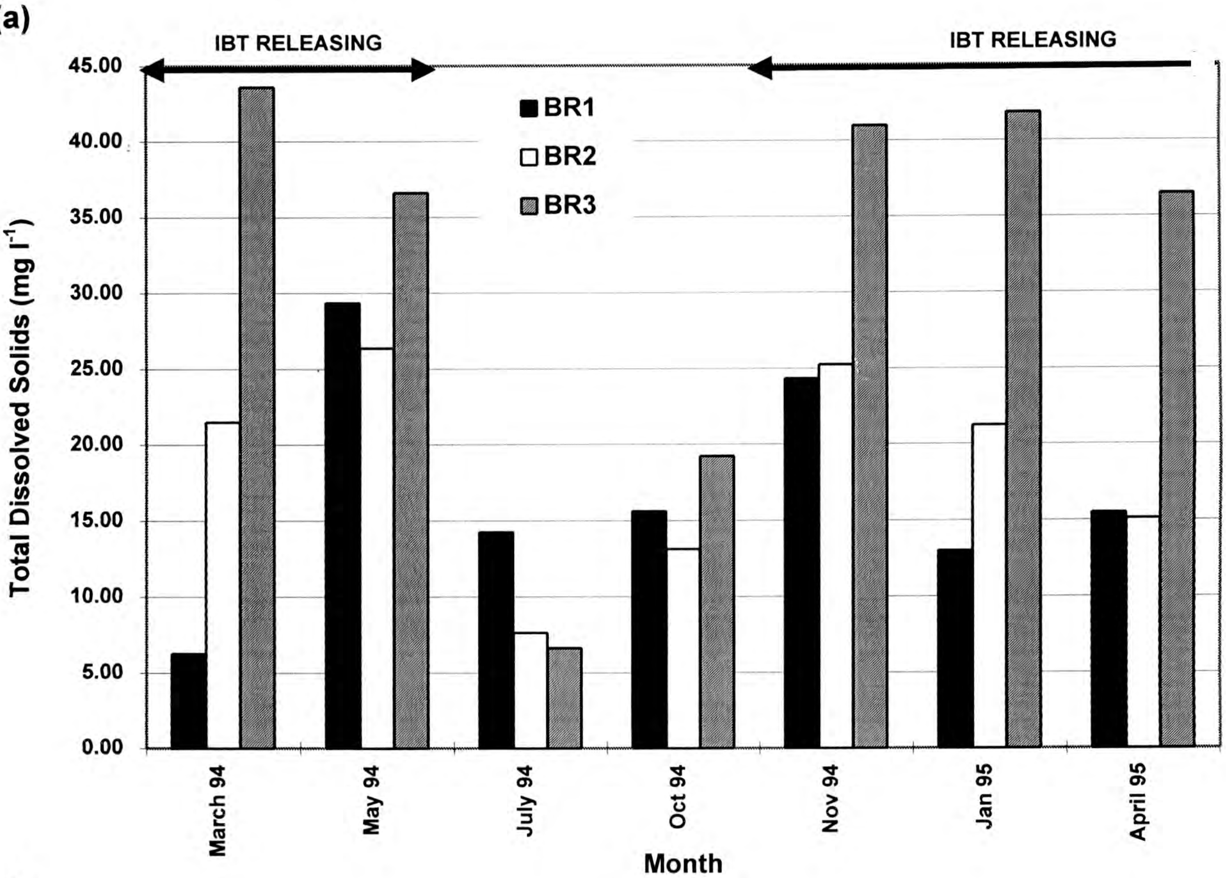


Figure 5.6 Conductivity measurements at all the sampling sites on (a) the Berg and (b) the Riviersonderend rivers, from March 1994 to April 1995.

(a)



(b)

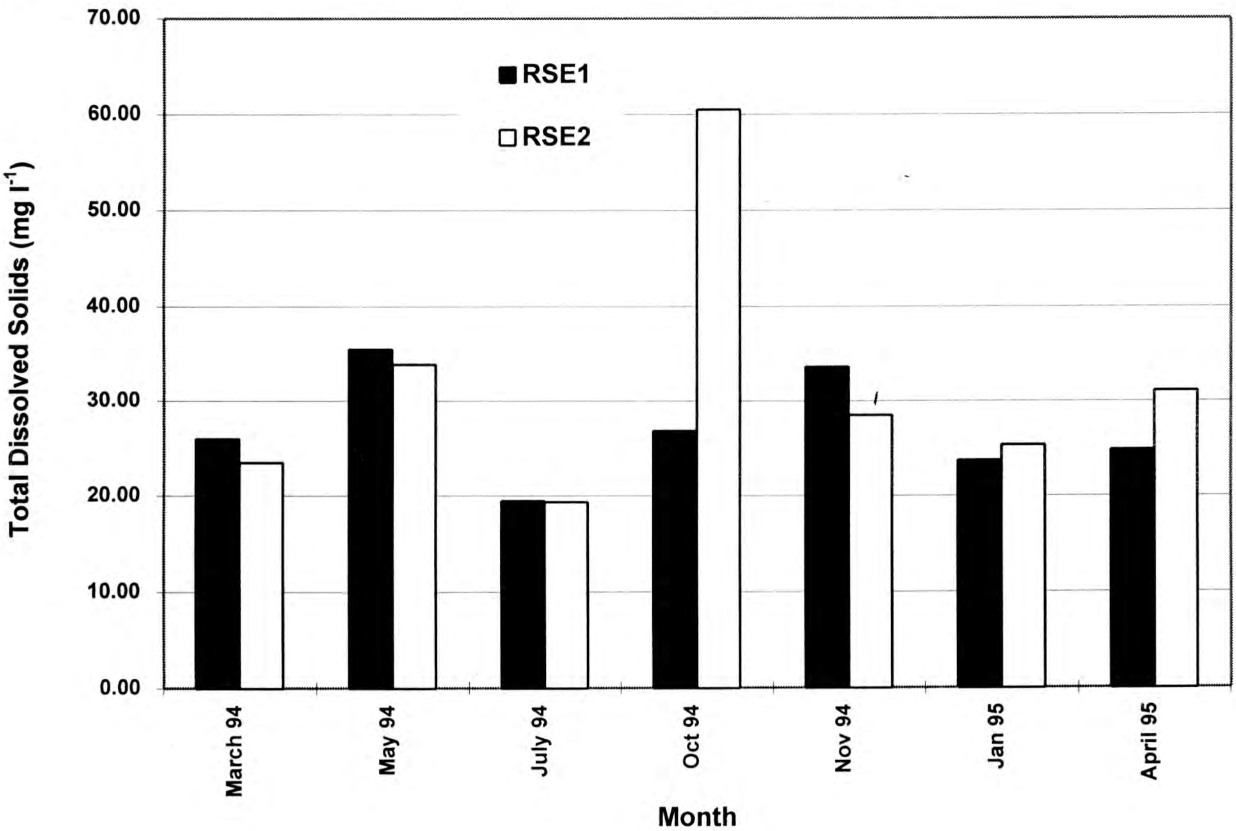
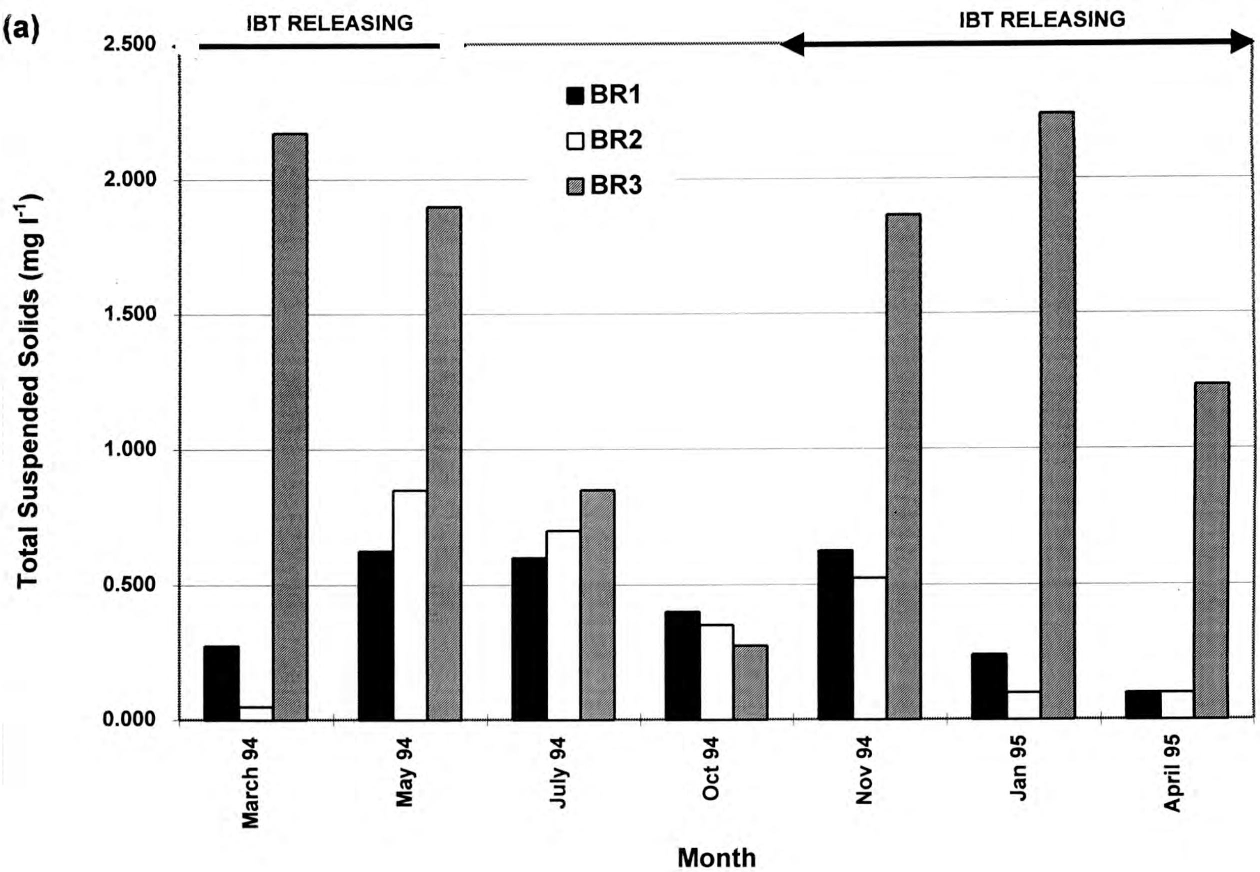


Figure 5.7 Total dissolved solids (TDS) measured at all sampling sites on (a) the Berg and (b) the Riviersonderend rivers, from March 1994 to April 1995.

(a)



(b)

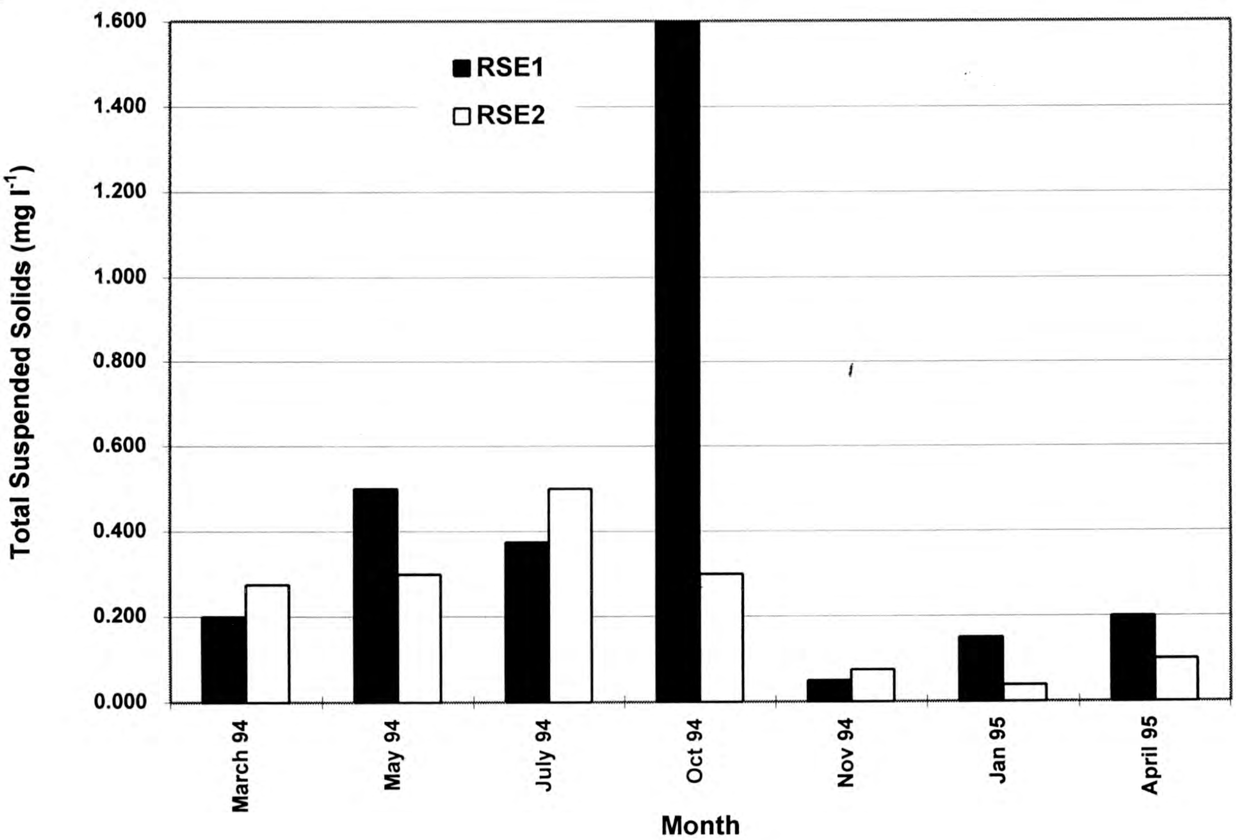


Figure 5.8 Total suspended solids (TSS) measured at all sampling sites on (a) the Berg and (b) the Riviersonderend rivers, from March 1994 to April 1995.

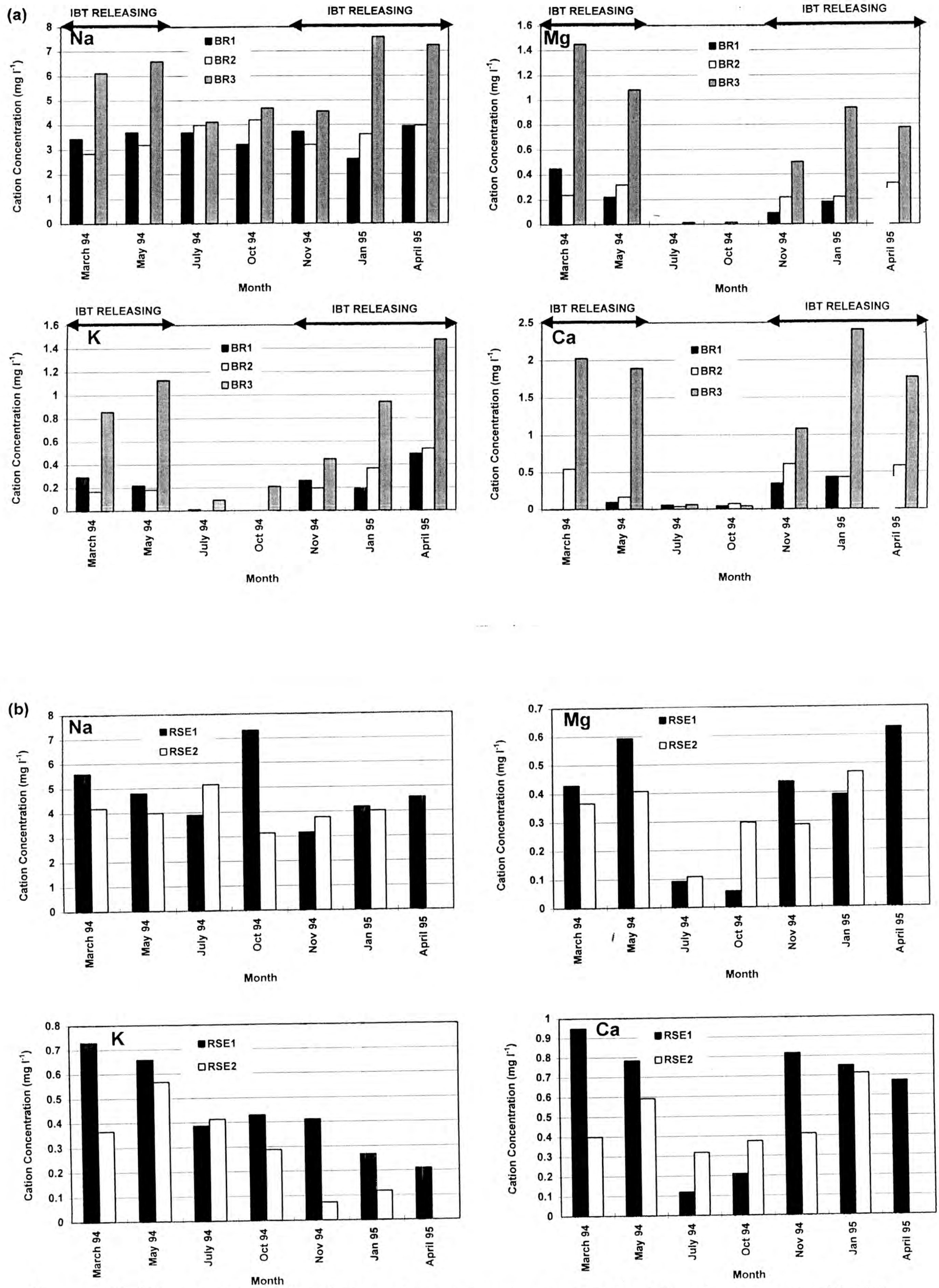


Figure 5.9 Concentrations of sodium, potassium, magnesium and calcium ions measured in (a) the Berg River and (b)the Riviersonderend over the sampling period, March 1994 to April 1995.

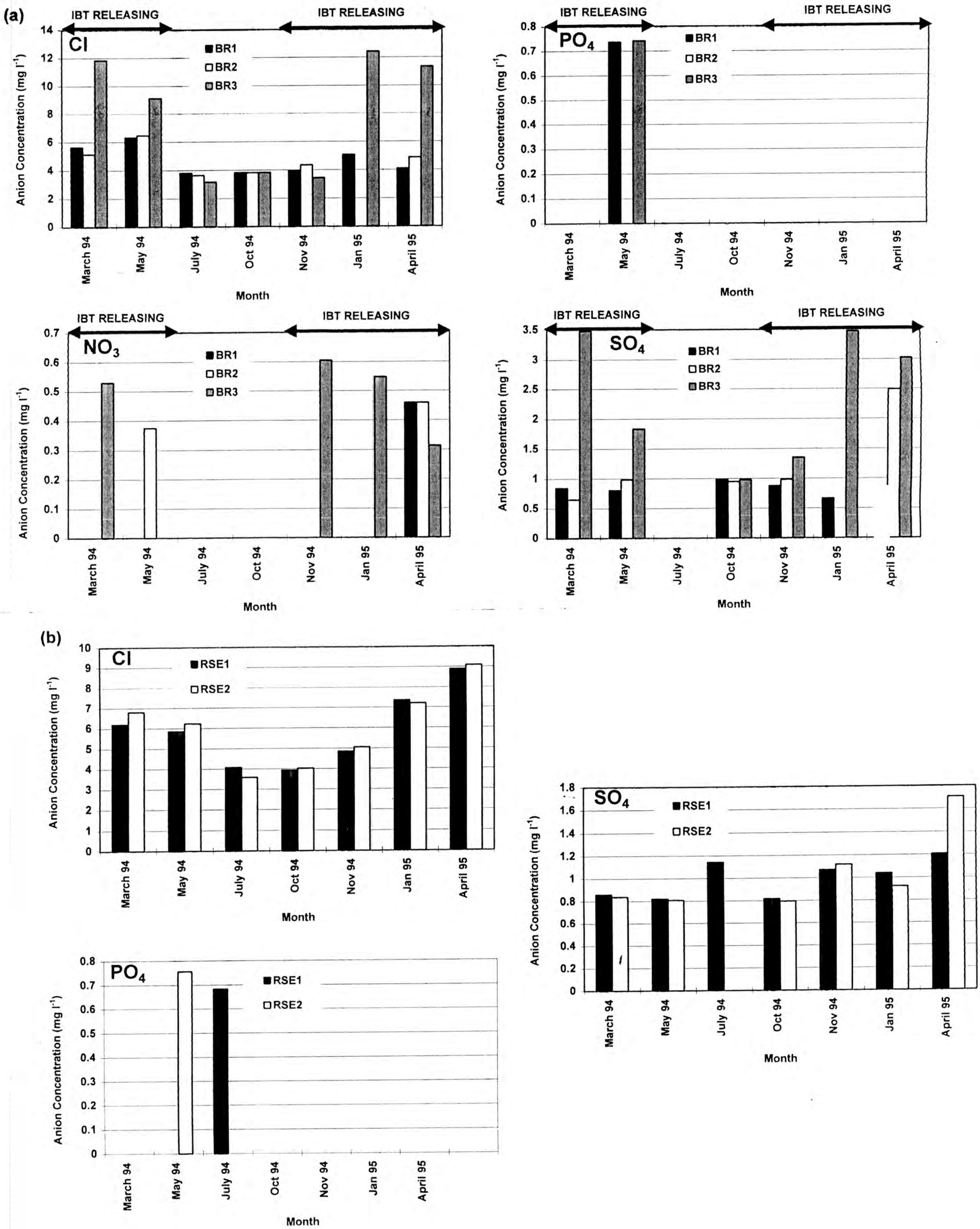


Figure 5.10 Concentrations of chloride, nitrate, phosphate and sulphate ions measured in (a) the Berg River and (b) the Riviersonderend over the sampling period, March 1994 to April 1995. Nitrate concentrations were zero in the Riviersonderend.

