AN ASSESSMENT OF THE ECOLOGICAL EFFECTS OF INTER-BASIN WATER TRANSFER SCHEMES (IBTs) IN DRYLAND ENVIROMENTS

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Motivation and Research Objectives

Globally, there are few studies that have assessed the ecological implications of IBTs, despite their common occurrence in dryland ("dryland" refers to regions or countries where rainfall is unevenly distributed and unpredictable) environments, as well as in the more humid regions of the world. Recently, a wide range of potential impacts of IBTs has been identified by Davies *et al.* (1992), but virtually none of them has been quantified. Furthermore, IBTs are recognised to be extremely variable in terms of their operational criteria, such as the type and timing of delivery and abstraction, the types of donor and recipient reaches involved, and the type of transfer route utilised. Of major concern is the potential role of IBTs in the transfer of species between catchments. Given South Africa's commitment to the protection of aquatic environments, and to the maintenance of biological diversity in particular, this aspect is of concern to ecologists.

This project aimed, therefore, to achieve the following:

1. Increase our knowledge concerning the distribution, operation and widely varying effects of inter-basin water transfer schemes (IBTs), at national, international (particularly with Australian and American collaboration), and global levels.
2. Develop a synthesis of the literature available on the ecological, and socio-economic impacts of IBTs.
3. Collect field data from a local IBT scheme (the Riviersonderend-Berg-Eerste River Government Scheme; RBEGS), in order to investigate aspects of water quality and the biological status of the donor and recipient rivers.
4. Provide guidelines and protocols for the better operation of extant schemes, for the design of future schemes, and for the operation of future schemes in South Africa, using knowledge gleaned both locally and internationally.
5. Develop national and international fora for the appraisal of IBTs on a world-wide scale, through workshops and conferences, and to disseminate information on the ecological impacts of IBTs.

Aims 1 and 2 have been addressed in a separate volume, which synthesises the global literature on the occurrence and ecological (physical, chemical and biological) effects of IBTs (Snaddon *et al.*, 2000). The executive summary of the literature review is included in this volume. Aims 3 and 4 are addressed in this volume. Aim 5 has been accomplished, to a certain extent, through several presentations at national and international workshops and conferences, and through collaboration with, in particular, Dr Martin Thoms of the University of Canberra (ACT, Australia) and Dr Mike Meador of the United Stated Geological Survey (Raleigh, North Carolina, U.S.A.). The researchers involved in this project have disseminated much of the information gathered during this project through a variety of publications and presentations given during the project period.

Results and Conclusions from the Field Work Component

The release of water from the RBEGS IBT into the Berg River had a marked effect on the physical, chemical and biotic attributes of the river reaches immediately below the transfer outlet. During summer, statistically significant and, at times, substantial increases in discharge and, consequently, stream depth were accompanied by significant alterations in water quality. It is likely that the recorded changes in water chemistry are associated with the impoundment of the donor river at Theewaterskloof. Although changes in water quality variables were significant, and exceeded the background variation recorded for the fynbos bioregion, it is unlikely that they would lead to acute or chronic physiological effects on organisms, but rather they could cause changes in ecosystem structure and function.

The benthic invertebrate communities recorded below the IBT during summer months (November to May, during which the IBT was generally releasing water) were markedly different from those occurring further upstream. The overall richness and number of taxa recorded here were lower than those recorded above the IBT. Diversity was not significantly different at the impacted sites in summer. An MDS ordination plot of all the benthic invertebrate data showed distinct and different summer, below-IBT communities in the Berg River. During the winter months, when the IBT was not in operation, however, this below-IBT community resembled the upstream, unimpacted communities sampled during the same period. When IBT releases recommenced in summer, the downstream benthic invertebrate community shifted away from its "recovered" state once again. "Recovery" was limited to winter, and the more diverse summer communities cannot establish below the IBT, thus representing a discontinuity in distribution of these taxa.

The dissimilarities between the impacted (below the IBT) and unimpacted (above the IBT) benthic invertebrate communities were attributed to shifts in community composition. The percentage contributions to these dissimilarities were highest for the Trichoptera and Ephemeroptera. Several taxa that are rare or sensitive to flow and water quality conditions and that were recorded above the IBT, were absent or reduced below it during summer months. These included the Notonemouridae (Plecoptera), Heptageniidae, Ephemerellidae and Leptophlebiidae (Ephemeroptera), Philopotamidae, Leptoceridae and Ecnomidae (Trichoptera), Corydalidae (Megaloptera), Elmidae adults, Hydraenidae adults, and Helodidae larvae (Coleoptera) and the Athericidae (Diptera).

In adddition, large quantities of zooplanktonic groups (the Cladocera and Copepoda) were transferred through the IBT tunnel, and released, live or fragmented, into the Berg. In response, collector-predator Trichoptera larvae, of the family Hydropsychidae, and the filter-feeding dipteran Simuliidae proliferated below the IBT during summer months. In summary, the summer invertebrate community below the IBT resembled one typical of lake or reservoir outlets. The release of water from the Theewaterskloof impoundment into the Berg River has led to changes in water quality that are similar to those recorded downstream of lakes or impoundments, with the introduction of dissolved and suspended organic loads, and fast, turbulent flow, all of which are characteristic of lake or dam outlets.

The IBT outlet into the Berg River clearly represents a discontinuity in the river continuum, with some recovery during winter. A number of the Serial Discontinuity Concept predictions for the downstream effects of impoundment of the upper reaches of a river hold true for IBTs, such as increased nutrient levels, increased input of fine particulate organic matter, introduction of plankton, increased numbers of filter-feeding guilds and, thus, altered trophic relationships (Ward & Stanford, 1983).

Matrix of Effects

The matrices presented in Tables i and ii provide summaries of the actual (recorded) and predicted ecosystem responses to the ecological effects of IBTs, based on the literature and the data collected from the Berg River. Table iii lists some of the ecological implications of the ecosystem responses listed in Tables I and ii. Significantly, most of the ecosystem responses listed in the tables lead to the loss of biodiversity. This has evolutionary significance, and is important in the light of the fact that South Africa is a signatory to the international biodiversity convention.

References for Tables 5.1 and 5.2:

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3. Schorr *et al.*, 1995
4. O'Keeffe & de Moor, 1988
5. Matthews *et al.*, 1996
6. Brown *et al.*, 1984
7. W. Croucamp, in litt.
8. Cross & Moss, 1987
9. Stevens *et al.*, 1985
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11. Day, 1985
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13. O'Keeffe & Ashton, 1991
14. Snaddon *et al.*, 2000 (Section 3.2.2)
15. Chessman *et al.*, 1992
16. Cugley, 1988
17. Guiver, 1976
18. Cambray & Jubb, 1977
19. Laurenson & Hocutt, 1984
20. National Rivers Authority, 1994
21. García De Jalón, 1987
22. Kubomoto *et al.*, 1974
23. Holmes & Whitton, 1977
24. Brittain *et al.*, 1984
25. Borgstrøm & Løkensgard, 1984
26. Roos-Collins, 1993
27. Rozengurt *et al.*, 1985

Table i A matrix of the ecological effects of IBTs and the associated ecosystem (riverine) responses in the **recipient system and transfer route**.

	Ecosystem responses									
IBT effects on recipient and transfer route	Decreased taxon richness	Altered taxon dominance	Altered trophic dynamics	Altered community composition	Mixing of previously isolated gene pools	Local extinctions	Altered channel morphology	Altered habitat/ biotopes	Altered riparian zone	Algal blooms
Unseasonal increase in discharge	**o** [1,2,3]	**o** [1,4]	**o** [1,4]	**o** [1,4,5,6]		**p**	**o** [7]	**o** [6,7]	**p**	
Decreased MAR below offtake										
Increased flooding	**p**	**p**	**p**	**p**		**p**	**p**	**p**	**p**	
Decreased lateral connectivity	**p**	**p**	**p**	**p**		**p**	**p**	**p**	**p**	
Decreased flow variability	**p**	**o** [4]	**o** [4]	**o** [4,8]		**p**	**p**	**o** [4]	**p**	
Increased flow variability	**o** [2,3]	**p**	**p**	**o** [2,3,5,9,10]		**p**	**p**	**p**	**p**	
Increased bank and bed erosion	**p**	**p**	**p**	**o** [11]		**p**	**o** [11,12,13,14,15]	**p**	**p**	
Increased light penetration	**p**	**p**	**o** [8]	**o** [8]		**p**				**p**
Increased turbidity	**p**	**p**	**p**	**p**		**p**	**p**	**p**	**p**	
Increased organic suspensoids	**o** [1]	**o** [1]	**o** [1]	**o** [1]		**p**			**p**	**p**
Decreased TDS/ conductivities	**p**	**o** [4]	**o** [4]	**o** [4]		**p**			**p**	
Increased TDS/ conductivities	**p**	**p**	**p**	**o** [1]		**p**			**p**	
Decreased pH	**p**	**p**	**p**	**p**		**p**			**p**	
Increased pH	**o** [1]	**o** [1]	**o** [1]	**o** [1]		**p**			**p**	
Enrichment (nutrients)	**p**	**p**	**p**	**p**		**p**			**p**	**o** [16]
Dilution of chemical constituents	**p**	**p**	**p**	**p**		**p**			**p**	
Transfer of new species or populations	**p**	**o** [17]	**p**	**o** [18,19,20,21,22,23]	**o** [18,19,20,21,22,23]	**p**				**o** [17]
Introduction of parasites/ disease vectors	**p**	**p**	**p**	**p**		**p**			**p**	
Transfer of exotic species	**p**	**p**	**p**	**p**		**p**			**p**	
Transfer of secondary metabolites						**p**				**p**
Increased freshwater inflow to estuaries	**p**	**p**	**p**	**p**		**p**	**p**	**p**	**p**	**p**
Reduced freshwater inflow to estuaries										

p: potential response; **o**: observed response.

Table ii A matrix of the ecological effects of IBTs and the associated ecosystem (riverine) responses in the **donor** system.

	Ecosystem responses									
IBT effects on donor	Decreased taxon richness	Altered taxon dominance	Altered trophic dynamics	Altered community composition	Mixing of previously isolated gene pools	Local extinctions	Altered channel morphology	Altered habitat/ biotopes	Altered riparian zone	Algal blooms
Unseasonal increase in discharge										
Decreased MAR below offtake	**o** [24]	**p**	**p**	**o** [11,24]		**p**	**p**	**o** [24,25,26]	**p**	**p**
Increased flooding	**p**	**p**	**p**	**p**		**p**	**p**	**p**	**p**	
Decreased lateral connectivity	**p**	**p**	**p**	**p**		**p**	**p**	**p**	**p**	
Decreased flow variability	**p**	**p**	**p**	**p**		**p**	**p**	**p**	**p**	
Increased flow variability	**p**	**p**	**p**	**p**		**p**	**p**	**p**	**p**	
Increased bank and bed erosion	**p**	**p**	**p**	**p**		**p**	**p**	**p**	**p**	
Increased light penetration	**p**	**p**	**p**	**p**		**p**				**p**
Increased turbidity	**p**	**p**	**p**	**p**		**p**	**p**	**p**	**p**	
Increased organic suspensoids	**p**	**p**	**p**	**p**		**p**			**p**	**p**
Decreased TDS/ conductivities	**p**	**p**	**p**	**p**		**p**			**p**	
Increased TDS/ conductivities	**p**	**p**	**p**	**p**		**p**			**p**	
Decreased pH	**p**	**p**	**p**	**p**		**p**			**p**	
Increased pH	**p**	**p**	**p**	**p**		**p**			**p**	
Enrichment (nutrients)	**p**	**p**	**p**	**p**		**p**			**p**	**p**
Dilution of chemical constituents										
Transfer of new species or populations	**p**	**p**	**p**	**o** [23]	**o** [23]	**p**				**p**
Introduction of parasites/ disease vectors	**p**	**p**	**p**	**p**		**p**			**p**	
Transfer of exotic species	**p**	**p**	**p**	**p**		**p**			**p**	
Transfer of secondary metabolites						**p**				**p**
Increased freshwater inflow to estuaries										
Reduced freshwater inflow to estuaries	**o** [27]	**p**	**p**	**o** [27]		**p**	**p**	**p**		

p: potential response; **o**: observed response.

Table iii Some of the ecological implications of the ecosystem responses listed in Tables 5.1 and 5.2. This list is not comprehensive, but it gives an indication of the probable consequences of IBTs. The reader is referred to Snaddon *et al.* (2000) for more information.

Ecosystem response	**Ecological Significance**
Decreased taxon richness	• loss of biodiversity • altered biotic interactions, such as competition and predation • loss of rare and sensitive species
Altered taxon dominance	• loss of biodiversity • pest or weedy species dominant
Altered trophic dynamics	• loss of biodiversity • altered ecological processes, such as decomposition, production and nutrient cycling
Altered community composition	• loss of biodiversity • altered biotic interactions • reduced abundance
Mixing of previously isolated gene pools	• loss of biodiversity through hybridisation • altered biotic interactions • local extinction or increased mortality • implications for long-term evolution and basin integrity • the isolation of gene pools may also occur as a result of the construction and operation of IBTs
Local extinctions	• loss of biodiversity • altered biotic interactions
Altered channel morphology	• loss of habitats suitable for establishment of certain species or communities • altered deposition or scouring processes • loss of riparian connectivity
Altered habitat/ biotopes	• local extinction and/or increased mortality • loss of biodiversity • shifts in trophic dynamics
Altered riparian zone	• altered community composition of riparian zone • loss of important food source • loss of lateral connectivity • decreased/increased shading • altered timing of food inputs • changes in CPOM:FPOM ratios • altered P:R ratios
Algal blooms	• loss of biodiversity • deoxygenation of water column and sediments • toxic/non-toxic exudates • altered water quality

Achievement of Objectives

It was felt that the research objectives were achieved, while contributing to our overall understanding of the ecological effects of the regulation of river systems through water transfers. It is hoped that the matrices of IBT effects developed for this project will provide a useful summary of IBT effects and the associated ecosystem responses, that can be used by ecologists and water managers

Recommendations for IBT Planning and Management

The following recommendations are made, based on both the global review of the literature on IBTs, and the results of the field work component of the project.

- The ecological consequences of IBTs are such that caution is warranted. Data are scarce, and thus, the collection of timely, objective and detailed information on the actual and potential effects of transfers is necessary, in order that all human communities affected by any scheme may assess the IBT in an informed and reasonable manner. The collection of pre-IBT data is essential, as the assessment of IBTs must occur on a case-specific basis.
- All projects should be monitored, in order that the assessment of the effects of IBT can continue through the operational phases.
- Extant schemes should be re-assessed in terms of their effects, so that detrimental impacts can be minimised through mitigation.
- 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 should be given equal weighting, and should be assessed to the same level for each basin.
- Greater public participation is required during the planning of IBTs, supported by appropriate legislation, and designed to ensure adequate consultation in both the donor and recipient catchments, and communities along the transfer route(s). In South Africa, this should be achieved by the appropriate catchment management agency or agencies where more than one water management area is involved.
- 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.
- 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, disease vectors and pests of economic importance, are likely consequences of IBTs. This requires great caution and extensive investigation during the assessment of the feasibility of such schemes. Once again, the availability of pre-transfer information from donor and recipient catchments would aid in the assessment of the likelihood of the transfer of exotic and indigenous fauna and flora, parasites and disease vectors.

- Similarly, the likelihood of the transfer of water quality problems, such as cyanophyte blooms, between catchments, should be assessed. The threat of such transfers could be reduced through flexibility in the operational criteria of an IBT scheme, thus allowing for a cease in transfer during periods of risk.

- A botanical investigation to predict the change in vegetative cover under various release strategies would be useful for IBT planning and management. A more definitive prediction of morphological response relies on an improved understanding of the effects of various durations of inundation on those species of trees, shrubs and grass prevalent in the riparian zone of a river.

- There is a need to shift the current focus on ecosystem responses to low flows, to include the effects of increases in flow on aquatic systems. This would allow ecologists to determine the upper limits to flow increases (or capping flows) as a result of river regulation, such as IBT releases, and the timing of such releases.

- The temporal "recovery" by the riverine macroinvertebrates observed in the Berg River, during months when the IBT was not operational, provides evidence of the extent to which an ecosystem is affected by the operational criteria of a transfer scheme. Releases from IBTs should be timed to allow the riverine communities to reset during all seasons, in order to maintain the diversity of populations inhabiting the impacted river reaches.

- The transfer of lentic fauna into the Berg River contributed significantly to the shifts in benthic invertebrate community composition below the outlet. It is preferable, therefore, for IBT releases to be received by a standing waterbody, such as an artificially created wetland or off-channels storage, rather than released directly into a river. The impacts of such solutions, however, require detailed assessment.

- It is essential to be able to monitor the water chemistry of the donor reservoir. This is especially important where nutrient enrichment can occur in the donor system, which may lead to algal blooms and associated water quality problems in the recipient and along the transfer route.

Resource-directed measures for the protection of water resources

The revision of the South African Water Act has generated a number of initiatives which aim to increase the efficiency with which freshwater resources are managed and thus, protected. One of these is the implementation of a water resources protection policy, which aims to develop resource-directed measures (RDM) for the protection of water resources. The RDM procedure involves the definition of a desired level of protection for a water resource, and on that basis, setting clear numerical or descriptive goals for the resource quality of the resource (the Resource Quality Objectives; RQOs). An RDM procedure has been designed to undertake preliminary determinations of the ecological management class, the Ecological Reserve and resource quality objectives for individual water resources, as specified in sections 14 and 17 of the South African National Water Act (Act 36 of 1998) (Mackay, 1999). This process was begun by the Department of Water Affairs and Forestry in 1997, and will continue into a final phase where the detailed procedures and a full resource classification will be finalised. The appropriate application of this procedure should ultimately provide effective protection for rivers, wetlands and groundwater resources affected by water resource development, such as IBTs, and should guide the future planning and management of such schemes.

One aspect of the biotic integrity of river systems is the state of the riverine macroinvertebrates. This can be measured using rapid bio-assessment techniques, such as the South African Scoring System (SASS). It is possible to set reference conditions for rivers within a particular bioregion, subregion or river type, and compare monitoring or impacted sites against these conditions, in order to determine their present and further, their desired management classes. Resource quality objectives can then be set in order to achieve the desired management class.

On request by the Steering Committee of this project, SASS data were generated from the Berg River data, although it should be noted that the correct SASS protocol was not followed. According to preliminary categories recently developed, all sites fell into the unimpaired – class A – or the moderately impaired category – class B. During summer months, when the tunnel was releasing water, the total scores and average scores per taxon at impacted sites (BR3 and BR4) indicated that these sites generally were of lower biotic integrity compared with the sites above the IBT. Although this trend was not consistent, and there was some seasonal variation, the deterioration in biotic integrity appeared to occur towards the end of each summer season, after a few months of water release.

It is recommended that such a procedure be followed for all water resources affected by IBTs and should, indeed, be repeated at the Berg River sites according to the correct sampling protocol. SASS4 data should be collected above and below all IBT outlets, where possible, and scores should then be compared between sites, to assess the deterioration in biotic integrity directly caused by the IBT, and also compared with reference conditions for the appropriate bioregion and subregion, or river type, in order to assess the impact in the context of the whole catchment.

Such data collection would dovetail with the RDM procedure described above, as it would provide data for the determination of present state classes and consequently, management classes, the Ecological Reserve, and the appropriate RQOs. Furthermore, this process would provide a national database on the effects of IBTs on biotic integrity. This would prove useful in the development of mitigatory measures for the improved management of IBTs. For example, an IBT that is operational for only a few months of the year, thus preventing a deterioration in the biotic integrity of the recipient river, is more acceptable than a scheme that operates throughout the year. Such operational rules could be applied elsewhere.

Further Research Needs

- Tables i and ii highlight many gaps in our knowledge of the actual ecological effects of IBTs. There is a need for more empirical evidence in order to confirm the links between the predicted ecological effects of IBTs and the associated ecosystem responses.
- This project failed to identify which of the physical and chemical changes recorded in summer below the IBT in the Berg River were more important determinants of the observed shifts in invertebrate community composition. This will require the use of experimental systems, where changes in water chemistry and flow regime can be manipulated. In this way, species-level responses to IBTs can be determined.
- Further research should attempt to investigate the effects of IBTs at the level of the whole river system or, at least, river zone. For example, the effects of the discontinuity in the establishment of the naturally diverse summer Berg River community requires further study (see Section 4.3). This would require the collection of seasonal data from the entire length of a recipient or donor river, or river zone. Such data collection should, indeed, be incorporated in the monitoring programmes instituted by all catchment management agencies throughout the country.

Introduction

In many dryland environments (semi-arid, arid and hyper-arid countries), where rainfall is unevenly distributed and unpredictable, inter-basin water transfers (IBTs) are frequently perceived as the most feasible solution to the problems associated with the skewed (in terms of human water demand) distribution of available aquatic resources in relation to human population centres and human needs. In many parts of the world, water transfers have become the lifeblood of developing and extant human settlements, for which no alternative is currently perceived to be available. The rationale for the development of IBTs varies according to climate and social needs. In countries such as Canada and Norway, which have comparatively high rainfalls, IBTs are primarily used to augment the power generation potential of hydro-electric schemes, while in drier countries, IBTs supply potable and irrigation water.

In international terms, water transfers have begun to assume increasing significance, while significant improvements in engineering have enabled IBTs to become far more important in terms of the volumes transferred and the distances traversed. However, assessments of the effects of IBTs on the aquatic systems and human communities involved are few. Such assessments usually are limited to the planning and construction phases, with no follow-through, post-construction, in order to assess functioning and effects. Ecological concerns surrounding IBTs have been virtually ignored world-wide.

Where ecological and environmental work has been undertaken on proposed IBTs, it is conceptual and empirical by nature, and pre- and post-transfer ecological work has rarely been undertaken. Thus, the recent upswing in the number proposals for IBTs, world-wide, has occurred despite a lack of adequate knowledge of the effects of such schemes. Furthermore, public opposition to transfers has begun to increase due to a growing awareness that water is a finite resource within any catchment, and that the initiation of a transfer out of a donor catchment may result in a permanent loss of water in that catchment, to the detriment of its future development. The benefits and costs of IBTs are rapidly being subjected to critical appraisal, particularly from communities living within the donor catchments.

Aims of the Report

The main aims of the report were to:

- Increase our knowledge concerning the distribution, operation and widely varying effects of inter-basin water transfer schemes (IBTs), at national, international (particularly with Australian and American collaboration), and global levels.

- Develop a synthesis of the literature available on the ecological (physical, chemical and biological), socio-political and socio-economic impacts of IBTs.

Definition of IBTs

Following Davies *et al.* (1992), we have adopted a definition of an inter-basin transfer of water 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 water transfers of all volumes, across any distance both within and between catchments. IBTs comprise a donor (or source) system, a transfer route and recipient system.

Recovery and Reset

It is likely that IBTs represent a disruption of the river continuum, similar to that caused by impoundment, and other forms of river regulation. Consequently, aquatic systems affected by IBTs can be expected to recover some distance from the scheme. In the temporal context, many IBTs cease for some part of the year, especially where the reason for transfer is irrigation. Thus, the rivers involved, if not impounded, will have the opportunity to recover to some extent, before transfer recommences. This recovery will depend on several factors such as the location of the water transfer, the robustness of the systems, the period during which the transfer ceases and the timing of releases. Given that these arguments are well-founded, the application of river ecosystem theory, including the River Continuum Concept (RCC; Vannote *et al.*, 1980) and Serial Discontinuity Concept (SDC; Ward & Stanford, 1983) is not only possible, but, coupled with the recovery aspects of the SDC, should have important implications for the design of future IBTs and for the design, operation and management of extant schemes.

Global review of IBTs and their ecological effects

The following section provides a summarised description of IBT schemes across the globe, and the effects (both predicted and observed) of these schemes on the ecological functioning of the aquatic systems involved, in terms of physical, chemical and biological characteristics, and on socio-economic and cultural aspects of human communities.

CONTINENTAL AFRICA

Egypt

Jonglei Project (incomplete); transfers water for irrigation within the White Nile Basin; maximum: 4×10^9 m^3 yr^{-1}.

- probable that snails, found to be intermediate hosts to trematode parasites causing several diseases in humans and stock, could colonise the margins of the main Jonglei Canal, and secondary, draw-off canals, and could populate the new wetland habitat created by the IBT.
- provision of suitable habitat within the Jonglei Canal has increased the spread of the invasive water hyacinth (*Eichhornia crassipes*) and the water cabbage (*Pistia stratiotes*).

Libya

Great Man-Made River Project, transfers water from Alkufrah/Assarir/Fezzan regions to the coastal region, for irrigation; maximum 820×10^9 m^3 yr^{-1}.

Israel

Jordan River Project; from the Jordan River (Sea of Galilee) to the Central and southern coastal plain; for irrigation and municipal water supply; maximum 1200×10^6 m^3 yr^{-1}.

- diversion of the Jordan River away from the Dead Sea has resulted in a reduction of surface area of this inland sea by 300 km^2, or 30% of its original area, leaving factories, that extracted potash and other salts from the seawater, stranded several kilometres from the edge of the sea

Iraq

Tharthar Development Project; transfers between the Tigris and Euphrates rivers; for irrigation; maximum 1100×10^6 m^3 yr^{-1}

- the coastal marshes of the Tigris and Euphrates rivers are threatened by the almost total diversion of the Euphrates and many of its tributaries – the marshes are globally important as wetland bird habitat

Jordan

Red-Dead Canal (proposed); between the Red Sea and Dead Sea; for replenishing the Dead Sea water source.

Lebanon

Proposed transfer from the Litani River to the Jordan River; for general water supply.

South Africa

Usutu-Vaal River Government Water Scheme; transfers water between the Vaal and Usutu rivers; for electric power and general water supply; maximum 500 x 10^6 m^3 yr^{-1}.

Grootdraai Dam Emergency Augmentation Scheme, within the Vaal catchment; for emergency general supply.

Usutu River Government Water Scheme; transfers within the Usutu River catchment; for power station cooling; maximum 103 x 10^6 m^3 yr^{-1}.

Komati River Government Water Scheme; transfers water from Komati River to Olifants River catchment; for power station cooling; maximum 131 x 10^6 m^3 yr^{-1}.

Slang River Government Water Scheme; from Slang River (Buffalo) to Perdewaterspruit and Schulpspruit (Vaal); for power station cooling.

Tugela-Vaal Scheme; transfers from Tugela to River to Nuwejaarsspruit (Vaal); for general supply; maximum 631 x 10^6 m^3 yr^{-1}.

- it was expected that the freshwater snail hosts for schistosomiasis, *Biomphalaria* sp. (intestinal bilharzia) and *Bulinus* (*Physopsis*) sp. (urinary bilharzia), would be transferred from the Tugela to the Vaal system – this has not occurred due to temperature differences between the two systems

Caledon-Modder Scheme; from Caledon to Modder River; for municipal and industrial supply; maximum 40 x 10^6 m^3 yr^{-1}.

Lesotho Highlands Water Project (Phase 1A); transfers from Malibamatso River to Ash River (Vaal); for general water supply; maximum 533 x 10^6 m^3 yr^{-1}.

- substantial reductions and fluctuations in flow are expected in the donor Malibamatso River; decreased channel size of Malibamatso is likely
- erosion, stream-bed armouring, and transport of sediment downriver expected in the Malibamatso (donor) and Ash/Liebenbergsvlei rivers, as a result of settling of sediment in Katse Reservoir
- reservoir-induced seismic activity has been recorded in Katse Reservoir area since filling commenced
- water in donor Katse Reservoir expected to be warmer in summer and cooler in winter than water in the recipient Ash River

- invertebrate communities in recipient Ash/Liebenbergsvlei system expected to be affected by loss of habitat and smothering by filamentous algae that will establish in the sediment-free water transferred from Katse Reservoir
- transfer of lentic taxa to recipient river is expected
- filter-feeding taxa expected to benefit in recipient system
- establishment of pest simuliids (blackflies) in recipient system is possible
- the clarity of the water released from Katse Dam could lead to increased light penetration, which will encourage the growth of various macrophytes, such as filamentous algae, *Potamogeton* spp (pondweed) and *Typha* spp (bulrush)
- it is also possible that the transfer of clear water to the Vaal River will lead to decreased turbidities in the Vaal Reservoir and thus, to increased plant production
- strong objection to the increase in water tariffs in South Africa, and lack of water-demand management strategies which would reduce the tariff increment

Lesotho Highlands Water Project (further phases); **proposed** transfers from Matsoku (Malibamatso) and Senqunyane to Ash River (Vaal); for general supply; maximum 2207 x 10^6 m^3 yr^{-1}

- 1 in 20 year flood equivalent expected in recipient Ash/Liebenbergsvlei system; inundation of land in recipient catchments

Mhlatuze Government Water Scheme; from Mhlatuze River to Lake Nseze (Mhlatuze catchment); for municipal and industrial supply.

Orange River Project; water transferred from Orange River to Teebus (Great Fish), and to Sundays River; for irrigation, maximum 1700 x 10^6 m^3 yr^{-1}.

- 500-800% increase in flow in recipient Great Fish River, converted from seasonal to perennial system
- reduction in salinities in recipient Great Fish River
- significant changes in riverine invertebrate species composition in recipient Great Fish River, with only 33% of taxa common to pre-and post-transfer communities
- significant shifts in the dominant dipteran Chironomidae and Simuliidae species, and trichopteran Hydropsychidae
- pest species of simuliid, *Simulium chutteri*, has outcompeted other species, and causes stock damage and losses
- increase in high-flow-velocity habitats favoured establishment of simuliid larvae
- transfer of four species to the recipient Great Fish River as a result of the IBT: smallmouth yellowfish, *Barbus aeneus*, the Orange River mudfish *Labeo capensis*, the sharptooth catfish, *Clarias gariepinus* and the rock barbel, *Gephyroglanis sclateri*
- also likely that new individuals have been added to recorded populations in the Great Fish

Riviersonderend-Berg-Eerste River Government Water Scheme; from Riviersonderend to Berg and Eerste rivers; for general supply; maximum 130 x 10^6 m^3 yr^{-1}.

- 570-4400% increase in flow in recipient Berg River
- transfer of geosmin, a non-toxic cyanophyte exudate, from donor reservoir to the recipient Berg River, affecting flesh of rainbow trout in a downstream farm
- increased pH, conductivity, total dissolved and suspended solids, sodium, magnesium, potassium and calcium cations, and sulphate and chloride anions in the recipient Berg River
- invertebrate communities below IBT had significantly greater numbers of individuals, and lower overall richness and numbers of taxa
- overall loss of sensitive invertebrate taxa
- filter-feeding taxa appeared to benefit from the water transfer
- transfer of lentic taxa from donor impoundment to recipient river

Mooi-Mgeni Scheme; from Mooi River (Tugela) to Mpofana River (Mgeni); for general water supply; maximum $30 \times 10^6\ m^3\ yr^{-1}$.

Mooi-Mgeni Scheme; **proposed** increased transfer from Mooi (Tugela) to Mpofana (Mgeni); for general supply; maximum $315 \times 10^6\ m^3\ yr^{-1}$.

- some reaches of the recipient Mpofana River likely to erode further

Amatole Scheme; transfer from Toise and Kubusi rivers to Yellowwoods and Nahoon rivers; for municipal supply; maximum $225 \times 10^6\ m^3\ yr^{-1}$.

- bank and bed erosion in recipient rivers

Palmiet River Scheme; from Palmiet to Steenbras River; for general supply; maximum $22 \times 10^6\ m^3\ yr^{-1}$.

Palmiet River Scheme; **proposed** transfer from Palmiet to Steenbras; for general supply; maximum $127 \times 10^6\ m^3\ yr^{-1}$.

Mzimkulu-Mkomaas-Illovo Scheme; **proposed** transfer from the Mzimkulu and Mkomaas rivers to the Mgeni and Illovo rivers; for municipal supply; maximum $375 \times 10^6\ m^3\ yr^{-1}$.

Sabie River Government Water Scheme; **proposed** transfer from the Marite to the Sand and Klein Sand; maximum $25 \times 10^6\ m^3\ yr^{-1}$.

The Zambezi Aqueduct; **proposed** transfer from the Zambezi River to Botswana and the Vaal River; for general supply; maximum $4000 \times 10^6\ m^3\ yr^{-1}$.

Tugela-Mhlatuze Transfer; **proposed** transfer from the Tugela River to the Mhlatuze River; for general supply.

Western Cape proposals; **proposed** transfers between the Berg, Olifants and Breede rivers; for general supply.

Namibia

Eastern National Water Carrier; from Swakop, Omatako rivers and Karstland boreholes to Windhoek area; maximum 15-20 x 10^6 m^3 yr^{-1}

- water table depression in Karstveld area
- growth of filamentous algae in open canals likely to cause decreases in water quality
- open canal is a trap for animals

Eastern National Water Carrier; ***proposed*** *transfer from the Kavango River to the Omatako River*; for general supply; maximum 9.5 x 10^6 m^3 yr^{-1}

- the transfer of schistosomiasis (bilharzia) from the Kavango River to recipient areas is expected

Botswana

North-South Water Carrier; from the Shashe River to Eastern Botswana; for general supply.

CONTINENTAL AMERICA

Canada

Long Lake Diversion; transfers water from the Kenogami River to the Aguasabon River; for pulpwood transport and hydro-electric power generation; maximum 1356 x 10^6 m^3 yr^{-1}.

- rapid erosion in diversion channels
- transfer of sea lamprey was avoided due to the presence of natural barriers and hydro-electric power dams
- increased fish-egg mortality in recipient systems, due to increased erosion

Ogoki Diversion; from Ogoki River to the Little Jackfish River; for hydro-electric power generation; maximum 3563 x 10^6 m^3 yr^{-1}.