5.3.2 Invertebrates of the stones-in-current biotope

Table 5.1 provides a list of those invertebrate taxa that consistently dominated the communities, or that showed marked changes in abundance between sites or seasons in both rivers. The actual numbers of individuals m^{-2} recorded within each taxon, are provided in Appendix 1.1. Looking first at the Riviersonderend, during summer months the communities at both sites were dominated by the following taxa, the abundances of which were classified as category 2 or greater in Table 5.1:

- Lumbriculidae (Oligochaeta)
- Hydrachnellae (Acarina)
- Notonemouridae (Plecoptera)
- Baetidae, Ephemerellidae and Leptophlebiidae (Ephemeroptera)
- Hydropsychidae, Leptoceridae, Hydroptilidae and Ecnomidae (Trichoptera)
- Elmidae and Helodidae larvae (Coleoptera)
- Chironomidae, Simuliidae and Athericidae (Diptera)

In winter, the dominant taxa (category 2 or greater) were:

- Lumbriculidae (Oligochaeta)
- Notonemouridae (Plecoptera)
- Baetidae (Ephemeroptera)
- Chironomidae and Simuliidae (Diptera)

There appeared to be few differences between RSE1 and RSE2 during both seasons (Table 5.1; Appendix 1.1).

Turning to the Berg River, in summer at BR1 and BR2 the taxa that contributed most to the total abundance of invertebrates were (category 3 or greater in Table 5.1):

- Baetidae, Heptageniidae and Leptophlebiidae (Ephemeroptera)
- Elmidae larvae (Coleoptera)
- Chironomidae and Simuliidae (Diptera)

During the same season at BR3, the following taxa were recorded in large numbers, indicating a response to the IBT, or survival despite its presence (category 3 or greater in Table 5.1):

- *Hydra* sp.
- Cladocera
- Copepoda
- Baetidae (Ephemeroptera)
- Hydropsychidae (Trichoptera)
- Chironomidae and Simuliidae and Chaoboridae (Diptera)

The zooplanktonic taxa - the Cladocera, Copepoda, and Chaoboridae - and the attached cnidarian, *Hydra* sp., were introduced into the river through the IBT tunnel, from the donor Theewaterskloof reservoir(Figure 4.1). These animals were sampled in large numbers, and many were observed to be alive on release into the river. Furthermore, there were a few taxa that were only recorded below the IBT in summer; these were the Caenidae (Ephemeroptera), Polycentropodidae (Trichoptera) and Libellulidae (Odonata).

On the other hand, there were several taxa that were present above the IBT in the Berg River in summer, but which were absent or present in greatly reduced numbers below it. These included:

- Notonemouridae (Plecoptera)
- Heptageniidae, Ephemerellidae and Leptophlebiidae (Ephemeroptera)
- Philopotamidae, Leptoceridae, Hydroptilidae and Ecnomidae (Trichoptera)
- Corydalidae (Megaloptera)
- Elmidae larvae and adults, Hydraenidae adults, and Helodidae larvae (Coleoptera)
- Athericidae (Diptera)

In addition, most of the hemipterans were absent from the communities sampled below the IBT. Several rare taxa that were recorded in very low numbers above the IBT were seldom found below it; these were the Sericostomatidae, Barbarochthonidae and Petrothrincidae (Trichoptera), Limnichidae and Gyrinidae (Coleoptera), and the Dixidae (Diptera).

The proportional abundances of various groups of invertebrates above and below the IBT in the Berg River are shown in Figures 5.11 and 5.12. In most months, the Diptera were the dominant order in the Berg River (Figure 5.11). The Ephemeroptera were dominant in October 1994 at BR2 and November 1994 at BR1 and BR2, while the Trichoptera dominated BR3 in March 1994. and in April 1995. Within the macroinvertebrate orders sampled in the Berg River, two families of filter-feeders that were encountered in large numbers below the IBT, the dipteran Simuliidae and the trichopteran Hydropsychidae, showed some interesting trends (Figure 5.12). In March 1994 and January 1995, lower numbers of simuliids were recorded at BR3, compared with BR1 and BR2, while in May and November 1994 and April 1995, this group was more abundant at BR3, with a peak in November 1994. Higher numbers of hydropsychids were always recorded below the IBT during summer, water-release months, with peaks in March 1994 and April 1995.

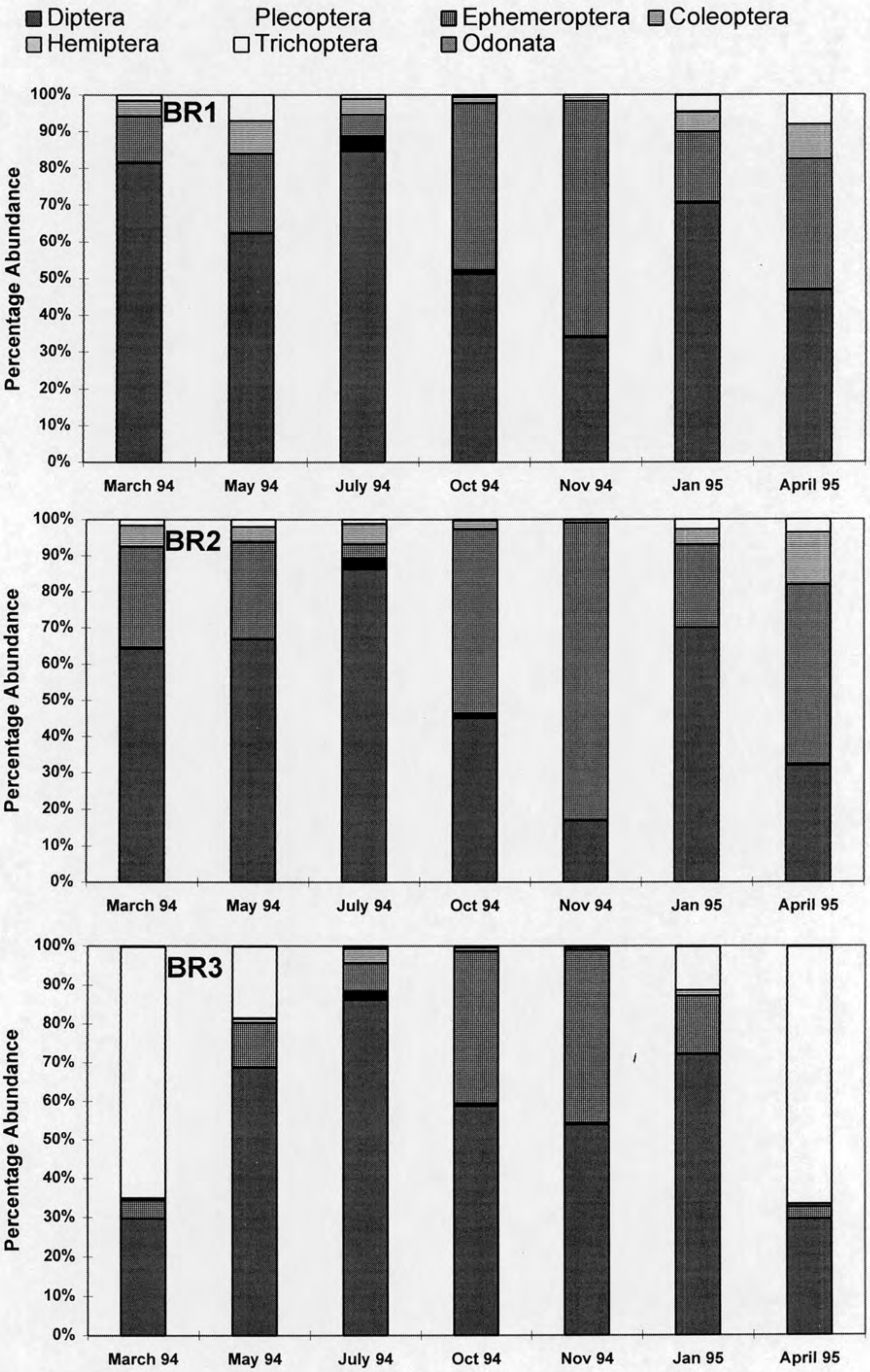


Figure 5.11 Percentage abundances of the insect orders recorded at the three sites on the Berg River, over the sampling period from March 1994 to April 1995.

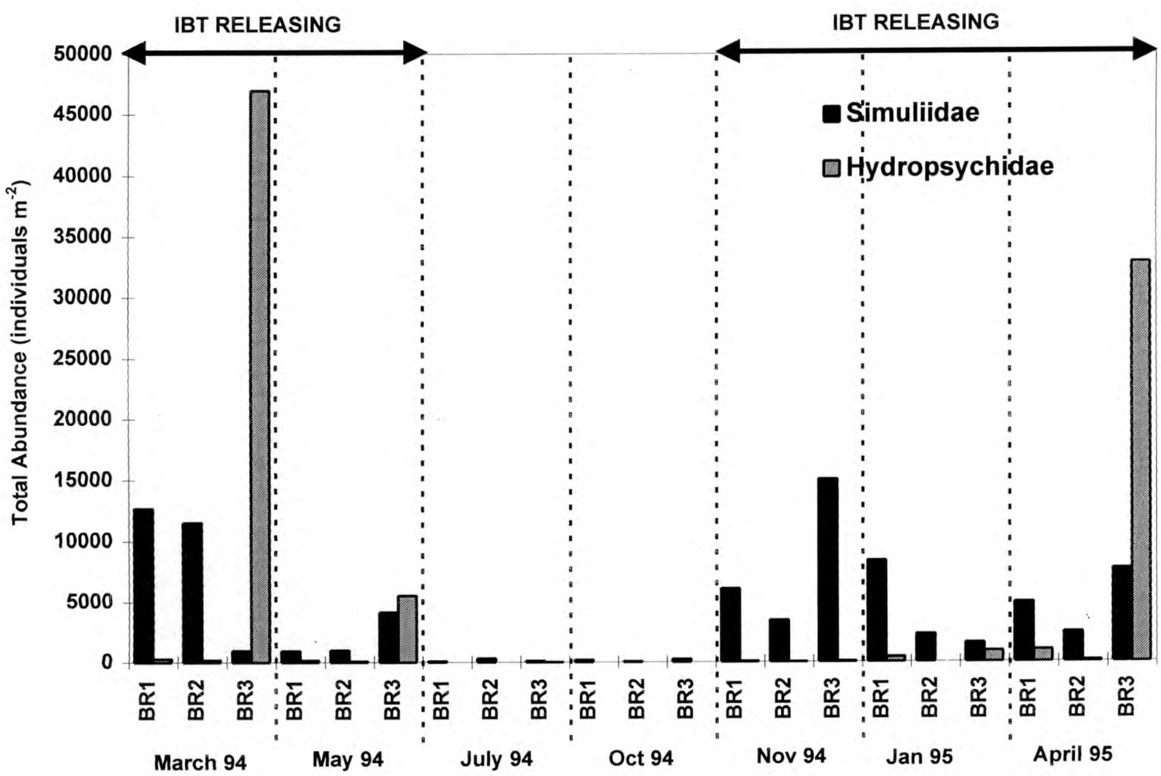


Figure 5.12 Abundances of two filter-feeding families, the Simuliidae (Diptera) and the Hydropsychidae (Trichoptera), recorded in the Berg River, over the sampling period

Table 5.1 A list of macroinvertebrate taxa that were dominant in the Berg River and Riviersonderend communities, or that showed marked differences in abundance between seasons and/or sites. Each taxon is classified according to the abundance recorded in each month: "-"=no individuals; category "**0**"=fewer than 10 individuals m^{-2}; 10<"**1**"<100; 100<"**2**"<1000; 1000<"**3**"<10000; 10 000<"**4**"<100 000; 100 000<"**5**"<1 000 000.

Taxon			March 1994					May 1994				
			BR1	BR2	BR3	RSE1	RSE2	BR1	BR2	BR3	RSE1	RSE2
Cnidaria												
Hydrozoa		*Hydra* sp.	-	-	4	-	-	-	-	2	-	-
Annelida												
OLIGOCHAETA	Lumbriculidae		1	2	-	2	3	2	1	-	2	2
Arthropoda												
Acarina	Hydrachnellae		2	2	2	2	2	2	2	2	2	2
Crustacea												
CLADOCERA			-	1	4	0	0	2	1	4	1	1
COPEPODA			-	1	5	-	1	0	0	4	1	1
PLECOPTERA	Notonemouridae		2	1	-	1	2	1	1	-	1	1
EPHEMEROPTERA	Baetidae		3	4	3	3	3	3	3	3	3	2
	Heptageniidae		1	2	-	0	-	1	2	-	-	-
	Ephemerellidae		1	1	0	2	2	1	2	-	1	1
	Caenidae		-	-	1	0	-	-	-	-	-	0
	Leptophlebiidae		2	3	1	2	3	2	2	1	1	2
TRICHOPTERA	Hydropsychidae		2	2	4	1	0	2	2	3	1	-
	Philopotamidae		1	2	-	-	0	2	1	-	1	0
	Leptoceridae		2	2	1	2	3	2	2	-	2	1
	Hydroptilidae		2	2	1	1	1	1	1	0	1	1
	Polycentropodidae		-	1	0	0	-	0	-	1	-	-
	Ecnomidae		1	1	-	2	1	2	1	-	1	0
ODONATA	Libellulidae		-	-	2	-	-	0	-	1	-	-
MEGALOPTERA	Corydalidae		0	1	1	-	1	1	0	-	1	0
COLEOPTERA	Elmidae	adults	1	2	0	0	1	2	1	1	1	0
	Elmidae	larvae	3	3	2	2	3	3	2	2	2	2
	Hydraenidae	adults	2	2	1	-	1	2	2	2	1	0
	Helodidae	larvae	1	1	0	1	2	2	1	1	2	2
DIPTERA	Chironomidae		4	4	4	4	5	4	4	4	3	3
	Simuliidae		4	4	2	2	2	2	3	3	2	2
	Athericidae		2	2	1	1	2	1	1	1	1	1
	Chaoboridae		-	-	2	-	-	-	-	1	-	-

Table 5.1 continued...

Taxon			July 1994					October 1994				
			BR1	BR2	BR3	RSE1	RSE2	BR1	BR2	BR3	RSE1	RSE2
Cnidaria												
Hydrozoa		*Hydra* sp.	-	-	-	-	-	-	-	-	-	-
Annelida												
OLIGOCHAETA	Lumbriculidae		1	2	1	2	1	2	2	2	1	1
Arthropoda												
Acarina	Hydrachnellae		1	2	1	1	1	2	2	1	1	1
Crustacea												
CLADOCERA			1	1	1	-	-	-	1	0	-	-
COPEPODA			1	2	1	0	1	0	1	0	-	-
PLECOPTERA	Notonemouridae		2	2	1	1	1	2	2	1	1	2
EPHEMEROPTERA	Baetidae		2	2	2	1	1	3	3	3	2	2
	Heptageniidae		-	-	0	-	0	0	1	-	-	-
	Ephemerellidae		1	1	1	1	1	1	2	1	1	1
	Caenidae		-	1	-	1	-	-	-	0	-	0
	Leptophlebiidae		1	1	1	0	1	1	1	1	1	1
TRICHOPTERA	Hydropsychidae		0	0	1	-	-	-	-	1	-	-
	Philopotamidae		1	1	-	1	-	-	-	-	0	0
	Leptoceridae		1	1	-	0	0	1	1	1	-	1
	Hydroptilidae		-	1	0	1	1	1	0	0	1	1
	Polycentropodidae		-	-	-	-	-	-	-	-	-	-
	Ecnomidae		-	0	-	-	-	1	0	-	0	1
	Libellulidae		-	-	-	-	-	-	-	-	-	-
MEGALOPTERA	Corydalidae		-	-	0	-	-	0	-	0	-	0
COLEOPTERA	Elmidae	adults	1	1	-	0	-	1	0	1	-	-
	Elmidae	larvae	1	2	1	1	1	2	2	1	1	1
	Hydraenidae	adults	0	1	1	-	0	1	1	1	-	0
	Helodidae		1	1	1	1	1	2	2	1	1	1
DIPTERA	Chironomidae		3	3	3	3	3	3	3	3	3	3
	Simuliidae		1	2	2	1	1	2	1	2	2	1
	Athericidae		-	-	-	-	-	-	-	-	-	-
	Chaoboridae		-	-	-	-	-	-	-	-	-	-

Table 5.1 continued...

Taxon			November 1994					January 1995				
			BR1	BR2	BR3	RSE1	RSE2	BR1	BR2	BR3	RSE1	RSE2
Cnidaria												
Hydrozoa		*Hydra* sp.	-	-	1	-	-	-	-	3	-	-
Annelida												
OLIGOCHAETA	Lumbriculidae		2	2	2	2	1	2	1	3	2	2
Arthropoda												
Acarina	Hydrachnellae		2	2	2	2	2	2	1	1	2	2
Crustacea												
CLADOCERA			2	2	5	1	-	2	1	4	-	0
COPEPODA			-	1	5	1	-	0	1	4	1	1
PLECOPTERA	Notonemouridae		2	2	2	2	1	1	-	-	1	1
EPHEMEROPTERA	Baetidae		4	4	4	3	3	3	2	2	3	3
	Heptageniidae		1	1	0	-	-	2	2	-	-	-
	Ephemerellidae		0	1	-	2	1	2	2	0	2	2
	Caenidae		1	-	1	-	-	-	-	2	-	-
	Leptophlebiidae		2	2	1	2	2	2	1	0	2	2
TRICHOPTERA	Hydropsychidae		1	1	1	2	-	2	-	2	-	1
	Philopotamidae		1	0	-	0	-	1	1	-	-	-
	Leptoceridae		1	1	-	2	1	2	1	1	2	2
	Hydroptilidae		-	-	-	1	1	1	-	2	1	1
	Polycentropodidae		-	-	1	-	0	-	-	0	-	0
	Ecnomidae		1	1	0	0	0	0	1	1	2	1
	Libellulidae		-	-	-	-	-	-	-	1	-	-
MEGALOPTERA	Corydalidae		1	0	-	-	-	0	-	-	0	1
COLEOPTERA	Elmidae	adults	2	2	2	2	1	2	1	-	1	1
	Elmidae	larvae	2	2	2	2	2	2	2	2	2	2
	Hydraenidae	adults	2	2	2	1	1	2	1	1	1	1
	Helodidae		2	2	1	2	2	1	1	-	2	1
DIPTERA	Chironomidae		4	4	4	4	3	3	3	3	3	3
	Simuliidae		3	3	4	2	2	3	3	3	1	2
	Athericidae		2	2	1	2	2	1	1	1	2	2
	Chaoboridae		-	0	3	-	-	-	-	2	-	-

Table 5.1 continued...

Taxon			April 1995				
			BR1	BR2	BR3	RSE1	RSE2
Cnidaria							
Hydrozoa		*Hydra* sp.	-	-	3	-	-
Annelida							
OLIGOCHAETA	Lumbriculidae		2	2	2	2	2
Arthropoda							
Acarina	Hydrachnellae		2	2	2	2	2
Crustacea							
CLADOCERA			1	-	4	1	-
COPEPODA			0	0	4	0	0
PLECOPTERA	Notonemouridae		1	1	-	2	1
EPHEMEROPTERA	Baetidae		3	3	3	2	2
	Heptageniidae		2	3	-	-	-
	Ephemerellidae		1	2	-	1	1
	Caenidae		-	1	1	-	-
	Leptophlebiidae		3	3	0	2	2
TRICHOPTERA	Hydropsychidae		2	2	4	1	1
	Philopotamidae		2	-	-	1	1
	Leptoceridae		2	2	-	2	1
	Hydroptilidae		1	1	-	2	1
	Polycentropodidae		1	0	-	1	-
	Ecnomidae		2	1	-	1	1
	Libellulidae		-	-	2	0	0
MEGALOPTERA	Corydalidae		1	1	-	0	-
COLEOPTERA	Elmidae	adults	2	2	0	1	1
	Elmidae	larvae	3	3	2	2	2
	Hydraenidae	adults	2	2	1	1	0
	Helodidae		1	1	-	2	1
DIPTERA	Chironomidae		3	3	3	3	3
	Simuliidae		3	3	3	2	2
	Athericidae		1	2	1	1	2
	Chaoboridae		-	-	2	-	-

Most of the taxa sampled in the Berg in summer, were present in winter, and the main difference between the seasons was the lower abundance of individuals in winter (Table 5.1; Appendix 1.1). Zooplanktonic groups were sampled at all sites in winter, but in very low numbers, and *Hydra* sp. was absent during this season. The ephemeropterans were present in lower numbers, especially the Heptageniidae, as were the trichopterans, the odonates, the coleopterans and some of the dipterans. There were some taxa which occurred in greater numbers in winter, including the Nematoda, the Collembola (springtails), the Notonemouridae (Plecoptera), and the

Ceratopogonidae (Diptera). There were no marked differences between the invertebrate communities sampled at the three sites during this season.

Univariate Analyses Figures 5.13 and 5.14 illustrate the changes in total abundances, total numbers of taxa, Margalef's richness indices and Shannon-Wiener diversity indices for each site over the sampling period, while Appendix 2.4 provides the values. The total number of taxa varied from 22 to 35 in the Berg River, and from 22 to 34 in the Riviersonderend. The variation in numbers of taxa appeared to be seasonal in the Riviersonderend, with lowest figures in winter, while this variation did not follow an obviously seasonal pattern in the Berg. In both rivers, the lowest abundances were recorded during the winter months (Riviersonderend: 2080 individuals m^{-2} in July; Berg: 4650 individuals m^{-2} in July; Figure 5.13, Appendix 2.4). Diversity and richness indices did not appear to follow a seasonal pattern in either river (Figure 5.14).

In the Berg River in summer, there were clear differences between indices calculated from BR1 and BR3, but no significant differences between BR1 and BR2. The invertebrate community at BR3 had significantly greater numbers of individuals ($t=-2.489$; $p<0.05$, one-tailed test), and lower overall richness ($t=5.494$; $p<0.005$, one-tailed test) and numbers of taxa ($t=2.875$; $p<0.05$; one-tailed test). Diversity, as measured by the Shannon-Wiener diversity index was not significantly different at BR3 ($t=1.639$; $p>0.05$; one-tailed test). In winter, there were no significant differences between invertebrate community indices at any of the sites on the Berg River. In addition, there were no significant differences between any of the indices measured at RSE1 and RSE2, over the sampling period. Lastly, a statistical comparison between the Riviersonderend and Berg River, above-IBT sites indicated that there were significantly fewer taxa and lower total abundances of invertebrates in the Riviersonderend (*taxa*: $t=-1.936$, $p<0.05$; *abundance*: $t=-2.334$, $p<0.05$; one-tailed tests). However, the two rivers were similar in terms of richness and diversity.

Multivariate Analyses

Testing for Significance In all cases, there were significant differences between sampling months, averaged across all sites, in both the Riviersonderend ($R=0.421$, $p<0.05$) and Berg rivers (*summer*: $R=0.592$, $p<0.05$; *winter*: $R=0.531$, $p<0.05$). In other words, there were clear temporal differences between the invertebrate communities sampled in both rivers. Furthermore, the results of the first ANOSIM test - summer samples from BR1, BR2 and BR3 - showed significant differences between sites ($R=0.459$, $p<0.05$). In this case, dissimilarities between BR1 and BR3, and BR2 and BR3 contributed significantly to differences between sites (*BR1* vs *BR3*: $R=0.850$, $p<<0.05$; *BR2* vs *BR3*: $R=0.900$, $p<<0.05$). In winter, there were no significant differences between any of the sites in the Berg ($R=-0.444$, $p>>0.05$). Similarly, there were no significant differences between the two Riviersonderend sites ($R=-0.059$, $p>>0.05$) over the sampling period. Lastly, both BR1 and BR2 were significantly different from RSE1 and RSE2, when the data were averaged across all months (*BR1* vs *RSE1*: $R=0.169$, $p<0.05$; *BR1* vs *RSE2*: $R=0.286$, $p<0.05$; *BR2* vs *RSE1*: $R=0.137$, $p<0.05$; *BR2* vs *RSE2*: $R=0.237$, $p<0.05$).

Multivariate Representation of the Data Many of the differences in community structure that have been described in the previous sections were synthesised into a more clear analysis by the multivariate statistical techniques used by the package PRIMER. Figures 5.15 to 5.18 provide dendrograms and MDS cluster plots that illustrate the changes in community structure in the Berg River, while Figure 5.19 provides similar results for the Riviersonderend, and Figure 5.20 combines the datasets from both rivers.

Community data analyses for the Berg River data, including the transferred zooplanktonic groups, resulted in a clear grouping of benthic invertebrate samples collected below the IBT (BR3) in summer, when water was being released from the tunnel. These samples were approximately 50% similar to the remaining samples (Figure 5.15). Winter samples taken above the IBT (BR1 and BR2) formed another group which separated from summer samples from the same sites, at approximately 58% similarity. Within the winter group, samples tended to separate according to the sampling month, with a discrete October 1994 group approximately 65% similar to the July 1994 samples, with the exception of two July samples which separated from the main group at approximately 63% similarity (Figure 5.15). In summer, there appeared to be a random grouping of samples, with the exception of the November 1994 samples which formed their own group which was, at most, 75% similar to the remaining summer samples. The MDS cluster plot summarises these groupings of samples, and illustrates a distinctive and discrete invertebrate community which occurred in summer below the IBT, while summer and winter communities above the IBT were more similar to each other, yet still distinct (Figure 5.15).

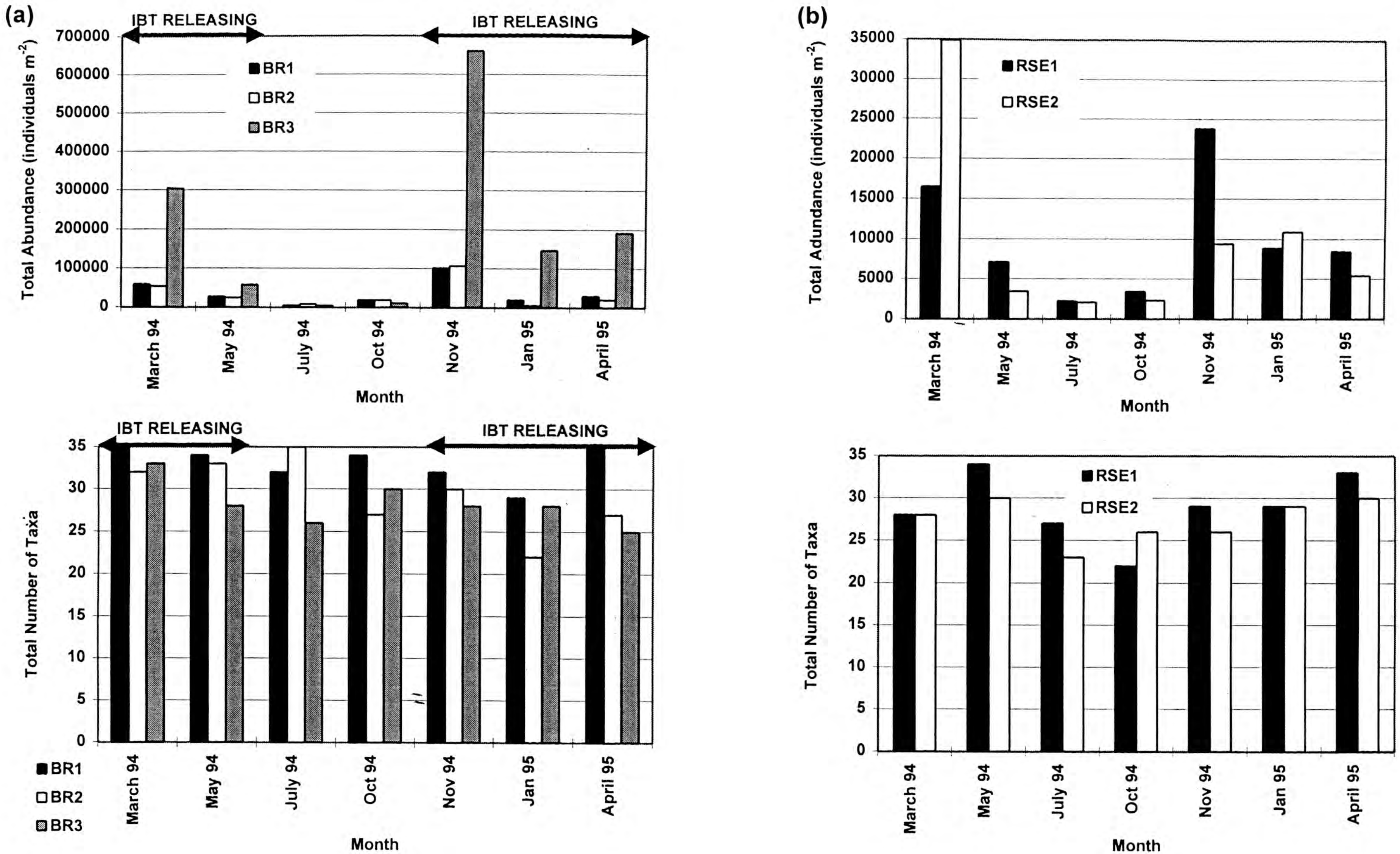


Figure 5.13 A composite diagram showing various indices calculated from the macroinvertebrate data from both rivers, including (a) total abundances of all individuals and total numbers of taxa in the Berg River, and (b) similar values for the Riviersonderend.

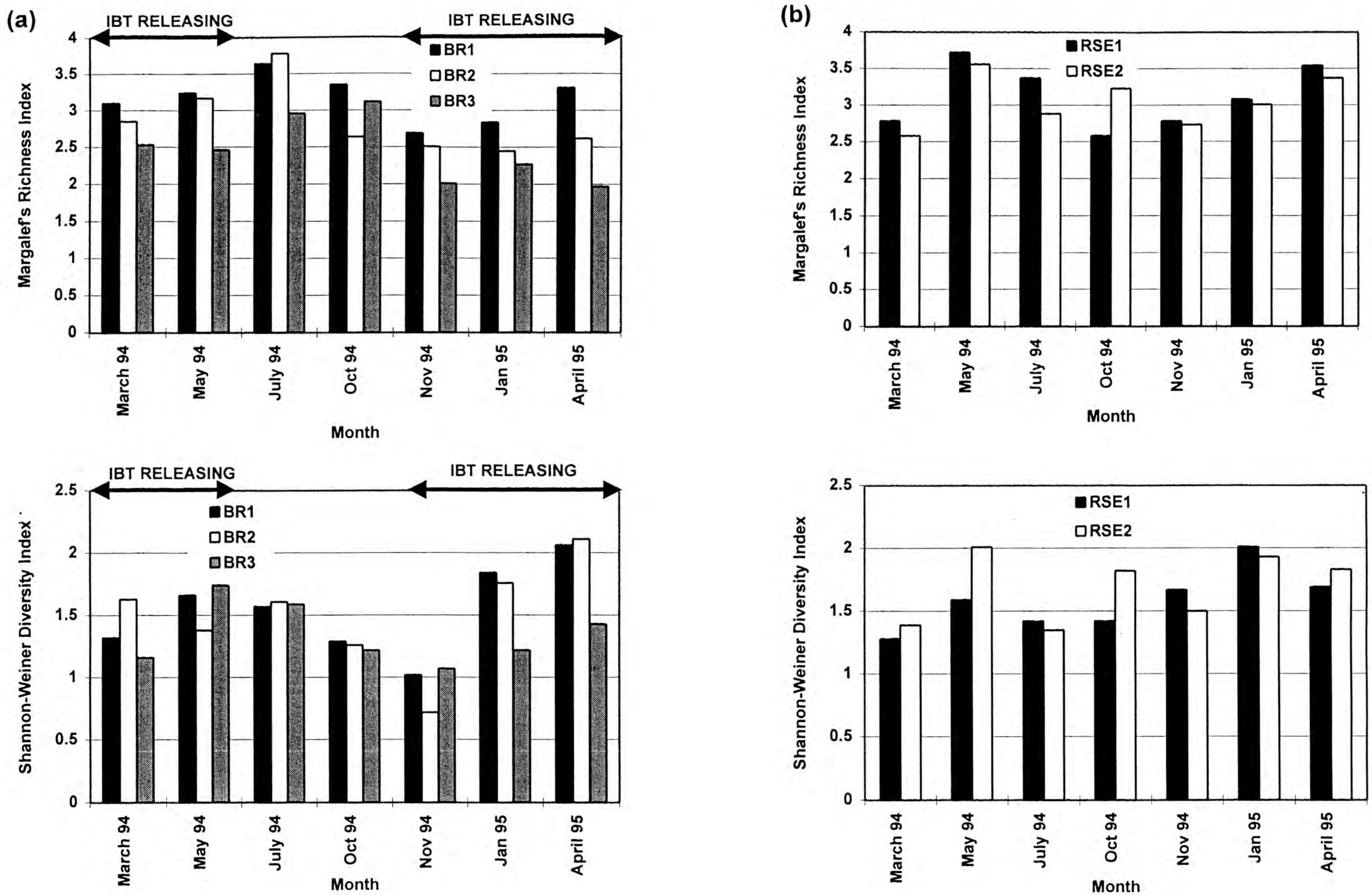


Figure 5.14 Margalef's richness indices and Shannon-Weiner diversity indices calculated for the invertebrate data from (a) the Berg River, and (b) the Riviersonderend.

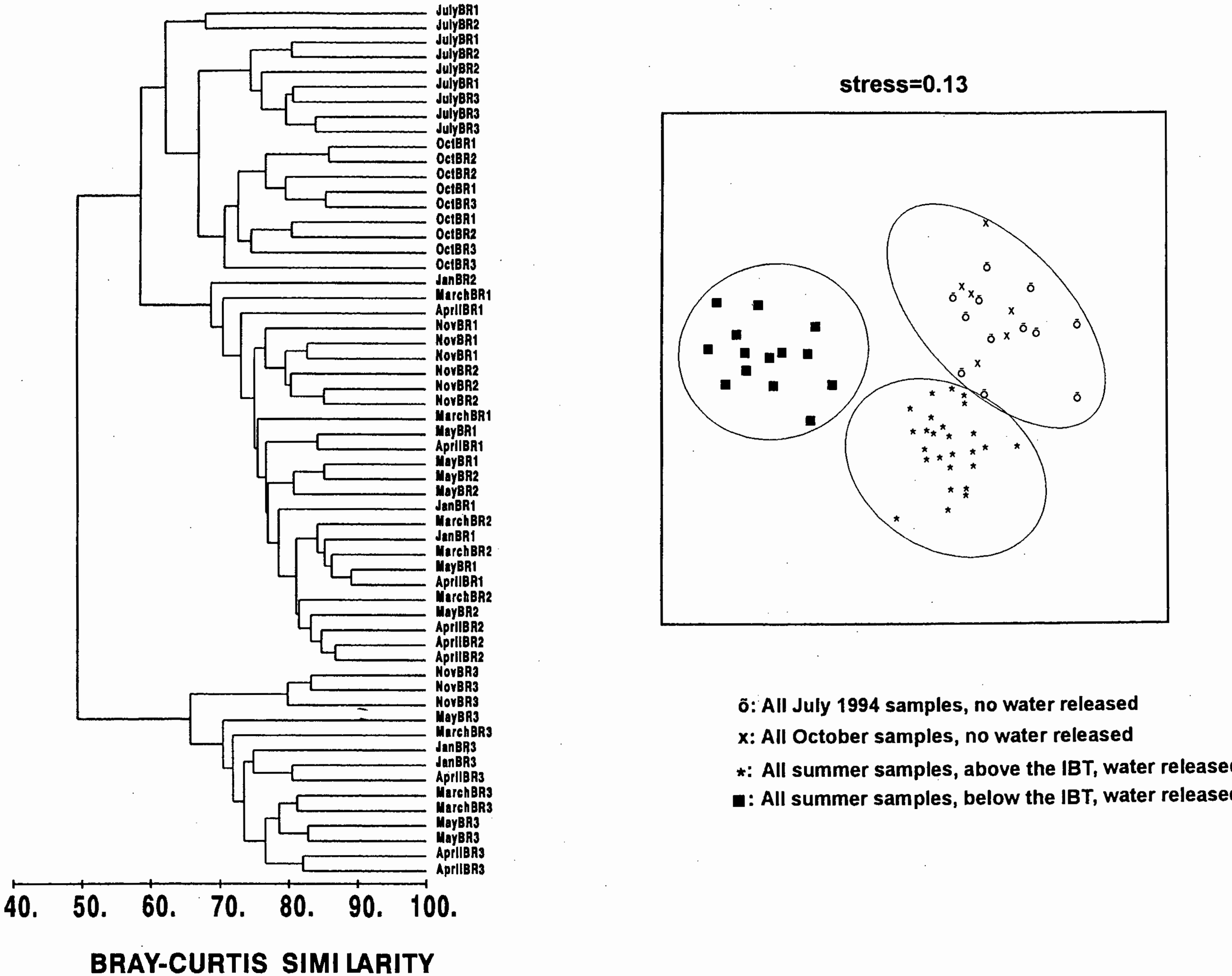


Figure 5.15 A Bray-Curtis similarity dendrogram and MDS-ordination plot of all the Berg River samples, including all the data. A stress of 0.13 indicates a potentially useful 2-dimensional plot.