- erosion in recipient Little Jackfish River

- transfer of sea lamprey was avoided due to the presence of natural barriers and hydro-electric power dams
- increased fish-egg mortality in recipient systems, due to increased erosion

Kemano Diversion; transfer from Nechako River to Kemano River; for hydro-electric power generation; maximum 3626 x 10^6 m^3 yr^{-1}.

- Nechako chinook salmon were adversely affected by flow disruptions from the closure of dams, warm-water releases and siltation in the donor Nechako River
- increased discharges in the recipient Kemano River have resulted in pink and chum salmon runs
- no compensation given for the loss of approximately 350 000 ha of Tweedsmuir Park, a popular tourist destination in Canada
- Cheslatta Band Indians living in the area to be inundated had to abandon traditional occupations of hunting, trapping and fishing, and were afforded meagre compensation
- the developer met with these communities three days before flooding began

Churchill River Diversion; from the Churchill River to the Rat, Burntwood and Nelson rivers; for hydro-electric power generation; maximum 24 440 x 10^6 m^3 yr^{-1}.

- 35-fold increase in flow in the Rat River, 7-fold increase in the Burntwood River; 34% increase expected in the lower Nelson River
- 75% reduction in flow in donor Churchill River
- decrease in water temperature in Southern Indian Lake, a receiving water body
- leaching of soils containing mercury led to increased mercury levels in fish, and reduced whitefish catches

James Bay Project; transfer from Eastmain Caniapiscau and Sakami rivers to the La Grande Rivière; for hydro-electric power generation; maximum 50 457 x 10^6 m^3 yr^{-1}.

- 87% reduction of flow in the donor Eastmain River; 43% reduction in the Caniapiscau River
- 2-fold increase in discharge in recipient, La Grande Rivière
- traditional sites for Indian and European settlements on river banks have been lost as a result of raised water levels in the recipient La Grande Rivière basin

McGregor Diversion; **proposed** transfer from Fraser River to the Peace River; to reduce flooding and for hydro-electric power generation; maximum 6 x 10^6 m^3 yr^{-1}.

- transfer of fish and their protozoan and metazoan parasites was expected between the Pacific-draining donor Fraser River and the Arctic-draining recipient Peace River

U.S.A.

Los Angeles Aqueduct (California); from Owens Lake to the Los Angeles metropolitan area; for general water supply.

- Owens sucker, *Catostomus fumeiventris*, was transferred to the Los Angeles Basin from northern donor rivers
- reduction in water levels endangered the donor Mono Lake's suitability as a nesting and feeding area for migratory waterfowl

Colorado to Los Angeles (California); transfer from Colorado River to the Los Angeles metropolitan area; for general supply; maximum 580×10^6 m^3 yr^{-1}.

Central Valley Project (California); from Trinity River to the Sacramento River; for general supply; maximum 3300×10^6 m^3 yr^{-1}.

California State Water Project (California); from the Sacramento and Feather rivers to Southern California; for general supply; maximum 5210×10^6 m^3 yr^{-1}.

- decreased numbers of the introduced striped bass, Morone saxatilis, the threatened delta smelt, Hypomesus transpacificus, and the endangered winter-run chinook salmon, Onchorynchus tshawytscha
- spring-run chinook salmon have also declined significantly, perhaps due to IBT-related flow fluctuations
- some of the algal species occurring in Silverwood Lake appear to have been transported from the north through the California Aqueduct
- reductions in freshwater flow to San Franscisco Bay have resulted in a 30% reduction in numbers of chinook salmon, Oncorhyncus tshawytscha, and an 80% reduction in numbers of striped bass, Morone saxatilis
- economic impact of these losses was estimated to be about US$1.3 billion
- led to damming of scenic Feather River
- possible further impoundment of sensitive Klammath, Trinity and Eel river systems, for southward diversion in order to maintain the San Francisco Bay delta

Truckee Canal (Nevada); transfer from Truckee River to Carson River; for irrigation.

Big Thompson Project (Colorado); from the Colorado River to the Platte; for irrigation and municipal supply; maximum 370×10^6 m^3 yr^{-1}.

Frying Pan-Arkansas Project (Colorado); from Colorado River to the Arkansas River; for irrigation.

Central Arizona Project (Arizona); transfer from the Colorado River to the Gila River; for irrigation; maximum 2650×10^6 m^3 yr^{-1}.

San Juan-Chama Transfer (New Mexico); from Colorado River to the Rio Grande; for irrigation and general supply.

- diversion from donor San Juan River threatened water availability at important American Indian religious sites
- reduced water availability to communal Hispanic irrigation ditches threatened the existence of these communities, and imposed a system of individual water-use priorities on a tradition of communal sharing of surpluses and shortages

Canadian River Project (Texas); from Canadian River to the Red, Brazos and Colorado rivers; for municipal and industrial use; maximum 190 x 10^6 m^3 yr^{-1}.

Lake Texoma-Lake Lavon Transfer (Texas/Oklahoma); from Red River to Trinity River; supply to Dallas.

- increased conductivity in recipient Sister Grove Creek
- substantial reductions in meiofaunal populations on fine sediment and woody substrata
- changes in abundance of individual fish species were recorded in the recipient Sister Grove Creek
- quantitative and qualitative changes in species composition of the fish assemblage were substantial at some sites in the recipient
- abundance of two minnows increased, and the abundance of one centrarchid species decreased, in Sister Grove Creek
- potential for decline in the recreationally important sport fisheries of Lake Texoma, valued at about US$57 million yr^{-1} in total business sales, US$23 million in personal income, and 718 jobs

New York supply (New York); from the Delaware River to New York City; for municipal supply.

Virginia Beach supply (Virginia); transfer from the Roanoke River to the Virginia coast; for municipal supply; maximum 83 x 10^6 m^3 yr^{-1}.

Santee-Cooper Project (S. Carolina); from the Santee to the Cooper River; for hydro-electric power

- increased sediment load in the Cooper River led to decreased navigational capacity, and increased ship-channel maintenance costs in harbour
- changed the estuary at the mouth of the Cooper River from a vertically well-mixed estuary to one that is stratified

Santee-Cooper Re-Diversion Project (S. Carolina); from the Cooper River to the Santee River; for dilution of sediment in recipient.

Texas Water Plan (Texas); **proposed** transfer from the Mississippi, Arkansas and White rivers to the Texas High Plains; for irrigation.

- the release of large quantities of water could lead to the proliferation of the encephalitis-carrying mosquito

U.S.A./Canada

North American Water and Power Alliance; **proposed** transfer from various rivers in Canada and the U.S.A. to rivers in the U.S.A. and Mexico; multi-purpose; maximum 136 000 x 10^6 m^3 yr^{-1}.

- donor waters cooler than recipient waters
- migration routes of the Caribou would be blocked by IBT canals and dams
- many of Canada's transcontinental transport links would have been cut by proposed reservoirs

Garrison Diversion Project; **proposed** transfer from the Missouri River in the U.S.A. to various recipients, including Lake Winnipeg, Canada; multi-purpose.

- Missouri River species are expected to establish in new habitat made available in Canada by the IBT, including the shovelnose sturgeon *Scaphirhynchus platorynchus*, paddlefish *Polyodon spathula*, shortnose gar *Lepisosteus platostomus*, gizzard shad *Dorosoma cepedianum*, rainbow smelt *Osmerus mordax*, river carpsucker *Carpiodes carpio*, smallmouth buffalo *Ictiobus bubalus*, Utah chub *Gila atraria*, and the endangered pallid sturgeon *Scaphirhynchus albus*
- existing populations of walleye *Stizostedion vitreum*, sauger *S. canadense*, lake whitefish *Coregonus clupeaformis* in lakes Manitoba and Winnipeg are expected to decline as a result

Grand Replenishment and Northern Development Canal; **proposed** transfer from the James Bay Basin to the Great Lakes of Canada; for hydro-electric power and flood control.

Mexico

National Water Plan; groundwater in the Lerma Basin, and water from the Cutzamala and Amacuzac rivers transferred to Mexico City; for municipal supply; maximum 1700 x 10^6 m^3 yr^{-1}.

- land subsidence and drying up of lakes has occurred

Water Plan for the Noroeste Region; from rivers of Sinaloa State to the Alamos River; multi-purpose.

National Water Plan; **proposed** transfer from Tecolutla and Oriental rivers to Mexico City; for municipal supply.

Brazil

Transfer from the Paraíba River to the Piraí and Lajes rivers; for hydro-electric power generation; maximum 5045 x 10^6 m^3 yr^{-1}.

Peru

Lima water supply; from the Mantaro River to the Rimac River; for hydro-electric power generation and municipal supply; maximum 1100 x 10^6 m^3 yr^{-1}.

CONTINENTAL ASIA

Russia

North Crimean Canal; transfer from the Dnieper River to various recipients; for irrigation

Kara Kum Canal; from the Amudar'ya to various recipients.

Severskiy Donets-Donbas Canal.

- growth of filamentous algae in open canals causes deteriorations in water quality

Kazakhstan

Irtysh-Karaganda Canal; from the Irtysh to the Karaganda River; for industry and irrigation.

Russia/Kazakhstan

Siberia-Central Asia Project; **proposed** transfer between the Ob River and the Syrdar'ya, Amudar'ya and Yenisey rivers; maximum 60 000 x 10^6 m^3 yr^{-1}.

- *flow fluctuations; 9-15% flow reduction in wet years and 19-31% for dry years in the Ob River;* flow reductions in the Irtysch
- reduced inundation of important floodplains
- improved conditions in the recipient Aral Sea
- leakage of water from canals likely to raise groundwater levels
- increased conductivity in Aral Sea
- potential transfer of water-borne diseases and parasites along open canals
- loss of feeding, breeding and wintering areas, pollution, altered hydrological regimes would lead to a 14 to 17% reduction in commercially exploited fish catches for the first phase alone
- *reduction in commercially exploited marsh-bird populations expected in donor areas*

European Transfer Project; **proposed** transfer from the Sukhona and Pechora rivers to the Volga; for irrigation and general supply; maximum 9800 x 10^6 m^3 yr^{-1}.

- PHASE I: *53% reduction in flow in donor Onega River; 43% reduction in Upper Sukhona River*, with sedimentation and tributary downcutting likely
- PHASE II: reduced flow in the Svir and Neva rivers
- FURTHER PHASE: 40-76% reduction in flow in the Pechora River, with effects on bank stability, navigation and on floodplain ecology
- OVERALL: increased freshwater input to Caspian Sea, the Sea of Azov, the Volga-Akhtuba floodplain and the Volga estuary
- increased concentration of pollutants in donor Onega and Sukhona rivers; increased pollutants in Gulf of Finland due to diversion of flow from the Neva River
- *the transfer of the intermediate parasite hosts Bithynia leachii* (a freshwater snail) and *Cyclops* sp. (a crustacean) to southern regions was expected where, at present, these invertebrates are almost absent; these invertebrates are the carriers of opisthorchiasis (liver-fluke disease) and diphyllobothriasis (Broad Fish tapeworm)
- the increased supply of water to the midland regions was also likely to encourage the establishment of water fever in recipient areas
- projected 50% reduction in salmon catch due to proposed third-stage diversion of water from the Pechora to the Volga
- in the proposed recipient Aral Sea past diversions have led to an 80% drop in the catches of bream, carp, zander and vobla, and now 80% of the fish caught are the less economically viable sprats; the transfer would alleviate this
- 43% reduction in flow in the donor Upper Sukhona River would have resulted in difficulties for navigation and for timber rafting on the river
- historically significant sites such as the Ferapontovo and Kirrilov-Belozerro monasteries, the Sophia Cathedral, and the settlements of Vologda, Kargopol', Beloozersk and Tot'ma, were thought to be threatened by the likely raising of groundwater levels and consequent flooding

Danube-Dnieper Water Resources Utilisation System; **proposed** transfer from the Danube to Southern Ukraine; for general supply.

Japan

Shin-Nippon Seitetsu Kabushiki Kaisha Scheme; from the Onga River to an iron-manufacturing plant; for industry; maximum 7×10^6 m^3 yr^{-1}.

Kagawa Irrigation Project; from the Yoshino River to irrigated lands; for irrigation.

Tama River water supply system; transfers between the Tama, Edo, Sagami and Tone rivers; for municipal supply to Tokyo City; maximum 1000×10^6 m^3 yr^{-1}.

Sakawa to Sagami; transfer from the Sakawa River to the Sagami River; for municipal supply.

China

Transfer from the Yangtze to the Yi-Shu-Si; multi-purpose.

South to North Water Project *(Western Route)*; **proposed** transfer from the Yangtze to Qaidam and Huang rivers and to Dingxi County; for general supply.

(Middle Route); from the Yangtze to Beijing District and the Han Jiang; for general supply.

(*East Route)*; from the Yangtze, Yellow and Huai rivers to Dawen and North China Plain; for general supply; maximum $30\ 000 \times 10^6$ m^3 yr^{-1}.

- leakage of water from canals likely to raise groundwater levels
- salinisation of recipient soils
- possibility of transfer if pollutants from donor to recipient catchments
- siltation in the lower reaches of the donor Yangtze River was expected to have detrimental effects on the Yangtze estuary, and estuarine and brackish water fish could also negatively be affected
- feared that productivity in shallow lakes to be used along transfer route, which serve as important freshwater fisheries and which are also rich growing areas for various economically important plant species, will be reduced as a result of high water levels
- altered flows, sediment transport and tidal gradients in the donor Yangtze estuary will adversely affect navigation in these parts of the river and estuary

- fears of substantial reduction in freshwater supplies to the industrial, agricultural and domestic water for Shanghai Municipality

India

Godavari-Krishna-Pennar Link; from the Godavari River to Krishna River, and from the Krishna to the Pennar River; for irrigation.

Narmada High Level Canal; **proposed** water transfer from the Narmada River to the Gujarat Region; general supply; maximum 34 690 x 10^6 m^3 yr^{-1}.

Brahmaputra-Ganga Link; **proposed** transfer from the Brahmaputra River to the Ganga; for flood control and general supply.

Sri Lanka

Mahaweli-Ganga Project; from the Mahaweli to the Ganga; for irrigation.

AUSTRALASIA

Australia

Snowy Mountains Scheme; transfers between the Murrumbidgee, Eucumbene, Tumut, Murray and Snowy rivers; for hydro-electric power generation; maximum 1130 x 10^6 m^3 yr^{-1} (Phase 1).

- flow fluctuations in recipient Murray River catchment; 45-99% reductions in flow in donor Snowy River
- water in the Snowy River is 6°C cooler than recipient Murray River water
- aquatic insects likely to be affected by the accumulation of fine sediments on the Snowy River bed
- flow reductions in the Snowy River are leading to a progressive upstream movement of the salt wedge in the lower reaches of the river
- a 99% reduction in suspended sediment yields below the IBT is likely to have effects on deposition and erosion at the river mouth

Lower River Murray transfers; from the Murray River to eight catchments in the Adelaide/Whyalla region; for municipal supply; maximum 157 x 10^6 m^3 yr^{-1}.

- increased salinity and turbidity in all recipient reservoirs in Adelaide/Whyalla area
- water transfers from the Murray River have elevated nutrient levels in recipient reservoirs in the Adelaide/Whyalla area, leading to increases in blooms of toxic cyanophytes

Nymboida River-Blaxlands Creek Scheme; transfer from Nymboida River (tributary of the Clarence River) to Goolang and Blaxland creeks; for hydro-electric power.

- erosion of channels in recipient Blaxland Creek
- river rehabilitation measures at a cost of A$0.5 million

Shoalhaven-Wingecarribee; from the Shoalhaven River to the Wingecarribee River; for municipal supply; maximum 1000 x 10^6 m^3 yr^{-1}.

- inundation of land in places; bank and bed erosion; overbank or in-channel sedimentation in places

Barnard-Hunter; from the Barnard River to the Hunter River; for hydro-electric power generation; maximum 20 x 10^6 m^3 yr^{-1}.

New Zealand

Waitaki Power Scheme; from Lake Tekapo (Waitaki Basin) to Lake Pukaki (Waitaki Basin); for hydro-electric power generation, irrigation and flood control; maximum 2340 x 10^6 m^3 yr^{-1}.

Rangitata Diversion Race; from the Rangitata and Ashbuton rivers to the Rakaia River (winter), and Hinds and Ashburton rivers, and coastal, spring-fed creeks (irrigation season); for hydro-electric power generation (winter) and irrigation (summer); maximum 730 x 10^6 m^3 yr^{-1}.

Tongariro Power Scheme; from the Wanganui, Rangitaiki and Whangaehu rivers tributaries to the Waikato River; for hydro-electric power generation; maximum 1000 x 10^6 m^3 yr^{-1}.

Managahao Power Scheme; transfer from the Mangahao River (Manawatu Basin) to the Tokomaru River (Manawatu Basin); for hydro-electric power generation; maximum 160 x 10^6 m^3 yr^{-1}.

Manapouri Power Scheme; from the Waiau River to the sea in the Doubtfull Sound; for hydro-electric power generation; maximum 10 000 x 10^6 m^3 yr^{-1}.

EUROPE

Norway

Glåma-Rena Transfer; transfer from the Glåma River to the Rena; for hydro-electric power generation.

- densities of Trichoptera, Ephemeroptera and Plecoptera considerably lower below donor impoundment

Veo-Smådøla Transfer; transfer from the River Veo to the Smådøla; for hydro-electric power generation.

- reduced recruitment and availability of important food organisms has resulted in reduced numbers of introduced brown trout, *Salmo trutta*, in the recipient Smådøla River

Aurland Hydropower Scheme; within the Aurland River catchment; for hydro-electric power generation.

Ulla-Førre hydro-electric power scheme; from Suldalslågen to Hylsfjorden; for hydro-electric power generation; maximum 410 x 10^6 m^3 yr^{-1}.

- the possibility exists that returning salmon will enter Hylsfjorden rather than Suldalslågen, as a result of the transfer of population-specific pheromones to Hylsfjorden

U.K.

Kielder Water Scheme; from the River Tyne to the Wear and Tees rivers; for hydro-electric power generation and compensation releases; maximum 200 x 10^6 m^3 yr^{-1}.

- flow was doubled in the River Wear, and velocities increased tenfold in some areas of the channel
- possible dilution of pollutants in recipient basins
- flow changes are expected to lead to shifts in fish spawning grounds
- live mayfly and chironomid taxa transferred to the River Wear, as well as dead and fragmented cladocerans
- populations of diatoms in recipient reservoirs owe their persistence to re-inoculation from IBTs
- disjunct distributions of some macrophyte species in the donor Tyne River, and recipient Wear and Tees rivers could be mixed as a result of the IBT

Ely Ouse to Essex Scheme; transfer from the Ely Ouse to the Stour, Colne and Blackwater rivers; maximum 180 x 10^6 m^3 yr^{-1}.

- transfer of pesticide from Ely Ouse to Essex catchment, where tomatoes were rendered unmarketable
- transfer of beet factory effluent from Ely Ouse to Essex
- dilution of sewage spill in a recipient, the River Stour
- screening and chlorination of Ely Ouse water is necessary to prevent the spread of the alien zebra mussel, *Dreissena polymorpha*, and diatom blooms
- transfer of the predatory fish, *Stizostedion lucioperca*, appears to have occurred, from Ely Ouse to the recipient River Stour
- reduced nesting success of birds in headwater region of the Stour, due to temporary drying up of the river during shut-downs
- transfers of *Stephanodiscus* blooms between the catchments gave rise to public complaints about potable water quality
- shift in dominance occurred, from a *Melosira* sp.-dominated system, to one dominated by *Stephanodiscus*

Severn-Thames Transfer; **proposed** transfer from the Severn to the Thames; for general supply; maximum 140 x 10^6 m^3 yr^{-1}.

Severn-Trent Transfer; **proposed** transfer from the Severn to the Trent; for general supply; maximum 100 x 10^6 m^3 yr^{-1}.

- establishment of new populations as a result of increased baseflows in the recipient Tame River (tributary of the Trent)
- reduced numbers in the River Severn, as a result of barriers to migration

Wye-Thames Transfer; **proposed** transfer from the the Wye to the Thames, via the Severn; for general water supply; maximum 140 x 10^6 m^3 yr^{-1}.

Slovakia/Hungary

Danube Diversion; transfer within the Danube River catchment; for navigation; maximum 158 000 x 10^6 m^3 yr^{-1}.

Germany

Danube-Main Scheme; *from the Danube to the Main River (Rhine River basin); maximum* $3000 \times 10^6\ m^3\ yr^{-1}$.

Greece

Acheloos Diversion; from the Acheloos River to the Plains of Thessaly; for irrigation.

France

Eaux de la Neste et de la Garonne; transfer from the Garonne River to the Le gave de Pau River (tributary of the Adour River); for hydro-electric power generation.

Eaux du Basin Artois Picardie; transfer from the Escaut and Canche rivers to the Lys River; for municipal supply.

Eaux de la Durance et du Verdon; from the Durance River (Rhone River basin) to the Marseilles/Toulon area; for industry and municipal supply.

Spain

Tajo-Segura Transfer; transfer from the Tajo River to the Segura River; for general supply; maximum $350 \times 10^6\ m^3\ yr^{-1}$.

- the gudgeon, *Gobio gobio*, has been introduced into the Segura River from the donor Tajo River

Recommendations for IBT Planning and Management

- A list of *recommendations for the planning and management of* IBT *schemes is presented.* They have been compiled from the literature, and are not listed in any order of priority. More detailed recommendations are presented in a final research report to the Water Research Commission (Snaddon and Davies, 2000).

- It is clear from the literature 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 collection of timely, objective and detailed information on the actual and potential effects of transfers is necessary, in order that all human communities affected by any scheme may assess the IBT in an informed and reasonable manner.
- In addition to the collection of information before the construction of an IBT scheme, all projects should be monitored, in order that the assessment of the effects of IBT can continue through the operational phases (e.g. Davies *et al.*, 1992).
- Extant schemes should be re-assessed in terms of their effects, so that detrimental impacts can be minimised through mitigation.
- 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.
- Greater public participation is required during the planning of IBTs, 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). 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 should usually be provided by the developer (e.g. in Lesotho, the Lesotho Highlands Development Authority; in South Africa, Department of Water Affairs and Forestry).
- 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.
- 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.
- Monitoring the water quality in transfer structures 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, disease vectors and pests of economic importance, are likely consequences of IBTs. There are examples in the literature of all of these threats (Chapters 3 and 4). This requires great caution and extensive investigation during the assessment of the feasibility of such schemes. Once again, the availability of pre-

transfer information from donor and recipient catchments would aid in the assessment of the likelihood of the transfer of fauna and flora.

- Similarly, the likelihood of the transfer of water quality problems, such as cyanophyte blooms, between catchments, should be assessed. The threat of such transfers could be reduced through flexibility in the operational criteria of an IBT scheme, thus allowing for a cease in transfer during periods of risk.
- A botanical investigation to predict the change in vegetative cover under various release strategies would be useful for IBT planning and management. A more definitive prediction of geomorphological response relies on an improved understanding of the effects of various durations of inundation on those species of trees, shrubs and grass prevalent in the riparian zone of a river.
- The results of work in South Africa indicate that the ecological effects of an IBT from a donor impoundment to a river, are similar to those occurring downstream of an impoundment (Snaddon, 1998). 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, methodologies developed for the determination of instream flow requirements (IFR) of rivers (e.g. King & Tharme, 1994; King *et al.*, 1995) should be applied to cases where water is transferred into a catchment.

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Table of Contents

Chapter 1 Introduction

The rapidly increasing human population in South Africa, its associated water demands, and the uneven distribution of riverine ecosystems across the country, has necessitated the transfer of water from areas with perceived surplus to those in deficit (Department of Water Affairs, 1986a; Davies *et al.*, 1992, 1993). Indeed, South Africa is one of the world leaders in the technology of inter-basin water transfer (IBT) construction. Due to the arid/semi-arid nature of most of the sub-continent, there are no naturally-occurring lentic surface water resources (such as lakes) available for human exploitation, and the availability of sites on rivers that are suitable for impoundment is rapidly decreasing. Impoundment and river regulation present major threats to the ecological functioning of our riverine resources. The rivers of South Africa require increasing research to manage them as sustainable resources, and to conserve them as valuable and valued ecosystems.

Globally, there are very few studies that have assessed the ecological implications of IBTs, despite their common occurrence in dryland ("dryland" refers to regions or countries where rainfall is unevenly distributed and unpredictable – see Davies *et al.*, 1992) environments, as well as in the more humid regions of the world (see Snaddon *et al.*, 2000.). Recently, a wide range of potential impacts of IBTs has been identified by Davies *et al.* (1992), but virtually none of them has been quantified. Furthermore, IBTs are recognised to be extremely variable in terms of their operational criteria, such as the type and timing of delivery and abstraction, the types of donor and recipient reaches involved, and the type of transfer route utilised. Of major concern is the potential role of IBTs in the transfer of species between catchments. Given South Africa's commitment to the protection of aquatic environments, and to the maintenance of biological diversity in particular, this aspect is of concern to ecologists.

1.1 AIMS OF THE PROJECT

The aims of the project described in this report, were to:

1. Increase our knowledge concerning the distribution, operation and widely varying effects of inter-basin water transfer schemes (IBTs), at national, international (particularly with Australian and American collaboration), and global levels.

2. Develop a synthesis of the literature available on the ecological, and socio-economic impacts of IBTs.

3. Collect field data from a local IBT scheme (the Riviersonderend-Berg-Eerste River Government Scheme), in order to investigate aspects of water quality and the biological status of the donor and recipient rivers.

4. Provide guidelines and protocols for the better operation of extant schemes, for the design of future schemes, and for the operation of future schemes in South Africa, using knowledge gleaned both locally and internationally.

5. Develop national and international fora for the appraisal of IBTs on a world-wide scale, through workshops and conferences, and to disseminate information on the ecological impacts of IBTs.

Aims 1 and 2 have been addressed in a separate volume, which synthesises the global literature on the occurrence and ecological (physical, chemical and biological) effects of IBTs (Snaddon *et al.*, 2000). Aims 3 and 4 are addressed in this volume, and the structure of the report is described below. Aim 5 has been accomplished, to a certain extent, through several presentations at national and international workshops and conferences, and through collaboration with, in particular, Dr Martin Thoms of the University of Canberra (ACT, Australia) and Dr Mike Meador of the United Stated Geological Survey (Raleigh, North Carolina, U.S.A.). The original intention of holding an international meeting in South Africa, which would draw together expertise on the ecological effects of IBTs, was not realised, due to time and budgetary constraints. It is felt, however, that the researchers involved in this project have disseminated much of the information gathered during this project (a list of publications and presentations given during the project is provided in Appendix A).

1.2 STRUCTURE OF THE REPORT

Chapter 1 **Introduction**

Chapter 2 **The Riviersonderend-Berg-Eerste River Government Water Scheme (RBEGS)**

A brief overview of IBTs in South Africa introduces this chapter, while the reader is referred to the Snaddon *et al.* (2000) literature review for more detail. The study area and IBT scheme chosen for detailed assessment is described in addition to the selection of study sites. The general approach and methodology that was developed for the research component of the project is outlined.

Chapter 3 **Physical and Chemical Effects of the RBEGS IBT**

Details are provided on materials, methods, data analysis and results of research on the physical and chemical effects of the IBT scheme described in Chapter 2. These include changes in discharge, width, depth, temperature, pH, conductivity, total dissolved/suspended solids and a range of ionic concentrations.

Chapter 4 **Biological Effects of the RBEGS IBT**

Similar to Chapter 3, this chapter provides details on the biological effects of the RBEGS IBT, including changes in benthic invertebrate community composition, diversity and richness.

Chapter 5 **Conclusions and Recommendations for IBT Planning and Management**

This chapter summarises and discusses the results presented in the previous two chapters, and evaluates the extent to which the aims of the project were achieved. A hierarchical matrix of the effects of IBTs is discussed in this chapter, in order to provide an ecological protocol for the assessment of IBTs in South Africa. Recommendations for improved IBT planning and management in South Africa arising from the literature review presented in Snaddon *et al.* (2000) are included here, in addition to recommendations arising from the results of the field work component of this study. Lastly, future research needs are listed.

References

Appendices: A, publications and presentations arising from the project; **B,** physical and chemical data; **C,** benthic invertebrate data.

2.1 INTRODUCTION

Most of southern Africa is water scarce (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) (Basson, 1996; van Niekerk *et al.*, 1996), although water availability and usage varies substantially throughout the subcontinent. 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 and generally low availability of water in South Africa, anthropogenic pressures such as over-abstraction, salinisation and pollution, are exacerbating problems of water supply (Snaddon *et al.*, 1998; Snaddon, 1999). 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 adequate sanitation (e.g. Palmer Development Group, 1995). Thus, in order to meet the water requirements of previously disadvantaged communities, water demands inevitably will increase over the next decade or so, 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 higher, and perceived to be of a greater priority, than use within its catchment of origin.

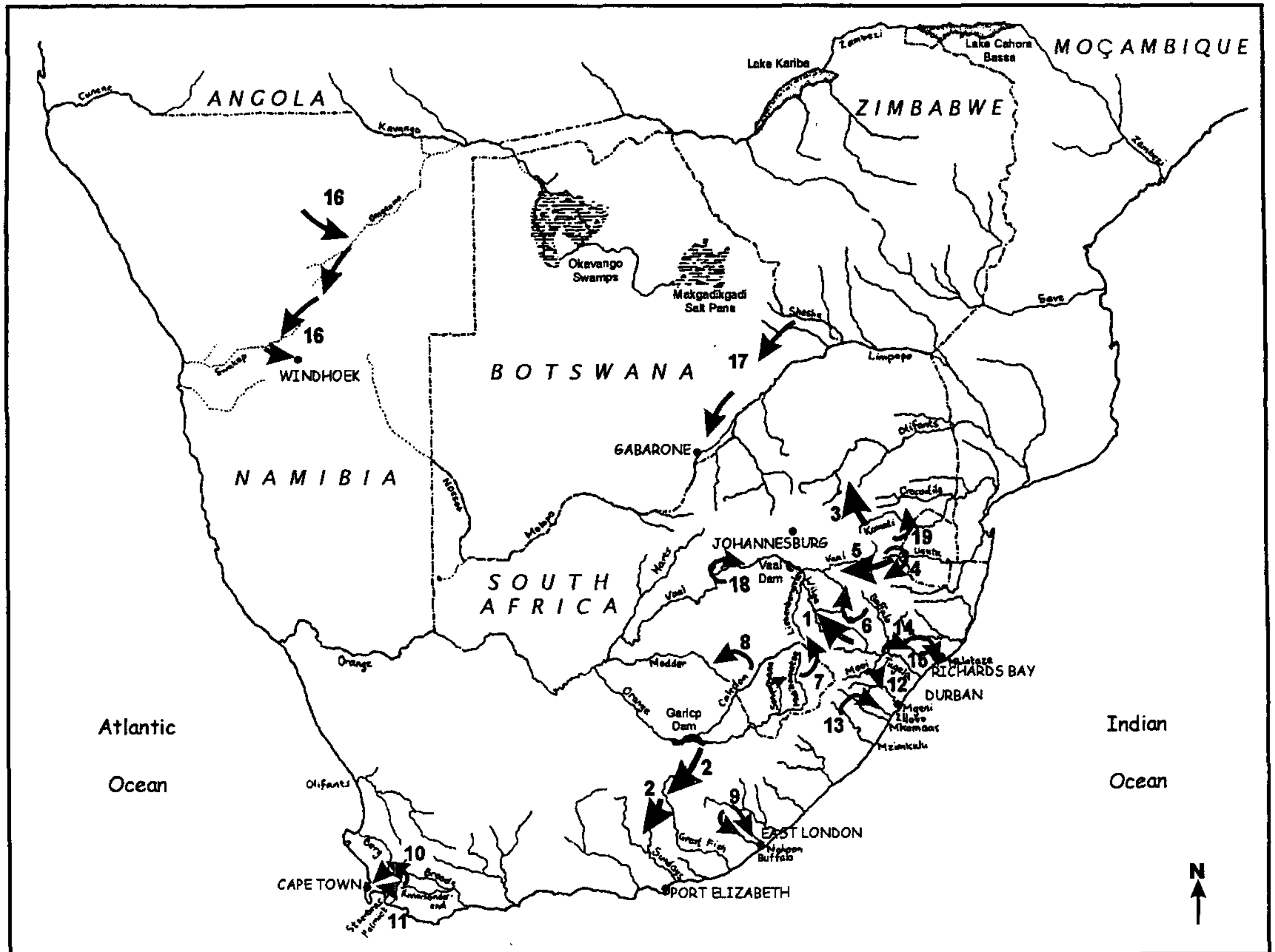


Figure 2.1 Major IBTs of southern Africa. 1, Tugela-Vaal Scheme; 2, Orange River Project; 3, Komati Scheme; 4; Usutu Scheme; 5, Usutu-Vaal Scheme; 6, Slang River Scheme; 7, Lesotho Highlands Water Project; 8, Caledon-Modder Scheme; 9, Amatole Scheme; 10, Riviersonderend-Berg-Eerste River Government Water Scheme; 11, Palmiet River Scheme; 12, Mooi-Mgeni Scheme; 13, Mzimkulu-Mkomaas-Illovo Scheme; 14; Tugela-Mhlatuze Transfer Scheme; 15, Mhlatuze Scheme; 16, Eastern National Water Carrier; 17, North-South Water Carrier; 18, Grootdraai Emergency Scheme; 19, Sabie River Government Water Scheme.

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) (Figure 2.1). 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, and both of these rivers are being investigated as potential sources of water, for countries such as Zimbabwe, Botswana, Namibia 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.

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.