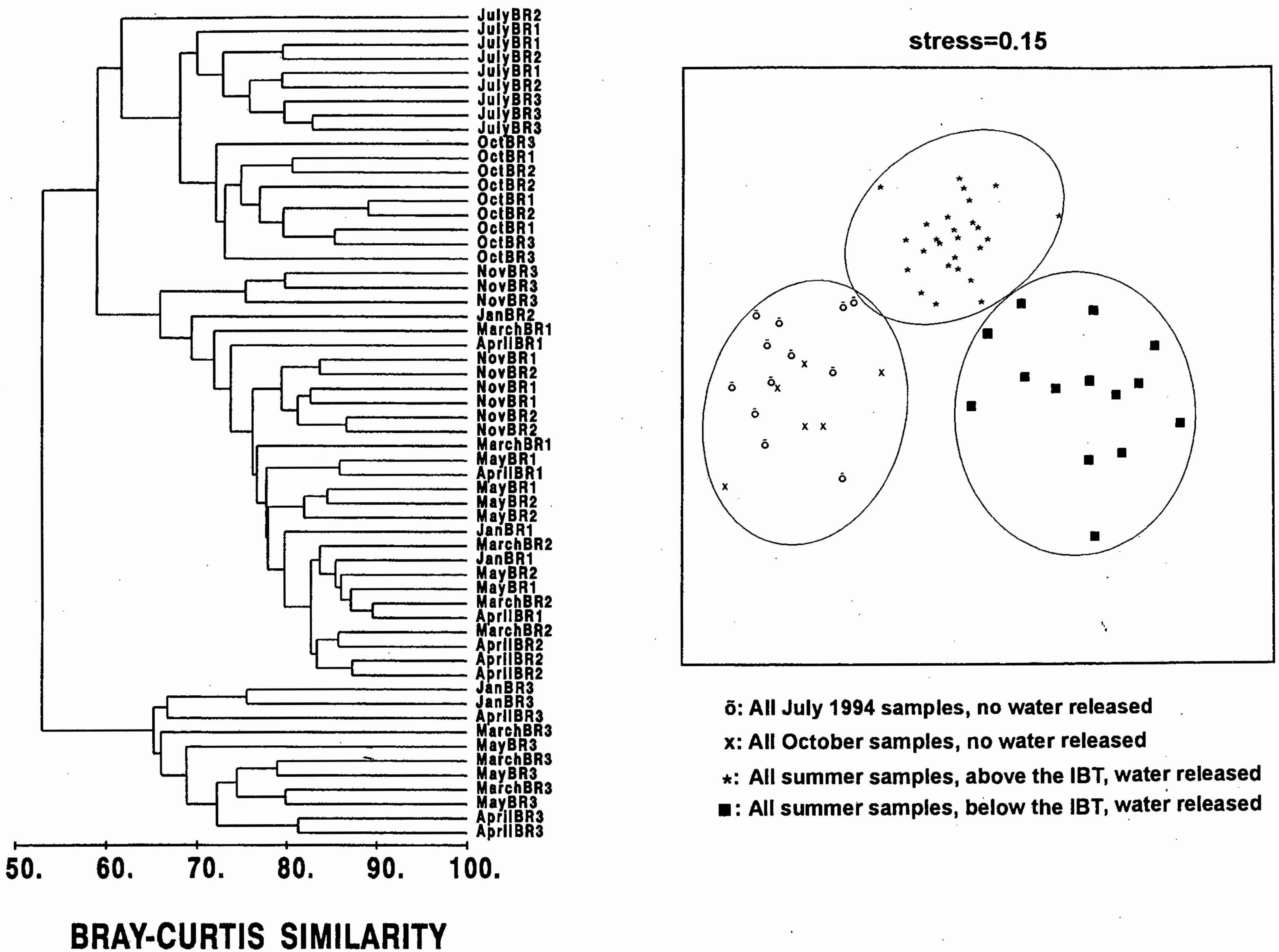


Figure 5.16 A Bray-Curtis similarity dendrogram and MDS-ordination plot of all the Berg River samples, excluding those taxa that were transferred into the river from Theewaterskloof reservoir. A stress of 0.15 indicates a potentially useful 2-dimensional plot.

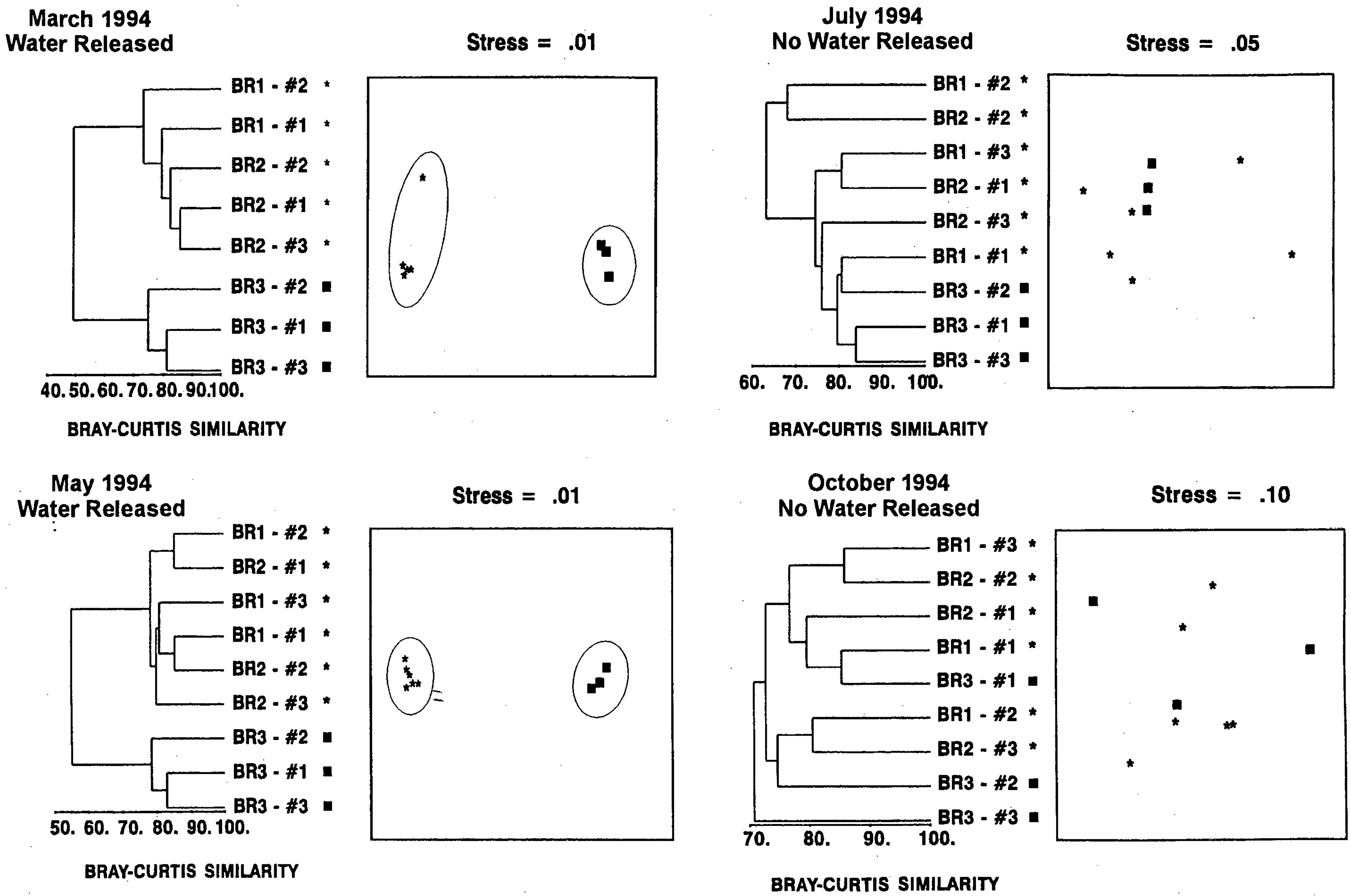


Figure 5.17 A composite diagram showing Bray-Curtis similarity dendrograms and MDS-ordination plots for the Berg River macroinvertebrate community data. The data are arranged by month, in order to illustrate the progressive dissimilarity between samples taken above and below the IBT over the period of water release. #1, #2 and #3 are the three replicate samples taken at each site. Stress values range from 0 (excellent representation of the data) to 0.1 (good ordination).

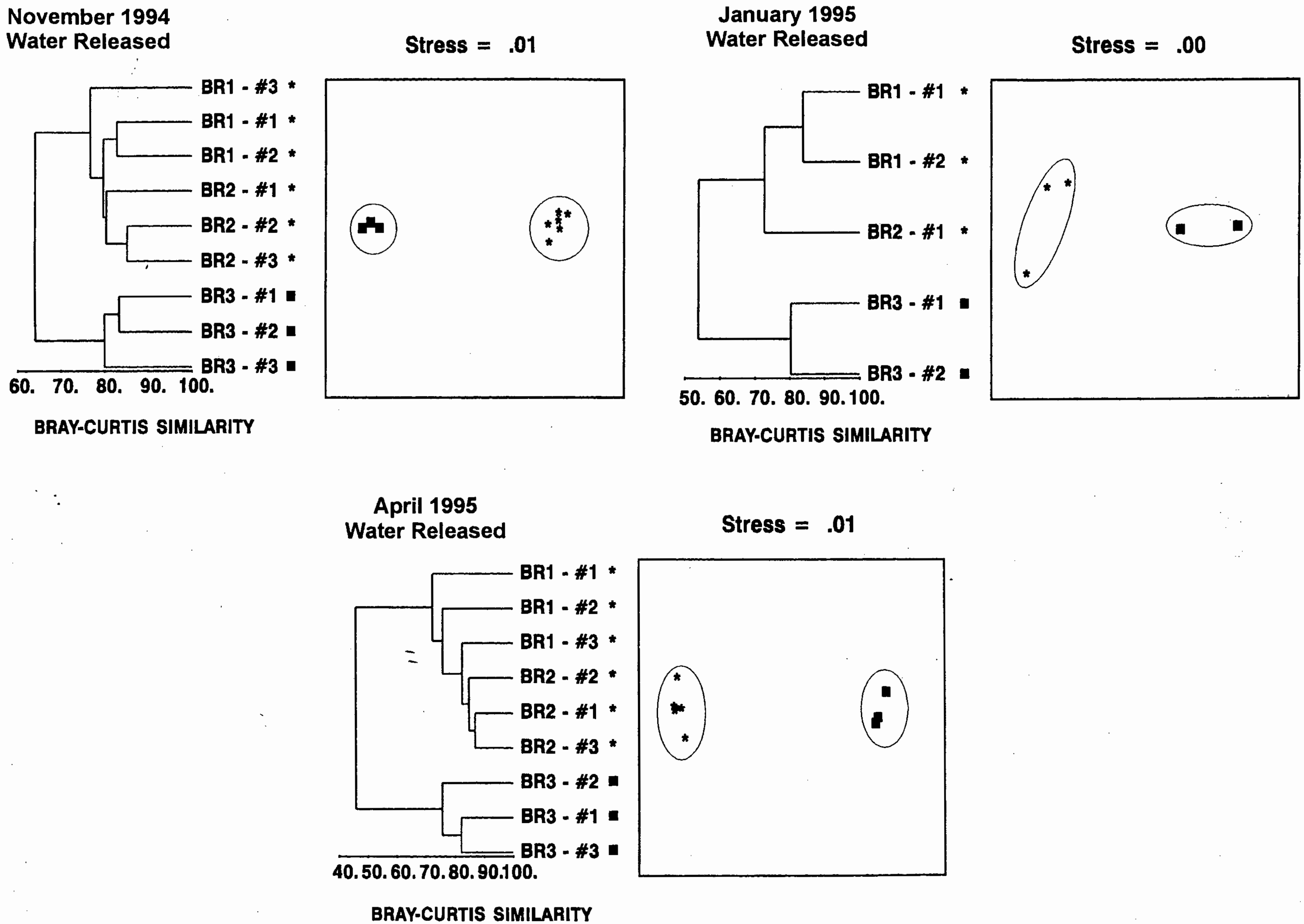


Figure 5.17 continued...

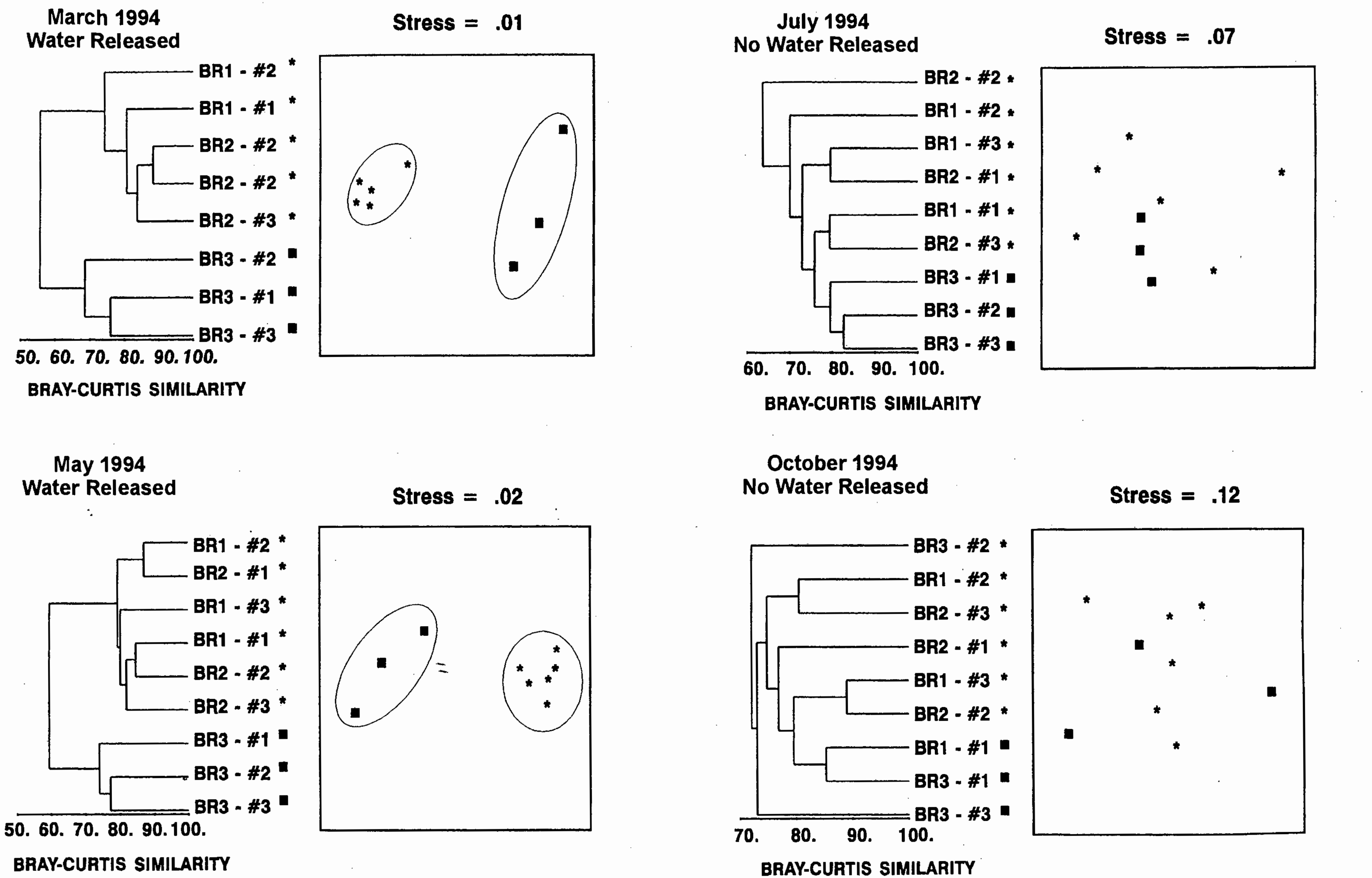


Figure 5.18 A composite diagram showing Bray-Curtis similarity dendrograms and MDS-ordination plots for the Berg River macroinvertebrate community data, without the taxa transferred from Theewaterskloof. The data are arranged per month, in order to illustrate the progressive dissimilarity between samples taken above and below the IBT over the period of water release. #1, #2 and #3 are the three replicate samples taken at each site. Stress values range from 0 (excellent representation) to 0.12 (potentially useful representation of the data).

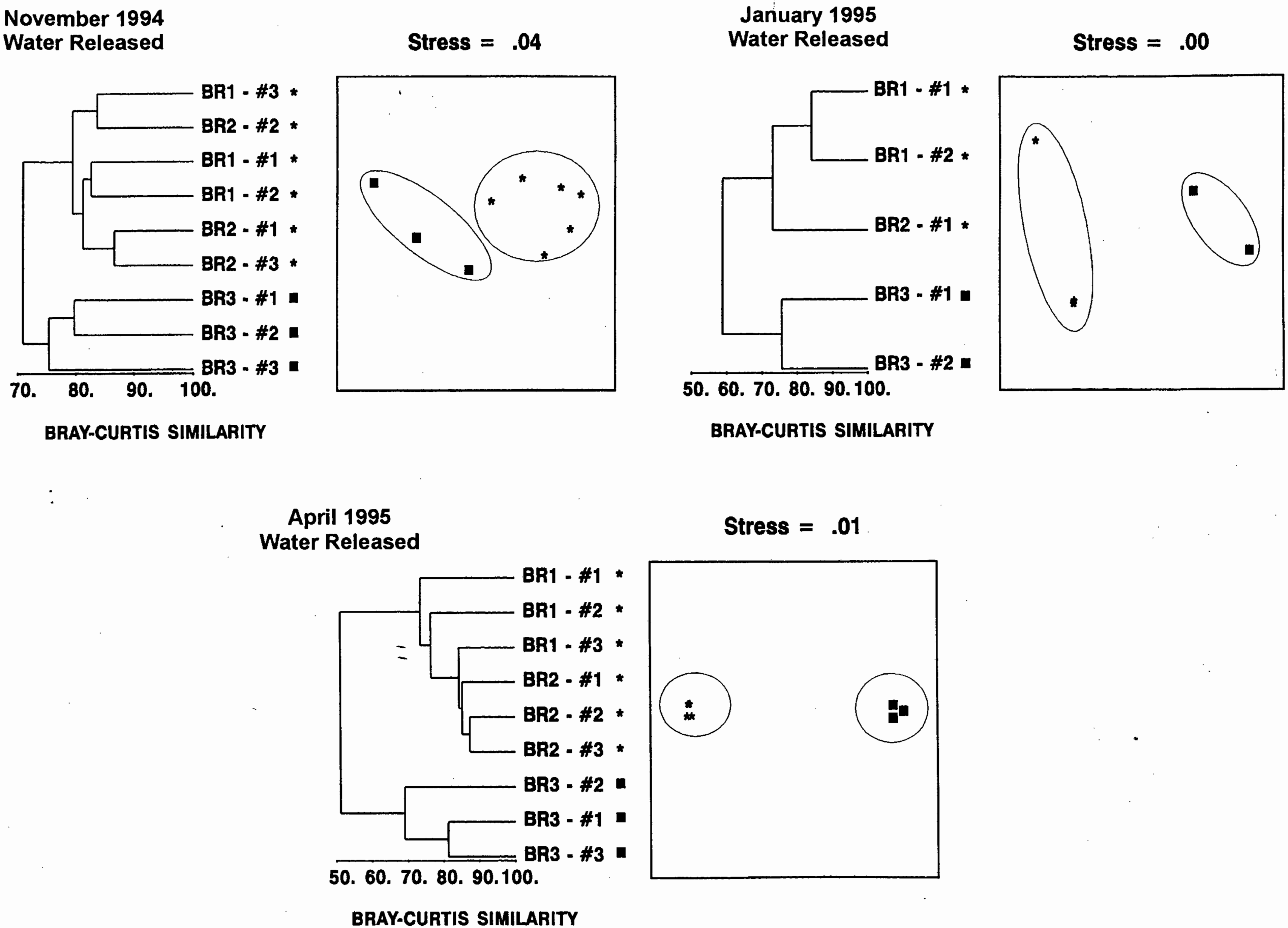


Figure 5.18 continued...

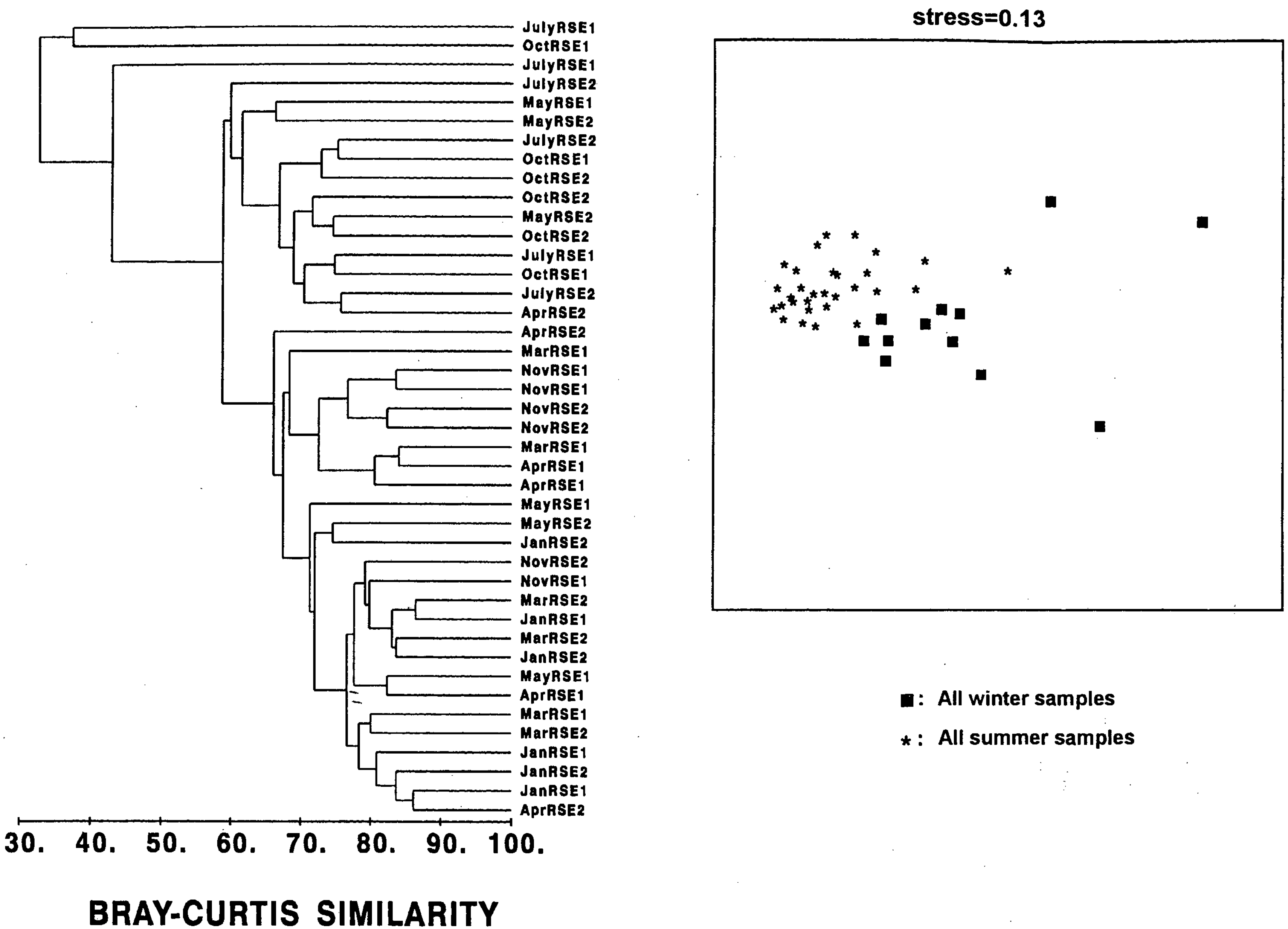


Figure 5.19 A Bray-Curtis similarity dendrogram and MDS-ordination plot of all the Riviersonderend samples, taken over the sampling period. A stress value of 0.13 indicates a potentially useful representation of the data.

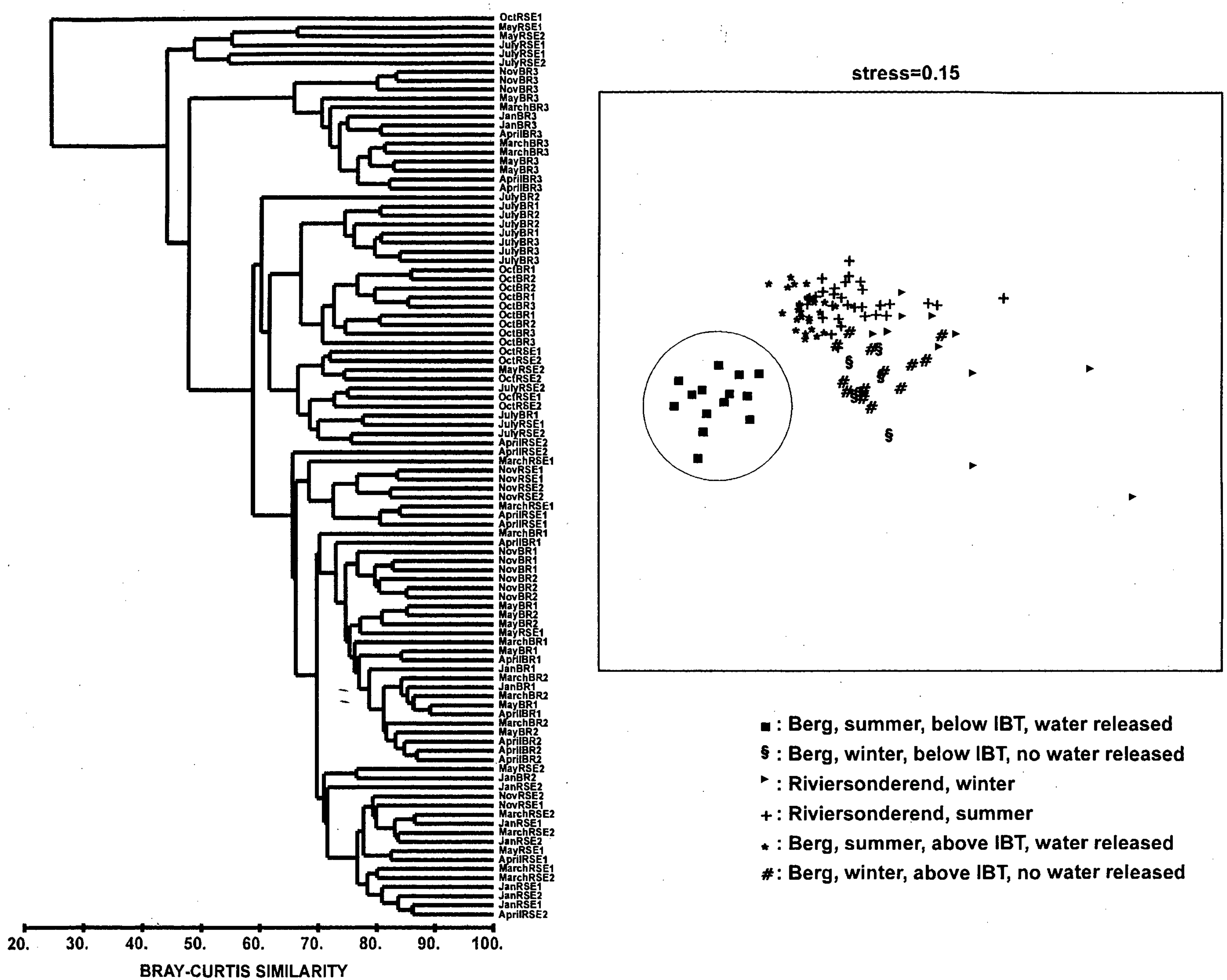


Figure 5.20 A dendrogram and MDS-ordination plot which combine all the macroinvertebrate data collected from the Riviersonderend and Berg rivers, over the sampling period. A stress of 0.15 indicates a potentially useful 2-dimensional plot.

On removal of taxa that were introduced into the river by the IBT, a similar clustering of samples occurred, but with one marked difference (Figure 5.16). The November samples, taken below the IBT (BR3) grouped with the summer samples taken above the IBT at just over the 65% similarity level. Thus, immediately after the tunnel commenced its summer releases of water at the end of October 1994, the invertebrate community still resembled that above the IBT, and the marked differences between sites only became apparent by January 1995.

A month-by-month analysis of the invertebrate data showed a more detailed interpretation of the changes in community structure in the Berg over the period of water releases. All summer months showed a clear grouping of BR3 samples, separate from the above-IBT samples, with stress values not exceeding 0.01 (Figure 5.17). Similarities between above-IBT and below-IBT groups of samples varied throughout the sampling period, from a maximum of approximately 63% in November 1994, to close to 45% in April 1994. The January 1995 results were less distinct, due to the paucity of data in this month. In July and October 1994, samples did not group according to site, and this became most apparent in October 1994, immediately before IBT releases re-commenced. Stress values were higher for these MDS plots - 0.05 in July, and 0.1 in October 1994, but were still indicative of a good ordination.

Once again, the removal of transferred taxa from the dataset resulted in similar groupings of samples (Figure 5.18). However, similarities were greater between above-IBT and below-IBT sample groups in all summer, release months. Furthermore, stress values for MDS-plots were higher for most months during the sampling period.

The Riviersonderend invertebrate data were fairly mixed, in terms of site and month (Figure 5.19). One distinct summer group separated from the remaining samples below the 60% similarity level, while three winter samples from RSE1 split off from the group below the 40% level. The remaining group of samples was a combination of winter and summer samples from both sites (Figure 5.19).

Lastly, a dendrogram and MDS-ordination plot showing all samples taken over the sampling period provided a comparison between invertebrate communities in the Riviersonderend and Berg rivers (Figure 5.20). According to the dendrogram in Figure 5.20, the summer BR3 samples were less than 50% similar to the rest of the samples, while three winter

Riviersonderend samples split from the rest of the samples just above the 30% similarity level. The remaining samples split into a summer group, containing all Berg River samples and most Riviersonderend samples, and a mixed group containing all winter Berg and Riviersonderend samples and some summer samples from the latter river. Within the summer group, samples from the two rivers formed three more or less discrete groups, with similarities ranging from approximately 65 to 70%. The winter Berg River samples were approximately 60% similar to the rest of the group. The MDS-ordination illustrates the grouping of samples based on the dendrogram (Figure 5.20).

Determining Discriminating Taxa A determination of families that contributed to the dissimilarity between Berg River sites produced a list of families that confirmed the observed differences between the sampled invertebrate communities, described in Section 5.3.2. Table 5.2 provides the results of this analysis, which excludes the transferred taxa - cladocerans, copepods, *Hydra* and chaoborids. The hydropsychid trichopterans contributed most to the dissimilarity between sites, with an extremely high average abundance of 18452 individuals m^{-2} below the IBT. Collectively, the Trichoptera made a 15.6% contribution to dissimilarities between samples. The Baetidae, Leptophlebiidae, Heptageniidae and Ephemerellidae (Ephemeroptera) occurred in markedly lower numbers below the IBT in summer, and these differences collectively accounted for over 18% of the dissimilarity between sites.

Table 5.2 A list of Berg River families that contributed to the dissimilarities between groups of samples illustrated in Figures 5.15 to 5.18. These values were generated by the SIMPER routine of the PRIMER package.

Order	Family	Average Abundance Above IBT	Average Abundance Below IBT	% Contribution to Dissimilarity	Cumulative Contribution
Trichoptera	Hydropsychidae	254.62	18 452.10	8.11	8.11
Ephemeroptera	Leptophlebiidae	863.46	15.00	5.38	13.49
Ephemeroptera	Baetidae	22 256.92	7 719.29	5.32	18.82
Trichoptera	Leptoceridae	267.31	5.71	4.24	23.06
Ephemeroptera	Heptageniidae	368.85	0.71	4.15	27.21
Annelida	Lumbriculidae	304.23	552.86	4.03	31.23
Ephemeroptera	Ephemerellidae	125.00	2.14	3.56	34.79
Diptera	Simuliidae	5 230.00	6 210.00	3.47	38.26
Coleoptera	Elmidae larvae	1 267.69	252.86	3.32	41.58
Diptera	Chironomidae	15 105.00	12 175.00	3.28	44.86
Trichoptera	Philopotamidae	187.31	0.00	3.25	48.10

5.4 Discussion

The effects of river regulation by impoundment are well-known (e.g. Ward & Stanford, 1983a,b; Lillehammer & Saltveit, 1984; Petts, 1984; Ward *et al.*, 1984; Craig & Kemper, 1987; Byren & Davies, 1989; Gore & Petts, 1989; Petts *et al.*, 1989). In order to provide reliable and constant sources of water to human users, river impoundment generally leads to a decrease in the magnitude and an increase in frequency and duration of downstream flows (e.g. Thoms & Walker, 1992; Davies *et al.*, 1993). Thus, impoundment results in the decreased variability of the flow regime. In the context of southern African semi-arid rivers, the effects of flow regulation on the riverine biota are exacerbated by the fact that they are adapted to natural conditions of hydrological variability and unpredictability (Davies *et al.*, 1993; Section 3.1). It is clear from the literature dealing with IBTs (Chapter 2), that flow alterations are a feature of these water developments. In order to understand the effects of IBTs on river systems, it is important to investigate the differences and similarities between IBTs and river impoundment. A major difference between IBTs and impoundment is presented in Figure 5.3a, which shows discharge peaks during summer, low-flow months, in the recipient reaches of the Berg River. In some cases these summer discharges exceeded winter volumes. Accordingly, the magnitude of summer discharges have increased, along with their frequency and duration.

The extent of flow alteration in the upper Berg River can best be described in terms of the percentage increase over the discharge measured upstream of the IBT. In March 1994, the discharge downstream of the IBT was over 4400% greater than that measured above it, 730% greater in May, 17 and 36% respectively in July and October, while in November 1994, once water releases recommenced, the increase was over 1700%, 2200% in January 1995, and 570% in April 1995 (Figure 5.3). The slight increases during the winter months probably could be ascribed to runoff entering the river between sampling sites. The summer discharge increases, however, were substantial, during months when the discharges in the Berg River should be at their lowest. These alterations in flow and, consequently, width (Figure 5.1b), represented a substantial local disturbance in the river.

The summer discharge peaks downstream of the IBT in the Berg are reflected in the measures of water chemistry in these reaches. In addition to substantial changes in hydrology, the IBT transferred water of significantly different quality into the upper Berg River. While temperature did not show any significant correlation with the water transfer, measures of pH, conductivity, total dissolved and suspended solids, sodium, magnesium, potassium and calcium cations, and sulphate and chloride anions, were significantly higher downstream of the IBT during summer. All of these variables are expected to increase downstream (e.g. Dallas, 1995; Harrison & Elsworth, 1958). The IBT, however, resulted in an unnaturally marked alteration in water quality conditions within the same river zone. There were significant differences in pH, conductivity, TDS and cation concentrations between the Riviersonderend and Berg rivers. Conductivity, TDS, sodium, magnesium and calcium ion concentrations were higher in the Riviersonderend, and this might explain the changes in water chemistry below the IBT. However, it was likely that chemical conditions in the donor Theewaterskloof reservoir had a greater influence on the quality of transferred water than inter-catchment differences. Evidence for this was the higher pH recorded below the IBT, which contrasted with the significantly lower pH in the Riviersonderend. A chemical profile of the Theewaterskloof reservoir is necessary in order to investigate this suggestion.

The nature of the physical and chemical environment, or the habitat template, within a river reach has been suggested to be the main determinant of the biotic community associated with it (Hildrew & Giller, 1994). Therefore, any disturbance which alters the habitat template will lead to changes in the composition and functioning of the biotic community. For example, in their work on the upper Berg River, Harrison & Elsworth (1958) observed that the distribution of invertebrate communities in these river reaches was influenced by temperature and silt loads. An increase in these variables downstream led to a loss of upper river communities, and seasonal changes were marked by winter floods and summer desiccation. The transfer of water into the Berg during summer months effectively has reversed these seasonal influences. It is not surprising therefore, that the biotic communities inhabiting these reaches of the Berg River, have responded to this disturbance, resulting in an invertebrate community that differs markedly from that further upstream (Figures 5.15 and 5.16). The below-IBT summer invertebrate communities had significantly higher numbers of individuals and lower overall richness (Figures 13 and 14). Furthermore, a lower number of taxa were recorded below the IBT than above it. However, the greatest differences between above- and below-IBT invertebrate communities were expressed in terms of composition, rather than richness. Such a result is similar to that detected by O'Keeffe & de Moor (1988) in the Great Fish River, where taxon numbers dropped slightly after water transfer but where only 33% of taxa were common to communities before and after water transfer (see Section 2.5). The shifts in taxon dominance were centred on the Trichoptera (Hydropsychidae only) and the Diptera (Chironomidae and Simuliidae).

Shifts in the composition of the Berg River invertebrate communities occurred in most of the orders recorded in the river (Table 5.1), but especially in the Trichoptera and Ephemeroptera which collectively accounted for 34% of the dissimilarity between the invertebrate communities above and below the IBT (Table 5.2). Within all the main invertebrate orders there was a general loss of sensitive taxa below the IBT. These included the notonemourid Plecoptera, which are known to occur in waters of good quality (Davies & Day, 1998), and the majority of ephemeropteran families were more abundant at the upstream sites than below the IBT. Amongst these were the Heptageniidae, Ephemerellidae and Leptophlebiidae, all of which are generally found in undisturbed mountain streams (Table 5.1). Furthermore, the sites upstream of the IBT supported a wide diversity of trichopterans, including some rare taxa (Sericostomatidae, Barbarochthonidae and Petrothrincidae; Appendix 1.1), but very few of these taxa were found below the transfer. The Philopotamidae, Hydroptilidae, Leptoceridae and Ecnomidae are sensitive to flow and water quality conditions: the philopotamids and ecnomids generally prefer low flow or sheltered habitats, while the leptocerids and hydroptilids are sensitive to changes in water chemistry (Scott, 1985; de Moor, 1992); few individuals of these taxa were found below the IBT. Other taxa that were represented above the IBT, and in lower numbers below it, were the elmid (adults and larvae), hydraenid (adults), limnichid (larvae), helodid (larvae) and gyrinid (larvae) Coleoptera. All of these taxa are common in mountain streams of the Western Cape, and fairly sensitive to changes in water quality and quantity (Freshwater Research Unit, unpublished data). The megalopteran family Corydalidae and the dipteran family Athericidae are common predators in mountain streams (Ractliffe *et al.*, 1996; Davies & Day, 1998) of the Western Cape, but these were virtually absent from the site below the IBT. The absence of most of the hemipterans from the site below the IBT could be attributed to the turbulent conditions immediately downstream of the outlet, as the semi-aquatic adults of the families recorded in the Berg River all prefer standing or slow-flowing waters (e.g. Jacobs, 1985; Davies & Day, 1998).

On the other hand, aquatic invertebrate taxa that are known to be more tolerant of a range of physical and chemical conditions persisted at the site below the IBT. The caenid Ephemeroptera predominantly were found below the IBT, and this is a fairly hardy group commonly found in mud and organic debris in slow-flowing waters (e.g. Agnew, 1967, 1985). Within the Trichoptera, the Hydropsychidae appeared to flourish below the IBT (Table 5.1), exceeding 45 000 individuals m^{-2} in March 1994, while the Polycentropodidae were also recorded in higher numbers at this site. The Hydropsychidae are the most important trichopteran family in southern Africa, in terms of numbers of individuals in various habitats, especially fast-flowing water with good food supply (Scott, 1985; Dallas & Day, 1993). Hydropsychid larvae frequently occur in regulated rivers in greater numbers than other trichopteran families (e.g. Chutter, 1969, 1970; Mackay & Waters, 1986). Lastly, the Libellulidae are a robust family of predatory odonates that prefers fast-flowing water, and muddy or sandy substrates (Pinhey, 1985; B. Wilmot, Albany Museum, Grahamstown, unpublished article); this family occurred in high numbers in summer below the IBT, possibly due to the increased discharges and availability of prey, such as the hydropsychids, simuliids and zooplanktonic taxa.

A few taxa were recorded both above and below the IBT in summer. Amongst these were the baetid Ephemeroptera, a fairly widespread group (Davies & Day, 1998), that was abundant at all sites (Table 5.1). In addition, the dipteran families Chironomidae and Simuliidae are both ubiquitous groups, occurring in most riverine ecosystems. The Simuliidae occur on hard substrata in fast-flow conditions (e.g. de Moor, 1982), and this group was abundant at all sites throughout the summer.