The significance of IBTs to water resources development in South Africa is clear from Figure 2.1, while Figure 2.2 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, 1991; Snaddon *et al.*, 1998) (Scheme 1, Figure 2.1). This scheme was soon followed by the Orange River Project (ORP; Scheme 2, Figure 2.1) and the Komati and Usutu schemes (Schemes 3 and 4, Figure 2.1). The major water demand in South Africa is located in the province of Gauteng, in the catchments of the Vaal (south), the Crocodile (north), and the Olifants rivers (east). Thus, as can be seen in Figure 2.1, 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).

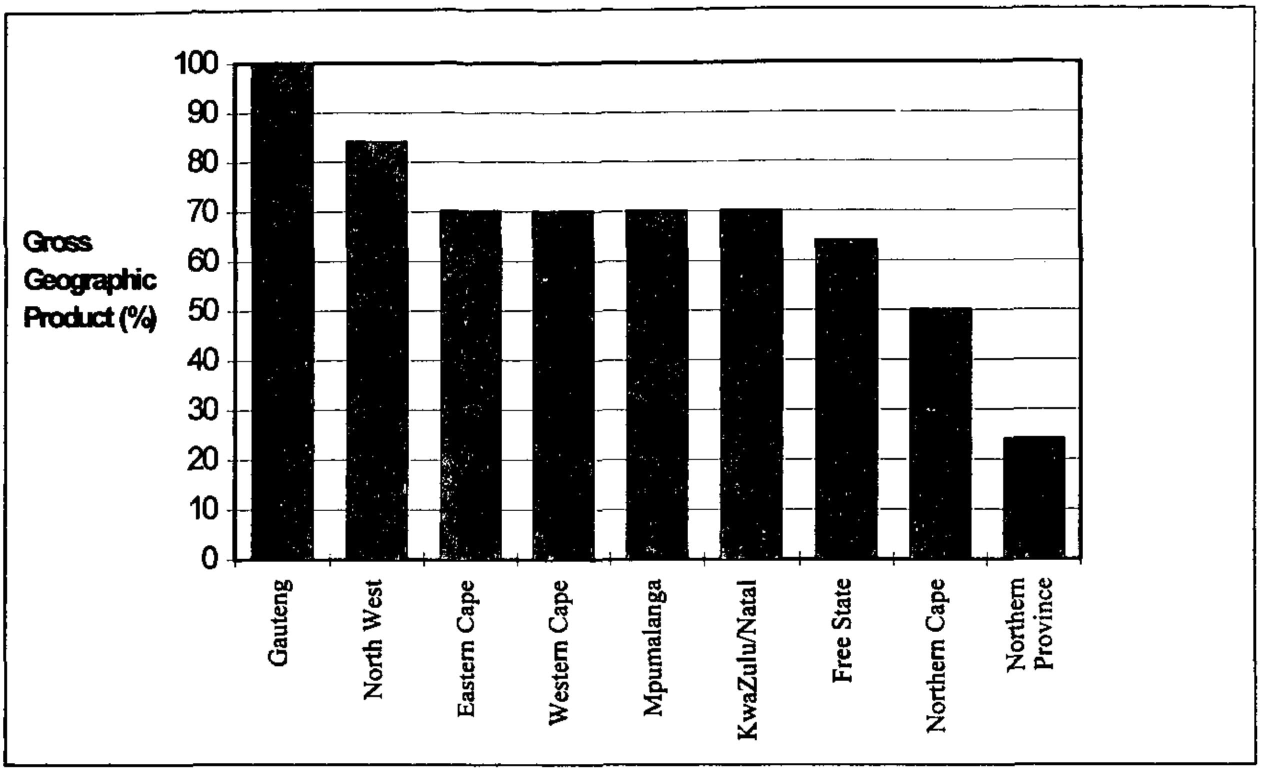


Figure 2.2 A graphic presentation of the percentage of Gross Geographic Product (GGP), as an indicator of productivity per province, that is at least partly reliant on inter-basin water transfer schemes (from van Niekerk *et al.* (1996)).

The IBT scheme chosen for study was the Riviersonderend-Berg-Eerste River Government Water Scheme (RBEGS), situated in the Western Cape Province (Scheme 10, Figure 2.1). The Western Cape is a winter rainfall area, with a mean annual precipitation (MAP) of *ca* 1250mm yr^{-1}. Intensive irrigation agriculture predominates, with viniculture, fruit and wheat farming forming the major activities. Recently there has been a rapid human population growth in the Cape 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 the city boundaries: demand is increasing at a rate of 4% yr^{-1} (Clayton, 1994, 1996).

The rivers now supplying this water include the Riviersonderend, Du Toits, Elands, Berg, Eerste, Palmiet and Steenbras systems. 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.

2.2 THE RIVIERSONDEREND (DONOR)

The Riviersonderend is a tributary of the Breede River in the Western Cape (Figure 2.3). The river flows eastwards from its source at an altitude of approximately 1590 mAMSL in the Groot Drakenstein Mountains. For the first 15 km it flows through the Riviersonderend Gorge within the Hottentots Holland Nature Reserve and State Forest (Figure 2.3). 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 such as *Acacia mearnsii*, and, in many places, is choked by the indigenous palmiet reed, *Prionium 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 good, and the waters are darkly stained by humic acids, which is typical of rivers that 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 is especially darkly stained in winter, when the increased runoff carries large quantities of humic acids into the river.

2.3 THE THEEWATERSKLOOF IMPOUNDMENT (DONOR)

Completed in 1980, Theewaterskloof Reservoir is situated in the Riviersonderend valley near the Western Cape town of Villiersdorp. The earthfill dam wall, 37.5 m high, impounds a gross storage volume of 482 x 10^6 m^3, and at full supply level the reservoir covers an area of 5100 ha (Department of Water Affairs and Forestry, 1989; Bath, 1993; Figure 2.3; Plate 3). Five rivers feed Theewaterskloof, namely the Riviersonderend, Du Toits, Elands, Waterkloof and Amandelhout rivers.

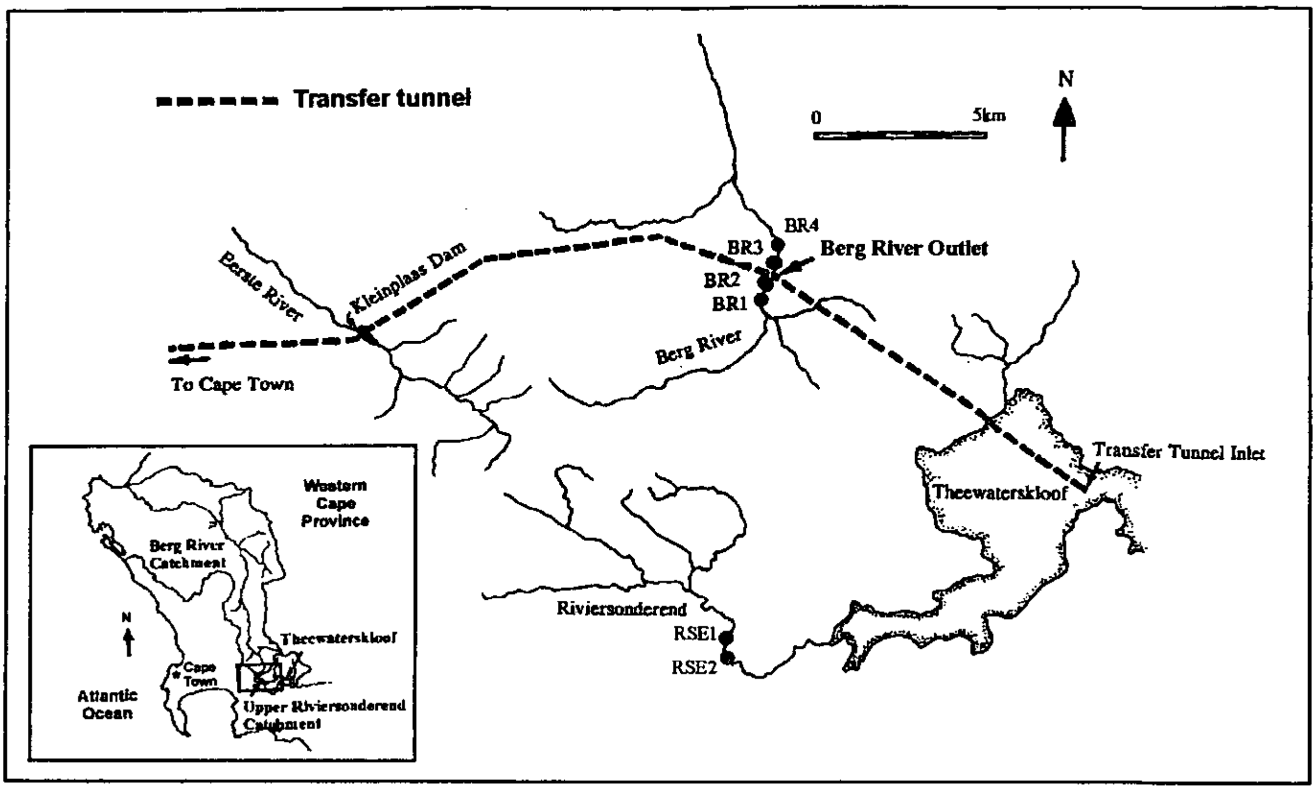


Figure 2.3 A map of the Rivierdonderend-Berg-Eerste River Government Water Scheme, situated in the Western Cape, indicating the location of the four sampling sites. The two Riversonderend sites were dropped from the analysis in 1995.

Chemical and physical data were not collected from the impoundment by the study team, due to equipment and time constraints, and also due to the comprehensive data-collection programme of the Scientific Services Department of the Cape Metropolitan Council. This Department collected physical and chemical data on a regular basis from the reservoir, from 1993 to 1997. Samples were taken from the shore of the reservoir and from a boat at various points.

2.4 THE BERG RIVER (RECIPIENT)

The Berg River rises at approximately 1500m AMSL, in the Groot Drakenstein Mountains, Western Cape (Figure 2.3). 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. 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, that 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 2.3; Plates 1 and 2). The riparian vegetation of the study area is dominated by exotic pines and acacias (respectively, *Pinus radiata* and *Acacia longifolia*), with a few indigenous species such as wild almond (*Brabejum stellatifolium*) and the lance-leafed myrtle (*Metrosideros angustifolia*). 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, *P. serratum*, 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 2.3) is 152.3 x 10^6 m^3 yr^{-1}.

Over the last few decades the Berg River has been well-studied, and, like the Great Fish River, the Berg River fortunately has a detailed historical database (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 exotic afforestation, agricultural runoff and trout farm effluent, and the transfer of water from Theewaterskloof (Dallas, 1992).

2.5 THE 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). The floodwaters of the upper reaches of the Riviersonderend are regulated and stored in Theewaterskloof Reservoir. From Theewaterskloof, water is transferred (maximum 30 m^3 s^{-1}) in a 12 km-long tunnel through the Franschhoek Mountains to the upper Berg River (Figure 2.3; 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 2.4), with an average discharge of *ca* 3.5 m^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 13 km westwards to a balancing dam (Kleinplaas Dam: capacity 3.77 x 10^5 m^3) on the Eerste River, and then 5 km through the Stellenbosch Mountain to Blackheath Purification Works (Figure 2.3), 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.

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 from the upper Berg River. However, the Skuifraam Dam is due to be built on the upper Berg River in the La Motte Forest Reserve in early 2000, in order to supply water to the Cape Metropolitan Area (CMA) (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.

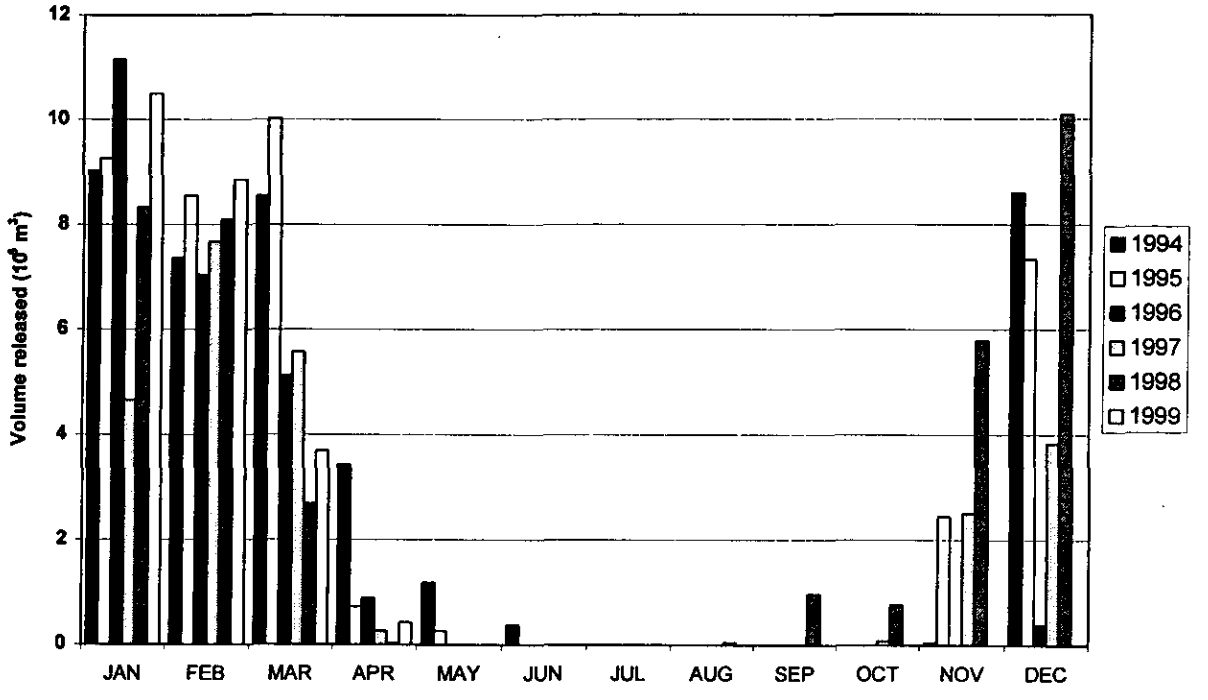


Figure 2.4 Water transfers through the RBEGS tunnel into the upper Berg River are seasonal, with releases reaching a maximum in mid-summer - January to March - and ceasing during winter.

2.6 SITE SELECTION

In order to investigate the overall ecological effects of the RBEGS IBT, five sampling sites were initially selected, three on the upper Berg River (recipient system) and two on the Riviersonderend (donor) (Figure 2.3). In August 1995, after preliminary analysis of the data collected in 1994 and early 1995, a further site was selected on the Berg River, in order to investigate spatial recovery of biotic communities below the IBT. Furthermore, the Riviersonderend sites were dropped from the analysis, as water transferred into the Berg River is taken from an impoundment, and not directly from the river, and thus, there is no link between the physical, chemical and biotic nature of the Riviersonderend sites and the receiving reaches of the Berg River. The chemical nature of water in the Theewaterskloof impoundment is of greater importance to this study. Data from the Riviersonderend sites have been analysed and presented in Snaddon (1998).

Two of the upper Berg River sites (BR1 and BR2) were situated respectively 500 m and 200 m above the siphon (Figure 2.3, Plate 4), both above a small weir, while the third (BR3) and fourth (BR4) sites were selected respectively approximately 100 m and 800 m below the siphon (Figure 2.3, Plate 2). The choice of two unimpacted or control sites was based on the advice of Underwood (1993), who suggested that the choice of several unimpacted control sites 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 to 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 and BR4) sites. Variations between the two impacted sites were assessed in separate analyses, in order to gauge the extent to which the river had recovered (i.e. returned to its unimpacted state) with distance from the IBT.

All four Berg River sites lie within the Upper-Foothill-Stony-Run-Zone of Harrison & Elsworth (1958), which is characterised by a gentle gradient and river reaches that comprise long runs and riffles, interspersed with pools (Plate 4). The substratum is dominated by cobble, with some sand, gravel and bedrock in places. 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 exceptionally pure quality. Below the siphon at site 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. The riverbanks and bed at site BR4 are undisturbed, with the exception of almost complete invasion of the riparian zone by exotic acacias.

The advantage of the location of the Berg River IBT outlet is that no other significant anthropogenic perturbations occur in these upper reaches of the river, apart from afforestation. Thus, the consequences of the IBT can be assessed without the confounding effects of other impacts. The effects of afforestation are assumed to be uniform throughout the study reaches and of little significance in terms of the physical, chemical and biotic comparisons made in this report.

2.7 SAMPLING FREQUENCY

All sites 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 August 1997, inclusive. The months during which the IBT was releasing water into the Berg River were termed "summer" months (generally November through to May each year), while the remaining months were termed "winter" months.

3.1 METHODS

3.1.1 Data collection – Physical variables

The physical variables measured at each site were discharge, depth profile, stream width and water temperature. Depth and width profiles were selected for the calculation of dicharges, and not to typify each site. 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 propeller-conversion equations to convert multiple flow/depth measurements along channel profiles of known width at each site. Stream width was measured using a standard tape measure. Discharge, depth and stream width were measured at only one site above the IBT (BR1), while measurements were taken at both BR3 and BR4, below the IBT. Averages were calculated for stream depth from readings taken at regular intervals across each profile. Stream widths and Water temperature was measured at all sites using an alcohol thermometer, accurate to within 0.5°C.

3.1.2 Data collection – Chemical variables

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) were measured by 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 1 mg, and TDS was recorded as mg l^{-1}. Total Suspended Solids (TSS) were 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 as 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.

3.1.3 Data analysis – Physical and chemical variables

The physical attributes and water chemistry of the 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). Due to the inclusion of two unimpacted sites in the sampling programme, t-tests were performed between unimpacted sites, and again between one unimpacted site (consistently chosen as BR1) and each of the impacted sites (BR3 and BR4). This was done in order to determine (a) whether significant differences between the impacted and unimpacted sites were due to the IBT, and not simply due to natural fluctuations in measured variables, and (b) whether there was any significant recovery, or reset, at BR4. The data were also tested for significant differences between the two impacted sites, BR3 and BR4.

3.2 RESULTS

The physical and chemical data collected over the sampling period from the upper Berg River are graphically presented in Figures 3.1 to 3.5. Table 3.1 provides the results of statistical tests between various sites, and Appendix B contains the raw data. Concentrations of phosphate and nitrate ions were almost negligible at all sites and thus, are not dealt with further.

Almost all of the physical and chemical variables measured below the IBT (BR3 and BR4) while it was in operation showed marked and significant deviations from the unimpacted conditions at sites BR1 and BR2 (Table 3.1; Figures 3.1 to 3.5). An exception was the width of the river at the sampling sites: this would be expected the width of the sites varied considerably between sites. Furthermore, temperature was not significantly different at BR3, and the concentration of potassium ions was not significantly different at BR4. Differences between the unimpacted site BR1 and the impacted site BR3 were, in most cases, highly significant, while differences between BR1 and BR4, further downstream, were generally significant (Table 3.1). In all significant comparisons, an increase in each variable was measured below the IBT (Figures 3.1 to 3.5). The two impacted sites, BR3 and BR4 were not significantly different in terms of most of the measured variables, except for pH, conductivity, temperature and TDS.

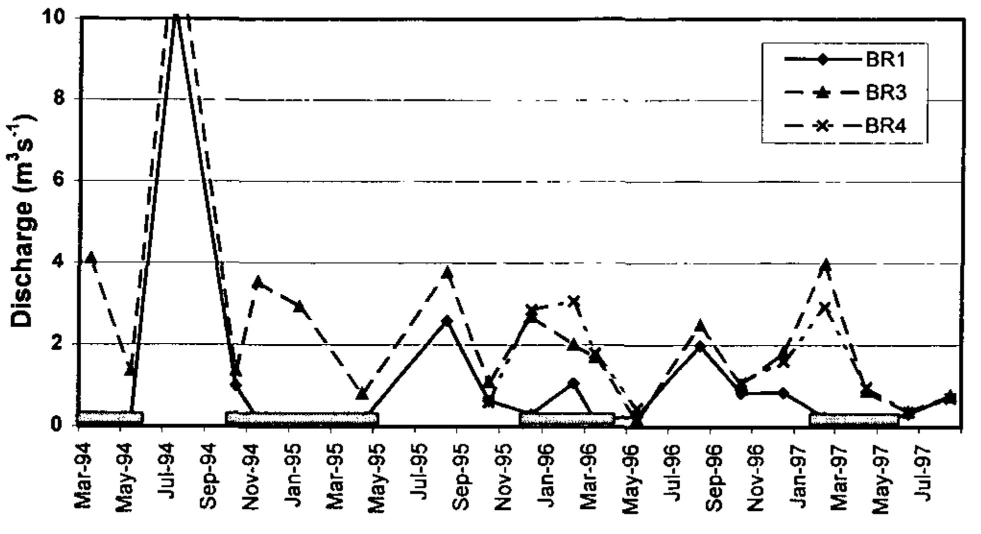
In winter, when the IBT was not releasing water, there were few significant differences between the impacted and unimpacted sites, with the exception of discharge (greater at BR3), depth (shallower at BR4), width (narrower at BR3 and BR4), pH (greater at BR3 and BR4) and temperature (lower at BR3, higher at BR4) (Table 3.1; Figures 3.1 and 3.2). There were no significant differences between the two impacted sites in winter.

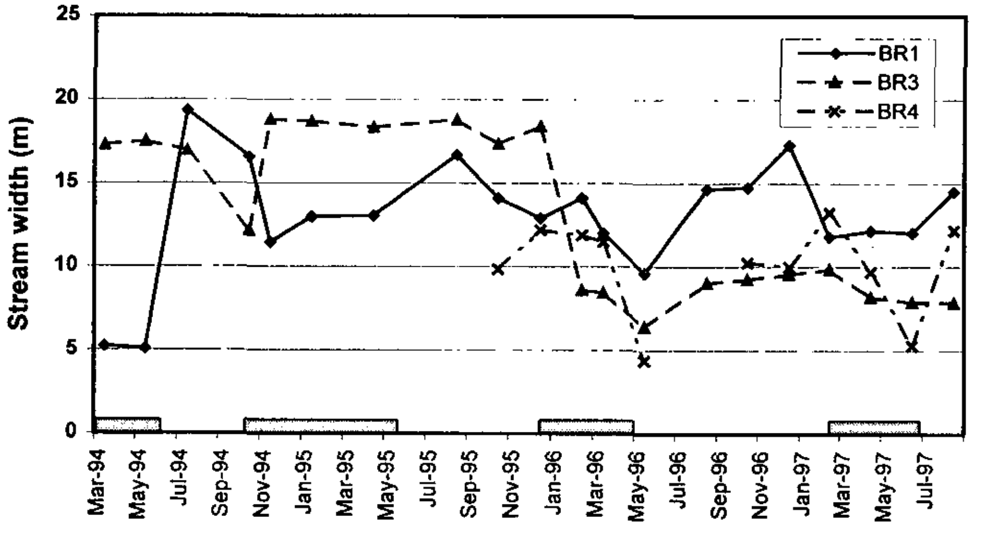
Physical and chemical conditions at sites BR1 and BR2 were similar in both seasons, with the exception of a significantly higher chloride ion concentration at BR2 in summer, and a significantly higher calcium ion concentration at BR1 in winter (Table 3.1).

Table 3.1 Statistical results for comparisons between physical and chemical variables measured at each site. Paired t-tests were used to test for significant differences between the means of each site. NS: not significant ($p>0.05$); *: significant ($0.05>p>0.001$); ***: highly significant ($p<0.001$). Discharge, depth and width were not measured at BR2.

SUMMER IBT ON	Variable	BR1 vs BR2		BR1 vs BR3		BR1 vs BR4		BR3 vs BR4	
		t-statistic	p	t-statistic	p	t-statistic	p	t-statistic	p
	discharge			-5.288	***	-5.850	***	-0.219	NS
	depth			-2.582	*	-4.711	*	1.556	NS
	width			-1.592	NS	1.256	NS	-0.550	NS
	pH	-1.329	NS	-7.509	***	-8.725	***	-3.422	*
	conductivity	-1.427	NS	-4.515	***	-7.568	*	-3.701	*
	temperature	0.993	NS	0.591	NS	-2.816	*	-3.513	*
	TSS	0.595	NS	-5.871	***	-4.605	*	-0.031	NS
	TDS	-1.103	NS	-4.851	***	-5.006	*	-4.337	*
	chloride	-2.147	*	-5.226	***	-7.851	*	-2.407	NS
	sulphate	-1.271	NS	-5.514	***	-8.624	*	-3.489	NS
	sodium	0.271	NS	-5.407	***	-3.217	*	0.556	NS
	potassium	0.882	NS	-3.249	*	-0.765	NS	-0.127	NS
	magnesium	0.617	NS	-5.113	***	-3.803	*	-0.900	NS
	calcium	-0.272	NS	-6.279	***	-4.184	*	-1.930	NS

WINTER IBT OFF	Variable	BR1 vs BR2		BR1 vs BR3		BR1 vs BR4		BR3 vs BR4	
		t-statistic	p	t-statistic	p	t-statistic	p	t-statistic	p
	discharge			-2.865	*	-1.577	NS	0.697	NS
	depth			0.910	NS	3.142	*	1.100	NS
	width			3.027	*	6.809	*	0.641	NS
	pH	0.081	NS	-2.765	*	-2.579	*	-0.944	NS
	conductivity	0.221	NS	1.305	NS	2.313	NS	-0.326	NS
	temperature	0.461	NS	2.553	*	-2.784	*	-1.887	NS
	TSS	-0.841	NS	1.230	NS	0.034	NS	-0.692	NS
	TDS	0.726	NS	-0.282	NS	-0.428	NS	-0.120	NS
	chloride	0.845	NS	1.136	NS	0.153	NS	-0.554	NS
	sulphate	0.435	NS	0.174	NS	-0.335	NS	-0.414	NS
	sodium	-0.244	NS	-1.050	NS	1.163	NS	0.816	NS
	potassium	0.009	NS	-1.361	NS	0.112	NS	-0.595	NS
	magnesium	1.612	NS	1.285	NS	1.819	NS	2.234	NS
	calcium	2.002	*	0.478	NS	1.412	NS	0.272	NS





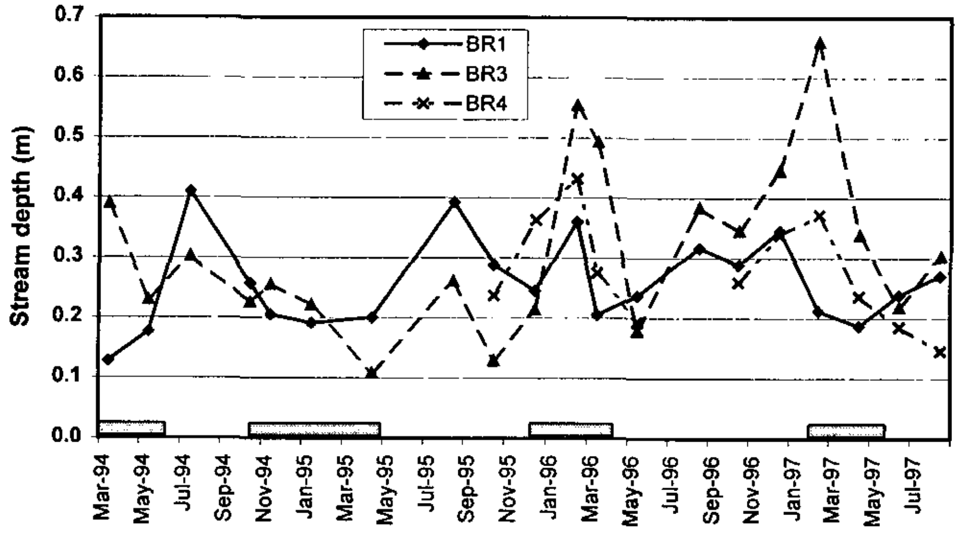
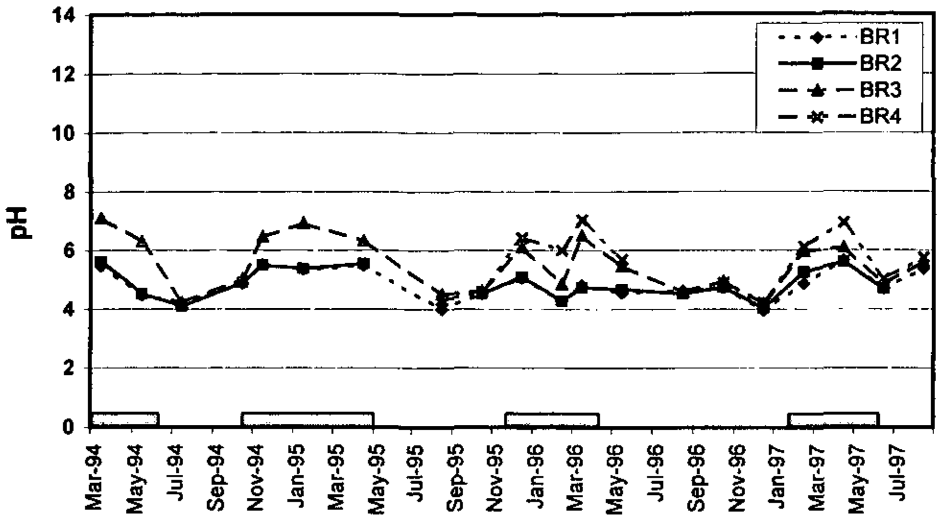
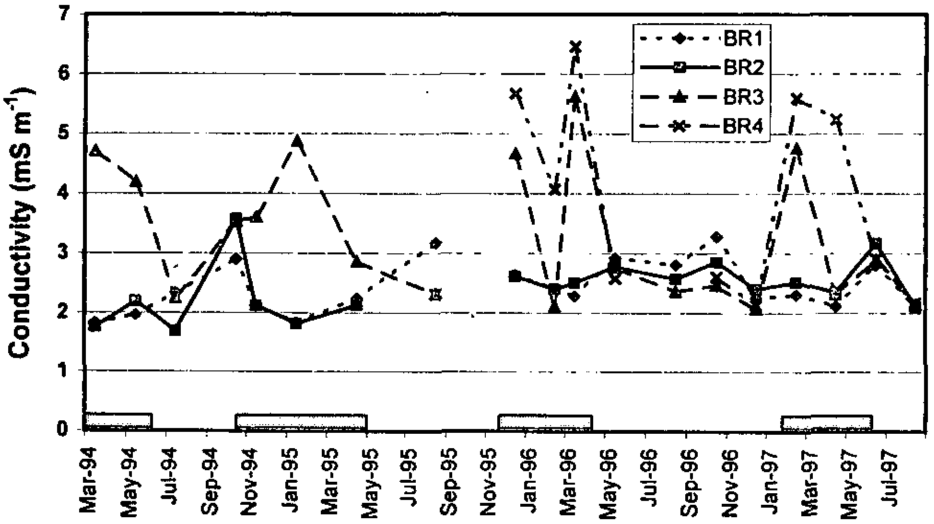


Figure 3.1 Discharge, stream depth and width at three of the Berg River sites over the study period, March 1994 to August 1997. Physical measurements were not taken at BR2. *The shaded bars indicate periods of water release.*





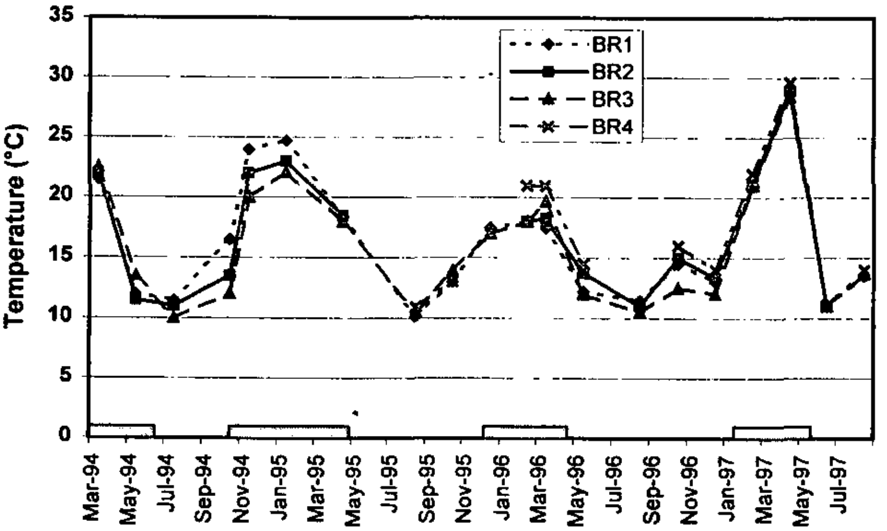
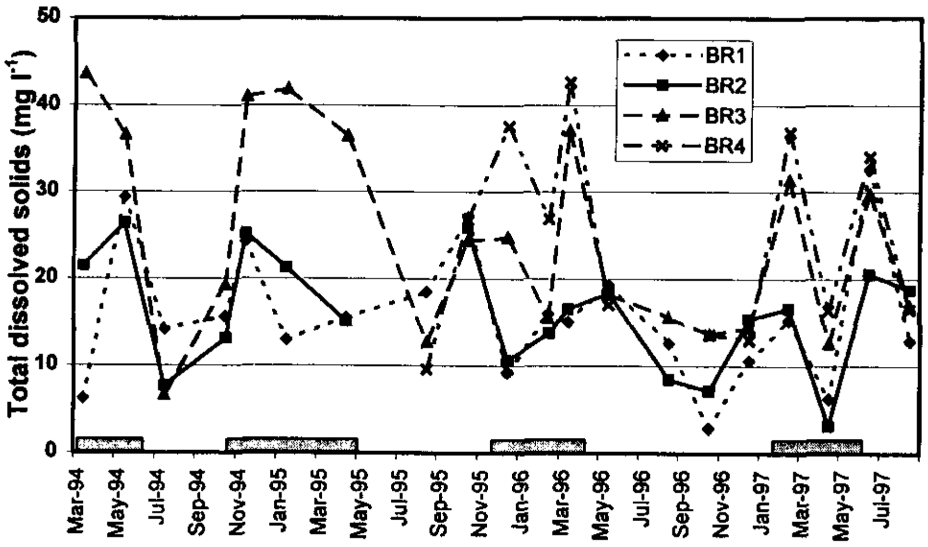


Figure 3.2 pH, conductivity and temperature measurements from each of the Berg River sites over the study period, March 1994 to August 1997. The shaded bars indicate periods of water release.