It therefore appears that, despite an increase in flow magnitudes, rather than a decrease characteristic of impounded river reaches, the summer invertebrate community below the Berg River IBT resembled one typical of lake outlets. Lake outlet communities have been described by several authors, and have been recorded not only below impoundments (e.g. Mackay & Waters, 1986; McDowell & Naiman, 1986; Valett & Stanford, 1987; Richardson & Mackay, 1991, Malmqvist & Eriksson, 1995; Palmer & O'Keeffe, 1995) but also downstream of a surface-release hydro-electric power plan in Finland (Nyman, 1995). Lake outlets are habitats in transition between lentic and lotic conditions, inhabited by species that are adapted to lakes, plus those normally found in rivers, in addition to species which are specifically adapted to live below outlets (Malmqvist & Eriksson, 1995). It has been suggested that lotic communities downstream of lake outlets are resource-limited, particularly in terms of availability of food (e.g. Malmqvist & Eriksson, 1995), and thus increased concentrations of particulate organic matter which are often discharged from reservoirs, provide a ready food source for such communities.

Furthermore, the often predictable discharges below an impoundment provide the velocities that ensure a high availability of food. The filter-feeding guilds, for example, are particularly well-adapted to take advantage of the increased availability of organic matter (Ward & Stanford, 1983a; Brittain *et al.*, 1984; Mackay & Waters, 1986; Boon, 1988; Richardson & Mackay, 1991; Schlosser, 1992; Malmqvist & Eriksson, 1995; Palmer & O'Keeffe, 1995). The release of water from the Theewaterskloof impoundment into the Berg River has led to similar changes in water quality, with the introduction of dissolved and suspended organic loads, and fast, turbulent flow, all of which are characteristic of lake outlets. In addition to this, the transfer of vast quantities of zooplanktonic material, both live and fragmented, into the river (Table 5.1) provided a ready food source to those animals which could take advantage of the high-flow conditions below the IBT. Indeed, during the summer months, the Berg River reaches below the IBT were inhabited by high densities of hydropsychids and simuliids, which are collector-filterers (e.g. Merritt & Cummins, 1984).

The above two groups have been observed to interact fairly strongly (Hemphill, 1988, 1991; Morin, 1991; Hildrew & Giller, 1994). Furthermore, hydropsychids are known to take a longer time to establish than other filter-feeder groups such as the simuliids (e.g. Schlosser, 1992). In the Berg, the hydropsychids appeared to peak towards the end of summer (Figure 5.12), although numbers were fairly low in May 1994, while the simuliids were recorded in fairly high numbers throughout the summer (Figure 5.12). Once established, the hydropsychids are believed to outcompete the simuliids, by colonising all available space, and may also prey on the simuliid larvae (Hemphill, 1988; Hemphill & Cooper, 1983; Richardson & Mackay, 1991). The wetted rock surfaces of the Berg River, downstream of the IBT during summer, were almost covered by the silk and sand cases of the hydropsychids. Simuliids are also known to establish on clean rock surfaces (e.g. Boon, 1988; Nyman, 1995), and competition for space might explain the lower numbers of simuliids recorded in May 1994 at BR3, than in March of the same year. Such interactions may serve to keep simuliids in check throughout the year. Dense populations of simuliids have been recorded below impoundments in many rivers in southern Africa, and downstream of the Orange-Fish transfer in the Eastern Cape Province (see Section 2.5), where pest species were favoured by physico-chemical conditions brought about by the IBT (O'Keeffe & de Moor, 1988). This was not the case in the Berg River.

Although invertebrates were identified to family level for this study, an interesting discontinuity in the distribution of two species of Hydropsychidae (Trichoptera) was observed during analysis of the Berg River invertebrate data. The Hydropsychidae were represented by two species in the Berg, *Cheumatopsyche afra* Mosely and *C. thomasseti* Ulmer. Both species were observed at BR1 and BR2 during summer, but *C. afra* dominated these sites, while, at BR3 below the IBT, the only species observed was *C. thomasseti*, often in very large numbers (Table 5.1; Appendix 1.1). During winter, the few hydropsychids that were recorded at any of the sites were *C. afra*. This split in distribution appears to be due to the changes in water chemistry below the IBT. The genus *Cheumatopsyche* comprises predatory omnivores that build shelters with attached silk nets for catching algae and small invertebrates, such as the crustaceans Copepoda and Cladocera, and insect larvae such as the simuliids and ephemeropterans (Chutter, 1968, 1970; Scott, 1985; Richardson & Mackay, 1991). The genus is fairly tolerant of mild pollution, and is found in a wide variety of freshwater conditions, throughout southern Africa. *Cheumatopsyche thomasseti* is found in a variety of hydraulic conditions and is more tolerant of fluctuations in climatic, chemical and physical conditions than other species within the genus, such as *C. afra* (Scott, 1983). In the Berg River, the preferred habitat of *C. thomasseti* is reaches where the river runs through farmlands, and where stony runs and riffles alternate with sandy, shallow pools. In terms of water chemistry, this species can tolerate the higher turbidities, pH, dissolved solids and temperatures of the lower river. Thus, *C. thomasseti* occurs in the lower reaches of the Berg during summer months, while *C. afra* is found higher up the system, but they can co-occur, in smaller numbers, in the lower reaches during summer (Scott, 1983). Harrison & Elsworth (1958) recorded *C. afra* in waters where the pH ranged from 4.7 to 6.8, where TDS ranged from 19-78mg l^{-1}, and temperature from 9-29°C while the correspondihg conditions for *C. thomasseti* were, pH: 6.4-8.2, TDS: 45-584mg l^{-1} and temperature: 10-31.7°C. Thus the shift in chemical attributes towards downstream conditions, especially in terms of pH which increased in summer from a range of 4.5-5.5 above the IBT, to 6.3-7.1 below it, was favourable for the establishment and success of *C. thomasseti*. Furthermore, the introduction of large quantities of zooplanktonic material provided a ready food source for hydropsychids.

Fortunately for this study, there is an historical database for the Berg River, which allows some comparison. Harrison & Elsworth (1958) produced a fairly detailed description of the chemical and biological status of the Berg River as it was in 1951, and Scott (1958) worked on the Chironomidae of the system. The work of Harrison & Elsworth (1958) is of particular interest, as they sampled the same reach of the Berg River selected for this study, and, furthermore, the same biotope was sampled, that of the stones-in-current. Thus a comparison similar to that performed by O'Keeffe & de Moor (1988) in the Great Fish River, is possible (Table 5.3).

Some 25 taxa were common to both studies, while only one taxa, the dipteran family Tabanidae, was only recorded by Harrison & Elsworth (1958). Thus, very few taxa have been lost from the system over the past four decades. However, there were many new records when compared to the 1950s database (Table 5.3). Most of these disparities probably were due to differences in mesh sizes between the two studies. For this study, analyses included invertebrates retained by a 250µm mesh, while Harrison & Elsworth (1958) recorded organisms retained by a 1mm mesh. This would explain the absence of many of the smaller-sized taxa from Harrison & Elsworth's (1958) records, such as the Platyhelminthes, Acarina, collembolans, Hydrophilidae and Limnichidae (Coleoptera), and Ceratopogonidae (Diptera). Other groups not recorded in the 1950s were the rare mountain stream taxa, whose capture could be a chance event.

The main differences between the two sampling records are the large numbers of individuals of lentic groups recorded during this study (*Hydra* sp., Cladocera, Copepoda, dipteran Chaoboridae), and this supports the finding that these groups were a specific introduction from the reservoir source of the IBT, Theewaterskloof. It is therefore important to look at the Berg River invertebrate data, without the introduced lentic groups in order to gain a true reflection of the benthic community changes in the river. The dendrograms and MDS-ordination plots provide a useful overview of changes in the communities sampled in the Berg. The monthly diagrams show clear dissimilarities between invertebrate communities above and below the IBT, which gradually increased as the summer water-release period progressed (Figures 5.17 and 5.18), with the overall MDS-plots in Figures 5.15 and 5.16 indicating a discrete and dissimilar summer, below-IBT community. This result supports the conclusion made above that, in summer, the receiving reaches of the RBEGS support invertebrate communities that resemble those found below lake outlets, or those occurring further down the Berg River. The IBT effectively represents a discontinuity in the river, similar to an impoundment (e.g. Ward & Stanford, 1983a,b; Byren & Davies, 1989; Davies & Day, 1998).

Table 5.3 A comparison of the taxa recorded by Harrison & Elsworth (1958) from the upper reaches of the Berg River in the vicinity of the Berg River siphon, with those recorded in this study, above and below the Berg River IBT outlet, over the sampling period.

TAXON		Above	Below	1950s
Cnidaria				
Hydrozoa	*Hydra* sp.		+	
Platyhelminthes				
Turbellaria		+	+	
Nematoda		+	+	+
Annelida				
OLIGOCHAETA	Lumbriculidae	+	+	+
Arthropoda				
Acarina	Hydrachnellae	+	+	
Crustacea				
CLADOCERA		+	++	
COPEPODA		+	++	
OSTRACODA		+	+	
AMPHIPODA		+		+
Insecta				
COLLEMBOLA		+	+	
PLECOPTERA	Notonemouridae	+	+	+
EPHEMEROPTERA	Baetidae	+	+	+
	Heptageniidae	+	+	+
	Ephemerellidae	+	+	+
	Caenidae	+	+	
	Leptophlebiidae	+	+	+
TRICHOPTERA	Hydropsychidae	+	++	+
	Philopotamidae	+		+
	Leptoceridae	+	+	+
	Hydroptilidae	+	+	+
	Polycentropodidae	+	+	
	Ecnomidae	+	+	
	Sericostomatidae	+		
	Barbarochthonidae	+		+
	Petrothrincidae	+		+

TAXON continued...		Above	Below	1950s
ODONATA	Aeshnidae	+	+	
	Libellulidae	+	+	
	Chlorolestidae	+		
	Anisopteran juveniles			+
MEGALOPTERA	Corydalidae	+	+	+
LEPIDOPTERA	Nymphulidae	+		
HYMENOPTERA		+	+	
HEMIPTERA	Veliidae	+	+	
	Corixidae	+	+	
	Hebridae	+		
	Pleidae	+		
	Mesoveliidae	+		
COLEOPTERA	Elmidae	+	+	+
	Dryopidae	+	+	+
	Hydraenidae	+	+	+
	Hydrophilidae	+	+	
	Limnichidae	+	+	
	Helodidae	+	+	+
	Gyrinidae	+	+	
	?Hydrochidae	+		
	?Staphylinidae	+		
DIPTERA	Chironomidae	+	+	+
	Simuliidae	+	+	+
	Athericidae	+	+	+
	Empididae	+	+	
	Blephariceridae	+	+	+
	Ceratopogonidae	+	+	
	Stratiomyidae	+		
	Tipulidae	+		+
	Psychodidae	+	+	
	Chaoboridae		+	
	Dixidae	+		
	"coneworm"	+		**Tabanidae**

Unlike most impounded rivers, however, the receiving reaches of the Berg River had an opportunity to recover, over the winter, no-release months. This is clear from the increased similarities between the invertebrate communities sampled above and below the IBT during winter (Figures 5.17 and 5.18). The recovery towards upstream conditions reached its maximum towards the end of winter (October 1994), when some between-site similarities were greater than within site values (Figures 5.17 and 5.18). The disturbance created by the commencement of water transfer, however, served to drive the invertebrate communities away from "natural" composition.

Invertebrate stream communities are believed to recover fairly rapidly from small-scale disturbance (e.g. Townsend, 1989). Further, these communities appear to be fairly resistant, at a larger scale of disturbance (Hildrew & Giller, 1994). In order to assess the ecological effects of the IBT on the receiving reaches of the Berg, it is necessary to describe the nature and scale of the disturbance it provides. Spatially, the disturbance appeared to be fairly localised, while temporally, the disturbance was almost constant over the summer and ceased in winter. On an annual basis, the invertebrate communities of these reaches of the Berg were fairly resistant to the disturbance of the IBT, as, each winter, the river has a chance to recover; while on a seasonal basis, the communities appear to be resilient, recovering by late winter. This is similar to the findings of Brittain *et al.* (1984), which are described in detail in Section 2.4.1. They found that the invertebrate communities below the donor impoundment of the transfer recovered fairly rapidly in summer when discharges increased, and after spring spates had facilitated the recolonisation of river reaches below the dam.

The concept of recovery distance was not addressed by this study, but this forms a component of the wider WRC research project. Data are currently being collected from a sample site situated a few hundred metres downstream of BR3, for comparisons between invertebrate communities at these two sites below the IBT.

The invertebrate community composition in the Riviersonderend was found to be significantly different from that in the Berg. However, it was felt that this had a minimal effect on the invertebrate communities downstream of the IBT as it is unlikely that lotic invertebrates from the Riviersonderend would survive the lentic conditions of Theewaterskloof. This conclusion was supported by the exclusive transfer of lake-adapted taxa from the reservoir, into the Berg River.

A concern which was not addressed during the fieldwork component of this project, was the transfer of cyanophyte exudates from Theewaterskloof into the Berg River. The growth of cyanophytes, commonly of the genus *Anabaena*, in Theewaterskloof is being encouraged by fertiliser-enriched runoff from farms within its catchment, increasing the nutrient loading of the reservoir and leading to the development of blooms, usually during the summer months (Harding, 1992, 1997). The Cape Metropolitan Council, responsible for water quality in the Cape Metropolitan Area (CMA), has received complaints from a rainbow trout farmer supplied by the RBEGS, that the transferred geosmin accumulates in the flesh of the fish, imparting an unpleasant odour and taste. This has resulted in significant economic losses. Furthermore, residents of the CMA often submit complaints about the taste of water transferred to the city during summer (Dr W.R. Harding, Cape Metropolitan Council, Department of Scientific Services, personal communication). The transfer of *geosmin* points towards the possible, and more harmful, introduction of toxic cyanophyte exudates, which could have severe ecological, social and economic implications.

Of additional importance in ecological terms, and a feature not found where rivers are impounded, is the transfer and mixing of previously isolated biota between catchments, and thus the mixing of genetic material. This occurred in the Eastern Cape Province, where at least four species appear to have been transferred into the Great Fish River from the Orange River *via* the Orange/Fish tunnel of the ORP (see Section 2.4.2). The long-term effects of these fish transfers have not been assessed, but in many cases the genetic differences between populations that occur in neighbouring catchments, or subcatchments, have been shown to be significant. For example, Hughes *et al.* (1996) studied populations of the freshwater shrimp, *Caradina zebra* (Decapoda, Atyidae) which occurs in rivers in Australia. They found a high level of genetic differentiation between populations occurring in sub-catchments of the Tully River, and similar differences were demonstrated between the Tully and Herbert rivers. A transfer of water has been proposed between the Tully and the Herbert systems, which will thus threaten the genetic integrity of the highly differentiated populations of *C. zebra* (Hughes *et al.*, 1996).

Furthermore, comprehensive legislation is in place in many of the states and provinces of the United States and Canada, in order to regulate the transfer of organisms between catchments (e.g. Wingate, 1991; Leach & Lewis, 1991; Kapuscinski & Hallerman, 1991). The invasion of many river systems in North America by the organisms such as Zebra Mussel (*Dreissena polymorpha*) and carp (*Cyprinus carpio*) has been a critical incentive for the development of these laws, however, such legislation will be severely compromised if there is a lack of similar controls over the transfer of species through IBTs. Globally there are the additional threats of the transfer of parasites, and disease-causing agents (human and livestock) and their vectors (e.g. Pitchford & Visser, 1975; Arthur *et al.*; 1976; Pretorius *et al.*, 1976; Arai & Mudry; 1983; Greer, 1983; Day, 1985). In southern Africa, introductions of exotic and invasive plant species, such as the floating macrophytes, water hyacinth (*Eichhornia crassipes*) and Kariba Weed (*Salvinia molesta*), and the invasive riparian tree species, *Sesbania punicea*, are becoming increasingly problematic.

It is clear from this project that organisms, such as the zooplanktonic taxa transferred from Theewaterskloof, can survive the rigours of transfer through tunnels and pipelines. Thus, the transfer of other biota, including exotic and invasive species, is a likely consequence of IBTs. This is an area that requires urgent and comprehensive attention.

5.5 Conclusion

It can be concluded that it is the manner in which IBTs differ from river impoundment that requires most caution when planning and assessing IBTs. The transfer of water into the Berg has resulted in significant and possibly problematic alterations in water quality, and the introduction of lentic taxa. Despite the technology associated with IBT schemes in South Africa, the transfer of organisms and water quality problems appears to be difficult to avoid. It is of critical importance, therefore, to gather detailed information on the donor system when assessing the effects of an IBT.

Furthermore, this study indicates that some of the major effects of IBTs on recipient rivers systems appear to be fairly similar to those resulting from river impoundment, in the case where water is transferred from a donor reservoir. This is despite the fact that receiving reaches experience increased rather than decreased discharges. This conclusion has implications for the feasibility assessment of IBT schemes. Currently, the instream flow requirements (IFR) methodology is being used to determine the flow needs of rivers throughout the country which are regulated or due to be regulated by impoundments (e.g. King & Tharme, 1994; King *et al.*, 1995). This methodology should be extended to include river reaches which are proposed to receive IBTs from donor impoundments.

In the absence of knowledge of the consequences of such transfers, the sustainable use of water within catchments should be maximised before any consideration of importation of water from other catchments.

Chapter 6 IBT Planning and Management

"A city may be moved, but not a well."
I Ching (the most ancient Chinese book of wisdom and philosophy)

6.1 Introduction

In the vast majority of cases, the initial emphasis in planning IBT projects has been placed on the engineering and economic aspects associated with them. Most water transfers have been perceived by governments and engineers as technological and economic challenges and, consequently, little or no environmental planning has accompanied most of the technical and economic feasibility studies that are carried out as a matter of course (e.g. Greer, 1983; Day, 1985; Shiklomanov, 1985; Platt, 1995). However, and inevitably, many problems have arisen from such limited planning and hasty development, many of which are associated with the environmental and socio-economic consequences of water transfers. Public, and often political opposition to IBTs is on the increase, and the need for more sensitive planning is rapidly becoming apparent worldwide (Ortalano, 1978; Day, 1985).

This chapter examines current IBT planning in various countries around the world, and summarises some of the problems and conflicts that have arisen, and the legislation that has been developed in association with the planning or construction of IBTs. Lastly, recommendations are made for improved and more comprehensive planning of IBTs in South Africa. These, however, should not be considered as a complete set of recommendations, for more are likely to arise after the completion of the Water Research Commission research contract, early in 1999. The aim of the chapter, therefore, is to place the conclusions of this project in the context of water-resources planning and development in South Africa. A substantial proportion of the information in this chapter was taken from a report written for the Department of Water Affairs and Forestry, during the water law review process in South Africa (Snaddon & Davies, 1997).

6.2 Problems associated with IBT Planning

Many of the problems associated with the implementation of water transfer projects relate to the planning phases of a scheme (e.g. Greer, 1983; Cox *et al.*, 1985; Shiklomanov, 1985), such as:

- a lack of comprehensive environmental assessments;
- subordination of environmental assessments to the engineering and economic aspects of IBTs;
- a lack of co-ordination between environmental assessments (where they occur), and other aspects of IBT planning, such as the technical and economic studies; and
- a general geographical bias towards recipient catchment/s, at the expense of donor system/s, while transfer routes are effectively ignored.

The first two of these problems were evident during the planning of the Texas Water Plan (see Section 2.4.4), in the USA, which was eventually abandoned. In this case, primary attention was given to the engineering and economic aspects of the scheme. Environmental studies were then initiated in response to completion of the engineering and economic feasibility phases, while a third step consisted of a revision of the initial plan to accommodate those environmental concerns that were perceived more as political obstacles than as ecologically necessary modifications (Greer, 1983). The Texas Water Plan also provides an example of the fourth problem listed above: water demand, economic and population projections were calculated for Texas, but not for the donor system, the Mississippi Basin (Greer, 1983).

Locally, evaluation of the Western Cape System Analysis (WCSA) uncovered the foolishness of excluding one of the Western Cape's major river basins, the Breede River catchment, from such a system analysis. Several of the donor reservoirs for the Cape Metropolitan Area are situated on the Breede River and its tributaries, and yet the consequences of these impoundments have not been assessed within the context of the WCSA (Zille Shandler Associates, 1996). Communities, and especially the agricultural community, of the Breede River catchment have called for a moratorium on damming of their rivers until a Breede River System Analysis has been completed.

Several IBT schemes have been abandoned or modified as problems have become apparent, while others have been postponed as the result of a lack of adequate environmental assessments during the feasibility planning stages (e.g. Guiver, 1976; Greer, 1983; Day, 1985; Shiklomanov, 1985). These include (see Chapter 2):

- The transfer from Lake Suldalsvatnet to Hylsfjorden in Norway ceases during June and July during salmon runs (Nordeng, 1977; Tòndevold, 1984).
- The McGregor Diversion, Canada, was shelved in 1978 as a result of the possible transfer of both fish and parasites from the Fraser River to the Arctic-draining Peace River (Arai & Mudry, 1983; Day, 1985).
- Flow into the Nechako River from the Kemano Diversion, Canada, had to be ensured by a British Columbia Supreme Court injunction, in order to protect Sockeye migration routes (Day, 1985).
- The Ely Ouse to Essex Scheme, United Kingdom, was temporarily halted due to the transfer of algal blooms from the point of abstraction (Guiver, 1976).
- The proposed diversion of Siberian rivers, the Ob and Irtyish, to Soviet Central Asia, was "indefinitely postponed" in 1986, as a result of pressure from environmentalists, and economists (Voropaev & Velikanov, 1985).
- The North American Water and Power Alliance (NAWAPA) scheme was shelved due to the numerous vocal and politically inspired public objections to the economic and environmental effects of the project (Micklin, 1976; Shiklomanov, 1985).

In the past, there has been a disturbing lack of transparency in IBT planning. The people who ultimately pay for these schemes, and whose environment is due to be altered, are often not informed of the plans, and are seldom given an opportunity to take part in the planning process. A comprehensive, and participatory process needs to be in place, supported by appropriate legislation, and designed to ensure adequate consultation in both the donor and recipient catchments, and communities along the transfer route(s) (e.g. Ortalano, 1978). For example, despite a number of calls for action (e.g. Petitjean & Davies, 1988a,b; Davies *et al.*, 1992), it was only in mid-1996 that a wide range of organisations and individuals affected by the Lesotho Highlands Water Project were brought together to discuss the project and its consequences (Group for Environmental Monitoring, 1996). Such a workshop should have taken place at the feasibility stage of the planning process (the early 1980s) and not after commencement of filling of Katse Dam (Phase 1a of the Project), in October 1995.

In this context, the decision to transfer water should not be made by engineers, or water resource managers alone. If a comprehensive and inclusive consultation process were followed, the decision would be reached over time, with the responsibility for determining the optimal solution spread throughout the communities to be affected by the scheme. This would avoid the situation where distant individuals make decisions for local communities, rather then allowing them that power.

6.3 Legislation associated with IBTs

Many authors have articulated the need to improve policy surrounding IBTs, and the institutions responsible for the planning and implementation of transfer schemes (e.g. Cox *et al.*, 1985; Day, 1985). This section briefly explores legislation specific to IBTs, in a variety of countries.

- Probably the earliest IBT proposal in England was rejected and shelved during its passage through the House of Commons, as a result of a public outcry concerning the consequences of the scheme for the donor system, the River Thames (Sheail, 1984). The 1855 Bill proposal for the transfer of one million gallons of springwater a day (4.5×10^3 m^3 d^{-1}) from the headwaters of the Thames to the Severn River Valley, raised fears that since none of the transferred water would return to the Thames, not only would the water level in the Thames drop, making navigation more difficult, but also that reduced flow would result in less efficient dilution of effluents, particularly sewage, which already entered the river. Interestingly, this was probably one of the earliest public objections to an IBT on the grounds of the potential environmental effects of the scheme. On a suggestion made by a Welsh Member of Parliament, during a parliamentary sitting in the late 1890s, that the central industrial complex of Birmingham and surrounds should financially compensate Wales for the water transferred to that region, the Right Honourable Member of Parliament for Birmingham, Sir Joseph Chamberlain, stated that the water "...comes from Heaven and goes to the sea, and no more belongs to Welshmen than anybody else who stands in need of it." Fortunately this attitude has changed and the "watershed doctrine" has since gained recognition, where those parties which are adversely affected by transfers, and other water developments, are compensated wherever possible for their losses.

- In the United States, detailed economic analyses of IBTs has been obligatory since the early 1930s, but evaluation of environmental effects has only been required in legislation since 1969 (Ortalano, 1978; Shiklomanov, 1985).

- In Massachusetts, the Interbasin Transfer Act (Massachusetts Laws 1983) provides a safeguard against unnecessary water transfers (Platt, 1995). The legislation stipulates that the minimum flow requirements (unspecified, but presumably "ecological requirements"; Platt, 1995) of the donor basin are met, and that the applicant basin must prove that it has used "*every reasonable method to develop and conserve its own in-basin sources before looking outward.*" (Massachusetts Water Resources Authority, 1990).

- In 1968, the United States Congress announced a ten-year moratorium on the investigation of any diversions from the Columbia River (Johnson, 1971). This was in order to give the Pacific Northwest (the donor) time to assess present and future water needs, and to determine whether or not there was a surplus of water available for export. During the ten years, one of the issues that was explored was the protection of the "area-of-origin".

- The "appropriation" system of the Western United States has never made any distinction between the use of water within or outside a catchment (Johnson, 1971; M. Meador, United States Geological Survey, Raleigh, North Carolina, personal communication). As a result, communities in many donor catchments have expressed concern over the lack of protection for their catchments. A variety of State statutes (California, Colorado, Nebraska, Texas and Oklahoma) and one inter-state compact (made in 1929, between the upper and lower basin states of the Colorado River), were created to give a degree of protection to catchments of origin (donors). In general, this legislation serves to protect, or to reserve water for use within the donor catchments, and also to guide the design of IBT projects in order to ensure that a donor catchment is left in a better position in terms of water-supply infrastructure than it was before diversion. This legislation is described in greater detail by Johnson (1971) and Cox (1991).

- In Canada, national water resources are protected by a federal policy preventing water transfer to the United States, until the national supply-demand scenario has adequately been addressed (Howe, 1978; Day, 1985). Towards the mid-1960s, Canada took a strong stance in opposing any transfer of water to the USA, stating that "...Canadian water is for Canadian development." (Howe, 1978). It is likely, however, that this situation will change if the North American Free Trade Agreement (NAFTA) is implemented. This legislation will remove virtually all of the international political obstacles to large-scale water transfers, and make it difficult to prevent the construction of projects like NAWAPA through national or provincial regulations.

- The governments of the states of the Northwest Pacific in the United States, accompanied by a powerful public lobby, have consistently opposed the transfer of water out of the Columbia River Basin (Howe, 1978). Senator Jackson of Washington has stated: "The people of the Northwest deeply believe that before any other region asks for a study of the diversion of the Columbia River, such region must first establish that it actually needs additional water....Can sufficient water be secured through conservation and reuse?...".

- The new Water Policy in South Africa states that after providing for the Reserve - this is an amount of water to be preserved for basic human requirements and the ecological requirements of the freshwater source - and international obligations, the basis for granting a licence to use water available in the area, will be to achieve beneficial use in the public interest which will include consideration of the need for programmes of corrective action. Any allocation of water for transfers to other basins will be subject to conditions which ensure that reasonable needs in the donor catchment are met (Department of Water Affairs and Forestry, 1997).

- Under the new Water Law, inter-basin transfers will have to meet special planning requirements and implementation procedures, which must involve agencies from both the donor and recipient catchments. Catchments to which water will be transferred will have to show that the water currently available in that catchment is being optimally used and that reasonable measures to conserve water are in force (Department of Water Affairs and Forestry, 1997).

6.5 Recommendations for IBT Planning and Management

In this final section, a list of recommendations is provided that should be considered during the planning and management of IBT schemes. These recommendations are based on the literature and on the results of this study. They are not listed in any order of priority.

- It is clear from the literature presented in Chapter 2 and the results discussed in Chapter 5 that the ecological consequences of inter-basin water transfers are such that great caution is warranted. Data are scarce, but the "precautionary principle" (Department of Environmental Affairs and Tourism, Environmental Policy Discussion Document, 1996) needs to be applied to water-resources planning, allowing for the gathering of data before the feasibility stage of any IBT project.
- The environmental aspects of IBTs should not be seen as subordinate to the technical and economic consequences.
- The needs (environmental, social and economic) of all basins concerned in any IBT must be given equal weighting, and must be assessed to the same level for each basin.
- There are already a large number of extant IBTs in our country, many of which are having negative effects on donors, recipients or transfer routes. Extant schemes should be re-assessed in terms of their effects, so that detrimental impacts can be minimised through mitigation.
- Water-demand management, as an holistic, multi-faceted and multi-disciplinary approach, must be the strategy underpinning all water resource management. Currently, there are few data that give evidence of effective water-demand management in South Africa. For example, water-resources planners in South Africa claim that only 10% of water use can be saved if water-demand procedures were to be implemented (Eric van der Berg, Project Planning, Department of Water Affairs and Forestry, personal communication). However, in Cape Town, over 20% of water provided to the city is known as "unaccounted for water" (Clayton, 1994, 1996), which is defined as water that is supplied but not billed. A substantial proportion of this water is lost through leaks. A reduction in this loss could postpone the next proposed IBT scheme to Cape Town.

- Greater public participation is required during the planning of IBTs. This has bearing on the environmental consequences of IBTs, as social and environmental issues are intimately linked. Again, the institutional framework for the effective management of this type of participation is crucial to the process. There are examples where public participation is limited to conflict resolution and/or mediation (e.g. Cox *et al.*, 1985), but, in most instances, true consultation is favoured (e.g. Platt, 1995). The funding for such participation is usually provided by the developer (e.g. in Lesotho, the Lesotho Highlands Development Authority; in South Africa, Department of Water Affairs and Forestry).

- The land-use implications of IBTs, such as effects on soils, waterlogging and groundwater levels can be severe and thus, the regulation and management of land use should be integrated into IBT planning. There is a need for holistic planning in South Africa. System analyses, rather than piecemeal solutions, should be initiated for all regions of the country.

- Long-term policy development is required for potential donors, recipients and transfer routes, in order to give some systems total protection.

- Monitoring the water quality in transfer tunnels during periods when an IBT is not active, is important. Equally important is the prevention of stagnant water flushing into recipient rivers.

- The transfer and mixing of previously isolated biota between catchments, and thus the mixing of genetic material and the transfer of exotic and invasive fauna and flora, are likely consequences of IBTs. This requires great caution and extensive investigation during the assessment of the feasibility of such schemes. In addition, it is recommended that further work should be done on the ecological effects of the mixing of biota between catchments.

- Similarly, the transfer of water quality problems, such as cyanophyte blooms, between catchments, should be avoided.

- The results of this study indicate that the ecological effects of an IBT from a donor impoundment to a river, are similar to those occurring downstream of an impoundment. Hence, the recommendation is made that such IBT schemes be assessed using methodologies similar to those utilised in the assessment of river impoundment. For example, instream flow requirement methodologies (IFR) should be applied to cases where water is transferred into a catchment, thereby increasing flow, rather than the usual case of decreased flow through water abstraction, as a result of impoundment.

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Appendix 1.1 Invertebrate data from all sites on the Berg River and Riviersonderend, over the sampling period. Three replicate samples were taken at most sites, and averaged to give an abundance figure for each site. The data are given as individuals m^{-2}.

TAXON			March 1994					May 1994				
			BR1	BR2	BR3	RSE1	RSE2	BR1	BR2	BR3	RSE1	RSE2
Cnidaria												
Hydrozoa		*Hydra* sp.	0	0	19213	0	0	0	0	193	0	0
Platyhelminthes												
Turbellaria			5	0	47	7	140	0	0	0	17	37
Nematoda			0	0	13	7	0	3	13	0	3	13
Annelida												
OLIGOCHAETA	Lumbriculidae		10	273	0	183	1930	467	33	0	243	167
Arthropoda												
Acarina	Hydrachnellae		860	613	467	530	907	150	177	133	150	143
Crustacea												
CLADOCERA			0	43	10153	3	3	200	40	11367	20	10
COPEPODA			0	13	200937	0	50	3	7	15057	27	33
OSTRACODA			65	0	43	0	0	0	0	7	0	0
AMPHIPODA			0	0	0	0	0	0	0	0	0	3
Insecta												
COLLEMBOLA			0	3	3	0	0	50	13	10	7	0
PLECOPTERA	Notonemouridae		100	83	0	43	117	23	30	0	17	20
EPHEMEROPTERA	Baetidae		6170	12327	3267	2383	1430	4570	5217	3500	1093	567
	Heptageniidae		55	673	0	7	0	27	213	0	0	0
	Ephemerellidae		65	83	7	243	263	50	343	0	67	57
	Caenidae		0	0	83	3	0	0	0	0	0	7
	Leptophlebiidae		990	1657	10	190	1647	917	703	10	53	190
TRICHOPTERA	Hydropsychidae		305	223	46993	23	3	173	100	5547	57	0
	Philopotamidae		35	150	0	0	3	970	97	0	33	3
	Leptoceridae		290	323	13	257	1310	473	223	0	137	47
	Hydroptilidae		115	100	20	47	33	17	10	7	27	10
	Polycentropodidae		0	17	7	3	0	7	0	53	0	0
	Ecnomidae		10	63	0	153	63	150	53	0	10	3
	Sericostomatidae		0	3	0	0	0	0	0	0	0	3
	Barbarochthonidae		0	0	0	7	7	0	0	0	13	0
	Petrothrincidae		0	0	0	0	0	0	0	0	0	0
	pupae		0	0	170	0	0	0	0	3	0	0
ODONATA	Aeshnidae		15	0	3	0	7	0	7	3	0	0
	Libellulidae		0	0	193	0	0	7	0	27	0	0
	Chlorolestidae		0	7	0	0	0	0	0	0	0	0
	Gomphidae		0	0	0	0	0	0	0	0	0	0
	Anisopteran juveniles		0	0	0	0	0	0	0	0	0	0
MEGALOPTERA	Corydalidae		5	13	13	0	33	27	7	0	30	7
LEPIDOPTERA	Nymphulidae		5	0	0	0	0	0	0	0	0	0
HYMENOPTERA			0	0	0	0	0	0	3	0	0	0

Appendix 1 continued...

TAXON			March 1994					May 1994				
			BR1	BR2	BR3	RSE1	RSE2	BR1	BR2	BR3	RSE1	RSE2
HEMIPTERA	Veliidae		10	0	0	3	0	0	3	0	0	0
	Corixidae		0	0	0	0	0	0	0	0	0	0
	Hebridae		0	0	0	0	0	0	0	0	0	0
	Pleidae		0	0	0	0	0	0	0	0	0	0
	Mesoveliidae		0	0	0	0	0	0	0	0	0	0
COLEOPTERA	Elmidae	adults	90	117	3	3	50	113	40	10	27	7
	Elmidae	larvae	2045	2140	350	767	1437	1293	637	233	230	120
	Dryopidae	adults	0	3	0	0	0	3	0	0	10	0
	Dryopidae	larvae	0	0	0	0	0	0	0	0	7	0
	Hydraenidae	adults	275	790	63	0	47	790	137	110	27	3
	Hydraenidae	larvae	0	0	0	0	0	7	63	13	7	0
	Hydrophilidae	adults	0	3	0	0	0	0	0	0	3	3
	Hydrophilidae	larvae	0	0	0	0	0	0	0	0	0	0
	Limnichidae		5	0	0	7	0	7	0	0	3	0
	Helodidae		10	27	3	63	753	117	93	10	143	343
	Gyrinidae		0	10	0	0	0	10	3	0	0	0
	?Hydrochidae		0	0	0	0	0	0	0	0	0	0
	?Staphylinidae		0	0	0	0	0	0	0	0	0	0
DIPTERA	Chironomidae		33950	22147	19680	11150	23767	14993	14820	16517	4317	1513
	Simuliidae		12690	11550	993	127	253	970	1040	4180	277	100
	Athericidae		100	103	63	30	310	33	53	17	13	10
	Empididae		10	0	33	0	3	3	13	13	7	3
	Blephariceridae		10	0	0	0	0	0	0	7	0	0
	Ceratopogonidae		5	3	0	30	20	0	0	0	13	7
	Stratiomyidae		5	0	0	0	0	0	0	0	0	0
	Tipulidae		0	0	0	0	0	0	0	0	0	0
	Psychodidae		0	0	0	0	0	0	0	0	0	0
	Chaoboridae		0	0	823	0	0	0	0	90	0	0
	Dixidae		0	0	0	0	0	0	0	0	0	0
	pupae		65	57	97	203	290	93	70	80	30	20
	"coneworm"		5	0	0	0	0	0	0	0	0	0

Appendix 1 continued...

TAXON			July 1994					October 1994				
			BR1	BR2	BR3	RSE1	RSE2	BR1	BR2	BR3	RSE1	RSE2
Cnidaria												
Hydrozoa		*Hydra* sp.	0	0	0	0	0	0	0	0	0	0
Platyhelminthes												
Turbellaria			0	0	0	0	0	0	0	0	7	0
Nematoda			30	80	67	3	0	47	73	13	3	3
Annelida												
OLIGOCHAETA	Lumbriculidae		47	157	57	157	20	233	170	133	43	10
Arthropoda												
Acarina	Hydrachnellae		77	167	90	47	10	140	173	37	53	70
Crustacea												
CLADOCERA			37	47	37	0	0	0	20	3	0	0
COPEPODA			47	153	53	3	23	3	20	7	0	0
OSTRACODA			7	0	13	0	3	7	0	13	0	0
AMPHIPODA			3	3	0	0	0	0	0	0	0	0
Insecta												
COLLEMBOLA			570	697	763	7	0	100	113	67	3	3
PLECOPTERA	Notonemouridae		170	217	80	87	43	197	240	80	83	110
EPHEMEROPTERA	Baetidae		197	167	197	60	37	7997	8907	4027	563	397
	Heptageniidae		0	0	3	0	3	3	10	0	0	0
	Ephemerellidae		27	53	33	17	77	73	103	57	80	87
	Caenidae		0	13	0	10	0	0	0	3	0	3
	Leptophlebiidae		20	30	20	7	27	83	27	20	23	43
TRICHOPTERA	Hydropsychidae		7	7	17	0	0	0	0	10	0	0
	Philopotamidae		20	10	0	13	0	0	0	0	3	3
	Leptoceridae		10	10	0	7	7	10	20	20	0	20
	Hydroptilidae		0	30	3	13	13	13	3	3	13	13
	Polycentropodidae		0	0	0	0	0	0	0	0	0	0
	Ecnomidae		0	3	0	0	0	23	7	0	3	10
	Sericostomatidae		0	0	0	0	0	0	0	0	0	10
	Barbarochthonidae		0	0	0	3	0	0	0	0	0	0
	Petrothrincidae		0	0	0	3	0	0	0	0	0	0
	pupae		0	0	0	0	0	0	0	0	0	0
ODONATA	Aeshnidae		0	0	0	0	0	3	0	0	3	0
	Libellulidae		0	0	0	0	0	0	0	0	0	0
	Chlorolestidae		0	0	0	0	0	0	0	0	0	0
	Gomphidae		0	0	0	0	0	0	0	0	0	0
	Anisopteran juveniles		0	0	0	0	0	0	0	0	0	0
MEGALOPTERA	Corydalidae		0	0	3	0	0	3	0	7	0	3
LEPIDOPTERA	Nymphulidae		3	3	0	0	3	0	0	0	0	0
HYMENOPTERA			10	0	0	3	3	83	80	27	0	40

Appendix 1 continued...