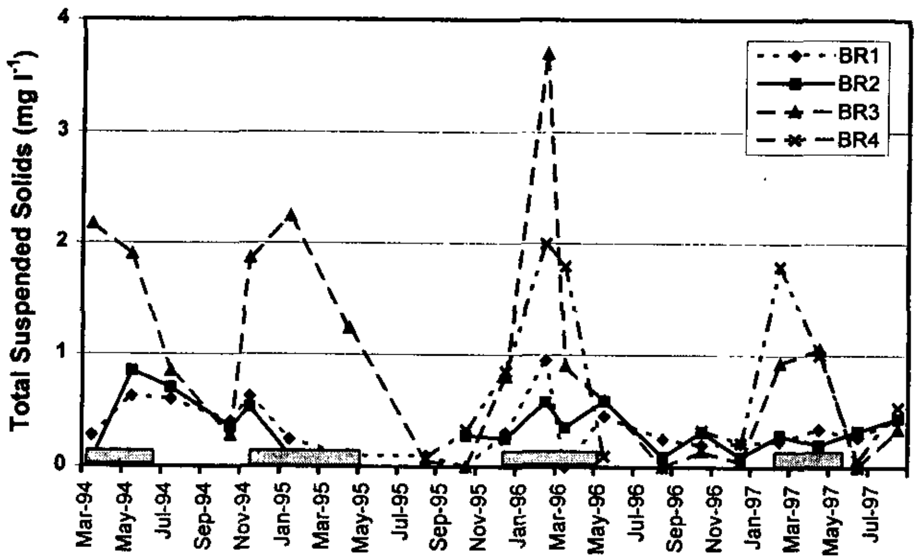
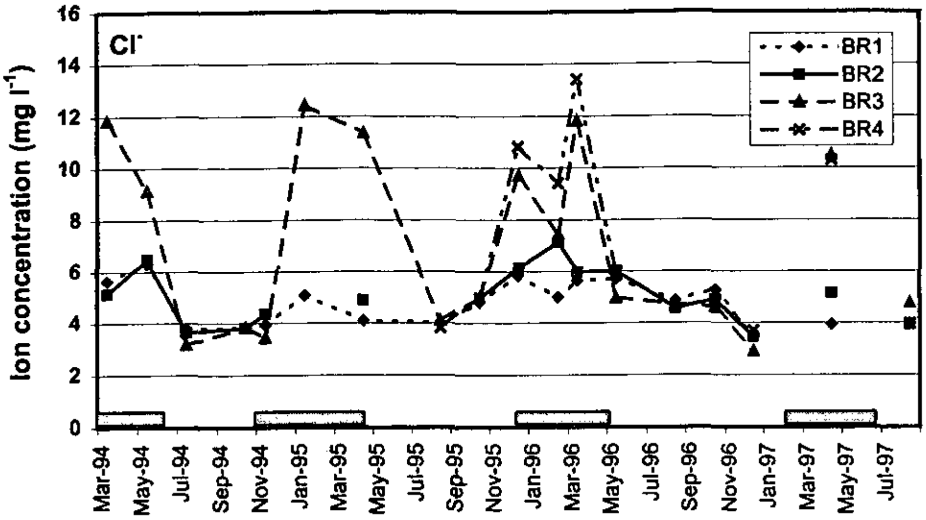


Figure 3.3 Concentrations of total dissolved and suspended solids (TDS and TSS) from each of the Berg River sites over the study period, March 1994 to August 1997. The shaded bars indicate periods of water release.



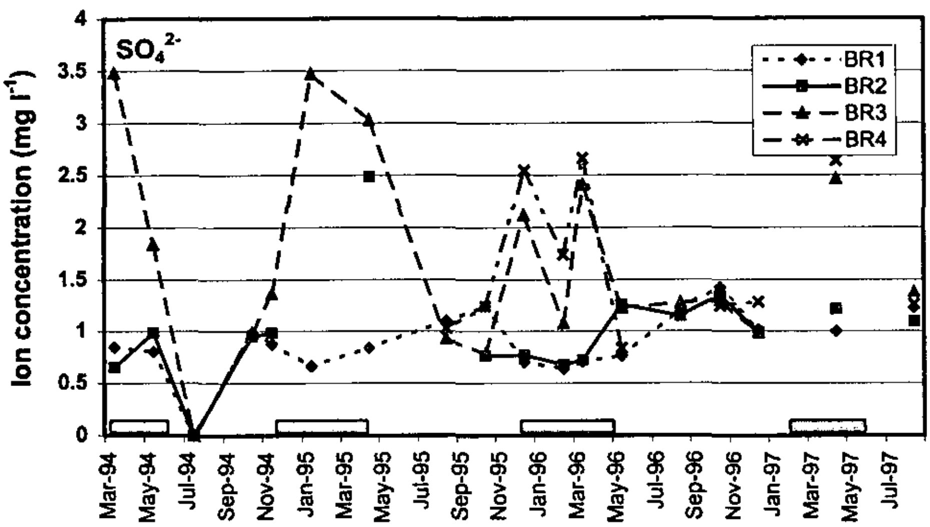


Figure 3.4 Anion concentrations at each of the Berg River sites over the study period. The shaded bars indicate periods of water release.

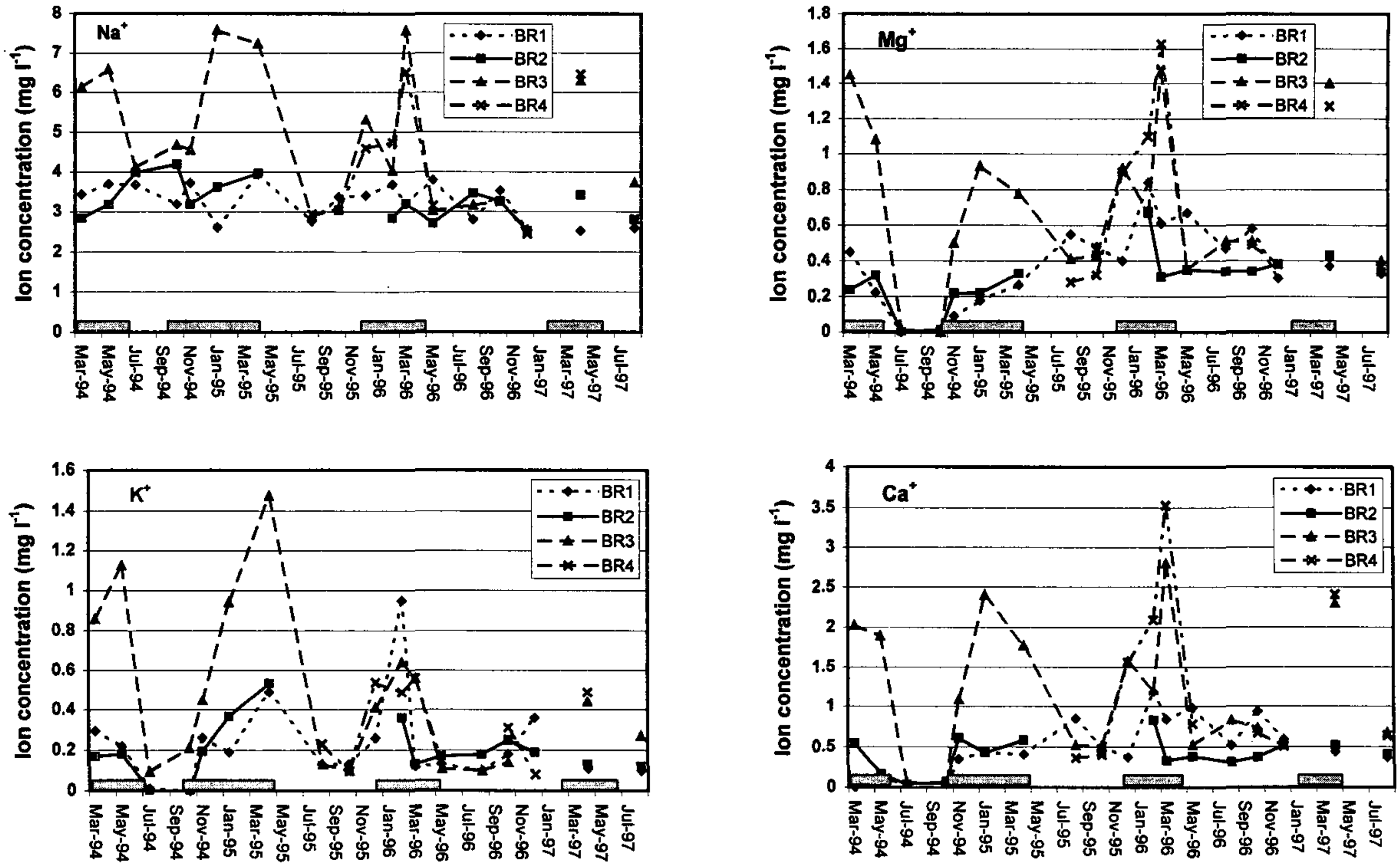


Figure 3.5 Cation concentrations at each of the Berg River sites over the study period. The shaded bars indicate periods of water release from the IBT.

3.3 DISCUSSION

The effects of river regulation by impoundment are well-known (e.g. Ward & Stanford, 1979, 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). It is clear from the literature dealing with IBTs, that flow alterations are a significant feature of these water developments (Snaddon *et al.*, 2000). 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 3.6, which illustrates daily flow data collected at a flow gauging station on the upper Berg River at Driefontein (station G1H004) during three periods of four years each: 1975 to 1978 (pre-IBT), 1981 to 1984 (immediately post-IBT construction) and 1994 to 1997 (the study period). Although this gauging station is a fair distance downstream of the IBT tunnel, the data provide a comparison of flow conditions before and after the construction of the IBT, and also an indication of the extent to which the discharges measured below the IBT during operation exceeded the natural variations in flow in the upper catchment. The data show that the regulated (post-IBT construction) daily discharges in summer exceed natural discharges recorded in the river before the IBT during the same season. In summary, the magnitude of summer discharges have increased, along with their frequency and duration.

The extent of flow alteration recorded in the recipient reaches of the upper Berg River during the study period can also be described in terms of the percentage increase over the discharge measured immediately upstream of the IBT (Figure 3.1). Discharges at BR3 were generally between 600 and 2000% greater than those measured at BR1, with the exception of March 1994, when the downstream discharge was over 4400% greater than that measured at BR1, and February 1996, when the difference was only 188%. Discharges at BR4 varied similarly, and were between 800 and 1200% greater than the upstream site, with the exception of February 1996, when the discharge was 288% greater. During winter months, discharges measured at the two downstream sites remained higher than at BR1; this could probably 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, depth (Figure 3.1; Table 3.1), represented a substantial local disturbance in the recipient reaches of the river.

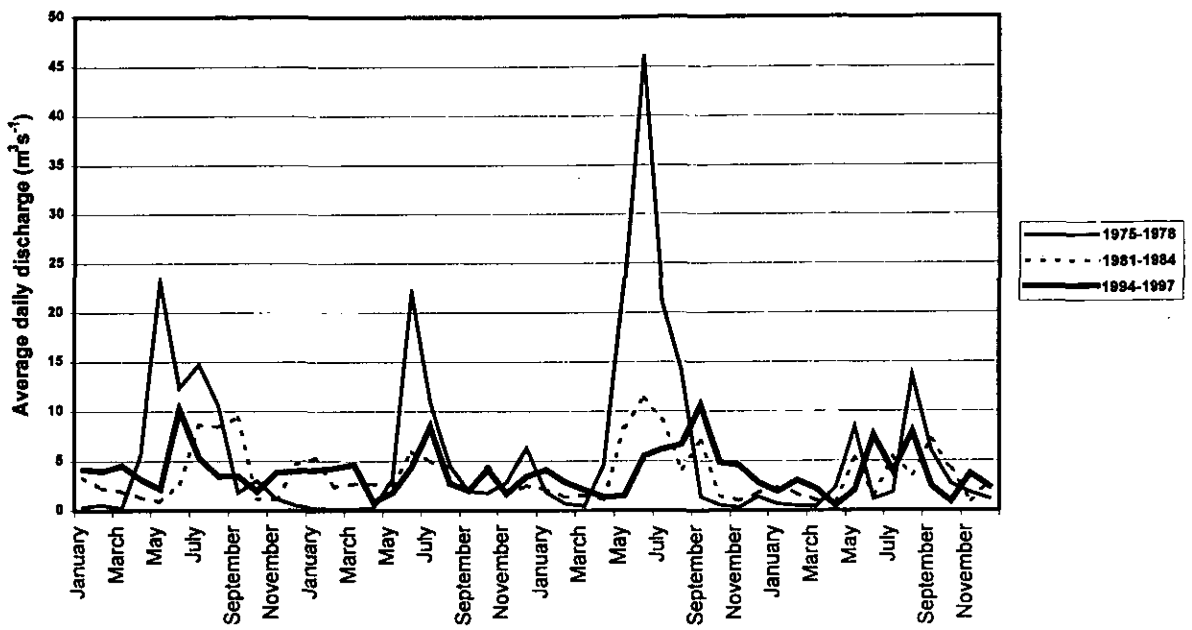


Figure 3.6 Average daily discharge data from a gauging station on the upper Berg River (G1H004), over three periods of four years each. 1975-1978: pre-IBT construction; 1981-1984: immediately after completion; 1994-1997: the study period.

The summer discharge peaks downstream of the IBT in the Berg River 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. With the exception of temperature, stream width and potassium ion concentrations, measurements of physical and chemical variables were elevated below the IBT during summer. While all of these variables are expected to increase downstream (e.g. Dallas, 1995; Harrison & Elsworth, 1958), water releases from the IBT resulted in an unnaturally marked alteration in water quality conditions within the same river zone.

The extent of the physical and chemical alterations in the Berg River attributable to the IBT appeared to be less severe at the downstream impacted site, BR4. Statistical comparisons between sites BR1 and BR4 were generally less significant than those between BR3 and BR1 (Table 3.1). This could be interpreted as a "recovery" in the physical and chemical conditions in the river to resemble upstream conditions, with distance from the release structure. However, in most cases, conditions at BR4 were not significantly different from those at BR3, thus contradicting this interpretation. Two exceptions were noted, however, in the cases of conductivity and TDS, both measures of the total amount of dissolved material. It can be concluded that there was a significant decrease in these variables with distance downstream of the IBT. The amount of dissolved solids in the water column is an important determinant of the biotic characteristics of any aquatic environment (e.g. Dallas & Day, 1993).

It is essential to place these changes in water quality in the context of natural fluctuations in some of the major variables. This is important in the assessment of the ecological significance of the statistically significant results obtained above. For pH, temperature, TDS and TSS, the background levels and fluctuations are taken to be those measured above the IBT, while the South African Water Quality Guidelines for Aquatic Ecosystems are used to determine the extent to which these levels and fluctuations can change without substantial deterioration in ecological functioning (Department of Water Affairs and Forestry, 1996) (Table 3.2). Water Quality Guidelines for the remaining water quality variables measured during this project have not been determined, and thus, these are not considered further.

Table 3.2 Some of the South African Target Water Quality Guidelines for Aquatic Ecosystems.

Variable	Target Water Quality Guideline
pH	should not change from natural by >0.5 unit, or 5%, whichever is more conservative
Conductivity	no guidelines provided
Temperature	should not be allowed to vary from the background average daily water temperature considered to be normal for a specific site and time, by >2°C, or by >10%, whichever is more conservative
TDS	should not be changed by >15% from the normal cycles of the waterbody under unimpacted conditions at any time of the year, and the amplitude and frequency of natural cycles in TDS concentrations should not be changed
TSS	should not change by >10% of background concentrations at a specific site and time

The release of water from the RBEGS IBT led to changes in pH, TDS and TSS that exceed the target water quality guidelines. Furthermore, pH, conductivity and TSS exceeded the average values and standard deviations for these variables measured throughout the southern and western coastal water quality management region (or fynbos bioregion) (Dallas *et al.*, 1998). It is unlikely that the changes in water quality varables measured here would lead to chronic or acute effects on aquatic organisms, but they may cause changes in ecosystem structure and functioning (Dallas *et al.*, 1998), and may be responsible for the altered community composition at the impacted sites, described in the following chapter.

In order to understand the changes in water chemistry recorded in the recipient reaches of the Berg River, it is important to take a look at the conditions prevailing in the donor reservoir. The CMC dataset for the Theewaterskloof Reservoir is extremely inconsistent, but it provides some comparison of water chemistry between the donor reservoir and the recipient reaches of the Berg River. Just over one full year (November 1993 to December 1994, inclusive) of data could be patched together for the purposes of this study, and the following variables could be compared with data obtained during this study: pH, temperature, conductivity (Figure 3.7). Data were collected from a boat near the inlet tunnel of the IBT, and also from the shoreline in the same area (Dr Bill Harding, Southern Waters Consulting, previously of Scientific Services Department, Cape Metropolitan Council, pers. comm.).

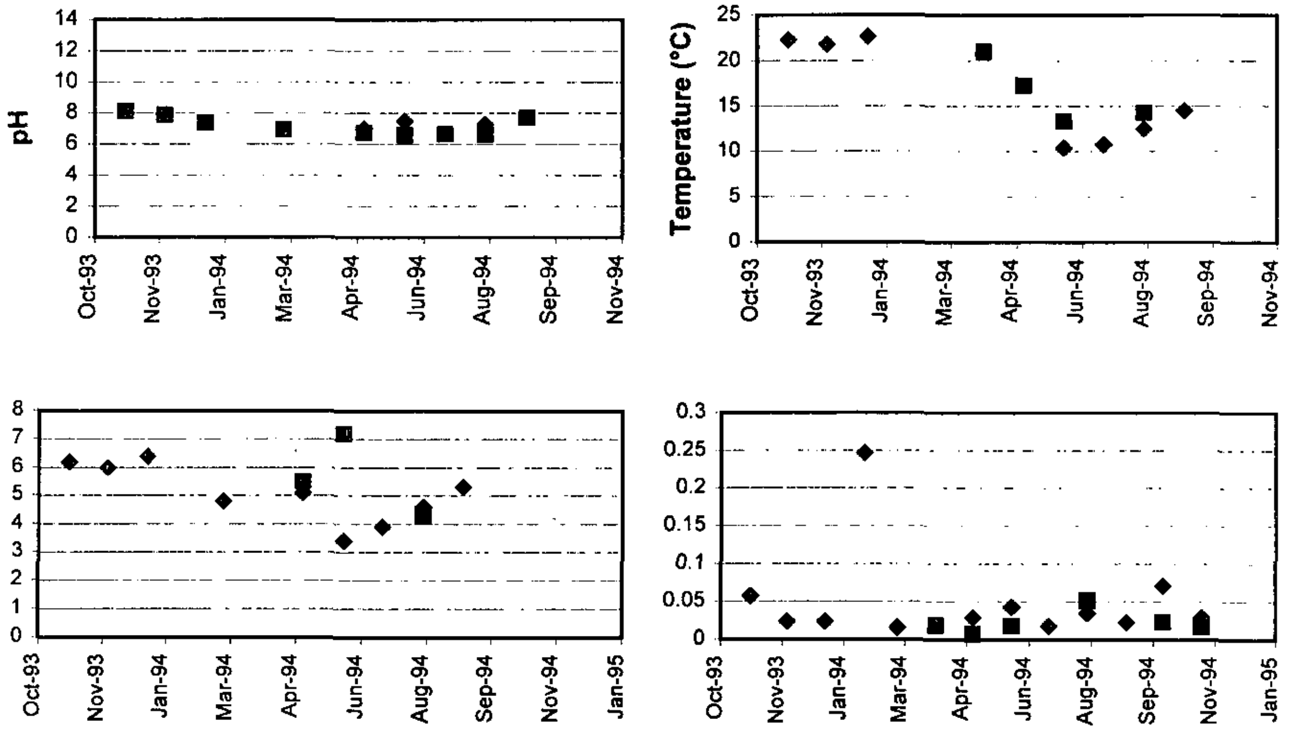


Figure 3.7 Cape Metropolitan Council dataset for the Theewaterskloof Reservoir, collected from the shoreline in the vicinity of the IBT inlet tower (solid squares) and from a boat in the same area (solid diamonds), from November 1993 to December 1994.

The pH of water in the reservoir varied between 6 and 8, which corresponds to the pH measured in the Berg River below the IBT during summer months (Figure 3.2). Similarly, conductivities varied between 3 and 8 mS m^{-1} in Theewaterskloof, which could explain the elevated conductivities at BR3 and BR4, in comparison with BR1 and BR2 where the conductivities did not exceed 3 mS m^{-1} during summer. Thus, elevations in pH and conductivity below the IBT could be linked to similar conditions in the donor reservoir. On the other hand, water temperature fluctuated on a seasonal basis in Theewaterskloof, from around 22°C in summer down to 10°C in winter, which matches the seasonal temperature fluctuations in the Berg River (Figure 3.2).

Another variable of interest in Theewaterskloof was total phosphorus (Figure 3.7), as an indication of the trophic status of the reservoir and the potential for eutrophication of the Berg River. With the exception of February 1994, total phosphorus concentrations did not reach levels of nutrient enrichment (i.e. all values <0.06 mg l^{-1}; Dallas & Day, 1993). In the past, however, inundation of vegetation during the filling of Theewaterskloof, and the subsequent decomposition, have increased the nutrient loading of the reservoir to the extent that populations of the alga, *Anabaena* sp., have been sustained in the reservoir (Dr W.R. Harding, Southern Waters Consulting, pers. comm.). The Cape Metropolitan Council, responsible for water quality in the Cape Metropolitan Area (CMA), has had taste and odour problems with the water supplied by the RBEGS. 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.

It is clear that it is essential to be able to monitor the water chemistry of the donor reservoir, in order to predict changes in the receiving system.

Finally, an aspect of water quality problems associated with IBTs, but not investigated here, is the change in water chemistry inside the transfer tunnels and canals. For instance, water quality in lined canals has been the object of study by several authors, in particular by Oksiyuk *et al.* (1979; 1981) on the Severskiy Donets-Donbas Canal in Russia, where filamentous algae were identified as the main cause of water quality deterioration in the canal. It has also been suggested that the Eastern National Water Carrier in Namibia is 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). This is an important aspect of IBTs which requires monitoring, especially for intermittent systems where water may be left to stagnate for considerable periods (e.g. Huntingdon & Armstrong, 1974; Snaddon *et al.*, 2000).

4.1 METHODS

4.1.1 Data collection

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 0.1 m^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 15 cm. 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% ethyl alcohol in the laboratory, within 7 days of collection. Laboratory treatment included fractionation into four size categories: >950 µm fraction, between 500 µm and 950 µm, between 250 µm and 500 µm, and a fraction between 80 µm and 250 µm, which was curated but not analysed. The invertebrate instars found in this size fraction generally are too immature for accurate identification. Some size 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 investigate the responses of the invertebrate communities of the Berg River to an IBT, and the extent to which these communities "recovered" to resemble upstream conditions, during months when the IBT ceased to operate. 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, but 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. However, the researchers are aware of the fact 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).

In addition to the identification of invertebrate taxa, the project team was requested by the Steering Committee to retrospectively apply the South African Scoring System, Version 4 (SASS4) of Chutter (1998) to the data collected using the box sampler, as described above. This was done in order to gain additional insights into the comparative differences in invertebrate communities between affected and unaffected sites within the recipient river reaches. The accepted SASS4 protocol enables a rapid, on-site assessment to be made, either from a single or a variety of biotopes, using a non-destructive kick-sampling technique that disturbs the stream bed so that invertebrates are dislodged and retained on a 950μm-mesh sieve. The sample is placed in a basin and each taxon recorded and then returned to the sample site. The SASS4 system allocates a predetermined score for each taxon according to its normal distribution and sensitivity to disturbance. Sensitive taxa are allocated high scores (maximum of 15) while taxa more common to degraded/disturbed systems receive low scores. In this manner, an accumulative total-SASS4 score is developed for each site, enabling direct and objective assessments to be made across sites. In addition to total-SASS4 scores per site, Average Scores Per Taxon (ASPTs) can also be generated: a low score will indicate that the community is dominated by species resistant to anthropogenic perturbations such as pollution and river regulation, while high scores indicate the occurrence of more sensitive and, often rare, species, that would be expected to occur in undisturbed systems. For this study, SASS4 analyses were applied to the data only after the collection of quantitative box-sampler material – scores were calculated from the >950μm fraction.

The SASS is designed primarily for the assessment of the effects of changes in water quality, but changes in habitat diversity can have a significant effect on the interpretation of results. In this instance, however, only one biotope was sampled – the stones-in-current or riffle biotope. Thus, it must be emphasised that the scores obtained here are not comparable with SASS4 data collected elsewhere and interpretation of the data should be done with caution.

4.1.2 Data analysis

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 be 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 to 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. The proportional abundances of some 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 can 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 paired t-test for differences between sample means. The t-test was used on the assumption that none of the river sites was independent of the other sites (see Section 3.1.3).

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 multi-dimensional scaling (MDS) (see below - Multivariate representation of the data). The null hypothesis was that there were no significant differences in benthic invertebrate community structure 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 samples between groups, r_W is the average of all rank similarities among groups, and where $M = n(n-1)/2$ 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 samples within groups are more similar to each other than to any samples from different groups. It will approximate zero when there are, in fact, no differences between groups of 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. A one-way 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. The ANOSIM routine was performed on two subsets of the invertebrate community data: (1) summer samples taken at BR1, BR2, BR3 and BR4, (2) winter samples taken at the same sites. In all cases, the critical significance level was fixed at 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. In all cases, only the benthic invertebrate were used.

Multivariate Representation of the Data Two multivariate methods were used in this study in order to develop graphic representations of the benthic 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. 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:

$$Sjk = 1 - \delta jk$$

where:

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

and Y_{ij} is the abundance of taxon *i* in the *j*th sample, Y_{ik} is the abundance of taxon *i* in the *k*th 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 similarity 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:

$$\delta_{jk}(i) = \frac{100.|Y_{ij} - Y_{ik}|}{\sum_{i=1}^{p}(Y_{ij} + Y_{ik})}$$

$\delta_{jk}(i)$ 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 summer (IBT releasing water) benthic invertebrate data from the Berg River, in order to investigate which invertebrate taxa in the Berg River were responding to the IBT.

4.2 RESULTS

In summer, the invertebrate communities of the unimpacted reaches of the Berg River (sites BR1 and BR2) were dominated by the following taxa (Table 4.1; Appendix C):

- Baetidae, Heptageniidae, Ephemerellidae and Leptophlebiidae (Ephemeroptera)
- Elmidae larvae and adults, Hydraenidae adults, Helodidae larvae (Coleoptera)
- Chironomidae and Simuliidae (Diptera)
- Acarina

The above taxa are typical of the diverse fauna occurring in the upper reaches of rivers in the Western Cape. During the same season, the Berg River sites below the IBT were dominated by the following taxa (Table 4.1):

- *Hydra* sp.
- Cladocera
- Copepoda
- Chironomidae and Simuliidae (Diptera) and, in some months, Chaoboridae (Diptera)

- Hydropsychidae (Trichoptera), especially during the summer months of 1994, 1995 and 1996.

The attached cnidarian, *Hydra* sp., and the zooplanktonic Cladocera, Copepoda and Chaoboridae are all zooplanktonic groups that are commonly found in the lentic waters of lakes and impoundments. These groups were introduced into the river through the IBT tunnel, from the donor Theewaterskloof Reservoir. These animals were sampled in large numbers, and many were observed to be alive on release into the river. Turning to the benthic communities, the Chironomidae and Simuliidae continued to dominate, but did not consistently decrease or increase in numbers below the IBT (Table 4.1). Furthermore, there were a few taxa that were only recorded or which were consistently more abundant below the IBT in summer; these were the Caenidae (Ephemeroptera), Polycentropodidae (Trichoptera) and Libellulidae (Odonata).

There were a number of taxa that were recorded above the IBT in summer, but which were noticeably reduced in numbers below it; these included (Table 4.1):

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

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, Petrothrincidae, Glossosomatidae and Lepidostomatidae (Trichoptera), Limnichidae and Gyrinidae (Coleoptera), and the Dixidae (Diptera).

During the winter months, there were no noticeable differences between the invertebrate communities sampled above and below the IBT, and these communities generally were dominated by the following taxa (Table 4.1):

- Chironomidae (Diptera)
- Baetidae and Leptophlebiidae (Ephemeroptera)
- Elmidae larvae and Helodidae larvae(Coleoptera)

Invertebrates were recorded in lower numbers at all sites during winter.

Table 4.1 (overleaf) A list of the benthic invertebrate taxa collected from the four sites on the upper Berg River, that were dominant, or that responded to the IBT. The abundance of each taxon is given as individuals per m^2. BR1 and BR2 were situated above the IBT, and BR3 and BR4 below it. BR4 was sampled from August 1995.