TAXON			July 1994					October 1994				
			BR1	BR2	BR3	RSE1	RSE2	BR1	BR2	BR3	RSE1	RSE2
HEMIPTERA	Veliidae		0	0	0	0	0	0	0	0	0	0
	Corixidae		0	3	0	0	0	0	0	0	0	0
	Hebridae		0	0	0	0	0	37	0	0	0	0
	Pleidae		0	0	0	3	0	0	0	0	0	0
	Mesoveliidae		0	0	0	0	0	0	0	0	0	0
COLEOPTERA	Elmidae	adults	10	33	0	3	0	10	3	10	0	0
	Elmidae	larvae	83	103	53	63	47	127	180	47	43	47
	Dryopidae	adults	3	3	0	0	0	0	0	0	0	10
	Dryopidae	larvae	0	0	0	0	0	0	0	0	0	0
	Hydraenidae	adults	7	73	10	0	3	17	30	13	0	3
	Hydraenidae	larvae	13	50	53	0	0	30	43	7	0	0
	Hydrophilidae	adults	3	0	0	0	3	0	0	0	0	0
	Hydrophilidae	larvae	0	27	3	0	0	0	0	0	0	0
	Limnichidae		0	0	0	0	0	0	0	0	0	3
	Helodidae		50	83	17	27	93	150	197	33	57	60
	Gyrinidae		0	0	0	0	0	3	0	0	0	0
	?Hydrochidae		0	0	0	0	0	7	0	0	0	0
	?Staphylinidae		3	0	0	0	0	0	0	0	0	0
DIPTERA	Chironomidae		3133	5247	2790	1537	1447	8757	7683	5707	2173	1207
	Simuliidae		97	347	137	33	50	210	73	267	117	80
	Athericidae		0	0	0	0	0	0	0	0	0	0
	Empididae		0	7	0	7	3	0	0	0	13	0
	Blephariceridae		0	0	10	0	0	0	0	0	0	0
	Ceratopogonidae		23	47	33	13	0	10	10	0	0	0
	Stratiomyidae		0	0	0	0	0	0	0	0	0	0
	Tipulidae		0	3	0	0	0	3	0	0	0	0
	Psychodidae		0	0	0	0	0	3	0	3	0	0
	Chaoboridae		0	0	0	0	0	0	0	0	0	0
	Dixidae		0	0	0	0	0	0	0	0	0	0
	pupae		3	10	13	3	3	133	100	133	57	17
	"coneworm"		0	0	0	0	0	0	0	0	0	0

Appendix 1 continued...

TAXON			November 1994					January 1995				
			BR1	BR2	BR3	RSE1	RSE2	BR1	BR2	BR3	RSE1	RSE2
Cnidaria												
Hydrozoa		*Hydra* sp.	0	0	10	0	0	0	0	6200	0	0
Platyhelminthes												
Turbellaria			0	0	0	3	3	0	20	0	23	77
Nematoda			0	0	0	13	0	25	0	5	0	107
Annelida												
OLIGOCHAETA	Lumbriculidae		247	380	190	587	33	205	40	3005	170	810
Arthropoda												
Acarina	Hydrachnellae		647	363	110	933	320	375	40	90	473	483
Crustacea												
CLADOCERA			133	233	314400	17	0	275	30	51040	0	3
COPEPODA			0	47	285547	53	0	5	30	77035	13	17
OSTRACODA			0	13	27	0	0	0	0	120	0	0
AMPHIPODA			0	0	0	0	0	0	0	0	0	3
Insecta												
COLLEMBOLA			13	0	13	0	0	0	0	0	0	0
PLECOPTERA	Notonemouridae		240	113	207	340	77	45	0	0	33	47
EPHEMEROPTERA	Baetidae		63867	86113	27183	5807	2777	2450	870	870	1580	1780
	Heptageniidae		20	30	3	0	0	315	150	0	0	0
	Ephemerellidae		7	40	0	133	30	215	110	5	327	217
	Caenidae		13	0	27	0	0	0	0	580	0	0
	Leptophlebiidae		320	227	43	857	107	615	40	5	940	357
TRICHOPTERA	Hydropsychidae		70	27	87	150	0	435	0	920	0	27
	Philopotamidae		23	3	0	3	0	50	20	0	0	0
	Leptoceridae		23	40	0	450	50	325	90	20	583	270
	Hydroptilidae		0	0	0	13	10	10	0	105	27	23
	Polycentropodidae		0	0	13	0	3	0	0	5	0	3
	Ecnomidae		33	27	3	7	3	5	30	10	100	23
	Sericostomatidae		0	0	0	13	0	0	0	0	3	0
	Barbarochthonidae		0	0	0	0	3	0	0	0	0	3
	Petrothrincidae		0	3	0	0	0	0	0	0	0	0
	pupae		0	0	0	0	0	0	0	20	0	0
ODONATA	Aeshnidae		10	0	13	0	3	25	0	5	7	0
	Libellulidae		0	0	0	0	0	0	0	15	0	0
	Chlorolestidae		0	0	0	0	0	0	0	0	0	0
	Gomphidae		0	0	0	0	0	0	0	0	3	0
	Anisopteran juveniles		0	0	0	0	0	15	0	0	0	3
MEGALOPTERA	Corydalidae		13	7	0	0	0	5	0	0	7	20
LEPIDOPTERA	Nymphulidae		0	3	0	40	43	0	0	0	10	0
HYMENOPTERA			3	0	0	7	3	0	0	0	0	0

Appendix 1 continued...

TAXON			November 1994					January 1995				
			BR1	BR2	BR3	RSE1	RSE2	BR1	BR2	BR3	RSE1	RSE2
HEMIPTERA	Veliidae		3	0	0	0	0	0	0	0	0	0
	Corixidae		0	0	0	0	0	0	0	0	0	0
	Hebridae		0	3	0	0	0	0	0	0	0	0
	Pleidae		0	0	0	0	0	0	0	0	0	0
	Mesoveliidae		0	0	0	0	0	0	0	0	3	0
COLEOPTERA	Elmidae	adults	150	147	127	153	60	110	80	0	27	47
	Elmidae	larvae	783	457	177	400	500	355	120	120	427	413
	Dryopidae	adults	0	0	0	0	0	0	0	0	0	0
	Dryopidae	larvae	0	0	0	0	0	0	0	0	0	0
	Hydraenidae	adults	390	167	153	40	10	480	10	20	13	33
	Hydraenidae	larvae	20	0	0	0	0	0	0	0	0	0
	Hydrophilidae	adults	0	0	0	0	0	0	0	0	0	0
	Hydrophilidae	larvae	0	0	0	0	0	0	0	0	0	0
	Limnichidae		3	0	0	0	0	0	0	0	0	0
	Helodidae		130	157	30	707	107	45	10	0	100	87
	Gyrinidae		0	13	7	0	3	5	0	0	3	0
	?Hydrochidae		0	0	0	0	0	0	0	0	0	0
	?Staphylinidae		0	0	0	0	0	0	0	0	0	0
DIPTERA	Chironomidae		27567	14063	11960	12037	4760	4535	1140	4290	3633	5157
	Simuliidae		6063	3447	15043	567	193	8390	2330	1580	60	180
	Athericidae		140	327	20	340	173	40	30	10	140	510
	Empididae		27	0	10	23	3	5	0	25	0	0
	Blephariceridae		0	0	0	0	0	0	0	0	0	0
	Ceratopogonidae		0	0	0	7	0	0	30	0	3	10
	Stratiomyidae		0	0	0	0	0	0	0	0	0	0
	Tipulidae		0	0	0	0	0	0	0	0	0	0
	Psychodidae		0	0	0	0	0	0	0	0	0	0
	Chaoboridae		0	3	4707	0	0	0	0	940	0	0
	Dixidae		3	0	0	0	0	0	0	0	0	0
	pupae		37	27	847	73	77	20	40	150	147	107
	"coneworm"		0	0	0	0	0	0	0	0	0	0

Appendix 1 continued...

TAXON			April 1995				
			BR1	BR2	BR3	RSE1	RSE2
Cnidaria							
Hydrozoa		*Hydra* sp.	0	0	1230	0	0
Platyhelminthes							
Turbellaria			0	0	30	7	13
Nematoda			3	0	30	0	0
Annelida							
OLIGOCHAETA	Lumbriculidae		883	197	387	240	383
Arthropoda							
Acarina	Hydrachnellae		403	170	130	243	100
Crustacea							
CLADOCERA			40	0	80927	10	0
COPEPODA			7	7	58610	7	3
OSTRACODA			0	0	443	0	0
AMPHIPODA			0	0	0	0	0
Insecta							
COLLEMBOLA			27	3	0	3	7
PLECOPTERA	Notonemouridae		47	43	0	123	30
EPHEMEROPTERA	Baetidae		7960	6803	1493	253	233
	Heptageniidae		330	1607	0	0	0
	Ephemerellidae		67	270	0	57	93
	Caenidae		0	13	17	0	0
	Leptophlebiidae		1470	1107	3	440	347
TRICHOPTERA	Hydropsychidae		997	123	32870	63	33
	Philopotamidae		317	0	0	10	17
	Leptoceridae		290	503	0	380	43
	Hydroptilidae		43	33	0	230	43
	Polycentropodidae		50	7	0	13	0
	Ecnomidae		577	60	0	37	10
	Sericostomatidae		0	0	0	0	0
	Barbarochthonidae		0	0	0	10	0
	Petrothrincidae		0	0	0	0	0
	pupae		0	0	247	0	0
ODONATA	Aeshnidae		7	0	0	7	3
	Libellulidae		0	0	103	3	3
	Chlorolestidae		0	0	0	0	0
	Gomphidae		0	0	0	0	0
	Anisopteran juveniles		0	0	0	0	0
MEGALOPTERA	Corydalidae		13	37	0	7	0
LEPIDOPTERA	Nymphulidae		0	0	0	0	0
HYMENOPTERA			17	0	0	7	0

Appendix 1 continued...

TAXON			April 1995				
			BR1	BR2	BR3	RSE1	RSE2
HEMIPTERA	Veliidae		0	0	0	0	0
	Corixidae		3	0	7	0	0
	Hebridae		0	0	0	0	0
	Pleidae		0	0	0	0	0
	Mesoveliidae		0	0	0	0	0
COLEOPTERA	Elmidae	adults	193	710	7	27	13
	Elmidae	larvae	2140	1897	340	377	487
	Dryopidae	adults	0	7	0	3	3
	Dryopidae	larvae	0	0	0	0	0
	Hydraenidae	adults	173	133	23	17	3
	Hydraenidae	larvae	0	0	0	0	0
	Hydrophilidae	adults	0	0	0	0	0
	Hydrophilidae	larvae	0	0	0	0	0
	Limnichidae		7	0	0	0	30
	Helodidae		77	63	0	123	13
	Gyrinidae		10	3	0	0	7
	?Hydrochidae		0	0	0	0	0
	?Staphylinidae		0	0	0	0	0
DIPTERA	Chironomidae		7863	3420	5800	5250	2927
	Simuliidae		4933	2493	7710	263	250
	Athericidae		90	237	17	33	243
	Empididae		3	0	7	7	7
	Blephariceridae		0	0	0	0	0
	Ceratopogonidae		0	0	0	20	3
	Stratiomyidae		0	0	0	0	0
	Tipulidae		0	0	0	0	0
	Psychodidae		0	0	0	0	0
	Chaoboridae		0	0	660	0	0
	Dixidae		3	0	0	0	0
	pupae		60	107	520	100	27
	"coneworm"		0	0	0	0	0

Appendix 2.1 Depth, width and discharge measurements made at the sampling sites, from March 1994 through to April 1995.

Month	Site	Depth (m)	Width (m)	Discharge ($m^3 s^{-1}$)
March 1994	BR1	0.13	5.25	0.090
	BR3	0.39	17.30	4.104
	RSE1	0.19	9.45	0.304
	RSE2	0.14	12.85	0.234
May 1994	BR1	0.18	5.10	0.163
	BR3	0.23	17.50	1.353
	RSE1	0.35	12.00	1.373
	RSE2	0.35	13.40	1.341
July 1994	BR1	0.41	19.35	10.311
	BR3	0.30	16.95	12.113
	RSE1	0.37	12.15	3.706
	RSE2	0.24	13.20	2.324
October 1994	BR1	0.26	16.60	0.999
	BR3	0.23	12.15	1.362
	RSE1	0.31	12.30	1.270
	RSE2	0.19	15.55	1.270
November 1994	BR1	0.21	11.45	0.202
	BR3	0.26	18.80	3.528
	RSE1	0.30	9.05	0.345
	RSE2	0.13	12.70	0.223
January 1995	BR1	0.19	13.00	0.127
	BR3	0.22	18.70	2.934
	RSE1	0.17	11.15	0.382
	RSE2	0.15	10.45	0.315
April 1995	BR1	0.20	13.05	0.119
	BR3	0.11	18.35	0.798
	RSE1	0.15	10.50	0.262
	RSE2	0.10	9.75	0.164

Appendix 2.2 Physical and chemical variables measured at the sampling sites, from March 1994 through to April 1995.

Month	Site	Temperature (°C)	pH	Conductivity (mS m^{-1})	TDS (mg l^{-1})	TSS (mg l^{-1})
March 1994	BR1	21.5	5.46	1.83	6.25	0.275
	BR2	22.0	5.63	1.76	21.50	0.050
	BR3	22.5	7.10	4.71	43.63	2.171
	RSE1	20.5	4.29	2.49	26.00	0.200
	RSE2	18.3	4.26	2.40	23.50	0.275
May 1994	BR1	12.0	4.47	1.97	29.38	0.625
	BR2	11.5	4.53	2.20	26.38	0.850
	BR3	13.5	6.31	4.20	36.63	1.900
	RSE1	11.5	4.18	3.13	35.50	0.500
	RSE2	11.0	4.28	2.99	33.88	0.300
July 1994	BR1	11.5	4.23	2.34	14.25	0.600
	BR2	11.0	4.09	1.70	7.63	0.700
	BR3	10.0	4.23	2.25	6.63	0.850
	RSE1	9.0	3.92	2.83	19.50	0.375
	RSE2	10.0	3.62	2.86	19.38	0.500
October 1994	BR1	16.5	4.87	2.91	15.63	0.400
	BR2	13.5	4.92	3.57	13.13	0.350
	BR3	12.0	5.03	3.53	19.25	0.275
	RSE1	12.1	3.74	3.83	26.88	1.600
	RSE2	13.1	*	4.33	60.50	0.300
November 1994	BR1	24.0	5.53	*	24.34	0.625
	BR2	22.0	5.49	2.12	25.25	0.525
	BR3	20.0	6.46	3.61	41.00	1.867
	RSE1	24.0	4.69	2.74	33.63	0.050
	RSE2	21.5	4.58	2.47	28.50	0.075
January 1995	BR1	24.5	5.34	1.84	13.00	0.240
	BR2	23.0	5.39	1.82	21.25	0.100
	BR3	22.1	6.93	4.88	41.88	2.240
	RSE1	20.1	4.74	2.47	23.75	0.150
	RSE2	21.7	4.77	2.54	25.38	0.039
April 1995	BR1	18.4	5.49	2.25	15.50	0.100
	BR2	18.5	5.55	2.13	15.13	0.100
	BR3	18.0	6.33	2.85	36.50	1.235
	RSE1	17.0	4.49	2.78	24.88	0.200
	RSE2	19.5	4.69	2.79	31.13	0.100

* no measurement was made, due to a malfunctioning conductivity meter.

Appendix 2.3 Ionic concentrations, measured at all the sampling sites on the Riviersonderend and Berg River, over the period March 1994 to April 1995. Analyses were performed by the Department of Geology, University of Cape Town.

		Cations (mg l^{-1})				**Anions (mg l^{-1})**			
Month	**Site**	Na	K	Mg	Ca	Cl	NO_3	PO_4	SO_4
March 1994	BR1	3.433	0.294	0.449	0.003	5.643	0	0	0.844
	BR2	2.840	0.169	0.237	0.546	5.142	0	0	0.650
	BR3	6.123	0.858	1.450	2.026	11.835	0.530	0	3.478
	RSE1	5.586	0.729	0.429	0.949	6.192	0	0	0.859
	RSE2	4.186	0.366	0.367	0.400	6.799	0	0.756	0.838
May 1994	BR1	3.696	0.220	0.221	0.096	6.311	0	0.737	0.803
	BR2	3.189	0.182	0.318	0.169	6.465	0.375	0	0.982
	BR3	6.592	1.127	1.083	1.893	9.131	0	0.742	1.837
	RSE1	4.786	0.657	0.593	0.784	5.853	0	0.684	0.821
	RSE2	3.987	0.566	0.408	0.592	6.230	0	0	0.809
July 1994	BR1	3.685	0.012	0	0.056	3.806	0	0	0
	BR2	3.986	0	0	0.036	3.645	0	0	0
	BR3	4.115	0.094	0.014	0.062	3.172	0	0	0
	RSE1	3.896	0.386	0.092	0.119	4.071	0	0	1.143
	RSE2	5.132	0.413	0.109	0.318	3.584	0	0	0
October 1994	BR1	3.198	0	0	0.042	3.833	0	0	0.986
	BR2	4.192	0	0.015	0.072	3.824	0	0	0.949
	BR3	4.678	0.210	0	0.043	3.839	0	0	0.979
	RSE1	7.322	0.429	0.057	0.209	3.934	0	0	0.818
	RSE2	3.137	0.288	0.297	0.375	4.021	0	0	0.796
November 1994	BR1	3.725	0.261	0.091	0.348	3.947	0	0	0.876
	BR2	3.197	0.195	0.218	0.608	4.359	0	0	0.982
	BR3	4.553	0.447	0.499	1.085	3.436	0.605	0	1.355
	RSE1	3.154	0.410	0.438	0.816	4.836	0	0	1.071
	RSE2	3.776	0.073	0.288	0.410	5.053	0	0	1.118
January 1995	BR1	2.610	0.190	0.179	0.432	5.085	0	0	0.665
	BR2	3.618	0.366	0.219	0.428	*	*	*	*
	BR3	7.587	0.942	0.934	2.404	12.423	0.548	0	3.468
	RSE1	4.186	0.267	0.392	0.751	7.355	0	0	1.038
	RSE2	4.029	0.118	0.470	0.713	7.206	0	0	0.923
April 1995	BR1	3.940	0.491	0.264	0.406	4.100	0.459	0	0.838
	BR2	3.969	0.536	0.328	0.582	4.875	0.459	0	2.486
	BR3	7.242	1.474	0.774	1.770	11.362	0.314	0	3.025
	RSE1	4.553	0.209	0.627	0.673	8.888	0	0	1.206
	RSE2	*	*	*	*	9.082	0	0	1.705

* Water samples for these analyses were contaminated or lost.

Appendix 2.4 Total number of taxa recorded at each site over the sampling period, and the total abundance of individuals, Margalef's richness indices and Shannon-Wiener diversity indices.

Month	Site	Number of Taxa	Total Abundance (individuals m^{-2})	Margalef's richness index	Shannon-Wiener diversity index
March 1994	BR1	35	58 500	3.10	1.32
	BR2	32	53 600	2.85	1.63
	BR3	33	304 000	2.53	1.16
	RSE1	28	16 500	2.78	1.28
	RSE2	28	34 900	2.58	1.39
May 1994	BR1	34	26 800	3.24	1.66
	BR2	33	24 400	3.17	1.38
	BR3	28	57 300	2.46	1.74
	RSE1	34	7 120	3.72	1.59
	RSE2	30	3 460	3.56	2.01
July 1994	BR1	32	4 960	3.64	1.57
	BR2	35	8 110	3.78	1.61
	BR3	26	4 650	2.96	1.59
	RSE1	27	2 240	3.37	1.42
	RSE2	23	2 080	2.88	1.35
October 1994	BR1	34	18 900	3.35	1.29
	BR2	27	18 600	2.64	1.26
	BR3	30	10 900	3.12	1.22
	RSE1	22	3 430	2.58	1.42
	RSE2	26	2 310	3.23	1.82
November 1994	BR1	32	101 000	2.69	1.02
	BR2	30	107 000	2.51	0.72
	BR3	28	661 000	2.01	1.07
	RSE1	29	23 800	2.78	1.67
	RSE2	26	9 420	2.73	1.50
January 1995	BR1	29	19 400	2.84	1.84
	BR2	22	5 270	2.45	1.76
	BR3	28	147 000	2.27	1.22
	RSE1	29	8 900	3.08	2.01
	RSE2	29	10 900	3.01	1.93
April 1995	BR1	35	29 200	3.31	2.06
	BR2	27	20 100	2.62	2.11
	BR3	25	192 000	1.97	1.43
	RSE1	33	8 430	3.54	1.69
	RSE2	30	5 420	3.37	1.83