TAXON			March 1994 BR1	BR2	BR3	May 1994 BR1	BR2	BR3	July 1994 BR1	BR2	BR3	October 1994 BR1	BR2	BR3	November 1994 BR1	BR2	BR3
Cnidaria																	
Hydrozoa	*Hydra* sp.		0	0	19213	0	0	193	0	0	0	0	0	0	0	0	10
Annelida																	
OLIGOCHAETA	Lumbriculidae		10	273	0	467	33	0	47	157	57	233	170	133	247	380	190
Arthropoda																	
Acarina			860	613	467	150	177	133	77	167	90	140	173	37	647	363	110
Crustacea (all zooplanktonic groups)																	
CLADOCERA			0	43	10153	200	40	11367	37	47	37	0	20	3	133	233	314400
COPEPODA			0	13	200937	3	7	15057	47	153	53	3	20	7	0	47	285547
OSTRACODA			65	0	43	0	0	7	7	0	13	7	0	13	0	13	27
Insecta																	
PLECOPTERA	Notonemouridae		100	83	0	23	30	0	170	217	80	197	240	80	240	113	207
EPHEMEROPTERA	Baetidae		6170	12327	3267	4570	5217	3500	197	167	197	7997	8907	4027	63867	86113	27183
	Heptageniidae		55	673	0	27	213	0	0	0	3	3	10	0	20	30	3
	Ephemerellidae		65	83	7	50	343	0	27	53	33	73	103	57	7	40	0
	Caenidae		0	0	83	0	0	0	0	13	0	0	0	3	13	0	27
	Leptophlebiidae		990	1657	10	917	703	10	20	30	20	83	27	20	320	227	43
TRICHOPTERA	Hydropsychidae		305	223	46993	173	100	5547	7	7	17	0	0	10	70	27	87
	Philopotamidae		35	150	0	970	97	0	20	10	0	0	0	0	23	3	0
	Leptoceridae		290	323	13	473	223	0	10	10	0	10	20	20	23	40	0
	Hydroptilidae		115	100	20	17	10	7	0	30	3	13	3	3	0	0	0
	Polycentropodidae		0	17	7	7	0	53	0	0	0	0	0	0	0	0	13
	Ecnomidae		10	63	0	150	53	0	0	3	0	23	7	0	33	27	3
	pupae		0	0	170	0	0	3	0	0	0	0	0	0	0	0	0
ODONATA	Libellulidae		0	0	193	7	0	27	0	0	0	0	0	0	0	0	0
MEGALOPTERA	Corydalidae		5	13	13	27	7	0	0	0	3	3	0	7	13	7	0
COLEOPTERA	Elmidae	larvae	2045	2140	350	1293	637	233	83	103	53	127	180	47	783	457	177
	Elmidae	adults	90	117	3	113	40	10	10	33	0	10	3	10	150	147	127
	Hydraenidae	adults	275	790	63	790	137	110	7	73	10	17	30	13	390	167	153
	Helodidae		10	27	3	117	93	10	50	83	17	150	197	33	130	157	30
DIPTERA	Chironomidae		33950	22147	19680	14993	14820	16517	3133	5247	2790	8757	7683	5707	27567	14063	11960
	Simuliidae		12690	11550	993	970	1040	4180	97	347	137	210	73	267	6063	3447	15043
	Athericidae		100	103	63	33	53	17	0	0	0	0	0	0	140	327	20
	Chaoboridae		0	0	823	0	0	90	0	0	0	0	0	0	0	3	4707
Collembola			0	3	3	50	13	10	570	697	763	100	113	67	13	0	13

TAXON		January 1995 BR1	BR2	BR3	April 1995 BR1	BR2	BR3	August 1995 BR1	BR3	BR4	October 1995 BR1	BR2	BR3	BR4	December 1995 BR1	BR2	BR3	BR4
Cnidaria																		
Hydrozoa	*Hydra* sp.	0	0	6200	0	0	1230	0	0	0	0	0	0	3	0	0	0	0
Annelida																		
OLIGOCHAETA	Lumbriculidae	205	40	3005	883	197	387	10	20	40	73	0	17	23	47	5	160	57
Arthropoda																		
Acarina		375	40	90	403	170	130	15	3	73	217	10	70	190	303	185	0	53
Crustacea (all zooplanktonic groups)																		
CLADOCERA		275	30	51040	40	0	80927	0	0	0	0	0	0	0	40	360	124365	210547
COPEPODA		5	30	77035	7	7	58610	5	10	0	0	0	7	40	400	665	278320	318280
OSTRACODA		0	0	120	0	0	443	0	0	0	0	0	0	0	0	0	0	0
Insecta																		
PLECOPTERA	Notonemouridae	45	0	0	47	43	0	220	143	347	797	70	213	293	603	205	260	110
EPHEMEROPTERA	Baetidae	2450	870	870	7960	6803	1493	3355	1113	2427	8103	720	13147	6567	26927	42845	9595	12400
	Heptageniidae	315	150	0	330	1607	0	25	7	110	20	140	7	0	20	15	5	13
	Ephemerellidae	215	110	5	67	270	0	405	70	130	140	470	93	130	60	110	10	53
	Caenidae	0	0	580	0	13	17	0	0	0	0	10	0	0	0	5	80	3
	Leptophlebiidae	615	40	5	1470	1107	3	90	80	160	417	30	53	10	170	155	320	43
TRICHOPTERA	Hydropsychidae	435	0	920	997	123	32870	0	0	23	13	10	3	20	100	60	15	3
	Philopotamidae	50	20	0	317	0	0	0	0	0	0	0	0	3	0	0	0	3
	Leptoceridae	325	90	20	290	503	0	0	7	10	57	20	57	43	37	55	0	10
	Hydroptilidae	10	0	105	43	33	0	0	0	0	123	20	7	0	7	0	0	0
	Polycentropodidae	0	0	5	50	7	0	0	3	7	0	10	3	3	0	0	0	0
	Ecnomidae	5	30	10	577	60	0	0	3	0	103	20	3	3	13	25	0	0
	pupae	0	0	20	0	0	247	0	0	0	0	0	0	0	0	0	0	0
ODONATA	Libellulidae	0	0	15	0	0	103	0	0	3	0	0	0	0	0	0	5	0
MEGALOPTERA	Corydalidae	5	0	0	13	37	0	0	0	0	13	0	0	3	50	10	10	3
COLEOPTERA	Elmidae larvae	355	120	120	2140	1897	340	10	20	77	853	90	87	137	923	640	175	570
	Elmidae adults	110	80	0	193	710	7	0	7	0	100	20	80	100	63	155	80	93
	Hydraenidae adults	480	10	20	173	133	23	0	3	3	90	0	43	60	307	175	85	57
	Helodidae	45	10	0	77	63	0	5	40	67	777	70	53	63	167	65	5	7
DIPTERA	Chironomidae	4535	1140	4290	7863	3420	5800	3280	2370	3687	15137	1930	12747	23530	16473	10825	18250	10293
	Simuliidae	*8390*	*2330*	*1580*	*4933*	*2493*	*7710*	*65*	*177*	*40*	*473*	*10*	*850*	*357*	*607*	*485*	*2360*	*1380*
	Athericidae	40	30	10	90	237	17	5	0	3	3	0	3	3	57	195	5	3
	Chaoboridae	0	0	940	0	0	660	0	0	0	0	0	0	0	0	0	130	43
Collembola		0	0	0	27	3	0	15	70	53	37	0	0	3	33	40	0	0

TAXON			February 1996				March 1996				May 1996				August 1996			October 1996				December 1996			
			BR1	BR2	BR3	BR4	BR1	BR2	BR3	BR4	BR1	BR2	BR3	BR4	BR1	BR2	BR3	BR1	BR2	BR3	BR4	BR1	BR2	BR3	BR4
Cnidaria																									
Hydrozoa	*Hydra* sp.		7	3	2973	18297	0	10	11617	10983	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Annelida																									
OLIGOCHAETA	Lumbriculidae		90	200	120	347	437	10	200	273	100	20	20	7	10	43	7	30	3	0	3	23	47	63	73
Arthropoda																									
Acarina			7	37	43	0	577	560	0	20	150	70	37	87	47	30	7	303	77	20	83	213	170	187	87
Crustacea (all zooplanktonic groups)																									
CLADOCERA			27	73	5707	6453	0	0	807	727	0	0	0	0	3	0	0	0	0	0	0	0	0	0	0
COPEPODA			33	20	12093	12027	0	0	7763	5183	0	3	0	40	7	0	0	0	0	0	0	0	0	0	0
OSTRACODA			0	0	0	0	0	30	0	0	0	0	0	0	0	0	0	0	0	0	7	0	0	0	0
Insecta																									
PLECOPTERA	Notonemouridae		10	47	10	0	177	95	0	0	103	50	10	10	357	120	30	1257	147	33	200	277	377	397	158
EPHEMEROPTERA	Baetidae		390	447	180	277	7850	18410	3173	3197	7827	10323	6363	3197	407	287	87	15887	8400	5543	6023	32587	35400	37523	28998
	Heptageniidae		157	167	3	3	273	665	0	7	163	517	47	0	0	3	0	20	0	0	0	20	0	0	0
	Ephemerellidae		933	437	23	140	63	255	27	53	27	303	17	23	200	110	33	130	100	27	77	17	53	47	47
	Caenidae		73	13	10	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	13	0	0	0
	Leptophlebiidae		60	180	0	0	860	440	17	20	887	730	3	13	77	67	20	60	0	7	23	47	97	53	53
TRICHOPTERA	Hydropsychidae		0	3	127	170	316	65	4227	3533	274	47	1197	990	7	7	0	7	0	0	0	3	0	27	60
	Philopotamidae		0	7	0	0	67	40	0	0	250	0	3	0	0	0	0	0	0	0	0	0	0	27	0
	Leptoceridae		150	67	0	30	187	265	20	13	420	323	23	7	13	10	7	53	13	0	13	7	17	17	3
	Hydroptilidae		257	133	73	93	13	45	13	13	10	3	0	0	3	0	0	20	0	3	0	3	0	7	0
	Polycentropodidae		0	0	87	3	0	0	0	0	220	10	0	7	0	0	0	0	0	3	3	7	0	0	0
	Ecnomidae		0	3	0	0	450	0	0	0	93	30	3	0	3	7	0	23	7	0	3	0	10	0	3
	pupae		0	0	3	0	0	0	433	73	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ODONATA	Libellulidae		0	0	33	0	0	0	20	7	0	0	0	7	0	0	0	0	0	0	0	0	0	0	0
MEGALOPTERA	Corydalidae		3	7	0	0	17	5	3	0	13	0	3	13	3	0	0	0	0	0	0	0	0	3	0
COLEOPTERA	Elmidae	larvae	257	500	63	237	1383	970	407	430	1360	733	353	353	43	10	27	153	7	87	123	80	137	173	265
	Elmidae	adults	0	57	0	57	157	220	0	0	167	30	33	7	3	7	3	20	13	3	20	7	7	7	27
	Hydraenidae	adults	7	10	3	17	433	85	23	37	473	80	137	157	0	3	0	50	0	30	13	50	17	150	23
	Helodidae		63	400	0	3	43	20	7	0	367	413	3	0	200	173	60	223	37	7	363	147	297	413	387
DIPTERA	Chironomidae		1557	3340	2270	3330	10857	8250	15737	7817	7620	2770	6777	6523	6217	7763	8732	10923	8117	10353	10467	1850	3067	2153	1050
	Simuliidae		810	600	9790	2003	2830	1445	680	180	957	327	33	280	97	37	67	340	30	12373	80	403	307	483	147
	Athericidae		13	17	0	10	43	75	20	10	87	47	23	13	0	0	0	0	0	0	0	3	50	7	3
	Chaoboridae		0	0	50	3	0	0	13	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Collembola			250	163	280	80	0	0	0	0	0	0	0	0	13	20	30	0	0	3	10	43	50	47	15

TAXON		February 1997 BR1	BR2	BR3	BR4	April 1997 BR1	BR2	BR3	BR4	June 1997 BR1	BR2	BR3	BR4	August 1997 BR1	BR2	BR3	BR4
Cnidaria																	
Hydrozoa	*Hydra* sp.	0	0	60	513	0	0	2853	7907	0	0	0	0	0	0	0	0
Annelida																	
OLIGOCHAETA	Lumbriculidae	235	160	0	3	10	26	0	170	40	0	36	37	27	20	3	0
Arthropoda																	
Acarina		945	277	53	17	703	303	40	43	210	65	17	27	37	60	33	23
Crustacea (all zooplanktonic groups)																	
CLADOCERA		*120*	13	*4180*	*793*	*0*	3	*8580*	633	13	*0*	*0*	27	*0*	*0*	*0*	3
COPEPODA		140	553	140443	33433	20	20	60560	4133	7	0	3	10	0	0	3	70
OSTRACODA		0	0	53	0	0	0	13	0	3	0	0	0	0	0	0	0
Insecta																	
PLECOPTERA	Notonemouridae	585	517	30	27	133	163	13	0	63	70	67	77	153	203	207	60
EPHEMEROPTERA	Baetidae	11235	14890	2587	753	3270	3747	1010	1133	4907	3615	1063	2683	507	2440	933	710
	Heptageniidae	305	67	0	0	403	517	0	0	30	115	13	0	0	0	13	0
	Ephemerellidae	470	177	173	190	157	443	160	200	173	375	37	87	83	163	193	100
	Caenidae	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*5*	*0*	*0*	*0*	*0*	*0*	*0*
	Leptophlebiidae	1030	193	7	27	1060	537	3	0	467	70	20	27	20	103	80	33
TRICHOPTERA	Hydropsychidae	135	70	13	10	210	73	120	240	137	5	7	7	0	3	0	10
	Philopotamidae	5	0	0	0	0	0	0	0	10	0	0	0	0	0	0	0
	Leptoceridae	65	170	0	0	293	147	13	10	293	20	27	54	7	27	3	3
	Hydroptilidae	0	3	0	0	37	3	350	37	40	10	0	3	7	13	7	0
	Polycentropodidae	20	0	23	0	60	20	13	0	13	5	0	0	0	0	0	0
	Ecnomidae	95	13	0	0	157	53	0	0	57	15	3	3	0	10	0	0
	pupae	0	0	0	0	0	0	7	0	0	0	3	0	0	0	0	0
ODONATA	Libellulidae	0	0	7	0	7	0	3	10	0	0	0	7	0	0	0	0
MEGALOPTERA	Corydalidae	60	13	0	0	7	10	0	0	7	0	0	3	3	0	0	0
COLEOPTERA	Elmidae larvae	555	390	27	153	713	593	290	480	843	225	123	533	40	77	20	50
	Elmidae adults	230	157	50	33	53	230	10	10	140	20	0	10	0	13	0	20
	Hydraenidae adults	815	450	13	7	230	47	3	40	37	10	7	73	7	7	10	0
	Helodidae	195	330	17	7	53	13	3	10	360	40	27	13	380	197	243	43
DIPTERA	Chironomidae	11445	5367	2773	2040	5743	4883	12363	5280	4863	6390	8933	13347	6530	7143	4637	6307
	Simuliidae	2030	2330	7277	2163	1090	233	8813	250	177	200	297	1307	70	33	190	33
	Athericidae	170	10	0	0	17	40	7	10	17	0	7	10	0	0	0	0
	Chaoboridae	30	0	1413	103	0	0	850	0	0	0	0	0	0	0	0	0
Collembola		0	0	0	0	0	0	20	0	0	0	7	13	17	0	10	3

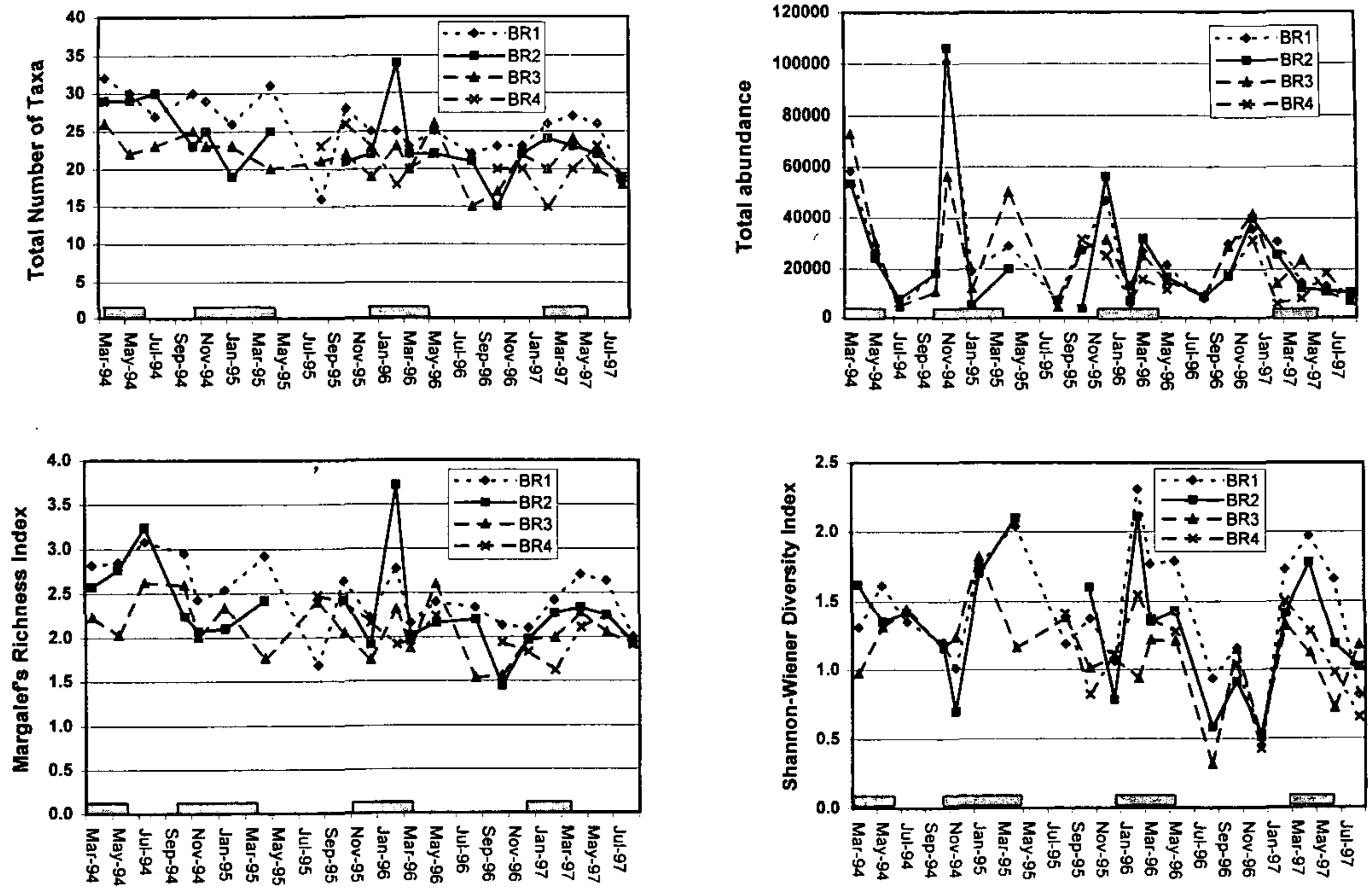


Figure 4.1 A variety of biotic indices, calculated from the benthic inverterbate data collected from four sites on the Berg River over the study period, March 1994 to August 1997. BR1 and BR2 are situated above the IBT, and BR3 and BR4 below it. The shaded bars indicate periods of water release.

Table 4.2 SASS4 total scores and ASPTs for all sites on the Berg River, sampled from March 1994 to August 1997. The shaded cells are occasions where the total score dropped below 125, and where the ASPT was close to or below 7 (see text for explanation).

	BR1		BR2		BR3		BR4	
Date	**Total score**	**ASPT**	**Total score**	**ASPT**	**Total score**	**ASPT**	**Total score**	**ASPT**
Mar-94	192	7.7	177	8.0	93	6.6		
May-94	172	8.2	173	7.9	82	6.8		
Jul-94	109	8.4	125	7.8	110	7.9		
Oct-94	117	7.8	114	8.1	100	7.7		
Nov-94	173	8.2	151	7.9	100	7.1		
Jan-95	136	8.0	86	8.6	95	6.3		
Apr-95	171	7.8	164	8.2	68	6.2		
Aug-95	95	8.6			96	8.0	135	7.9
Oct-95	158	7.9	85	8.5	148	8.2	129	8.6
Dec-95	153	7.7	135	8.4	111	7.9	150	7.9
Feb-96	113	8.1	131	7.7	93	7.2	78	7.8
Mar-96	147	8.2	132	8.8	90	6.9	83	6.4
May-96	172	8.2	144	9.0	127	8.5	115	7.7
Aug-96	106	7.6	115	7.7	88	8.0		
Oct-96	128	7.5	75	8.3	115	8.8	100	8.3
Dec-96	108	8.3	93	7.8	112	8.0	96	8.0
Feb-97	155	8.2	131	7.7	72	8.0	50	7.1
Apr-97	156	8.2	115	8.2	102	7.3	104	6.9
Jun-97	149	8.3	98	8.2	118	8.4	126	7.4
Aug-97	80	8.0	105	7.5	89	8.9	66	9.4

The total SASS4 scores varied between sites and sampling occasions, while ASPTs were fairly consistent and remained high. This may be due to the fact that only one biotope was sampled (stones in current or riffles); total scores are more affected by a lack of habitat diversity (Chutter, 1998). Thus, the interpretation of results is limited to water quality conditions, and scores cannot be used for comparison. According to Chutter (1998), a SASS4 score that exceeds 125, combined with an ASPT greater than 7, is an indication of natural water quality conditions and a high habitat diversity. Thus, most of the Berg River sites attained these conditions on most sampling occasions (Table 4.2). Lower total scores and ASPTs were recorded at BR3, in March and May 1994, January and April 1995 and March 1996, and BR4 in March 1996, and April 1997. All of these occasions were during periods of water release from the IBT. The total scores recorded on these sampling occasions were all between

60 and 125, while ASPTs were less than 7, which is indicative of some deterioration in water quality. During February 1997 at BR4, the total score dropped below 60, while the ASPT remained above 7 – according to Chutter (1998) this is indicative of a major deterioration in water quality.

On several occasions total scores were less than 125, but the ASPTs remained above 7 (Table 4.2). This is indicative of natural water quality conditions, but reduced habitat diversity (Chutter, 1998). Due to the fact that invertebrates were sampled only in the riffle biotope, these lower scores must be attributed to some deterioration in water quality, or perhaps to a reduction in the suitability of the riffle biotope.

Univariate Analyses In summer, there were two measures of biotic richness that were significantly different between the invertebrate communities above and below the IBT; these were the total numbers of taxa and Margalef's richness index. Both measures were significantly higher above the IBT than below it (Table 4.3; Figure 4.1). With the exception of a comparison between sites BR3 and BR4, total abundance of invertebrates was not significantly different between sites in summer, and biotic diversity, measured as the Shannon-Wiener diversity index, was significantly different between sites. Thus, diversity and invertebrate abundance were not affected by the IBT.

Table 4.3 Statistical results for comparisons between various biotic indices at each site. Paired t-tests were used to test for significant differences between the means of each site. NS: not significant ($p>0.05$); *: significant ($0.05>p>0.001$); ***: highly significant ($p<0.001$).

SUMMER IBT ON	Variable	BR1 vs BR2		BR1 vs BR3		BR1 vs BR4		BR3 vs BR4	
		t-statistic	p	t-statistic	p	t-statistic	p	t-statistic	p
	Number of taxa	1.584	NS	6.194	***	3.721	*	1.136	NS
	Total abundance	0.728	NS	0.495	NS	2.606	NS	5.333	*
	Margalef's Richness	1.275	NS	6.043	***	3.147	*	0.540	NS
	Shannon-Wiener Diversity	2.461	*	2.815	*	2.714	*	-2.242	NS
WINTER IBT OFF	**Variable**	**BR1 vs BR2**		**BR1 vs BR3**		**BR1 vs BR4**		**BR3 vs BR4**	
		t-statistic	p	t-statistic	p	t-statistic	p	t-statistic	p
	Number of taxa	2.507	*	2.440	*	0.824	NS	-0.764	NS
	Total abundance	1.171	NS	1.210	NS	0.998	NS	0.482	NS
	Margalef's Richness	2.791	*	1.835	NS	0.526	NS	-0.676	NS
	Shannon-Wiener Diversity	1.088	NS	1.386	NS	2.101	*	0.855	NS

In winter, few of the comparisons between sites revealed significant differences (Table 4.3). The numbers of taxa were significantly greater at BR1 than at BR2 and BR3, and Margalef's richness index was similarly greater at BR1 than at BR2. Lastly, invertebrate diversity was greater at BR1 than at BR4.

Multivariate Analyses

Testing for Significance The results of the tests for significant differences between the benthic invertebrate communities at each of the Berg River sites in summer and in winter are provided in Table 4.4. There were marked between-site differences in summer, when the IBT was releasing water, but not in winter, when the IBT ceased to operate. In summer, highly significant differences were recorded between the sites above and below the IBT, while the two sites above the IBT, BR1 and BR2, and the two sites below it, BR3 and BR4, were not significantly different from each other.

In winter, although the global test indicated no significant differences between sites, BR2 and BR3 were significantly different.

Table 4.4 Statistical results of the one-way ANOSIM analysis, providing global and pair-wise results. NS: not significant ($p>0.05$); ***: highly significant ($p<0.001$).

SUMMER – IBT ON Global R-statistic = 0.462; ***		
BR1 vs BR2	0.018	NS
BR1 vs BR3	0.646	***
BR1 vs BR4	0.702	***
BR2 vs BR3	0.716	***
BR2 vs BR4	0.759	***
BR3 vs BR4	0.209	NS
WINTER – IBT OFF Global R-statistic = 0.037; NS		
BR1 vs BR2	-0.009	NS
BR1 vs BR3	0.039	NS
BR1 vs BR4	0.037	NS
BR2 vs BR3	0.123	*
BR2 vs BR4	0.084	NS
BR3 vs BR4	-0.051	NS

Multivariate Representation of the Data Figures 4.2 to 4.7 illustrate the results of the cluster and ordination analyses of the aquatic invertebrate community data. In an initial procedure in which all of the invertebrate community data were compared with each other, two strong patterns emerged. A seasonal separation of communities is clear from the dendrogram, where most of the invertebrate samples taken from the four Berg River sites during winter formed a cluster that was approximately 67% similar to the remaining samples (Figure 4.2). This pattern is illustrated on the ordination plot, where samples taken during winter were coded E to H (Figure 4.2). A further separation of invertebrate samples occurred on a spatial basis during the summer, when the IBT was releasing water into the river – samples taken at BR3 and BR4 below the IBT were generally 60% similar to the remaining samples (Figure 4.2). On the ordination plot, these samples (coded C and D) were clearly separated from summer samples taken above the IBT (coded A and B).

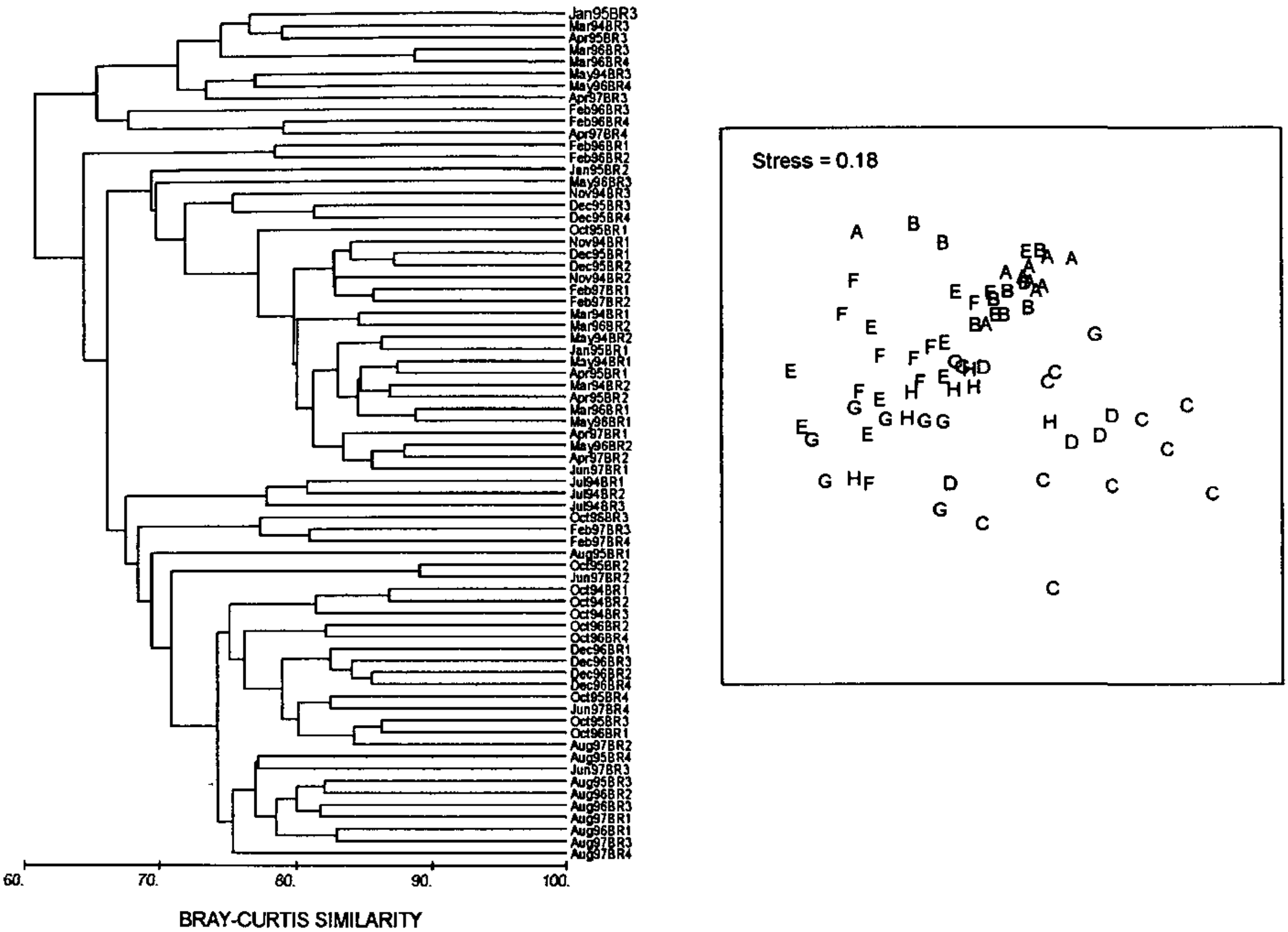


Figure 4.2 A dendrogram and ordination plot displaying all of the invertebrate samples collected from four sites on the Berg River over the study period, March 1994 to August 1997. BR1 (A, summer; E, winter) and BR2 (B, summer; F, winter) are situated above the IBT, and BR3 (C, summer; G, winter) and BR4 (D, summer; H, winter) below it. The stress value of 0.18 indicates a potentially useful 2-dimensional representation of the data.

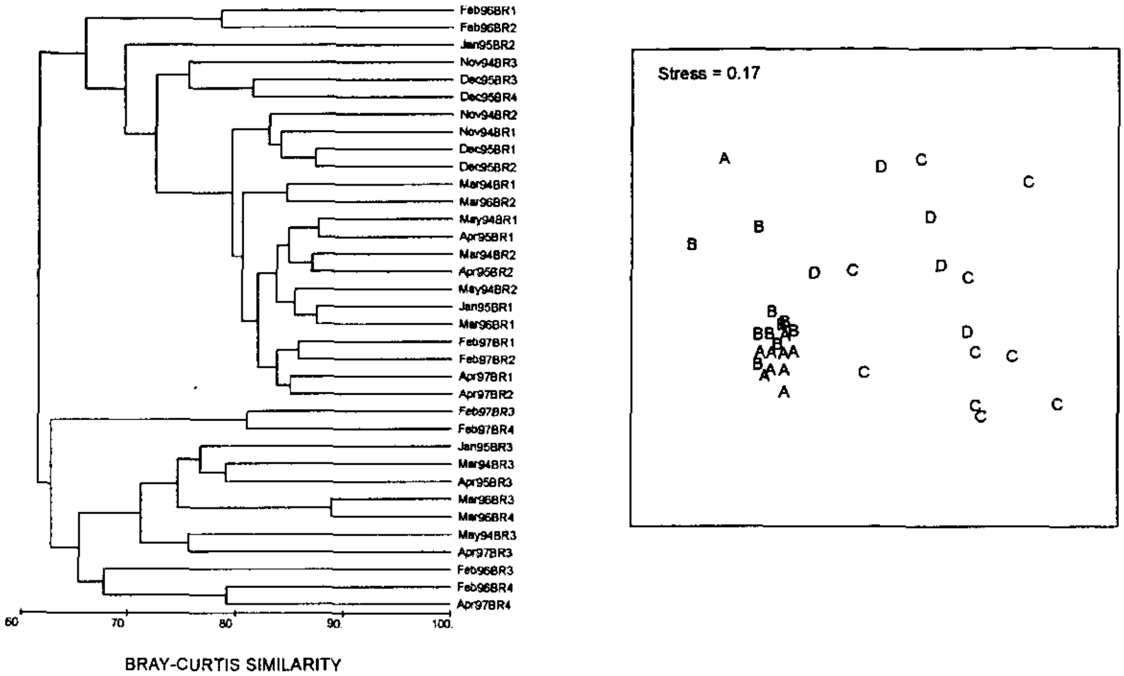


Figure 4.3 A dendrogram and ordination plot displaying the invertebrate samples collected from four sites on the Berg River in summer only, over the study period March 1994 to August 1997. BR1 (A) and BR2 (B) are situated above the IBT, and BR3 (C) and BR4 (D) below it. The stress value of 0.17 indicates a potentially useful 2-dimensional representation of the data.

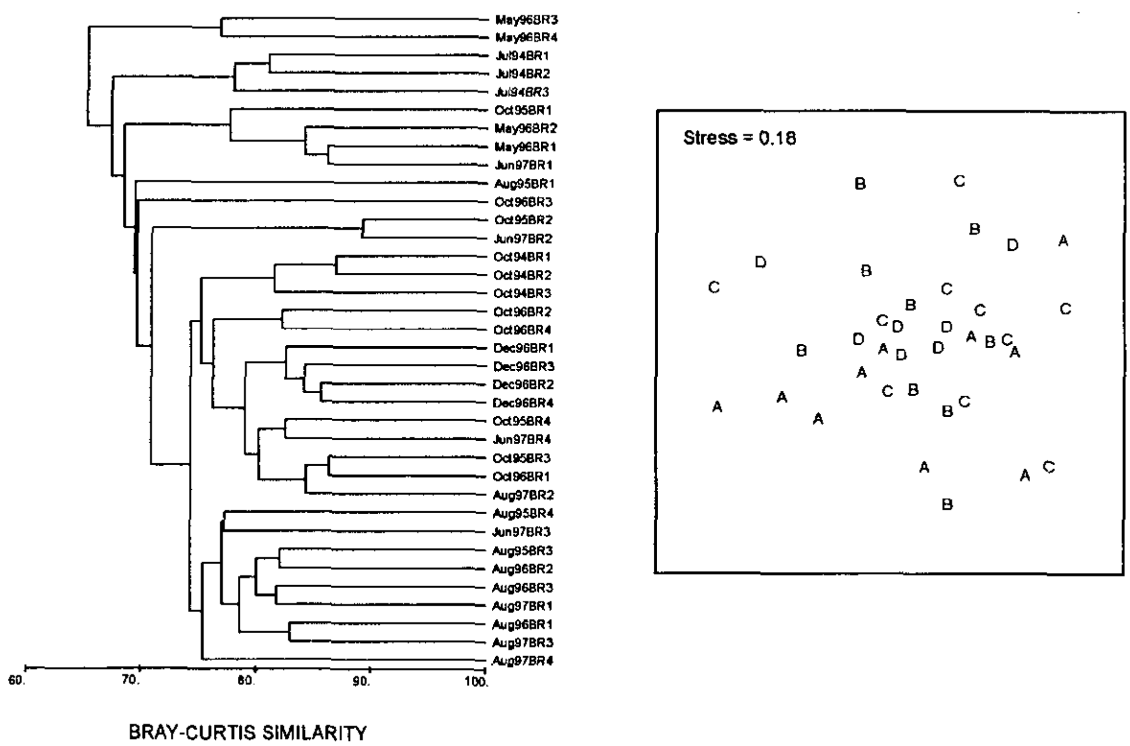


Figure 4.4 A dendrogram and ordination plot displaying the invertebrate samples collected from four sites on the Berg River in winter only, over the study period March 1994 to August 1997. BR1 (A) and BR2 (B) are situated above the IBT, and BR3 (C) and BR4 (D) below it. The stress value of 0.18 indicates a potentially useful 2-dimensional representation of the data.

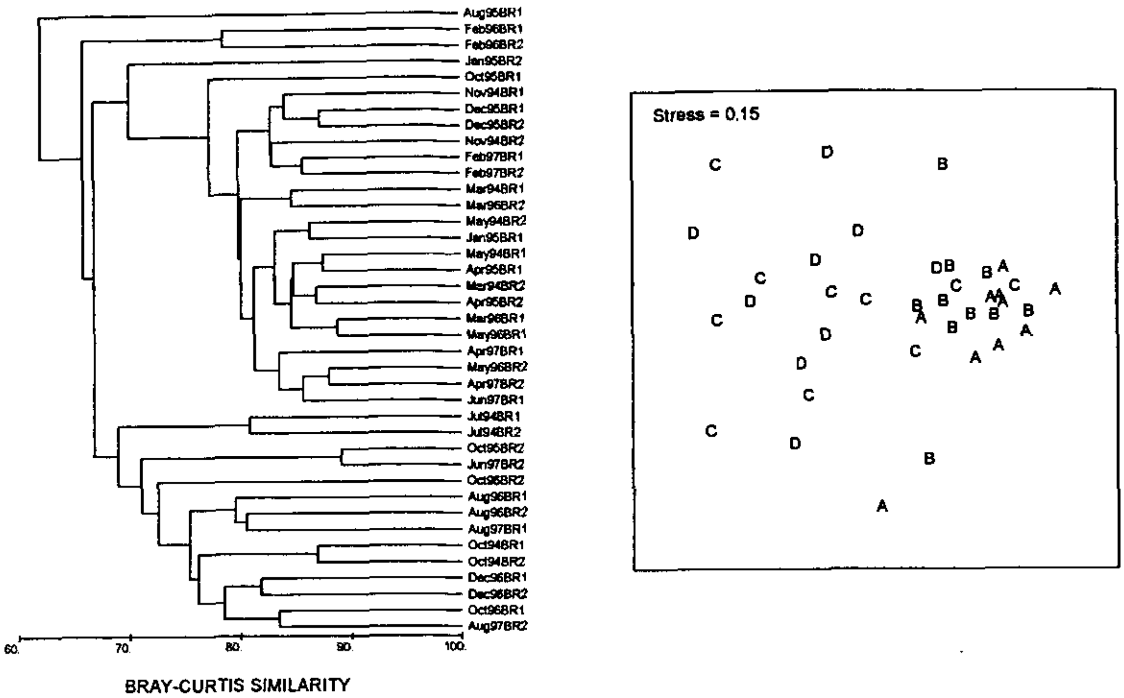


Figure 4.5 A dendrogram and ordination plot displaying the invertebrate samples collected from the two sites situated above the IBT on the Berg River, over the study period March 1994 to August 1997. BR1 is coded A in summer and C in winter; and BR2 is coded B in summer, and D in winter. The stress value of 0.15 indicates a potentially useful 2-dimensional representation of the data.

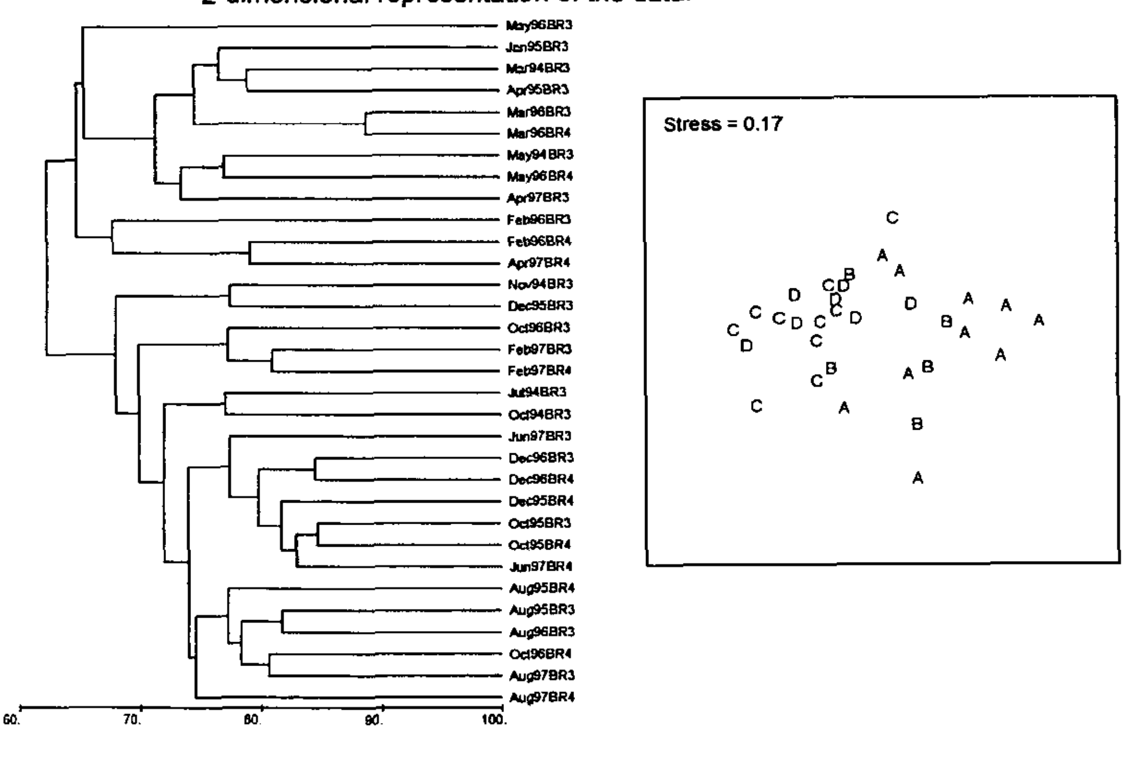


Figure 4.6 A dendrogram and ordination plot displaying the invertebrate samples collected from the two sites situated below the IBT on the Berg River, over the study period March 1994 to August 1997. BR3 is coded A in summer and C in winter; and BR4 is coded B in summer, and D in winter. The stress value of 0.17 indicates a potentially useful 2-dimensional representation of the data.

In order to elucidate these patterns in greater detail, the summer invertebrate samples were analysed as a subset (Figure 4.3). The results showed the same patterns described above – with few exceptions, invertebrate samples collected above the IBT separated from those collected below the transfer at the 60% similarity level. The exceptions were one sample taken in November 1994 at BR3, and two samples taken in December 1995 at BR3 and BR4. A further interesting point to note from Figure 4.3 is the dense clustering of above-IBT samples compared to the more loose array of below-IBT samples.

In winter, there were no clear groupings of invertebrate samples, although samples taken during the same months over the three-year sampling period appeared to form clusters (Figure 4.4). There was no spatial separation of invertebrate samples and thus, in winter, the invertebrate communities below the IBT resembled the unimpacted communities occurring above it.

Further subsets of invertebrate samples were investigated, including all samples taken above the IBT over the three years (Figure 4.5) and those collected below the IBT over the same period (Figure 4.6), in order to investigate seasonal differences. Figure 4.5 shows a repeat of the seasonal separation of samples illustrated in Figure 4.2; thus, with few exceptions, winter samples were approximately 67% similar to summer samples. Summer samples were clustered more closely than winter samples. Figure 4.6 shows a similar seasonal separation of samples, where winter samples were approximately 63% similar to summer samples. All samples were similarly placed on the ordination plot, with no dense clusters. The exceptions to the separation of samples below the IBT were two May 1996 samples, which were collected when the IBT was not in operation but which clustered with the summer samples (Figure 4.6). Similarly, a November 1993 sample, a December 1995 sample and two February 1997 samples, all taken while the IBT was releasing water, clustered with the winter samples.

The smallest subset of invertebrate samples is illustrated in Figure 4.7. The summer samples taken from below the IBT (sites BR3 and BR4) were treated separately in order to investigate whether there were significant differences in community composition with distance from the IBT. No such differences are visible from the results of the multivariate analyses, and this supports the findings of the univariate analyses. Thus, there was no spatial recovery in community composition recorded at site BR4.

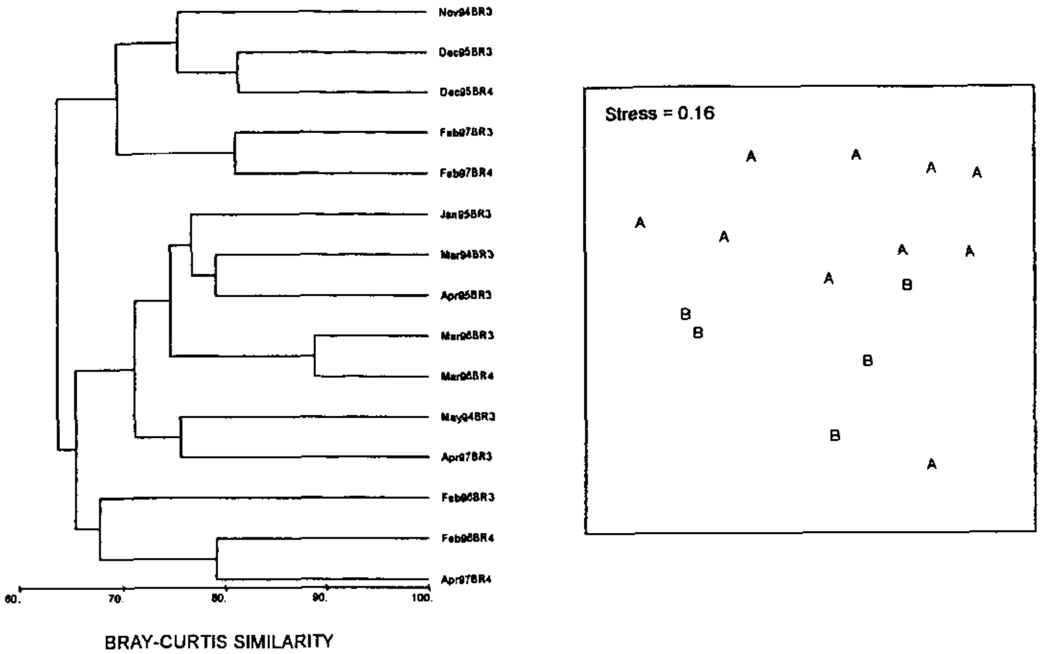


Figure 4.7 A dendrogram and ordination plot displaying the invertebrate samples collected from the two sites situated below the IBT on the Berg River, in summer only, over the study period March 1994 to August 1997. BR3 is coded A; and BR4 is coded B. The stress value of 0.16 indicates a potentially useful 2-dimensional representation of the data.

Determining Discriminating Taxa The taxa that contributed to the dissimilarity between the invertebrate communities below and above the IBT outlet are listed in Table 4.5. Most of the taxa listed here are from the orders Trichoptera and Ephemeroptera, and in general, the dissimilarity between communities could be attributed to a significant decrease in the abundance of contributing taxa below the IBT (Table 4.5). For example, the ephemeropteran families Heptageniidae, Leptophlebiidae, Ephemerellidae and Baetidae all occurred in greater numbers above the IBT, as did the trichopteran families Leptoceridae, Ecnomidae and Philopotamidae. Furthermore, the sensitive Notonemouridae (Plecoptera), the Helodidae, Elmidae adults and Hydraenidae adults (Coleoptera), the Corydalidae (Megaloptera), the Acarina, the Collembola and the Athericidae (Diptera) were all more numerous above the IBT in summer than below it (Table 4.5).

However, a number of the contributing taxa were found in greater numbers below the IBT – these included the Hydropsychidae, Hydroptilidae and Polycentropodidae (Trichoptera) and trichopteran pupae (these were generally of the family Hydropsychidae), the Lumbriculidae (Oligochaeta), the Libellulidae (Odonata), Ostracoda, and the Caenidae (Ephemeroptera).

Table 4.5 A list of taxa that contributed to the dissimilarity between invertebrate communities sampled above and below the IBT in summer months. A 75% cumulative contribution to dissimilarity was used arbitrarily as the cutoff point.

Taxon	Average abundance per m^2 above the IBT	Average abundance per m^2 below the IBT	% contribution to dissimilarity	Cumulative % contribution to dissimilarity
Heptageniidae (EPHEMEROPTERA)	300.42	2.33	5.66	5.66
Leptophlebiidae (EPHEMEROPTERA)	636.50	33.89	5.02	10.68
Leptoceridae (TRICHOPTERA)	200.83	8.67	4.65	15.32
Ecnomidae (TRICHOPTERA)	90.92	0.89	4.21	19.53
Notonemouridae (PLECOPTERA)	163.00	43.78	4.05	23.58
Hydropsychidae (TRICHOPTERA)	174.30	6325.00	3.90	27.48
Helodidae (COLEOPTERA)	103.92	6.78	3.49	30.97
ACARINA	384.75	80.00	3.33	34.30
Elmidae adults (COLEOPTERA)	153.60	32.00	3.32	37.62
Lumbriculidae (OLIGOCHAETA)	197.72	327.44	3.22	40.84
Philopotamidae (TRICHOPTERA)	89.17	0.22	3.13	43.96
Hydroptilidae (TRICHOPTERA)	41.33	47.44	2.91	46.88
Ephemerellidae (EPHEMEROPTERA)	217.75	69.44	2.90	49.77
Athericidae (DIPTERA)	89.50	12.78	2.85	52.62
Hydraenidae adults (COLEOPTERA)	295.17	43.67	2.83	55.45
Libellulidae (ODONATA)	0.67	28.22	2.70	58.15
Corydalidae (MEGALOPTERA)	15.42	2.00	2.65	60.80
COLLEMBOLA	29.83	27.11	2.61	63.41
Baetidae (EPHEMEROPTERA)	16317.84	4707.89	2.52	65.93
Trichoptera pupae (TRICHOPTERA)	0.00	63.78	2.49	68.42
OSTRACODA	5.42	47.11	2.40	70.82
Caenidae (EPHEMEROPTERA)	5.92	53.33	2.34	73.16
Polycentropodidae (TRICHOPTERA)	9.00	13.67	2.17	75.33

4.3 DISCUSSION

A comparison of invertebrate and water quality data collected from the Berg River 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 of the river, which is where the IBT tunnel outlet is located (Figure 2.3). 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 (*Prionium serratum*). It was also suggested that the water transferred from Theewaterskloof could have had a negative impact on the faunal communities of the Berg (Dallas, 1992).

The invertebrate data collected during the study period reported on here clearly support this suggestion, and show some significant alterations in community structure as a result of the release of water from the IBT. During summer, when the IBT was in operation, there was an overall decrease in the taxon richness of the invertebrate communities inhabiting the river reaches below the IBT, compared with the communities occurring in the unimpacted reaches further upstream (Table 4.3; Figure 4.1). Invertebrate diversity and the total abundance of individuals at the impacted sites appeared not to be affected by the IBT. Instead, the major differences between the communities sampled above and below the IBT during summer were expressed in terms of changes in community composition (Table 4.1). This ecological response to the IBT was clearly illustrated by the dendrograms and MDS ordination plots generated by the multivariate analyses of the data (Figures 4.2 to 4.7). These show seasonal differences between winter (no water release) and summer (water release) invertebrate communities and, more importantly, clear spatial differences between invertebrate communities sampled above and below the IBT during summer. 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 below the IBT in summer occurred in most of the orders recorded in the river (Table 4.1), but especially in the Trichoptera and Ephemeroptera which collectively accounted for approximately 42% of the dissimilarity between the invertebrate communities above and below the IBT (Table 4.5). 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 (Tables 4.1 and 4.4). Amongst these were the Heptageniidae, Ephemerellidae and Leptophlebiidae, all of which are generally found in undisturbed mountain streams. Furthermore, the sites upstream of the IBT supported a wide diversity of trichopterans, including some rare taxa (Sericostomatidae, Barbarochthonidae, Petrothrincidae, Glossosomatidae and Lepidostomatidae), but very few of these taxa were found below the IBT. The Philopotamidae, 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 are sensitive to changes in water chemistry (Scott, 1985; de Moor, 1992); fewer individuals of these taxa were found below the IBT compared to above it. 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 sites below the IBT. The low occurrence of most of the hemipterans at the sites 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 (Tables 4.1 and 4.4). These taxa were able to withstand the unseasonal increases in discharge and the associated changes in water chemistry below the IBT in summer (see Chapter 3). 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 (Tables 4.1 and 4.4), often occurring in numbers over 1000 individuals m^{-2}, while the Polycentropodidae were also recorded in higher numbers at these sites. The Hydropsychidae are the most important trichopteran family in southern Africa, in terms of numbers of individuals in various habitats, especially in fast-flowing water with a 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). An interesting increase in the abundance of the trichopteran family Hydroptilidae was noted below the IBT. These generally are regarded as a family that is sensitive to increases in flow, however, genera known to occur in the rivers of the Western Cape prey on the larvae and eggs of the simuliids (Scott, 1985). Thus, the availability of food might explain the increase in the abundance of the hydroptilids. Lastly, the Libellulidae are a robust family of predatory odonates that prefer fast-flowing water, and muddy or sandy substrata (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 (Table 4.1).

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 4.1). 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.

According to the SASS4 total scores and ASPTs, the invertebrate communities sampled below the IBT in summer months, and especially those months towards the end of the water release period, were affected by at least some deterioration in water quality (Table 4.2). The interpretation of SASS4 data is discussed further in Section 5.4.4.

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 plant 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 that 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 provided a ready food source for those animals which could take advantage of the high-flow conditions below the IBT (numbers exceeded 10 000 individuals per m^2 in some months; see Table 4.1). 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).

An interesting interaction was noticed between these two main groups of filter-feeders that inhabited the recipient reaches of the Berg River in summer. In 1994/5, a peak in the Simuliidae below the IBT in November 1994 was followed by a peak in the Hydropsychidae numbers in May 1995 at the same sites. Similarly in 1996, a peak in the Simuliidae in February 1996, was followed by a peak in the Hydropsychidae in March 1996 (Figure 4.8). Only the Simuliidae were observed in substantial numbers at both impacted sites in 1997. These two groups have been observed to interact fairly strongly (Hemphill, 1988, 1991; Morin, 1991; Hildrew & Giller, 1994), and hydropsychids are known to take a longer time to establish than other filter-feeder groups such as the simuliids (e.g. Schlosser, 1992). 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 the summer months of 1994/5 and 1995/1996, were almost covered by the silk and sand cases of the hydropsychid larvae and pupae. Simuliids are known to establish on clean rock surfaces (e.g. Boon, 1988; Nyman, 1995), and competition for space might explain the fact that peaks in the numbers of simuliids and hydropsychids did not coincide, and that simuliid numbers were kept in check throughout the study period. 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, 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.

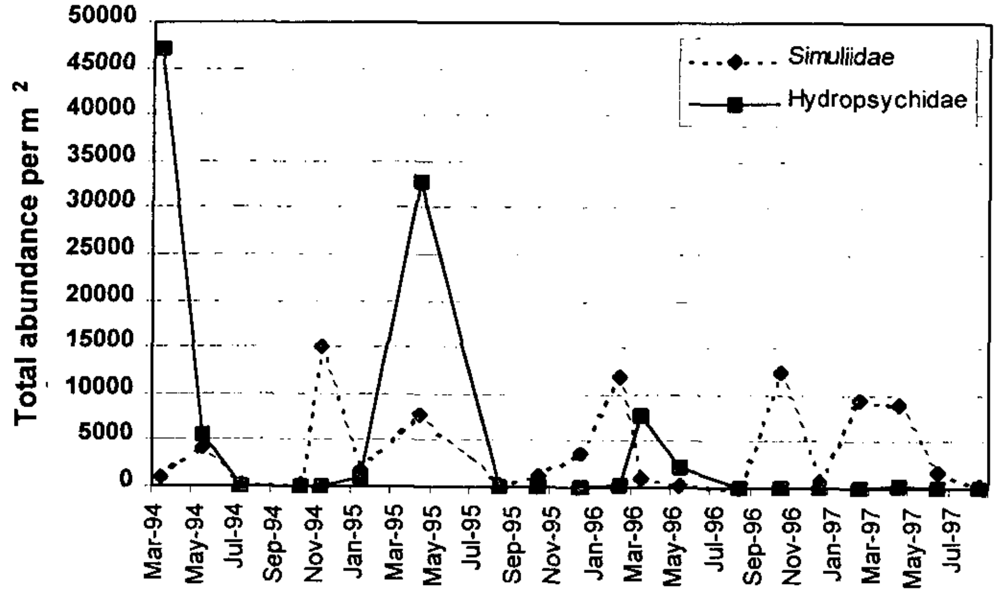


Figure 4.8 A comparison between the total abundance of the Simuliidae and of the Hydropsychidae, below the IBT over the study period.

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 and BR4 below the IBT, the only species observed was *C. thomasseti*, often in very large numbers (Table 4.1). 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 corresponding 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.

An additional difference between above-IBT and below-IBT invertebrate communities was noted on the ordination plot which illustrated the similarities between summer samples (Figure 4.3). Samples taken above the transfer were densely clustered, and were mostly over 80% similar to each other, while the samples collected from below the transfer were more scattered on the ordination plot. This could be attributed to the fact that the unseasonal water releases from the IBT caused short-term fluctuations in community composition, while the unimpacted invertebrate communities further upstream were more consistent in composition.

There appeared to be no significant differences between the two impacted sites, BR3 and BR4. Thus, there was no evidence of recovery with distance from the IBT release tunnel. In winter, however, when the IBT ceased to operate, the invertebrate communities below the IBT resembled the unimpacted communities upstream and thus, these sites were allowed to recover from the impacted condition. In some instances, the invertebrate communities sampled below the IBT immediately after the IBT commenced releasing water (November 1994 and December 1995) were more similar to winter communities, and conversely, communities sampled soon after the IBT ceased to operate (May 1996) were more similar to summer communities. Thus, temporal recovery occurred gradually throughout the winter. This recovery was limited to the winter season, when a naturally more depauperate community established at the sites below the IBT. The more diverse summer invertebrate communities could not establish at the impacted sites and thus, the "recovery" is artificial. The downstream effects of this discontinuity in the establishment of the summer community requires investigation.

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), who found that the invertebrate communities below the donor impoundment of the transfer recovered fairly rapidly in summer when discharges increased naturally, and after spring spates had facilitated the recolonisation of river reaches below the dam.

It can be concluded, therefore, that the RBEGS IBT effectively represents a discontinuity in the river, similar to an impoundment (e.g. Ward & Stanford, 1983a,b; Byren & Davies, 1989), which disrupts both the spatial and temporal continuity of the river system.

Transfer of organisms 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 has 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 (Cambray and Jubb, 1977; Laurenson and Hocutt, 1984, 1986). 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 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.

5.1 GENERAL DISCUSSION OF RESULTS AND CONCLUSIONS

The release of water from the RBEGS IBT into the Berg River had a marked effect on the physical, chemical and biotic attributes of the river reaches immediately below the transfer outlet. During summer, statistically significant and, at times, substantial increases in discharge and, consequently, stream depth were accompanied by significant alterations in water quality. It is likely that these recorded changes in water chemistry are associated with the impoundment of the donor river at Theewaterskloof.

The benthic invertebrate communities recorded below the IBT during summer months were markedly different from those occurring further upstream. The overall richness and number of taxa recorded here were lower than those recorded above the IBT. Diversity was not significantly different at the impacted sites in summer. An MDS ordination plot of all the benthic invertebrate data showed distinct and different summer, below-IBT communities in the Berg River. During the winter months, when the IBT was not in operation, however, this below-IBT community "recovered" to resemble the invertebrate communities above the IBT sampled during the same season. When IBT releases recommenced in summer, the downstream benthic invertebrate community shifted away from its "recovered" state once again.

The dissimilarities between the impacted (below the IBT) and unimpacted (above the IBT) benthic invertebrate communities were attributed to shifts in community composition. The percentage contributions to these dissimilarities were highest for the Trichoptera and Ephemeroptera. Several taxa that are rare or sensitive to flow and water quality conditions and that were recorded above the IBT, were absent or reduced below it during summer months. These included the Notonemouridae (Plecoptera), Heptageniidae, Ephemerellidae and Leptophlebiidae (Ephemeroptera), Philopotamidae, Leptoceridae and Ecnomidae (Trichoptera), Corydalidae (Megaloptera), Elmidae adults, Hydraenidae adults, and Helodidae larvae (Coleoptera) and the Athericidae (Diptera).

In adddition, large quantities of zooplanktonic groups (the Cladocera and Copepoda) were transferred through the IBT tunnel, and released, live or fragmented, into the Berg. In response, collector-predator Trichoptera larvae, of the family Hydropsychidae, and the filter-feeding dipteran Simuliidae proliferated below the IBT during summer months. In summary, the summer invertebrate community below the IBT resembled one typical of lake or reservoir outlets. The release of water from the Theewaterskloof impoundment into the Berg River has led to changes in water quality that are similar to those recorded downstream of lakes or impoundments, with the introduction of dissolved and suspended organic loads, and fast, turbulent flow, all of which are characteristic of lake or dam outlets.

The IBT outlet into the Berg River clearly represents a discontinuity in the river continuum, with some recovery during winter. A number of the Serial Discontinuity Concept predictions for the downstream effects of impoundment of the upper reaches of a river hold true for IBTs, such as increased nutrient levels, increased input of fine particulate organic matter, introduction of plankton, increased numbers of filter-feeding guilds and, thus, altered trophic relationships (Figure 5.1) (Ward & Stanford, 1983).

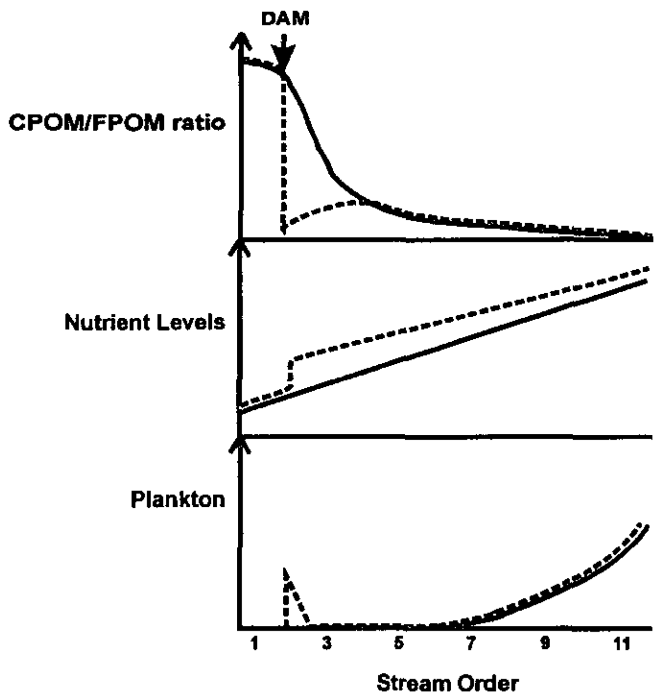


Figure 5.1 Some of the predictions of the Serial Discontinuity Concept of Ward & Stanford (1983). The graphs show theoretical data for an unimpounded (solid line) and for an impounded (dashed line) river.

5.2 MATRIX OF EFFECTS

The ecological effects of IBTs on rivers are varied and numerous (e.g. Snaddon *et al.*, 2000), while the rivers themselves are naturally dynamic and complex systems. These complexities present the water manager with the challenge of deciding how to manage and conserve the natural systems, how to prioritise and consequently minimise the negative effects of river manipulation, and, importantly, whether an IBT scheme should be constructed at all. A technique used by several authors to synthesise available information on ecosystem responses to anthropogenic impacts, is the development of a matrix or table. The "Leopold Matrix" is one such method, where impacts are listed in rows, and ecosystem responses in columns. In this way, the major effects of an impact, such as an

IBT, and the array of significant and known ecosystem responses can be viewed at a glance, and management priorities can be determined (e.g. Begg, 1986; Allanson, 1995).

Leopold matrices were developed for both the recipient and donor systems (Tables 5.1 and 5.2). The IBT effects on and ecosystem responses of aquatic systems along the transfer route were assumed to similar to those of the recipient (Table 5.1). The matrices contain information gleaned from the literature on IBTs - references are included which refer to observed ("O") IBT effects and the reader is referred to Snaddon *et al.* (2000) for details - and also include the potential effects of IBTs based on the literature and on the authors' experience as freshwater ecologists. The effects considered here are at the level of the catchment, and do not address large-scale effects, such as climate change, or the effects of impoundment, which have been well-studied. In compiling these matrices, the assumption was made that the transfer of water occurs in one direction only, and thus, where this is not the case, both matrices will apply to all affected systems.

The ecological significance of the ecosystem responses listed in Tables 5.1 and 5.2 are briefly summarised in Table 5.3, in order to provide further details on the ecological consequences of IBTs, and to highlight the importance of prevention or mitigation of the effects of water transfer. Significantly, almost all of the ecosystem responses listed in Table 5.3 lead to the loss of biodiversity. This has evolutionary significance, and also has legal implications relating to the fact that South Africa is a signatory to the Biodiversity Convention.

References for Tables 5.1 and 5.2:

1. Snaddon & Davies, this volume.
2. Golladay & Hax, 1995
3. Schorr *et al.*, 1995
4. O'Keeffe & de Moor, 1988
5. Matthews *et al.*, 1996
6. Brown *et al.*, 1984
7. W. Croucamp, in litt.
8. Cross & Moss, 1987
9. Stevens *et al.*, 1985
10. Campbell & Moyle, 1990
11. Day, 1985
12. Day *et al.*, 1982
13. O'Keeffe & Ashton, 1991
14. Snaddon *et al.*, 2000 (Section 3.2.2)
15. Chessman *et al.*, 1992
16. Cugley, 1988
17. Guiver, 1976
18. Cambray & Jubb, 1977
19. Laurenson & Hocutt, 1984
20. National Rivers Authority, 1994
21. García De Jalón, 1987
22. Kubomoto *et al.*, 1974
23. Holmes & Whitton, 1977
24. Brittain *et al.*, 1984
25. Borgstrøm & Løkensgard, 1984
26. Roos-Collins, 1993
27. Rozengurt *et al.*, 1985

Table 5.1 A matrix of the ecological effects of IBTs and the associated ecosystem (riverine) responses in the **recipient system and transfer route**.

	Ecosystem responses									
IBT effects on recipient and transfer route	Decreased taxon richness	Altered taxon dominance	Altered trophic dynamics	Altered community composition	Mixing of previously isolated gene pools	Local extinctions	Altered channel morphology	Altered habitat/ biotopes	Altered riparian zone	Algal blooms
Unseasonal increase in discharge	**o** [1,2,3]	**o** [1,4]	**o** [1,4]	**o** [1,4,5,6]		**p**	**o** [7]	**o** [6,7]	**p**	
Decreased MAR below offtake										
Increased flooding	**p**	**p**	**p**	**p**		**p**	**p**	**p**	**p**	
Decreased lateral connectivity	**p**	**p**	**p**	**p**		**p**	**p**	**p**	**p**	
Decreased flow variability	**p**	**o** [4]	**o** [4]	**o** [4,8]		**p**	**p**	**o** [4]	**p**	
Increased flow variability	**o** [2,3]	**p**	**p**	**o** [2,3,5,9,10]		**p**	**p**	**p**	**p**	
Increased bank and bed erosion	**p**	**p**	**p**	**o** [11]		**p**	**o** [11,12,13,14,15]	**p**	**p**	
Increased light *penetration*	**p**	**p**	**o** [8]	**o** [8]		**p**				**p**
Increased turbidity	**p**	**p**	**p**	**p**		**p**	**p**	**p**	**p**	
Increased organic suspensoids	**o** [1]	**o** [1]	**o** [1]	**o** [1]		**p**			**p**	**p**
Decreased TDS/ conductivities	**p**	**o** [4]	**o** [4]	**o** [4]		**p**			**p**	
Increased TDS/ conductivities	**p**	**p**	**p**	**o** [1]		**p**			**p**	
Decreased pH	**p**	**p**	**p**	**p**		**p**			**p**	
Increased pH	**o** [1]	**o** [1]	**o** [1]	**o** [1]		**p**			**p**	
Enrichment (nutrients)	**p**	**p**	**p**	**p**		**p**			**p**	**o** [16]
Dilution of chemical constituents	**p**	**p**	**p**	**p**		**p**			**p**	
Transfer of new species or populations	**p**	**o** [17]	**p**	**o** [18,19,20,21,22,23]	**o** [18,19,20,21,22,23]	**p**				**o** [17]
Introduction of parasites/ disease vectors	**p**	**p**	**p**	**p**		**p**			**p**	
Transfer of exotic species	**p**	**p**	**p**	**p**		**p**			**p**	
Transfer of secondary metabolites						**p**				**p**
Increased freshwater inflow to estuaries	**p**	**p**	**p**	**p**		**p**	**p**	**p**	**p**	**p**
Reduced freshwater inflow to estuaries										

p: potential response; **o**: observed response.

Table 5.2 A matrix of the ecological effects of IBTs and the associated ecosystem (riverine) responses in the **donor** system.

IBT effects on donor	**Ecosystem responses**									
	Decreased taxon richness	Altered taxon dominance	Altered trophic dynamics	Altered community composition	Mixing of previously isolated gene pools	Local extinctions	Altered channel morphology	Altered habitat/ biotopes	Altered riparian zone	Algal blooms
Unseasonal increase in discharge										
Decreased MAR below offtake	**o** [24]	**p**	**p**	**o** [11,24]		**p**	**p**	**o** [24,25,26]	**p**	**p**
Increased flooding	**p**	**p**	**p**	**p**		**p**	**p**	**p**	**p**	
Decreased lateral connectivity	**p**	**p**	**p**	**p**		**p**	**p**	**p**	**p**	
Decreased flow variability	**p**	**p**	**p**	**p**		**p**	**p**	**p**	**p**	
Increased flow variability	**p**	**p**	**p**	**p**		**p**	**p**	**p**	**p**	
Increased bank and bed erosion	**p**	**p**	**p**	**p**		**p**	**p**	**p**	**p**	
Increased light penetration	**p**	**p**	**p**	**p**		**p**				**p**
Increased turbidity	**p**	**p**	**p**	**p**		**p**	**p**	**p**	**p**	
Increased organic suspensoids	**p**	**p**	**p**	**p**		**p**			**p**	**p**
Decreased TDS/ conductivities	**p**	**p**	**p**	**p**		**p**			**p**	
Increased TDS/ conductivities	**p**	**p**	**p**	**p**		**p**			**p**	
Decreased pH	**p**	**p**	**p**	**p**		**p**			**p**	
Increased pH	**p**	**p**	**p**	**p**		**p**			**p**	
Enrichment (nutrients)	**p**	**p**	**p**	**p**		**p**			**p**	**p**
Dilution of chemical constituents										
Transfer of new species or populations	**p**	**p**	**p**	**o** [23]	**o** [23]	**p**				**p**
Introduction of parasites/ disease vectors	**p**	**p**	**p**	**p**		**p**			**p**	
Transfer of exotic species	**p**	**p**	**p**	**p**		**p**			**p**	
Transfer of secondary metabolites						**p**				**p**
Increased freshwater inflow to estuaries										
Reduced freshwater inflow to estuaries	**o** [27]	**p**	**p**	**o** [27]		**p**	**p**	**p**		

p: potential response; **o**: observed response.

Table 5.3 Some of the ecological implications of the ecosystem responses listed in Tables 5.1 and 5.2. This list is not comprehensive, but it gives an indication of the probable consequences of IBTs. The reader is referred to Snaddon *et al.* (2000) for more information.

Ecosystem response	Ecological Significance
Decreased taxon richness	• loss of biodiversity • altered biotic interactions, such as competition and predation • loss of rare and sensitive species
Altered taxon dominance	• loss of biodiversity • pest or weedy species dominant
Altered trophic dynamics	• loss of biodiversity • altered ecological processes, such as decomposition, production and nutrient cycling
Altered community composition	• loss of biodiversity • altered biotic interactions • reduced abundance
Mixing of previously isolated gene pools	• loss of biodiversity through hybridisation • altered biotic interactions • local extinction or increased mortality • implications for long-term evolution and basin integrity • the isolation of gene pools may also occur as a result of the construction and operation of IBTs
Local extinctions	• loss of biodiversity • altered biotic interactions
Altered channel morphology	• loss of habitats suitable for establishment of certain species or communities • altered deposition or scouring processes • loss of riparian connectivity
Altered habitat/ biotopes	• local extinction and/or increased mortality • loss of biodiversity • shifts in trophic dynamics
Altered riparian zone	• altered community composition of riparian zone • loss of important food source • loss of lateral connectivity • decreased/increased shading • altered timing of food inputs • changes in CPOM:FPOM ratios • altered P:R ratios
Algal blooms	• loss of biodiversity • deoxygenation of water column and sediments • toxic/non-toxic exudates • altered water quality

5.3 ACHIEVEMENT OF OBJECTIVES

The two objectives that were addressed in this report were as follows (see Chapter 1):

1. Collect field data from a local IBT scheme (the Riviersonderend-Berg-Eerste River Government Scheme), in order to investigate aspects of water quality and the biological status of the donor and recipient rivers.

2. Provide guidelines and protocols for the better operation of extant schemes, for the design of future schemes, and for the operation of future schemes in South Africa using knowledge gleaned both locally and internationally.

It was felt that these objectives were achieved, while contributing to our overall understanding of the ecological effects of the regulation of river systems through water transfers. It is hoped that the matrices developed in Section 5.2 above will provide a useful summary of IBT effects and the associated ecosystem responses, that can be used by ecologists and water managers.

5.4 RECOMMENDATIONS FOR IBT PLANNING AND MANAGEMENT

5.4.1 *Introduction*

In the vast majority of cases, the 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 have been 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 (Snaddon *et al.*, 2000). Public, and often political opposition to IBTs, and other large-scale river regulation, is on the increase, and the need for more sensitive planning is rapidly becoming apparent worldwide (Loh & Gomez, 1996; Davies & Day, 1998).

In terms of the social and economic effects of IBTs, the donor catchment is often disadvantaged, with most, if not all, benefits going to the recipient. However, as far as environmental effects are concerned, the effects of water transfer on the recipient, the donor and often on the transfer route, can be equally severe, but are seldom given equal consideration, if any at all (Snaddon *et al.*, 1998). In order to address the concerns associated with IBTs, many of which have been raised in the previous chapters, the planning phases of a project must take all of the components into account and give them equal weighting.

5.4.2 *Problems associated with IBT Planning*

Many of the problems associated with the implementation of water transfer projects relate to the planning phases of these schemes (e.g. Thomas and Box, 1969; Greer, 1983; Cox *et al.*, 1985; Shiklomanov, 1985; Davies *et al.*, 1992), such as:

- planning priorities do not take the ecological significance of IBTs into account as these are often not understood;
- a lack of comprehensive environmental assessments;

- the subordination of environmental assessments to the technical and economic aspects of IBTs;
- a lack of co-ordination between environmental assessments (where they do 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 effectively are ignored.

The first two of these problems were evident during the planning of the Texas Water Plan in the USA, which was eventually abandoned. In this case, primary attention was given to the technical and economic aspects of the scheme, which involves the transfer of water from the Mississippi River, to western Texas and New Mexico. Environmental studies were initiated in response to completion of the engineering and economic feasibility phases, while a third step consisted of a revision of the initial technical 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).

In retrospect, a similar oversight was committed in the Western Cape Province of South Africa, where a regional analysis of available water resources was commissioned in 1984 by the Department of Water Affairs and Forestry. An evaluation of the final analysis, the Western Cape System Analysis (WCSA), uncovered the problem associated with excluding one of the Western Cape's major river basins, the Breede River catchment, from such a system analysis. At least three IBTs have been proposed which would transfer water from the Breede River and its tributaries to the Cape Metropolitan Area, and yet the consequences of these IBTs 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 the impoundment of their rivers until a Breede River Basin Study has been completed. The Basin Study commenced in mid-1999.

A further example can be drawn from southern Africa, where the downstream impacts of an IBT were not adequately considered at the planning stage. Impoundment of the upper Orange River in Lesotho, in order to supply water to South Africa, *via* the Lesotho Highlands Water Project (LHWP), could ultimately reduce the yield of a downstream IBT, the Orange River Project (ORP) (Davies *et al.*, 1992), and reduce the water available in the middle and lower reaches of the Orange River system (Benade, 1993). The Orange River is the most important and the largest single aquatic ecosystem in the Northern Cape Province of South Africa. Half of the catchment lies within the boundaries of the Province, which contributes only 2% to the MAR of the system. The maintenance of the ecological functioning of the river is thus crucial to the Province, and there is considerable concern that upstream schemes such as the LHWP will divert water from the Province (e.g. Benade, 1993).

Globally, several IBT schemes have been abandoned or modified as problems have become apparent, while others have been postponed as a result of the lack of adequate environmental assessments during the feasibility planning stages (e.g. Guiver, 1976; Greer, 1983; Day, 1985; Shiklomanov, 1985). These include (see Snaddon *et al.* (2000) for more details on these transfer schemes):

- A water transfer from Lake Suldalsvatnet to Hylsfjorden in Norway ceases during June and July during salmon runs in order to protect these economically important species (Nordeng, 1977; Tøndevold, 1984).
- The McGregor Diversion, Canada, was shelved in 1978 as a result of the possible transfer of both fish and their parasites from the Fraser River to the Arctic-draining Peace River (Arai and 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 salmon migration routes (Day, 1985).
- The Ely Ouse to Essex Scheme, in the U.K., 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 Irtysh, to Soviet Central Asia (SCAP), was "indefinitely postponed" in 1986, as a result of pressure from environmentalists, and economists who were concerned about the feasibility of such a project (Voropaev and Velikanov, 1985).
- The North American Water and Power Alliance (NAWAPA) scheme was shelved due to the numerous public and political objections to the economic and environmental effects of the project (Micklin, 1977; Shiklomanov, 1985).
- The transfer of water from Lesotho to South Africa *via* the LHWP, mentioned above, was recently halted until the end of October 1999, in order to undertake erosion protection work in the Ash River catchment, the recipient of the LHWP (Willie Croucamp, spokesperson for the Trans-Caledon Tunnel Authority, 1 October 1999). South Africa will have to compensate Lesotho for the loss of income as a result of the interruption in delivery. The potential for substantial erosion below the outlet of the LHWP was highlighted by several authors (e.g. Chutter, 1993).

In the past, there has been a 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. For example, despite a number of calls for action (e.g. Petitjean and Davies, 1988a,b; Davies *et al.*, 1992), it was only in mid-1996 that a wide range of organisations and individuals affected by the LHWP in southern Africa 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 South Africa, the power of authority for the planning of IBTs lies with the National Department of Water Affairs and Forestry. The Water Act (1998) states in Chapter 2, Part 1:

> "6(1): The national water resource strategy must...
> (f) state water management area (WMA) surpluses or deficits;
> (g) provide for inter-catchment water transfers between surplus WMA's and deficit WMA's;
> ..."

The management of catchments, however, will occur through catchment management agencies, that will be representative of the users in the catchment. Thus, although the national water resource strategy will be developed by national government, opportunities for consultation and participation in the actual planning of water transfer projects has been mandated by law (Sections 5 and 8, Water Act 1998).

5.4.3 *Recommendations for IBT Planning and Management*

In this section, a list of recommendations for the planning and management of IBT schemes is presented. These have been compiled from the literature, and are not listed in any order of priority.

- It is clear from the literature that the ecological consequences of IBTs are such that great caution is warranted. Data are scarce, but the "precautionary principle" needs to be applied to water-resources planning, allowing for the gathering of data before the feasibility stage of any IBT project. The collection of timely, objective and detailed information on the actual and potential effects of transfers is necessary, in order that all human communities affected by any scheme may assess the IBT in an informed and reasonable manner. Such data collection should form one of the tasks of the appropriate catchment management agency, or agencies, where more than one water management area is affected.

  The development of catchment management plans or strategies for all catchments in South Africa will provide a framework for the collection of data before further river regulation takes place, and for the monitoring of the effects of current IBTs.

- Extant schemes should be re-assessed in terms of their effects, so that detrimental impacts can be minimised through mitigation (see Section 5.4.4 for more detail).

- The environmental aspects of IBTs should not be viewed 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. It is assumed that this level of detail will be achieved through the process of developing a catchment management plan.

- Greater public participation is required during the planning of IBTs, 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). 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 should usually be provided by the developer (e.g. in Lesotho, the Lesotho Highlands Development Authority; in South Africa, Department of Water Affairs and Forestry).

  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.

- 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.

- 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, disease vectors and pests of economic importance, are likely consequences of IBTs. There are examples in the literature of all of these threats. This requires great caution and extensive investigation during the assessment of the feasibility of such schemes. Once again, the availability of pre-transfer information from donor and recipient catchments would aid in the assessment of the likelihood of the transfer of exotic and indigenous fauna and flora, parasites and disease vectors.

  There are no examples of water transfer schemes that have successfully prevented the transfer of organisms between catchments. In the UK, it was originally thought that the pumps and high water pressures associated with the Ely Ouse to Essex 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 probable that both fry and eggs are transferred, the installation of electric screens will most likely be of minimal efficiency (Guiver, 1976). In Canada, biologists proposed the addition of a fish screen and a sand filtration system to inhibit transfer of biota through the Garrison Diversion Project (Keys, 1984). This was not considered to be efficient, and was not implemented. Research on technical and ecological solutions to the prevention of biotic transfers is urgently required.

- Similarly, the likelihood of the transfer of water quality problems, such as cyanophyte blooms, between catchments, should be assessed. The threat of such transfers could be reduced through flexibility in the operational criteria of an IBT scheme, thus allowing for a cease in transfer during periods of risk.

- A botanical investigation to predict the change in vegetative cover under various release strategies would be useful for IBT planning and management. A more definitive prediction of morphological response relies on an improved understanding of the effects of various durations of inundation on those species of trees, shrubs and grass prevalent in the riparian zone of a river.

The following recommendations for IBT planning and management are based on the results of the fieldwork component of this project.

- A great deal of research and attention has and continues to be focused on the effects of reductions in flow on aquatic ecosystems. The methods used for the determination of instream flow requirements (e.g. the building-block methodology) are based on our knowledge of the effects of flow reduction (e.g. King & Tharme, 1994; King *et al.*, 1995). There is a need to shift this focus to include the effects of increases in flow on aquatic systems. In these cases, a "capping flow recommendation" is required, in order to prevent constant releases from reducing the natural variability of the flow regime, thus creating artificially stable flows throughout the year (O'Keeffe, 1999). The capping flow recommendation should aim to prevent the reversal of seasonal flow variation (such as occurred in the Berg River during summer months), to maintain as much as possible of the vairability of the flow regime; and to prevent constant elevated base flows during low flow seasons.

- The temporal "recovery" by the riverine macroinvertebrates observed in the Berg River, during months when the IBT was not operational, provides evidence of the extent to which an ecosystem is affected by the operational criteria of a transfer scheme. Releases from IBTs should be timed to allow the riverine communities to reset during all seasons, in order to maintain the diversity of populations inhabiting the impacted river reaches. It is not enough to cease operation during one season only, thus allowing an unimpacted riverine community to establish during this season but continuing the perturbation over the remaining months.

- In terms of water quality, the South African Water Quality Guidelines for Aquatic Systems provide adequate guidance on the acceptable changes in water quality of a receiving waterbody. These relate to comparisons with unimpacted reaches of the same river system during the same period. For example, TDS concentrations should not be changed by more than 15% of unimpacted levels, and the amplitude and frequency of natural cycles should not be changed. The Guidelines should be applied to all river systems affected by IBTs.

- The transfer of lentic fauna into the Berg River contributed significantly to the shifts in benthic invertebrate community composition below the outlet. It is preferable, therefore, for water to be received by a standing waterbody rather than released directly into a river. A filtering wetland system, or a weir on the recipient river could solve this problem. The impacts of such solutions, however, require detailed assessment. An ideal solution would be the off-channel storage of transferred water, and delivery to the user. Again, this recommendation would require detailed technical assessment, in order to solve associated problems such as water pressure build-up within the transfer tunnels.

- It is essential to be able to monitor the water chemistry of the donor reservoir. This is especially important where nutrient enrichment can occur in the donor system, which may lead to algal blooms and associated water quality problems in the recipient and along the transfer route.

5.4.4 Resource-directed measures for the protection of water resources

The revision of the South African Water Act has generated a number of initiatives which aim to increase the efficiency with which freshwater resources are managed and thus, protected. One of these is the implementation of a water resources protection policy, which aims to develop resource-directed measures (RDM) for the protection of water resources. The RDM procedure involves the definition of a desired level of protection for a water resource, and on that basis, setting clear numerical or descriptive goals for the resource quality of the resource (the Resource Quality Objectives; RQOs). An RDM procedure has been designed to undertake preliminary determinations of the ecological management class, the Ecological Reserve and resource quality objectives for individual water resources, as specified in sections 14 and 17 of the South African National Water Act (Act 36 of 1998) (Mackay, 1999). This process was begun by the Department of Water Affairs and Forestry in 1997, and will continue into a final phase where the detailed procedures and a full resource classification will be finalised. The appropriate application of this procedure should ultimately provide effective protection for rivers, wetlands and groundwater resources affected by water resource development, such as IBTs, and should guide the future planning and management of such schemes.

The Resource Quality Objectives for a water resource are defined as "...a numerical or descriptive statement of the conditions which should be met in the receiving water resource, in terms of resource quality, in order to ensure that the water resource is protected." (Mackay, 1999). The RQOs address four important aspects of a water resource, including:

- requirements for water quantity, i.e. instream flow requirements (IFR) for a river or estuary, or water level requirements for lentic water or groundwater;
- requirements for water quality, which are based on the guidelines and procedures outlined in the South African Water Quality Guidelines;
- requirements for habitat integrity, which encompass the physical structure of instream and riparian habitats, as well as the botanical aspects;
- requirements for biotic integrity, which encompass community structure, distribution and overall state of the aquatic biota.

In the case of an IBT, it is recommended that RQOs be set for all affected water resources, thus including the donor, transfer and receiving (or recipient) water resource. It is clear from the literature and from this project (e.g. Tables 5.1 and 5.2) that all of the aspects listed above are affected by the transfer of water between catchments.

In order to determine the RQOs for a river reach or complete catchment, reference conditions need to be established for the water resources concerned. Reference conditions (or reference sites) provide the basis for comparison with impacted resource units (or monitoring sites). Once these are determined, the present state of all resource units can be established, according to ecological status and resource quality, water and land uses, and socio-economic conditions. The next step is the determination of the desired management class for resource units, taking into account the importance and sensitivity of the unit and the achievability of the desired management class. Once these are set, the Reserve can be quantified for each resource unit, integrating water quantity and quality requirements, and assessing the river, wetland and groundwater components. Finally, the RQOs are set for each resource unit. An appropriate resource monitoring programme will ensure continual re-assessment of objectives and the strategies for achieving the desired management class, the Reserve and the RQOs.

Preliminary reference conditions have been set for the fynbos bioregion (*sensu* Eekhout *et al.*, 1996), in terms of the macroinvertebrate fauna (Dallas *et al.*, 1998). Unimpacted mountain streams and foothill rivers are thus defined as attaining total SASS4 scores greater than 140 and ASPTs greater than 8. Three preliminary categories of biotic integrity have been defined in relation to these reference conditions (Helen Dallas, Southern Waters Consulting, pers. comm.); these categories are currently being developed and should not be extrapolated from this report.

- *unimpaired* (class A): SASS4 score > 140; OR ASPT > 8;
- *moderately impaired* (class B): SASS4 score > 100, and < 140; OR 6 < ASPT < 8;
- *considerably impaired* (class C): SASS4 score < 100; OR ASPT < 6.

Table 5.4 *A hypothetical determination of present state biotic integrity classes for the four sites sampled on the upper Berg River. The description of each class is provided in the text above. BR1 and BR2 were situated above the IBT, and BR3 and BR4 below it.*

	Class A		Class B		Class C

Status of water releases	Date	BR1 Total score	BR1 ASPT	BR2 Total score	BR2 ASPT	BR3 Total score	BR3 ASPT	BR4 Total score	BR4 ASPT
ON	Mar-94	192	7.7	177	8.0	93	6.6		
ON	May-94	172	8.2	173	7.9	82	6.8		
OFF	Jul-94	109	8.4	125	7.8	110	7.9		
OFF	Oct-94	117	7.8	114	8.1	100	7.7		
ON	Nov-94	173	8.2	151	7.9	100	7.1		
ON	Jan-95	136	8.0	86	8.6	95	6.3		
ON	Apr-95	171	7.8	164	8.2	68	6.2		
OFF	Aug-95	95	8.6			96	8.0	135	7.9
OFF	Oct-95	158	7.9	85	8.5	148	8.2	129	8.6
ON	Dec-95	153	7.7	135	8.4	111	7.9	150	7.9
ON	Feb-96	113	8.1	131	7.7	93	7.2	78	7.8
ON	Mar-96	147	8.2	132	8.8	90	6.9	83	6.4
OFF	May-96	172	8.2	144	9.0	127	8.5	115	7.7
OFF	Aug-96	106	7.6	115	7.7	88	8.0		
OFF	Oct-96	128	7.5	75	8.3	115	8.8	100	8.3
OFF	Dec-96	108	8.3	93	7.8	112	8.0	96	8.0
ON	Feb-97	155	8.2	131	7.7	72	8.0	50	7.1
ON	Apr-97	156	8.2	115	8.2	102	7.3	104	6.9
OFF	Jun-97	149	8.3	98	8.2	118	8.4	126	7.4
OFF	Aug-97	80	8.0	105	7.5	89	8.9	66	9.4

Although the SASS4 data presented here were not collected according to accepted protocols (see Section 4.1.1), the invertebrate communities sampled at the four Berg River sites over the sampling period are assigned to a biotic integrity category (class) as described above, as a hypothetical "test case". Such a procedure could form the basis of the RDM procedure for the Berg River (see Table 5.4), and could be applied to other rivers affected by IBTs. According to the categories described above, all sites fell into the unimpaired – class A – or the moderately impaired category – class B (Table 5.4). During summer months, when the tunnel was releasing water, the total scores and ASPTs at impacted sites (BR3 and BR4) indicated that these sites generally were of lower biotic integrity compared with the sites above the IBT. Although this trend was not consistent, and there was some seasonal variation, the deterioration in biotic integrity appeared to occur towards the end of each summer season, after a few months of water release. It is also clear from Table 5.4 that biotic integrity, in terms of macroinvertebrates, was generally lower at the two sites below the IBT.

It is recommended that such a procedure be followed for all water resources affected by IBTs and should, indeed, be repeated at the Berg River sites according to the correct sampling protocol. SASS4 data should be collected above and below all IBT outlets, where possible, and scores should then be compared between sites, to assess the deterioration in biotic integrity directly caused by the IBT, and also compared with reference conditions for the appropriate bioregion and subregion, or river type, in order to assess the impact in the context of the whole catchment.

Such data collection would dovetail with the RDM procedure described above, as it would provide data for the determination of present state classes and consequently, management classes, the Ecological Reserve, and the appropriate RQOs. Furthermore, this process would provide a national database on the effects of IBTs on biotic integrity. This would prove useful in the development of mitigatory measures for the improved management of IBTs. For example, an IBT that is operational for only a few months of the year, thus preventing a deterioration in the biotic integrity of the recipient river, is more acceptable than a scheme that operates throughout the year. Such operational rules could be applied elsewhere.

5.5 RESEARCH NEEDS

It is clear from Tables 5.1 and 5.2 that empirical evidence is required in order to confirm the links between the predicted ecological effects of IBTs and the associated ecosystem responses. The tables should be used to prioritise further research on the effects of IBTs. For example, the effects of changes in flow, particularly increases in discharge and reductions in the variability of flow throughout the year, and the implications for the riverine communities and ecosystem functioning, should be investigated. Such research would add to the body of work on the instream flow requirements of aquatic communities and ecosystems.

Further, this project failed to identify which of the physical and chemical changes recorded in summer below the IBT in the Berg River were more important determinants of the observed shifts in invertebrate community composition. This will require the use of experimental systems, such as those developed and described by Palmer *et al.* (1996), where changes in water chemistry and flow regime can be manipulated. In this way, species-level responses to IBTs can be determined.

The Berg River sites selected for this project were necessarily close to the IBT outlet, due to the proximity of a trout farm offtake point and, further downstream, effluent release, to the outlet. This limited our assessment of downstream effects to only two sites. Further research should attempt to investigate the effects of IBTs at the level of the whole river system or, at least, river zone. For example, the effects of the discontinuity in the establishment of the naturally diverse summer Berg River community requires further study (see Section 4.3). This would require the collection of seasonal data from the entire length of a recipient or donor river, or river zone. Such data collection should, indeed, be incorporated in the monitoring programmes instituted by all catchment management agencies throughout the country.

Lastly, one of the recommendations listed in Section 5.4 indicated that many of the negative effects on the recipient system of the RBEGS IBT, and other similar schemes, could be minimised if the transferred water were to be stored in an off-channel storage, and directly provided to the user, or allowed to enter a filtering wetland. The feasibility of this suggestion requires investigation

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1. Snaddon, C.D. and B.R. Davies, 1998. A preliminary assessment of the effects of a small inter-basin water transfer, the Riviersonderend-Berg River Transfer Scheme, Western Cape, South Africa, on discharge and invertebrate community structure. *Regulated Rivers: Research and Management*, **14**: 421-441.

2. Snaddon, C.D., M.J. Wishart and B.R. Davies, 1998. Some Implications of Inter-Basin Water Transfers for River Ecosystem Functioning and Water Resources Management in Southern Africa. *Journal of Aquatic Ecosystem Health and Management*, **1**: 159-182.

3. Snaddon, C.D., 2000. The ecological implications of invertebrate community changes below a small inter-basin water transfer in the Western Cape Province, South Africa. Proceedings of the 1998 Symposium of the International Association of Theoretical and Applied Limnology.

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4. Davies, B.R., C.D. Snaddon, M.J. Wishart, M. Thoms and M. Meador, 2000. Implications of inter-basin transfers of water for river conservation and management. In P.J. Boon, B.R. Davies and G.E. Petts (Eds), *Global Perspectives of River Conservation: Science, Policy and Practice.* John Wiley & Sons, Chichester, UK. (in press).

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5. Snaddon, C.D. and B.R. Davies, 1997. *An Analysis of Effects of Inter-Basin Water Transfers in Relation to the New Water Law.* A report submitted to DWA&F, January 1997, 50 pp.

6. Snaddon, C.D., 1998. Some of the Ecological Effects of a Small Inter-Basin Water Transfer on the Receiving Reaches of the Upper Berg River, Western Cape. Unpublished MSc Thesis, University of Cape Town, 130 pp.

7. Snaddon, C.D., Davies, B.R & Wishart, M.J. 2000. A Global Overview of Inter-Basin Water Transfers Schemes, with an Appraisal of their Ecological, Socio-Economic and Socio-Political Implications, and Recommendations for their Management. Report for the Water Research Commission, Pretoria.

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8. Davies, B.R., C.D. Snaddon and M. Wishart, 1996. Some implications of inter-basin water transfers for river functioning and water resources management in Southern Africa. *Victoria Falls Conference on Aquatic Ecosystems*, Elephant Hills Hotel, Zimbabwe, July, 1996.

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Appendix B

A complete list of physical and chemical data recorded from the upper Berg River, over the study period, March 1994 to August 1997.

Discharge ($m^3 s^{-1}$)	BR1	BR3	BR4
Mar-94	0.09	4.10	
May-94	0.16	1.35	
Jul-94	10.31	12.11	
Oct-94	1.00	1.36	
Nov-94	0.20	3.53	
Jan-95	0.13	2.93	
Apr-95	0.12	0.80	
Aug-95	2.60	3.77	
Oct-95	0.64	1.10	0.60
Dec-95	0.31	2.70	2.86
Feb-96	1.07	2.01	3.07
Mar-96	0.16	1.70	1.79
May-96	0.27	0.18	0.46
Aug-96	1.99	2.48	
Oct-96	0.82	1.03	1.08
Dec-96	0.84	1.82	1.60
Feb-97	0.24	3.98	2.93
Apr-97	0.11	0.87	0.96
Jun-97	0.32	0.36	0.36
Aug-97	0.75	0.77	0.71

Depth (m)	BR1	BR3	BR4
Mar-94	0.13	0.39	
May-94	0.18	0.23	
Jul-94	0.41	0.30	
Oct-94	0.26	0.23	
Nov-94	0.21	0.26	
Jan-95	0.19	0.22	
Apr-95	0.20	0.11	
Aug-95	0.39	0.26	
Oct-95	0.29	0.13	0.24
Dec-95	0.25	0.21	0.36
Feb-96	0.36	0.55	0.43
Mar-96	0.21	0.49	0.28
May-96	0.24	0.18	0.19
Aug-96	0.32	0.38	
Oct-96	0.29	0.35	0.26
Dec-96	0.35	0.45	0.34
Feb-97	0.21	0.66	0.37
Apr-97	0.19	0.34	0.24
Jun-97	0.24	0.22	0.19
Aug-97	0.27	0.30	0.15

Width (m)	BR1	BR3	BR4
Mar-94	5.25	17.30	
May-94	5.10	17.50	
Jul-94	19.35	16.95	
Oct-94	16.60	12.15	
Nov-94	11.45	18.80	
Jan-95	13.00	18.70	
Apr-95	13.05	18.35	
Aug-95	16.70	18.80	
Oct-95	14.10	17.35	9.85
Dec-95	12.90	18.40	12.20
Feb-96	14.10	8.60	11.90
Mar-96	12.00	8.45	11.50
May-96	9.60	6.40	4.40
Aug-96	14.65	9.03	
Oct-96	14.74	9.25	10.25
Dec-96	17.30	9.55	10.00
Feb-97	11.80	9.85	13.25
Apr-97	12.15	8.20	9.70
Jun-97	12.05	7.90	5.30
Aug-97	14.50	7.85	12.20

Appendix B continued...

pH	BR1	BR2	BR3	BR4
Mar-94	5.5	5.6	7.1	
May-94	4.5	4.5	6.3	
Jul-94	4.2	4.1	4.2	
Oct-94	4.9	4.9	5.0	
Nov-94	5.5	5.5	6.5	
Jan-95	5.3	5.4	6.9	
Apr-95	5.5	5.6	6.3	
Aug-95	4.0		4.5	4.3
Oct-95	4.5	4.5	4.6	4.6
Dec-95	5.1	5.1	6.1	6.4
Feb-96	4.3	4.3	4.9	6.0
Mar-96	4.8	4.7	6.5	7.1
May-96	4.5	4.7	5.5	5.7
Aug-96	4.6	4.5	4.6	
Oct-96	4.7	4.7	4.9	4.9
Dec-96	4.0	4.0	4.2	4.2
Feb-97	4.9	5.3	6.0	6.1
Apr-97	5.7	5.6	6.1	7.0
Jun-97	4.7	4.7	4.9	5.0
Aug-97	5.4		5.6	5.8

Conductivity (mS m^{-1})	BR1	BR2	BR3	BR4
Mar-94	1.83	1.76	4.71	
May-94	1.97	2.20	4.20	
Jul-94	2.34	1.70	2.25	
Oct-94	2.91	3.57	3.53	
Nov-94	2.12	2.12	3.61	
Jan-95	1.84	1.82	4.88	
Apr-95	2.25	2.13	2.85	
Aug-95	3.18		2.32	2.32
Oct-95				
Dec-95	2.64	2.61	4.67	5.67
Feb-96	2.41	2.40	2.11	4.07
Mar-96	2.29	2.51	5.62	6.47
May-96	2.92	2.77	2.77	2.58
Aug-96	2.81	2.58	2.36	
Oct-96	3.28	2.86	2.47	2.61
Dec-96	2.26	2.40	2.07	2.14
Feb-97	2.31	2.52	4.76	5.60
Apr-97	2.12	2.35	2.31	5.24
Jun-97	2.80	3.17	2.89	2.91
Aug-97	2.11	2.11	2.14	2.18

Temperature (°C)	BR1	BR2	BR3	BR4
Mar-94	21.5	22	22.5	
May-94	12	11.5	13.5	
Jul-94	11.5	11	10	
Oct-94	16.5	13.5	12	
Nov-94	24	22	20	
Jan-95	24.75	23	22.1	
Apr-95	18.4	18.5	18	
Aug-95	10.1		10.5	10.9
Oct-95	13.0		13.9	13.0
Dec-95	17.5		17.0	
Feb-96	18.0	18.0	18.0	21.0
Mar-96	17.5	18.3	19.7	21.0
May-96	12.2	13.7	12.0	14.5
Aug-96	11.5	11.0	10.5	
Oct-96	14.5	15.0	12.5	16.0
Dec-96	13.0	13.3	12.0	14.0
Feb-97	21.0	21.1	21.0	22.0
Apr-97	29.0	28.9	28.4	29.6
Jun-97	11.0	11.0	11.0	11.0
Aug-97	13.5		13.7	14.0

Appendix B continued...

Chloride (mg l^{-1})	BR1	BR2	BR3	BR4	Sulphate (mg l^{-1})	BR1	BR2	BR3	BR4
Mar-94	5.643	5.142	11.835		Mar-94	0.844	0.65	3.478	
May-94	6.311	6.465	9.131		May-94	0.803	0.982	1.837	
Jul-94	3.806	3.645	3.172		Jul-94	0	0	0	
Oct-94	3.833	3.824	3.839		Oct-94	0.986	0.949	0.979	
Nov-94	3.947	4.359	3.436		Nov-94	0.876	0.982	1.355	
Jan-95	5.085		12.423		Jan-95	0.665		3.468	
Apr-95	4.1	4.875	11.362		Apr-95	0.838	2.486	3.025	
Aug-95	4.030		4.099	3.825	Aug-95	1.096		0.925	1.026
Oct-95	4.804	4.938	4.985	4.981	Oct-95	1.238	0.758	0.775	1.237
Dec-95	5.842	6.123	9.722	10.841	Dec-95	0.701	0.767	2.117	2.546
Feb-96	5.006	7.137	7.443	9.414	Feb-96	0.639	0.677	1.072	1.741
Mar-96	5.642	5.977	11.828	13.411	Mar-96	0.706	0.719	2.410	2.663
May-96	5.718	6.009	4.976	5.846	May-96	0.762	1.254	1.223	0.836
Aug-96	4.865	4.552	4.713		Aug-96	1.210	1.144	1.271	
Oct-96	5.269	4.873	4.594	4.843	Oct-96	1.425	1.344	1.290	1.242
Dec-96	3.486	3.456	2.912	3.713	Dec-96	1.013	0.990	0.981	1.280
Feb-97					Feb-97				
Apr-97	3.944	5.148	10.476	10.300	Apr-97	1.000	1.210	2.464	2.640
Jun-97					Jun-97				
Aug-97	3.978	3.927	4.762	3.982	Aug-97	1.222	1.092	1.377	1.259
Sodium (mg l^{-1})	BR1	BR2	BR3	BR4	**Potassium (mg l^{-1})**	BR1	BR2	BR3	BR4
Mar-94	3.433	2.84	6.123		Mar-94	0.294	0.169	0.858	
May-94	3.696	3.189	6.592		May-94	0.22	0.182	1.127	
Jul-94	3.685	3.986	4.115		Jul-94	0.012	0	0.094	
Oct-94	3.198	4.192	4.678		Oct-94	0	0	0.21	
Nov-94	3.725	3.197	4.553		Nov-94	0.261	0.195	0.447	
Jan-95	2.61	3.618	7.587		Jan-95	0.19	0.366	0.942	
Apr-95	3.94	3.969	7.242		Apr-95	0.491	0.536	1.474	
Aug-95	2.760		2.880	2.940	Aug-95	0.130		0.130	0.230
Oct-95	3.370	3.040	3.100	3.040	Oct-95	0.130	0.100	0.100	0.100
Dec-95	3.410		5.310	4.590	Dec-95	0.260		0.410	0.540
Feb-96	3.680	2.840	4.030	4.730	Feb-96	0.950	0.360	0.640	0.490
Mar-96	3.200	3.210	7.580	6.520	Mar-96	0.120	0.130	0.560	0.560
May-96	3.800	2.710	3.030	3.160	May-96	0.130	0.170	0.110	0.150
Aug-96	2.810	3.470	3.170		Aug-96	0.100	0.180	0.100	
Oct-96	3.530	3.270	3.270	3.270	Oct-96	0.180	0.250	0.140	0.310
Dec-96	2.560	2.520		2.460	Dec-96	0.360	0.190		0.080
Feb-97					Feb-97				
Apr-97	2.530	3.420	6.290	6.470	Apr-97	0.110	0.130	0.440	0.490
Jun-97					Jun-97				
Aug-97	2.600	2.810	3.730	2.820	Aug-97	0.100	0.120	0.270	0.120

Appendix B continued...

Magnesium (mg l^{-1})	BR1	BR2	BR3	BR4	Calcium (mg l^{-1})	BR1	BR2	BR3	BR4
Mar-94	0.449	0.237	1.45		Mar-94	0.003	0.546	2.026	
May-94	0.221	0.318	1.083		May-94	0.096	0.169	1.893	
Jul-94	0	0	0.014		Jul-94	0.056	0.036	0.062	
Oct-94	0	0.015	0		Oct-94	0.042	0.072	0.043	
Nov-94	0.091	0.218	0.499		Nov-94	0.348	0.608	1.085	
Jan-95	0.179	0.219	0.934		Jan-95	0.432	0.428	2.404	
Apr-95	0.264	0.328	0.774		Apr-95	0.406	0.582	1.77	
Aug-95	0.550		0.410	0.280	Aug-95	0.850		0.520	0.360
Oct-95	0.480	0.430	0.430	0.320	Oct-95	0.500	0.430	0.510	0.400
Dec-95	0.400		0.920	0.900	Dec-95	0.370		1.570	1.560
Feb-96	0.840	0.680	0.670	1.100	Feb-96	1.190	0.830	1.210	2.090
Mar-96	0.610	0.310	1.470	1.630	Mar-96	0.840	0.330	2.800	3.530
May-96	0.670	0.350	0.350	0.350	May-96	0.980	0.380	0.520	0.780
Aug-96	0.470	0.340	0.510		Aug-96	0.520	0.310	0.830	
Oct-96	0.580	0.340	0.510	0.490	Oct-96	0.940	0.370	0.730	0.670
Dec-96	0.300	0.380		0.380	Dec-96	0.590	0.520		0.510
Feb-97					Feb-97				
Apr-97	0.370	0.430	1.400	1.270	Apr-97	0.440	0.520	2.300	2.410
Jun-97					Jun-97				
Aug-97	0.330	0.340	0.400	0.370	Aug-97	0.370	0.410	0.670	0.640

Appendix C

A complete list of the aquatic macroinvertebrate taxa, and the abundance of individuals m^{-2} at the four sites on the upper Berg River, over the sampling period. **Pages viii – xv.**

TAXON		March 1994 BR1	March 1994 BR2	March 1994 BR3	May 1994 BR1	May 1994 BR2	May 1994 BR3	July 1994 BR1	July 1994 BR2	July 1994 BR3	October 1994 BR1	October 1994 BR2	October 1994 BR3	November 1994 BR1	November 1994 BR2	November 1994 BR3
Cnidaria																
Hydrozoa	*Hydra* sp.	0	0	19213	0	0	193	0	0	0	0	0	0	0	0	10
Platyhelminthes																
Turbellaria		5	0	47	0	0	0	0	0	0	0	0	0	0	0	0
Nematoda		0	0	13	3	13	0	30	80	67	47	73	13	0	0	0
Annelida																
OLIGOCHAETA	Lumbriculidae	10	273	0	467	33	0	47	157	57	233	170	133	247	380	190
Arthropoda																
Acarina		860	613	467	150	177	133	77	167	90	140	173	37	647	363	110
Crustacea (all zooplanktonic groups)																
CLADOCERA		0	43	10153	200	40	11367	37	47	37	0	20	3	133	233	314400
COPEPODA		0	13	200937	3	7	15057	47	153	53	3	20	7	0	47	285547
OSTRACODA		65	0	43	0	0	7	7	0	13	7	0	13	0	13	27
AMPHIPODA	*Paramelita* sp.	0	0	0	0	0	0	3	3	0	0	0	0	0	0	0
Insecta																
PLECOPTERA	Notonemouridae	100	83	0	23	30	0	170	217	80	197	240	80	240	113	207
EPHEMEROPTERA	Baetidae	6170	12327	3267	4570	5217	3500	197	167	197	7997	8907	4027	63867	86113	27183
	Heptageniidae	55	673	0	27	213	0	0	0	3	3	10	0	20	30	3
	Ephemerellidae	65	83	7	50	343	0	27	53	33	73	103	57	7	40	0
	Caenidae	0	0	83	0	0	0	0	13	0	0	0	3	13	0	27
	Leptophlebiidae	990	1657	10	917	703	10	20	30	20	83	27	20	320	227	43
TRICHOPTERA	Hydropsychidae	305	223	46993	173	100	5547	7	7	17	0	0	10	70	27	87
	Philopotamidae	35	150	0	970	97	0	20	10	0	0	0	0	23	3	0
	Leptoceridae	290	323	13	473	223	0	10	10	0	10	20	20	23	40	0
	Hydroptilidae	115	100	20	17	10	7	0	30	3	13	3	3	0	0	0
	Polycentropodidae	0	17	7	7	0	53	0	0	0	0	0	0	0	0	13
	Ecnomidae	10	63	0	150	53	0	0	3	0	23	7	0	33	27	3
	Sericostomatidae	0	3	0	0	0	0	0	0	0	0	0	0	0	0	0
	Barbarochthonidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Petrothrincidae	0	0	0	0	0	0	0	0	0	0	0	0	0	3	0
	Glossosomatidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Lepidostomatidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	pupae	0	0	170	0	0	3	0	0	0	0	0	0	0	0	0
ODONATA	Libellulidae	0	0	193	7	0	27	0	0	0	0	0	0	0	0	0
	Aeshnidae	15	0	3	0	7	3	0	0	0	3	0	0	10	0	13
	Chlorolestidae	0	7	0	0	0	0	0	0	0	0	0	0	0	0	0
	Anisopteran juveniles	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Gomphidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
MEGALOPTERA	Corydalidae	5	13	13	27	7	0	0	0	3	3	0	7	13	7	0

TAXON			March 1994 BR1	BR2	BR3	May 1994 BR1	BR2	BR3	July 1994 BR1	BR2	BR3	October 1994 BR1	BR2	BR3	November 1994 BR1	BR2	BR3
COLEOPTERA	Elmidae	larvae	2045	2140	350	1293	637	233	83	103	53	127	180	47	783	457	177
	Elmidae	adults	90	117	3	113	40	10	10	33	0	10	3	10	150	147	127
	Hydraenidae	larvae	0	0	0	7	63	13	13	50	53	30	43	7	20	0	0
	Hydraenidae	adults	275	790	63	790	137	110	7	73	10	17	30	13	390	167	153
	Hydrophilidae	larvae	0	0	0	0	0	0	0	27	3	0	0	0	0	0	0
	Hydrophilidae	adults	0	3	0	0	0	0	3	0	0	0	0	0	0	0	0
	Dryopidae	larvae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Dryopidae	adults	0	3	0	3	0	0	3	3	0	0	0	0	0	0	0
	Limnichidae		5	0	0	7	0	0	0	0	0	0	0	0	3	0	0
	Helodidae		10	27	3	117	93	10	50	83	17	150	197	33	130	157	30
	Gyrinidae		0	10	0	10	3	0	0	0	0	3	0	0	0	13	7
	Hydrochidae		0	0	0	0	0	0	0	0	0	7	0	0	0	0	0
	Staphylinidae		0	0	0	0	0	0	3	0	0	0	0	0	0	0	0
	Noteridae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
HEMIPTERA	Veliidae		10	0	0	0	3	0	0	0	0	0	0	0	3	0	0
	Corixidae		0	0	0	0	0	0	0	3	0	0	0	0	0	0	0
	Hebridae		0	0	0	0	0	0	0	0	0	37	0	0	0	3	0
	Pleidae		0	0	0	0	0	0	0	0	0	0	3	3	0	0	0
	Mesoveliidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Notonectidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Belostomatidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
DIPTERA	Chironomidae		33950	22147	19680	14993	14820	16517	3133	5247	2790	8757	7683	5707	27567	14063	11960
	Simuliidae		12690	11550	993	970	1040	4180	97	347	137	210	73	267	6063	3447	15043
	Athericidae		100	103	63	33	53	17	0	0	0	0	0	0	140	327	20
	Empididae		10	0	33	3	13	13	0	7	0	0	0	0	27	0	10
	Blephariceridae		10	0	0	0	0	7	0	0	10	0	0	0	0	0	0
	Ceratopogonidae		5	3	0	0	0	0	23	47	33	10	10	0	0	0	0
	Stratiomyidae		5	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Tipulidae		0	0	0	0	0	0	0	3	0	3	0	0	0	0	0
	Psychodidae		0	0	0	0	0	0	0	0	0	3	0	3	0	0	0
	Chaoboridae		0	0	823	0	0	90	0	0	0	0	0	0	0	3	4707
	Dixidae		0	0	0	0	0	0	0	0	0	0	0	0	3	0	0
	Tabanidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Ephydridae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	pupae		65	57	97	93	70	80	3	10	13	133	100	133	37	27	847
	coneworm		5	0	0	0	0	0	0	0	0	0	0	0	0	0	0
LEPIDOPTERA	Nymphulidae		5	0	0	0	0	0	3	3	0	0	0	0	0	3	0
HYMENOPTERA			0	0	0	0	3	0	10	0	0	83	80	27	3	0	0
Collembola			0	3	3	50	13	10	570	697	763	100	113	67	13	0	13

TAXON		January 1995 BR1	BR2	BR3	April 1995 BR1	BR2	BR3	August 1995 BR1	BR3	BR4	October 1995 BR1	BR2	BR3	BR4	December 1995 BR1	BR2	BR3	BR4
Cnidaria																		
Hydrozoa	*Hydra* sp.	0	0	6200	0	0	1230	0	0	0	0	0	0	3	0	0	0	0
Platyhelminthes																		
Turbellaria		0	20	0	0	0	30	0	0	0	0	0	0	0	3	0	0	0
Nematoda		25	0	5	3	0	30	0	3	3	0	0	0	0	0	0	0	0
Annelida																		
OLIGOCHAETA	Lumbriculidae	205	40	3005	883	197	387	10	20	40	73	0	17	23	47	5	160	57
Arthropoda																		
Acarina		375	40	90	403	170	130	15	3	73	217	10	70	190	303	185	0	53
Crustacea (all zooplanktonic groups)																		
CLADOCERA		275	30	51040	40	0	80927	0	0	0	0	0	0	0	40	360	124365	210547
COPEPODA		*5*	*30*	*77035*	*7*	*7*	*56610*	*5*	*10*	*0*	*0*	*0*	*7*	*40*	*400*	*665*	*278320*	*318280*
OSTRACODA		0	0	120	0	0	443	0	0	0	0	0	0	0	0	0	0	0
AMPHIPODA	*Paramelita* sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Insecta																		
PLECOPTERA	Notonemouridae	45	0	0	47	43	0	220	143	347	797	70	213	293	603	205	260	110
EPHEMEROPTERA	Baetidae	2450	870	870	7960	6803	1493	3355	1113	2427	8103	720	13147	6567	26927	42845	9595	12400
	Heptageniidae	315	150	0	330	1607	0	25	7	110	20	140	7	0	20	15	5	13
	Ephemerellidae	215	110	5	67	270	0	405	70	130	140	470	93	130	60	110	10	53
	Caenidae	0	0	580	0	13	17	0	0	0	0	10	0	0	0	5	80	3
	Leptophlebiidae	615	40	5	1470	1107	3	90	80	160	417	30	53	10	170	155	320	43
TRICHOPTERA	Hydropsychidae	435	0	920	997	123	32870	0	0	23	13	10	3	20	100	60	15	3
	Philopotamidae	50	20	0	317	0	0	0	0	0	0	0	0	3	0	0	0	3
	Leptoceridae	325	90	20	290	503	0	0	7	10	57	20	57	43	37	55	0	10
	Hydroptilidae	10	0	105	43	33	0	0	0	0	123	20	7	0	7	0	0	0
	Polycentropodidae	0	0	5	50	7	0	0	3	7	0	10	3	3	0	0	0	0
	Ecnomidae	5	30	10	577	60	0	0	3	0	103	20	3	3	13	25	0	0
	Sericostomatidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Barbarochthonidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Petrothrincidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Glossosomatidae	0	0	0	0	0	0	0	0	0	7	0	0	3	0	0	0	0
	Lepidostomatidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	pupae	0	0	20	0	0	247	0	0	0	0	0	0	0	0	0	0	0
ODONATA	Libellulidae	0	0	15	0	0	103	0	0	3	0	0	0	0	0	0	5	0
	Aeshnidae	25	0	5	7	0	0	0	0	0	0	0	0	0	0	0	0	0
	Chlorolestidae	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*0*	0	0	0	0	0	0
	Anisopteran juveniles	15	0	0	0	0	0	0	0	0	7	0	0	0	0	0	0	0
	Gomphidae	0	0	0	0	0	0	0	0	7	0	0	0	0	0	0	0	0
MEGALOPTERA	Corydalidae	5	0	0	13	37	0	0	0	0	13	0	0	3	50	10	10	3

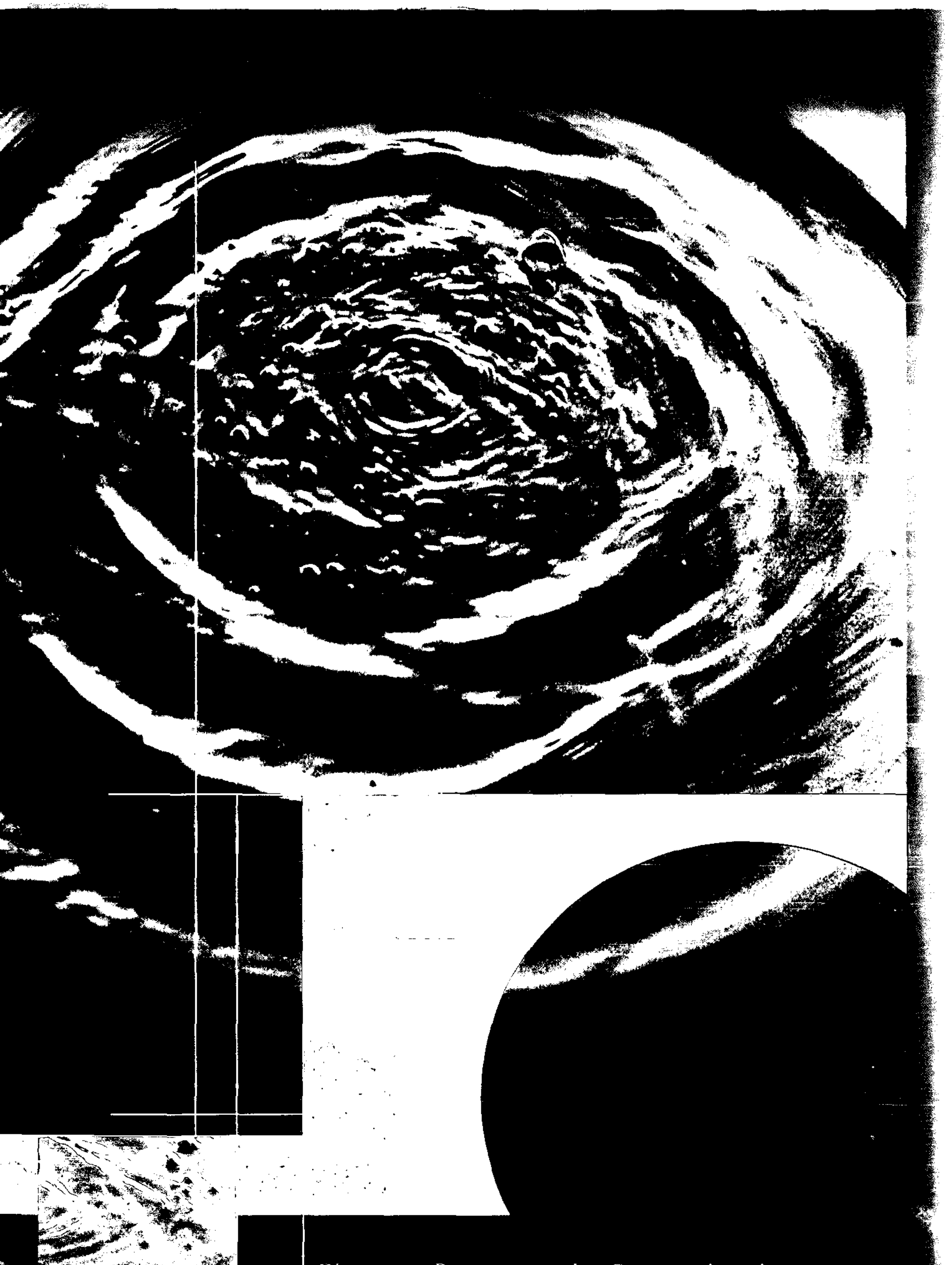
TAXON			January 1995 BR1	BR2	BR3	April 1995 BR1	BR2	BR3	August 1995 BR1	BR3	BR4	October 1995 BR1	BR2	BR3	BR4	December 1995 BR1	BR2	BR3	BR4
COLEOPTERA	Elmidae	larvae	355	120	120	2140	1897	340	10	20	77	853	90	87	137	923	640	175	570
	Elmidae	adults	110	80	0	193	710	7	0	7	0	100	20	80	100	63	155	80	93
	Hydraenidae	larvae	0	0	0	0	0	0	0	50	13	17	0	3	7	3	0	0	0
	Hydraenidae	adults	480	10	20	173	133	23	0	3	3	90	0	43	60	307	175	85	57
	Hydrophilidae	larvae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Hydrophilidae	adults	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Dryopidae	*larvae*	*0*	*0*	*0*	*0*	*0*	*0*	*15*	*0*	*0*	*17*	*0*	*0*	*0*	*0*	*0*	*0*	*0*
	Dryopidae	adults	0	0	0	0	7	0	0	3	0	0	0	0	0	0	0	0	0
	Limnichidae		0	0	0	7	0	0	0	0	0	0	0	0	0	0	0	0	0
	Helodidae		45	10	0	77	63	0	5	40	67	777	70	53	63	167	65	5	7
	Gyrinidae		5	0	0	10	3	0	0	0	0	0	0	3	0	0	0	0	0
	Hydrochidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Staphylinidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Noteridae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
HEMIPTERA	Veliidae		0	0	0	0	0	0	0	0	0	0	10	0	0	0	0	0	17
	Corixidae		0	0	0	3	0	7	0	0	0	0	0	0	0	0	0	0	0
	Hebridae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Pleidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Mesoveliidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Notonectidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Belostomatidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
DIPTERA	Chironomidae		4535	1140	4290	7863	3420	5800	3280	2370	3687	15137	1930	12747	23530	16473	10825	18250	10293
	Simuliidae		8390	2330	1580	4933	2493	7710	65	177	40	473	10	850	357	607	485	2360	1380
	Athericidae		40	30	10	90	237	17	5	0	3	3	0	3	3	57	195	5	3
	Empididae		5	0	25	3	0	7	0	0	3	13	10	0	17	10	0	15	3
	Blephariceridae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Ceratopogonidae		0	30	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0
	Stratiomyidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Tipulidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Psychodidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Chaoboridae		0	0	940	0	0	660	0	0	0	0	0	0	0	0	0	130	43
	Dixidae		0	0	0	3	0	0	0	0	0	0	0	0	0	0	0	0	0
	Tabanidae		0	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0
	Ephydridae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	pupae		20	40	150	60	107	520	35	33	50	70	30	140	137	17	45	60	63
	coneworm		0	0	0	0	0	0	0	0	0	17	0	0	0	0	0	0	0
LEPIDOPTERA	Nymphulidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
HYMENOPTERA			0	0	0	17	0	0	5	0	0	23	10	0	10	0	15	0	13
Collembola			0	0	0	27	3	0	15	70	53	37	0	0	3	33	40	0	0

TAXON		February 1996 BR1	BR2	BR3	BR4	March 1996 BR1	BR2	BR3	BR4	May 1996 BR1	BR2	BR3	BR4	August 1996 BR1	BR2	BR3	October 1996 BR1	BR2	BR3	BR4	December 1996 BR1	BR2	BR3	BR4
Cnidaria																								
Hydrozoa	*Hydra* sp.	7	3	2973	18297	0	10	11617	10983	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Platyhelminthes																								
Turbellaria		0	0	0	0	0	0	3	7	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nematoda		0	0	3	0	0	0	0	0	0	0	0	0	7	0	0	60	0	0	7	0	7	0	3
Annelida																								
OLIGOCHAETA	Lumbriculidae	90	200	120	347	437	10	200	273	100	20	20	7	10	43	7	30	3	0	3	23	47	63	73
Arthropoda																								
Acarina		7	37	43	0	577	560	0	20	150	70	37	87	47	30	7	303	77	20	83	213	170	187	87
Crustacea (all zooplanktonic groups)																								
CLADOCERA		27	73	5707	6453	0	0	807	727	0	0	0	0	3	0	0	0	0	0	0	0	0	0	0
COPEPODA		33	20	12093	12027	0	0	7763	5183	0	3	0	40	7	0	0	0	0	0	0	0	0	0	0
OSTRACODA		0	0	0	0	0	30	0	0	0	0	0	0	0	0	0	0	0	0	7	0	0	0	0
AMPHIPODA	*Paramelita* sp.	0	0	0	0	0	0	0	0	0	0	7	0	0	0	0	0	0	0	0	0	0	0	0
Insecta																								
PLECOPTERA	Notonemouridae	10	47	10	0	177	95	0	0	103	50	10	10	357	120	30	1257	147	33	200	277	377	397	158
EPHEMEROPTERA	Baetidae	390	447	180	277	7850	18410	3173	3197	7827	10323	6363	3197	407	287	87	15887	8400	5543	6023	32587	35400	37523	28998
	Heptageniidae	157	167	3	3	273	665	0	7	163	517	47	0	0	3	0	20	0	0	0	20	0	0	0
	Ephemerellidae	933	437	23	140	63	255	27	53	27	303	17	23	200	110	33	130	100	27	77	17	53	47	47
	Caenidae	73	13	10	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	13	0	0	0
	Leptophlebiidae	60	180	0	0	860	440	17	20	887	730	3	13	77	67	20	60	0	7	23	47	97	53	53
TRICHOPTERA	Hydropsychidae	0	3	127	170	316	65	4227	3533	274	47	1197	990	7	7	0	7	0	0	0	3	0	27	60
	Philopotamidae	0	7	0	0	67	40	0	0	250	0	3	0	0	0	0	0	0	0	0	0	0	27	0
	Leptoceridae	150	67	0	30	187	265	20	13	420	323	23	7	13	10	7	53	13	0	13	7	17	17	3
	Hydroptilidae	257	133	73	93	13	45	13	13	10	3	0	0	3	0	0	20	0	3	0	3	0	7	0
	Polycentropodidae	0	0	87	3	0	0	0	0	220	10	0	7	0	0	0	0	0	3	3	7	0	0	0
	Ecnomidae	0	3	0	0	450	0	0	0	93	30	3	0	3	7	0	23	7	0	3	0	10	0	3
	Sericostomatidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Barbarochthonidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Petrothrincidae	0	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Glossosomatidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Lepidostomatidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	7	0	3	0	0	0	0	0
	pupae	0	0	3	0	0	0	433	73	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ODONATA	Libellulidae	0	0	33	0	0	0	20	7	0	0	0	7	0	0	0	0	0	0	0	0	0	0	0
	Aeshnidae	0	0	3	0	0	10	10	3	0	0	0	7	0	0	0	0	0	0	0	0	0	0	0
	Chlorolestidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Anisopteran juveniles	0	13	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Gomphidae	0	0	0	0	0	0	0	0	3	0	10	0	0	0	0	0	0	0	0	0	0	0	0
MEGALOPTERA	Corydalidae	3	7	0	0	17	5	3	0	13	0	3	13	3	0	0	0	0	0	0	0	0	3	0

TAXON			February 1996				March 1996				May 1996				August 1996			October 1996				December 1996			
			BR1	BR2	BR3	BR4	BR1	BR2	BR3	BR4	BR1	BR2	BR3	BR4	BR1	BR2	BR3	BR1	BR2	BR3	BR4	BR1	BR2	BR3	BR4
COLEOPTERA	Elmidae	larvae	257	500	63	237	1383	970	407	430	1360	733	353	353	43	10	27	153	7	87	123	80	137	173	265
	Elmidae	adults	0	57	0	57	157	220	0	0	167	30	33	7	3	7	3	20	13	3	20	7	7	7	27
	Hydraenidae	larvae	3	73	0	0	0	0	0	0	0	7	0	7	0	7	27	3	13	3	7	0	3	3	0
	Hydraenidae	adults	7	10	3	17	433	85	23	37	473	80	137	157	0	3	0	50	0	30	13	50	17	150	23
	Hydrophilidae	larvae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Hydrophilidae	adults	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Dryopidae	larvae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0
	Dryopidae	adults	0	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0	3	0
	Limnichidae		0	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0	0	0	0	0	0
	Helodidae		63	400	0	3	43	20	7	0	367	413	3	0	200	173	60	223	37	7	363	147	297	413	387
	Gyrinidae		0	0	0	0	0	0	0	0	0	0	0	3	0	0	0	3	0	0	0	0	0	0	0
	Hydrochidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Staphylinidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Noteridae		7	0	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0	0	0	0	0
HEMIPTERA	Veliidae		7	17	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Corixidae		0	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Hebridae		0	0	0	0	0	0	0	0	3	0	7	0	0	0	0	0	0	0	0	0	0	0	0
	Pleidae		27	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Mesoveliidae		0	0	0	0	0	0	0	0	0	0	7	0	0	0	0	0	0	0	0	0	0	0	0
	Notonectidae		0	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Belostomatidae		0	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
DIPTERA	Chironomidae		1557	3340	2270	3330	10857	8250	15737	7817	7620	2770	6777	6523	6217	7763	8732	10923	8117	10353	10467	1850	3067	2153	1050
	Simuliidae		810	600	9790	2003	2830	1445	680	180	957	327	33	280	97	37	67	340	30	12373	80	403	307	483	147
	Athericidae		13	17	0	10	43	75	20	10	87	47	23	13	0	0	0	0	0	0	0	3	50	7	3
	Empididae		0	0	0	0	37	0	7	3	0	0	23	13	13	0	0	7	3	0	0	13	13	0	7
	Blephariceridae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Ceratopogonidae		320	177	20	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0	7	0	0
	Stratiomyidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Tipulidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Psychodidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Chaoboridae		0	0	50	3	0	0	13	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Dixidae		0	33	10	53	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Tabanidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Ephydridae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	pupae		90	17	27	13	20	80	217	140	7	7	47	40	57	10	57	370	260	290	350	27	33	40	63
	coneworm		0	3	0	0	3	0	0	0	10	0	0	0	10	0	0	0	0	0	0	0	0	0	0
LEPIDOPTERA	Nymphulidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	7	0	0
HYMENOPTERA			13	0	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0	0
Collembola			250	163	280	80	0	0	0	0	0	0	0	0	13	20	30	0	0	3	10	43	50	47	15

TAXON		February 1997 BR1	BR2	BR3	BR4	April 1997 BR1	BR2	BR3	BR4	June 1997 BR1	BR2	BR3	BR4	August 1997 BR1	BR2	BR3	BR4
Cnidaria																	
Hydrozoa	*Hydra* sp.	0	0	60	513	0	0	2853	7907	0	0	0	0	0	0	0	0
Platyhelminthes																	
Turbellaria		0	0	0	0	0	0	3	3	0	0	0	0	0	0	0	0
Nematoda		0	0	3	0	0	0	0	0	17	0	0	0	0	0	3	0
Annelida																	
OLIGOCHAETA	Lumbriculidae	235	160	0	3	10	26	0	170	40	0	36	37	27	20	3	0
Arthropoda																	
Acarina		945	277	53	17	703	303	40	43	210	65	17	27	37	60	33	23
Crustacea (all zooplanktonic groups)																	
CLADOCERA		120	13	4180	793	0	3	8580	633	13	0	0	27	0	0	0	3
COPEPODA		140	553	140443	33433	20	20	60560	4133	7	0	3	10	0	0	3	70
OSTRACODA		0	0	53	0	0	0	13	0	3	0	0	0	0	0	0	0
AMPHIPODA	*Paramelita* sp.	0	0	0	0	0	3	0	0	0	0	0	0	0	0	0	0
Insecta																	
PLECOPTERA	Notonemouridae	585	517	30	27	133	163	13	0	63	70	67	77	153	203	207	60
EPHEMEROPTERA	Baetidae	11235	14890	2587	753	3270	3747	1010	1133	4907	3615	1063	2683	507	2440	933	710
	Heptageniidae	305	67	0	0	403	517	0	0	30	115	13	0	0	0	13	0
	Ephemerellidae	470	177	173	190	157	443	160	200	173	375	37	87	83	163	193	100
	Caenidae	0	0	0	0	0	0	0	0	0	5	0	0	0	0	0	0
	Leptophlebiidae	1030	193	7	27	1060	537	3	0	467	70	20	27	20	103	80	33
TRICHOPTERA	Hydropsychidae	135	70	13	10	210	73	120	240	137	5	7	7	0	3	0	10
	Philopotamidae	5	0	0	0	0	0	0	0	10	0	0	0	0	0	0	0
	Leptoceridae	65	170	0	0	293	147	13	10	293	20	27	54	7	27	3	3
	Hydroptilidae	0	3	0	0	37	3	350	37	40	10	0	3	7	13	7	0
	Polycentropodidae	20	0	23	0	60	20	13	0	13	5	0	0	0	0	0	0
	Ecnomidae	95	13	0	0	157	53	0	0	57	15	3	3	0	10	0	0
	Sericostomatidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Barbarochthonidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Petrothrincidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Glossosomatidae	0	0	0	0	13	0	0	0	0	0	0	0	0	0	0	0
	Lepidostomatidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	pupae	0	0	0	0	0	0	7	0	0	0	3	0	0	0	0	0
ODONATA	Libellulidae	0	0	7	0	7	0	3	10	0	0	0	7	0	0	0	0
	Aeshnidae	15	7	3	3	27	0	0	0	0	0	0	0	7	0	0	0
	Chlorolestidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Anisopteran juveniles	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Gomphidae	0	0	3	0	0	0	0	0	0	0	0	0	0	0	0	0
MEGALOPTERA	Corydalidae	60	13	0	0	7	10	0	0	7	0	0	3	3	0	0	0

TAXON			Feb 1997 BR1	Feb 1997 BR2	Feb 1997 BR3	Feb 1997 BR4	Apr 1997 BR1	Apr 1997 BR2	Apr 1997 BR3	Apr 1997 BR4	Jun 1997 BR1	Jun 1997 BR2	Jun 1997 BR3	Jun 1997 BR4	Aug 1997 BR1	Aug 1997 BR2	Aug 1997 BR3	Aug 1997 BR4
COLEOPTERA	*Elmidae*	*larvae*	555	390	27	153	713	593	290	480	843	225	123	533	40	77	20	50
	Elmidae	adults	230	157	50	33	53	230	10	10	140	20	0	10	0	13	0	20
	Hydraenidae	larvae	0	0	0	0	0	0	0	0	0	0	0	20	17	0	0	3
	Hydraenidae	adults	815	450	13	7	230	47	3	40	37	10	7	73	7	7	10	0
	Hydrophilidae	larvae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	7
	Hydrophilidae	adults	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Dryopidae	*larvae*	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0	0
	Dryopidae	adults	0	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Limnichidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Helodidae		195	330	17	7	53	13	3	10	360	40	27	13	380	197	243	43
	Gyrinidae		0	3	0	0	0	0	0	3	0	0	0	0	0	3	0	0
	Hydrochidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Staphylinidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Noteridae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
HEMIPTERA	Veliidae		0	0	0	0	0	0	0	0	0	5	0	0	0	0	0	3
	Corixidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Hebridae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Pleidae		0	0	0	0	0	0	3	0	0	0	0	0	0	0	0	0
	Mesoveliidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Notonectidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Belostomatidae		5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
DIPTERA	Chironomidae		11445	5367	2773	2040	5743	4883	12363	5280	4863	6390	8933	13347	6530	7143	4637	6307
	Simuliidae		2030	2330	7277	2163	1090	233	8813	250	177	200	297	1307	70	33	190	33
	Athericidae		170	10	0	0	17	40	7	10	17	0	7	10	0	0	0	0
	Empididae		25	0	0	0	0	3	13	0	0	5	0	3	0	3	3	0
	Blephariceridae		0	0	0	0	0	0	0	0	0	0	10	0	7	0	0	3
	Ceratopogonidae		5	3	0	0	3	0	0	0	0	0	0	0	0	0	0	0
	Stratiomyidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Tipulidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Psychodidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Chaoboridae		30	0	1413	103	0	0	850	0	0	0	0	0	0	0	0	0
	Dixidae		0	0	0	0	7	0	0	13	0	0	0	0	0	0	0	0
	Tabanidae		5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Ephydridae		0	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0
	pupae		70	40	723	287	23	30	270	43	7	25	60	57	30	27	120	23
	coneworm		0	0	0	0	3	0	0	0	3	0	0	0	0	0	0	0
LEPIDOPTERA	Nymphulidae		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
HYMENOPTERA			0	0	10	0	0	0	0	0	0	5	0	0	0	0	0	0
Collembola			0	0	0	0	0	0	20	0	0	0	7	13	17	0	10	3



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